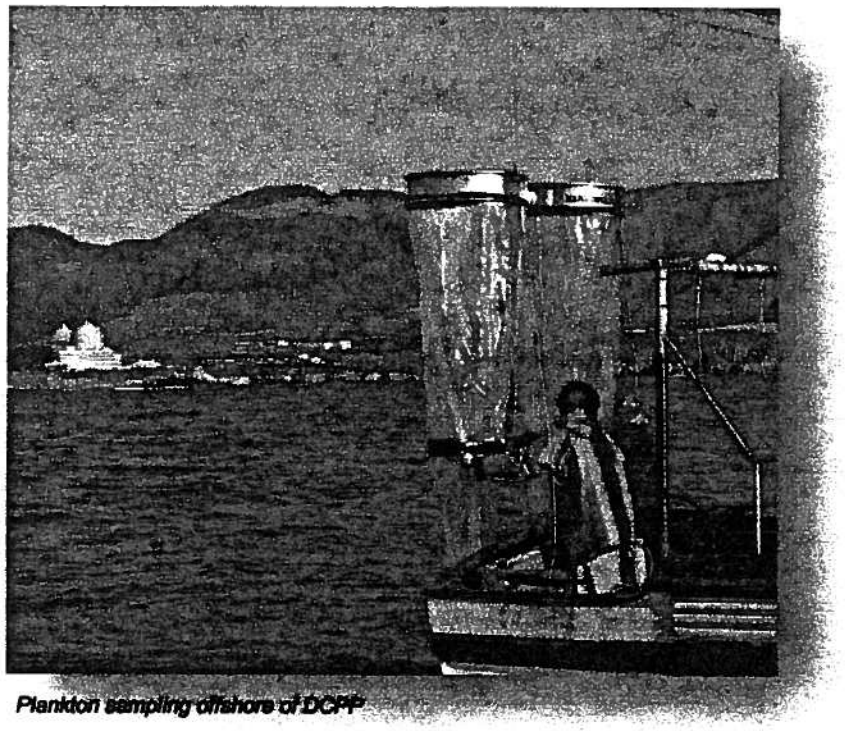
Diablo Canyon Power Plant

316(b) Demonstration Report



Plankton sampling offshore of DCPP

March 1, 2000

Document No. E9-055.0

Prepared for:

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Re: PG&E Diablo Canyon - Clean Water Act Section 308 Request Submittal -

Diablo Canyon 316(b) Demonstration Report

Dear Ms. Stuber and Mr. Mahanes:

Pursuant to Alexis Strauss' letter dated September 22, 1999, Pacific Gas & Electric (PG&E) is hereby submitting the Diablo Canyon Power Plant (DCPP) 316(b) Demonstration Report. The DCPP 316(b) study was initiated in October, 1996 with sampling completed in June, 1999. The sampling design and methods of the study, selection of target taxa, and formulation of the three assessment methods (Tenera 1997a; Tenera 1997b; Tenera 1998a) were overseen by the Entrainment Technical Work Group (ETWG) established by the Central Coast Regional Water Quality Control Board (CCRWQCB).

The enclosed DCPP 316(b) Demonstration report follows two draft preliminary assessment reports dated August 2nd, 1999, and December 10th, 1999. The purpose of these preliminary reports was to provide the CCRWQCB, Environmental Protection Agency (EPA), and the ETWG an opportunity to review a comprehensive application of the three assessment approaches (adult equivalent loss, fecundity hindcasting, and proportional entrainment) on the selected target species. These draft reports contained the life history information, assumptions, and values for each target species, as well as the methodology to be used in applying the three assessment approaches. They were provided to the ETWG with the goal of obtaining agreement on the information, methodology and assumptions that would be applied in this demonstration report.

Additionally, included with this submittal is a CD ROM which contains the following information:

- Appendix G
- Appendix H
- Appendix I
- Current Meter Data
- Entrainment and Grid Length Data
- Entrainment and Grid Survey Data
- Adjusted Inshore Volume
- Intake Cove Plankton Tow Data
- Power Plant Intake Flow
- Wind Data

I certify, under penalty of law, that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to ensure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system, or those persons directly responsible for gathering the information, I certify that the information submitted is, to the best of my knowledge and belief, true, accurate, and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations.

If you have any questions, please contact Anne Jackson of my staff at (415) 973-6523.

Sincerely,

R. A. Walters for
David H. Oatley

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March 1, 2000

PG&E Letter DCL-2000-514

Mr. Roger Briggs, Executive Officer
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Re: PG&E Diablo Canyon Submittal -

Diablo Canyon Power Plant - 316(b) Demonstration Report

Dear Mr. Briggs:

In accordance with your letter dated July 26, 1999, Pacific Gas & Electric (PG&E) is hereby submitting the Diablo Canyon Power Plant (DCPP) 316(b) Demonstration Report. The DCPP 316(b) study was initiated in October, 1996 with sampling completed in June, 1999. The sampling design and methods of the study, selection of target taxa, and formulation of the three assessment methods (Tenera 1997a; Tenera 1997b; Tenera 1998a) were overseen by the Entrainment Technical Work Group (ETWG) established by the Central Coast Regional Water Quality Control Board (CCRWQCB).

The enclosed DCPP 316(b) Demonstration report follows two draft preliminary assessment reports dated August 2nd, 1999, and December 10th, 1999. The purpose of these preliminary reports was to provide the CCRWQCB, Environmental Protection Agency (EPA), and the ETWG an opportunity to review a comprehensive application of the three assessment approaches (adult equivalent loss, fecundity hindcasting, and proportional entrainment) on the selected target species. These draft reports contained the life history information, assumptions, and values for each target species, as well as the methodology to be used in applying the three assessment approaches. They were provided to the ETWG with the goal of obtaining agreement on the information, methodology and assumptions that would be applied in this demonstration report.

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I certify, under penalty of law, that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to ensure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system, or those persons directly responsible for gathering the information, I certify that the information submitted is, to the best of my knowledge and belief, true, accurate and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations.

If you have any questions, please contact Anne Jackson of my staff at (415) 973-6523.

Sincerely,

R.A. Walter for
David H. Oatley

Enclosure

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Diablo Canyon Power Plant

316(b) Demonstration Report

March 1, 2000

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GLOSSARY OF TERMS AND ABBREVIATIONS

AEI	Adverse Environmental Impact: According to USEPA (1977), "adverse aquatic environmental impacts occur whenever there will be entrainment or impingement damage as a result of the operation of a specific cooling water intake structure. The critical question is the magnitude of any adverse impact."
AEL	Adult Equivalent Loss: Forecasts the number of adults that would have resulted from the number of entrained larvae, had they survived entrainment. Calculated using estimates of natural mortality rates applied to various life stages.
ambient	Describes the conditions and factors that are present in the natural environment.
ASW	Auxiliary seawater pump supply cooling water to the safety related component cooling water heat exchangers that maintain the operating temperature of vital plant equipment.
BTA	Best Technology Available: In USEPA regulations, refers to intake technology that is the best available to minimize AEI.
°C	Degrees Centigrade (or Celsius)
CalCOFI	California Cooperative Oceanic Fisheries Investigations
compensation	The ability of a [fish] population to offset, in whole or part, reductions in numbers caused by impacts from natural and/or man-induced stresses.
CV	Coefficient of variation: A statistical quantity computed as the standard error of a sample divided by the mean.
CTD	A data recording instrument used to measure ocean water conductivity [salinity] and temperature as a function of depth.
CWP	Circulating water pumps supply cooling water from the Pacific Ocean to the condensers to condense exhaust steam from the low pressure turbines.
CWS	Cooling water system: Includes portions of the DCPP intake structure, circulating water pumps, conduit tunnels, condensers, and discharge structure.
Davidson Current	Northward flowing current along the western US coast, usually during the fall and winter.
deme	A local population of organisms which is geographically isolated but not genetically isolated (i.e., through larval dispersal).
demography	The study of population statistics; specifically density, distribution, and vital rates of a population of organisms.
density dependence	Adjustment of vital rates (e.g., fecundity, mortality, somatic growth) in response to changes in population density; see *compensation*.

GLOSSARY (continued)

ENSO	El Niño–Southern Oscillation: Naturally occurring climatologic and oceanographic condition consisting of anomalous cooling and warming events.
El Niño	Ocean warming condition in the eastern Pacific that occurs when trade winds weaken over the central and western Pacific. Coastal upwelling is reduced resulting in widespread declines in nearshore productivity; an extreme phase of the ENSO phenomenon.
entrainment	Passage of small planktonic marine life through the power plant CWS.
ETM	Empirical Transport Model: A mathematical model that estimates the total annual probability of mortality (P_M) associated with entrainment using PE estimates.
ETWG	Entrainment Technical Work Group
FH	Fecundity Hindcasting: The number of larvae entrained are hindcast to the number of eggs by applying mortality estimates; the number of eggs is then used to estimate the number of adult females that would have produced that quantity of eggs.
gpm	Gallons per minute (0.0038 m^3/min)
growth overfishing	Fishing pressure that eliminates young fishes before they can spawn, as in the live-fish fishery.
impingement	Entrapment of macroscopic organisms on the intake traveling screens.
km	Kilometer (0.62 miles)
La Niña	An oceanographic phenomenon that occurs when trade winds strengthen over the tropical Pacific, increasing the amount of cold, nutrient rich water brought to the surface in upwelling currents; generally results in high primary productivity; an extreme phase of the ENSO.
m	Meter (3.28 feet)
m^3/d	Cubic meters per day (measure of flow rate; 264 gallons/ m^3)
MLLW	Mean lower low water; the elevation defined as 0.0 m (ft)
MT	Metric ton; (1,000 kg; 2,205 pounds)
North Pacific Gyre	Clockwise gyre in the, north Pacific Ocean that creates a southward movement of water along the western coast of the United States (i.e., California Current).
NPDES	National Pollutant Discharge Elimination System
NPV	Net Present Value of cash flow: The economic measure of evaluating proposed projects including capital and maintenance costs, and the cumulative lost revenue, discounted to a present value using an after-tax discount rate.

GLOSSARY (continued)

$\#/m^3$	Number of organisms per cubic meter of water (density of organisms)
PE	Proportional Entrainment: A mathematical value comparing the number of larvae entrained to the number of larvae available in the source water body.
P_M	Total annual mortality probability due to entrainment
population	The individuals of a given species or taxon within a defined geographical area.
P_s	The proportion of the population of inference represented by the number of larvae in the study grid, (i.e., number of larvae at risk in the study grid) / (number of larvae in the population of inference).
r	Instantaneous rate of increase
recruitment overfishing	Fishing pressure that limits the amount of larvae a population produces by taking primarily older, more fecund adults and leaving behind young, less fecund individuals.
RWQCB	Regional Water Quality Control Board (Central Coast)
species	A group of similar organisms that produce fertile offspring.
SST	Sea surface temperature
stock	A managed unit of a fish subpopulation available to a fishery.
subpopulation	A group of individuals of a species which interbreeds but is reproductively isolated from other such groups.
surplus production	In fisheries biology, production above and beyond an equilibrium level that can be harvested without substantially depleting a population; a population in equilibrium occurs when recruitment (or emigration) balances mortality (or immigration); see *compensation*.
traveling screens	A series of rotating meshed baskets at the intake structure designed to remove material and debris in the cooling water flow.
USEPA	United States Environmental Protection Agency
upwelling	Offshore transport of surface waters that results from steady northwest winds and causes cold, deep, nutrient-rich water to rise to the surface.
Z	Instantaneous mortality rate

1.0 EXECUTIVE SUMMARY

Section 316(b) of the Federal Water Pollution Control Act Amendments of 1972 (Clean Water Act) requires that Diablo Canyon Power Plant (DCPP) cooling water intake structure represents the best technology available (BTA) to minimize adverse environmental impacts (AEI) as defined by the USEPA. This report provides information to the Central Coast Regional Water Quality Control Board (RWQCB) in response to the requirements in Section 316(b). The report follows previous reports that described the sampling design and methods of the study, selection of target taxa, and formulations of the three assessment approaches (Tenera 1997a and b; Tenera 1998a) for approval by the RWQCB. PG&E also submitted two preliminary assessment reports to the Central Coast RWQCB, USEPA, and the ETWG (Tenera 1999a and b) that provide a comprehensive application of the three assessment approaches to entrainment effects on the selected target taxa.

Entrainment Technical Work Group

The Entrainment Technical Work Group (ETWG) was assembled by the RWQCB to assist their staff in reviewing the design and implementation of the 316(b) demonstration at DCPP. The ETWG was composed of PG&E and their consultants, RWQCB and their consultants, a consultant to the League for Coastal Protection, the California Department of Fish and Game, and USEPA. The ETWG assisted in developing criteria to assess the effects of the DCPP intake.

Impingement

The cooling water intake system at DCPP entrains and impinges larval, juvenile, and adult fishes and invertebrates. Entrainment occurs when organisms smaller than the 0.95 cm mesh of the traveling screens pass through the screens and enter the cooling water system. Impingement occurs when organisms too large to pass through the mesh are held against the screens by the water pumped through the plant. Once entrained, larval mortality was assumed to be 100%. Estimates of impingement were very low based on a review of the 1985–86 DCPP impingement

study (Tenera 1988a). The ETWG determined that additional impingement studies were not required at DCPP because of the low levels of impingement.

Assessment Approaches

Pacific Gas and Electric (PG&E) and its consultants, in cooperation with other members of the ETWG, employed three population assessment approaches to determine the entrainment and impingement effects of the DCPP intake. Estimates of entrainment loss, and the demographic data collected from the scientific literature, were used to parameterize two demographic approaches to estimating entrainment effects: Adult Equivalent Loss (*AEL*) and Fecundity Hindcasting (*FH*). The *AEL* method forecasts the number of adults that would have survived from the estimated number of entrained larvae, as if they had survived entrainment, using knowledge of mortality rates other than from entrainment by the power plant (i.e., natural mortality). The *FH* method is similar except that the number of larvae entrained are hindcast to the number of eggs, and the number of eggs is then used to estimate the number of female adults that would have produced them.

A third method, the Empirical Transport Model (*ETM*), estimates an average annual larval mortality due to entrainment per individual taxon, using estimates of proportional entrainment (*PE*) that compare the number of larvae entrained in one day to the number of available larvae in the source water body. Larval mortality is calculated after *PE* is weighted by the estimated fraction of the total population affected and compounded by the time larvae are susceptible to entrainment. Data collected during the study period from a current meter located offshore of DCPP was used to estimate this fraction (P_S) using either alongshore currents (for nearshore species) or both alongshore and onshore currents (for those more widely distributed species).

Target Organism Selection

The ETWG selected the organisms targeted for study: larval fishes, *Cancer* spp. crab zoea and megalops larvae, and metamorphosing larval and juvenile *Strongylocentrotus* spp. sea urchins. The individual taxa were chosen based on a set of criteria following USEPA Draft Guidelines

(USEPA 1977) and described in Tenera (1997b), that included constraints imposed by the availability of data to estimate population-level effects.

Sixteen target taxa were identified for assessment of entrainment effects using criteria based on USEPA guidance on 316(b) studies, and recommendations of the ETWG and other experts:

- Brown rock crab
- Slender crab
- Pacific sardine
- Northern anchovy
- Blue rockfish complex
- KGB rockfish complex
- Painted greenling
- Smoothhead sculpin
- Snubnose sculpin
- Cabezon
- White croaker
- Monkeyface prickleback
- Clinid kelpfishes
- Blackeye goby
- Sanddabs
- California halibut

Sampling

Entrainment and study grid sampling provided data for estimating entrainment abundance and abundance in the nearshore area defined by a study grid centered on the DCPP intake. Collection of the DCPP entrainment samples took place once per week from four permanently moored sampling stations located directly in front of the intake structure. The study grid contained 64 sampling cells in an eight cell by eight cell pattern and extended a distance of 17.4 km alongshore (8.7 km to the north and south) and an average distance of 3.0 km offshore.

The sampling surveys were conducted during 1996–1999 and divided into annual periods for analyses. Where it was possible, the estimates of annual entrainment were scaled to better represent long-term trends for a taxon by using data from surface plankton tows collected at the mouth of the DCPP Intake Cove starting in 1990. The analysis periods provided two annual estimates for comparing *FH* and *AEL* and two years of paired entrainment and study grid survey for comparing annual estimates of larval mortality $\widehat{P_{AI}}$ using *ETM*.

Results of Assessment

Variation of *FH* and *AEL* estimates between the years 1996–97 and 1997–98 appeared to be taxon-specific, but was likely also influenced by the El Niño event that occurred along the central California coast during the fall, winter, and spring seasons of 1997–98. For example, estimates of *FH* and *AEL* for both Pacific sardine and northern anchovy increased markedly between the first and second year of the study. The spawning biomass for both of these species is typically centered south of Point Conception, and was likely displaced northward with the north-flowing El Niño currents present in the second year. White croaker *FH* and *AEL* also appeared to increase slightly, which may also be due to additional transport northward during the El Niño year. Other species appeared to be somewhat less affected by El Niño-related transport since *FH* and *AEL* remained relatively constant or changed only slightly between years (e.g., KGB rockfishes, blue rockfish complex, blackeye goby, and sanddabs). These fishes probably have more localized spawning populations that were less affected by the El Niño event.

For some taxa, multiple assessment methods were applied to a known reference population or to fishery harvest data. For other taxa, assessment of impacts was limited to reporting the number of individuals lost to the population, or to a mortality estimate that was specific to an area where there is little knowledge of the adult population size. For example, demographic data on egg and larval mortality for blackeye goby allowed the application of multiple assessment approaches, but the absence of a stock assessment or fishery data precluded a context for the estimates.

For taxa where all three assessment methods were completed, comparison among the results illustrated the usefulness of the multiple assessment approach. While the range and variance of life history parameters and population estimates were sources of uncertainty in our estimates, we determined our areas of uncertainty through model comparison. Where both *FH* and *AEL* estimates exist, assuming a 50:50 sex ratio and aligning the ages of recruitment and maturation, the models were compared directly using the relationship $AEL \equiv 2FH$ presented in previous assessment reports (Tenera 1999a, b). The results from the demographic approaches were in similar units of either adult females (*FH*) or adults (*AEL*).

The *FH* and *AEL* model estimates, when ages were aligned, were in close agreement for northern anchovy and the blue rockfish complex. Results for these taxa provided assurance that the parameters used in the models (and thus the assessments) for these two taxa are reasonably accurate for central California populations. In the example of the blue rockfish, the *FH* and *AEL* estimates were used to extrapolate population-level effects and then were compared to local fishery landings. Applying the range of P_M estimates for this taxon (0.001–0.02) calculated using alongshore currents and extrapolating density offshore to these same fishery data provided an estimate of loss to the fishery of 15–292 kg of whole fish. Using an estimate of approximately 0.5 kilogram per live fish, the percentage catch equaled 29–584 adult fish. This estimate also agreed with aligned estimates of 2*FH* (114–273) and *AEL* (142–353).

Age at assessment needed to be considered when interpreting the large *FH* estimate for brown rock crab. The estimated number of reproductive adults using the *FH* model was extrapolated to known fishery-sized animals using estimates of adult mortality because females are reproductive before they enter the fishery. This reduced the *FH* estimate by an order of magnitude.

Assessments of entrainment effects on taxa that had very little demographic information were limited to *ETM* estimates. However, the availability of data on populations from sport or commercial fisheries provided context for assessments on some of these taxa (e.g., California halibut, cabezon, and monkeyface prickleback). The absence of catch data or stock assessments for other taxa limited the ability to assess entrainment effects on their populations with any methodology. Limited species' life history information imposed limitations on which methods could be used to assess entrainment effects on a particular taxon.

Taxa that are not commercially or recreationally harvested were primarily small, nearshore fishes. As might be expected due to the shallow water, shoreline location of the DCPP intake structure, several of these taxa were entrained in relatively high numbers, resulting in large P_M estimates. For example, adult kelpfishes that occur in rocky, nearshore habitat had one of the highest P_M estimates (ca. 30–40%). Other taxa with nearshore distributions, including smoothhead sculpin, snubnose sculpin, and monkeyface prickleback, also had relatively large

P_{AI} estimates. Their greatest larval abundance occurred in the cells closest to shore in the study grid; a distribution that was consistent with their adult distributions.

Estimates of P_{AI} for some of the species that have broader adult distributions (e.g., blackeye goby and California halibut) were also high. Blackeye goby larvae (P_{AI} ca. 20%), distributed throughout the survey grid, also occurred as adults from the intertidal zone out to depths of 30m. The high P_{AI} for blackeye goby, in part, may have resulted from the high density of adults directly in front of the intake structure. Although California halibut P_{AI} estimates were high, particularly during the July 1998–June 1999 period, the estimates are based on a limited number of larvae. Although low in abundance in entrainment California halibut was included in the list of target taxa because of its sport and commercial fishery importance.

Estimates of P_{AI} for species with more widespread, pelagic larvae (e.g., Pacific sardine and northern anchovy) were low. Impacts on commercially and recreationally harvested species with subtidal or pelagic distributions were minimal. For cases where all three assessment approaches were applied, the effects detected were relatively small, appeared to be localized, and thus would not affect the overall adult populations. These conclusions were supported by results for taxa such as KGB rockfishes where the estimated low levels of entrainment effects were corroborated by corollary data that showed no consequent adult abundance or larval production decreases.

Several small, nearshore fishes were included in the assessment because of their high abundance in entrainment samples. There was very little available information describing the demography of these taxa which limited the applicable assessment approaches to *ETM*. The *ETM* results showed that entrainment could remove over 10% of the annual larval production of several of these nearshore and intertidal taxa in an area from one to eight times the study grid area (52 km^2). In some of these cases, corollary data indicated that entrainment only removed surplus production since there were no apparent long-term declines in local adult populations, or in the weekly intake cove plankton tows conducted since 1990. For those nearshore taxa that did not display decreasing adult abundance, it was apparent from their length-frequency distributions that they were exposed to entrainment mortality for only a brief portion of their early

development. In contrast, for kelpfish and snubnose sculpin, whose adult abundance appeared to be decreasing, the length-frequency distributions indicated a longer exposure to entrainment. The combination of length-frequency analyses, *ETM* estimates, and other corroborating data support the conclusion that the local subpopulations of most nearshore taxa are not experiencing long-term declines in abundance due to entrainment.

Conclusion

United States Environmental Protection Agency (USEPA) criteria for determining the extent of adverse environmental impacts (AEI) focus on population-level effects. The results presented in this 316(b) study provide the necessary assurance that the DCPP cooling water intake system is not causing widespread or long-term population-level effects on the target taxa. Since DCPP is not causing population-level effects, no consideration of alternative cooling water intake structure technologies for minimizing AEI is necessary. However, this report does include an assessment of alternative technologies in response to a requirement by the Central Coast RWQCB. The conclusion of this alternative technologies assessment is that the existing cooling water intake structure is the best technology available for DCPP.

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2.0 INTRODUCTION

The purpose of this study is to determine if the Diablo Canyon Power Plant (DCPP) intake is causing an adverse environmental impact and, if so, what type of intake structure represents the best technology available to minimize that impact. The results will also be used to project the long-range effects on taxa where estimates of their populations are available. The estimates of entrainment and impingement effects on target taxa presented in this report will be used in recommending any action that may be required to comply with Section 316(b) of the Federal Water Pollution Control Act Amendments of 1972 (Clean Water Act).

2.1 USEPA Section 316(b) Background

Section 316(b) of the Clean Water Act regulates cooling water intake structures and requires that "the location, design, construction, and capacity of cooling water intake structures reflect the best technology available [BTA] for minimizing adverse environmental impact [AEI]." Although the EPA attempted to establish regulations in the mid-1970s, these rules were set aside by the courts on procedural grounds. As a result there are currently no regulations in effect for implementing Section 316(b).

To date, USEPA has not reissued Section 316(b) regulations and, instead, has allowed state and federal permit writers to implement Section 316(b) on a case-by-case basis. To make Section 316(b) decisions, permit writers have relied on other cases and on USEPA's (1977) informal draft "Guidance for Evaluating the Adverse Impact of Cooling Water Intake Structures on the Aquatic Environment: Section 316(b) P.L. 92-500."

In California, the State Water Resources Control Board (SWRCB) and the Regional Water Quality Control Boards (RWQCB) are authorized to implement the Section 316(b) requirement. In the case of DCPP, the permit applicant is obligated to provide the Central Coast RWQCB with

the "best information reasonably available"[1] to assist it in fulfilling its decision-making responsibility.

As is clear from the statute, the permit writer must consider two basic issues in making a finding that an intake technology meets BTA criteria for minimizing AEI:

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

The usual approach for a 316(b) demonstration is to consider the question of BTA only if a determination has been made that a facility is causing an AEI. However, in response to a requirement from the Central Coast RWQCB, PG&E is providing an alternative technology assessment as a part of this demonstration report (**Section 6**) and discusses BTA in **Section 7** of this report, even though a determination of AEI has not been made.

2.1.1 Adverse Environmental Impact (AEI) Standard

Since there are no regulations defining AEI, permit decisions must be based on the USEPA's long-standing AEI interpretations and previous 316(b) decisions. In several guidance documents issued since the 1970's, the USEPA has indicated that assessment of AEI should be based on an evaluation of population level effects, not just losses of individual organisms. In its 1975 Draft BTA Guidelines, the USEPA stated that "[a]dverse environmental impacts occur when the ecological function of the organism(s) of concern is impaired or reduced to a level which precludes maintenance of existing populations....".[2] Additionally, in the 1976 Development Document, released in conjunction with the EPA's previous Section 316(b) rules, the USEPA said that "[t]he major impacts related to cooling water use are those affecting the aquatic ecosystems. Serious concerns are with population effects that...may interfere with the

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[1] Boston Edison Company (Pilgrim Station), Case No. 78-7 (May 3 1978).

[2] USEPA, Draft Guidelines to Determine Best Available Technology for the Location, Design, Construction, and Capacity of Cooling Water Intake Structures for Minimizing Adverse Environmental Impact Section 316(b) P.L. 92-500 (December 5, 1975), p 8.

maintenance or establishment of optimum yields to sport or commercial fish and shellfish, decrease populations of endangered organisms, and seriously disrupt sensitive ecosystems."[3] The EPA has looked to population level effects in evaluating AEI in specific permit decisions. One such example was in a decision on Seabrook Power Station in New Hampshire. In Seabrook I, the Administrator determined that the estimated loss of 100 billion *Mya* clam larvae through entrainment would not have a "significant effect" on the local clam population.[4] On remand, the USEPA Administrator in Seabrook II determined that entrainment mortality would not have an adverse impact on abundance and distribution of plankton "even though the total biomass...killed may amount to hundreds or thousands of tons per year." He cited the ability of clams to reproduce rapidly, their wide distribution in the area beyond the immediate vicinity of the plant, and their transient nature.[5] Upon review of the Seabrook II decision, a circuit court upheld the Administrator's decision and indicated that the issue in assessing AEI is whether intake losses would "affect the ability [of fish species] to propagate and survive."[6]

2.1.2 Best Technology Available (BTA) Standard

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

- consideration of the technical and engineering feasibility of alternative intake technologies,

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[3] Development Document for Best Technology Available for the Location, Design, Construction, and Capacity of Cooling Water Intake Structures on the Aquatic Environment: Section 316(b). (1976 Development Document) USEPA Effluent Guidelines Division, Office of Water and Hazards Materials, April 1976.

[4] In the Matter of Public Service Co. of New Hampshire: Seabrook Station I and II, NPDES Appeal No. 76-7 (Decision of the Administrator, June 10, 1977) (Seabrook I).

[5] In the Matter of Public Service Co. of New Hampshire, et al.: Seabrook I and II. NPDES Permit Application No. NH0020338 Case No. 76-7 (Decision on Remand August 4, 1978 Seabrook II).

[6] Seacoast Anti-Pollution League v. Costle, 597 F.2d 306,309,311 (1st Cir. 1979).

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

Although no reference to cost is made in Section 316(b), legislative history suggests that Congress intended that costs be considered in 316(b) determinations. Specifically, a statement by the spokesman for the House Conferees indicates that Congress intended the "best technology available" to be interpreted to mean the "best technology available commercially at an economically practicable cost."[7] Additionally, in responding to comments during the drafting of its 1977 Draft Guidance, the USEPA said that BTA is the technology or group of technologies that minimize adverse impacts to the greatest possible degree at a cost that is not "wholly disproportionate" to the environmental benefits. This standard was also applied by the USEPA Regional Administrator in the Pilgrim decision that states, "a decision regarding the required degree of minimization calls for a determination that the costs involved are not wholly out of proportion to the adverse environmental impact being avoided."[8]

2.1.3 Status of Future 316(b) Regulations

Currently, the USEPA is developing regulations for implementing Section 316(b) of the Clean Water Act. This effort is the result of a 1993 lawsuit brought by the Hudson Riverkeeper and several environmental groups alleging that the USEPA had failed to perform a "non-discretionary duty" to promulgate Section 316(b) regulations.[9] In its effort to draft these

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[7] 188 Cong. Rec. H 9130 (daily ed. Oct. 4, 1972) (remarks of Congressman Clausen).

[8] "Determination Regarding Issuance of Proposed NPDES Permit No. MA 0025135", dated March 11, 1977, regarding Boston Edison's Pilgrim Units 1 and 2.

[9] Cronin v. Browner, 898 F. Supp. 1052, 1063 (S.D.N.Y. 1995).

regulations, the USEPA has held several public meetings to obtain input from interested parties and has also requested some facilities to respond to questionnaires to provide data that the USEPA deems relevant. As a result of a settlement agreement reached by the USEPA and the Hudson Riverkeeper plaintiffs, the USEPA will issue final 316(b) regulations by August 2001. However, the USEPA recently requested from the court that they be given a time extension and allowed to bifurcate the regulations. If the request is granted, the USEPA would issue final regulations for new facilities by May 2002 and presumably less stringent regulations for existing facilities by April 2004.

2.2 DCPP Study Development Process

The Central Coast RWQCB assembled a team of experts to assist their staff in reviewing the design and implementation of the 316(b) demonstration at DCPP. This team, the Entrainment Technical Work Group (ETWG), has met periodically since 1996 to discuss topics relevant to assessing entrainment and impingement effects at DCPP. The ETWG approved the design for the DCPP 316(b) demonstration, including the selection of the organisms targeted for study (larval fishes, *Cancer* spp. crabs, and metamorphosing and juvenile *Strongylocentrotus* spp. sea urchins). The individual taxa were chosen by the ETWG for assessment based on criteria agreed to and described by Tenera (1997b) that included the statistical qualities of the data for a taxa and the availability of suitable life-history information to meet assessment model requirements. These criteria were, in turn, based on criteria described in USEPA Draft Guidelines (USEPA 1977). The ETWG and the RWQCB's consultants will present their final recommendations regarding the 316(b) demonstration study for consideration by the Central Coast RWQCB at their July 14, 2000 meeting.

Several approaches to assessment were developed by PG&E, its consultants, and additional members of the Entrainment Technical Work Group (ETWG). These assessment approaches are fecundity hindcasting (*FH*: Alec MacCall, NOAA/NMFS, Tiburon Laboratory, pers. comm.; Tenera 1998a), adult equivalent loss (*AEL*: Horst 1975; Goodyear 1978), and empirical transport modeling (*ETM*: Boreman et al. 1978, 1981). Previous reports submitted to the ETWG for

review and approval detailed the sampling design and methods of the study, selection of target taxa, and formulations of the three assessment methods (Tenera 1997a; Tenera 1997b; Tenera 1998a). Two preliminary assessment reports were also presented using these three approaches. The first (Tenera 1999a) presented an assessment for three of the target fish taxa for the period July 1997 through June 1998, while the second presented results for all 16 target taxa (crabs and fishes) for the same one year period (Tenera 1999b). Results in these reports demonstrated how available life history information could be incorporated into assessment calculations, what influence life history information would have on results, and how results of the three approaches could be interpreted together. In this report, the methods are described, the principles embodied in these methods are summarized, and the results are presented using one or more of these assessment approaches, as appropriate, for the 16 target taxa studied.

2.3 Standards for Assessment

The USEPA (1977) draft guidelines acknowledge that the determination of the extent of AEI when it is occurring is difficult to assess. They state:

> *Adverse aquatic environmental impacts occur whenever there will be entrainment or impingement damage as a result of the operation of a specific cooling water intake structure. The critical question is the magnitude of any adverse impact. The exact point at which adverse aquatic impact occurs at any given plant site or water body segment is highly speculative and can only be estimated on a case-by-case basis...*

In light of these obvious difficulties, the document (USEPA 1977) also provides some general guidelines for determining the extent of AEI. These involve determining the "relative biological value of the source water body zone of influence for selected species and determining the potential for damage by the intake structure" based on the following considerations of the value of a given area to a particular species:

- principal spawning (breeding) ground,
- migratory pathways,
- nursery or feeding areas,
- numbers of individuals present, and

- other functions critical during the life history.

Following this general approach provided by the USEPA (1977), additional criteria can be evaluated that are specific to the marine environment around Diablo Canyon, unique to marine aquatic animals, and directly applicable to the present 316(b) entrainment study:

- distribution (pelagic, subtidal, nearshore subtidal & intertidal);
- range, density, and dispersion of population;
- population center (source or sink);
- magnitude of effects;
- long-term abundance trends (e.g., Intake Cove plankton tows, subtidal fish observations, fishery catch data);
- long-term environmental trends (climatological or oceanographic); and
- life history strategies (e.g., longevity and fecundity).

By assessing the relative value of each of these criteria for a particular target taxon examined in the present study, we will be able to better assess the extent of the impact that the loss of these animals has on the local environment and the population at large.

2.4 Report Organization

Section 3 of this report presents a background on the power plant and its environmental setting. Methods used for data collection and the formulation of each assessment models are contained in **Section 4**. **Section 5** presents the results of the study, including assessments for each of the target taxa as separate subsections. Taxa are organized in phylogenetic order from the most primitive to the more evolutionarily advanced species, a standard form of organization in scientific publications. Each taxon's subsection provides a summary of its demography, entrainment estimates, and assessment results. An evaluation of alternative intake technologies available to minimize entrainment and impingement is presented in **Section 6**. A summary discussion of all results and their interpretations is provided in **Section 7**, which is followed by **Section 8**: Conclusions. A listing of the literature cited is presented in **Section 9**. Fourteen

appendices present previous reports, results summary tables, and other supporting information.

These appendices are as follows:

Appendix A: DCPP 316(b) Demonstration Study: Phase 1-Entrainment Study Design, I. Sampling Location

Appendix B: DCPP 316(b) Demonstration Study: Phase 1-Entrainment Study Design, II. Selection of Target Organisms, Sampling Methods, and Gear Testing

Appendix C: DCPP 316(b) Study: Phase 3-Sampling Plan and Modelling Evaluation

Appendix D: DCPP 316(b) Entrainment Study, Field and Laboratory Procedures

Appendix E: DCPP 316(b) Study: Comparison of Entrainment Larval Density in Paired Samples: One-vs. Two-nets

Appendix F: Estimating Total Entrainment Mortality using the Delta method

Appendix G: Number of Samples Collected and Laboratory Processed

Appendix H: Estimated Mean Density of Larvae Collected at DCPP

Appendix I: Estimated Number of Target Taxa Larvae Entrained per Week

Appendix J: Results of t-tests and Kolmogorov-Smirnov tests on length frequency data from paired entrainment and study grid samples

Appendix K: DNA Analysis of Larval *Sebastes* spp.

Appendix L: Technologies that do not have demonstrated commercial operability or reliability at power plants of a scale similar to DCPP

Appendix M: Technologies that are currently available but would not reduce impingement or entrainment to levels lower than observed at DCPP

Appendix N Technologies and methods currently available and proven effective at facilities of the same size as DCPP but determined to not be effective at DCPP

3.0 BACKGROUND

Diablo Canyon Power Plant, owned and operated by Pacific Gas and Electric Company (PG&E), is a nuclear-powered, steam-turbine power plant with a rated output of 2,200 megawatts of electricity. Commercial operation of Unit 1 began in May 1985, Unit 2 in March 1986. DCPP is located on a coastal terrace midway between the communities of Morro Bay and Avila Beach on the central California coast (**Figure 3-1**). The local coast is a steep and rugged rocky shoreline that is exposed to heavy wave activity. The study area supports a rich community of marine life that is a biogeographical extension of similar marine communities extending many hundreds of miles to the north. Except for the DCPP, the coast is largely uninhabited and undeveloped along the 16 km (10 mi) between the cities of Morro Bay and Avila Beach.

The power plant draws in seawater from a constructed intake cove through a cooling water system (CWS) to provide cooling for power plant operations. The intake structure was sited behind breakwater structures to mitigate the effect of wave action. Use of the breakwater structures reduced the rise and fall of pump suction levels due to wave action by 75% and reduced the potential of storm-uprooted kelp and suspended sands entering the intake. The common unit 1 and unit 2 intake structure was sited at the north end of the cove created by the breakwaters to accommodate the potential addition of units 3 through 6 (Lillevang 1969).

As the water passes through the plant's condensers, it causes the steam contained within the secondary reactor loop to recondense. After passing through the plant's CWS, the heated water is discharged into Diablo Cove. The discharge is approximately 11°C (20°F) warmer than ambient ocean waters under normal operating conditions.

3.1 DCPP Circulating Water System Description

The DCPP intake structure entrains and impinges larval, juvenile, and adult fishes and invertebrates. Entrainment occurs when organisms smaller than the 0.95 cm mesh of the traveling screens pass through the screens and enter the cooling water system. Impingement

occurs when organisms too large to pass through the mesh are held against the screens by the velocity of the water pumped through the plant's intake structure.

The intake for DCPP units is a shoreline structure that houses bar racks, vertical traveling screens, auxiliary cooling water systems, and main circulating water pumps (**Figure 3-2**). On the ocean side of the intake structure, a concrete curtain wall extends approximately 2.4 m below mean sea level to prevent floating debris from entering the structure. Seawater entering the intake structure passes through one of 16 sets of bar racks designed to exclude large debris from the forebays. The bar racks are either 1.5 or 3.1 m wide and consist of vertical rows of approximately 8 cm x 1 cm steel bars spaced about 8 cm apart. There are seven vertical traveling screens per unit that are designed to remove debris that passes through the bar racks. The screens extend from the upper deck of the intake structure to the bottom at a depth of approximately 10 m below sea level. The six wider traveling screens filter seawater to each unit's two main circulating water pumps (CWP), and the one narrower traveling screen filters seawater to each unit's two auxiliary seawater (ASW) pumps. Each CWP traveling screen is composed of 57 baskets that are approximately 3 m wide by 61 cm tall. The ASW traveling screens also have 57 baskets that are 1.5 m wide by 61 cm tall. The interior of each basket is covered with 0.95 cm mesh designed to prevent material from entering the conduits and clogging the 2.5 cm diameter condenser tubes. Objects small enough to pass through the bar racks and larger than the 0.95 cm mesh of the traveling screens may be impinged.

Each CWP has a manufacturer's estimated average pumping flow rate of 1,641 m^3/min (PG&E 1998a). The calculated DCPP total daily intake volume is 9.45 million m^3/d when all four CWPs (two per unit) are operating. The flow rates of the ASW pumps are 240,000 m^3/d. The volume of cooling water can vary daily due to a variety of factors that include changes in tidal and swell height, as well as resistance caused by occlusion of condenser tubes. During the period June 1996 through June 1999, pumping stopped during three refueling outages and other shorter periods of time when one or more of the DCPP CWPs were not in operation (**Figure 3-3**).

The traveling screen assemblies are equipped with a high pressure seawater wash system, and screens are rotated either automatically or manually. When the screens rotate, impinged debris, fishes, and invertebrates are rinsed from the screens into a trough that slopes to a central refuse sump area (**Figure 3-4**). In Fall 1997, a grinder system was installed to decrease the size of all material before it entered the sump. All material in the sump is then pumped back to the ocean at the landward end of the west breakwater. Automatic operation of the screens occurs in one of two ways: by timed cycles or by hydrostatic pressure. Timers are typically set to initiate a 40-minute screen wash once every four hours. The screens also rotate automatically when a height differential of approximately 20 cm across the screen surface is detected. Manual operation of the traveling screens occurs whenever necessary, especially when heavy accumulations of kelp threaten the safe operation of the intake system. During these times continuous screen washing is usually necessary.

3.2 Environmental Setting

3.2.1 Bathymetry

The bathymetry of the nearshore region between Point Buchon and Point San Luis is characterized by sloping bedrock and soft-bottom flats, with steeper relief generally increasing from the south to the north (**Figure 3-5**). The majority of the nearshore region near Pt. San Luis, from the shoreline to ~2 km offshore, is less than 40 m in depth, while the corresponding nearshore region off Point Buchon is 60–80 m in depth. Rocky pinnacles are relatively common out to the 40 m contour, in contrast to the relatively flat bottom typical of the 40–100 m region. Within the geographic area bounded by Point Buchon and Point San Luis, several prominent rocky ridges extend from the shoreline out to about the 20 m contour, especially noticeable at Point Buchon, Lion Rock, and Pecho Rock.

3.2.2 Ocean Current Patterns

The nature and origin of processes structuring the nearshore currents in the vicinity of Diablo Canyon are fairly complex, reflecting dynamics of seasonal currents, winds, and tidal cycles. The general current pattern near Diablo Canyon is composed of three currents: the constant current, the smoothed current, and the residual current (Safaie 1986). The constant current has a period of greater than 30 d and results from large-scale, southward and northward flows related to the California and Davidson currents, respectively. The smoothed current, with a period of 1–30 d, is primarily driven by wind; the residual current, with a period of less than 1 d, is controlled largely by both tide fluctuation and wind.

The following paragraphs discuss the mechanics of the constant and smoothed current components summarized in Safaie (1986). The residual current component was not discussed as completely in Safaie (1986), presumably because the magnitude and duration of this current is relatively minor compared to that of the constant and smoothed currents. In addition to these regularly-occurring currents, the mechanics of periodic El Niño and La Niña oceanographic events are described. Finally, the hydrology and water chemistry of the nearshore region of Diablo Canyon during the study period October 1996–June 1999 are summarized.

In general, two major types of currents exist off the coast of California: shore parallel and shore normal (perpendicular). The shore parallel currents include constant currents: the southward-flowing California Current and the northward-flowing Davidson Current. The California Current originates from the clockwise North Pacific Gyre, which creates a southward flow along the western coast of North America. This current is present year-round along the California coast, but is typically displaced offshore by the northward flowing Davidson Current in the fall and winter. The Davidson Current is formed by a deeper-water, counter-clockwise gyre in the California Current present between Cape Mendocino (Mendocino County) and Point Conception (Santa Barbara County). The Davidson Current is sometimes referred to as the California Undercurrent, particularly during the spring and summer when it is a deeper-water phenomenon. Of the two, the Davidson Current is weaker and more diffuse than the California Current. As such, current reversals can, and do, commonly occur during the winter.

Smoothed currents in the vicinity of Diablo Canyon (Safaie 1986) include shoreward and seaward currents that originate from seasonal onshore and offshore winds, respectively. These seasonal, perpendicular currents are of minor magnitude relative to the shore-parallel currents, but their consequent downwelling and upwelling events are important to the nutrient cycling and productivity of the Diablo Canyon nearshore region.

The California Current ecosystem is characterized by seasonally high levels of primary production when northwesterly winds predominate and cause coastal upwelling to occur, typically in the spring and summer of each year. Upwelling occurs because the northwesterly winds generate Ekman transport of surface waters due to the Coriolis Force, resulting in a net movement of surface waters perpendicular to the wind direction: to the right in the northern hemisphere and offshore relative to the California coast.

3.2.3 El Niño Southern Oscillation and Ocean Water Temperatures

The El Niño Southern Oscillation (ENSO) in the eastern Pacific Ocean typically consists of a warm water El Niño phase followed by cooler water temperatures during a subsequent La Niña. However, the effects of the ENSO are not limited to ocean waters as they also affect the atmospheric/oceanic circulation (Rasmusson and Wallace 1983). Much of the interannual ocean variability on decadal and sub-decadal time scales can be attributed to El Niño events that have significant impacts on the global ocean's heat budget and ecosystems (Barber and Chavez 1986; Cane et al. 1986). Its global impacts (Rasmusson and Wallace 1983) are most apparent in the eastern Pacific (Barber and Chavez 1983) where the influence of El Niño can be detected from the southern tip of Chile to Alaska (Enfield and Allen 1980). It remains unclear whether the primary impacts of the ENSO arrive via ocean or atmospheric circulation, but the fact remains that these impacts are significant in the coastal waters of Central California (e.g., Lenarz et al. 1995; Chavez 1996; Schwing et al. 1997; Lynn et al. 1998).

Moderate to weak La Niña conditions prevailed along the central California coast from 1995 through early 1997 (Schwing et al. 1997) when one of the strongest El Niño events recorded this

century began to develop (Lynn et al. 1998). During the first year of field sampling conducted at DCPP (October 1996–September 1997), this weak La Niña may have, atypically, contributed to higher than normal sea surface temperatures (SST) observed off California and Mexico as the result of anomalous wind relaxation events (Schwing et al. 1997). The El Niño that had been developing in the western tropical Pacific since early Spring 1997 arrived along the coast of California in Fall 1997, bringing higher than normal SST and suppressing upwelling activity along the length of the coast. This El Niño event continued into the summer of 1998 (Lynn et al. 1998) and was followed by another La Niña beginning in Fall 1998 and extending at least to the end of the field sampling operations at DCPP in June 1999.

The continuing cycle of ENSO events has also had dramatic effects on the primary and secondary production of the California Current ecosystem. Primary production, measured as chlorophyll concentration, decreased during warmer El Niño years—1992 and 1997—and rose in the cooler La Niña years—1991 and 1995 (Chavez 1999). Total phytoplankton biomass demonstrated similar trends: declining in 1992 and 1997 and rising in 1995 when the average temperature was close to the climatological mean, 13.5°C. Chavez (1999) also notes that over the 10-year period 1988–1998, SST have gradually risen while levels of nitrate, chlorophyll, primary production, and certain phytoplankters (e.g., centric diatoms) have declined, effectively reducing the overall productivity of the California Current ecosystem. The changes to the ecosystem at these fundamental levels may have dramatic consequences for the remainder of the ecosystem, which supports a diverse assemblage of marine birds, mammals, and fishes. Lynn et al. (1998) also notes that "mean macrozooplankton abundance during the spring of 1998 was the lowest in the 50-year CalCOFI time series," further confirming the findings of Chavez and MBARI.

3.2.4 Ocean Currents and Temperatures during the Entrainment Study Period (October 1996 – June 1999)

The DCPP 316(b) study period was characterized by a series of climatological and meteorological ENSO events (**Table 3-1**) that are known to have dramatic effects on ocean ecosystems, oceanic and atmospheric currents, and on productivity in the California Current.

Field sampling was initiated during an anomalously warm La Niña event that occurred during the fall of 1996 and continued through the fall of 1997 (Schwing et al. 1997). In the fall of 1997, one of the strongest El Niño events recorded this century occurred along the California coast and continued through the summer of 1998 (Lynn et al. 1998). The fall and winter of 1998–99 were subject to a more typical cold water La Niña phase that continued through the end of field sampling operations in June 1999.

Data collected during the study period from the single current meter and meteorological observation station at DCPP showed that nearshore currents were largely controlled by prevailing wind conditions (**Figure 3-6**). During those infrequent periods when wind speeds were negligible (i.e., less than 2 nautical miles per hour), longshore constant currents predominated.

Mean monthly water temperatures near DCPP during the study period October 1996 through June 1999 ranged from 17.5°C in November 1997 to 10.5°C in April 1999 (**Figure 3-7**). While maximum temperatures occurred at various times throughout the year (e.g., November 1997 and August/September 1998), annual temperature minimums were consistently present during the month of April. These temperature minimums generally corresponded to seasonal offshore winds and consequent coastal upwelling. The continuing influence of the ENSO throughout the study period may explain the inconsistent interannual timing of recorded maximum temperatures. Overall, nearshore seawater temperatures during the study period deviated appreciably from long-term patterns. The mean seawater temperature recorded for November 1997 was the warmest since records began in 1976 (Figure 3-7). This was rivaled only by temperatures recorded in the summer/fall of 1983 and 1984, another strong El Niño event.

Table 3-1. Presence and relative magnitude of El Niño (W) and La Niña (C) events from 1996 through 1999 in the tropical Pacific: relatively strong events are noted with (+), relatively weak events with (-), and C* denotes anomalously warm La Niña conditions postulated by Schwing et al. (1997). Blank cell in 4th quarter of 1999 represents non-ENSO (= normal) period. Table combines data from Schwing et al. (1997) and the NOAA ensoyears.html page at www.noaa.gov.

Year	1st Quarter	2nd Quarter	3rd Quarter	4th Quarter
1996	C*	C*	C*	C*
1997	C*	W	W+	W+
1998	W+	W	C-	C
1999	C+	C	C-	

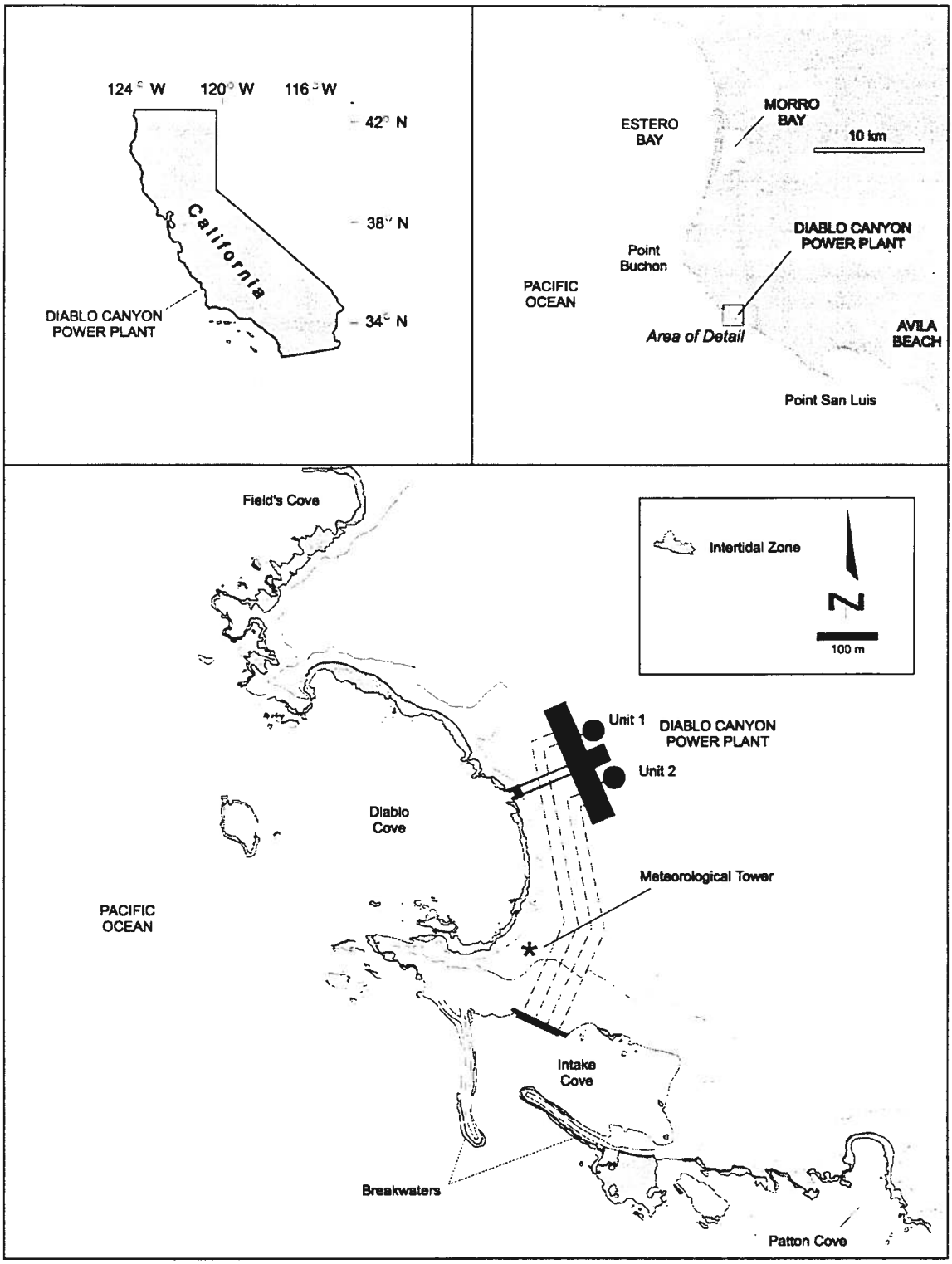


Figure 3-1. Location of Diablo Canyon Power Plant.

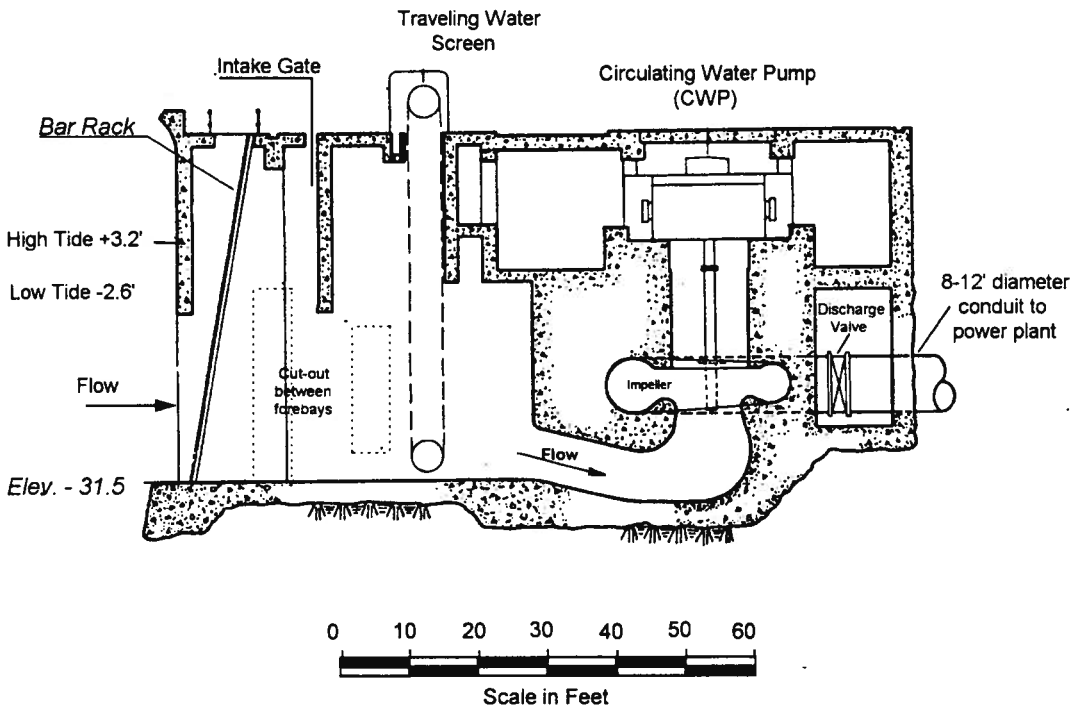


Figure 3-2. Cross-section diagram of DCPP intake structure showing water flow path. Elevations are based on mean sea level (modified from PG&E 1988a).

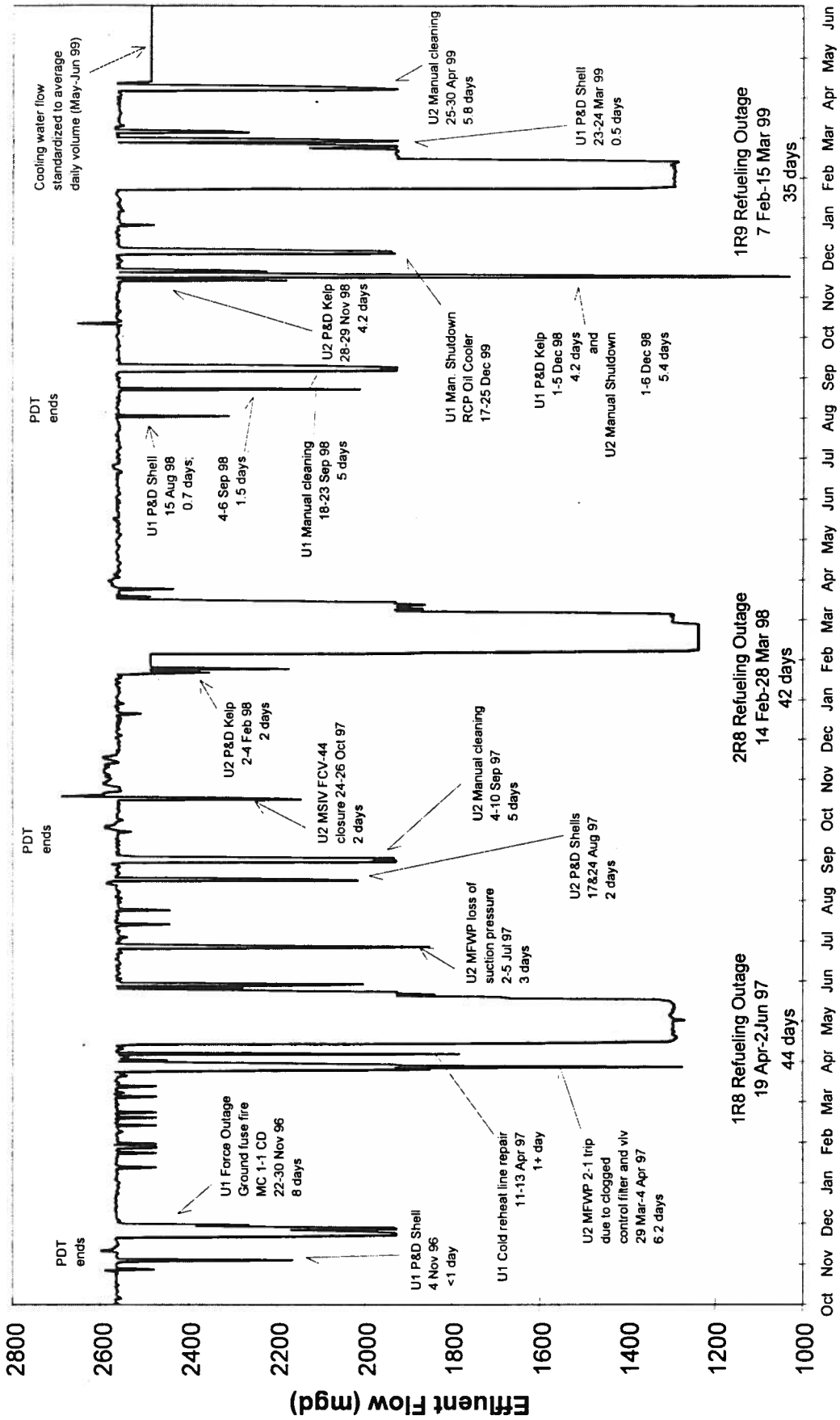
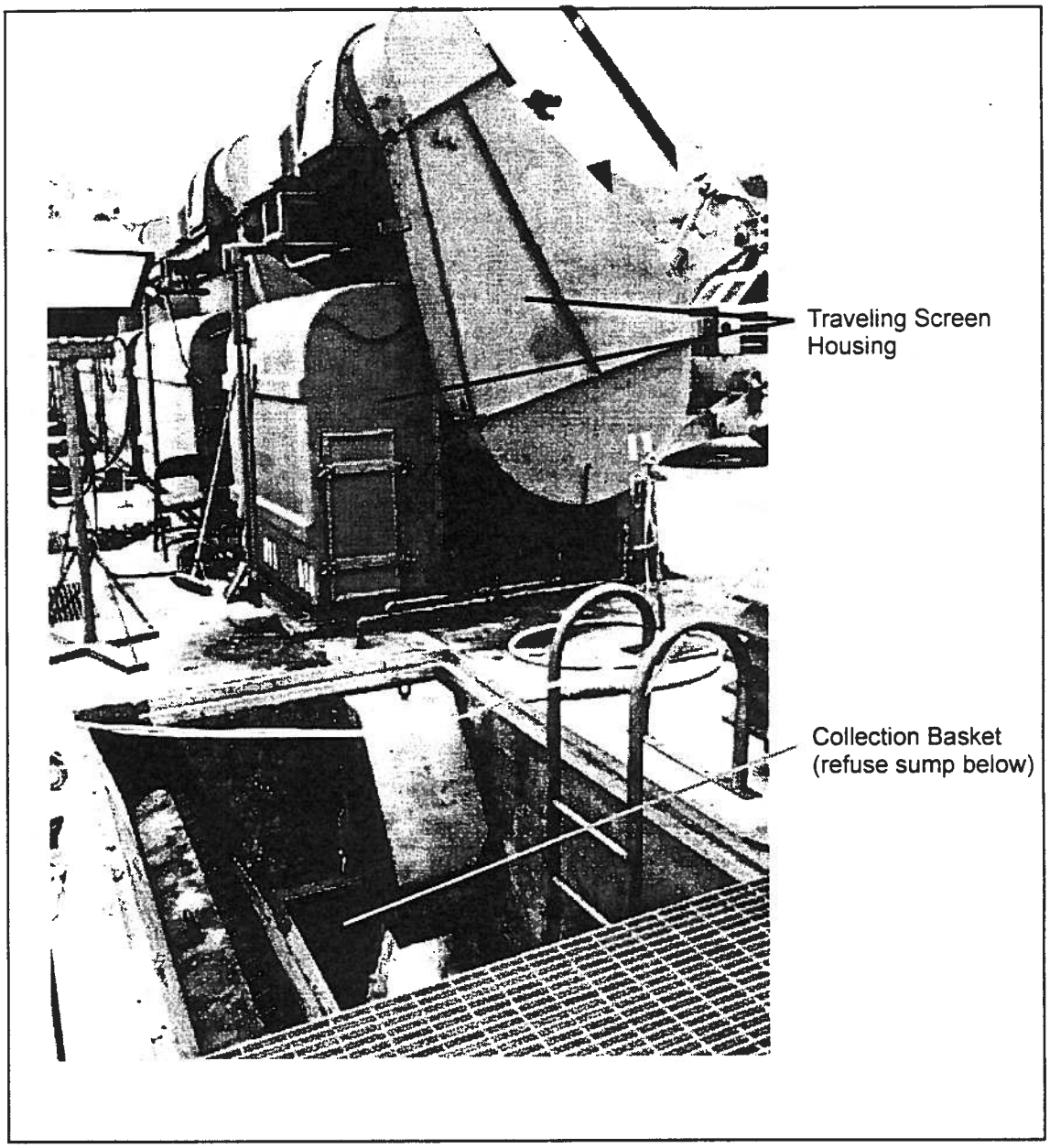


Figure 3-3. DCPP daily cooling water flow - October 1996-June 1999.

Figure 3-4. Unit 1 traveling screen assemblies photographed in 1985 during an impingement collection.



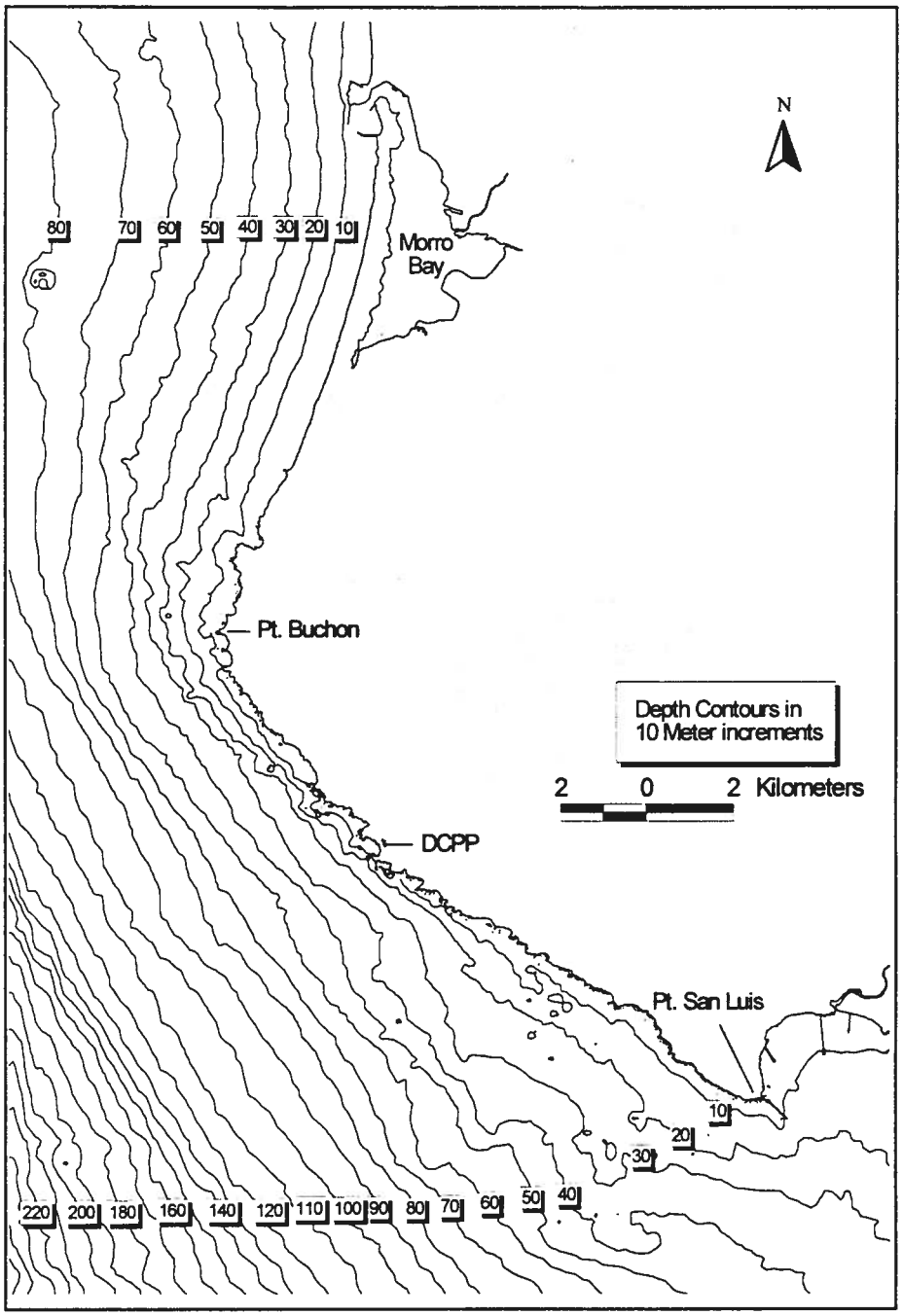


Figure 3-5. Bathymetry of nearshore region between Point Buchon and Point San Luis.

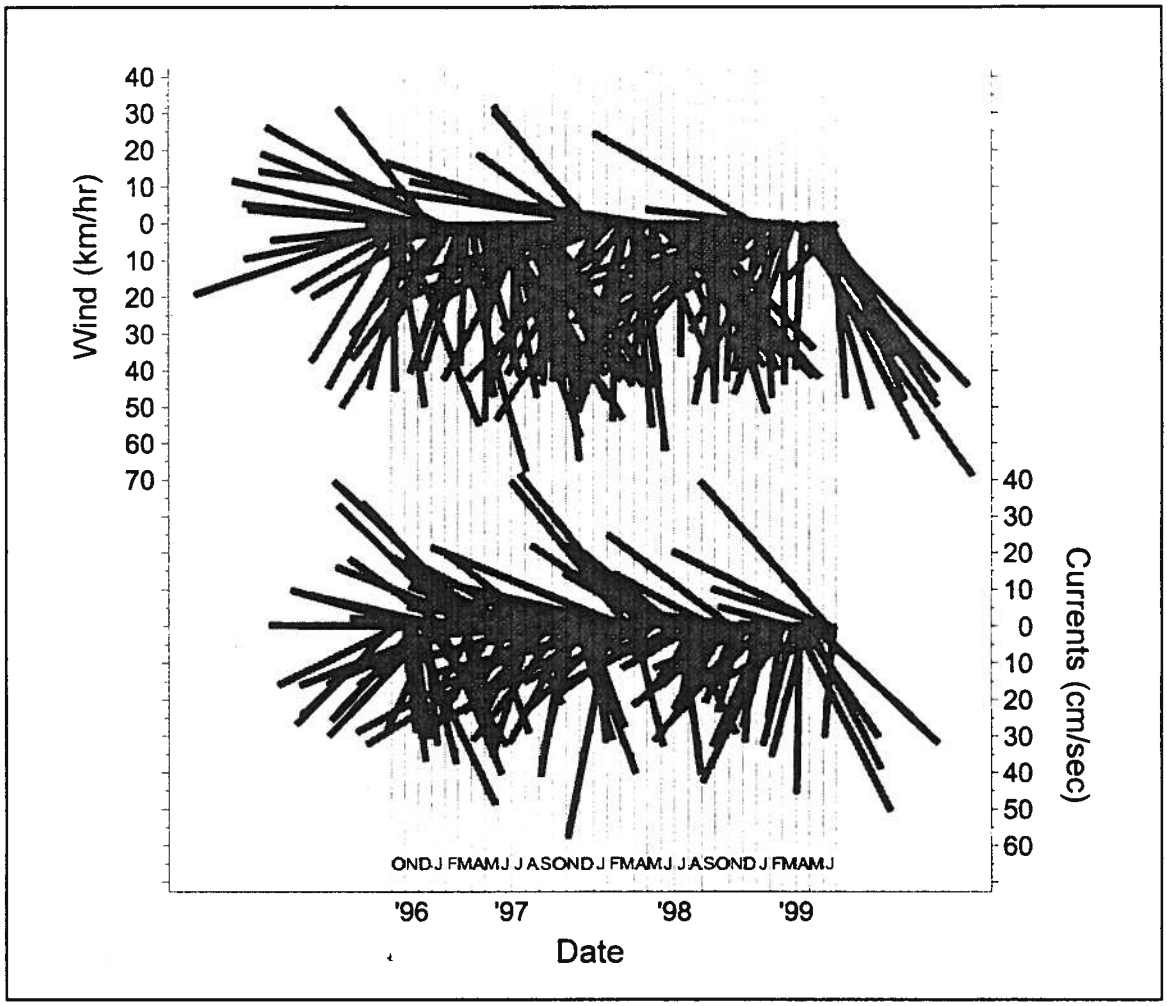


Figure 3-6. DCPP wind and water current patterns October 1996 through June 1999.

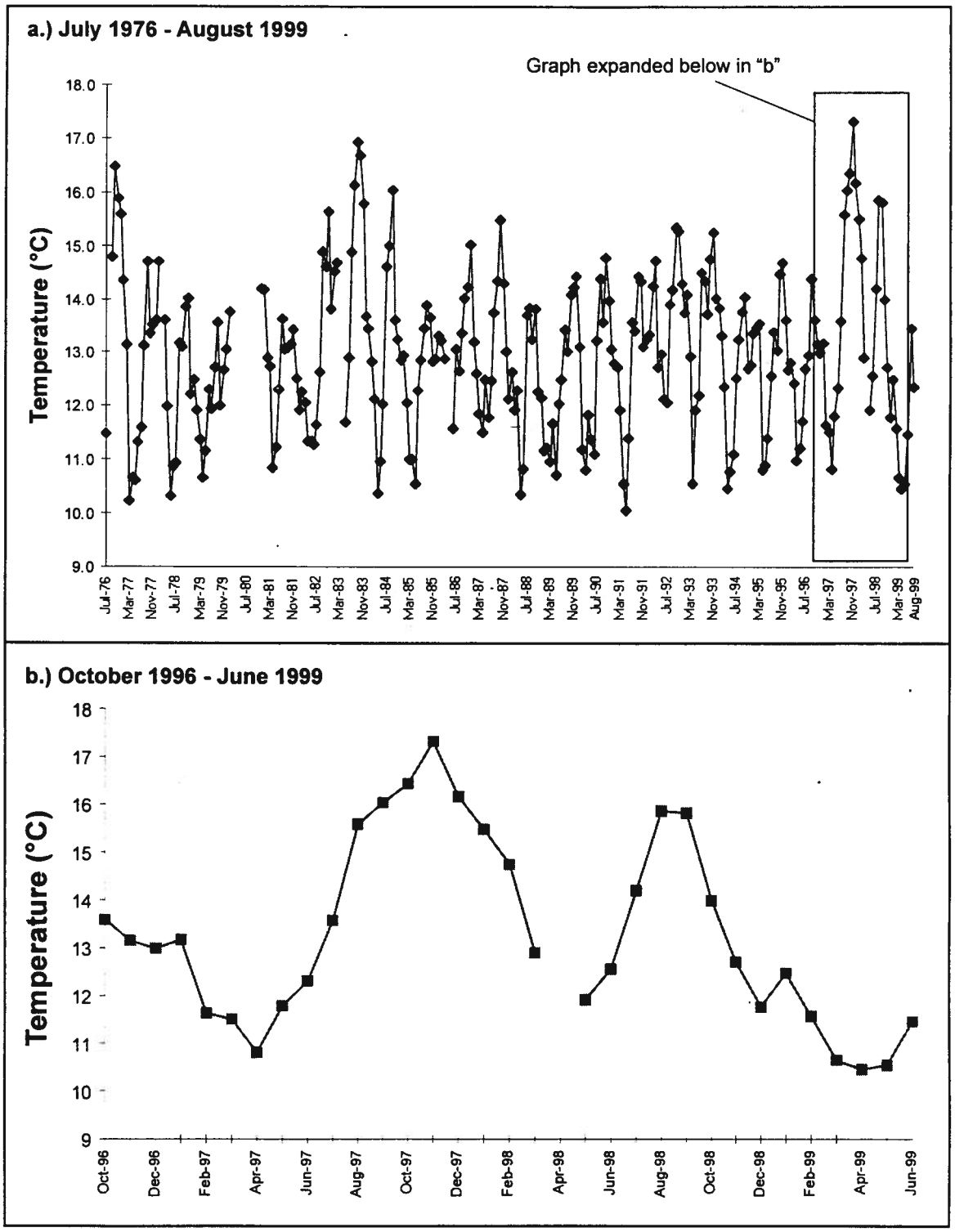


Figure 3-7. Mean monthly seawater temperatures near DCPP.

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4.0 METHODS

The Environmental Protection Agency EPA guidance on Section 316(b) demonstrations recognizes that it is not practical to analyze the impacts on all organisms that may be entrained by cooling water intake operations (USEPA 1977). Generally only some groups or species of organisms in the vicinity of a facility are targeted for collection and analysis. Target organisms are those having certain characteristics that make them suitable for detecting or forecasting impacts to an ecosystem under investigation (Cairns and Pratt 1989; Jones and Kaly 1996).

4.1 Criteria for Selection of Target Organisms

The initial criteria used to select appropriate target organisms at DCPP were developed by the Environmental Protection Agency (USEPA 1977). Discussions with the DCPP Entrainment Technical Working Group (ETWG) were instrumental in guiding the use of the criteria at DCPP. Meetings with the ETWG and other invited experts at theTiburon Impact Assessment Colloquia (October 1996 and October 1998) also helped refine these criteria for application at DCPP. USEPA guidance on 316(b) defines target organisms as those that meet at least one or more of the following criteria:

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

The DCPP ETWG added three additional criteria for selecting target organisms for the entrainment study at DCPP:

8. identifiable to the species level;
9. entrained in sufficient abundance to allow for impact assessment, i.e. allowing the model(s) constraints to be met and confidence intervals to be calculated; and
10. having local adult and larval populations (i.e., source not sink species). For example, northern lampfish (*Stenobrachius leucopsarus*), relatively abundant as entrained larvae, is a sink species because it is an offshore and deep water species whose larvae become widely separated from their viable environment and enter coastal waters.

Target organisms meeting these criteria included larval fishes, larval *Cancer* spp. crabs, and larval sea urchin. The final list of target organism taxa (**Table 4-1**) was determined by the ETWG (May 1999) based mainly on the data collected during this study and the above listed criteria.

Other potential target organism groups reviewed by the ETWG for possible inclusion in this study but subsequently rejected by the ETWG included kelp spores, fish eggs, abalone larvae, squid, and Pismo clams. 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 and most of the taxa included in the final entrainment evaluation have egg stages that are not likely to be entrained: they either have demersal/adhesive eggs or are internally fertilized and extrude free-swimming larvae. Abalone larvae were not included because they are at low risk from entrainment and cannot be effectively sampled or identified during early life stages when they would be susceptible to entrainment (Tenera 1997b). Young squid are also unlikely to be entrained because they are competent swimmers immediately after hatching. Pismo clam larvae were not included because there is no suitable substrate for settlement in the vicinity of DCPP, and the duration of their pelagic phase is too short to enable them to reach appropriate settling habitat in significant numbers after spawning in the Diablo Canyon region (Sandra Owen, CDF&G, Long Beach, CA, pers. comm.).

4.2 Entrainment

This section presents a summary of the sample collection and laboratory processing methods. A more detailed presentation of the entrainment and study grid collection methods can be found in

- Appendix A—DCPP 316(b) Demonstration Study: Phase 1, Entrainment Study Design, I. Sampling Location
- Appendix B—DCPP 316(b) Demonstration Study: Phase 1, Entrainment Study Design, II. Selection of Target Organisms, Sampling Methods, and Gear Testing
- Appendix C—DCPP 316(b) Demonstration Study: Phase 3, Sampling Plan and Modelling Evaluation
- Appendix D— DCPP 316(b) Entrainment Study, Field and Laboratory Procedures

Sea urchin larvae were originally chosen by the ETWG as a target organism, but laboratory experiments showed that the early stages of larval urchins are small enough to be pass through the 335 μm net mesh. The only urchin developmental stage that appeared to be effectively captured during sampling were metamorphosing and recently metamorphosed individuals. Therefore, the results of the entrainment sampling are presented for metamorphosing and recently metamorphosed sea urchins, but no assessment on the effects to the population is made based on agreement with the ETWG.

4.2.1 Sample Collection

Entrainment and study grid sampling provided data for estimating entrainment abundances and abundances in the nearshore area defined by the study grid. The sampling plans were tailored to the methodology chosen for estimating future adult loss and population mortality. Three methods were chosen, developed fully in **Section 4.4**, and briefly described below.

The estimate of entrainment loss and demographic data, collected from the scientific literature, was used to parameterize two demographic models of entrainment effects: Adult Equivalent Loss (*AEL*) and Fecundity Hindcasting (*FH*). The *AEL* method forecasts the number of adult survivors from the estimated number of entrained larvae, as if they had survived entrainment, using knowledge of mortality rates other than from entrainment by the power plant (i.e., natural

mortality). The *FH* method is similar except that the number of larvae entrained are hindcast to number of eggs, and the number of eggs is then used to estimate the number of female adults that would have produced them. A third method, the Empirical Transport Model (*ETM*), estimates an average annual larval mortality per individual due to entrainment, using estimates of Proportional Entrainment (*PE*), which compares the number of larvae entrained in one day to the number of available larvae in the source water body. A larval mortality is calculated after *PE* is weighted by an estimated fraction of the total population affected and compounded by the time larvae are susceptible to entrainment. The Phase 3 Sampling Plan (Table 2-1 in Tenera 1998a) describes the data required for *AEL, FH,* and *PE*.

4.2.1.1 Entrainment Sampling

Collection of the DCPP entrainment samples took place once per week from four permanently moored sampling stations located directly in front of the intake structure (**Figures 4-1** and **4-2**). Entrainment sampling occurred from October 1996 through June 1999, with details about the number of samples collected per survey presented in **Appendix G**. Surveys 01 and 02 were not processed because they were used to finalize sampling protocols and only 8 and 34 subsamples were collected, respectively. Therefore, the first complete survey was conducted on October 23, 1996. The four fixed stations were sampled in a random order during each 3-hour survey cycle. Samples were collected over a 24-hour period from a boat moored approximately 10 m from the intake structure using a 0.71 m diameter bongo frame with two 1.8 m long, 335 µm white Nitex™ mesh nets similar to the nets used by the California Cooperative Oceanic Fisheries Investigations (CalCOFI). The first 9 surveys were collected with 505 µm, but due to possible extrusion of larval fishes observed in Surveys 8 and 9, subsequent surveys were collected with 335 µm mesh (Tenera 1997b). Surveys 9 and 10 were completed on consecutive days, but only the data from Survey 10 was used in the analyses because samples were collected using the smaller mesh size. The bongo frame and nets were fished from the top to the bottom and back to the surface a total of eight times to filter approximately 40–50 m^3 per net. At the surface, the net was turned as the upper portion of the frame hit the surface and was turned within approximately 13–25 cm of the bottom. The upward vertical lift speed of the nets was held constant at approximately 0.3 m/s

(~0.6 knots) using an electric winch. The downward speed of the nets (0.3–0.45 m/s) was determined primarily by gravity acting on the mass of the bongo frame and the drag resistance of the nets. Each 24-hour sample period was divided into eight 3-hour sampling cycles, with each station being sampled once per cycle. The sample contents of each net were initially preserved separately in either a solution of 5% buffered formalin or 70–80% ethanol. All samples were later transferred to ethanol before laboratory processing. A total of 64 subsamples (32 samples) was collected per day with a total of approximately 2,560–3,200 m^3 of water filtered per 24-hour survey period.

4.2.1.2 Study Grid Sampling

Sampling of the study grid was designed to provide the information on abundance and distribution of target organisms in the vicinity of DCPP that is used in the *ETM* assessment. The data from each study grid survey were paired with the corresponding entrainment survey to estimate an index of entrainment loss (*PE*): the ratio between the number of larvae entrained in one day to the number estimated in the study grid over a three day period. The boundaries and shape of the study grid were chosen based on the following criteria:

- the grid would be large enough to characterize the larvae from the target groups potentially influenced by DCPP's cooling water intake operations, and
- the grid would include a representative variety of local nearshore habitats.

In establishing the study grid, the ETWG also considered that ocean currents in the area generally move both up and down the coast past DCPP. Currents also show inshore/offshore oscillations, but these occur less frequently and generally at a lower magnitude (**Section 3.2.2**). The study grid contained 64 cells in an eight cell by eight cell pattern (**Figure 4-3**) and was centered on the Intake Cove at DCPP. The northern extent of the grid was near Point Buchon and the southern half, a mirror image of the northern portion, extended to Point San Luis. The coastline between Point Buchon and Point San Luis makes a slight bend (approximately 20°) northward at DCPP. Tangents along the coastline from the tip of the west breakwater at DCPP defined the inshore margins of the study grid at 321° True (NW) and 121° True (ESE). The grid extends a distance of

8.7 km to both the north and south at an average distance of 3 km offshore. Lines dividing the study grid into eight partitions ran SSW (221° True) to a distance of 2.7 km offshore of the inshore margin. Regions inshore of the study grid dimensions were in shallow water with partially submerged rocks, making the areas unsafe for boat operations and sampling.

The study grid was sampled monthly from July 1997 through June 1999. Two randomly positioned stations within each of the 64 cells of the grid were sampled using a 0.71 m diameter standard CalCOFI-style bongo frame with two 3.3 m long, 335 μm white Nitex™ mesh nets. 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 (**Figure 4-4**). 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 (e.g., cell D1 in **Figure 4-4**), one of the two adjacent cells in the direction of travel (E1 or E3) was randomly selected to be sampled next.

Nets were fished in an oblique fashion following CalCOFI protocol (Smith and Richardson 1977). The sampling depth was limited to within 3 m of the bottom depth to avoid damage or loss of sampling equipment and for safe operation of the vessel. Had larval fishes been concentrated in the 3 m near bottom, this sampling method would not detect them leading to underestimates of larval density that would increase estimates of *PE*. Two stations were sampled per grid cell with the station locations for each monthly survey being randomly-positioned, latitude/longitude coordinates within each cell. The random position of latitude and longitude for each grid cell station was generated using Microsoft Excel™ macro code, designed to randomly select polar coordinates and to convert these coordinates to a Mercator Projection system used for navigational purposes. Latitude and longitude positions were located in the field using a Micrologic™ Global Positioning System (GPS) plotter.

A total of 256 subsamples was collected during each 72-hr survey. At the end of each haul, the material collected in one bongo net was fixed in 5% buffered formalin for abundance and

composition analyses. The material in the second net was fixed in about 70–80% ethanol (ETOH) to be archived for possible future use in larval fish age and growth analyses. To minimize temporal variation between entrainment and study grid sampling, grid surveys were scheduled to bracket the 24-hour entrainment survey, overlapping by one day before and after the collection of entrainment samples. The paired sampling from July 1997 was an exception to this design, i.e. entrainment sampling occurred during the first 24 hours of the 72-hour grid survey.

4.2.1.3 Intake Cove Surface Plankton Tow Sampling

Weekly plankton samples were collected in the DCPP Intake Cove during the following periods: February 1990 through June 1990, January 1991 through June 1991, and then continuously from September 13, 1991 to the present. Three samples were collected at dawn once per week by towing a 0.5 m diameter, 335 µm mesh standard plankton net in front of the intake structure to a location approximately even with the outer end of the west breakwater (**Figure 4-5**). The net was towed at a depth of about 1 m below the surface. Water flowing into the net was measured by a calibrated flowmeter. Approximately 40–50 m^3 of water was filtered per tow. The samples were preserved in 5% buffered formalin and later transferred to 80% ethanol. These samples were only processed for larval fishes.

4.2.2 Laboratory Processing

Laboratory sample processing procedures used during this study are presented in **Appendix D**. All larval fishes and the megalopal stages of *Cancer* spp. were sorted from either 32 or 64 subsamples (four to eight per cycle) from each entrainment survey and from the formalin-preserved grid survey samples (two per cell). One of the two paired bongo nets represents a dependent subsample such that both to the paired nets comprise the complete sample or tow. The estimate of the mean larval density calculated after processing only 32 subsamples was on average similar to the estimate for all 64 subsamples (Tenera 1998b: **Appendix E**). All *Cancer* zoeal stages and juvenile urchins were sorted from one randomly selected entrainment sample from each of the 3-hour cycles, resulting in eight samples processed per sampling day. The zoea

and urchins, sorted from the samples collected in the grid, were processed from one sample per cell for only two of the total number of surveys performed in the grid.

Due to the large number of zoea and urchins, most samples were split before processing. Density estimates were calculated for each sample based on the number of individuals of each taxon and the volume of water filtered. For split samples, the density was mathematically adjusted for the level of splitting. For example, when a sample was divided into eight parts and one part was sorted, the number of each of the larval stages identified and enumerated was multiplied by eight to estimate the number in the sample.

Not all of the target organism groups were sorted from all of the entrainment and grid survey samples, per agreement with the ETWG (**Figure 4-6**). Larval fishes were sorted from weekly entrainment samples from surveys collected from October 1996 through September 1998 and from surveys that paired with the grid surveys for the period of October 1998 through June 1999. *Cancer* crab megalops were sorted from entrainment samples collected from October 1996 through June 1999. *Cancer* crab zoea and metamorphosing and recently metamorphosed sea urchins were sorted from entrainment surveys collected from December 1996 through June 1998. Samples during October and November 1996 were not sorted for zoea and urchins because these samples were collected with 505 μm mesh nets and these organisms could possibly have been extruded through the mesh, resulting in underestimates of their density. These samples were used for larval fish and megalops because their generally larger size would lessen the chance of extrusion. Larval fishes and *Cancer* crab megalops were processed from two samples per cell during all of the grid surveys. As previously mentioned, *Cancer* crab zoea and metamorphosing and recently metamorphosed sea urchins were processed only from one sample per cell during surveys collected in March and April 1998.

An on-site quality control program ensured that each of the sorters maintained a high level of accuracy in identifying target taxa in the samples. This program employed both on-site and off-site personnel to verify the identification and enumeration of the collected organisms. All data pertaining to the collection and laboratory processing of each of the samples were tracked using a

computer database. The procedures used in laboratory processing, identification, and quality control are included in **Appendix D**.

A variety of general and specific references were used to identify the collected individuals. The larvae of several species in the DCPP area are at present poorly known or undescribed. Larvae for several taxa could only be identified into higher taxonomic categories, such as genus (e.g. *Gibbonsia* spp.) or family (e.g. Cottidae). Identification for larvae of the rockfishes (genus *Sebastes*) was especially troublesome. At least 72 species and 11 subgenera of rockfish live along the eastern Pacific coast of North America (Moreno 1993). Because of relatively recent speciation in this genus, many larval characteristics are shared over a wide range of species, making identification within this genus problematic (Moser et al. 1977; Moser and Ahlstrom 1978; Baruskov 1981; Kendall and Lenarz 1987; Moreno 1990; Nishimoto in prep.). Identification of larval rockfish to the species level relies heavily on pigment patterns that change as the larvae develop. Currently, at least 5 of these 72 species can be reliably identified to the species level at larval developmental stages. They are the blue rockfish (*Sebastes mystinus*), shortbelly rockfish (*S. jordani*), cowcod (*S. levis*), bocaccio (*S. paucispinis*; Yoklavich et al. 1996), and stripetail rockfish (*S. saxicola*; Laidig et al. 1995). Other species within this genus can only be resolved to broad sub-generic groupings based on pigment patterns (Nishimoto in prep.). **Table 4-2** presents the rockfish pigment groups used in this study and the species reported in each group.

Crab larvae also are sometimes difficult to separate to species. The first zoeal stage of brown rock crab (*C. antennarius*), yellow crab (*C. anthonyi*), and slender crab (*C. gracilis*) is similar in appearance; consequently, these species could not be separated at this stage. When these species metamorphose to the second zoeal stage, however, individual characteristics develop that allow their correct classification. Therefore, in calculating *FH* models for the two species of *Cancer* crabs, the proportion of the two species for zoeal stage II was assumed to be the same for the first zoeal stage. This ratio was used to estimate the numbers of zoeal stage I larvae for the two species. The species of a few individual megalops and zoea could not be determined; therefore, identification of these individuals was left at the genus level, *Cancer* spp.

Lengths of the larval fish target taxa were measured using a computer imaging system. A quality assurance program for the operators of the system was conducted. Larvae from both entrainment and study grid samples were measured.

4.3 Impingement

An impingement study at DCPP was conducted from February 1985 through March 1986 to provide information on

- taxonomic composition and abundance of impinged organisms
- size of impinged fishes and selected macroinvertebrates
- diel and seasonal patterns of impinged organisms, and
- sex ratio and degree of gonadal maturity of selected impinged species.

Impingement samples were collected once per week for a continuous 24-hr period. A requirement for sampling was that at least one of the two cooling water pumps (CWP) per unit be operating for the entire 24 hr. At times, collections were postponed because of equipment testing or maintenance that may have prevented the collection of a complete 24-hr period. When possible, the sampling was re-scheduled for later the same week. Samples were not collected when kelp accumulation caused the collection baskets to overflow, potentially leading to a loss of organisms and inaccurate estimates of impingement rates.

On a collection day, the traveling screens were rotated initially to ensure that they were clean before the sampling began. The troughs were also cleaned of all accumulated debris and organisms. The screens remained stationary for approximately 3.75 hr and then rotated and rinsed for 15 min. The impinged material was rinsed into the troughs and collected in metal baskets made of 0.635 cm steel mesh. All impinged material was removed from the baskets after each of these 4-hr periods (cycles), except during times of heavy kelp accumulation when the screens operated continuously. During continuous screen rotation the screen wash system was stopped, when possible, for 15 min per cycle to allow for the removal of the samples from the baskets. The

impinged material from each unit was kept separate. All organisms were removed from the impinged material and returned to the laboratory for processing. A quality control program verified that all organisms were removed during the sorting process.

All fishes and selected invertebrates were identified to the lowest practical taxonomic category, counted, measured, and weighed. Invertebrates that were measured and weighed included caridean shrimps, decapod and pelagic red crabs, cephalopod molluscs, rock scallops, and sea urchins. All other invertebrates were recorded as either individual counts or as present and were not measured nor weighed. A quality control program ensured the accuracy of the data.

invertebrates not in this report

It was assumed that all impinged organisms would not survive. Studies at other power plants where impingement losses are high normally combine impingement losses for a taxon with other projected adult losses from estimated entrainment. We did not combine impingement with estimates of projected entrainment losses because of the low impingement rate at DCPP and the time difference between the two studies. The results of the impingement study are included in the DCPP 316(b) Study Phase 3: Sampling Plan and Modelling Evaluation (Tenera, 1998a; **Appendix C**), and impingement methodology is not described further.

4.4 Description of Entrainment Analytical Methods

The entrainment and study grid surveys were divided into annual periods for analysis (**Figure 4-6**). The entrainment surveys were divided into three analysis periods:

- Period 1—October 23, 1996 through September 30, 1997
- Period 2—October 1, 1997 through September 30, 1998
- Period 3—July 1, 1997 through June 30, 1998

Data for Period 3 were previously analyzed by Tenera (1999b), Preliminary Assessment of Entrainment Effects on Target Fish and Crab Taxa (July 1, 1997–June 30, 1998). Analysis Period 3 overlaps Periods 1 and 2. It is presented here as the entrainment data for this time interval is used in the calculation of *PE* which is used in the *ETM*, which allows a comparison of *FH*, *AEL*

and *ETM* calculations for the same 12 month time interval. Analysis for zoea for Period 1 was adjusted to December 1, 1996 through November 30, 1997 because the samples from October and November 1996 were collected with a larger mesh net and not analyzed for crab larvae.

Larval densities from samples collected during Periods 1, 2, and 3 were used to calculate the total number of larvae entrained and project adult losses using *FH* and *AEL* methodology. The grid surveys were divided into two analysis periods for the purpose of providing two annual estimates of larval mortality based on the *ETM* methodology. The monthly grid and paired entrainment surveys for these 2 analysis periods are as follows:

- Period 3—July 1, 1997 through June 30, 1998
- Period 4—July 1, 1998 through June 30, 1999

The Intake Cove surface plankton tows were processed only for the consecutive seven month period of December through June, per agreement with the ETWG (**Figure 4-6**), as the peak in larval ichthyoplankton is generally during these months. These samples were processed only for larval fishes. These data were used to adjust the number of larvae entrained during a year to a long-term average that was more likely to represent average effects.

4.4.1 Total Annual Entrainment

The density of each of the target organisms in the entrainment samples was used to estimate the total annual larval entrainment for each taxa ($\widehat{E_T}$). Daily entrainment estimates and their variances were derived from the mean density of larvae (number of larvae per cubic meter of water filtered) and calculated from the 32 samples collected during each 24-hr entrainment survey, using the methods described in the Phase 3 report (Tenera 1998a). In the few instances where the measured volumes in each paired net differed by more than 20%, the higher value was used under the assumption that one of the paired flowmeters had slowed due to kelp fouling, improper lubrication, or misalignment. Density estimates were multiplied by the DCPP daily intake flow volume to obtain the number of larvae entrained per day. The daily intake flow volume was calculated by multiplying the number of hours of operation of each of the pumps by

their rated design flow rate. One hundred percent mortality was assumed for all entrained organisms. To estimate weekly entrainment, the number of days that each sampling event represented was determined by setting the sampling date at the mid-point between sample collections. The daily cooling water intake volumes were then used to calculate entrainment for the period by summing the product of the entrainment estimates and the daily intake volumes for each approximate weekly period. These weekly estimates and their associated variances were then added to obtain annual estimates of entrainment and variance for each taxa.

The annual estimate of entrainment ($\widehat{E_T}$) was calculated by adding a weekly entrainment estimate that was the daily entrainment estimate multiplied by the weekly volume of intake cooling water:

$$\widehat{E_T} = \sum_{i=1}^{52} \left(\frac{V_i}{v_i} \right) \widehat{E_i} \tag{1}$$

where

v_i = intake volume on the survey day of the ith weekly period (i=1,...,52)
V_i = total intake volume for the ith weekly period (i=1,...,52)
$\widehat{E_i}$ = the estimate of daily entrainment during the entrainment survey of the ith weekly period.

The variance of $\widehat{E_T}$ was estimated as

$$\widehat{Var}(\widehat{E_T}) = \sum_{i=1}^{52} \left(\frac{V_i}{v_i} \right)^2 \widehat{Var}(\widehat{E_i}), \tag{2}$$

using the sampling variances of entrainment on the survey day of the ith weekly period, $\widehat{Var}(\widehat{E_i})$. Equation 2 will underestimate the true variance because it does not incorporate the variance associated with the within-week ratio estimator

$$\frac{V_i}{v_i} = \frac{\widehat{E_{T_i}}}{\widehat{E_i}},$$

where $\widehat{E_{T_i}}$ is the estimated total entrainment for the *i*th week. It also does not incorporate daily variations in intake flow due to waves, tide, and other factors not measured by the power plant.

As recommended by the ETWG, the estimates of annual entrainment were scaled to better represent long-term trends for a taxa by using data from Intake Cove surface plankton tows collected at DCPP, starting in 1990. These data provide an index of annual trends in larval abundance for the period of 1990–1998. This multi-year annualized index consists of five months (February–June) of larval fish density estimates in 1990, six months (January–June) in 1991, and seven months (December–June) in all subsequent years. The estimated annual entrainment was adjusted to the long-term average by the equation

$$\hat{E}_{Adj-T} = \widehat{E_T} \cdot \left(\frac{\overline{I}}{I_i} \right) \tag{3}$$

where

$\hat{E}_{Adj-T}$ = adjusted estimate of total annual entrainment to a long-term average, 1990–1998,

I_i = index value from DCPP Intake Cove surface plankton tows in the *i*th year, and

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

The variance for adjusted annual entrainment can then be expressed as

$$\widehat{Var}(\hat{E}_{Adj-T}) = \widehat{Var}(\widehat{E_T}) \cdot \left(\frac{\overline{I}}{I_i} \right)^2, \tag{4}$$

assuming the indices are sampled without error. Ignoring the sampling error of the indices will underestimate the true variance, but Equation 4 will account for the change in scale associated with multiplying the annual entrainment estimate by a scalar. The variance of $\hat{E}_{Adj-T}$, however, does not take into account the between-day, within-stratum variance, interannual variation, nor the variance associated with the indices used in the adjustment. Hence, the actual variance of the $\hat{E}_{Adj-T}$ estimate is greater than the value expressed above.

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

$$E(I_i) = C \cdot E_i$$

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, then the interannual variance in the index has the relationship

$$Var(I_i) = C^2 Var(E_i).$$

Therefore, the coefficients of variation (CV) for $\bar{I}$ and $\bar{E}$ across n years have the relationship

$$CV(\bar{I}) = \frac{\sqrt{\frac{Var(I)}{n}}}{\bar{I}} \doteq \frac{\sqrt{\frac{C^2 Var(E)}{n}}}{C\bar{E}} = CV(\bar{E}).$$

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 estimates of $\hat{E}_i$ and $\hat{I}_i$, the variances include sampling errors that may not be equal. Therefore, the CV of $\hat{\bar{I}}$ was used to express the possible variation in entrainment across years.

The use of adjusted entrainment in the *FH* and *AEL* models provides results that represent average long-term effects. The CV for $\hat{\bar{I}}$ was used in propagating the variances of the entrainment effects and calculating the ranges of possible values for *FH* and *AEL* in sensitivity analyses. Sensitivity analyses were used to examine the effects of different input parameters on model results.

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* are valid, then the estimate already represents average long-term entrainment effects. Using adjusted entrainment in the *FH* and *AEL* models provides a better basis to compare results from all three models when

they are converted into a common currency through the use of population or fishery stock assessments.

The length frequency distributions for the measured larvae from the paired entrainment and study grid surveys were tested for differences using a Kolmogorov-Smirnov 2 sample test (Zar 1984). Large differences between the two distributions may indicate that the age distribution of entrained larvae is significantly different from the source population in the study grid. The mean larval lengths from the two distributions were tested for differences using a two-sample *t*-test. When variances were unequal between the two groups, Satterthwaite-adjusted degrees of freedom were used in the test (SAS Institute 1990). The statistical power of these tests to detect very small differences between the two groups will be very high because of the large sample sizes (Zar 1984).

4.4.2 Fecundity Hindcasting (*FH*)

The *FH* approach combines larval entrainment losses with adult fecundity to estimate the amount of adult female reproductive output eliminated by entrainment, assuming no compensatory reserve of the population. The loss of reproductive output (i.e., numbers of larvae) was used to hindcast the numbers of adult females possibly removed from the reproductively active population. These *FH* estimates require data on age-specific mortality from the egg and early larval stages up to entrainment. The fact that *FH* requires survivorship from a relatively short period of time (i.e., egg to larva) is an advantage of this approach. Fecundity and mortality from sources other than entrainment are integrated into an estimate of loss by converting the estimated number of entrained larvae backwards to reproductively active females (i.e., hindcasting). *FH* does not require additional sampling other than from the entrainment intake water, for estimates of larval abundance. These advantages may be offset by the need to (1) obtain or model age-specific mortality rates and total lifetime fecundities to predict the adult losses, and (2) secure information on the size of the adult population of interest to estimate population-level effects (i.e., relative losses). Therefore, the method was only used for those taxa where the minimal data of item (1), above, were available. With the broad generalization of ignoring any population

reserve (compensation), entrainment losses can be interpreted as population level impacts when *FH* estimates are applied as direct losses to an inferred population.

4.4.2.1 Formulation

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

$$\widehat{FH} = \frac{1}{\widehat{\overline{F_T}}} \sum_{i=1}^{w} \frac{\widehat{E_{T_i}}}{S_i} \qquad (5)$$

where

w = number of weeks the larvae are vulnerable to entrainment,
$\widehat{E_{T_i}}$ = estimated total entrainment for the ith weekly survey period ($i = 1,\ldots, w$),
S_i = survival rate from eggs to larvae of the stage present in the ith weekly survey period, and
$\widehat{\overline{F_T}}$ = average total life time fecundity for females, equivalent to the average numberof eggs spawned per female over their reproductive years.

Equation 5 is based on the simplified case of a single synchronized spawning by a species. For species with overlapping or continuous spawning, larval abundance would have to be specified by week and age class (i.e., $\widehat{E_{ij}}$). However, we used the mean size of all larvae entrained at DCPP to estimate a representative age of larvae, and then estimated a survival rate to this representative age. Two input parameters in Equation 5 that may not be available for many species, and thus may limit the method, are average fecundity ($\widehat{\overline{F_T}}$) and survival rates (S_i) from spawning to entrainment.

Larval age (duration from release or hatching to entrainment), needed for estimating survival, was calculated from growth rates and the minimum and mean size at entrainment. Estimates of growth rate appear in **Table 4-3**, and their use in survival estimation is discussed further when applied to specific taxa.

Adjusted annual entrainment can be incorporated in the *FH* method (Equation 5), assuming that average rates of survival are the same among years and study periods:

$$\widehat{FH} = \frac{\hat{E}_{Adj-T}}{\prod_{j=1}^{n} \widehat{S}_j \cdot \widehat{\overline{F}_T}} \tag{6}$$

where

$\widehat{S}_j$ = age specific survival of eggs and larvae for the *j*th age class (j = 1, ..., n) and

$\widehat{\overline{F}_T}$ = expected number of eggs produced in a reproductive lifetime.

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 total lifetime fecundity $\overline{F}_T$ was approximated by the expression

$$\begin{aligned}\widehat{\overline{F}_T} &= \text{Average eggs/year} \cdot \text{Average number of years of reproductive life} \\ &= \text{Average eggs/year} \cdot \left(\frac{\text{Longevity - Age at maturation}}{2} \right).\end{aligned} \tag{7}$$

The expected length of reproductive life was approximated as the midpoint between the times of maturation and longevity. This approximation was based on the assumption of a linear survivorship curve between these events (i.e., uniform survival). For exploited species such as northern anchovy and sardine, the expected number of years of reproductive life may be much less than predicted using this assumption. Therefore, the estimated longevity was based on the oldest observed individual caught by the fishery, rather than by the oldest recorded fish. Simulation, comparing exponential survival, shows that the calculation of $\widehat{\overline{F}_T}$ will be negatively biased for species with short reproductive lifespans, and positively biased for those with longer durations.

The variance of $\widehat{FH}$ was approximated by the Delta method (Seber 1982):

$$\widehat{Var}(\widehat{FH}) = (\widehat{FH})^2 \left[CV^2(\hat{E}_{Adj-T}) + \sum_{j=1}^{n} CV^2(\widehat{S}_j) + CV^2(\widehat{\bar{F}}) + \left(\frac{\widehat{Var}(A_L) + \widehat{Var}(A_M)}{(A_L - A_m)^2} \right) \right]$$

where

$CV(\hat{E}_{Adj-T})$= CV of adjusted estimated entrainment (estimated by $CV(\hat{\bar{I}})$ when available),

$CV(\widehat{S}_j)$ = CV of estimated survival of eggs and larvae up to entrainment,

$CV(\widehat{\bar{F}})$ = CV of estimated average annual fecundity,

A_M = age at maturation, and

A_L = age at maturity.

The behavior of estimator 7 for *FH* appears log-linear, suggesting that an approximate confidence interval can be based on the assumptions that $\ln(\widehat{FH})$ is normally distributed and uses the pivotal quantity

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

A 90% confidence interval for *FH* was estimated by solving for *FH* and setting Z equal to +/-1.645, i.e.

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

4.4.2.2 Sensitivity Analysis

In addition to calculating a confidence interval, a sensitivity analysis was performed where $\widehat{FH}$ was recomputed by varying each input parameter $(\hat{\theta})$ and using an additive error model for the entrainment parameter ($\widehat{E_7}$ or $\hat{E}_{Adj-T}$) of ±1.645 $\widehat{SE}(\theta)$ and using a multiplicative error model for the remaining parameters. The CV for $\hat{\bar{I}}$ (i.e. the average surface tow index across years) was used in propagating the variances of the adjusted entrainment estimate for calculating the ranges of possible values for $\widehat{FH}$ in the sensitivity analyses. An assumed CV of 1.0 (100%) was used in

calculating the other parameter estimates as $\hat{\theta} e^{\pm 1.645 CV(\hat{\theta})}$. This approach helps to identify how the parameters' variability influences $\widehat{FH}$. In addition, the following constraints to the modified parameter estimates were applied to assure reasonable ranges:

- Survival ≤ 1
- Age at maturity ≥ one-half the original estimate of age at maturity
- Age at maturity ≤ the age half way between the original estimates of age at maturity and longevity
- Longevity ≥ the age half way between the original estimates of age at maturity and longevity
- Longevity ≤ twice the original estimated longevity

4.4.2.3 Assumptions

Fecundity hindcasting calculations assume the following:

- Values of parameters from the literature represent the population parameters for the years and location of the Diablo Canyon study.
- No population reserve or compensation counters entrainment mortality.
- Reported values of egg mass are lifetime averages in order to calculate an unbiased estimate of lifetime fecundity.
- Reproductive life expectancy can be accurately calculated by assuming that time of death is uniformly distributed between age-at-maturation and age-of-longevity.
- Juvenile and egg survival rates are constant over time.
- The loss of the reproductive potential of one female is equivalent to the loss of an adult female.

4.4.3 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. The approach requires survival estimates (had the larvae not been entrained) from entrainment to an age at recruitment to the adult population. This model and the *FH* model both translate larval entrainment mortality into adult fish losses, which are familiar units to resource managers. Although *AEL* methodology does not require estimates of larval abundance in water volumes other than at the power plant intake, this advantage may be offset by (1) the need to obtain or model age-specific survival rates to predict

the adult equivalent losses and by (2) the need for information on the size of the adult population of interest to estimate relative population effects. Similar to *FH* methodology, with an assumption that any population reserve (compensation) that could account for the entrainment loss is lacking, entrainment losses yield population level impacts when *AEL* estimates are interpreted as direct losses to an inferred population.

4.4.3.1 Formulation

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

$$\widehat{AEL} = \sum_{j=1}^{n} \widehat{E_j}\widehat{S_j} \tag{8}$$

where

n = number of age classes,
$\widehat{E_j}$ = estimated number of larvae lost in age class j, and
$\widehat{S_j}$ = survival rate for the jth age class to adulthood (Goodyear 1978).

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

When age-specific survival rates from larval stage to recruitment into the fishery were available, *AEL* was calculated using survival from a representative age of the entrained larvae at DCPP. This age was calculated by dividing the average larval length at entrainment (minus hatch length) by a literature-based growth rate (**Table 4-3**). Age-specific survivorship for any interval of time (t) was then calculated following the formula (Ricker 1975)

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

where

$N_t =$ number of animals in the population at time t,
$N_0 =$ number of animals in the population at time $t = 0$,
$\frac{N_t}{N_0} = S$ (finite survivorship to time t),
$e = 2.71828...$(base of the natural log), and
$Z =$ instantaneous mortality rate.

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

$$\widehat{AEL} = \hat{E}_{Adj-T} \prod_{j=1}^{n} \widehat{S}_j \tag{9}$$

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where

$n =$ number of age classes from entrainment to recruitment and
$\widehat{S}_j =$ survival rate from the beginning to end of the jth age class.

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

$$\widehat{Var}(\widehat{AEL}) = \widehat{AEL}^2 \left(CV^2(\hat{E}_{Adj-T}) + \sum_{j=1}^{n} CV^2(\widehat{S}_j) \right). \tag{10}$$

An alternative analysis would be to compare $\widehat{AEL}$ with the size of the adult population of interest or with fishery harvest data. This method converts numbers of adult losses into fractional loss of the population of interest (e.g., stock assessment). However, information describing adult stocks is limited for many species, and independent field estimates of survival from time of entrainment to adulthood are not available for some species. For some species where such information is unavailable, we can estimate this parameter by assuming a stationary population where an adult

female must produce two adults (i.e., one male and one female). Overall survival (S_T) can then be estimated from total lifetime fecundity ($\overline{F}_T$) by the quantity

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

which leads to

$$\hat{S}_{adult} = \frac{2}{\widehat{\overline{F}_T} \cdot \hat{S}_{egg} \cdot \hat{S}_{larvae}}. \tag{11}$$

Substituting Equation 11 into the overall form of the *AEL* equation where

$$\widehat{AEL} = \hat{E}_{Adj-T} \cdot \hat{S}_{adult} \tag{12}$$

yields

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

where

$$\widehat{AEL} \equiv 2\widehat{FH}. \tag{13}$$

Without independent adult survival rates and assuming a 50:50 sex ratio, $\widehat{AEL}$ and $\widehat{FH}$ are deterministically related according to Equation 13, with an associated standard error of $\widehat{SE}(\widehat{AEL}) = 2\widehat{SE}(\widehat{FH})$. Equation 13 should be aligned so that the average female age is also the age of recruitment used in computing $\widehat{AEL}$. This alignment is accomplished by solving the simple exponential survival equation (Ricker 1975)

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

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

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

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

The estimates of mortality calculated from the AEL and FH approaches can be compared for the same time periods for taxa where independent estimates are available for (1) survival from entrainment to recruitment into the fishery and (2) entrainment back to hatching. These comparisons serve as a method of cross-validation for the demographic approaches to impact assessment.

4.4.3.2 Sensitivity Analysis

A sensitivity analysis for $\widehat{AEL}$ was computed in order to identify which parameters' estimates could be improved to increase the reliability of the estimate of adult losses. The analysis varied each input parameter ($\hat{\theta}$), using (1) an additive error model for the entrainment parameter ($\widehat{E_T}$ or $\hat{E}_{Adj-T}$) by $\pm 1.645\widehat{SE}(\hat{\theta})$ and (2) a multiplicative error model ($\hat{\theta}e^{\pm 1.645CV(\hat{\theta})}$) for survival parameters. These survival estimates used an assumed CV of 1.0 (100%) and were constrained $\leq$ 1. When estimates of survival to adult are unavailable, the sensitivity analysis performed for $\widehat{FH}$ was applicable. The CV for $\hat{\bar{I}}$ was used in propagating the variances of the adjusted entrainment estimate for calculating the ranges of possible values for $\widehat{AEL}$ in the sensitivity analyses. This approach helps to identify how a parameter's variability influences $\widehat{AEL}$.

4.4.3.3 Assumptions

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

- Values in the literature on life history parameters represent the fish population in the years and location of the Diablo Canyon study.

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

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

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

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

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

4.4.4 Empirical Transport Model *(ETM)*

The *ETM* estimates the total annual probability of mortality (P_M) associated with entrainment and requires an estimate of proportional entrainment (*PE*) as an input. Proportional entrainment estimates daily mortality imposed by DCPP on local larval populations (i.e., in the study grid), assuming no other types of mortality exist. In this context, *PE* is called conditional mortality (Ricker 1975). *PE* was calculated using the ratio based on intake samples and study grid samples. In previous entrainment studies using the *ETM* method, intake densities were assumed from weighted population concentrations (Boreman et al. 1981). The *PE* index is used in the *ETM* to estimate mortality rates resulting from cooling water withdrawals. 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 annual, conditional mortality due to entrainment, while accounting for the spatial and temporal variability in distribution and vulnerability of each life stage to cooling water withdrawals. The generalized form of the *ETM* incorporates many time-, space-, and age-specific estimates of mortality as well as information regarding spawning periodicity and larval duration. Most of this information is limited or unknown for the taxa being investigated at DCPP. Thus, the applicability of the *ETM* to the study at DCPP will be limited by the absence of empirically derived or reported demographic parameters needed as input to the model. Specifically, the length of time a larva is susceptible to entrainment must be estimated. However, the concept of summarizing *PE* over time with the *ETM* can be used to estimate entrainment effects over appropriate temporal scales, either by modeling or making assumptions about species-specific life histories. At DCPP, we employ a *PE* approach similar to the method described by MacCall et al. (1983) and used by Murdoch et al. (1989) in (Parker and DeMartini 1989) their final Marine Review Committee (MRC) report to the California Coastal Commission for SONGS. The annual estimate of *ETM* is formed from the weighted average of 12 approximately monthly surveys which pair entrainment and study grid larval samples.

4.4.4.1 Formulation of *PE* and *ETM*

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

$$\widehat{PE} = \frac{\widehat{N_E}}{\widehat{N_G}}$$

where

$\widehat{N_E}$ = estimated number of larvae entrained during the day, calculated as (estimated density of larvae in the water entrained that day) × (design specified daily cooling water intake volume),

$\widehat{N_G}$ = estimated number of larvae in the study grid that day, calculated as $\sum_{i=1}^{64}$[(estimated average density of larvae per cell)· (cell volume)] for i = 1, ..., 64 grid cells

where the estimated cell densities are obtained from the 72-hour grid survey that contains the 24-hour entrainment sampling period.

The *PE* estimate for days when grid surveys were not done is the estimate for the nearest day to the monthly survey. 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 4-3**). The estimated volume of the water directly inshore of the study grid was multiplied by the density of larvae collected in the row 1 cells, except for cells A1, D1, and E1.

Cell A1 is more offshore than the rest of the cells in row 1, due to the bend in the coastline at Point Buchon. An adjustment was not done for the volume of water inshore of cell A1 because a substantial volume would be added and the composition and abundance sampled in cell A1 would not adequately represent inshore volumes. Columns D and E are directly offshore from the Intake Cove where entrainment samples were collected. The density estimate from entrainment sampling was used for the areas inshore of cells D1 and E1. This estimated number of larvae inshore of the grid was added to the grid abundance number estimated for the cells in row 1 before *PE* and *ETM* calculations were done.

The ETWG recommended using entrainment samples in adjusting only the volumes inshore of cells D1 and E1. The estimated number of larvae inshore of the grid cells D1 and E1 was somewhat larger than the estimated entrainment numbers used in *PE* because the sum of water volumes inshore of D1 and E1 (4.90 and 5.95 million m^3, respectively) is slightly greater than the daily flow through the power plant (9.60 million m^3). However, the estimated entrainment density could be used to represent the remaining inshore volumes (total inshore volume excluding A1 is 38.2 million m^3), in which case the *PE* value (and therefore estimated mortality) would decrease especially for those nearshore species not found commonly in the study grid. While not appropriate for taxa with larvae dispersed throughout the study grid, an adjustment to the unsampled inshore volumes may be reasonable for some nearshore species whose larvae are potentially more abundant closer to shore.

The *PE* value represents the effects of a number of processes operating over a day and is estimated in two year groups, each year having 12 approximately monthly surveys. Proportional entrainment was calculated for the 12 paired nearshore study grid and entrainment surveys conducted between July 1, 1997 and June 30, 1998, and for another 12 pairs from July 1, 1998 to June 30, 1999. To estimate specific *PE* ratios, each 24-hour entrainment abundance estimate was temporally paired with the abundance estimate from the 72-hour nearshore grid survey.

The purpose of the DCPP study is to estimate entrainment effects within the study grid and attempt to extrapolate those effects to a population of inference. Boreman et al. (1981) point out that if any members of the population are located outside the study grid area, then the ETM will overestimate the conditional entrainment mortality for the entire population. In their study of entrainment by SONGS, Parker and DeMartini (1989) incorporated the inference population (which was an extrapolation to the entire Southern California Bight, an area extending about 500 km) directly into their estimate of *PE*. In the following *ETM* analyses, *PE* is multiplied by the fraction of the population in the study grid area. Let P_S = (number of larvae at risk in the study grid) / (number of larvae in the population of inference). The fraction of larva being entrained from the population of inference on a given day is then the product

$$PE_i \cdot P_S \ .$$

The boundaries of the population affected by entrainment could vary from local (e.g., a subarea of the study grid) to regional (e.g., fishery management units, zoogeographic range). For some species the area of the study grid may represent the population of inference and, in these cases, $P_S = 1$. For other species, the population of inference will likely be larger than the study grid. In fact, the population of inference depends both on the species and on what appeals usefully to intuition. Therefore, *ETM* is computed over a range of values of P_S for each of the target taxa. The curves from these functions can be used to determine the *ETM* at any value of P_S. The curves can also be interpreted as a continuous probability function representing the risk of entrainment to the larvae at different values of P_S. This representation replaces the need for sensitivity analysis for P_M to the population of inference implied by P_S. Several approaches will be used to define the proportion of the population of inference for the different taxa groups.

These point estimates of P_S (and their ranges) will also be represented on the curves for each of the target taxa. This representation of the population at risk is more biologically meaningful than sensitivity analyses using confidence intervals around a single estimate of P_M.

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

$$(1-\widehat{PE_i}\cdot\widehat{P_S})^{\hat{d}}.$$

Larval duration from hatching to entrainment was calculated from growth rates using either the mean or maximum size at entrainment. The value for d was computed by dividing an estimate of growth rate into the change in length based on either the mean or maximum size at entrainment (**Tables 4-3** and **4-4**). Although the maximum larval size produces the longest duration of susceptibility, larval entrainment probabilities may be age dependent. The mean size is used to illustrate the sensitivity of the estimate of population-wide mortality to larval duration of susceptibility. The mean length of target taxa larvae from all entrainment samples was calculated, and length-frequency histograms were used to determine the central 98% of the distribution. The minimum and maximum sizes used for analyses were determined after removing the largest and smallest 1% of the values. This procedure helped account for 1) outliers that were the result of larger post-larval fishes, 2) inadvertently measured larval fish fragments from the samples, 3) shrinkage in preservation fluid, and 4) variations in hatch length. As a result, the larval lengths that were used to estimate ages were the 98% center interval of all the lengths.

It is possible that aging was biased, even though standard lengths of larval fishes (i.e., measurements of minimum, mean and maximum), and larval growth rates were applied to estimate the ages of the entrained larvae. It was assumed that larvae shorter than the minimum length of the central 98% of the distribution were just hatched and aged zero days. Subsequent ages were estimated using this length. Other reported data for various species suggests that hatching length can be either smaller or larger than estimated from DCPP samples, and indicates that the smallest observed larvae represent either natural variation in hatch lengths within the population or the phenomenon of shrinkage following preservation (Theilacker 1980). The

possibility remains that all larvae from the observed minimum length to the greatest reported hatching length (or to some other size) could have just hatched, leading to overestimation of ages for all larvae. Both *FH* and *AEL* estimation use the mean and minimum sizes to estimate age at entrainment and are subject to similar biases.

Twelve monthly survey periods were used to estimate larval mortality (P_M) due to entrainment for 2 one-year study periods. P_M was estimated by the equation

$$\widehat{P_M} = 1 - \sum_{i=1}^{12} \hat{f}_i (1 - \widehat{PE}_i \cdot \widehat{P_S})^{\hat{d}} \tag{15}$$

where

$\widehat{PE}_i$ = estimate of proportional entrainment for the ith monthly survey (i = 1,...,12),
$\widehat{P_S}$ = estimated proportion of the fish population in the study grid,
$\hat{f}_i$ = annual proportion of the total larvae hatched during the ith monthly survey, and
$\hat{d}$ = the estimated number of days of larval life.

To establish independent survey estimates, it is assumed that during each survey a new and distinct cohort of larvae is subject to entrainment. Each of the twelve surveys was weighted by $\hat{f}_i$ and estimated as the proportions of the yearly entrainment in all the days of the ith survey period. The survey dates (from July 1997 through June 1998 and July 1998 through June 1999) of the paired entrainment samples and number of days in the survey periods are found in **Table 4-5**. For each of the two years, the sum of the proportions equals one:

$$\hat{f}_i = \frac{\widehat{E_i}}{\sum_{j=1}^{12} \widehat{E_j}} \text{ and } \sum_{i=1}^{12} \hat{f}_i = 1.$$

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

The estimate of the population-wide probability of entrainment ($\widehat{PE}_i \cdot \widehat{P_S}$) is the central feature of the *ETM* approach (Boreman et al. 1981; MacCall et al. 1983). If a population is stable and

stationary, then $\widehat{P_M}$ also estimates the effects on the fully-recruited adult age classes when uncompensated natural mortality from larva to adult is assumed.

4.4.4.2 Formulation of $\widehat{P_S}$

The relationship between $\widehat{P_M}$ and $\widehat{P_S}$ is represented by the sets of curves presented for each of the target taxa. Discrete values of $\widehat{P_S}$ are also presented for each of the target taxa. These values are computed differently for three groups of taxa identified by the distribution of adults:

- Offshore - Pacific sardine and northern anchovy
- Subtidal - rockfishes, flatfishes, crabs, cabezon, white croakers, and blackeye gobies
- Intertidal and shallow subtidal or nearshore sculpins, kelpfishes, pricklebacks, and greenlings

The incorporation of P_S into the *ETM* is typically defined by the ratio of the area or volume of the study grid 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, $\widehat{P_S}$ can also be computed using the estimate of the larval or adult population in the study grid, 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 $\widehat{P_S}$ for the proportion of the population will be the same as the value computed using area or volume. For offshore and subtidal target taxa whose larval distribution extends to the offshore edge of the study grid, $\widehat{P_S}$ will be calculated as the ratio

$$P_S = N_G / N_P$$

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 is presented is section **4.4.4.1** in the calculation of *PE*, i.e.

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

where

A_{G_k} = area of grid (G) cell k,

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

$\rho_{i,k}$ = density (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 densities, using water current measurements. First, a conceptual model was formulated to extrapolate larval densities (per m^3) offshore of the grid

$$\widehat{P_S} = \frac{\widehat{N_G}}{\widehat{N_P}} = \frac{\sum_{j=1}^{K_G} L_{G_j} \cdot W_j \cdot \bar{D}_j \cdot \overline{\text{density}_j}}{\sum_{j=1}^{K_P} L_{P_j} \cdot W_j \cdot \bar{D}_j \cdot \overline{\text{density}_j}} \tag{16}$$

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 the population in the *j*th stratum based on current data,

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

$\overline{\text{density}_j}$ = 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 $K_P > K_G$ alongshore strata.

Formula (16) above describes discrete values in intervals of a continuous function. Therefore, to ease implementation, an essentially equivalent formula incorporates the use of grid cell densities during the *i*th sampling period, $\rho_{i,k}$, and integrates 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:

$$\widehat{P_{S_i}} = \frac{\widehat{N_{G_i}}}{\widehat{N_{P_i}}} = \frac{\widehat{N_{G_i}}}{\widehat{N_{G_i}}\left(\frac{L_{P_i}}{L_G}\right) + L_{P_i} \cdot \int_{W_O}^{W_{max}} \rho(w)\,dw}$$

where

L_{P_i} = alongshore length of the population (P) in the *i*th study period based on current data.

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 can occur outside the study grid area.

Note 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 4-7**).

The alongshore and onshore current velocities were measured at a current meter positioned approximately 1 km west of the DCPP intake at a depth of approximately 6 m (**Figure 4-3**). 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 4-8** 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 (average of two tangents, **Section 4.2.1.2**). The movement of water was then tracked during the period from April 1997 through June 1999. A total alongshore length or displacement in kilometers can be calculated from these data using the range of both upcoast and downcoast 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 movement. Onshore movement used to extrapolate densities offshore can also be calculated from this graph by excluding periods of offshore movement. Within this scenario, there are two subclasses:

1. For species where the regression of density versus offshore distance has a negative slope, the offshore distance predicted where density is zero (i.e., integral of zero) will be calculated. The alongshore distance would be calculated from the water current data.

2. For species where the regression of density versus offshore distance has a slope of ≥ 0, either the offshore distance from the water current data or an average distance based on the depth distribution of the adults offshore will be used. Literature values (e.g., CalCOFI) will be used to place a ceiling on both the distance and density values used in the offshore extrapolation.

Parameter values needed in performing the extrapolation were obtained by using analysis of covariance based on all of the data from the surveys within the period July 1997 through June

1998. Another set of parameter values was calculated for the period July 1998 through June 1999. A quadratic model was tested in the analysis

$$\delta_{ij} = \alpha_i + \beta w_{ij} + \gamma w_{ij}^2 + \varepsilon_{ij}$$

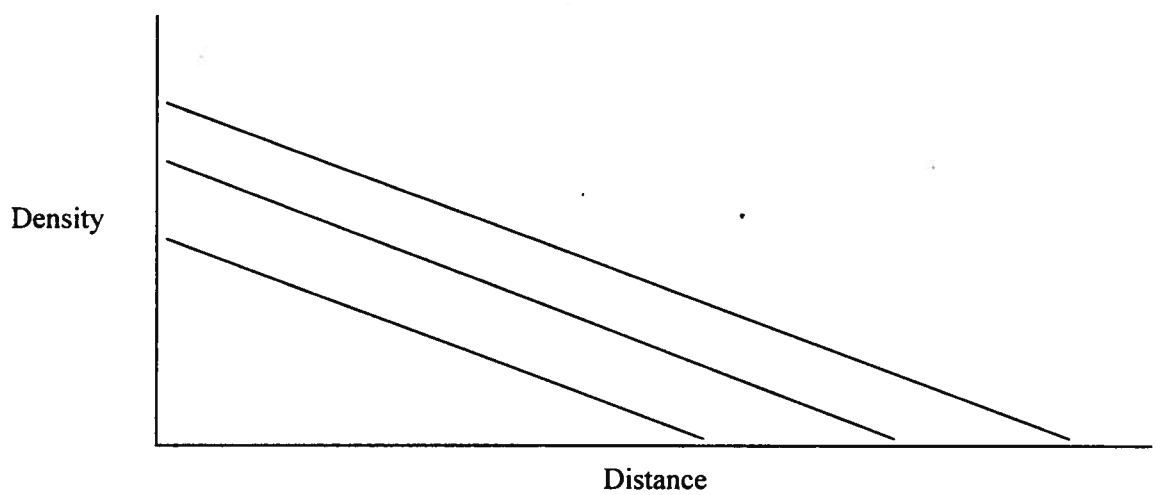
where

ε_{ij} = normally distributed error term with mean of zero,
w_{ij} = distance for the ith observation in the jth survey,
δ_{ij} = larval density for the ith observation in the jth survey and
α_i, β, γ = regression coefficients.

A linear model produced a better fit in all cases:

$$\delta_{ij} = \alpha_i + \beta w_{ij} + \varepsilon_{ij}.$$

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. The illustration below depicts three hypothetical survey periods.



Note that unique distances (i.e., intercepts) are used from each survey period when calculating N_P. The value of P_S computed using this method to extrapolate increasing or decreasing densities offshore will result in non-uniform densities over the extrapolated area of inference.

Therefore, this value would not be equal to a P_S that is computed using the ratio of area or volume from the grid and the population of inference. The estimate of P_M obtained using the extrapolated value of P_S will be compared for the Pacific sardine and northern anchovy to their fishery management areas and existing biomass estimates. The ratio of the extrapolated area to the fishery management area will be used to estimate the effects of entrainment on these populations.

A P_S for nearshore taxa will be calculated using the alongshore length of the sampling grid, L_G, and the total alongshore displacement, L_P, to predict a coastwise fraction of the population of inference (**Figure 4-7**). The total alongshore displacement in the *i*th survey, L_{P_i}, includes both upcoast and downcoast movement calculated during a period equal to the larval duration before each survey. No onshore or offshore current movement was incorporated in the calculated population fraction:

$$\widehat{P_S} = \frac{L_G}{\frac{1}{n}\sum_{i=1}^{n} L_{P_i}} = \frac{L_G}{\overline{L_P}} .$$

4.4.4.3 Assumptions

Assumptions associated with the estimation of P_M include the following:

- The samples at each survey period represent a new and independent cohort of larvae.
- The monthly estimates of larval abundance represent a proportion of total annual larval production during that month.
- The conditional probability of entrainment (PE_i) is constant within monthly survey periods.
- P_S accurately characterizes the fraction of the target population represented by the sampling grid.
- Lengths and applied growth rates of larvae accurately estimate larval duration

Table 4-1. List of target taxa for assessment in DCPP 316(b) Demonstration. Target taxa, listed in phylogenetic order, were selected at the May 7, 1999 ETWG meeting.

Common Name	Scientific Name
Brown rock crab	*Cancer antennarius*
Slender crab	*Cancer gracilis*
Pacific sardine	*Sardinops sagax*
Northern anchovy	*Engraulis mordax*
Blue rockfish complex	†*Sebastes* spp. V / *S. mystinus*
KGB rockfish complex	†*Sebastes* spp. V_De/V_D_
Painted greenling	*Oxylebius pictus*
Smoothhead sculpin	*Artedius lateralis*
Snubnose sculpin	*Orthonopias triacis*
Cabezon	*Scorpaenichthys marmoratus*
White croaker	*Genyonemus lineatus*
Monkeyface prickleback	*Cebidichthys violaceus*
Clinid kelpfishes	*Gibbonsia* spp.
Blackeye goby	*Coryphopterus nicholsi*
Sanddab	*Citharichthys* spp.
California halibut	*Paralichthys californicus*

† *Sebastes* spp. V/ *S. mystinus*, and *Sebastes* spp. V_De/V_D_ refer to pigmentation groups within the larval rockfish complex that were used as a taxonomic group in this study.

Table 4-2. Pigment groups, parturition peak periods (PPM), and seasonality of 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 stop)
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)

LETTER CODE	SPECIES	COMMON NAME	SUBGENUS	'77-'84 PPM	Parturition period N and C CA (Wyllie Echeverria 1987)
V_P	**Long ventral series, no dorsal, pectoral pigment**				
	S. chlorostictus	greenspotted	Sebastomus	?	Apr-Sep
	S. ensifer	swordspine	Sebastomus	?	Uncommon in central CA (Miller and Lea 1972)
V_D	**Long ventral series, short dorsal series, no pectoral pigment**				
	S. saxicola	stripetail	Allosebastes	Jan	Jan-Mar (Nov-Mar, Phillips 1964)
V_D_	**Long ventral series, short dorsal series, no pectoral pigment**				
	S. atrovirens	kelp	Mebarus	ND	ND
	S. chrysomelas	black and yellow	Pteropodus	Feb	Feb-Mar (Jan-May (Larson 1980))
	S. maliger	quillback	Pteropodus	Apr	Apr-Jul
	S. nebulosus	China	Pteropodus	?	?
	S. semicinctus	halfbanded	Allosebastes	ND	Jan-Apr for southern CA (Love et al. 1990)
V_De	**Long ventral series, elongating dorsal series, pectoral pigment**				
or	*S. auriculatus*	brown	Auctospina	June	Dec-Jan, principally May-Jul (May, Larson 1980)
V_DeP	*S. carnatus*	gopher	Pteropodus	Mar	Mar-May (May, Larson 1980)
or	*S. caurinus*	copper	Pteropodus	Feb	Feb
V_dep	*S. dalli*	calico	Allosebastes	?	Uncommon in central CA (Miller and Lea 1972)
	S. rastrelliger	grass	Pteropodus	ND	ND

Table 4-2. (continued). Pigment groups, parturition peak periods (PPM), and seasonality of preflexion *Sebastes* spp. larvae from Nishimoto (in prep.).

LETTER CODE	SPECIES	COMMON NAME	SUBGENUS	'77-'84 PPM	Parturition period N and C CA (Wyllie Echeverria 1987)
V	**Short ventral series, no dorsal series, no pectoral**				
	S. aleutianus	rougheye	Zalopyr	ND	
	S. alutus	POP	Acutomentum	Mar	Jan-Mar (?)
	S. brevispinis	silvergrey	Acutomentum	ND	
	S. crameri	darkblotched	Eosebastes	Jan	Nov-Mar (Nov-Mar, Larson 1980)
	S. diploproa	splitnose	Allosebastes	Jul	Jan-Sep (Feb-Jul, Phillips 1964)
	S. elongatus	greenstriped	Hispaniscus	May	May-Jul
	S. macdonaldi	Mexican	Acutomentum	ND	
	S. miniatus	vermilion	Rosicola	Sep	Sep (Nov, Moser 1967; Nov-Mar, Phillips 1964)
	S. nigrocinctus	tiger	Sebastichtys	ND	
	S. proriger	redstripe	Allosebastes	?	Jul-Aug
	S. rosaceus	rosy	Sebastomus	Jun	Apr-Jul
	S. ruberrimus	yelloweye	Sebastopyr	Jun	Apr-Jul (?)
	S. serriceps	treefish	Sebastocarus	ND	
	S. umbrosus	honeycomb	Sebastomus	ND	ND
	S. wilsoni	pygmy	Acutomentum	ND	
	S. zacentrus	sharpchin	Allosebastes	?	May-Jun
VP	**Short ventral series, no dorsal series, various patterns of pectoral pigmentation** (At smaller size might fit into group **V** above)				
	S. constellatus	starry	Sebastodes	Apr	Apr-May
	S. eos	pink	Sebastomus	ND	ND
	S. goodei	chilipepper	Sebastodes	Jan	Nov-Jun (Nov-Mar, Phillips 1964)
	S. helvomaculatus	rosethorn	Sebastomus	?	May-Jun
	S. levis	cowcod	Hispaniscus	Dec	Dec-Feb
	S. melanostomus	blackgill	Eosebastes	Feb	Feb-Apr
	S. paucispinis	bocaccio	Sebastodes	Feb	Jan-May (Nov-Mar, Moser 1967)
	S. rosenblatti	greenblotched	Sebastomus	ND	ND
	S. rubrivinctus	flag	Hispaniscus	?	Jul
Vdp	**Short ventral series, develops dorsal series, develops various patterns of pectoral pigmentation** (At stage 1-2 can confuse with **V** above due to lack of dorsal and pectoral pigmentation.)				
	S. entomelas	widow	Acutomentum	Feb	Dec-Apr (Nov-Mar, Phillips 1964)
	S. flavidus	yellowtail	Sebastosomus	Feb	Jan-Jul (Nov-Mar, Phillips 1964)
	S. melanops	black	Sebastosomus	Feb	Jan-May
	S. mystinus	blue	Sebastosomus	Jan	Nov-Jan (Nov-Jan, Wales 1952)
	S. rufus	bank	Acutomentum	Feb	Dec-May
	S. serranoides	olive	Sebastosomus	Feb	Jan-Mar
VD	**Short ventral series, short dorsal series**				
	S. aurora	aurora	Eosebastes	Apr	Mar-May
	S. babcocki	redbanded	Rosicola	May	May
	S. gilli	bronzespotted	Sebastosomus	?	Uncommon in central CA (Miller and Lea 1972)
	S. hopkinsi	squarespot	Acutomentum	Mar	Feb-Mar
	S. jordani	shortbelly	Sebastodes	Feb	Feb-Apr (Nov-Apr, Phillips 1964)
	S. ovalis	speckled	Acutomentum	May	May
	S. pinniger	canary	Rosicola	Dec	Dec-Mar (Nov-Mar. Moser 1967)
Species without descriptions or illustrations					
	S. phillipsi	chameleon	?	?	Uncommon in central CA (Miller and

Table 4-3. Larval growth estimates for entrainable fish larvae at DCPP based on literature citations. Growth estimates are used to estimate larval durations in **Table 4-4**. Growth rates are not listed for *Cancer* crabs because of step-wise growth increments between larval stages. Larval durations for crabs were taken directly from laboratory studies and are listed in **Table 4-4**.

Family *Species*	Larval Growth Rate (mm/d)	Reference
Cancridae		
Cancer antennarius	—	Roesijadi 1976
Cancer gracilis	—	Ally 1975
Clupeidae		
Sardinops sagax	0.238	Miller 1952
Engraulididae		
Engraulis mordax	0.445	Methot and Kramer 1979
Scorpaenidae		
Sebastes spp. KGB	0.140	Love and Johnson 1999 and Yoklavich pers. comm.
Sebastes spp. V/*S. mystinus*	0.140	Yoklavich et al. 1996 and Yoklavich pers. comm.
Hexagrammidae		
Oxylebius pictus	0.083	Freeman et al. 1985
Cottidae		
Artedius lateralis	0.083	Freeman et al. 1985
Orthonopias triacis	0.083	Freeman et al. 1985
Scorpaenichthys marmoratus	0.300	O'Connell 1953; Moser 1996
Sciaenidae		
Genyonemus lineatus	0.200	Murdoch et al. 1989
Stichaeidae		
Cebidichthys violaceus	0.248	Stepien 1986
Clinidae		
Gibbonsia spp.	0.248	Stepien 1986
Gobiidae		
Coryphopterus nicholsi	0.273	Steele 1997
Paralichthyidae		
Citharichthys spp.	0.200	Kendall 1992; Sakuma 1995
Paralichthys californicus	0.280	Allen et al. 1990

Table 4-4. Mean and maximum larval stage durations in days from hatching to entrainment estimated for fish larvae collected at DCPP. Maximum duration is based on the central 98% of the length frequency distribution and growth rates from the scientific literature (Table 4-3). Crab megalopal duration estimated from Roesijadi 1976 and Ally 1975.

Family *Genus and Species*	Shortest larva entrained (mm)	Mean larva entrained (mm)	Longest larva entrained (mm)	Estimated larval durations Mean-Max (d)
Cancridae				
Cancer antennarius (megalops)	—		—	42.7
Cancer gracilis (megalops)	—		—	41.6
Clupeidae				
Sardinops sagax	2.4	5.5	19.1	13.3–70.6
Engraulididae				
Engraulis mordax	2.1	6.1	24.9	9.0–51.2
Scorpaenidae				
Sebastes spp. KGB	3.3	4.2	5.6	6.2–16.4
Sebastes spp. V / *S. mystinus*	2.7	3.6	4.5	6.4–12.9
Hexagrammidae				
Oxylebius pictus	3.2	4.1	5.2	10.4–24.1
Cottidae				
Artedius lateralis	2.4	3.1	5.3	8.9–34.9
Orthonopias triacis	2.6	3.6	6.1	11.4–42.2
Scorpaenichthys marmoratus	3.7	4.8	6.1	3.6–8.0
Sciaenidae				
Genyonemus lineatus	1.1	2.4	5.5	6.7–22.0
Stichaeidae				
Cebidichthys violaceus	5.7	2.6	12.0	6.6–25.4
Clinidae				
Gibbonsia spp.	4.1	6.4	12.0	9.0–31.6
Gobiidae				
Coryphopterus nicholsi	2.0	2.6	3.4	2.2–5.2
Paralichthyidae				
Citharichthys spp.	1.3	2.6	3.5	6.7–11.0
Paralichthys californicus	1.6	3.3	7.8	6.0–22.1

Table 4-5. Study grid survey start dates and the period of days used in calculations for each survey, for two sampling periods: June 1997 through July 1998, and June 1998 through July 1999. The start date of the entrainment surveys that paired with the grid surveys generally started 24 hours after the beginning of each 72–hour grid survey.

	Survey Start Date	Period Start	Period End	Days
Analysis Period 3				
Surveys	Jul 21, 1997	Jul 4, 1997	Aug 7, 1997	35
	Aug 25, 1997	Aug 8, 1997	Sep 11, 1997	35
	Sep 29, 1997	Sep 12, 1997	Oct 9, 1997	28
	Oct 20, 1997	Oct 10, 1997	Nov 2, 1997	24
	Nov 17, 1997	Nov 3, 1997	Nov 28, 1997	26
	Dec 10, 1997	Nov 29, 1997	Dec 31, 1997	33
	Jan 22, 1998	Jan 1, 1998	Feb 8, 1998	39
	Feb 26, 1998	Feb 9, 1998	Mar 7, 1998	27
	Mar 18, 1998	Mar 8, 1998	Mar 31, 1998	24
	Apr 15, 1998	Apr 1, 1998	May 1, 1998	31
	May 18, 1998	May 2, 1998	May 28, 1998	27
	Jun 8, 1998	May 29, 1998	Jul 3, 1998	36
Analysis Period 4				
Surveys	Jul 21, 1998	Jul 3, 1998	Aug 7, 1998	36
	Aug 26, 1998	Aug 8, 1998	Sep 5, 1998	29
	Sep 16, 1998	Sep 6, 1998	Sep 25, 1998	20
	Oct 6, 1998	Sep 26, 1998	Oct 23, 1998	28
	Nov 11, 1998	Oct 24, 1998	Nov 24, 1998	32
	Dec 9, 1998	Nov 25, 1998	Dec 25, 1998	31
	Jan 12, 1999	Dec 26, 1998	Jan 22, 1999	28
	Feb 3, 1999	Jan 23, 1999	Feb 23, 1999	32
	Mar 17, 1999	Feb 24, 1999	Mar 30, 1999	35
	Apr 14, 1999	Mar 31, 1999	May 3, 1999	34
	May 24, 1999	May 4, 1999	Jun 7, 1999	35
	Jun 23, 1999	Jun 8, 1999	Jul 2, 1999	25

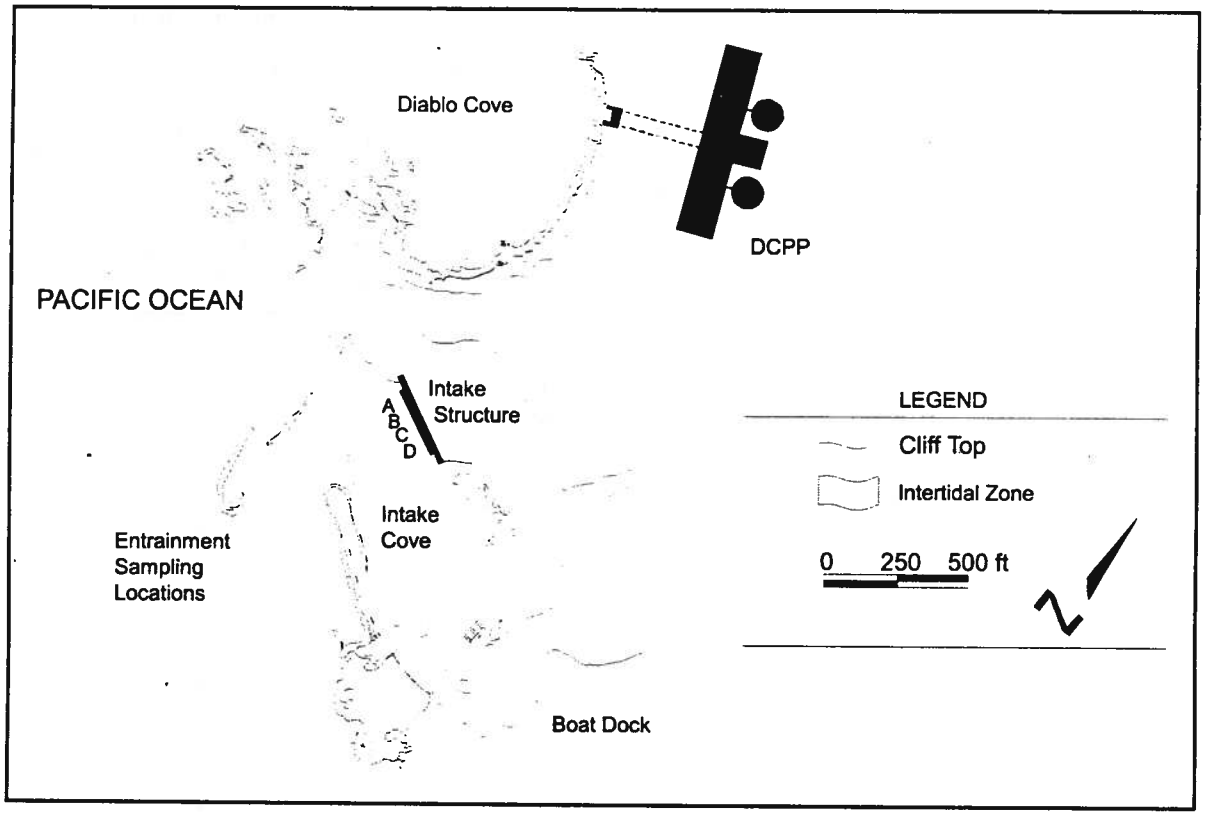


Figure 4-1. Diagram of DCPP Power Plant, Intake Cove and Entrainment Sampling Locations.

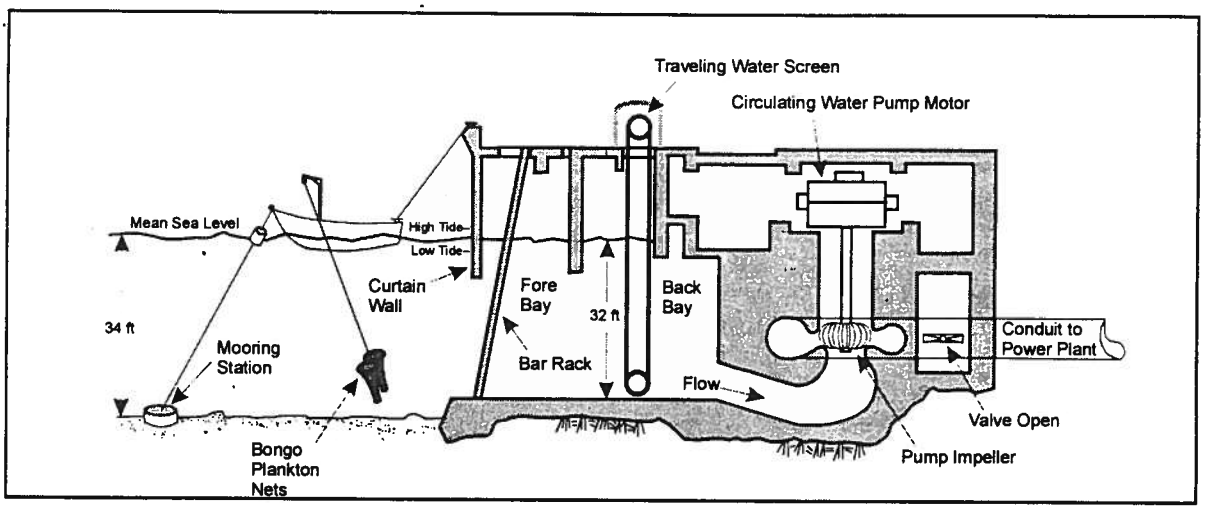


Figure 4-2. Cross-section view of the DCPP intake structure illustrating the location of the entrainment sampling sites.

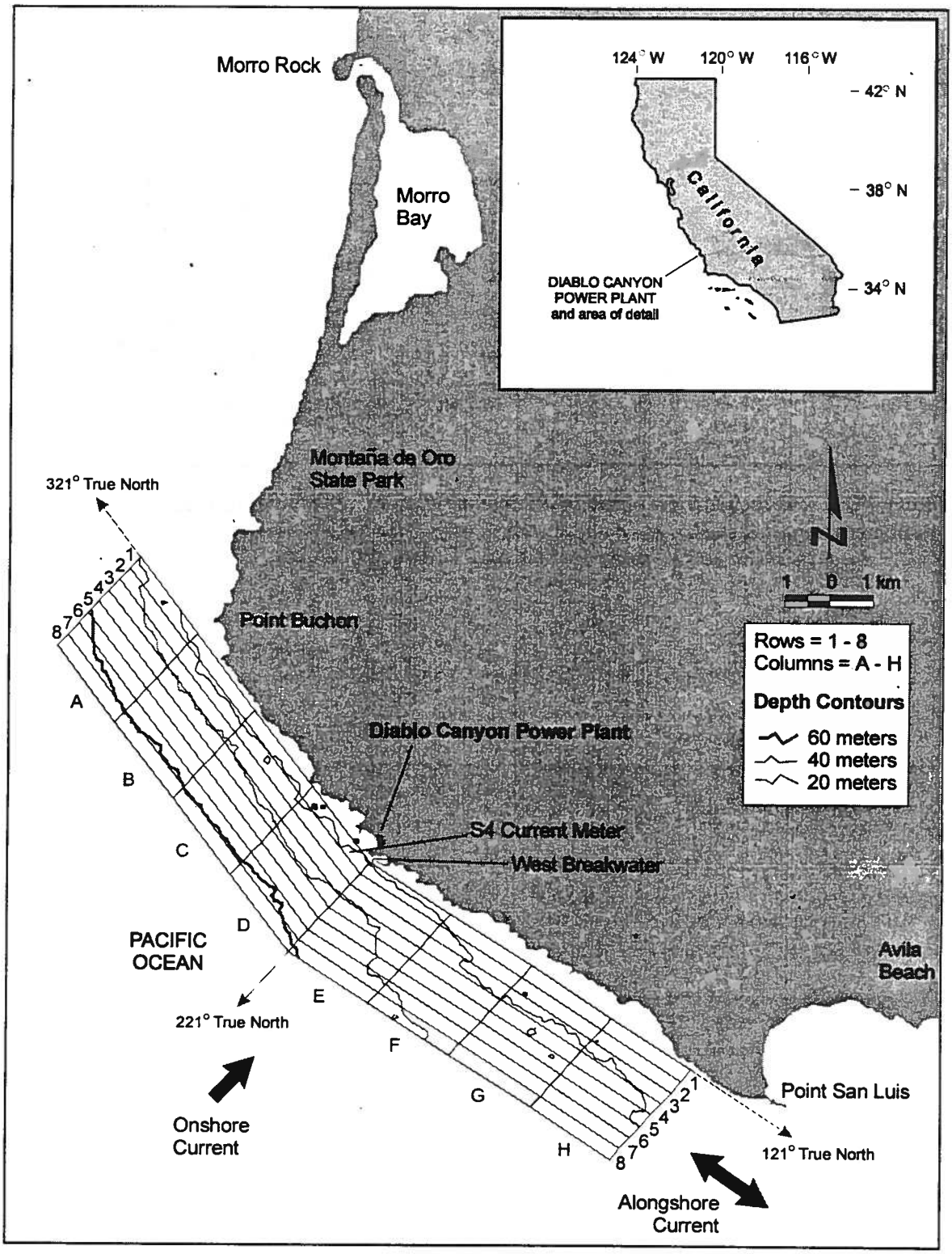


Figure 4-3. DCPP 316(b) study grid and depth contours.

H	G	F	E	D	C	B	A
8	8	8	8	8	8	8	8
7	7	7	7	7	7	7	7
6	6	6	6	6	6	6	6
5	5	5	5	5	5	5	5
4	4	4	4	4	4	4	4
3	3	3	3	3	3	3	3
2	2	2	2	2	2	2	2
1	1	1	1	1	1	1	1

DCPP Intake Cove

Figure 4-4. An example of the "ping-pong" sampling track employed in grid cell sampling; the starting cell (F1) and the initial southward direction of the transect were randomly selected. All 64 cells are sampled during the 72-hour survey period, weather permitting. DCPP's Intake Cove is located east of the juncture between cells E1 and D1.

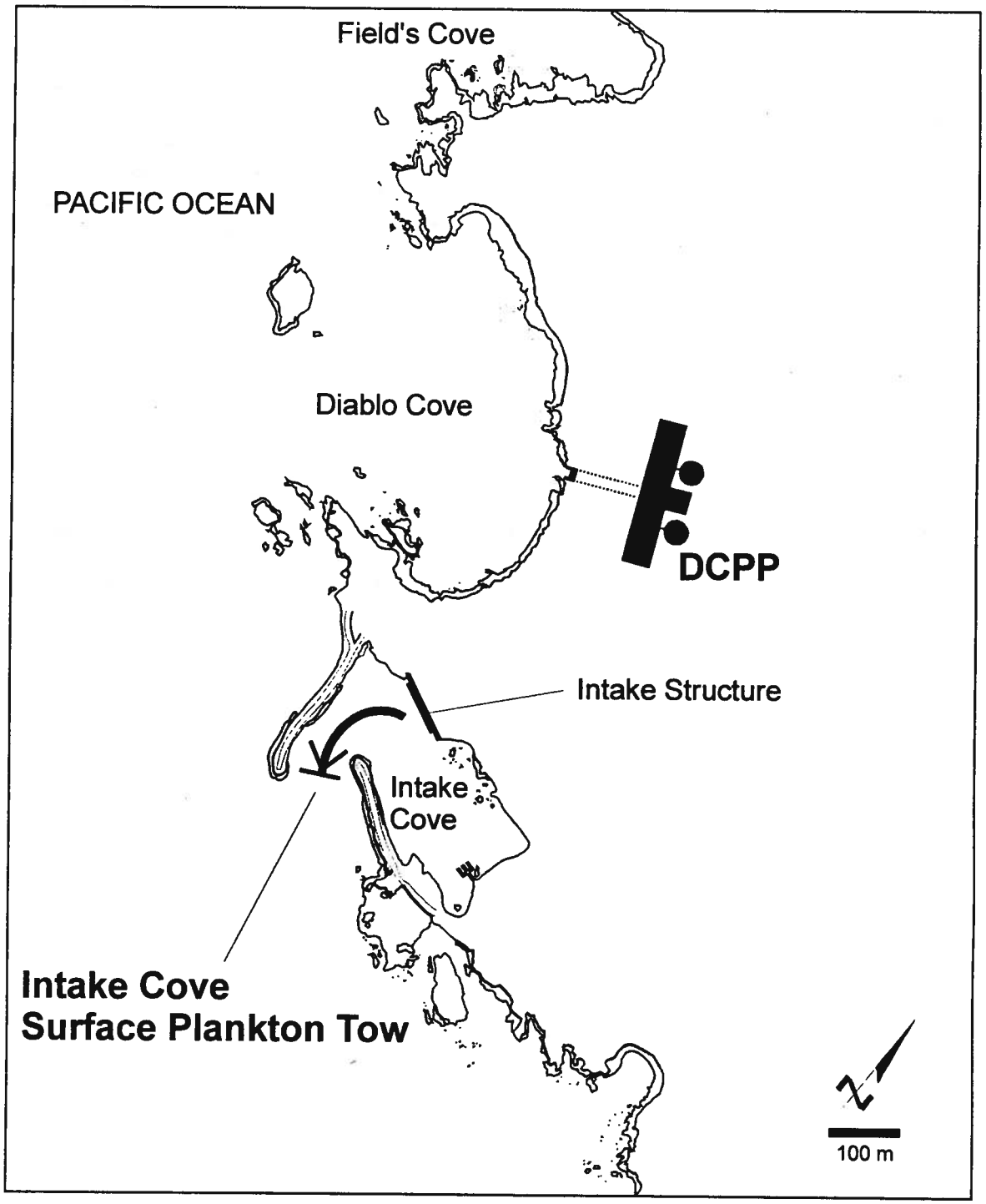
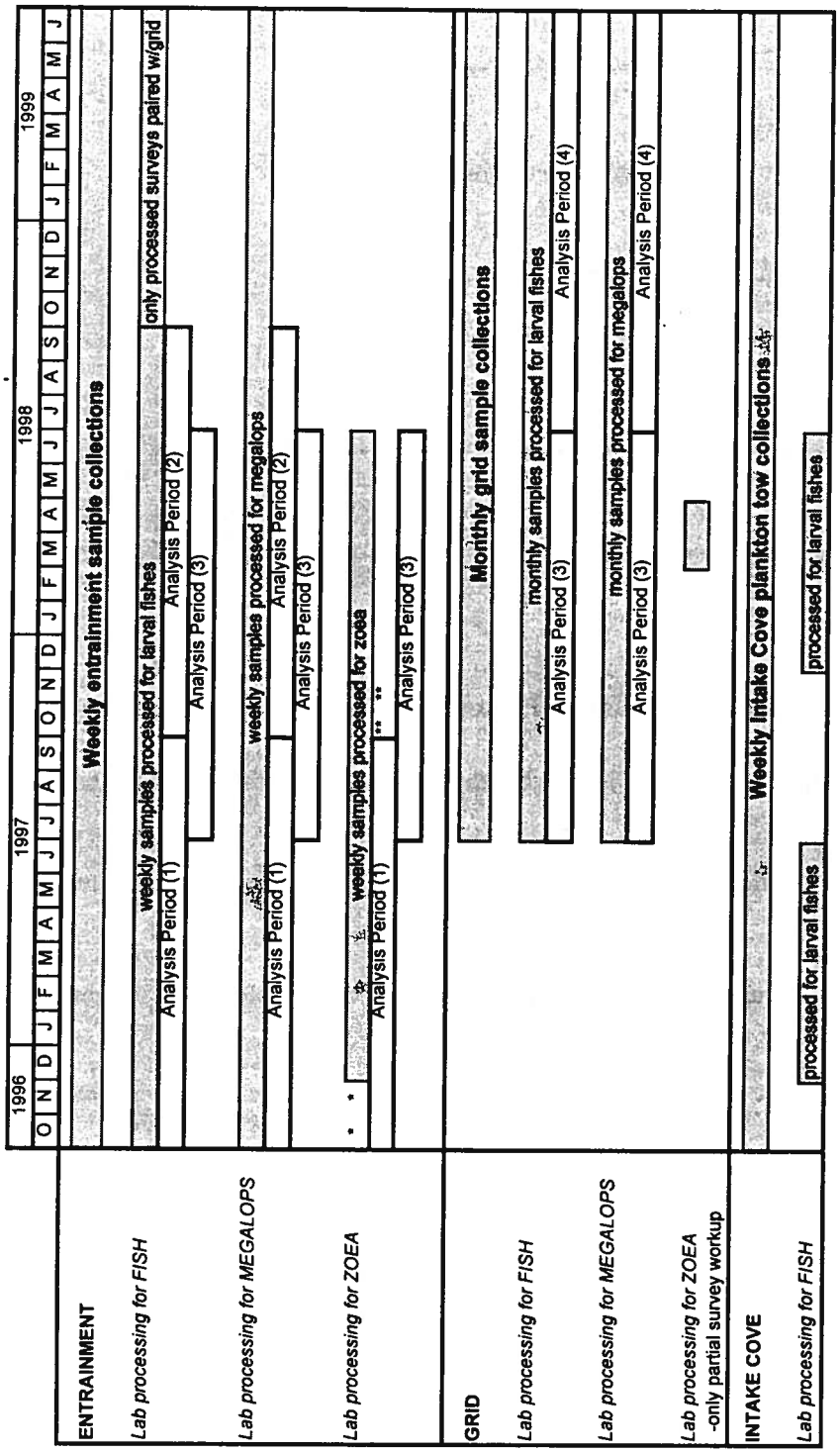


Figure 4-5. Diagram of DCPP Intake Cove surface plankton tow sampling location.



* zoea not removed from these samples.
** zoea density substituted for October and November 1996
Analysis Period (1): October 23, 1996 - September 30, 1997
Analysis Period (2): October 1, 1997 - September 30, 1998
Analysis Period (3): July 1, 1997 - June 30, 1998
Analysis Period (4): July 1, 1998 - June 30, 1999

Figure 4-6. Summary of entrainment, study grid, and Intake Cove surface plankton tow sample collection, laboratory processing, and analysis periods.

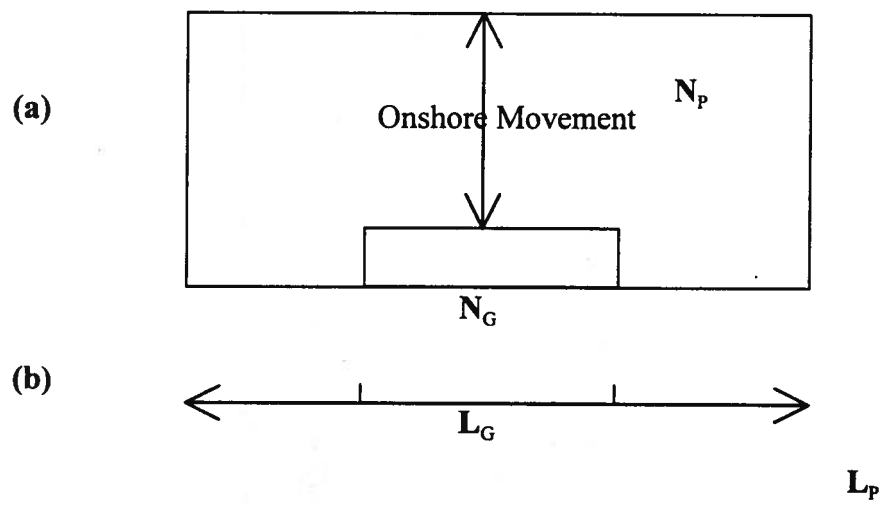


Figure 4-7. Two approaches for calculating $\widehat{P_S}$ based upon water current measurements: (a) For species dispersed throughout the sampling grid (small rectangle), both alongshore and onshore components are used to calculate $\widehat{P_S} = \frac{\widehat{N_G}}{\widehat{N_P}}$, where $\widehat{N_G}$ is the larval population estimate of the offshore grid and $\widehat{N_P}$ is the estimate of larval abundance of the population of inference; (b) For nearshore species, only the alongshore component is used to calculate $\widehat{P_S} = \frac{L_G}{L_P}$, where L_G is the length of the offshore grid and L_P is the estimated alongshore movement through the offshore grid.

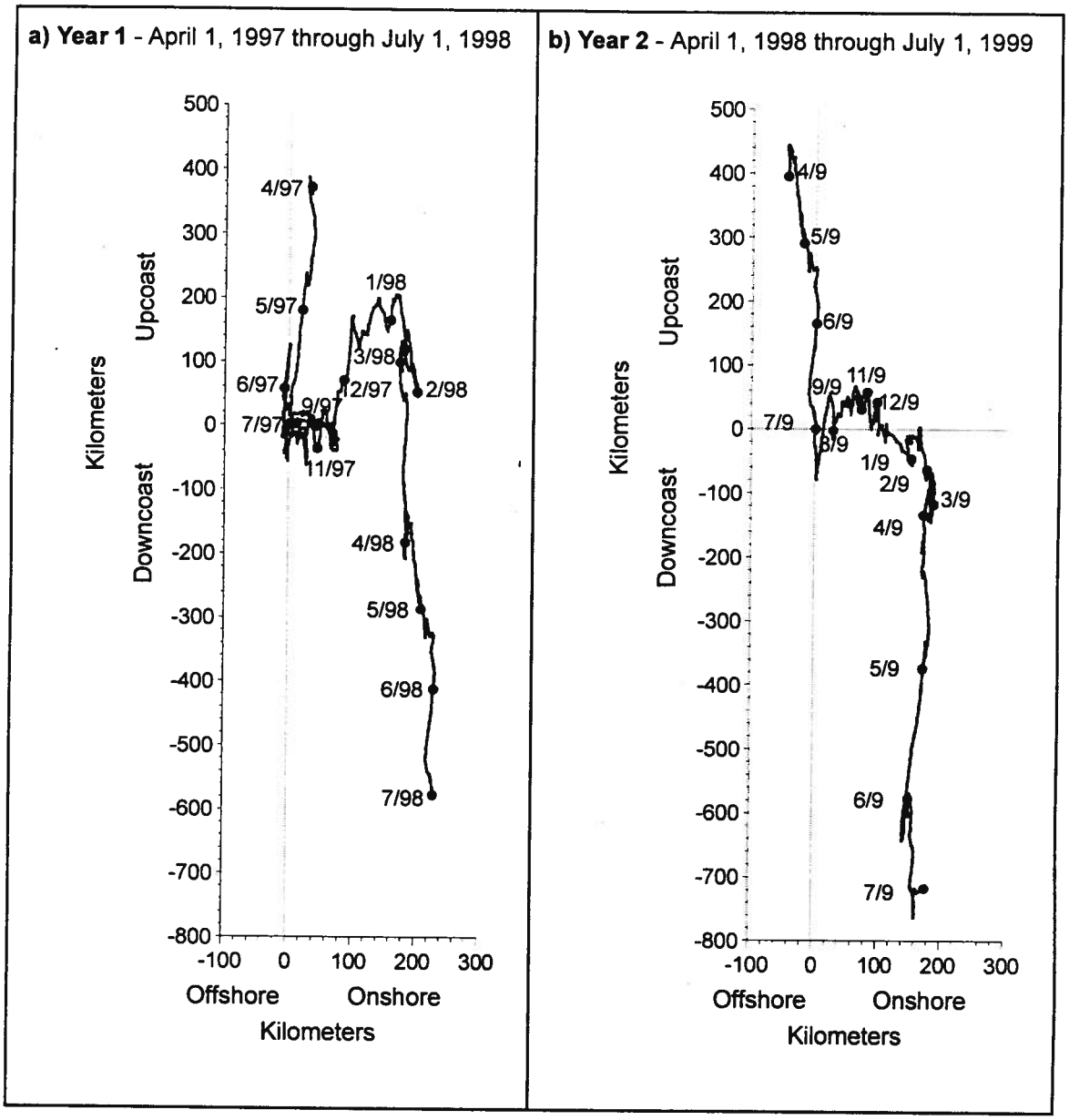


Figure 4-8. Cumulative upcoast/downcoast and onshore/offshore movement of water at the Diablo Canyon current meter station (current meter station is located at the cross-hairs).

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5.0 RESULTS

This section is divided into a general overview of the data collected during the entrainment and impingement studies at DCPP, results of various modeling analyses on selected target taxa, and a summary of the taxa assessments.

5.1 Overview

The following is a general overview of the data collected and analyzed for the DCPP Section 316(b) Demonstration. Density estimates of target organisms were determined from plankton collections at the intake structure, in the Intake Cove, and in the study grid. The sections that follow present and discuss the results of analyses of individual target taxa chosen by the ETWG.

5.1.1 Entrainment

Entrainment is defined as the capture and inclusion of organisms in the cooling water of power plants (Schubel and Marcy 1978). Organisms become entrained in the cooling water system at DCPP once they pass through the traveling screens and cannot escape. Larval stages of fish and invertebrates are usually weak swimmers and generally enter the flow passively.

5.1.1.1 Weekly Abundance Estimates of Entrainment

There were 8,804 subsamples collected at the entrainment sampling location from October 23, 1996 through June 1999. In accordance with the ETWG agreement, larval fishes, *Cancer* crab larvae, and sea urchin larvae were identified and enumerated from a portion of these subsamples. Cancer crab megalops were processed from 5,524 subsamples, larval fish from 4,693 subsamples, and *Cancer* crab zoea and strongylocentrotid sea urchins from 636 subsamples during laboratory processing. The data for sea urchins were collected for informational purposes only and were not used in assessing entrainment effects per agreement with the ETWG.

Cancer *spp. crabs*

Zoeal stages of *Cancer antennarius*, *C. anthonyi*, *C. gracilis*, *C. productus*, and two *Cancer* spp. identifications left at the generic level were found in the entrainment subsamples. One unidentified *Cancer* species (sp. A) was confirmed late in the study to be *Cancer jordani* (P. Reilly, CDF&G, Monterey, CA). There were 96,295 zoeae identified and enumerated from the 636 entrainment subsamples processed. The mean density of each stage by species is presented in **Appendix H**. In general, zoeal stage 1 individuals were most abundant, and zoeal stage 5 individuals were least abundant.

The megalopal stage of *Cancer* spp. crabs was also identified and enumerated from entrainment subsamples. Megalops of *Cancer antennarius*, *C. anthonyi*, *C. gracilis*, *C. magister*, *C. productus*, and at least two generic identifications of *Cancer* spp. and *C. jordani* were recorded. In the 5,524 entrainment subsamples processed for megalops, 4,887 individuals were identified and enumerated. *Cancer antennarius* and *C. gracilis* megalops were more abundant than other *Cancer* species megalops (**Appendix H**).

The species chosen by the ETWG for analysis were *Cancer antennarius* and *C. gracilis* (Tenera 1997b). Additional information on the demography and results of the three assessment models for these two species is presented in **Section 5.2.1**.

Fishes

There were 98,593 larval fishes identified and enumerated from 4,693 entrainment subsamples (**Appendix H**). Of these, ~98% were placed into 178 different taxonomic categories ranging from ordinal to specific classifications (**Table 5.1-1**) while the remaining ~2% were placed into one of three categories: unidentified larval fish (0.4%), unidentified damaged larval fish (0.2%), or unidentified larval fish fragment (1%). 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 were those whose adults were generally found close to shore, in shallow water. One exception was the thirteenth most abundant taxon, the northern lampfish (*Stenobranchius leucopsarus*), whose adults are found to depths of 3,000 m (Miller and Lea 1972).

The ETWG chose 14 larval fish taxa (**Table 4-1**) for detailed assessment using the *FH*, *AEL*, and *ETM* approaches. Results of these analyses are presented separately in subsequent sections.

Sea Urchins

The ETWG agreed that only metamorphosing and recently metamorphosed juvenile sea urchins would be removed, identified, and enumerated from the entrainment subsamples, but that no assessment of entrainment effects would be made. Due to the small size of the echinopluteus larvae, it was assumed that many would be extruded through the net mesh used, resulting in an underestimate of their abundance.

A total of 5,966 sea urchin larvae was found in the 636 subsamples processed. The weekly mean density of *Strongylocentrotus fransicanus* and *S. purpuratus* individuals collected at the intake during the period October 1996 through June 1998 is presented in **Figure 5.1-1.** The density of *S. purpuratus* was greater than *S. fransicanus* during most surveys where individuals of both species were present. Both species were most abundant during the spring and summer, with highest densities of *S. fransicanus* present in March 1997 and *S. purpuratus* peak densities observed during June 1997. A greater density of both species occurred during 1997 than in 1998. Generally, only a few individuals were collected during the fall or winter. A summary of the data for all entrainment surveys is presented in **Appendix H**.

5.1.1.2 Monthly Abundance Estimates in the Study Grid

A total of 6,141 subsamples was collected during the monthly study grid surveys conducted from July 1997 through June 1999. As agreed with the ETWG, larval fish were identified and enumerated from 3,061 subsamples, megalops were identified and enumerated from 3,066 subsamples, and zoea and urchins were identified and enumerated from 128 subsamples during laboratory processing (**Appendix G**).

Cancer *spp. crabs*

The five *Cancer* crab zoeal stages were removed from the half of the subsamples preserved in formalin and collected during the March and April 1998 grid surveys. The ETWG decided not to require that zoea be removed from the remainder of the grid subsamples collected during this study.

Zoeal stages of *Cancer antennarius*, *C. gracilis*, *C. anthonyi*, *C. productus*, *C. jordani*, and at least one unidentified *Cancer* spp. were found in the subsamples from the two study grid surveys that were processed. A total of 27,270 zoea was removed and identified from the 128 grid samples processed. The mean density of each stage by species is presented in **Appendix H**. Generally, recently hatched stage 1 zoea were the most abundant and older stage 5 zoea were least abundant.

The megalops stage of *Cancer antennarius*, *C. gracilis*, *C. anthonyi*, *C. productus*, *C. magister*, *C. jordani*, and at least one unidentified *Cancer* spp. were found in the study grid samples. In the 3,066 grid samples processed for megalops, 6,599 individuals were removed and identified. The mean density of megalops by species for each survey is presented in **Appendix H**. *Cancer antennarius* and *C. gracilis* megalops were more abundant than the megalops of other *Cancer* species. The *ETM* assessment of these two species only uses megalops data because zoea were not processed from all of the paired study grid surveys per agreement by the ETWG.

Fishes

There were 43,785 larval fishes identified and enumerated from the 3,163 study grid subsamples that 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 grid subsamples were those whose adults were typically pelagic or subtidal; more intertidally or nearshore distributed species were found in lower abundance in the study grid.

Sea Urchins

Sea urchins were identified from the half of the subsamples preserved in formalin during only the March and April 1998 study grid surveys. The ETWG decided not to require that sea urchins be sorted from the remainder of the study grid subsamples and agreed that the data would not be assessed using a mathematical modeling approach (Tenera 1997b). There were 764 *Strongylocentrotus* spp. found in the 128 subsamples processed. The mean density of *S. purpuratus* in March and April of 1998 appeared to be greater than *S. fransicanus* during those months (**Appendix H**).

5.1.1.3 Weekly Intake Cove Surface Tows

There were 27,062 larval fishes identified and enumerated from 660 Intake Cove surface plankton tow samples collected between 1990 and 1998. The ETWG agreed that only the samples from the 7 mo period of December through June of each year would be processed and analyzed for each year because these months are generally the period of peak larval abundance for most species. The taxa present in these samples were similar to those found in the entrainment and study grid samples (**Appendix H**). As mentioned previously, these data were used to generate a nine-year average of larval abundance. This index calculated from this long-term average was then used to scale the estimates of annual entrainment for each taxon.

5.1.2 Impingement

There were 66 taxa representing 29 families of bony fishes, sharks, rays, and eels identified from impinged material collected during the 1985–86 DCPP study (**Table 5.1-1**). Some taxa, such as thornback rays, were collected in higher numbers and biomass at Unit 2 despite more frequent sampling at Unit 1. The densities of impinged fishes for Units 1 and 2 were compared during a 10 d period when pump and traveling screen operations between units were similar (PG&E 1988). The densities of impinged skates and rays were greater on the Unit 2 screens than at Unit 1 (PG&E 1988). Impingement rates for both units were similar for rockfishes and total bony fishes (PG&E 1988).

cooling water pump

The CWP did not operate continually during the impingement sampling (**Figure 5.1-2**). Equipment repairs on Unit 1 resulted in periods when pumps or traveling screens were not in full operation. Unit 2 was in the final stages of construction and testing during the DCPP Section 316(b) impingement study, which also limited the number of days when the pumps were operational and traveling screens were sampled. Although impingement sampling was scheduled to occur on the same day each week, it was adjusted several times due to operations at the intake. A total of 51 d was sampled for Unit 1, and 24 d were sampled for Unit 2. Unit 2 began operating on a more regular basis by October 1985. There were eight days of sampling when both Units 1 and 2 were fully operational for 24 hr (**Figure 5.1-2**).

Many fishes live within the intake forebays without becoming impinged. Diver observations and underwater video of the DCPP traveling screens during pre-operation (Behrens and Larsson 1979) and operation (PG&E 1988) have recorded both large and small fishes freely swimming throughout the forebays in front of the traveling screens. During the impingement study, divers in the Intake Cove on August 21, 1985, observed over 100 juvenile yellowtail/olive rockfishes in front of the Unit 1 bar racks (PG&E 1988), yet only one rockfish was collected during the two August impingement surveys.

The number of fishes, their combined weight, and the amount of debris collected during each of the 24-hour surveys were compared for samples collected at Units 1 and 2 (**Figure 5.1-3**). There does not appear to be a relationship between the number or weight of fishes and the amount of debris collected by Unit 1. The largest number of fishes was collected from Unit 1 during late June 1985 when the amount of debris collected was low. The total weight of fishes was low because most of the individuals were young-of-the-year rockfishes ca. 60–80 mm in length. At Unit 2, the greatest number and weight of fishes impinged did not consistently occur when the amount of debris was heaviest. The largest amount of debris was collected during late December 1985. During that period, the weight of the fishes in the collection was high, but the number of individuals was low.

A total of 1,314 *Cancer antennarius* was collected in impingement samples from February 1985 through March 1986. Ninety-seven percent of the crabs were juveniles, having an average weight of about 3 g. **Table 5.1-2** presents the standardized biomass (g/million m^3 of water entrained) of *Cancer* crab species impinged at DCPP. Among the various species of *Cancer* crabs, *C. antennarius* were impinged at the highest rate. The overall impingement rate of *Cancer* crabs was similar for both power plant units.

A report reviewing the 1985–86 DCPP impingement study was prepared and submitted to the ETWG (Tenera 1988). The ETWG determined that additional impingement studies were not required at DCPP based on this report that showed that the plant had generally low rates of impingement.

Table 5.1-1. Phylogenetic list of fishes collected during the Diablo Canyon Power Plant 316(b) study following primarily the AFS list of Common and Scientific Fish Names (Robins et al. 1991) and secondarily the classification adopted by Moser (1996).

Order Suborder Family	Genus & Species	Common Name
Clupeiformes		
Clupeidae	herrings	
	Etrumeus teres (DeKay 1842)	round herring
	Sardinops sagax (Jenyns 1842)	Pacific sardine
Engraulididae	anchovies	
	Anchoa compressa (Girard 1858)	deepbody anchovy
	Engraulis mordax Girard 1854	northern anchovy
Salmoniformes		
Argentinoidei		
Argentinidae	Argentine	
	Argentina sialis Gilbert 1890	Pacific argentine
Bathylagidae	blacksmelt and smoothtongues	
	Bathylagus ochotensis Schmidt 1938	popeye blacksmelt
	Bathylagus pacificus Gilbert 1890	Pacific blacksmelt
	Leuroglossus spp.	smooth tongues
	Leuroglossus stilbius Gilbert 1890	California smoothtongue
Microstomatidae	pencilfishes	
	Microstoma spp.	dusky pencilsmelt
Salmonoidei		
Osmeridae	smelts	
Stomiiformes		
Gonostomatidae	lightfishes	
	Cyclothone spp.	bristlemouths
	Cyclothone signata Garman 1899	showy bristlemouth
Sternoptychidae	hatchetfishes	
	Sternoptyx spp.	hatchetfish
Phosichthyidae	highseas lightfishes	
	Vinciguerria lucetia (Garman 1899)	Pacific lightfish
	Vinciguerria poweriae (Cocco 1838)	highseas lightfish
Chauliodontidae	dragonfishes	
	Chauliodus macouni Bean 1890	Pacific viperfish
Stomiidae	scaly dragonfishes	
	Stomias atriventer Garman 1899	blackbelly dragonfish
Aulopiformes		
Alepisaurioidei		
Synodontidae	lizardfishes	
	Synodus lucioceps (Ayres 1855)	California lizardfish
Paralepididae	barracudinas	
	Lestidiops ringens (Jordan & Gilbert 1880)	slender barracudina

(continued)

Table 5.1-1. (cont'd)

Order Suborder Family	Genus & Species	Common Name
Myctophiformes		
Myctophidae	lanternfishes	
	Ceratoscopelus townsendi (Eigenmann & Eigenmann 1889)	dogtooth lampfish
	Diaphus theta Eigenmann & Eigenmann 1890	California headlight fish
	Diogenichthys atlanticus (Tåning 1928)	longfin lanternfish
	Hygophum atratum (Garman 1899)	thickhead lanternfish
	Nannobrachium regalis (Gilbert 1892)	pinpoint lampfish
	Nannobrachium ritteri (Gilbert 1915)	broadfin lampfish
	Nannobrachium spp.	lanternfishes
	Protomyctophum crockeri (Bolin 1939)	California flashlightfish
	Stenobrachius leucopsarus (Eigenmann & Eigenmann 1890)	northern lampfish
	Symbolophorus californiensis (Eigenmann & Eigenmann 1889)	California lanternfish
	Tarletonbeania crenularis (Jordan & Gilbert 1880)	blue lanternfish
	Triphoturus mexicanus (Gilbert 1890)	Mexican lampfish
Gadiformes		
Gadidae	cods	
	Microgadus proximus (Girard 1854)	Pacific tomcod
Merlucciidae	hake	
	Merluccius spp.	hake
	Merluccius productus (Ayres 1855)	Pacific hake
Ophidiiformes		
Ophidiidae	cusk-eels	
	Chilara taylori (Girard 1858)	spotted cusk-eel
	Ophidion scrippsae (Hubbs 1916)	basketweave cusk-eel
Bythitidae	brotulas	
	Brosmophycis marginata (Ayres 1854)	red brotula
	Cataetyx rubirostris Gilbert 1890	rubynose brotula
Batrachoidiformes		
Batrachoididae	toadfishes	
	Porichthys notatus Girard 1854	plainfin midshipman
Gobiesociformes		
Gobiesocidae	clingfishes	
	Gobiesox spp.	clingfish
	Gobiesox maeandricus (Girard 1858)	northern clingfish
	Rimicola spp.	kelp clingfish
Atheriniformes		
Atherinidae	silversides	
	Atherinops affinis (Ayres 1860)	topsmelt
	Atherinopsis californiensis Girard 1854	jacksmelt
	Leuresthes tenuis (Ayres 1860)	California grunion
Scomberesocidae	sauries	
	Cololabis saira (Brevoort 1856)	Pacific saury

(continued)

Table 5.1-1. (cont'd)

Order Suborder Family	Genus & Species	Common Name
Lampridiformes		
Trachipteroidei		
Trachipteridae	ribbonfishes	
	Trachipterus altivelis Kner 1858	king-of-the-salmon
Beryciformes		
Stephanoberycoidei		
Melamphaidae	bigscales	
	Melamphaes spp.	bigscales
	Melamphaes parvus Ebeling 1962	little bigscale
	Poromitra crassiceps (Günther 1878)	crested bigscale
Gasterosteiformes		
Gasterosteoidei		
Gasterosteidae	sticklebacks	
	Aulorhynchus flavidus Gill 1861	tube-snout
Syngnathoidei		
Syngnathidae	pipefishes	
	Syngnathus spp.	pipefishes
	Syngnathus californiensis Storer 1845	kelp pipefish
	Syngnathus leptorhynchus Girard 1854	bay pipefish
Scorpaeniformes		
Scorpaenoidei		
Scorpaenidae	scorpion fishes, rockfishes, and thornyheads	
	Sebastes aurora (Gilbert 1890)	aurora rockfish
	Sebastes diploproa (Gilbert 1890)	splitnose rockfish
	Sebastes jordani (Gilbert 1896)	shortbelly rockfish
	Sebastes levis (Eigenmann & Eigenmann 1889)	cow cod
	Sebastes mystinus (Jordan & Gilbert 1881)	blue rockfish
	Sebastes paucispinis Ayres 1854	bocaccio
	Sebastes saxicola (Gilbert 1890)	stripetail rockfish
	Sebastes spp.	rockfishes
	Sebastes spp. 1 / V_DeP	rockfishes
	Sebastes spp. D	rockfishes
	Sebastes spp. V	rockfishes
	Sebastes spp. V_	rockfishes
	Sebastes spp. V_D	rockfishes
	Sebastes spp. V_D_	rockfishes
	Sebastes spp. V_De	rockfishes
	Sebastes spp. VD	rockfishes
	Sebastes spp. VP	rockfishes
	Sebastes VDp	rockfishes
	Sebastolobus spp.	thornyheads

(continued)

Table 5.1-1. (cont'd)

Order Suborder Family	Genus & Species	Common Name
Scorpaeniformes (cont'd)		
Hexagrammoidei		
Hexagrammidae	greenlings	
	Hexagrammos spp.	greenlings
	Ophiodon elongatus Girard 1854	lingcod
	Oxylebius pictus Gill 1862	painted greenling
	Zaniolepis spp.	combfishes
	Zaniolepis frenata Eigenmann & Eigenmann 1889	shortspine combfish
	Zaniolepis latipinnis Girard 1858	longspine combfish
Cottoidei		
Cottidae	sculpins	
	Artedius spp.	sculpin
	Artedius harringtoni (Starks 1896)	scalyhead sculpin
	Artedius lateralis (Girard 1854)	smoothhead sculpin
	Chitonotus / Icelinus	sculpin
	Chitonotus pugetensis (Steindachner 1876)	roughback sculpin
	Clinocottus spp.	sculpins
	Clinocottus analis (Girard 1858)	wooly sculpin
	Clinocottus embryum (Jordan & Starks 1895)	calico sculpin
	Enophrys spp.	sculpin
	Hemilepidotus spinosus (Ayres 1854)	brown Irish lord
	Icelinus spp.	sculpin
	Icelinus quadriseriatus (Lockington 1880)	yellowchin sculpin
	Leptocottus armatus Girard 1854	staghorn sculpin
	Nautichthys oculofasciatus (Girard 1858)	sailfin sculpin
	Oligocottus spp.	sculpin
	Oligocottus maculosus Girard 1856	tidepool sculpin
	Oligocottus snyderi Greeley 1898	fluffy sculpin
	Orthonopias triacis Starks & Mann 1911	snubnose sculpin
	Radulinus spp.	sculpin
	Ruscarius creaseri (Hubbs 1926)	roughcheek sculpin
	Ruscarius meanyi Jordan & Starks 1895	Puget Sound sculpin
	Scorpaenichthys marmoratus (Ayres 1854)	cabezon
	Synchirus gilli Bean 1890	manacled sculpin
Agonidae	poachers	
	Odontopyxis trispinosa Lockington 1880	pygmy poacher
	Stellerina xyosterna (Jordan & Gilbert 1880)	pricklebreast poacher
Cyclopteridae	snailfishes	
	Liparis spp.	snailfishes
Perciformes		
Percoidei		
Serranidae	sea basses	
	Paralabrax spp.	sand basses
	Paralabrax clathratus (Girard 1854)	kelp bass

(continued)

Table 5.1-1. (cont'd)

Order Suborder	Family	Genus & Species	Common Name
Perciformes (cont'd)			
Percoidei			
	Carangidae	jacks	
		Trachurus symmetricus (Ayres 1855)	jack mackerel
	Haemulidae	grunts	
		Xenistius californiensis (Steindachner 1875)	salema
	Sciaenidae	drums	
		Atractoscion nobilis (Ayres 1860)	white seabass
		Cheilotrema saturnum (Girard 1858)	black croaker
		Genyonemus lineatus (Ayres 1855)	white croaker
		Menticirrhus undulatus (Girard 1854)	California corbina
		Roncador stearnsi (Steindachner 1875)	spotfin croaker
		Seriphus politus Ayres 1860	queenfish
		Umbrina roncador Jordan & Gilbert 1882	yellowfin croaker
	Kyphosidae	sea chubs	
		Girella nigricans (Ayers 1860)	opaleye
	Pomacentridae	damselfishes	
		Chromis punctipinnis (Cooper 1863)	blacksmith
	Sphyraenidae	barracudas	
		Sphyraena argentea Girard 1854	Pacific barracuda
Labroidei			
	Labridae	wrasses	
		Oxyjulis californica (Günther 1861)	senorita
		Semicossyphus pulcher (Ayres 1854)	California sheephead
Zoarcoidei			
	Bathymasteridae	ronquils	
		Rathbunella spp.	ronquils
	Stichaeidae	pricklebacks	
		Cebidichthys violaceus (Girard 1854)	monkeyface eel
		Chirolophis nugator (Jordan & Williams 1895)	mosshead warbonnet
		Plectobranchus evides Gilbert 1890	bluebarred prickleback
	Pholididae	gunnels	
Blennioidei			
	Clinidae	clinid kelpfishes	
		Gibbonsia spp.	clinid kelpfishes
		Heterostichus rostratus Girard 1854	giant kelpfish
	Chaenopsidae	tube blennies	
		Neoclinus spp.	fringehead
	Blenniidae	combtooth blennies	
		Hypsoblennius spp.	blennies
		Hypsoblennius gilberti (Jordan 1882)	rockpool blenny
Icosteoidei			
	Icosteidae	ragfishes	
		Icosteus aenigmaticus Lockington 1880	ragfish

(continued)

Table 5.1-1. (cont'd)

Order Suborder Family	Genus & Species	Common Name
Perciformes (cont'd)		
Percoidei		
Trachinoidei		
Ammodytidae	sand lances	
	Ammodytes hexapterus Pallas 1814	Pacific sand lance
Gobioidei		
Gobiidae	gobies	
	Coryphopterus nicholsi (Bean 1882)	blackeye goby
	Eucyclogobius newberryi (Girard 1856)	tidewater goby
	Lepidogobius lepidus (Girard 1858)	bay goby
	Lythrypnus spp.	gobies
	Lythrypnus dalli (Gilbert 1890)	bluebanded goby
	Lythrypnus zebra (Gilbert 1890)	zebra goby
	Typhlogobius californiensis Steindachner 1879	blind goby
Scombroidei		
Scombridae	mackerels	
	Scomber japonicus Houttuyn 1782	Pacific mackerel
Stromateoidei		
Centrolophidae	medusafishes	
	Icichthys lockingtoni Jordan & Gilbert 1880	medusa fish
Tetragonuridae	squaretails	
	Tetragonurus cuvieri Risso 1810	smalleye squaretail
Stromateidae	butterfishes	
	Peprilus simillimus (Ayres 1860)	Pacific butterfish
Pleuronectiformes		
Pleuronectoidei		
Paralichthyidae	lefteye flounders	
	Citharichthys spp.	sanddab
	Citharichthys sordidus (Girard 1854)	Pacific sanddab
	Citharichthys stigmaeus Jordan & Gilbert 1882	speckled sanddab
	Hippoglossina stomata Eigenmann & Eigenmann 1890	bigmouth sole
	Paralichthys californicus (Ayres 1859)	California halibut
	Xystreurys liolepis Jordan & Gilbert 1880	fantail sole
Pleuronectidae	righteye flounders	
	Embassichthys bathybius (Gilbert 1890)	deepsea sole
	Eopsetta exilis (Jordan & Gilbert 1880)	slender sole
	Errex zachirus (Lockington 1879)	rex sole
	Hypsopsetta guttulata (Girard 1856)	diamond turbot
	Lepidopsetta bilineata (Ayres 1855)	rock sole
	Microstomus pacificus (Lockington 1879)	Dover sole
	Parophrys vetulus (Girard 1854)	English sole
	Platichthys stellatus (Pallas 1787)	starry flounder

(continued)

Table 5.1-1. (cont'd)

Order Suborder Family	Genus & Species	Common Name
Pleuronectiformes		
Pleuronectoidei		
Pleuronectidae	righteye flounders	
	Pleuronectes spp.	righteye flounders
	Pleuronectes isolepis (Lockington 1880)	butter sole
	Pleuronichthys spp.	turbots
	Pleuronichthys coenosus Girard 1854	c-o turbot
	Pleuronichthys ritteri Starks & Morris 1907	spotted turbot
	Pleuronichthys verticalis Jordan & Gilbert 1880	hornyhead turbot
	Psettichthys melanostictus Girard 1854	sand sole
Soleioidei		
Cynoglossidae	tonguefishes	
	Symphurus atricauda (Jordan & Gilbert 1880)	California tonguefish
Unidentified Larval Fishes:		
	Whole larval fishes	
	Damaged larval fishes	
	Larval fish fragments	

Table 5.1-2. Total abundance, weight (grams), and average biomass (grams/million m^3 flow) of impinged fishes at Diablo Canyon Power Plant (DCPP) during 1985–86. Abundance and weight are totals for the sampling periods; biomass is the number of grams per million m^3 of water flow during the collection periods.

x7?

Taxon	Scientific Name	Common Name	02/05/85 - 03/31/85 Abun.	Unit 1 Weight	Bio.	Unit 1 Abundance	Unit 2 Abundance	04/01/85 - 03/31/86 Unit 1 Weight	Unit 2 Weight	Unit 1 Biomass	Unit 2 Biomass
		Total All Fishes:	**11**	**1421**	**6.49**	**262**	**140**	**14441.3**	**12746.2**	**65.64**	**148.21**
BONY FISHES (Osteichthyes)											
Engraulididae	*Engraulis mordax*	northern anchovy					1		24.0		0.28
Batrachoididae	*Porichthys notatus*	plainfin midshipman				11	8	1056.6	658.2	4.80	7.65
Ophididae	*Chilara taylori*	spotted cusk-eel	1	23.8	0.11						
Gobiesocidae	*Gobiesox maeandricus*	northern clingfish				2	1	2.6	7.7	0.01	0.09
Atherinidae	*Atherinops affinis*	topsmelt				1	2	11.0	55.5	0.05	0.65
Gasterosteidae	*Aulorhynchus flavidus*	tubesnout				7	6	14.5	23.2	0.07	0.27
Syngnathidae	*Syngnathus californiensis*	kelp pipefish	1	4.8	0.02	8	4	143.5	40.2	0.65	0.47
	Syngnathus spp.	unidentified pipefish				2	1	6.9	5.3	0.03	0.06
	Syngnathidae unid.	unidentified pipefish				1		0.7		<0.01	
Scorpaenidae	*Sebastes atrovirens*	kelp rockfish				6	3	509.2	21.9	2.31	0.25
	Sebastes carnatus	gopher rockfish				2	1	8.4	2.0	0.04	0.02
	Sebastes flavidus	yellowtail rockfish				34	14	207.3	144.5	0.94	1.68
	Sebastes jordani	shortbelly rockfish				1		1.6		0.01	
	Sebastes melanops	black rockfish				1	1	3.9	3.4	0.02	0.04
	Sebastes mystinus	blue rockfish				5	3	326.4	152.1	1.48	1.77
	Sebastes paucispinis	boccacio				2		7.8		0.04	
	Sebastes rastrelliger	grass rockfish	1	70.2	0.32						
	Sebastes serranoides	olive rockfish				19	9	484.7	543.4	2.20	6.32
	Sebastes serranoides/flavidus	olive/yellowtail rockfish				7	3	46.3	16.2	0.21	0.19
	Sebastes spp.	unidentified rockfish				6	2	21.4	6.5	0.10	0.05
Hexagrammidae	*Hexagrammos decagrammus*	kelp greenling	1	143.1	0.65	1		5.9		0.03	
	Oxylebius pictus	painted greenling				1		29.3		0.13	
	Zaniolepis latipinnis	longspine combfish					1		29.0		0.34
Cottidae	*Artedius creaseri*	roughcheek sculpin				1		2.4		0.01	
	Artedius corallinus	coralline sculpin				8	2	74.4	6.2	0.34	0.07
	Artedius lateralis	smoothhead sculpin	1	5.8	0.03	7	3	37.7	12.8	0.17	0.15
	Artedius notosplitotus	bonyhead sculpin				1		2.3		.0.01	
	Cottidae	unidentified sculpin				1		3.5		0.02	
	Oligocottus maculosus	tidepool sculpin				1		3.1		0.01	
	Oligocottus rubellio	rosy sculpin				2		9.2		0.04	
	Orthonopias triacis	snubnose sculpin				5	4	29.2	22.7	0.13	0.26
	Scorpaenichthys marmoratus	cabezon				1		4.9		0.02	
Agonidae	*Agonopsis sterletus*	southern poacher					1		7.8		0.09
Cyclopteridae	*Liparis mucosus*	slimy snailfish				1		3.0		0.01	
Carangidae	*Trachurus symmetricus*	jack mackerel	1	273.4	1.24	1	3	329.4	386.8	1.50	4.50
Sciaenidae	*Seriphus politus*	queenfish				2	7	8.2	40.5	0.04	0.47
Embiotocidae	*Amphistichus argenteus*	barred surfperch				2		10.0		0.05	
	Brachyistius frenatus	kelp surfperch				6	3	79.8	29.1	0.36	0.34
	Cymatogaster aggregata	shiner surfperch				17		212.3		0.97	
	Embiotoca jacksoni	black surfperch	1	296.3	1.35	2		517.4		2.35	
	Embiotoca lateralis	striped surfperch			0.03	3		190.7		0.87	
	Embiotocidae	unidentified surfperch	1	6.4	0.03	3		45.3		0.21	
	Hyperprosopon argenteum	walleye surfperch				2	3	23.0	103.8	0.10	1.21
	Micrometrus minimus	dwarf surfperch				1	1	7.7	4.1	0.04	0.05
	Phanerodon furcatus	white surfperch				1		3.6		0.02	
Labridae	*Oxyjulis californica*	senorita					1		16.0		0.19
Pomacentridae	*Chromis punctipinnis*	blacksmith				1	2	109.5	66.7	0.50	0.78
Clinidae	*Gibbonsia erythra*	scarlet kelpfish					2		12.8		0.15
	Gibbonsia metzi	striped kelpfish				8	1	145.0	4.7	0.66	0.05
	Gibbonsia montereyensis	crevice kelpfish				4		16.4		0.07	
	Gibbonsia spp.	unidentified kelpfish					2		55.5		0.65
Stichaeidae	*Anoplarchus purpurescens*	high cockscomb				2	2	26.5	23.4	0.12	0.27
Pholidae	*Ulvicola sanctaerosae*	kelp gunnel				1		3.3		0.02	
	Xerepes ficorum	rockweed gunnel				1	1	8.5	4.9	0.04	0.06
Scombridae	*Scomber japonicus*	Pacific mackerel				2	2	801.7	733.4	3.64	8.53
Cynoglossidae	*Symphurus atricauda*	California tonguefish				2		6.8		0.03	
Paralichthyidae	*Citharichthys stigmaeus*	speckled sanddab	1	20.7	0.09	4		4.7		0.02	
	Xystreurys liolepis	fantail sole				9		869.2		3.95	
Pleuronectidae	Pleuronectidae unid.	unidentified turbot or sole				1		1.7		0.01	
	Microstomus pacificus	dover sole				1		32.8		0.15	
	Pleuronichthys coenosus	C-O turbot				1		74.8		0.34	
		Total Fishes:	**9**	**844.5**	**3.87**	**223**	**100**	**6586.0**	**3261.8**	**29.94**	**37.93**
Sharks and Rays (Chondrichthyes)											
Rajidae	*Raja binoculata*	big skate				3	2	86.0	73.6	0.39	0.86
Torpedinidae	*Torpedo californica*	Pacific electric ray	1	374.4	1.70	10	4	1830.2	693.2	8.32	8.06
Platyrhinidae	*Platyrhinoidis triseriata*	thornback ray	1	202.1	0.92	23	32	4687.4	7475.9	21.31	86.93
Dasyatididae	*Urolophus halleri*	round stingray					1		407.9		4.74
Chimeridae	*Hydrolagus collei*	spotted ratfish				3	1	1251.7	833.8	5.69	9.70
		Total Sharks and Rays:	**2**	**576.5**	**2.62**	**39**	**40**	**7855.3**	**9484.4**	**35.71**	**110.28**

Table 5.1-3. Biomass (grams/million m^3 flow) of impinged *Cancer* spp. crabs at DCPP from April 1985 through March 1986.

Species	Unit 1	Unit 2
Cancer antennarius	27.43	21.69
Cancer anthonyi	0.02	0.88
Cancer productus	0.47	1.97
Cancer jordani	0.42	0.27
Cancer magister	0.01	-
Cancer spp.	0.03	0.01
Cancer gracilis	0.29	-
TOTAL	28.67	24.82

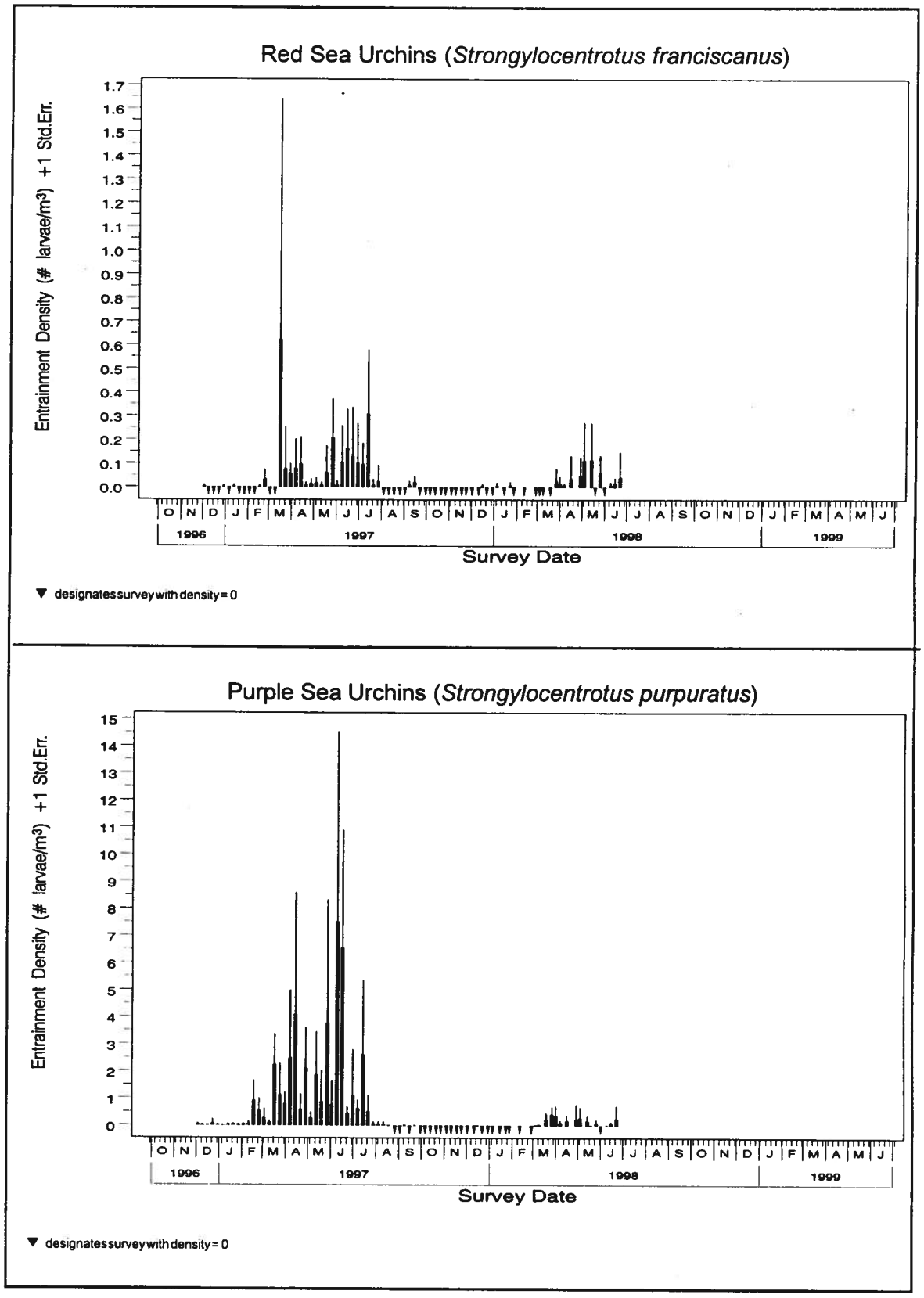

Figure 5.1-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake. Y-axis scale varies between graphs.

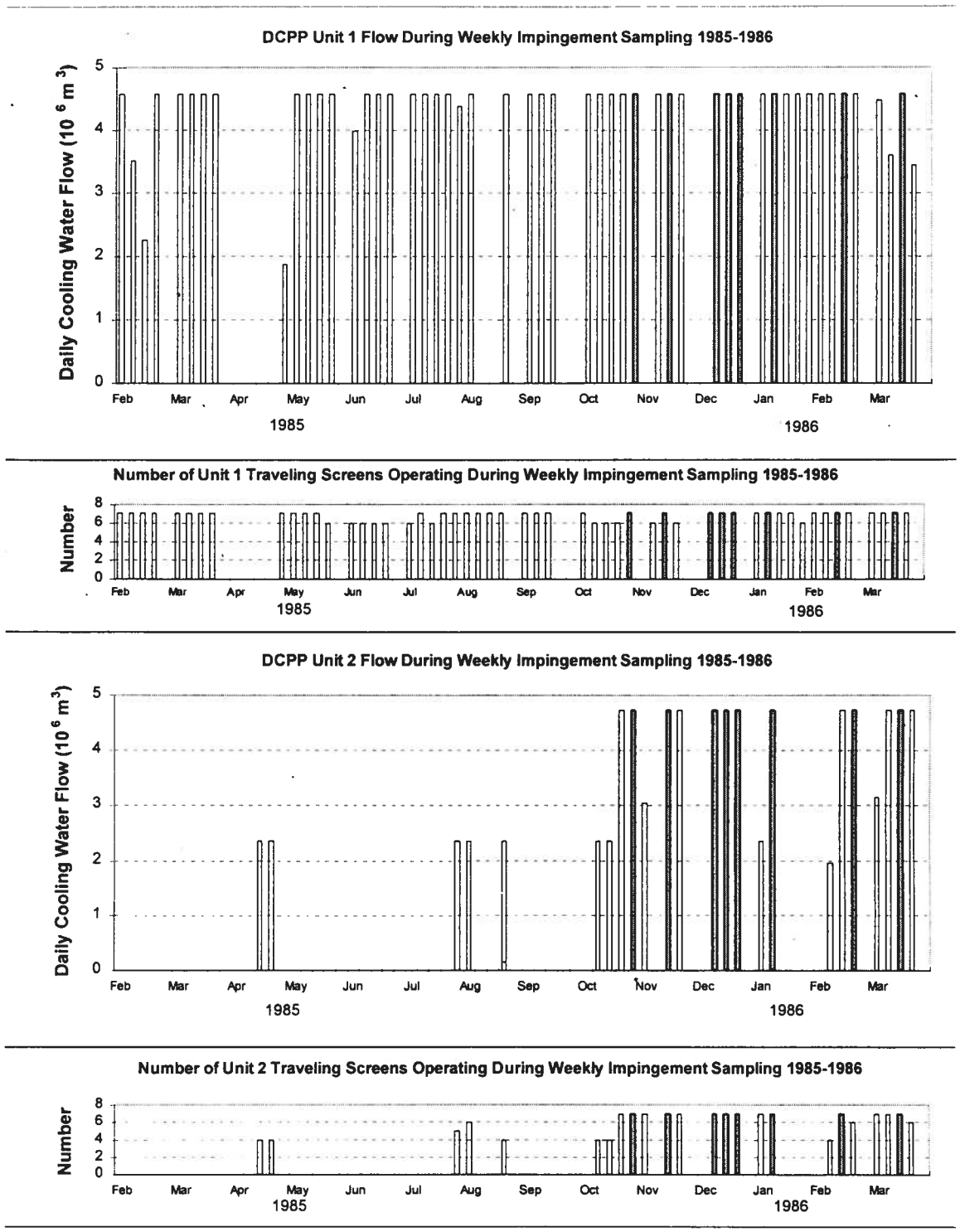

Figure 5.1-2. DCPP intake operational status: cooling water pump (CWP) daily water flow and number of traveling screens operating on impingement sampling days. **Bold** bars indicate days when all pumps and traveling screens operated concurrently. Each unit has two pumps and seven traveling screens (six for the CWP and one for the auxiliary seawater system).

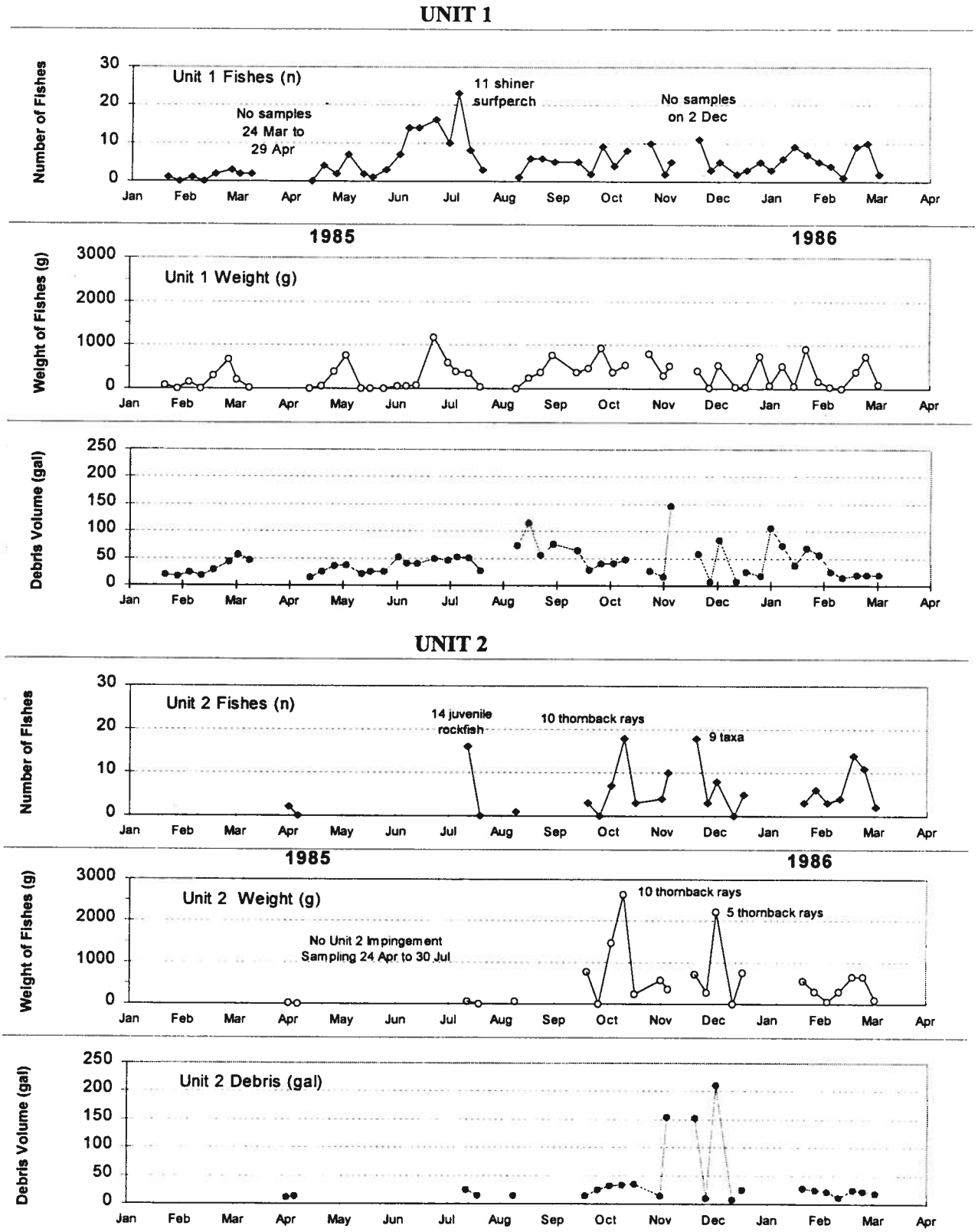

Figure 5.1-3 Comparison of fish abundance (n) and weight (g), and seaweed debris volume (gallons) collected in DCPP Unit 1 and 2 impingement samples.

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5.2 Assessment of Effects on Target Taxa

The effects of larval entrainment mortality caused by the DCPP cooling water intake system were assessed for 16 target taxa. In the following sections, life history and distribution information describing the adult and early life stages of these taxa were summarized to give context to the subsequent estimates of entrainment effects. The data generated from the various sampling programs were then used to estimate effects of larval entrainment mortality using demographic data for hindcasting (*FH*) or prediction (*AEL*) of adult losses, or were translated directly into proportional annual losses to the local larval population (P_M).

5.2.1 *Assessment of Target Taxa in the Crab Genus* Cancer

Crabs of the genus *Cancer* are widely distributed in coastal waters of the west coast of North America (Nations 1975). They occur in intertidal and shallow subtidal habitats on both rock and sand substrate. Of the nine species known to occur in the northeast Pacific, four species contribute to economically significant fisheries. Dungeness crab (*Cancer magister*) has the highest economic value among these, and three species of rock crabs (*C. anthonyi, C. antennarius,* and *C. productus*) comprise the remainder of the catches. Seven species of *Cancer* larvae have been collected in the DCPP entrainment study and are listed here (in order of relative abundance):

Common name	Scientific Name
Brown rock crab	*C. antennarius*
Slender crab	*C. gracilis*
Hairy rock crab	*C. jordani*
Red rock crab	*C. productus*
Yellow crab	*C. anthonyi*
Dungeness crab	*C. magister*
unknown sp. A	*Cancer* spp.

The two most abundant species (brown rock crab and slender crab) were selected for a detailed impact assessment in this report. The other five species were not consistently abundant enough in the study to be analyzed.

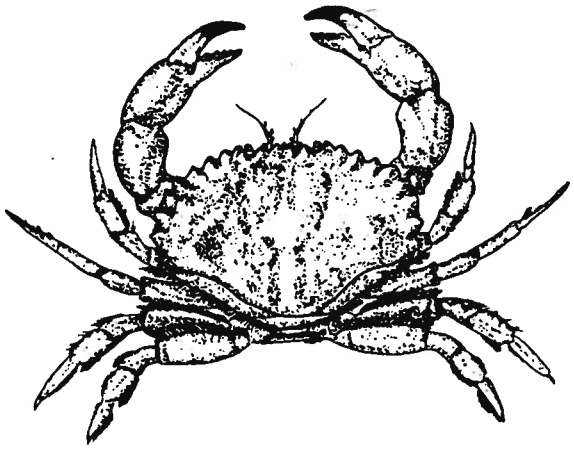
All species of *Cancer* crabs share certain fundamental life history traits. Eggs are extruded from the ovaries through an oviduct and are carried in a sponge-like mass beneath the abdominal flap of the adult female. After a development period of several weeks, the eggs hatch and a pre-zoea larva emerges,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

Each species in the genus has characteristic differences in distribution, preferred habitat, growth rates, and demographic parameters. For example, brown rock crab is a relatively large species (carapace width >200 mm) that lives primarily on sand and mud substrates in estuarine and coastal shelf areas. Slender crab is a smaller species (carapace width >130 mm) associated with mixed rock-sand substrates in shallow outer coast habitats. Maximum clutch sizes in *Cancer* crabs can range from as many as 5,000,000 eggs in *C. anthonyi* to approximately 50,000 in *C. oregonensis,* one of the smaller species (Hines 1991). These types of differences imply that specific information on life history parameters cannot readily be generalized among *Cancer* species.

Rock crabs are fished along the entire California coast (Leet et al. 1992). Three species are harvested commercially: brown rock crab, red rock crab, and yellow crab. There is no commercial fishery for the slender crab. The rock crab fishery is most important in southern California (from Morro Bay south), which produces a majority of the landings, and of lesser

importance in northern areas of California where a fishery for the more desirable Dungeness crab takes place. Recreational crabbing is popular in many areas and is often conducted in conjunction with other fishing activities. The commercial harvest has been difficult to assess on a species-by-species basis because the fishery statistics are combined into the general "rock crab" category. Rock crab landings in California in 1990 were 818 MT, including the landings of crab claws only that were converted to whole weight (Leet et al. 1992). Rock crab landings from five ports near the Monterey Bay National Marine Sanctuary averaged 92 MT/yr from 1980–1995 (Starr et al. 1998).

5.2.1.1 Brown rock crab (*Cancer antennarius*)



Cancer antennarius Stimpson 1856; brown rock crab; carapace width to 15.5 cm (6.1 in); Queen Charlotte Sound, British Columbia to Cabo San Lucas, Baja California; intertidal to >100 m (328 ft); mottled dark brown dorsally with red spotting over a white background ventrally (Jensen 1995; Carroll and Winn 1989).

The brown rock crab primarily inhabits rocky shores and rocky subtidal reefs but may bury in coarse to silty sands adjacent to preferred habitat. Ovigerous brown rock crabs have been observed buried in sand at the base of rocks in shallow water and are found more commonly in water less than 18 m (59 ft) deep in southern California. Brown rock crab females can extrude between approximately 156,000 and 5 million eggs per batch (Hines 1991; **Table 5.2.1-1**). Females on average produce a single batch per year; however, due to occasional multiple spawnings, the average number of batches per year may be greater than one (Carroll 1982).

Eggs require a development time of approximately 7–8 wk from extrusion to hatching (Carroll 1982). Larval development in the brown rock crab was described by Roesijadi (1976). Eggs hatch into pre-zoea larvae that molt to first stage zoea in less than 1 h. Average larval development time (from hatching through completion of the fifth stage) was 36 d at 13.8 °C. Although some crabs molted to the megalops stage, none molted to the first crab instar stage, so the actual duration of the megalops stage is unknown. A reasonable estimate can be derived from studies of slender crab by Ally (1975), who found an average duration of megalops stage of 14.6 d. Therefore, the estimated length of time from hatching to settling for brown rock crab is approximately 50 d.

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

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

There are no published estimates of brown rock crab larval mortality. However, data from the abundance of zoea and megalops in the DCPP entrainment subsamples (**Appendix I**) can be used to estimate mortality between stages. First stage zoea of the taxa *Cancer antennarius, C. anthonyi,* and *C. gracilis* (combined because of uncertainties in identification) were substantially

more abundant, on average, than all other stages combined. The proportions of each species of zoea stage 1 were derived by using the proportions of each species in zoea stage 2 that could be identified to species. Larval mortality of brown rock crab was estimated from the entrainment data by comparing adjusted abundances of zoea stage 1 to abundance of megalops. Intermediate stages 2, 3, 4, and 5 were not used in the estimate because they were significantly affected by offshore transport out of the study grid and would, therefore, be under-represented in the subsamples (Paul Reilly, California Department of Fish and Game, pers. comm.). Megalops of all species were more abundant than zoea stage 5 in the entrainment subsamples, indicating that many of the developing megalops were being transported back toward shore.

An instantaneous larval mortality of 0.158/d was estimated by fitting an exponential curve to the estimated numbers of entrained zoea stage 1 (December 1996 through June 1998) and megalops (mid January 1997 through mid August 1998) and using 38 d as the time between stages (i.e., 5 d and 43.3 d, respectively). This procedure is fully described in the following section concerning fecundity hindcasting. Because brown rock crab megalops were more abundant close to shore than slender crab (Graham 1989), the mortality rate developed for brown rock crab is also applied to slender crab larvae (**Section 5.2.1.2**) that also have a duration of 38 d between zoea and megalops stages.

Estimated weight at age was used in interpreting adult equivalent losses to the population. Brown rock crabs mature at an age of about 18 mo post-settlement with a size of approximately 60 mm carapace width and a weight of 73 g (Carroll 1982). Faster growth rates may occur in highly productive environments such as on the supporting members of offshore oil platforms (Dan Dugan, Tenera Environmental, pers. comm.). A growth curve for brown rock crab was estimated based on asymptotic carapace width and weight (155 mm and 1,050 g for males, 145 mm and 683 g for females), and a growth rate between instars from tag recovery data (ranging from 7–26 % with smaller crabs growing at the highest rate), for a maximum of 12–13 post-larval instars. Brown rock crabs can probably live to a maximum age of about 6 yr. Size at recruitment to the fishery is approximately 125 mm carapace width at age 4 yr for males and 4.5 yr for females.

Summary of Field Collections

Brown rock crab zoeae were removed from subsamples collected at the DCPP intake structure from December 1996 through June 1998 while megalopae were removed for the period October 1996 through June 1999. Zoeae were removed from only some of the subsamples collected during two grid surveys in 1998. Megalopae were removed from subsamples from the study grid from July 1997 until June 1999. Estimated density (numbers per volume of water filtered) of zoeae and megalopae from the processed subsamples are presented in **Appendix H**. The estimated weekly number of zoeae and megalopae entrained (**Appendix I**) was calculated by multiplying the estimated larval density by the volume of water drawn through the CWS. Abundance of both zoea and megalops was greatest in the spring although zoea 1 began to appear in winter months (**Figure 5.2.1-1**).

Larval densities were estimated from bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999; for a daily sampling event there were generally 32 to 64 subsamples processed for estimating megalops densities and 8 subsamples processed for estimating zoea densities. Of a possible 5,524 entrainment subsamples processed for megalopae, 665 (12%) were positive for brown rock crab. Megalops densities in entrainment subsamples were highest during 1997 and lowest in 1999 (**Figure 5.2.1-1**). Of 636 subsamples processed for zoea, the percentages of subsamples by stage were: 83% zoea stage 1, 26% zoea 2, 14% zoea 3, 6 % zoea 4 and 2% zoea 5.

Megalops densities in the DCPP study grid were highest in May and June of 1998 and 1999, but much higher in 1998 than in 1999 (**Figure 5.2.1-2**). Of 3,168 bongo net subsamples collected in the grid, 13% contained brown rock crab megalopae.

Estimating Total Annual Entrainment

Estimates of annual entrainment ($\widehat{E_T}$) of the five zoea and one megalops stages of brown rock crab during the two analysis periods (December 1, 1996–November 30, 1997 and July 1, 1997–June 30, 1998) are presented in **Table 5.2.1-2**. The date of the first analysis period for zoea was adjusted from October 1996 through September 1997 to December 1996 through November 1997

because the entrainment subsamples collected during October and November 1996 were not processed for zoea. Generally there were fewer individuals of each older stage. Although estimates of zoea stages 1–4 abundances were approximately equal between the two analysis periods, the greater abundance of zoea stage 5 during 1998 for Analysis Period 3 ($\widehat{E_T}$ = 111,000,000) did not result in an increased megalops abundances for that period. Estimated megalops abundance was actually greater during 1997, Analysis Period 1 ($\widehat{E_T}$ = 56,500,000) than during 1998, Analysis Period 3 ($\widehat{E_T}$ = 24,500,000). The estimates of annually-entrained brown rock crab were not adjusted to a long-term average because crab larvae were not enumerated in the Intake Cove surface plankton tow samples.

*Fecundity Hindcasting (*FH*)*

The *FH* method requires age-specific mortality and fecundity to assess entrainment effects. A single value for instantaneous mortality, Z, was estimated for the two analysis periods by fitting an exponential curve to the zoea 1 and megalops entrainment estimates. The survival rate to 43.3 d post hatching was estimated for all larval stages using an exponential survival model:

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

where the estimated entrainment of zoea stage 1 (N_0 at time t_0 =5) and megalops (N_t at time t=43.3) stages were used to estimate the rate Z. Zoea 1 abundance was estimated as the fraction of brown rock crab stage 2 larvae times the number of the unidentified zoea 1. The instantaneous mortality rate, Z, can then be solved from the two larval stage abundances (zoea 1, December 1996 through June 1998 and megalops, mid January 1997 through mid August 1998) as

$$Z = \frac{\ln\left(\frac{34{,}600{,}000{,}000}{80{,}100{,}000}\right)}{43.3-5} = 0.158 \text{ per day}.$$

An estimated survival through larval stages can then be calculated as $\hat{S}_{larvae} = e^{(-0.158)t}$, where t is reported in **Table 5.2.1-3** as mean cumulative duration. Cumulative survival estimates to stage midpoint were

total days = 56

.158
56
948
7900
8.848

1) zoea 1 0.454
2) zoea 2 0.128
3) zoea 3 0.0497
4) zoea 4 0.0193
5) zoea 5 0.00637
6) megalops 0.00107 .

The *FH* approach combines larval entrainment losses with survival and adult fecundity to hindcast the numbers of adult females effectively removed from the reproductively active population. An estimate of mean annual fecundity is 1,756,450 eggs. The number of reproductive years was estimated by halving the range between age of maturity of 1.5 yr and longevity of 4 yr. A longevity of 4 yr was chosen because of low abundance of older animals found in a trapping study (Jay Carroll, Tenera, pers. comm.). Coefficients of variation of survival, fecundity, and ages of maturation and longevity are not well known and are therefore reasonably assumed to be 100% with a lognormal error structure.

The number of female brown rock crabs estimated to produce the entrained larvae was calculated for the five zoea and one megalops stage for the two. All larval stages were hindcast and the estimated numbers of females and variances of estimates were added to compute an estimate of total number of brown crab females and the variance of the estimate.

The estimated FH values for zoea and megalops stages for the period December 1, 1996 through November 30, 1997 from **Table 5.2.1-2** were

$$\widehat{FH}_1 = 17{,}500 \text{ females with } \widehat{SE}(\widehat{FH}_1) = 39{,}000$$
$$\widehat{FH}_2 = 16{,}300 \text{ females with } \widehat{SE}(\widehat{FH}_2) = 38{,}100$$
$$\widehat{FH}_3 = 43{,}300 \text{ females with } \widehat{SE}(\widehat{FH}_3) = 104{,}000$$
$$\widehat{FH}_4 = 15{,}700 \text{ females with } \widehat{SE}(\widehat{FH}_4) = 37{,}400$$
$$\widehat{FH}_5 = 239 \text{ females with } \widehat{SE}(\widehat{FH}_5) = 626$$
$$\widehat{FH}_{Megs} = 24{,}100 \text{ females with } \widehat{SE}(\widehat{FH}_{Megs}) = 53{,}400$$

The total estimated number of breeding females needed to produce the total number of larvae entrained during the period was calculated by summing the $\widehat{FH}_i$ over all stages:

$$\widehat{FH} = 117{,}000 \text{ females}$$

with $\widehat{SE}(\widehat{FH}) = \sqrt{\sum_{i=1}^{6} \widehat{SE}_i \left(\widehat{FH}_i\right)^2} = 134{,}000$. An approximate 90% confidence interval for *FH*, based on a lognormal error structure, is estimated as 17,700 to 773,000 females.

The estimated *FH* values for zoea and megalops stages for the period July 1, 1997 through June 30, 1998 from **Table 5.2.1-2** were

$$\widehat{FH}_1 = 18{,}600 \text{ females with } \widehat{SE}(\widehat{FH}_1) = 41{,}700$$
$$\widehat{FH}_2 = 13{,}400 \text{ females with } \widehat{SE}(\widehat{FH}_2) = 30{,}500$$
$$\widehat{FH}_3 = 22{,}100 \text{ females with } \widehat{SE}(\widehat{FH}_3) = 51{,}700$$
$$\widehat{FH}_4 = 18{,}500 \text{ females with } \widehat{SE}(\widehat{FH}_4) = 46{,}300$$
$$\widehat{FH}_5 = 7{,}950 \text{ females with } \widehat{SE}(\widehat{FH}_5) = 20{,}100$$
$$\widehat{FH}_{Megs} = 10{,}400 \text{ females with } \widehat{SE}(\widehat{FH}_{Megs}) = 23{,}100$$

The total estimated number of breeding females needed to produce the total number of larvae entrained during this period was calculated by summing the stage specific estimates as

$$\widehat{FH} = 91{,}000 \text{ females}$$

with $\widehat{SE}(\widehat{FH}) = \sqrt{\sum_{i=1}^{6} \widehat{SE}_i \left(\widehat{FH}_i\right)^2} = 91{,}800$. An approximate 90% confidence interval for *FH*, based on a lognormal error structure, is estimated as 17,300 to 478,000 females.

Sensitivity Analysis

In addition to calculating a confidence interval, a sensitivity analysis **(Table 5.2.1-2)** for all larval stages was performed in which $\widehat{FH}$ was recomputed by varying the input parameter of estimated annual entrainment, $\widehat{E_T}$, by $\pm 1.645 \widehat{SE}\left(\widehat{E_T}\right)$ and survival, $\hat{S}$, by using a multiplicative error structure (i.e. $\hat{S} e^{\pm 1.645 CV(\hat{S})}$) and using $CV\left(\hat{S}\right) = 1.0$. The range of survival estimates for all stages produced the same relative changes in $\widehat{FH}$ as expected based upon a constant multiplier. However, $\widehat{FH}$ was less sensitive to entrainment than survival variation. This difference was greatest for the megalops stage and can be explained by the use of sampling variance for

estimating entrainment variability for all stages with *CV*s of close to or less than 100%. Therefore, improvement in the precision of $\widehat{FH}$ can result from improved estimates of the larval stages' survival.

Females that reach reproductive size are not subject to fishery capture due to their small size. Therefore, the $\widehat{FH}$ should be extrapolated to fishery size because it will provide more meaningful results for subsequent comparisons. Adult survival can be estimated for brown rock crab as 0.088 yr^{-1} (based on data in Carroll 1982). Hankin et al. (1989) estimated adult survival for dungeness crab exceeding 155 mm as 0.11 yr^{-1}. Therefore, the estimates of *FH* (117,000 and 90,900 female crabs) can be reasonably extended to full fishery recruitment from the representative female age of 2.3 yr (one third between 1.5 and 4 yr) by applying a survival of 0.1 for an additional year. This would result in a population-wide loss of 11,700 and 9,100 as the two yearly estimates of fishery recruit-size females lost due to entrainment.

*Adult Equivalent Loss (*AEL*)*

The *AEL* approach uses estimates of the abundance of entrained organisms to project the loss of equivalent numbers of adults based on mortality schedules and age at recruitment. Survival rates from entrainment of megalopae to recruitment into fishery are not available. However, assuming a 50:50 sex ratio, $\widehat{AEL}$ and $\widehat{FH}$ can be compared as $\widehat{AEL} \equiv 2\widehat{FH} = 234{,}000$ and 182,000 adults, with an associated standard error of $\widehat{SE}(\widehat{AEL}) = 2\widehat{SE}(\widehat{FH}) = 269{,}000$ and 184,000.

To reiterate, females that reach reproductive size are not subject to fishery capture due to their small size. Therefore, the $\widehat{AEL}$ also should be extrapolated to fishery size. Adult survival can be estimated for brown rock crab as 0.088 yr^{-1} (Carroll 1982). Hankin et al. (1989) estimated adult survival for dungeness crab exceeding 155 mm as 0.11 yr^{-1}. Therefore, the estimate of *AEL* (234,000 and 182,000 crabs) can be reasonably extended to full fishery recruitment from the representative female age of 2.3 yr (one third between 1.5 and 4 yr) by applying a survival of 0.1 for an additional year. This would result in a population-wide loss of 23,400 and 18,200 as the two yearly estimates of fishery recruits lost due to entrainment.

Empirical Transport Model (ETM)

The *ETM* assessment requires estimates of larval stage duration and larval density in entrainment and grid subsamples. The crab data were collected for only the megalops larval stage in the grid subsamples and will be compared to megalops stage larvae in the entrainment subsamples.

Two entrainment mortality estimates are presented: for July 1, 1997 through June 30, 1998 and for July 1, 1998 through June 30, 1999. Brown rock crab larvae may be susceptible to entrainment by the plant for approximately 43 d, the average age of a megalop. Although the duration is more than one month, each monthly survey period will be assumed to represent discrete groups of larvae, permitting two independent weighted estimates of total entrainment mortality from 24 subsamples. Total annual entrainment mortality (P_M) is estimated from the values of PE_i and f_i for each survey period (**Table 5.2.1-4**), based on larval densities from entrainment (**Figure 5.2.1-1**) and grid surveys (**Figure 5.2.1-2**).

The proportion of the population within the nearshore study grid was based on a linear extrapolation of densities in the grid in the offshore direction and displacement in the alongshore direction. **Table 5.2.1-4** shows monthly estimates of entrainment proportions, of survey period to survey grid abundance ($\widehat{PE_i}$; based upon paired surveys) and of survey period to yearly entrainment ($\hat{f}_i$; based upon weekly subsamples), for brown rock crab. $\widehat{PE_i}$ from the two years sampled ranged from a minimum of zero to a maximum of 0.0254±0.0219 ($\pm 1\widehat{SE}(\widehat{PE_i})$) in October 1997. Because estimated per period survival is weighted to form an annual mortality estimate, the largest values of $\hat{f}_i$ (and of associated $\widehat{PE_i}$) have the largest effects on the estimate (July and June 1998). No larvae were collected at either the DCPP intake or the study grid when both $\widehat{PE_i}$ and the annual proportion of larvae hatched ($\hat{f}_i$) in the *i*th survey period were equal to zero (September 1998, February 1999, and March 1999). When $\widehat{PE_i}$ =0 and $\hat{f}_i$>0, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hr study grid survey. Low values of $\hat{f}_i$ indicate periods of the year when brown rock crab megalops are least abundant in weekly entrainment subsamples. While June and July 1998 entrainment fractions were highest overall, May and June 1998 densities were highest in the study grid (**Figure 5.2.1-2**).

Total annual entrainment mortality from the *ETM* is estimated as $\widehat{P_M}$ = 0.0000186 for the period July 1, 1997 through June 30, 1998 and $\widehat{P_M}$ = 0.000146 for the period July 1, 1998 through June 30, 1999 with associated standard errors of $\widehat{SE}(\widehat{P_M})$ = 0.2327 and 0.0325, respectively (**Figure 5.2.1-3**). The sensitivity analysis considers the effect of varying values of $\widehat{P_S}$ on the outcome of the *ETM* calculations shown in **Figure 5.2.1-3** for the two periods. The values of $\widehat{P_M}$ resulting from calculations using only the alongshore displacement were higher (0.00563 and 0.00652) than those reported for the offshore extrapolated densities (0.0000186 and 0.000146). Values of $\widehat{P_S}$ ranged from 0.000386 to 0.00239 for $\widehat{P_M}$ extrapolated offshore and ranged from 0.11 to 0.13 for alongshore only.

Interpretation of Assessment Results

The *ETM* estimate of entrainment mortality of megalopae can be assumed to apply to all stages of larval mortality at DCPP and, by extension, can be applied to the adult crab population or to harvest assuming no compensation. Estimates of stock size and density that could be used to convert $\widehat{P_M}$ into an estimate of adult equivalent loss (assuming no compensatory mortality) are not available. Although $\widehat{P_M}$ was estimated using both alongshore current displacement and density extrapolation offshore, the estimate using alongshore currents was used in assessing fishery value because it was more representative of the areas actively fished for rock crab. The rock crab fishery does not extend into the offshore areas included in the extrapolation.

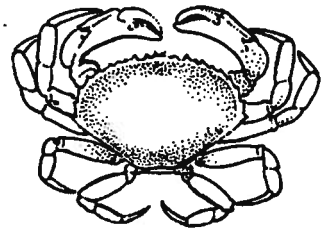
Brown rock crabs have both commercial and recreational fishery value. An area that contains the population at risk, predicted by using the average alongshore displacements of 140 km to 160 km (**Figure 5.2.1-3**), is limited to the region where crabs are landed at the ports of Morro Bay and Avila. From 1975 to 1998 an average of 135,200 kg of rock crabs were landed at Morro Bay and Avila (**Figure 5.2.1-4**). An ex-vessel price of $2.32 per kg and average weight of 0.34 and 0.45 kg for females and males, respectively, is reported for the Morro Bay area catch (Deborah Johnston, California Department of Fish and Game, pers. comm.). The estimated mortality rates from the *ETM* using alongshore extension ($\widehat{P_M}$ = 0.00563 and 0.00652) are applied to the average landings. Assuming that the proportional effect of entrainment mortality on the larval

population also acts on the adult population, then the average yearly historical Morro Bay area catch is affected by the same proportion, which equates to 760 and 880 kg of crabs for the two years, respectively. In terms of monetary value, the yearly valuation would be $1,760 and $2,040 for the two years.

The *AEL* model estimates are assigned to population-wide losses but can be compared to the fishery using the ex-vessel price and average weight of 0.4 kg. Crabs that reach reproductive size are not subject to fishery capture due to their small size. Therefore, the number of estimated adults lost to the population is extrapolated to fishery size, and the resulting two yearly estimates of 23,400 and 18,200 crabs lost to the population per yr represent $21,700 and $16,900, 7% and 5% of individuals in an average annual catch (based on an average annual catch of 135,200 kg and 0.4 kg per crab). These values overestimate fishery losses since proportional reductions of numbers due to catchability, trapping selectivity, and fishing effort are not considered. In an experimental trapping study by Carroll (1982), tag returns on over 9,000 brown rock crab along the Diablo Canyon coastline averaged 6.3%. Further, fewer than 0.1% of the tagged crabs were recovered in the local commercial fishery over the course of the 5-year study. Although there were no data available on commercial fishing effort during that period, a conservative estimate that 10% of the fishery-sized crabs in the population were actually landed per year would reduce the valuation of the *FH* and *AEL* losses by one order of magnitude.

The *ETM*-based loss assessments of $1,760 to $2,040 use a more appropriate valuation method because it considers catch as a proportion of the population and is also conservative because it is calculated using the larger values of $\widehat{P_M}$ (i.e., based on an alongshore-only translation of the survey grid numbers). Because of the low estimated population mortality by power plant entrainment (alongshore and offshore-based $\widehat{P_M}$ = 0.0000186 and 0.000146), effects on the brown rock crab population are probably low.

5.2.1.2 Slender crab (*Cancer gracilis*)



Cancer gracilis Dana 1852; slender crab; carapace width to 11.5 cm (4.5 in); Prince William Sound, Alaska to Bahia Playa Maria, Mexico; low intertidal to 143 m (470 ft); carapace purple with white edging; claws with white tips, legs purple (Jensen 1995).

The slender crab is commonly found on mud flats and in beds of eelgrass although it is usually not found intertidally south of central California (Morris et al. 1980). Although seasonally found in bays, the slender crab does not tolerate brackish conditions. Slender crabs can extrude between approximately 143,000 and 1,000,000 eggs per batch (Hines 1991; **Table 5.2.1-5**). Females, on average, produce a single batch per yr. Total lifetime spawning occurs over a maximum of three seasons and more commonly only two (Orensanz and Gallucci 1988). In Elkhorn Slough, mating is common in November, and ovigerous females were noted in July and August (Morris et al. 1980). Farther north, in Puget Sound, animals held in the laboratory bore eggs from December to April, and a few females produced a small second batch. Graham (1989) recorded both spring and fall spawning periods in Monterey Bay.

Larval development in the slender crab was described by Ally (1975). Eggs hatch into pre-zoea larvae, which quickly molt to first stage zoea. Average larval development time (from hatching through completion of the megalops stage) was 48.9 d at 17 °C, with most stages lasting approximately one week (**Table 5.2.1-6**).

During their planktonic existence, crab larvae can become widely distributed in nearshore waters. In a study in Monterey Bay, Graham (1989) found that slender crab stage 1 zoeae were very abundant close to shore (within 6 km) during March and April. During an autumn spawning period in August, he found stage 1 zoeae concentrated approximately 11 km from shore. Later

stage larvae, including megalopae, were found further from shore during all times of the year. This offshore larval distribution, compared to the nearshore distribution of brown rock crab, reflects the fact that adult slender crabs are widely distributed in coastal shelf areas, further offshore than brown rock crabs. The megalops larvae and juvenile crabs are frequently found crawling unharmed on and under the bells, and even in the stomachs, of larger jellyfishes, especially *Pelagia colorata* (Morris et al. 1980).

There are no published estimates of larval mortality in slender crab, except for in laboratory culture (Ally 1975). From December 1996 until July 1997, megalopae of *C. gracilis* were more abundant than all zoea stages 2–5 in the entrainment subsamples, indicating that many of the developing megalopae were being transported back toward shore. However, slender crab megalopae are less abundant close to shore than brown rock crab (Graham 1989). As a result, it is difficult to estimate slender crab larval survival from entrainment sampling. Therefore, data from the abundance of stage 1 zoeae and of megalopae for brown rock crab in the DCPP entrainment were used to estimate the larval mortality of slender crab. An instantaneous larval mortality of 0.158/d, estimated for another species, the brown rock crab, and described in **Section 5.2.1.1**, was applied for estimating the number of females that produced the entrained slender crab.

Based on field growth studies, it was estimated that slender crabs matured at an age of about 10 months post-settlement to a size of approximately 60 mm carapace width (Orensanz and Gallucci 1988). Growth occurs through 11–12 instars, with crabs attaining an estimated maximum age of 4 yr post-settlement.

Summary of Field Collections

Cancer gracilis zoea were removed from subsamples collected at the DCPP intake structure from December 1996 through June 1998, while megalops were removed for the period October 1996 through June 1999. Zoea were removed only from some of the subsamples collected during two grid surveys in 1998. Megalops were removed from subsamples from the study grid from July 1997 until June 1999. Estimated density (numbers per volume of water filtered) of zoea and megalops from the processed subsamples are presented in **Appendix H**. The estimated weekly

number of zoea and megalops entrained (**Appendix I**) was calculated by multiplying the estimated larval density by the volume of water drawn through the CWS. Zoea abundance was greatest in the spring although zoea 1 began to appear in the entrainment subsamples during the winter months (**Figure 5.2.1-5**). Megalops were found throughout the entire year, with peak abundances in winter 1996–97, fall 1998, and spring 1999.

Larval densities were estimated from bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999. For a daily sampling event there generally were 32 to 64 subsamples processed for estimating megalops densities and eight subsamples processed for counting zoea. Of a possible 5,524 subsamples processed for megalopae, slender crab were found in 590 (11%). Megalops densities appear relatively constant with peaks in late fall and winter (**Figure 5.2.1-5**). Of 636 subsamples processed for zoea, the frequencies of subsamples by stage were 83% zoea 1, 6% zoea 2, 3% zoea 3, 1% zoea 4, and no zoea 5.

Megalops densities in the DCPP offshore study grid were highest in May and June 1998 and frequently encountered at all other times, especially July 1997, April 1998, August 1998, September 1998, March 1999, and May 1999 (**Figure 5.2.1-6**). There were 3,168 bongo net subsamples processed for megalops, of which 15% contained slender crab megalops.

Estimating Total Annual Entrainment

As previously mentioned, estimates of entrainment of zoea 1 of the taxa brown rock crab, yellow crab, and slender crab were combined because of uncertainties in identification. The proportions of slender crab zoea stage 1 were derived by using the proportions of slender crab in zoea stage 2 that could be identified to species.

Estimates of annual entrainment ($\widehat{E_T}$) of the five zoea and one megalops stages of brown rock crab during the two analysis periods (December 1, 1996–November 30, 1997 and July 1, 1997–June 30, 1998) are presented in **Table 5.2.1-7.** The date of the first analysis period for zoea was adjusted from October 1996 through September 1997 to December 1996 through November 1997 because the entrainment subsamples collected during October and November 1996 were not

processed for zoea. Generally there were fewer individuals of each older zoea stage, although no zoea stage 4 were collected during 1997 in Analysis Period 1, and no zoea stage 5 were collected in either period. Although zoea abundance overall was estimated to be less in 1997 during Analysis Period 1, the estimate of megalops abundance ($\widehat{E_T}$ = 20,300,000) was greater than 1998 during Analysis Period 3 ($\widehat{E_T}$ = 4,870,000). The estimates of annually-entrained slender rock crab were not adjusted to a long-term average because crab larvae were not enumerated in the Intake Cove surface plankton tow samples.

Fecundity Hindcasting (FH)

An instantaneous daily mortality rate, -0.158, calculated for brown rock crab is used for slender crab. Estimated survival through larval stages can then be calculated as $\hat{S}_{larvae} = e^{(-0.158)t}$, where t is reported in **Table 5.2.1-6** as mean cumulative duration. Survival estimates to stage midpoint were

Full stage duration on Table 5.2.1-6

1) zoea 1 0.584
2) zoea 2 0.375
3) zoea 3 0.063
4) zoea 4 0.0222
5) zoea 5 0.0077
6) megalops 0.0014 .

The *FH* approach combines larval entrainment losses with survival and adult fecundity to hindcast the numbers of adult females effectively removed from the reproductively active population. Mean annual fecundity, using data through age three (**Table 5.2.1-5**), was estimated as 555,583 eggs. The number of reproductive years was estimated by halving the range between age of maturity of 1 yr and assumed effective longevity of 3 yr (supported by linear survivorship model for adults). Coefficients of variation of survival, fecundity, and ages of maturation and longevity are not well known and are, therefore, reasonably assumed to be 100% with a lognormal error structure.

The number of female slender crabs estimated to produce the entrained larvae was calculated for the five zoea and one megalops stage for the two periods. All larval stages were hindcast, and the estimated numbers of females and variances of estimates were added to compute an estimate of total number of slender crab females and the variance of the estimate. The estimated FH for zoea and megalops stages for the period December 1, 1996 through November 30, 1997 from **Table 5.2.1-7** were

$$\widehat{FH}_1 = 538 \text{ females with } \widehat{SE}(\widehat{FH}_1) = 1{,}150$$
$$\widehat{FH}_2 = 220 \text{ females with } \widehat{SE}(\widehat{FH}_2) = 563$$
$$\widehat{FH}_3 = 221 \text{ females with } \widehat{SE}(\widehat{FH}_3) = 528$$
$$\widehat{FH}_4 = 143 \text{ females with } \widehat{SE}(\widehat{FH}_4) = 392$$
$$\widehat{FH}_{Megs} = 26{,}200 \text{ females with } \widehat{SE}(\widehat{FH}_{Megs}) = 55{,}500$$

No slender crab zoea stage 5 were collected during this period. The total estimated number of breeding females needed to produce the total number of larvae entrained was calculated by summing the $\widehat{FH}$ over all stages:

$$\widehat{FH} = 27{,}300 \text{ females,}$$

with $\widehat{SE}(\widehat{FH}) = \sqrt{\sum_{i=1}^{5} \widehat{SE}_i \left(\widehat{FH}_i\right)^2} = 55{,}500$. An approximate 90% confidence interval for FH, based on a lognormal error structure, is estimated as 959 to 776,000 females.

The estimated FH for slender crab zoea and megalops stages for the period July 1, 1997 through June 30, 1998 from **Table 5.2.1-7** were

$$\widehat{FH}_1 = 1{,}480 \text{ females with } \widehat{SE}(\widehat{FH}_1) = 3{,}180$$
$$\widehat{FH}_2 = 469 \text{ females with } \widehat{SE}(\widehat{FH}_2) = 1{,}050$$
$$\widehat{FH}_3 = 729 \text{ females with } \widehat{SE}(\widehat{FH}_3) = 1{,}670$$
$$\widehat{FH}_{Megs} = 6{,}270 \text{ females with } \widehat{SE}(\widehat{FH}_{Megs}) = 13{,}300$$

No slender crab zoea stage 4 or 5 were collected during this period. The total estimated number of breeding females needed to produce the total number of larvae entrained during this period was calculated by summing the stage specific estimates as

$$\widehat{FH} = 8{,}950 \text{ females}$$

with $\widehat{SE}(\widehat{FH}) = \sqrt{\sum_{i=1}^{5} \widehat{SE}_i \left(\widehat{FH}_i\right)^2} = 13{,}800$. An approximate 90% confidence interval for *FH*, based on a lognormal error structure, is estimated as 703 to 114,000 females.

Sensitivity Analysis

In addition to calculating a confidence interval, a sensitivity analysis (**Table 5.2.1-7**) for all larval stages was performed, in which $\widehat{FH}$ was recomputed by varying the input parameter of estimated annual entrainment, $\widehat{E_T}$, by $\pm 1.645\widehat{SE}\left(\widehat{E_T}\right)$ and survival, $\hat{S}$, by using a multiplicative error structure (i.e. $\hat{S}e^{\pm 1.645 CV(\hat{S})}$) and using $CV\left(\hat{S}\right) = 1.0$. The range of survival estimates for all stages produced the same relative changes in $\widehat{FH}$, as expected based upon a constant multiplier. However, $\widehat{FH}$ was less sensitive to entrainment than survival variation. This difference was especially pronounced for the zoea 1 and megalops stages and can be explained by the use of sampling variance for estimating entrainment variability, with *CV*s of less than 100%. Therefore, improvement in the precision of $\widehat{FH}$ can result from improved estimates of the larval stages' survival.

Adult Equivalent Loss (AEL)

The *AEL* approach uses estimates of the abundance of entrained organisms to project the loss of equivalent numbers of adults based on mortality schedules and age at recruitment. Survival rates from entrainment of megalopae to recruitment into fishery are not available. However, assuming a 50:50 sex ratio, $\widehat{AEL}$ and $\widehat{FH}$ can be compared as $\widehat{AEL} \equiv 2\widehat{FH} = 54{,}600$ and 17,900 adults, with an associated standard error of $\widehat{SE}(\widehat{AEL}) = 2\widehat{SE}(\widehat{FH}) = 111{,}000$ and 27,700.

5.2.1.2e Empirical Transport Model (ETM)

The *ETM* assessment requires estimates of larval stage duration and larval density in entrainment and grid subsamples. The crab data were collected for only the megalops larval stage in the grid subsamples and will be compared to megalops stage larvae in the entrainment subsamples.

Two entrainment mortality estimates are presented: for July 1, 1997 through June 30, 1998 and for July 1, 1998 through June 30, 1999. Slender crab megalops larvae may be susceptible to entrainment by the power plant for approximately 42 d, based on the average estimated larval duration (Ally 1975). Although the duration is more than one month, each monthly survey period will be assumed to represent discrete groups of larvae, permitting two independent weighted estimates of total entrainment mortality from 24 subsamples. Total annual entrainment mortality (P_M) is estimated from the values of PE_i and f_i for each survey period (**Table 5.2.1-8**), based on larval densities from entrainment (**Figure 5.2.1-5**) and grid surveys (**Figure 5.2.1-6**).

The proportion of the population within the nearshore study grid was based on a linear extrapolation of densities in the grid in the offshore direction and displacement in the alongshore direction. Monthly estimates of proportional entrainment ($\widehat{PE_i}$) for slender crab in each *i*th survey from the two years sampled ranged from a minimum of zero to a maximum of 0.859±0.911 ($\pm 1\,\widehat{SE}\left(\widehat{PE_i}\right)$) in August 1997 (**Table 5.2.1-8**). Periods when both $\widehat{PE_i}$ and the annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) were equal to zero (December 1997 and January 1998) indicate that no larvae were collected at either the DCPP intake or the study grid. When $\widehat{PE_i}$ =0 and $\hat{f}_i$>0, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. Low values of $\hat{f}_i$ indicate periods of the year when slender crab megalops are least abundant in weekly entrainment subsamples. June 1998 and October 1998 were highest for the two annual periods. May and June 1998 densities were highest in the study grid during the first period, and August and September 1998 were highest in the second period (**Figure 5.2.1-6**).

Total annual entrainment mortality from the *ETM* is estimated as $\widehat{P_M}$ = 0.0107 for the period July 1, 1997 through June 30, 1998 and $\widehat{P_M}$ = 0.000784 for the period July 1, 1998 through June 30, 1999 with associated standard errors of $\widehat{SE}(\widehat{P_M})$ 0.0515 and 0.0373, respectively (**Figure 5.2.1-7**). The sensitivity analysis, which considers the effect of varying values of $\widehat{P_S}$ on the outcome of the *ETM* calculations, is shown in **Figure 5.2.1-7**. The values of $\widehat{P_M}$ resulting from calculations using only the alongshore displacement were higher (both years $\widehat{P_M}$ = 0.09) than those reported

for the offshore extrapolated densities (0.0107 and 0.000784). Values of $\widehat{P_S}$ ranged from 0.00422 to 0.00875 for $\widehat{P_M}$ extrapolated offshore and 0.125 for both years using alongshore only.

Interpretation of Assessment Results

Interpretation of assessment results for slender crabs is limited by the absence of any information on population size. The *ETM* estimates of 1.07% and 0. 0784% probably represent low population impacts. Values of $\widehat{P_S}$ that were calculated based on extrapolation offshore were also larger than for brown crabs. As a result (comparing $1/\widehat{P_S}$), the population of slender crab at risk proportional to the study grid was estimated to be approximately 5–10 times smaller than the population of brown rock crabs relative to the study grid. However, values of $1/\widehat{P_S}$ are nearly the same for both slender and brown rock crabs, from the calculation using alongshore currents only. There is no fishery value for the estimates of slender crab adult equivalent loss (54,600 and 17,900 crabs).

Table 5.2.1-1. Schedule of estimated fecundity by age for brown rock crab (*Cancer antennarius*; from Carroll 1982 and Hines 1991).

Age (yr post-settlement)	Estimated carapace width (mm)	Estimated no. eggs per batch	Number of Batches
1.5	62	156,400	1.0
2	83	513,100	1.5
3	105	1,401,800	1.5
4	123	2,664,700	1.5
5	135	3,994,600	1.5
6	143	5,004,000	1.5

Table 5.2.1-2. Brown rock crab (*Cancer antennarius*): Annual estimated number of zoeae and megalopae entrained, estimated *FH*, and sensitivity of zoea- and megalops-based *FH* estimates for two analysis periods. Sensitivity estimates recalculated for model parameters (θ) of entrainment ($\hat{\theta} \pm 1.645 \cdot \widehat{SE}(\hat{\theta})$), and survival ($\hat{\theta} e^{\pm 1.645\, CV(\hat{\theta})}$ where $CV(\hat{\theta}) = 1.0$). $\widehat{E_T}$ = estimated annual # crab larvae entrained (by stage), *FH* = number of adult female crabs, and *S* = finite survivorship.

rep of: Oct 96 - Sept 97

a) Analysis Period 1 (date adjusted): December 1, 1996–November 30, 1997

Larval Stage	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Sensitivity Analysis Parameters of Recalculated $\widehat{FH}$: $\widehat{E_T}$ $+1.645\widehat{SE}(\hat{\theta})$	$\widehat{E_T}$ $-1.645\widehat{SE}(\hat{\theta})$	$\hat{S}$ $\times e^{+1.645\, CV(\hat{\theta})}$	$\hat{S}$ $\times e^{-1.645\, CV(\hat{\theta})}$
Zoea 1	17,400,000,000	3,940,000,000	17,500	39,000	24,000	11,000	3,370	90,600
Zoea 2	4,580,000,000	3,410,000,000	16,300	38,100	36,200	0	3,140	84,400
Zoea 3	4,720,000,000	4,370,000,000	43,300	104,000	109,000	0	8,350	224,000
Zoea 4	665,000,000	572,000,000	15,700	37,400	38,000	0	3,040	81,500
Zoea 5	3,480,000	4,140,000	249	626	735	0	48	1,290
Megalops	56,500,000	4,480,000	24,100	53,400	27,200	20,900	4,650	125,000
TOTAL	27,424,980,000		117,000					

b) Analysis Period 3: July 1, 1997–June 30, 1998

Larval Stage	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Sensitivity Analysis Parameters of Recalculated $\widehat{FH}$: $\widehat{E_T}$ $+1.645\widehat{SE}(\hat{\theta})$	$\widehat{E_T}$ $-1.645\widehat{SE}(\hat{\theta})$	$\hat{S}$ $\times e^{+1.645\, CV(\hat{\theta})}$	$\hat{S}$ $\times e^{-1.645\, CV(\hat{\theta})}$
Zoea 1	18,500,000,000	6,260,000,000	18,600	41,700	28,900	8,270	3,590	96,400
Zoea 2	3,770,000,000	1,900,000,000	13,400	30,500	24,500	2,310	2,590	69,400
Zoea 3	2,420,000,000	1,760,000,000	22,100	51,700	48,700	0	4,280	115,000
Zoea 4	781,000,000	910,000,000	18,500	46,300	53,900	0	3,570	95,700
Zoea 5	111,000,000	135,000,000	7,950	20,100	23,800	0	1,530	41,200
Megalops	24,500,000	2,200,000	10,400	23,100	12,000	8,880	2,010	54,000
TOTAL	25,606,500,000		91,000					

Table 5.2.1-3. Duration of laboratory-reared zoea stages (@ 13.8°C) of brown rock crab (*Cancer antennarius*; Roesijadi 1976) and megalops stage duration from. Ally (1975). Mean cumulative duration is cumulative duration minus half of the stage duration.

Larval Stage	Stage Duration (d)	Cumulative duration	Mean Cumulative duration
Zoea 1	10	10	5
Zoea 2	6	16	13
Zoea 3	6	22	19
Zoea 4	6	28	25
Zoea 5	8	36	32
Megalops	14.6	50.6	43.3
Total	50.6		

Table 5.2.1-4. Brown rock crab (*Cancer antennarius*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors ($\widehat{SE}$) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}(\widehat{PE_i})$	$\hat{f}_i$	$\widehat{SE}(\hat{f}_i)$
Jul 21, 1997	0	0	0.00600	0.00134
Aug 25, 1997	0	0	0.00586	0.00167
Sep 29, 1997	0	0	0.00169	0.000640
Oct 20, 1997	0.0254	0.0219	0.0127	0.00206
Nov 17, 1997	0.0224	0.0313	0.00163	0.000620
Dec 10, 1997	0	0	0.00106	0.000470
Jan 22, 1998	0	0	0.00370	0.00103
Feb 26, 1998	0	0	0.00201	0.000660
Mar 18, 1998	0	0	0.00251	0.000830
Apr 15, 1998	0.00248	0.00146	0.0528	0.00504
May 18, 1998	0.000260	0.0000700	0.289	0.0157
Jun 8, 1998	0.000890	0.000390	0.621	0.0162

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}(\widehat{PE_i})$	$\hat{f}_i$	$\widehat{SE}(\hat{f}_i)$
Jul 21, 1998	0.00114	0.00117	0.650	0.0232
Aug 26, 1998	0	0	0.0132	0.00384
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0.00807	0.00903	0.0302	0.00637
Nov 11, 1998	0.00644	0.00723	0.0156	0.00435
Dec 9, 1998	0.0108	0.00917	0.0274	0.00582
Jan 12, 1999	0	0	0.0179	0.00483
Feb 3, 1999	0	0	0	0
Mar 17, 1999	0	0	0	0
Apr 14, 1999	0	0	0.0212	0.00535
May 24, 1999	0	0	0.171	0.0164
Jun 23, 1999	0.000820	0.000520	0.0532	0.00963

Table 5.2.1-5. Schedule of estimated fecundity by age for slender crab (*Cancer gracilis*) (from Orensanz and Gallucci 1988; Hines 1991).

Age (yr post-settlement)	Estimated carapace width (mm)	Estimated no. eggs per batch	Number of Batches
1.5	47	143,800	1.0
2	62	342,400	1.5
3	77	672,900	1.5
4	86	953,400	1.0

Table 5.2.1-6. Duration of planktonic zoea and megalops stages of slender crab (*Cancer gracilis*) reared in the laboratory (Ally 1975).

Larval Stage	Mean duration (d)	Lower limit (d)	Upper limit (d)	Cumulative duration	Mean Cumulative duration
Zoea 1	6.8	5	11	6.8	3.4
Zoea 2	7.2	5	12	14.0	6.2
Zoea 3	7.0	5	10	21.0	17.5
Zoea 4	6.2	5	8	27.2	24.1
Zoea 5	7.1	6	8	43.3	30.8
Megalops	14.6	12	17	48.9	41.6
Total	48.9	38	66		

Table 5.2.1-7. Slender crab (*Cancer gracilis*): Annual estimated number of zoeae and megalopae entrained, estimated *FH*, and sensitivity of zoea- and megalops-based *FH* estimates for two analysis periods. Sensitivity estimates recalculated for model parameters (θ) of entrainment ($\hat{\theta} \pm 1.645 \cdot \widehat{SE}(\hat{\theta})$), and survival ($\hat{\theta} e^{\pm 1.645\, CV(\hat{\theta})}$ where $CV(\hat{\theta}) = 1.0$). $\widehat{E_T}$ = estimated # crab larvae entrained (by stage), FH = number of adult female crabs, and S = finite survivorship.

a) Analysis Period 1 (date adjusted): December 1, 1996–November 30, 1997

Larval Stage	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Sensitivity Analysis Parameters of Recalculated $\widehat{FH}$			
					$\widehat{E_T}$ $+1.645\widehat{SE}(\hat{\theta})$	$\widehat{E_T}$ $-1.645\widehat{SE}(\hat{\theta})$	$\hat{S}$ $\times e^{+1.645\,CV(\hat{\theta})}$	$\hat{S}$ $\times e^{-1.645\,CV(\hat{\theta})}$
Zoea 1	175,000,000	39,600,000	538	1,150	739	338	104	2,790
Zoea 2	46,000,000	65,600,000	220	563	737	0	43	1,140
Zoea 3	7,740,000	8,510,000	221	528	621	0	43	1,150
Zoea 4	1,770,000	3,040,000	143	392	549	0	28	742
Zoea 5	0	0	0	.	0	0	0	0
Megalops	20,300,000	1,140,000	26,200	55,500	28,600	23,700	5,050	136,000
TOTAL			27,300					

b) Analysis Period 3: July 1, 1997–June 30, 1998

Larval Stage	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Sensitivity Analysis Parameters of Recalculated $\widehat{FH}$			
					$\widehat{E_T}$ $+1.645\widehat{SE}(\hat{\theta})$	$\widehat{E_T}$ $-1.645\widehat{SE}(\hat{\theta})$	$\hat{S}$ $\times e^{+1.645\,CV(\hat{\theta})}$	$\hat{S}$ $\times e^{-1.645\,CV(\hat{\theta})}$
Zoea 1	480,000,000	162,000,000	1,480	3,180	2,300	658	286	7,670
Zoea 2	97,700,000	67,200,000	469	1,050	999	0	90	2,430
Zoea 3	25,500,000	21,800,000	729	1,670	1,760	0	141	3,780
Zoea 4	0	0	0	.	0	1	0	0
Zoea 5	0	0	0	.	0	2	0	0
Megalops	4,870,000	633,000	6,270	13,300	7,610	4,930	1,210	32,500
TOTAL			8,950					

Table 5.2.1-8. Slender crab (*Cancer gracilis*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors ($\widehat{SE}$) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}(\widehat{PE_i})$	$\hat{f}_i$	$\widehat{SE}(\hat{f}_i)$
Jul 21, 1997	0	0	0.0602	0.00969
Aug 25, 1997	0.859	0.911	0.0726	0.0131
Sep 29, 1997	0	0	0.0754	0.0158
Oct 20, 1997	0.0230	0.0137	0.0961	0.0114
Nov 17, 1997	0.00102	0.00105	0.0644	0.00974
Dec 10, 1997	0	0	0	0
Jan 22, 1998	0	0	0	0
Feb 26, 1998	0	0	0.000680	0.000680
Mar 18, 1998	0	0	0.00422	0.00173
Apr 15, 1998	0.000860	0.000890	0.144	0.0152
May 18, 1998	0.000350	0.000130	0.151	0.0160
Jun 8, 1998	0.000190	0.000140	0.332	0.0259

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}(\widehat{PE_i})$	$\hat{f}_i$	$\widehat{SE}(\hat{f}_i)$
Jul 21, 1998	0	0	0.0442	0.00675
Aug 26, 1998	0.000310	0.000310	0.0508	0.00591
Sep 16, 1998	0.000910	0.000670	0.0267	0.00471
Oct 6, 1998	0.00439	0.00288	0.302	0.0231
Nov 11, 1998	0.000900	0.000910	0.242	0.0193
Dec 9, 1998	0.00432	0.00334	0.0535	0.00609
Jan 12, 1999	0.00490	0.00557	0.0226	0.00433
Feb 3, 1999	0	0	0.00923	0.00261
Mar 17, 1999	0.000670	0.000490	0.0434	0.00567
Apr 14, 1999	0	0	0.0869	0.0141
May 24, 1999	0.00106	0.000770	0.0859	0.0083
Jun 23, 1999	0.00278	0.00153	0.0325	0.00488

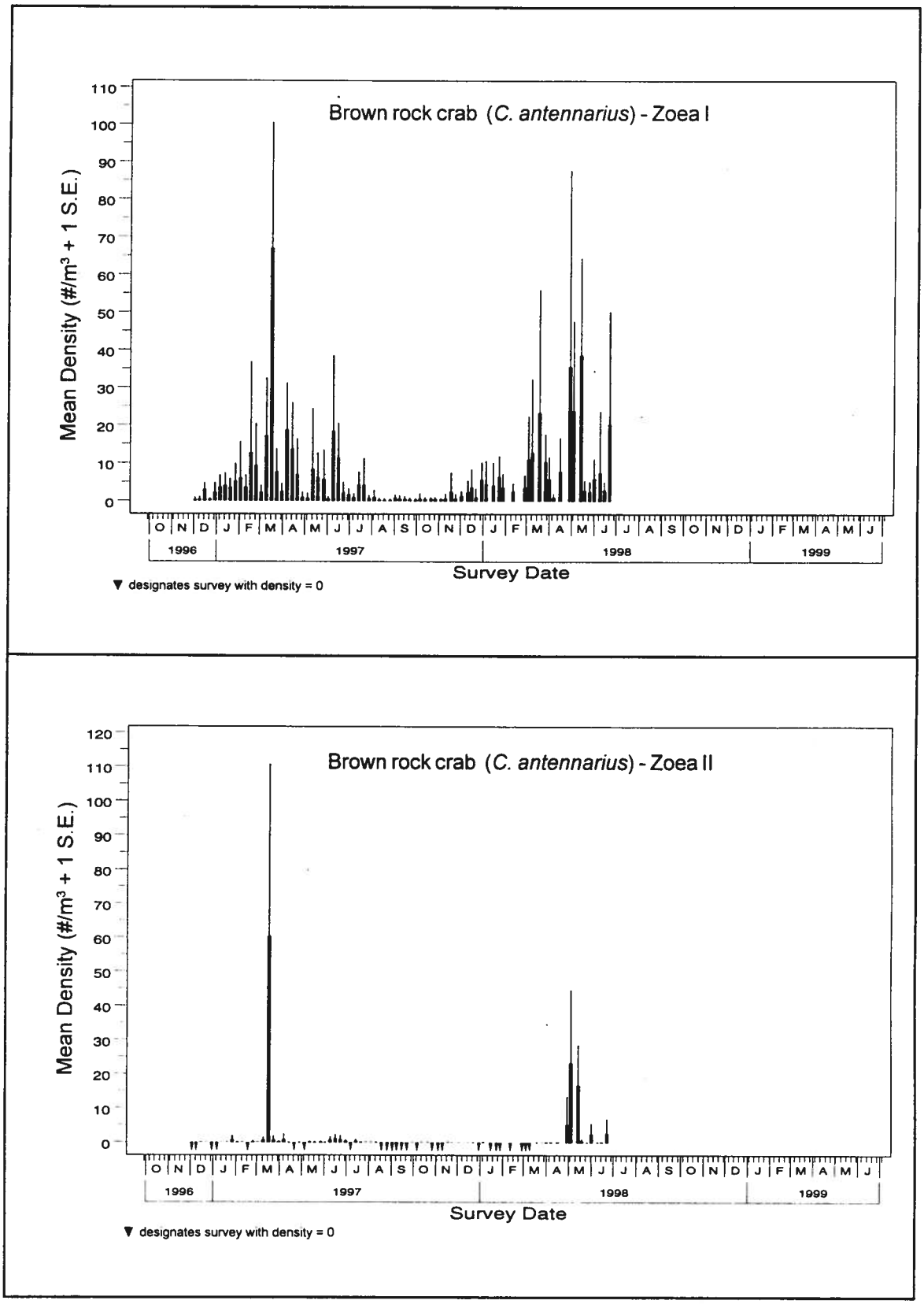


Figure 5.2.1-1. Weekly mean zoeae and megalops larval density (#/m³ + 1 S.E.) at the DCPP intake. Y-axis scale varies between graphs. Zoea not sorted from July 1998 through June 1999 samples.

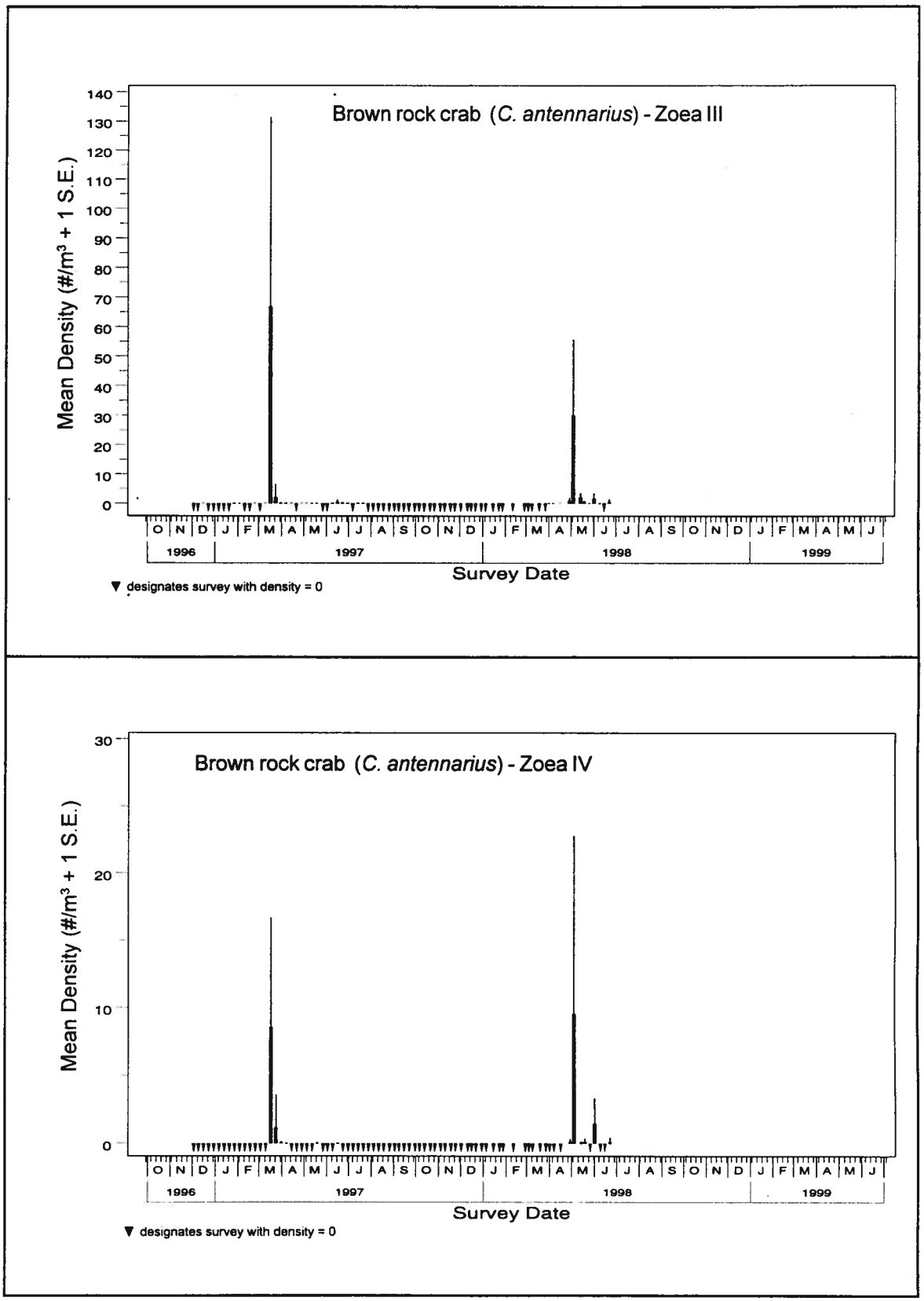


Figure 5.2.1-1 (continued). Weekly mean zoeae and megalops larval density (#/m³ + 1 S.E.) at the DCPP intake. Y-axis scale varies between graphs. Zoea not sorted from July 1998 through June 1999 samples.

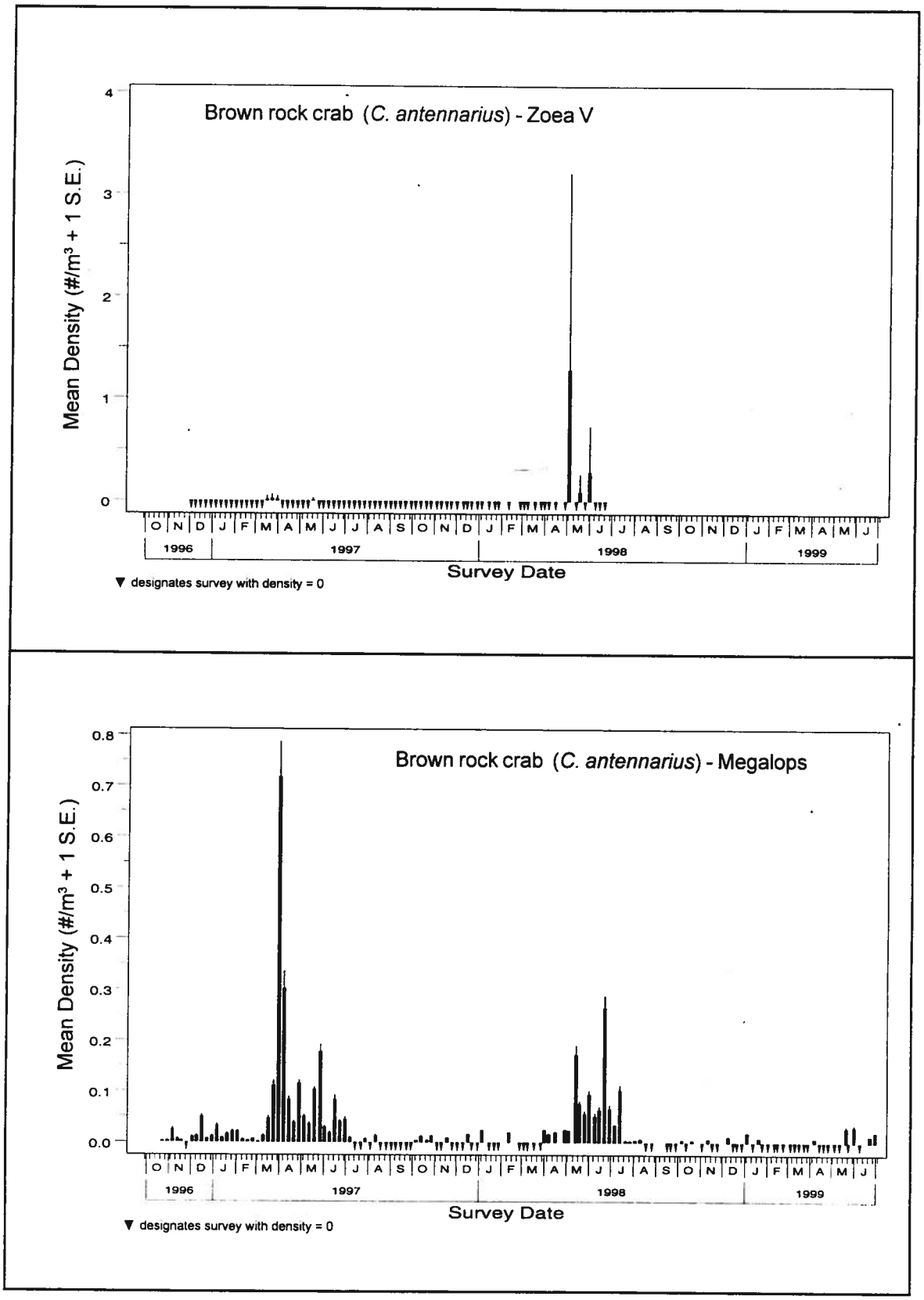


Figure 5.2.1-1 (continued). Weekly mean zoeae and megalops larval density (#/m³ + 1 S.E.) at the DCPP intake. Y-axis scale varies between graphs. Zoea not sorted from July 1998 through June 1999 samples.

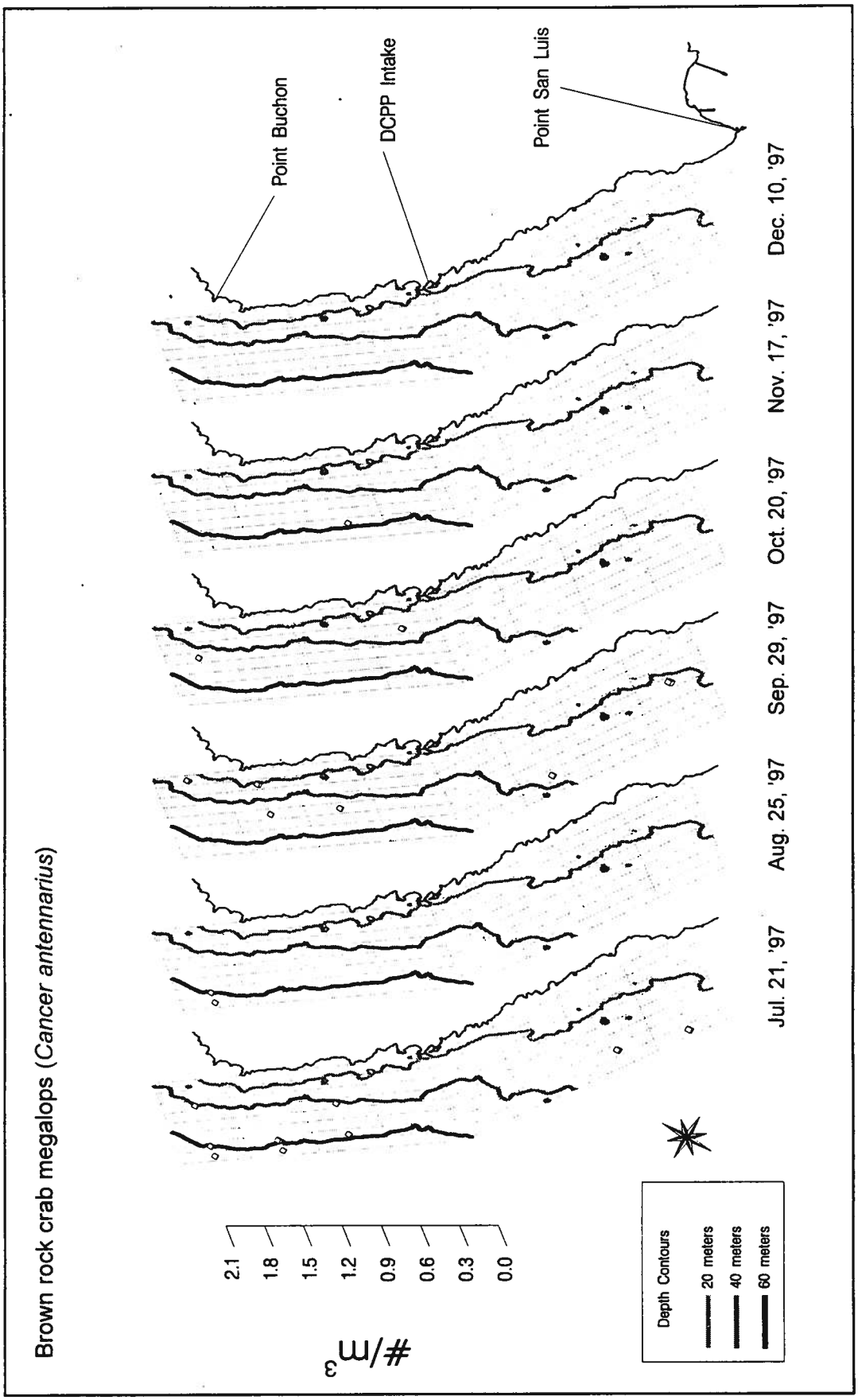


Figure 5.2.1-2a . Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

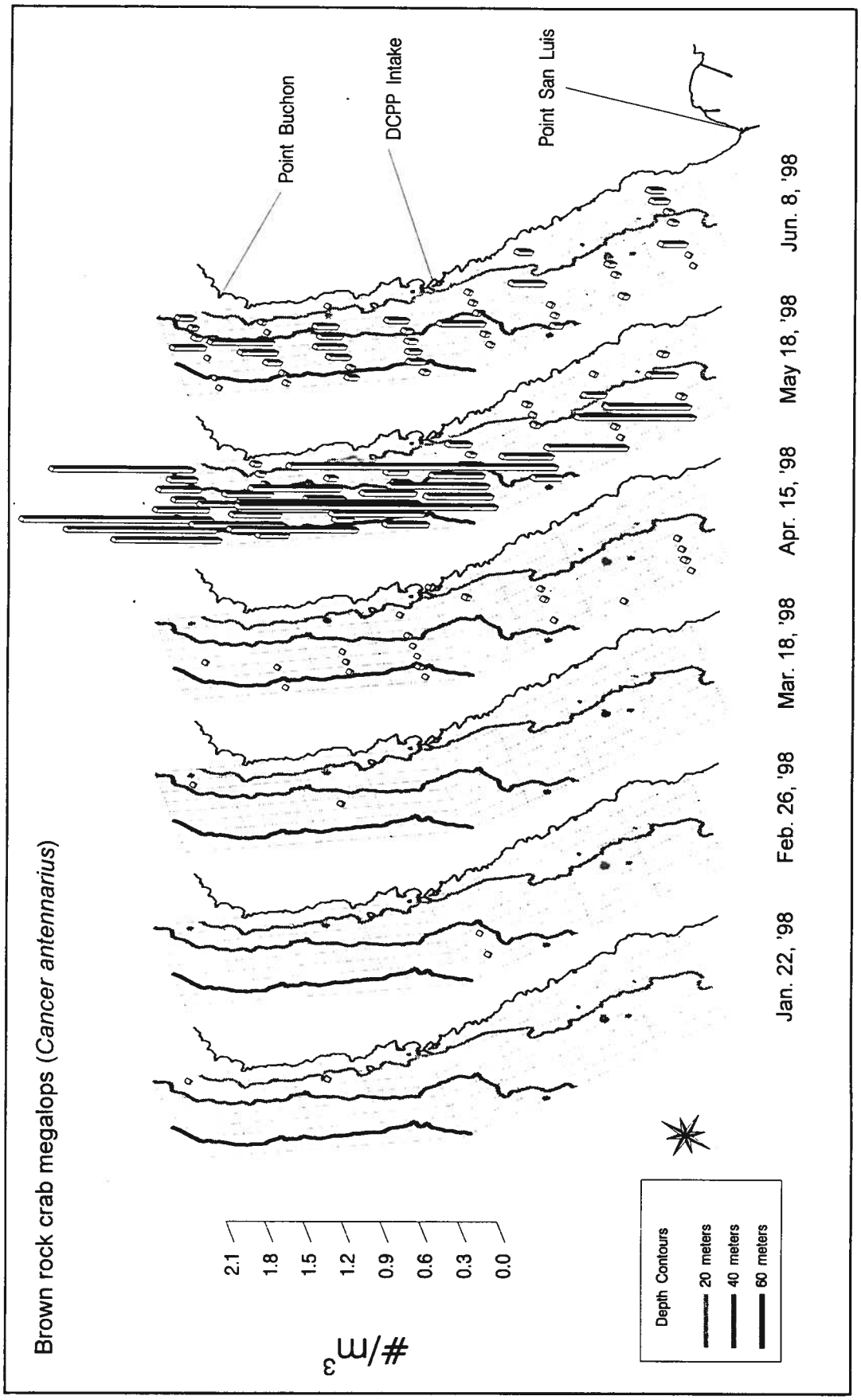


Figure 5.2.1-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

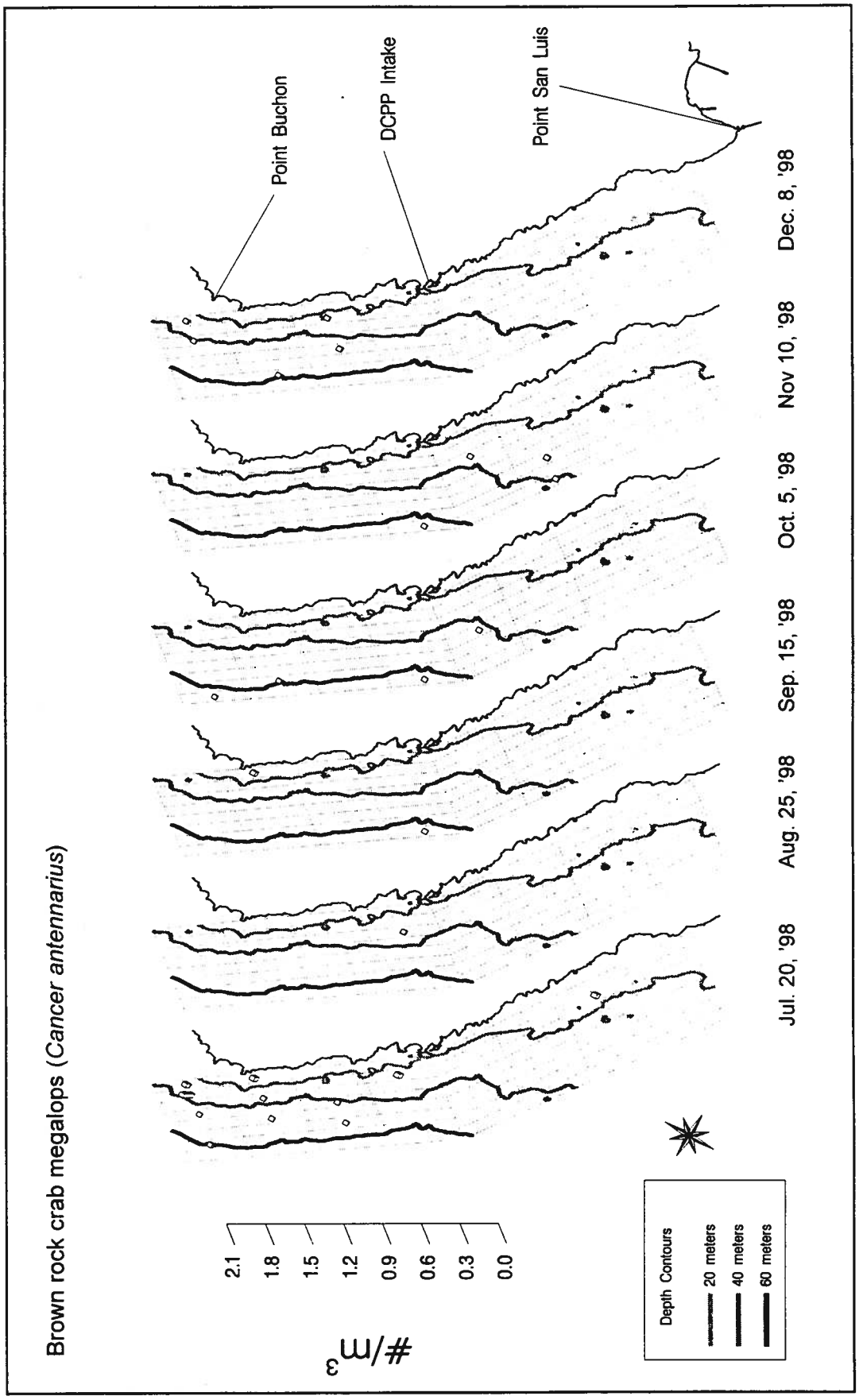


Figure 5.2.1-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

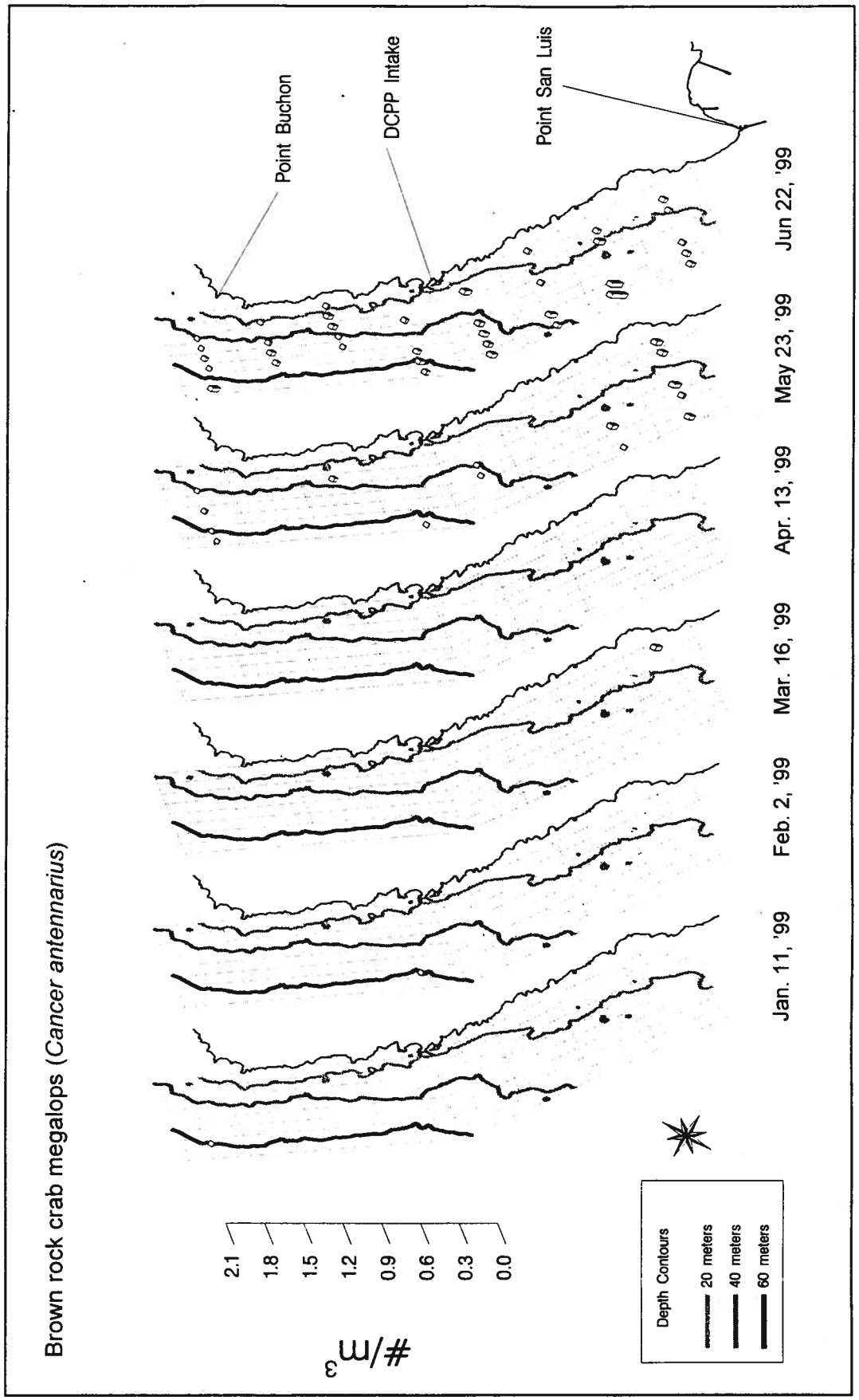


Figure 5.2.1-2d (continued). Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

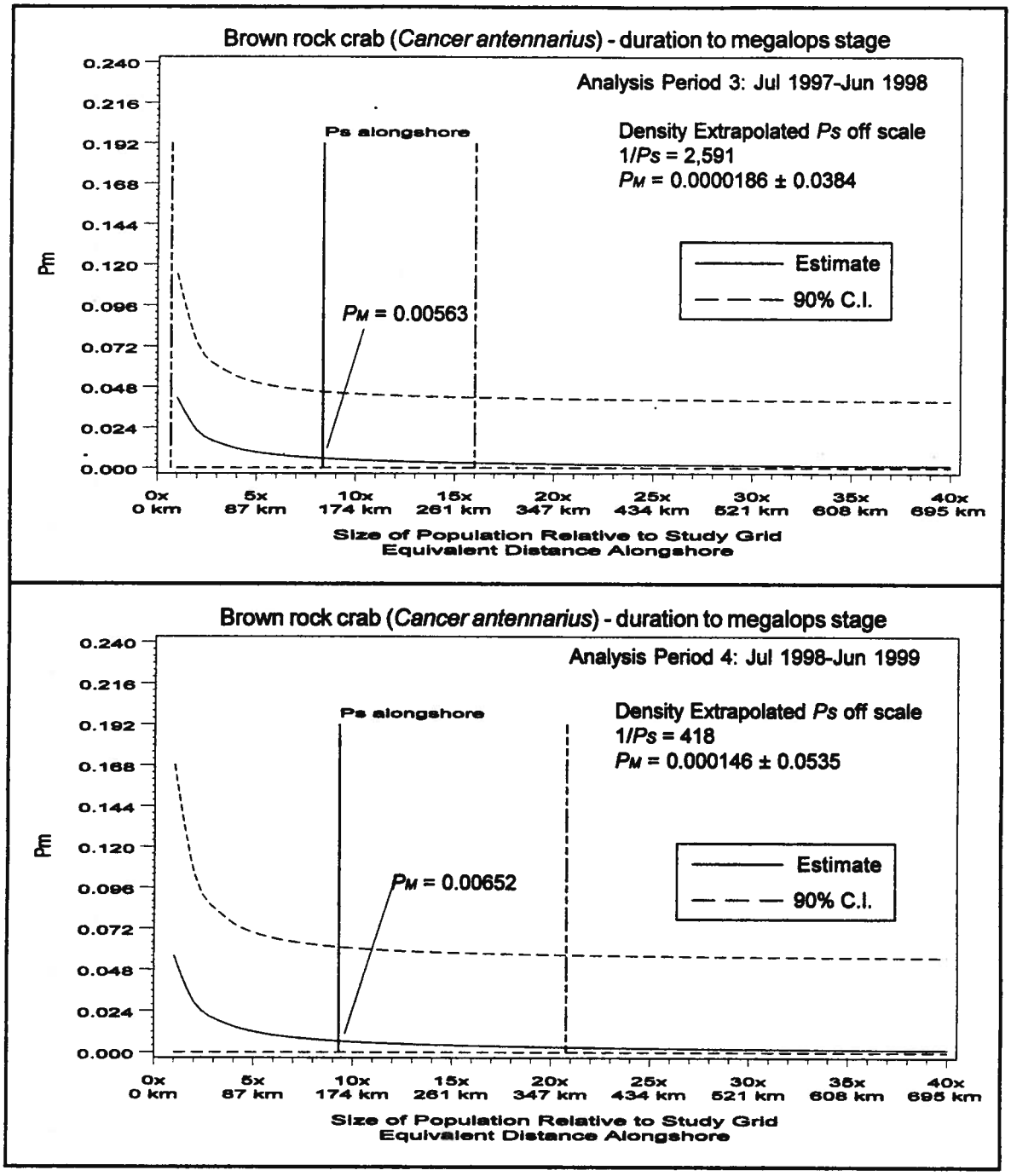


Figure 5.2.1-3. Total annual entrainment mortality (P_M) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid (P_S) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

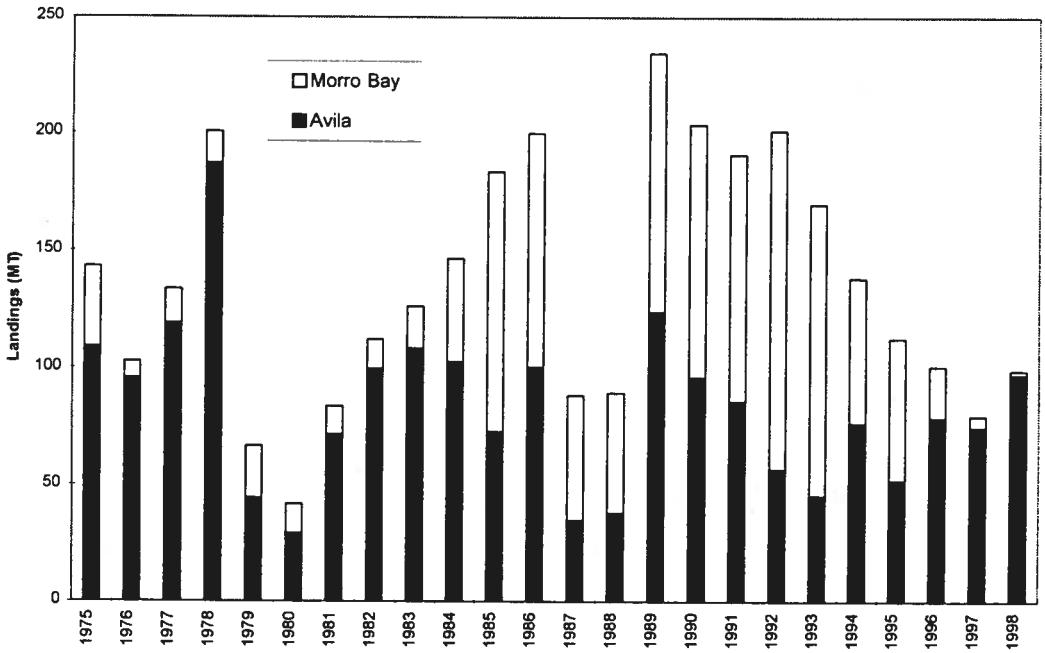


Figure 5.2.1-4. Rock crab (*Cancer* spp.) landings (MT) at Morro Bay and Port San Luis between the years 1975–1998 (source: CDF&G database).

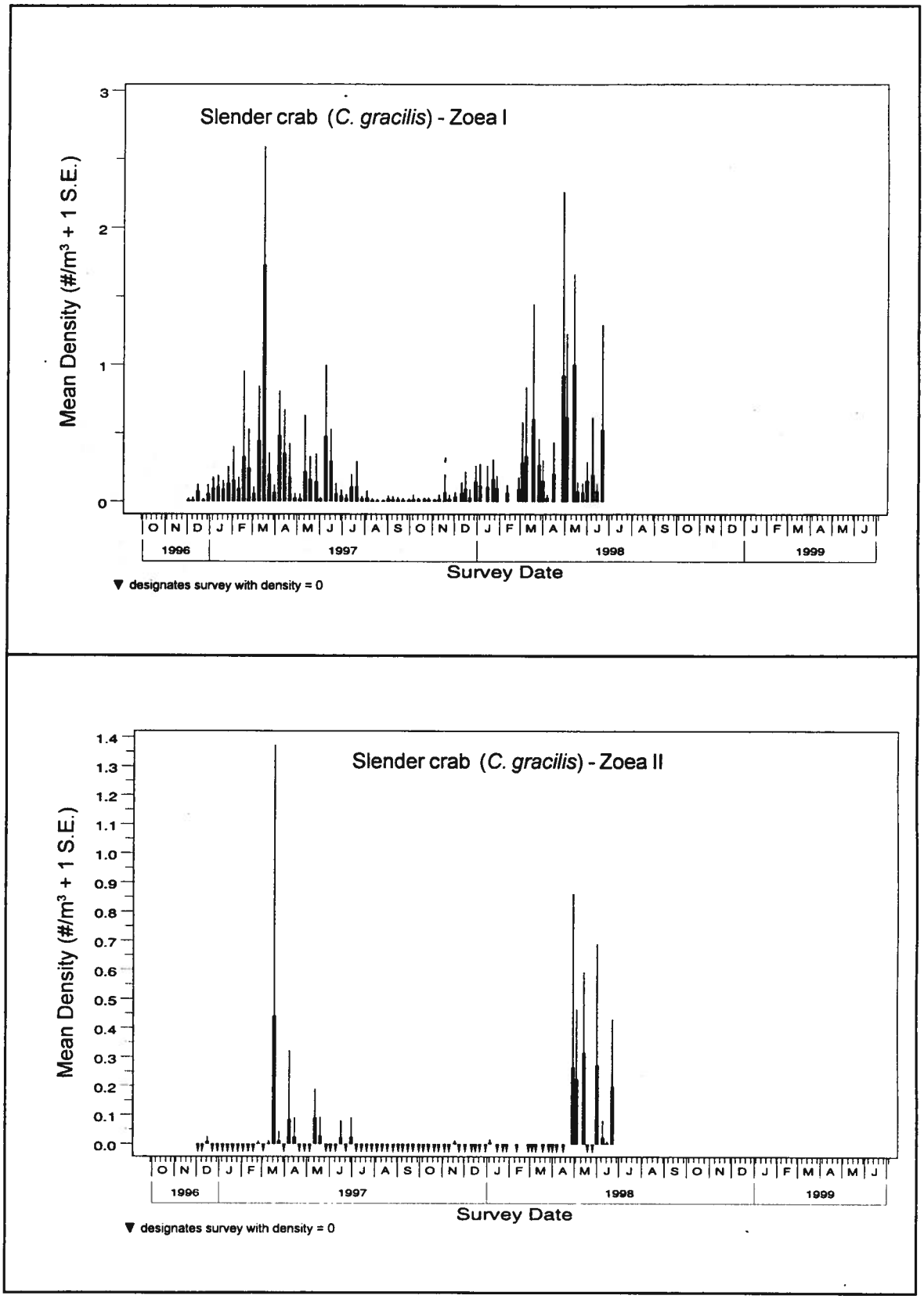


Figure 5.2.1-5. Weekly mean zoeae and megalops larval density (#/m³ + 1 S.E.) at the DCPP intake. Y-axis scale varies between graphs. Zoea not sorted from July 1998 through June 1999 samples.

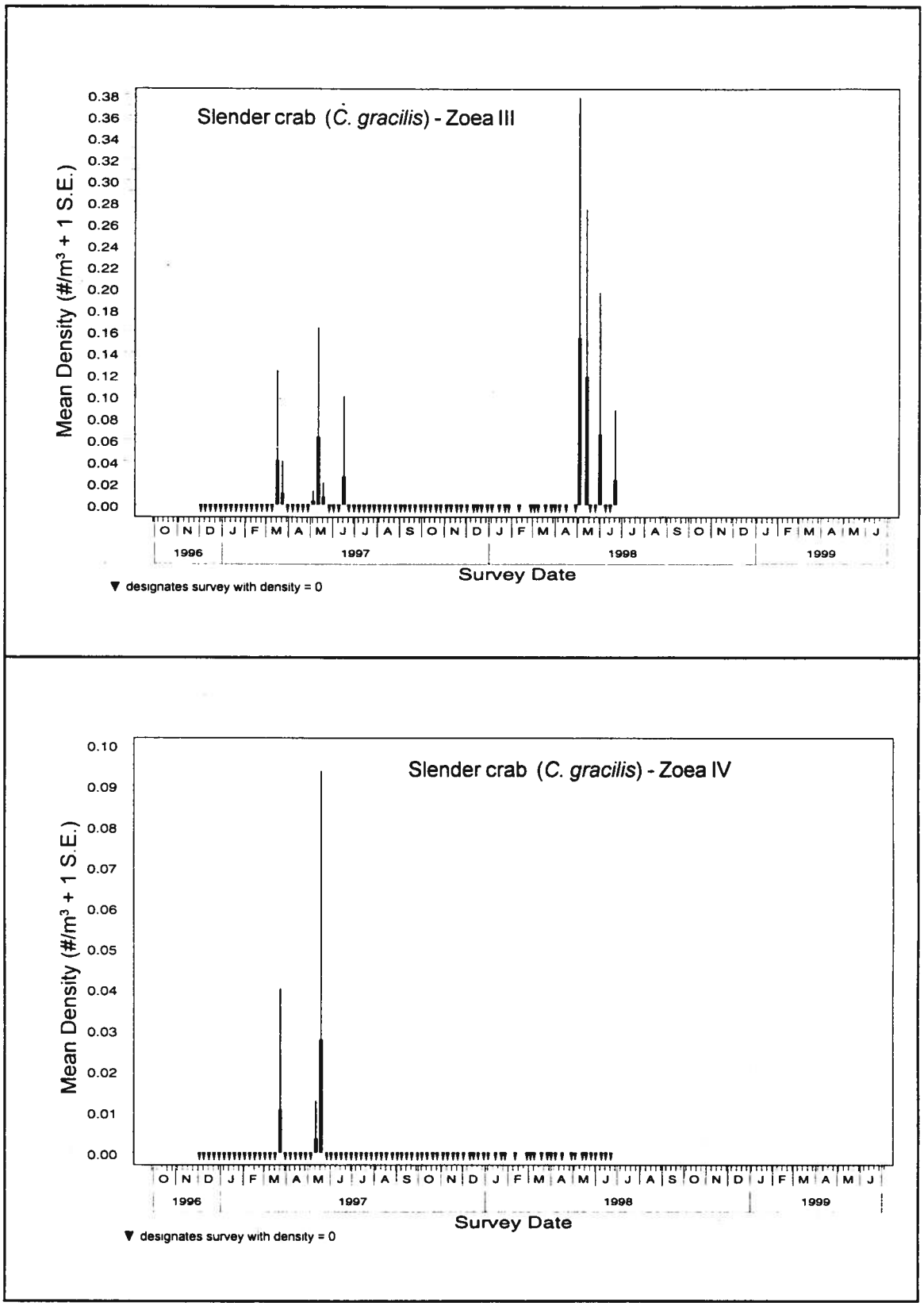


Figure 5.2.1-5 (continued). Weekly mean zoeae and megalops larval density (#/m³ + 1 S.E.) at the DCPP intake. Y-axis scale varies between graphs. Zoea not sorted from July 1998 through June 1999 samples.

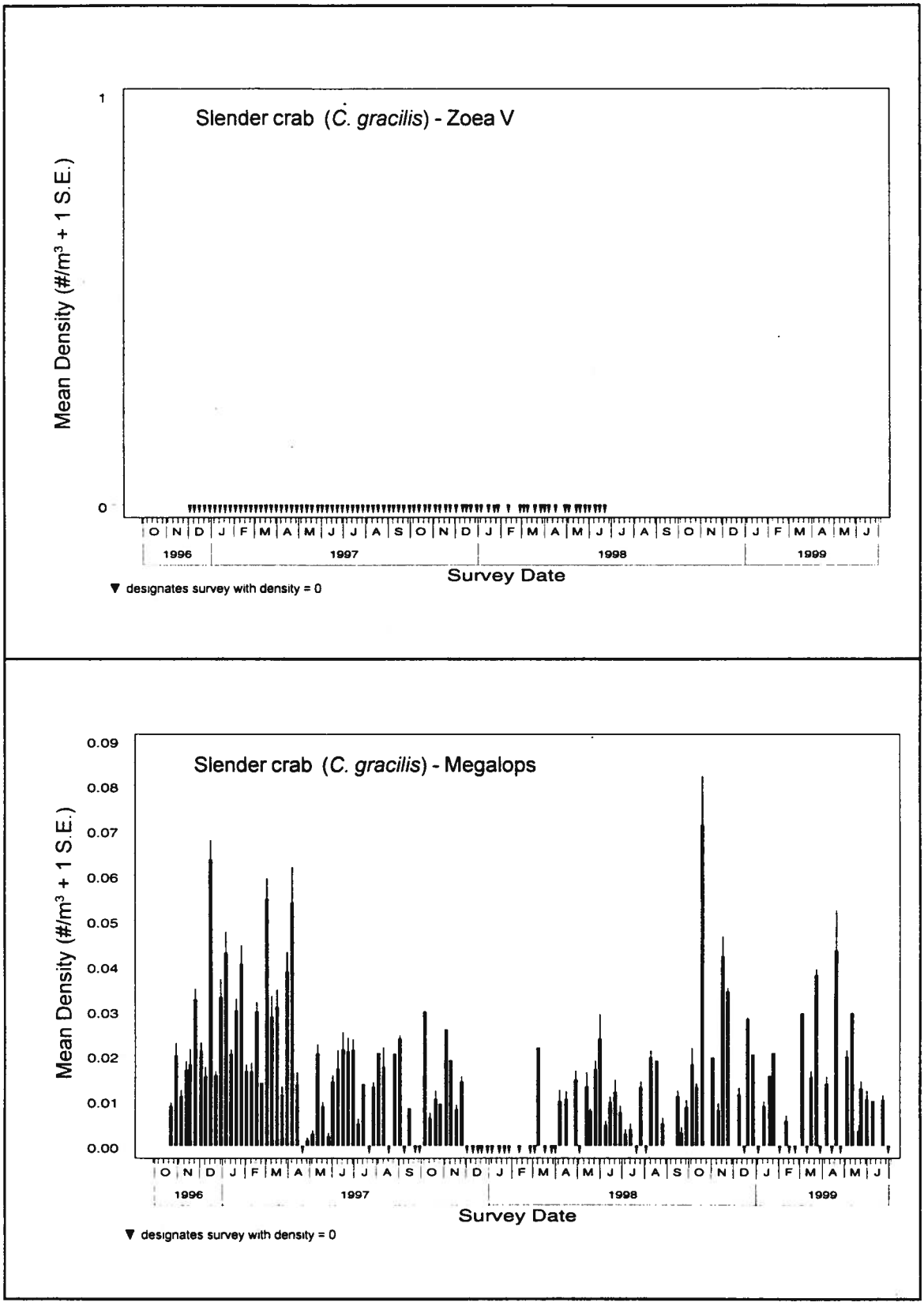


Figure 5.2.1-5 (continued). Weekly mean zoeae and megalops larval density (#/m³ + 1 S.E.) at the DCPP intake. Y-axis scale varies between graphs. Zoea not sorted from July 1998 through June 1999 samples.

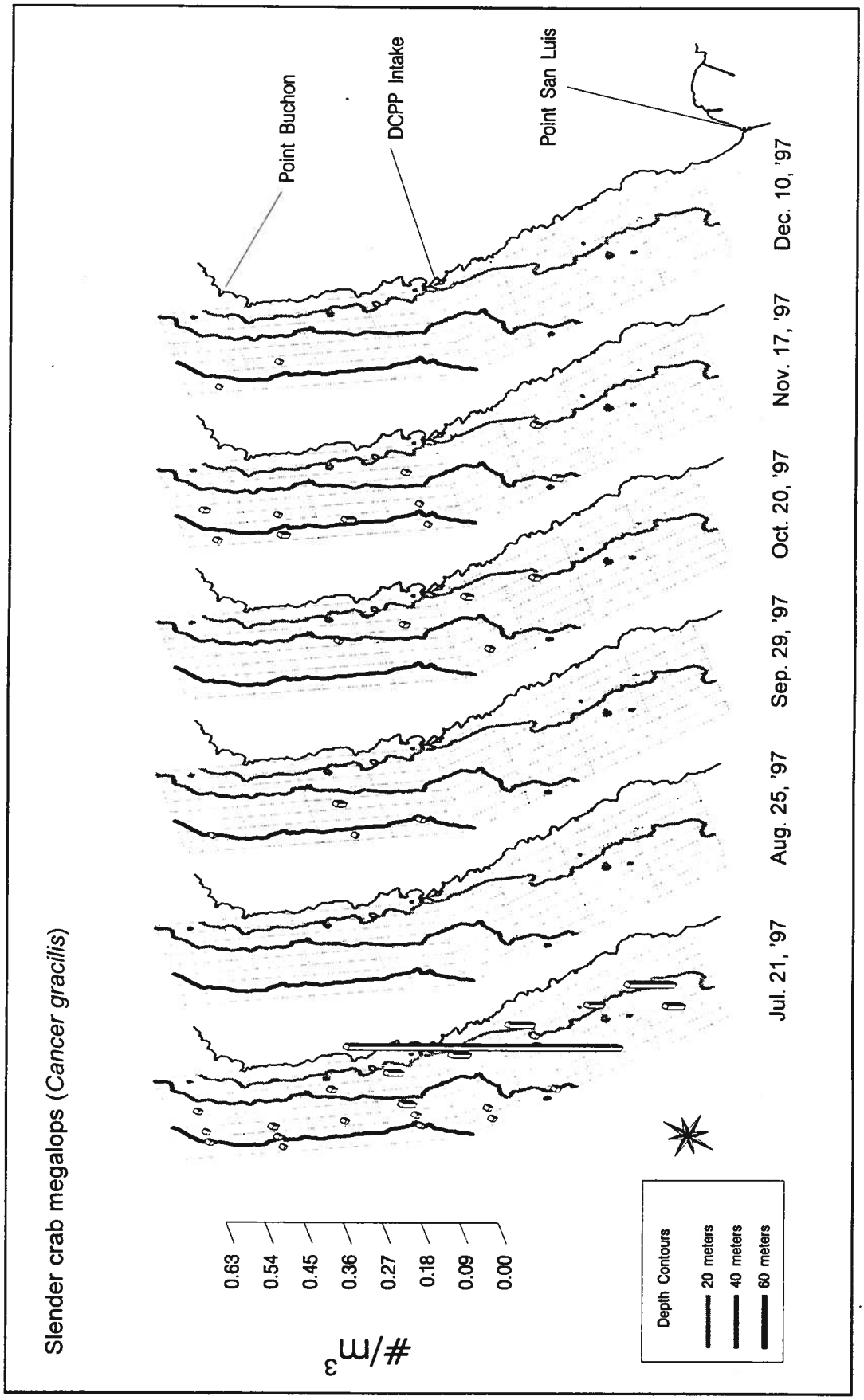


Figure 5.2.1-6a . Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

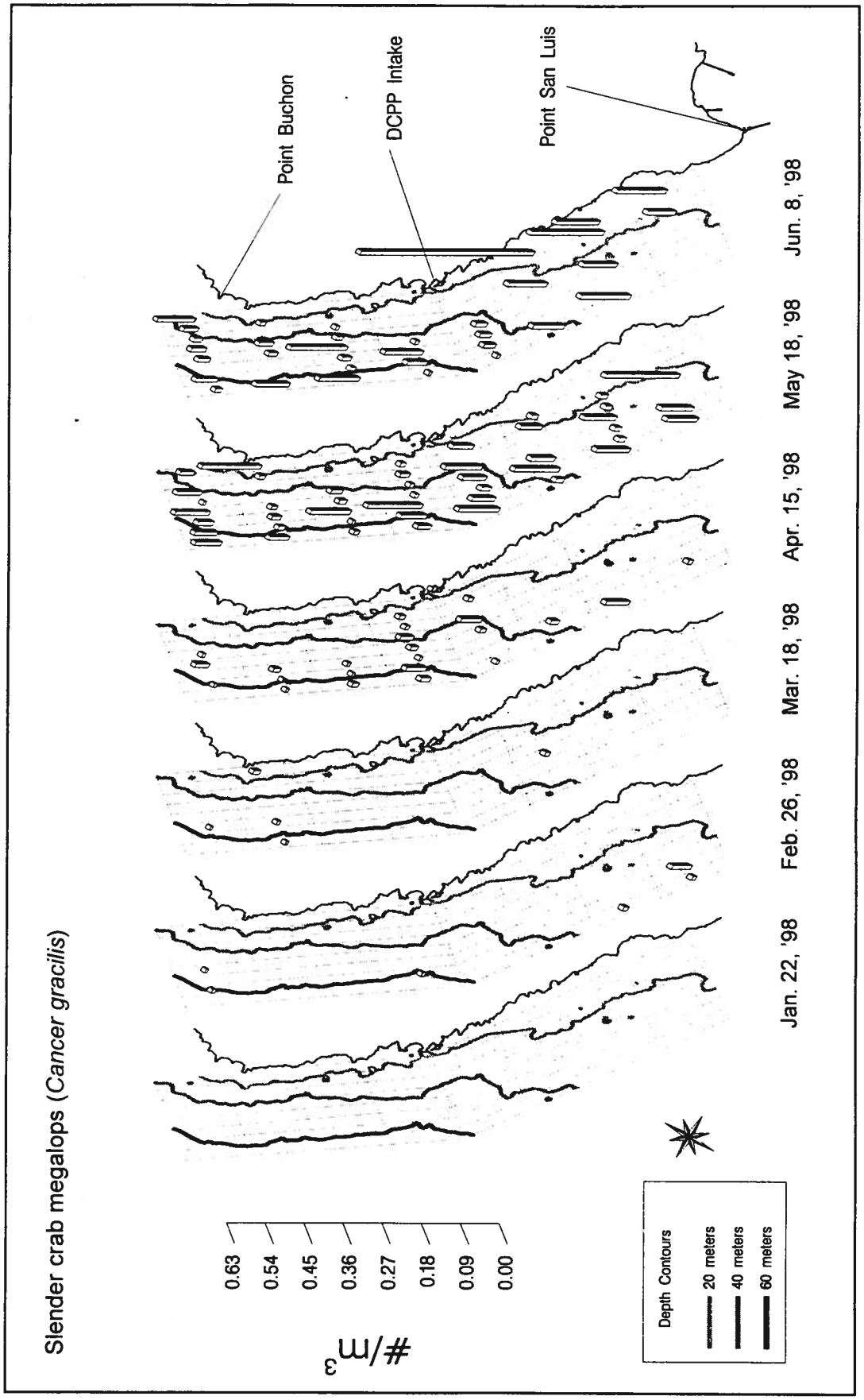


Figure 5.2.1-6b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

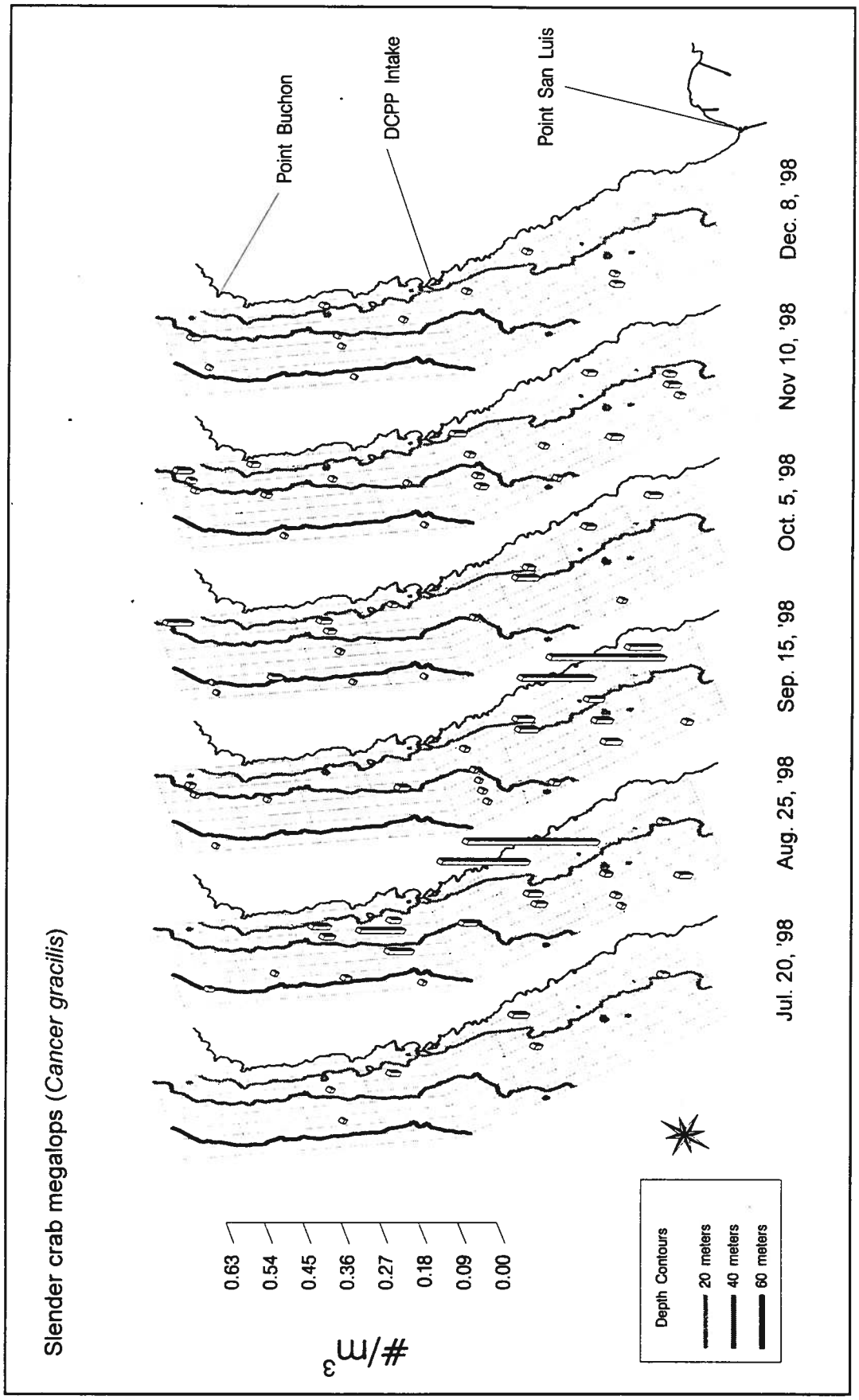


Figure 5.2.1-6c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

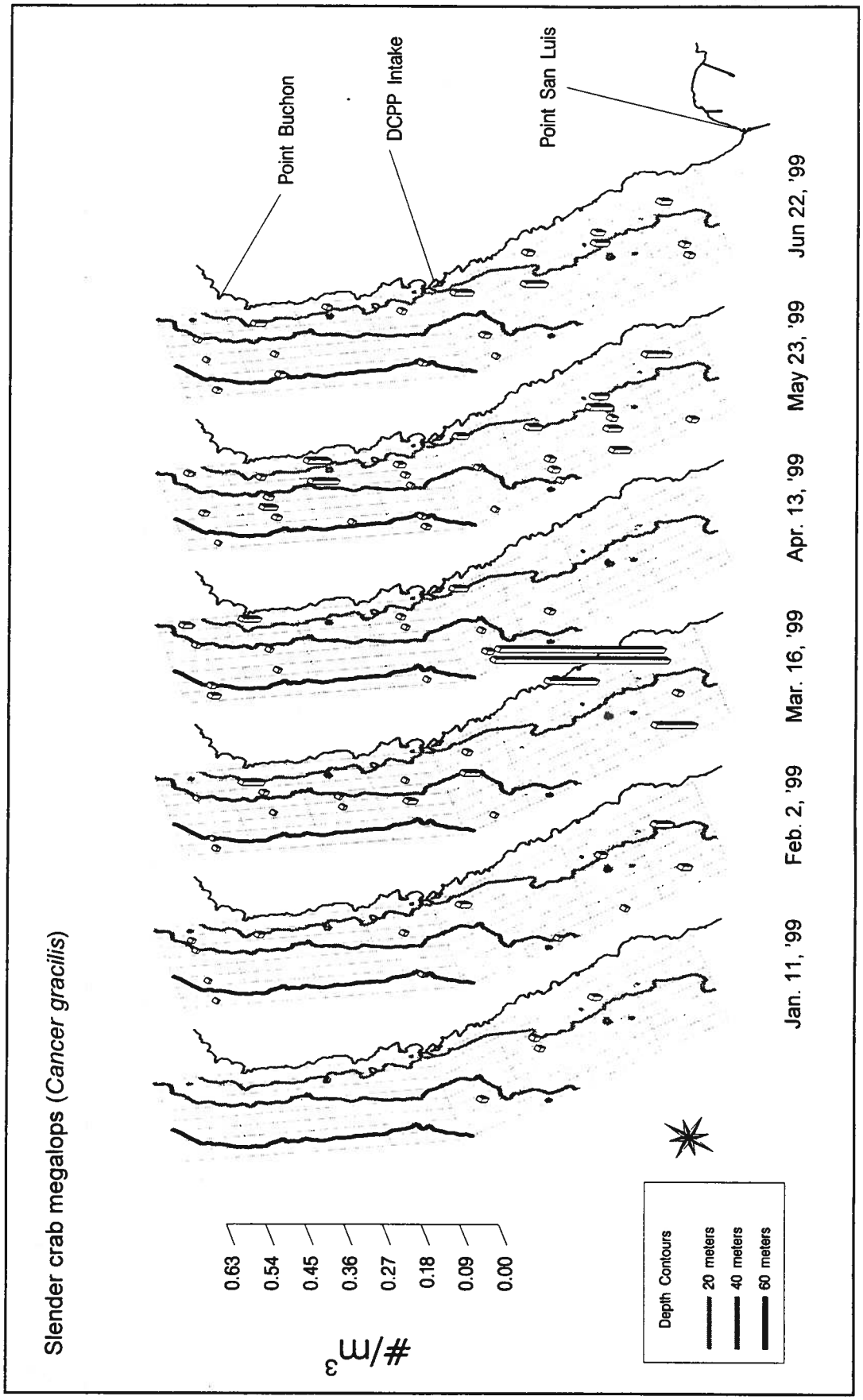


Figure 5.2.1-6d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

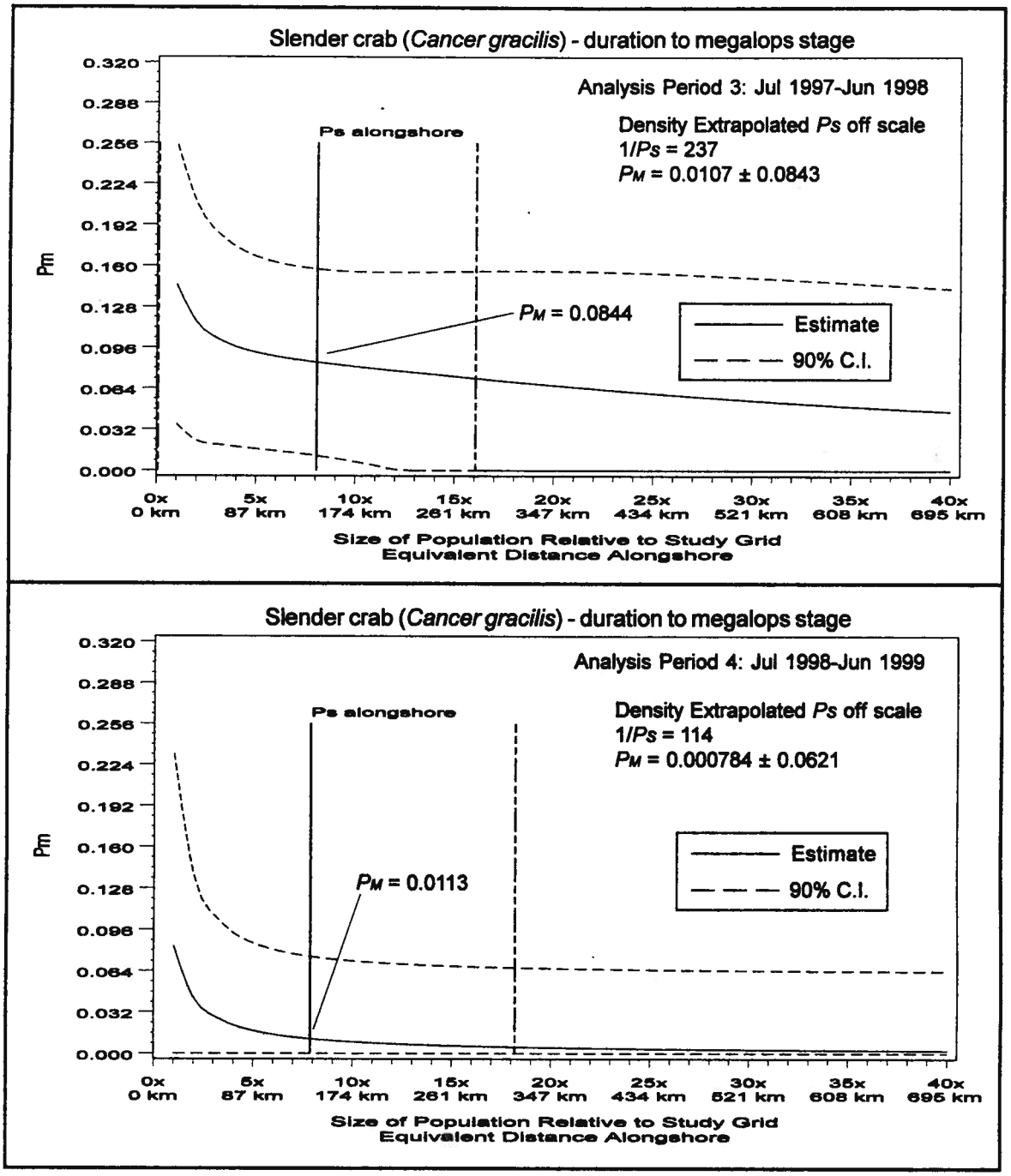


Figure 5.2.1-7. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

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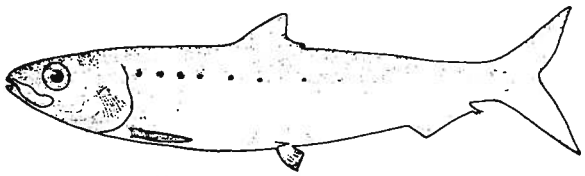
5.2.2 *Assessment of Pacific Sardine* (Sardinops sagax)



Sardinops sagax (Jenyns 1842); Pacific sardine; length to 41 cm but usually less than 30 cm; Kamchatka, Russia to southeast Alaska and Guaymas, Mexico to Peru and Chile; schools over continental shelf, often near shore (Eschmeyer et al. 1983). Blue-green above, white below, series of black spots on back (Miller and Lea 1972).

Pacific sardine is a member of the family Clupeidae (herrings), which is also represented in the waters around Diablo Canyon by American shad (*Alosa sapidissima*), Pacific herring (*Clupea pallasi*), threadfin shad (*Dorosoma petenense*), and round herring (*Etrumeus teres*). The sharp decline of the Pacific sardine population in the mid-1940's led to the demise of the world's largest commercial fishery and to the establishment of the California Cooperative Oceanic Fisheries Investigations (CalCOFI) program (originally named the Cooperative Sardine Research Program) in 1947 (Moser 1996). Recently, the California Department of Fish and Game (CDF&G) issued a press release (January 15, 1999) indicating that the Pacific sardine resource has now fully recovered. Their most recent stock assessment indicated the catch quota be increased from the 1998 quota of 43,574 metric tons (MT) to a 1999 harvest of 120,556 MT (Hill et al. in press). Pacific sardine larvae were not among the ten most abundant larval fish taxa collected in the DCPP study, but are a species of interest because of their recovery status, their recent inclusion in the Fishery Management Plan for Coastal Pelagic Species (PFMC 1998), and their potentially growing fishery.

Pacific sardine spawn pelagic eggs and larvae year round with a fall/winter minimum and a spring/summer maximum (Moser 1996) primarily south of Point Conception to the south of San Diego (Hart 1973). Reproduction is temperature dependent, and the spawning biomass may move north during El Niño years. Age at maturity also may be temperature dependent, with 50% of females maturing at about 16 cm standard length (SL) in southern California (Macewicz et al.

1996) and 50% of the females maturing at about 13 cm off Ensenada, Baja California Norte, Mexico in 1958 during an El Niño year (Ahlstrom 1960). Relatively large proportions of fish at age class 0-yr have reached maturity in both the Southern California Bight and Monterey Bay (**Table 5.2.2-1**; Deriso et al. 1996).

Estimates from previous studies of sardine fecundity range widely. Hart (1973) estimated 30,000–65,000 eggs/batch with large individuals producing 200,000 eggs/yr. Fitch and Lavenberg (1971) reported an estimate of sardine fecundity of 90,000–200,000 eggs/yr. Lo et al. (1996) estimated an average batch fecundity of 24,282 (CV=11%). The highest estimates of annual fecundity from Butler et al. (1993) indicate that Pacific sardine fecundity ranged from 146,754 eggs/two-yr-old female to as many as 2,156,600 eggs for ten-yr and older females (**Table 5.2.2-2**). Fitch and Lavenberg (1971) indicated that Pacific sardine can live to 25 yr, but longevity is more likely about 13 yr according to Butler et al. (1993); however, Leet et al. (1992) indicated that Pacific sardine older than five years were seldom encountered in the fishery; this was supported by data on catch-at-age presented in Hill et al. (in press).

Each year sardines migrate northward early in summer and return south in fall, migrating farther with each year of life. The timing and extent of these migrations are complex and may be affected by oceanographic conditions. Age stratification of the adult population does appear to occur over a latitudinal gradient, with the larger, older fish occurring farther north (Hart 1973). The adult population off the central coast of California generally consists of young adults (2–4 yr) that have migrated from the primary spawning grounds in southern California to feeding grounds in the waters near Diablo Canyon (PFMC 1998).

Age and growth characteristics of Pacific sardine at all life stages have been well described. Larval growth estimated from otoliths has been measured in several temperature regimes (Miller 1952), from which we are able to derive an approximate growth rate for larvae collected in the DCPP study. Growth of the adults has been described with a von Bertalanffy growth function (VBGF: L_∞=205.4 mm ± 1.6 mm SE, k=1.19 ± 0.04 SE, t_0 = 0) by Butler et al. (1996).

Pacific sardine are among the few fishes with age- and stage-specific mortality estimates from the egg stage through later life stages reported in the scientific literature. Instantaneous egg mortality has been estimated as 0.13/d off of Oregon with a CV=243% (Barnes et al. 1992). Lo et al. (1996) produced a similar estimate of embryonic (yolk-sac) mortality of 0.12/d, but with a CV=97%. Butler et al. (1993) modeled the demography of Pacific sardine from the egg stage through the late adult stages with estimates of instantaneous daily natural mortality, the estimated duration of each stage, and daily fecundity. Deriso et al. (1996) modeled the annual fishing mortality of Pacific sardine for the years 1983–1995 (**Table 5.2.2-3**).The natural adult mortality rate in fished populations has been assumed to be 0.4/yr (Murphy 1966; MacCall 1979).

A recent stock assessment of Pacific sardine incorporated fishery harvest statistics collected through 1998 (Hill et al. in press). They indicate that in 1998 the ex-vessel revenue of the commercial sardine fishery (i.e., wetfish) in the state of California totaled $3.5 million, which is down from the $4.1 million generated in 1997. Directed fishery harvests in southern and central California are mostly canned for human consumption and sold overseas, with a small proportion sold fresh for human consumption or animal food (PFMC 1998). The ex-vessel price for sardines in the wetfish fishery ranged from $55–91 per MT, averaging $73.70/MT for January through September of 1998. In addition to the wetfish fishery, a small live bait fishery exists that usually takes <4,540 MT/yr. The overall revenue generated by the live bait fishery exceeds that of the directed fishery because of the higher dollar value for live bait, averaging $817/MT in 1998.

5.2.2.1 Summary of Field Collections

Pacific sardine larvae were present intermittently in the entrainment subsamples during the years 1997–1999 (**Appendix H**), with their greatest abundance occurring during March–May, 1998 (**Figure 5.2.2-1**). There was a total of 2,191 larval Pacific sardines in 230 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999, representing 5% of the subsamples collected and processed from the sampling locations in front of the DCPP intake structure. An El Niño event began during the spring of 1997, influencing oceanographic conditions along the California coast in the fall/winter season of 1997–98 (Lynn et al. 1998;

NOAA 1999), and corresponds with the observed peak in larval Pacific sardine abundance in DCPP subsamples collected in 1998. The spread of warm water during the El Niño may have shifted the sardine spawning biomass north of their primary spawning grounds from south of Point Conception in the Southern California Bight (Hart 1973; Hill et al. in press) to the vicinity of Diablo Canyon.

Larval Pacific sardine occurred in the DCPP study grid during February–July 1998, in December 1998, and again during April–June 1999 (**Appendix H**), with the highest abundance occurring in March and April of 1998 (**Figure 5.2.2-2**). The patterns observed in the study grid further support the supposition that the 1997–98 El Niño may have displaced the Pacific sardine spawning biomass center further north than its distribution in a cooler water year. There was 5,122 larval Pacific sardines identified from 368 bongo net subsamples, representing 12% of the study grid subsamples collected and processed from July 1997–June 1999.

Standard lengths of all Pacific sardine larvae collected at the DCPP intake structure between October 1996 and June 1999 ranged from a minimum of 1.5 mm to a maximum of 25 mm, with corresponding age estimates ranging from 0–94 d (**Figure 5.2.2-3**). The growth rate of larval Pacific sardine (0.24 mm/d; Miller 1952) was used to estimate ages of entrained larvae. The central 98% of this length-frequency distribution resulted in minimum and maximum lengths used for the analyses of 2.4 mm and 19.1 mm, respectively. We assumed that larvae shorter than the minimum length of the central 98% of the distribution were just hatched and, therefore, age 0 d, and we calculated estimated ages from this point. However, reported hatching size for this species ranges from 3.5–3.8 mm (Moser 1996), indicating that the smallest larvae observed represent either natural variation in hatch lengths within the population or the phenomenon of shrinkage following preservation (Theilacker 1980). The possibility remains that all larvae from the observed minimum length of 1.5 mm to the greatest reported hatching length of 3.8 mm (Moser 1996) could have just hatched, leading to gross overestimation of ages for all larvae <3.8 mm. The mean larval length in this distribution was 5.5 mm and approximately 13 d of age based on a minimum length of 2.4 mm.

Mean lengths of Pacific sardine larvae sampled at the DCPP intake structure and from the study grid (**Figure 5.2.2-4**) were not different (*t*-test: p=0.185), but the Kolmogorov-Smirnov goodness of fit test detected a significant difference between the distributions of the two collections (p=0.003). Since mean lengths were not significantly different between the two locations, it was determined that the estimated mean ages were not significantly different either (about 14 d). The length-frequency distributions of Pacific sardine larvae were based on 155 larvae from the intake and 2,882 larvae from the study grid mirroring their adult distribution. It was concluded from these data that the Pacific sardine larvae entrained came from the same population available in the study grid, but that there was a greater proportion of small individuals present in the study grid subsamples as compared with the entrainment subsamples.

5.2.2.2 Estimating Total Annual Entrainment

Annual estimated numbers of Pacific sardine larvae entrained at DCPP increased by greater than three orders of magnitude between Analysis Period 1 (October 1996–September 1997; $\widehat{E_T}$ = 368,000) and Analysis Period 2 (October 1997–September 1998; $\widehat{E_T}$ = 104,000,000: **Table 5.2.2-4**). Estimates for Analysis Periods 2 and 3 (July 1997–June 1998) were the same because they included the same period of 1998 when peak abundance of Pacific sardine larvae occurred. Most of the spawning biomass of Pacific sardine is located south of Point Conception in the Southern California Bight (Hill et al. in press) during non-El Niño years. However, during El Niño events (e.g., fall and winter of 1997–98), warm water moves up the California coast from the south (NOAA 1999), probably carrying southern California spawning stocks north with it. This appeared to be indicated in the waters near Diablo Canyon by the increase in larval Pacific sardine abundance in 1997 and 1998 during Analysis Period 2.

The estimates of annually entrained Pacific sardine larvae were adjusted (**Table 5.2.2-5**) to the long-term average Intake Cove surface plankton tow index ($\overline{I} / I_i$) calculated as the ratio between the 9 yr average ($\overline{I}$) of Intake Cove sampling (**Figure 5.2.2-5**) and the average annual index estimated from these same tows during the year being adjusted (I_i). The average indices for Analysis Period 1 and Analysis Periods 2 and 3 were <0.001 and 0.070 larvae/m^3, respectively,

and the long-term average index for 1990–98 was 0.0152 larvae/m^3. Thus, the ratios used to adjust the Analysis Period 1 and Analysis Period 2 estimates of larvae entrained were 23.0 and 0.217, respectively. The same trends in overall abundance as noted for unadjusted entrainment values are apparent in the adjusted values: larval Pacific sardine were less abundant in Analysis Period 1 ($\hat{E}_{Adj-T}$ = 8,470,000) than in Analysis Periods 2 and 3 ($\hat{E}_{Adj-T}$ = 22,600,000). However, the adjustment to the long-term average had the effect of reducing the difference in abundance between years from three orders of magnitude to less than one. The adjusted CV ranged from 28% in Analysis Period 1, reflecting the low and patchy abundance during the period, to 7% in Analysis Periods 2 and 3 when the larvae were found in greater abundance and were more evenly distributed.

5.2.2.3 Fecundity Hindcasting (*FH*)

Fecundity hindcasting requires age-specific fecundity (M_X) and mortality (L_X) to assess entrainment effects. Butler et al. (1993) modeled egg and larval survivorship of Pacific sardine for several life stages (**Table 5.2.2-2**). Their 'best' estimate is derived by fitting the range of field-estimated mortalities to the assumption of a stable and stationary population. Instantaneous mortality rates from Butler et al. (1993) were used to describe survivorship of egg (0.720), yolk-sac (0.670), and early larval stages (0.242) of Pacific sardine. Applying these rates over their respective average stage durations (Butler et al. 1993), they can then be used to estimate finite survivorship for each developmental stage: 0.165 (egg), 0.125 (yolk-sac), and 0.077 (early larvae). The survival rate from egg stage to entrainment (0.002) was then calculated from an average age at entrainment of 16 d, estimated using a growth rate of 0.24 mm/d (Miller 1952), and an average and smallest size at entrainment of 5.5 mm and 1.5 mm, respectively.

The *FH* approach combines larval entrainment losses and adult fecundity to hindcast the numbers of adult females effectively removed from the reproductively active population. For Pacific sardine, there is substantial regional variation in size-at-age, and ages generally increase from north to south (Phillips 1948). Size and age at maturity may decline with a decrease in biomass, but latitude and temperature are also important (Butler 1987). At low biomass levels, MacCall

(1979) reports that all 1 yr old sardines appear to be reproductively mature, whereas at high biomass levels some 2 yr old fish have yet to reach maturity. Butler et al. (1993) also developed models using age at first maturity as 2 yr. Geographic distribution, however, influenced maturity as Deriso et al. (1996) estimated that 58% of age 0 sardines (<1 yr) were mature in Monterey Bay and 65% mature in the Southern California Bight in 1995 (**Table 5.2.2-1**). In subsequent calculations, we will assume that age at first maturity is 2 yr as modeled in Butler et al. (1993).

Pacific sardines may live as long as 13 yr (Butler et al. 1993; Love 1996), but Deriso et al. (1996) and Hill et al. (in press) report that the oldest fish in their samples from off the California coast were aged 9 yr. Leet et al. (1992) reports that fish older than 5 yr are seldom caught in the fishery, and this is supported by the catch-at-age data reported in Hill et al. (in press) that extends through June 1998.

Since both longevity and age at maturation are biomass dependent (Hart 1973) and likely changing in this recovering population (Hill et al. in press), we will use what appears to be the fishery longevity (i.e., 5 yr; Leet et al. 1992; Hill et al. in press) to calculate total lifetime fecundity (*TLF*). If 5 yr is an underestimate of the actual reproductive life span of this fish, then it represents an assumption that reduces the total lifetime fecundity and thus may overestimate the impact of entrainment losses on adult females. The average fecundity from age-2 to age-5 modeled by Butler et al. (1993) is 496,018 eggs per female per year. This fecundity is multiplied by one half the difference between the fishery-modified longevity and age at maturation yielding

$$\widehat{TLF} = 496{,}018 \cdot \left(\frac{5-2}{2}\right) = 744{,}027 .$$

The adjusted total larval entrainment for Pacific sardine ($\hat{E}_{Adj-T}$) was used to estimate the number of breeding females needed to produce the number of larvae entrained (**Table 5.2.2-5**). The estimated number of breeding females ($\widehat{FH}$) whose fecundity equals the estimated total loss of entrained larvae is calculated assuming age of maturation is 2 yr and longevity in the fishery is 5 yr (**Table 5.2.2-6**). The number of adult females hindcast from the larvae entrained at DCPP during Analysis Period 1 ($\widehat{FH} = 3{,}170$) was less than one-half of the adult females hindcast during Analysis Periods 2 and 3 ($\widehat{FH} = 8{,}460$).

Sensitivity Analysis

Two parameters, survivorship and fecundity, had the greatest leverage on recalculated values of $\widehat{FH}$ (**Table 5.2.2-7**). Varying either of these two parameters in the model resulted in an $\widehat{FH}$ range of 613–43,800 adult females for Analysis Period 1 and Analysis Periods 2 and 3, compared to the original estimate of 3,170–8,460 adult females. Maturation and longevity had the next greatest amount of leverage on recalculated $\widehat{FH}$.

5.2.2.5 Adult Equivalent Loss (*AEL*)

Similar to the $\widehat{FH}$ results, estimates of adult equivalents lost (AEL) due to larval entrainment were smaller during Analysis Period 1 (2,630) than during Analysis Periods 2 and 3 (7,000) (**Table 5.2.2-8**). The $\widehat{AEL}$ of 2,630 adults predicted from $\hat{E}_{Adj-T}$ at DCPP during Analysis Period 1 reflects the low abundance of Pacific sardine larvae during this period. As indicated from the long term Intake Cove surface tows, these low abundance estimates represent a more typical year relative to the following years that appeared to be influenced by El Niño conditions. $\widehat{AEL}$ values were almost three times higher during Analysis Periods 2 and 3 (i.e., 7,000 adult equivalents) when spawning stocks generally found south of Point Conception were likely displaced northward as warm El Niño waters spread up the California coast.

Sensitivity Analysis

Late larval survivorship had the greatest leverage on recalculated values of $\widehat{AEL}$ (**Table 5.2.2-9**). Varying this parameter in the model resulted in an $\widehat{AEL}$ ranges of 507–36,300 adult equivalents for Analysis Period 1 and Analysis Periods 2 and 3, compared to the original estimate of 2,630–7,000 adult equivalents. Early juvenile survivorship had the next greatest leverage on $\widehat{AEL}$, while varying the estimate of entrainment did not substantially change the original estimate of $\widehat{AEL}$.

5.2.2.6 Empirical Transport Model (*ETM*)

Monthly estimates of proportional entrainment (PE_i) in each *i*th survey from the two years sampled ranged from a minimum of zero to a maximum of 0.859±0.911 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in January 1999 (**Table 5.2.2-10**). When both $\widehat{PE_i}$ and the annual proportion of larvae hatched in the *i*th

survey period ($\hat{f}_i$) were equal to zero, no larvae were collected at either the DCPP intake nor from the study grid. When $\widehat{PE}_i$ =0 and $\hat{f}_i$>0, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hr study grid survey. For Pacific sardine, the relatively large $\widehat{PE}_i$ for January 1999 was accompanied by an f_i = 1 since this was the only month in which Pacific sardine were collected during Analysis Period 4. In addition, there were no Pacific sardine larvae collected in the study grid during the paired 72-hour sampling period. A *PE* was calculated for the survey period because the larval density from the weekly entrainment sample (**Appendix H**) was used to estimate the densities in the unsampled volume inshore of the two cells nearest the intake (D1 and E1). As a result no estimates of larval mortality due to entrainment (P_M) were calculated for Analysis Period 4.

By varying the estimated proportion of the sardine population in the study grid (P_S) over the mean and maximum larval durations, estimates of $\widehat{P_M}$ ranged over nearly an order of magnitude during Analysis Period 3 (**Figure 5.2.2-6**). The differences in $\widehat{P_M}$ resulted from longer larval duration and consequently greater alongshore and onshore transport during the period which decreased P_S, although it increased time the larvae were susceptible to entrainment. The $\widehat{P_M}$ values calculated from offshore extrapolated abundance extended alongshore were small regardless of the larval duration used (i.e., 0.0000669 and 0.000284 for the maximum and mean durations, respectively). The extrapolations developed in this analysis indicated that sardine larvae were at risk over an area ranging from 938 km^2 to 14,800 km^2 (based on 1997–1998 data of current velocities, gradients in larval distribution, and durations at risk for mean and maximum larval sizes entrained, respectively). These areas represent 0.6–9.1%, respectively, of the total area of the 1998 spawning population of Pacific sardine extending from San Francisco Bay to Ensenada, Mexico (based on values reported by Hill et al. [in press] and the PSMFC database; **Figure 5.2.2-7**).

5.2.2.7 Interpretation of Assessment Results

Estimates of entrainment losses using *AEL* estimates can be used to estimate effects on local harvest of Pacific sardine. An estimated annual average equivalent loss of approximately 5,000 adult sardines (*AEL*) at DCPP translates into 0.625 MT/yr using an average weight of 125 g for

two year old sardines (Hill et al. in press). Assuming, conservatively, that all of these sardines were vulnerable to the fishery, approximately 10% of this biomass would be directed to the live bait fishery (average value of $817/MT) and the remainder would be directed to the wetfish fishery (average value of $74/MT). These amounts translate into $51/yr from the live bait fishery and $42/yr from the wetfish fishery.

Alternatively, the estimates of annual entrainment mortality probability ($\widehat{P_M}$ ranges from 0.0000669–0.000284) can be applied to the proportion of the recent stock size estimates at risk to entrainment assuming no compensation. Hill et al. (in press) report that approximately 337,596 MT of Pacific sardine were present in the area from Monterey Bay to San Diego. The proportion of this amount to which $\widehat{P_M}$ would be applied would be 16,900 MT, which is approximately 5% (the average area of the spawning population potentially affected by entrainment, based on the two estimates using maximum and mean larval durations) of the total biomass within the area from San Diego to Monterey. Using the larger more conservative P_M estimate yields a potential loss of 4.8 MT of Pacific sardine as a result of entrainment. By apportioning the valuation of catches as in the *AEL* loss estimates, 4.8 MT represents direct annual dollar value losses of $390 to the live bait fishery and $320 to the wetfish fishery.

These estimates indicate that DCPP entrainment effects represent minimal risk to local or regional populations of Pacific sardine. During the period of power plant operations that began in 1985, adult stocks of Pacific sardine have fully recovered (Hill et al. in press) from their population crash in the late 1940's and early 1950's (PFMC 1998), and commercial catches have steadily increased (Starr et al. 1998). Pacific sardine larvae occurred episodically in plankton samples collected at DCPP during the present study and in data sets spanning a larger time frame. There is little possibility that any additional larval mortality due to entrainment of this species at DCPP is affecting the adult stocks when one considers the uncommon occurrence of Pacific sardine larvae in plankton tows near DCPP combined with the trends of increasing abundance and catch.

Table 5.2.2-1. Percent maturity of female Pacific sardines (*Sardinops sagax*) in the Southern California Bight and in Monterey Bay (Deriso et al. 1996).

Age (yr)	% Mature (Southern California Bight)	% Mature (Monterey, California)
0	65	58
1	87	79
2	96	91
3	99	97
4	100	99
5+	100	100

Table 5.2.2-2. Life table for Pacific sardine (*Sardinops sagax*): a) Age-specific fecundity schedule (M_X=natality rate; L_X=survivorship) and b) stage-specific survivorship schedule (Z=instantaneous daily mortality; S=finite survival rate) modified from Butler et al. (1993).

a) Age-specific fecundity

Age (yr)	M_X	L_X	M_XL_X
1	0	1,000	0
2	146,754	670	98,325,180
3	388,188	449	174,296,412
4	599,640	301	180,491,640
5	849,490	202	171,596,980
6	1,167,457	135	157,606,695
7	1,487,528	91	135,365,048
8	1,617,450	61	98,664,450
9	1,887,025	41	77,368,025
10	2,156,600	27	58,228,200
11	2,156,600	18	38,818,800
12	2,156,600	12	25,879,200
13	2,156,600	8	17,252,800

(continued)

fecundity

Table 5.2.2-2. (continued)

b) Stage-specific survivorship

Stage	Z_{min}	Z_{best}	Z_{max}	Duration (d)	Cumulative	Duration (d)	S_{max}	S_{best}	S_{min}	CV_{best}
Egg	0.3100	0.7200	2.1200	3		3	0.4607	0.1653	0.0050	0.4595
Yolk-sac larva	0.3940	0.6698	0.9710	3	6	3	0.2948	0.1254	0.0493	0.3264
Early larva	0.1423	0.2417	0.3502	11	17	7.26	0.356	0.173	0.0788	0.267
Survivorship from egg to entrainment:								0.0036		
Early larva	0.1423	0.2417	0.3502	11	17	3.74	0.587	0.4047	0.270	0.131
Late larva	0.0570	0.0964	0.1390	35	52	35	0.1360	0.0343	0.0077	0.6243
Early juvenile	0.0290	0.0560	0.0810	25	77	25	0.4843	0.2466	0.1320	0.2381
Juvenile I	0.0116	0.0197	0.0285	50	127	50	0.5599	0.3734	0.2405	0.1425
Juvenile II	0.0023	0.0040	0.0058	110	237	110	0.7765	0.6440	0.5283	0.0642
Juvenile III	0.0016	0.0028	0.0040	146	383	146	0.7917	0.6644	0.5577	0.0587
Juvenile IV	0.0012	0.0022	0.0032	170	553	170	0.8155	0.6880	0.5804	0.0569
Pre-recruit	0.0006	0.0011	0.0015	175	728	175	0.9003	0.8249	0.7691	0.0265
Survivorship from entrainment to recruitment:								0.0003		

Table 5.2.2-3. Estimated fishing mortality rates (*F*) on Pacific sardine (*Sardinops sagax*) from 1983–1995. Data from Deriso et al. (1996).

Age (yr)	Estimated *F* Range
0	0.00–0.04
1	0.01–0.30
2	0.04–0.71
3	0.04–1.08
4	0.03–1.05
5	0.03–0.96

Table 5.2.2-4. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for Pacific sardine (*Sardinops sagax*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	368,000	103,000
2) Oct 1, 1997–Sep 30, 1998	104,000,000	7,430,000
3) Jul 1, 1997–Jun 30, 1998	104,000,000	7,430,000

Table 5.2.2-5. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for Pacific sardine (*Sardinops sagax*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	8,470,000	2,360,000
2) Oct 1, 1997–Sep 30, 1998	22,600,000	1,610,000
3) Jul 1, 1997–Jun 30, 1998	22,600,000	1,610,000

Table 5.2.2-6. Estimated number of Pacific sardine (*Sardinops sagax*) adult females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{FH})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	3,170	12,100	1,690,000	6
2) Oct 1, 1997–Sep 30, 1998	8,460	32,300	4,490,000	16
3) Jul 1, 1997–Jun 30, 1998	8,460	32,300	4,490,000	16

Table 5.2.2-7. Pacific sardine (*Sardinops sagax*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 3,170				
$\hat{E}_{Adj-T}$	8,470,000	1,720	4,630	2,910
$\hat{S}_{egg}$	0.165	613	16,400	15,800
$\hat{S}_{yolk\text{-}sac}$	0.125	613	16,400	15,800
$\hat{S}_{larvae}$	0.173	613	16,400	15,800
# Eggs/yr	496,000	613	16,400	15,800
Longevity	5	1,190	6,350	5,160
Maturation	2	2,380	9,520	7,140

Table 5.2.2-7 (continued). Pacific sardine (*Sardinops sagax*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 8,460				
$\hat{E}_{Adj-T}$	22,600,000	7,460	9,450	1,990
$\hat{S}_{egg}$	0.165	1,630	43,800	42,200
$\hat{S}_{yolk\text{-}sac}$	0.125	1,630	43,800	42,200
$\hat{S}_{larvae}$	0.173	1,630	43,800	42,200
# Eggs/yr	496,000	1,630	43,800	42,200
Longevity	5	3,170	16,900	13,700
Maturation	2	6,340	25,400	19,000

c) Analysis Period 3: July, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 8,460				
$\hat{E}_{Adj-T}$	22,600,000	7,470	9,450	1,990
$\hat{S}_{egg}$	0.165	1,630	43,800	42,200
$\hat{S}_{yolk\text{-}sac}$	0.125	1,630	43,800	42,200
$\hat{S}_{larvae}$	0.173	1,630	43,800	42,200
# Eggs/yr	496,000	1,630	43,800	42,200
Longevity	5	3,170	16,900	13,700
Maturation	2	6,350	25,400	19,000

Table 5.2.2-8. Estimated number of equivalent Pacific sardine (*Sardinops sagax*) adults ($\widehat{AEL}$) equal to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{AEL})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{AEL}$	$\widehat{SE}(\widehat{AEL})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	2,630	10,300	1,650,000	4
2) Oct 1, 1997–Sep 30, 1998	7,000	27,400	4,390,000	11
3) Jul 1, 1997–Jun 30, 1998	7,000	27,400	4,390,000	11

Table 5.2.2-9. Pacific sardine (*Sardinops sagax*): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645$ $\widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 2,630				
$\hat{E}_{Adj-T}$	8,470,000	1,420	3,830	2,410
$\hat{S}$ *early larvae*	0.405	507	6,490	5,980
$\hat{S}$ *late larvae*	0.0343	507	13,600	13,100
$\hat{S}$ *early juvenile*	0.247	507	10,700	10,100
$\hat{S}$ *juv. I*	0.373	507	7,030	6,530
$\hat{S}$ *juv. II*	0.644	507	4,080	3,570
$\hat{S}$ *juv. III*	0.664	507	3,950	3,450
$\hat{S}$ *juv. IV*	0.688	507	3,820	3,310
$\hat{S}$ *pre-recruit*	0.825	507	3,180	2,680

Table 5.2.2-9 (continued). Pacific sardine (*Sardinops sagax*): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL} = 7{,}000$				
$\hat{E}_{Adj-T}$	22,600,000	6,180	7,820	1,650
$\hat{S}$ *early larvae*	0.405	1,350	17,300	15,900
$\hat{S}$ *late larvae*	0.0343	1,350	36,300	34,900
$\hat{S}$ *early juvenile*	0.247	1,350	28,400	27,000
$\hat{S}$ *juv. I*	0.373	1,350	18,700	17,400
$\hat{S}$ *juv. II*	0.644	1,350	10,900	9,520
$\hat{S}$ *juv. III*	0.664	1,350	10,500	9,180
$\hat{S}$ *juv. IV*	0.688	1,350	10,200	8,820
$\hat{S}$ *pre-recruit*	0.825	1,350	8,490	7,130

Table 5.2.2-9 (continued). Pacific sardine (*Sardinops sagax*): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 7,000				
$\hat{E}_{Adj-T}$	22,600,000	6,180	7,830	1,650
$\hat{S}$ *early larvae*	0.405	1,350	17,300	16,000
$\hat{S}$ *late larvae*	0.0343	1,350	36,300	34,900
$\hat{S}$ *early juvenile*	0.247	1,350	28,400	27,000
$\hat{S}$ *juv. I*	0.373	1,350	18,800	17,400
$\hat{S}$ *juv. II*	0.644	1,350	10,900	9,520
$\hat{S}$ *juv. III*	0.664	1,350	10,500	9,190
$\hat{S}$ *juv. IV*	0.688	1,350	10,200	8,830
$\hat{S}$ *pre-recruit*	0.825	1,350	8,490	7,140

Table 5.2.2-10. Pacific sardine (*Sardinops sagax*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) within an analysis period (e.g., July 1, 1997–June 30, 1998) with associated standard errors (*SE*) used in estimating entrainment mortality for two years of paired entrainment and study grid surveys.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}(\widehat{PE_i})$	$\hat{f}_i$	$\widehat{SE}(\hat{f}_i)$
Jul 21, 1997	0	0	0	0
Aug 25, 1997	0	0	0	0
Sep 29, 1997	0	0	0.00091	0.00024
Oct 20, 1997	0	0	0	0
Nov 17, 1997	0	0	0	0
Dec 10, 1997	0	0	0	0
Jan 22, 1998	0	0	0.00043	0.00015
Feb 26, 1998	0	0	0	0
Mar 18, 1998	0.00092	0.00011	0.066	0.00254
Apr 15, 1998	0.00084	0.00021	0.123	0.00538
May 18, 1998	0.00096	0.00032	0.802	0.00676
Jun 8, 1998	0.00199	0.0007	0.00784	0.00107

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}(\widehat{PE_i})$	$\hat{f}_i$	$\widehat{SE}(\hat{f}_i)$
Jul 21, 1998	0	0	0	0
Aug 26, 1998	0	0	0	0
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0	0	0	0
Jan 12, 1999	0.859	0.911	1	–
Feb 3, 1999	0	0	0	0
Mar 17, 1999	0	0	0	0
Apr 14, 1999	0	0	0	0
May 24, 1999	0	0	0	0
Jun 23, 1999	0	0	0	0

"–" = no estimate of standard error because only one fish was collected during this survey period

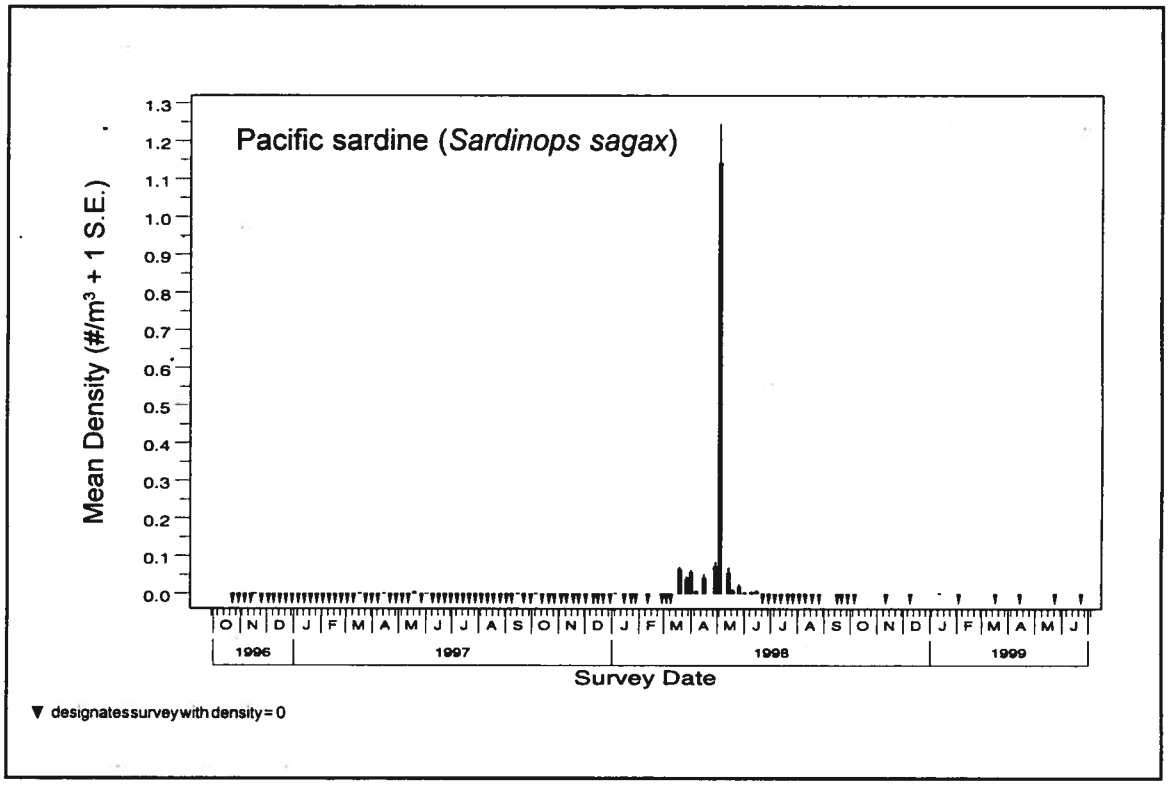


Figure 5.2.2-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

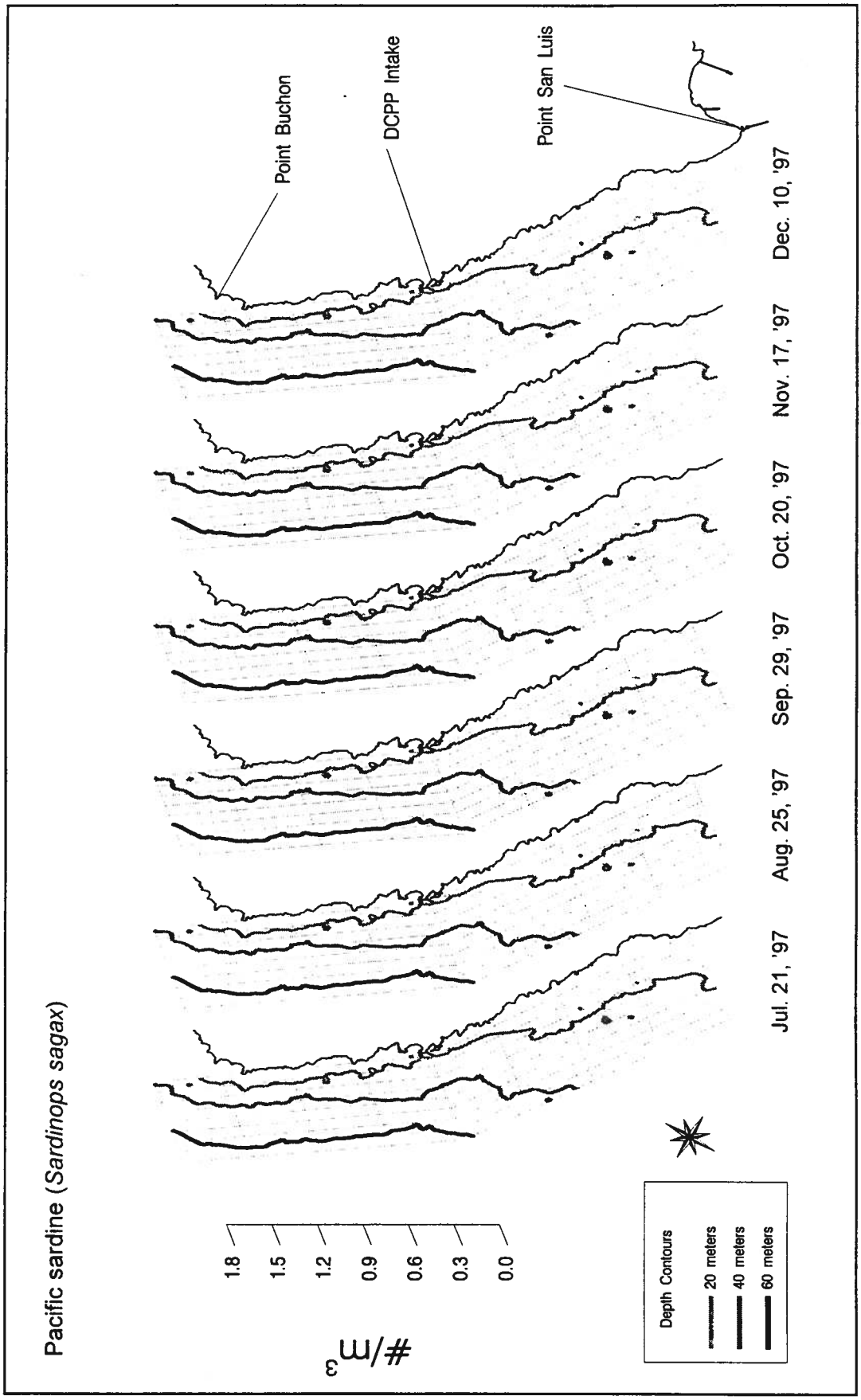


Figure 5.2.2-2a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

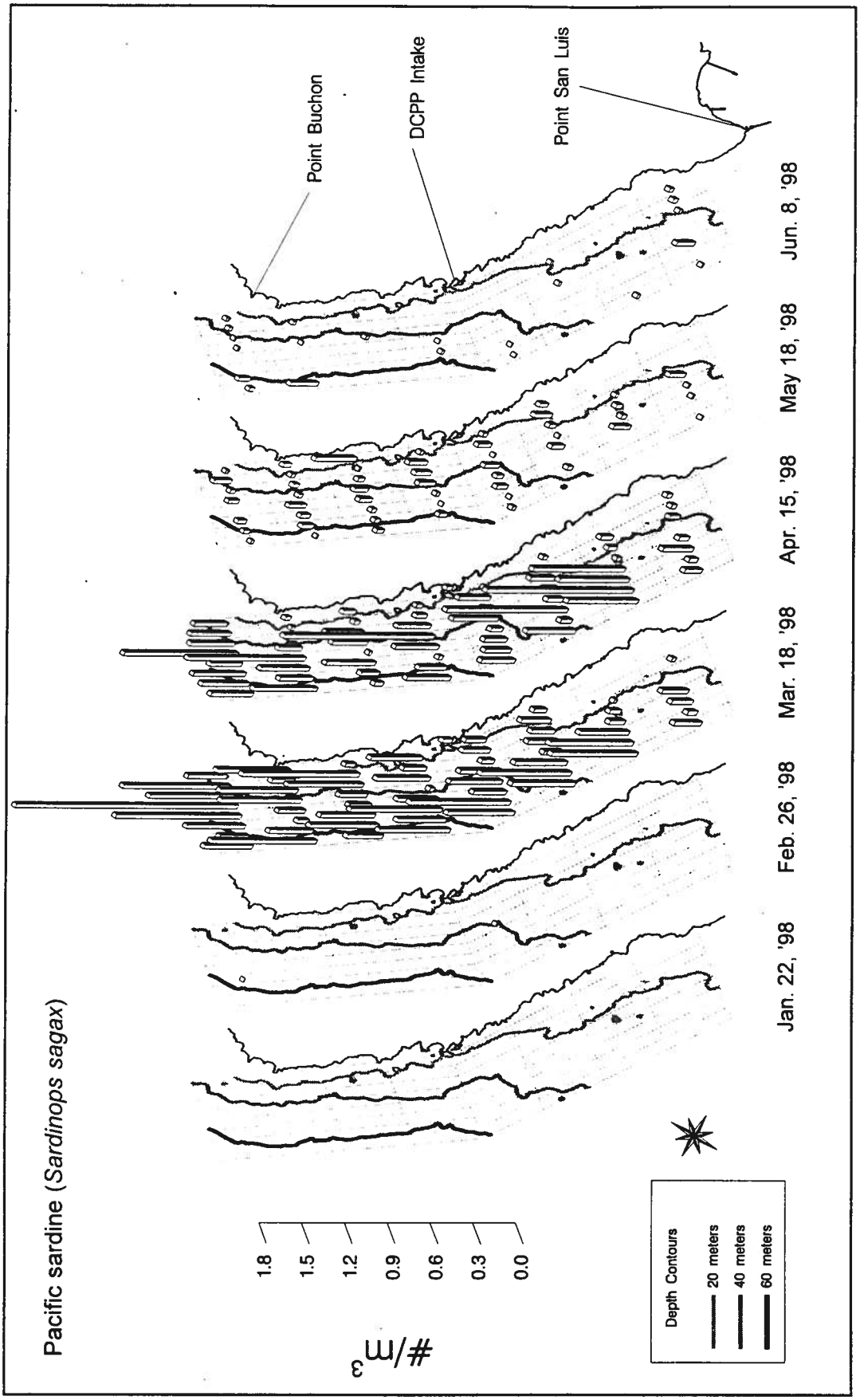


Figure 5.2.2-2b (continued). Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

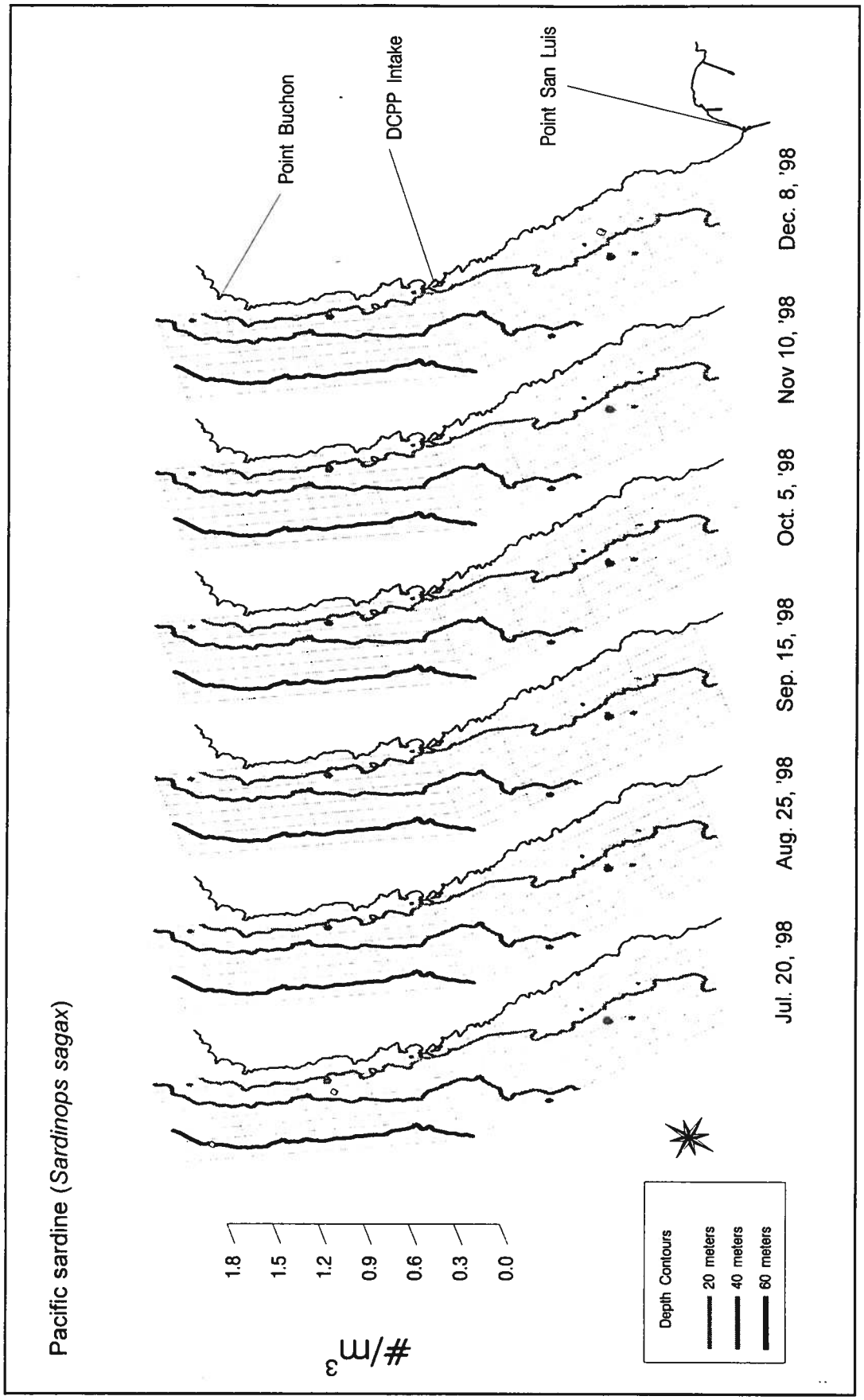


Figure 5.2.2-2c (continued). Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

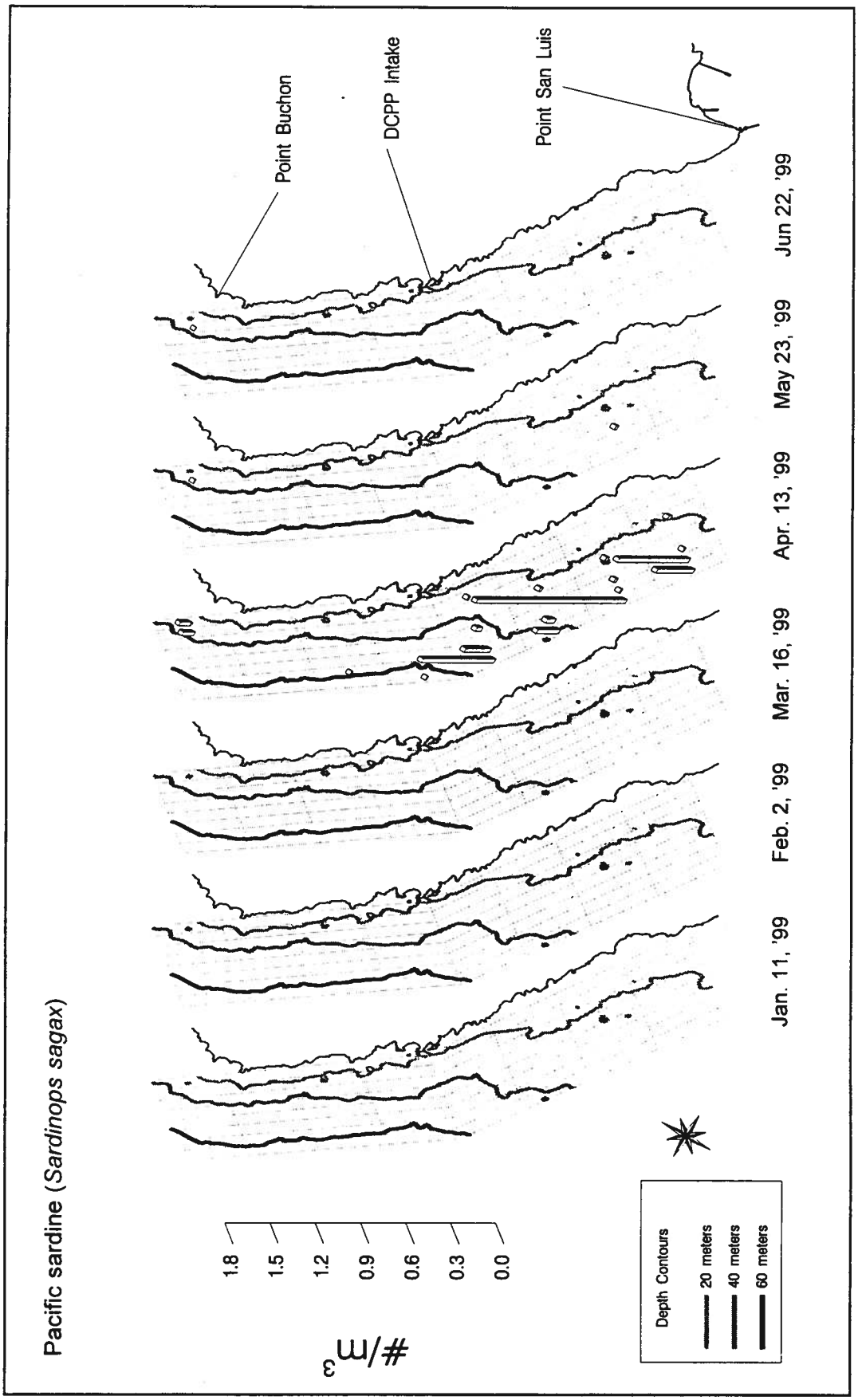


Figure 5.2.2-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

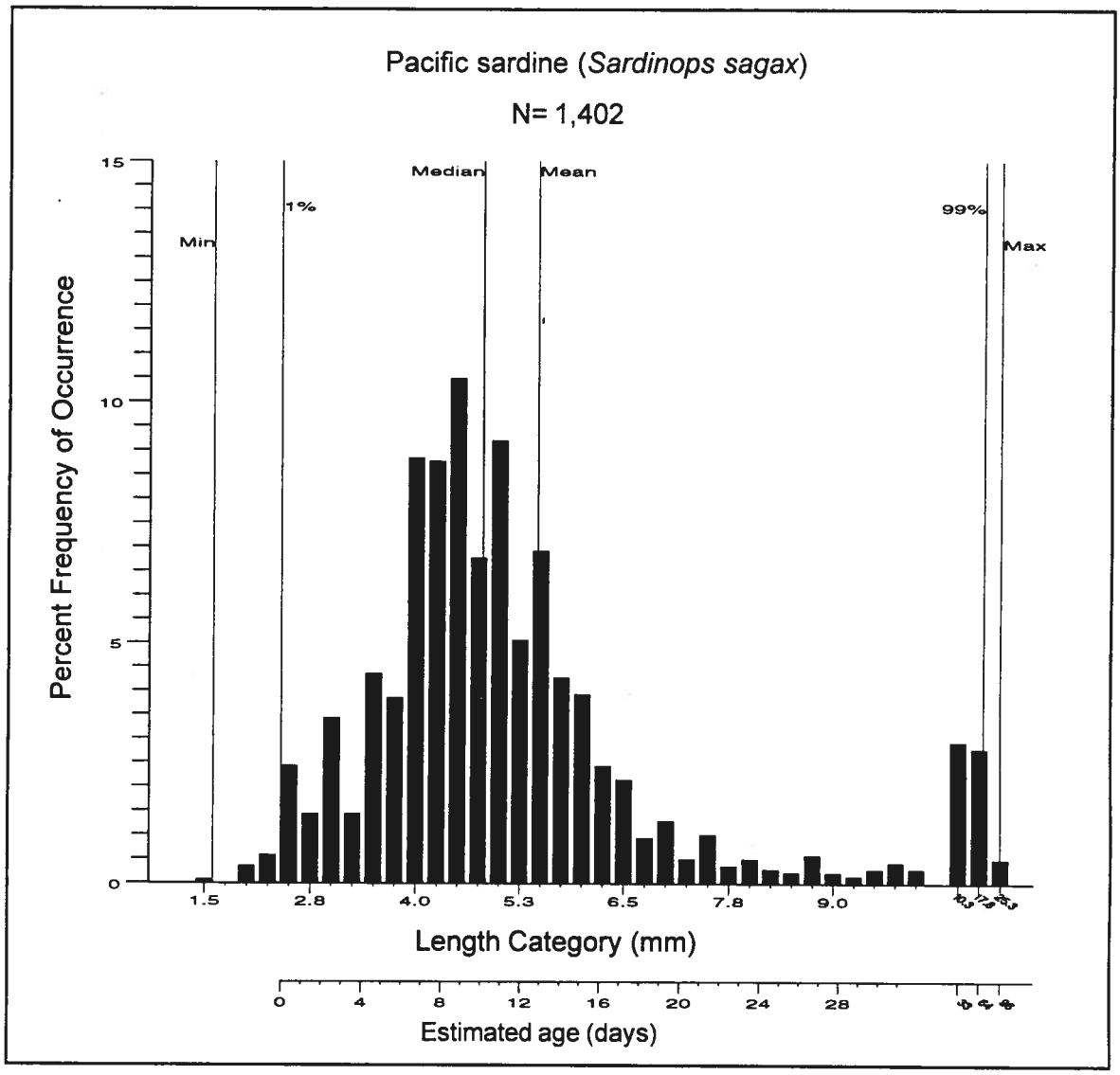


Figure 5.2.2-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

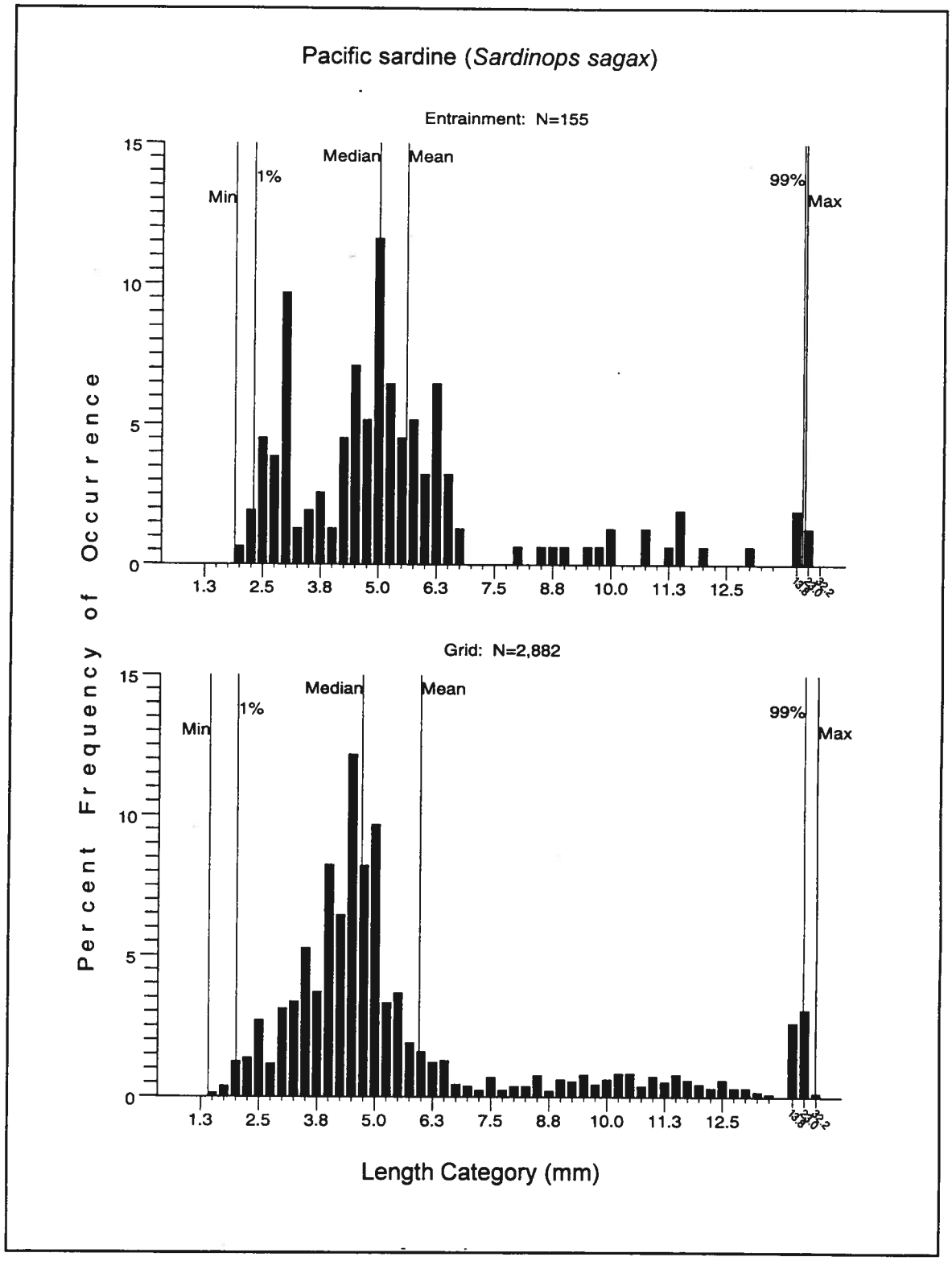


Figure 5.2.2-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

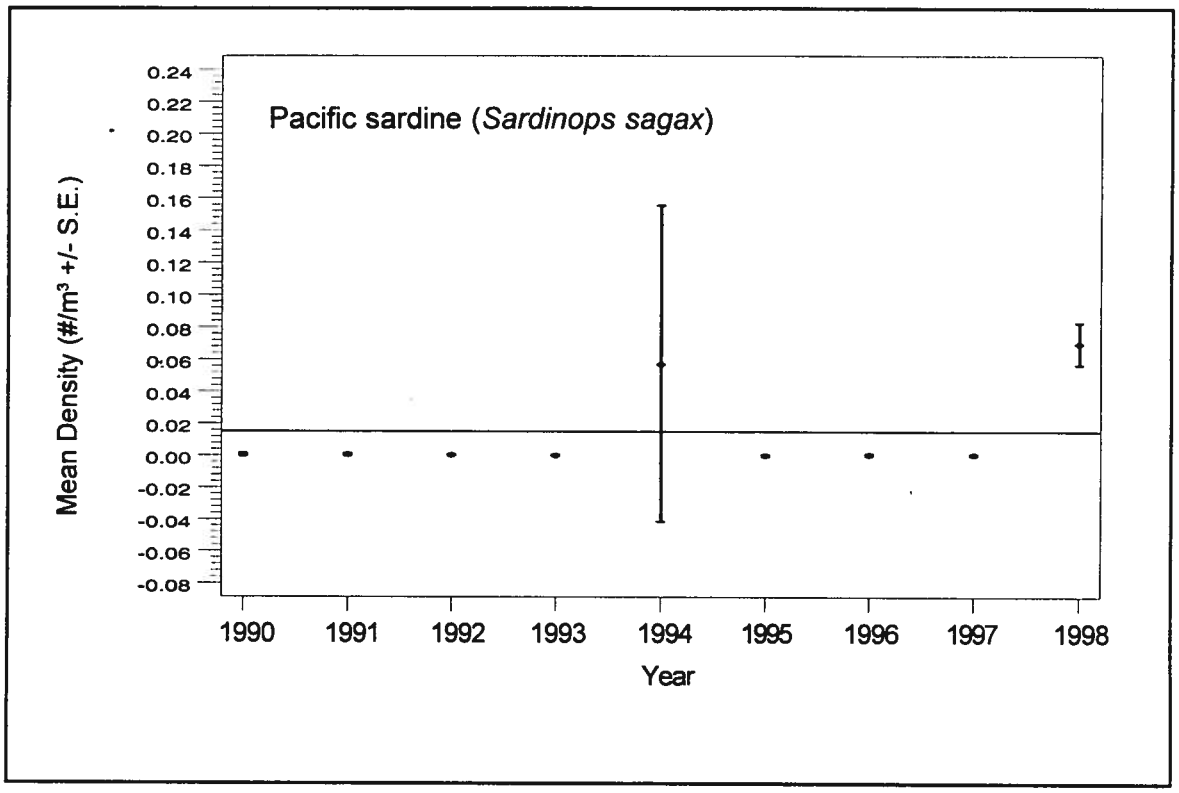


Figure 5.2.2-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

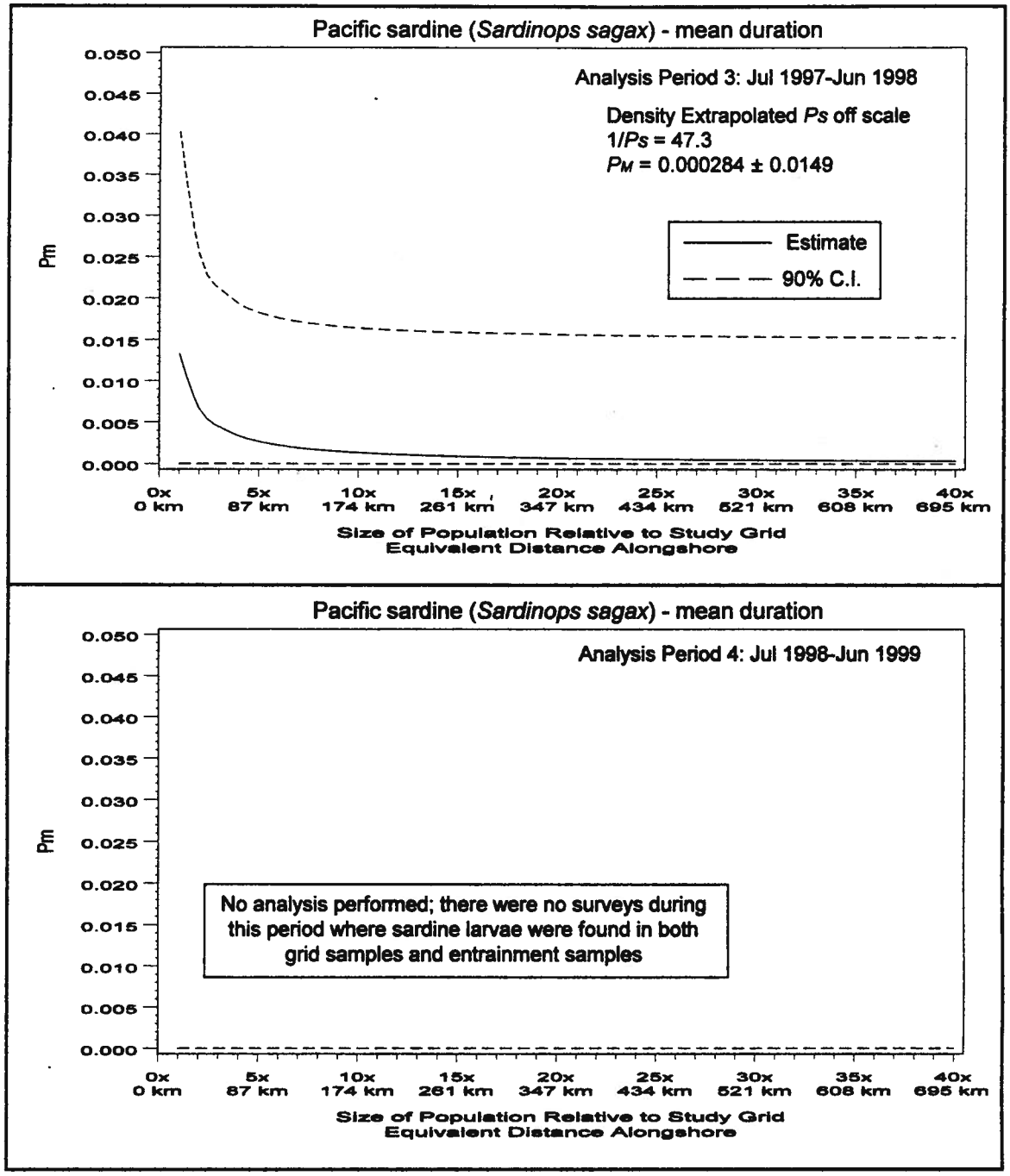


Figure 5.2.2-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the extent of the study grid population based on offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with the 90% C.I. indicated.

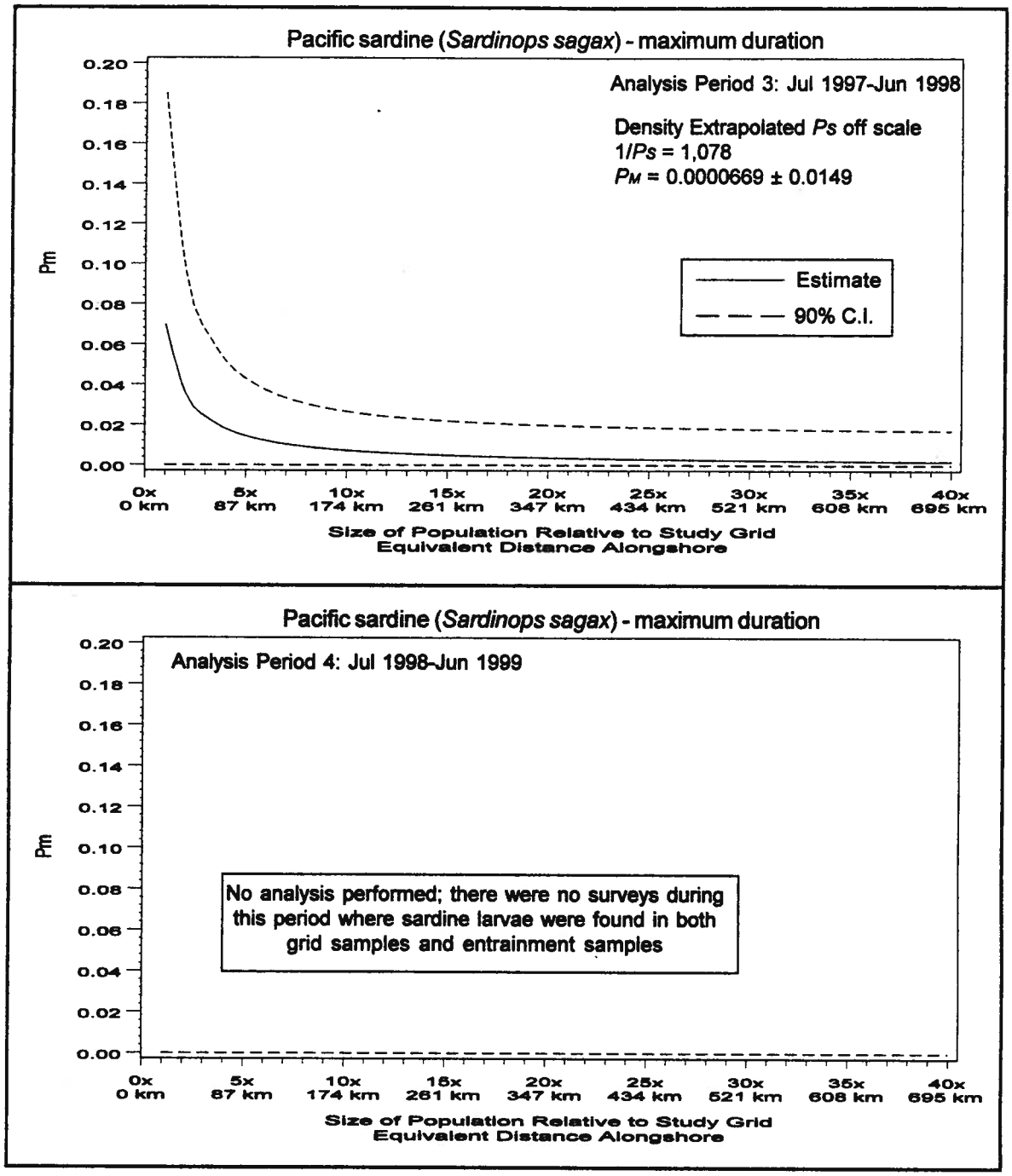


Figure 5.2.2-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the extent of the study grid population based on offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with the 90% C.I. indicated.

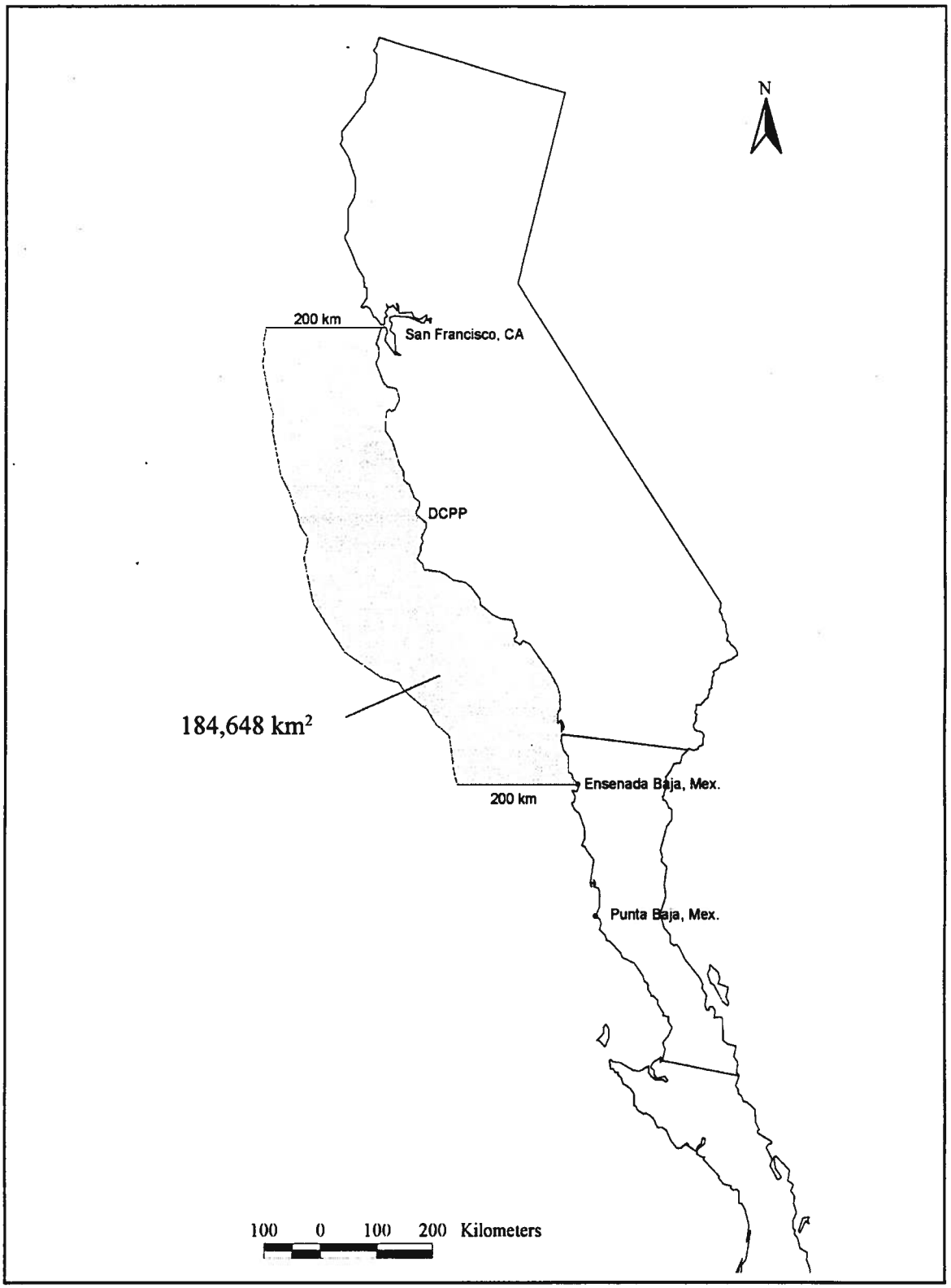


Figure 5.2.2-7. PFMC management area containing the spawning population of the Pacific sardine (Hill et al. in press).

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5.2.3 Assessment of Northern Anchovy (Engraulis mordax)

Engraulis mordax (Girard 1854); northern anchovy; length to 23 cm; Queen Charlotte Island in British Columbia to Cabo San Lucas, Baja California Sur and lower Gulf of California; surface and inshore to 310 m; blue to greenish on back, silvery below; adults with faint silvery side stripe (Miller and Lea 1972; Eschmeyer et al. 1983).

The northern anchovy is one of four species in the family Engraulididae (the anchovies) reported from California coastal waters (Miller and Lea 1972). Other representatives of this family are the deepbody anchovy (*Anchoa compressa*), slough anchovy (*Anchoa delicatissima*), and the anchoveta (*Centengraulis mysticetus*; Eschmeyer et al. 1983; Love et al. 1996). Potentially three distinct sub-populations of northern anchovy live along the Pacific coast of the United States (Lo 1985; PFMC 1990; Love 1996). One group is found from British Columbia to north of San Francisco, another is found from San Francisco to northern Baja California, and the last is found along the southern coast of Baja. This grouping of adult sub-populations has been accepted as part of the Fishery Management Plan (FMP) for Coastal Pelagic Species (CPS) by the Pacific Fishery Management Council (PFMC 1998).

Northern anchovy in the central subpopulation are harvested commercially in Mexico and California for human consumption, live bait, dead bait, and other commercial uses (PFMC 1998). Landings of northern anchovy in California between 1916 and 1997 varied from a low of 72 metric tons (MT) in 1926 to a high of 143,799 MT in 1975 (PFMC 1998). Although northern anchovy are fished throughout the state, commercial landings are usually made in San Francisco, Monterey, and Los Angeles; therefore, landing records for local ports are limited. The most recent annual (1997) local landings data for northern anchovy reported by CDF&G are from Port San Luis (22.6 MT), but the average annual landing from Avila during the period 1990–1998 is

2.7 MT, and the most recent northern anchovy landing at the port of Morro Bay occurred in 1981 (**Figure 5.2.3-1**).

Reproduction of northern anchovy varies with time and location. Northern anchovy off southern and central California can reach sexual maturity by the end of their first year, with all individuals maturing by 4 yr of age; off Oregon and Washington they do not mature until their third year (Love 1996). An early estimate of northern anchovy fecundity (Baxter 1967) indicates an annual range of 20,000–30,000 eggs per female. More recent data from Love (1996) indicates that females can release from 2,700–16,000 eggs per batch, with annual fecundity as high as 130,000 eggs in southern California and around 35,000 eggs per year in northern populations. Parrish et al. (1986) indicate that total annual fecundity from the first to the fourth-plus spawning seasons ranges from 32,514 to 322,957 eggs per female, respectively.

In southern California, anchovy spawn year-round with peaks during late winter to spring (Love 1996; Moser 1996). In Oregon and Washington, spawning can occur from mid-June to mid-August (Love 1996). During the peak of the spawning season, females can spawn every 6–8 d (Love 1996; Schlotterbeck and Connally 1982). Spawning normally occurs at night in the upper layers of the water column (Hart 1973). The most recent stock assessment for northern anchovy estimated spawning biomass at 388,000 MT (CV 38%) during the middle of February 1995 (Jacobson et al. 1995). Although no more recent stock assessments have been made, a qualitative analysis of the available data indicates that this remains the best estimate of current spawning biomass (Jacobson et al. 1997).

Females are oviparous, producing planktonic eggs and larvae (Moser 1996). Eggs occur from the surface to 75 m depth but usually are found in the upper 50 m (Emmett et al. 1991). The eggs hatch in 2–4 d, depending on the water temperature, and release 2.5–3.0 mm long relatively undeveloped larvae (Moser 1996). The larvae remain largely inactive, floating over 90% of the time, with intense swimming events taking place about once per minute and lasting from one to two seconds (Hunter 1976). Larvae begin schooling at 11–12 mm and transform into juveniles at 35–40 mm in approximately 70 d (Hart 1973).

5.2.3.1 Summary of Field Collections

Northern anchovy larvae were collected intermittently at the DCPP intake structure during the years 1996–1999 (**Appendix H**), with their greatest abundance tending to occur between December and May and their highest abundance in January 1998 (**Figure 5.2.3-2**). These larval density estimates represented a total of 3,443 larval northern anchovies in 879 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999, representing 19% of the subsamples collected and processed from that location during that period. An El Niño event began during the spring of 1997 (NOAA 1999) and was detected along the California coast in fall/winter of 1997–98 (Lynn et al. 1998). This climatological and oceanographic event may account for the observed peak in larval northern anchovy abundance during 1997–98. The primary spawning grounds of northern anchovy are south of Point Conception (Hart 1973; PFMC 1998), but the warm water that moved up the coast as a result of this El Niño could have carried the anchovy spawning biomass to the north of their normal southerly distribution.

Larval northern anchovy occurred in the DCPP study grid during all but one month (October 1998) of the sampling period (**Appendix H**), with their highest abundance during March–May 1998 (**Figure 5.2.3-3**). The patterns observed in the study grid further support the supposition that the 1997–98 El Niño may have displaced their spawning biomass center further north than its expected distribution in a cooler water year. The density estimates from the study grid represented 8,464 larval northern anchovies identified from 1,134 bongo net subsamples, representing 37% of the study grid subsamples processed from July 1997–June 1999.

Standard lengths of all northern anchovy larvae collected at the DCPP intake structure between October 1996 and June 1999 ranged from a minimum of 1.6 mm to a maximum of 37.1 mm (**Figure 5.2.3-4**). The central 98% of this length-frequency distribution resulted in a minimum length of 2.1 mm and a maximum length of 25 mm. Reported hatching size for this species ranges from 2.5–3.0 mm (Moser 1996). The fact that we observed some larvae smaller than the reported hatching lengths can be explained partly by natural variation of hatch lengths within the population and partly by the phenomenon of shrinkage following preservation (Theilacker 1980).

The mean larval length in this distribution was 6.1 mm. The growth rate of larval northern anchovy (0.45 mm/d) reported by Methot and Kramer (1979) was used to estimate ages of entrained larvae. Assuming that larvae less than the minimum 1% length (2.1 mm) were immediately post-hatch and aged zero days, then the estimated age of larvae entrained could range from zero days up to 51 d post-hatching for the longest larva measured (25 mm). On average, the estimated age of northern anchovy larvae entrained at DCPP was ca. 9 d post-hatching.

No significant differences were detected between the bi-modal distributions of larval northern anchovy lengths at the DCPP intake and from the study grid (**Figure 5.2.3-5**) using a Kolmogorov-Smirnov test ($p \geq 0.299$; **Appendix J**). Additionally, no significant difference between mean lengths at the DCPP intake (8.8 mm) and study grid (9.2 mm) was detected by a t-test ($p \geq 0.358$). These results, combined with the similarity of the length-frequency distributions, indicate that the subsamples in the two locations came from the same population.

5.2.3.2 Estimating Total Annual Entrainment

The annual estimated mean entrainment and associated standard error for northern anchovy larvae are presented in **Table 5.2.3-1**. Estimates of total annual entrainment ($\widehat{E_T}$) were nearly four times higher in 1997–98 Analysis Period 2 ($\widehat{E_T}$ = 104,000,000) than in 1996–97 Analysis Period 1 ($\widehat{E_T}$ = 26,600,000). A value of $2 \cdot \widehat{SE}(\widehat{E_T})$ can be used to approximate a 95% confidence interval around the point estimates for each analysis period. The confidence intervals for Analysis Periods 1 and 2 do not overlap, indicating that there were probably statistically significant differences between the estimates from the two years (although not all variance terms are considered). The larger estimate for 1997–98 may have been due to the El Niño event that occurred during this time period. The warm water transported northward likely brought more northern anchovy into the waters around Diablo Canyon from the center of their spawning distribution that typically occurs south of Point Conception.

Estimates of annual northern anchovy entrainment for all three analysis periods were adjusted (**Table 5.2.3-2**) to the long-term average larval fish abundance index from the Intake Cove

plankton tows (**Figure 5.2.3-6**). The average index (I_i) was 0.015 larvae/m^3 in 1997 and 0.021 larvae/m^3 in 1998. The long-term average index ($\bar{I}$) was 0.077 larvae/m^3 for the years 1990–1998. The ratio used to adjust total annual entrainment to the long-term average ($\bar{I} / I_i$) was calculated as 5.11 for 1997 and 3.62 for 1998, indicating that larval abundance during those years was lower than the long-term average. Although the correction resulted in larger estimates for both periods, the difference between $\hat{E}_{Adj-T}$ for 1996–97 ($\hat{E}_{Adj-T}$ = 136,000,000) and 1997–98 ($\hat{E}_{Adj-T}$ = 376,000,000) was reduced from a factor of 4 for $\widehat{E_T}$ to less than a factor of 3. The adjusted entrainment estimates for analysis periods 2 and 3 were very close in value because both periods overlap for much of the period of peak larval abundance for northern anchovy.

5.2.3.4 Fecundity Hindcasting *(FH)*

Fecundity hindcasting requires age-specific fecundity (M_X) and mortality (L_X) to assess entrainment effects. Butler et al. (1993) modeled egg and larval survivorship of northern anchovy (**Table 5.2.3-3**). Their 'best' estimate is derived by fitting the range of mortality estimates from field collections to the assumption of a stable and stationary population. Instantaneous daily mortality estimates were used from Butler et al. (1993) for egg (0.231), yolk-sac (0.366), and early larval stages (0.286). These rates can be converted, over their average stage durations, to finite survivorship rates for each developmental stage: egg (0.512), yolk-sac (0.268), and early larval stages (0.414). Survival rates for eggs and larvae were then calculated for an estimated average age at entrainment of ca 9 d, using a growth rate of 0.45 mm/d (Methot and Kramer 1979) and an average (6.1 mm) and 1% smallest (2.1 mm) size at entrainment. Survival to entrainment for average-aged larvae was thus estimated as

$$S_{Egg-8.99days} = 0.0660 .$$

The *FH* approach combines larval entrainment losses, adult fecundity, age at maturity, and longevity to hindcast the numbers of adult females effectively removed from the reproductively active population. Clark and Phillips (1952) report age at sexual maturity as 1–2 yr. In subsequent calculations, we will assume the mid-value of 1.5 yr and assume the reported range

corresponds to 99% of a normal distribution. For longevity, Hart (1973) reports a value of 7 yr, but Leet et al. (1992) states that northern anchovy in the fished population rarely exceed 4 yr of age. We will use the value of 4 yr to represent the most likely reproductive life span. Parrish et al. (1986) estimates an annual fecundity in the second spawning season of 102,174 eggs per female.

The adjusted estimate of annually entrained northern anchovy larvae ($\hat{E}_{Adj-T}$) was used to estimate the number of breeding females needed to produce those larvae in analysis periods 1, 2, and 3 (**Table 5.2.3-4**). Estimates of *FH* for the 1996–97 analysis period ($\widehat{FH}$ = 16,100) were less than half of those estimated for the 1997–98 Analysis Period 2 ($\widehat{FH}$ = 44,600). Similar to the results seen with entrainment, these differences are a result of the entrainment estimates for those periods postulated to be due to the El Niño event that occurred during the second sampling year.

Sensitivity Analysis

Fecundity had the greatest leverage on recalculated values of $\widehat{FH}$ (**Table 5.2.3-5**). Varying this parameter in the model resulted in an $\widehat{FH}$ range of 3,110–232,000 adult females among the three analysis periods, compared to the original estimate of 16,100–44,700 adult females. Survivorship (specifically, yolk-sac survivorship) had the next greatest amount of leverage on recalculated $\widehat{FH}$.

5.2.3.5 Adult Equivalent Loss *(AEL)*

The *AEL* approach used estimates of the abundance of entrained organisms to project the loss of equivalent numbers of adults based on mortality schedules and age at recruitment into the fishery. The instantaneous survival rate from the entrained larval stage to recruitment into the northern anchovy fishery (through juvenile and early adult stages to 2 yr of age) is 0.000318 estimated from the life table produced by Butler et al. (1993; **Table 5.2.3-4**). Average age at entrainment for northern anchovy larvae (9 d) was calculated by dividing the average larval length at entrainment (minus length at hatching) by a literature-based growth rate of 0.45 mm per day (Methot and Kramer 1979). Butler et al. (1993) apportioned survivorship to recruitment into

several developmental stages, and AEL was calculated using the entrainment of a single age class having the average age at recruitment of 2 yr.

The numbers of equivalent adult northern anchovies predicted from adjusted total annual entrainment ranged from 43,200 2 yr-old recruits in 1996–97 to 120,000 in 1997–98 (**Table 5.2.3-6**). The estimates nearly tripled between years, possibly due to El Niño-induced increases in larval abundance during the second year. However, the 90% confidence intervals overlap, which may indicate that these estimates are not significantly different between sampling periods.

Sensitivity Analysis

Early and late larval survivorship had the greatest leverage on recalculated values of $\widehat{AEL}$ (**Table 5.2.3-7**). Varying either of these two parameters in the model resulted in an $\widehat{AEL}$ range of 8,330–621,000 adult equivalents among the three analysis periods, compared to the original estimate of 43,200–120,000 adult equivalents. Pre-recruit and early juvenile survivorship had the next greatest leverage on estimates of $\widehat{AEL}$.

5.2.3.6 Empirical Transport Model *(ETM)*

There was large variation in the estimates of proportional entrainment ($\widehat{PE}$) for each *i*th survey during the two analysis periods (July 1997–June 1998 and July 1997–June 1998; **Table 5.2.3-8**). Estimates of proportional entrainment ranged from zero to as high as 0.0102. Maximum values of $\widehat{PE_i}$ in the two periods occurred during January 1998 (0.00186) and February 1999 (0.0102). During Analysis Period 3, the largest proportion of larvae were collected during January 1998 ($\hat{f}_i$ = 0.468), the same monthly survey period as the maximum $\widehat{PE_i}$. In the second period, the largest proportion of larvae were collected during August 1998 ($\hat{f}_i$= 0.431). The February 1999 $\hat{f}_i$ was 0.055, indicating that the maximum $\widehat{PE_i}$ was weighted less than others during the study period in calculating the larval mortality estimate $\widehat{P_M}$.

When both $\widehat{PE_i}$ and $\hat{f}_i$ were equal to zero, no larvae were collected at either the DCPP intake or from the study grid. When $\widehat{PE_i}$ = 0 and $\hat{f}_i$>0, larvae were collected at the DCPP intake during the

survey period but not during the entrainment survey paired with the 72-hour study grid survey. This occurred in February and again in September of 1998 and, as expected, were accompanied by low $\hat{f}_i$ values.

Larval entrainment mortality ($\widehat{P_M}$) was estimated for the two, year-long periods, using two estimates of the duration of time that northern anchovy larvae may be susceptible to entrainment. These durations were estimated as 9 d and 51.2 d using the mean and maximum lengths, respectively, of entrained larvae. Northern anchovy larvae were dispersed throughout the study grid (**Figure 5.2.3-3**), but they are typically found in their greatest density offshore (Moser et al. 1993). Therefore, $\widehat{P_S}$ was calculated by extrapolating study grid abundance patterns offshore and then extending them alongshore. Estimates of total entrainment mortality ($\widehat{P_M}$) were 0.000575 for the 1997–1998 analysis period and 0.00187 for the 1998–1999 analysis period, using the mean larval duration. Smaller values of $\widehat{P_M}$ were estimated using the maximum larval duration: 0.0000786 in the 1997–1998 period and 0.000184 in the 1998–1999 period. Standard errors of $\widehat{P_M}$ were large relative to the estimates: 0.0168 in the first period and 0.0385 in the second period.

$\widehat{P_S}$ was calculated as a proportion of larval numbers in the study grid to an extrapolated number using the grid density and a larger area defined by alongshore and offshore current measurements. These areas were a function of the estimated duration of larval susceptibility to entrainment, and for the periods 1997–1998 and 1998–1999, averaged 639 and 370 km^2 for the mean larval duration, and 6,670 and 6,140 km^2 for the maximum larval duration, respectively.

For the plots showing the sensitivity analysis for each analysis period, $\widehat{P_M}$ declined rapidly with increasing $1/\widehat{P_S}$, using either average or maximum length to calculate larval duration (**Figure 5.2.3-7**). As duration increased within an analysis period, $\widehat{P_M}$ decreased and $1/\widehat{P_S}$ increased. There were large differences between analysis periods in the point estimates of $\widehat{P_M}$ for both larval durations. The large error associated with calculating $\widehat{P_M}$ may account for some of the differences and contributes to the level of uncertainty associated with these estimates.

5.2.3.7 Interpretation of Assessment Results

Population-level losses, using the estimated larval mortality $\widehat{P_M}$ due to entrainment at DCPP, can be compared, as a small fraction, to recent spawning stock estimates of 388,000 MT of northern anchovy in the region from San Francisco to Punta Baja, Mexico (Jacobson et al. 1997). The estimated losses by weight are calculated, using the average areas to determine $\widehat{P_M}$ (**Section 5.2.3.6**), as a proportion of the larger stock assessment area of 231,192 km^2 (**Figure 5.2.3-8**). The spawning stock biomass is multiplied by the product of this proportion and $\widehat{P_M}$. For the 1997–1998 and 1998–1999 analysis periods, these losses are conservatively estimated, assuming no compensation, as 0.61 MT and 1.16 MT (based on mean larval duration) and 0.88 MT and 1.90 MT (based on maximum larval duration), respectively.

By comparison, approximately 1,185 MT of northern anchovy were landed in the Monterey Bay National Marine Sanctuary-wetfish (whole fish) fishery in 1995 (Starr et al. 1998). During 1980 to 1988, the price paid for anchovy landed by the U.S. non-reduction wetfish (other than live bait) fishery averaged $288/MT (Leet et al. 1992). In 1988, the price for anchovy sold in the reduction wetfish (processed for meal) fishery was $32/MT (Leet et al. 1992).

Estimates of *AEL* can be converted to direct reductions of the standing stock by converting them to units of biomass (e.g., kilograms of northern anchovy). Estimating that the average weight of a northern anchovy from the live bait and wetfish fisheries is 19 g (estimated from Clark and Phillips 1952) and assuming that this represents an average weight of an adult in the population at large, *AEL* estimates (43,200 in 1996–97 and 120,000 in 1997–98) translate to a maximum estimated weight of 1.92 MT. These can be further translated into an annualized reduction of the estimated standing stock (388,000 MT; Jacobson et al. 1995, 1997) of <0.001%. These adult equivalents are assigned the average age of a recruit (2 yr), so this annualized reduction necessarily assumes that all of the standing stock was 2 yr of age. Since it is likely that the average age in the standing stock is somewhat greater than the youngest recruits, this percent reduction is likely an underestimate of the loss to the 2 yr old cohort present in the standing stock.

It is noteworthy that not only are *AEL* estimated losses similar to $\widehat{P_M}$ -based losses in northern anchovy biomass, but also, when estimates are aligned to age 2 (**Section 4.4.3.1**), $\widehat{FH}$ gives similar results. Assuming a linear population decline, an estimated average age of female anchovies ($\widehat{FH}$) is one third of the age from maturity to longevity, 2.33 yr. Using a natural mortality estimate of Z = 0.0031 d^{-1}, the numbers ($AEL = 2\widehat{FH} = 89{,}400$) and biomass lost is estimated as 130,000 fish and 2.08 MT.

One method of interpreting population lost to entrainment converts their estimated weight into dollars using the fishery value of $288/MT reported in Leet et al. (1992). The predicted equivalent adult losses using 2 MT would then be valued at $576 annually. Another approach would be to compare the weight of equivalent adults (approximately 2 MT) with that of the most recent catch landed in the Morro Bay area, which was approximately 23 MT in 1997 (**Figure 5.2.3-1**). [CDF&G records included no northern anchovy catch at either Morro Bay area port for either 1995–96 or 1998 although northern anchovy were caught and sold as live bait during these years]. A fractional comparison of annual *AEL* with the 1997 catch statistics indicates that potential losses as a result of DCPP, conservatively assuming 100% catchability of the equivalent adults, equaled approximately 10% of the 1997 Morro Bay area catch. Although $\widehat{P_M}$ -based results produce a similar proportion of Morro Bay area catch, when applied to the population from San Francisco to Punta Baja, Mexico, the loss is insignificant, less than 0.001%. In addition, harvesting parameters such as selectivity, catchability, and fishing effort indicate that this estimate is likely conservative further reducing the impact to the fishery.

Table 5.2.3-1. Estimated total annual entrainment ($\widehat{E_T}$) of northern anchovy larvae at the DCPP intake structure for the three analysis periods with their standard errors ($\widehat{SE}(\widehat{E_T})$).

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	26,600,000	1,510,000
2) Oct 1, 1997–Sep 30, 1998	104,000,000	7,320,000
3) Jul 1, 1997–Jun 30, 1998	104,000,000	7,320,000

Table 5.2.3-2. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) of northern anchovy larvae at the DCPP intake structure for the three analysis periods with their standard errors ($\widehat{SE}(\hat{E}_{Adj-T})$).

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	136,000,000	7,690,000
2) Oct 1, 1997–Sep 30, 1998	376,000,000	26,500,000
3) Jul 1, 1997–Jun 30, 1998	377,000,000	26,500,000

Table 5.2.3-3. Survivorship of northern anchovy (*Engraulis mordax*): a) Age-specific fecundity schedule (M_X = natality rate; L_X = survivorship) and b) stage-specific life-history parameters (Z=instantaneous daily mortality; S=finite survival rate) modified from Butler et al. (1993).

a)

Age (yr)	M_X	L_X	M_XL_X
1	22,500	1,000	22,500,000
2	93,500	468	43,800,000
3	195,000	216	42,000,000
4	280,000	102	28,600,000
5	328,000	48	15,700,000
6	328,000	22	7,210,000
7	328,000	10	3,280,000

b)

Stage	Z_{best}	Stage duration (d)	Age (d)	Duration modified by DCPP data	S_{best}	CV_{best}
Egg	0.231	2.9		2.9	0.512	0.142
Yolk-sac larva	0.366	3.6	6.5	3.6	0.268	0.240
Early larva	0.286	12	18.5	2.56	0.481	0.071
Survivorship from egg to entrainment:					0.0660	
Early larva	0.286	12	18.5	9.44	0.0671	0.329
Late larva	0.0719	45	63.5	45	0.0393	0.427
Early juvenile	0.0141	62	125.5	62	0.417	0.239
Late Juvenile	0.00440	80	205.5	80	0.703	0.0328
Pre-recruit	0.00310	287	492.5	287	0.411	0.0882
Survivorship from entrainment to recruitment:					0.000318	

combine - "larvae"

combine "juvenile"

$S = e^{-Zt}$

Table 5.2.3-4. Estimated number of northern anchovy adult females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{FH})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	16,100	57,300	5,630,000	46
2) Oct 1, 1997–Sep 30, 1998	44,700	159,000	15,600,000	128
3) Jul 1, 1997–Jun 30, 1998	44,700	159,000	15,600,000	128

Table 5.2.3-5. Northern anchovy (*Engraulis mordax*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 16,100				
$\hat{E}_{Adj-T}$	136,000,000	14,600	17,600	3,000
$\hat{S}_{egg}$	0.512	8,240	83,400	75,200
$\hat{S}_{yolk\text{-}sac}$	0.268	4,310	83,400	79,100
$\hat{S}_{larvae}$	0.481	7,750	83,400	75,700
# Eggs/yr	102,000	3,110	83,400	80,300
Longevity	4	6,190	32,200	26,000
Maturation	1.5	12,400	40,300	27,900

Table 5.2.3-5 (continued). Northern anchovy (*Engraulis mordax*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 44,600				
$\hat{E}_{Adj-T}$	376,000,000	39,500	49,800	10,300
$\hat{S}_{egg}$	0.512	22,800	231,000	208,000
$\hat{S}_{yolk\text{-}sac}$	0.268	12,000	231,000	219,000
$\hat{S}_{larvae}$	0.481	21,500	231,000	210,000
# Eggs/yr	102,000	8,620	231,000	223,000
Longevity	4	17,200	89,300	72,100
Maturation	1.5	34,300	112,000	77,300

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 44,700				
$\hat{E}_{Adj-T}$	377,000,000	39,500	49,900	10,300
$\hat{S}_{egg}$	0.512	22,900	232,000	209,000
$\hat{S}_{yolk\text{-}sac}$	0.268	12,000	232,000	220,000
$\hat{S}_{larvae}$	0.481	21,500	232,000	210,000
# Eggs/yr	102,000	8,620	232,000	223,000
Longevity	4	17,200	89,400	72,200
Maturation	1.5	34,400	112,000	77,300

Table 5.2.3-6. Estimated number of equivalent northern anchovy adults ($\widehat{AEL}$) equal to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{AEL})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{AEL}$	$\widehat{SE}(\widehat{AEL})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	43,200	160,000	19,300,000	97
2) Oct 1, 1997–Sep 30, 1998	120,000	444,000	53,400,000	268
3) Jul 1, 1997–Jun 30, 1998	120,000	444,000	53,500,000	268

Table 5.2.3-7. Northern anchovy (*Engraulis mordax*): Sensitivity analysis of $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL} = 43{,}200$				
$\hat{E}_{Adj-T}$	136,000,000	39,100	47,200	8,050
$\hat{S}$ *early larvae*	0.0671	8,330	224,000	215,000
$\hat{S}$ *late larvae*	0.0393	8,330	224,000	215,000
$\hat{S}$ *early juvenile*	0.417	8,330	103,000	95,100
$\hat{S}$ *juv. I*	0.703	8,330	61,400	53,000
$\hat{S}$ *pre-recruit*	0.411	8,330	105,000	96,800

Table 5.2.3-7 (continued). Northern anchovy (*Engraulis mordax*): Sensitivity analysis of $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 120,000				
$\hat{E}_{Adj-T}$	376,000,000	106,000	134,000	27,700
$\hat{S}$ *early larvae*	0.0671	23,100	620,000	597,000
$\hat{S}$ *late larvae*	0.0393	23,100	620,000	597,000
$\hat{S}$ *early juvenile*	0.417	23,100	287,000	264,000
$\hat{S}$ *juv. I*	0.703	23,100	170,000	147,000
$\hat{S}$ *pre-recruit*	0.411	23,100	291,000	268,000

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 120,000				
$\hat{E}_{Adj-T}$	377,000,000	106,000	134,000	27,700
$\hat{S}$ *early larvae*	0.0671	23,100	621,000	598,000
$\hat{S}$ *late larvae*	0.0393	23,100	621,000	598,000
$\hat{S}$ *early juvenile*	0.417	23,100	287,000	264,000
$\hat{S}$ *juv. I*	0.703	23,100	170,000	147,000
$\hat{S}$ *pre-recruit*	0.411	23,100	292,000	269,000

Table 5.2.3-8. Northern anchovy (*Engraulis mordax*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$), the annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$), and their associated standard errors ($\widehat{SE}$).

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.000310	0.000320	0.00474	0.000730
Aug 25, 1997	0.000310	0.000310	0.0102	0.000790
Sep 29, 1997	0.000730	0.000330	0.00822	0.000870
Oct 20, 1997	0.000840	0.000320	0.00768	0.00150
Nov 17, 1997	0.000270	0.000190	0.00574	0.00169
Dec 10, 1997	0.000620	0.000260	0.0238	0.00155
Jan 22, 1998	0.00186	0.000970	0.468	0.0131
Feb 26, 1998	0	0	0.00219	0.000310
Mar 18, 1998	0.000080	0.0000400	0.155	0.00584
Apr 15, 1998	0.000380	0.000140	0.103	0.00412
May 18, 1998	0.000390	0.0000800	0.198	0.00697
Jun 8, 1998	0.00148	0.000390	0.0131	0.00101

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.000400	0.000210	0.170	0.0176
Aug 26, 1998	0.0010	0.000220	0.431	0.0204
Sep 16, 1998	0	0	0.00863	0.00353
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0.000580	0.000590	0.0582	0.0103
Jan 12, 1999	0.00200	0.00153	0.137	0.0175
Feb 3, 1999	0.0102	0.0117	0.0547	0.00987
Mar 17, 1999	0	0	0	0
Apr 14, 1999	0	0	0	0
May 24, 1999	0.00208	0.00156	0.140	0.0154
Jun 23, 1999	0	0	0	0

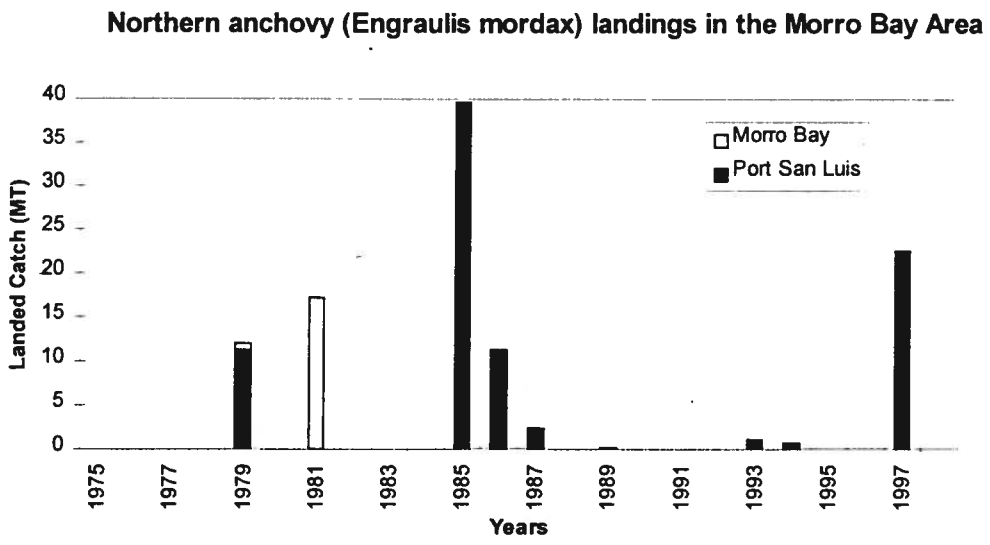

Figure 5.2.3-1. Commercial landings of northern anchovy (*Engraulis mordax*) at the ports in the Morro Bay Area between 1975 and 1997 (source: CDF&G Database).

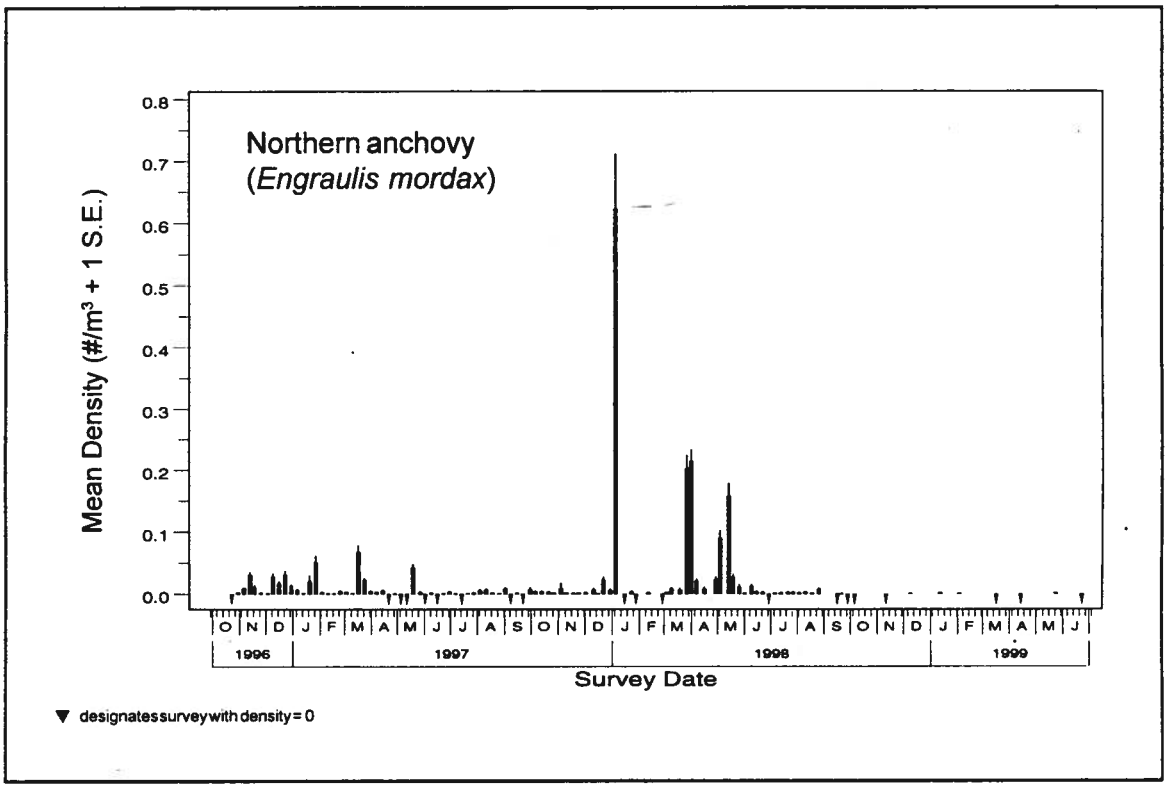

Figure 5.2.3-2. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

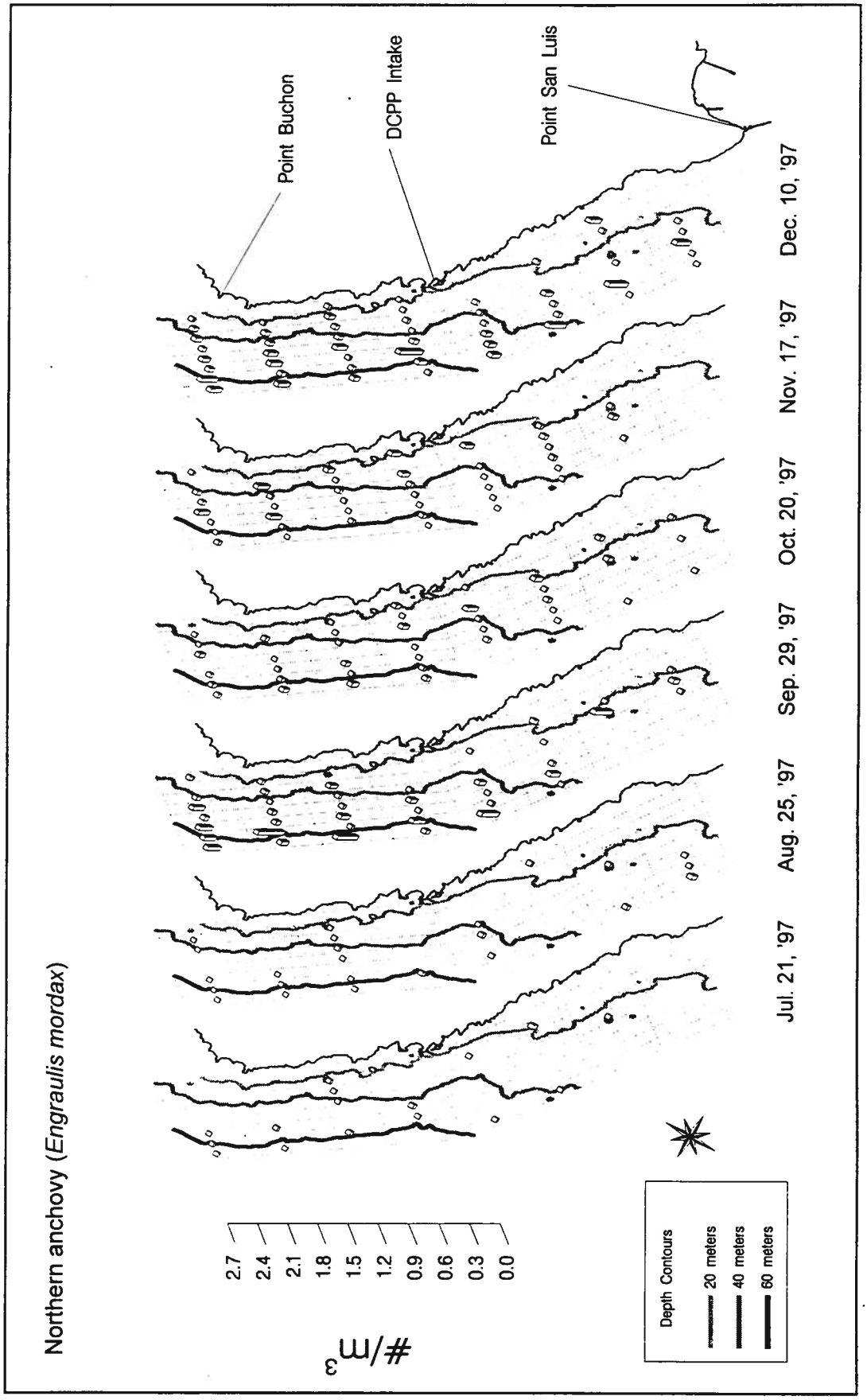

Figure 5.2.3-3a. Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

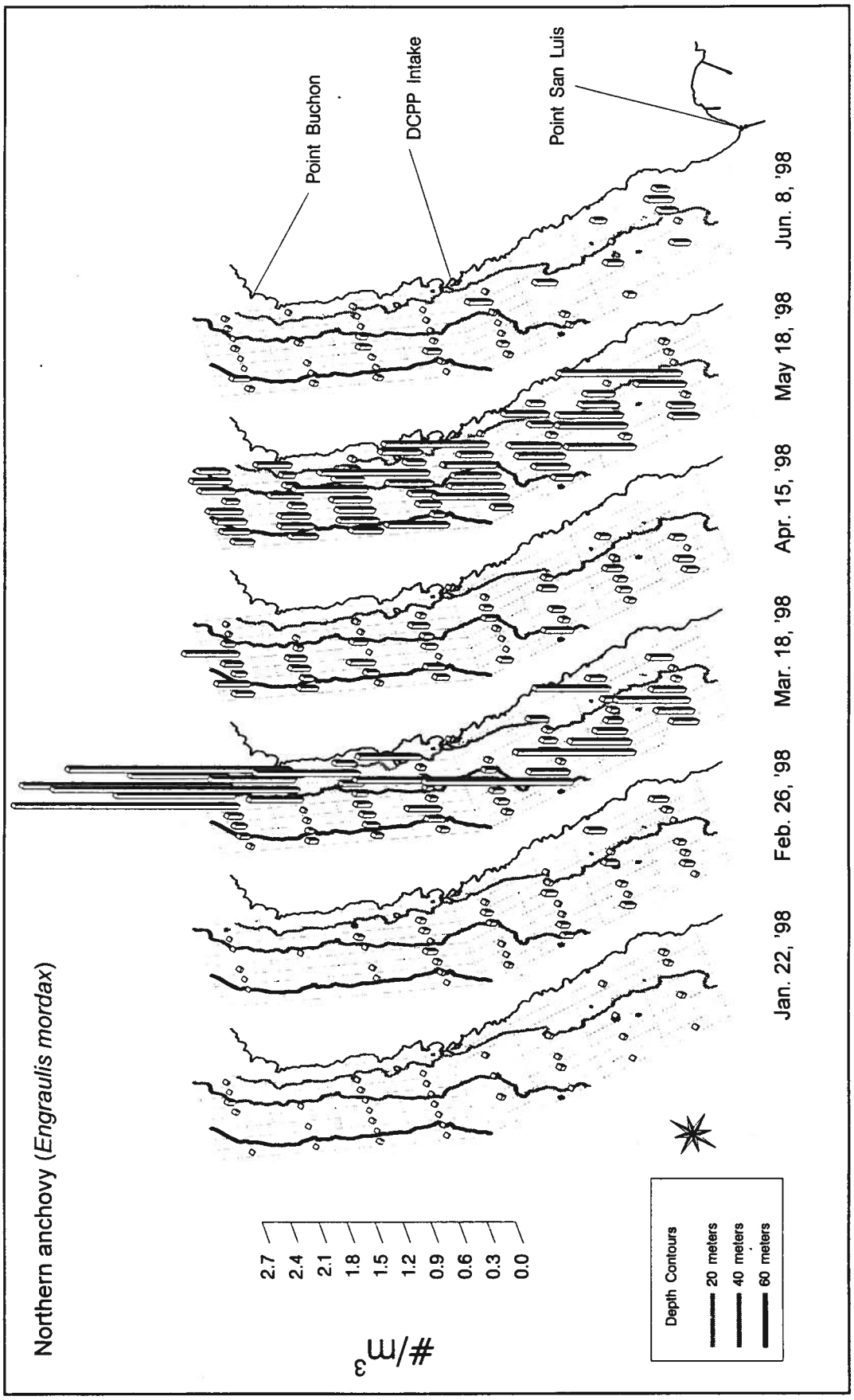

Figure 5.2.3-3b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

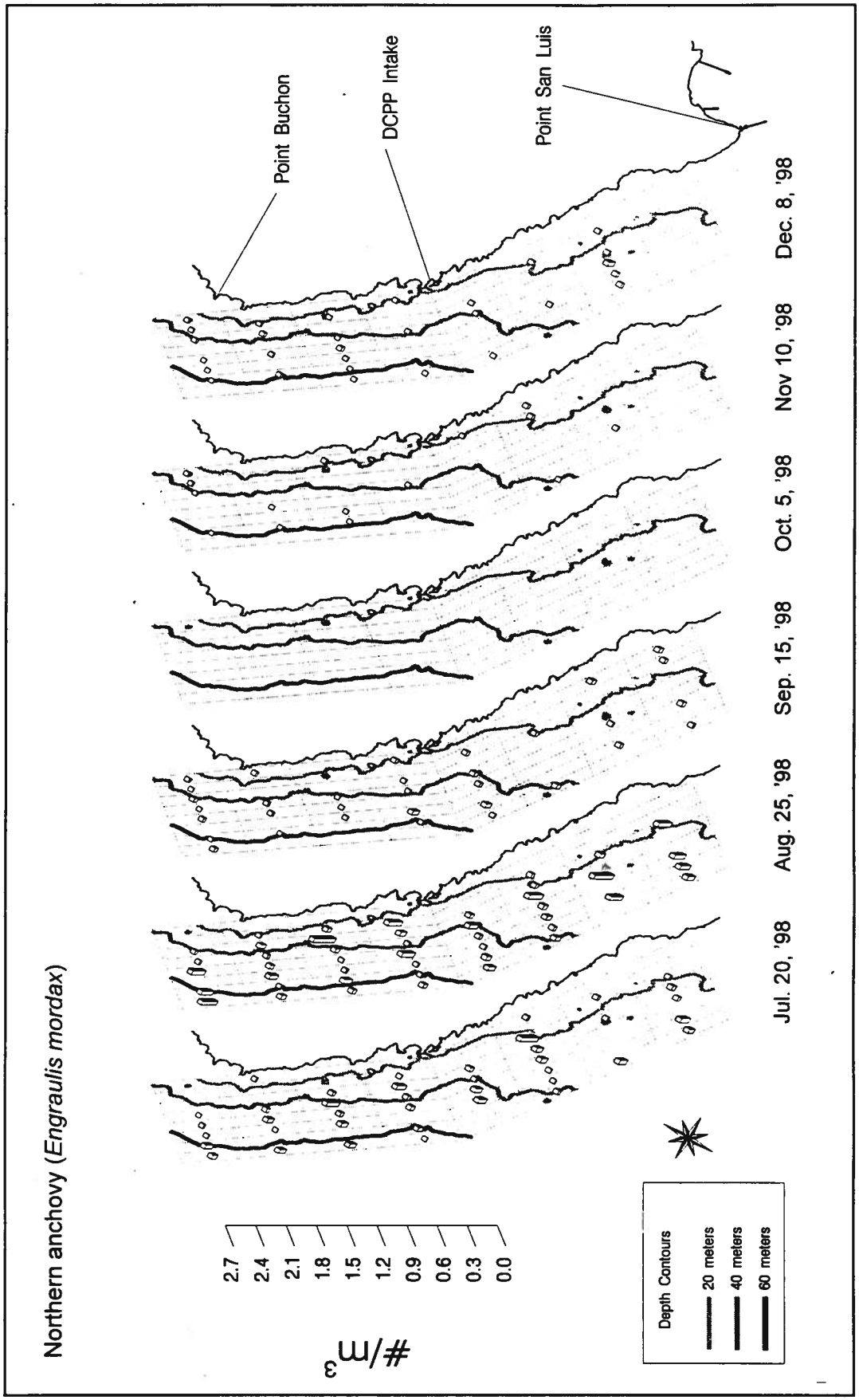

Figure 5.2.3-3c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

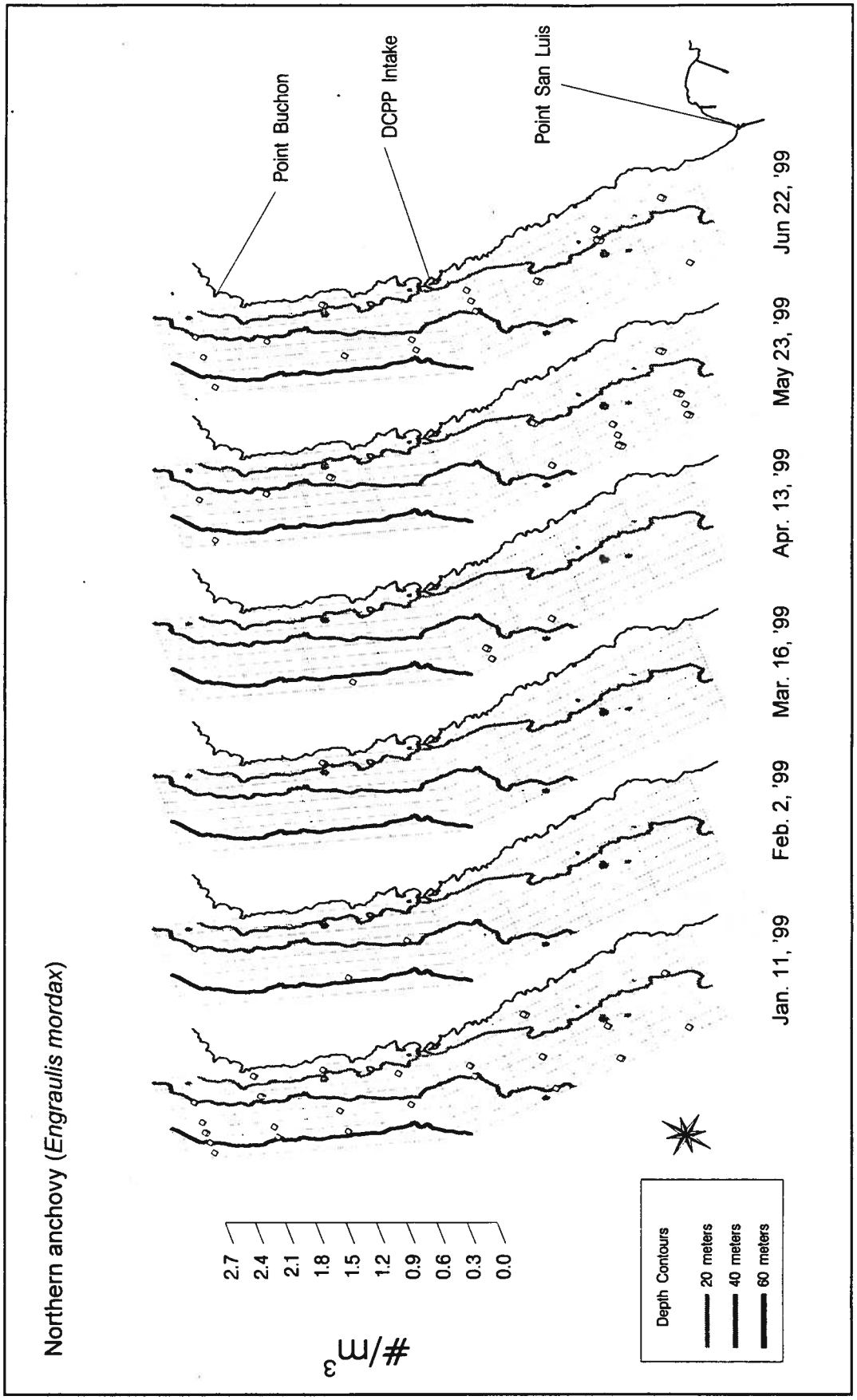

Figure 5.2.3-3d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

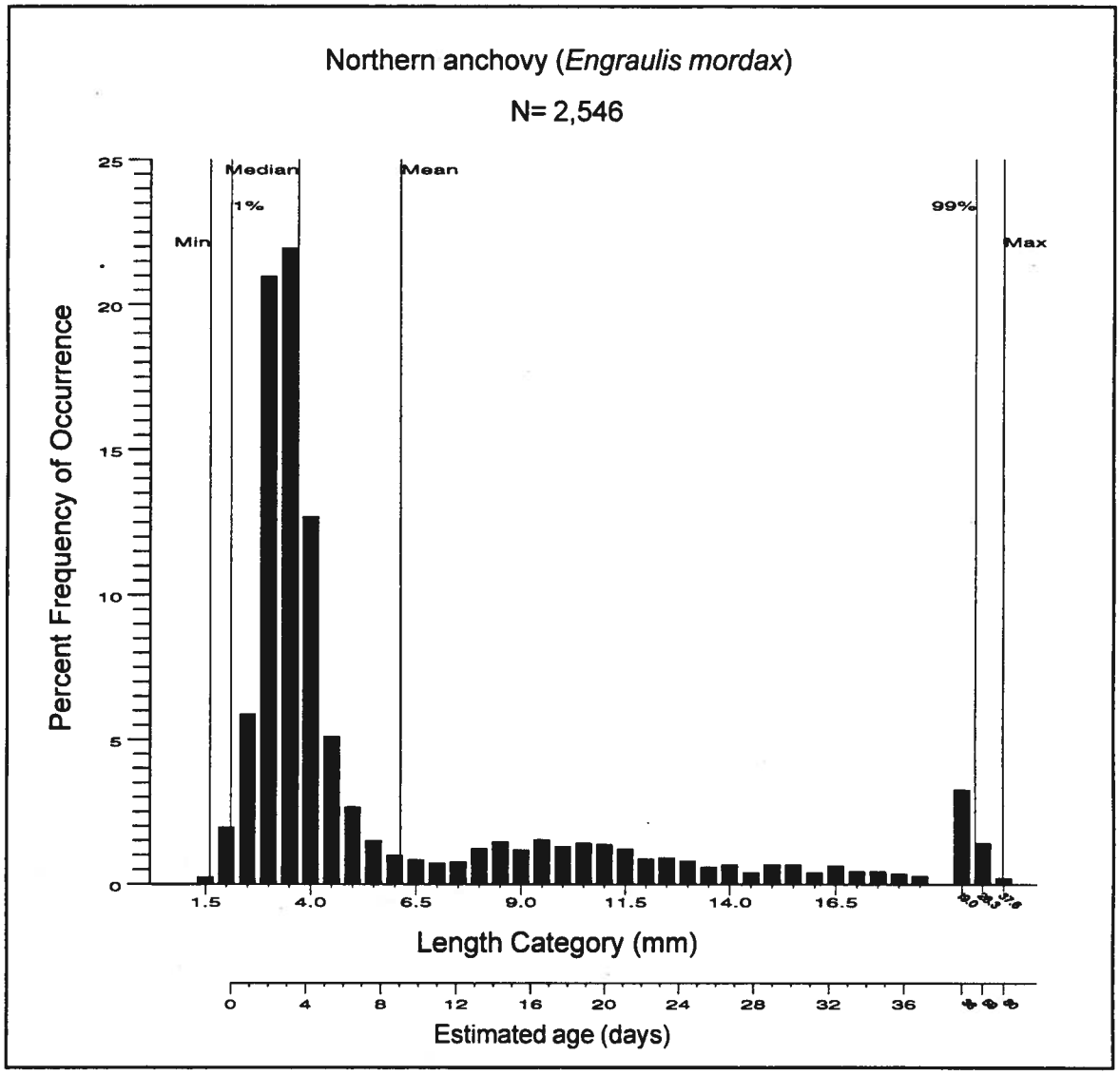

Figure 5.2.3-4. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

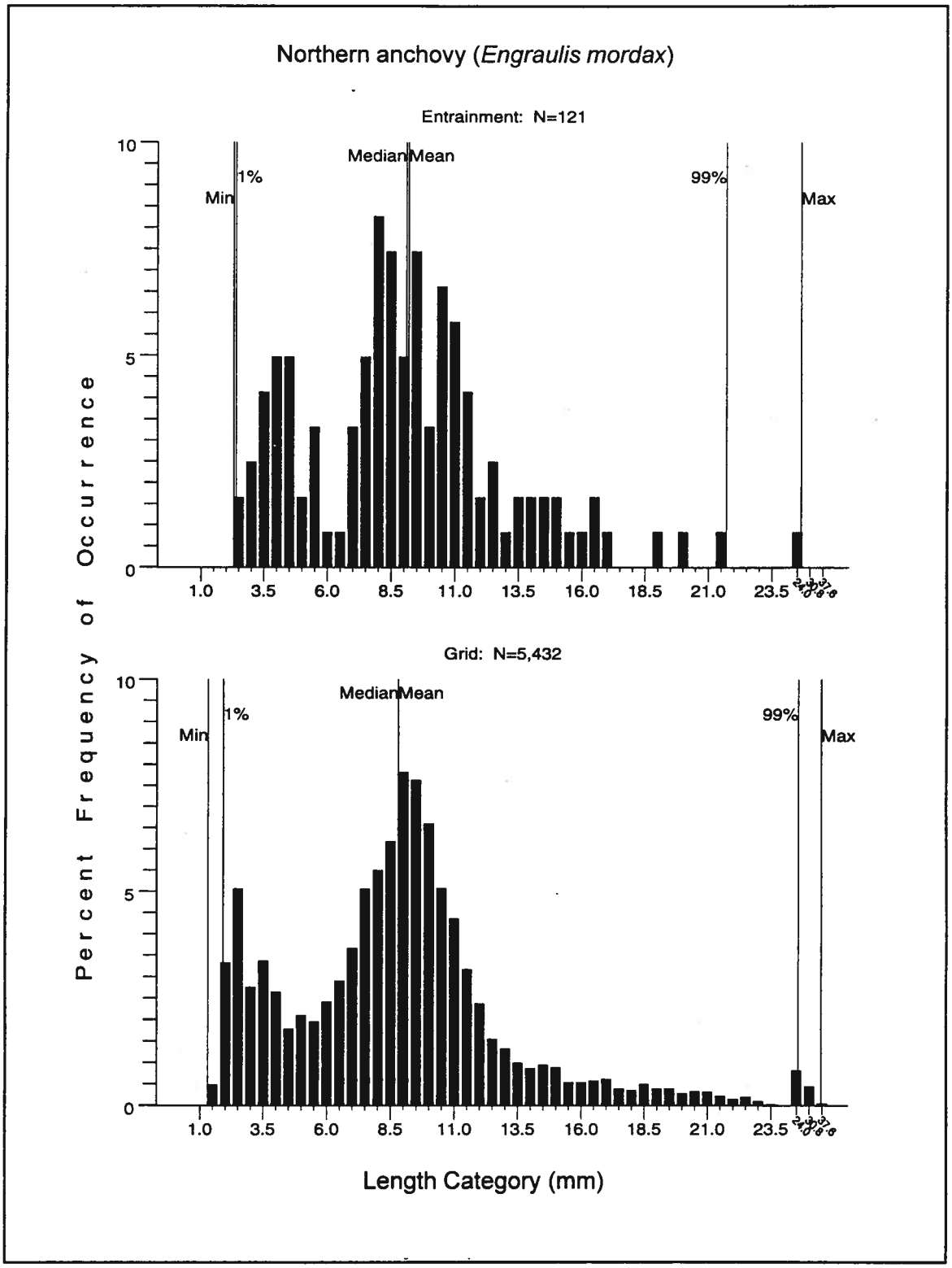

Figure 5.2.3-5. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

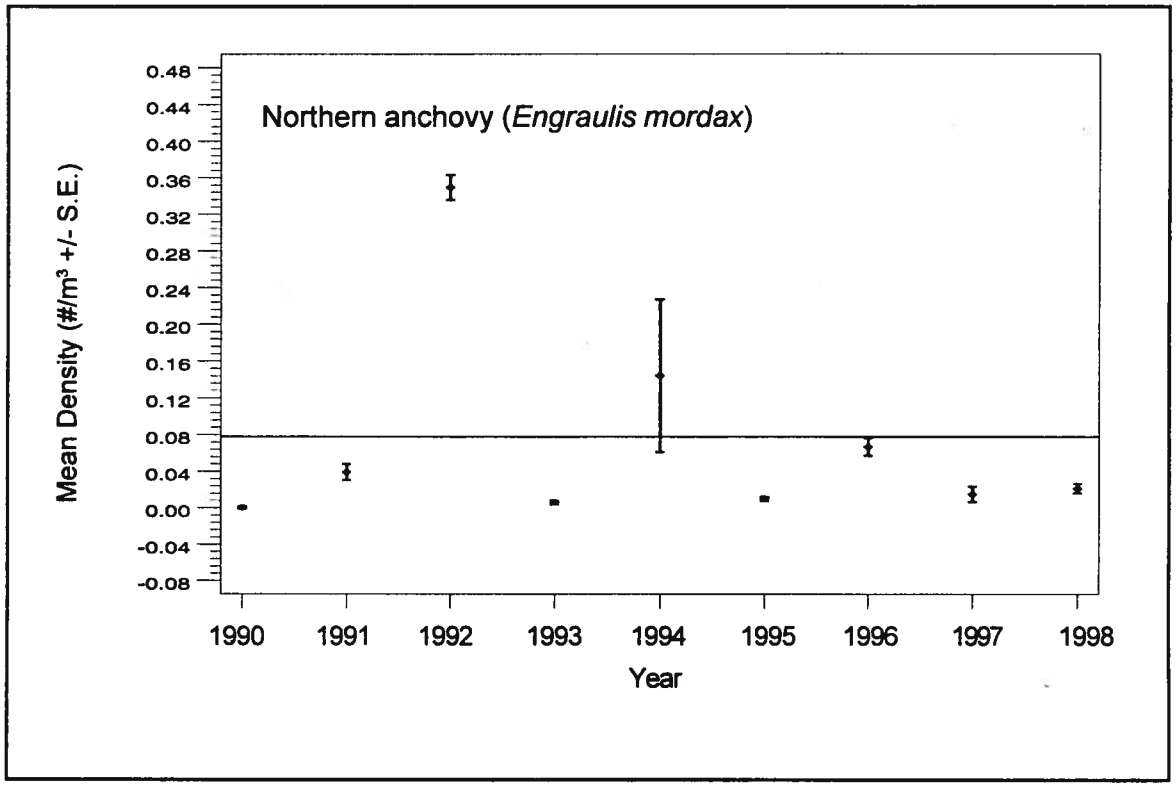

Figure 5.2.3-6. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

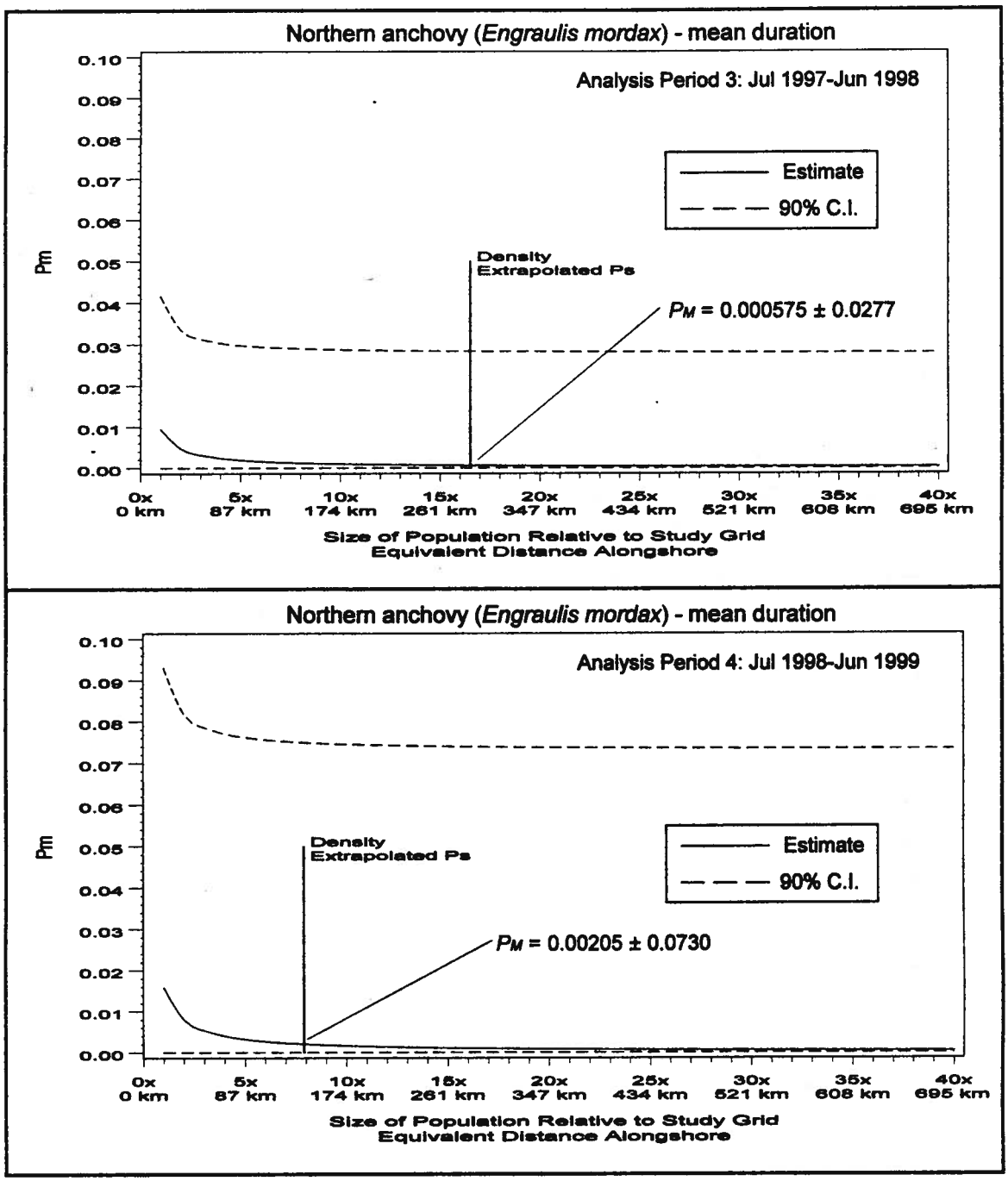

Figure 5.2.3-7a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the extent of the study grid population based on offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with the 90% C.I. indicated.

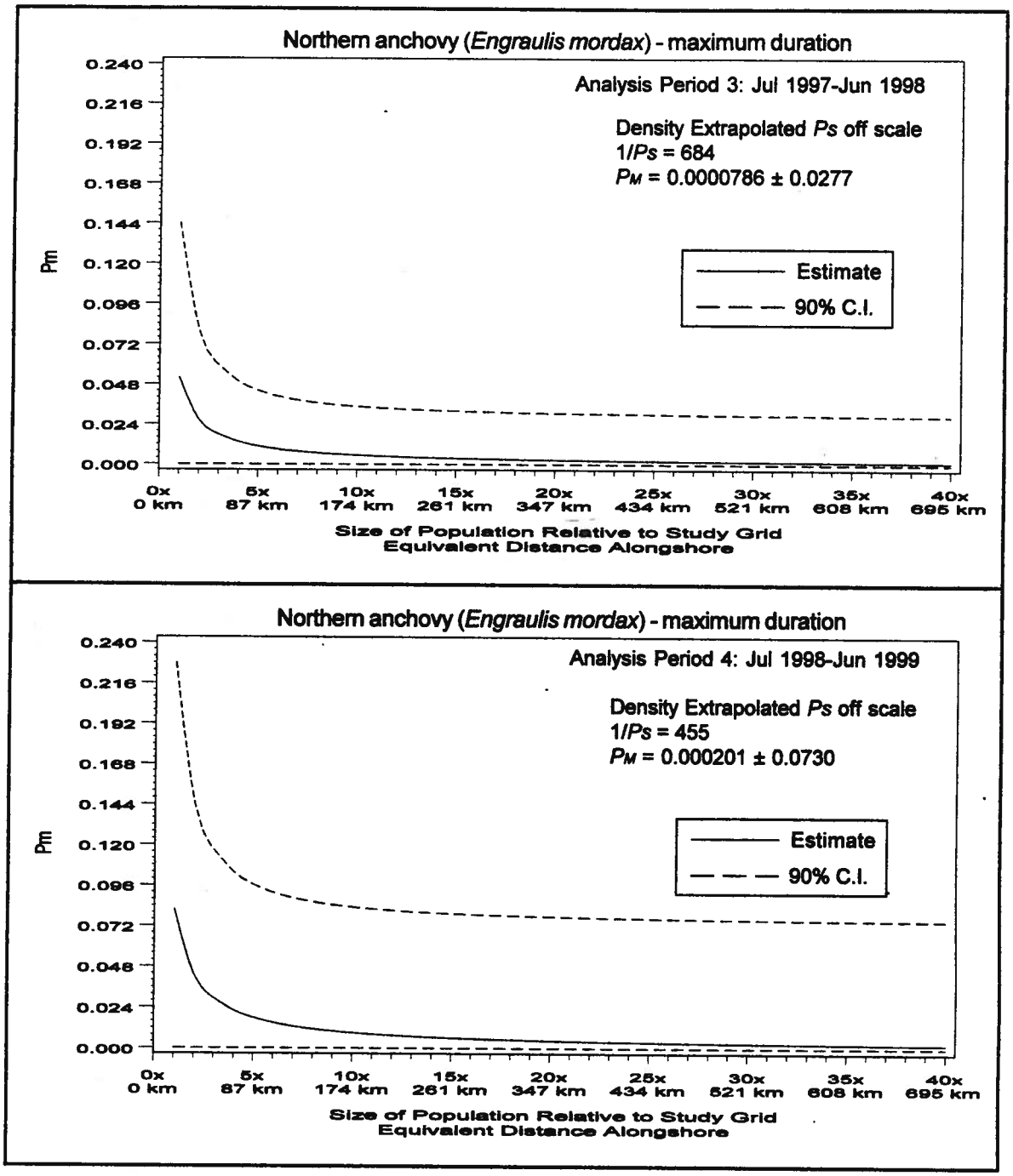

Figure 5.2.3-7b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the extent of the study grid population based on offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with the 90% C.I. indicated.

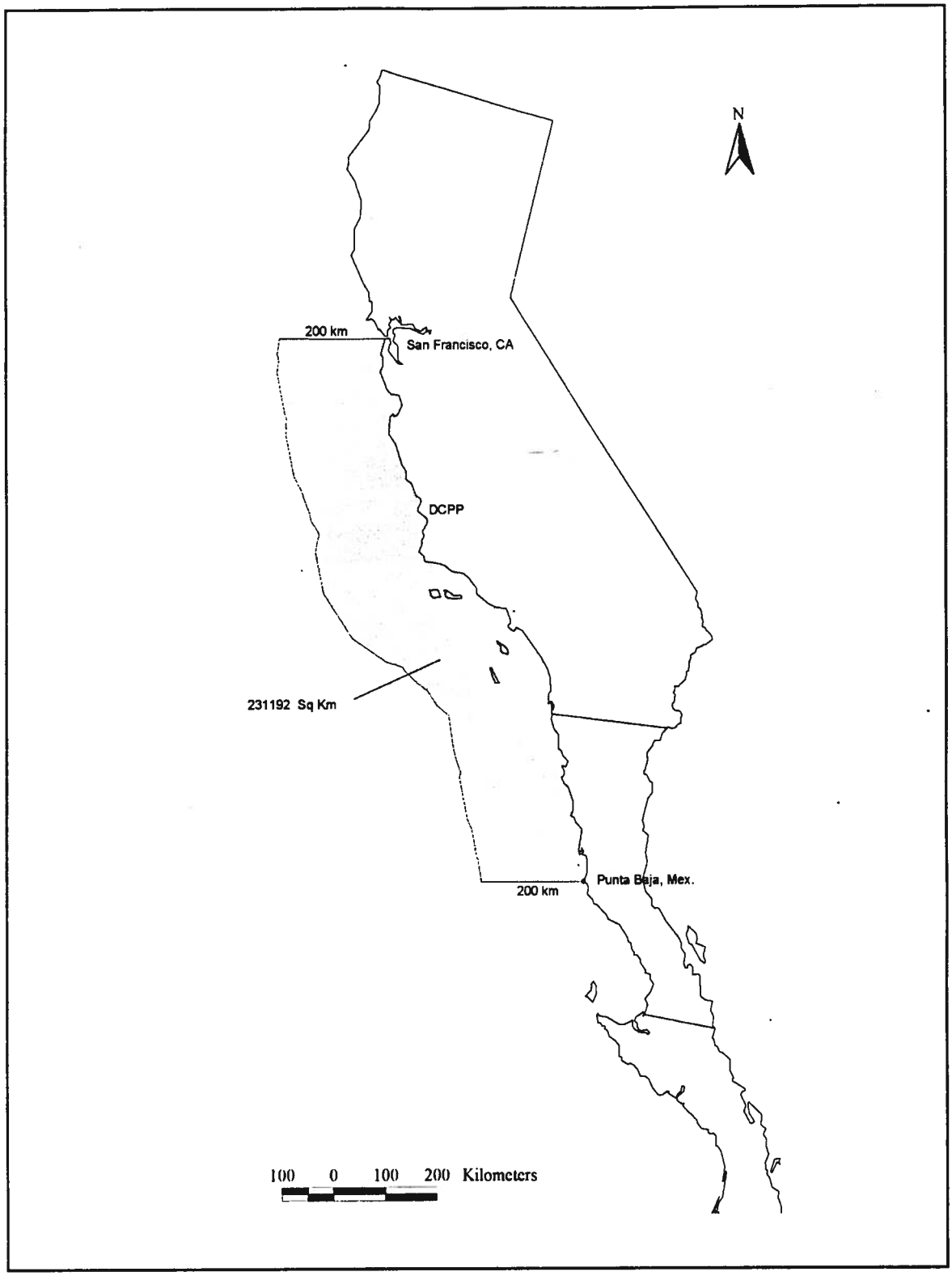

Figure 5.2.3-8. PFMC management area containing the spawning biomass of the central subpopulation of the northern anchovy .

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5.2.4 Assessment of Target Taxa in the Family Scorpaenidae

Rockfishes (*Sebastes* spp.) belong to the family Scorpaenidae that contains two other genera: the scorpionfishes (*Scorpaena* spp.) and the thornyheads (*Sebastolobus* spp.). Scorpaenidae comprise the largest number of commercially and recreationally important California marine fish species. They are also abundant in nearshore California habitats and play important trophic and ecological roles in these communities. They comprise a large component of the shallow subtidal fish community, ranging from nearshore coastal habitats (e.g., kelp forests) to the continental shelf. Adult California scorpionfish (*Scorpaena guttata*) are reported as far north as Santa Cruz, California, but adults are most common in waters south of Point Conception where they are an important component of the sport and commercial catch between Santa Monica Bay and San Diego, California (Leet et al. 1992; Love 1996). No California scorpionfish larvae were entrained at DCPP during the study. Commercial landings of rockfishes (*Sebastes* spp.) have historically included fishes in the genus *Sebastolobus* (thornyheads) represented by two species: the shortspine thornyhead (*Sebastolobus alascanus*) and the longspine thornyhead (*S. altivelis*). Until recently (mid-1980's), most rockfishes and thornyheads were landed together in a single market category, but they now are reported separately (Starr et al. 1998). Few thornyheads were entrained during the study at DCPP (**Appendix H**). The rockfishes (*Sebastes* spp.) are the most diverse genus in the Scorpaenidae with some 62 species reported from California coastal waters (Starr et al. 1998), approximately 85% of which are harvested in California commercial or sport fisheries.

Reproductive capacity of rockfishes is directly related to size, with larger females carrying significantly more eggs than smaller females. Rockfishes are viviparous with internal fertilization (Yoklavich et al. 1996), and the female retains the eggs until she extrudes hundreds to millions (e.g., *Sebastes paucispinis*; Moser 1967) of eyed, live larvae (Bloeser 1999). The larvae and juveniles can remain in the plankton from 1 mo to approximately 1 yr before settling into primarily benthic habitats as juveniles (Matarese et al. 1989; Moser 1996; Starr et al. 1998). This extended planktonic period makes environmental variation an important determinant of the

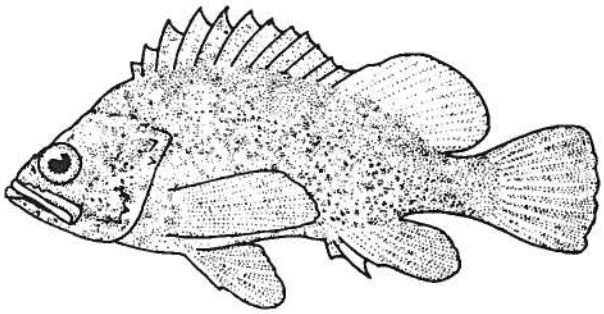
population abundance of many rockfish species since their vulnerable life stages are exposed to potentially adverse conditions for greater periods of time. Once on the bottom, individuals of many species migrate to deeper water as they mature.

Many rockfish species are closely related, and the larvae share many morphological and meristic characteristics, making it difficult to visually identify the individual larvae to species (Moser et al. 1977; Moser and Ahlstrom 1978; Baruskov 1981; Kendall and Lenarz 1987; Moreno 1993; Nishimoto in prep.). To standardize the identification of *Sebastes* spp. larvae, we collaborated with Mary Nishimoto at the University of California at Santa Barbara in developing a system of grouping rockfish larvae by pigment characters (**Table 5.2.4-1**; Nishimoto in prep.).

Using this system the two most abundant pigment groupings entrained at DCPP (*Sebastes* spp. V_De and *Sebastes* spp. V) were genetically identified to determine the species composition of each proposed pigment grouping (Vetter and Stannard 1999; **Appendix J**). Results from these analyses indicated that the species contained in the *Sebastes* spp. V_De grouping (brown, gopher, copper, calico, and grass rockfishes) were genetically indistinguishable from the species contained in the *Sebastes* spp. V_D_ grouping (kelp, black-and-yellow, quillback, China, and halfbanded). Therefore, these two groups were combined into a single complex called the *Sebastes* spp. KGB Complex. This complex has historically been used to describe indistinguishable young-of-the-year (YOY) kelp forest rockfishes thought to be either kelp (K), gopher (G), or black-and-yellow (B) rockfishes in subtidal surveys (David VenTresca, CDF&G, Monterey, CA, pers. comm.). Results of the genetic analyses on the *Sebastes* spp. V grouping indicated that more than half of the larvae selected for analysis were blue rockfish (*Sebastes mystinus*) leading to the promotion of blue rockfish from their original pigment grouping of *Sebastes* spp. Vdp to the *Sebastes* spp. V group and to the subsequent designation of a second complex, the Blue Rockfish Complex, used in this report to describe the *Sebastes* spp. V / *S. mystinus* pigment grouping.

5.2.4.1 KGB Complex (*Sebastes* spp. V_De & V_D_)

The larval rockfishes that comprise the KGB complex share pigment characteristics that make individual larvae difficult to identify to species (e.g., V_De and V_D_; **Table 5.2.4-1**). The primary pigment characters distinguishing this grouping from other rockfish larvae are a long ventral series (V_), either a long or developing dorsal series (D_ or De, respectively), and lack of pectoral pigmentation. Fishes that are classified into these pigment groupings also appear to be genetically similar (Vetter and Stannard 1999; **Appendix J**). Since most of the species in this complex have similar life histories and share the same adult habitats, the KGB complex can be considered a guild of nearshore, benthic, or epi-benthic rockfishes sharing similar ecological roles that form a basis for their combination into this complex above and beyond the morphometric and meristic similarities of their larval forms.



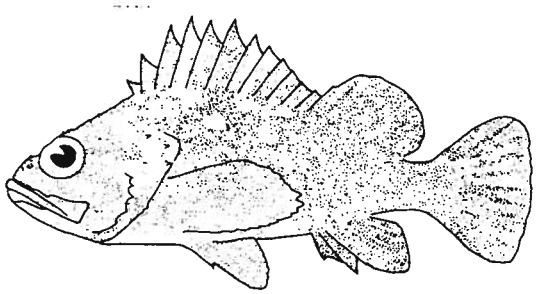
Sebastes atrovirens (Jordan and Gilbert 1880); kelp rockfish; length to 42 cm; Timber Cove, northern California to Punta San Pablo, central Baja California; inshore to 46 m; olive-brown to gray brown, with darker brown mottling, sometimes pinkish below (Miller and Lea 1972; Eschmeyer et al. 1983).

The kelp rockfish is typically found near the bottom in kelp beds or among the canopy blades living in rocky areas of nearshore waters (common at 10 m; Moser 1996). While they may occasionally move into deeper water during storms (Love 1996), they generally demonstrate high site fidelity throughout adult life (Miller and Geibel 1973; Lea et al. 1999). Principal food items are crustaceans and juvenile fishes, with some of those being other rockfish juveniles (Lea et al. 1999).

Kelp rockfish are important in both recreational and commercial fisheries. Recreational fishermen regularly take kelp rockfish on hook-and-line in and around kelp beds, and sport divers can easily spear them because of their docile nature (Love 1996; Lea et al. 1999). Commercially, this species is harvested on hook-and-line, in traps, and in gill nets (Love 1996). Many of them are sold as whole fish, sometimes live, in fish markets (Leet et al. 1992; Starr et al. 1998; Bloeser 1999). For the ports in the Monterey Bay National Marine Sanctuary (MBNMS) and San Francisco, kelp rockfish landings have increased markedly in the last ten years, with landings exceeding 14,500 MT in 1995. This equals an estimated average number of fish taken per year for the period 1980–1995 of 24,270 (Starr et al. 1998).

Kelp rockfish fecundity ranges from 344 to 403 eggs/g (female body weight), and spawning occurs once during late winter to spring (MacGregor 1970; Love et al. 1990; Moser 1996). The reproductive period lasts about 7 mo (Lea et al. 1999) and parturition occurs in April and May (Moreno 1993). Larval kelp rockfish are extruded at around 4.0 mm (Moser 1996). Young-of-the-year (YOY) first appear under nearshore kelp canopies from July through August and then as schooling fish in the water column from August through October. Lengths of YOY range from 20 to 40 mm total length (TL; Lea et al. 1999).

Longevity for the kelp rockfish is estimated at 15–20 yr (Burge and Schultz 1973; Lea et al. 1999; Ralph Larson, CSU San Francisco, pers. comm.). These ages are validated for fish up to about 5 yr and assumed to be accurate for older fish (Lea et al. 1999). The smallest sexually mature male was 246 mm TL at 4 yr, and the largest immature male was 338 mm TL (not aged; Lea et al. 1999). The smallest sexually mature female was 218 mm TL at 5 yr, and the largest immature female was 320 mm TL at 7 yr (Lea et al. 1999). Females attain 50% maturity at 4–5 yr and 100% maturity at 6–7 yr (Bloeser 1999).



Sebastes carnatus (Jordan and Gilbert 1880); gopher rockfish; length to 30 cm; Eureka, northern California to San Roque, central Baja California; inshore to 55 m; brownish to olive, mottled with pale areas, flesh-colored to slightly whitish areas on back (Miller and Lea 1972; Eschmeyer et al. 1983).

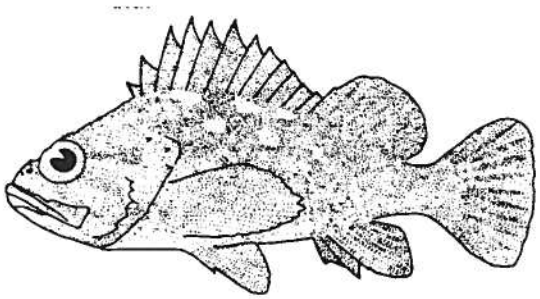
Gopher rockfish are benthic as adults, often found in crevices down to 55 m depth (Moser 1996). They appear to be very residential (Lea et al. 1999), but other evidence shows that they occasionally move over short distances to inhabit artificial reefs (Matthews 1986). Small fish eat zooplankton and larger fish eat benthic items like crustaceans, fishes (up to 80% YOY rockfishes), and octopi (Love 1996; Lea et al. 1999).

Gopher rockfish have both recreational and commercial fishery value. Recreational and commercial fishermen typically take gopher rockfish on hook-and-line while fishing for other species (Love 1996). Many of them are sold whole, sometimes live, in fish markets (Leet et al. 1992; Starr et al. 1998; Bloeser 1999). For the MBNMS ports and San Francisco, the gopher rockfish has shown variable landings over the past two decades. Landings exceeded 100,000 MT from 1980–81, declined to almost negligible landings by 1991, and then rapidly increased to greater than 118,200 MT in 1992. By 1995, landings exceeded 136,400 MT (Starr et al. 1998) possibly reflecting the increased effort in the live-fish fishery (Bloeser 1999). An estimated average number of gopher rockfish taken in recreational fisheries per year for the period 1980–1995 was 95,200 (Starr et al. 1998).

Gopher rockfish fecundity ranges from 176–307 eggs/g female weight, and spawning occurs once per season in spring (MacGregor 1970; Wyllie Echeverria 1987; Moser 1996). The reproductive period lasts 10 mo (Lea et al. 1999), and parturition occurs in March–May (Moreno 1993). Planktonic duration is approximately 2–3 mo (Larson 1980). Metamorphosing juveniles

first appear in nearshore habitats in mid- to late-June (Larson 1980). YOY first appear associated with nearshore reefs in July and August at 20 to 40 mm TL (Lea et al. 1999).

Longevity for the gopher rockfish was estimated at 24 yr (Lea et al. 1999). Age estimates were validated for fish up to about 5 yr and assumed to be accurate for older fish (Lea et al. 1999). The 24 yr old (316 mm TL) tagged fish reported by Lea et al. (1999) grew only 4 mm in nearly 11 years between capture dates. A 15 yr old tagged fish (282 mm TL) grew 10 mm TL in 6.7 yr between capture dates (Lea et al. 1999). The smallest sexually mature male in their study was 237 mm TL at 10 yr, and the largest immature male was 237 mm TL at 10 yr (Lea et al. 1999). The smallest sexually mature female was 207 mm TL (not aged), and the largest immature female was 306 mm TL at 9 yr (Lea et al. 1999). Females are estimated to attain 50% maturity at 4 yr (Wyllie Echeverria 1987; Bloeser 1999).



Sebastes chrysomelas (Jordan and Gilbert 1881); black-and-yellow rockfish; length to 39 cm; Eureka, northern California to Isla Natividad, central Baja California; intertidal to 37 m; mostly blackish or olive-brown, with large irregular yellow areas on back, paler below (Miller and Lea 1972; Eschmeyer et al. 1983).

Black-and-yellow rockfish typically inhabit holes and crevices in rocky areas and can be found up into the intertidal zone (Eschmeyer et al. 1983). They appear to be very residential (Lea et al. 1999), and short-range homing has been demonstrated (Hallacher 1984). Benthic organisms comprise a majority of the diet with crustaceans making up the largest proportion (89%; Lea et al. 1999). Of the rockfishes studied by Lea et al. (1999), black-and-yellow rockfish exhibited the greatest diversity of prey items.

Black-and-yellow rockfish are important to both recreational and commercial fisheries. Recreational and commercial fishermen typically take black-and-yellow rockfish on hook-and-line while fishing for other species (Love 1996). Many of them are sold whole, sometimes live, in fish markets (Leet et al. 1992; Starr et al. 1998; Bloeser 1999). For the MBNMS ports and San Francisco, the commercial landings for black-and-yellow rockfish increased from no landings between 1980 and 1991 (except 1987 where 30.8 MT were landed) to greater than 24.5 MT by 1995 (Starr et al. 1998). This increase may reflect the increasing effort directed at the live-fish fishery along the California coast (Bloeser 1999). The estimated average number of black-and-yellow rockfish taken annually in the northern California sport fishery during the period 1980–1995 was 24,860 (Starr et al. 1998).

Parturition timing and early development of black-and-yellow rockfish is similar to that of other species in the KGB complex. Black-and-yellow rockfish spawn between February and May (Larson 1980; Wyllie Echeverria 1987), and larvae are released annually (Lea et al. 1999). YOY have been observed in kelp beds in July and August at ca. 20 to 30 mm TL (Lea et al. 1999).

Longevity for the black-and-yellow rockfish was estimated at 21 yr (Lea et al. 1999). Age estimates were validated for fish up to about 5 yr and assumed to be accurate for older fish (Lea et al. 1999). Highest assigned age was 15 yr at 252 mm TL by Burge and Schultz (1973). The smallest sexually mature male was 239mm TL at 4 yr, while the largest immature male was 301 mm TL at 9 yr (Lea et al. 1999). The smallest sexually mature female was 243 mm TL at 6 yr and the largest immature female was 270 mm TL at 7 yr (Lea et al. 1999). Females are estimated to attain 50% maturity at 3 yr and 100% maturity at 4 yr (Wyllie Echeverria 1987; Bloeser 1999).

Other members of the KGB complex

Most members of the KGB complex dwell on or near the bottom of nearshore kelp beds and rocky reefs with peak abundance found at less than 50 to 100 m depth (Love 1996). The notable exception to this distribution is the halfbanded rockfish (*Sebastes semicinctus*), which is commonly observed on hard and soft, flat bottom habitat in waters up to 402 m deep (Miller and

Lea 1972; Eschmeyer et al. 1983; Love 1996). Geographic ranges for all members of this group begin off central Baja California, Mexico, with the exception of quillback and China rockfishes (Miller and Lea 1972; Eschmeyer et al. 1983; Love 1996). These latter two species begin their distribution near San Miguel Island off southern California (Miller and Lea 1972; Eschmeyer et al. 1983; Love 1996). The northern distribution of this group ranges from Monterey Bay and San Francisco, California for halfbanded and calico, and to the northern Gulf of Alaska for brown, copper, and China rockfishes (Miller and Lea 1972; Eschmeyer et al. 1983; Love 1996). Fishes with the most northerly distributions in this group typically attain the greatest total lengths and ages for the complex. Brown, copper, quillback, and grass rockfishes can attain maximum lengths of >50 cm (Miller and Lea 1972; Eschmeyer et al. 1983). This is also true for estimated longevity. Copper and quillback rockfishes may reach 41 yr and 76 yr, respectively, in the Canadian fishery (Yamanaka and Kronlund 1997). The smallest and shortest living rockfish of this group is the calico rockfish that attains a total length of 25 cm and has an estimated longevity of about 12 yr (Chen 1971; Miller and Lea 1972; Eschmeyer et al. 1983). The calico rockfish also has the lowest fecundity recorded in the KGB complex at about 2,000 eggs per female at 50% maturity but ranging to as high as 113,000 eggs per female (Haldorson and Love 1991). The most fecund rockfish from this group is the grass rockfish with about 760,000 eggs for a 26 cm female (Love and Johnson 1999). The highest age range at 50% maturity is 6–11 yr for quillback rockfish (Wyllie Echeverria 1987; Yamanaka and Kronlund 1997).

Little is known about the planktonic duration or natural mortality of the fish in the KGB complex. Planktonic duration was estimated for brown, calico, and gopher rockfishes at about 3 mo, 1–2 mo, and 2–3 mo, respectively (Larson 1980; Moser and Butler 1981; Matarese et al. 1989; D. Woodbury, NOAA, Tiburon Laboratories, pers. comm.). There are no published estimates of egg or larval survivorship for the species in the KGB complex.

The KGB complex formed a large part of the rockfish numbers in both the larval entrainment surveys (**Appendix H**) and subtidal fish surveys (Tenera 1997c) along the Diablo Canyon coastline. However, compared to the overall rockfish catch in California, which is comprised of both nearshore and offshore species, the KGB complex is a relatively minor, but increasing,

component (Leet et al. 1992; Starr et al. 1998; Bloeser 1999). Nearly all members of the KGB complex, except for halfbanded rockfish, are components of the live-fish fishery that targets nearshore habitats (Leet et al. 1992; Starr et al. 1998; Bloeser 1999). Brown rockfish landings averaged ca. 134 MT from 1979 to 1986, or 2–4% of the total recreational rockfish catch in California (Leet et al. 1992; Starr et al. 1998). The copper rockfish has followed a similar trend at about 2–3% of the recreational rockfish landings (Leet et al. 1992; Starr et al. 1998). The average number of brown and copper rockfishes landed per year at MBNMS ports and San Francisco over the period 1980–1995 was 160,670 and 106,390, respectively (Starr et al. 1998). The only KGB complex species listed by Bloeser (1999) within the top ten rockfishes landed on trolling gear in 1997 was the China rockfish, which ranked ninth. Similarly, the only KGB complex species listed within the top ten rockfishes landed in 1997 using hook-and-line gear was the gopher rockfish, which also ranked ninth. However, in this latter gear category, unspecified rockfishes ranked highest. In contrast, no KGB complex members were listed in the top ten trawl-caught rockfish landings that made up 88% of the west coast landings in 1997. In 1997, the average price for rockfish from commercial landings was varied by gear type with the lowest value ($0.93/kg) attributed to trawl gear and the highest ($3.15/kg) from pots and traps (Bloeser 1999). The KGB rockfishes that ranked in the top ten landings at Morro Bay, California in 1997 were grass (2), gopher (6), and black-and-yellow (8) rockfishes (ranks given in parentheses; Bloeser 1999). The average price in 1999 for all KGB complex rockfishes landed in the Morro Bay area (black-and-yellow, brown, China, copper, gopher, grass, and kelp) was $7.65/kg (Source: PSMFC Database).

Summary of Field Collections

Larval rockfishes in the KGB complex showed distinct seasonal peaks of abundance at the DCPP intake structure during the years 1996–1999 (**Appendix H**), with their greatest abundance tending to occur between March and July (**Figure 5.2.4-1**). There were 17,863 larval KGB rockfishes identified from 774 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 20% of the entrainment subsamples collected and processed during that period. 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 affected density in 1997 compared with the previous year. This warm water event may also have delayed spawning slightly during the 1997–98 spawning season (Woodbury and Ralston 1991).

Larval KGB rockfishes generally occurred in the DCPP study grid with similar seasonality to that observed at the DCPP intake structure (**Appendix H**) with peak abundance occurring in May of both 1998 and 1999 (**Figure 5.2.4-2**). There were 5,377 KGB rockfish larvae identified from 701 bongo net subsamples representing 23% of the study grid subsamples collected and processed from July 1997–June 1999. Mean density in May of each sampling year was very similar (May 1998: 0.29/m^3; May 1999: 0.28/m^3), indicating little change in abundance between the El Niño and subsequent La Niña season.

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 a minimum of 2.4 mm to a maximum of 8.0 mm (**Figure 5.2.4-3**). The central 98% of this length-frequency distribution resulted in minimum and maximum lengths for the analyses of 3.3 mm and 5.6 mm, respectively. The mean larval length in this distribution was 4.2 mm. The growth rate applied to the KGB complex (0.14 mm/d) derived from Yoklavich et al. (1996) was used to estimate ages of entrained larvae. Assuming that the shortest larvae of the central 98% of the length-frequency distribution was immediately post-extrusion and aged zero days, then the estimated ages of larvae entrained could range from zero up to ca. 34 d post-extrusion for the largest larva measured. The average estimated age of KGB larvae entrained at DCPP was estimated as 6 d post-extrusion.

There were significant differences between mean lengths (*t*-test: $p \leq 0.004$) and between length-frequency distributions (Kolmogorov-Smirnov test: $p < 0.001$) of larval KGB rockfishes from the DCPP intake structure and study grid despite the appearance of similarity between the two unimodal distributions (**Figure 5.2.4-4**). The large sample size (1,282 larvae from the intake structure and 2,850 larvae from the study grid) could cause the *t*-test to be highly sensitive to small differences between the means (Zar 1984). The mean standard lengths from the intake

structure and study grid were 4.2 mm and 4.1 mm, respectively. The maximum difference between the length-frequency distributions detected by the Kolmogorov-Smirnov goodness of fit test occurred at 4.1 mm, further indicating that a greater number of small individuals were collected from the study grid. While these differences were statistically detectable and significant, we believe that the biological significance of these differences is low.

Reported extrusion size for species in this complex ranges from 4.0–5.5 mm (Moser 1996). The fact that we observed many larvae smaller than the reported extrusion lengths can be explained partly by natural variation of extrusion lengths within the population and partly by the phenomenon of shrinkage following preservation (Theilacker 1980). If larvae less than 5.5 mm were zero days old, then the presently assumed aged-0 d length of 2.4 mm would lead to an overestimate of larval ages for those individuals with a consequent overestimation of entrainment mortality probability.

Estimating Total Annual Entrainment

Annual estimated numbers of KGB rockfish larvae entrained at DCPP varied relatively little between 1996–97 Analysis Period 1 ($\widehat{E_T}$ = 268,000,000) and 1997–98 Analysis Period 2 ($\widehat{E_T}$ = 199,000,000) (**Table 5.2.4-2**). 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. Values of $2 \cdot \widehat{SE}(\widehat{E_T})$ can be used to approximate 95% confidence intervals around the point estimates. The confidence intervals overlap for the estimates indicating that the differences between them were probably not statistically significant and that entrainment of these rockfish larvae was relatively constant between years. Therefore, reproductive output in the KGB rockfish complex did not appear to be substantially affected by the 1997–98 El Niño event (**Figure 5.2.4-1**).

The estimates of annually entrained KGB rockfish larvae above were adjusted (**Table 5.2.4-3**) to the long-term average Intake Cove surface plankton tow index ($\bar{I} / I_i$), calculated as the ratio between the 9 yr average ($\bar{I}$) of Intake Cove sampling (**Figure 5.2.4-5**) and the average annual

index estimated from these same tows during the year being adjusted (I_i). The average indices for the years 1997 and 1998 were 0.0703 and 0.0647 larvae/m^3, respectively, and the long-term average index for 1990–98 was 0.0722 larvae/m^3. Thus, the ratios used to adjust the 1997 and 1998 estimates of larvae entrained were 1.027 and 1.13, respectively, indicating that larval density was slightly lower than the long-term average during those years. The adjustments resulted in an estimate of 275,000,000 for 1996–97 Analysis Period 1 and 222,000,000 for 1997–98 Analysis Period 2. The same trends in overall abundance as noted for unadjusted entrainment values are 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.

Fecundity Hindcasting (FH)

The parameters required for the formulation of *FH* estimates for the KGB rockfish complex were compiled from references on different species within the group. Rockfish within the KGB complex spawn once per year. From multiple references on fecundity for various species within the KGB complex an average annual fecundity estimate of 213,158 eggs per female was used to calculate *FH* (DeLacy 1964: 52,000–339,000; MacGregor 1970: 44,118–104,101 and 143,156–182,890; Love and Johnson 1999: 80,000–760,000), with a CV of 100%. From the various sources of information on KGB complex rockfishes **(Table 5.2.4-4)**, longevity of 15 yr and age at maturation of 5 yr were used to calculate *FH*.

Survival of larvae from the time of release to entrainment was estimated using an instantaneous mortality rate of 0.14/d from blue rockfish (Mary Yoklavich, NOAA/NMFS/PFEG, Pacific Grove, CA, pers. comm.) over 6.21 d since no independent estimates of larval KGB rockfish mortality were available. This was calculated from a mean length at entrainment (4.17 mm) minus the minimum length of the central 98% of the length frequency distribution (3.3 mm) and divided by a larval growth rate for brown rockfish of 0.14 mm/d (Love and Johnson 1999; Yoklavich et al. 1996), as

$$\hat{S}_{larvae} = e^{(-0.14)(6.21)} = 0.419.$$

The estimated number of adult KGB rockfish females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year at DCPP ranged from 617 in 1996–97 to 497 in 1997–98 (**Table 5.2.4-5**). Hindcast adult estimates differed minimally among the analysis periods. These fairly low values also reflected the relatively high fecundity of the adults and the young average entrainment age for the larvae in this complex.

Sensitivity Analysis

Fecundity had the greatest leverage on recalculated values of $\widehat{FH}$ (**Table 5.2.4-6**). Varying this parameter in the model resulted in an $\widehat{FH}$ range of 96–3,200 adult females among the three analysis periods, compared to the original estimate of 497–617 adult females. Larval survivorship had the next greatest amount of leverage on recalculated $\widehat{FH}$.

Adult Equivalent Loss (AEL)

Larval blue rockfish survival estimates were used in *AEL* estimates for the KGB complex because of the absence of data on later-stage survival rates in KGB rockfishes (**Table 5.2.4-7**). 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. 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.

Similar to the results seen with $\widehat{FH}$, estimates of adult equivalents lost (*AEL*) due to larval entrainment were fairly similar among survey periods (**Table 5.2.4-8**). The $\widehat{AEL}$ of 1,120 adults predicted from $\hat{E}_{Adj-T}$ at DCPP during 1996–97 reflects the slightly higher abundance of KGB rockfish larvae present during this year when compared to 1997–98 Analysis Period 2 ($\widehat{AEL}$ = 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.

Sensitivity Analysis

Larval and juvenile survivorship had the greatest leverage on recalculated values of $\widehat{AEL}$ (**Table 5.2.4-9**). Varying these parameters in the model resulted in an $\widehat{AEL}$ range of 175–5,820 adult equivalents for the 1996–97 and 1997–98 analysis periods, compared to the original estimate of 905-1,120 adult equivalents. Pre-recruit survivorship had the next greatest leverage on estimates of $\widehat{AEL}$. Varying the estimate of entrainment did not substantially change the original value of calculated $\widehat{AEL}$.

Empirical Transport Model (ETM)

The lengths of entrained KGB larvae, excluding the largest 1% and smallest 1% of all measurements, ranged from 3.3 to 5.6 mm. Thus, KGB larvae may have been susceptible to entrainment for approximately 16.4 d based on a growth rate from larval brown rockfish of 0.14 mm/d (Love and Johnson 1999; Yoklavich et al. 1996). On average, however, entrained KGB rockfish larvae were estimated to be ca. 6.2 d old.

Monthly estimates of proportional entrainment (PE_i) in each *i*th survey ranged from a minimum of zero to a maximum of 0.587±0.297 ($\pm 1\,\widehat{SE}\left(\widehat{PE_i}\right)$) in March 1998 (**Table 5.2.4-10**). When both $\widehat{PE_i}$ and the annual proportion of larvae extruded in the *i*th survey period ($\hat{f}_i$) were equal to zero, no larvae were collected at either the DCPP intake or from the study grid. Highest $\widehat{PE_i}$ values occurred in March during the peak parturition of many species in the KGB complex (Moser 1996). However, the highest $\hat{f}_i$ values typically occurred a couple of months later in May. The greatest $\widehat{PE_i}$ occurred during the 1997–98 El Niño event when onshore water movement was also greatest.

Due to the broad dispersion of KGB rockfish larvae across the study grid (**Figure 5.2.4-2**), P_S for this complex was calculated using both alongshore and onshore current movements. Current speed and direction, measured at the single current meter near DCPP, were used to extend the estimates of P_S beyond the bounds of the study grid (see **Figure 4-3**). Density of KGB larvae within the study grid was extended alongshore on the basis of current speed and direction

constrained by average and maximum estimates of larval duration and the width of the study grid perpendicular to the shore. The pattern of larval KGB density within the study grid was extrapolated offshore using onshore current movement inferred from the current meter and then extended alongshore. The offshore extrapolation of abundance was also modeled on the basis of average and maximum estimated larval durations, yielding a range of possible entrainment probability values.

Total entrainment mortality probability ($\widehat{P_M}$) was calculated for estimated average and maximum larval duration in days based on current speed and direction measured at the single DCPP current meter (**Figure 5.2.4-6**). Both larval duration and currents were used to determine the magnitude of $\widehat{P_S}$. Alongshore extended estimates of P_M ranged from 0.04–0.05 over both mean and maximum larval duration and between years indicating that $\widehat{P_M}$ represented a nearly constant probability of larval entrainment alongshore. Offshore extrapolation of the study grid abundance of KGB rockfish larvae resulted in larger $1/\widehat{P_S}$ values with longer larval durations and consequently lower $\widehat{P_M}$ values. Estimates of P_M using offshore extrapolated abundance extended alongshore and based on the two larval durations ranged from approximately 0.02 in both years using mean duration to 0.01 in Analysis Period 3 (July 1997–June 1998) and 0.005 in Analysis Period 4 (July 1998–June 1999) using maximum larval duration. The differences between the offshore density extrapolated estimates of P_M showed that while onshore transport did not play a significant role in the variation of $1/\widehat{P_S}$ over short larval durations, its combination with different annual values of PE and alongshore flow did result in larger differences over longer durations. The average areas over which larval densities were extrapolated were more similar between 1997–98 and 1998–99 based on mean duration (5 and 4 times the grid study area, respectively) than on maximum duration (23 and 30 times the grid study area, respectively).

Interpretation of Assessment Results

Abundance of larval, juvenile, and adult KGB rockfishes varies between years, but there are no apparent decreasing trends in abundance over time. Subtidal fish abundance estimates collected at the DCPP South Control station from 1978 to 1998 (Tenera 1997c) varied seasonally between

years with occasional peaks (**Figure 5.2.4-7**). The relatively stable adult and YOY abundance noted in the DCPP subtidal fish observation dataset is also reflected by fairly stable larval abundance in Intake Cove plankton tows (**Figure 5.2.4-5**). Thus, local abundance of all life stages of KGB rockfishes is not declining, indicating that larval losses due to entrainment are not adversely impacting the local population.

Rockfishes in the KGB complex have both commercial and recreational fishery value (Starr et al. 1998; Bloeser 1999; Lea et al. 1999). Commercial groundfish landings from all gear types reported by Pacific States Marine Fishery Council (PSMFC) in their PacFIN database for the years 1993–1999 show landings of black-and-yellow, gopher, and grass rockfishes in the Morro Bay area increasing from 1993–1996 but then decreasing and remaining relatively stable through 1999 (**Figure 5.2.4-7**). These three species are the members of the KGB complex that ranked in the top ten landings at Morro Bay, California in 1997 (Bloeser 1999). The majority of the peak landings in 1996 were accounted for by a single species; the gopher rockfish (*Sebastes carnatus*), but this relationship has alternated between gopher and grass rockfishes (*S. rastrelliger*) over recent years. Bloeser (1999) indicates that catches of KGB rockfishes in California have risen over the last 10–15 yr. Starr et al. (1998) note that while catches were stable or increasing between 1980–1995, abundance of many species was much higher before 1980.

The PSMFC database provides ex-vessel prices of rockfishes landed at the ports in the Morro Bay area during 1999. The members of the KGB complex that were landed at these ports in 1999 were the black-and-yellow rockfish ($7.05/kg), brown rockfish ($7.38/kg), gopher rockfish ($6.84/kg), grass rockfish ($11.22/kg), kelp rockfish ($6.53/kg), China rockfish ($11.36/kg), and copper rockfish ($3.15/kg). Thus, the average price-per-kilogram for KGB complex rockfishes landed at Morro Bay area ports in 1999 was $7.65. In 1999, 64 MT (ex-vessel revenue ≅ $500,000) of these rockfishes were landed in the Morro Bay area (PSMFC PacFIN database). Grass, gopher, black-and-yellow, and copper rockfishes represented 93% of the catch by weight and, consequently, produced the most revenue within this group of landings. Copper, grass, calico, and kelp rockfish accounted for 85% of the southern California recreational catch by number.

Estimates of *AEL* were compared to harvest data assuming 100% catchability of the adult equivalents and assuming no compensatory mortality. These assumptions will cause an overestimate when directly converting *AEL* into 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 *AEL* estimate of 977 rockfishes translates to a potential direct economic loss of $7,474 based on the average price of $7.65/kg. This value represents 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).

Alternatively, estimates of proportional entrainment mortality could be translated into losses to local fisheries. The probable effect of entrainment losses at DCPP on fisheries is likely localized to the ports within the Morro Bay area since most fishes in this complex demonstrate high site fidelity (Heilprin 1992; Lea et al. 1999). In addition, extension of effects based on alongshore currents and larval duration indicate that the area potentially affected is three to seven times the size of the study grid, which is likely within the range of fishers from either Port San Luis or Morro Bay. The estimate of entrainment mortality ($\widehat{P_M}$) is between 4–5% for this area. Applying this range of proportional reduction to the local catch from the Morro Bay area in 1999 yields estimated dollar losses to the Morro Bay area fishery of approximately $20,000.

5.2.4.2 Blue Rockfish (*Sebastes* spp. V/*S. mystinus*) Complex

The species that comprise the blue rockfish complex are those rockfish larvae that have a short ventral pigment series and no dorsal series or pectoral pigmentation. This group currently consists of 16 rockfish species (**Table 5.2.4-1**). The blue rockfish (*S. mystinus*) is being placed in this group despite its designation as a member of the Vdp group. The pigment characteristics of young blue rockfish place them in this complex because they cannot be distinguished from other species in the group until they develop pectoral and dorsal pigmentation.

Vetter and Stannard (1999; **Appendix J**) genetically identified larvae from the *Sebastes* spp. V pigment group. Their conclusions indicated that more than half of the larvae in this pigment group were actually blue rockfish larvae. Additional evidence supporting the fact that many of the individuals in this complex would be blue rockfish is that they have the earliest parturition that occurs in January (Wyllie Echeverria 1987; Love 1996) and corresponds to the larval abundance data collected at DCPP in the present study (**Appendix H**). These data, combined with the fact that young blue rockfish larvae are indistinguishable from other rockfish larvae in this pigment group, led to the designation given of Blue Rockfish Complex.

Sebastes spp. V

Three species from the blue rockfish complex that commonly occur off central California are used here to represent the demography of other species from the complex: splitnose rockfish (*S. diploproa*), greenstriped rockfish (*S. elongatus*), and vermilion rockfish (*S. miniatus*). As a late addition to this grouping, the blue rockfish is treated separately as a representative of this complex as well. In the earliest larval stages, blue rockfish resemble the other larvae in this pigment grouping.

The splitnose rockfish is demersal and usually seen on soft substrata, often singly in small depressions or in near bottom aggregations, with a peak population density between depths of 92 and 460 m (Love 1996; Moser 1996). Pelagic juveniles congregate around drifting kelp and settle in shallow water (Love 1996). Greenstriped rockfish are demersal and semi-pelagic and found at depths of 25–500 m, but are most common between 46–245 m (Matarese et al. 1989; Love 1996). They are typically solitary on mud or cobble (Love 1996), and juveniles recruit in depths of 30–89 m depths (Love et al. 1990).

Vermilion rockfish are demersal and semi-pelagic and aggregate near or slightly above the bottom, often over high relief rocky substrata. They are most abundant in depths of 60 to at least 239 m within the Southern California Bight (Love et al.1990). Adults have been found in depths as shallow as 7 m near Diablo Canyon (Burge and Schultz 1973), and they appear to have a high

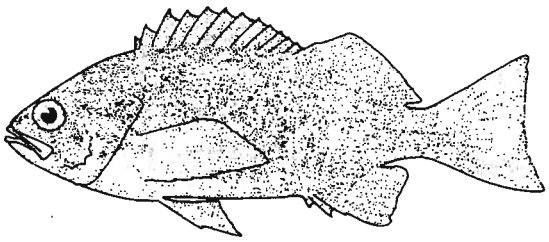
degree of site fidelity (Lea et al. 1999). Juveniles settle in water depths of 5–30 m and peak abundance of juveniles and small adults occurs in 90–149 m depths (Love et al. 1990).

Splitnose rockfish ranging in size from 195–365 mm produce 14,400–303,700 eggs/female (Phillips 1964). Their spawning period is from January through September (Phillips 1964; Wyllie Echeverria 1987; Moser 1996) with possibly two broods per year (Westrheim 1975). Growth for wild caught larvae was 0.21–0.96 mm/d (Boehlert 1981). Pre-juvenile splitnose remain pelagic for about 1 yr and reach sizes of 55 mm prior to settling to benthic habitat (Boehlert 1977). A planktonic duration of ca. 2.5 mo was estimated from growth rates and from extrusion and transformation lengths (Boehlert 1981; Moser 1996). Estimates of age at 50% maturity of splitnose rockfish vary from 4–7 yr (Phillips 1964; Chen 1971; Wyllie Echeverria 1987) with 100% maturity estimated to occur at 10 yr (Bloeser 1999). Greenstriped fecundity ranges from 26,000 at 50% maturity to 344,000 at ca. 100% maturity (Haldorson and Love 1991). Spawning occurs two or more times per year (Love et al. 1990) from January through June or July (Hart 1973; Wyllie Echeverria 1987). Female greenstriped rockfish reach 50% maturity at 7 yr (Wyllie Echeverria 1987) and their estimated longevity is 28 yr for females and 37 yr for males in the Southern California Bight (Love et al. 1990). The fecundity of vermilion rockfish ranging in length from 315 to 550 mm SL ranged from 3,300–1,625,600 eggs (Phillips 1964). Vermilions ranging in length from 460 to 680 mm SL produced 158,915–2,683,768 eggs per individual in the Southern California Bight (Love et al. 1990). At 50% maturity they are estimated to produce 151,000 eggs with a maximum possible production of 5,602,000 (Haldorson and Love 1991). Vermilion rockfish spawn once per year from September through March (Phillips 1964; Wyllie Echeverria 1987; Love et al. 1990). Age at 50% maturity for vermilion rockfish is estimated to be 3–5 yr for females (Chen 1971; Wyllie Echeverria 1987) and 8 yr at 100% maturity (Bloeser 1999). Young-of-the-year are common over sandy pockets in and near kelp beds (Lea et al. 1999).

Rockfishes in this species complex exhibit a wide range of estimates of longevity. Longevity of splitnose rockfish was estimated at 18 yr in early studies (Phillips 1964; Chen 1971). Radiometric age validation, however, indicates a longevity exceeding 80 yr (Bennett et al. 1982). Subsequent

studies utilizing otolith sections support these findings with estimates of 81–84 yr (Wilson and Boehlert 1990). No estimates of longevity exist for greenstriped rockfish. In an early study, the longevity of vermilion rockfish was estimated to be 22 yr (Chen 1971). More recently they have been aged as high as 43 yr (Paul Reilly, CDF&G, Monterey, CA, pers. comm.). The oldest fish reported in a study of whole vermilion rockfish otoliths was 29 yr for a 597 mm TL individual (Lea et al. 1999).

Each of these three species has varying levels of commercial and recreational fishery importance. Splitnose rockfish are primarily a commercially harvested fish with the major fishery occurring from central California northward to at least Washington; it was ranked 9th in the trawled rockfishes off the U.S. west coast in 1997 (Love 1996; Bloeser 1999). Splitnose rockfish ranked 4th in Monterey Bay, California landings, but did not place in the top ten in Morro Bay landings (Bloeser 1999). For the five major ports of the MBNMS and San Francisco, the splitnose rockfish has shown average landings of about 235.5 MT/yr, with catches declining since 1983 (Starr et al. 1998). Greenstriped rockfish are typically taken in the deep-water recreational fishery but are incidentally caught in commercial fisheries (Love 1996). This species is noted as exhibiting reductions in mean length over the period of 1950 to 1994 (Starr et al. 1998). For the five major ports of the MBNMS and San Francisco, the average number of greenstriped rockfish taken per year was 23,070 (Starr et al. 1998). Annual landings for the U.S. west coast, however, do not rank in the top ten for any fishing gear types used to harvest rockfishes (Bloeser 1999). Vermilion rockfish are an important component of both recreational and commercial fisheries and a majority of the catch occurs south of Monterey Bay (Love 1996). These fish make up 2–4% of the total marine recreational fishery catch in areas where they are abundant (Love 1996). Landings of vermilion rockfish in the MBNMS have varied widely over the period 1980–1995 but have averaged about 67.7 MT/yr (Starr et al. 1998). For the five major ports of the MBNMS and San Francisco, the average number of vermilion rockfish taken per year was 59,490 (Starr et al. 1998).



Sebastes mystinus (Jordan and Gilbert 1880); blue rockfish; length to 53 cm; northern limit uncertain, at least Vancouver I. (possibly Aleutian Is.) to Pt. Santo Tomas, northern Baja California; surface to 549 m; dark blue with light blue mottling (Miller and Lea 1972; Eschmeyer et al. 1983).

The typical habitat for blue rockfish is schooling in midwater or near kelp forests, at the surface near shallow to deep reefs, where adults can range to depths of 500 m (Moser 1996). Blue rockfish appear to have a high degree of site fidelity (Lea et al. 1999), and homing ability has been demonstrated (Heilprin 1992). Primary food items are midwater organisms (Gotshall et al. 1965 from Lea et al. 1999).

Blue rockfish are viviparous with planktonic larvae and juveniles (Moser 1996). Miller and Geibel (1973) estimated that their fecundity ranges from 50,000–300,000 eggs per female per year. However, a female that measured 405 mm TL had 524,000 young (Wales 1952). Spawning (extrusion of larvae) occurs November through March with a peak in January through February (Miller and Geibel 1973; Wyllie Echeverria 1987; Moser 1996), making this one of the earliest species of rockfish larvae to be released seasonally. Larvae are released once annually (Lea et al. 1999). However, Moreno (1993) found that this species may produce multiple spawns. Larvae are about 3.5 mm at parturition (Miller and Geibel 1973), with an average planktonic duration of 129 d, as calculated from observations of nine larvae (Dave Woodbury, NOAA/NMFS, Tiburon Laboratories, CA, pers. comm.). A period of 3–5 mo was observed for pelagic juveniles to settle to the nearshore benthos (Adams and Howard 1996). Young-of-the-year (YOY) were first observed in nearshore kelp beds in May and June at 40–60 mm TL (Lea et al. 1999). In April, juveniles of about 45–50 mm TL concentrate in shallow rocky areas and in kelp canopies. By October these fish range from 65–90 mm TL (Miller and Geibel 1973). Estimated instantaneous

mortality for juveniles in their first year of life ranged from 0.001 to 0.008 (Adams and Howard 1996).

Longevity for the blue rockfish was estimated at 17 yr for males and 24 yr for females. Scales were used to estimate age, however, and no validation was performed (Miller and Geibel 1973). Growth of 0.23 to 0.35 mm/d was observed for 85 mm individuals (Miller and Geibel 1973), and mean monthly growth from tag returns on adults was 2.46 mm (Wales 1952). The smallest sexually mature male Lea et al. (1999) collected was 219 mm TL, and the largest immature male was 332 mm TL. The smallest sexually mature female was 196 mm TL, and the largest immature female was 293 mm TL. Females were estimated to attain first maturity at 4–5 yr, 50% maturity at 6 yr, and 100% maturity at 11 yr (Wales 1952; Wyllie Echeverria 1987; Bloeser 1999).

Blue rockfish are one of the most important rockfish in recreational sport fishery along the California coast. In some years, at some locations, up to 31% of all fishes taken in the marine recreational fishery were blue rockfish (Love 1996). Blue rockfish are taken on hook-and-line or while diving (Love 1996). The commercial fishery is typically small with a few exceptions (Starr et al. 1998). For the five major ports of the MBNMS and San Francisco, the blue rockfish has had variable but high landings from 1980 to 1995 (Starr et al. 1998). The calculated average number of fish taken per year was the second highest of all rockfishes at 752,000 (Starr et al. 1998). In 1996, the average price for rockfish from commercial hook-and-line landings was approximately $0.43/kg (Bloeser 1999).

Other members of the V-group complex

All but four of the rockfishes in this group have their centers of distribution in the northern Pacific Ocean. The northernmost range of the rosy rockfish is Puget Sound, Washington, while the treefish, honeycomb, and Mexican rockfishes' northernmost range is off central California. The Mexican rockfish extends all the way into the Gulf of California. The rougheye and yelloweye rockfishes are the largest and longest living rockfishes in this group. The rougheye can attain a length of 96 cm TL and an age of 95 yr (Miller and Lea 1972; Nelson and Quinn II

1986). The yelloweye rockfish can attain a similar length of 91 cm TL and a longevity of 118 yr, one of the highest ever recorded in fishes (Kris Munk, Alaska Department of Fish and Game, Sitka, AK, pers. comm.). The rosy and honeycomb rockfishes appear to be the shortest lived (18 and 17 yr, respectively; Chen 1971) and are among the smallest (36 cm and 27 cm TL, respectively; Chen 1971) in this group. Age and longevity information is not available for the Mexican, darkblotched, pygmy, and treefish rockfishes. The most fecund species for which information was available is the yelloweye rockfish that can produce up to 2.7 million eggs (Clemens and Wilby 1961). However, there is a paucity of information for many of the remaining species regarding fecundity. The latest ages at 50% maturity belongs to the rougheye rockfish at 20 yr (Gunderson 1997) and the yelloweye rockfish at 18 yr (Yamanaka and Kronlund 1997). While these figures may represent the highest ages at maturity for the group, given that these two species reach the greatest length and age, there is a lack of information in this regard for several of the other long-lived group members, namely, silvergrey and darkblotched rockfishes.

Summary of Field Collections

Larvae in the blue rockfish complex showed distinct seasonal peaks in abundance at the DCPP intake structure during the years 1996–1999 (**Appendix H**), with their greatest abundance tending to occur between January and March (**Figure 5.2.4-8**). Larvae in this complex appeared in the water column sooner than did those in the KGB complex. There were 2,731 larvae identified from 537 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 11% of the subsamples collected and processed from that location during that period. An El Niño that began during the spring of 1997, was detected along the coast of California in the fall of 1997 (Lynn et al. 1998; NOAA 1999), and may account for lowered density in 1998 compared with the previous season. The warm water El Niño event may also have delayed spawning slightly in 1997–98 (Woodbury and Ralston 1991). Additionally, VenTresca et al. (1995) demonstrated that blue rockfish in post- El Niño years do not have sufficient reserves to produce large amounts of young, so these years typically have low larval abundance.

Blue rockfish complex larvae generally occurred in the DCPP study grid and at the DCPP intake structure with similar seasonality (**Appendix H**), and their peak abundance occurred in January and February (**Figure 5.2.4-9**). As with the entrainment data, density appears to be depressed during the 1997–98 El Niño relative to density estimates from 1999. There were 2,965 larvae identified from 792 bongo net subsamples representing 26% of the study grid subsamples collected and processed from July 1997–June 1999. Mean density in the peak season of blue rockfish complex larval abundance appears to be greater during the Winter/Spring of 1996–97 than that measured during 1997–98 (the El Niño year).

Standard lengths of all blue rockfish complex larvae collected from the DCPP intake structure between October 1996 and June 1999 and measured (2,407 larvae) ranged from a minimum of 1.7 mm to a maximum of 8.9 mm (**Figure 5.2.4-10**). The central 98% of this length-frequency distribution yielded a truncated distribution with minimum and maximum lengths for the analyses of 2.7 mm and 4.5 mm, respectively. The mean larval length in this distribution was 3.6 mm. The growth rate applied to the blue rockfish complex (0.14 mm/d) derived for brown rockfish from Yoklavich et al. (1996) and reported in Love and Johnson (1999) was used to estimate ages of entrained larvae. Assuming that the shortest larva of the central 98% of the length-frequency distribution were immediately post-extrusion and aged zero days, the estimated ages of larvae entrained could range from zero days up to 44 d post-extrusion for the largest larva measured. The average estimated age of blue rockfish larvae entrained at DCPP was 6 d post-extrusion.

Significant differences were detected when comparing mean larval lengths (*t*-test: $p < 0.001$) and length-frequency distributions (Kolmogorov-Smirnov test: $p < 0.001$) of blue rockfish complex larvae from the DCPP intake structure and study grid despite the apparent similarity of the distributions in the two locations (**Figure 5.2.4-11**). It is likely that the large number of individuals measured and tested with the *t*-test led to the ability of the statistical test to detect a difference between the two means of 3.6 mm at the intake structure and 3.5 mm from the study grid. The maximum difference between the two distributions measured by the Kolmogorov-Smirnov test occurred at 3.9 mm and indicated that more smaller individuals were found in the grid than at the intake structure, confirming the result of the *t*-test. While slight differences were

detected as statistically significant, it appears that the larvae found in these two locations were from very similarly structured populations.

Reported extrusion size for species in this complex ranges from 3.8–5.2 mm (Moser 1996). The fact that we observed some larvae smaller than the reported extrusion lengths can be explained partly by natural variation of extrusion lengths within the population and partly by the phenomenon of shrinkage following preservation (Theilacker 1980). If all larvae less than 5.2 mm were assumed to be aged-0 d, then the present $\widehat{P_M}$ will be an underestimate of the probability of entrainment since larval duration could have been overestimated by approximately 17 d.

Estimating Total Annual Entrainment

Annual estimates at DCPP of the number of entrained larvae from the blue rockfish complex were greatest during 1996–97 Analysis Period 1 ($\widehat{E_T}$ = 49,700,000) when compared to 1997–98 Analysis Period 2 ($\widehat{E_T}$ = 5,410,000) (**Table 5.2.4-11**). In contrast to the relative similarity among annual estimates of entrained KGB rockfish larvae, there was a large difference between the sampling years for larvae of fishes in the blue rockfish complex. Non-overlapping 95% confidence intervals approximated by $2 \cdot \widehat{SE}(\widehat{E_T})$ indicated that these differences were probably statistically significant. The second year of sampling, when entrained numbers of larvae from the blue rockfish complex were lower, was during the El Niño event. This climatological and oceanographic event could have had an effect on the reproductive output of rockfishes in this complex.

Annual estimates of the number of blue rockfish complex larvae entrained were adjusted (**Table 5.2.4-12**) to a long-term mean based on annual indices using weekly Intake Cove surface plankton tows (**Figure 5.2.4-12**). The average indices for the years 1997 and 1998 (I_i) were 0.0181 and 0.0049 larvae/m^3, respectively, while the long-term average index ($\overline{I}$) was 0.0307 larvae/m^3 for the years 1990–1998. The ratio combining these values ($\overline{I} / I_i$) yielded the correction factor used to adjust the annual entrainment estimates to the long-term average. These

factors were 1.69 for 1997 and 6.26 for 1998. Numbers of entrained larvae for each of the analysis periods were increased by these corrections, indicating that larval abundance of the blue rockfish complex was below the long-term average during both sampling years. Non-overlapping 95% confidence intervals approximated by $2 \cdot \widehat{SE}(\hat{E}_{Adj-T})$ indicated that the differences between the point estimates for 1996–97 Analysis Period 1 ($\hat{E}_{Adj-T}$ = 84,000,000) and 1997–98 Analysis Period 2 ($\hat{E}_{Adj-T}$ = 33,800,000) were statistically significant even after the adjustment.

Fecundity Hindcasting (FH)

The parameters required for the formulation of *FH* estimates for the blue rockfish complex were compiled from references on different species within the group. Rockfishes within this group spawn once or twice per year. From multiple references on fecundity for various species within the complex, an annual fecundity estimate of 508,607 eggs per female, with assumed CV=100%, was used to calculate *FH* (Wales 1952: 524,000; Phillips 1964: 14,400–303,700 and 3,300–1,625,600; Miller and Geibel 1973: 50,000–300,000, and 524,000; MacGregor 1970: 69,599; Love et al. 1990: 158,915–2,683,768; Haldorsen and Love 1991: 26,000–344,000). For blue rockfish complex fishes, longevity estimates of 24.7 yr and age at maturation of 6 yr were used in *FH* calculations (longevity- Chen 1971: 22 yr; Miller and Geibel 1973: 24 yr; Love et al. 1990: 28 yr; Age at 50% maturity - Wyllie Echeverria 1987: 6 yr; Bloeser 1999: 6 yr; Lea et al. 1999: 6 yr). Larval survivorship was estimated from extrusion to entrainment using an instantaneous blue rockfish larval mortality of 0.14/d (Mary Yoklavich, NOAA/NMFS/PFEG, Pacific Grove, CA, pers. comm.), over 6.43 d based upon the minimum 1% larval length and mean larval length of 2.7 to 3.6 mm and a larval brown rockfish growth rate of 0.14 mm/d (Yoklavich et al. 1996; Love and Johnson 1999):

$$\hat{S}_{larvae} = e^{(-0.14)(6.43)} = 0.407 .$$

The estimated number of adult blue rockfish complex females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year at DCPP ranged from 43 in 1996–97 to ca. 20 in 1997–98 (**Table 5.2.4-13**). The reduction in the number of adult females

estimated from the second sampling season reflects the lowered larval abundance during this time period. These fairly low values also reflected the relatively high fecundity of the adults and the young average entrainment age for the larvae from this complex.

Sensitivity Analysis

Fecundity and larval survival had the greatest leverage on recalculated values of $\widehat{FH}$ (**Table 5.2.4-14**). Varying fecundity or larval survival in the model resulted in an $\widehat{FH}$ range of ca. 3–225 adult females, compared to the original estimate of 18–43 adult females. Longevity had the next greatest amount of leverage on recalculated $\widehat{FH}$, while varying either entrainment or maturation had no effect on $\widehat{FH}$.

*Adult Equivalent Loss (*AEL*)*

The parameters used to formulate *AEL* estimates for the blue rockfish complex were survival estimates for the blue rockfish (**Table 5.2.4-15**). Early mortality was provided by Mary Yoklavich (NOAA/NMFS/PFEG, Pacific Grove, CA, pers. comm.) for 6.43 to 20 d (Z=0.14/d), 20 to 60 d (Z=0.8/d), 60 to 80 d (Z=0.04/d), 180 to 365 d (Z= 0.0112/d). The survival rate from entrainment of the larval stage to recruitment into the fishery also included a fifth stage prior to recruitment into the fishery (Z= 0.000548/d). Survivorship, to an assumed age at recruitment into the fishery of three years, was apportioned into these stages, and *AEL* was calculated assuming the entrainment of a single age class having the average age of recruitment.

Estimated numbers of equivalent adults of the blue rockfish complex decreased by more than half between the 1996–97 analysis period (353 adults) and the 1997–98 analysis period (164 adults: **Table 5.2.4-16**). These values directly reflect the changing abundance of larvae during this time. Despite the large differences in the estimates of larvae for the blue rockfish complex between these two periods (**Table 5.2.4-11**), the relatively small difference in estimated adults may indicate that year-to-year recruitment may be less variable.

Sensitivity Analysis

Larval and juvenile survivorship had the greatest leverage on recalculated values of $\widehat{AEL}$ (**Table 5.2.4-17**). Varying these parameters in the model resulted in an $\widehat{AEL}$ range of 27–1,830 adult equivalents, compared to the original estimate of 142–353 adult equivalents. Pre-recruit survivorship had the next greatest leverage on estimates of $\widehat{AEL}$. Varying the estimate of entrainment did not substantially change the original value of calculated $\widehat{AEL}$.

*Empirical Transport Model (*ETM*)*

The lengths of entrained blue rockfish complex larvae, excluding the largest 1% and smallest 1% of all measurements, ranged from 2.7 to 4.5 mm. Thus, larvae from the central 98% of the length-frequency distribution may have been susceptible to entrainment for approximately 12.8 d based on a growth rate for brown rockfish of 0.14 mm/d (Love and Johnson 1999; Yoklavich et al. 1996). On average, entrained blue rockfish complex larvae were ca. 6.4 d old.

Monthly estimates of proportional entrainment (PE_i) in each *i*th survey ranged from a minimum of zero to a maximum of 0.037±0.022 (±1 $\widehat{SE}\left(\widehat{PE_i}\right)$) in April 1999 (**Table 5.2.4-18**). When both $\widehat{PE_i}$ and the annual proportion of larvae extruded in the *i*th survey period ($\hat{f}_i$) were equal to zero, no larvae were collected at either the DCPP intake structure or from the study grid. When $\widehat{PE_i}$ =0 but $\hat{f}_i$>0, larvae were collected at the DCPP intake structure during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. The highest $\widehat{PE_i}$ values occurred in April of both years which typically had the higher associated $\hat{f}_i$ values. The next highest $\widehat{PE_i}$'s occurred in January or February. This latter time frame corresponds to the expected parturition for blue rockfish (Wyllie Echeverria 1987; Moser 1996), but a peak later in the year also contained blue rockfish larvae (**Appendix J**).

Due to the broad dispersion of blue rockfish complex larvae across the study grid (**Figure 5.2.4-9**), P_S for this complex was calculated using both alongshore and onshore current movement. Current speed and direction measured at the single current meter near DCPP were used to extend the estimates of P_S beyond the bounds of the study grid. Abundance of blue rockfish complex larvae within the study grid was extended alongshore on the basis of current

speed and direction constrained by average and maximum estimates of larval duration and the width of the study grid perpendicular to the shore. The pattern of larval blue rockfish complex abundance within the study grid was extrapolated offshore using onshore current movement inferred from the current meter. The offshore extrapolation of abundance was also modeled on the basis of mean and maximum estimated larval durations to yield a range of possible entrainment probability values.

Total entrainment mortality probability ($\widehat{P_M}$) was calculated for estimated average and maximum larval duration based on current speed and direction measured at the single current meter (**Figure 5.2.4-13**). There were large differences for $\widehat{P_M}$ between the two analysis periods, but little change in $\widehat{P_M}$ when varying larval duration within an analysis period. Estimates of $\widehat{P_M}$ using alongshore current movement only varied from ~0.004 (both mean and maximum durations) in the 1997–1998 analysis period to ~0.05 during 1998–1999. When both alongshore and onshore currents were considered, $\widehat{P_M}$ ranged from ~0.001 in the 1997–1998 analysis period regardless of duration to 0.02 for mean duration and 0.004 for maximum duration in the 1998–1999 analysis period. Blue rockfish complex larvae were in much greater abundance in both entrainment and study grid subsamples during 1998–1999 when compared to 1997–1998. El Niño conditions may have contributed to the reduced numbers of larvae during 1997–1998.

Interpretation of Assessment Results

Fishes in the blue rockfish complex have both commercial and recreational fishery value and can be relatively abundant in both larval and YOY surveys near DCPP (**Figure 5.2.4-14**). These data can be compared with local fishery catch trends from the Morro Bay area to determine if there are any concurrent trends in these two indices of rockfish abundance (**Figure 5.2.4-14**). The probable effect on fisheries is likely localized to the ports within the Morro Bay area since most fishes in this complex demonstrate high site fidelity (Heilprin 1992), and, in general, most local catches are landed locally. Long-term subtidal fish observations at Diablo Canyon's South Control site (Tenera 1997c) show *Sebastes* spp. V group abundance characterized by periods of stability (e.g., 1980–1990) punctuated by rapid changes (e.g., 1991–1993) again followed by a period of stability (1993–1998). The overall trend from 1978–1998 appears to be one of declining abundance. Similar trends were also observed in the Intake Cove plankton tow data (**Figure 5.2.4-12**). Local catch records from Morro Bay area ports also show a generally decreasing trend in landings since 1993. While there is a possibility that larval entrainment losses at DCPP play a part in these declines, it seems more likely, based on the estimates of equivalent adult losses and annual entrainment probabilities for this complex, that general declines in northerly distributed species (e.g., blue rockfish) are attributed to a climate regime shift that began in the late 1970's as noted by Holbrook et. al. (1997).

In 1999, 14.6 MT of blue rockfish (*Sebastes mystinus*), greenstriped rockfish (*S. elongatus*), treefish (*S. serriceps*), vermilion rockfish (*S. miniatus*), and yelloweye rockfish (*S. rubberimus*) represented an ex-vessel revenue of $61,500 landed in the Morro Bay area. Blue and vermilion rockfish represented the majority of the catch by weight and also produced most of the revenue. Vermilion and blue rockfish accounted for 70% of the individuals captured in the southern California recreational fishery in 1998 (PSMFC PacFin Database).

The estimates of *AEL* (142–353 fish) represent the estimated loss of 3 yr old fish to the population. Converting these adult equivalents to direct losses from a fishery requires assumptions of 100% catchability of the individuals and no compensatory mortality. Both of

these assumptions will lead to an overestimate of the loss to the fishery. Using an estimate of the weight per adult blue rockfish in the fishery of 0.5 kg, we can estimate that 343 adults represent 172 kg of 3 yr old recruits that represent ca. 1% of the catch landed in the Morro Bay area in 1999. This 172 kg of biomass potentially lost to the fishery could have been worth approximately $740 in 1999 based on the average ex-vessel price paid in Morro Bay for blue, greenstriped, vermilion, and yelloweye rockfishes, as well as treefish.

Fecundity hindcast estimates can also be aligned to the age at recruitment used in *AEL* calculations. This is accomplished by hindcasting the number of females aged 12.2 yr (i.e., one third of the duration between maturity and longevity) to the age of *AEL* recruits (3 yr). The estimates of *FH* from the three analysis periods ranged from 18–43 adult females aged 12.2 yr. Using an instantaneous natural mortality rate (Z=0.000548/d), the relationship $2FH \equiv AEL$ yields 114–273 three year-old recruits. This aligned estimate, when compared with the *AEL* estimate above (142–353), is similar and confirms the validity of the parameter estimates used in both *FH* and *AEL* estimates.

Alternately, estimates of proportional larval entrainment mortality to the local area can be converted to proportionate losses to local fisheries assuming no compensatory mortality. Entrainment effects ($\widehat{P_M}$) ranged over extrapolated areas from 4–6 times and 17–21 times the study grid area, using mean and maximum larval durations, respectively, as the time over which current flows were measured. These extrapolations resulted in $\widehat{P_M}$ estimates that ranged from around 0.001 to 0.02. This range of proportional loss, applied to the 1999 ex-vessel revenue from Morro Bay area ports ($61,500) could represent between $62–1,230 during that year.

Table 5.2.4-1. Pigment groups, parturition peak periods (PPM), and seasonality of preflexion *Sebastes* spp. 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 stop)
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)

LETTER CODE	SPECIES	COMMON NAME	SUBGENUS	'77-'84 PPM	Parturition period N and C CA (Wyllie Echeverria 1987)
V_P	**Long ventral series, no dorsal, pectoral pigment**				
	S. chlorostictus	greenspotted	Sebastomus	?	Apr-Sep
	S. ensifer	swordspine	Sebastomus	?	Uncommon in central CA (Miller and Lea 1972)
V_D	**Long ventral series, short dorsal series, no pectoral pigment**				
	S. saxicola	stripetail	Allosebastes	Jan	Jan-Mar (Nov-Mar, Phillips 1964)
V_D_	**Long ventral series, short dorsal series, no pectoral pigment**				
	S. atrovirens	kelp	Mebarus	ND	ND
	S. chrysomelas	black and yellow	Pteropodus	Feb	Feb-Mar (Jan-May (Larson 1980))
	S. maliger	quillback	Pteropodus	Apr	Apr-July
	S. nebulosus	China	Pteropodus	?	?
	S. semicinctus	halfbanded	Allosebastes	ND	Jan-Apr for southern CA (Love et a. 1990)
V_De	**Long ventral series, elongating dorsal series, pectoral pigment**				
or	*S. auriculatus*	brown	Auctospina	June	Dec-Jan, principally May-July (May, Larson 1980)
V_DeP	*S. carnatus*	gopher	Pteropodus	Mar	Mar-May (May, Larson 1980)
or	*S. caurinus*	copper	Pteropodus	Feb	Feb
V_dep	*S. dalli*	calico	Allosebastes	?	Uncommon in central CA (Miller and Lea 1972)
	S. rastrelliger	grass	Pteropodus	ND	ND

Table 5.2.4-1. (continued). Pigment groups, parturition peak periods (PPM), and seasonality of preflexion *Sebastes* spp. larvae from Nishimoto (in prep.).

LETTER CODE	SPECIES	COMMON NAME	SUBGENUS	'77-'84 PPM	Parturition period N and C CA (Wyllie Echeverria 1987)
V	**Short ventral series, no dorsal series, no pectoral**				
	S. aleutianus	rougheye	Zalopyr	ND	
	S. alutus	POP	Acutomentum	Mar	Jan-Mar (?)
	S. brevispinis	silvergrey	Acutomentum	ND	
	S. crameri	darkblotched	Eosebastes	Jan	Nov-Mar (Nov-Mar, Larson 1980)
	S. diploproa	splitnose	Allosebastes	July	Jan-Sep (Feb-July, Phillips 1964)
	S. elongatus	greenstriped	Hispaniscus	May	May-July
	S. macdonaldi	Mexican	Acutomentum	ND	
	S. miniatus	vermilion	Rosicola	Sep	Sep (Nov, Moser 196; Nov-Mar, Phillips 1964)
	S. nigrocinctus	tiger	Sebastichtys	ND	
	S. proriger	redstripe	Allosebastes	?	July-Aug
	S. rosaceus	rosy	Sebastomus	June	Apr-July
	S. ruberrimus	yelloweye	Sebastopyr	June	Apr-July (?)
	S. serriceps	treefish	Sebastocarus	ND	
	S. umbrosus	honeycomb	Sebastomus	ND	ND
	S. wilsoni	pygmy	Acutomentum	ND	
	S. zacentrus	sharpchin	Allosebastes	?	May-June
VP	**Short ventral series, no dorsal series, various patterns of pectoral pigmentation** (At smaller size might fit into group **V** above)				
	S. constellatus	starry	Sebastodes	Apr	Apr-May
	S. eos	pink	Sebastomus	ND	ND
	S. goodei	chilipepper	Sebastodes	Jan	Nov-June (Nov-Mar, Phillips 1964)
	S. helvomaculatus	rosethorn	Sebastomus	?	May-June
	S. levis	cowcod	Hispaniscus	Dec	Dec-Feb
	S. melanostomus	blackgill	Eosebastes	Feb	Feb-Apr
	S. paucispinis	bocaccio	Sebastodes	Feb	Jan-May (Nov-Mar, Moser 1967)
	S. rosenblatti	greenblotched	Sebastomus	ND	ND
	S. rubrivinctus	flag	Hispaniscus	?	July
Vdp	**Short ventral series, develops dorsal series, develops various patterns of pectoral pigmentation** (At stage 1-2 can confuse with **VP** above due to lack of dorsal pigmentation.)				
	S. entomelas	widow	Acutomentum	Feb	Dec-Apr (Nov-Mar, Phillips 1964)
	S. flavidus	yellowtail	Sebastosomus	Feb	Jan-July (Nov-Mar, Phillips 1964)
	S. melanops	black	Sebastosomus	Feb	Jan-May
	S. mystinus	blue	Sebastosomus	Jan	Nov-Jan (Nov-Jan, Wales 1952)
	S. rufus	bank	Acutomentum	Feb	Dec-May
	S. serranoides	olive	Sebastosomus	Feb	Jan-Mar
VD	**Short ventral series, short dorsal series**				
	S. aurora	aurora	Eosebastes	Apr	Mar-May
	S. babcocki	redbanded	Rosicola	May	May
	S. gilli	bronzespotted	Sebastosomus	?	Uncommon in central CA (Miller and Lea 1972)
	S. hopkinsi	squarespot	Acutomentum	Mar	Feb-Mar
	S. jordani	shortbelly	Sebastodes	Feb	Feb-Apr (Nov-Apr, Phillips 1964)
	S. ovalis	speckled	Acutomentum	May	May
	S. pinniger	canary	Rosicola	Dec	Dec-Mar (Nov-Mar, Moser 1967)
Species without descriptions or illustrations					
	S. phillipsi	chameleon	?	?	Uncommon in central CA (Miller and Lea 1972)

Table 5.2.4-2. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for KGB rockfish larvae (*Sebastes* spp. V_De/V_D_) from three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	268,000,000	24,000,000
2) Oct 1, 1997–Sep 30, 1998	199,000,000	25,900,000
3) July 1, 1997–June 30, 1998	199,000,000	25,900,000

Table 5.2.4-3. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for KGB rockfish larvae (*Sebastes* spp. V_De/V_D_) from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	275,000,000	24,700,000
2) Oct 1, 1997–Sep 30, 1998	222,000,000	28,900,000
3) July 1, 1997–June 30, 1998	222,000,000	28,900,000

Table 5.2.4-4. Age at sexual maturation and longevity (years) for three rockfish species within the KGB complex.

Rockfish species	Maturation age	Longevity
kelp	5 yr[4] 4–5 yr[3]	15 yr[1,4,5]
black and yellow	3–4 yr[2] 6 yr[4]	15 yr[1]
gopher	4 yr[2] 9 yr[4]	

[1]Burge and Schultz 1973
[2]Wyllie Echeverria 1987
[3]Bloeser 1999
[4]Lea et al. 1999
[5]Ralph Larson, CSU San Francisco, personal communication

Table 5.2.4-5. Estimated number of adult KGB rockfish (*Sebastes* spp. V_De/V_D_) females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{FH})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	617	1,470	31,500	12
2) Oct 1, 1997–Sep 30, 1998	497	1,190	25,400	10
3) Jul 1, 1997–Jun 30, 1998	497	1,190	25,400	10

Table 5.2.4-6. KGB rockfishes (*Sebastes* spp. V_De/V_D_): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH} = 617$				
$\hat{E}_{Adj-T}$	275,000,000	526	708	182
$\hat{S}_{larvae}$	0.419	258	3,200	2,940
# Eggs/yr	213,000	119	3,200	3,080
Longevity	15	247	1,230	987
Maturation	5	493	1,230	740

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH} = 497$				
$\hat{E}_{Adj-T}$	222,000,000	391	603	213
$\hat{S}_{larvae}$	0.419	208	2,570	2,370
# Eggs/yr	213,000	95.9	2,570	2,480
Longevity	15	199	994	795
Maturation	5	398	994	596

Table 5.2.4-6 (continued). KGB rockfishes (*Sebastes* spp. V_De/V_D_): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T}$ ± 1.645 $\widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 497				
$\hat{E}_{Adj-T}$	222,000,000	391	604	213
$\hat{S}$ *larvae*	0.419	208	2,580	2,370
# Eggs/yr	213,000	96	2,580	2,480
Longevity	15	199	995	796
Maturation	5	398	995	597

Table 5.2.4-7. Three-year survival for the KGB rockfish complex larvae (*Sebastes* spp. V_De/V_D_), based on blue rockfish data. Survival was estimated from release as $\hat{S} = e^{(-Z)(Day(end)-Day(start))}$. Daily instantaneous mortality rates (Z) up to 1 yr of blue rockfish, *Sebastes mystinus*, larvae that were used to calculate KGB larval survivorship were provided by Mary Yoklavich (NOAA/NMFS/PFEG, Pacific Grove, CA, personal communication). Annual instantaneous mortality was assumed as 0.2/yr after 1 yr. Average age of entrainment was estimated as 6.21 d based on average size at entrainment and a growth rate of 0.14 mm/d (Yoklavich et al. 1996).

Day (start)	Day (end)	Instantaneous Natural Mortality (Z)	Survival ($\hat{S}$)
0	6.21	0.14	0.419
6.21	20	0.14	0.145
20	60	0.08	0.041
60	180	0.04	0.008
180	365	0.0112	0.126
365	1,095	0.0006	0.670

early larv
late larv
early juv
juv I
pre-recruit

Strata 3.11 e^-6 larv-age1 Π = 5.99 e^-6 Π = 4.015 e^-6

Table 5.2.4-8. Estimated number of equivalent KGB rockfish (*Sebastes* spp. V_De/V_D_) adults ($\widehat{AEL}$) equal to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{AEL})$) and 90% confidence limits (C.L.).

Strata larv to age 2

Analysis Period	$\widehat{AEL}$	$\widehat{SE}(\widehat{AEL})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	1,120	3,410	166,000	8
2) Oct 1, 1997–Sep 30, 1998	905	2,750	134,000	6
3) Jul 1, 1997–Jun 30, 1998	906	2,750	134,000	6

Table 5.2.4-9. KGB rockfishes (*Sebastes* spp. V_De/V_D_): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645 \cdot \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 1,120				
$\hat{E}_{Adj-T}$	275,000,000	958	1,290	331
$\hat{S}$ *early larvae*	0.145	217	5,820	5,600
$\hat{S}$ *late larvae*	0.0408	217	5,820	5,600
$\hat{S}$ *early juvenile*	0.00823	217	5,820	5,600
$\hat{S}$ *juv. I*	0.125	217	5,820	5,600
$\hat{S}$ *pre-recruit*	0.670	217	1,680	1,460

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 905				
$\hat{E}_{Adj-T}$	222,000,000	712	1,100	387
$\hat{S}$ *early larvae*	0.145	175	4,690	4,520
$\hat{S}$ *late larvae*	0.0408	175	4,690	4,520
$\hat{S}$ *early juvenile*	0.00823	175	4,690	4,520
$\hat{S}$ *juv. I*	0.125	175	4,690	4,520
$\hat{S}$ *pre-recruit*	0.670	175	1,350	1,180

Table 5.2.4-9 (continued). KGB rockfishes (*Sebastes* spp. V_De/V_D_): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL} = 906$				
$\hat{E}_{Adj-T}$	222,000,000	712	1,100	387
$\hat{S}$ *early larvae*	0.145	175	4,700	4,520
$\hat{S}$ *late larvae*	0.0408	175	4,700	4,520
$\hat{S}$ *early juvenile*	0.00823	175	4,700	4,520
$\hat{S}$ *juv. I*	0.125	175	4,700	4,520
$\hat{S}$ *pre-recruit*	0.670	175	1,350	1,180

Table 5.2.4-10. KGB rockfishes (*Sebastes* spp. V_De/V_D_): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae extruded in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997 - June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.0107	0.0151	0.00257	0.000560
Aug 25, 1997	0	0	0	0
Sept 29, 1997	0	0	0	0
Oct 20, 1997	0	0	0	0
Nov 17, 1997	0	0	0	0
Dec 10, 1997	0	0	0	0
Jan 22, 1998	0.000810	0.000880	0.00122	0.000190
Feb 26, 1998	0.00207	0.00127	0.00403	0.000370
Mar 18, 1998	0.0587	0.0297	0.0816	0.00961
Apr 15, 1998	0.00762	0.00348	0.130	0.0102
May 18, 1998	0.00337	0.000760	0.487	0.0249
Jun 8, 1998	0.0353	0.00842	0.293	0.0166

b) Analysis Period 4: July 1, 1998 - June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.00330	0.00345	0.000780	0.000130
Aug 26, 1998	0	0	0	0
Sept 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0	0	0	0
Jan 12, 1999	0	0	0	0
Feb 3, 1999	0.000460	0.000460	0.000630	0.000120
Mar 17, 1999	0.0327	0.0198	0.175	0.0156
Apr 14, 1999	0.0137	0.00752	0.204	0.0154
May 24, 1999	0.0115	0.00262	0.545	0.0166
Jun 23, 1999	0.0170	0.0125	0.0754	0.00533

Table 5.2.4-11. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for larvae of the blue rockfish complex (*Sebastes* spp.V/*S. mystinus*) from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	49,700,000	7,940,000
2) Oct 1, 1997–Sep 30, 1998	5,410,000	855,000
3) Jul 1, 1997–Jun 30, 1998	6,220,000	868,000

Table 5.2.4-12. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) of blue rockfish complex larvae (*Sebastes* spp.V/*S. mystinus*) and their standard errors ($\widehat{SE}(\hat{E}_{Adj-T})$) from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	84,040,000	13,400,000
2) Oct 1, 1997–Sep 30, 1998	33,800,000	5,350,000
3) Jul 1, 1997–Jun 30, 1998	38,900,000	5,430,000

Table 5.2.4-13. Estimated number of adult blue rockfish complex (*Sebastes* spp.V/*S. mystinus*) females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{FH})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	43	97	1,700	1
2) Oct 1, 1997–Sep 30, 1998	18	39	684	0
3) Jul 1, 1997–Jun 30, 1998	20	45	787	1

Table 5.2.4-14. Blue rockfish complex (*Sebastes* spp.V/*S. mystinus*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH} = 43$				
$\hat{E}_{Adj-T}$	84,000,000	54.9	32.0	23
$\hat{S}$ *larvae*	0.407	17.7	225	208
# Eggs/yr	509,000	8.39	225	217
Longevity	24.7	18.7	86.9	68.2
Maturation	6	64.0	37.5	26.5

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH} = 18$				
$\hat{E}_{Adj-T}$	33,800,000	22.1	13.0	9.10
$\hat{S}$ *larvae*	0.407	7.12	90.7	83.6
# Eggs/yr	509,000	3.38	90.7	87.3
Longevity	24.7	7.54	35.0	27.5
Maturation	6	25.8	15.1	10.7

Table 5.2.4-14 (continued). Blue rockfish complex (*Sebastes* spp.V/*S. mystinus*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH} = 20$				
$\hat{E}_{Adj-T}$	38,900,000	24.7	15.5	9.23
$\hat{S}$ *larvae*	0.407	8.18	104	96.1
# Eggs/yr	509,000	3.88	104	100
Longevity	24.7	8.67	40.3	31.6
Maturation	6	29.6	17.3	12.3

Table 5.2.4-15. Three year survival for the blue rockfish complex (*Sebastes* spp. V/*S. mystinus*) larvae. Survival was estimated from release as $\hat{S} = e^{(-Z)(Day(end) - Day(start))}$. Daily instantaneous mortality rates (*Z*) up to 1 yr of blue rockfish, *Sebastes mystinus*, larvae that were used to calculate larval survivorship were provided by Mary Yoklavich (NOAA/NMFS/PFEG, Pacific Grove, CA, personal communication). Annual instantaneous mortality was assumed as 0.2/yr after 1 yr. Average age of entrainment was estimated as 6.43 d, based on average size at entrainment and a growth rate of 0.14 mm/d.

Day (start)	Day (end)	Instantaneous Natural Mortality (Z)	Survival ($\hat{S}$)
0	6.43	0.14	0.4066
6.43	20	0.14	0.150
20	60	0.08	0.041
60	180	0.04	0.008
180	365	0.011	0.126
365	1,095	0.0006	0.6703

Π 6.15 e^{-3}

Table 5.2.4-16. Estimated number of equivalent blue rockfish complex (*Sebastes* spp.V/*S. mystinus*) adults ($\widehat{AEL}$) equal to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{AEL})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{AEL}$	$\widehat{SE}(\widehat{AEL})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	353	1,100	51,000	2
2) Oct 1, 1997–Sep 30, 1998	164	494	23,500	1
3) Jul 1, 1997–Jun 30, 1998	142	430	20,000	1

Table 5.2.4-17. Blue rockfish complex (*Sebastes* spp. V/*S. mystinus*): Sensitivity analysis of $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645 \cdot SE(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 \cdot CV(\hat{\theta})}$ where CV(θ) = 1.0 or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 353				
$\hat{E}_{Adj-T}$	84,000,000	260	446	186
$\hat{S}$ *early larvae*	0.150	68	1,830	1,760
$\hat{S}$ *late larvae*	0.0408	68	1,830	1,760
$\hat{S}$ *early juvenile*	0.00823	68	1,830	1,760
$\hat{S}$ *juv. I*	0.125	68	1,830	1,760
$\hat{S}$ *pre-recruit*	0.670	68	527	459

Table 5.2.4-17 (continued). Blue rockfish complex (*Sebastes* spp. V/*S. mystinus*): Sensitivity analysis of $\widehat{AEL}$ for the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645 \cdot SE(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 \cdot CV(\hat{\theta})}$ where CV(θ) = 1.0 or 100%.

c) Analysis Period 3: July 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 164				
$\hat{E}_{Adj-T}$	38,900,000	126	201	75
$\hat{S}$ *early larvae*	0.150	32	848	816
$\hat{S}$ *late larvae*	0.0408	32	848	816
$\hat{S}$ *early juvenile*	0.00823	32	848	816
$\hat{S}$ *juv. I*	0.125	32	848	816
$\hat{S}$ *pre-recruit*	0.670	32	244	213

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL}$ = 142				
$\hat{E}_{Adj-T}$	33,800,000	105	179	74
$\hat{S}$ *early larvae*	0.150	27	737	710
$\hat{S}$ *late larvae*	0.0408	27	737	710
$\hat{S}$ *early juvenile*	0.00823	27	737	710
$\hat{S}$ *juv. I*	0.125	27	737	710
$\hat{S}$ *pre-recruit*	0.670	27	212	185

Table 5.2.4-18. Blue rockfish complex (*Sebastes* spp./*S. mystinus*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae extruded in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997 - June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.00105	0.000510	0.127	0.0113
Aug 25, 1997	0.00125	0.000800	0.0439	0.00606
Sep 29, 1997	0	0	0	0
Oct 20, 1997	0	0	0.0129	0.00347
Nov 17, 1997	0	0	0	0
Dec.10, 1997	0	0	0	0
Jan 22, 1998	0.000770	0.000300	0.161	0.0133
Feb 26, 1998	0	0	0.151	0.0130
Mar 18, 1998	0.000900	0.000920	0.0546	0.00892
Apr 15, 1998	0.00416	0.00294	0.238	0.0281
May 18, 1998	0.000400	0.000210	0.129	0.0141
Jun 8, 1998	0.000810	0.000590	0.0826	0.0150

a) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.000420	0.000430	0.00389	0.000550
Aug 26, 1998	0	0	0	0
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0	0	0	0
Jan 12, 1999	0.00521	0.00291	0.181	0.0169
Feb 3, 1999	0.00822	0.00213	0.495	0.0209
Mar 17, 1999	0	0	0	0
Apr 14, 1999	0.0370	0.0218	0.308	0.0225
May 24, 1999	0.000590	0.000310	0.00791	0.000760
Jun 23, 1999	0.000250	0.000250	0.00352	0.000750

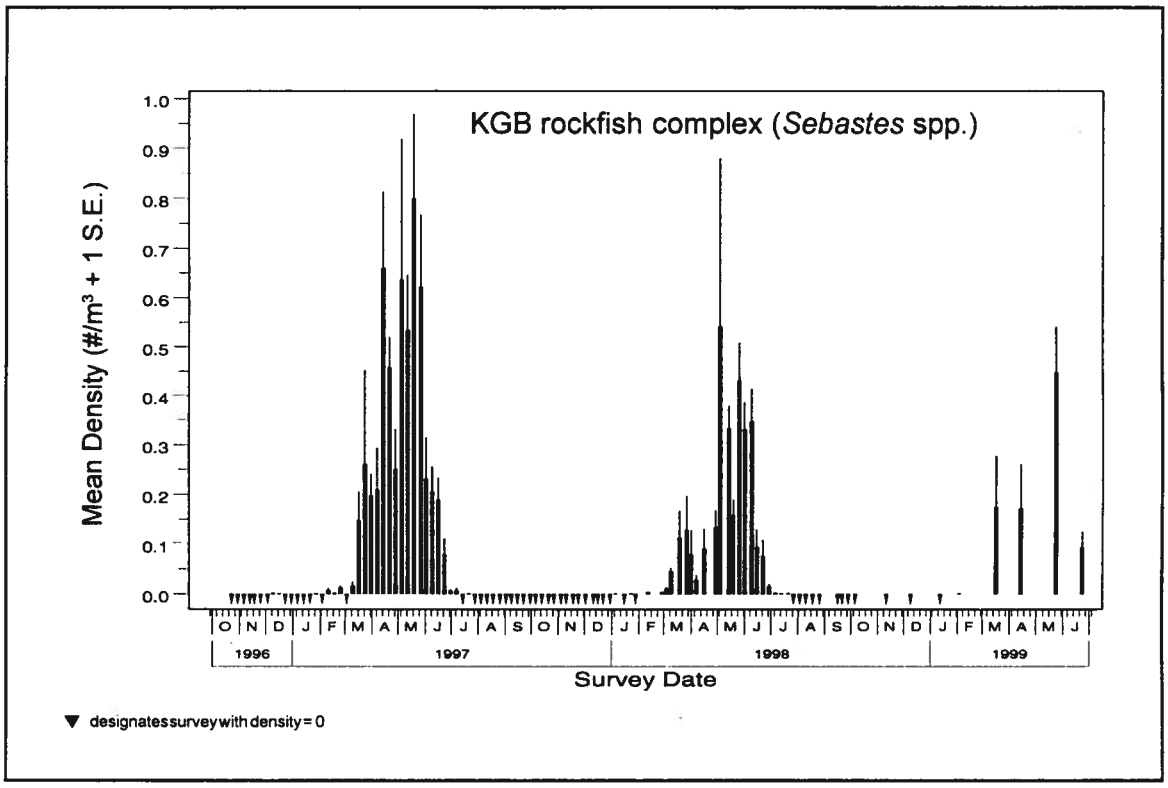


Figure 5.2.4-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

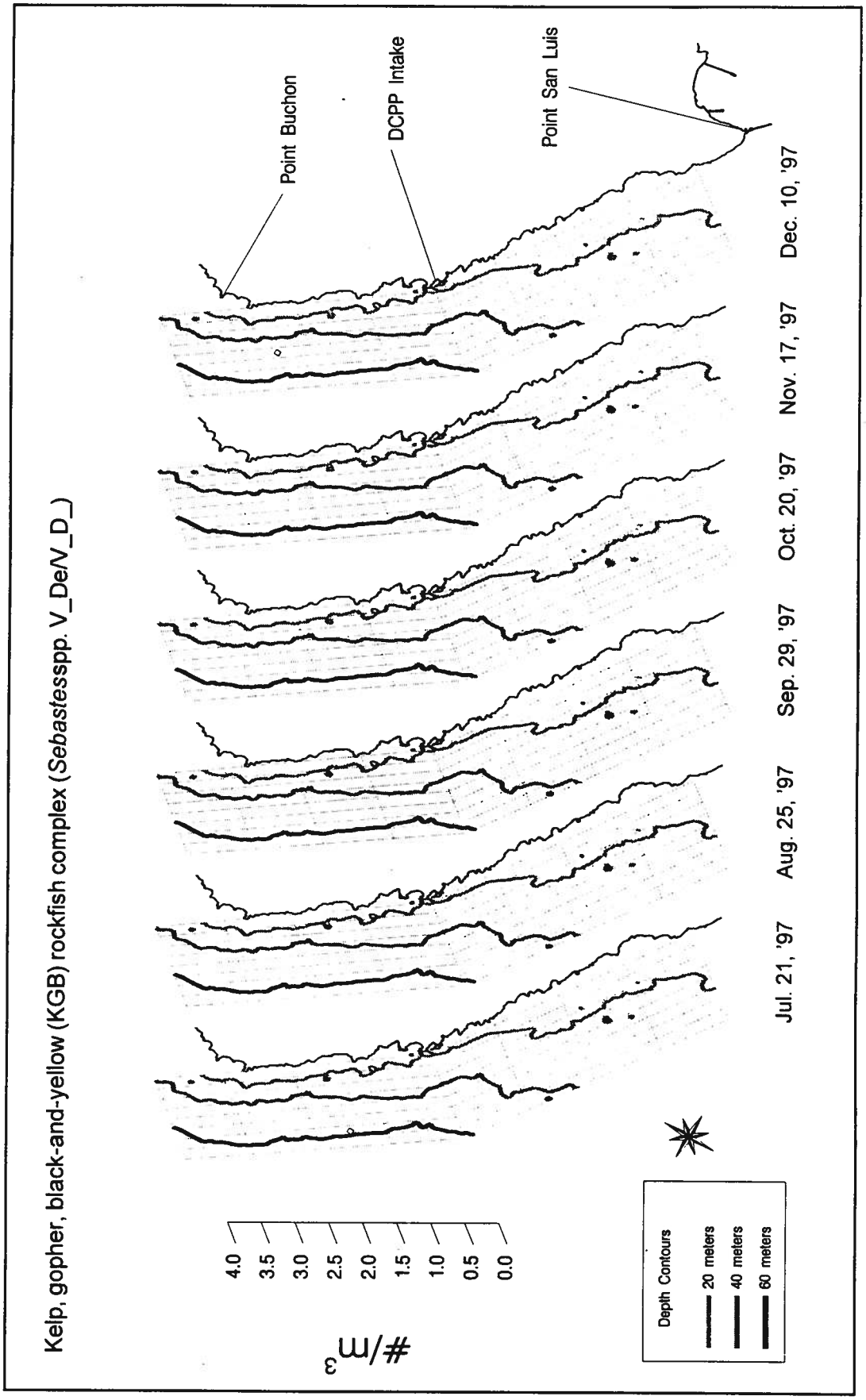


Figure 5.2.4-2a. Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

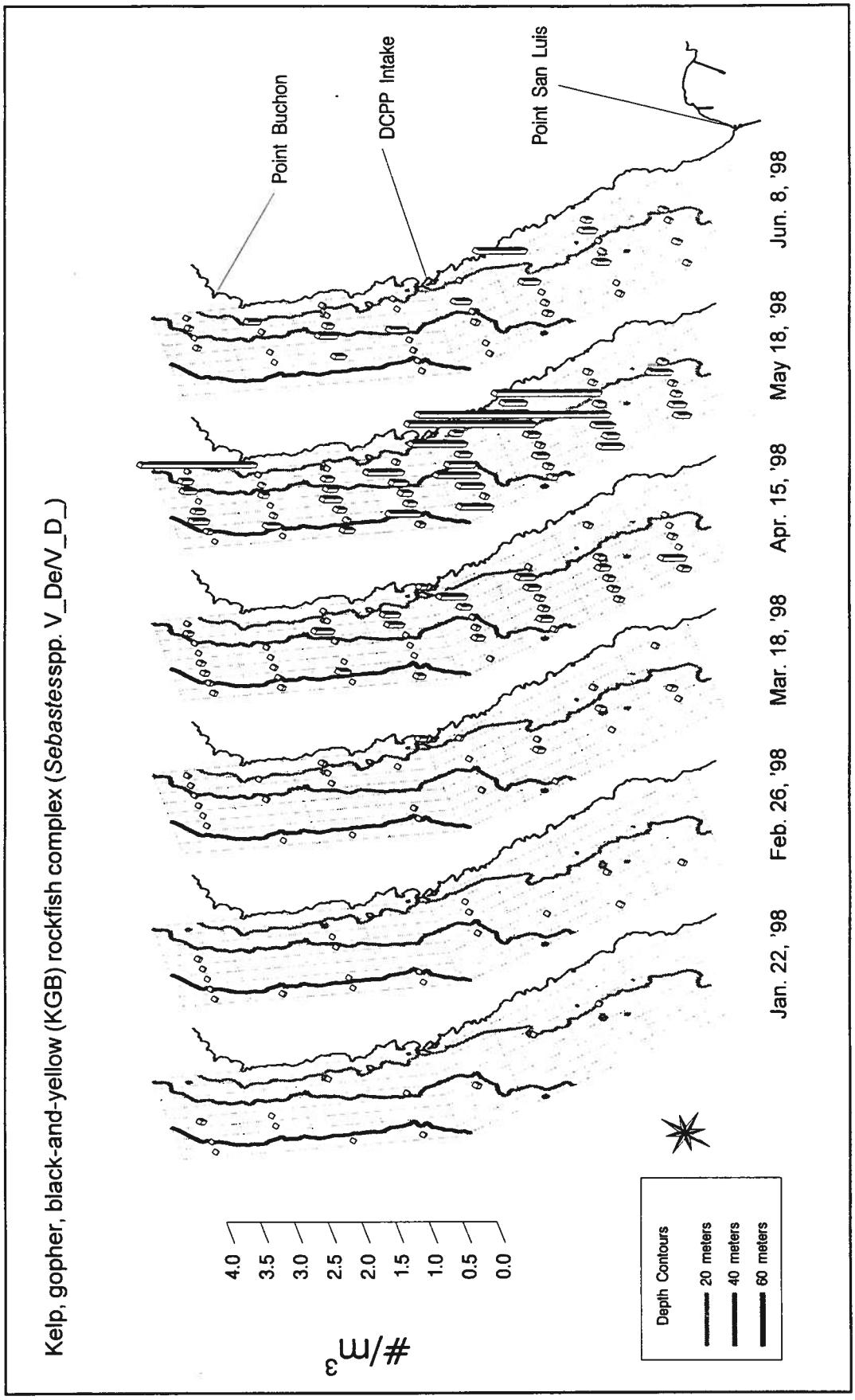


Figure 5.2.4-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

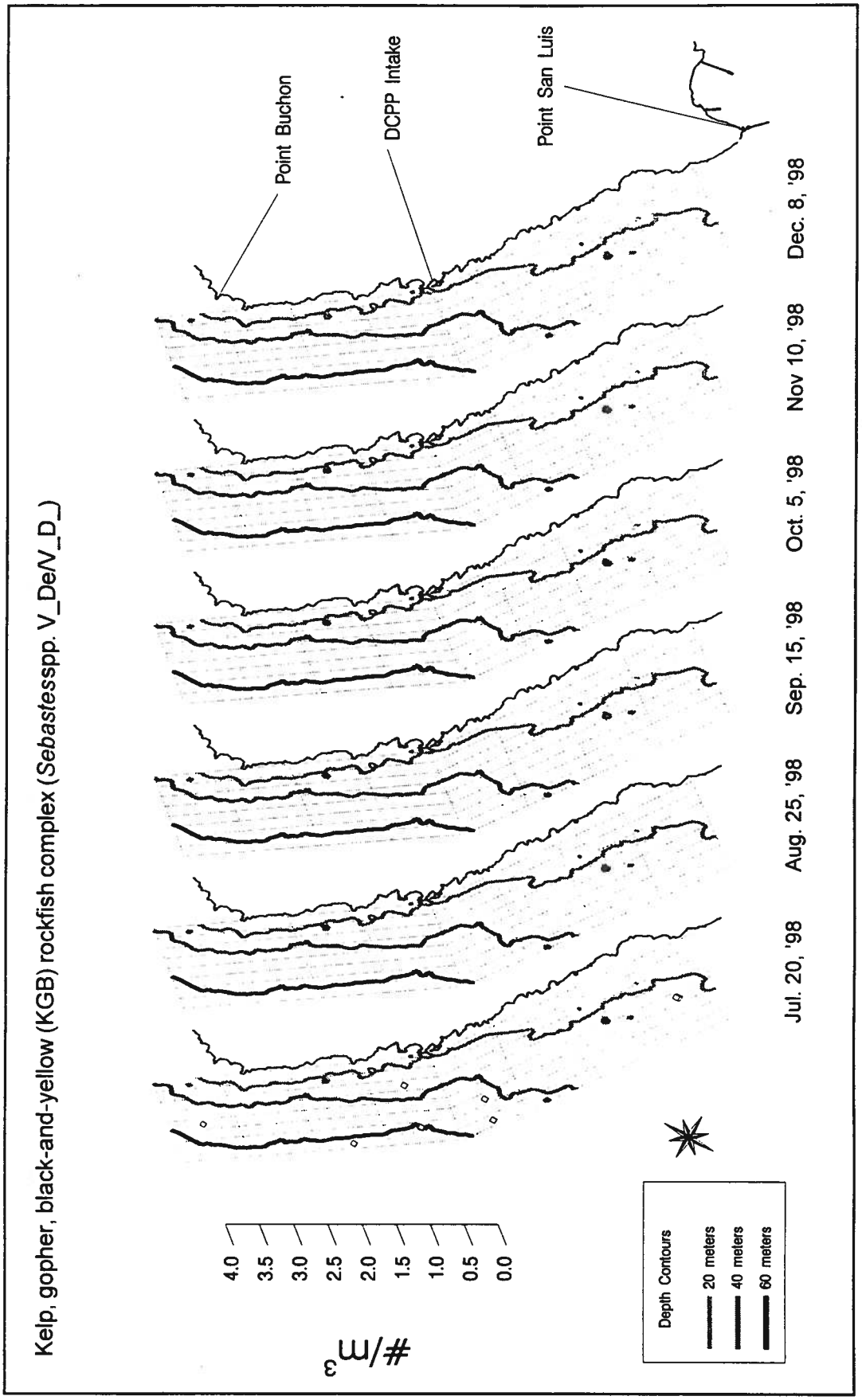


Figure 5.2.4-2c (continued). Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

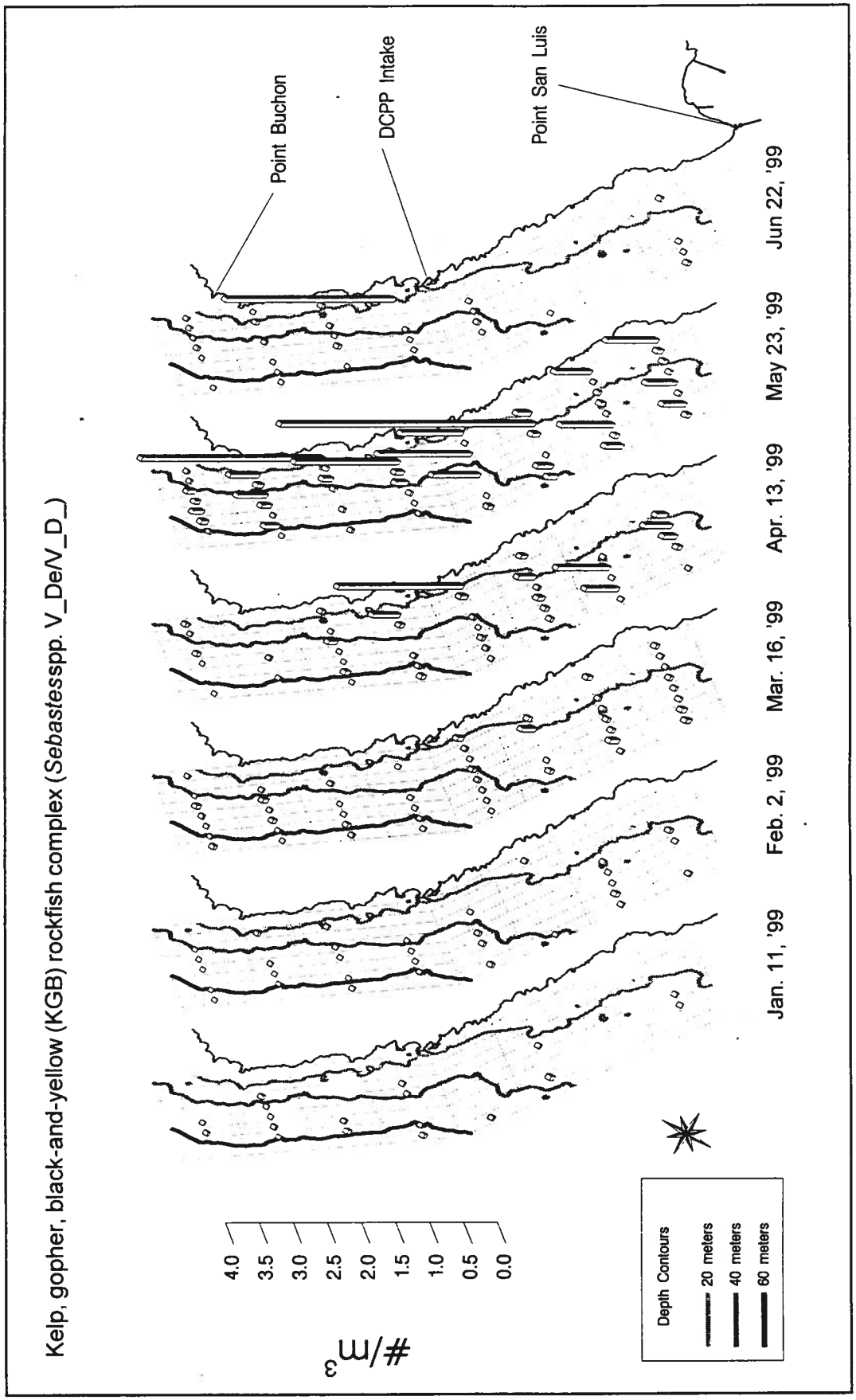


Figure 5.2.4-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

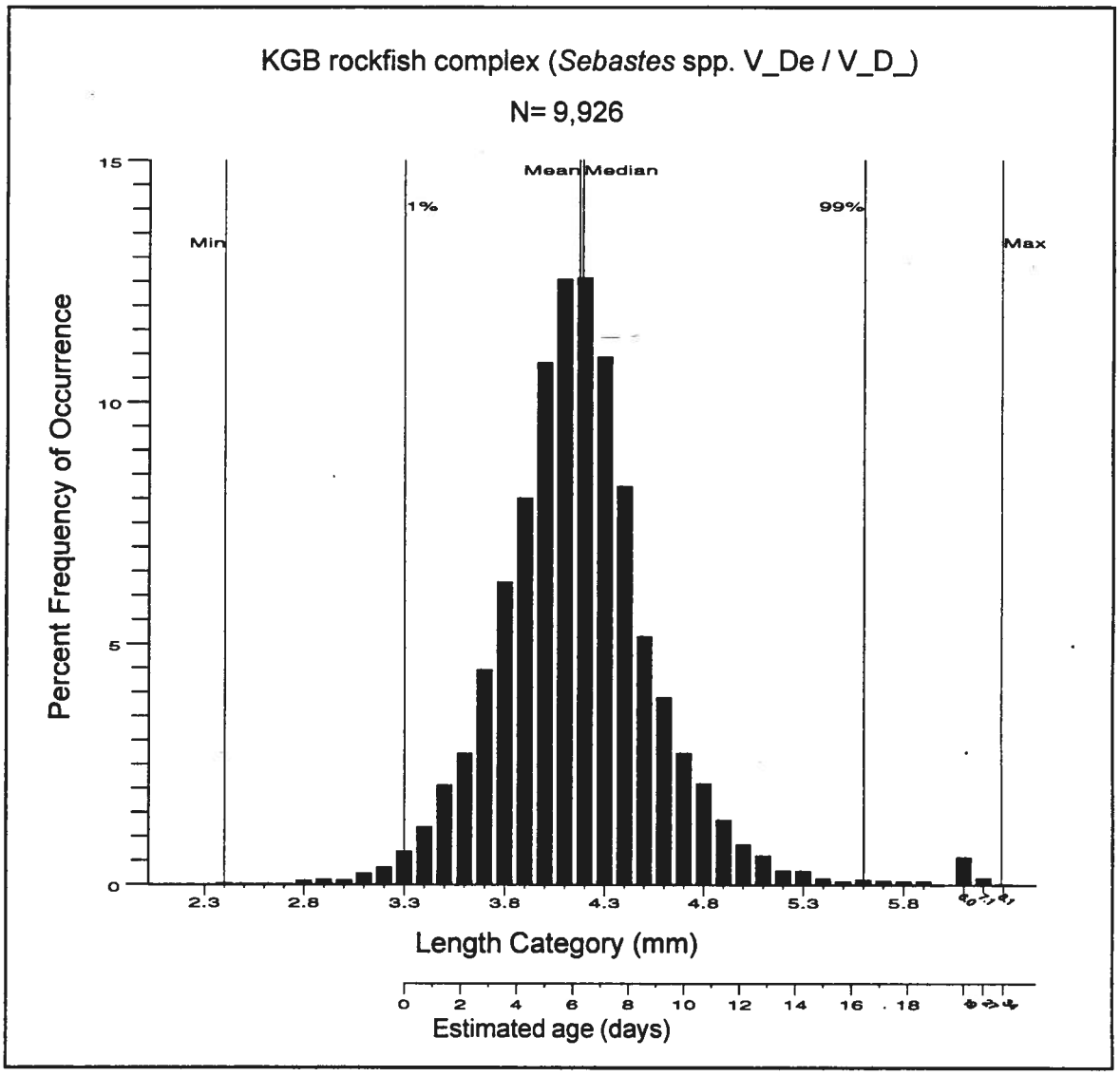


Figure 5.2.4-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

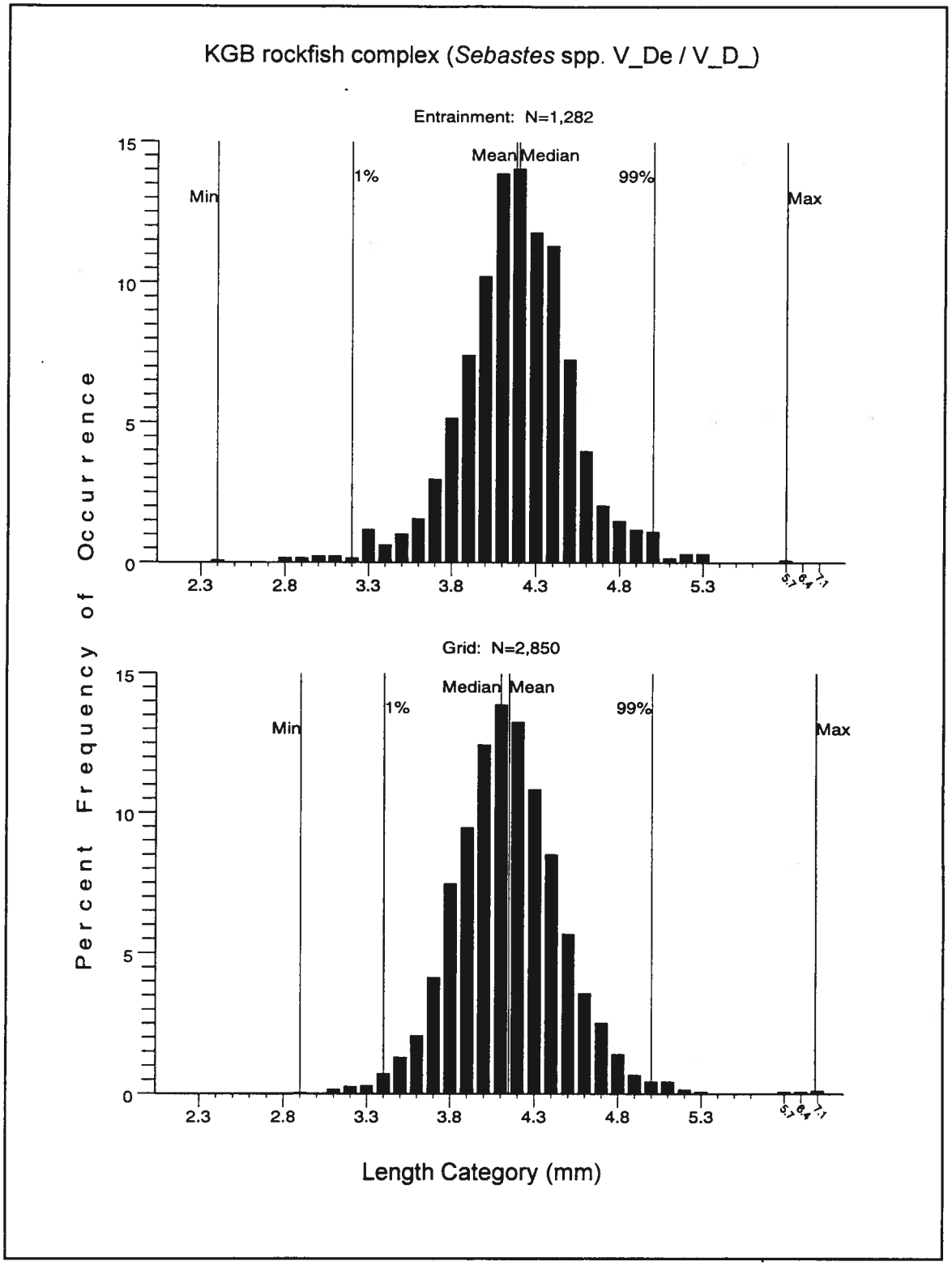


Figure 5.2.4-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

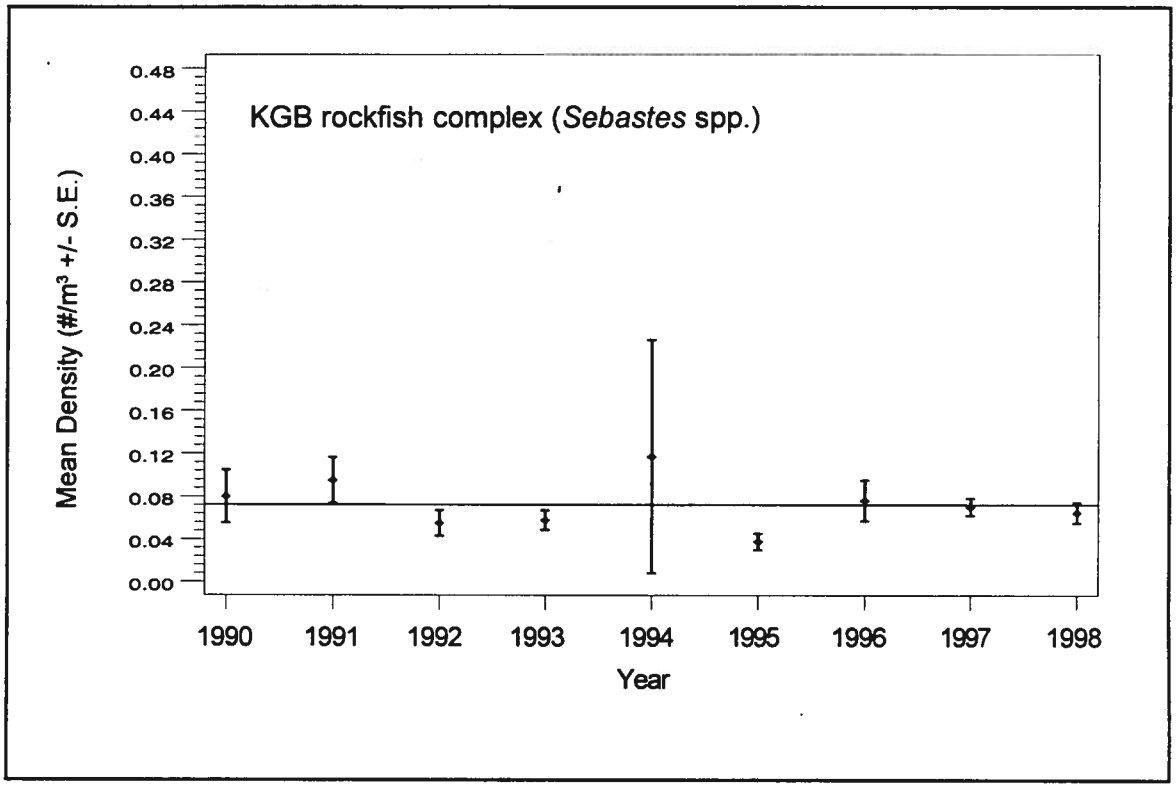


Figure 5.2.4-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

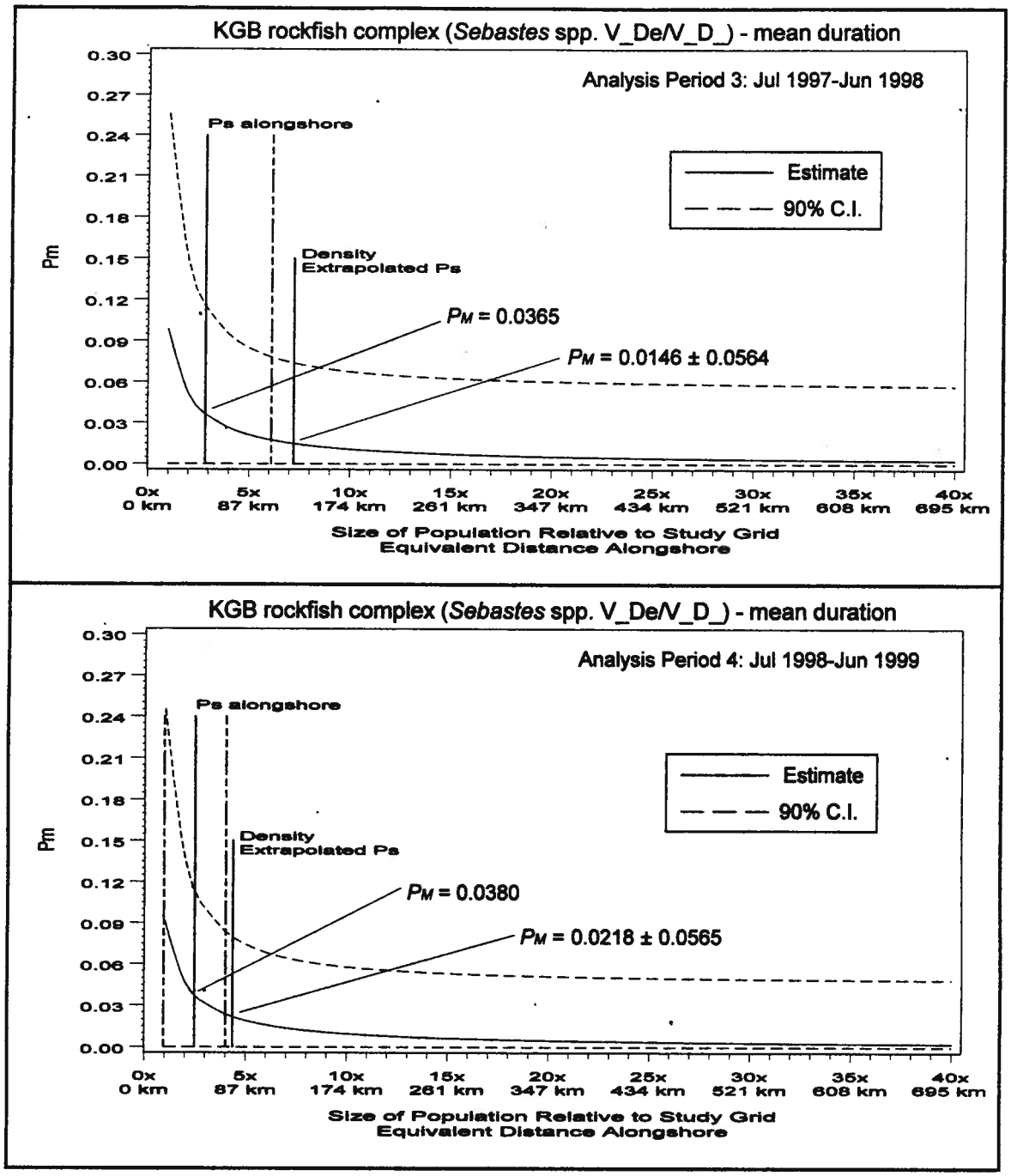


Figure 5.2.4-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

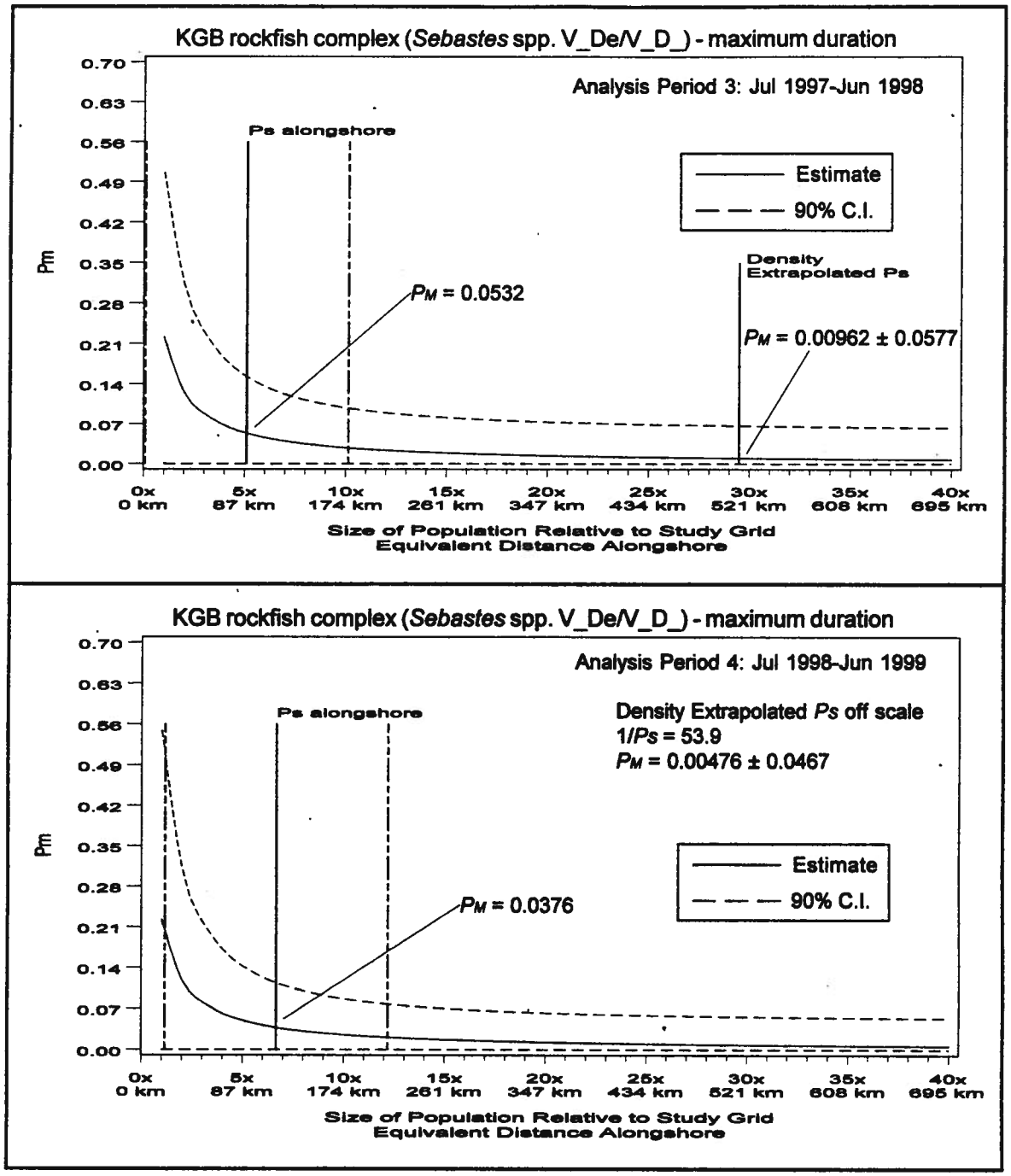


Figure 5.2.4-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

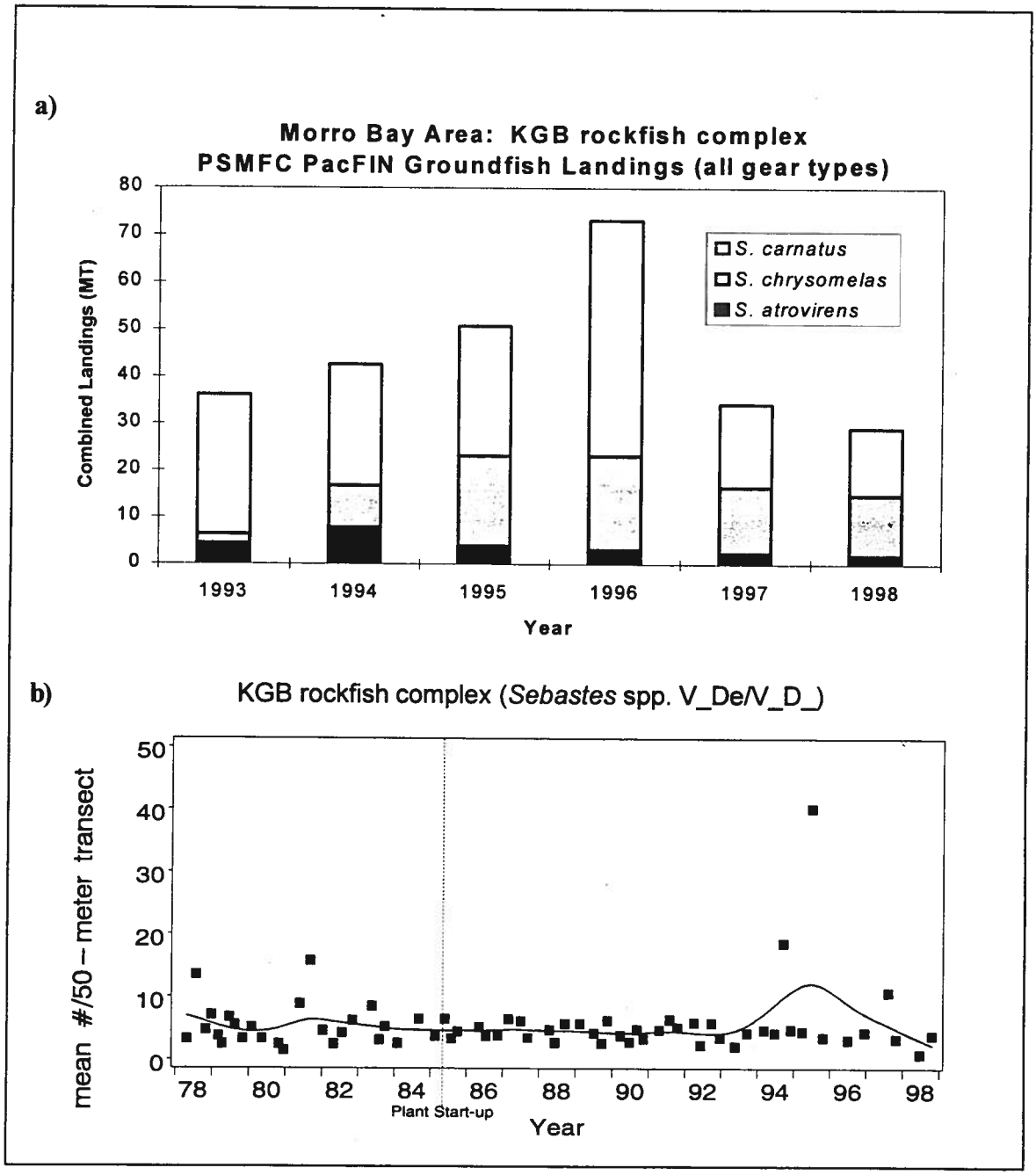


Figure 5.2.4-7. Yearly abundance of KGB rockfish complex measured by two independent methods: a) Commercial fishery landings of adults from Morro Bay area and, b) Mean number of adults and YOY observed per 50-meter subtidal benthic transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to fit curve through points.

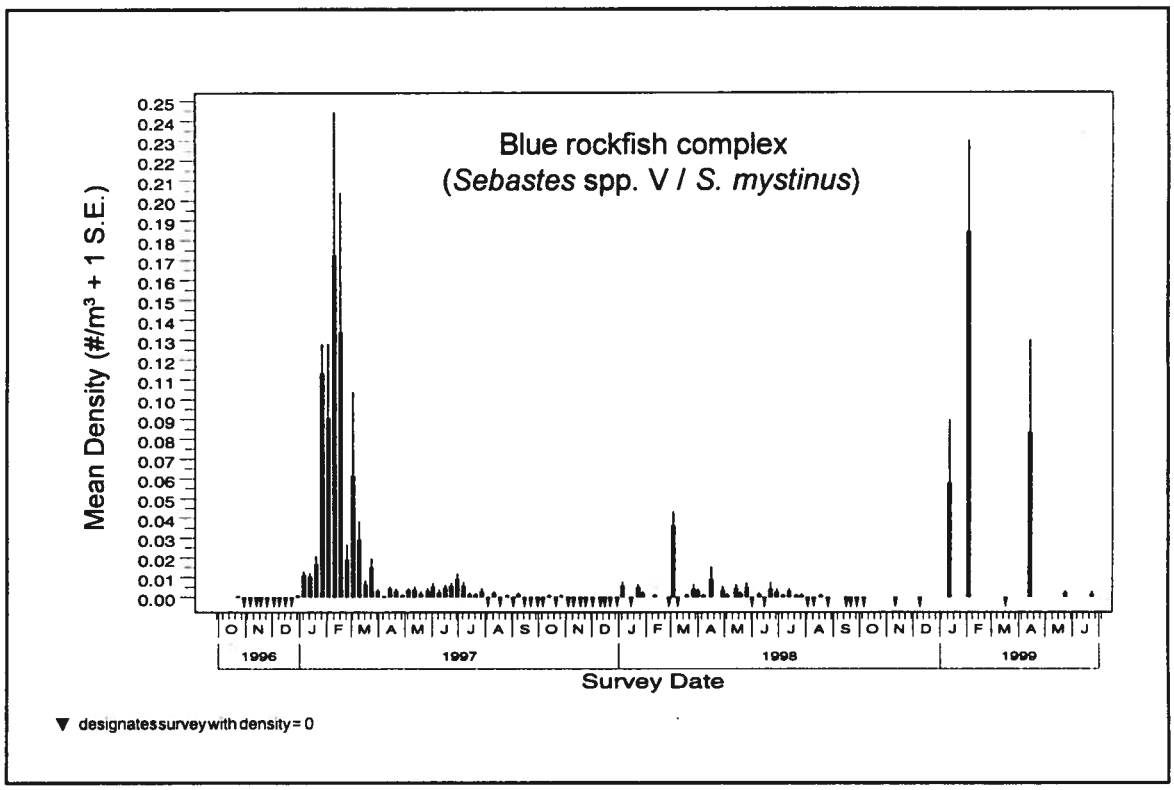


Figure 5.2.4-8. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

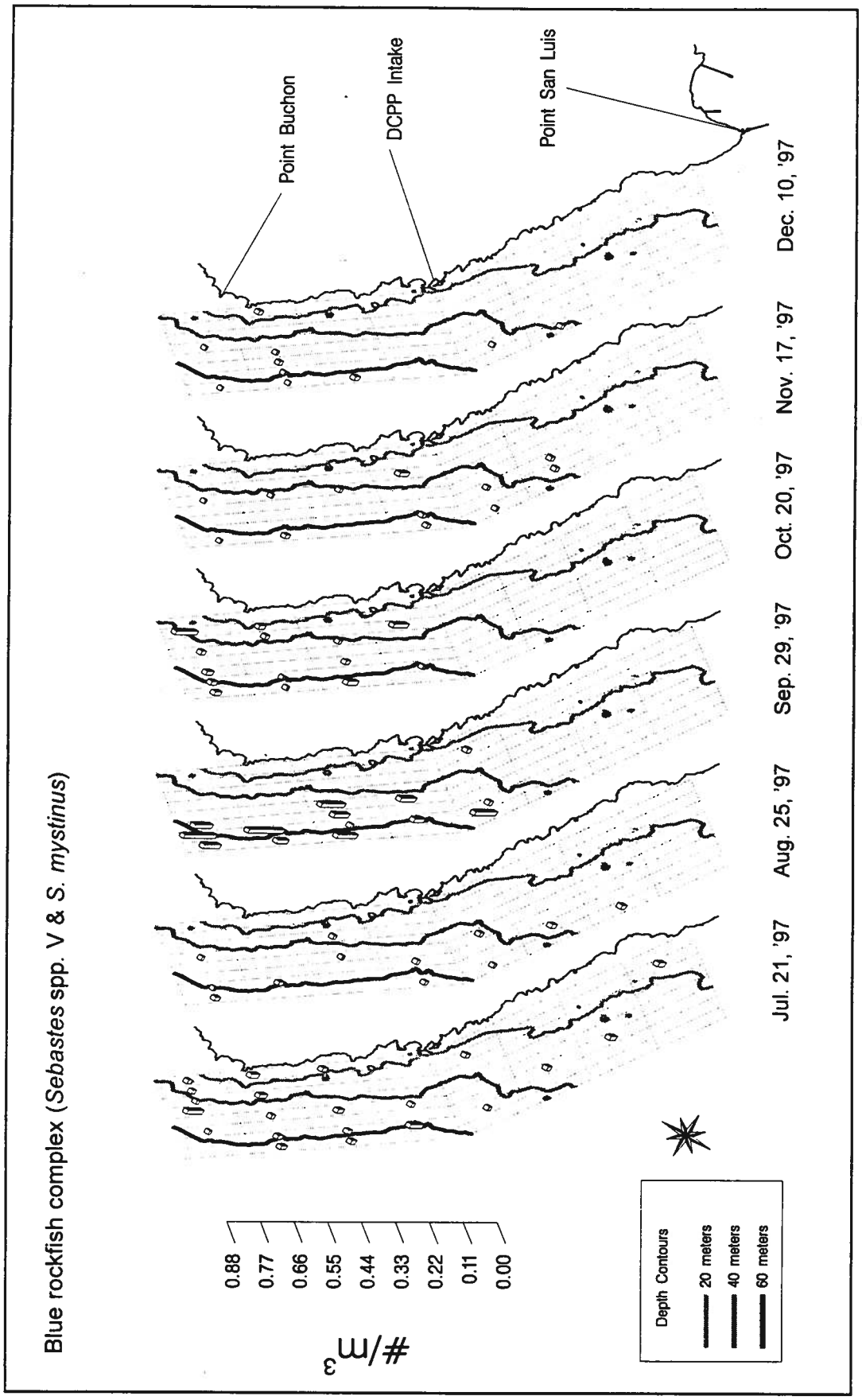


Figure 5.2.4-9a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

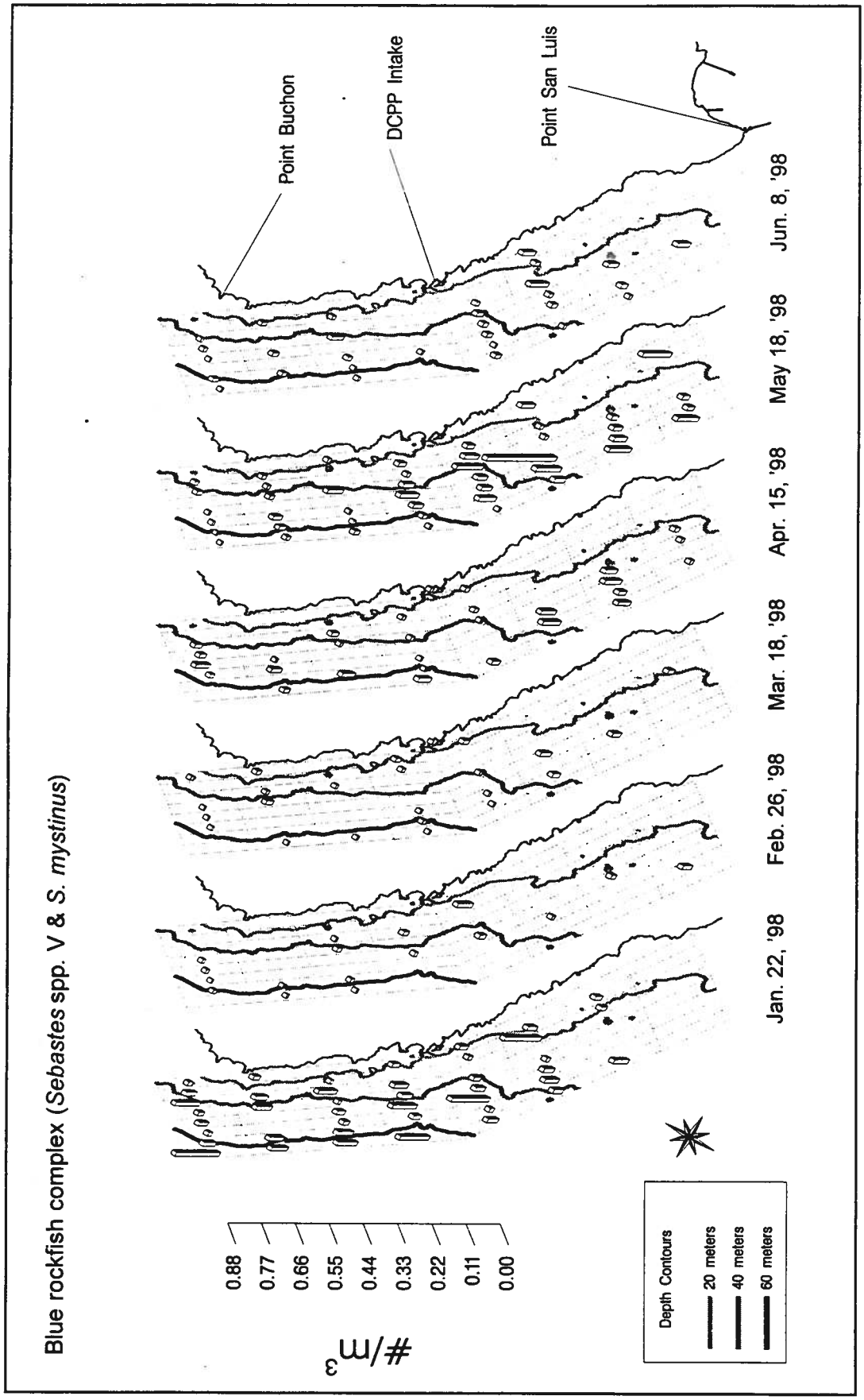


Figure 5.2.4-9b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

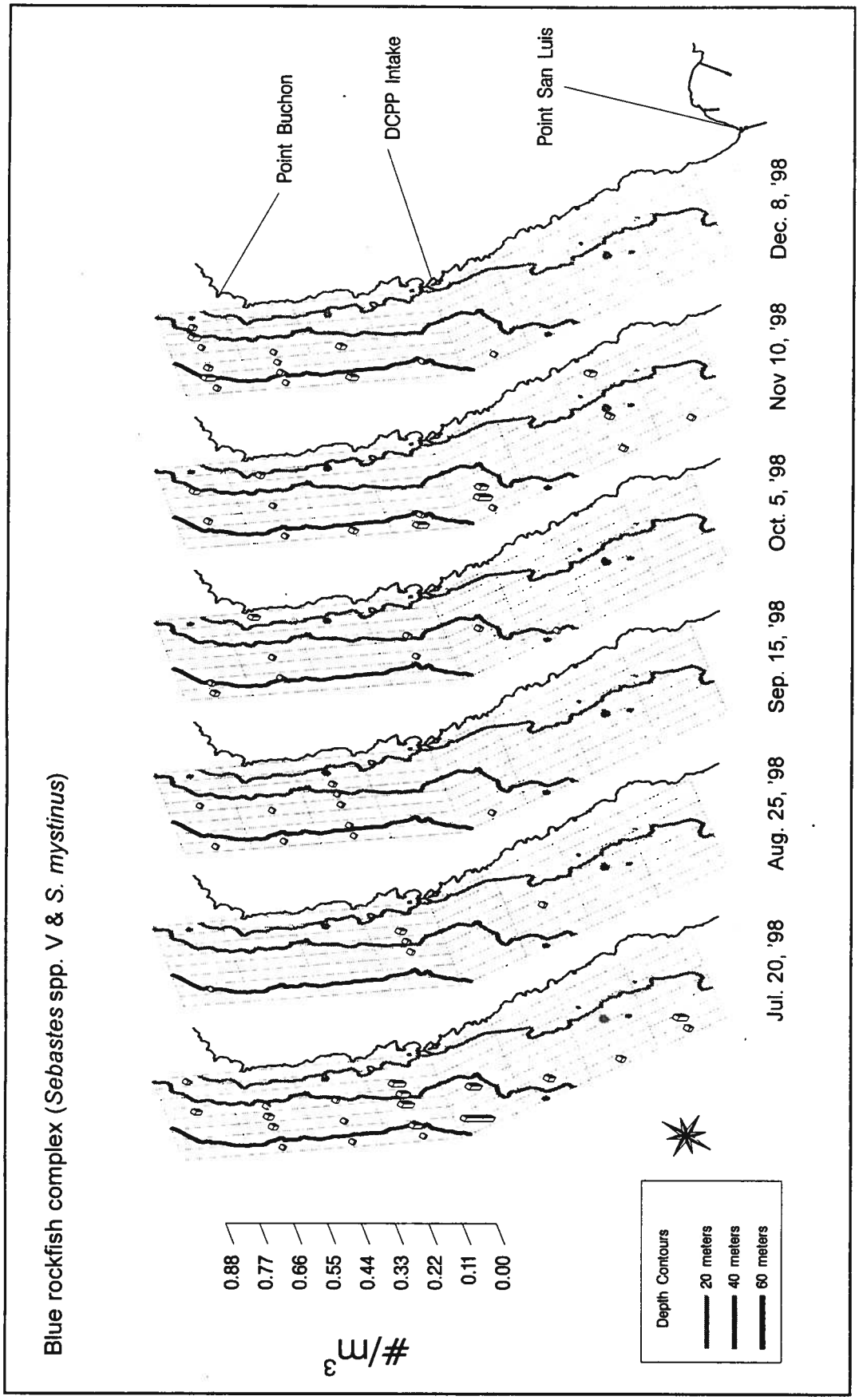


Figure 5.2.4-9c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

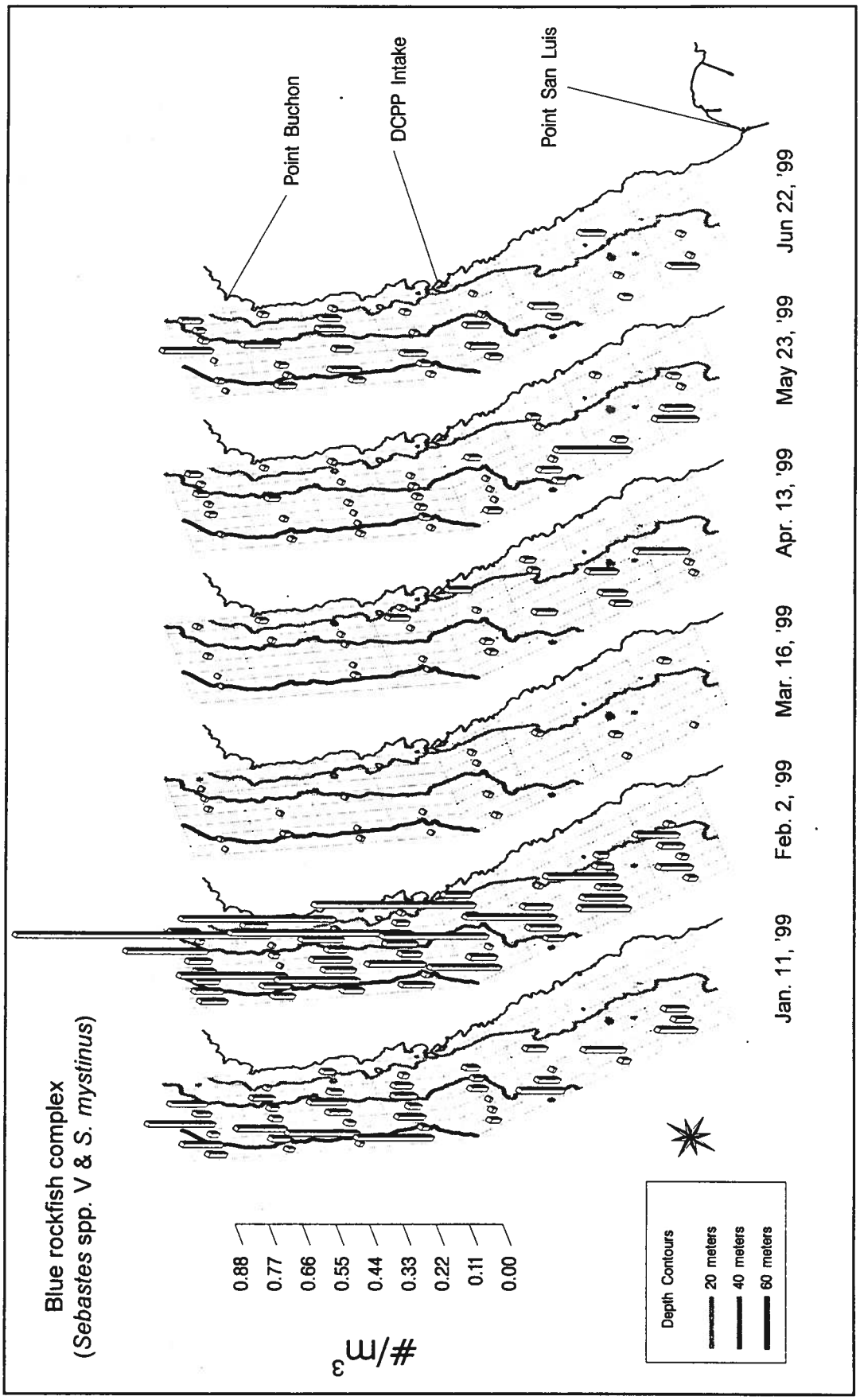


Figure 5.2.4-9d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

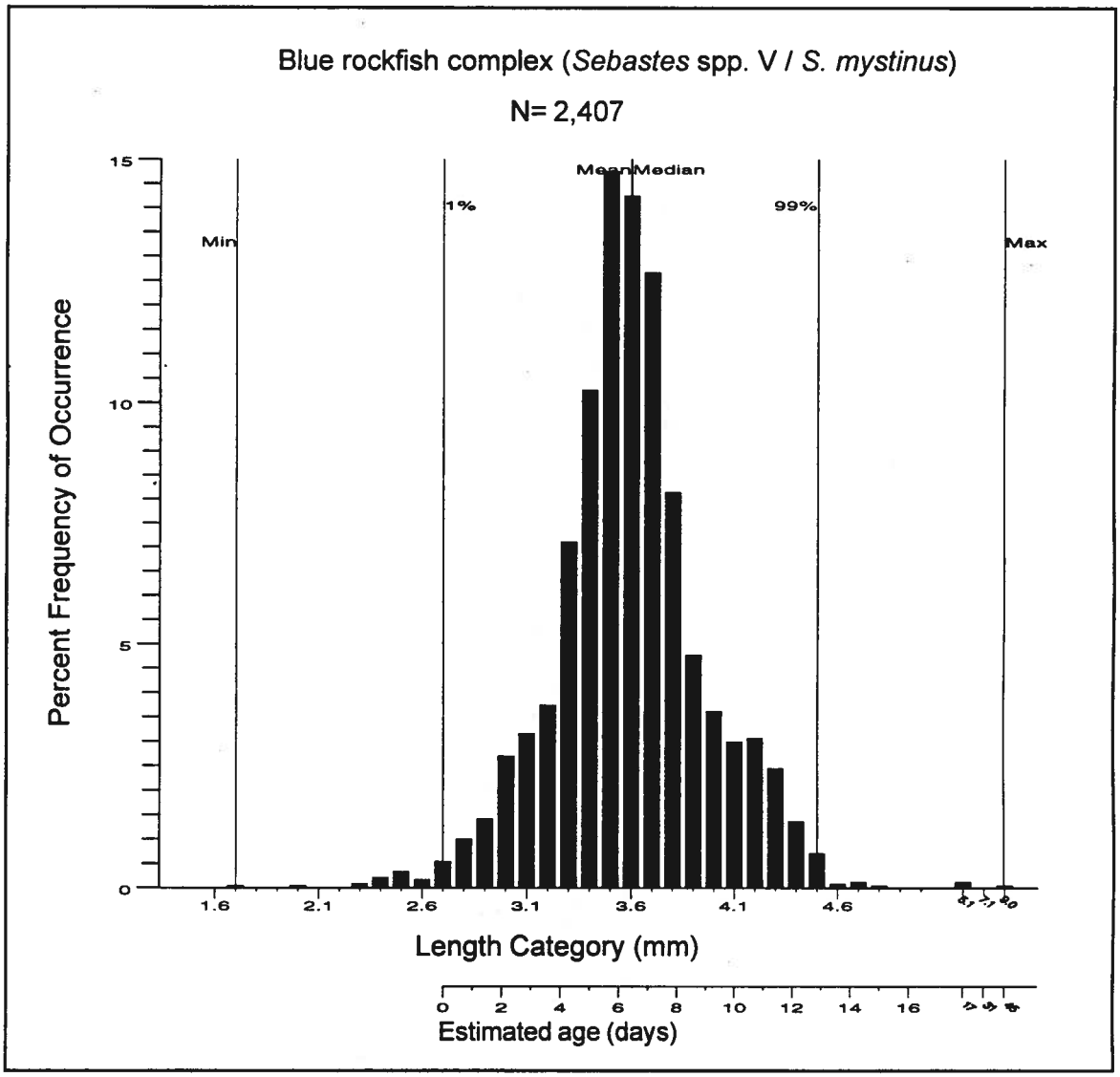


Figure 5.2.4-10. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

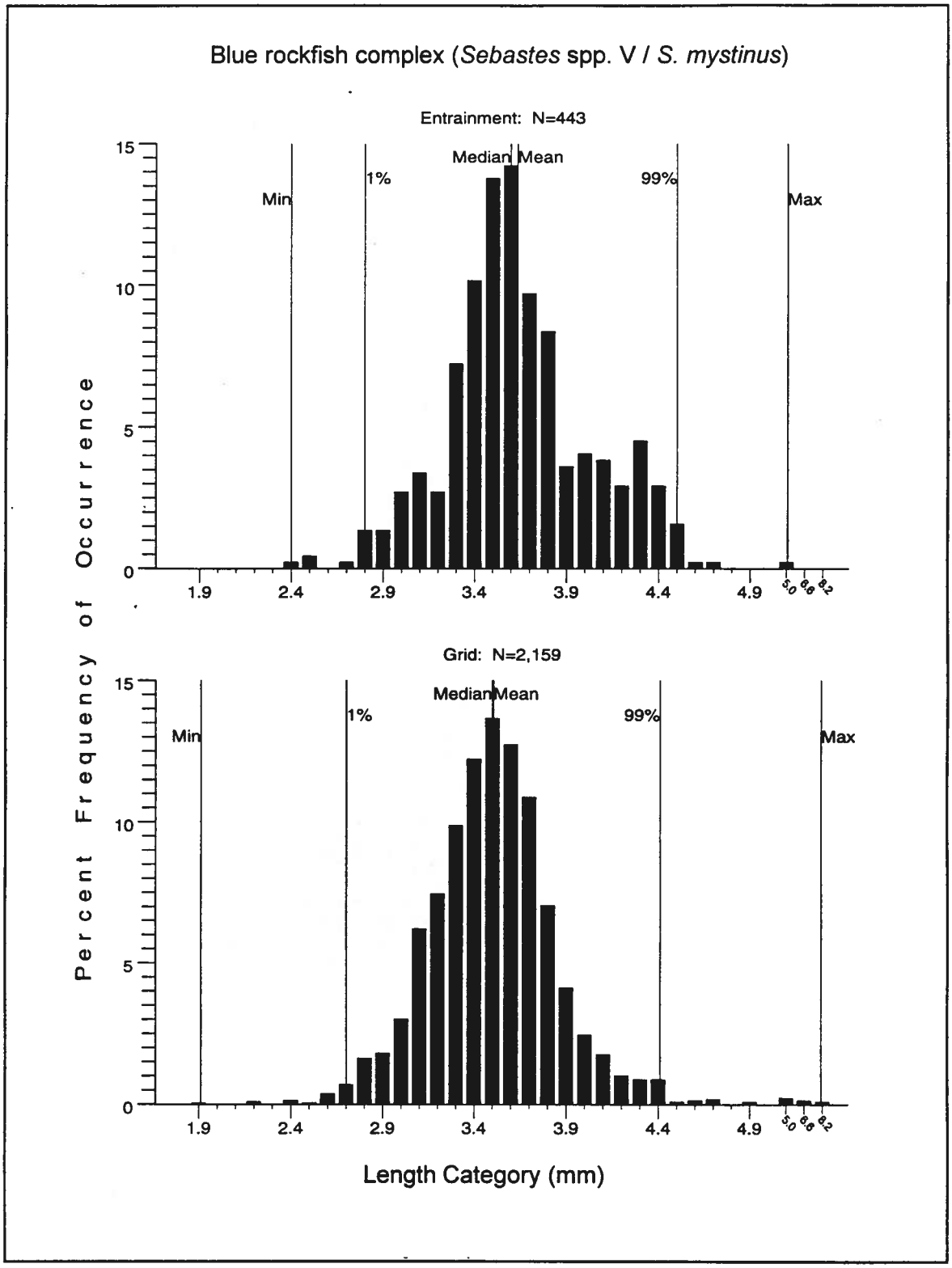


Figure 5.2.4-11. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

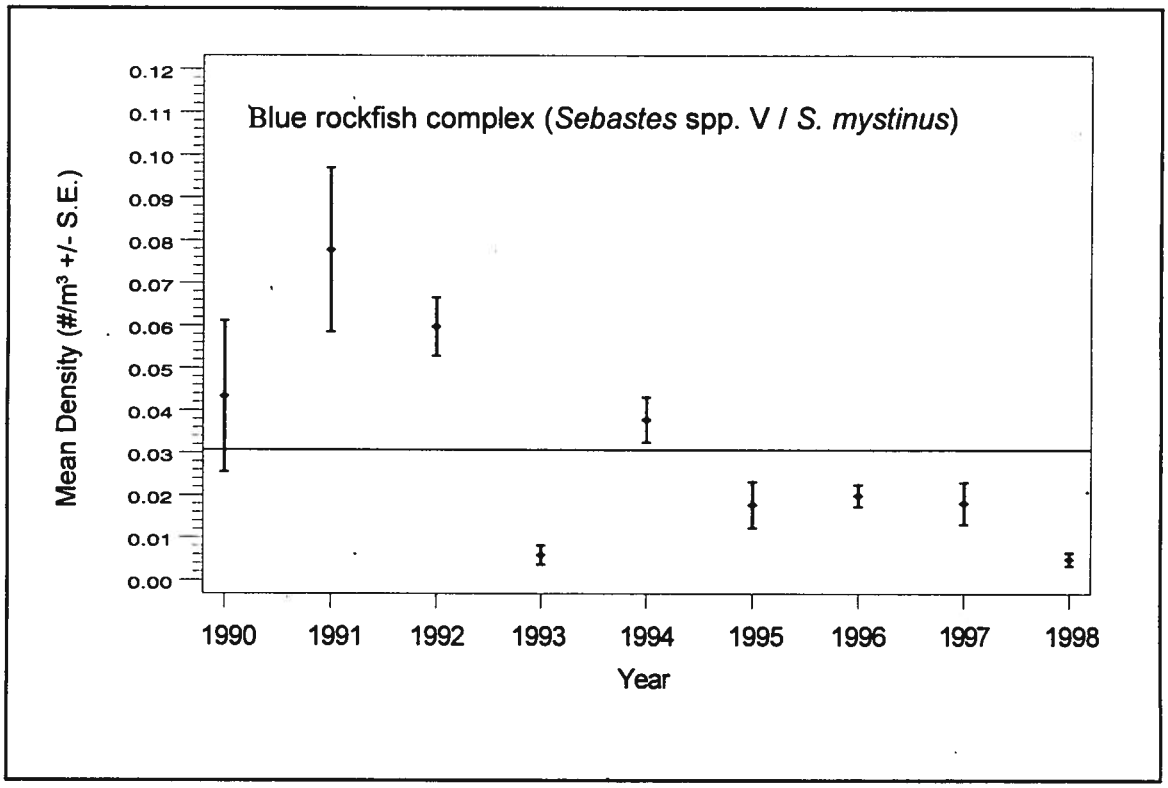


Figure 5.2.4-12. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

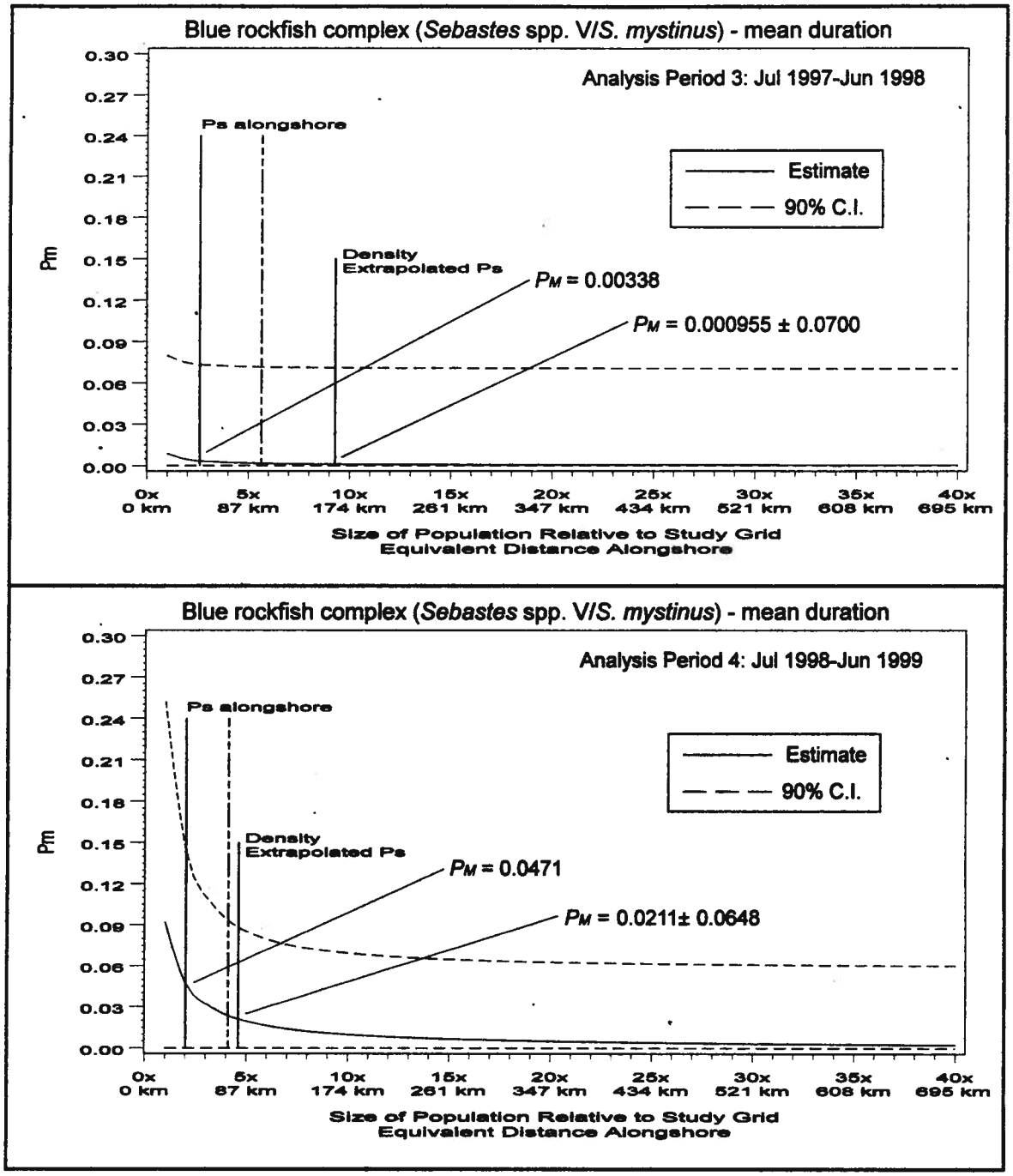


Figure 5.2.4-13a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

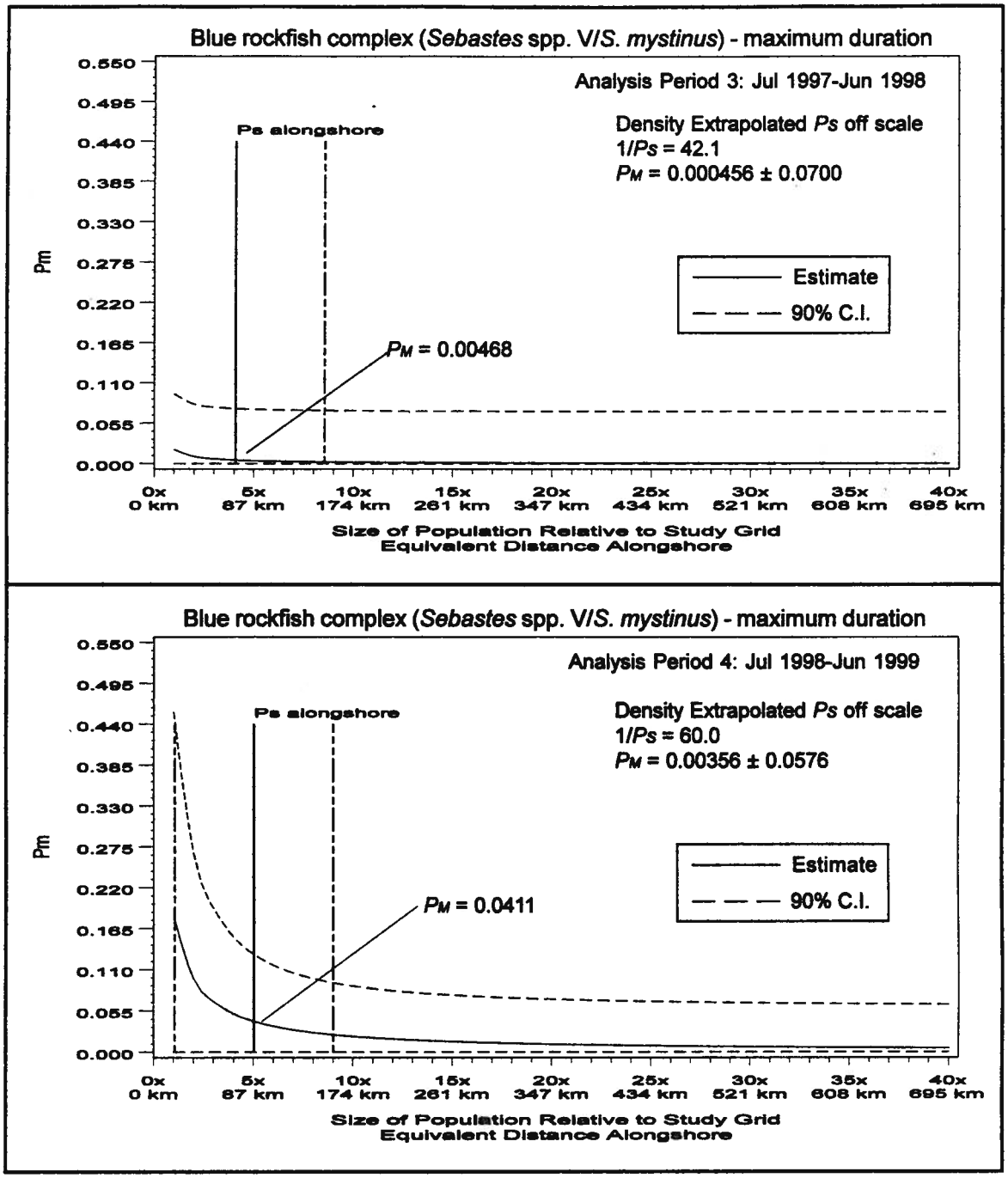


Figure 5.2.4-13b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

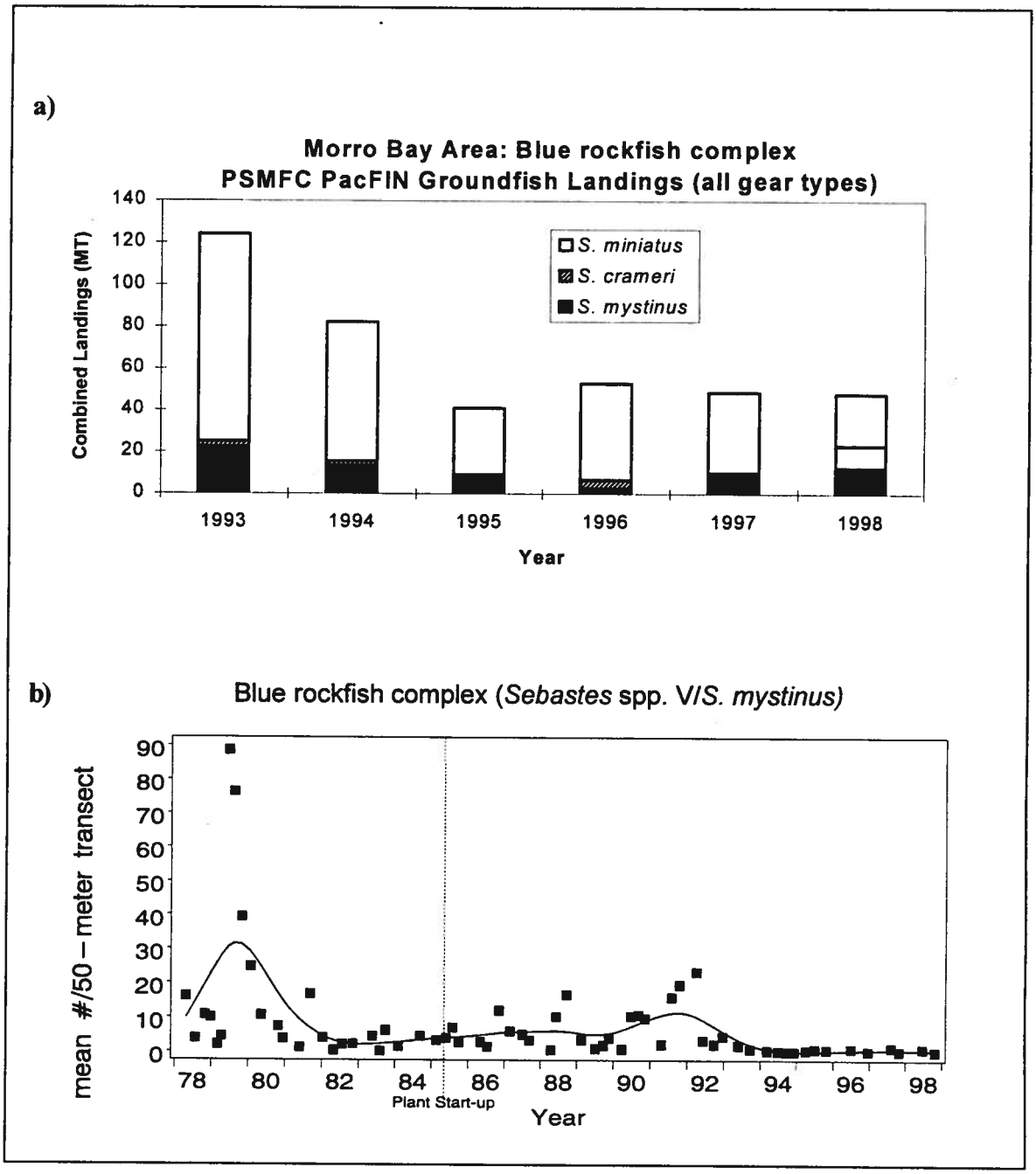
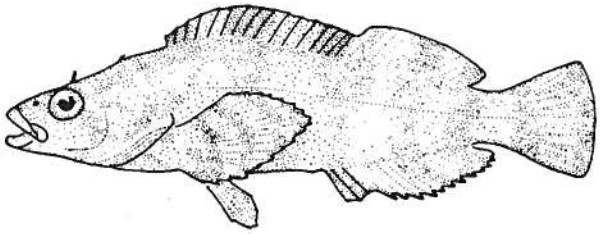


Figure 5.2.4-14. Yearly abundance of blue–rockfish complex measured by two independent methods: a) Commercial fishery +landings of adults from Morro Bay area and, b) Mean number of adults and YOY observed per 50-meter subtidal benthic transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to fit curve through points.

5.2.5 *Assessment of Painted Greenling* (Oxylebius pictus)



Oxylebius pictus Gill 1862; painted greenling; length to 26 cm; Prince William Sound, Alaska to Islas San Benito, central Baja California; intertidal to 94 m; grayish to brown, sometimes white spotted; throat usually spotted and three dark bands radiate from eye; dark spots on caudal, pectoral and pelvic fins (Miller and Lea 1972; Eschmeyer et al. 1983; Love 1996).

The painted greenling (*Oxylebius pictus*) is a member of the family Hexagrammidae, which is found exclusively in the northern Pacific Ocean (Nelson 1994). Three of the four subfamilies occur along the California coast: Hexagramminae (one genus, *Hexagrammos*), Ophiodontinae (one species, *Ophiodon elongatus*), and Oxylebiinae (one species, *O. pictus*; Nelson 1994). Painted greenling are solitary bottom dwellers that inhabit the intertidal zone down to depths of 94 m among or near cobble and boulder substrata, tending to mingle with the benthic flora in areas of surge (Miller and Lea 1972; Burge and Schultz 1973; Eschmeyer et al. 1983; Love 1996). Crustaceans and polychaetes are important food items (Burge and Schultz 1973; Love 1996). In turn, painted greenling are an important food item of Brandt's cormorant (Love 1996). Painted greenling was one of the most common fish in surveys of Diablo Cove and the vicinity in 1970 and 1971 (Burge and Schultz 1973). In southern California they are most abundant at depths ranging from 15 to 31 m, but in central California the peak distribution moves inshore to a range of 5 to 22 m (Love 1996). Painted greenling are territorial and seem to have high site affinity (Love 1996).

Painted greenling are oviparous and lay adhesive batches of demersal eggs on exposed rock surfaces with low lying algae, with males guarding the eggs (Fitch and Lavenberg 1975; Love 1996). Reports of the spawning period and duration vary among sources with Moser (1996) reporting spawning activity from October–July and Fitch and Lavenberg (1975) indicating

spawning takes place February–November. Females collected from Diablo Cove and the surrounding area in May and September contained developing and mature eggs (Burge and Schultz 1973). Females spawn three times during a 3 mo (summer) season in Puget Sound and at least three times over 7 mo (September–March) in Monterey Bay, California (DeMartini and Anderson 1980). The number of eggs produced annually ranges from ca. 12,000–28,500 (DeMartini and Anderson 1980; Love 1996). Larvae are planktonic and can remain in the water column for 1–3 mo (DeMartini and Anderson 1980; Moser 1996). The larvae transform to juveniles at a size of approximately 15 mm (Kendall and Vinter 1984).

Scientific literature gives no estimates of natural mortality rates for either early life stages or later (juvenile or adult) stages of painted greenling. Mortality rates have not been calculated for adults using age-frequency analyses because age distributions in the populations sampled were not stationary (DeMartini and Anderson 1980). Hatching success per brood ranged from 31%± 9% to 72% ± 6% (DeMartini 1987). Laboratory mortality rates during the yolk-sac stage for another family member, the greenling (*Hexagrammos otakii*), were calculated from survival curves as 0.001~0.003 (Hamai and Kyushin 1964). Survival rates at the end of the yolk absorption period ranged from 13% to 38%. During the feeding period of the larvae, the mortality rate (0.018~0.083) depends on temperature, with higher temperatures corresponding to higher mortality rates (Hamai and Kyushin 1964). These laboratory-derived mortality rates were measured in the absence of natural mortality (e.g., predation) and, therefore, were not appropriate to use as the larval mortality rates of painted greenling.

Longevity and age at maturation of painted greenling appear to vary with location. Their longevity was estimated at 8 yr from a 173 mm total length (TL) specimen collected in the vicinity of Diablo Canyon (Burge and Schultz 1973). In Monterey Bay, California the maximum longevity was estimated as 5 yr at 160 mm TL for males and 6 yr at 175 mm TL for females. Fish in Puget Sound had a maximum longevity of 6 yr at 200 mm TL for males and 8 yr at 215 mm TL for females (DeMartini and Anderson 1980). Age at 50% maturity was 3 yr for females and 2 yr for males in Monterey Bay (DeMartini 1976; DeMartini and Anderson 1980).

Painted greenling have limited fishery value. They are typically taken incidentally by recreational and commercial fisheries while targeting other fishes (Fitch and Lavenberg 1975; Love 1996). Thus there are no stock assessments or catch records with which to compare losses attributable to power plant entrainment.

5.2.5.1 Summary of Field Collections

Painted greenling larvae were collected almost year round at the DCPP intake structure during the years 1996–1999 (**Appendix H**), with their highest abundance occurring in late March and May 1997 (**Figure 5.2.5-1**). It appears there were fewer painted greenling larvae during 1998 than during either 1997 or 1999, especially during February and March 1998. This might have been due to the El Niño event that was detected along the central California coast in the fall of 1997 (Lynn et al. 1998; NOAA 1999). There were 1,133 larval painted greenling in 553 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 12% of the subsamples collected and processed from that location during that period.

Painted greenling larvae occurred in the DCPP study grid during most months except September 1997, and August and September 1998 (**Appendix H**). The highest density occurred during May and June 1999 (**Figure 5.2.5-2**) and in the grid cells closest to shore. There were 372 larval painted greenling identified from 261 bongo net subsamples representing 9% of the study grid subsamples collected and processed from July 1997–June 1999.

Standard lengths of all painted greenling larvae collected at the DCPP intake structure between October 1996 and June 1999 ranged from a minimum of 2.1 mm to a maximum of 7.9 mm (**Figure 5.2.5-3**). The central 98% of this length-frequency distribution resulted in minimum and maximum lengths for the analyses of 3.2 mm and 5.2 mm, respectively. The mean larval length in this distribution was 4.1 mm. The growth rate of larval painted greenling (0.083 mm/d) reported by Freeman et al. (1985) was used to estimate ages of entrained larvae. We assumed that the shortest larva of the central 98% of the length-frequency distribution was immediately post-hatch and aged zero days. Consequently, the estimated ages of larvae entrained ranged from zero

days up to 24–70 d post-hatching for the largest larvae measured. On average, the estimated ages of painted greenling larvae entrained at DCPP ranged from 10–24 d post-hatching. Reported hatching size for painted greenling is less than 3.5 mm (Moser 1996). The fact that we observed some larvae smaller than the reported hatching lengths may be partly explained by natural variation of hatch lengths within the population and partly by the phenomenon of shrinkage following preservation (Theilacker 1980).

Significant differences were detected between mean lengths (*t*-test: $p \leq 0.041$) and between length-frequency distributions (Kolmogorov-Smirnov test: $p \leq 0.002$) of larval painted greenling from the DCPP intake structure and study grid despite the appearance of similarity between the two distributions (**Figure 5.2.5-4**). The mean standard lengths from the intake and study grid were 4.1 mm and 4.0 mm, respectively. The maximum difference between the length-frequency distributions detected by the Kolmogorov-Smirnov goodness of fit test occurred at 4.1 mm, indicating that smaller individuals were collected from the DCPP intake. While these differences were statistically detectable and significant, we believe that the biological significance of these differences was low.

5.2.5.2 Estimating Total Annual Entrainment

The annual estimated mean entrainment for painted greenling larvae ranged from a minimum of 8,410,000 ($\widehat{SE}$ =800,000) in Analysis Period 2 to a maximum of 22,000,000 ($\widehat{SE}$ =1,600,000) in Analysis Period 1 (**Table 5.2.5-1**). Values of $2 \cdot \widehat{SE}(\widehat{E_T})$ can be used to approximate 95% confidence intervals around the point estimates. These values for October 1996–September 1997 and October 1997–September 1998 would not overlap, indicating that the differences between them is statistically significant.

Estimates of Annual estimates of entrained painted greenling larvae adjusted to a long-term mean based on annual indices using weekly Intake Cove surface plankton tows (**Figure 5.2.5-5**) ranged from a minimum of 9,610,000 ($\widehat{SE}$ =914,000) in Analysis Period 2 to a maximum of 24,200,000 ($\widehat{SE}$ =1,760,000) in Analysis Period 1 (**Table 5.2.5-2**). The average index for the years 1997 and 1998 were 0.0023 and 0.0022 larvae/m^3, respectively. The long-term average index was 0.0025

larvae/m^3 for the years 1990–1998, yielding the ratio ($\bar{I} / I_i$) of 1.0990 for 1997 and 1.143 for 1998 and indicating that larval greenling abundance was lower than the long-term average during 1997 and 1998. The adjustment increased the differences between the estimates for Analysis Periods 1 and 2.

NOTE: Entrainment estimates for painted greenling were not used to calculate *FH* or *AEL* models because no demographic data (e.g., larval survivorship) were available to parameterize these approaches.

5.2.5.3 Empirical Transport Model (*ETM*)

Monthly estimates of proportional entrainment (PE_i) in each *i*th survey from the two years sampled ranged from a minimum of zero to a maximum of 0.0423±0.0393 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in February 1998 (**Table 5.2.5-3**). This maximum $\widehat{PE_i}$ was not associated with the maximum $\hat{f}_i$ (0.252) which occurred in July 1997 and thus was proportionally less important during that analysis period. No larvae were collected at either the DCPP intake or from the study grid when both $\widehat{PE_i}$ and the proportion of larvae present in the *i*th survey period ($\hat{f}_i$) were equal to zero. When $\widehat{PE_i}$ =0 but $\hat{f}_i$>0, it indicated that larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. During the Analysis Period 3, the highest $\widehat{PE_i}$'s were all approximately 0.04, but only January 1998 had a relatively large $\hat{f}_i$ weighting factor (0.17) which meant it had proportionally greater influence on the annual entrainment mortality estimate. In Analysis Period 4, one of the lowest $\widehat{PE_i}$ values carried the greatest weighting factor $\hat{f}_i$=0.23.

Estimates of $\widehat{P_M}$ between analysis periods and for both larval durations are relatively similar and ranged from 0.032–0.067 (**Figure 5.2.5-6**). Since painted greenling larvae were dispersed throughout the study grid during most surveys (**Figure 5.2.5-2**), P_S was calculated using alongshore current movement and extrapolating study grid abundance offshore using onshore current movement. The estimates calculated using offshore extrapolation of study grid abundance were also similar between the two analysis periods and two larval durations and ranged from 0.004–0.011. Estimates of P_M for both analysis periods using either mean or maximum length in

calculating larval duration either declined are remained approximately the same when $1/\widehat{P_S}$ increased.

5.2.5.4 Interpretation of Assessment Results

Painted greenling has very little commercial or recreational fishery value. Because of the absence of any commercial fishery for this species, there are no catch data or stock assessments with which to compare entrainment mortality rates ($\widehat{P_M}$). Results of the *ETM* modeling show that the power plant may annually entrain approximately 4–5% of the painted greenling larvae over an area 6 to 7 times the area of the study grid (**Figure 5.2.5-6**).

Abundance of adult painted greenling was examined using data from the DCPP Receiving Water Monitoring Program studies on subtidal fishes (**Figure 5.2.5-7**). There was considerable variation in greenling abundance over the period of observation with one obvious decline in abundance from 1992 through 1994. This may have resulted from the extended El Niño events that occurred from 1991 through 1993 (Yoklavich et al. 1996) and again in 1994 (NOAA 1999). Annual mean density of painted greenling larvae from Intake Cove surface plankton tows show high larval production in 1992 and 1995, but no coherent trends (**Figure 5.2.5-5**). The absence of any evidence of long-term effects (>5 yr) on painted greenling abundance from these two sources indicates that entrainment of larvae by the power plant CWS is only removing excess larval production that does not affect local adult abundance or larval production.

The lack of evidence of declining trends in adult abundance or larval production could also be the result of compensation for the additional larval mortality resulting from entrainment. If, as reported, painted greenling larvae transform into juveniles at a size of 15 mm and are planktonic for 1–3 mo (DeMartini and Anderson 1980; Moser 1996), then our results indicate that the larvae are only subject to entrainment for a very limited period of time. The mean of the size frequency distribution for entrained larvae was 4.0 mm, and 99% of all the larvae ranged from 3.2 to 5.3 mm (**Figure 5.2.5-3**). The period of time represented by this size range is much less than the reported larval duration (up to 90 d) and may even be shorter if, as suspected, the lower sizes represent variation in hatch size. Only one larva was collected that was within the range of

Moser's (1996) reported length at flexion (7–9 mm). The absence of flexion and post-flexion length larvae in the samples may indicate that more developed larvae are able to avoid capture or that behavioral changes result in the larvae moving out of the plankton and into other habitats where they are not subject to capture or entrainment. These observations may also indicate that painted greenling larvae quickly settle out of the plankton and may have mechanisms in later stages that compensate for mortality of planktonic larvae since there do not appear to be declines in larval abundance through time.

Table 5.2.5-1. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for painted greenling (*Oxylebius pictus*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	22,000,000	1,600,000
2) Oct 1, 1997–Sep 30, 1998	8,410,000	800,000
3) Jul 1, 1997–Jun 30, 1998	11,100,000	970,000

Table 5.2.5-2. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for painted greenling (*Oxylebius pictus*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	24,200,000	1,760,000
2) Oct 1, 1997–Sep 30, 1998	9,610,000	914,000
3) Jul 1, 1997–Jun 30, 1998	12,100,000	1,110,000

Table 5.2.5-3. Painted greenling (*Oxylebius pictus*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.00894	0.00593	0.252	0.016
Aug 25, 1997	0.00954	0.0134	0.0329	0.0046
Sep 29, 1997	0	0	0	0
Oct 20, 1997	0.0398	0.0474	0.0310	0.0041
Nov 17, 1997	0.00751	0.0106	0.0191	0.0037
Dec 10, 1997	0.0125	0.0144	0.0333	0.0050
Jan 22, 1998	0.0405	0.0206	0.1703	0.0118
Feb 26, 1998	0.0423	0.0393	0.0296	0.0035
Mar 18, 1998	0.0361	0.0255	0.0345	0.0044
Apr 15, 1998	0.00440	0.00341	0.101	0.0112
May 18, 1998	0.00780	0.00318	0.089	0.0075
Jun 8, 1998	0.0162	0.00515	0.207	0.0123

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.00285	0.00305	0.0262	0.00366
Aug 26, 1998	0	0	0	0
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0.00741	0.00827	0.0108	0.00196
Jan 12, 1999	0.0153	0.00478	0.124	0.00499
Feb 3, 1999	0.0181	0.00464	0.168	0.00562
Mar 17, 1999	0.0116	0.00362	0.236	0.00960
Apr 14, 1999	0.00924	0.00450	0.0775	0.00524
May 24, 1999	0.00997	0.00338	0.233	0.00962
Jun 23, 1999	0.00747	0.00208	0.125	0.00601

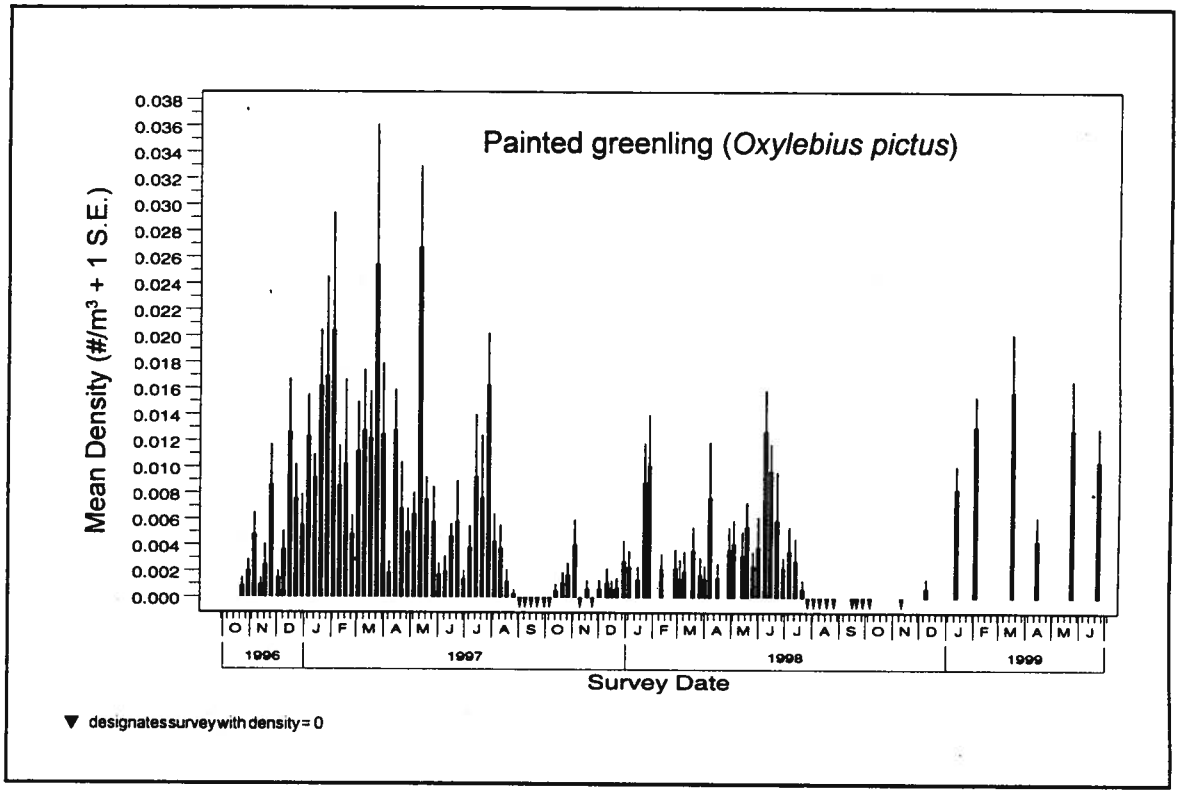


Figure 5.2.5-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

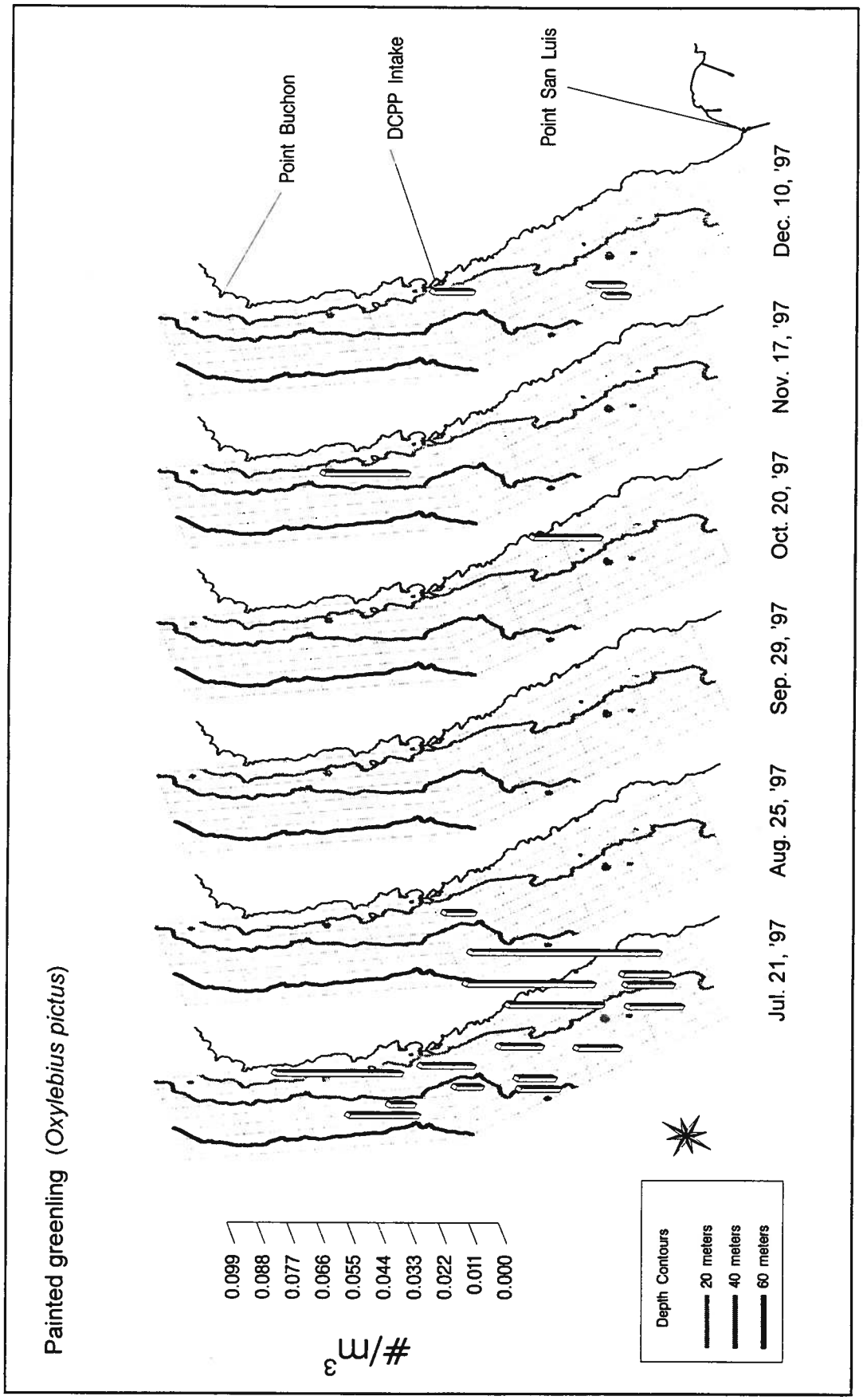


Figure 5.2.5-2a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

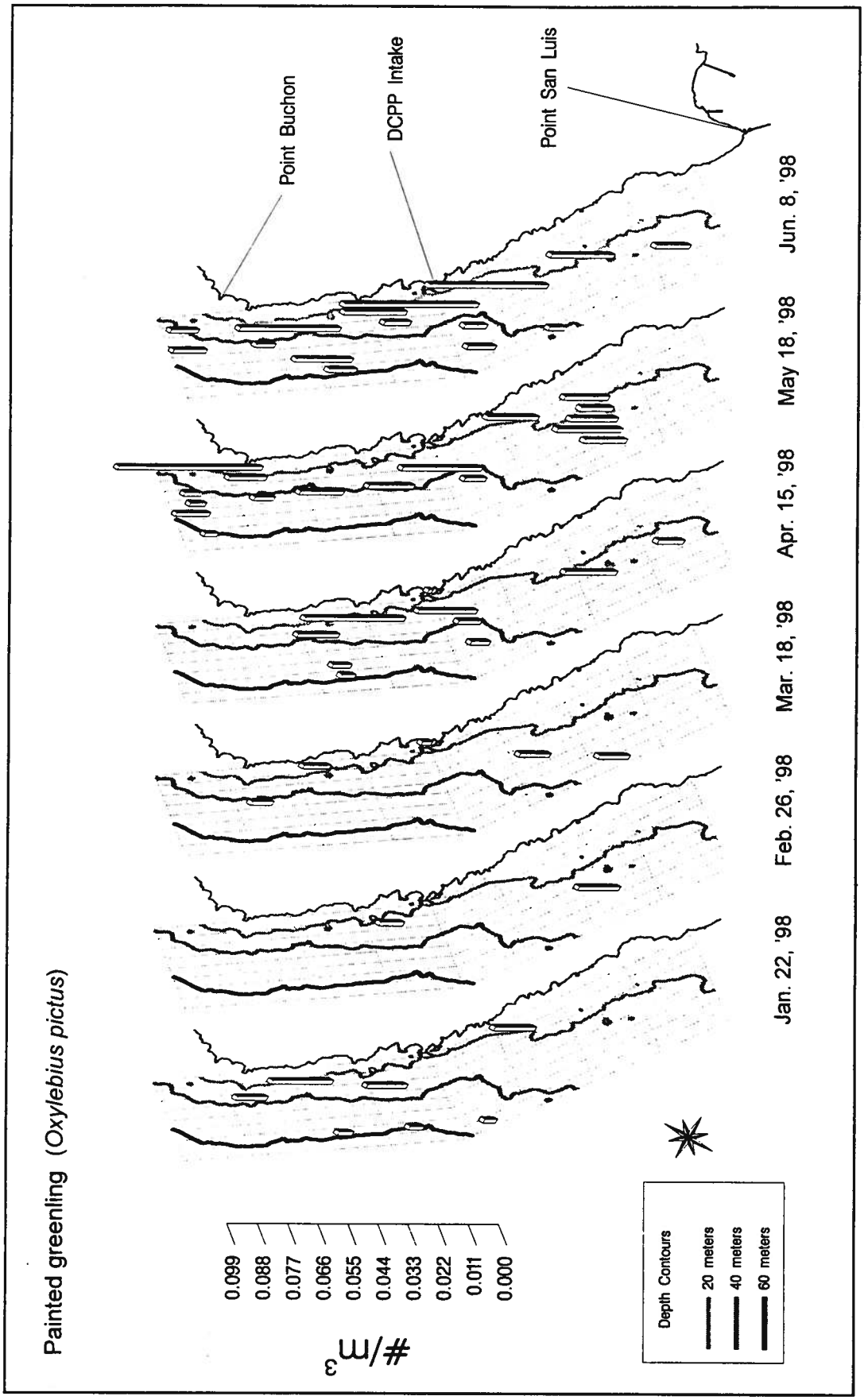


Figure 5.2.5-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

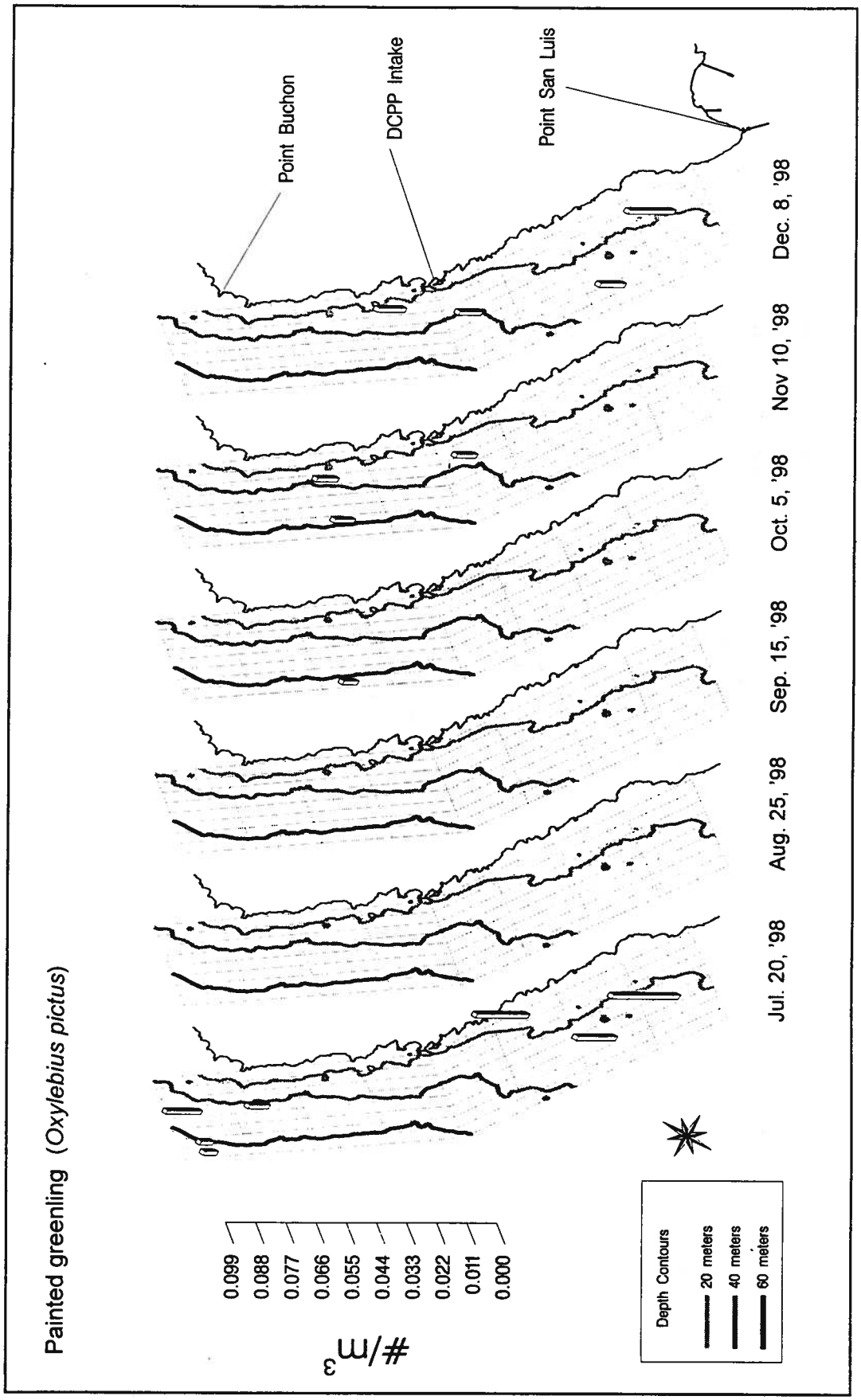


Figure 5.2.5-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

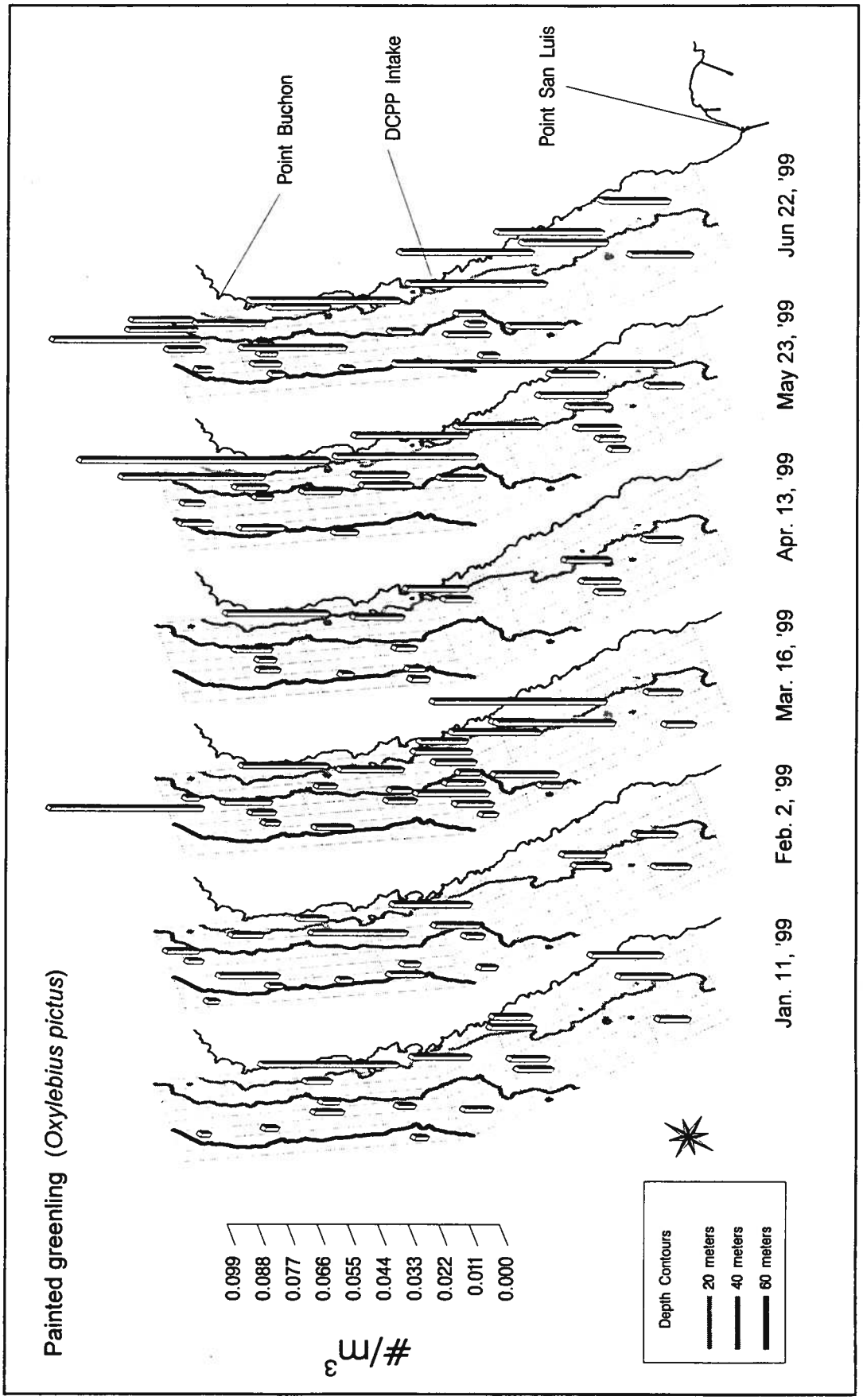


Figure 5.2.5-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

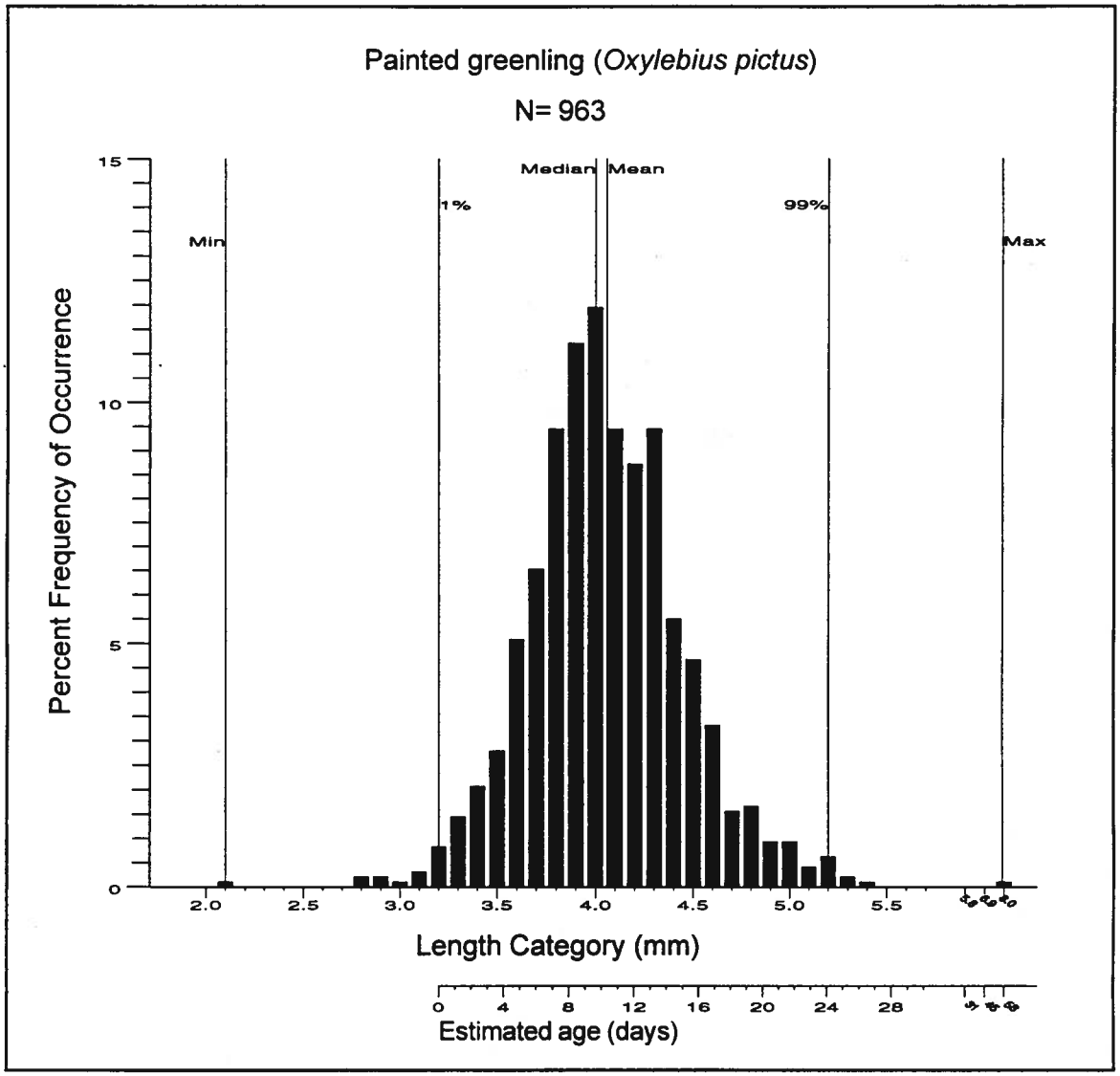


Figure 5.2.5-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

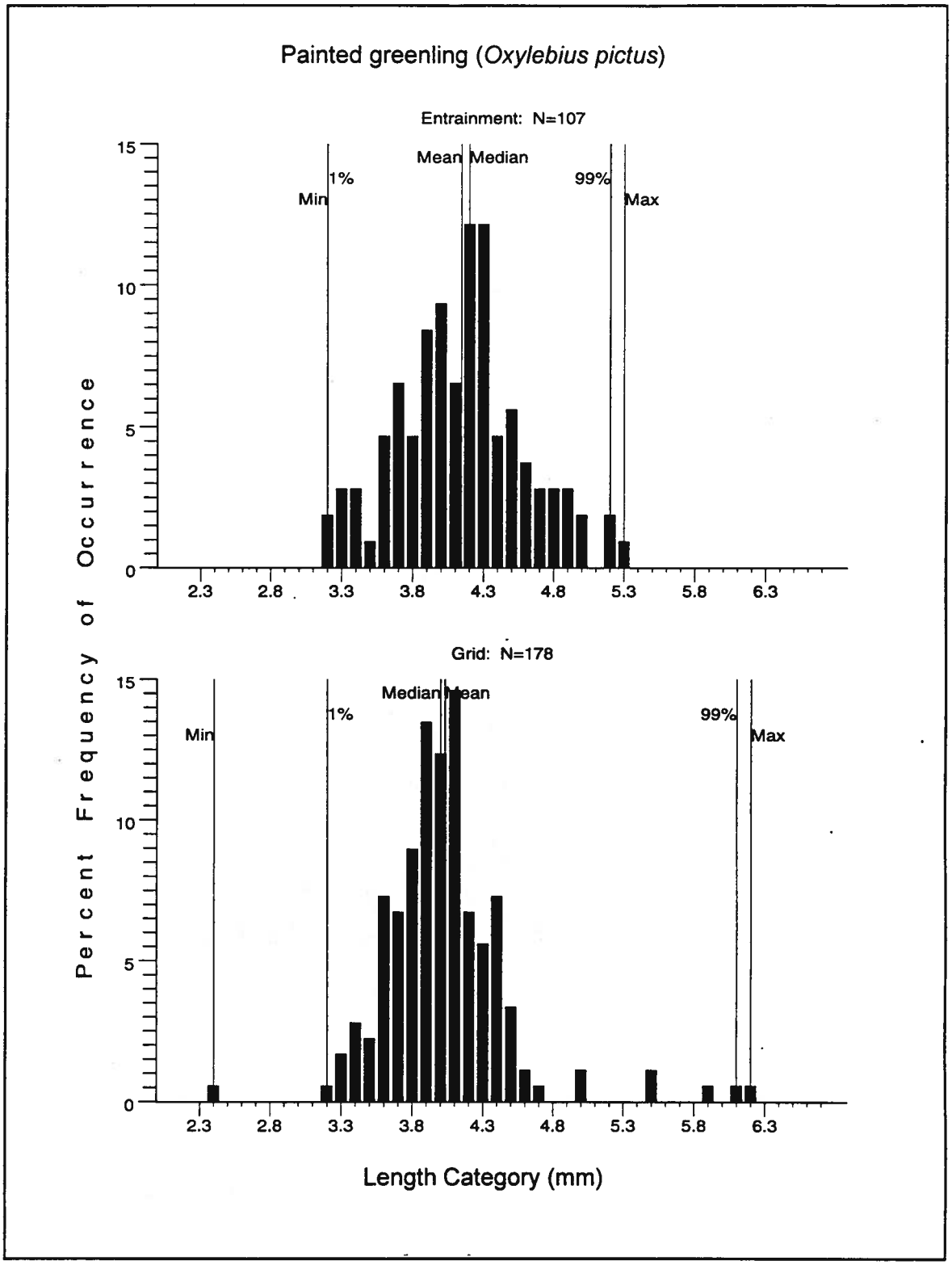


Figure 5.2.5-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

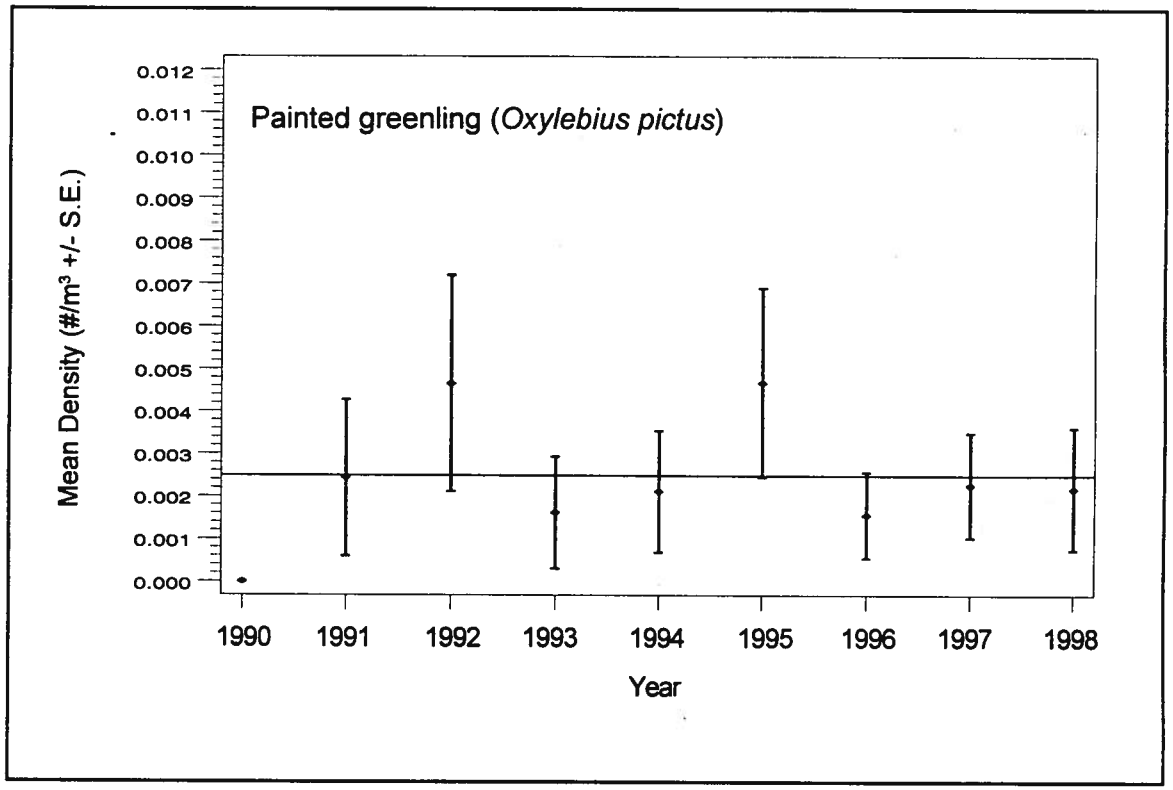


Figure 5.2.5-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

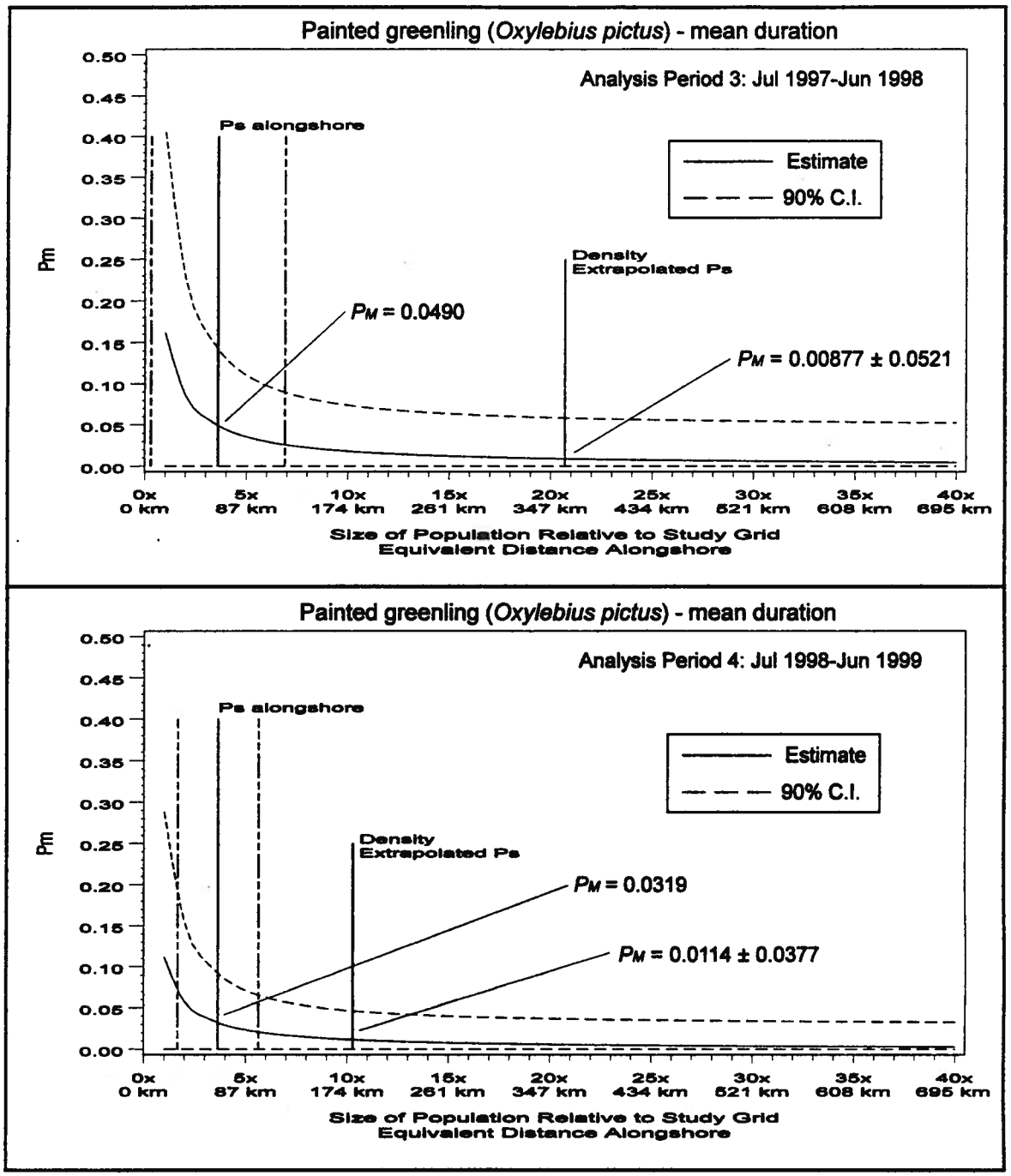


Figure 5.2.5-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

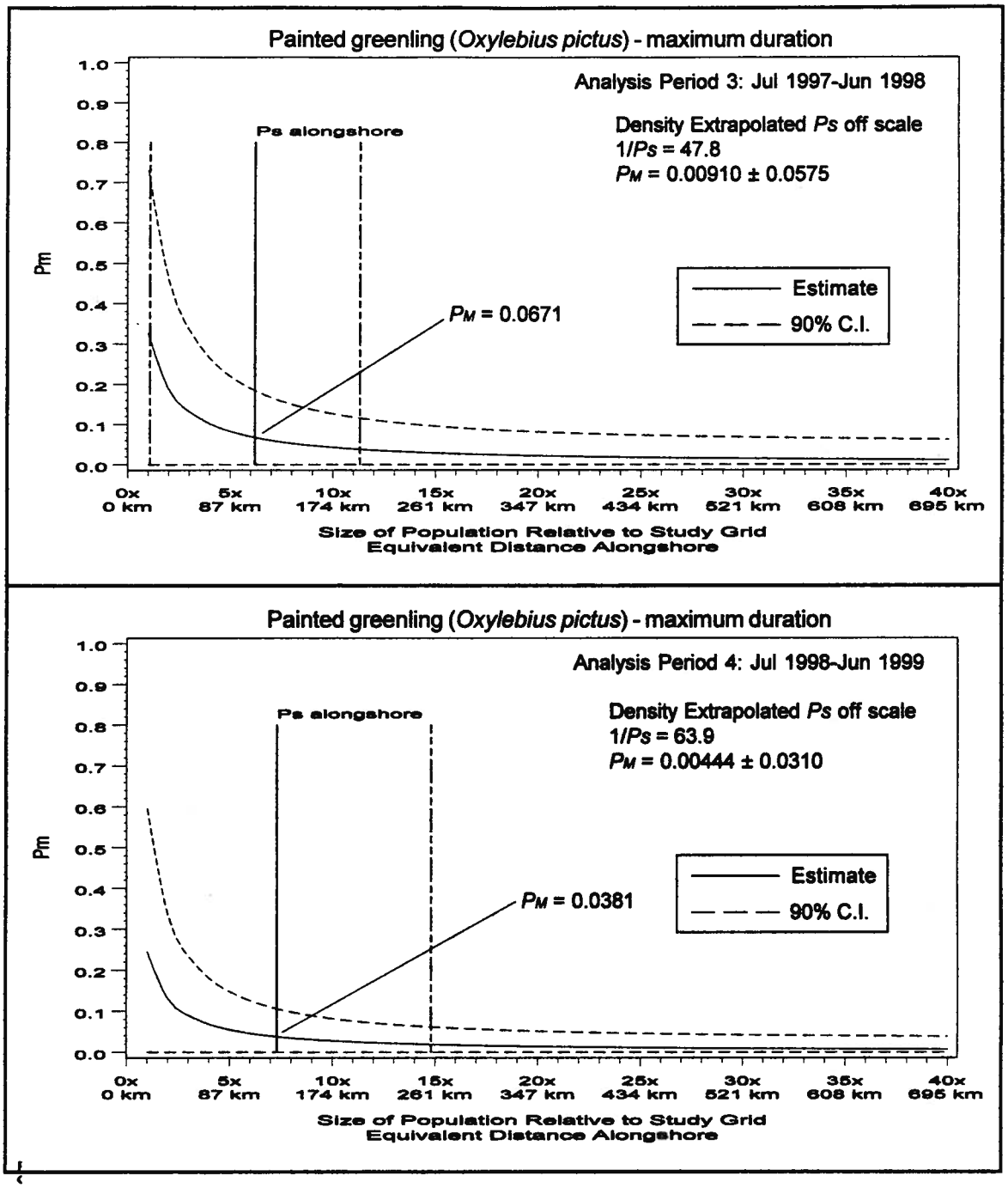


Figure 5.2.5-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

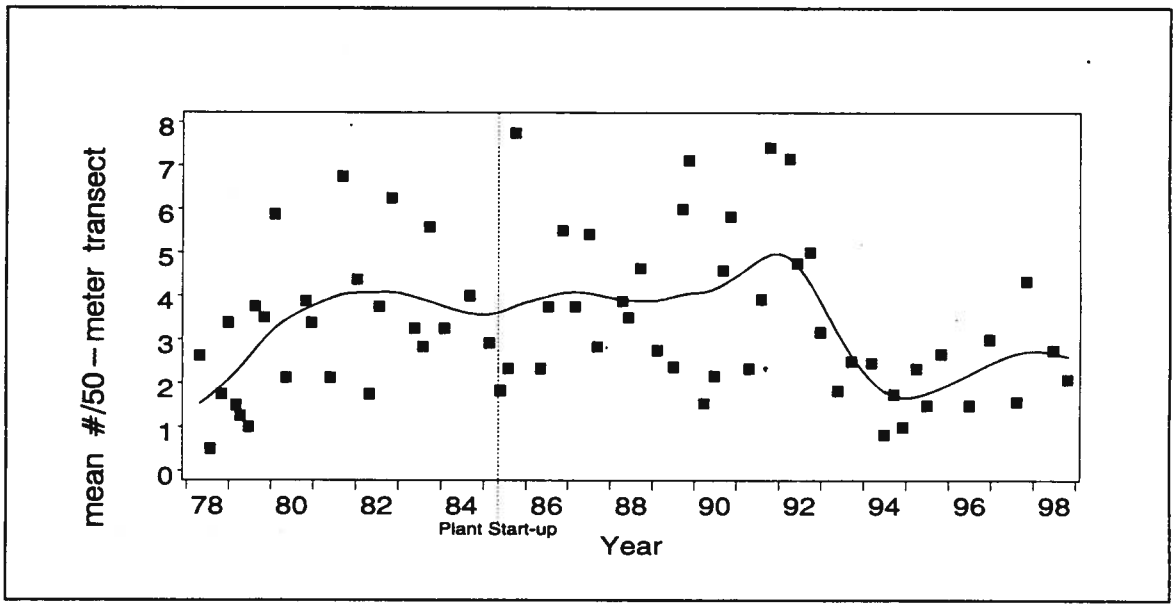


Figure 5.2.5-7. Mean number of painted greenlings observed per 50-meter subtidal benthic transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to fit curve through points.

5.2.6 Assessment of Sculpins (Family Cottidae)

Cottidae is a scorpaeniform family that comprises 70 genera worldwide (Nelson 1994). Forty-two species of sculpin occur along the California coast (Miller and Lea 1972), primarily in intertidal or shallow subtidal habitats. The ETWG selected three cottid species for assessment in the DCPP entrainment study: smoothhead sculpin (*Artedius lateralis*), snubnose sculpin (*Orthonopias triacis*), and cabezon (*Scorpaenichthys marmoratus*).

Intertidal and shallow subtidal fishes, including the cottids, display a wide range of life histories that defy broad demographic generalization. Species within the family Cottidae display a variety of life history patterns ranging from relatively short-lived to longer-lived species (Gibson 1969, 1982; Miller 1979). The large staghorn sculpin (*Leptocottus armatus*), common in Pacific coast bays and estuaries, is known to live to 3 yr and reach sexual maturity after 1 yr (Jones 1962; Tasto 1975). The fluffy sculpin (*Oligocottus snyderi*) and the tidepool sculpin (*O. maculosus*) have short lifespans and early maturation. This conclusion is based on growth rate data coupled with the scarcity of individuals older than 1.5 yr and on data showing early maturation and high reproductive output for fluffy sculpin (deVlaming et al. 1982). Cabezon, the largest of the North American cottids, may live 13 yr but only inhabits tidepools during its first or second year of life (O'Connell 1953). The demography of cottids is not well known.

A detailed review of the demography of several species of cottids closely related to smoothhead sculpin (*Artedius lateralis*) will show why substitution of early life history characteristics from related taxa is not possible for target taxa from the Cottidae. Begle (1989) proposed a phylogenetic classification placing the genera *Oligocottus* and *Clinocottus* as the nearest relatives to the genus *Artedius*. As with smoothhead sculpin, sculpins in the genera *Oligocottus* and *Clinocottus* are oviparous; lay demersal, adhesive egg batches; show varying degrees of parental care of the eggs; and produce pelagic larvae.

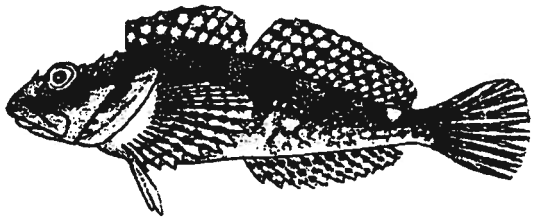
Though the demography of local species from these closely related genera (i.e., tidepool sculpin, fluffy sculpin, and woolly sculpin [*Clinocottus analis*]) is similar in many respects to that of smoothhead sculpin, information necessary to compute *AEL* and *FH* for this latter species is

lacking (**Table 5.2.6-1**). The tidepool sculpin achieves a maximum size of 8.9 cm, becomes sexually mature by 12 mo (Pierce and Pierson 1990), and primarily occurs in the intertidal zone. Using unvalidated ages, Pierce and Pierson (1990) reported mean annual fecundity ranges from 103 eggs for an age class I female to 699 eggs for an age class III female. The fluffy sculpin also reaches a maximum size of ca. 8.9 cm (Eschmeyer et al. 1983) and a maximum age of 1.5 yr at Dillon Beach, California (Freeman et al. 1985). It may spawn more than once during its approximately 8 mo reproductive period (Grossman and deVlaming 1984) and is found in tidepools and shallow rocky areas from Sitka, Alaska to Rio Socorro, northern Baja California (Miller and Lea 1972). Instantaneous growth rates of wild-caught fluffy sculpin adults, estimated by length-frequency analyses, varied between males and females and range from 0.063 for an age 1+ female to 1.059 for an age 0+ male (Freeman et al. 1985).

The wooly sculpin (*Clinocottus analis*) is larger and longer lived than either the smoothhead sculpin or the two representatives of the genus *Oligocottus*. It attains a maximum size of 18 cm (Eschmeyer et al. 1983) and a maximum age of 8 yr for males and 6 yr for females (Wells 1986). This sculpin is found from the intertidal zone to 18 m depths and from Cape Mendocino, northern California to Punta Asuncíon, central Baja California (Miller and Lea 1972). It spawns in tidepools at Point Fermin, southern California from September through November. Young-of-the-year (YOY) recruit to tidepools in southern California from November to February (Wells 1986). Egg incubation takes ca. 18–30 d in the laboratory (Eigenmann 1892; Budd 1940; Hubbs 1966). Wells (1986) indicates that all specimens over 60 mm TL and between zero and 1 yr of age appear sexually mature with batch fecundity described by the linear function $F=11.6TL-620.6$ ($n=45$; $r=0.940$; $P\leq0.01$). Males of this species appear to grow faster and to a larger size than females (Wells 1986). The largest female collected (110 mm) contained 784 eggs; the mean number of eggs was 242 per average reproductive female (74 mm; Wells 1986). Number of spawns per season is unknown, but multiple spawns are implied by tri-modal size distribution of ova present in gonads (Wells 1986). Based on Budd (1940) examining wooly sculpin and Morris (1951) examining bald sculpins, 11–25 mm fishes were surmised to be newly settled from the pelagic larval phase and hatched approximately 6–8 wk prior (Wells 1986). Despite these life

history descriptions, there are no estimates of larval survivorship with which to parameterize the *FH* or *AEL* models for members of the Cottidae.

5.2.6.1 Smoothhead sculpin (*Artedius lateralis*)



Artedius lateralis (Girard 1854); smoothhead sculpin; length to 14 cm; Kodiak Island, Alaska to Cabo San Quintin, northern Baja California; intertidal to 13 m; greenish to brown on top, cream to light brown below (Miller and Lea 1972; Eschmeyer et al. 1983).

The smoothhead sculpin is a common nearshore, intertidal cottid (Miller and Lea 1972). Spawning varies between locations: winter–spring in British Columbia (Marliave 1977) and June in Puget Sound (Matarese et al. 1989). Their eggs hatch into pelagic larvae in about 16 d at 15.5°C (Budd 1940; Matarese et al. 1989). Larvae of this species have been collected in DCPP entrainment samples nearly year-round (**Figure 5.2.6-1**). Love (1996) indicates that these sculpins likely mature within their first year of life and probably live as long as 3 yr.

The demographic data available for smoothhead sculpin and its close relatives (**Table 5.2.6-1**) do not provide sufficient information to compute *AEL* or *FH*. Estimates of fecundity and spawning periodicity are available for closely related species that likely compare favorably with *A. lateralis* based on their similar ecological roles, adult habitats, and close phylogenetic relationships. However, in the absence of any estimates of egg or larval survivorship, *FH* or *AEL* cannot be computed for this species.

Recreational and commercial fisheries do not target most small intertidal and shallow subtidal cottids. This is one reason that the literature on these species is limited. The absence of any

population estimates from fishery management plans or other sources will make assessment of impacts on the smoothhead sculpin population difficult.

Summary of Field Collections

Estimates of smoothhead sculpin larval abundance showed distinct seasonal peaks at the DCPP intake structure during the years 1996–1999 (**Appendix H**), with greatest abundance occuring between April and July (**Figure 5.2.6-1**). There were 5,598 larval smoothhead sculpin identified from 1,320 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 28% of the samples collected and processed from the intake structure during that period. With the exception of one very high peak density in April 1998, the distribution of abundance and magnitude of density did not appear to be different between 1997 and 1998. Although only entrainment surveys that were paired with study grid surveys were processed after September 1998, the abundance patterns appear to be similar to previous years.

Larval smoothhead sculpin generally occurred with similar seasonality in the DCPP study grid and at the DCPP intake structure (**Appendix H**). The largest densities in the study grid occurred in July 1997, April 1999, and May 1999 (**Figure 5.2.6-2**). There were 676 smoothhead larvae identified from 312 bongo net subsamples representing 10% of the study grid samples collected and processed from July 1997–June 1999. Larval abundance appeared to track the 20 m isobath, with larvae likely concentrated inshore of this contour (Marliave 1986). Apparently, the higher density of this species in the southern portion of the study grid results from the fact that more of the area in the southern half of the grid is within the 0–20 m depth range.

Standard lengths of 4,929 smoothhead sculpin larvae measured from samples collected at the DCPP intake structure between October 1996 and June 1999 ranged from 1.6 mm to 11.8 mm (**Figure 5.2.6-3**). The central 98% of this length-frequency distribution resulted in minimum and maximum lengths of 2.4 mm and 5.3 mm, respectively. The mean larval length in this distribution was 3.1 mm. Reported hatch size for this species ranged from 3.9–4.5 mm (Moser 1996). The fact that we observed some larvae smaller than the reported hatching lengths can be explained partly by natural variation of hatch lengths within the population and partly by the phenomenon

of shrinkage following preservation (Theilacker 1980). The growth rate applied to this species (0.08 mm/d) from Freeman et al. (1985) was used to estimate ages of entrained larvae. Assuming that the shortest larvae (either 1.6 or 2.4 mm) were immediately post-hatch and aged zero days, the estimated ages of larvae entrained could range from zero days up to 35–123 d post-hatching for the largest larvae measured. The average estimated ages of smoothhead sculpin larvae entrained at DCPP ranged from 9–19 d post-hatching.

Significant differences exist between mean lengths (*t*-test: $p < 0.001$) and between length-frequency distributions (Kolmogorov-Smirnov test: $p < 0.001$) of larval smoothhead sculpin from the DCPP intake structure and study grid, despite the appearance of similarity between the two distributions (**Figure 5.2.6-4**). The mean standard lengths from the intake and study grid were 3.1 mm and 3.7 mm, respectively. The large sample size (655 larvae from the intake and 280 larvae from the study grid) causes the *t*-test to be highly sensitive to small differences between the means (Zar 1984). Median lengths were similar at 3.1 mm and 3.2 mm at the intake structure and in the grid, respectively. The maximum difference between the length-frequency distributions detected by the Kolmogorov-Smirnov goodness of fit test occurred at 3.9 mm, indicating that more large individuals were collected from the grid. While these differences were statistically detectable and significant, we believe that the biological significance of these differences is low.

Estimating Total Annual Entrainment

Annual mean entrainment estimates for smoothhead sculpin larvae ranged from 88,900,000 ($\widehat{SE}$ =3,660,000) in 1996–97 Analysis Period 1 to 96,100,000 ($\widehat{SE}$ =9,410,000) in 1997–98 Analysis Period 2 (**Table 5.2.6-2**). The estimates for 1997–98 Analysis Period 3 were slightly different than Analysis Period 2 because smoothhead sculpin larvae were collected during the summer and fall months of 1997 when the two periods did not overlap. Values of $2 \cdot \widehat{SE}(\widehat{E_T})$ can be used to approximate 95% confidence intervals around the point estimates. Confidence intervals for the 1996–97 and 1997–98 estimates would overlap, indicating that the differences among them were probably not significant.

Annual entrainment estimates for smoothhead sculpin larvae adjusted to a long-term mean based on annual indices using weekly Intake Cove surface plankton tows (**Figure 5.2.6-5**) ranged from 57,700,000 ($\widehat{SE}$ =2,370,000) in Analysis Period 1 to 115,000,000 ($\widehat{SE}$ =11,300,000) in Analysis Period 2 (**Table 5.2.6-3**). The average index values for 1997 and 1998 were 0.0734 and 0.0397 larvae/m^3, respectively, while the long-term average index was 0.0476 larvae/m^3 for the years 1990–1998, yielding the ratio $\overline{I}/I_i$ of 0.649 for 1997, and 1.20 for 1998. Thus, the adjusted values increase the differences between the 1996–1997 and the 1997–1998 analysis periods.

NOTE: Estimates of *FH* or *AEL* for smoothhead sculpin using annual entrainment were not calculated because no demographic data exists for this species.

Empirical Transport Model (ETM)

Monthly estimates of proportional entrainment (PE_i) in the surveys from the two analysis periods ranged from a minimum of zero to a maximum of 0.0729±0.0181 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in June 1999 (**Table 5.2.6-4**). Both April and June 1999 demonstrated similar $\widehat{PE_i}$ values around 0.07, but the proportional entrainment estimate in April had greater weight with an annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) equal to 0.30 as compared with an $\hat{f}_i$=0.18 from June of 1999. When both $\widehat{PE_i}$ and $\hat{f}_i$ were equal to zero, there were no larvae collected at either the DCPP intake or the study grid. When $\widehat{PE_i}$ = 0 but $\hat{f}_i$>0 (e.g., Sept.–Nov. 1997), larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. The largest values of $\hat{f}_i$ occurred April through June in both analysis periods indicating that these were times of peak hatching for this species.

Smoothhead sculpin adults are found in the shallow nearshore area (Miller and Lea, 1972). Study grid samples showed that the larvae were also distributed primarily in nearshore areas (**Figure 5.2.6-2**). Therefore, P_S was calculated using only alongshore current movement to calculate point estimates of $\widehat{P_M}$. Results for smoothhead sculpin show how large differences in estimated larval durations (9 vs. 35 d) can affect estimates of $\widehat{P_M}$ (**Figures 5.2.6-6**). Estimates of total

entrainment mortality increased about 5% with increasing larval transport duration in both periods (Period 3, mean duration $\widehat{P_M} \cong 11\%$ and max. duration $\widehat{P_M} \cong 15\%$: Period 4, mean duration $\widehat{P_M} \cong 15\%$ and max. duration $\widehat{P_M} \cong 20\%$) with the second year being higher overall. Values for $\widehat{P_M}$ were greater in the second analysis period. The probability of entrainment is less for the shorter duration in both analysis periods because of the reduced time over which larval transport occurred. Estimates from longer larval durations showed that weaker alongshore currents resulting from the El Niño conditions during the 1997–98 analysis period resulted in a larger estimate of $\widehat{P_S}$ and smaller areas of inference compared to the 1998–99 analysis period.

Interpretation of Assessment Results

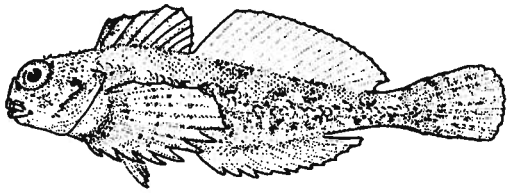
Smoothhead sculpin has neither commercial nor recreational fishery value, and there is little information on its ecological role in the community. Because of the absence of any fishery for this species, no catch data can be used to compare harvest mortality rates to entrainment mortality rates ($\widehat{P_M}$). There are also no estimates of stock size or density that could be used to convert $\widehat{P_M}$ into an estimate of adult equivalent loss, assuming no compensatory mortality.

Annual mean densities of smoothhead sculpin larvae from Intake Cove surface plankton tows did not show any long-term trends in abundance from 1990–1998 (**Figure 5.2.6-5**). Although the estimate for 1998 is close to the long-term mean density, the data overall show a large amount of variation among years. The data do not provide any evidence of a reduction in larval production that would affect recruitment and eventually adult population density. Trends in adult populations of smoothhead sculpin were examined using data from the DCPP Receiving Water Monitoring Program studies on subtidal fishes. Mean abundance from three 50 m transects in an area approximately 1 km south of Diablo Cove and not contacted by the plant's thermal discharge combine data for smoothhead sculpin with data for other cottids because of the difficulties in identifying these small, cryptic fishes underwater (**Figure 5.2.6-7**). The data vary considerably among years, but the fitted curve shows a declining trend from 1992 through 1998. Although these data are variable the timing of the apparent decline is consistent with the life-span (~3 yr)

and age of sexual maturity (~1–2 yr) for this species, since any reductions in adult density of smoothhead sculpin would only be expected to be detectable after several generations.

The sizes of the majority of the entrained larvae ranged from 2.0 mm to 4.5 mm (**Figure 5.2.6-3**). Although larvae as large as 11.9 mm were entrained, there were few larvae greater than the reported size at flexion of 5.0–6.3 mm (Moser 1996). In fact, most of the larvae were less than the maximum hatch size of 4.5 mm reported by Moser (1996). This may indicate that smoothhead sculpin larvae are only subject to entrainment over a limited period of their larval development. It may also indicate that larger larvae are avoiding capture or that behavioral changes result in the larvae moving out of the plankton and into other habitats where they are not subject to entrainment or capture using our sampling methods. The considerable variation in adult densities, the absence of a decreasing trend in larval abundance, and the relatively short developmental period over which larvae are entrained indicates that the removal of approximately 10–20% of the smoothhead sculpin larvae from an area 7–8 times that of the study grid does not negatively affect local adult populations or larval abundance.

5.2.6.2 Snubnose sculpin (*Orthonopias triacis*)



Orthonopias triacis (Starks and Mann 1911); snubnose sculpin; length to 10 cm. Farallon Islands, northern California to Isla San Geronimo, northern Baja California; intertidal to 30 m; green to reddish brown or orange above, with dark and light mottling; white below (Miller and Lea 1972; Eschmeyer et al. 1983; Long 1992).

Despite the common occurrence of snubnose sculpin in nearshore rocky subtidal and intertidal habitats, their life history remains relatively undescribed. Females are oviparous and spawn year round with peaks between February and October. The eggs are demersal and adhesive and hatch planktonic larvae (Feeney 1992; Moser 1996). Bolin (1941) conducted studies of the embryology

and development of early larval stages of *O. triacis* in laboratory rearing experiments. Egg incubation took 16–19 d at about 13°C, after which the larvae hatched at sizes ranging from 2.9–3.8 mm. All larvae died within 10 d of hatching despite several regimes of aeration and nutrition, increasing by about 0.2 mm in length over that time (i.e., growth rate of 0.02 mm/d). Yolk sacs were exhausted by about 5 d. The growth described above, representing the first 5–6 d of life until the yolk stores were exhausted, probably underestimates the growth rate in the wild where they can feed successfully. The growth rate of 0.083 mm/d used to estimate larval age of entrained *O. triacis* is from Freeman et al.'s (1985) description of the early life history of another intertidal cottid (*O. snyderi*).

The demographic data available for *O. triacis* do not provide sufficient information for computation of *AEL* or *FH*. Phylogenetic analyses indicate that the relationship of *O. triacis* to sister cottids found in similar habitats remains unresolved (Begle 1989). Recreational and commercial fisheries do not target small intertidal and shallow subtidal cottids. This is one reason that the literature on these species is limited. In the absence of egg or larval survivorship estimates, the impact of entrainment on the smoothhead sculpin population will be assessed using only the *ETM*.

Summary of Field Collections

Estimates of snubnose sculpin larval abundance showed seasonal peaks in late spring and early summer at the DCPP intake structure during the years 1997 and 1998 (**Figure 5.2.6-8**). There were 4,533 snubnose sculpin larvae identified from 1,422 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 31% of the samples collected and processed from the intake structure during that period. There appeared to be relatively compressed period of larval abundance in April though July of 1998 compared to a more protracted spawning season in1997 that lasted from February through September. This may have been related to the El Niño event that occurred during late 1997 (Lynn et al. 1998; NOAA 1999) as substantial increases in water temperatures and lowered productivity negatively affected

larval production and survival. Monthly samples collected in 1999 show larval densities similar in timing with 1997.

Larval snubnose sculpin generally occurred in the DCPP study grid with similar seasonality to that observed at the DCPP intake structure (**Appendix H**). The 447 snubnose larvae identified from 201 bongo net subsamples represented 7% of the fishes collected and processed from July 1997–June 1999 study grid samples. Thus, snubnose larvae were less common in the study grid than at the intake structure where their larvae were found in 31% of the samples. The peak abundances in the study grid occurred in July 1997 (**Figure 5.2.6-9**). During July 1997 and other surveys, snubnose sculpin larvae were typically more abundant in the nearshore areas of the grid. Similar to smoothhead sculpin, spatial distribution of larval snubnose abundance appeared to follow the 20 m isobath. Marliave (1986) found that larvae of rocky intertidal and nearshore fishes may resist offshore transport, and possibly even alongshore transport, to remain concentrated against the shore. Larvae of this species occurred more often in the southern portion of the study grid because more of the area in the southern half of the grid is within the 0–20 m depth range.

Standard lengths of all measured snubnose sculpin larvae collected at the DCPP intake structure between October 1996 and June 1999 (3,750 larvae) ranged from 1.7 mm to 8.5 mm (**Figure 5.2.6-10**). The central 98% of this length-frequency distribution resulted in minimum and maximum lengths used for analyses of 2.6 mm and 6.1 mm, respectively. The mean larval length in this distribution was 3.6 mm. Reported hatching size for this species ranged from 2.6–3.8 mm (Moser 1996). The fact that we observed some larvae smaller than the reported hatching lengths can be explained partly by natural variation of hatch lengths within the population and partly by the phenomenon of shrinkage following preservation (Theilacker 1980). The growth rate applied to this species (0.08 mm/d) from Freeman et al. (1985) was used to estimate ages of entrained larvae. Assuming that the shortest larvae (either 1.7 or 2.6 mm) were immediately post-hatch and aged zero days, then the estimated ages of larvae entrained could range from zero days up to 42–82 d post-hatching for the largest larvae measured. The average estimated ages of snubnose larvae entrained at DCPP ranged from 11–22 d post-hatching.

Distributions of snubnose sculpin larval lengths at the DCPP intake structure and in the study grid were compared using a Kolmogorov-Smirnov test (**Figure 5.2.6-11**). The test did not detect a significant difference between the two distributions ($p > 0.14$). Larvae in the study grid showed a bimodal size distribution, with a greater proportion of larger individuals represented in the grid relative to collections at the intake structure. A *t*-test detected a significant difference ($p \leq 0.0134$) between mean lengths for the two locations: 3.5 mm at the DCPP intake and 3.7 mm in the study grid. The large sample size (380 larvae from the intake and 161 larvae from the study grid) causes the *t*-test to be highly sensitive to small differences between the means (Zar 1984). While the difference between the means was statistically significant, we believe that the biological significance of the difference is low.

Estimating Total Annual Entrainment

The annual estimated mean entrainment of snubnose sculpin larvae ranged from a minimum of 59,800,000 ($\widehat{SE}$ =4,120,000) in 1997–98 Analysis Period 2 to a maximum of 85,600,000 ($\widehat{SE}$ =3,490,000) in 1996–97 Analysis Period 1 (**Table 5.2.6-5**). The estimates for 1997–98 Analysis Period 3 were slightly different than Analysis Period 2 because snubnose sculpin larvae were collected during the summer and fall months of 1997 when the two periods did not overlap. Values of $2 \cdot \widehat{SE}(\widehat{E_T})$ can be used to approximate 95% confidence intervals around the point estimates. These confidence intervals for October 1996–September 1997 and October 1997–September 1998 would not overlap, indicating that the differences between them were probably statistically significant.

The adjusted estimates of annual entrainment of snubnose sculpin larvae ranged from 83,500,000 ($\widehat{SE}$ =5,750,000) in 1997–98 Analysis Period 2 to 110,000,000 ($\widehat{SE}$ =4,480,000) in 1996–97 Analysis Period 1 (**Table 5.2.6-6**) based on annual indices using weekly Intake Cove surface plankton tows (**Figure 5.2.6-12**). The average index values for the years 1997 and 1998 are 0.028 and 0.025 larvae/m^3, respectively, while the long-term average index is 0.036 larvae/m^3 for the years 1990–1998, yielding the ratio $\overline{I} / I_i$ of 1.28 for 1997 and 1.40 for 1998. The adjustments

increased the estimates for all three periods but did not decrease the relative differences among them.

NOTE: Estimates of *FH* or *AEL* for snubnose sculpin using annual entrainment were not calculated because no demographic data exists for this species.

Empirical Transport Model (ETM)

Monthly estimates of proportional entrainment (PE_i) in the surveys from the two analysis periods ranged from a minimum of zero to a maximum of 0.859±0.911 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in December 1997 (**Table 5.2.6-7**). However, this maximum value was accompanied by a relatively low estimate of the annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) indicating that relatively few snubnose sculpin larvae were collected at the intake during this month. When both $\widehat{PE_i}$ and $\hat{f}_i$ were equal to zero, no larvae were collected at either the DCPP intake or from the study grid. When $\widehat{PE_i} = 0$ but $\hat{f}_i > 0$, larvae were collected at the DCPP intake during the monthly survey period but not during the entrainment survey paired with the 72-hour study grid survey. The largest values of $\hat{f}_i$ occurred during April through July in both analysis periods. During the 1996–1997 analysis period, the largest values of $\widehat{PE_i}$ occurred during periods when the $\hat{f}_i$ were lowest, indicating that the high $\widehat{PE_i}$ values resulted from low entrainment densities used to estimate abundance in nearshore areas of the study grid. During the 1997–1998 analysis period these surveys were also associated with the largest values of $\widehat{PE_i}$.

Results of *ETM* for snubnose sculpin showed how differences in estimated larval durations (11 vs. 42 d) can affect estimation of $\widehat{P_M}$ (**Figures 5.2.6-13**). Study grid samples showed that snubnose sculpin larvae were primarily distributed in nearshore areas (**Figure 5.2.6-9**) despite the fact that adults have been found from the intertidal zone out to depths of 30 m (Miller and Lea 1972). Based on observed larval and reported adult distributions, P_S was calculated using both alongshore current movement and extrapolating study grid abundance offshore using onshore current movement before extending alongshore. The probability of entrainment was typically less for the shorter duration within an extrapolation class because of the reduced time period over which larval transport occurred. Estimates of $\widehat{P_M}$ using alongshore transport were larger for the

1998–1999 analysis period (0.24: **Figure 5.2.6-13b)** than for the 1997–1998 analysis period (0.14) and were greater than within-period differences between estimates based on different larval durations. Estimates of $\widehat{P_M}$ using offshore abundance extrapolation and maximum transport duration were very similar between periods (i.e., 0.021 for 1998–1999 and 0.023 for 1997–1998). The increased onshore transport caused by the El Niño conditions during the 1997–1998 period reduced the differences in the estimates by compensating for the large differences in alongshore transport between the two periods.

Interpretation of Assessment Results

Snubnose sculpin has neither commercial nor recreational fishery value and there is little information on its ecological role in the community. Because of the absence of any fishery for this species, no catch data can be used to compare harvest mortality rates to entrainment mortality rates ($\widehat{P_M}$). There also are no stocks or adult density estimates that could be used to convert $\widehat{P_M}$ into equivalent adult losses (assuming no compensatory mortality).

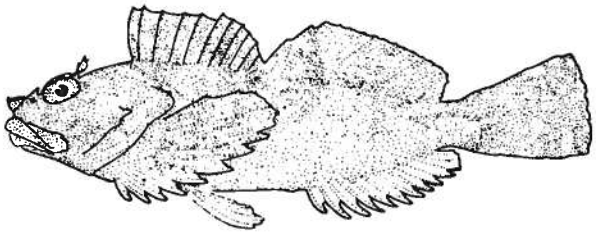
Annual mean density of snubnose sculpin larvae from Intake Cove surface plankton tows shows a declining trend in abundance over time, although the estimates for 1997 and 1998 appear to have leveled off (**Figure 5.2.6-12**). Reduced larval production may affect recruitment and eventually adult population densities. Reductions in adult density due to entrainment of snubnose sculpin larvae could be expected to appear only after several generations. There were no estimates in the scientific literature of the life-span or age at sexual maturity for snubnose sculpin, though they are likely to be similar to other small, nearshore cottids: 1–3 yr for sexual maturity and 2–7 yr for longevity (Gibson 1969; Miller 1979; deVlaming et al. 1982; Grossman and deVlaming 1984; Freeman et al. 1985; Wells 1986; Pierce and Pierson 1990).

Adult abundance of snubnose sculpin observed in the DCPP Receiving Water Monitoring Program (RWMP) studies on subtidal fishes varies considerably through time, but shows a downward trend from the early 1990s through 1998 (**Figure 5.2.6-14**). These data were collected along three 50 m transects in an area approximately 1 km south of Diablo Cove not contacted by the plant's thermal discharge. Since the potential life-span and age at sexual maturity of snubnose

sculpin are less than the time since power plant operation began, there has been sufficient time for losses in the adult population to occur as a result of a decrease in larvae. Thus, it appears that larval entrainment mortality of snubnose sculpin may be a factor in the reduction of adult abundance in the vicinity of DCPP.

Snubnose sculpin have a smaller geographical distribution (Farallon Islands, northern California to Isla San Geronimo, northern Baja California) than other central California sculpins (Eschmeyer et al. 1983; Love et al. 1996). Although the results of the *ETM* evaluated using extrapolated abundance implying a large population area, the more restricted distribution of this taxon indicates that an estimate of P_M closer to the values estimated using alongshore transport alone may be more appropriate. These estimates range from approximately 10% to over 20%, depending upon larval duration and sampling period (**Figures 5.2.6-13**). Based on Moser's (1996) reported size at flexion of 4.2–4.7 mm, the length range of larvae entrained shows that post-flexion larvae are still subject to entrainment (**Figure 5.2.6-10**). This also indicates that snubnose sculpin larvae have a prolonged exposure to entrainment. The prolonged exposure period and the levels of entrainment mortality estimated by P_M may partially explain other results that show long-term declines in larval abundance in intake samples (**Figure 5.2.6-12**) and declines in local adult abundance (**Figure 5.2.6-14**).

5.2.6.3 Cabezon (*Scorpaenichthys marmoratus*)



Scorpaenichthys marmoratus (Ayres 1854); cabezon; length to 99 cm; Sitka, Alaska to Punta Abreojos, central Baja California; intertidal to approx. 85 m; brown, reddish, or greenish above whitish or greenish below (Miller and Lea 1972; Eschmeyer et al. 1983).

The cabezon (*Scorpaenichthys marmoratus*) is the largest North American species of marine cottid and occurs over the nearshore continental shelf from depths of 85 m up to the intertidal zone (O'Connell 1953; Matarese et al. 1989). Cabezon are a popular sport fish and are also landed commercially (Fitch and Lavenberg 1971; Lamb and Edgell 1986). Females are oviparous and lay demersal, adhesive eggs in rocky crevices or on algae; males guard the egg nest until the pelagic larvae hatch (Burge and Schultz 1973; Feder et al. 1974; Matarese et al. 1989). Moser (1996) indicates that cabezon larvae hatch at 3–6 mm.

Larvae appear in the water column around November or December and recruit to tidepools at around 40 mm SL in March off Moss Beach, California (R.R. Harry unpubl. data cited in O'Connell 1953), implying a 3–4 mo planktonic duration. Females begin to mature in their third year between 25–48 cm SL (Fitch and Lavenberg 1971), and all are mature by year five (Starr et al. 1998). Fecundity for this species has been reported in several sources: 45,000 eggs for a 43 cm SL specimen and 95,000 eggs for a 65 cm SL specimen (Hart 1973); mean fecundity of 48,700 eggs for a 1.4 kg female and 97,600 eggs for a 4.6 kg female (O'Connell 1953; Bane and Bane 1971); and up to 152,000 eggs from a 76 cm SL female (Starr et al. 1998). O'Connell (1953) states that females spawn more than a single batch of eggs per year. Females live to 13 yr and males to 9 yr (O'Connell 1953).

As with other members of the Cottidae, there are insufficient data to parameterize the *FH* and *AEL* approaches for impact assessment. No independent estimates of survivorship for early life stages prior to entrainment or later life stages are available from the literature. Thus, the impact of entrainment on this species' population will be assessed using only the *ETM*.

Summary of Field Collections

Estimates of cabezon larval abundance showed distinct seasonal peaks at the DCPP intake structure during the years 1996–1999 (**Appendix H**) with an apparent trend of decreasing abundance at the intake over the same period (**Figure 5.2.6-15**). There were 1,938 larval cabezon identified from 575 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 12% of the entrainment samples collected and processed during that period. The spawning period for cabezon in winter 1996–1997 was longer and had higher abundances than the winter 1997–1998 spawning period. This difference may have been related to the El Niño event that occurred during late 1997 (Lynn et al. 1998; NOAA 1999) as substantial increases in water temperatures and lowered productivity negatively affected larval production and survival. The limited samples available from the winter 1998–1999 spawning period appear to show similar timing as previous years.

The seasonality of larval cabezon in the DCPP study grid was similar to the timing observed from the samples at the DCPP intake structure (**Figure 5.2.6-16**). There were 887 cabezon larvae identified from 326 bongo net subsamples (**Appendix H**) representing 11% of the study grid samples collected and processed from July 1997–June 1999. The spatial distribution of larval cabezon abundance in the study grid during 1997–1998 Analysis Period 3 appeared primarily constrained against the shoreward side of the study grid. During 1998–1999 Analysis Period 4, cabezon larvae appeared to be more widely distributed in the study grid. The stronger alongshore currents during the second analysis period may account for this difference. Larval dispersal may have been more limited during the El Niño conditions present during the 1997–1998 analysis period due to weaker alongshore and stronger onshore currents.

Standard lengths of all cabezon larvae collected at the DCPP intake structure between October 1996 and June 1999 and measured (1,537 larvae) ranged from a minimum of 2.3 mm to a maximum of 8.4 mm (**Figure 5.2.6-17**). The central 98% of this length-frequency distribution resulted in minimum and maximum lengths of 3.7 mm and 6.1 mm, respectively, that were used for calculating larval duration. The mean length of this distribution was 4.8 mm. Reported hatching size for this species ranged from 3–6 mm (Moser 1996). The fact that we observed some larvae smaller than the reported hatching lengths can be explained partly by natural variation of hatch lengths within the population and partly by the phenomenon of shrinkage following preservation (Theilacker 1980). The growth rate applied to this species (0.3 mm/d) was derived from O'Connell (1953) and Moser (1996) and was used to estimate ages of entrained larvae. Assuming that the shortest larvae (either 2.3 or 3.7 mm) were immediately post-hatch and aged zero days, the estimated ages of larvae entrained could range from zero days up to 8–20 d post-hatching for the largest larvae measured. The average estimated ages of cabezon larvae entrained at DCPP ranged from 4–8 d post-hatching.

Distributions of cabezon larval lengths at the DCPP intake structure and in the study grid were compared using a Kolmogorov-Smirnov test (**Figure 5.2.6-18**). The test detected a significant difference between the two distributions ($p \leq 0.001$). Larvae in the study grid had a greater proportion of smaller individuals relative to collections at the intake structure. A *t*-test also detected a significant difference ($p \leq 0.002$) between the mean lengths for the two locations: 4.7 mm at the DCPP intake and 4.8 mm in the study grid. While these differences were statistically significant, we believe that the biological significance of these differences was low.

Estimating Total Annual Entrainment

The annual estimated mean entrainment of cabezon larvae ranged from 14,700,000 ($\widehat{SE}$ =1,460,000) in 1997–98 Analysis Periods 2 and 3 to 35,700,000 ($\widehat{SE}$ =2,720,000) in 1996–97 Analysis Period 1 (**Table 5.2.6-8**). Values of $2 \cdot \widehat{SE}(\widehat{E_T})$ can be used to approximate 95% confidence intervals around the point estimates. The confidence intervals for the 1996–97 and 1997–98 periods would not overlap, indicating that the differences between them were probably

statistically significant. The estimates for Analysis Period 3 are the same as Analysis Period 2 because all of the cabezon were collected during the October–June period when the two periods overlapped (**Figure 5.2.6-15**).

Annual estimates of entrained cabezon larvae, adjusted to a long-term mean based on annual indices using weekly Intake Cove surface plankton tows (**Figure 5.2.6-19**) ranged from 36,300,000 ($\widehat{SE}$ =3,600,000) in Analysis Periods 2 and 3 to 51,900,000 ($\widehat{SE}$ =3,950,000) in Analysis Period 1 (**Table 5.2.6-9**). The average index for the years 1997 and 1998 are 0.0076 and 0.0045 larvae/m^3, respectively, while the long-term average index is 0.011 larvae/m^3 for the years 1990–1998, yielding the ratio $\overline{I}/I_i$ of 1.45 for 1997 and 2.47 for 1998. The adjustments increases the estimates of abundance and decreases the differences among them.

NOTE: Estimates of *FH* or *AEL* for cabezon using annual entrainment were not calculated because there were no demographic data available for this species.

Empirical Transport Model (ETM)

Monthly estimates of proportional entrainment (PE_i) in the surveys from the two analysis periods ranged from a minimum of zero to a maximum of 0.106±0.103 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in October 1997 (**Table 5.2.6-10**). However, this maximum $\widehat{PE_i}$ value was not accompanied by a maximal estimate of the annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) so it was proportionally less important to the estimate of annual entrainment mortality than other values with higher $\hat{f}_i$'s. Periods when both $\widehat{PE_i}$ and $\hat{f}_i$ were equal to zero indicated that no larvae were collected at either the DCPP intake or from the study grid. When $\widehat{PE_i} = 0$ but $\hat{f}_i > 0$, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. The largest values of $\hat{f}_i$ occurred from December–February in both analysis periods indicating that these were times of high larval abundance at the intake structure.

Estimates of $\widehat{P_M}$ for either alongshore current transport or offshore abundance extrapolation during both analysis periods were relatively similar and ranged from 0.006 to 0.03 (**Figures**

5.2.6-20). Cabezon adults have been found to depths of 75 m (Miller and Lea 1972) and their larvae were distributed throughout the study grid (**Figure 5.2.6-16**). Therefore, P_S was calculated using both alongshore current movement and extrapolating study grid abundance offshore using onshore current movement before extending it alongshore. Larval durations calculated using the mean and maximum larval lengths (central 98% of length distribution) only differed by 4 d (4 vs. 8 d). This similarity in duration resulted in relatively small values of $\widehat{P_M}$ and rapid decline in their value with increasing P_S. Thus, it appears that cabezon larvae were entrainable over a relatively short period of their development.

Interpretation of Assessment Results

Cabezon has both commercial and recreational fishery value. Combined reported landings at the ports of San Luis and Morro Bay have varied during 1975–1998 from zero in 1975 to a high of 78 MT in 1997 with an average of 9.7 MT (**Figure 5.2.6-21**). From 1995–1998, the combined average catch increased dramatically to over 60 MT/yr as a result of increased fishing effort stimulated by a high demand for cabezon in the 'live-fish' fishery. Juvenile and adult cabezon abundance examined using data from the DCPP Receiving Water Monitoring Program (RWMP) studies on subtidal fishes appears to be a declining since 1993 (**Figure 5.2.6-21**). Mean abundance of juvenile and adult cabezon was estimated from three 50 m transects in an area approximately 1 km south of Diablo Cove and not contacted by the plant's thermal discharge. This decline likely reflects the increased 'live-fish' fishery activity targeting young, nearshore species. Increases in the activity of fisherman in the 'live-fish' fishery near DCPP has been noted since approximately 1995. There does not appear to be a similar long-term abundance decline for cabezon larvae from the Intake Cove surface plankton tows; mainly due to high abundance in 1995 (**Figure 5.2.6-19**). In addition, the narrow size range of larvae entrained by the power plant (central 98% of length-frequency distribution ranged from 3.7–6.1 mm) indicates that cabezon larvae are exposed to entrainment for a relatively short period of time (**Figure 5.2.6-17**). Additionally, most larvae entrained were shorter than the reported length at flexion of 7.0 mm (Moser 1996). While it remains unclear whether local cabezon populations are in decline after

examining these data, the size range of entrained larvae and relatively low estimates of $\widehat{P_M}$ can lead to the conclusion that larval entrainment mortality is not affecting local adult populations.

Empirical transport modeling results indicate that the power plant may annually entrain approximately 1.5–3.5% of the cabezon larvae in an area 2–3 times the size of the study grid (assuming alongshore transport only) but less than 1% of larvae in an area from 4–22 times the size of the study grid (assuming both alongshore and offshore transport). Using this range of values, an estimate of $\widehat{P_M} = 0.015$ or 1.5% was used to estimate entrainment effects on cabezon landings assuming no compensatory mortality. Based on landings and revenues of cabezon in the Morro Bay area from 1998 (75.2 MT and $592,300; PSMFC PacFIN database), the average dollar loss to the cabezon fishery (assuming a 1.5% proportional loss) due to larval entrainment at DCPP would be approximately $9,000/yr.

Table 5.2.6-1. Summary of known demography for four closely related cottid species; X indicates data available from published sources; * indicates length-frequency data to estimate total adult mortality is available from this study; — indicates no data available.

Species	S_{egg}	S_{larvae}	S_{adult}	Fecundity	Age @ maturity	Spawning	Longevity	Age & Growth	Stage Duration
A. lateralis	—	—	—	—	X	—	X	—	X[1]
O. maculosus	—	—	*	X	X	—	—	—	—
O. snyderi	—	—	*	—	—	X	X	X	—
C. analis	—	—	—	X	X	X	X	X	X[1,2]

[1] Egg stage duration
[2] Planktonic larval duration

Table 5.2.6-2. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for smoothhead sculpin (*Artedius lateralis*) larvae from the three analysis periods.

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Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	88,900,000	3,660,000
2) Oct 1, 1997–Sep 30, 1998	96,100,000	9,410,000
3) Jul 1, 1997–Jun 30, 1998	107,000,000	9,490,000

Table 5.2.6-3. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for smoothhead sculpin (*Artedius lateralis*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	57,700,000	2,370,000
2) Oct 1, 1997–Sep 30, 1998	115,000,000	11,300,000
3) Jul 1, 1997–Jun 30, 1998	129,000,000	11,400,000

Table 5.2.6-4. Smoothhead sculpin (*Artedius lateralis*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.00600	0.00248	0.112	0.00534
Aug 25, 1997	0.0236	0.00797	0.0175	0.00111
Sep 29, 1997	0	0	0.00235	0.000440
Oct 20, 1997	0	0	0.00016	0.0000900
Nov 17, 1997	0	0	0.00022	0.000110
Dec 10, 1997	0	0	0	0
Jan 22, 1998	0	0	0.00251	0.000440
Feb 26, 1998	0	0	0.00173	0.000240
Mar 18, 1998	0.0166	0.00453	0.0317	0.00165
Apr 15, 1998	0.0431	0.0115	0.453	0.0163
May 18, 1998	0.0354	0.00750	0.161	0.00646
Jun 8, 1998	0.0380	0.00685	0.219	0.00861

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.00466	0.00229	0.0301	0.00155
Aug 26, 1998	0.00583	0.00619	0.00198	0.000360
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0	0	0	0
Jan 12, 1999	0.0194	0.0104	0.0157	0.00133
Feb 3, 1999	0.0284	0.010	0.0511	0.00214
Mar 17, 1999	0.0165	0.00483	0.0915	0.00351
Apr 14, 1999	0.0716	0.0222	0.300	0.00759
May 24, 1999	0.0440	0.00973	0.329	0.00788
Jun 23, 1999	0.0729	0.0181	0.180	0.00471

Table 5.2.6-5. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for snubnose sculpin (*Orthonopias triacis*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	85,600,000	3,490,000
2) Oct 1, 1997–Sep 30, 1998	59,800,000	4,120,000
3) Jul 1, 1997–Jun 30, 1998	75,500,000	4,094,000

Table 5.2.6-6. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for snubnose sculpin (*Orthonopias triacis*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	110,000,000	4,480,000
2) Oct 1, 1997–Sep 30, 1998	83,500,000	5,750,000
3) Jul 1, 1997–Jun 30, 1998	105,000,000	5,720,000

Table 5.2.6-7. Snubnose sculpin (*Orthonopias triacis*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.0139	0.00482	0.246	0.00837
Aug 25, 1997	0.0182	0.00671	0.0589	0.00296
Sep 29, 1997	0.859	0.288	0.0322	0.00173
Oct 20, 1997	0.859	0.527	0.00588	0.000700
Nov 17, 1997	0.00474	0.00625	0.00343	0.000510
Dec 10, 1997	0.859	0.911	0.00303	0.000510
Jan 22, 1998	0.0143	0.00760	0.0133	0.00108
Feb 26, 1998	0	0	0.00472	0.000500
Mar 18, 1998	0	0	0.00648	0.000810
Apr 15, 1998	0.0556	0.0201	0.216	0.0106
May 18, 1998	0.0323	0.00894	0.117	0.00480
Jun 8, 1998	0.0467	0.0132	0.293	0.00877

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.0129	0.00582	0.148	0.0112
Aug 26, 1998	0.00885	0.00783	0.00908	0.00109
Sep 16, 1998	0.00445	0.00293	0.00420	0.000600
Oct 6, 1998	0.0199	0.0103	0.0148	0.00121
Nov 11, 1998	0.0292	0.0270	0.0160	0.00175
Dec 9, 1998	0.00470	0.00509	0.00375	0.000680
Jan 12, 1999	0.0726	0.0407	0.0257	0.00218
Feb 3, 1999	0.0532	0.0255	0.0439	0.00297
Mar 17, 1999	0.123	0.0414	0.168	0.00570
Apr 14, 1999	0.104	0.0420	0.251	0.00736
May 24, 1999	0.0357	0.00896	0.143	0.00448
Jun 23, 1999	0.150	0.0527	0.173	0.00745

Table 5.2.6-8. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for cabezon (*Scorpaenichthys marmoratus*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	35,700,000	2,720,000
2) Oct 1, 1997–Sep 30, 1998	14,700,000	1,460,000
3) Jul 1, 1997–Jun 30, 1998	14,700,000	1,460,000

Table 5.2.6-9. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for cabezon (*Scorpaenichthys marmoratus*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	51,900,000	3,950,000
2) Oct 1, 1997–Sep 30, 1998	36,300,000	3,600,000
3) Jul 1, 1997–Jun 30, 1998	36,300,000	3,600,000

Table 5.2.6-9. Cabezon (*Scorpaenichthys marmoratus*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0	0	0	0
Aug 25, 1997	0	0	0	0
Sep 29, 1997	0	0	0	0
Oct 20, 1997	0.106	0.103	0.0935	0.00830
Nov 17, 1997	0.00608	0.00340	0.0596	0.00605
Dec 10, 1997	0.0140	0.00395	0.289	0.0139
Jan 22, 1998	0.00317	0.00244	0.428	0.0185
Feb 26, 1998	0	0	0.108	0.00854
Mar 18, 1998	0	0	0.00924	0.00197
Apr 15, 1998	0	0	0.0107	0.00219
May 18, 1998	0	0	0.00280	0.00115
Jun 8, 1998	0	0	0	0

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0	0	0	0
Aug 26, 1998	0	0	0	0
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0.00140	0.00072	0.104	0.00895
Jan 12, 1999	0.00630	0.00209	0.587	0.0175
Feb 3, 1999	0.00359	0.00127	0.177	0.0112
Mar 17, 1999	0.00380	0.00181	0.103	0.00768
Apr 14, 1999	0.00679	0.00774	0.028	0.00486
May 24, 1999	0	0	0	0
Jun 23, 1999	0	0	0	0

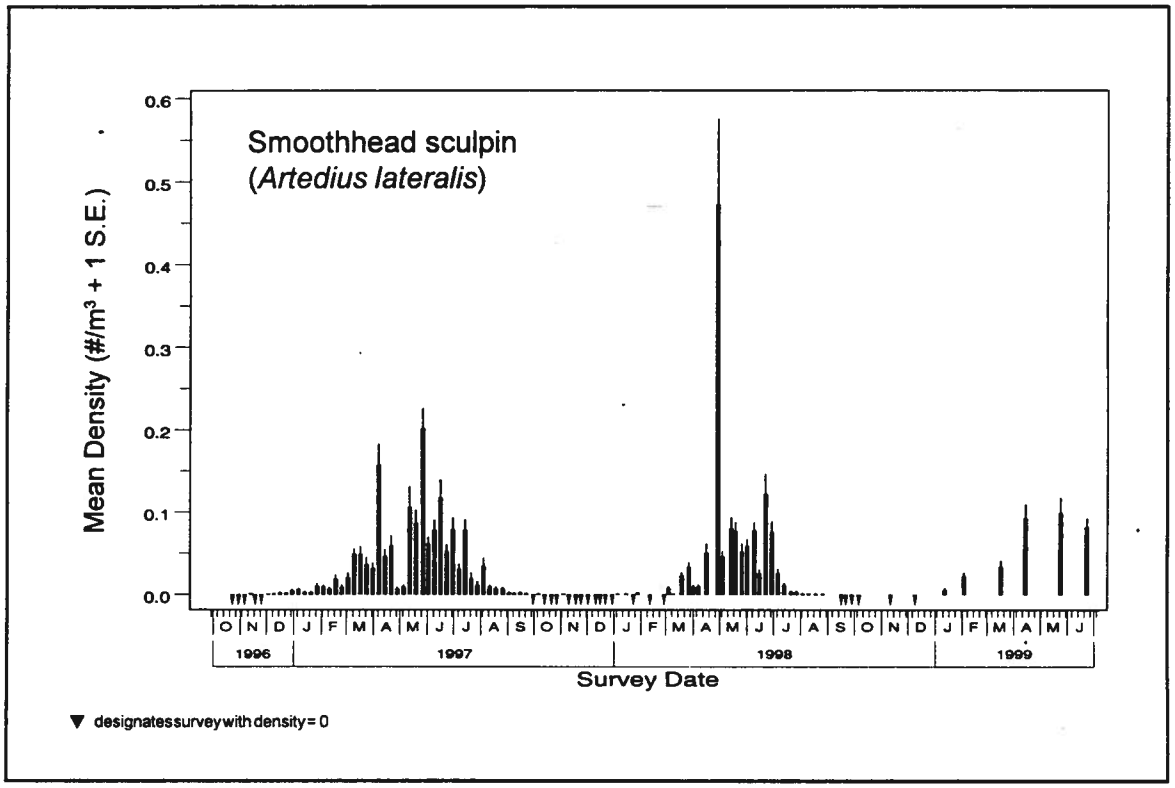


Figure 5.2.6-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

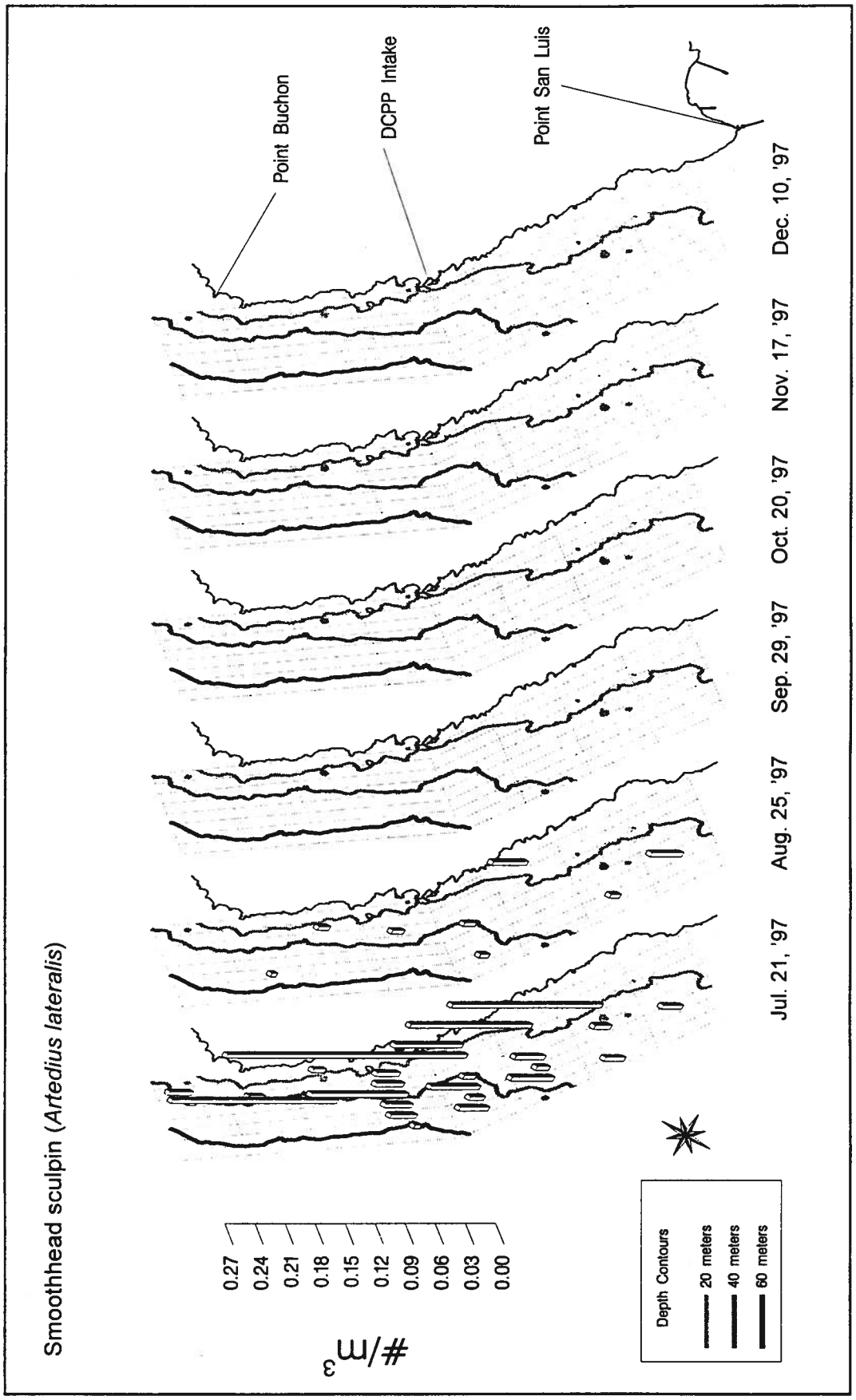


Figure 5.2.6-2a. Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

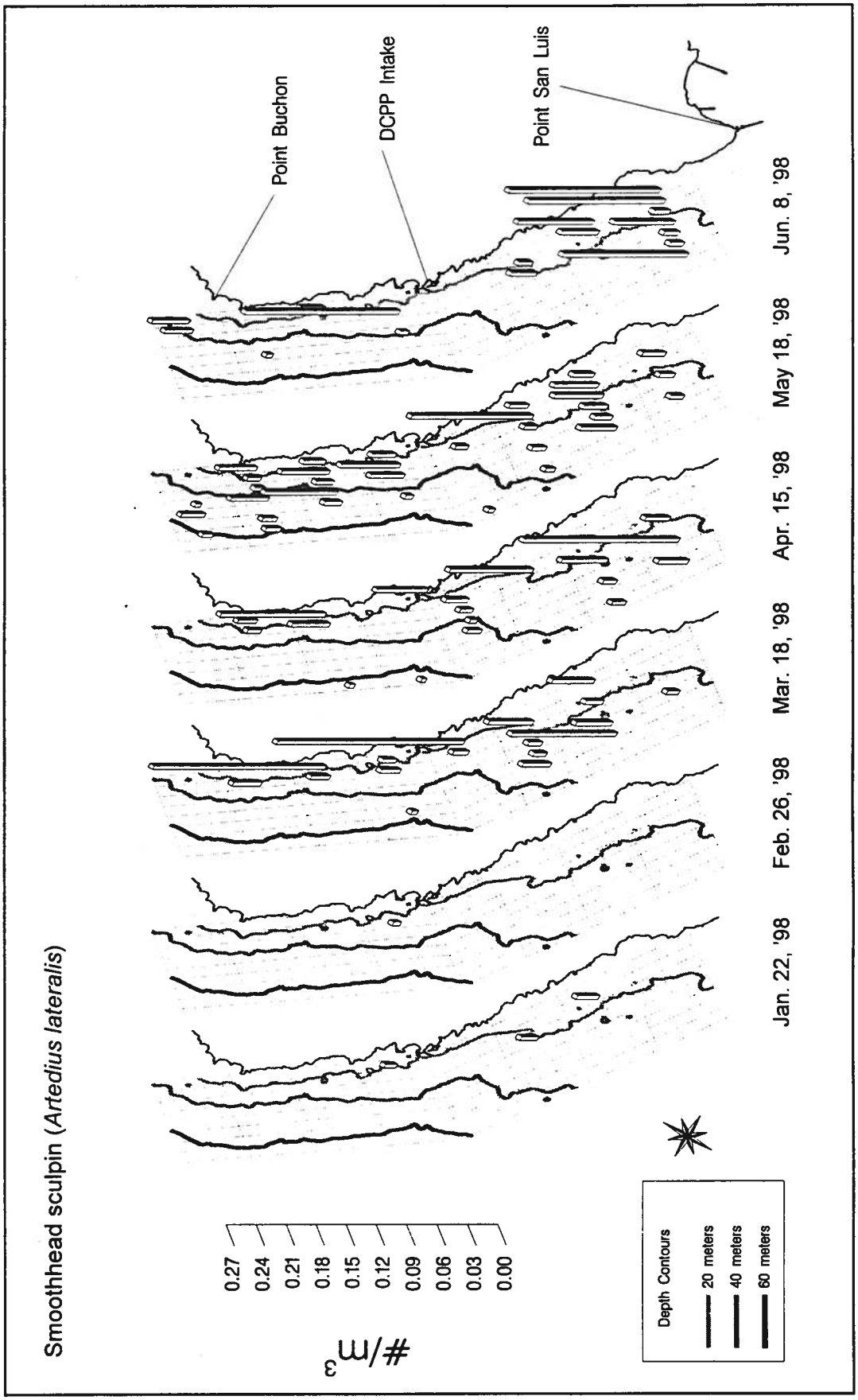


Figure 5.2.6-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

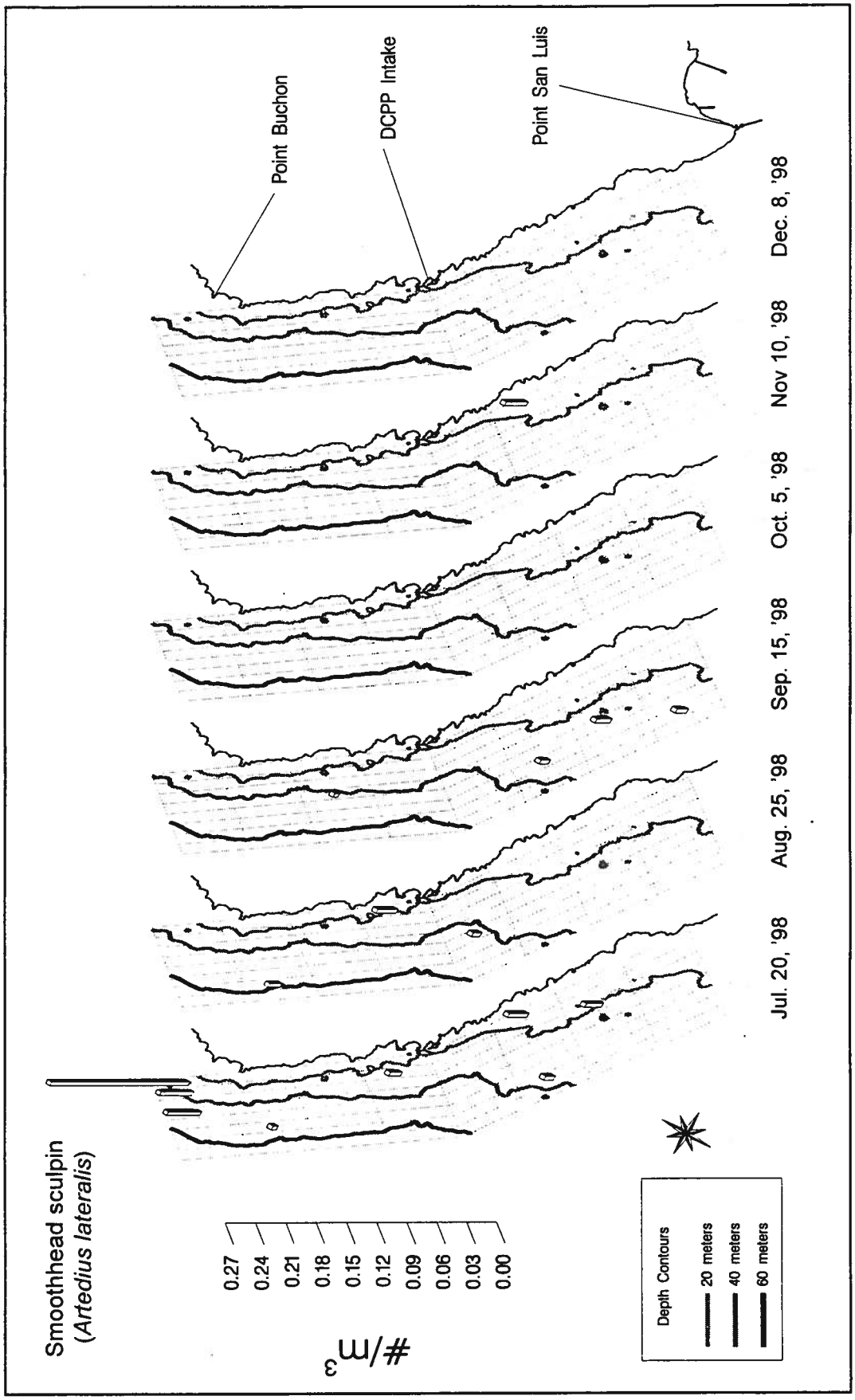


Figure 5.2.6-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

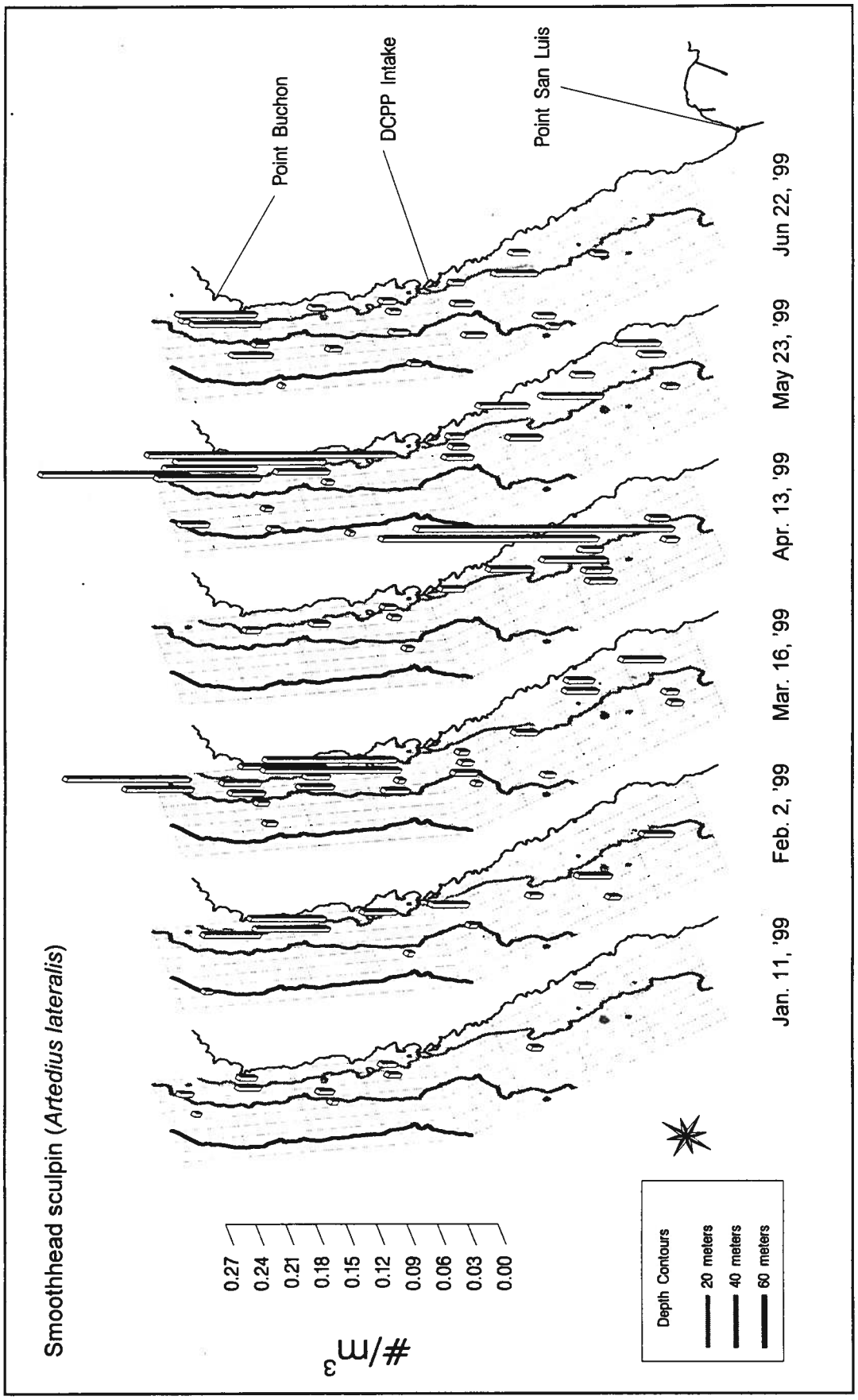


Figure 5.2.6-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

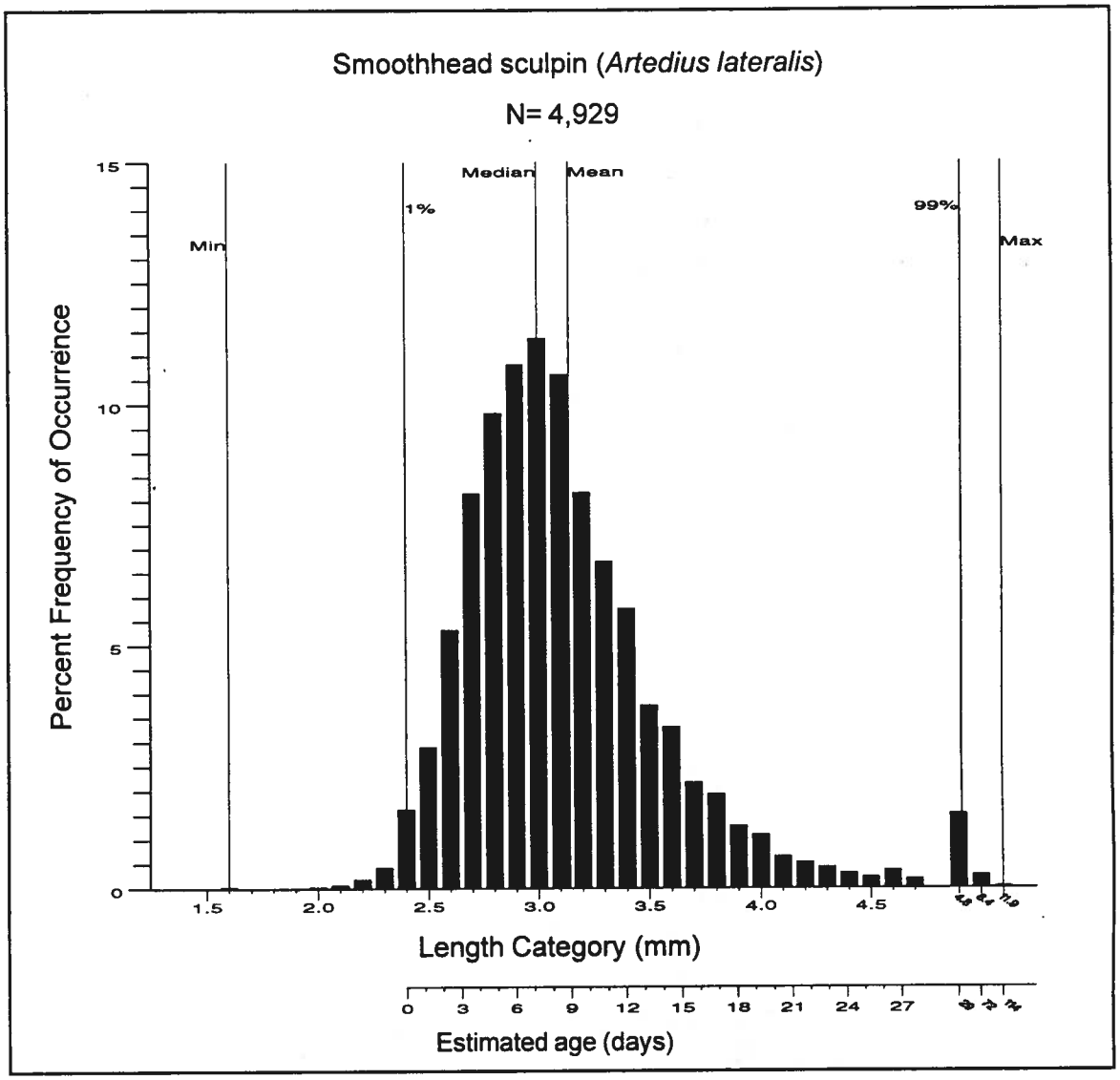


Figure 5.2.6-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

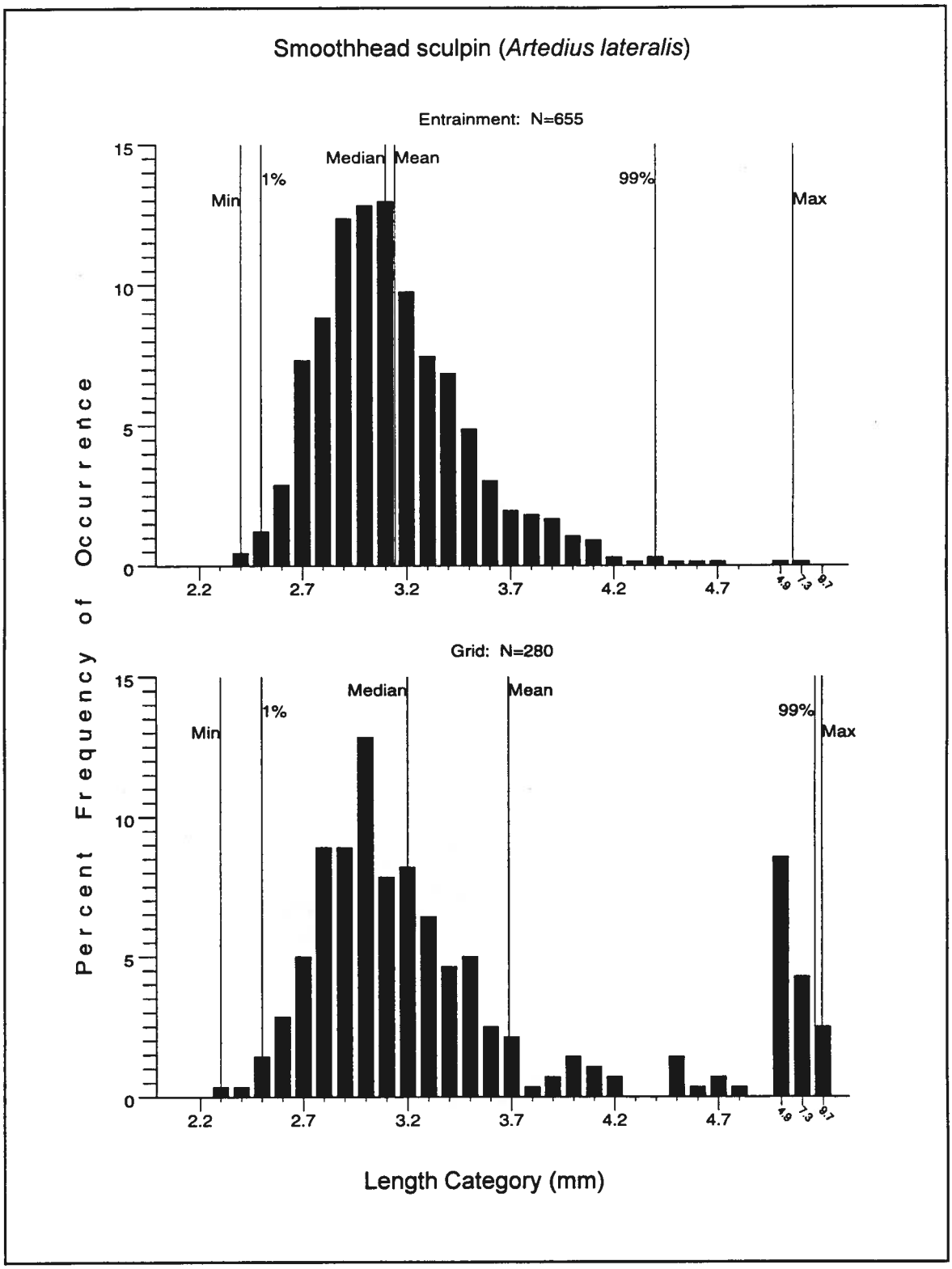


Figure 5.2.6-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

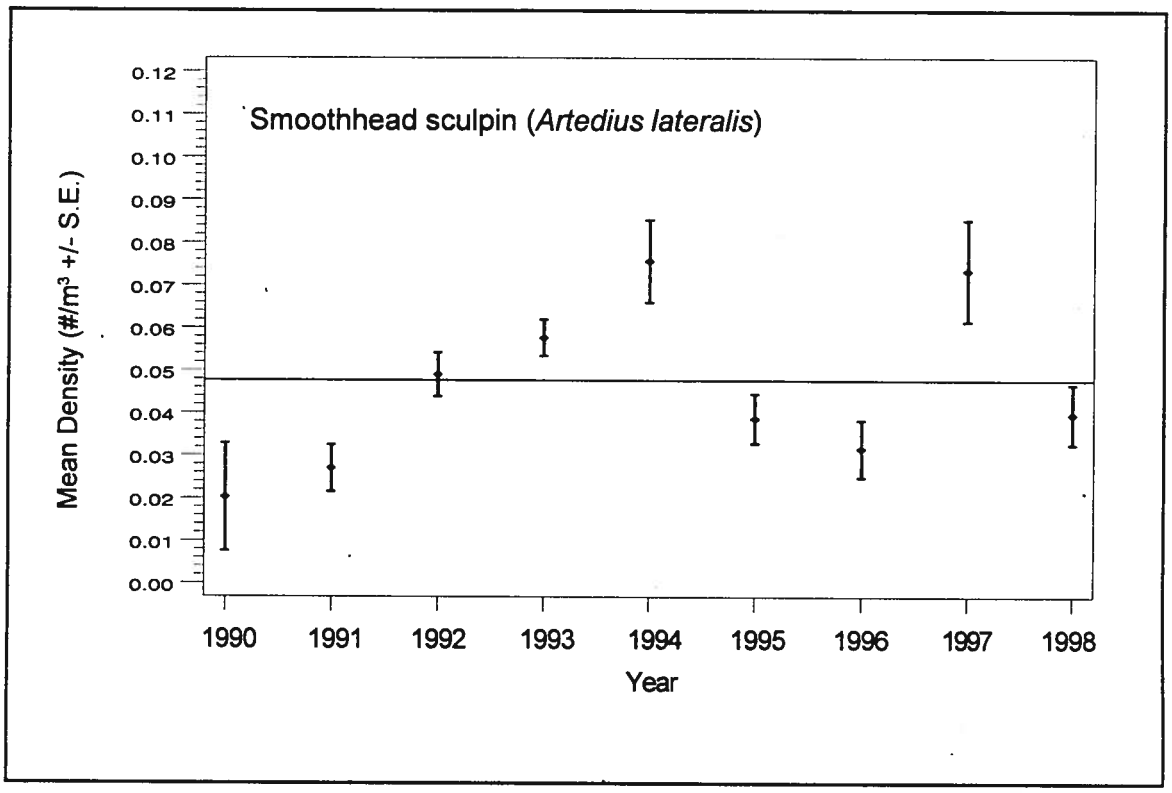


Figure 5.2.6-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

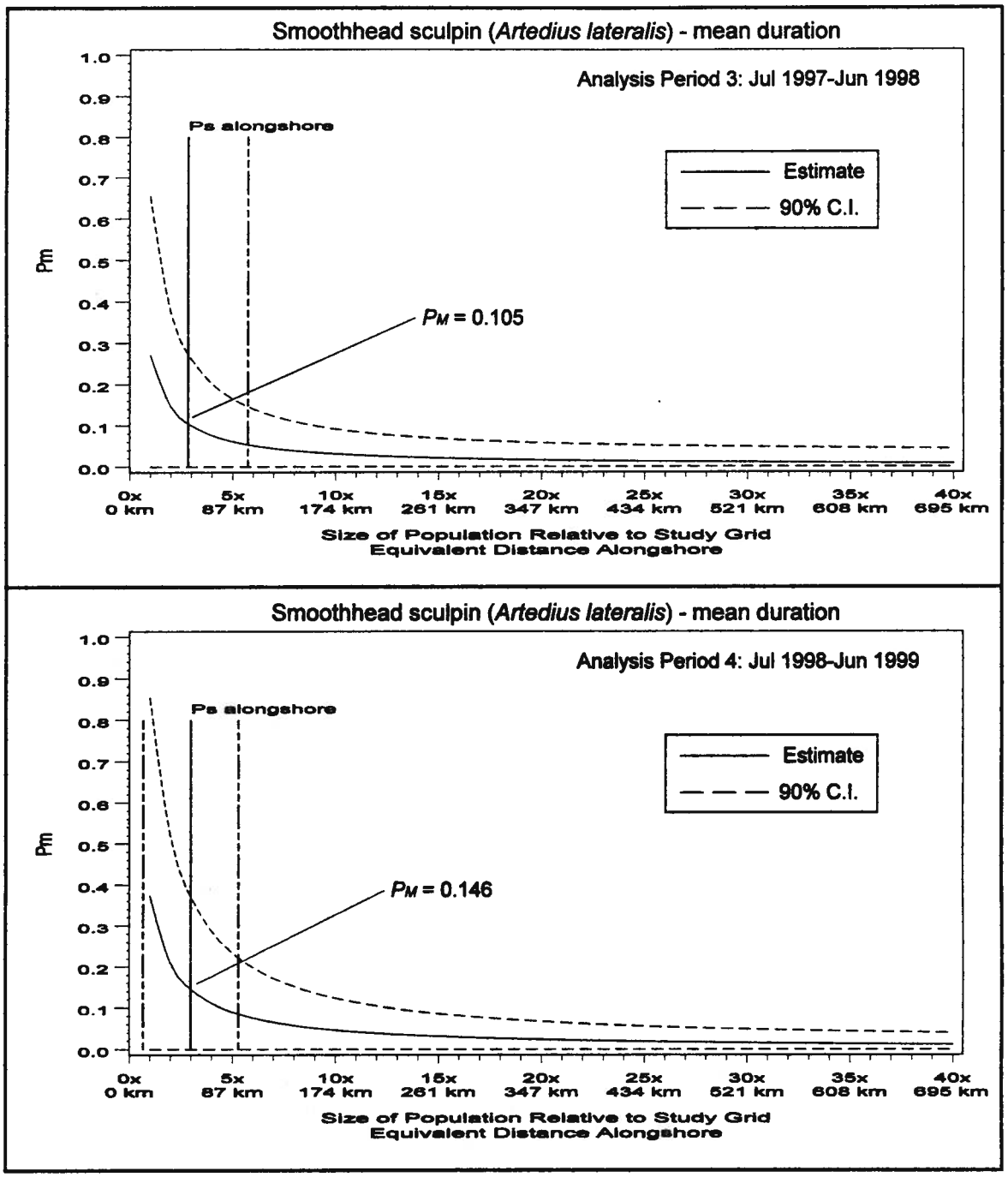


Figure 5.2.6-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements with the 90% C.I. indicated.

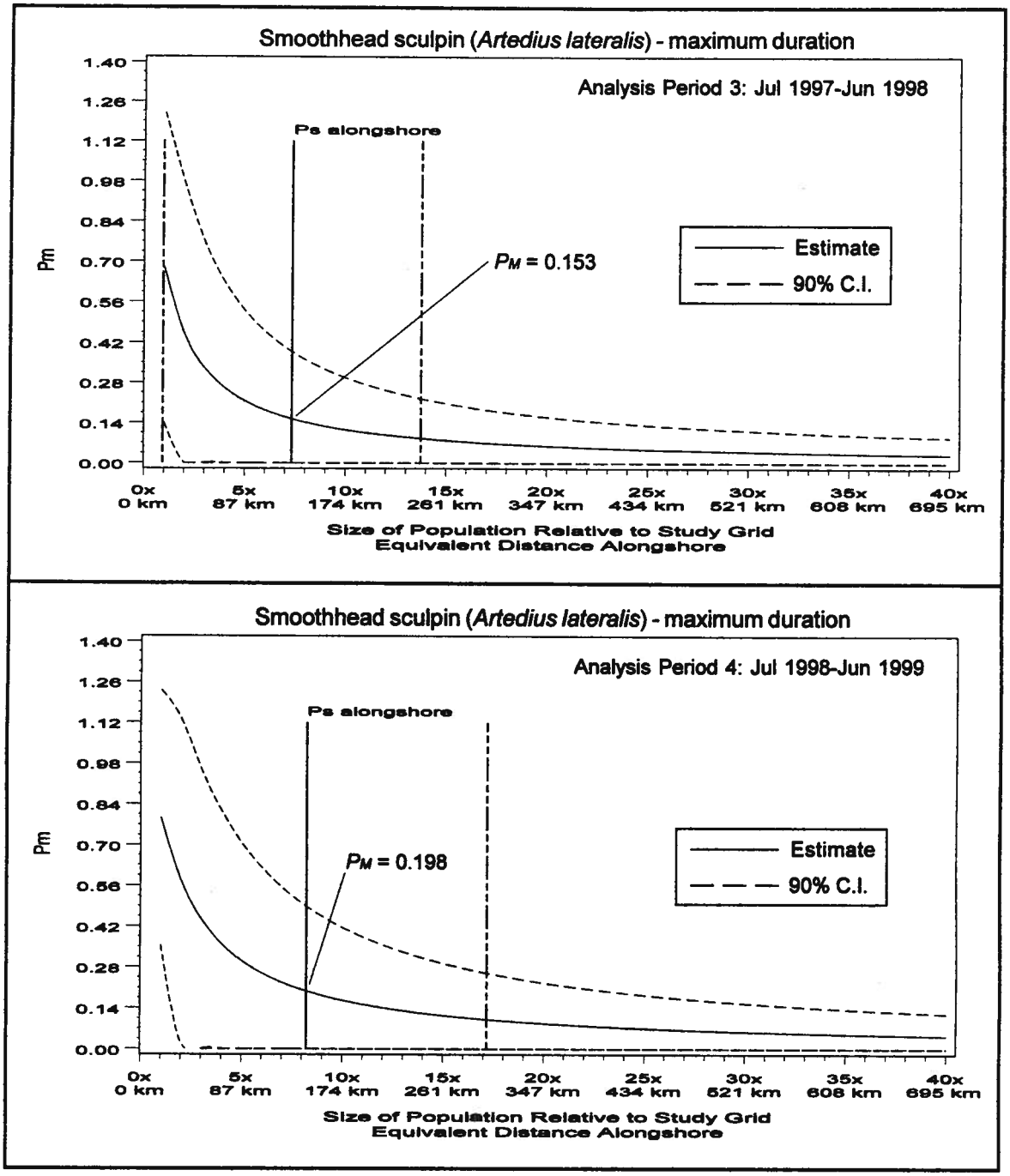


Figure 5.2.6-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements with the 90% C.I. indicated.

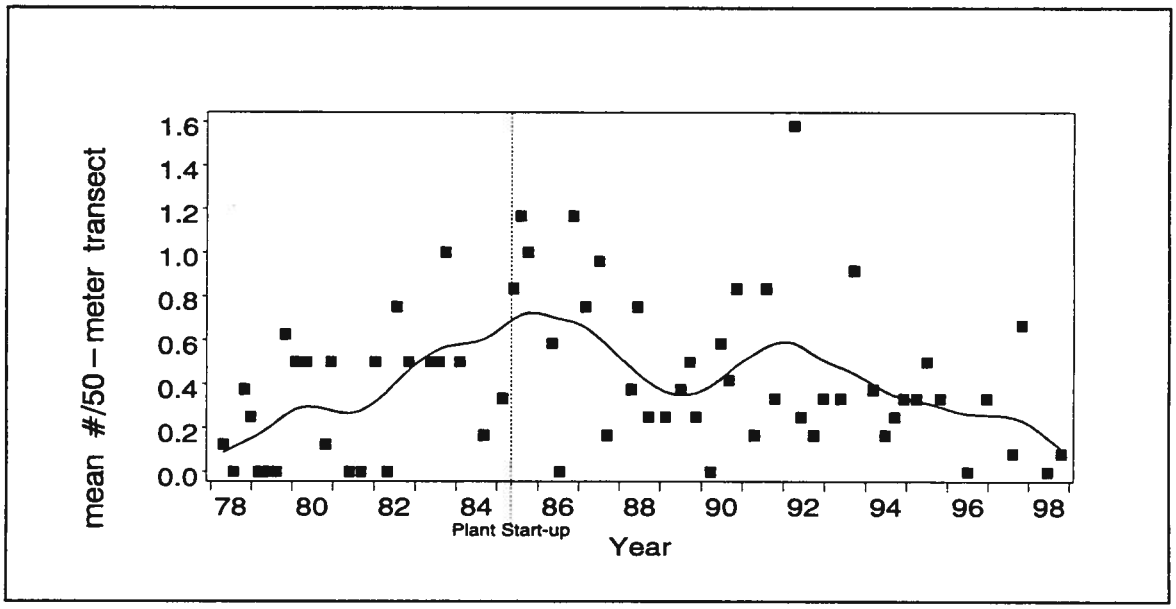


Figure 5.2.6-7. Mean number of sculpins (Cottidae) observed per 50-meter subtidal transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to fit curve through points.

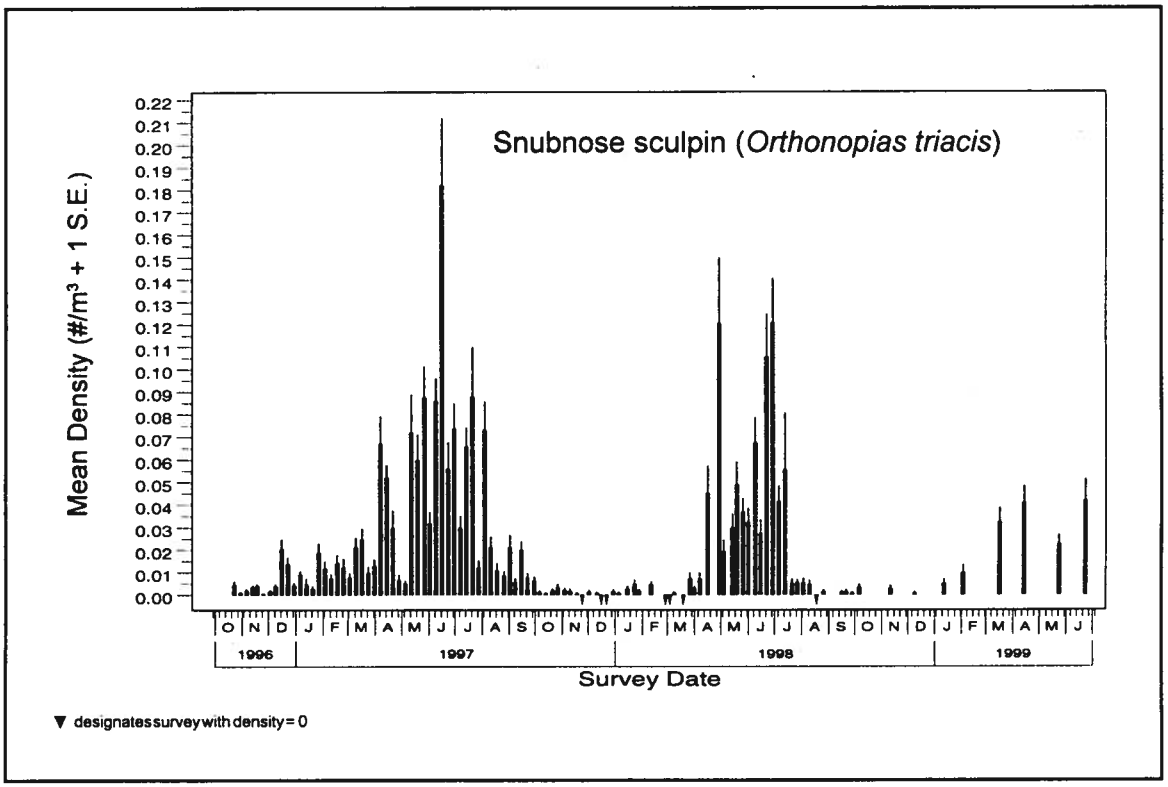


Figure 5.2.6-8. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

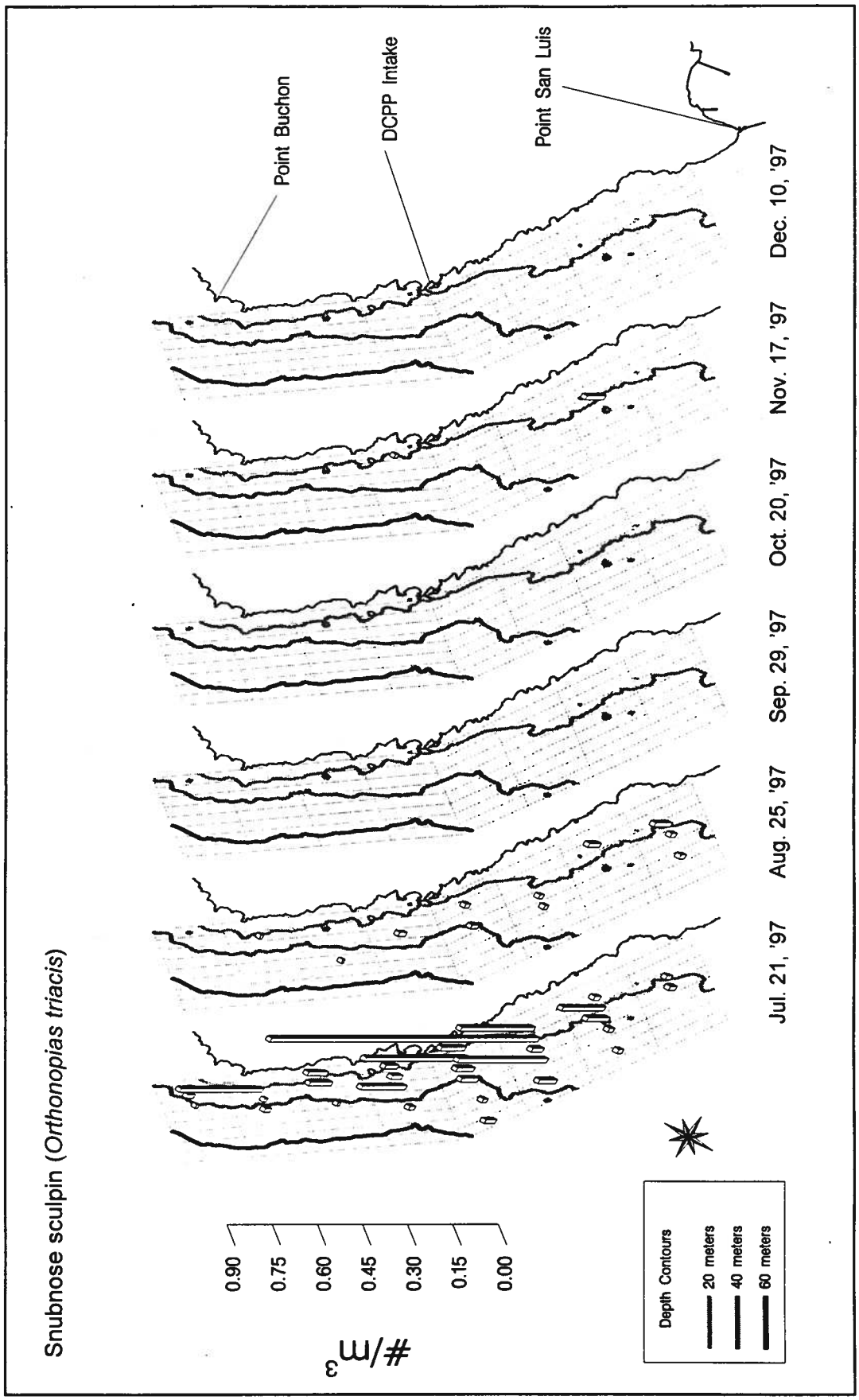


Figure 5.2.6-9a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

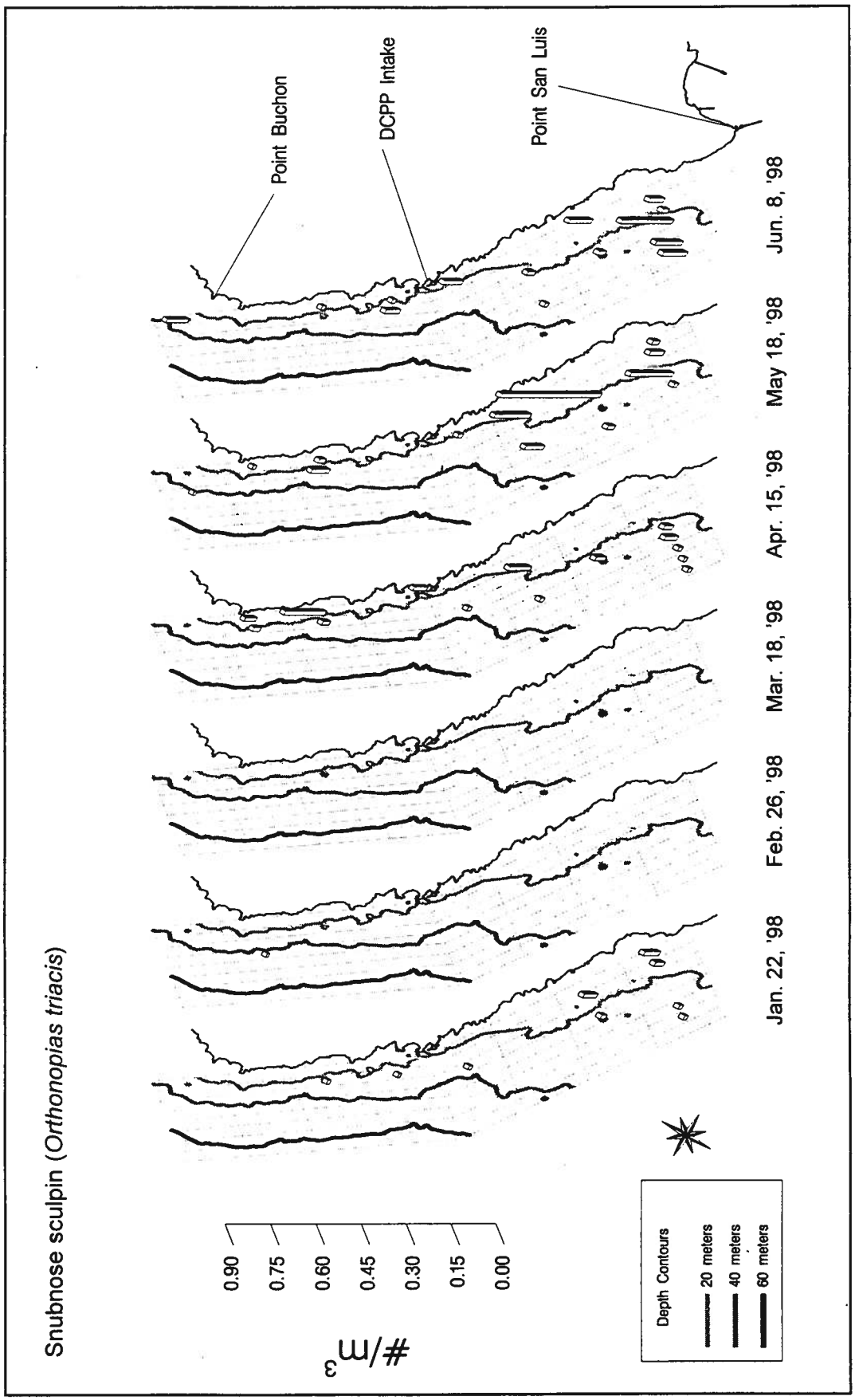


Figure 5.2.6-9b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

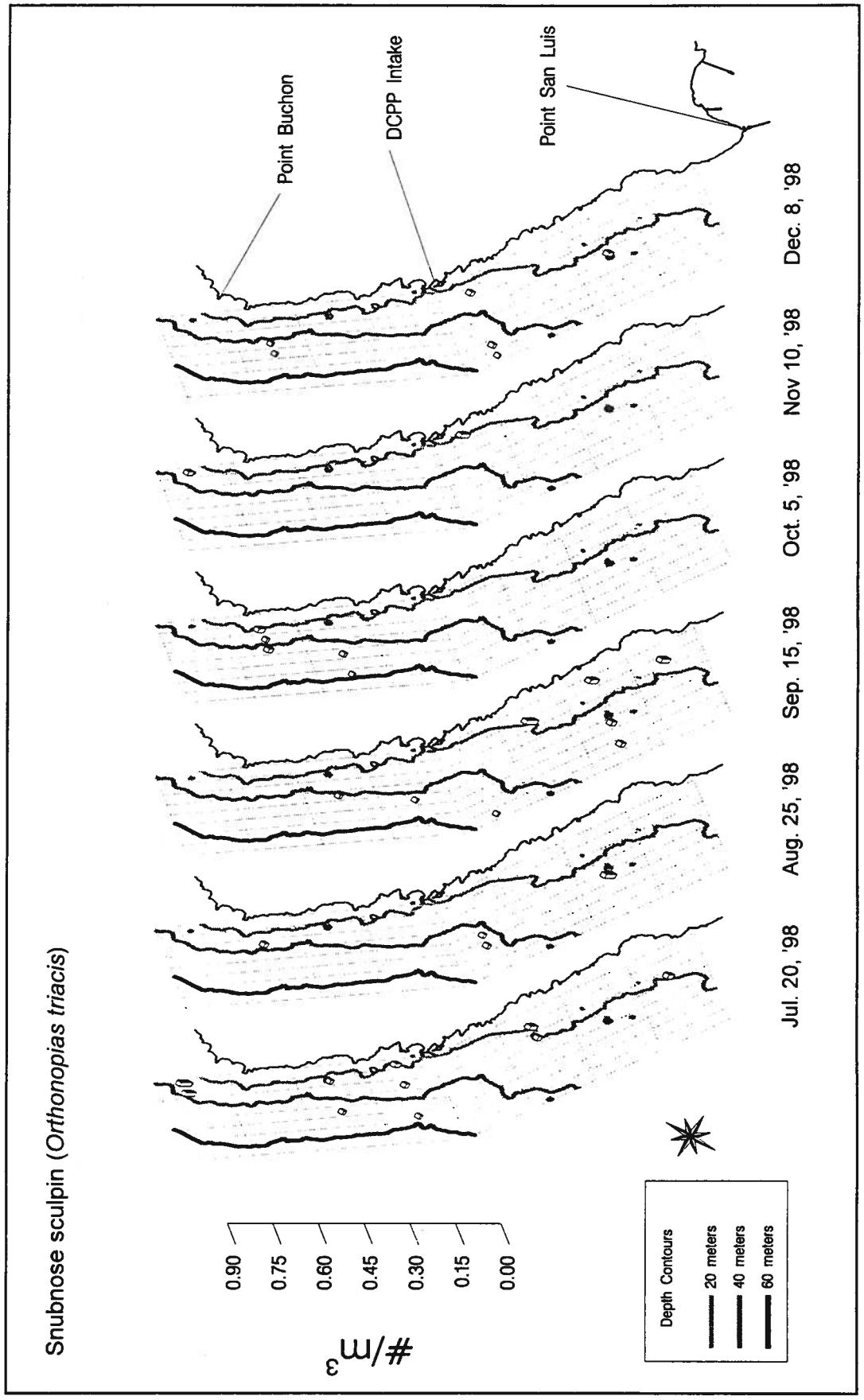


Figure 5.2.6-9c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

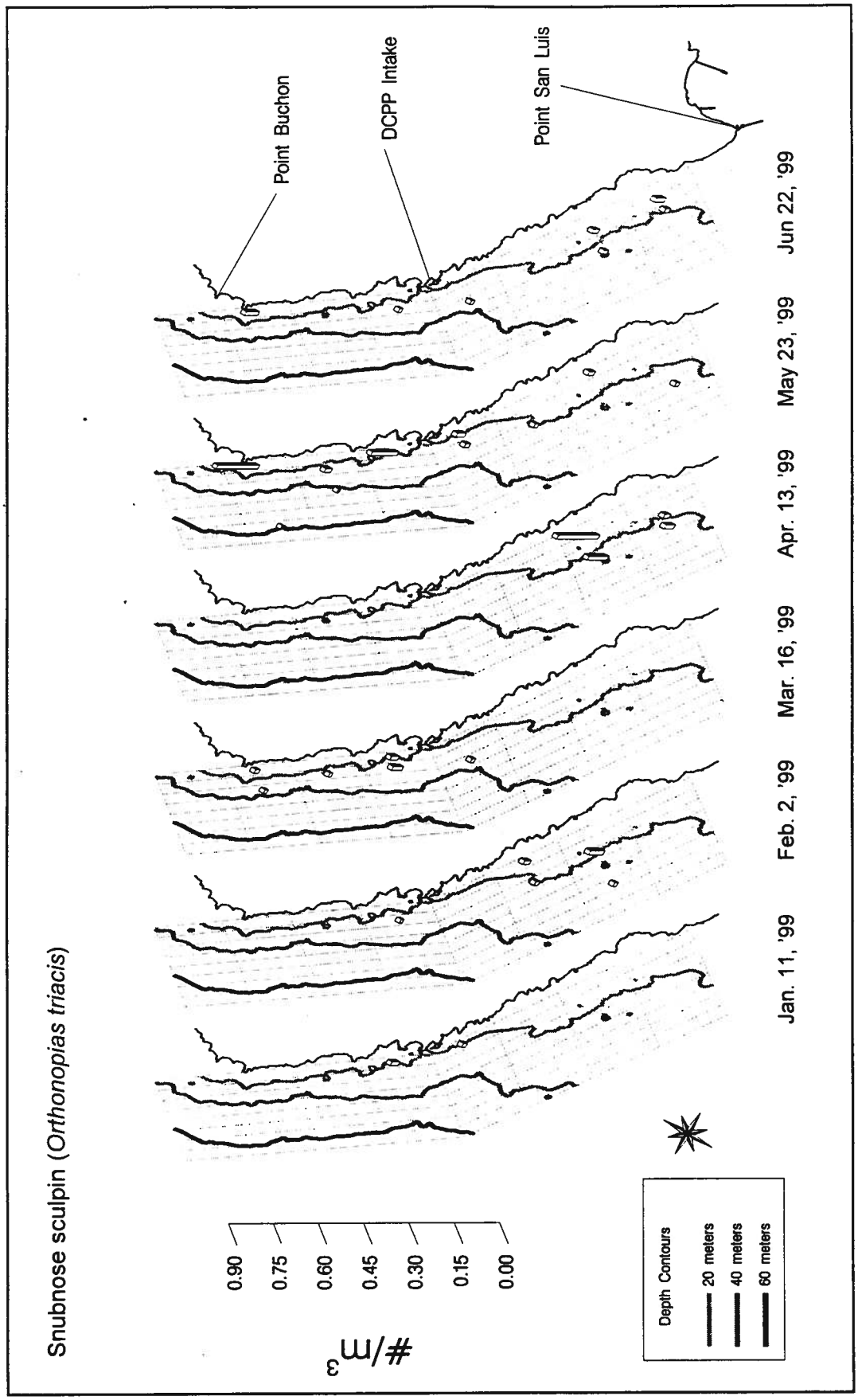


Figure 5.2.6-9d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

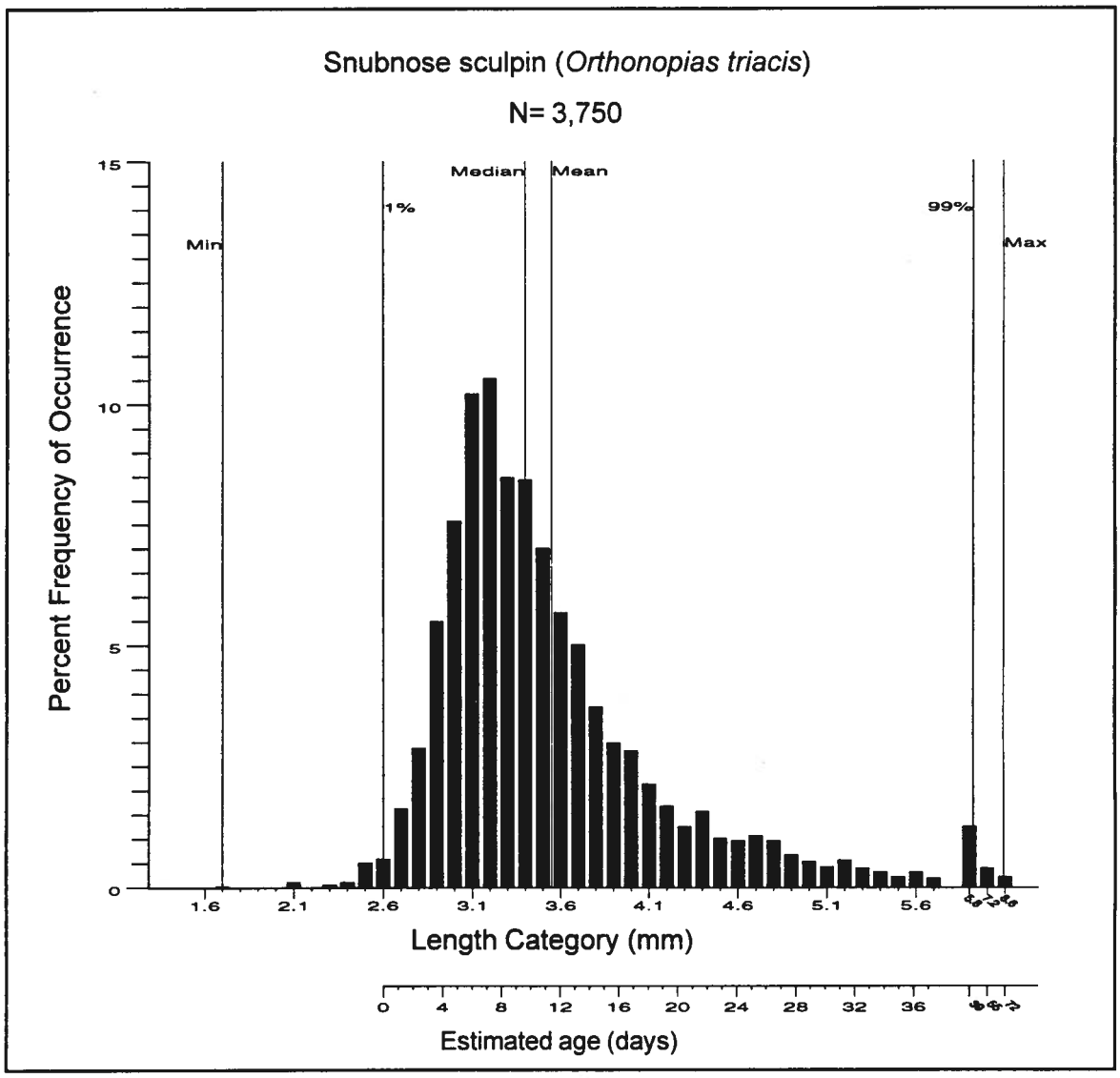


Figure 5.2.6-10. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

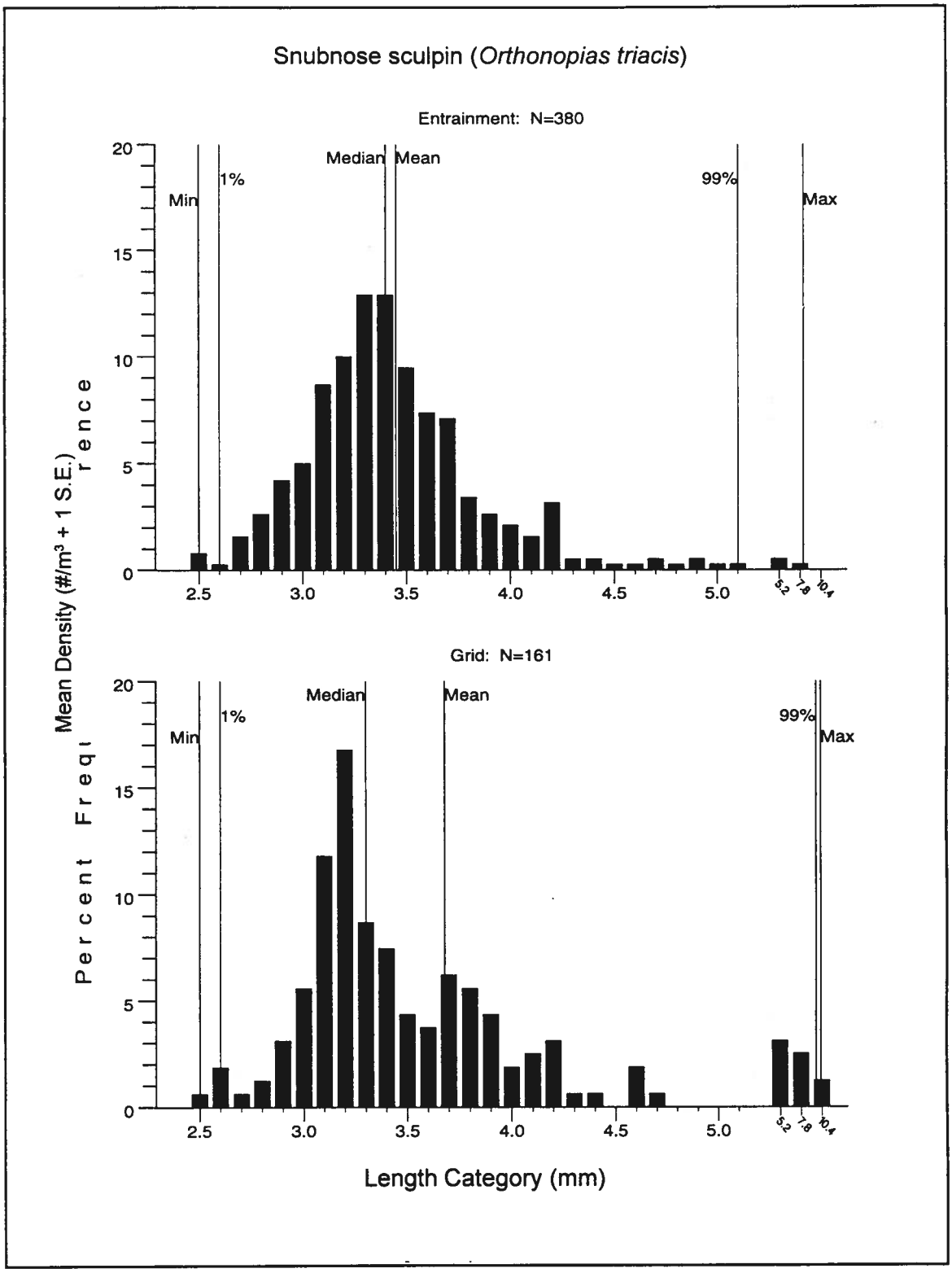


Figure 5.2.6-11. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

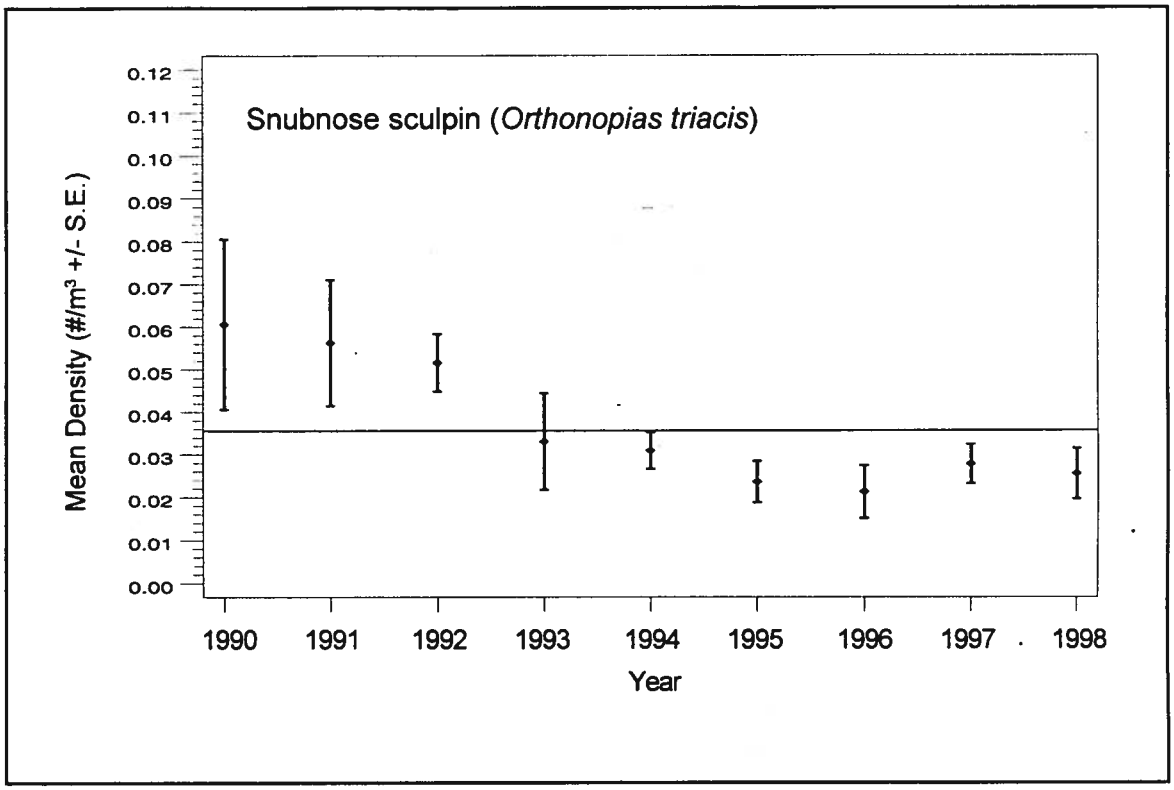


Figure 5.2.6-12. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

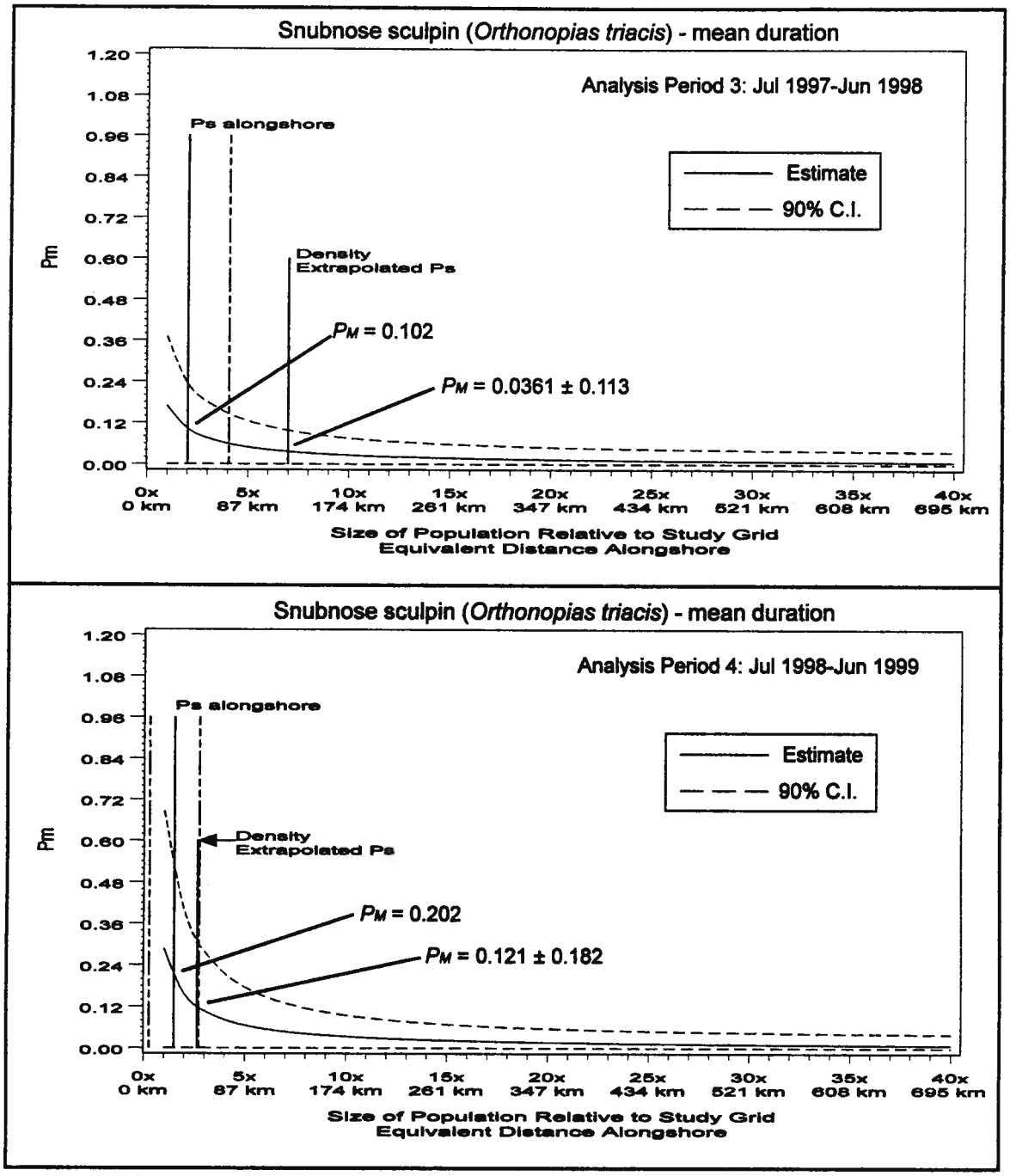


Figure 5.2.6-13a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

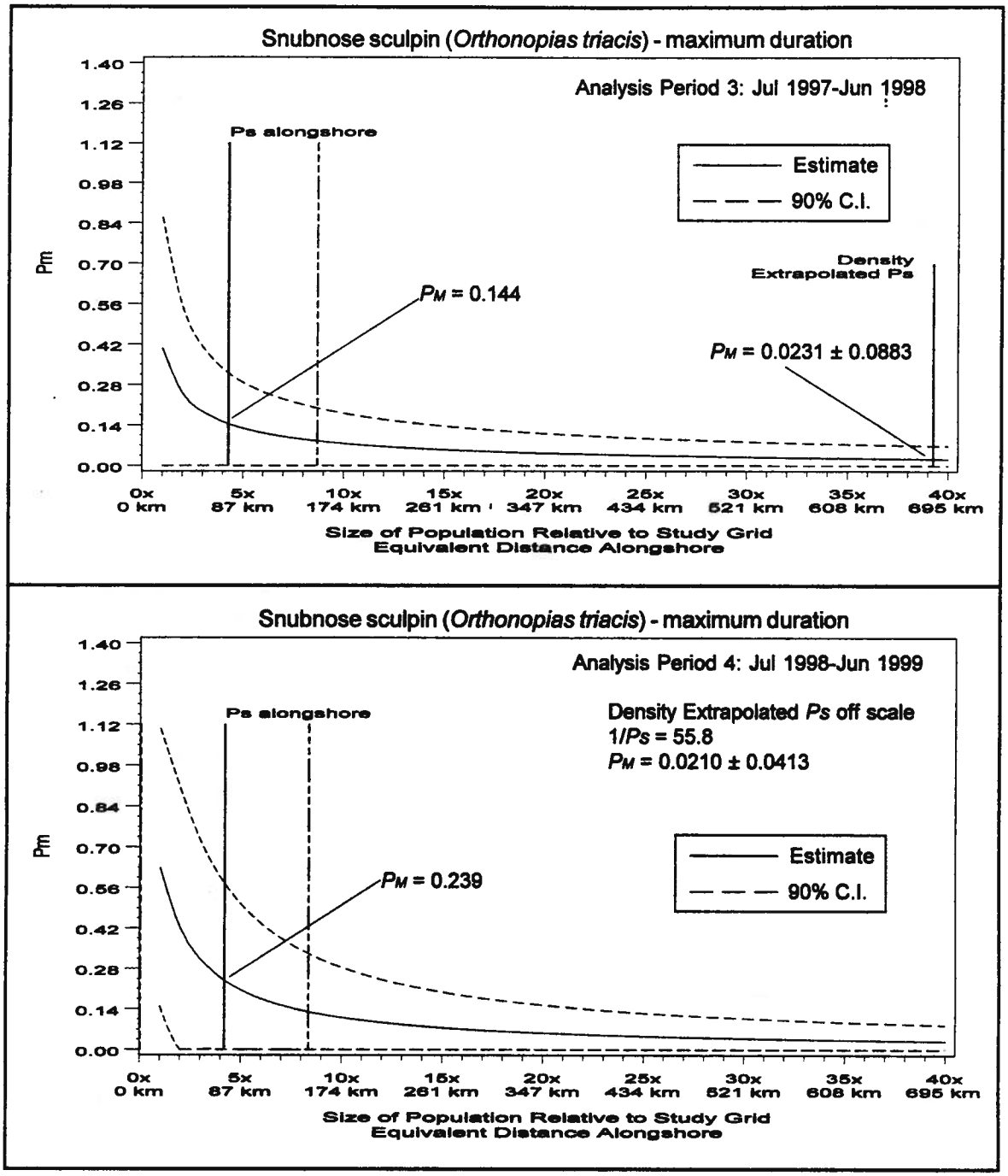


Figure 5.2.6-13b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

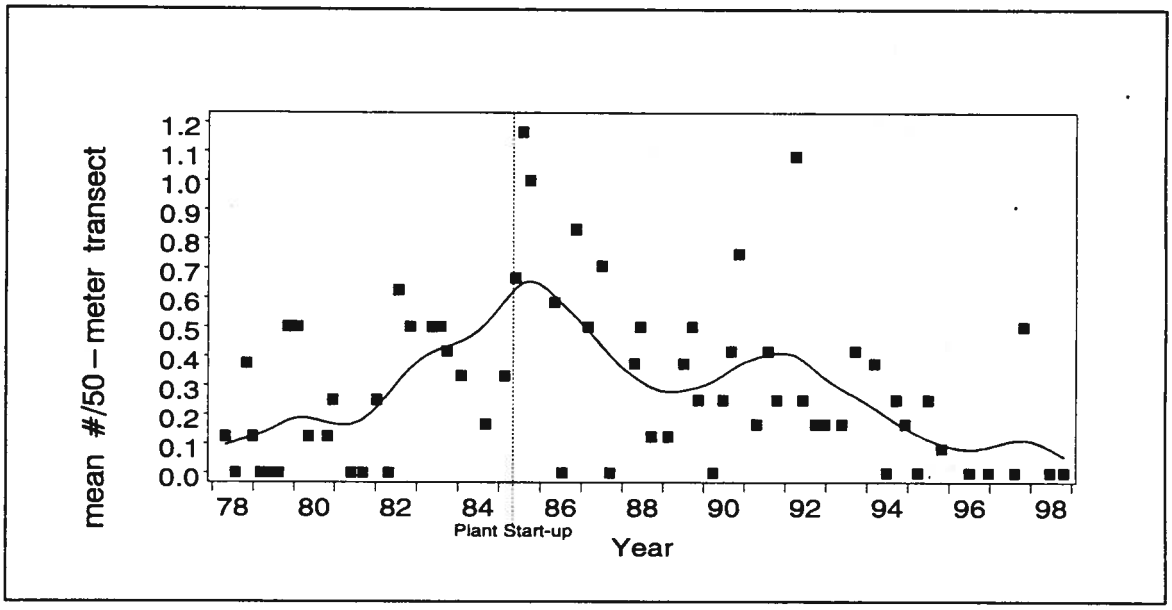


Figure 5.2.6-14. Mean number of snubnose sculpins observed per 50-meter subtidal transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to fit curve through points.

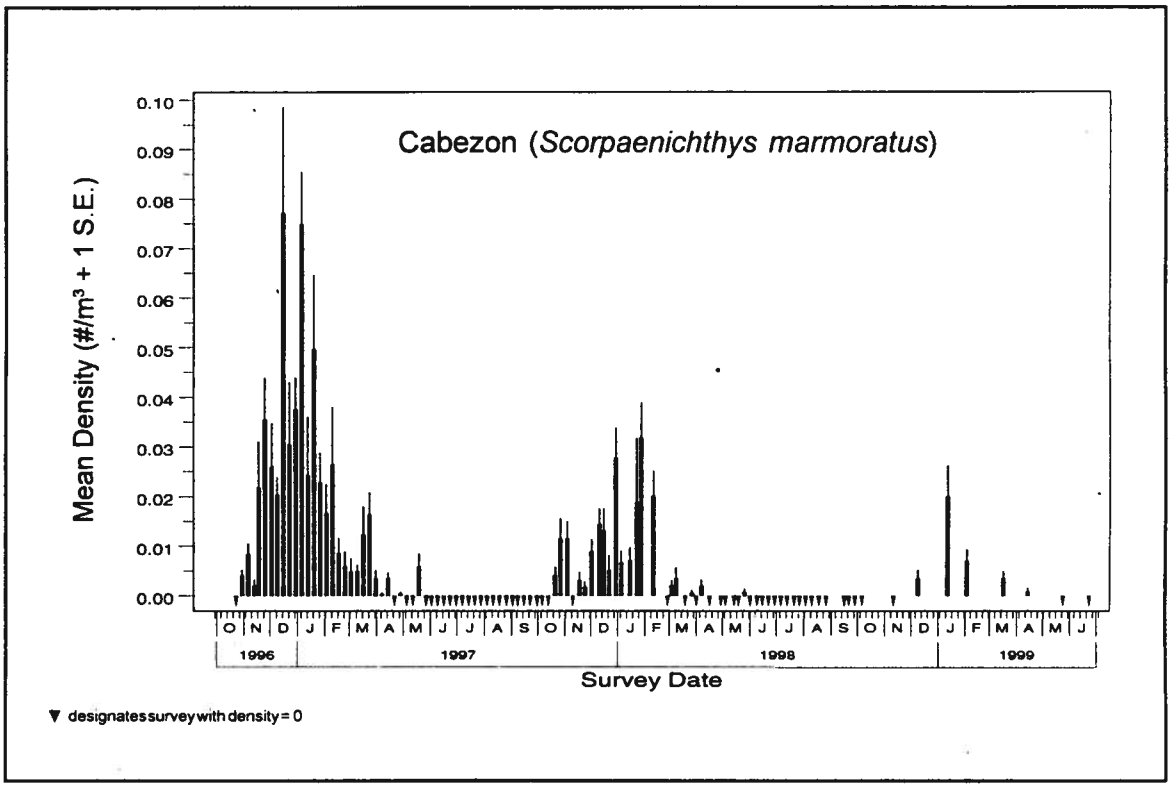


Figure 5.2.6-15. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

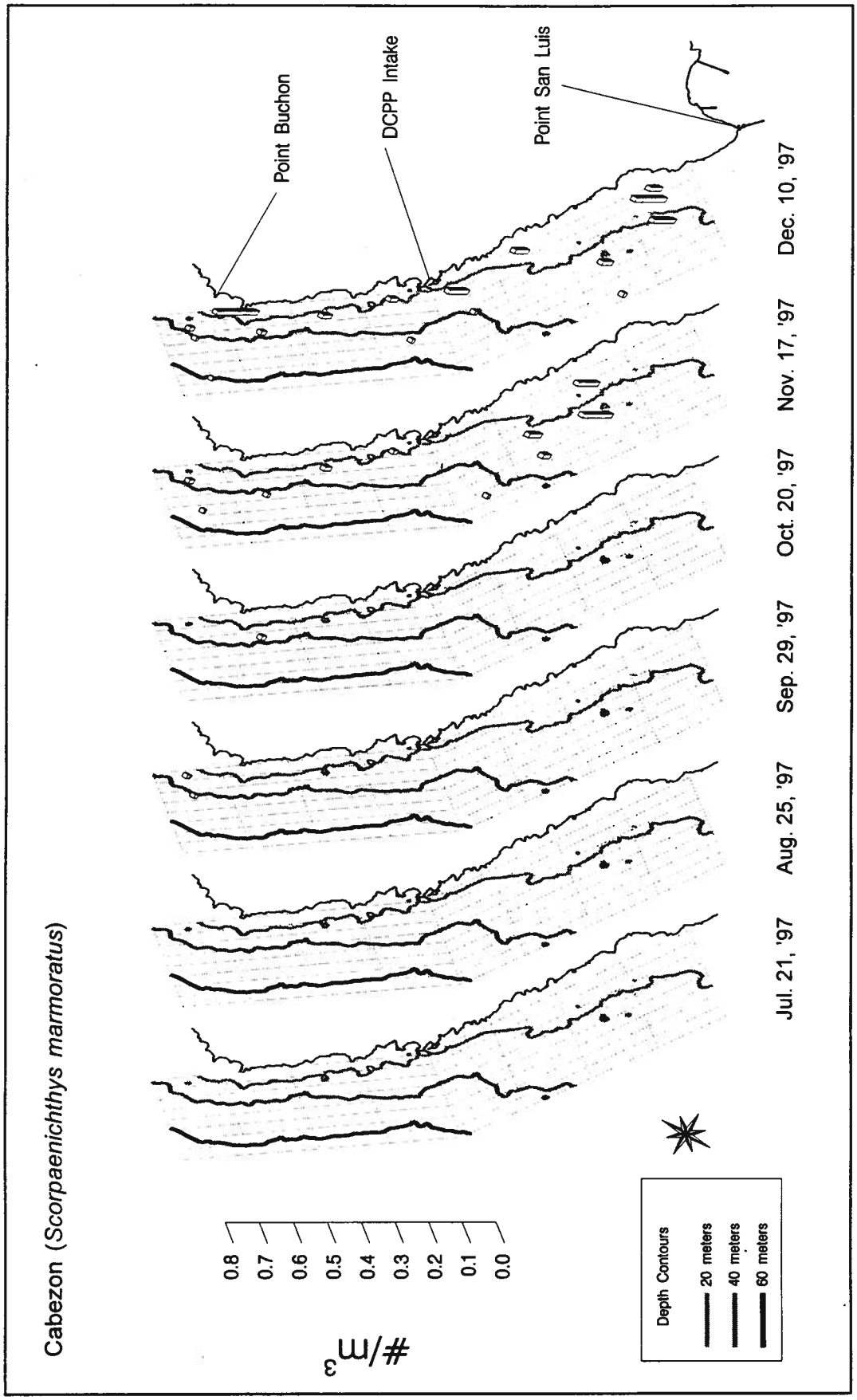


Figure 5.2.6-16a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

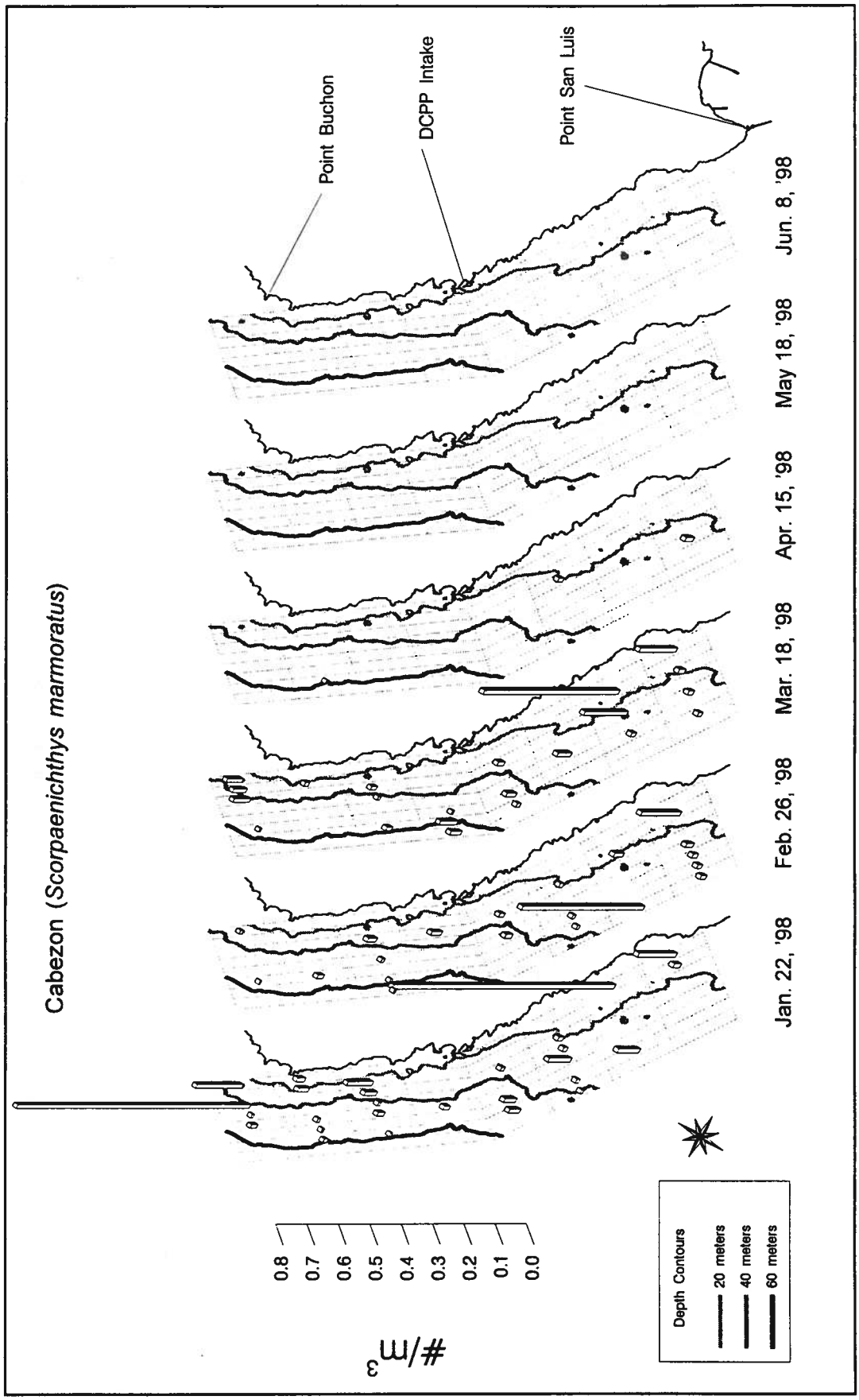


Figure 5.2.6-16b (continued). Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

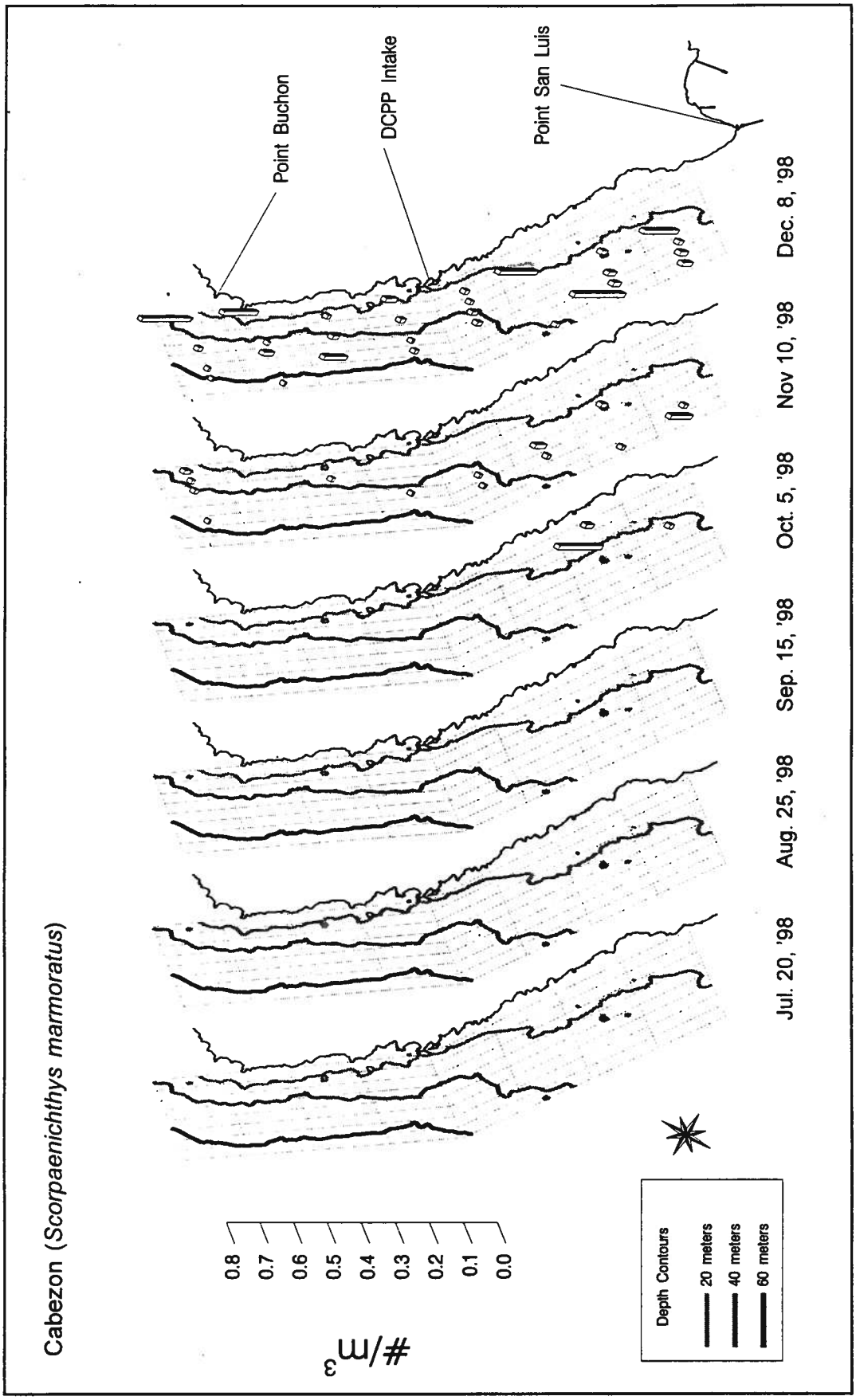


Figure 5.2.6-16c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

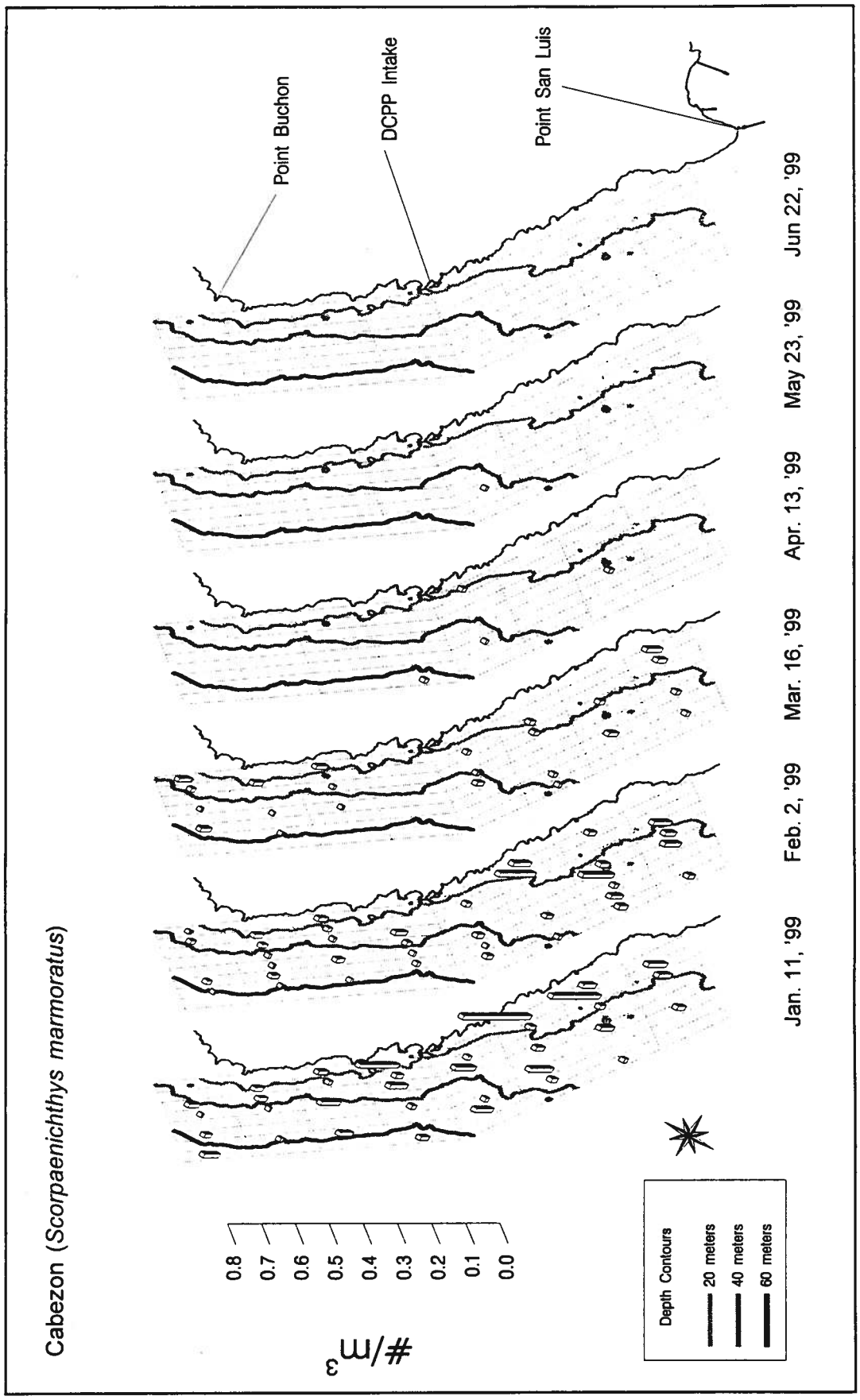


Figure 5.2.6-16d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

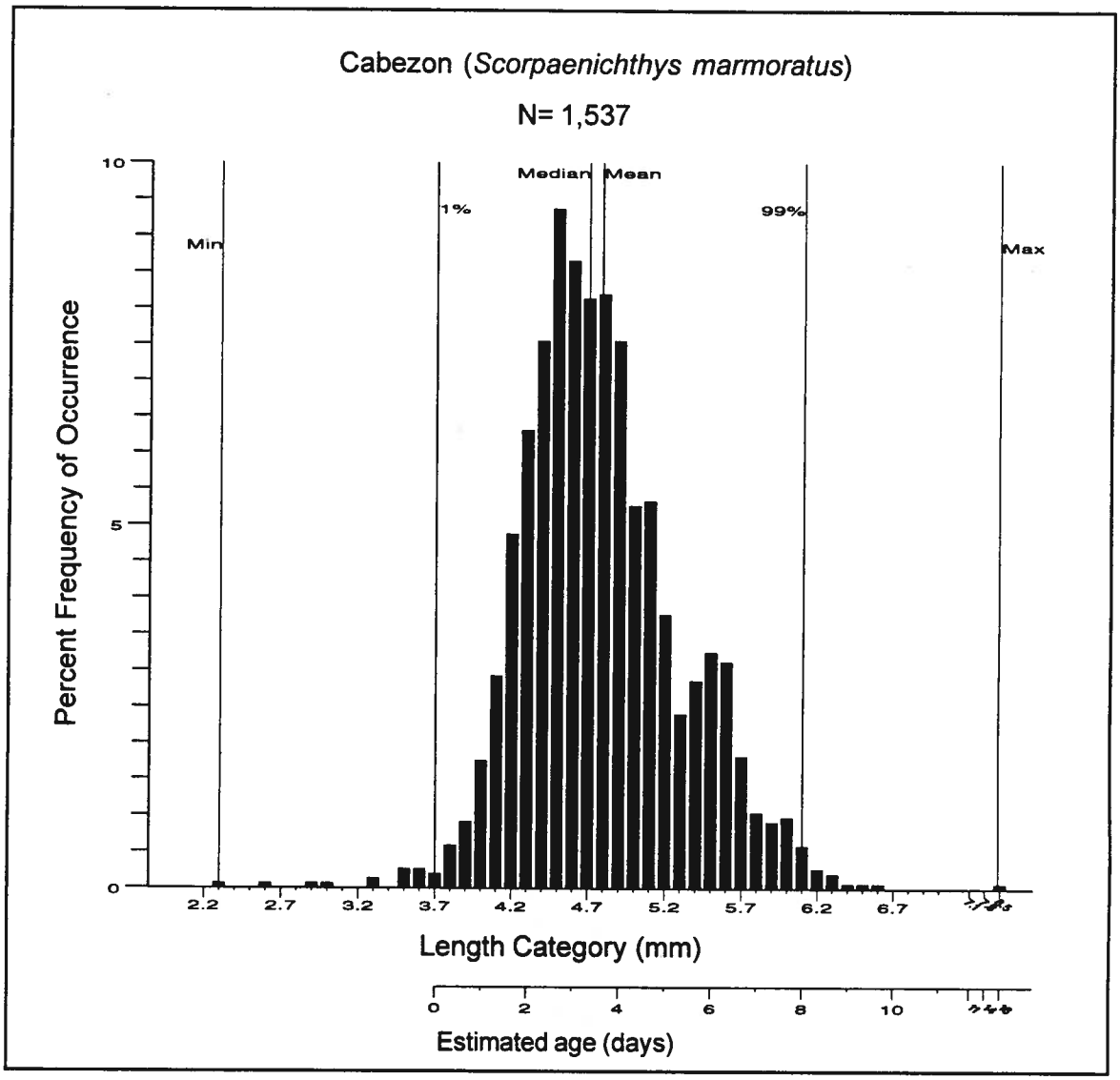


Figure 5.2.6-17. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

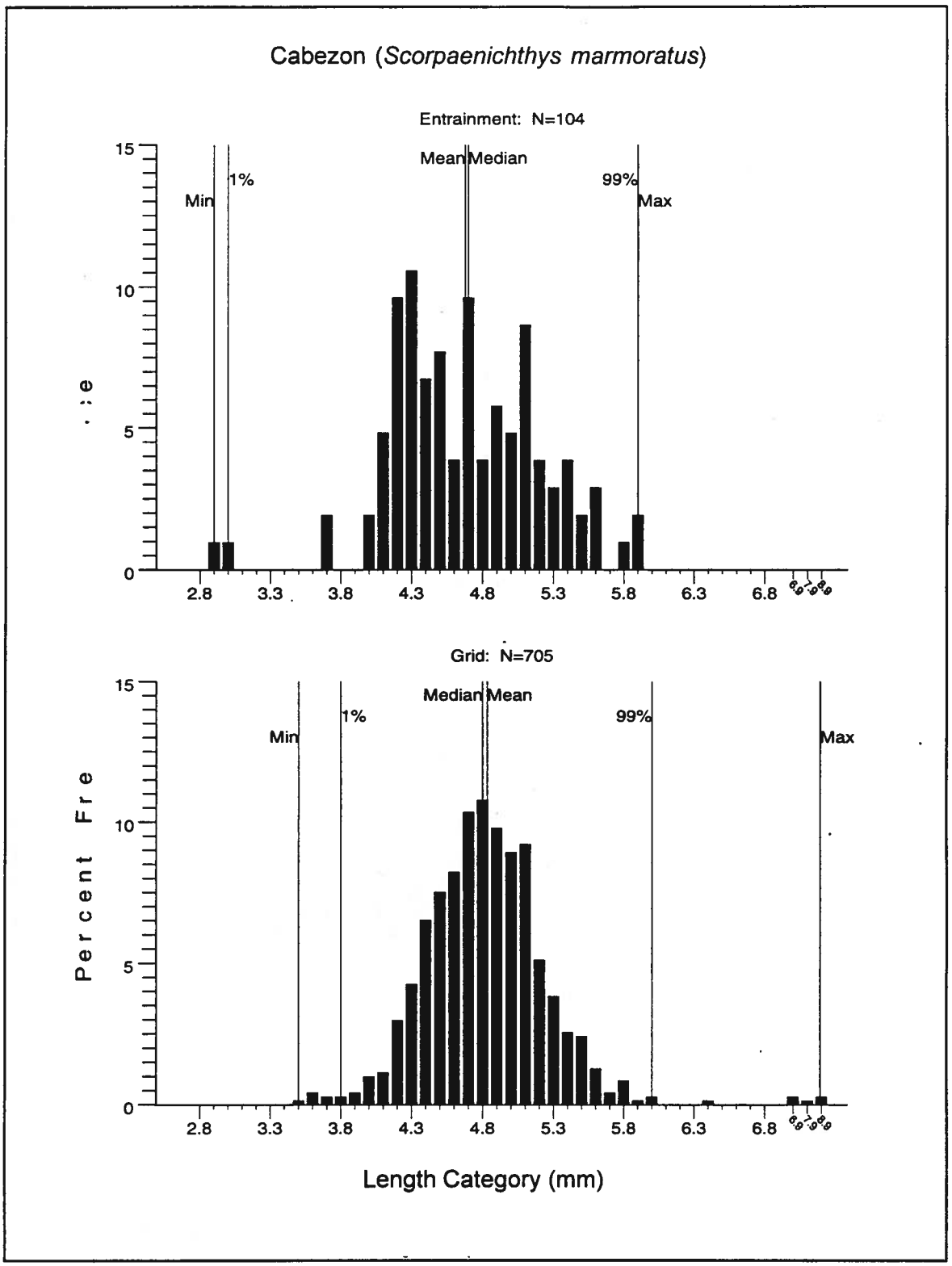


Figure 5.2.6-18. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

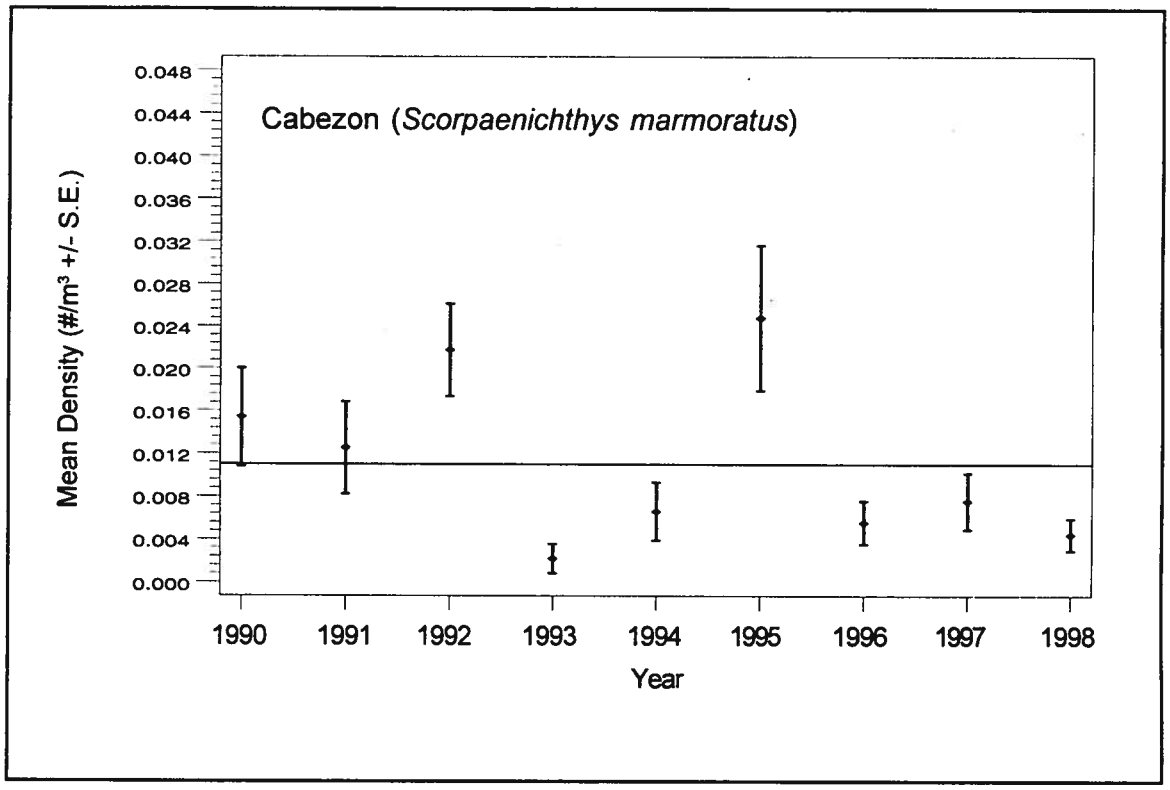


Figure 5.2.6-19. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

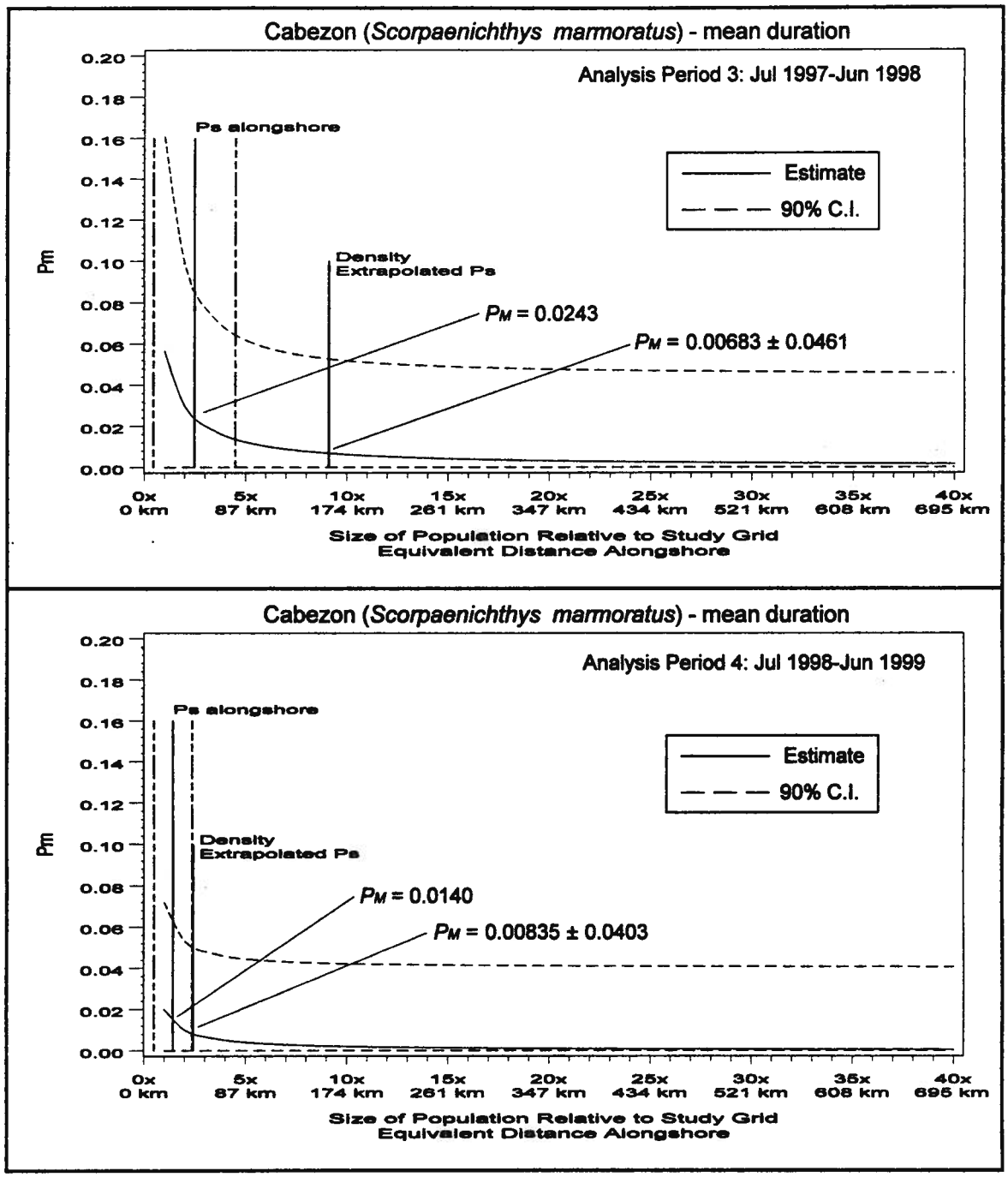


Figure 5.2.6-20a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

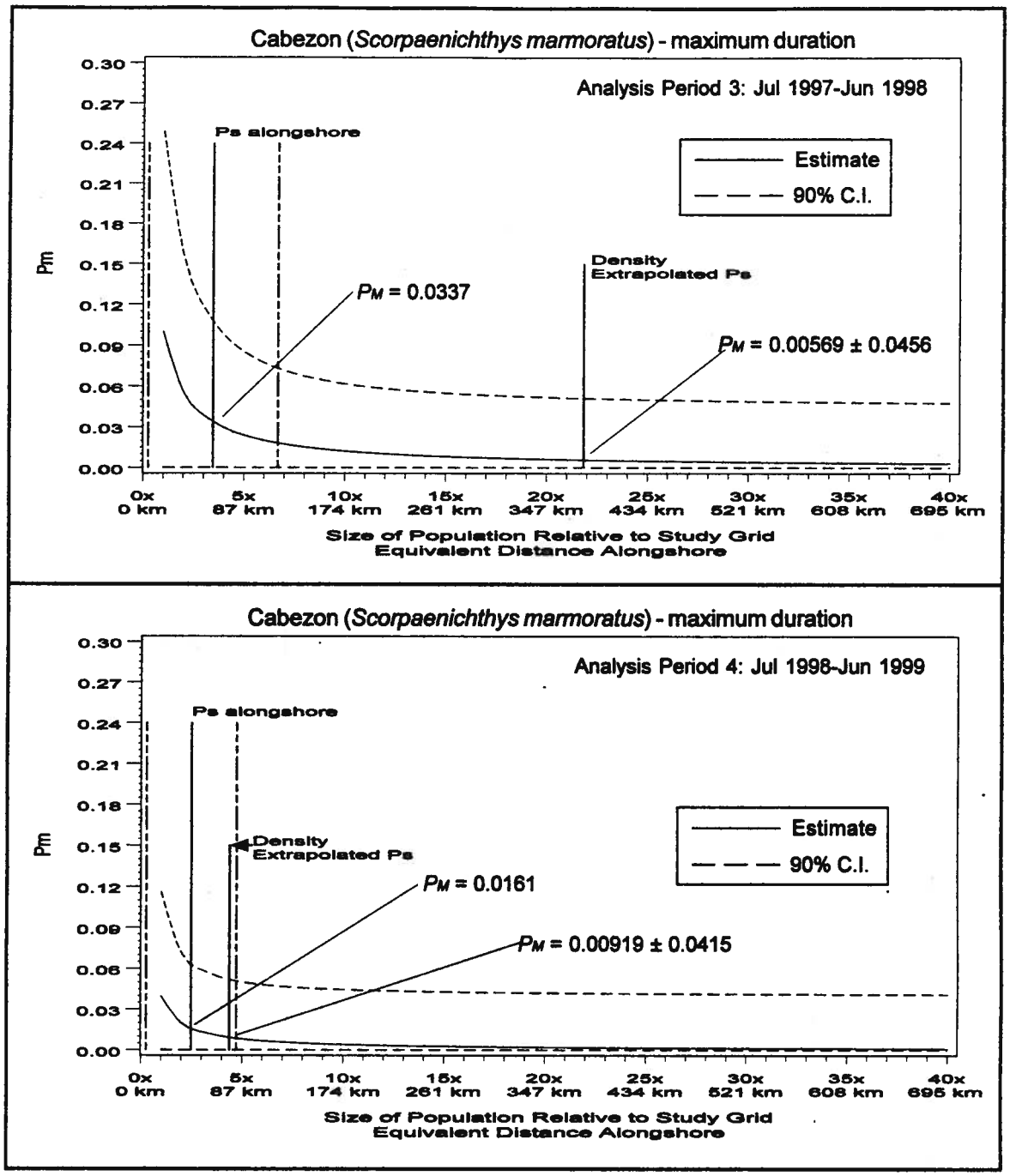


Figure 5.2.6-20b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

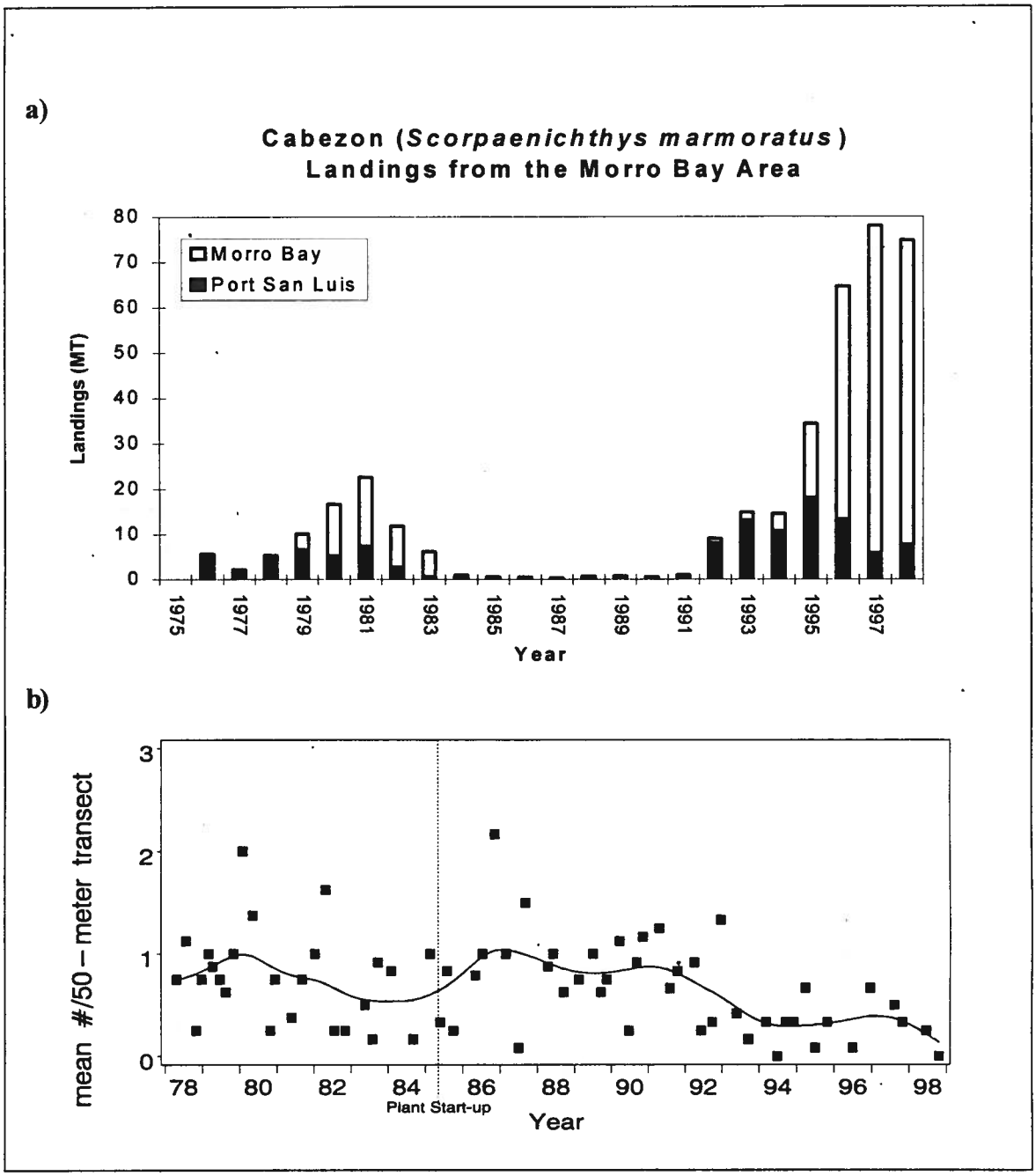
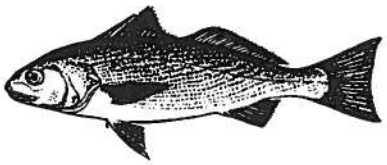


Figure 5.2.6-21. Yearly abundance of cabezon measured by two independent methods: a) Commercial landings of adults from Morro Bay area and, b) Mean number of adults and juveniles observed per 50-meter subtidal benthic transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to fit curve through points.

5.2.7 *Assessment of White Croaker* (Genyonemus lineatus)



Genyonemus lineatus (Ayres 1855); white croaker; length to 41 cm; Barkley Sound, British Columbia to Bahia Magdalena, southern Baja California; inshore to 236 m; incandescent brownish to yellowish on back, silver below; fins yellow to white (Miller and Lea 1972; Eschmeyer et al. 1983).

White croaker is one of eight species in the family Sciaenidae (the drums) occurring in California coastal waters. Other representatives of this family are white seabass (*Atractoscion nobilis*), shortfin corvina (*Cynoscion parvipinnis*), black croaker (*Cheilotrema saturnum*), spotfin croaker (*Roncador stearnsi*), California corbina (*Menticirrhus undulatus*), queenfish (*Seriphus politus*), and yellowfin croaker (*Umbrina roncador*). Two of these, shortfin corvina and black croaker, are neither commercially nor recreationally important fishes in California.

White croaker are most abundant in southern and central California at depths from 6 m to 75 m (Love 1996). They tend to form schools, often over sand or mud bottoms. White croaker spawn year-round in central California. Love et al. (1984) state that

> Batch fecundities ranged from 800 eggs in a 15.5 cm long female to 37,200 eggs in a 26 cm long female. During the spawning period about 19% of all mature female white croaker sampled contained hydrated eggs, implying that a female spawned about once every 5 d. Females of ages 1 and 2 (13–18 cm) have a spawning season of 3 mo and spawn about 18 times per season, whereas older fish (19 cm and larger) spawn over a period of 4 mo, about 24 times per season.

Love et al. (1984) also indicate that eggs and larvae are pelagic, and post-flexion larvae settle to the bottom as they develop. Juveniles are found near the bottom in 3–6 m of water, then migrate to deeper water as they mature. By one year after settlement, half of all male and female white

croaker are reproductively mature, with 100% cohort maturity attained by the third to fourth year. White croakers live 12–13 yr.

Studies conducted at the San Onofre Nuclear Generating Station (SONGS) provided estimates of adult equivalent losses for white croaker that were entrained and impinged by the cooling water intake system (Murdoch et al. 1989). They estimated the duration of each larval stage (e.g., yolk-sac, pre-flexion, flexion, post-flexion) at risk by dividing the range in body length of a larva in a given stage by the estimated daily growth rate of 0.20 mm/d (Murdoch et al. 1989). This method assumed linear growth rates throughout the larval stages. In the present study we used the same growth rate with the same assumption.

5.2.8.1 Summary of Field Collections

White croaker larvae were found seasonally at the DCPP intake structure during the years 1996–1999, with their highest densities occurring during the winter and spring (**Appendix H**). There were 4,300 larval white croaker identified from 1,114 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 24% of the subsamples collected and processed from that location during that period. The highest densities of larval croaker occurred in March 1997 and May 1998 (**Figure 5.2.7-1**). The density of this species was variable between weeks, with one or two weeks of rather high density followed by several consecutive weeks of lower density. The highest entrainment densities of white croaker larvae in 1998 occurred in May, while only a few were seen during that month in 1997. This shift in abundance peaks between years may have been affected by the El Niño event that began during the fall/winter season of 1997–98 (Lynn et al. 1998; NOAA 1999).

White croaker larvae generally occurred in the DCPP study grid during December through May of each year (**Appendix H**), with their highest recorded abundance occurring during March 1998 (**Figure 5.2.7-2**). When present, the highest white croaker larval densities occurred in the southern half of the nearshore grid sampling area. There were 1,710 larval white croaker identified from 422 bongo net subsamples representing 14% of the study grid subsamples collected and processed from July 1997–June 1999.

Standard lengths of all white croaker larvae collected at the DCPP intake structure between October 1996 and June 1999 ranged from a minimum of 0.90 mm to a maximum of 8.2 mm (**Figure 5.2.7-3**). The central 98% of this length-frequency distribution resulted in minimum and maximum lengths for the analyses of 1.1 mm and 5.5 mm, respectively. The mean larval length in this distribution was 2.4 mm. Hatching size for white croaker is reported between 1.5 and 1.8 mm (Moser 1996). The fact that we observed larvae smaller than the reported hatching lengths can be explained by natural variation of hatch lengths within the population and by the phenomenon of shrinkage following preservation (Theilacker 1980).

The growth rate of larval white croaker (0.20 mm/d) reported by Murdoch et al. (1989) was used to estimate ages of entrained larvae. Assuming that the shortest larvae were immediately post-hatch and aged zero days, then the estimated ages of larvae entrained could have ranged from zero days up to 22–37 d post-hatching for the longest larvae measured. On average, the estimated ages of white croaker larvae entrained at DCPP ranged from 7–8 d post-hatching.

Differences between the distributions of white croaker larval lengths at the DCPP intake structure and the study grid (**Figure 5.2.7-4**) were tested using a Kolmogorov-Smirnov test. The test detected a significant difference between the two distributions ($p < 0.001$). The largest difference between the distributions was detected at a length of approximately 2.7 mm, the size separating two modes of the distribution from the DCPP intake. A single mode predominated in the study grid distribution. A *t*-test also detected a significant difference ($p < 0.001$) between the mean lengths for the two locations: 2.7 mm at the DCPP intake and 2.3 mm in the study grid. Differences in the mean are partly due to a larger differences in sample size between the study grid (n=1,263) and the intake location (n=150). A large sample size also causes the *t*-test to be highly sensitive to small differences between the means (Zar 1984). The length frequency of white croaker (n=3,529) in all entrainment subsamples (**Figure 5.2.7-3**) yielded a distribution similar to that of the nearshore grid subsamples. While these differences were statistically detectable and significant, both distributions were heavily weighted with recently hatched (yolk-sac) and pre-flexion larvae. Therefore, we believe that the entrainment subsamples and grid subsamples represent the same population of available larvae.

5.2.7.2 Estimating Total Annual Entrainment

The annual estimated mean entrainment white croaker larvae ranged from a minimum of 65,100,000 ($\widehat{SE}$ =3,880,000) in Analysis Period 2 to a maximum of 70,500,000 ($\widehat{SE}$ =2,620,000) in Analysis Period 1 (**Table 5.2.7-1**). A 95% confidence interval around these estimates approximated by $2 \cdot \widehat{SE}(\widehat{E_T})$ does not overlap indicating that the estimates were probably not statistically different.

Annual estimates of the number of white croaker larvae entrained, adjusted to a long-term mean based on annual indices using weekly Intake Cove surface plankton tows (**Figure 5.2.7-5**), ranged from a minimum of 305,000,000 ($\widehat{SE}$ =11,300,000) in Analysis Period 1 to a maximum of 447,000,000 ($\widehat{SE}$ =26,300,000) in Analysis Period 3 (**Table 5.2.7-2**). The average index for the years 1997 and 1998 are 0.0251 larvae/m^3 and 0.0161 larvae/m^3, respectively, while the long-term average index is 0.109 larvae/m^3 for the years 1990–1998, yielding the ratio $\overline{I} / I_i$ of 4.33 for 1997 and 6.77 for 1998. The adjustment increases the difference between the estimates. A 95% confidence interval, approximated by $2 \cdot \widehat{SE}(\hat{E}_{Adj-T})$ would show a statistically significant difference between the 1996–1997 estimates and the two 1997–1998 estimates.

5.2.7.3 Fecundity Hindcasting (*FH*)

Fecundity hindcasting requires age-specific fecundity and mortality to assess entrainment effects. White croaker spawn multiple times within a year. Females of 1–2 yr (i.e., 13–18 cm) undergo a 3 mo spawning season and spawn ca. 18 times per season. Older fish (i.e., 19 cm and larger) spawn over a period of 4 mo, ca. 24 times per season (Love et al. 1984). In our calculations we will assume an average of 21 egg batches per yr. The average number of eggs per batch will be based on Love et al. (1984) who showed a modal age of 3 yr for fish lengths of 18–18.9 cm. This permits extrapolation to a batch fecundity of approximately 5,000 eggs for age 3 yr.

Love et al. (1984) found that adult white croaker have a maximum longevity of 12 yr, and Love (1996) stated that the species matures between 1 and 4 yr, with half spawning after 1 yr. The expected average age of maturation for the purpose of calculations was estimated as 2 yr.

Love (1996) reported that white croaker eggs hatch in 2 d, while Murdoch et al. (1989) suggested an instantaneous egg mortality rate of $Z = 0.25$ (survival ≈78%/day). Egg survival can therefore be estimated as

$$S_{Egg} = e^{-2(0.25)} = 0.6065.$$

An estimate of survival for larvae from the time of egg hatching to entrainment was based on information on larval lengths collected from DCPP entrainment subsamples. The central 98% lengths of the larvae ranged from 1.1–5.5 mm with a mean of 2.44 mm. Murdoch et al. (1989) states that white croaker larvae grow at a rate of approximately 0.2 mm per day. The average time to entrainment from hatching (hatch length is assumed to be the shortest length recorded in DCPP subsamples) is then estimated as 6.7 d. Assuming that the instantaneous larval survival rate is the same as for eggs, larval survival is estimated as

$$S_{Larvae} = e^{-6.7(0.25)} = 0.1873.$$

It is likely that the larval survival rate is equal to or greater than the survival rate of eggs. Thus, the survival rate for larvae is probably underestimated, resulting in an overestimate of *FH*.

As an example of the *FH* calculation, when these parameters are modeled for the period October 1, 1997 through September 30, 1998, the resulting number of adult females that would have produced the number of entrained larvae is given as

$$\widehat{FH} = \frac{440{,}198{,}000}{(0.6065)(0.1873)(5{,}000 \cdot 21)\left(\frac{12-2}{2}\right)} = 7{,}380 \text{ adult females per yr.}$$

Estimates of *FH* were different between the two years of sampling (**Table 5.2.7-3**). Numbers of hindcast adult females from the three analysis periods ranged from 5,000–7,500. The least number of hindcast individuals were estimated from Analysis Period 1 (October 1996–September 1997) while the greatest were estimated from Analysis Period 3 (July 1997–June 1998).

Sensitivity Analysis

Longevity had the greatest leverage on recalculated values of $\widehat{FH}$ (**Table 5.2.7-4**). Varying this parameter in the model resulted in an $\widehat{FH}$ range of 2,320–237,000 adult females for the three periods, compared to the original estimate of 5,110–7,500 adult females. Survivorship had the next greatest amount of leverage on recalculated $\widehat{FH}$.

5.2.7.4 Adult Equivalent Loss (*AEL*)

For white croaker, no independent estimate of survival from entrainment to age of adult recruitment was found in the literature. To calculate $\widehat{AEL}$ as $2 \cdot \widehat{FH}$, we must first align the ages at maturity (~5 yr) and recruitment into the fishery (3–4 yr) so that the *FH* females are of the same age as the *AEL* recruits. As such, $\widehat{AEL}$ ranged from 14,700 to 21,600 adult white croaker per year across the three analysis periods.

5.2.7.5 Empirical Transport Model (*ETM*)

Monthly estimates of proportional entrainment (PE_i) in each *i*th survey from the two years sampled ranged from a minimum of zero to a maximum of 0.0340±0.008 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in January 1998 (**Table 5.2.7-5**). No larvae were collected at either the DCPP intake or from the study grid when both $\widehat{PE_i}$ and the proportion of larvae present in the *i*th survey period ($\hat{f}_i$) were equal to zero (e.g., August–November 1998 and April–June 1999). When $\widehat{PE_i}$ =0 and $\hat{f}_i$>0, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. Since $\widehat{PE_i}$ was weighted by $\hat{f}_i$, some of the larger $\widehat{PE_i}$ values were less important to the annual estimate of entrainment mortality.

The lengths of entrained white croaker larvae, excluding the largest 1% and smallest 1% of all measurements, ranged from 1.1 to 5.5 mm. Assuming a growth rate of 0.2 mm/day (Murdoch et al. 1989), larvae may be susceptible to entrainment by the plant for approximately 22 days. The frequent, repeated breeding of white croaker suggests that the entrainment at Diablo Canyon occurs over multiple larval batches. Literature on the life history of white croaker would support this contention (Love et al. 1984; Love 1996).

Estimates of $\widehat{P_M}$ differed markedly when calculated using the mean and maximum larval durations and were generally higher during the second year of sampling (**Figures 5.2.7-6**). The larval durations calculated using the mean and maximum larval lengths differed by 15 d (7 vs. 22 d). For $\widehat{P_S}$ calculated using only alongshore currents, differences in duration did not cause much change in $\widehat{P_M}$ between mean and maximum duration; ca. 2–3% in Analysis Period 3 and ca. 4–6% in Analysis Period 4. Although white croaker larvae were more common in the southern half of the study grid when they were abundant (March 1998), they generally occurred throughout the study grid and were not restricted to the nearshore grid cells. Therefore, P_S was calculated using both alongshore current movement and extrapolating study grid abundance offshore based on onshore current movement. For $\widehat{P_S}$ calculated using alongshore currents and offshore abundance extrapolation, changing larval duration did not change $\widehat{P_M}$ much in Analysis Period 3 (ca. 0.1–0.3% for both durations), but yielded a change from 2% (mean duration) to 1% (maximum duration) in Analysis Period 4.

5.2.7.6 Interpretation of Assessment Results

White croaker have both commercial and recreational fishery value. Love et al. (1984) stated that fishing for white croaker in Monterey Bay occurs on a daily basis year round. The daily catch can range from 400–900 kg (ca. 0.4–0.9 MT) with a maximum catch of 1,800 kg (ca. 1.8 MT) of white croaker. The annual harvest of white croaker in Monterey Bay can then be estimated at approximately 248 MT. However, available evidence suggests that commercial catches of white croaker have been declining since around 1985 in the Monterey Bay area (Starr et al. 1998).

Payment to the fishermen in 1990 was ca. $0.19/kg depending on catch size and fish condition (Leet et al. 1992). The total recreational catch in both northern and southern California in 1998 was estimated at 443,000 fishes (PSMFC RecFIN database). Data from the California Department of Fish & Game show that local commercial landings of white croaker during the period 1975–1998 averaged 9.1 MT at Port San Luis and 3.8 MT at Morro Bay (**Figure 5.2.7-7**). Because of the low economic value of the species in comparison to other targeted species, some

of the white croaker landings probably represent incidental catches as a result of these other fisheries.

Love et al. (1984) gives the length mode for white croaker caught in the commercial gillnet fishery (28 cm SL) and from skiff sportfishing (22.5 cm SL). The weight of a 28 cm white croaker is approximately 200–225 g, depending on gender, with females generally heavier. Assuming a daily natural mortality rate of 0.0005 (Ricker 1975), we can align the approximately 5-yr old *FH* females to an age of recruitment to the fishery of 3.5 yr (Love et al. 1984). In this manner, assuming a 50:50 sex ratio in the population, $2 \cdot FH$ becomes equivalent to *AEL*. Thus, the approximate *AEL* based on *FH* ranged from 14,700 3.5-yr old adults in 1996–1997 to a maximum of 21,600 in 1997–1998. Using an average weight of 0.213 kg/adult in the fishery derived from Love et al. (1984), the price per kg from Leet et al. (1992), and assuming 100% catchability, these equivalent adult losses could equate to 3.1–4.6 MT annually, approximately $594–872/yr. Alternatively, this conservative estimate of loss represents 24–36% of the average annual catch from the Morro Bay area between 1975–1998. However, estimates of annual proportional larval entrainment mortality ($\widehat{P_M}$) that assume no compensation, suggest losses of 0.1% to perhaps as high as 6% of the local population (over areas 4–41 times the study grid area), providing a lower limit to these proportional effects.

Adult white croaker are widely distributed over sand and mud bottoms in shelf waters and are not typically found in association with rock substratum. Combining this fact with the fact that white croaker adults are rarely observed in the vicinity of the power plant (Tenera 1997c) leads to the conclusion that their abundance is centered elsewhere than around Diablo Canyon. Therefore, the greater abundance of soft bottom habitat relative to rock substrata in the Morro Bay and Avila areas, combined with relatively low larval mortality probabilities of $\widehat{P_M}$ (0.001–0.06), suggests that entrainment impacts on local white croaker populations are not appreciable.

Table 5.2.7-1. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for white croaker (*Genyonemus lineatus*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	70,500,000	2,620,000
2) Oct 1, 1997–Sep 30, 1998	65,100,000	3,880,000
3) Jul 1, 1997–Jun 30, 1998	66,100,000	3,890,000

Table 5.2.7-2. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for white croaker (*Genyonemus lineatus*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	305,000,000	11,300,000
2) Oct 1, 1997–Sep 30, 1998	440,000,000	26,300,000
3) Jul 1, 1997–Jun 30, 1998	447,000,000	26,300,000

Table 5.2.7-3. Estimated number of white croaker adult females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{FH})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	5,110	12,300	268,000	97
2) Oct 1, 1997–Sep 30, 1998	7,380	17,800	387,000	141
3) Jul 1, 1997–Jun 30, 1998	7,500	18,100	394,000	143

Table 5.2.7-4. White croaker (*Genyonemus lineatus*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: Oct 1, 1996–Sept 30, 1997

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 5,110				
$\hat{E}_{Adj-T}$	305,000,000	4,800	5,430	626
$\hat{S}_{egg}$	0.607	3,100	26,500	23,400
$\hat{S}_{larvae}$	0.187	987	26,500	25,500
# Eggs/yr	105,000	987	26,500	25,500
Longevity	12	2,320	162,000	159,000
Maturation	2	4,650	6,390	1,740

Table 5.2.7-4. White croaker (*Genyonemus lineatus*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%

b) Analysis Period 2: Oct 1, 1997–Sept 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 7,380				
$\hat{E}_{Adj-T}$	440,000,000	6,660	8,110	1,450
$\hat{S}_{egg}$	0.607	4,480	38,200	33,800
$\hat{S}_{larvae}$	0.187	1,420	38,200	36,800
# Eggs/yr	105,000	1,420	38,200	36,800
Longevity	12	3,360	233,000	230,000
Maturation	2	6,710	9,230	2,520

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 7,500				
$\hat{E}_{Adj-T}$	447,000,000	6,780	8,230	1,450
$\hat{S}_{egg}$	0.607	4,550	38,900	34,300
$\hat{S}_{larvae}$	0.187	1,450	38,900	37,400
# Eggs/yr	105,000	1,450	38,900	37,400
Longevity	12	3,410	237,000	234,000
Maturation	2	6,820	9,380	2,560

Table 5.2.7-5. White croaker (*Genyonemus lineatus*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.0244	0.0293	0.00704	0.000890
Aug 25, 1997	0	0	0.00447	0.000620
Sep 29, 1997	0	0	0.00584	0.000690
Oct 20, 1997	0	0	0.000570	0.000220
Nov 17, 1997	0	0	0.00135	0.000370
Dec 10, 1997	0	0	0.0628	0.00457
Jan 22, 1998	0.0340	0.00858	0.139	0.00489
Feb 26, 1998	0.000630	0.000450	0.00780	0.000750
Mar 18, 1998	0.000390	0.000120	0.0532	0.00277
Apr 15, 1998	0.00203	0.000720	0.293	0.00932
May 18, 1998	0.00526	0.00146	0.425	0.0107
Jun 8, 1998	0.00948	0.0116	0.000590	0.000200

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0	0	0.00221	0.000810
Aug 26, 1998	0	0	0	0
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0.00545	0.00325	0.0348	0.00338
Jan 12, 1999	0.00492	0.000830	0.673	0.00931
Feb 3, 1999	0.0122	0.00558	0.104	0.00606
Mar 17, 1999	0.0302	0.0104	0.186	0.00728
Apr 14, 1999	0	0	0	0
May 24, 1999	0	0	0	0
Jun 23, 1999	0	0	0	0

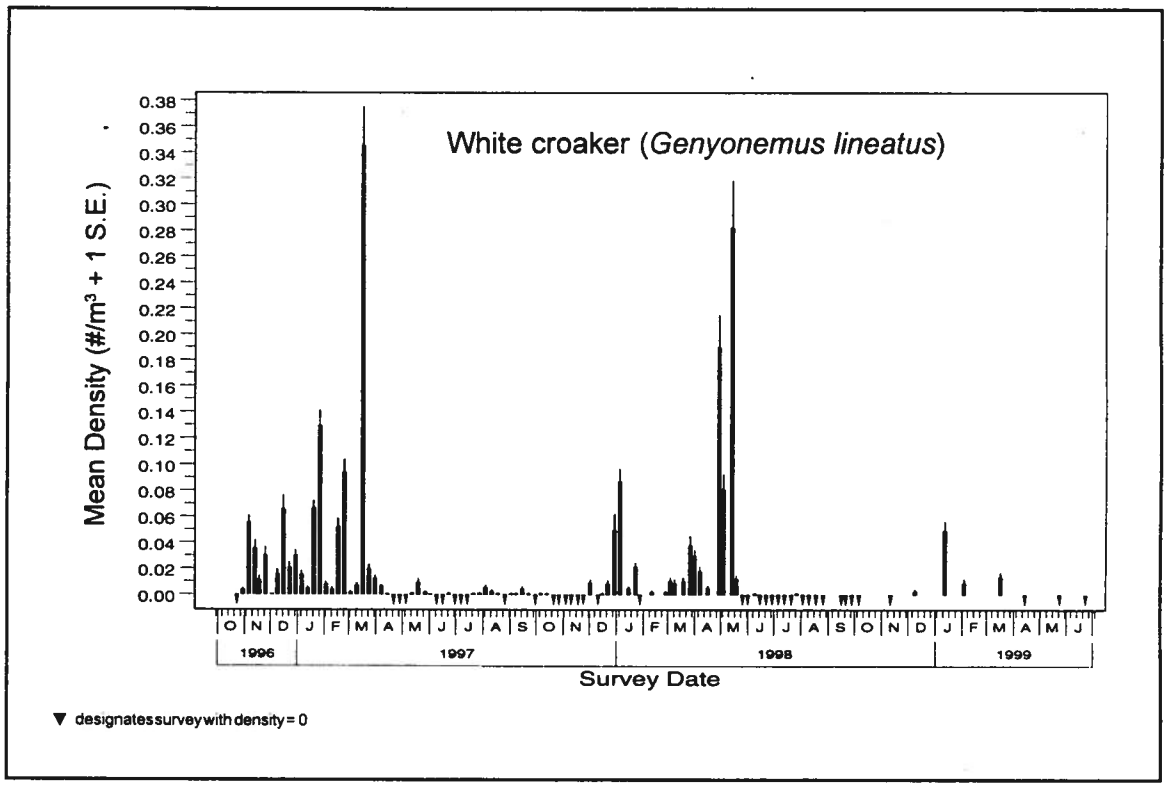


Figure 5.2.7-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

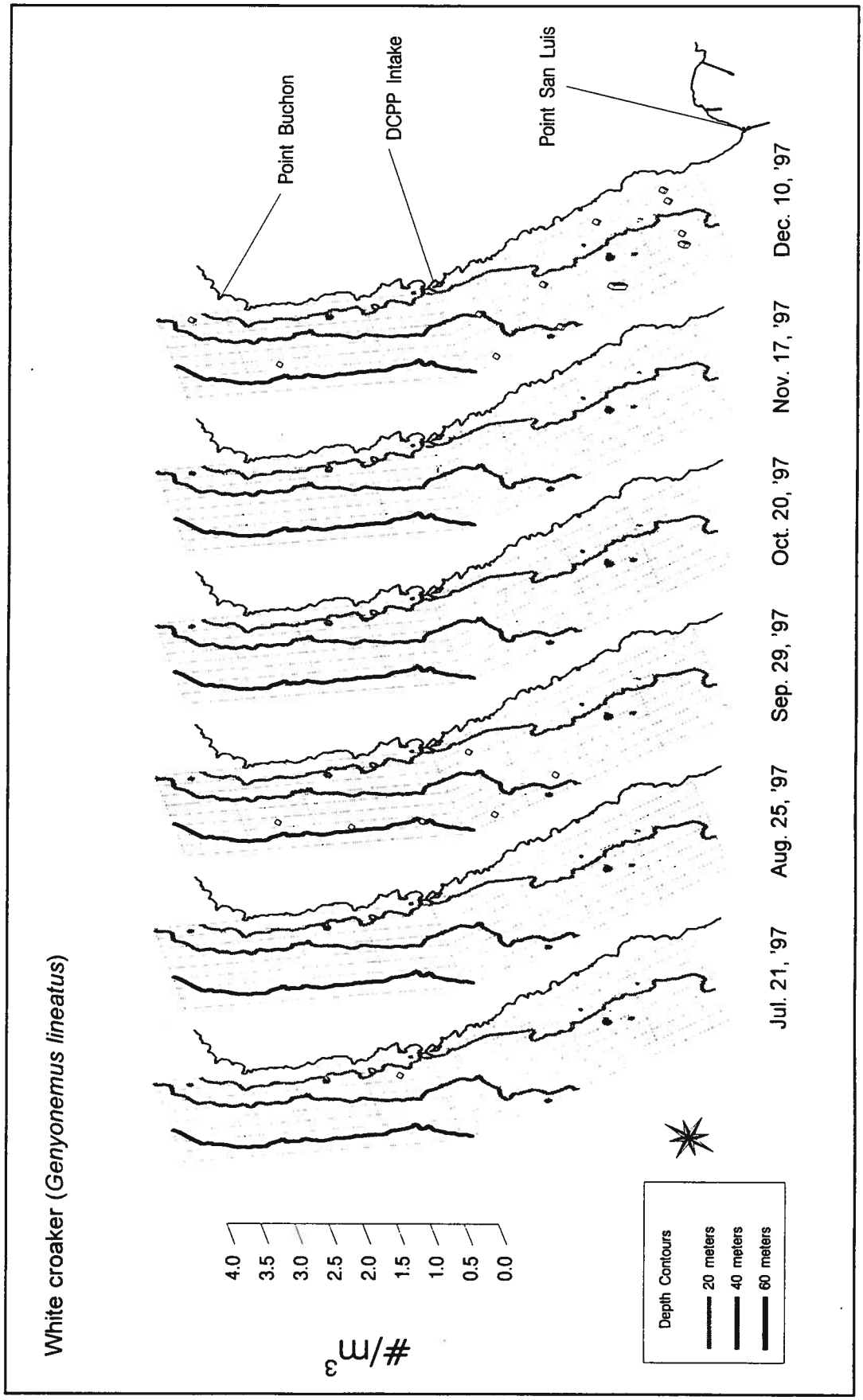


Figure 5.2.7-2a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

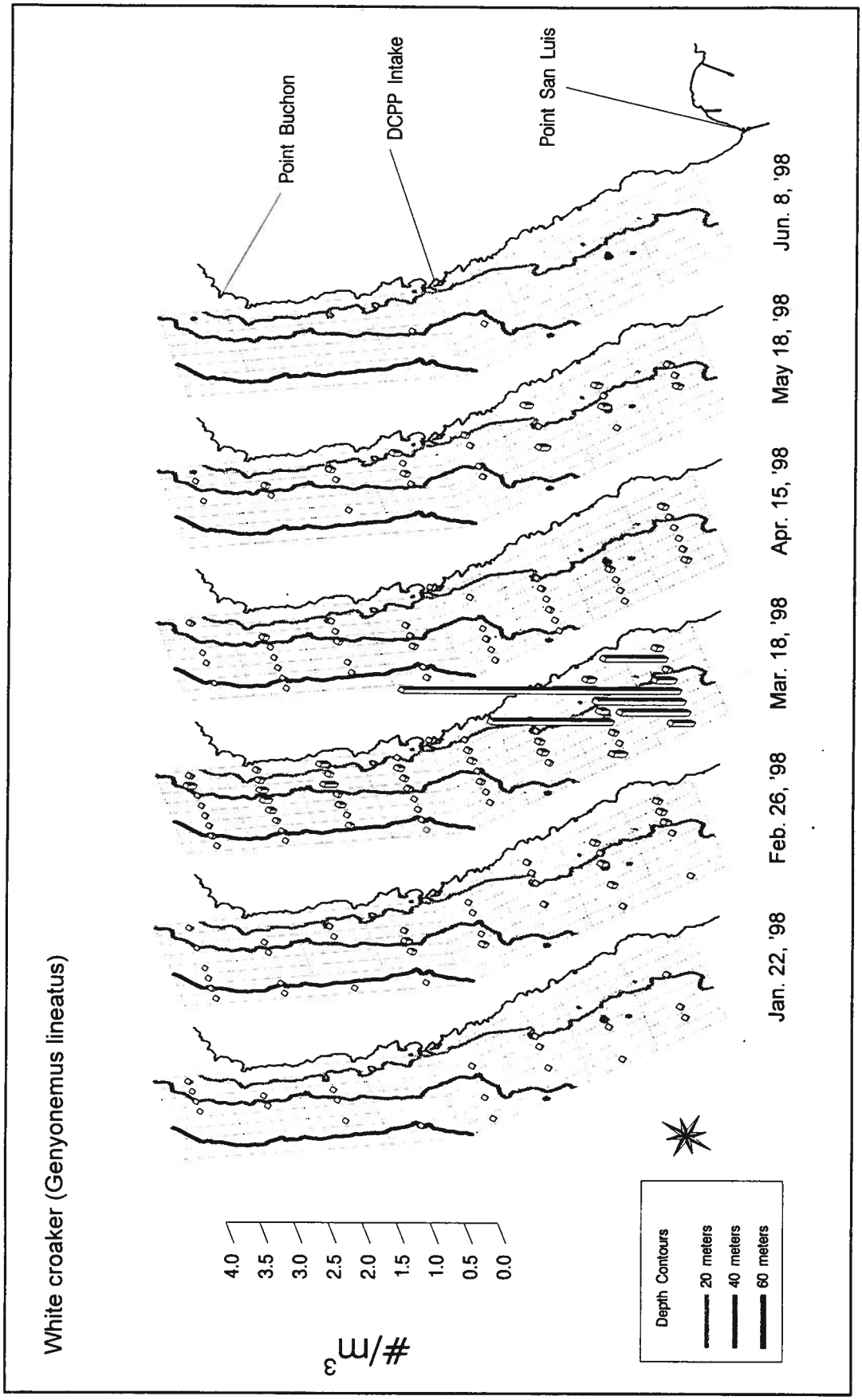


Figure 5.2.7-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

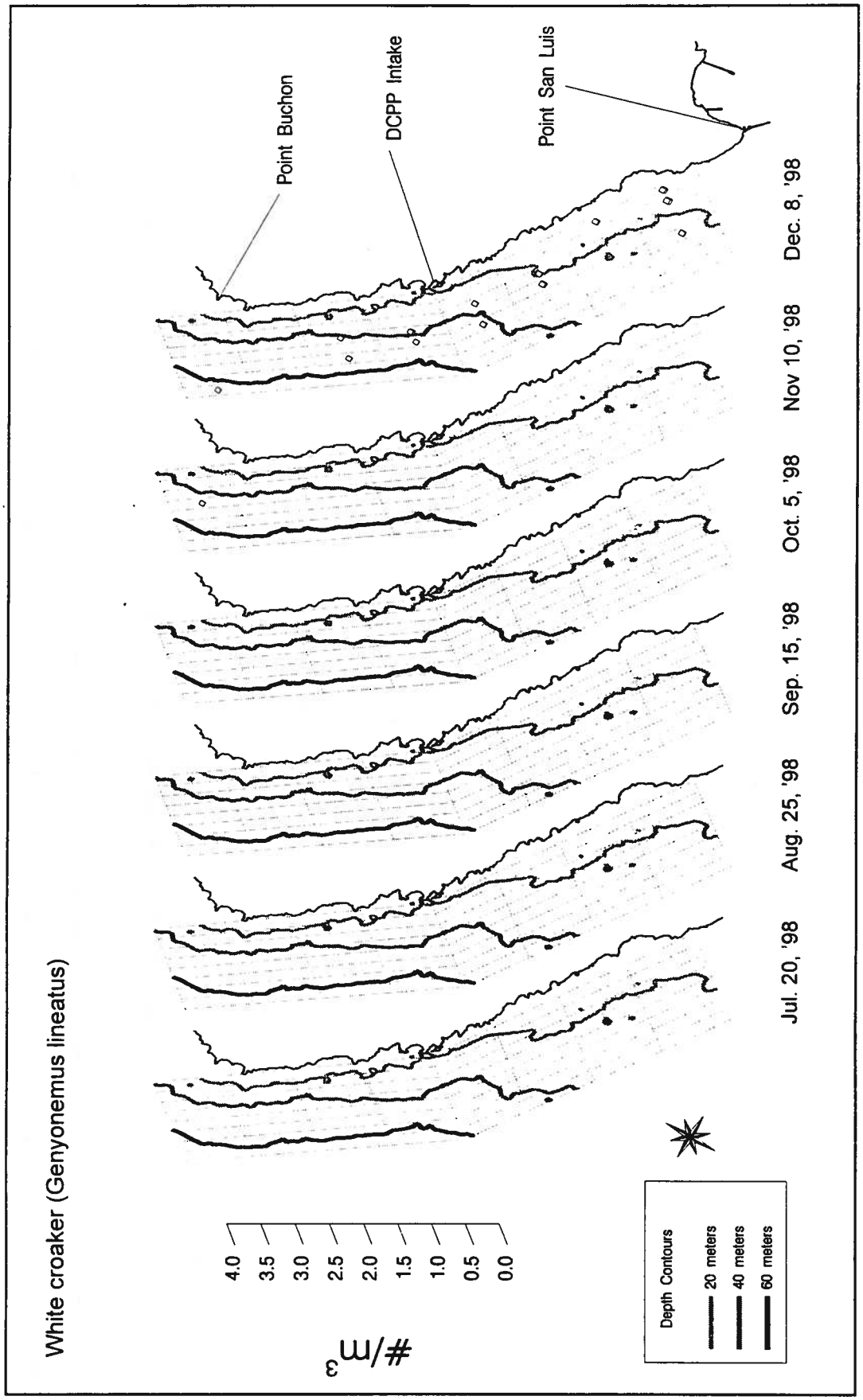


Figure 5.2.7-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

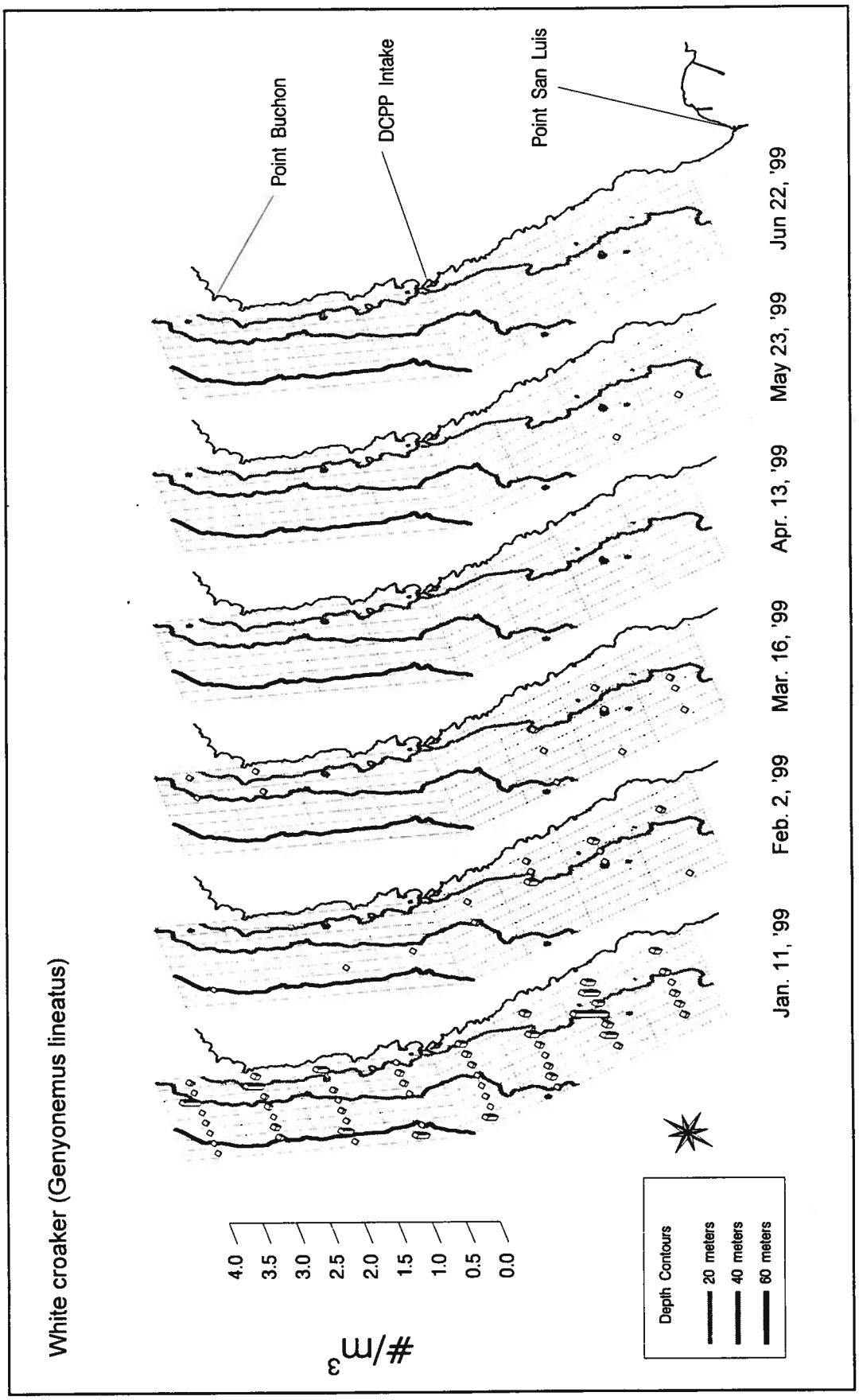


Figure 5.2.7-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

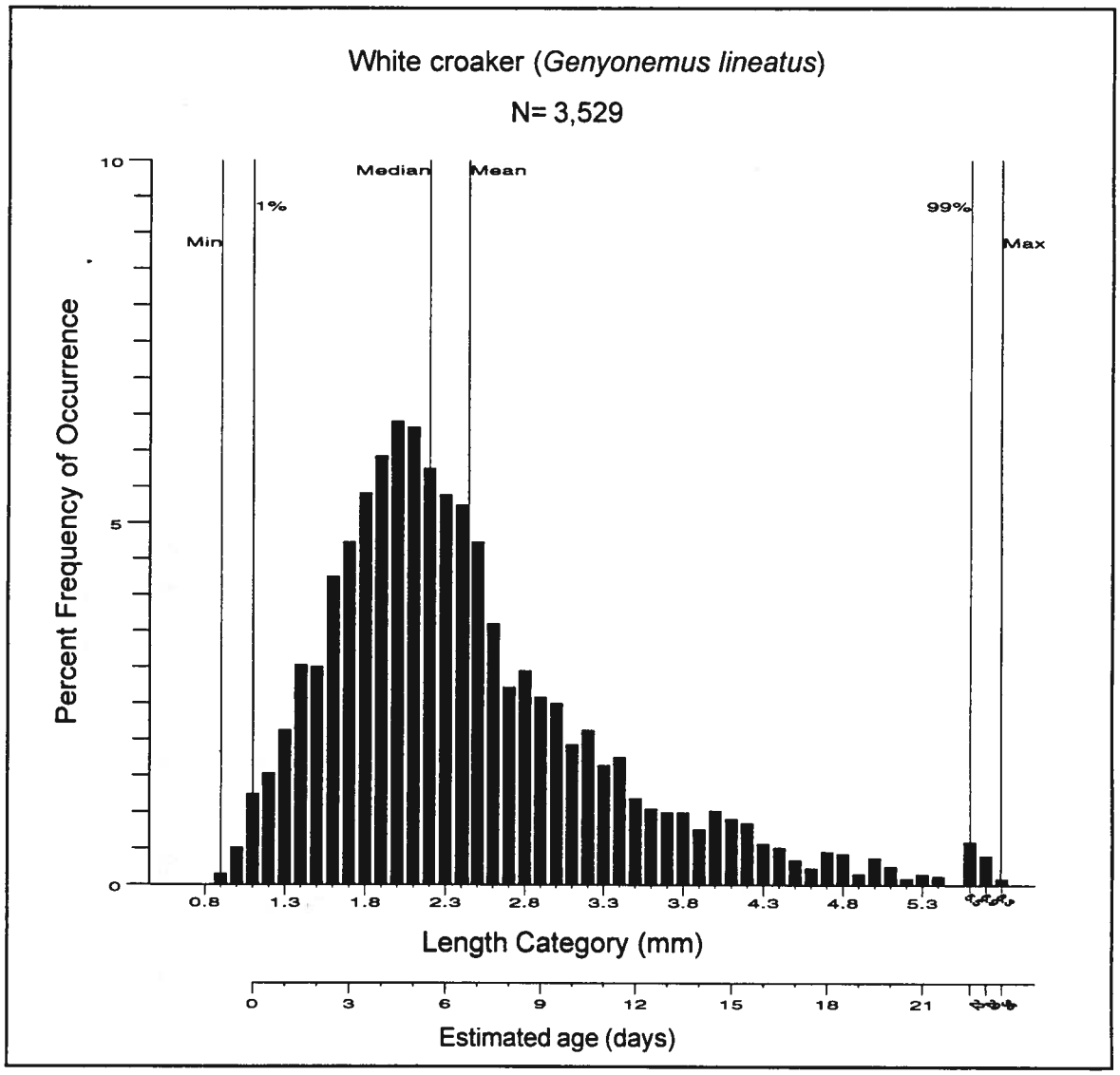


Figure 5.2.7-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

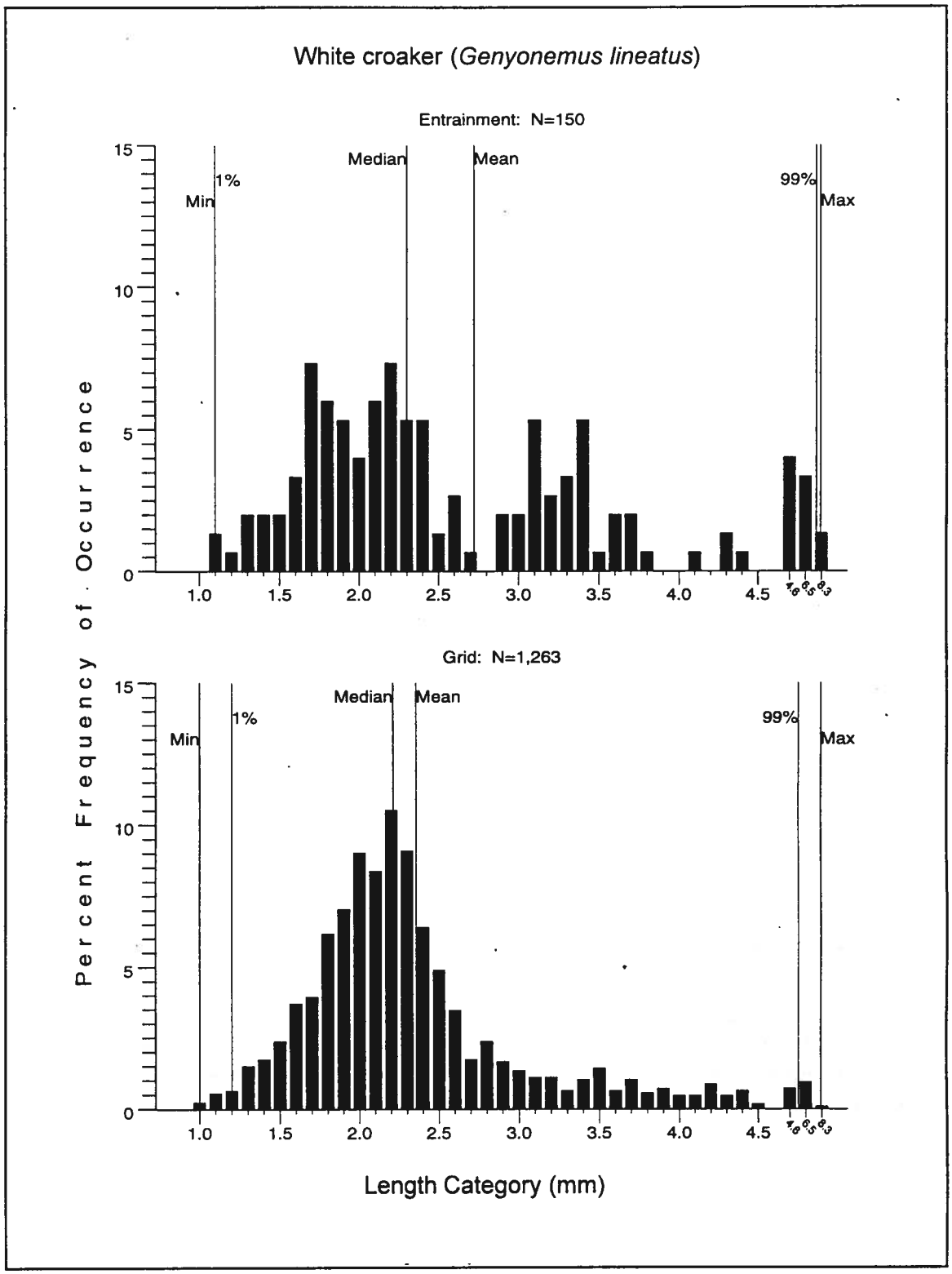


Figure 5.2.7-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

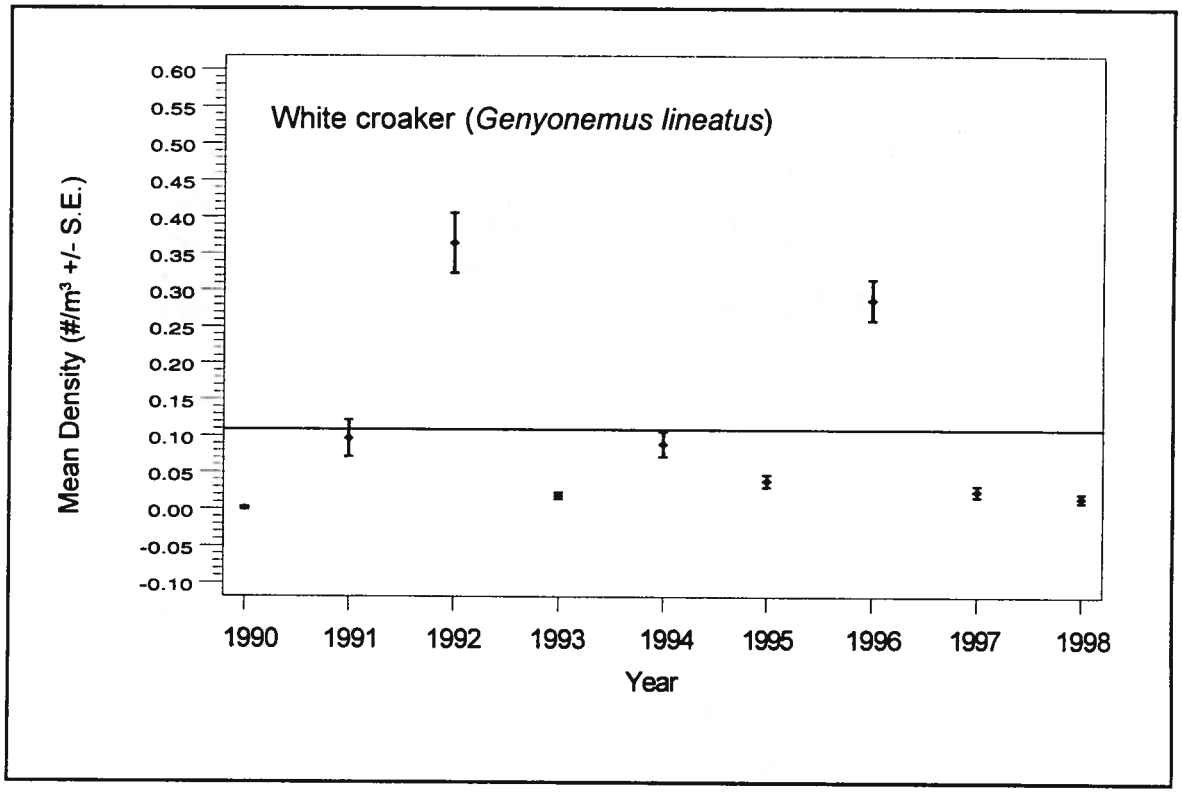


Figure 5.2.7-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

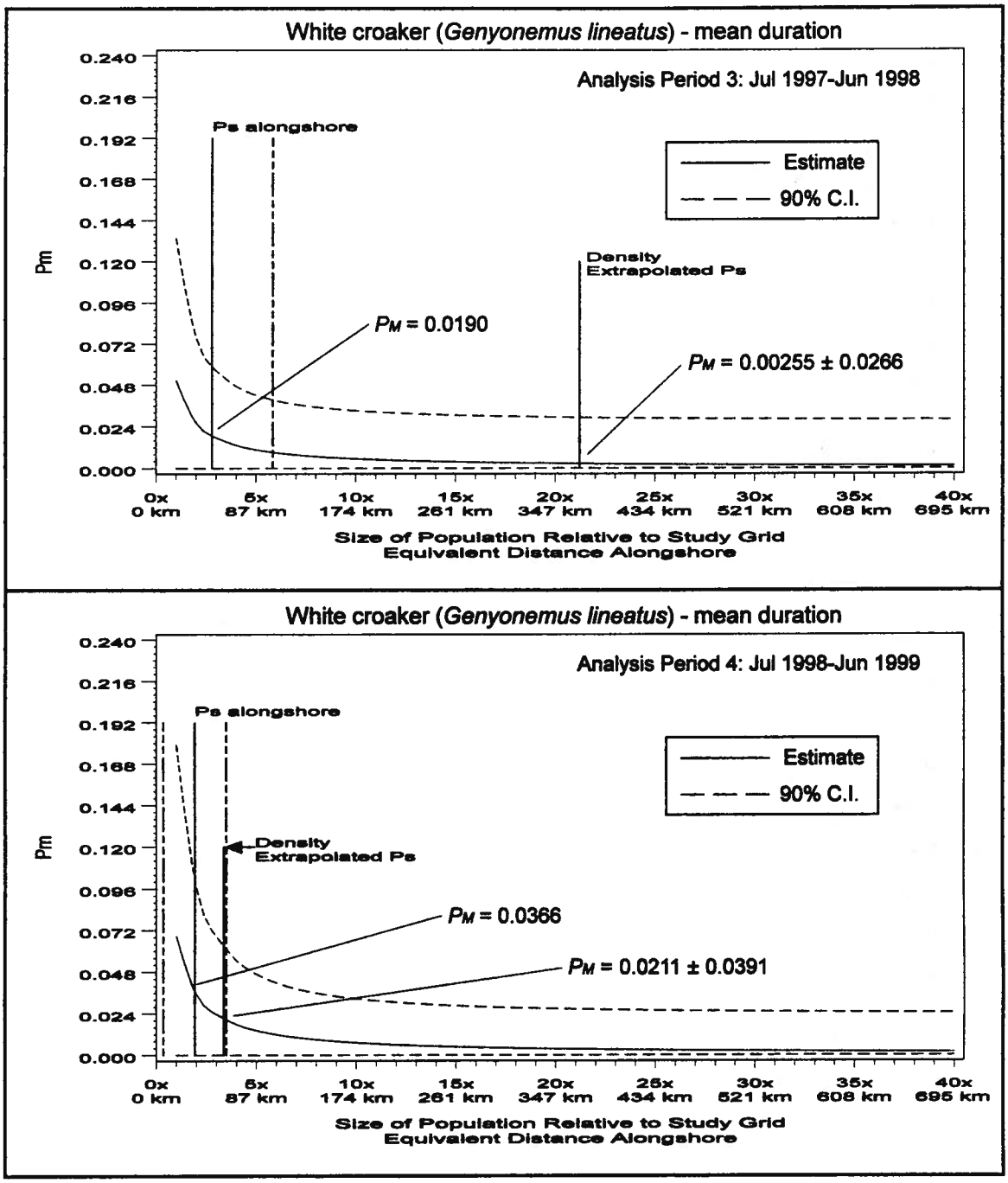


Figure 5.2.7-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

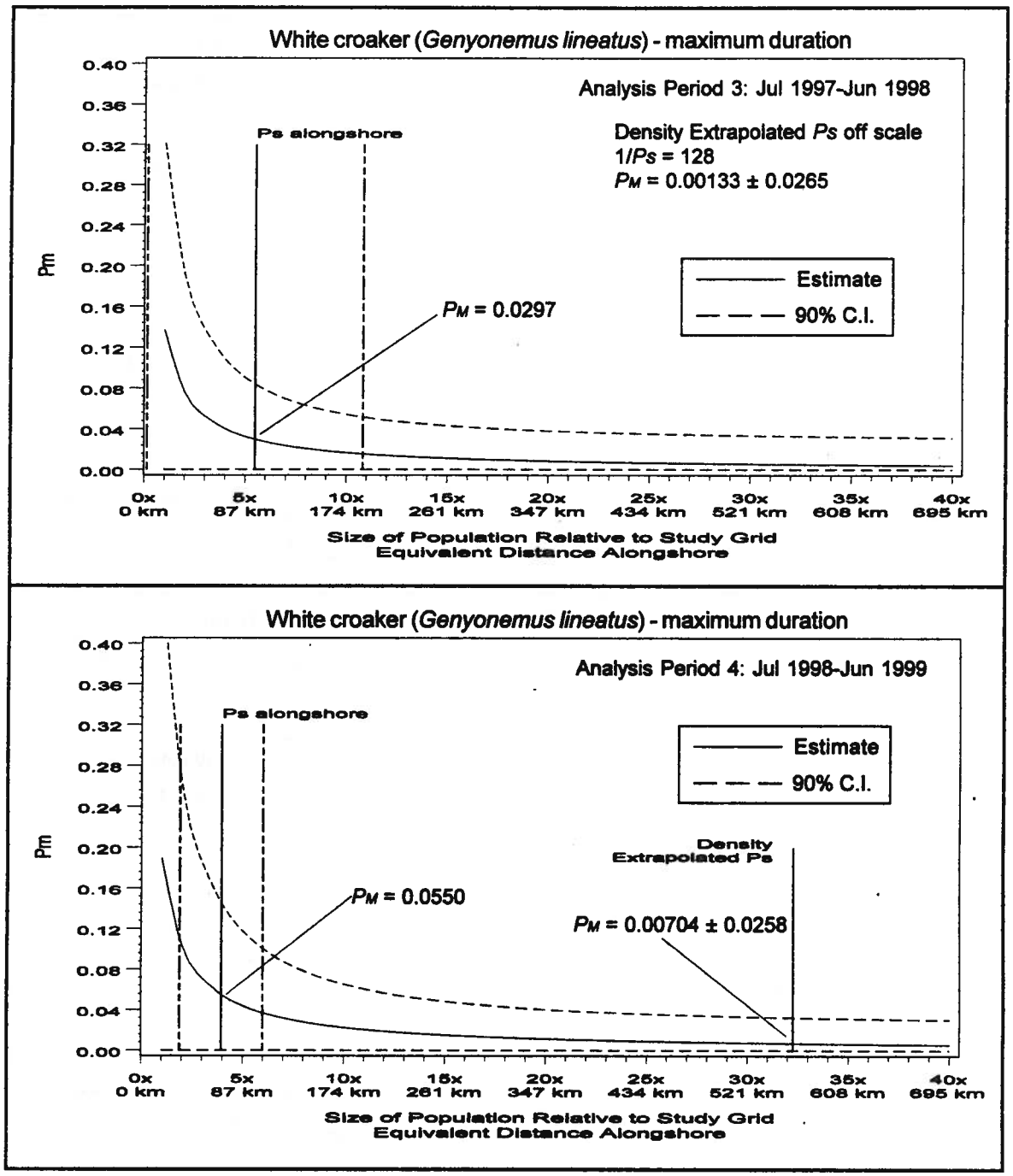


Figure 5.2.7-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

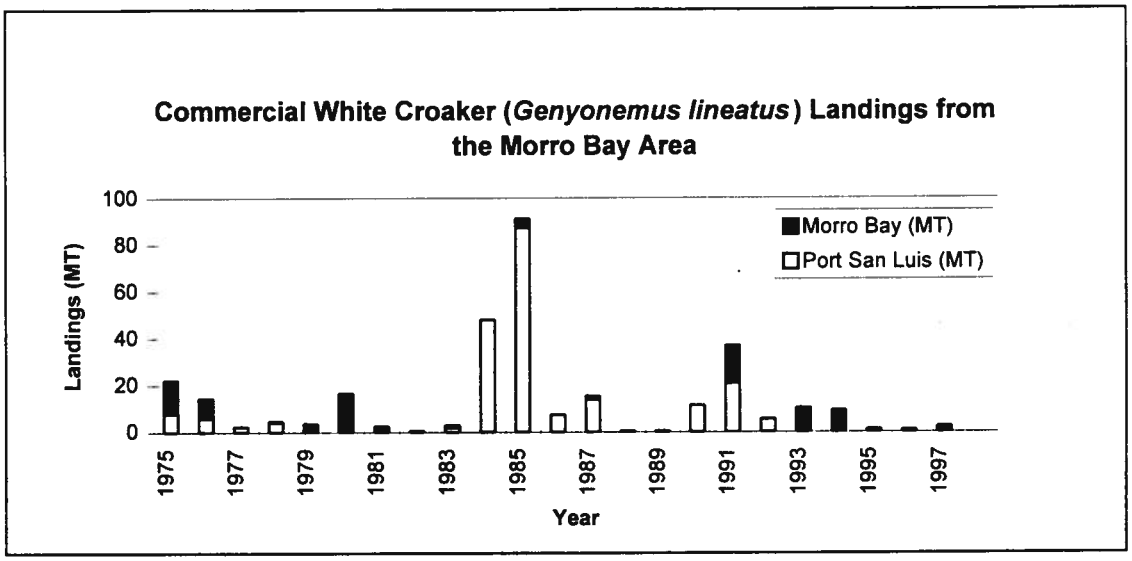


Figure 5.2.7-7. Annual commercial landings (MT) of white croaker (*Genyonemus lineatus*) at ports in the Morro Bay area (source: California Department of Fish and Game Database).

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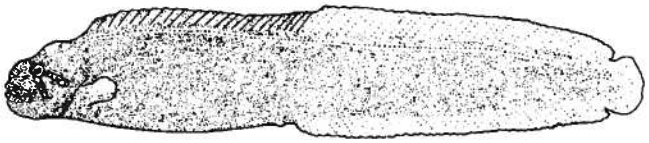
5.2.8 Assessment of Monkeyface Prickleback (*Cebidichthys violaceus*)



Cebidichthys violaceus (Girard 1854); monkeyface prickleback; length to 76 cm; southern Oregon to Bahia San Quintin, north-central Baja California; intertidal to 24 m; uniform black, olive, or gray, except for black streaks at eye (Miller and Lea 1972; Eschmeyer et al. 1983).

The monkeyface prickleback (*Cebidichthys violaceus*) is a member of the family Stichaeidae, which is primarily distributed in the northern Pacific Ocean (Nelson 1994). The family consists of three subfamilies, 31 genera, and about 60 species (Nelson 1994). The monkeyface prickleback belongs to the subfamily Xiphisterinae, which contains 15 species, several of which are common to the California coast (Miller and Lea 1972; Nelson 1984). The monkeyface prickleback is found as far south as central Baja California, Mexico but is rare south of Point Conception (Burge and Schultz 1973; Love 1996). They are common in crevices and rocks from the upper intertidal to shallow rocky reefs at depths of about 24 m and appear to be highly territorial (Wang 1986; Love 1996). They are not commonly sighted subtidally because of their cryptic nature (Burge and Schultz 1973). In central California, juveniles are most abundant in water about 0.5 m above mean lower low water (MLLW) tidal height and commonly found under rocks at low tide (Love 1996). Investigations into the diet of this fish in Diablo Cove reveal it is an herbivore feeding largely on red algae (Burge and Schultz 1973).

Monkeyface prickleback lay demersal, adhesive eggs (Wang 1986; Fitch and Lavenberg 1971; Baxter 1974) and exhibit parental egg-guarding behavior (Fitch and Lavenberg 1971; Bane and Bane 1971). Spawning has been reported from January to May (Fitch and Lavenberg 1971; Baxter 1974; Wang 1986; Love 1996). In Diablo Cove, females were full of eggs in January, contained small undeveloped eggs in July, and had spent ovaries in September (Burge and Schultz 1973). This evidence, coupled with the presence of young-of-the-year (YOY) in fall, suggests late summer spawning (Burge and Schultz 1973). Additionally, maturing ovaries were

observed from December through June, mature ovaries from March through May, and spent ovaries from February through August with one in December. Older fish appear to spawn earlier in the season than younger fish (Marshall and Wyllie Echeverria 1991).

Fecundity increases with age and length. Smaller fish produce 6,000 to 8,000 eggs (Fitch and Lavenberg 1971; Baxter 1974) while larger females produce more. A 41 cm SL female aged 7 yr produced 17,500 eggs, and a 61 cm SL female aged 11 yr had 46,000 eggs (Marshall and Wyllie Echeverria 1991).

Monkeyface prickleback larvae are planktonic (Wang 1986), but little else is known about the early life history of this species. A family member, the black prickleback (*Xiphister atropurpureus*), was shown to have marked positive phototaxis (attraction to light) for 3–5 d, after which time they become negatively phototactic (Peppar 1965 in Coleman 1992). There were no data on monkeyface prickleback larval growth rates in the literature. The larval growth rate (Stepien 1986) for an ecologically similar species, the giant kelpfish (*Heterostichus rostratus*, family Clinidae), was substituted for the missing *C. violaceus* growth rate. The accuracy of this growth rate as it applies to monkeyface prickleback is unknown, but it represents the best available data from the current scientific literature.

Longevity of the monkeyface prickleback was estimated at 18 yr from a 67 cm SL fish (Marshall and Wyllie Echeverria 1991). Females 20 cm in length from Monterey Bay, California were aged at 12 to 15 yr (Fitch and Lavenberg 1971). The oldest age estimated from fish collected in Diablo Cove was 14 yr based on two individuals that were approximately 49 cm SL and 64 cm SL (Burge and Schultz 1973). Considering this species reaches 76 cm SL (Eschmeyer et al. 1983), it probably lives longer. Fitch and Lavenberg (1971) reported the age at 50% maturity at 3–4 yr. However, Marshall and Wyllie Echeverria (1991) determined that age at first maturity is 4 yr, age at 50% maturity is 5 yr, and age at 100% maturity is 7 yr.

The fishery for monkeyface prickleback is largely recreational although some are sold commercially (Wang 1986; Love 1996). They are usually caught using a fishing method called

"poke-poling" among intertidal rocks. A small commercial fishery has been underway in the Morro Bay area since 1993 (California Department of Fish and Game Database).

5.2.8.1 Summary of Field Collections

Abundances of monkeyface prickleback larvae at the DCPP intake structure were seasonal during the years 1996–1999 (**Appendix H**), with highest densities occurring during the spring and summer (**Figure 5.2.8-1**). Few larvae of this species were collected during the months of August through December. It appears that the initiation of reproduction in monkeyface prickleback might have been delayed by a month or two during 1998. This might have been due to the El Niño event that began in 1997 (Lynn et al. 1998; NOAA 1999). There were 7,090 larval monkeyface prickleback identified from 1,253 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 27% of the intake subsamples collected and processed during that period.

Monkeyface prickleback larvae were generally present in the DCPP study grid during January through July of each year (**Appendix H**), with their highest abundance occurring during May 1998 and April–May 1999 (**Figure 5.2.8-2**). They were distributed in higher density in the cells closest to shore in the southern half of the grid. There were 904 larval monkeyface prickleback identified from 294 bongo net subsamples representing 10% of the study grid subsamples collected and processed from July 1997–June 1999.

Standard lengths of all monkeyface prickleback larvae collected at the DCPP intake structure between October 1996 and June 1999 ranged from a minimum of 2.8 mm to a maximum of 24.6 mm; corresponding age estimates ranged from zero days (just hatched) to about 88 d (**Figure 5.2.8-3**). The growth rate of larval giant kelpfish (0.25 mm/d) reported by Stepien (1986) was used to estimate ages of these monkeyface prickleback larvae. The central 98% of this length-frequency distribution resulted in minimum and maximum lengths for the analyses of 5.7 mm and 12 mm, respectively. The mean larval length in this distribution was 7.33 mm, which corresponded to an estimated age of 6.5 d. We assumed that larvae shorter than the minimum length of the central 98% of the distribution had just hatched and were aged zero days. Subsequent ages were estimated

from this assumed hatch length. No reported hatching length for monkeyface prickleback was found in the scientific literature. Reported hatching size for bluebarred prickleback (*Plectobranchus evides)* is <7.4 mm (Moser 1996). Individuals that appeared to be different ages based on length might be the same age as a result of natural variation in hatching size or variable shrinkage after preservation. Thus, it is possible that at least some of the smaller-sized larvae, at least up to 5.7 mm and possibly larger, had just hatched before being caught.

Distributions of monkeyface prickleback larval lengths at the DCPP intake structure and in the study grid were compared using a Kolmogorov-Smirnov test (**Figure 5.2.8-4**). The test did not detect a significant difference between the two distributions ($p > 0.288$). A *t*-test also did not detect a significant difference ($p > 0.75$) between the mean lengths from the two locations. On average, these larvae were estimated to be about 6.5 d old, indicating that the power plant entrains relatively young monkeyface prickleback larvae. In addition, these data indicate that the age and size structure of the population entrained is nearly identical to that of the population at risk.

5.2.8.2 Estimating Total Annual Entrainment

The annual estimates of the average entrainment for 1996–97 Analysis Period 1 ($\widehat{E_T}$ = 106,000,000) and 1997–98 Analysis Period 2 ($\widehat{E_T}$ = 119,000,000) were similar (**Table 5.2.8-1**). The small differences in the estimated entrainment numbers are not statistically significant because 95% confidence intervals approximated by $\pm 2 \cdot \widehat{SE}(\widehat{E_T})$ would not overlap.

Annual entrainment estimates of monkeyface prickleback larvae, adjusted to the long-term average from nine years of Intake Cove plankton sampling (**Figure 5.2.8-5**), ranged from 61,500,000 ($\widehat{SE}$ =5,580,000) in Analysis Period 2 to 83,100,000 ($\widehat{SE}$ =8,160,000) in Analysis Period 1 (**Table 5.2.8-2**). The adjustment index ($\bar{I} / I_i$) was calculated as the ratio between the nine year average ($\bar{I}$) of Intake Cove sampling and the average annual index estimated from these same tows during the year being adjusted (I_i). The average index for the years 1997 and 1998 (I_i) were 0.019 and 0.029 larvae/m^3, respectively, while the long-term average index ($\bar{I}$) was 0.0148 larvae/m^3 for the years 1990–1998, yielding the ratios $\bar{I} / I_i$ of 0.783 for 1997 and

0.517 for 1998. The differences between the adjusted estimates were not statistically significant because 95% confidence intervals approximated by $\pm 2 \cdot \widehat{SE}(\hat{E}_{Adj-T})$ did not overlap.

NOTE: Entrainment estimates for monkeyface prickleback larvae were not used to calculate annual adult losses (*FH* or *AEL*) because there were insufficient demographic data with which to parameterize these approaches.

5.2.8.3 Empirical Transport Model (*ETM*)

Monthly estimates of proportional entrainment (PE_i) in the surveys from the two analysis periods ranged from a minimum of zero to a maximum of 0.859±0.464 ($\pm 1\widehat{SE}(\widehat{PE_i})$) in January 1999 (**Table 5.2.8-3**). Periods when both $\widehat{PE_i}$ and the annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) were equal to zero indicated that no larvae were collected at either the DCPP intake or from the study grid. When $\widehat{PE_i} = 0$ but $\hat{f}_i > 0$, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. The largest values of $\hat{f}_i$ occurred between April and June in both analysis periods indicating that these were periods of high monkeyface prickleback larval abundance at the intake structure. These surveys were also associated with the largest values of $\widehat{PE_i}$ except for the estimate in January 1999. Since $\hat{f}_i$ was used to weight individual estimates of PE_i, those with higher $\hat{f}_i$ values were proportionally more important to the annual entrainment mortality estimates.

Larval monkeyface prickleback were distributed close to shore in the study grid subsamples (**Figure 5.2.8.2**) and similar to observed adult distributions in shallow nearshore areas up to depths of 25 m (Miller and Lea 1972). Therefore, P_S was only calculated using alongshore current movement and not offshore extrapolation of study grid abundance. *ETM* results for monkeyface prickleback show the differences in how estimated larval durations can affect calculation of $\widehat{P_M}$ (**Figures 5.2.8-6a** and **b**). As expected, values for $\widehat{P_M}$ increase with increasing values of $\widehat{P_S}$ (i.e., decreasing $1/\widehat{P_S}$) for shorter larval duration between periods. The probability of entrainment is less for the shorter duration because of the reduced time period for larval transport to occur. The difference between years for $\widehat{P_M}$ is much less for the shorter larval

duration than the longer duration (0.11 to 0.16 vs. 0.11 to 0.23). The larger confidence intervals around $\widehat{P_S}$ for the longer larval duration reflect the larger uncertainty that results from increased alongshore transport. The point estimates for $\widehat{P_M}$ for both larval durations are greater during 1997–1998 because of weaker alongshore currents associated with El Niño conditions.

5.2.8.4 Interpretation of Assessment Results

A limited fishery exists for monkeyface prickleback. Monkeyface prickleback landings were first recorded in 1993 at Port San Luis and Morro Bay, peaked in 1995, and declined through 1998 (**Figure 5.2.8-7**). Landings ranged from a low of 3 kg at Port San Luis in 1993 to a high of 180 kg at Morro Bay in 1995. The average combined annual catch between 1993 and 1998 from the Morro Bay area was 75 kg/yr. Because of the limited fishery for this species, there have been no estimates of stock size or adult density. Therefore, the catch data will not be used to compare harvest mortality rates to $\widehat{P_M}$.

The highest annual densities of monkeyface prickleback larvae from Intake Cove surface plankton tows occurred in 1990. Abundances in 1997 and 1998 were slightly above the long-term mean density (**Figure 5.2.8-5**). Adult monkeyface prickleback abundance was low and variable on the transects surveyed during the DCPP Receiving Water Monitoring Program fish observations from 1978 to 1998 (**Figure 5.2.8-7**). Monkeyface prickleback have a relatively long lifespan that has been estimated at up to 18 yr from a 67 cm SL fish (Marshall and Wyllie Echeverria 1991). Results of the *ETM* show that the power plant may annually entrain 11–23% of the monkeyface prickleback larvae from an area 7 to 8 times the area of the study grid (**Figure 5.2.8-6**). The wide range in the size of larvae entrained (<3 mm to >24 mm) indicate that monkeyface prickleback larvae are vulnerable to entrainment over a large portion of their larval life (**Figure 5.2.8-3**). This could indicate an increased potential for impacts from entrainment on this taxon. In the absence of other estimates of adult population abundance, there is no definitive way to determine whether this level of larval mortality is sufficient to cause a long-term local population decline. However, this species is widely distributed from southern Oregon to Baja,

and it is unlikely that larval losses from entrainment that may affect local adult abundances would affect the larger population.

Table 5.2.8-1. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for monkeyface prickleback (*Cebidichthys violaceus*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sept 30, 1997	106,000,000	10,400,000
2) Oct 1, 1997–Sep 30, 1998	119,000,000	10,800,000
3) Jul 1, 1997–Jun 30, 1998	116,000,000	10,700,000

Table 5.2.8-2. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for monkeyface prickleback (*Cebidichthys violaceus*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	83,100,000	8,160,000
2) Oct 1, 1997–Sep 30, 1998	61,500,000	5,580,000
3) Jul 1, 1997–Jun 30, 1998	60,200,000	5,520,000

Table 5.2.8-3. Monkeyface prickleback (*Cebidichthys violaceus*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.0247	0.00723	0.0437	0.00248
Aug 25, 1997	0	0	0	0
Sep 29, 1997	0	0	0	0
Oct 20, 1997	0	0	0	0
Nov 17, 1997	0	0	0	0
Dec 10, 1997	0	0	0	0
Jan 22, 1998	0	0	0.00320	0.000510
Feb 26, 1998	0.00621	0.00369	0.0122	0.00101
Mar 18, 1998	0.0217	0.0108	0.0379	0.00490
Apr 15, 1998	0.139	0.0481	0.403	0.0164
May 18, 1998	0.0135	0.00301	0.188	0.00981
Jun 8, 1998	0.0542	0.0123	0.313	0.0126

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.0108	0.00972	0.0580	0.00483
Aug 26, 1998	0	0	0	0
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0	0	0	0
Jan 12, 1999	0.859	0.464	0.0116	0.00114
Feb 3, 1999	0.0384	0.0245	0.0389	0.00348
Mar 17, 1999	0.0223	0.00636	0.119	0.00538
Apr 14, 1999	0.0401	0.0146	0.434	0.0140
May 24, 1999	0.0155	0.00440	0.178	0.00729
Jun 23, 1999	0.0615	0.0170	0.161	0.00667

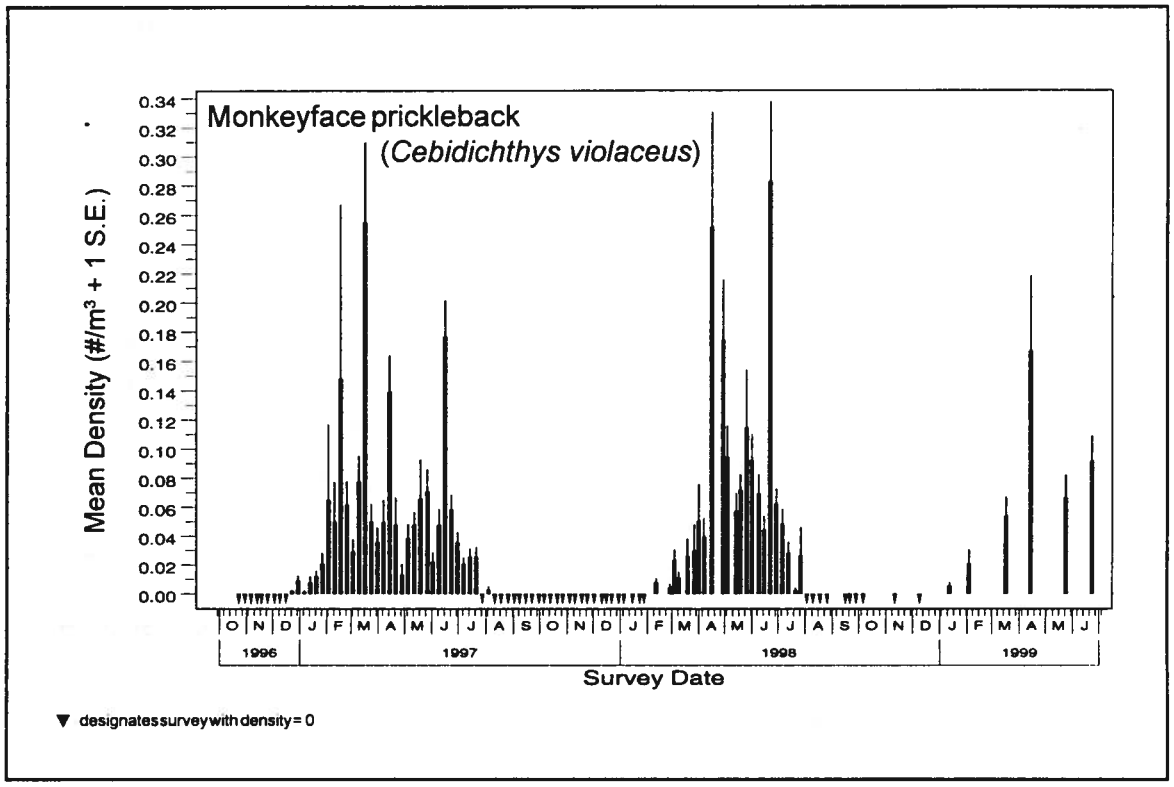


Figure 5.2.8-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

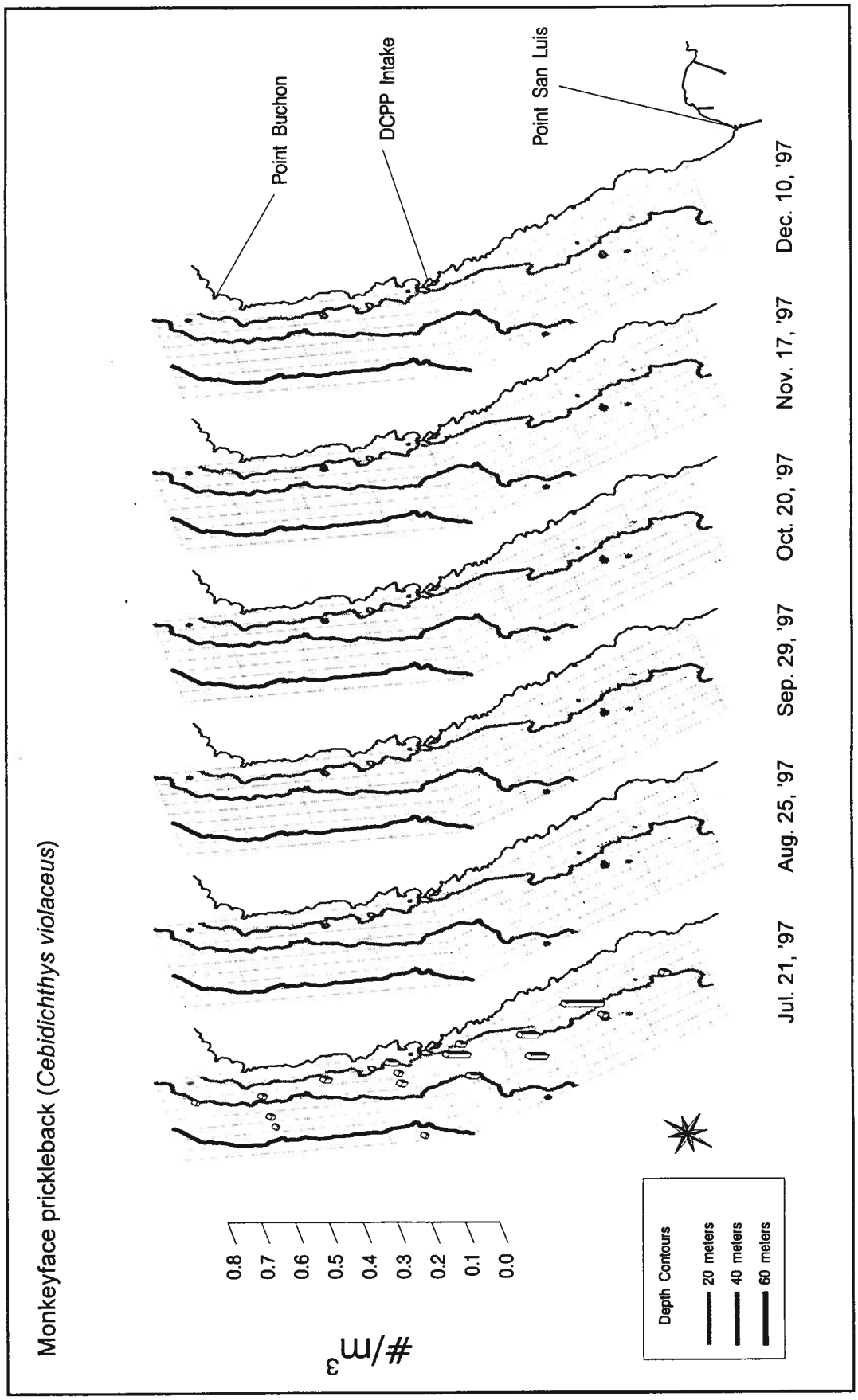


Figure 5.2.8-2a. Mean larval density (#/m³) collected in the DCPP study grid cells and at the entrainment sampling location.

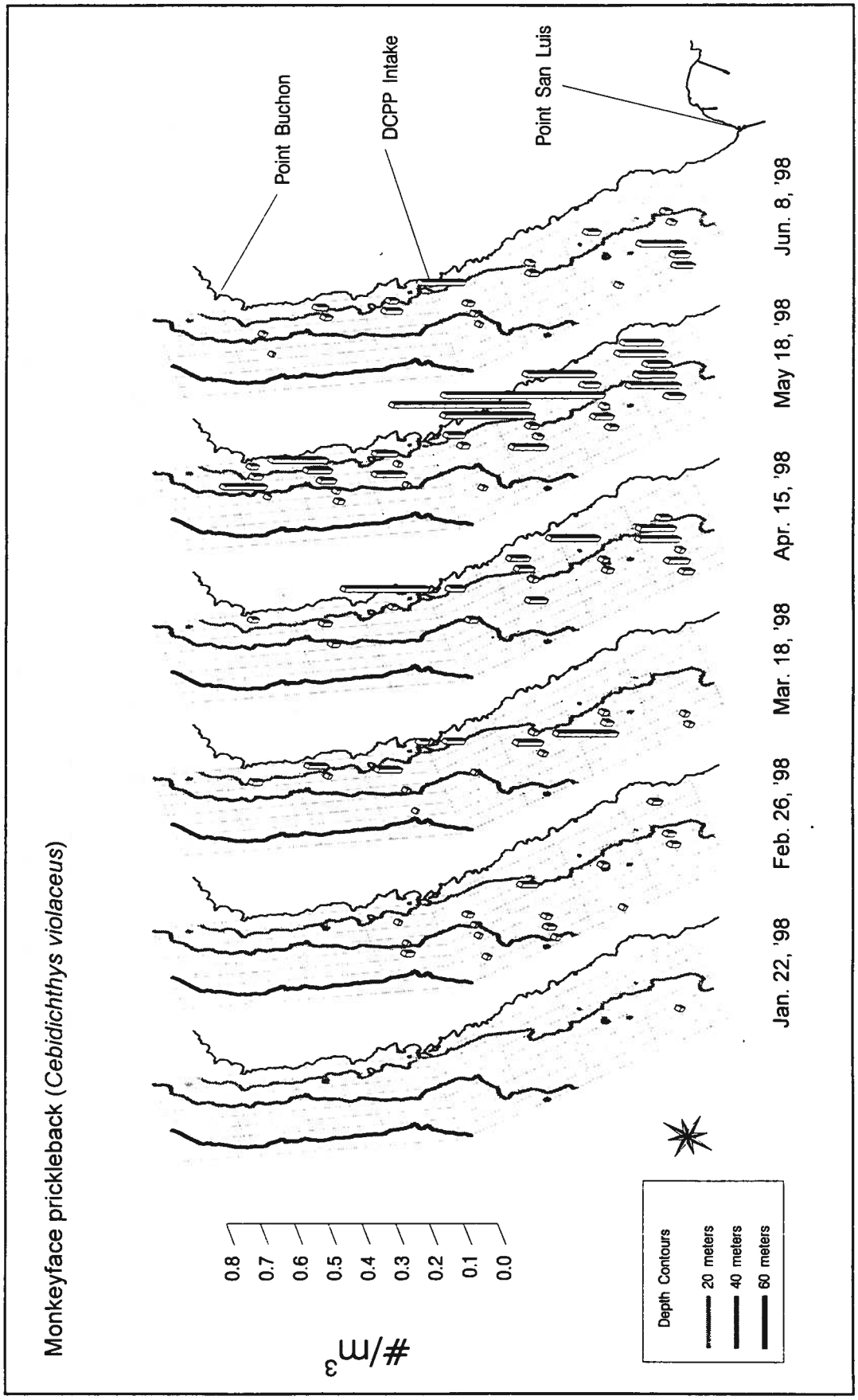


Figure 5.2.8-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

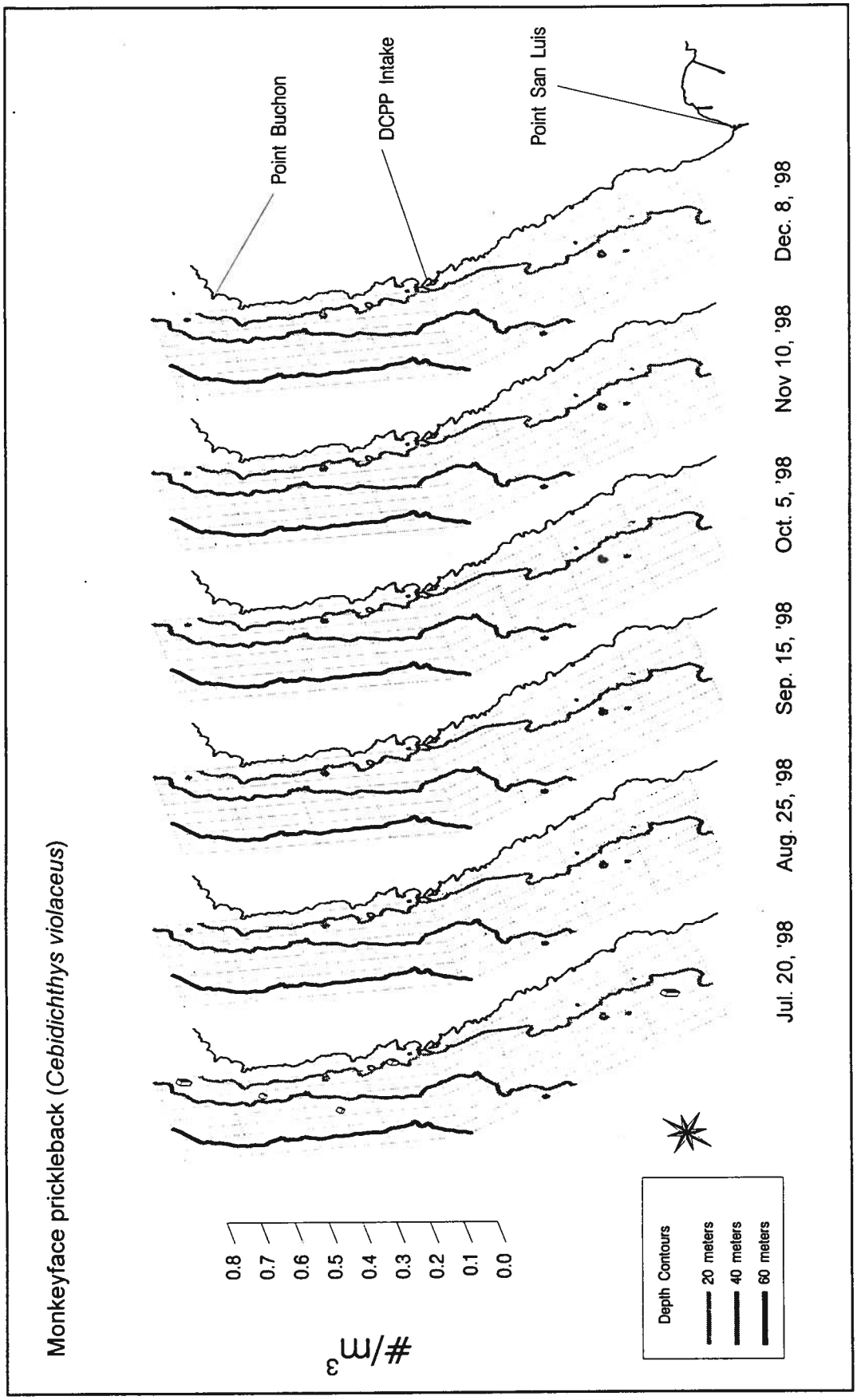


Figure 5.2.8-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

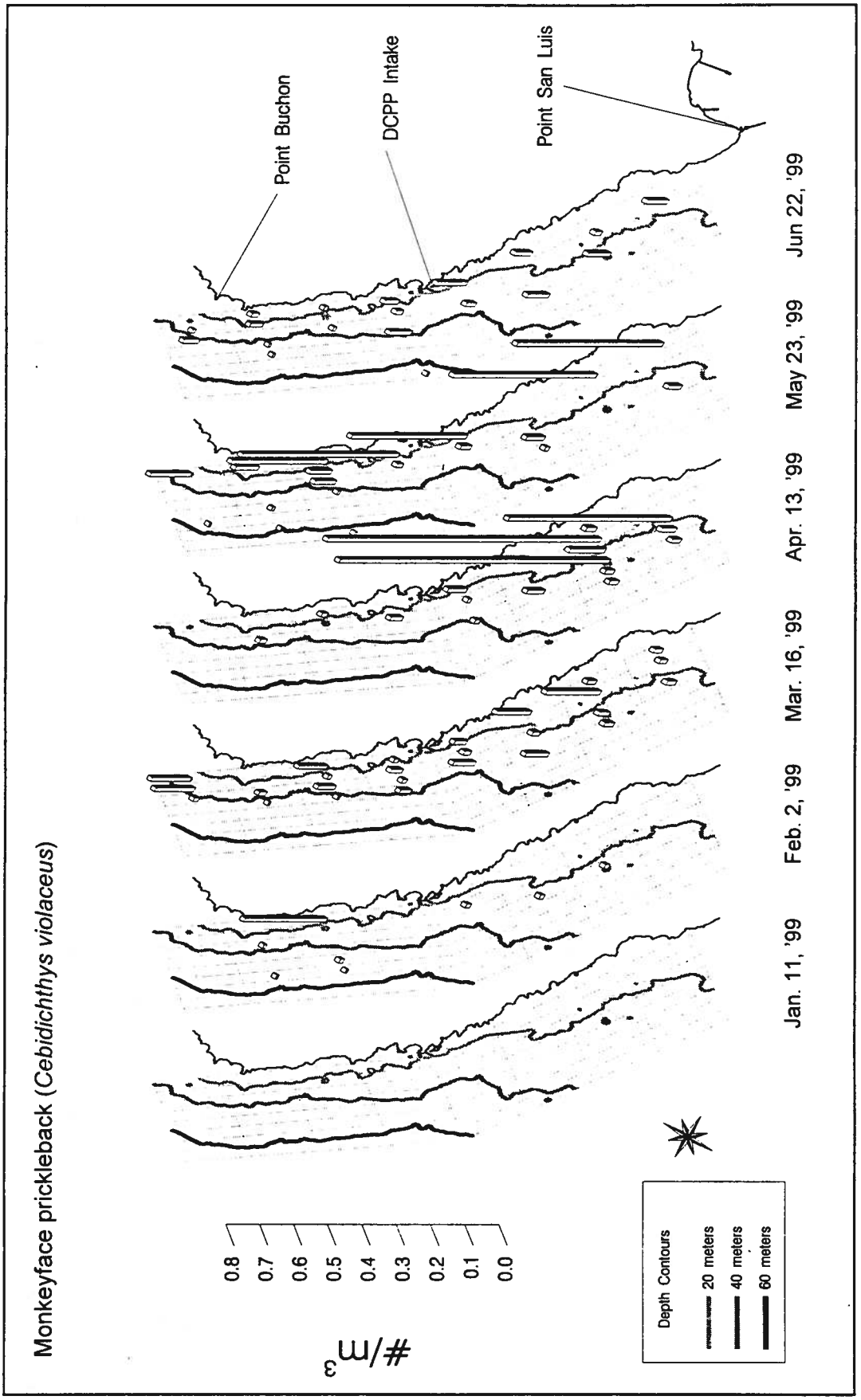


Figure 5.2.8-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

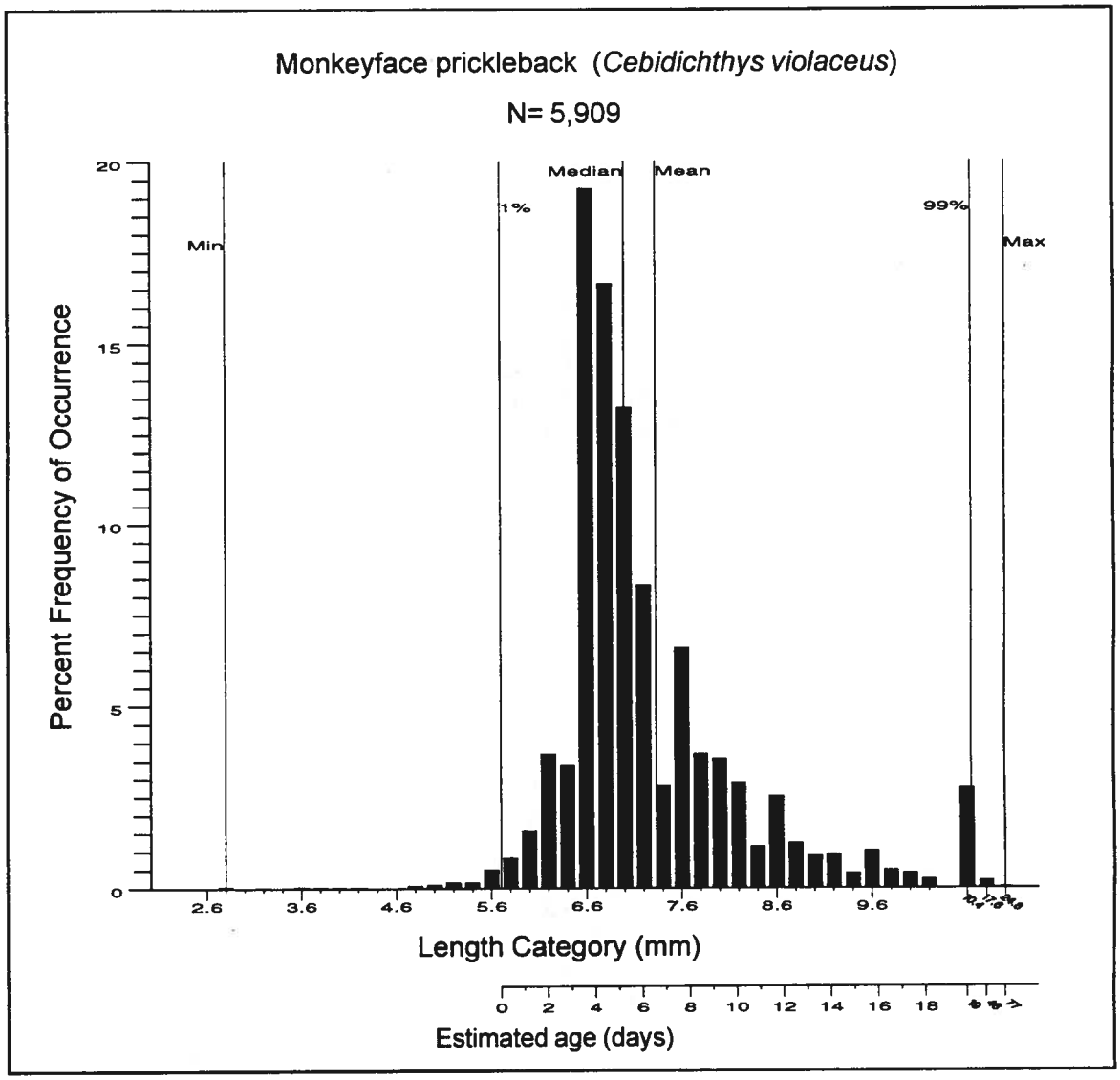


Figure 5.2.8-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

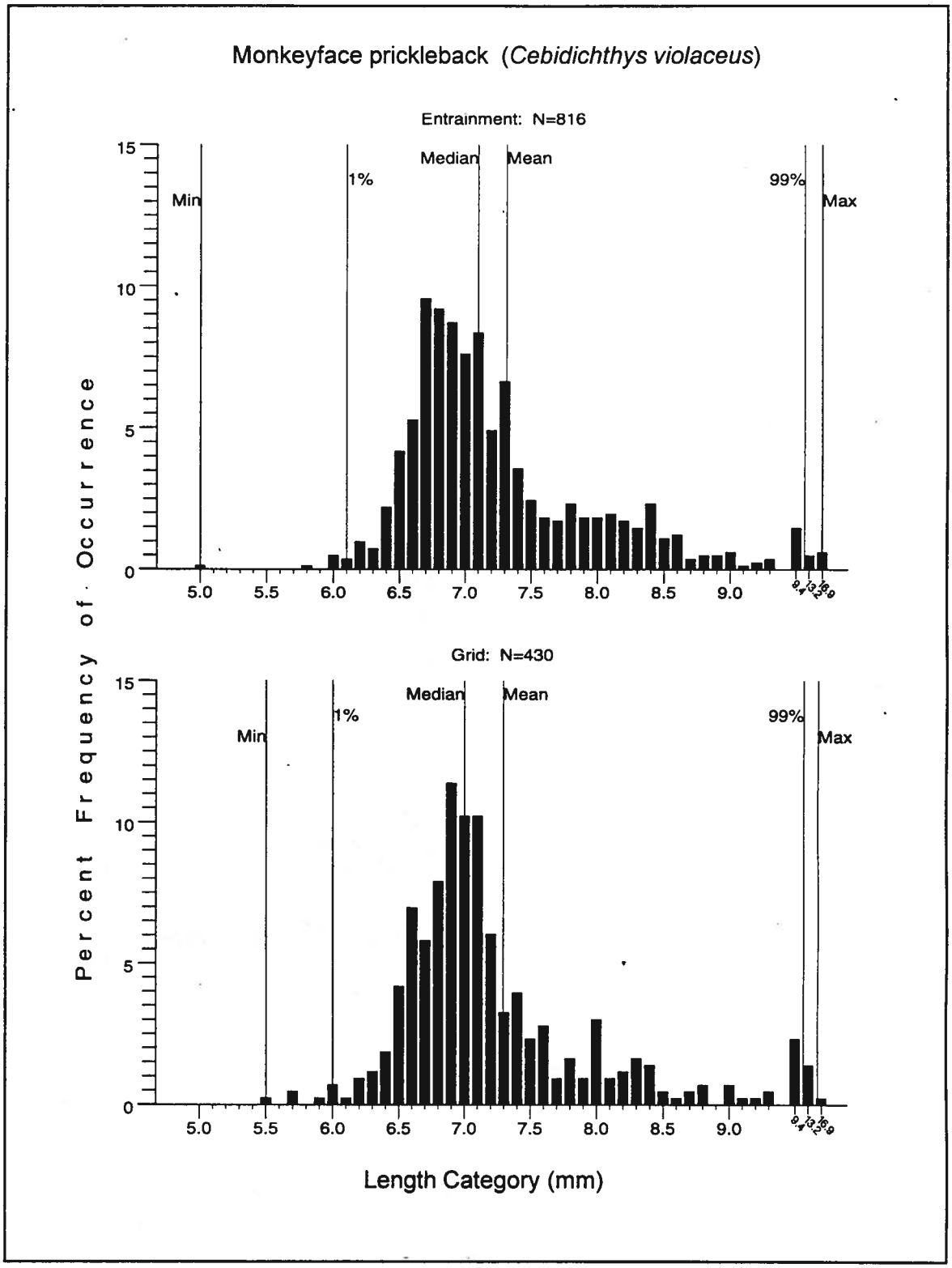


Figure 5.2.8-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

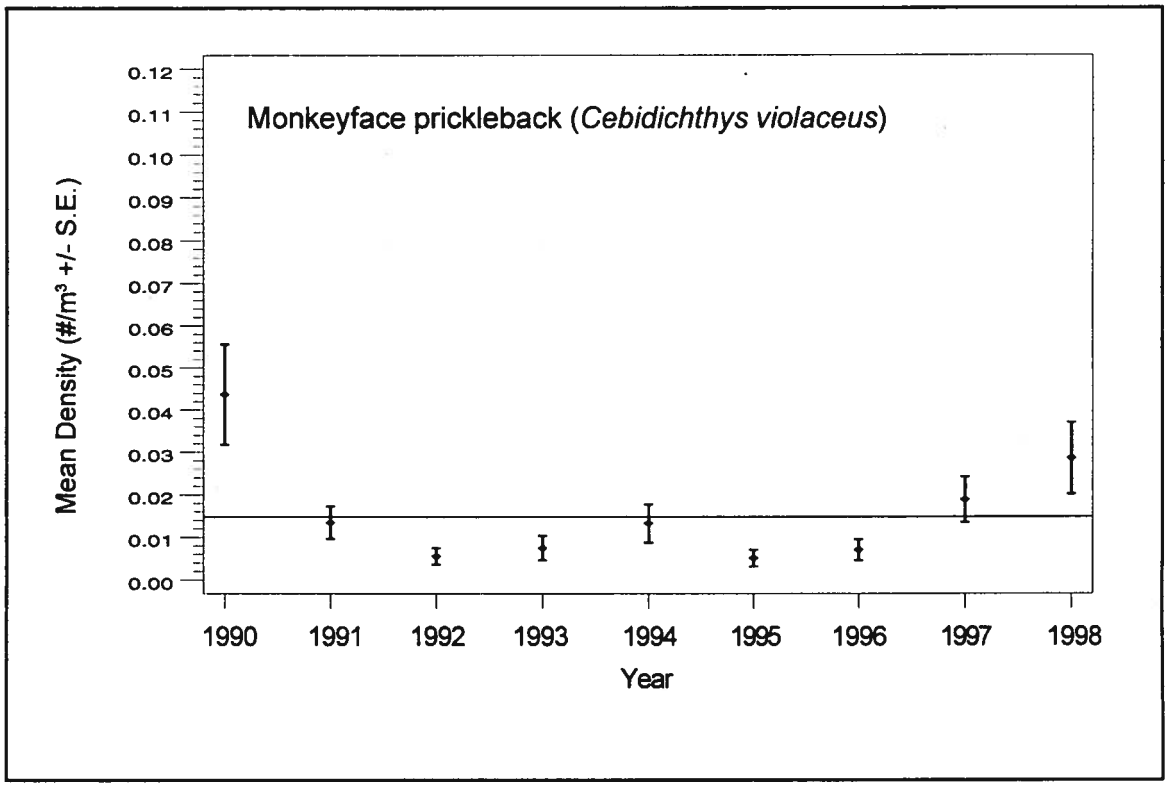


Figure 5.2.8-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

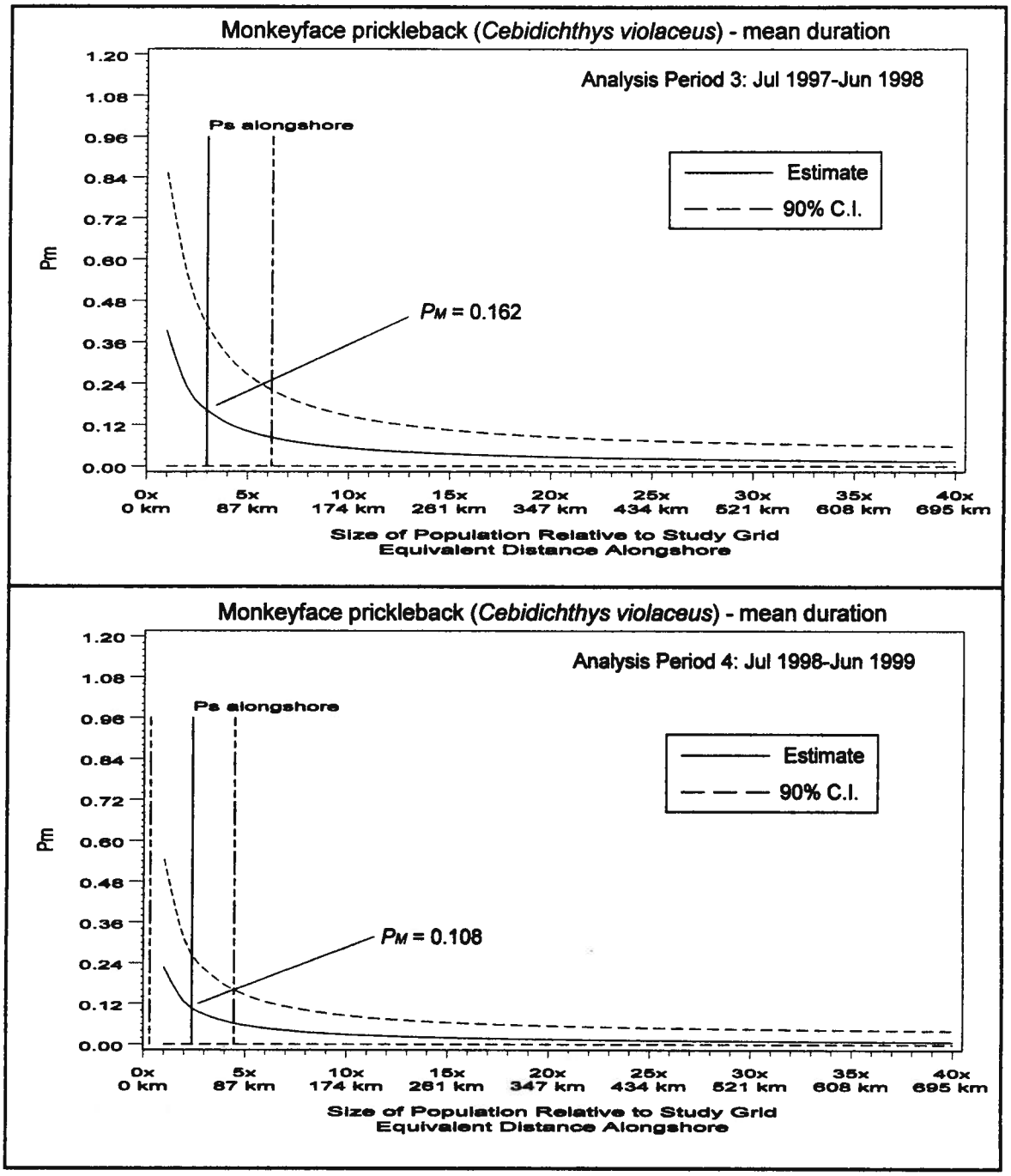


Figure 5.2.8-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements with the 90% C.I. indicated.

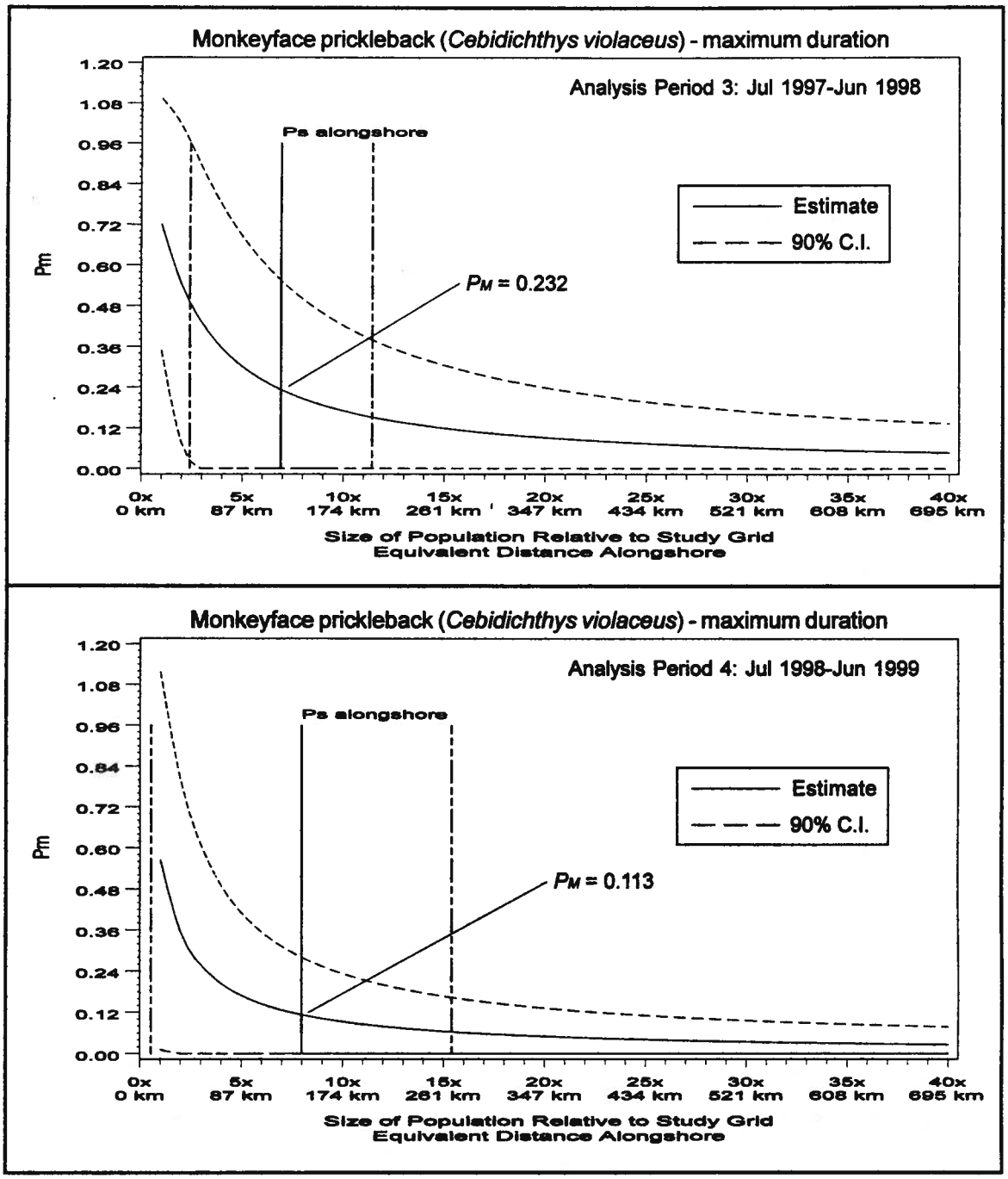
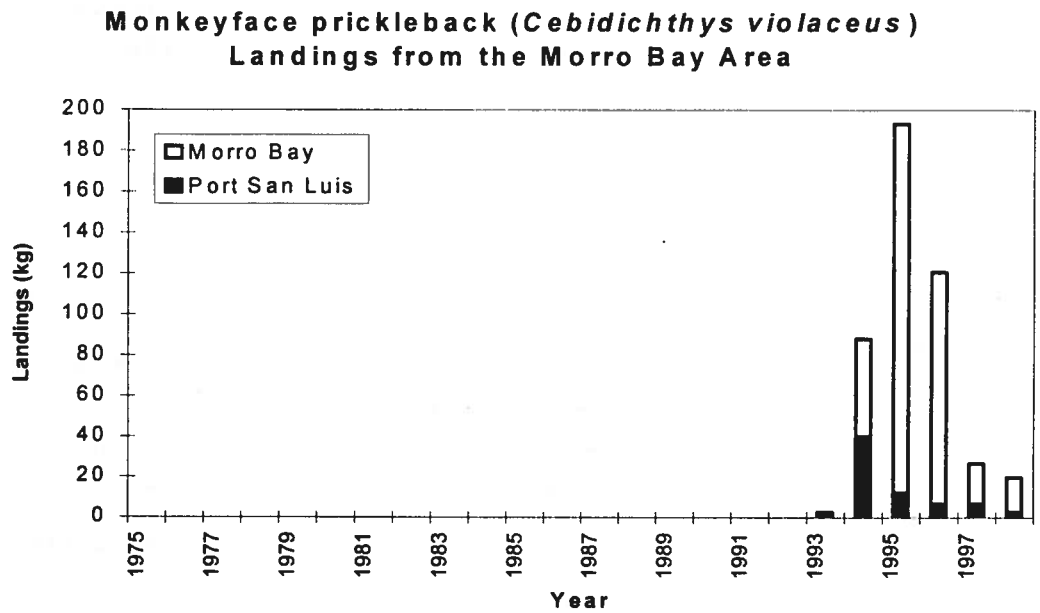


Figure 5.2.8-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements with the 90% C.I. indicated.

a)



b)

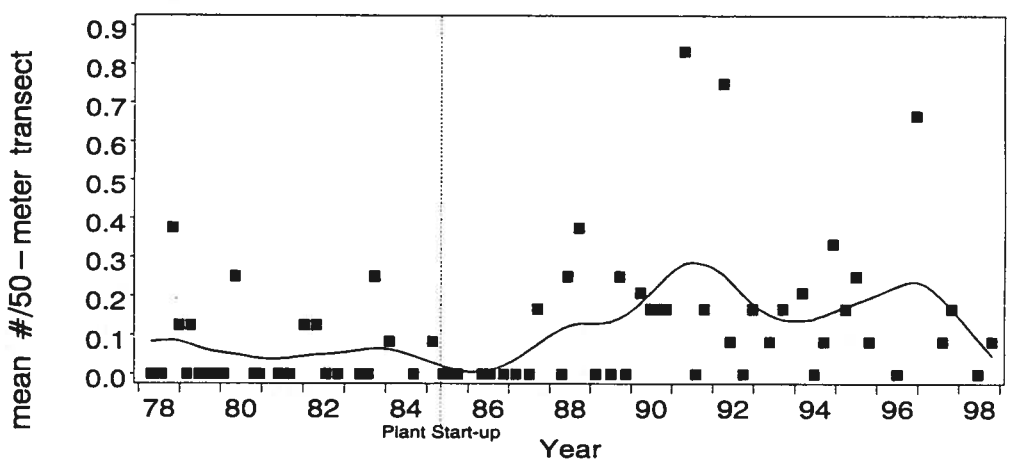
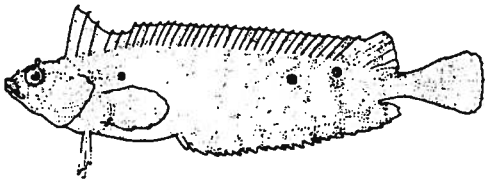


Figure 5.2.8-7. Yearly abundance of monkeyface prickleback measured by two independent methods: a) Commercial landings of adults from Morro Bay area, b) Mean number of fish observed per 50-meter subtidal benthic transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to draw the curve through the points.

5.2.9 Assessment of Clinid Kelpfishes (Clinidae)



Gibbonsia elegans (Cooper 1864); spotted kelpfish; length to 16 cm; Piedras Blancas Pt., central California to Bahia Magdalena, southern Baja California, including Isla Guadalupe; to 56 m (Miller and Lea 1972; Eschmeyer et al. 1983); green to brown or tan or reddish, often blotched or streaked (Eschmeyer et al. 1983).

Gibbonsia metzi Hubbs 1927; striped kelpfish; length to 24 cm; Vancouver Is., British Columbia to Punta Rompiente, central Baja California; intertidal to 9.1 m (Miller and Lea 1972; Eschmeyer et al. 1983); reddish to light brown with stripes or darker mottling (Eschmeyer et al. 1983).

Gibbonsia montereyensis Hubbs 1927; crevice kelpfish; length to 15 cm; British Columbia to Rio Santo Tomas, northern Baja California. 15–37 m. (Miller and Lea 1972; Eschmeyer et al. 1983); reddish to brown or lavender, plain colored to spotted or striped (Eschmeyer et al. 1983). *G. erythra* is a synonym (Stepien and Rosenblatt 1991).

Heterostichus rostratus Girard 1854; giant kelpfish; length to 61 cm; British Columbia to Cabo San Lucas, southern Baja California, including Isla Guadalupe; intertidal to 40 m (Miller and Lea 1972; Eschmeyer et al. 1983); giant kelpfish come in as many as three color forms (red, brown, or green) usually associated with the plant habitats from which they were collected (Stepien 1987).

In the family Clinidae there are three species of kelpfishes in the genus *Gibbonsia* and the giant kelpfish (*Heterostichus rostratus*) that occur along the west coast of North America from Baja to British Columbia (George and Springer 1980). Nelson (1994) indicates that worldwide there are three groups of clinids with about 20 genera and 73 species. Clinids are small (generally <25 cm SL; exception *H. rostratus* ca. 60 cm SL), primarily demersal residents of nearshore rocky reefs and kelp and seaweed beds in temperate marine waters (Lamb and Edgell 1986; Moser 1996).

There are no catch statistics for these species because they are not commercially or recreationally harvested.

All four representatives of this family in central California are oviparous (Nelson 1994), probably spawning demersal adhesive eggs (Fitch and Lavenberg 1971; Moser 1996) although Bane and Bane (1971) report striped kelpfish (*Gibbonsia metzi*) as having pelagic eggs. Larval forms are pelagic and are only identifiable to genus in the case of *Gibbonsia* spp. although giant kelpfish can be identified to species at most life stages. Yolk-sac duration in this latter species appears to be 2–3 d (Shiogaki and Dotsu 1972). Bane and Bane (1971) report a fecundity of around 2,300 eggs/female for a spotted kelpfish (*G. elegans*). Kelpfishes of *Gibbonsia* spp. first spawn at 2 yr, may spawn more than once per year, and live to around 7 yr (Fitch and Lavenberg 1975).

There is very little demographic information relating to the early life stages of clinids in general or specifically to the four representatives of the family found in central California. Growth of larval giant kelpfish was estimated by linear regression from data on lab-reared specimens (Stepien 1986) as 0.25 mm/d ± 0.013 (slope ± 1 SE). A similar estimate of growth rate is not available for *Gibbonsia* spp., but if the growth rate for giant kelpfish is substituted for *Gibbonsia*, it allows us to estimate larval ages at entrainment and calculate *ETM* for kelpfishes. There are no available data to indicate how well the larval growth rate of giant kelpfish describes the larval growth of *Gibbonsia* spp. Therefore, using giant kelpfish larval growth is considered only as an approximation to the growth of *Gibbonsia* spp. larvae. Additionally, there are no literature estimates of early life stage survivorship for any of the clinids treated here, so *FH* and *AEL* are not calculated for the clinid kelpfishes. The relatively few identified larval giant kelpfish (42 at the DCPP intake and one from the study grid) were incorporated into the clinid kelpfish group for the analyses.

5.2.9.1 Summary of Field Collections

Estimates of kelpfish larval abundance at the DCPP intake structure during the years 1996–99 indicate that species in this genus are continuous spawners (**Appendix H**) exhibiting at least two peaks in larval abundance per year. During the 1996–97 season, larval abundance at the intake

structure peaked in January and again in June–August (**Figure 5.2.9-1**). In the 1997–98 season, the January peak seen in the previous year did not occur, and the later peak appeared displaced by about one month and occurred in May–June. Although there are fewer processed subsamples from the 1998–99 spawning season, the peaks appear to conform with the pattern for the 1996–97 period. There were 9,361 larval kelpfishes identified from 2,808 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 60% of the subsamples collected and processed from the intake structure during that period.

Larval kelpfishes were relatively scarce in the DCPP study grid (**Appendix H**), but the temporal distribution of abundance roughly corresponded to that observed at the DCPP intake structure. There were 467 kelpfish larvae identified from 213 bongo net subsamples in study grid between July 1997–June 1999 representing 7% of the study grid subsamples collected and processed. Kelpfish larval abundance was generally distributed within the 20 m isobath, and larvae were likely concentrated in unsampled areas inshore of the study grid (Marliave 1986). The apparently higher density of this species in the southern portion of the study grid results from the relationship postulated with depth above and the fact that more of the area in the southern half of the grid is within the 0–20 m depth range.

Standard lengths of the *Gibbonsia* spp. larvae (7,331 larvae) measured from subsamples at the DCPP intake structure between October 1996 and June 1999 ranged from a minimum of 2.4 mm to a maximum of 23 mm (**Figure 5.2.9-3**). The minimum and maximum lengths, 4.1 mm and 12 mm, from the central 98% of this length-frequency distribution were used in calculating larval duration. A larval growth rate from giant kelpfish of 0.25 mm/d (Stepien 1986) was applied to the range of lengths to estimate the ages of the entrained larvae. The mean larval length in this distribution was 6.4 mm at approximately 9 d of age. We assumed that larvae shorter than the minimum length of the central 98% of the distribution were just hatched and aged zero days. Subsequent ages were estimated from this length and up. Other reported data for this species suggests that hatching length is approximately 4.5 mm (Moser 1996), indicating that the smallest larvae observed represent either natural variation in hatch lengths within the population or the phenomenon of shrinkage following preservation (Theilacker 1980). The possibility remains that

all larvae from the observed minimum length of 2.4 mm to the greatest reported hatching length of 4.5 mm (Moser 1996) could have just hatched, leading to overestimation of ages for all larvae <4.5 mm.

Distributions of *Gibbonsia* spp. larval lengths at the DCPP intake structure and in the study grid (**Figure 5.2.9-4**) were compared using a Kolmogorov-Smirnov test. The test detected a significant difference between the two distributions ($p \leq 0.001$). The length-frequency distributions in both sampling locations were unimodal, with a few larger individuals. A *t*-test did not detect a significant difference ($p > 0.786$) between mean lengths at the DCPP intake and in the study grid. The estimated age for an average length larva was about 9 d. These data indicate that entrained larvae were very young and that differences between age and length of the population entrained and the population at risk are very small.

5.2.9.2 Estimating Total Annual Entrainment

There was considerable variation of annual estimated mean entrainment of larval kelpfishes among years (**Table 5.2.9-1**). Approximately 186,000,000 ($\widehat{SE}$ =4,740,000) kelpfish larvae were entrained in Analysis Period 1 while entrainment estimates ranged from 83,600,000 ($\widehat{SE}$ =4,150,000) to 124,000,00 ($\widehat{SE}$ =5,000,000) between the two analysis periods in the 1997–98 sampling season. All periods appeared to be different from one another because 95% confidence intervals approximated by $2 \cdot \widehat{SE}(\widehat{E_T})$ did not overlap.

Annual estimates of the number of kelpfish larvae entrained, adjusted to the nine year average estimated from Intake Cove surface plankton tow sampling (**Figure 5.2.9-5**), were 181,000,000 ($\widehat{SE}$ =4,610,000) for Analysis Period 1 and 308,000,000 ($\widehat{SE}$ =15,300,000) for Analysis Period 2 (**Table 5.2.9-2**). The adjustment index ($\overline{I} / I_i$) was calculated as the ratio between the average ($\overline{I}$) of Intake Cove and the average annual index estimated from these same tows during the year being adjusted (I_i). The average index for 1997 and 1998 (I_i) was 0.0682 and 0.0181 larvae/m^3, respectively, while the long-term average index ($\overline{I}$) is 0.0664 larvae/m^3 for the years 1990–1998, yielding a ratio $\overline{I} / I_i$ of 0.974 for 1997 and 3.68 for 1998. The adjusted estimate for the 1997–1998 period is larger than the unadjusted entrainment estimate, and was significantly

greater than the 1996–1997 adjusted estimate because 95% confidence intervals approximated by $2 \cdot \widehat{SE}(\hat{E}_{Adj-T})$ did not overlap.

NOTE: *FH* and *AEL* estimates were not calculated from the estimates of entrainment for kelpfishes because there were no demographic data available to parameterize these models.

5.2.9.3 Empirical Transport Model (*ETM*)

Monthly estimates of proportional entrainment (PE_i) for kelpfishes from the surveys during the two years sampled ranged from a minimum of 0.00144±0.00156 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in March 1998 to a maximum of 0.346±0.189 in May 1998 (**Table 5.2.9-3**). There were other months when the $\hat{f}_i$ was larger than the value for May 1998, which would cause the $\widehat{PE_i}$ during that month to contribute more to the total P_M value calculated for the *ETM*. There were no periods when $\widehat{PE_i}$ and $\hat{f}_i$ were both equal to zero indicating that larvae were always present during the pasired entrainment and 72-hour study grid surveys.

Kelpfish larvae from the study grid subsamples were primarily distributed in nearshore areas (**Figure 5.2.9-2**) and similar to reported adult distributions (Miller and Lea 1972). Therefore, P_S was calculated using only alongshore current movement and not by extrapolating study grid abundance offshore. The large differences in estimated ages of larval kelpfish at entrainment (9 vs. 31 d) can affect calculation of $\widehat{P_M}$ (**Figure 5.2.9-6**). As expected, values for $\widehat{P_M}$ were less for shorter larval durations because of the reduced period of larval transport. Estimates of $\widehat{P_M}$ for either larval duration are relatively similar within an analysis period for a given duration

5.2.9.4 Interpretation of Assessment Results

Kelpfish do not have any commercial or recreational fishery value, and there is little information on their ecological role in the community. Because of the limited fishery harvest data for these species (ca. 0.14 MT at Port San Luis in 1990), entrainment mortality rates ($\widehat{P_M}$) could not be compared to harvests. Additionally, there are no estimates of population size or adult density that could be used to convert $\widehat{P_M}$ into an estimate of adult equivalent loss. Annual mean density of

kelpfish larvae from Intake Cove surface plankton tows shows considerable variation in abundance relative to the long-term mean for the period 1995–1998 (**Figure 5.2.9-5**). Abundance of the local adult kelpfish population appears to decline from 1993–1998 (**Figure 5.2.9-7**). Results of the *ETM* show that the power plant may annually entrain approximately 40% of the kelpfish larvae over an area six to seven times the area of the study grid (**Figure 5.2.9-6**). The length frequency distribution of entrained larvae shows a wide range of lengths, indicating that the larvae are subject to entrainment for a longer duration than other nearshore taxa (**Figure 5.2.9-3**). If Moser's (1996) estimate of size at flexion for spotted kelpfish *Gibbonsia elegans* is used for comparison, then the length frequency data indicates that post-flexion larvae are also being entrained. Although $\widehat{P_M}$ for kelpfishes is likely to be overestimated because of grid sampling that is biased for nearshore taxa, the relatively high probability of entrainment by the power plant intake as well as additional evidence of declines in local adult and larval abundance indicate a high likelihood for local impacts on this taxon. Additionally, local habitat for kelpfishes (the rocky habitat between Point Buchon and Point San Luis) is bordered to the north and south by large expanses of sandy beach that do not favor adult kelpfishes and may imply a geographically constrained subpopulation. However, the extent of larval duration estimated from our data indicates that kelpfish larvae could be transported sufficient distances to connect the local rocky habitat to areas north and south of the headlands mentioned above. The most common member of this genus found locally as an adult, crevice kelpfish (*Gibbonsia montereyensis*), is distributed from British Columbia to Baja California, making it unlikely that local impacts could affect the larger population.

Table 5.2.9-1. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for kelpfish larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	186,000,000	4,740,000
2) Oct 1, 1997–Sep 30, 1998	83,600,000	4,150,000
3) Jul 1, 1997–Jun 30, 1998	124,000,000	5,000,000

Table 5.2.9-2. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for kelpfish larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	181,000,000	4,610,000
2) Oct 1, 1997–Sep 30, 1998	308,000,000	15,300,000
3) Jul 1, 1997–Jun 30, 1998	458,000,000	18,400,000

Table 5.2.9-3. Kelpfishes: Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.0347	0.0146	0.224	0.00635
Aug 25, 1997	0.250	0.122	0.172	0.00471
Sep 29, 1997	0.0298	0.022	0.0668	0.00447
Oct 20, 1997	0.0357	0.0181	0.0329	0.00162
Nov 17, 1997	0.0239	0.0117	0.0307	0.00155
Dec 10, 1997	0.0160	0.00762	0.0206	0.00116
Jan 22, 1998	0.00836	0.00291	0.0466	0.00194
Feb 26, 1998	0.0148	0.00473	0.0229	0.00100
Mar 18, 1998	0.00144	0.00156	0.00948	0.000850
Apr 15, 1998	0.168	0.114	0.0468	0.00177
May 18, 1998	0.346	0.189	0.0815	0.00372
Jun 8, 1998	0.216	0.122	0.246	0.00863

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.152	0.104	0.0847	0.00336
Aug 26, 1998	0.168	0.0937	0.0532	0.00270
Sep 16, 1998	0.0417	0.0174	0.0286	0.00199
Oct 6, 1998	0.0612	0.0241	0.0560	0.00259
Nov 11, 1998	0.116	0.0412	0.129	0.00283
Dec 9, 1998	0.0389	0.00762	0.183	0.00426
Jan 12, 1999	0.0137	0.00293	0.0946	0.00253
Feb 3, 1999	0.0802	0.0331	0.103	0.00286
Mar 17, 1999	0.0131	0.00592	0.0351	0.00140
Apr 14, 1999	0.190	0.0940	0.0762	0.00200
May 24, 1999	0.240	0.113	0.0989	0.00288
Jun 23, 1999	0.277	0.149	0.0580	0.00240

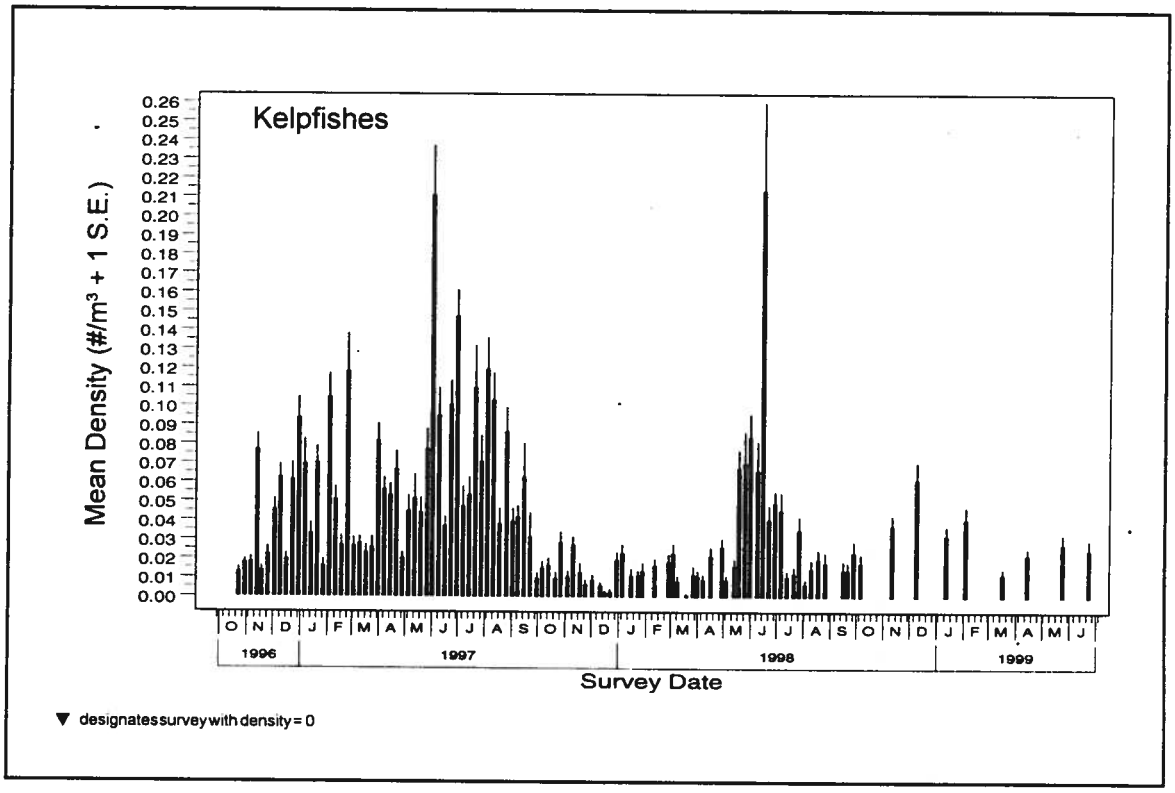


Figure 5.2.9-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

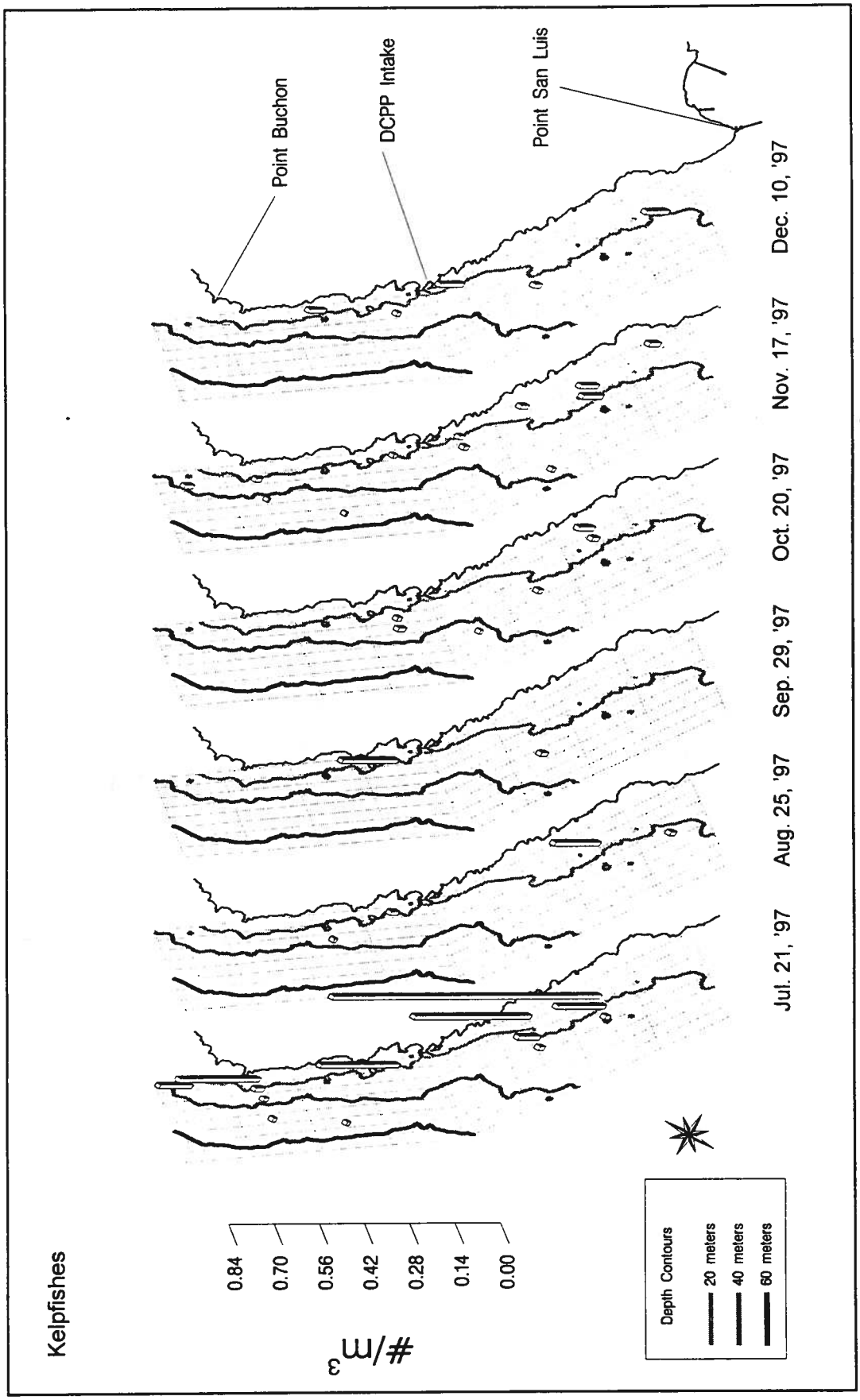


Figure 5.2.9-2a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

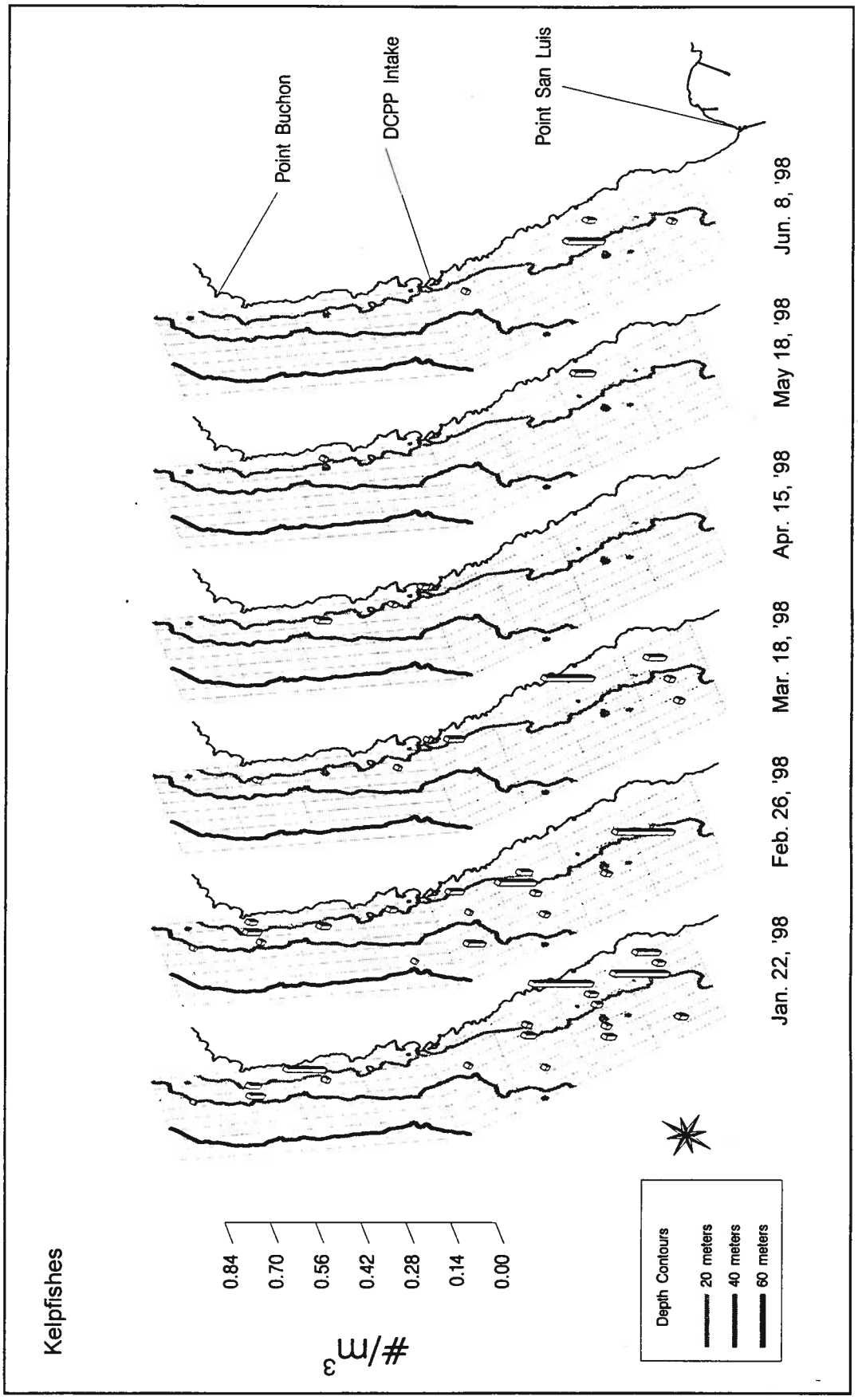


Figure 5.2.9-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

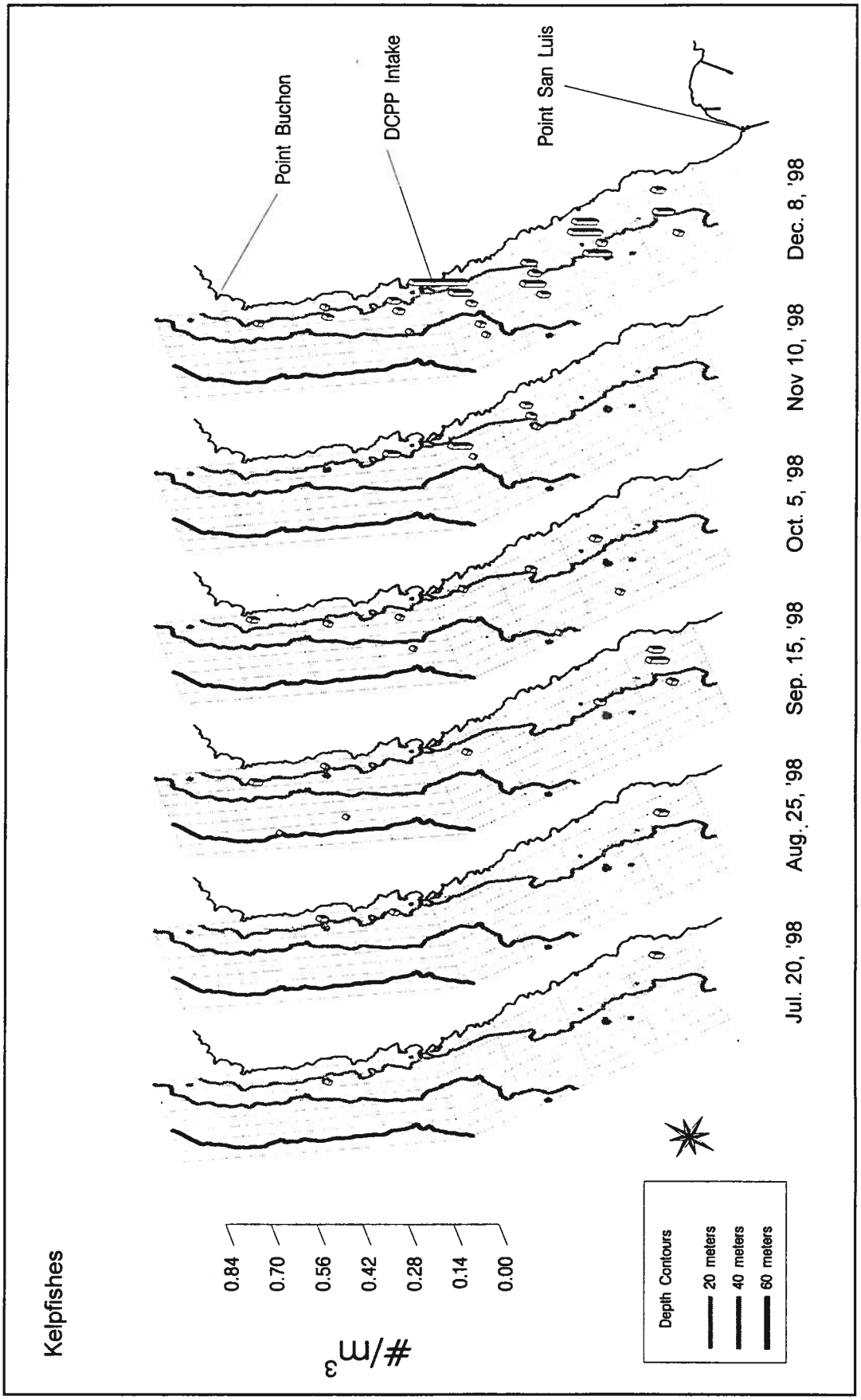


Figure 5.2.9-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

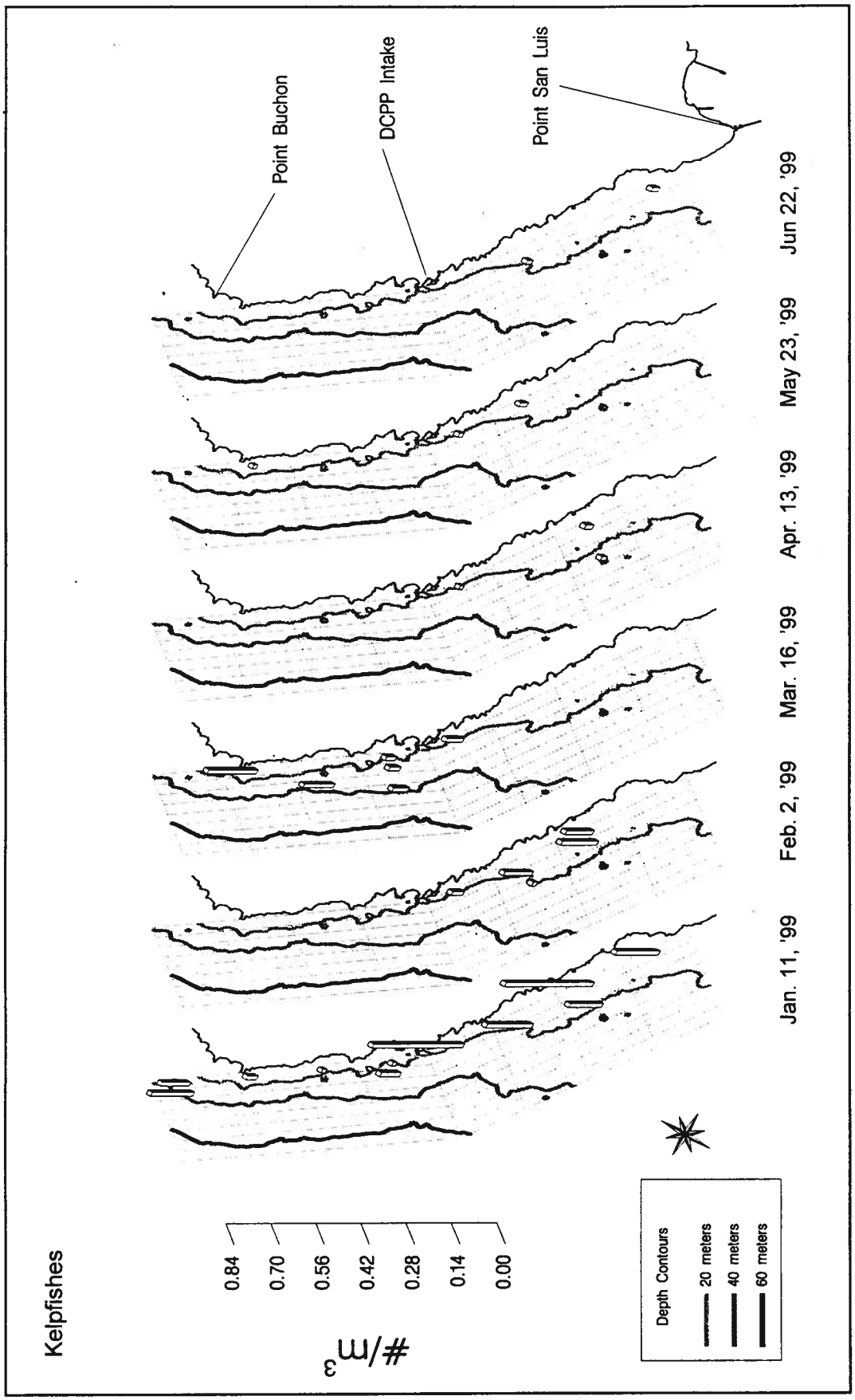


Figure 5.2.9-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

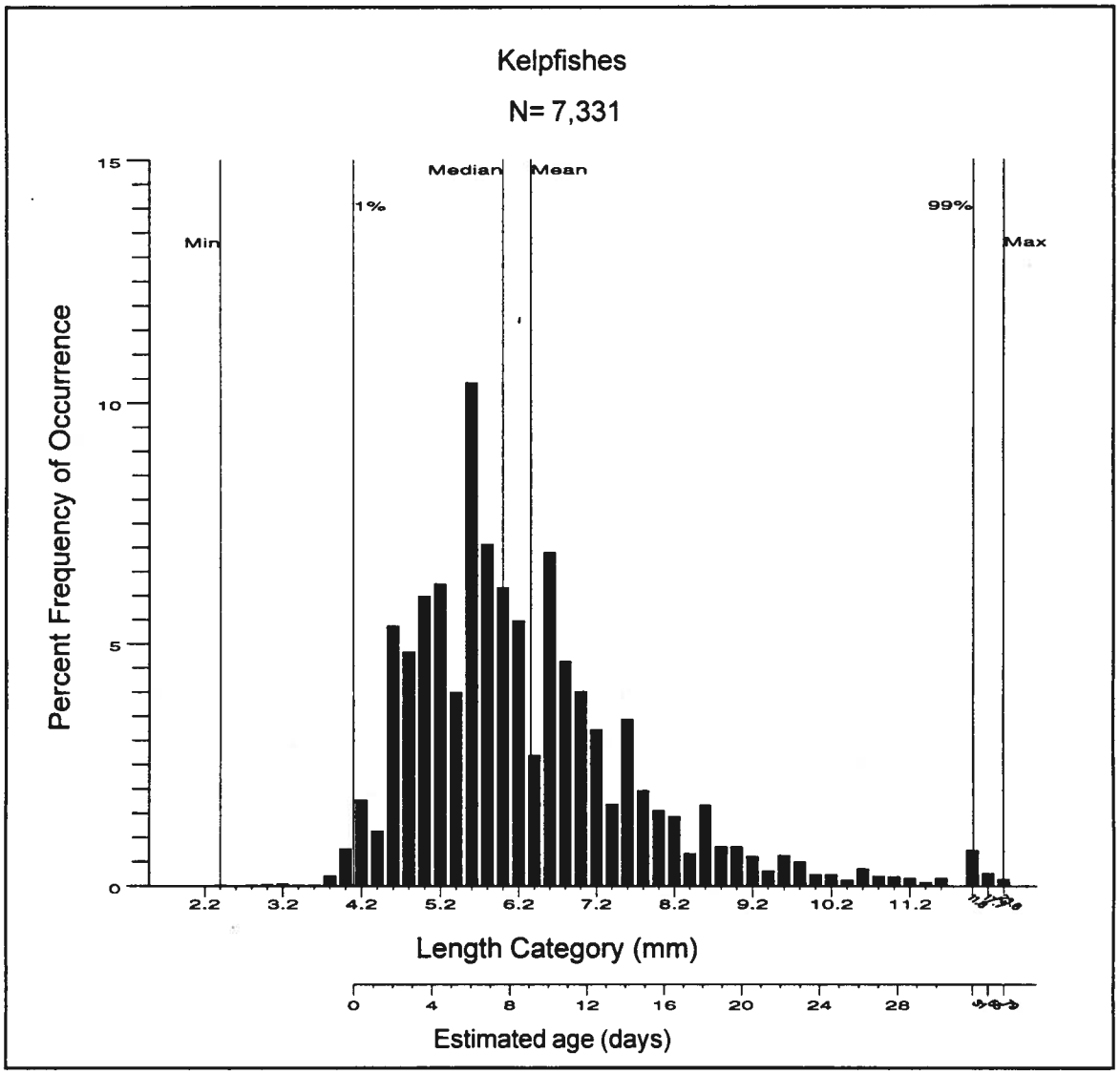


Figure 5.2.9-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

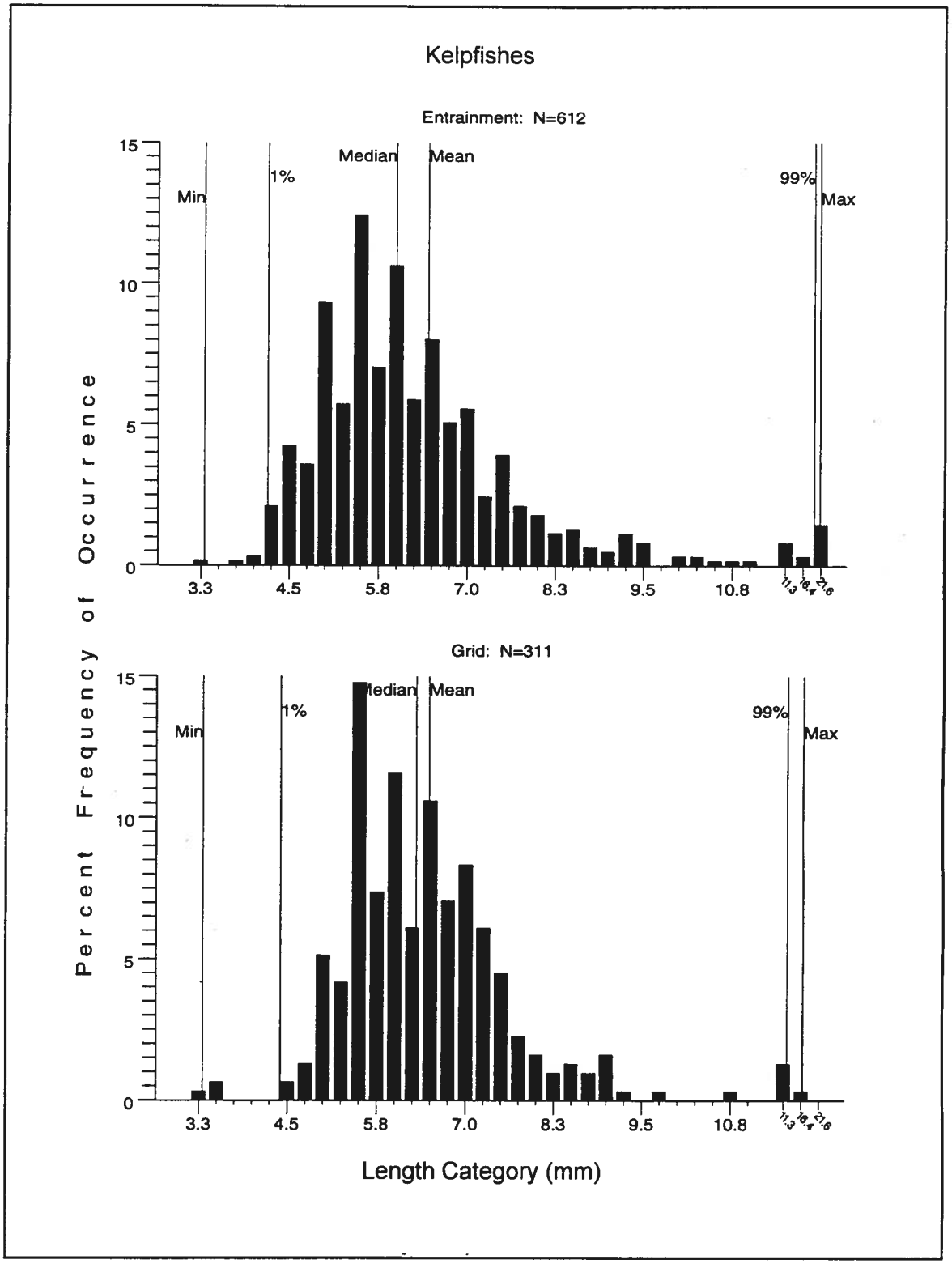


Figure 5.2.9-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

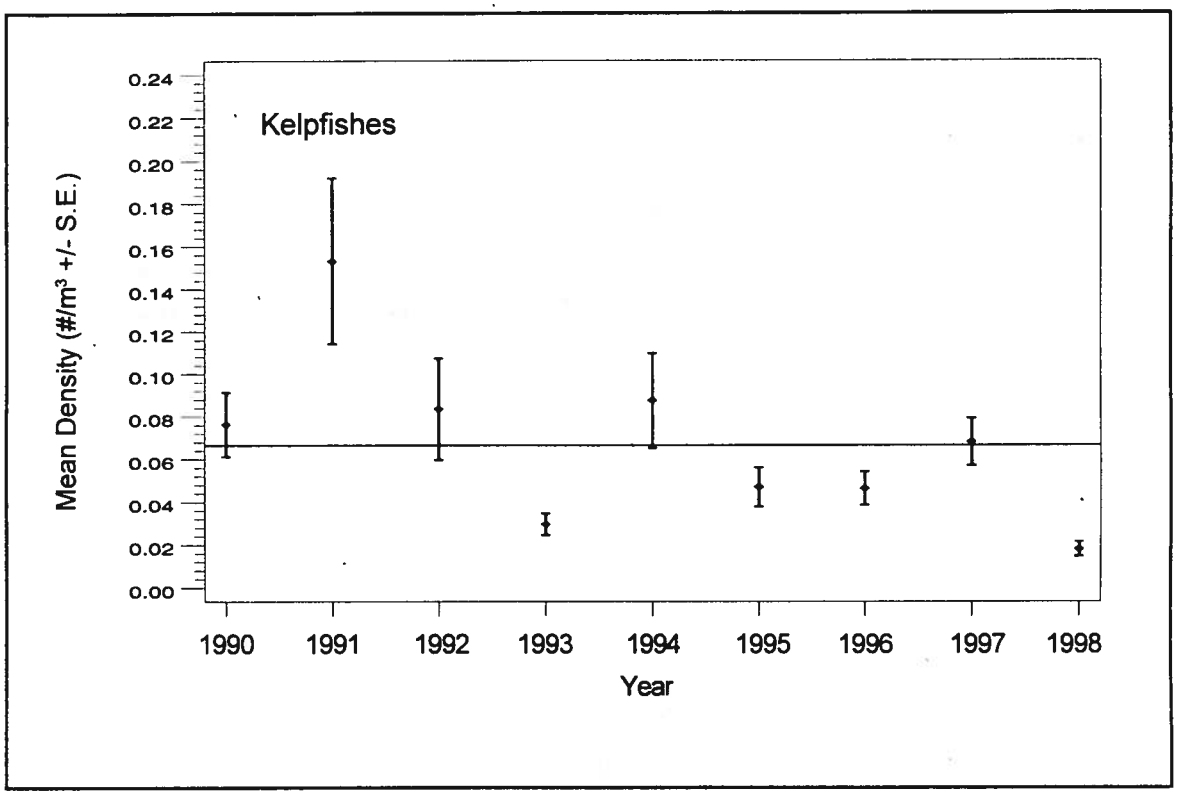


Figure 5.2.9-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

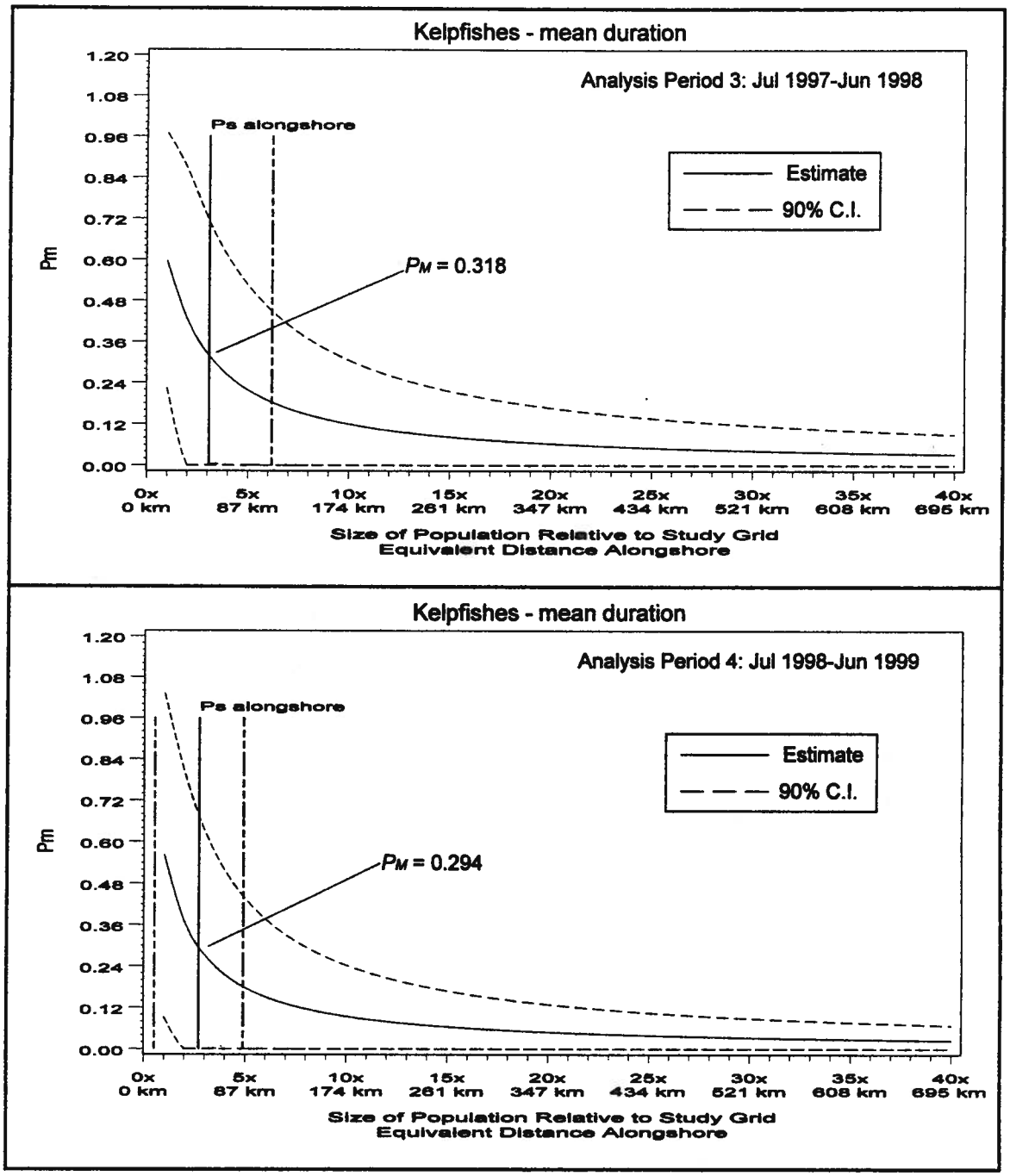


Figure 5.2.9-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements with the 90% C.I. indicated.

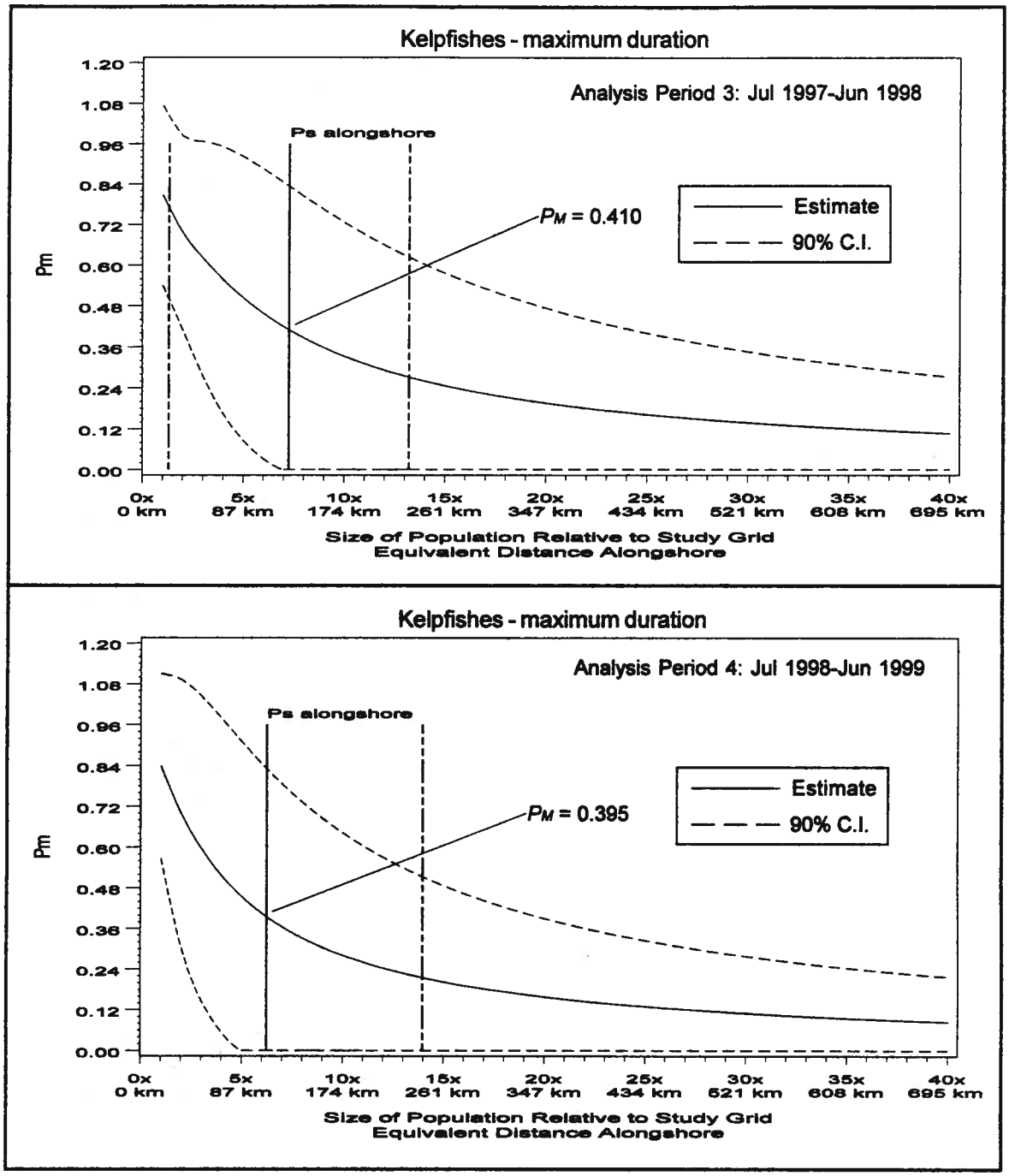


Figure 5.2.9-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements with the 90% C.I. indicated.

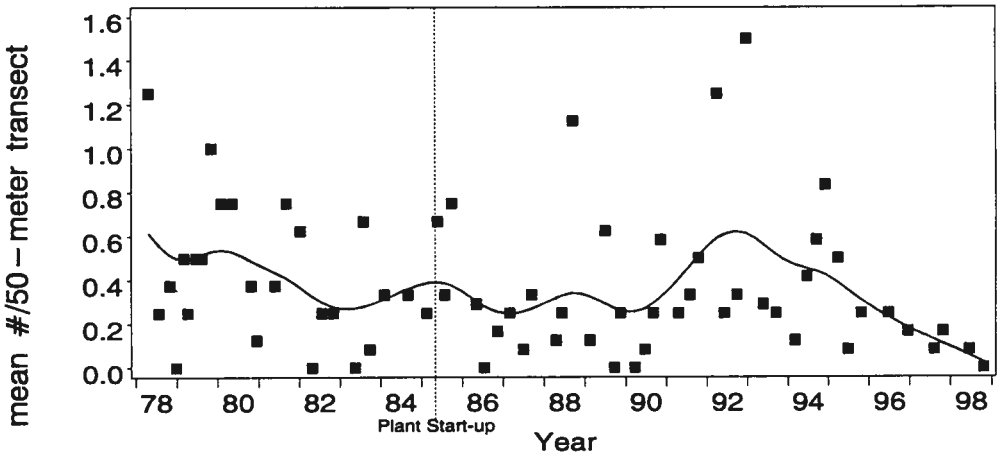


Figure 5.2.9-7. Mean number of kelpfishes observed per 50-meter subtidal transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to draw the curve through the points.

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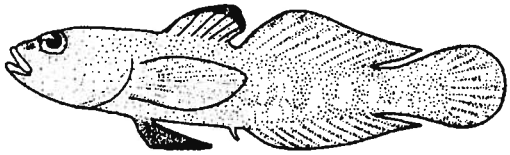
5.2.10 Assessment of Blackeye Goby (*Coryphopterus nicholsi*)



Coryphopterus nicholsi (Bean 1882); blackeye goby; length to 15 cm; northern British Columbia to south of Punta Rompiente, central Baja California; intertidal to 106 m; pale tan with some brown or greenish speckling; small blue dot below eye (Miller and Lea 1972; Eschmeyer et al. 1983).

The Gobiidae is the most speciose family of marine fishes with 1,875 species in 212 genera occurring worldwide in temperate to tropical climates and in a range of habitats from freshwater to brackish and marine environments (Brothers 1975; Nelson 1994). About 21 species and 16 genera occur in the CalCOFI study area from the Oregon-California border to south of Cabo San Lucas, Baja California Sur, Mexico (Moser 1996). Adult blackeye goby (*Coryphopterus nicholsi*) is common in benthic nearshore marine environments in the vicinity of Diablo Canyon, generally at the rock-sand interface (Tenera unpubl. data; Miller and Lea 1972; Love 1996).

The early life history of blackeye goby is similar to other members of the family Gobiidae. Females are oviparous, laying demersal, adhesive eggs under rocks that the males guard until planktonic larvae hatch (Love 1996; Moser 1996). Nests containing eggs are found from April to August in southern California (Wiley 1973). Larvae are reported to hatch at 2.8–3 mm (Moser 1996), with planktonic durations of approximately 75 d (Steele 1997). Larval transformation occurs at 16–25 mm (Moser 1996), which is within the range of lengths at settlement (15–29 mm) reported by Steele (1997). Blackeye goby demonstrate protogynous hermaphroditism (Cole 1983; Breitburg 1987; Cole and Shapiro 1992), with all animals beginning life as females and transforming to males at around 6–7.5 cm (Love 1996). They live approximately 5 yr and mature around 0.5 yr at approximately 4.5 cm (Steele 1997). Spawning occurs year-round, peaking between February and October (Matarese et al. 1989; Moser 1996). Females may spawn several times per year with fecundity estimates ranging in southern

California from 1,700 eggs/nest (Ebert and Turner 1962) to 3,274–4,788 eggs in Orange County (Wiley 1973).

While the size at age for post-larval blackeye goby has been described (Wiley 1973), size at age for the larvae estimated from planktonic duration and size at settlement are 2–3 mo and 15–29 mm, respectively (Steele 1997). Additionally, larval growth has been characterized for three gobiid species (arrow goby: *Clevelandia ios*; cheekspot goby: *Ilypnus gilberti*; shadow goby: *Quietula y-cauda*) from Mission Bay, California (Brothers 1975). Brothers (1975) described the growth coefficient of *C. ios* using the VBGF as k=0.96 (L_∞= 36 mm), that of *I. gilberti* as k=0.18 (L_∞= 60 mm), and that of *Q. y-cauda* as k=0.16 (L_∞= 70 mm).

No species-specific larval survivorship estimates have been reported for blackeye goby. However, Brothers (1975) calculated larval mortality up to settlement (to age 0 yr) for the three sympatric gobiids (arrow goby, cheekspot goby, and shadow goby) from Mission Bay, California (**Table 5.2-10-1**). These estimates were used to approximate blackeye goby mortality for early life stages as well as post settlement juvenile and adult stages. Substituting the survivorship information for these closely-related species into the assessment methods allowed us to estimate the number of lost spawning females using *FH* and to project losses into the future as equivalent adults, using *AEL*.

5.2.10.1 Summary of Field Collections

Blackeye goby larvae were collected almost year round at the DCPP intake structure during the years 1996–1999 (**Appendix H**), with their highest densities occurring primarily between May and September (**Figure 5.2.10-1**). Blackeye goby larvae were more abundant in 1997 than in 1998. This might have been due to the El Niño event that began during the spring of 1997 (Lynn et al. 1998; NOAA 1999). There were 7,658 larval blackeye goby identified from 1,119 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 24% of the subsamples processed from that location during that period.

Larval blackeye goby occurred in the DCPP study grid during most months except between February and March (**Appendix H**), with their highest density occurring during July 1997 (**Figure 5.2.10-2**). Blackeye goby larvae were dispersed throughout the study grid. However, during some months (e.g., October and November 1997, March 1999) blackeye goby larvae were abundant at the DCPP intake, but in low abundance in the study grid. There were 1,255 larval blackeye gobies identified from 505 bongo net subsamples representing 16% of the study grid subsamples collected and processed from July 1997–June 1999.

Standard lengths of all blackeye goby larvae collected at the DCPP intake structure between October 1996 and June 1999 ranged from a minimum of 1.5 mm to a maximum of 5.1 mm (**Figure 5.2.10-3**). The growth rate of larval blackeye goby (0.27 $mm \cdot d^{-1}$) reported by Steele (1997) was used to estimate larval ages. Examining the central 98% of this length-frequency distribution resulted in minimum and maximum lengths for the analyses of 2 mm and 3.4 mm, respectively. The mean larval length in this distribution was 2.6 mm and was estimated to be approximately 2 d old. We assumed that larvae shorter than the minimum length of the central 98% of the distribution were just hatched and assigned an age of 0 d. Reported hatching size range for this species is 2.8–3.0 mm (Moser 1996) however, indicating that the smallest larvae observed at DCPP represented either natural variation in hatch lengths within the population or the phenomenon of shrinkage following preservation (Theilacker 1980). If the observed lengths are the result of natural variation in hatching lengths, then the possibility exists that all larvae from the observed minimum length of 1.5 mm to the greatest reported hatching length of 3.0 mm (Moser 1996) could have just hatched (aged-0 d), leading to overestimation of ages for all larvae <3.0 mm. The average estimated age at entrainment of ca. 2 d for blackeye goby larvae indicated that the power plant entrains relatively young larvae.

There were significant differences between mean lengths (*t*-test: $p < 0.001$) and between length-frequency distributions (Kolmogorov-Smirnov test: $p \leq 0.037$) of larval blackeye goby from the DCPP intake structure and study grid despite the apparent similarity between the two distributions (**Figure 5.2.10-4**). The mean standard lengths from the intake and study grid were 2.6 mm and 2.8 mm, respectively. The disparity between sample sizes (699 larvae from the intake

and 144 larvae from the study grid) may cause the *t*-test to be overly sensitive to slight differences between the means (Zar 1984). While these differences were statistically detectable and significant using the *t*-test and K-S test, we believe the biological significance of these differences was probably low.

5.2.10.2 Estimating Total Annual Entrainment

The annual estimated mean entrainment of larval blackeye gobies for the 1996–97 Analysis Period 1 ($\widehat{E_T}$ = 138,000,000 ; $\widehat{SE}$ =14,100,000) and 1997–98 Analysis Period 2 ($\widehat{E_T}$ = 133,000,000 ; $\widehat{SE}$ =13,300,000) were extremely close in value (**Table 5.2.10-2**). The estimate for Analysis Period 3 differs from the other estimate for 1997–98, Analysis Period 2, because larval blackeye gobies were collected throughout the year, including the time period that does overlap between the two periods (**Figure 5.2.10-1**). Values of $\pm 2 \cdot \widehat{SE}(\widehat{E_T})$ can be used to approximate 95% confidence intervals around the point estimates. These approximate confidence intervals overlapped, indicating that the apparent differences between them were probably not statistically significant.

Annual estimates of the number of entrained blackeye goby larvae, adjusted to the nine year average estimated from Intake Cove surface plankton tow sampling (**Figure 5.2.10-5**), were 128,000,000 ($\widehat{SE}$ =13,100,000) for Analysis Period 1 and 109,000,000 ($\widehat{SE}$ =10,900,000) for Analysis Period 2 (**Table 5.2.10-3**). The adjustment index ($\bar{I} / I_i$) was calculated as the ratio between the nine year average ($\bar{I}$) of Intake Cove plankton samples and the average annual index estimated from these same tows during the year being adjusted (I_i). The average index for the years 1997 and 1998 (I_i) are 0.0181 and 0.0206 larvae/m^3, respectively, while the long-term average index ($\bar{I}$) is 0.0168 larvae/m^3 for the years 1990–1998, yielding the ratio $\bar{I} / I_i$ = 0.925 for 1997 and 0.816 for 1998. The confidence intervals, estimated from $\pm 2 \cdot \widehat{SE}(\hat{E}_{Adj-T})$, around the adjusted entrainment abundance overlap, indicating that the differences between the estimates are probably not statistically significant.

5.2.10.3 Fecundity Hindcasting *(FH)*

Larval mortality rates were not available for blackeye goby, but Brothers (1975) indicates that 99% mortality over two months is reasonable for low fecundity gobies. Finite survivorship for blackeye goby larvae was estimated as $0.926 \cdot d^{-1}$ from Brothers (1975) who examined three sympatric goby species (arrow, cheekspot, and shadow goby). Average age at entrainment of blackeye gobies at DCPP was estimated as 2.12 d, calculated using minimum size of the central 98% distribution (2 mm), mean size at entrainment (2.6 mm), and a growth rate of $0.27\ mm \cdot d^{-1}$ (Steele 1997). Survival to entrainment is then estimated as $0.926^{2.12} = 0.850$. Lifetime fecundity ($\widehat{TLF}$) is calculated using a longevity of 3.6 yr estimated from the von Bertalanffy growth function (VBGF; Ricker 1975) described below, an age at first maturation of 0.5 yr (Steele 1997), and an estimated annual fecundity of 8,062 eggs (2 batches of 4,031 annually; Wiley 1973). The VBGF was used to estimate longevity of females at their transformation length from female to male. Steele (1997) indicates that blackeye gobies mature at 0.5 yr (ca. 4.5 cm), and Love (1996) indicates that the largest fish recorded was ca. 15 cm; longevity is five years (Wiley 1973). The VBGF can be formulated as

$$L_t = L_\infty (1 - e^{-kt}), \text{ cm} \quad \times \frac{10\ mm}{1\ cm}$$

where

L_t = fish length at time t,
L_∞ = theoretical maximum fish length (L_∞ assumed to be 15 cm), and
k = growth coefficient.

Using these #s gives time @ maturity = 2 years (not 0.5!) ?

Then, using a length at maturity of approximately 4.5 cm and $L_\infty = 15$ cm, $k = 0.1724$, the VBGF estimated an age of 3.6 yr based on a length of transformation from female to male of 7 cm SL.

The adjusted total larval entrainment for blackeye goby ($\hat{E}_{Adj-T}$) was used to estimate the number of breeding females needed to produce the number of larvae entrained. The estimated number of breeding females ($\widehat{FH}$) whose fecundity equals the estimated total loss of entrained larvae is calculated assuming age of maturation of 0.5 yr and longevity value of 3.6 yr. The number of adult females hindcast from the larvae entrained at DCPP was greater for Analysis Period 1

($\widehat{FH}$ = 12,000; $\widehat{SE}$ =23,900) than for Analysis Period 2 ($\widehat{FH}$ = 10,300; $\widehat{SE}$ =20,300) (**Table 5.2.10-4**).

Sensitivity Analysis

Longevity had the greatest leverage on recalculated values of $\widehat{FH}$ (**Table 5.2.10-5**). Varying this parameter in the model resulted in an $\widehat{FH}$ range of 4,750–192,000 adult females for 1996–97 and 1997–98 analysis periods, compared to the original estimate of 10,300–12,100 adult females. Fecundity and larval survivorship had the next greatest amount of leverage on recalculated $\widehat{FH}$.

5.2.10.4 Adult Equivalent Loss *(AEL)*

larval survival 75 = 0.926

No independent estimate of survival of blackeye goby from entrainment to age of adults was found in the literature. Therefore, we used survival estimates for $\widehat{AEL}$ from two goby species (cheekspot and shadow goby) presented by Brothers (1975). Blackeye goby larval survival from entrainment up to settlement was estimated starting at estimated average entrainment age (2.12 d) to 75 d at settlement (Steele 1997), using daily survival of 0.926 derived from Brothers (1975). Larval survivorship over this period was calculated as $0.926^{72.9} \approx 0.004$. Juvenile and adult survivorship was estimated to 2 yr of age, which is the average age of adults assuming linear survival from 0.5 (age of first reproduction) to 5 yr (longevity). Daily finite survivorship was estimated as $e^{-1.02/365} = 0.997$ using annual instantaneous mortality = 1.02 (Brothers 1975). Survivorship from settlement to adulthood (2 yr less the 75 d of planktonic duration) is then $0.997^{656} \approx 0.159$.

The adjusted total larval entrainment for blackeye goby ($\hat{E}_{Adj-T}$) was used to estimate the number of equivalent adults lost to the population based on the estimated number of larvae entrained (**Table 5.2.10-6**). The estimated number of equivalent adults corresponding to the number of larvae entrained at DCPP was 75,200 ($\widehat{SE}$ =220,000) for Analysis Period 1 and 64,100 ($\widehat{SE}$ =188,000) for Analysis Period 2. Because blackeye goby are protogynous hermaphrodites, we do not assume that *AEL* is related to *FH* by a factor of two since we cannot assume a 50:50

sex ratio in the adult population. The magnitude of the $\widehat{SE}$ for the two estimates indicates that the differences between analysis periods are not statistically significant.

Sensitivity Analysis

Early larval and early juvenile survivorship had the greatest leverage on recalculated values of $\widehat{AEL}$ (**Table 5.2.10-7**). Varying either of these parameters in the model resulted in an $\widehat{AEL}$ range of 12,400-390,000 adult equivalents for the 1996–97 and 1997–98 analysis periods, compared to the original estimate of 64,100–75,400 adult equivalents. The range of estimates of entrainment had less effect on estimated $\widehat{AEL}$.

5.2.10.5 Empirical Transport Model *(ETM)*

Monthly estimates of proportional entrainment (PE_i) for blackeye goby in each *i*th survey from the two years sampled ranged from a minimum of zero to a maximum of 0.859±0.543 ($1\ \widehat{SE}\left(\widehat{PE_i}\right)$) in February 1998 and March 1999 (**Table 5.2.10-8**). No larvae were collected at either the DCPP intake or from the study grid when both $\widehat{PE_i}$ and the proportion of larvae present in the *i*th survey period ($\hat{f}_i$) were equal to zero (e.g., February 1999). When $\widehat{PE_i} = 0$ and $\hat{f}_i > 0$ (e.g., March 1998), larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. For blackeye goby, the relatively large $\widehat{PE_i}$ for February 1998 and March 1999 resulted from the fact that no blackeye goby larvae were collected in the study grid. This $\widehat{PE_i}$ was calculated using the larval density from the weekly entrainment sample (**Appendix H**) multiplied by the unsampled volume inshore of the two cells nearest the intake (D1 and E1) to approximate the number of larvae in this volume. This number was then added to the number of larvae found in cells D1 and E1, zero in this case as well as elsewhere in the grid, to approximate the number of larvae in the whole study grid. The low f_i values associated with these $\widehat{PE_i}$'s indicated that few larval blackeye gobies were collected at the intake structure during the survey period and that these estimates had low weights attributed to them in calculating the annual estimate of entrainment mortality. The greatest abundances of blackeye goby larvae at the intake structure occurred from July through September or October of each year as indicated by the higher values of f_i during those months.

Empirical transport modeling results for blackeye goby ($\widehat{P_M}$) show the effects of low estimated larval durations (**Figure 5.2.10-6**). Blackeye goby larvae were collected throughout the study grid, and thus, P_S was calculated by using alongshore and onshore current movement over estimated larval entrainment ages; study grid estimates of the number of blackeye goby larvae were extended alongshore or were extrapolated offshore, based on study grid abundance patterns offshore, and then extended alongshore. Alongshore larval transport over the short larval durations estimated from entrained larvae (average of 2 d to maximum of 5 d old at entrainment) resulted in low estimates of $\widehat{P_S}$. Consequently, estimates of $\widehat{P_M}$ over these small extrapolated areas were high. Estimates of $\widehat{P_M}$ were reasonably similar between the two analysis periods for either larval duration (mean or max.), whether extending $\widehat{P_S}$ only alongshore or extrapolating offshore and then extending alongshore. The alongshore $\widehat{P_M}$ value ranged from 0.174 to 0.228, while the extrapolated $\widehat{P_M}$ values range from 0.079 to 0.179.

5.2.10.6 Interpretation of Assessment Results

Blackeye goby does not support a commercial or recreational fishery, and there is little information on their ecological role in the community. However, there are some indications of their importance as forage for other species (e.g., cormorants: Love 1996). Because of the absence of any fishery for this species, there are no catch data that can be used to compare harvest mortality rates to entrainment mortality rates ($\widehat{P_M}$) or to estimated *FH* and *AEL* values. Results of *ETM* modeling show that the power plant may annually entrain approximately 17–23% of the blackeye goby larvae from an area approximately twice that of the study grid. Fecundity hindcast estimates indicate that these values of $\widehat{P_M}$ may be equal to predicted losses of the reproductive output of about 10,000–12,000 adult females. An independent estimate of *AEL* yields a predicted loss of 64,000–75,000 adults (males and females combined). Blackeye gobies are protogynous hermaphrodites (Cole 1983; Breitburg 1987; Cole and Shapiro 1992) and there is no reason to assume that the adult population for this species reflects a 50:50 sex ratio that would to required for the relationship of $AEL \equiv 2FH$. Although both of these estimates are relatively large the local population of adult blackeye goby does not exhibit any long-term declines based on DCPP Receiving Water Monitoring Program studies on subtidal fishes (**Figure**

5.2.10-7: Tenera 1997c). The time period since power plant operation began is much greater than the relatively short life-span (~5 yr) of blackeye gobies, indicating that there has probably been sufficient time for power plant effects to become apparent for this species.

The Intake Cove surface plankton tows were also analyzed to determine if any declining trends in larval production could be detected. These data were only analyzed for the period December through June. Blackeye goby larvae were present in the intake cove samples collected during December through June, but were also abundant in the.entrainment subsamples collected from July through September. Therefore, the index based on these data that was used to adjust annual entrainment estimates to the long-term average may not be a good estimator for blackeye goby. There is no way to estimate how the index would have changed if based on an entire 12 mo period. Nonetheless, the Intake Cove surface plankton tows data do not show any declining trends in annual mean larval blackeye goby densities and larval densities in 1997 and 1998 were greater than the nine year average(**Figure 5.2.10-5**).

Estimates of total annual entrainment mortality ($\widehat{P_M}$) for this species were relatively high among the target fish taxa examined (ca. 13–23%). These values are based on extending the larval abundance within the study grid alongshore (or extrapolating that abundance offshore and then extending it alongshore) using estimated larval duration at entrainment. The short estimated larval durations for this taxa resulted in relatively small estimated areas for the population at risk These small population areas are representative of the reflects the small size range for the majority of entrained larvae (2.0–3.4 mm). This size range is well below the reported range at flexion of 5.8–6.8 mm (Moser 1996). In fact, our observed minimum larval lengths were less than reported hatching lengths in the literature, which may be due to natural variation in hatching lengths plus an undetermined contribution of larval shrinkage due to preservation (Theilacker 1980). This may indicate that our assumption that age-0 d larvae correspond to our minimum 99% larval length is inaccurate and that all larvae less than maximum reported hatching length of 3.0 mm (Moser 1996) were aged-0 d. This would further reduce our estimate of larval duration and population area. While the small areas based on these short larval durations and alongshore and onshore current movements lead, in part, to the high estimates of entrainment mortality, the

size of the entrained larvae indicate that the time the larvae are exposed to entrainment is very short. The lack of apparent declines in the local adult and larval populations over several generations since power plant startup indicates that entrainment mortality may only be affecting surplus larval production that would not lead to a reduction in the local population. This could also indicate that compensation is occurring at some later stage in development. In either case, it does not appear the DCPP cooling water intake is causing a long-term decline in the local population of blackeye goby.

Table 5.2.10-1. Larval mortality statistics for three sympatric gobiids from Mission Bay, California (Brothers 1975).

Species	Two-Month Larval Mortality Rates (to settlement) % (Finite)	Annual Post-Settlement Mortality Rates % (Finite)	(Instantaneous)
Clevelandia ios	98.3	89–91	-2.39
Ilypnus gilberti	98.6	66–74	-1.08
Quietula y-cauda	99.2	62–69	-0.96

Table 5.2.10-2. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for blackeye goby (*Coryphopterus nicholsi*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	138,000,000	14,100,000
2) Oct 1, 1997–Sep 30, 1998	133,000,000	13,300,000
3) Jul 1, 1997–Jun 30, 1998	157,000,000	14,600,000

Table 5.2.10-3. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for blackeye goby (*Coryphopterus nicholsi*) larvae from the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	128,000,000	13,100,000
2) Oct 1, 1997–Sep 30, 1998	109,000,000	10,900,000
3) Jul 1, 1997–Jun 30, 1998	128,000,000	11,900,000

Table 5.2.10-4. Estimated number of blackeye goby adult (*Coryphopterus nicholsi*) females ($\widehat{FH}$) whose reproductive output was equivalent to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{FH})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	12,000	23,900	315,000	461
2) Oct 1, 1997–Sep 30, 1998	10,300	20,300	268,000	393
3) Jul 1, 1997–Jun 30, 1998	12,100	24,000	316,000	462

Table 5.2.10-5. Blackeye goby (*Coryphopterus nicholsi*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

		Recalculated $\widehat{FH}$		
Parameter	Estimate	Minimum	Maximum	Range
$\widehat{FH}$ = 12,000				
$\hat{E}_{Adj-T}$	128,000,000	10,000	14,100	4,050
S_{larvae}	0.850	10,200	62,400	52,200
# Eggs/yr	8,060	2,320	62,400	60,100
Longevity	3.60	5,570	192,000	186,000
Maturation	0.5	11,100	14,400	3,210

Table 5.2.10-5 (continued). Blackeye goby (*Coryphopterus nicholsi*): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

b) Analysis Period 2: October 1, 1997–September 30, 1998

		Recalculated $\widehat{FH}$		
Parameter	Estimate	Minimum	Maximum	Range
$\widehat{FH}$ = 10,300				
$\hat{E}_{Adj-T}$	109,000,000	8,580	11,900	3,370
S_{larvae}	0.850	8,710	53,100	44,400
# Eggs/yr	8,060	1,980	53,100	51,200
Longevity	3.60	4,750	163,000	158,000
Maturation	0.5	9,490	12,200	2,740

c) Analysis Period 3: July 1, 1997–June 30, 1998

		Recalculated $\widehat{FH}$		
Parameter	Estimate	Minimum	Maximum	Range
$\widehat{FH}$ = 12,100				
$\hat{E}_{Adj-T}$	128,000,000	10,200	13,900	3,690
S_{larvae}	0.850	10,300	62,600	52,300
# Eggs/yr	8,060	2,330	62,600	60,200
Longevity	3.60	5,590	192,000	187,000
Maturation	0.5	11,200	14,400	3,220

Table 5.2.10-6. Estimated number of equivalent blackeye goby (*Coryphopterus nicholsi*) adults ($\widehat{AEL}$) equal to the adjusted number of larvae entrained per year ($\hat{E}_{Adj-T}$) at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{AEL})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{AEL}$	$\widehat{SE}(\widehat{AEL})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	75,200	220,000	9,260,000	611
2) Oct 1, 1997–Sep 30, 1998	64,100	188,000	7,890,000	521
3) July 1, 1997–June 30, 1998	75,400	221,000	9,290,000	613

Table 5.2.10-7. Blackeye goby (*Coryphopterus nicholsi*): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

		Recalculated $\widehat{AEL}$		
Parameter	Estimate	Minimum	Maximum	Range
$\widehat{AEL}$ = 75,200				
$\hat{E}_{Adj-T}$	128,000,000	87,900	62,600	25,300
$\hat{S}$ *early larvae*	0.00372	390,000	14,500	375,000
$\hat{S}$ *early juvenile*	0.158	390,000	14,500	375,000

Table 5.2.10-7 (continued). Blackeye goby (*Coryphopterus nicholsi*): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

b) Analysis Period 2: October 1, 1997–September 30, 1998

		Recalculated $\widehat{AEL}$		
Parameter	Estimate	Minimum	Maximum	Range
$\widehat{AEL}$ = 64,100				
$\hat{E}_{Adj-T}$	109,000,000	74,600	53,600	21,000
$\hat{S}$ *early larvae*	0.00372	332,000	12,400	320,000
$\hat{S}$ *early juvenile*	0.158	332,000	12,400	320,000

c) Analysis Period 3: July 1, 1997–June 30, 1998

		Recalculated $\widehat{AEL}$		
Parameter	Estimate	Minimum	Maximum	Range
$\widehat{AEL}$ = 75,400				
$\hat{E}_{Adj-T}$	128,000,000	87,000	63,900	23,100
$\hat{S}$ *early larvae*	0.00372	391,000	14,600	376,000
$\hat{S}$ *early juvenile*	0.158	391,000	14,600	376,000

Table 5.2.10-8. Blackeye goby (*Coryphopterus nicholsi*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.0141	0.00798	0.206	0.0187
Aug 25, 1997	0.0353	0.0130	0.219	0.0140
Sep 29, 1997	0.0398	0.0172	0.143	0.0127
Oct 20, 1997	0.452	0.288	0.0390	0.00500
Nov 17, 1997	0.291	0.185	0.0524	0.00554
Dec 10, 1997	0.0658	0.0332	0.0174	0.00160
Jan 22, 1998	0.0512	0.0270	0.0447	0.00556
Feb 26, 1998	0.859	0.543	0.0140	0.00150
Mar 18, 1998	0	0	0.00172	0.000310
Apr 15, 1998	0.859	0.361	0.0368	0.00285
May 18, 1998	0.291	0.130	0.0854	0.00667
Jun 8, 1998	0.202	0.111	0.140	0.0119

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.0377	0.0167	0.160	0.0115
Aug 26, 1998	0.0130	0.00539	0.182	0.0131
Sep 16, 1998	0.00747	0.00370	0.0687	0.0122
Oct 6, 1998	0.0186	0.00953	0.153	0.0236
Nov 11, 1998	0.167	0.148	0.0903	0.0118
Dec 9, 1998	0.0575	0.0514	0.00879	0.00118
Jan 12, 1999	0.0297	0.0413	0.00163	0.000310
Feb 3, 1999	0	0	0	0
Mar 17, 1999	0.859	0.504	0.0755	0.00730
Apr 14, 1999	0.395	0.217	0.0837	0.00785
May 24, 1999	0.0164	0.00620	0.0806	0.00555
June 23, 1999	0.0125	0.00555	0.0955	0.00862

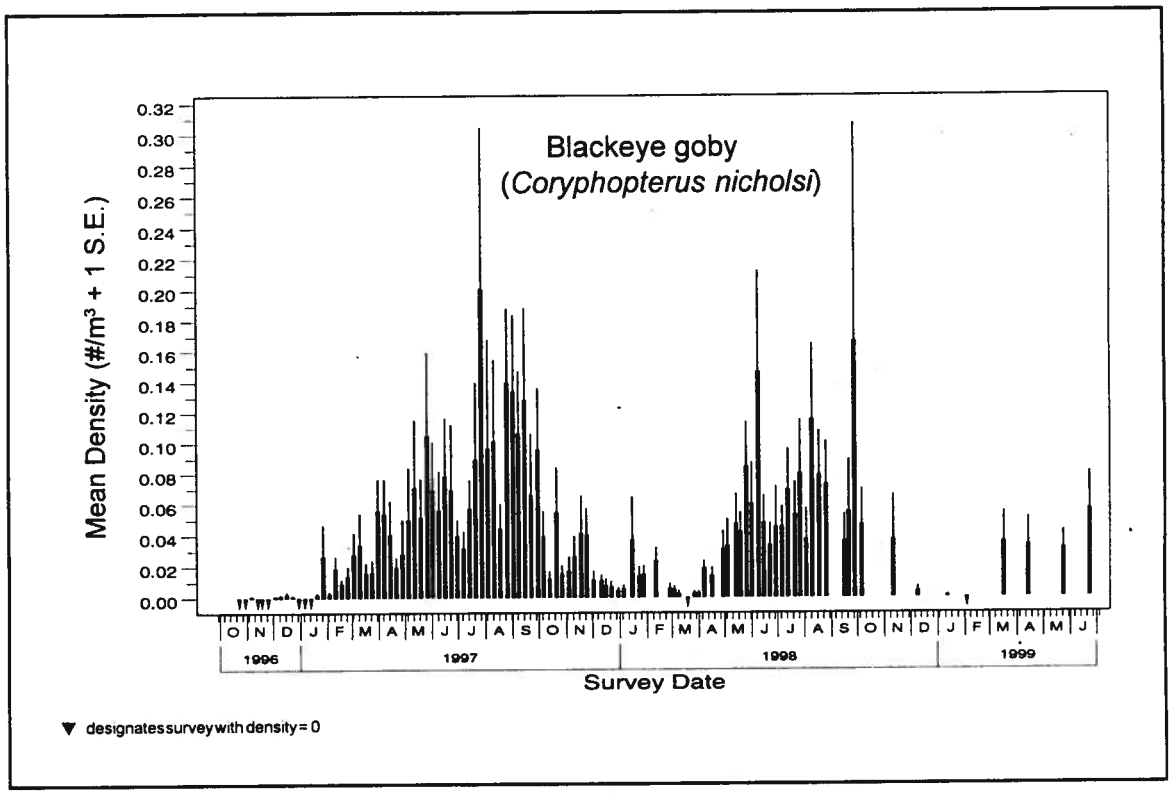


Figure 5.2.10-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

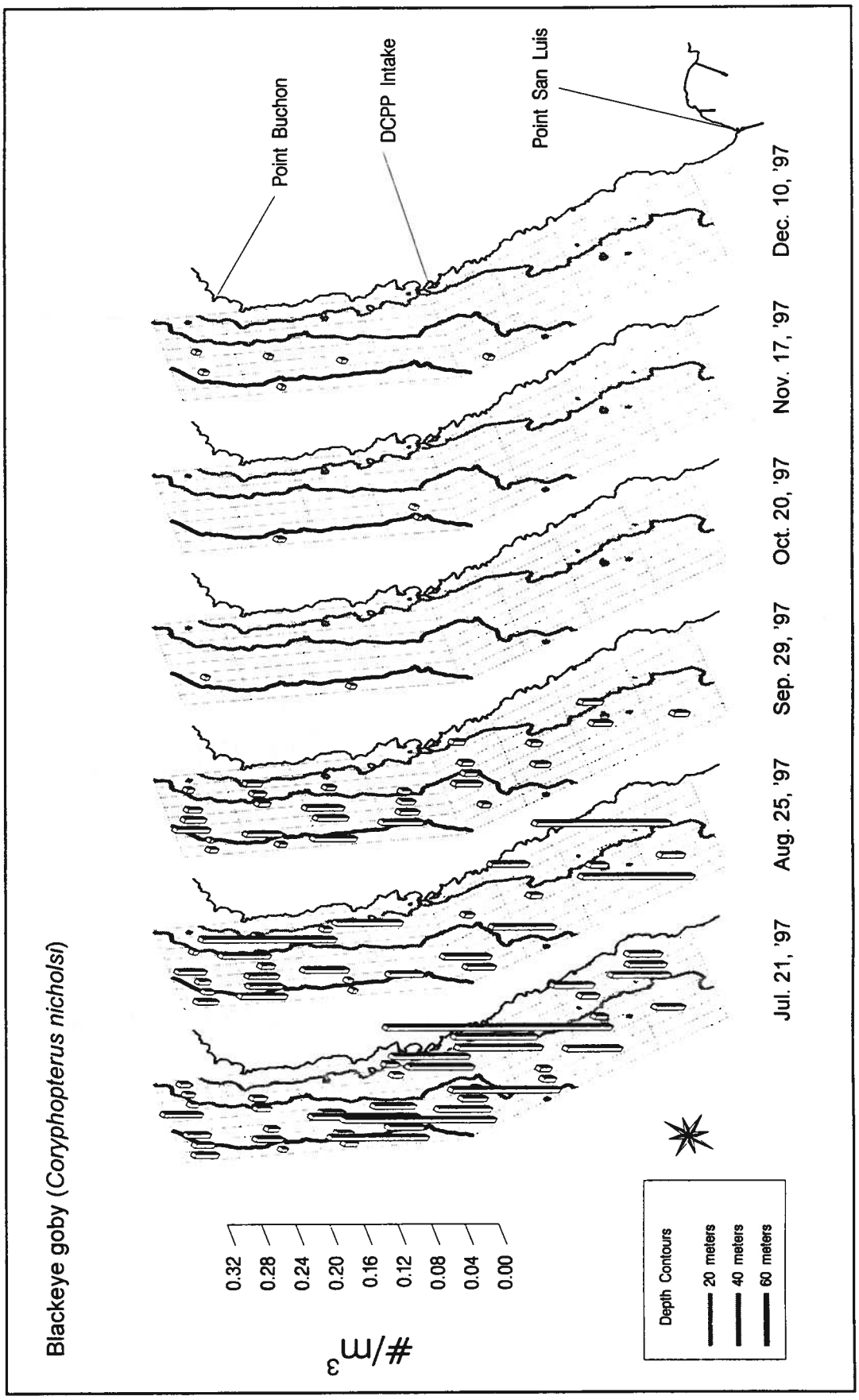


Figure 5.2.10-2a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

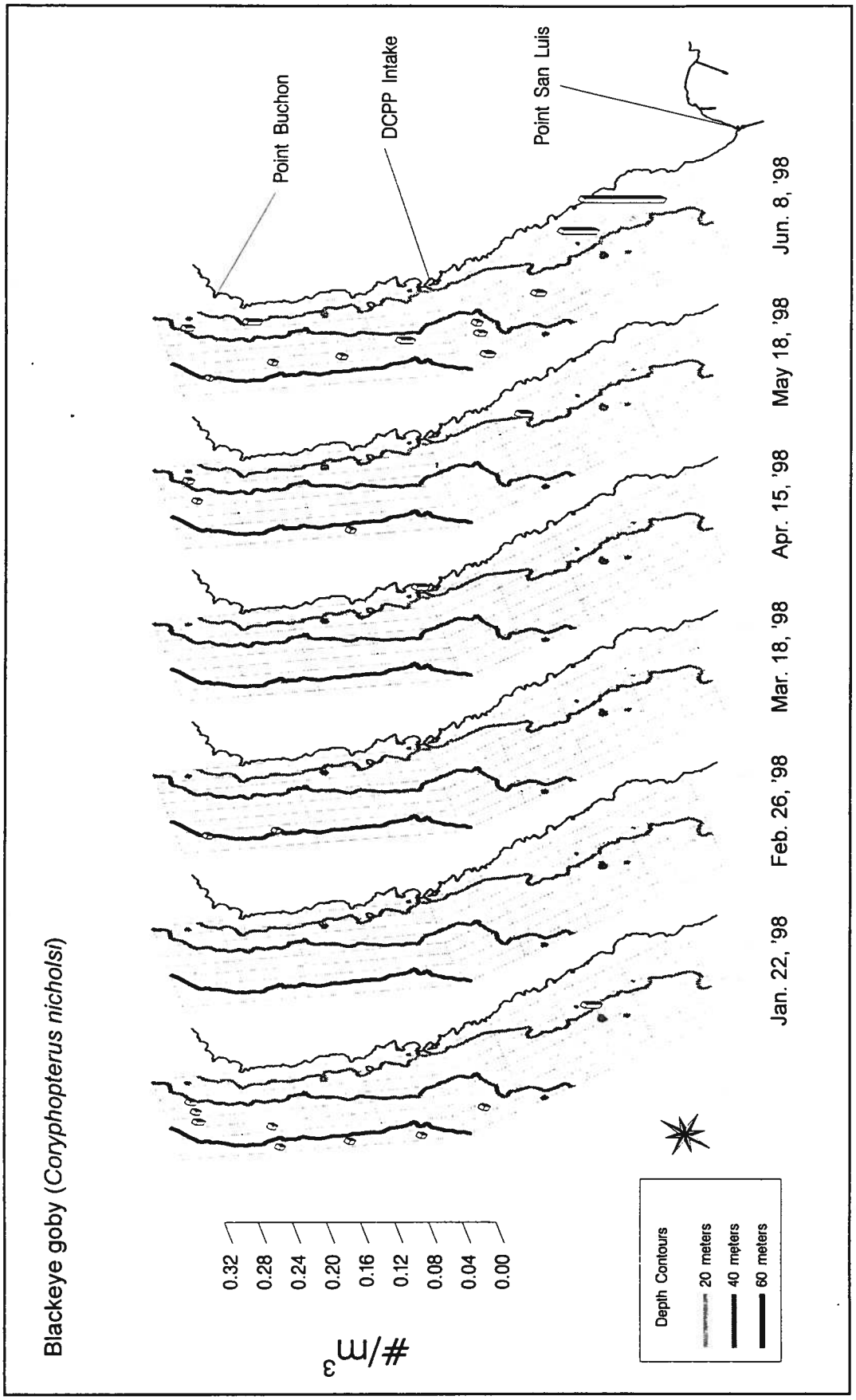


Figure 5.2.10-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

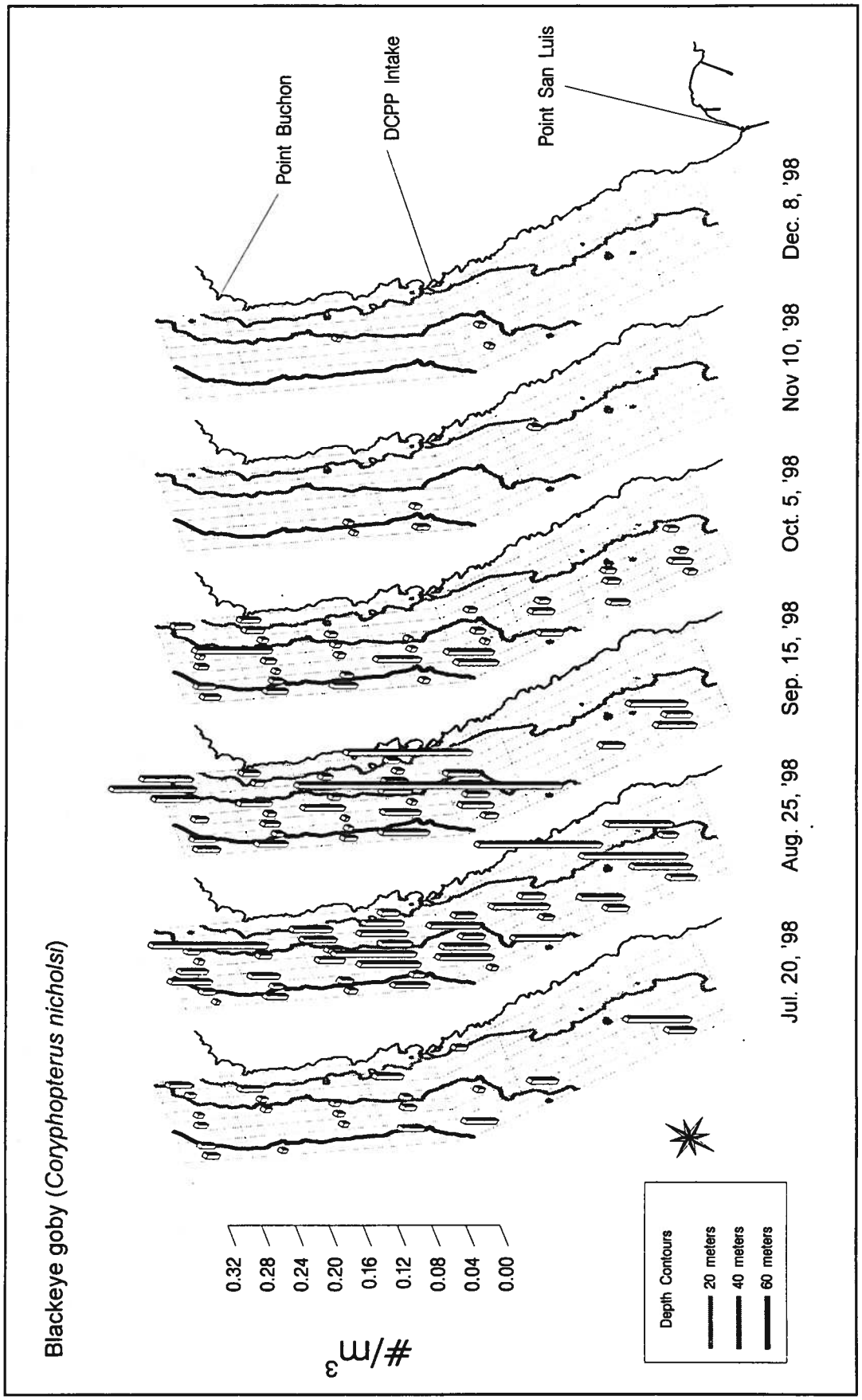


Figure 5.2.10-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

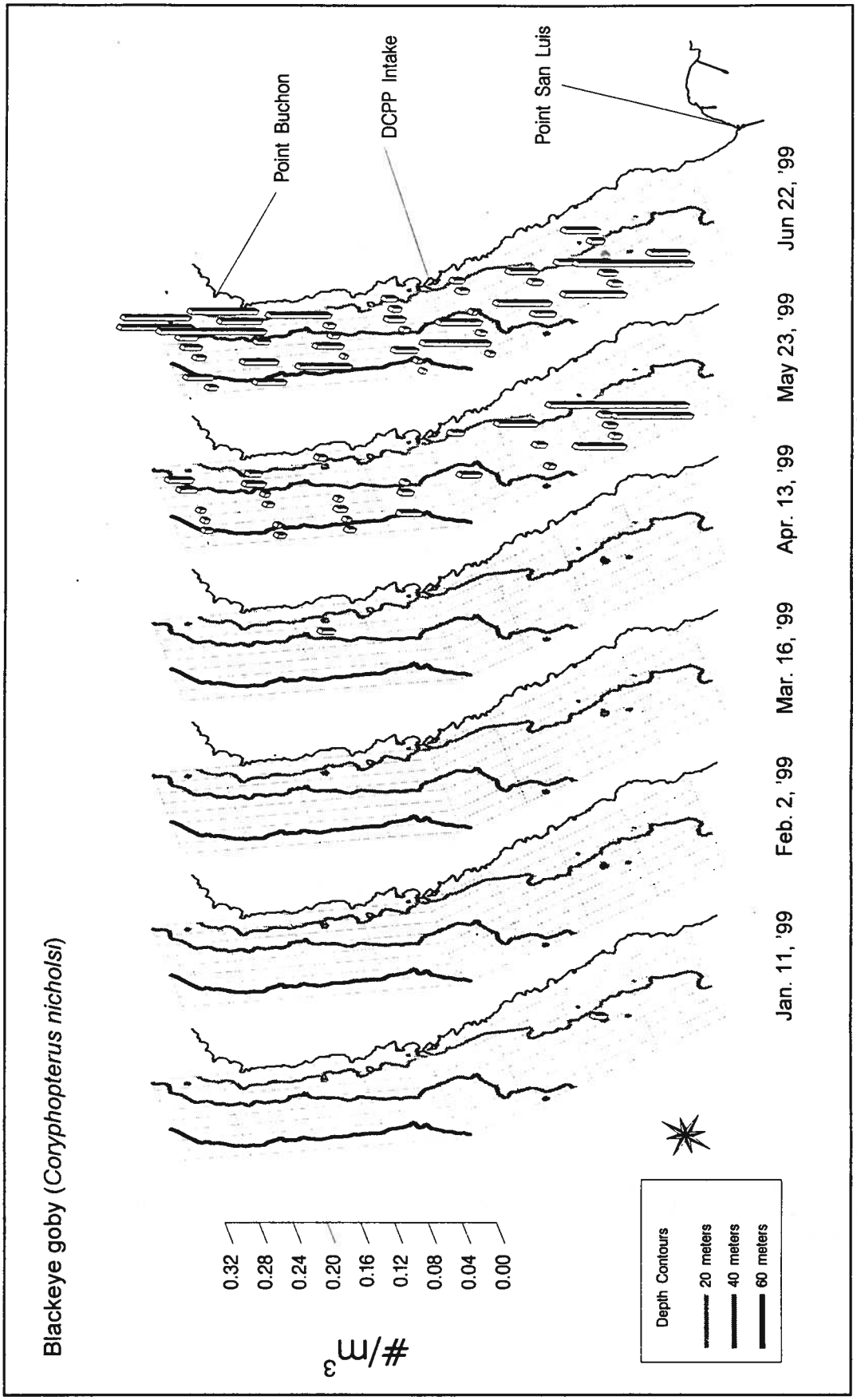


Figure 5.2.10-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

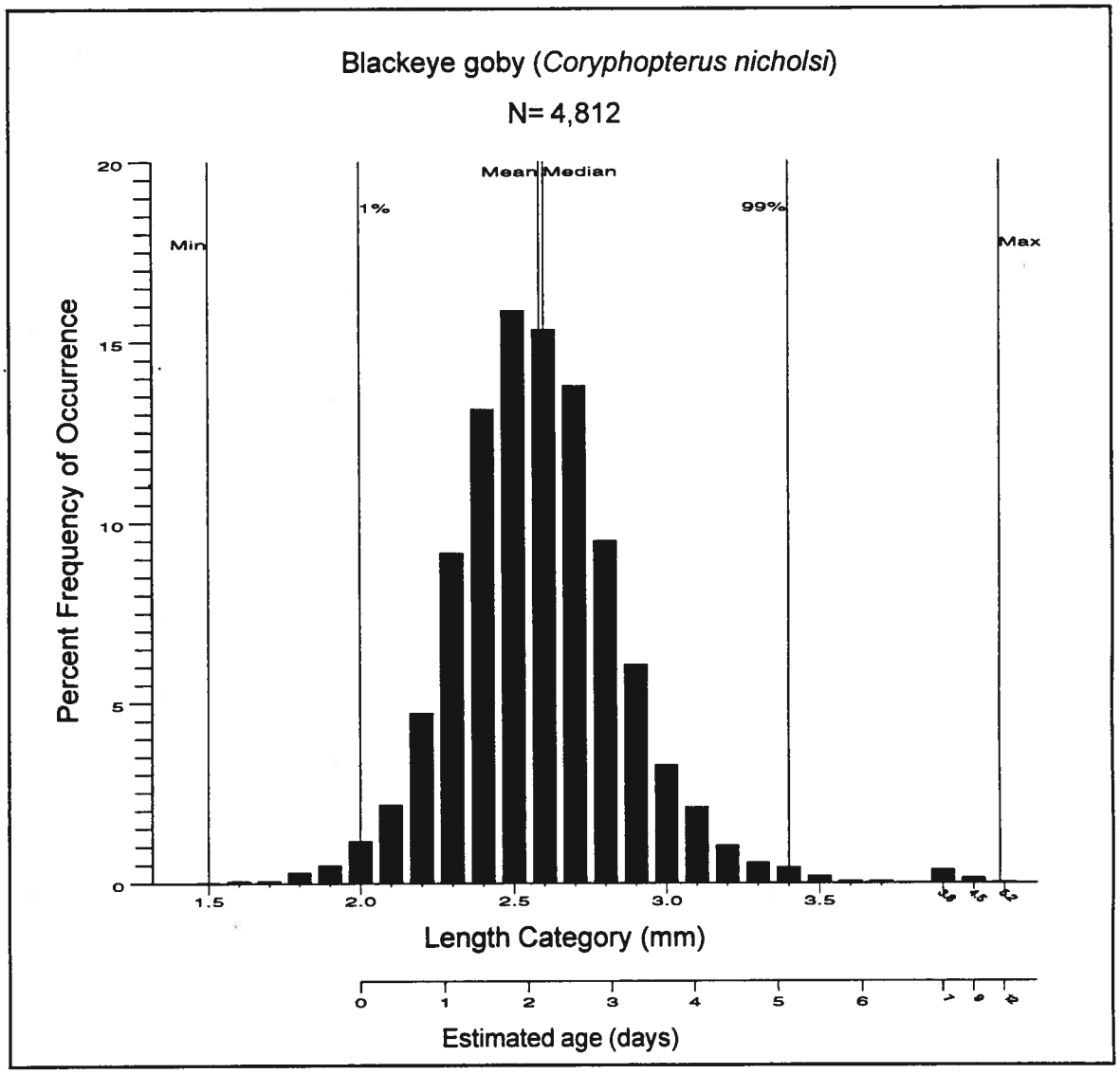


Figure 5.2.10-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

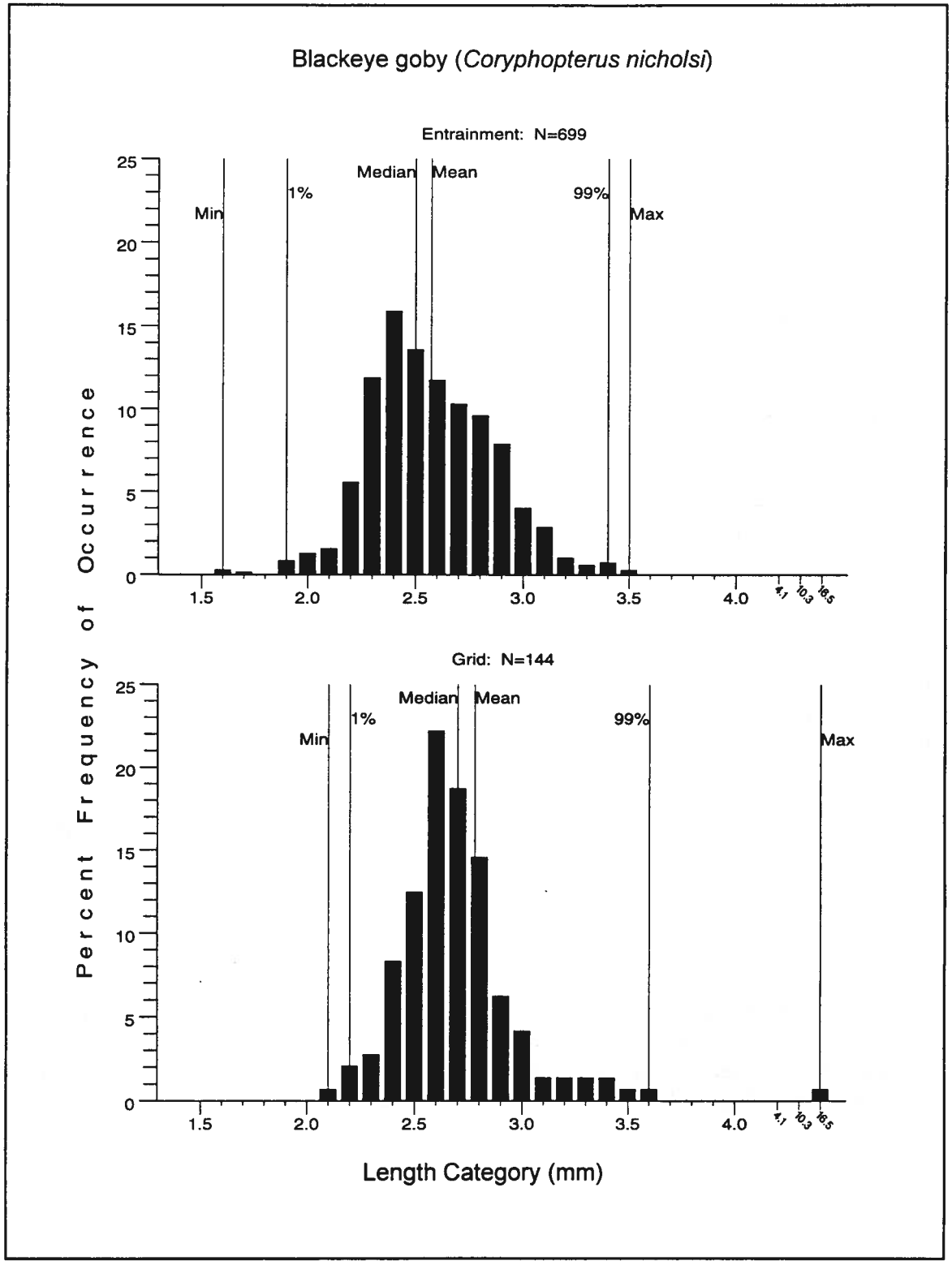


Figure 5.2.10-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

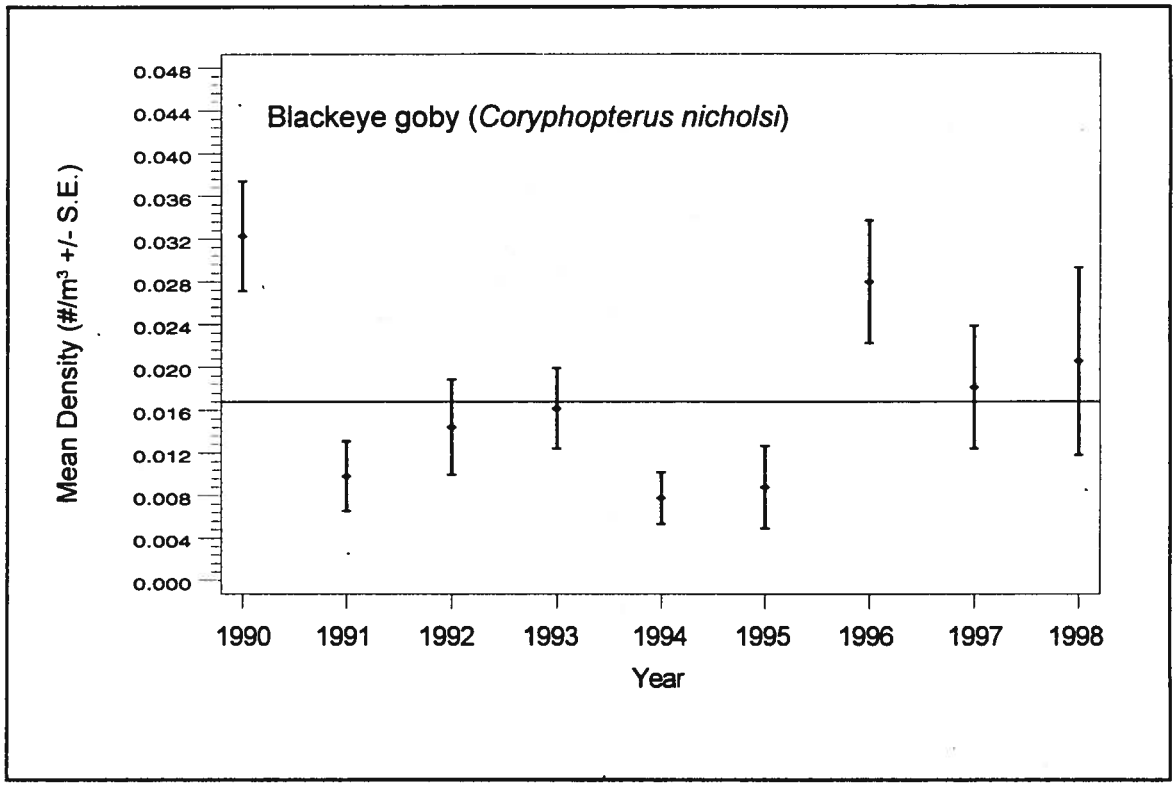


Figure 5.2.10-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

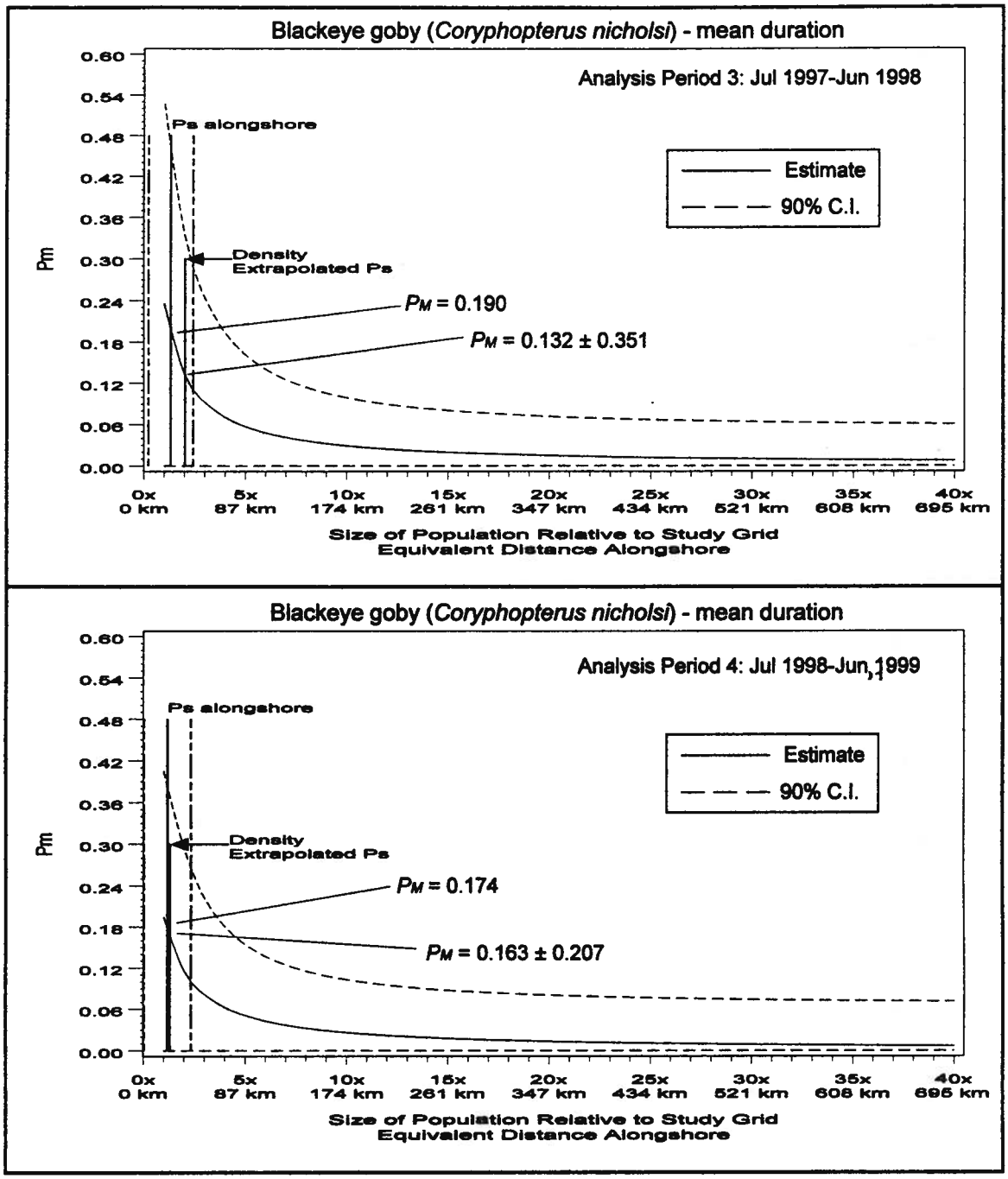


Figure 5.2.10-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

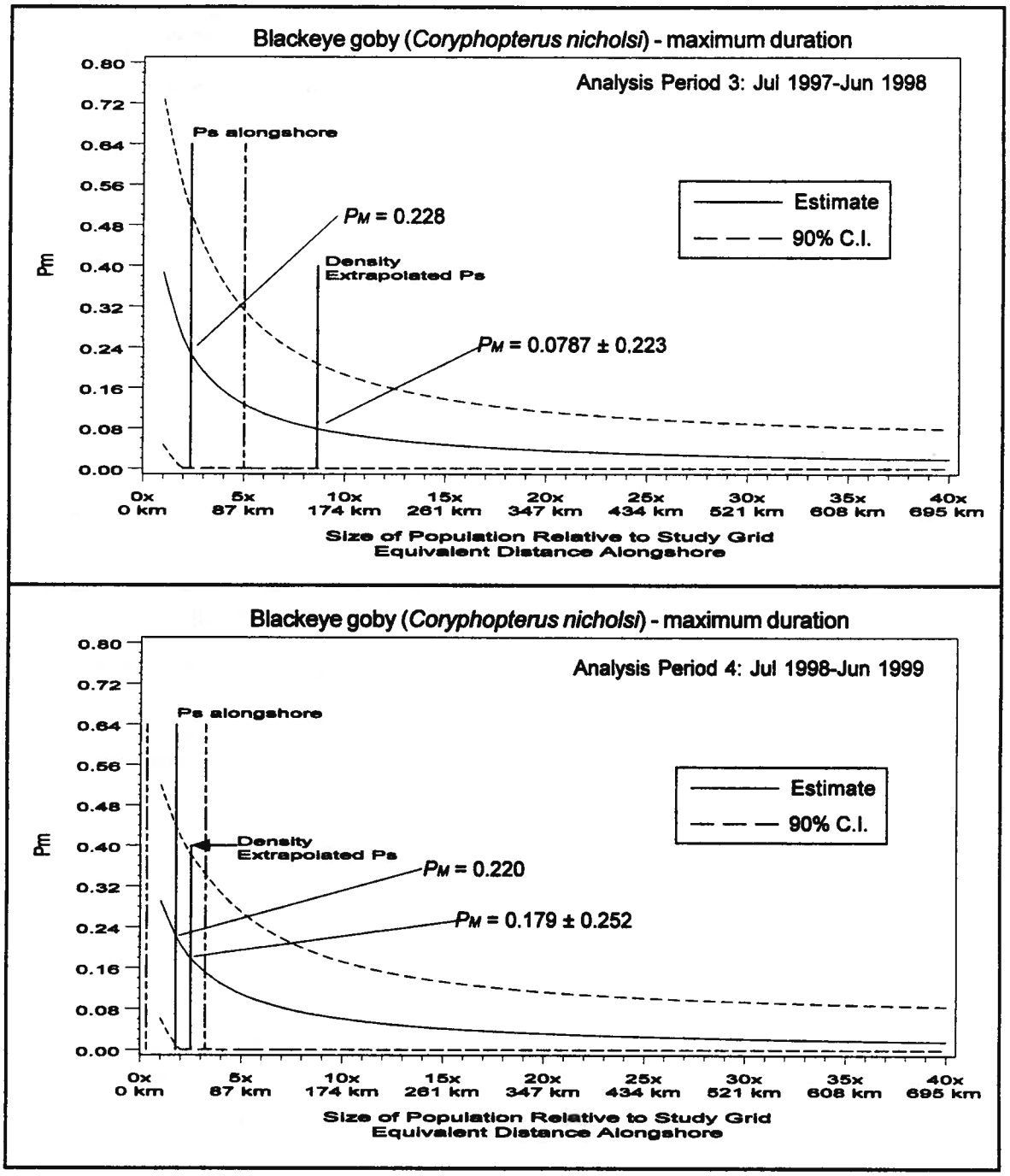


Figure 5.2.10-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

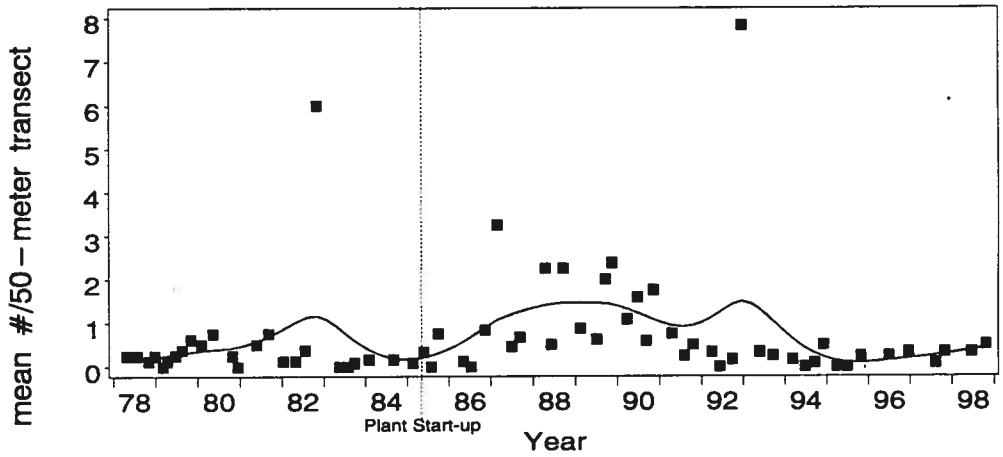


Figure 5.2.10-7. Mean number of blackeye gobies observed per 50-meter subtidal transect in the DCPP RWMP South Control area. Spline smoothing algorithm used to draw the curve through the points.

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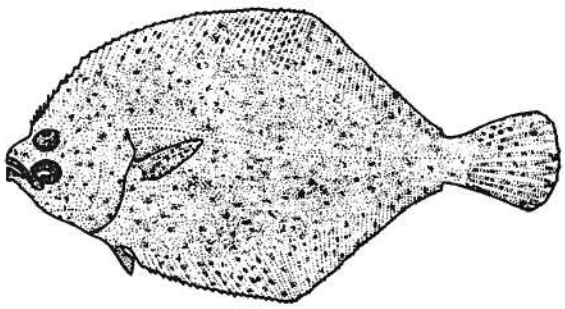
5.2.11 Assessment of Flatfishes (Family Paralichthyidae)

Left-eyed flatfishes (Paralichthyidae) are represented in the California Current region by 8 genera and 19 species (Nelson 1994; Moser and Sumida 1996). The larvae of four sanddab species (genus *Citharichthys*) are abundant in CalCOFI tows up to 180 km offshore, while other paralichthyid flatfish such as the California halibut (*Paralichthys californicus*) are primarily found within 40 km of shore. Females in this family are oviparous with large batch sizes and multiple spawns. Spawning takes place throughout the year with seasonal peaks. DCPP entrainment effects on larvae were assessed for sanddabs (*Citharichthys* spp.) and for California halibut.

5.2.11.1 Sanddabs (*Citharichthys* spp.)



Citharichthys stigmaeus Jordan and Gilbert 1882; speckled sanddab; length to 17 cm; Prince William Sound, Alaska to Bahia Magdalena, southern Baja California; nearshore to 607 m; eyed side brown or tan with black speckles and spots; blind side white (Miller and Lea 1972; Eschmeyer et al. 1983).

Citharichthys sordidus (Girard 1854); Pacific sanddab; length to 41 cm; Sea of Japan to Bering Sea to Cabo San Lucas, southern Baja California; 0–549 m (Miller and Lea 1972; Eschmeyer et al. 1983); eyed side light brown, mottled with dark brown sometimes with yellow and orange spots; blind side whitish or light tan (Eschmeyer et al. 1983).

Citharichthys species, including the speckled and Pacific sanddabs, are commercially fished and generally found over sandy bottom, with the speckled sanddab generally found in shallow water bays and estuaries (Moser 1996; Rackowski and Pikitch 1989). Sanddab females are oviparous,

producing planktonic eggs and larvae (Moser 1996). According to Moser (1996), both species spawn year-round in the CalCOFI area with their highest abundance during the periods August–December for speckled sanddab, and January–February plus August–October for Pacific sanddab. In southern California, Goldberg and Pham (1987) found that speckled sanddab spawn March–October, while Ford (1965) reported that Pacific sanddabs spawn mainly from April–September.

Longevity and reproduction have been investigated in both species of sanddabs. Ford (1965) found that female speckled sanddabs begin to spawn in their second year and can produce up to three batches of eggs per year. They may produce 1,000–6,000 eggs per batch and may live over 4 yr (Ford 1965; **Table 5.2.11-1**). Some female Pacific sanddabs first mature at 2 yr (Smith 1936). Half of the older females are mature by 3 yr and all are mature by 4 yr (Arora 1951). Total lifespan may be over 11 yr (Love 1996). No estimates of annual egg production have been reported.

Both species of sanddabs can have extended planktonic durations. Sakuma and Larson's (1995) review reports that speckled sanddab (*Citharichthys stigmaeus*) larvae may be planktonic up to 324 d, while the Pacific sanddab (*Citharichthys sordidus*) may remain in the plankton for 271 d. Kendall (1992) found that speckled sanddabs probably settle after 113–324 d. Settlement sizes for speckled and Pacific sanddabs are 35 mm and 40 mm, respectively (Moser 1996). Larval growth rate of *Citharichthys* spp., mainly Pacific sanddab, for age-0 yr individuals was 0.19–0.30 mm/d, and the monthly instantaneous mortality rate was 0.437 in 1983 and 0.176 in 1984 (Rogers 1985). Speckled sanddab mortality during the egg and larval period is nearly 99.9% (Ford 1965; **Table 5.2.11-1**).

Summary of Field Collections

Sanddab larvae were collected at the DCPP intake structure during two periods: a minor pulse during December–March and a stronger pulse during July–September in all years 1996–1999 (**Appendix H**). Their highest density occurred in early August 1997 (**Figure 5.2.11-1**). The density during July–September 1998 was lower than during the same months in 1997, possibly

due to the El Niño event that occurred during the fall–winter seasons spanning 1997–98 (Lynn et al. 1998; NOAA 1999). There were 254 sanddabs identified from 170 bongo net subsamples representing 3.5% of the entrainment subsamples collected and processed from that location during this period. Of the total number identified from the entrainment subsamples, 113 were Pacific sanddabs, 71 were speckled sanddabs, and 70 were only identifiable to the generic level.

Sanddab larvae, notably *C. stigmaeus*, occurred year-round in the study grid (**Appendix H**), with their greatest density occurring during July 1997 and 1998 (**Figure 5.2.11-2**). The spatial distribution of sanddab larvae in the grid was inconsistent between surveys. Generally, they were collected farther from shore, except large numbers collected inshore in July 1997. There were 361 larval sanddabs identified from 43 bongo net subsamples representing 1% of the study grid subsamples collected and processed. Of the total number identified from the grid subsamples, 142 were Pacific sanddabs, 156 were speckled sanddabs, and 63 were identifiable to only the generic level.

Standard lengths of sanddab larvae in entrainment subsamples between October 1996 and June 1999 ranged from 1.3 mm to 32 mm (**Figure 5.2.11-3**). Only one larvae measuring 32 mm was collected; the next largest individual was 3.5 mm. The central 98% of the sanddab length-frequency distribution resulted in minimum and maximum lengths for the analyses of 1.3 mm and 3.5 mm, respectively. The growth rate of larval sanddabs (0.20 mm/d) reported by Kendall (1992) and Sakuma (1995) was used to estimate larval ages. The mean larval length in this distribution was 2.61 mm and approximately 6.5 d of age. The reported hatch length for *C. stigmaeus* is about 1.3 mm and less than 2.6 mm for *C. sordidus* (Moser 1996).

Lengths of sanddab larvae were collected for paired DCPP intake and study grid subsamples (**Figure 5.2.11-4**). However, very few measurable larvae occurred in the coincident subsamples: 4 at the DCPP intake and 52 in the study grid. These low sample sizes precluded statistical analyses and render difficult even qualitative comparisons between the two length distributions.

Estimating Total Annual Entrainment

The annual estimated mean entrainment and associated standard error for sanddab larvae show that few larval sanddabs were entrained (**Table 5.2.11-2**). Estimated entrainment was greater in 1996–97 Analysis Period 1 ($\widehat{E_T}$ = 7,160,000; $\widehat{SE}$ = 702,000) than in 1997–98 Analysis Period 2 ($\widehat{E_T}$ = 1,540,000; $\widehat{SE}$ = 283,000). The high larval abundance observed in summer 1997 produced the highest entrainment estimates. The apparent difference between the estimates for Analysis Period 2 (October 1997–September 1998) and Analysis Period 3 (July 1997–June 1998) illustrates the sensitivity of unadjusted annual entrainment estimates to the time period selected and to episodic oceanographic events. The estimates for these two periods were identical or very close in value for most target taxa. The estimated entrainment values for the analysis periods were not adjusted by the long-term weekly Intake Cove surface plankton tows (December–July), because sanddab spawning occurred mainly after the 7 mo surface tow period used for analysis (**Figure 5.2.11-5**).

Fecundity Hindcasting (FH)

Both *FH* and *AEL* methods required age-specific mortality, and *FH* required fecundity to assess entrainment effects. Egg and larval survival rates were estimated using Ford's (1965) estimate of 0.0015 survival through the first 90 d based on examination of fecundity, number of females, and transformed juveniles (**Table 5.2.11-1**). The calculated egg and larval daily survival was 0.930/d. An estimated age at entrainment of sanddab larvae was ca. 6.6 d, based on a growth rate of 0.2 mm/d and an average length at entrainment of 2.6 mm. Survival to entrainment was estimated as $0.93^{6.6} = 0.62$.

The *FH* approach combines larval entrainment losses and adult fecundity to hindcast the numbers of adult females effectively removed from the reproductively active population. An estimate of mean annual fecundity was 15,414 eggs/female based on a range of fecundity data for females measuring 85 to 148 mm in total length (**Table 5.2.11-3**; Ford 1965).

Age of maturation was assumed to be equal to 2.5 yr and longevity was assumed to be 6 yr from Ford (1965) and Arora (1951). The estimated total larval entrainment was used to estimate the

number of breeding females needed to produce the number of larvae entrained during the three analysis periods. A comparison of Analysis Periods 1 and 2 show that the value for 1996–97 ($\widehat{FH} = 426$; $\widehat{SE} = 996$) to be greater than the same time period for 1997–98 ($\widehat{FH} = 92$; $\widehat{SE} = 215$) **(Table 5.2.11-4)**.

Sensitivity Analysis

Fecundity had the greatest effect on recalculated values of $\widehat{FH}$ **(Table 5.2.11-5)**. Varying this parameter in the model resulted in an $\widehat{FH}$ range of ca. 18–2,210 adult females for Analysis Periods 1 and 2, compared to the original estimate of 92–426 adult females. Larval survivorship had the next greatest effect on recalculated $\widehat{FH}$.

Adult Equivalent Loss (AEL)

The *AEL* approach uses estimates of the abundance of entrained organisms to project the loss of equivalent numbers of adults based on mortality schedules and age at recruitment. Survival rates were estimated from entrainment of larval stage to recruitment into the fishery including juvenile to 2 yr adults **(Table 5.2.11-1)**. Larval survival was estimated from the average age at entrainment through settlement as

$$S_{Larvae} = 0.930^{(90-6.6)} = 0.002.$$

Survivorship to recruitment was apportioned into several age stages from Ford (1965; **Table 5.2.11-1**), and $\widehat{AEL}$ was calculated using the entrainment of a single age class having an average age of 6.6 d. Values of $\widehat{AEL}$ for Analysis Periods 1 and 2 show that the value for 1996–97 ($\widehat{AEL} = 2{,}370$; $\widehat{SE} = 6{,}710$) to be greater than the same time period for 1997–98 ($\widehat{AEL} = 511$; $\widehat{SE} = 1{,}450$) **(Table 5.2.11-6)**.

Sensitivity Analysis

Early larval survivorship had the greatest effect on recalculated values of $\widehat{AEL}$ **(Table 5.2.11-7)**. Varying this parameter in the model resulted in an $\widehat{AEL}$ range of 99–12,300 adult equivalents among the three analysis periods, compared to the original estimate of 511–2,372 adult

equivalents. Late larval through pre-recruit survivorship had the next greatest effect on $\widehat{AEL}$. The range of entrainment estimates did not substantially change the calculated values of $\widehat{AEL}$.

Empirical Transport Model (ETM)

Monthly estimates of proportional entrainment (PE_i) for sanddabs in the surveys from the two analysis periods ranged from a minimum of zero to a maximum of 0.859±0.911 ($\pm 1\widehat{SE}\left(\widehat{PE_i}\right)$) in April 1998 (**Table 5.2.11-8**). When both $\widehat{PE_i}$ and the annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) were equal to zero, no larvae were collected at either the DCPP intake or from the study grid. When $\widehat{PE_i} = 0$ but $\hat{f}_i > 0$, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. For sanddabs, the relatively large $\widehat{PE_i}$ for April 1998 resulted from the fact that no sanddab larvae were collected in the study grid. This $\widehat{PE_i}$ was calculated using the larval density from the weekly entrainment sample (**Appendix H**) multiplied by the unsampled volume inshore of the two cells nearest the intake (D1 and E1) to approximate the number of larvae in this volume. This number was then added to the number of larvae found in cells D1 and E1, zero in this case as well as elsewhere in the grid, to approximate the number of larvae in the whole study grid. The accompanying low value of $\hat{f}_i$ (0.00893) indicated that there were few sanddab larvae in the water column and, therefore, the high $\widehat{PE_i}$ had relatively low weight in the annual estimate of entrainment mortality. July and August of the first study year and May and June of the second year had higher values of f_i due to greater larval abundance in the subsamples from these months.

Estimates of $\widehat{P_M}$ between years showed the greatest differences in point estimates (**Figures 5.2.11-6a** and **b**). Estimated mean and maximum larval durations (7 or 11 d, respectively) were relatively similar, causing minimal change in point estimates of $\widehat{P_M}$ within-years. Since sanddab larvae were dispersed throughout the study grid, P_S was calculated using both onshore and alongshore current movements. Density was extrapolated offshore, with a common slope for all surveys of 0.0000342/m for Analysis Period 3 and 0.0000661/m for Analysis Period 4. The offshore extrapolation of study grid abundance, followed by an alongshore extension, yielded a

range of $\widehat{P_M}$ from both analysis periods of 0.4–4.5% of the larvae contained in an estimated population of approximately 3 to 22 times that in the study grid. The area of extrapolation ranged between 3 to 20 times the 52 km^2 area study grid. The extrapolated area was slightly less than the population because of the slightly increasing trend in density. In comparison, using only alongshore extension of study grid abundance, $\widehat{P_M}$ ranged from 1–7% of the larvae in an area two to four times the study grid alongshore or around 30–60 km of coastline distance.

Interpretation of Assessment Results

Estimates of adult equivalent loss, both *FH* and *AEL*, of sanddabs depend on estimated entrainment. Comparisons of the annual estimates of *FH* (92–426) and *AEL* (511–2,370) from Analysis Periods 1 and 2, that have comparable time periods, reflects the effects of the high entrainment estimates from summer 1997. The estimate of *AEL* assumes a recruitment age of 2.3 yr, while females in the fecundity hindcasting estimate range between 2.5 and 6 yr (3.7 yr assuming linear survival). The *FH* and *AEL* estimates are similar ($2 \cdot \widehat{FH}$ /0.25 = 736–3,408) when accounting for additional survival (**Table 5.2.11-1**) and equal number of males as females.

Sanddabs have both commercial and recreational fishery value. Catch data are available to compare harvest mortality rates to entrainment mortality rates. In 1998, an estimated 51,000 sanddabs were caught by recreational anglers in southern California waters (PSMFC RecFIN database). Commercial ex-vessel value in 1998 from ports between San Diego and Mendocino totaled $420,100, representing 595.6 MT (PSMFC PacFIN database). Value of sanddab landings in the port area of Morro Bay, that includes Port San Luis, in 1998 totaled $7,000, representing 12.7 MT (PSMFC PacFIN database).

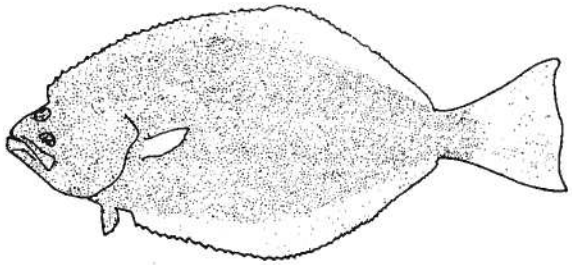
The estimated larval duration for sanddabs (7–11 d) produced a potential area of effects between 3–9 times the study area (for 7 d duration) and 9–20 times (for 11 d duration) that was used in computing an offshore extrapolation-based P_S (**Figure 5.2.11-6**). This indicates that only the commercial catch from the local ports of Morro Bay and Avila could potentially be affected by entrainment. An estimate of 60 g per average adult (Arora 1951) can be applied to the largest mortality predicted by the survival-weighted *FH* estimate. If we assume no compensatory

mortality and directly apply the *AEL*-based loss to the fishery catch, 3,400 fish represents 0.2 MT or $110. However, *FH* and *AEL* models do not consider selectivity, catchability, and fishing effort when extrapolated to fishery losses, so this estimate of monetary value is likely an overestimate of the loss to the fishery because it assumes all equivalent adults were harvested.

Alternately, estimates of annual entrainment mortality ($\widehat{P_M}$) could be used to determine the effect of larval entrainment on Morro Bay area catches. Proportional entrainment mortality ranged from ~1–7% over areas ranging from 3–22 times the study grid area (larger area associated with smaller mortality). These proportional losses can be applied to the 1993–1999 annual average revenue for the Morro Bay area sanddab catch of ca. $8,200/yr (PSMFC PacFIN Database) assuming no compensation. This results in an average dollar value impact on the Morro Bay area sanddab fishery of $82/yr, by translating a 1% larval mortality to adult losses (using the larger extrapolated areas still conservatively less than the fishery area).

Long-term trends, shown by Intake Cove plankton tows (**Figure 5.2.11-5**), show wide variation of larval abundance from 1990–1998 with no apparent trend. Since the startup of power plant operations, sanddabs have increased in Diablo Cove compared to pre-operation observations (Figure 4-39 in PG&E 1997c). Fishery harvest information indicates that catches in the MBNMS have generally increased between 1980 and 1995 (Starr et al. 1998), but there may be a declining trend for catches in the Morro Bay area since 1994 (**Figure 5.2.11-7**). Declining catches in the Morro Bay area could be due to many factors, but without data describing fishing effort it is difficult to attribute this apparent decline to any feature of the fishery. Since larval abundance does not appear to be declining in the Intake Cove, it seems likely that entrainment mortality is not reducing larval production locally or that compensation in birth rates is occurring.

5.2.11.2 California halibut (*Paralichthys californicus*)



Paralichthys californicus (Ayres 1859); California halibut: length to 152 cm; Quillayute River, northern Washington to Bahia Magdalena, southern Baja California; nearshore to 183 m; eyed side mostly uniform dark brown to black, but often mottled with light and dark; blind side usually white (Miller and Lea 1972; Eschmeyer et al. 1983).

California halibut is an important part of California's commercial and recreational fisheries (Leet et al. 1992; Starr et al. 1998). It is generally found on sandy substrata near rocks, kelp holdfasts, or other bottom structures (Love 1996). One-year-old individuals can be greater than 80 cm SL (Allen et al. 1990). Halibut can live to 30 yr (Frey 1971), with 50% of females mature by 4 yr and 100% mature by 7 yr (Love and Brooks 1990). They are oviparous, producing planktonic eggs and larvae (Oda 1991; Moser 1996) with external fertilization (Allen 1990). Halibut can spawn year-round, with peak spawning taking place during the late winter to spring and usually in water shallower than 75 m (Lavenberg et al. 1986). Spawning for the fish under natural environmental conditions was investigated in the laboratory during three different spawning years (Caddell et al. 1990; **Table 5.2.11-9**). They found during their laboratory study that halibut spawned 57 million eggs per year in an artificial environmental setting.

Little information on the population dynamics of larval California halibut was found in the scientific literature. In southern California, halibut less than 10 cm SL are found in greater numbers in bays than in coastal waters (Plummer et al. 1983). Peak densities of California halibut larvae occurred during February in CalCOFI samples (Moser et al. 1993). Moser and Watson (1990) found that less than 1% of the halibut larvae within the CalCOFI study area were collected in the central California zone that extends from just north of Point Conception to just north of San Francisco Bay. Allen and Jensen (unpubl. data in Allen et al. 1990) stated that newly settled larvae are about 8–12 mm and about 30 d old. Kramer (1990) stated that halibut 6–10 mm larvae

grew <0.3 mm/d, while larger 70–120 mm halibut grew about 1.0 mm/d. In a laboratory study, halibut held at 16°C grew to a length of 11.1 mm ± 2.61 (SD) in 2 mo from an initial hatch length of 1.9 mm (Gadomski et al. 1990). Instantaneous mortality rates (Z_{l_i}) were greatest for the smallest length classes, followed by lower, relatively stable rates for subsequent length classes up to 70 mm (**Table 5.2.11-10**).

In the 1985–1988 commercial catch, 63% of the California halibut were 6–7 yr (Sunada et al. 1990). Fish over 8 yr averaged about 18% of the catch, while a few 3 yr fish were large enough to be included in the fishery. No stock assessment has yet been published for this species (Bob Leos, CDF&G, Monterey, pers. comm.).

Summary of Field Collections

California halibut larvae were collected seasonally in the DCPP intake entrainment surveys during the years 1996–1999 (**Appendix H**). Halibut larvae at the intake occurred in two periods: March–May and July–August (**Figure 5.2.11-8**). The greatest apparent densities occurred during late April 1998. There were 378 California halibut larvae identified from 157 bongo net subsamples collected at the DCPP intake structure between October 1996 and June 1999 representing 3% of the entrainment subsamples collected and processed.

Within the DCPP study grid, California halibut larvae occurred sporadically during the two years surveyed (**Appendix H**), with their highest densities occurring during April–May and July 1998 (**Figure 5.2.11-9**). No clear distribution pattern was observed in the grid. The density estimates from the study grid represented 251 larval California halibut identified from 164 bongo net subsamples: 5% of the study grid subsamples collected and processed.

Standard lengths of California halibut larvae collected at the DCPP intake structure ranged from 1.5 mm to 8.0 mm (**Figure 5.2.11-10**). The growth rate of larval California halibut (0.28 mm/d) reported by Allen et al. (1990) was used to estimate ages of entrained larvae. The central 98% of this larval length-frequency distribution resulted in minimum and maximum lengths for the analyses of 1.6 mm and 7.8 mm, respectively. The mean larval length in this distribution was

3.28 mm and approximately 6 d post-hatch. Reported hatching size for California halibut ranges 1.5–1.8 mm (Moser 1996,) and it is one of the smaller larvae entrained at DCPP. The collection of just a few California halibut larvae from the paired entrainment and study grid surveys precluded statistical comparisons of larval length (**Figure 5.2.11-11**).

Estimating Total Annual Entrainment

Annual estimated entrainment of California halibut larvae was greater in 1997–98 Analysis Period 2 ($\widehat{E_T}$ = 14,300,000; $\widehat{SE}$ = 2,130,000) than in 1996–97 Analysis Period 1 ($\widehat{E_T}$ = 2,270,000; $\widehat{SE}$ = 296,000) (**Table 5.2.11-11**). These estimates represent a total of only 378 individual larval California halibut collected at the DCPP intake structure. Although the number of larval California halibut entrained was low it was chosen for assessment because of its importance in sport and commercial fisheries.

Estimated annual entrainment of California halibut larvae was adjusted (**Table 5.2.11-12**) to the long-term average index ($\bar{I} / I_i$) estimated from Intake Cove surface plankton tows (**Figure 5.2.11-12**). The index was calculated as the ratio between the nine year average ($\bar{I}$) of Intake Cove sampling and the average annual index estimated from the plankton tows during the year being adjusted (I_i). The average indices for the years 1997 and 1998 (I_i) were 0.0005 and 0.0016 larvae/m^3, respectively. The long-term average index ($\bar{I}$) was 0.0017 larvae/m^3 for the years 1990–1998, yielding the ratio $\bar{I} / I_i$ of 3.64 for 1997 and 1.10 for 1998 indicating that Intake Cove density in these years was lower than the long-term average. Annual adjusted entrainment of California halibut larvae was estimated to be 8,260,000 in 1996–97 Analysis Period 1 ($\widehat{SE}$ = 1,080,000) and 15,700,000 in 1997–98 Analysis Period 2 ($\widehat{SE}$ = 2,340,000) (**Table 5.2.11-12**).

NOTE: Annual entrainment estimates of California halibut larvae were not used to estimate *FH* or *AEL* because no larval survival data were available to parameterize these models. However, an approximate *FH* estimate is made in the interpretation section below.

Empirical Transport Model (ETM)

Monthly estimates of proportional entrainment (PE_i) for California halibut in each *i*th survey from the two years sampled ranged from a minimum of zero to a maximum of 0.859±0.911 (±1 $\widehat{SE}\left(\widehat{PE_i}\right)$) in April 1999 (**Table 5.2.11-13**). During periods when no larvae were collected at either the DCPP intake or from the study grid, both the annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and $\widehat{PE_i}$ were equal to zero. When $\widehat{PE_i} = 0$ but $\hat{f}_i > 0$, larvae were collected at the DCPP intake during the survey period but not during the entrainment survey paired with the 72-hour study grid survey. The maximum $\widehat{PE_i}$, recorded in April 1999, was the result of using study grid estimates based only on inshore volumes as no larvae were found in study grid cells. That is, *PE* for this survey period was calculated using the larval density from the weekly entrainment sample (**Appendix H**) multiplied by the unsampled volume inshore of the two cells nearest the intake (D1 and E1) to approximate the number of larvae within this volume. The product was added to the abundance in cells D1 and E1, zero in this case as well as elsewhere in the grid, to approximate the number of larvae in the whole study grid. The low value of f_i during April 1999 indicated that California halibut were not as prevalent as they were during August 1998 or March 1999. In comparison with $\widehat{PE_i}$ a month earlier, their distribution was more inshore during April 1999.

Estimates of $\widehat{P_M}$ showed greater differences between years than between larval durations during the same year (**Figures 5.2.11-13**). Since halibut larvae were dispersed throughout the study grid, P_S was calculated using both onshore and alongshore current movements with a common slope used for offshore extrapolation of study grid abundance of 0.0000252/m for Analysis Period 3 and 0.0000127/m for Analysis Period 4. Using offshore extrapolation of study grid abundance pattern followed by alongshore extension yielded a range of $\widehat{P_M}$ from both analysis periods of 0.07–0.08% (1997–1998) and 5–12% (1998–1999) of the larvae contained in a population area approximately 4 to 39 times the study grid area (and 4 to 35 times the grid's estimated larval population size). In comparison, $\widehat{P_M}$ results using only alongshore extension of study grid abundance also reflected differences between the years' results, ranging from 0.3–25% of the

larvae in an area two to six times the study grid alongshore or around 41–105 km of coastline distance.

Interpretation of Assessment Results

California halibut have both commercial and recreational fishery value. Commercial vessel landing prices in 1998 from ports between San Diego and Mendocino totaled $2,746,700 representing 532.1 MT (PSMFC PacFIN database). During the same year, landings in the Morro Bay area totaled about 10% of the catch from this larger area (54.8 MT and $264,000; PSMFC PacFIN database). In 1998, recreational fishers in southern California caught 66,000 adult California halibut (PSMFC RecFIN database).

The estimated age of larval California halibut entrained at DCPP averaged six days with a maximum estimated age of 22 d. The average duration, combined with current meter data, produced an extrapolated reference population that ranged from 4–8 times the study area between sampling years, while the maximum duration produced a reference population contained in an area 23–39 times the study grid. The population contained within the study grid and extended alongshore using current meter data and both average and maximum larval age at entrainment ranged over shoreline distances of 10–170 km. Therefore, we assumed that the commercial catch likely affected by entrainment was limited to the local ports of Morro Bay and Port San Luis.

Estimates of stock size or adult density that could be used to convert $\widehat{P_M}$ numbers of adult fishes were not available for this species. However, proportional losses predicted by *ETM* can be applied to the 1993–1999 average value of the local catch of $161,000/yr (PSMFC PacFIN Database) assuming no compensation. Thus, the proportional dollar loss to this fishery in the Morro Bay area could range from ca. $120–20,000 based on $\widehat{P_M}$ ranging from 0.00076–0.123 from the combined offshore extrapolated and alongshore extension of the study grid abundance.

Between-year differences in larval California halibut density at DCPP could be the result of changes in nearshore, coastal transport processes or may reflect some other factors. Relatively higher abundance observed in the study grid during the 1997–98 El Niño event may have been

due to increased shoreward transport at DCPP (out of the south) or could be related to suppressed upwelling during that period. However, the nine years of Intake Cove plankton tow data indicate that larval California halibut density does not vary greatly from the long-term average with the exception of a single strong year-class in 1992. Additionally, catches in the MBNMS appear to be relatively stable or only slightly decreasing between 1980 and 1995 (Starr et al. 1998); local Morro Bay area catches show a similarly stable catch between 1993 and 1999 (**Figure 5.2.11-14**). One explanation for the apparent between-year differences of entrainment effects is high interannual variability within the study period due to low sample size: only 378 California halibut larvae were collected over the course of the study. The large 1998–99 estimated larval mortality is based on a limited number of California halibut larvae.

Using the adjusted entrainment estimated over 1996–97, an estimate of *FH* can be made assuming a relatively small larval survival of 0.5/d during six days, average yearly fecundity of 4.9 million eggs (**Table 5.2.11-9**), and reproductive life of 11.5 yr as

$$\widehat{FH} = \frac{\hat{E}_{Adj-T}}{\hat{S} \times \hat{\bar{F}}_T} = \frac{8.3 \times 10^6}{0.5^6 \times \left(4.9 \times 10^6\right) \times \left(\frac{30-7}{2}\right)} = 9 \text{ females}$$

An estimate similarly can be made for 1997–98 as 18 females. These small adult female losses indicate that while $\widehat{P_M}$ is large for 1998–99, the population impact is probably slight.

Table 5.2.11-1. Life table for speckled sanddab (*C. stigmaeus*) covering the probable lifespan, where *x* is the age in months, *bx* the number of eggs spawned per female within the interval and *lx* the number surviving at the start of the interval. Separate *lx* columns are given for the 1962–63 and 1963–64 survival data. Values of *lx* for age group 3 were estimated by assuming the same mortality rate for age 2 and 3 fish. Intrinsic rate of natural increase, intrinsic birth and mortality rate, and generation length, based on these data, are given for 1962–63 and 1963–64 (Table 25 in Ford 1965).

Life stage/Age (yrs)	Age (mo)	eggs per time interval per female *bx*	1962–63 *lx*	1963–64 *lx*	Survival
Eggs and larvae	0–3	0	100,000	100,000	0.00150
0	3–6	0	150	150	0.563
	6–9	0	82	87	0.567
1	9–12	0	45	51	0.861
	12–15	3,500	44	38	0.859
	15–18	700	42	29	0.867
	18–21	0	41	22	0.851
2	21–24	0	40	16	0.788
	24–27	1,010	33	12	0.742
	27–30	2,000	27	8	0.782
	30–33	0	22	6	0.742
3	33–36	0	18	4	0.764
	36–39	1,860	14	3	0.762
	39–42	3,700	12	2	0.667
	42–45	0	10	1	

S = 0.9316 (p. 5-367)

S = 0.31 for age 0

Ave = 0.731

$S_{age\,03} \approx (0.731)^4$

Table 5.2.11-2. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for sanddab (*Citharichthys* spp.) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	7,160,000	702,000
2) Oct. 1, 1997–Sep 30, 1998	1,540,000	283,000
3) Jul 1, 1997–Jun 30, 1998	6,610,000	704,000

Table 5.2.11-3. Estimated number of eggs spawned per season by speckled sanddabs (*Citharichthys stigmaeus*) collected during April and May 1963 (Ford 1965)

Total length	Total # mature and maturing eggs
85.0	4,300
90.5	4,100
109.5	15,200
110.5	9,000
129.0	20,900
130.0	23,600
148.0	30,800

Table 5.2.11-4. Estimated number of adult sanddab (*Citharichthys* spp.) females ($\widehat{FH}$) whose reproductive output was equivalent to the number of larvae entrained per year at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{FH})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{FH}$	$\widehat{SE}(\widehat{FH})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	426	996	19,900	9
2) Oct. 1, 1997–Sep 30, 1998	92	215	4,320	2
3) Jul 1, 1997–Jun 30, 1998	393	919	18,400	8

Table 5.2.11-5. Sanddabs (*Citharichthys* spp.): Sensitivity analysis for $\widehat{FH}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 426				
$\hat{E}_{Adj-T}$	7,160,000	357	495	137
$\hat{S}$ *larvae*	0.623	265	2,210	1,940
# Eggs/yr	15,400	82.3	2,210	2,130
Longevity	6	157	852	695
Maturation	2.5	314	1,490	1,180

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 92				
$\hat{E}_{Adj-T}$	1,540,000	64.0	119	55.4
$\hat{S}$ *larvae*	0.623	57.1	475	418
# Eggs/yr	15,400	17.7	475	457
Longevity	6	33.8	183	150
Maturation	2.5	67.6	321	253

c) Analysis Period 3: July 1, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{FH}$ Minimum	Maximum	Range
$\widehat{FH}$ = 393				
$\hat{E}_{Adj-T}$	6,610,000	324	462	138
$\hat{S}$ *larvae*	0.623	245	2,040	1,790
# Eggs/yr	15,400	75.9	2,040	1,960
Longevity	6	145	787	642
Maturation	2.5	290	1,380	1,090

Table 5.2.11-6. Estimated number of equivalent sanddab (*Citharichthys* spp.) adults ($\widehat{AEL}$) equal to the number of larvae entrained per year at Diablo Canyon Power Plant including the standard error of the estimate ($\widehat{SE}(\widehat{AEL})$) and 90% confidence limits (C.L.).

Analysis Period	$\widehat{AEL}$	$\widehat{SE}(\widehat{AEL})$	Upper 90% C.L.	Lower 90% C.L.
1) Oct 23, 1996–Sep 30, 1997	2,370	6,710	249,000	23
2) Oct 1, 1997–Sep 30, 1998	511	1,450	54,000	5
3) Jul 1, 1997–Jun 30, 1998	2,190	6,200	18,400	8

Table 5.2.11-7. Sanddab (*Citharichthys* spp.): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

a) Analysis Period 1: October 1, 1996–September 30, 1997

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL} = 2{,}372$				
$\hat{E}_{Adj-T}$	162,000	1,990	2,750	765
$\hat{S}$ *early larvae*	0.00241	458	12,300	11,800
$\hat{S}$ *late larvae*	0.563	458	4,210	3,750
$\hat{S}$ *early juvenile*	0.567	458	4,180	3,720
$\hat{S}$ *juv. I*	0.861	458	2,750	2,300
$\hat{S}$ *juv. II*	0.859	458	2,760	2,300
$\hat{S}$ *juv. III*	0.867	458	2,730	2,280
$\hat{S}$ *juv. IV*	0.851	458	2,790	2,330
$\hat{S}$ *pre-recruit*	0.788	458	3,010	2,550

Table 5.2.11-7 (continued). Sanddab (*Citharichthys* spp.): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

b) Analysis Period 2: October 1, 1997–September 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL} = 511$				
$\hat{E}_{Adj-T}$	1,540,000	356	665	308
$\hat{S}$ *early larvae*	0.00241	99	2,650	2,550
$\hat{S}$ *late larvae*	0.563	99	906	808
$\hat{S}$ *early juvenile*	0.567	99	900	801
$\hat{S}$ *juv. I*	0.861	99	593	494
$\hat{S}$ *juv. II*	0.859	99	594	496
$\hat{S}$ *juv. III*	0.867	99	589	490
$\hat{S}$ *juv. IV*	0.851	99	600	501
$\hat{S}$ *pre-recruit*	0.788	99	648	550

Table 5.2.11-7 (continued). Sanddab (*Citharichthys* spp.): Sensitivity analysis for $\widehat{AEL}$ from the three analysis periods recalculated for each model input parameter (θ): $\hat{E}_{Adj-T} \pm 1.645\ \widehat{SE}(\hat{\theta})$; other parameters are $\theta \cdot e^{\pm 1.645 CV(\hat{\theta})}$ where $CV(\theta) = 1.0$ or 100%.

c) Analysis Period 3: July, 1997–June 30, 1998

Parameter	Estimate	Recalculated $\widehat{AEL}$ Minimum	Maximum	Range
$\widehat{AEL} = 2{,}190$				
$\hat{E}_{Adj-T}$	6,610,000	1,810	2,570	768
$\hat{S}$ *early larvae*	0.00241	423	11,300	10,900
$\hat{S}$ *late larvae*	0.563	423	3,890	3,460
$\hat{S}$ *early juvenile*	0.567	423	3,860	3,440
$\hat{S}$ *juv. I*	0.861	423	2,540	2,120
$\hat{S}$ *juv. II*	0.859	423	2,550	2,130
$\hat{S}$ *juv. III*	0.867	423	2,520	2,100
$\hat{S}$ *juv. IV*	0.851	423	2,570	2,150
$\hat{S}$ *pre-recruit*	0.788	423	2,580	2,360

Table 5.2.11-8. Sanddab (*Citharichthys spp.*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.00291	0.000750	0.645	0.0172
Aug 25, 1997	0.00101	0.00107	0.192	0.0127
Sep 29, 1997	0	0	0.0235	0.00452
Oct 20, 1997	0	0	0.00528	0.00200
Nov 17, 1997	0	0	0.000970	0.000970
Dec 10, 1997	0	0	0.0246	0.00516
Jan 22, 1998	0	0	0.0966	0.00966
Feb 26, 1998	0	0	0.00259	0.00116
Mar 18, 1998	0	0	0	0
Apr 15, 1998	0.859	0.911	0.00893	0.00269
May 18, 1998	0	0	0	0
Jun 8, 1998	0	0	0	0

b) Analysis Period 4: July 1, 1998–June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.00115	0.000840	0.126	0.0189
Aug 26, 1998	0.000740	0.000750	0.0467	0.0112
Sep 16, 1998	0.000540	0.000550	0.0453	0.0103
Oct 6, 1998	0.00271	0.00279	0.128	0.0167
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0	0	0	0
Jan 12, 1999	0	0	0	0
Feb 3, 1999	0	0	0	0
Mar 17, 1999	0	0	0	0
Apr 14, 1999	0	0	0	0
May 24, 1999	0.00855	0.00502	0.328	0.0277
Jun 23, 1999	0.0430	0.0416	0.327	0.0387

Table 5.2.11-9. Average spawn per female California halibut (*Paralichthys californicus*). Spawning frequency is average days between spawning events for each year. Data from Caddell et al. (1990).

Years	No. spawns	Average number eggs per spawn	Number of eggs spawned per year	Spawning frequency (d)
1985–86	12	455,000	5,460,000	7.0
1986–87	5	313,000	1,565,000	7.2
1987–88	13	589,000	7,657,000	14.0

Average

Table 5.2.11-10. Mortality rates for juvenile California halibut (*Paralichthys californicus*) for the 1988 year-class up to 115 d. Data from Kramer (1990)

instantaneous

Length class (mm)	Age class (d)	$z(t_i)$
<=10	30.3	
11–15	43.3	0.044
16–20	53.3	0.016
21–25	61.8	0.015
26–30	69.3	0.013
31–35	76.2	0.013
35–40	82.5	0.013
41–45	88.4	0.011
46–50	94.1	0.014
51–55	99.5	0.012
56–60	104.8	0.011
61–65	109.9	0.013
66–70	114.9	0.012

t_i 13d 10d 8.5 7.5 6.9 6.3 5.9 5.7 5.4 5.3 5.1 5.0

$S = e^{-(\sum z_i t_i)}$

Table 5.2.11-11. Estimated total annual entrainment ($\widehat{E_T}$) and standard error ($\widehat{SE}(\widehat{E_T})$) for California halibut (*Paralichthys californicus*) larvae from the three analysis periods.

Analysis Period	$\widehat{E_T}$	$\widehat{SE}(\widehat{E_T})$
1) Oct 23, 1996–Sep 30, 1997	2,270,000	296,000
2) Oct 1, 1997–Sep 30, 1998	14,300,000	2,130,000
3) Jul 1, 1997–Jun 30, 1998	14,100,000	2,130,000

Table 5.2.11-12. Estimated total annual adjusted entrainment ($\hat{E}_{Adj-T}$) and standard error ($\widehat{SE}(\hat{E}_{Adj-T})$) for California halibut (*Paralichthys californicus*) larvae estimated for the three analysis periods.

Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{SE}(\hat{E}_{Adj-T})$
1) Oct 23, 1996–Sep 30, 1997	8,260,000	1,080,000
2) Oct 1, 1997–Sep 30, 1998	15,700,000	2,340,000
3) Jul 1, 1997–Jun 30, 1998	15,500,000	2,330,000

Table 5.2.11-13. California halibut (*Paralichthys californicus*): Monthly estimates of proportional entrainment ($\widehat{PE_i}$) and annual proportion of larvae hatched in the *i*th survey period ($\hat{f}_i$) and associated standard errors (*SE*) used in estimating entrainment mortality for two analysis periods.

a) Analysis Period 3: July 1, 1997–June 30, 1998

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1997	0.000700	0.000530	0.0186	0.00284
Aug 25, 1997	0	0	0.0421	0.00582
Sep 29, 1997	0	0	0.00283	0.00107
Oct 20, 1997	0	0	0	0
Nov 17, 1997	0	0	0	0
Dec 10, 1997	0	0	0.00425	0.00127
Jan 22, 1998	0.0127	0.0155	0.0115	0.00240
Feb 26, 1998	0	0	0	0
Mar 18, 1998	0	0	0.000850	0.000600
Apr 15, 1998	0.00143	0.000650	0.761	0.0146
May 18, 1998	0	0	0.156	0.0109
Jun 8, 1998	0	0	0.00273	0.00104

b) Analysis Period 4: July 1, 1998– June 30, 1999

Survey Start Date	$\widehat{PE_i}$	$\widehat{SE}\left(\widehat{PE_i}\right)$	$\hat{f}_i$	$\widehat{SE}\left(\hat{f}_i\right)$
Jul 21, 1998	0.00136	0.000690	0.0910	0.00984
Aug 26, 1998	0	0	0.0249	0.00376
Sep 16, 1998	0	0	0	0
Oct 6, 1998	0	0	0	0
Nov 11, 1998	0	0	0	0
Dec 9, 1998	0	0	0	0
Jan 12, 1999	0	0	0	0
Feb 3, 1999	0	0	0	0
Mar 17, 1999	0.0751	0.0372	0.861	0.0112
Apr 14, 1999	0.859	0.911	0.0227	0.00392
May 24, 1999	0	0	0	0
Jun 23, 1999	0	0	0	0

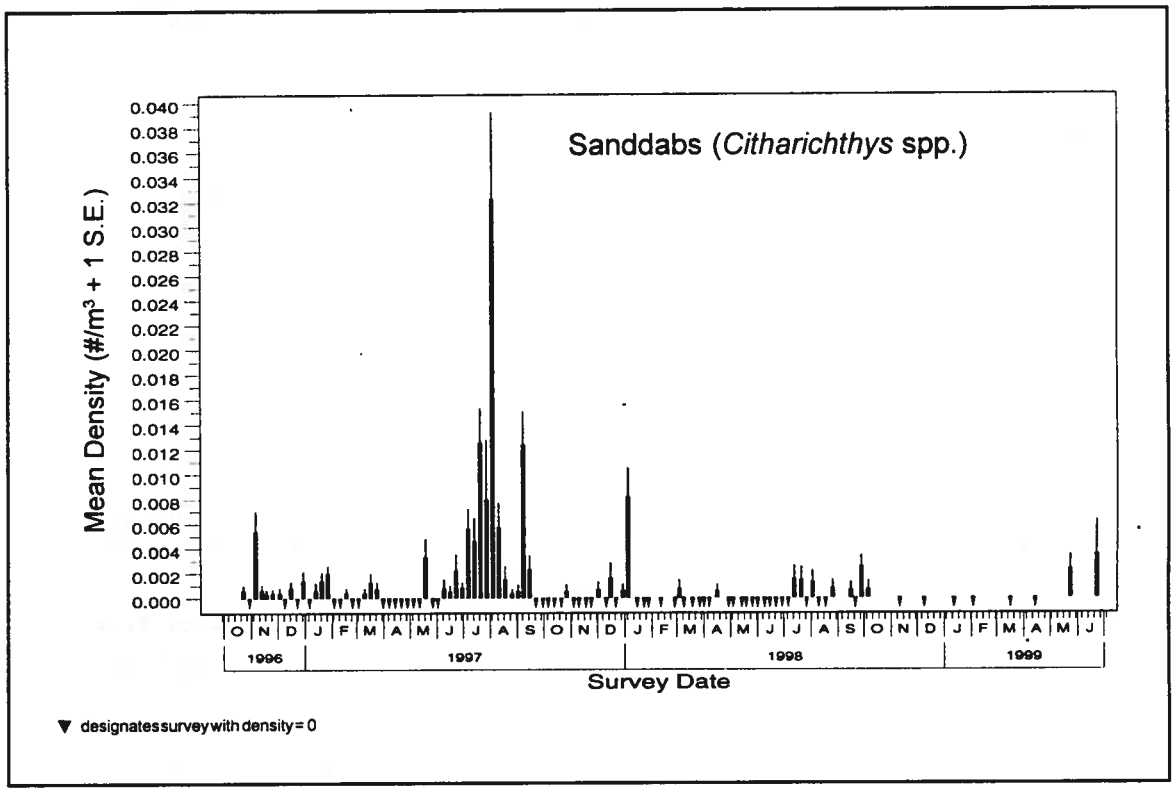


Figure 5.2.11-1. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

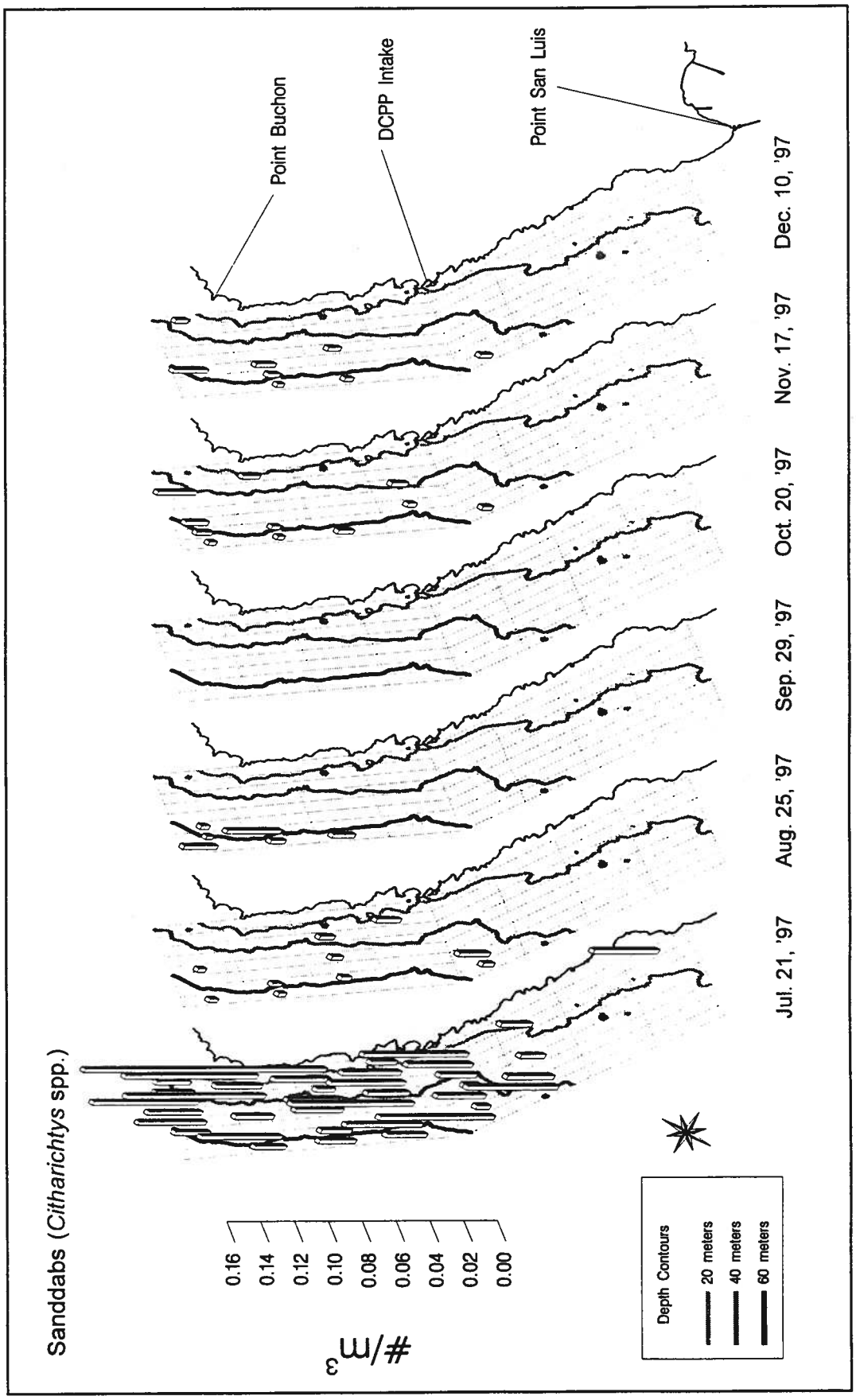


Figure 5.2.11-2a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

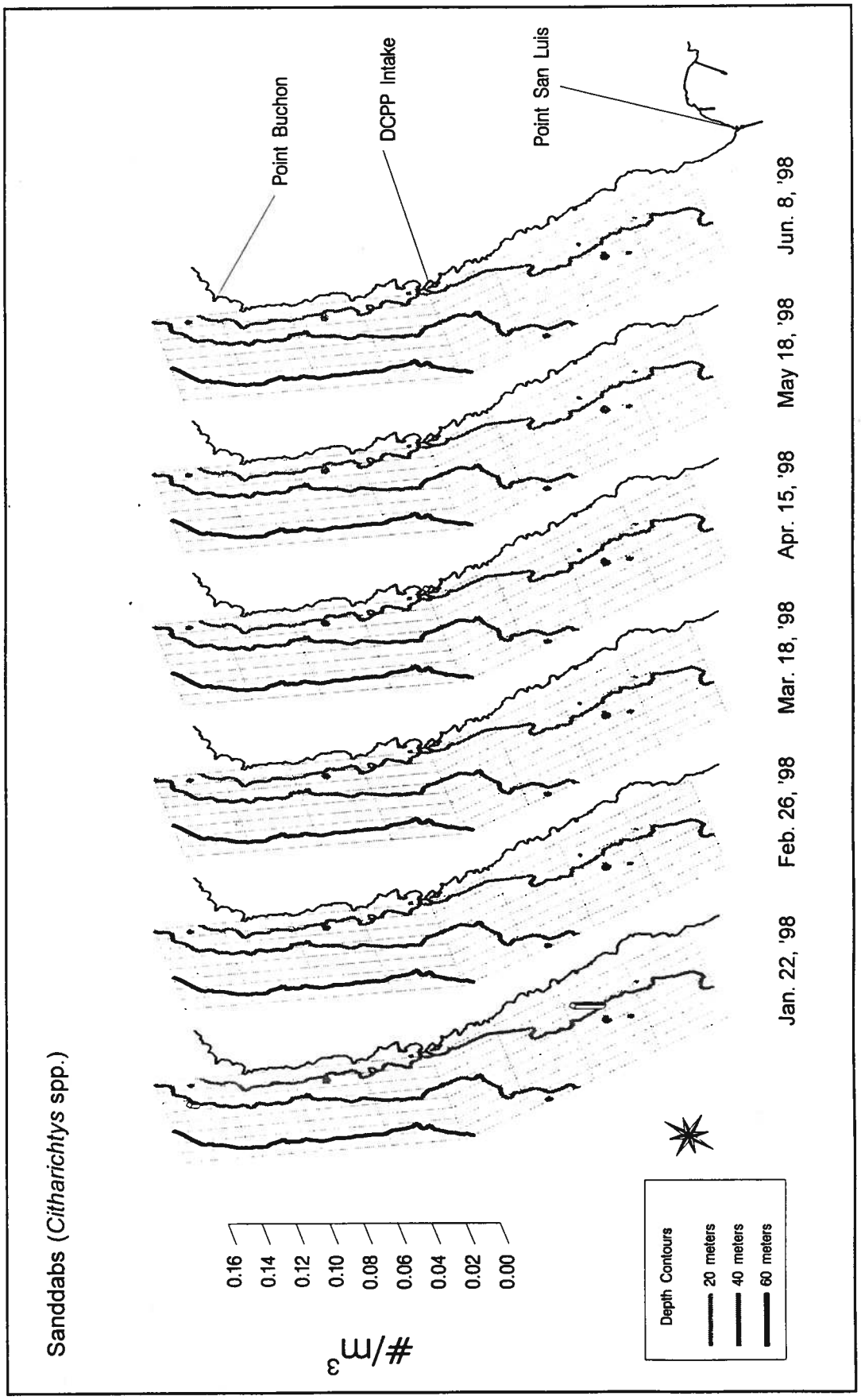


Figure 5.2.11-2b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

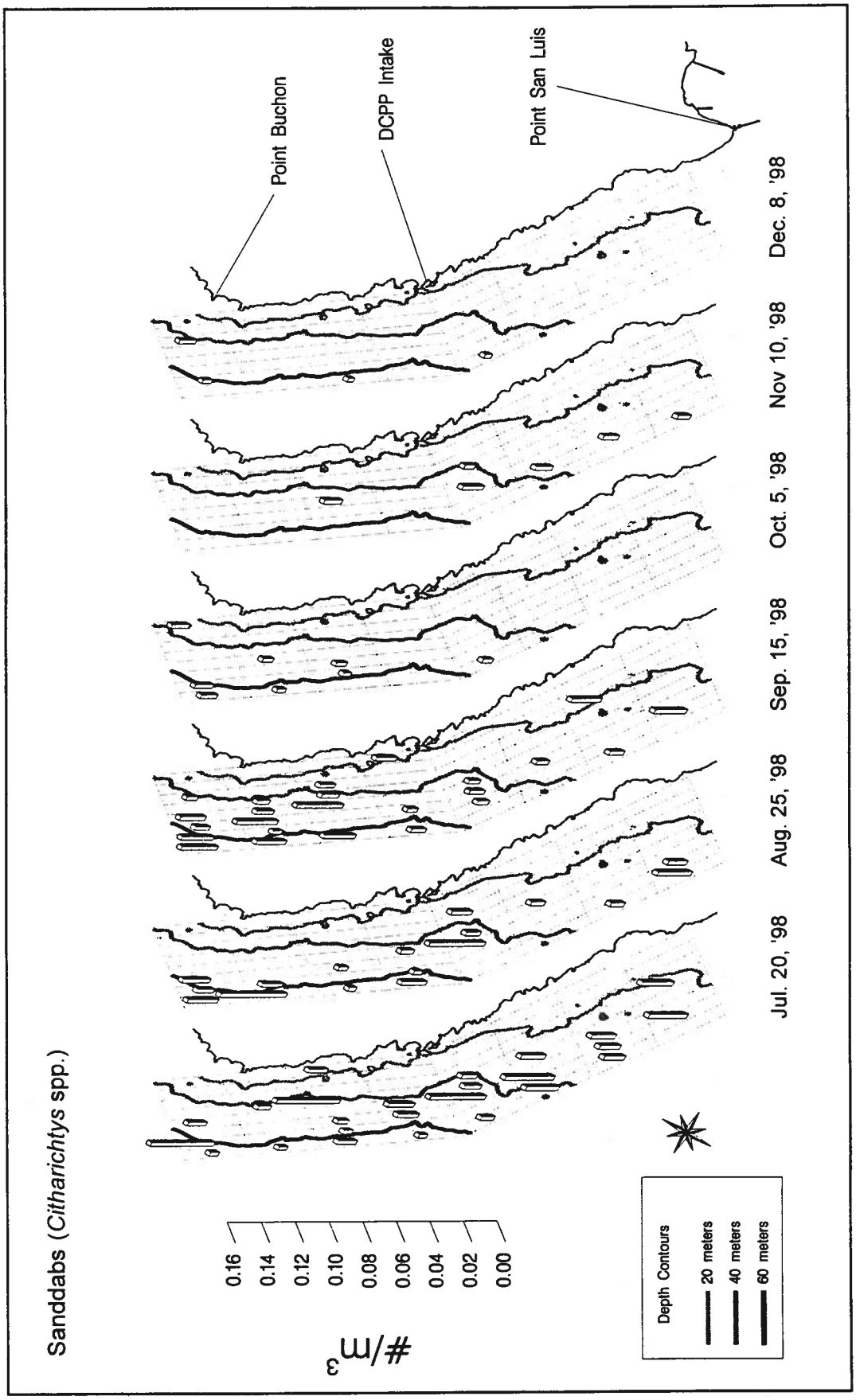


Figure 5.2.11-2c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

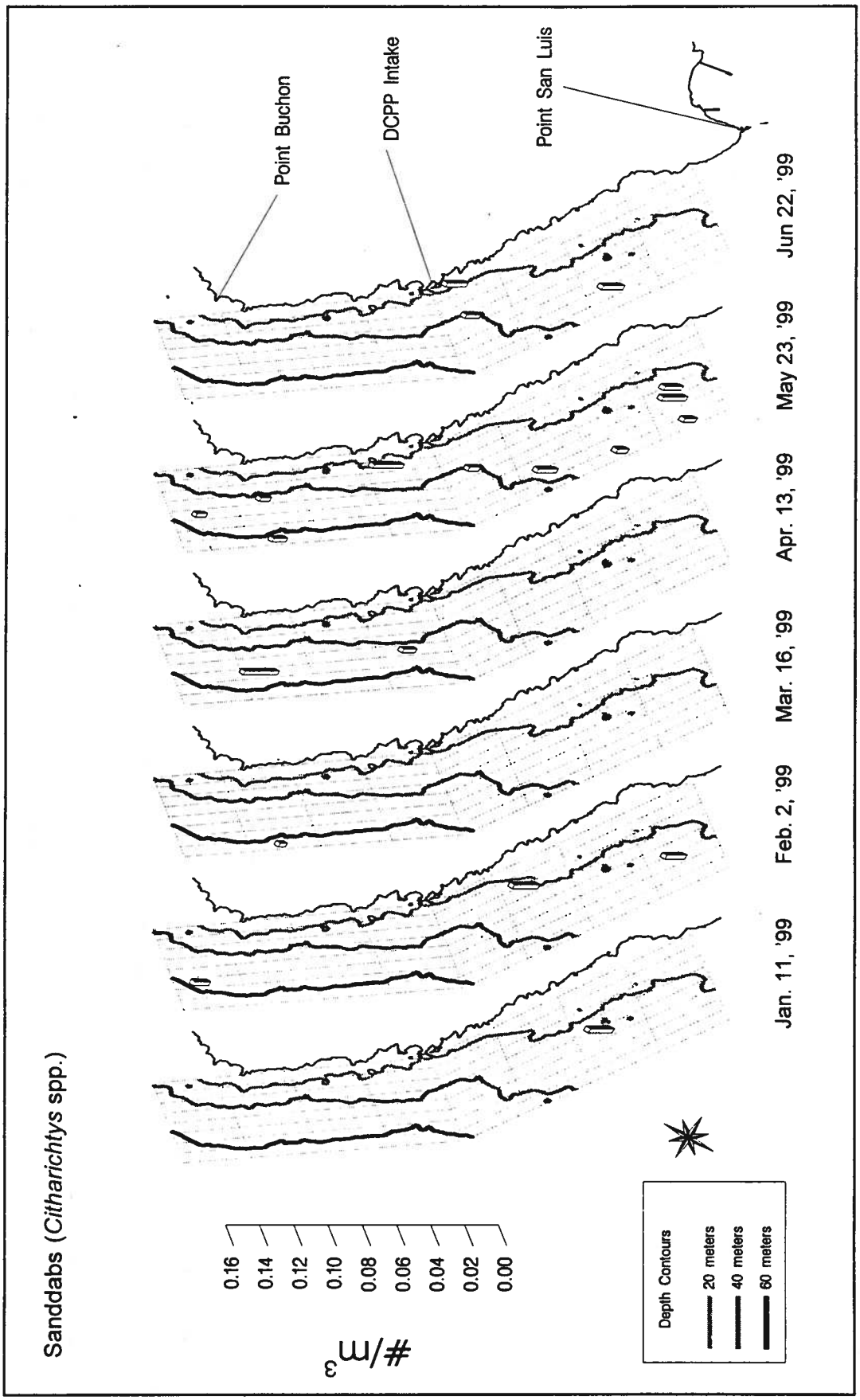


Figure 5.2.11-2d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

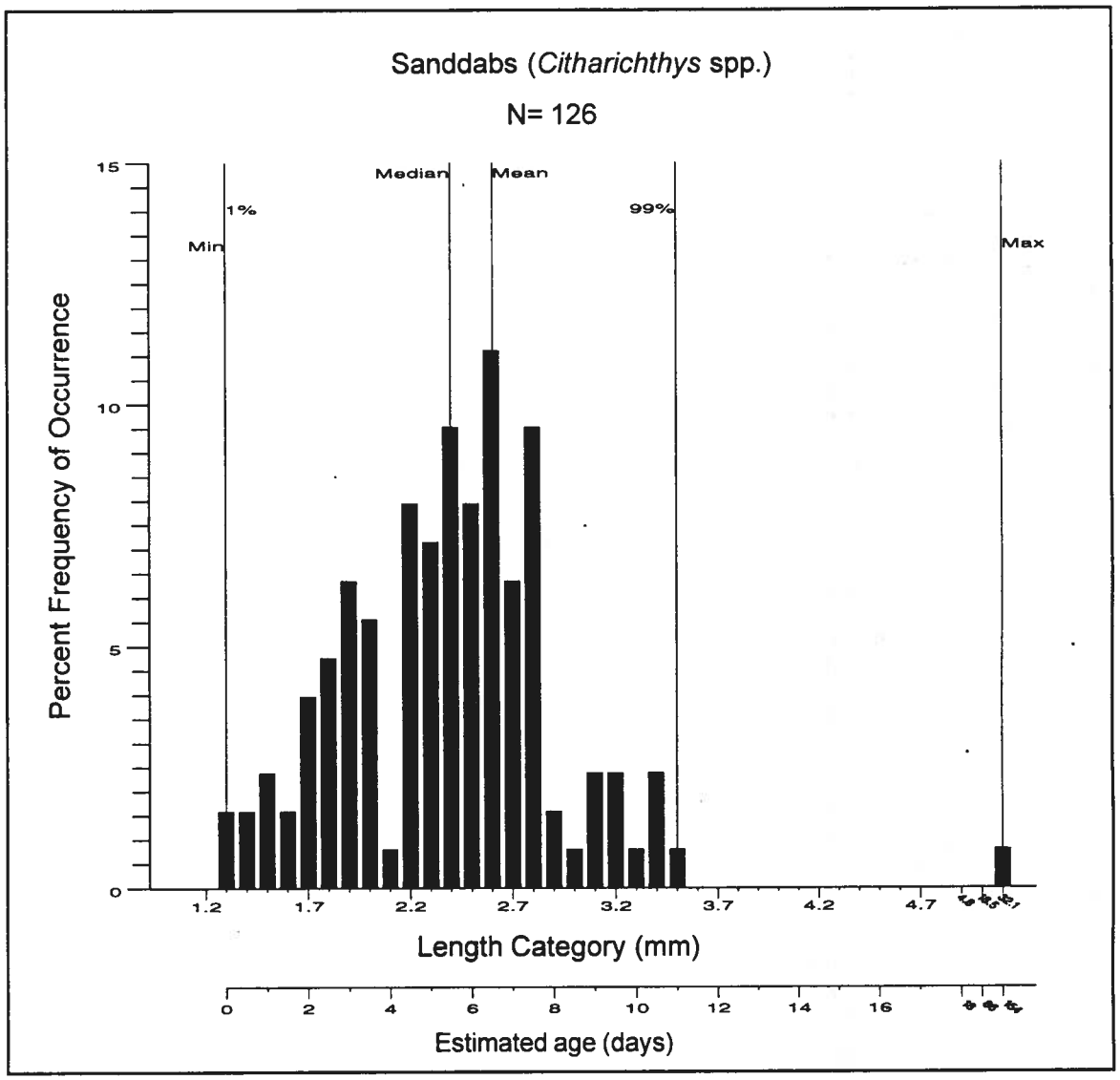


Figure 5.2.11-3. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

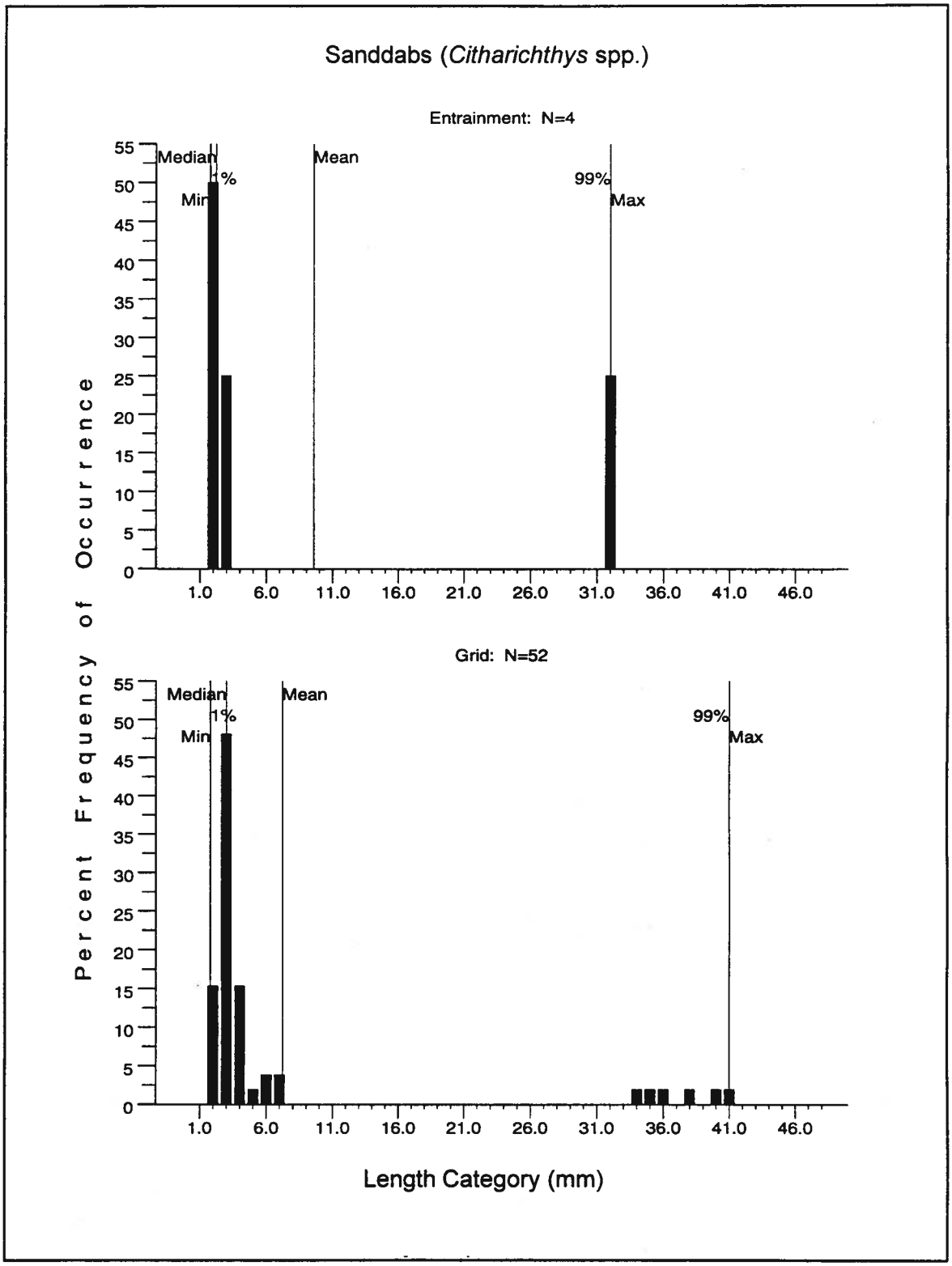


Figure 5.2.11-4. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

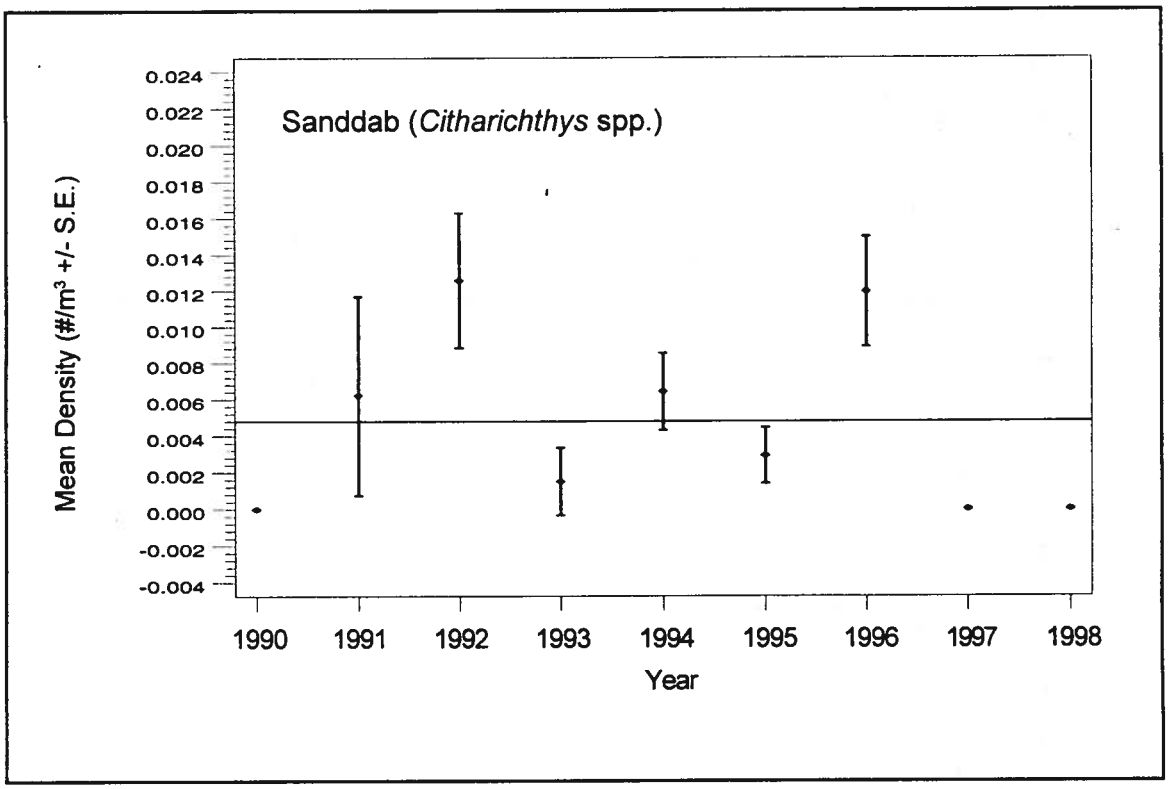


Figure 5.2.11-5. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

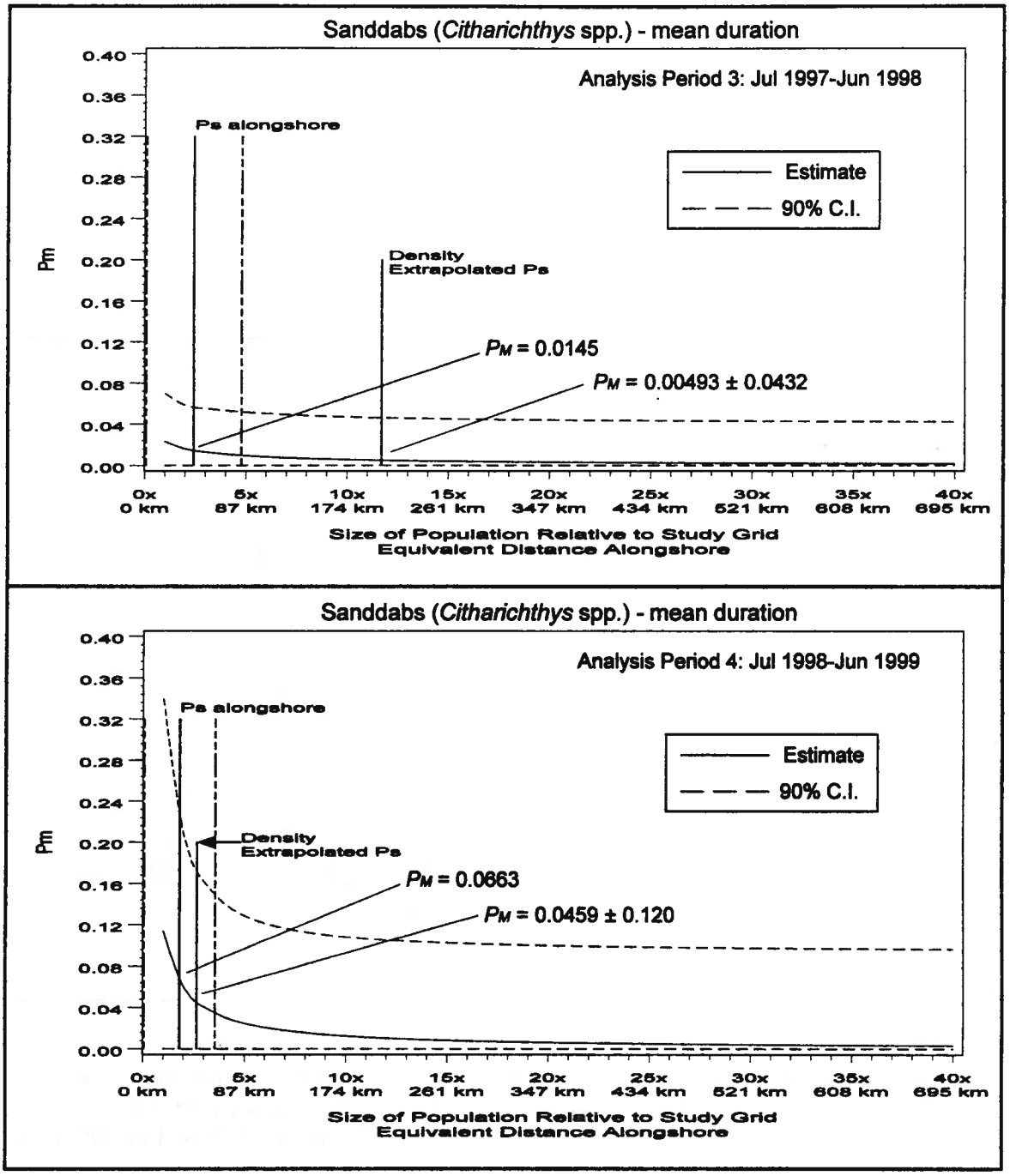


Figure 5.2.11-6a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

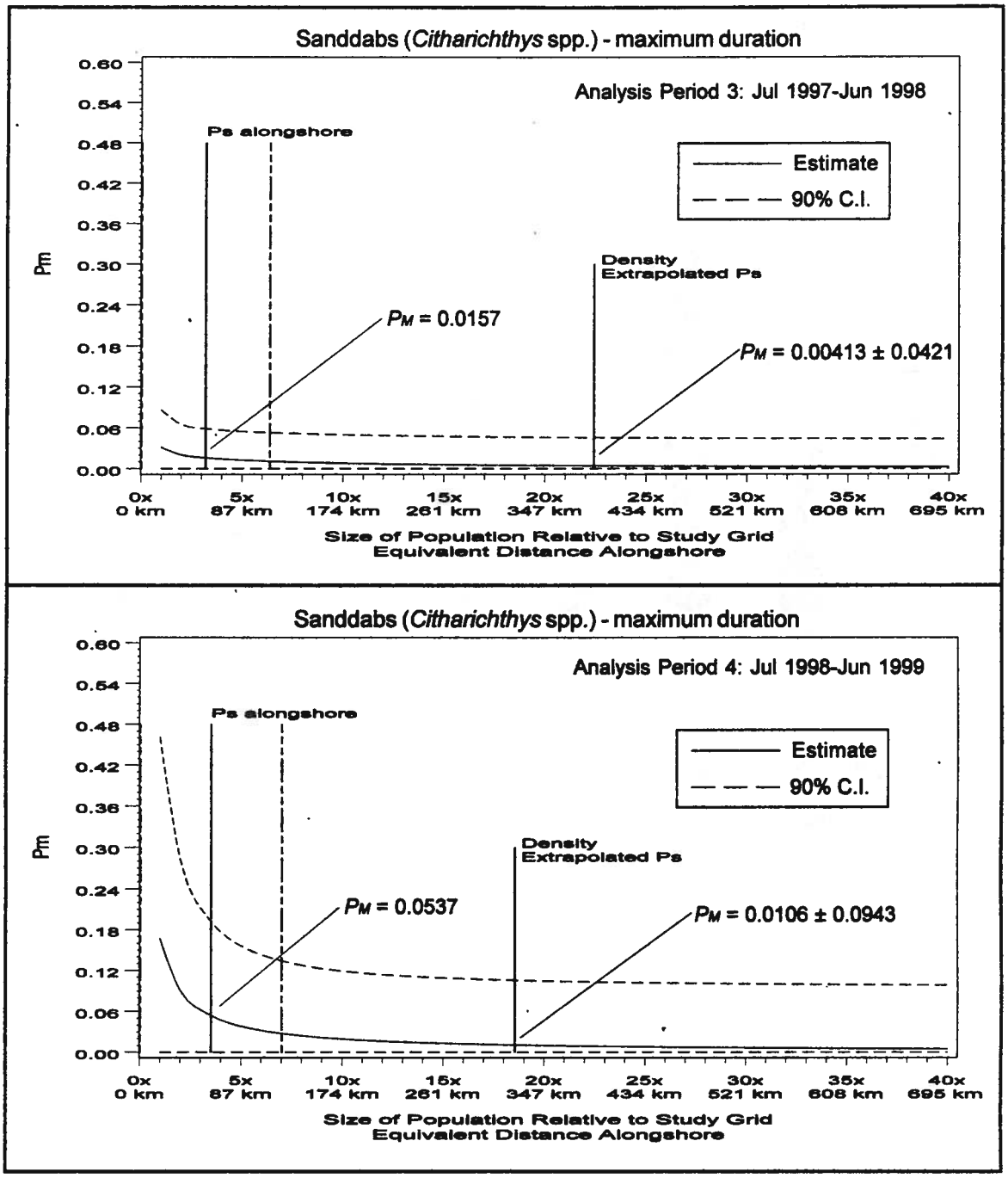


Figure 5.2.11-6b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

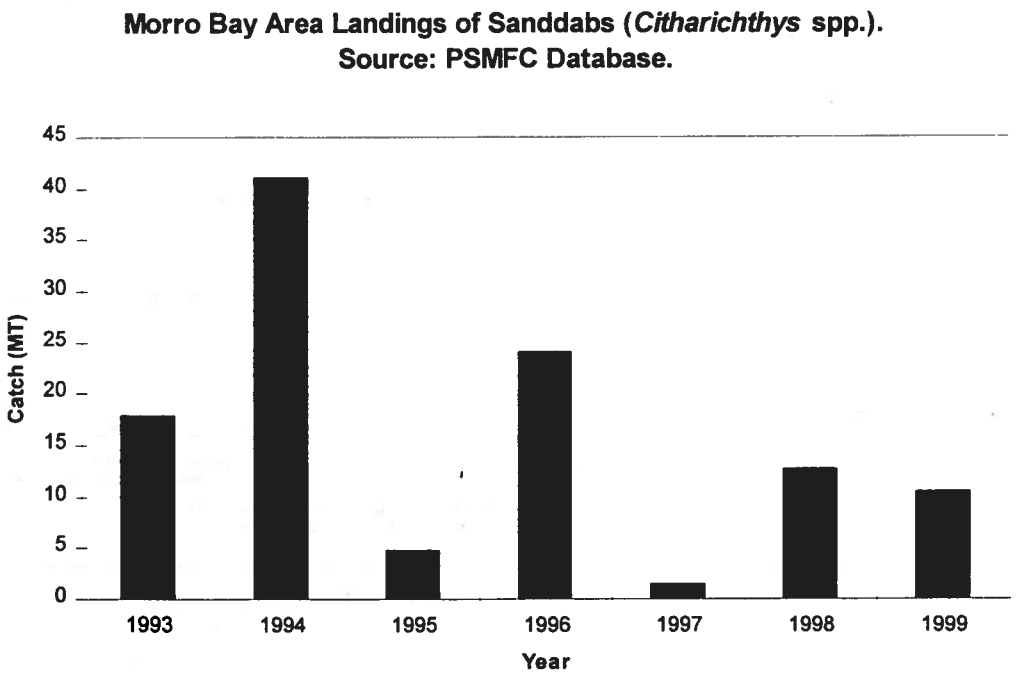


Figure 5.2.11-7. Commercial landings of sanddab species (*Citharichthys* spp.) in the Morro Bay area between 1993 and 1999 (Source: PSMFC Database).

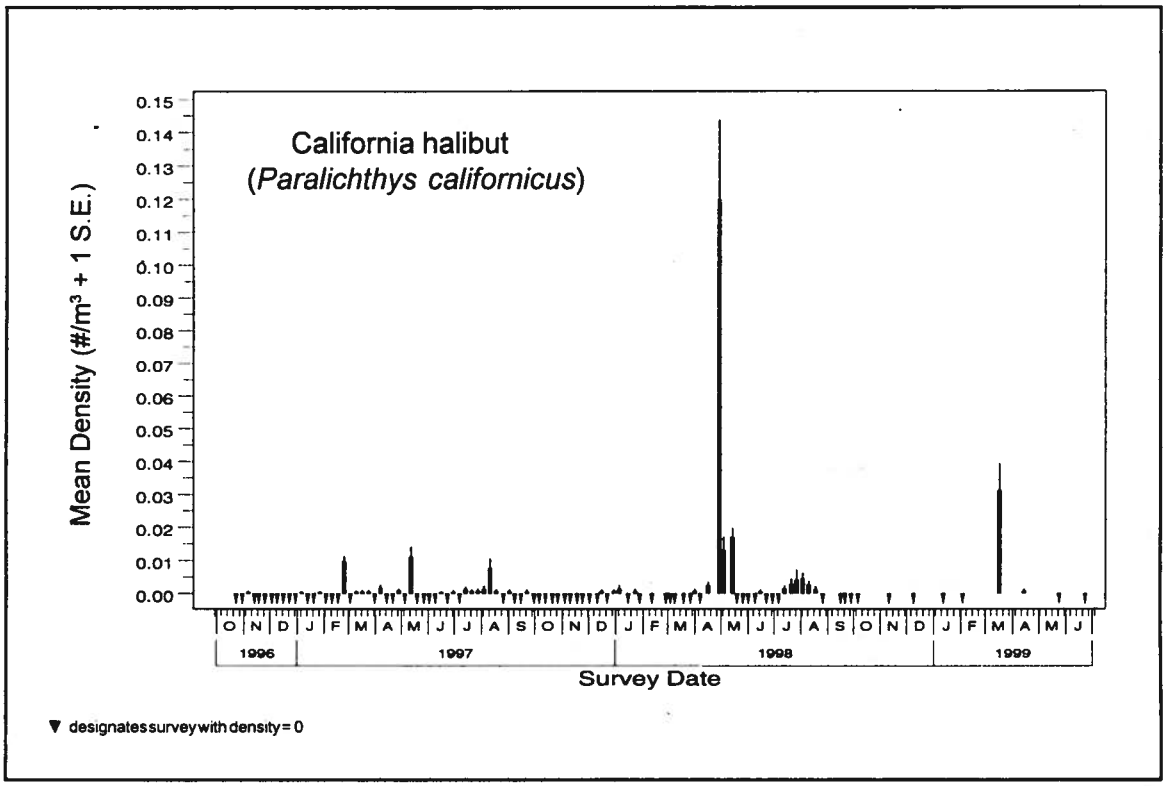


Figure 5.2.11-8. Weekly mean larval density (#/m³ + 1 S.E.) at the DCPP intake.

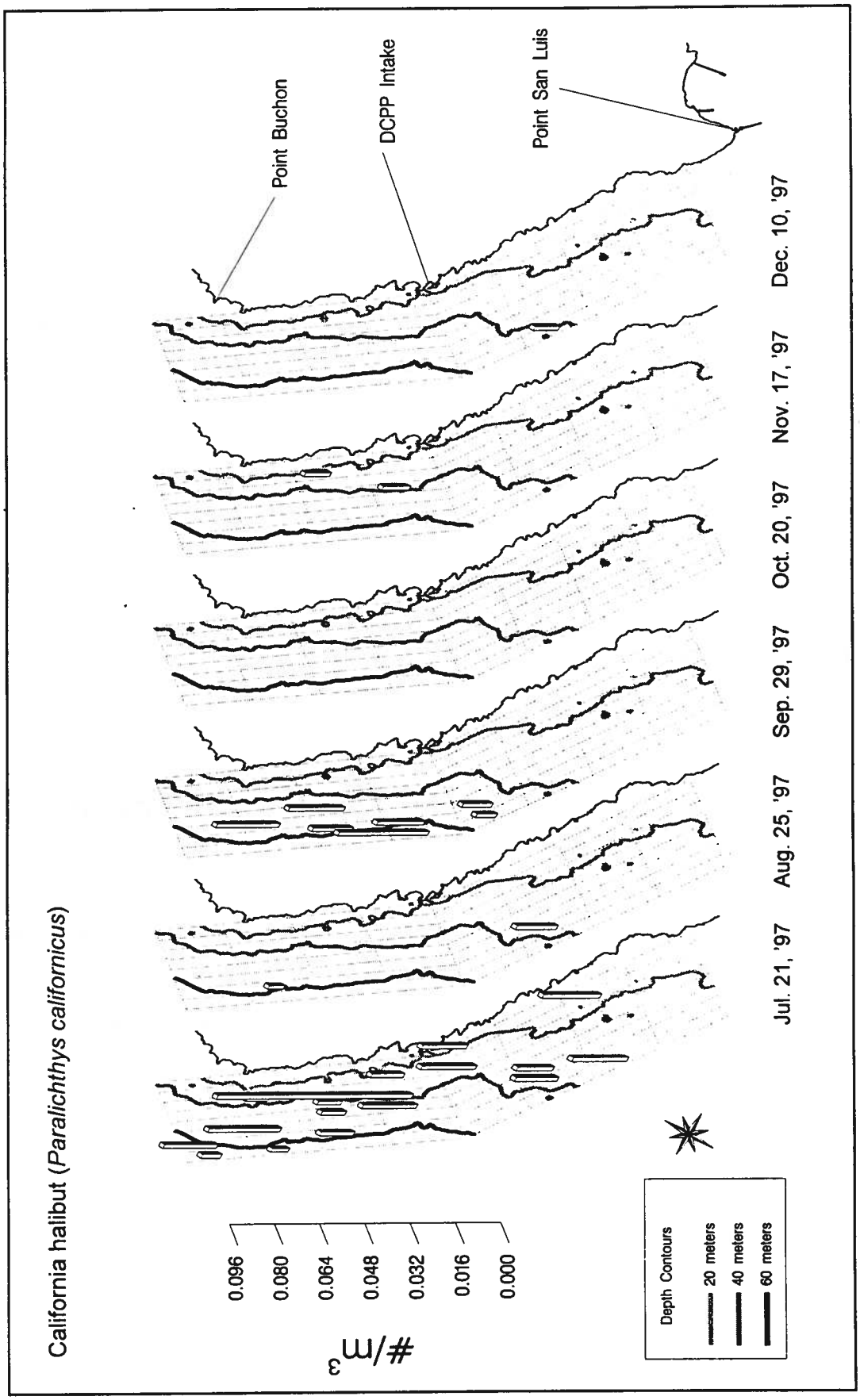


Figure 5.2.11-9a. Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

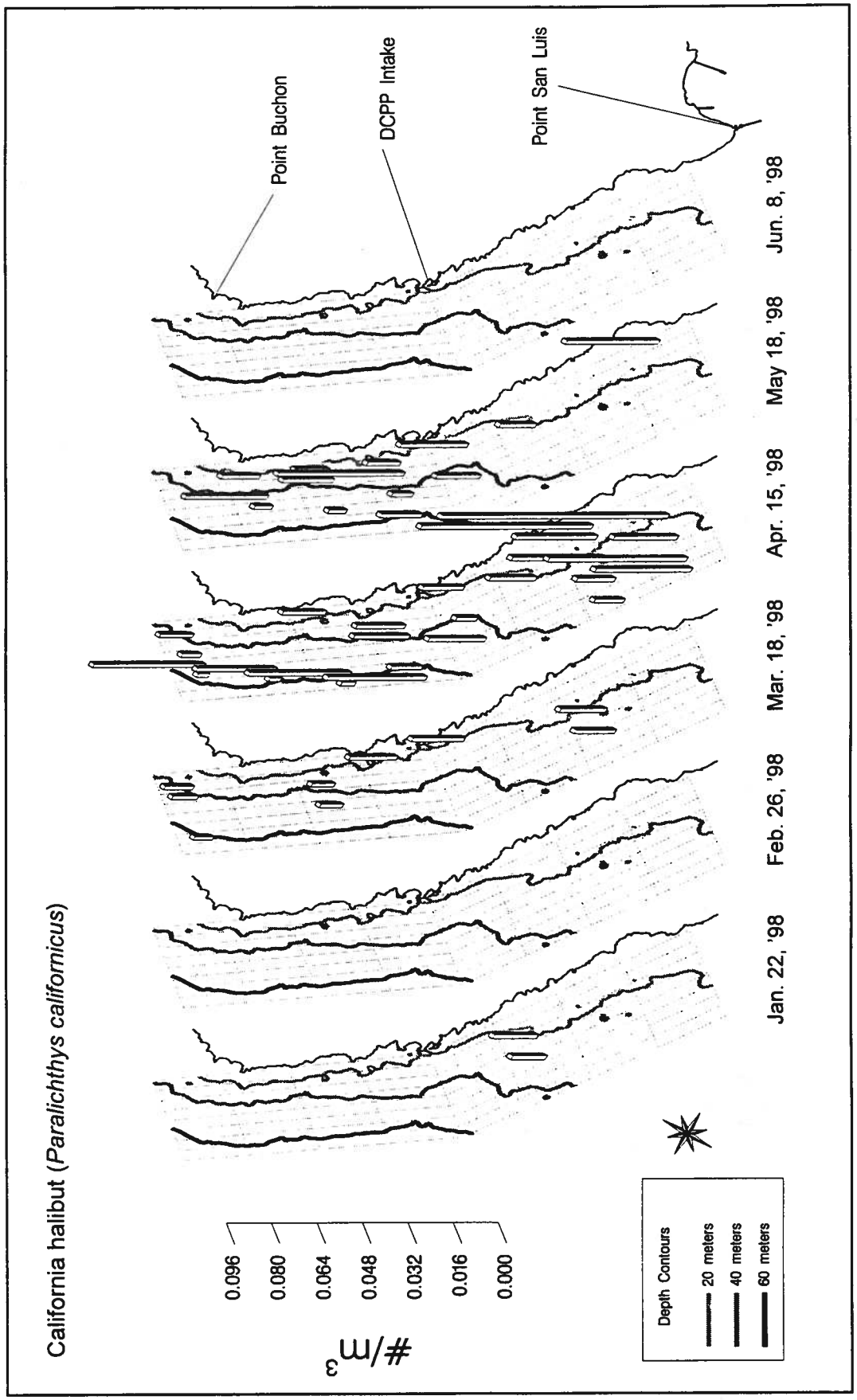


Figure 5.2.11-9b (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

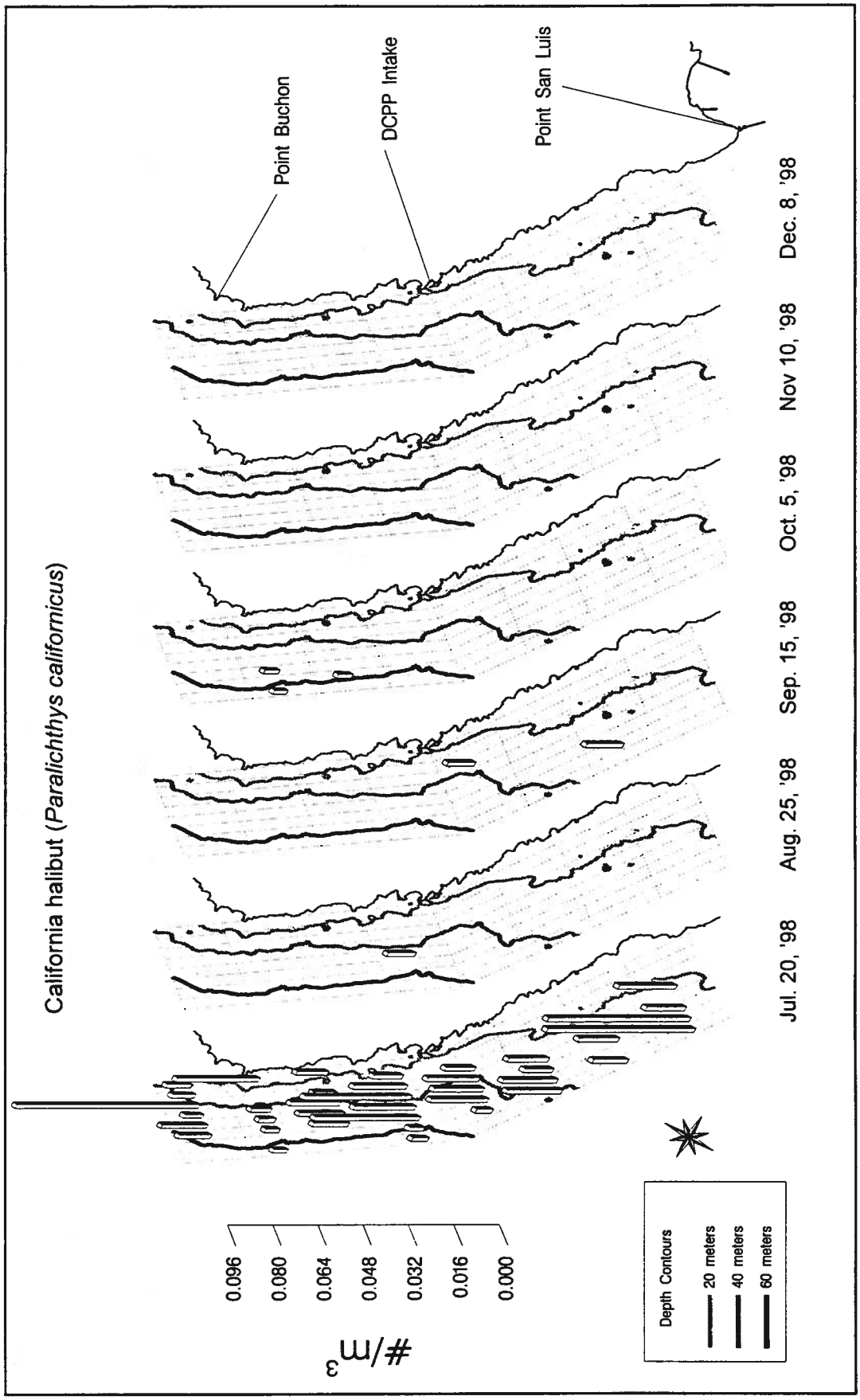


Figure 5.2.11-9c (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

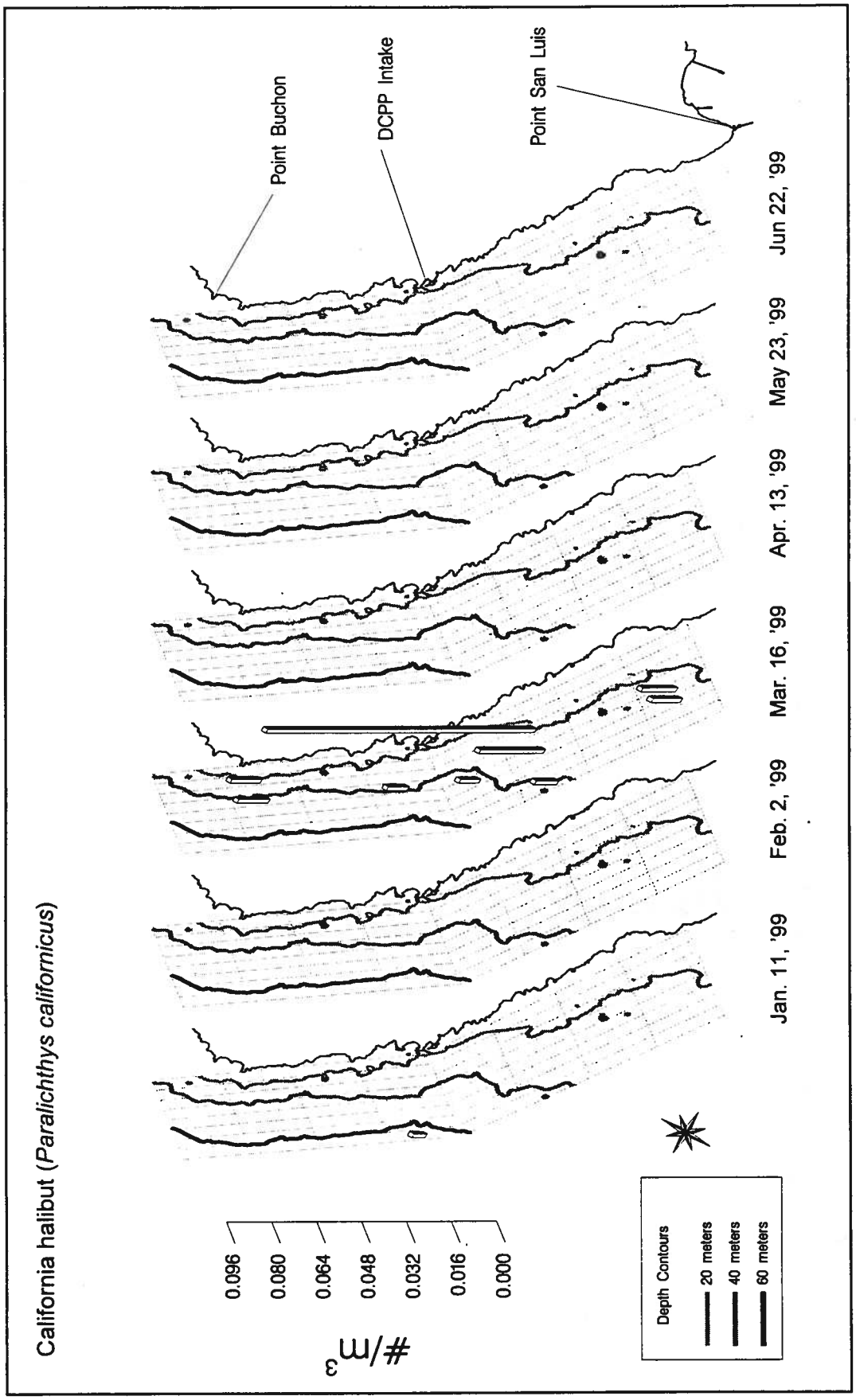


Figure 5.2.11-9d (continued). Mean larval density (#/m^3) collected in the DCPP study grid cells and at the entrainment sampling location.

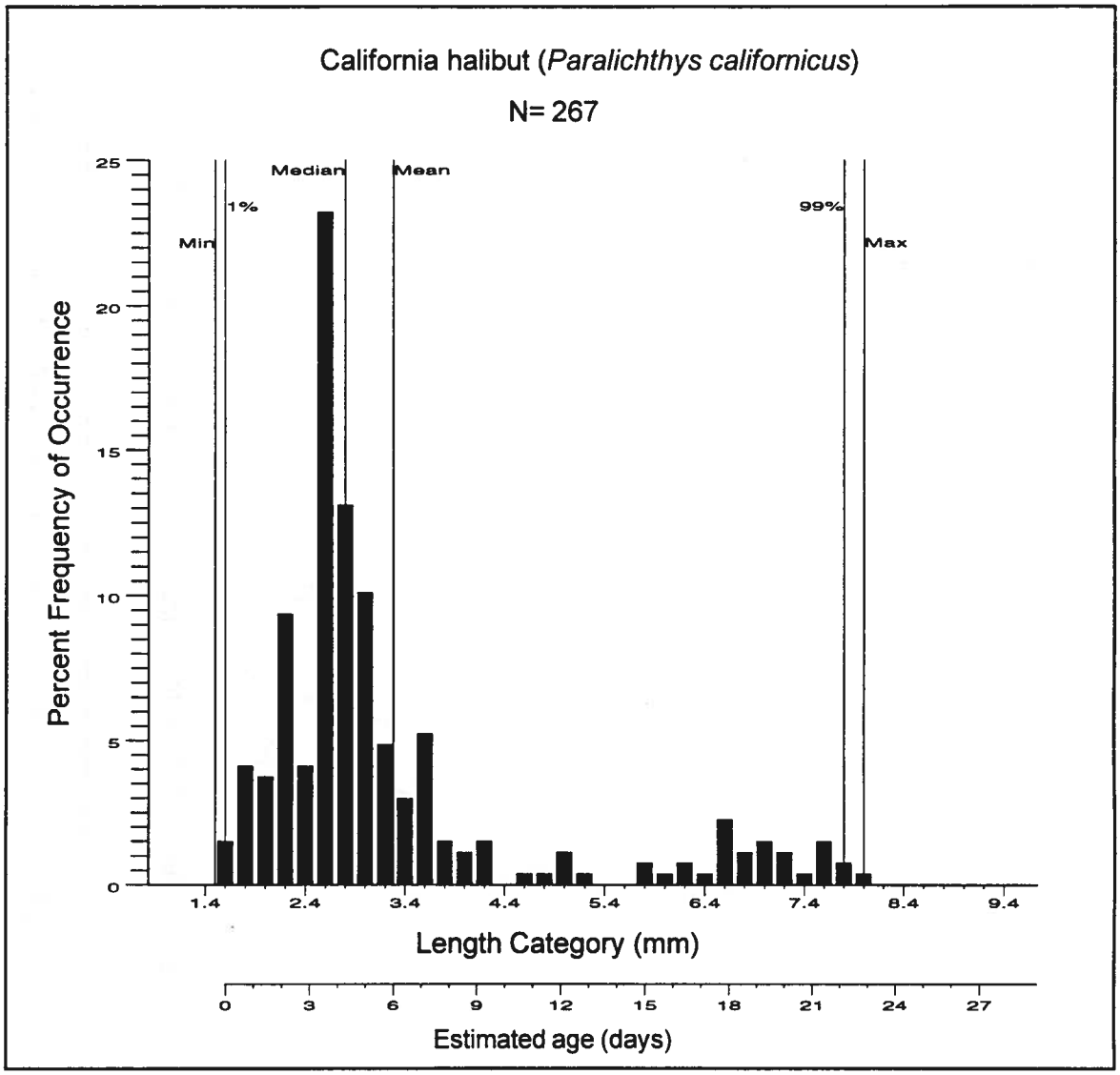


Figure 5.2.11-9. Length frequency of all measured larval fish collected in the entrainment samples. Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

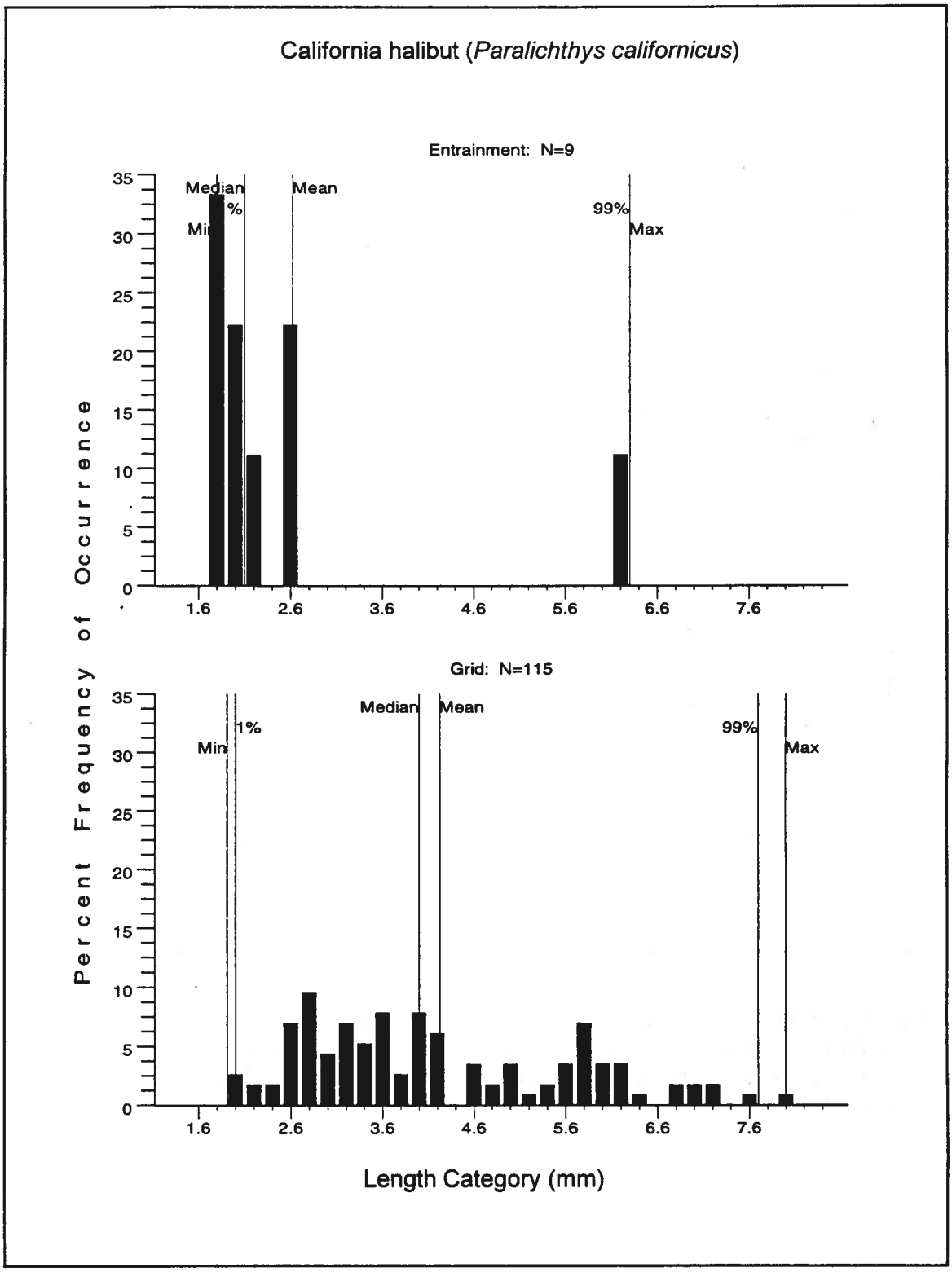


Figure 5.2.11-11. Length frequency of measured fish from entrainment surveys that paired with grid surveys (both years combined). Minimum, median, mean, maximum, and central 98% of the larval lengths are noted. X-axis scale is not continuous at larger lengths.

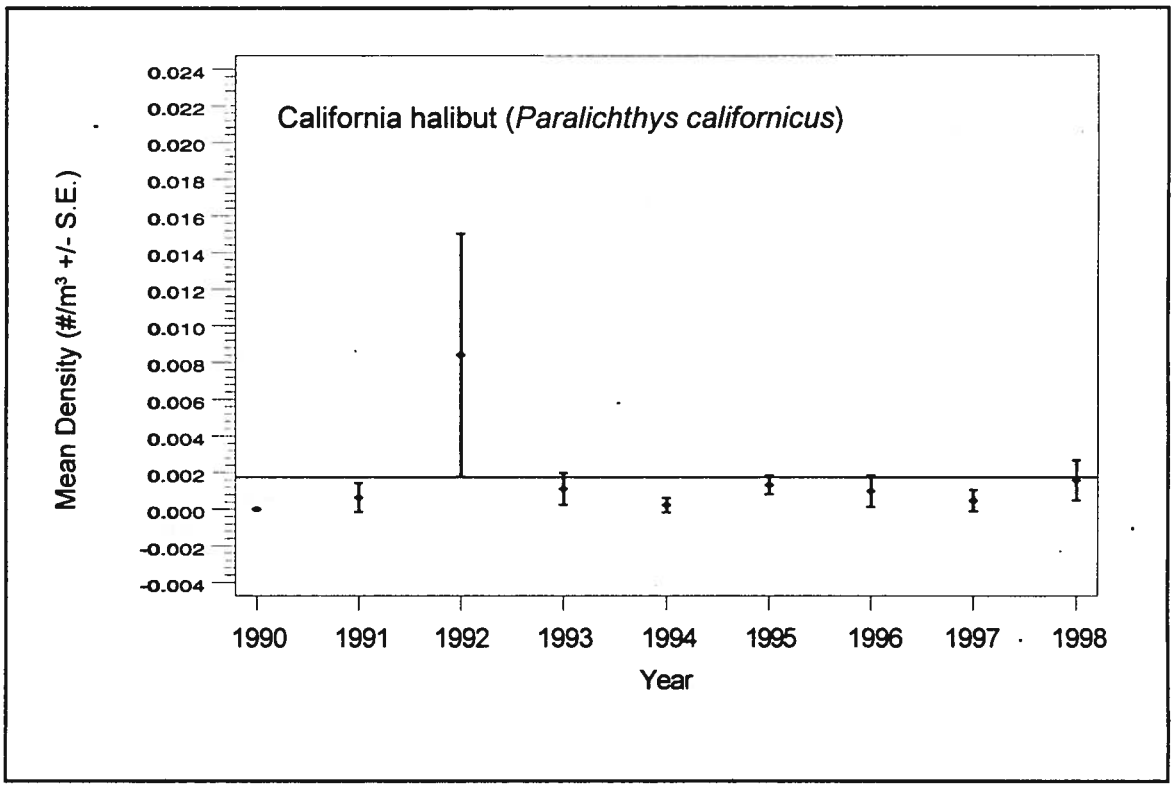


Figure 5.2.11-12. Annual mean density +/- 2 standard errors (vertical lines) and grand mean density for all years combined (horizontal line) for the Intake Cove surface plankton tows. The annual mean densities are based on seven consecutive months of data (December through June) except for 1990, which had only five months (February through June).

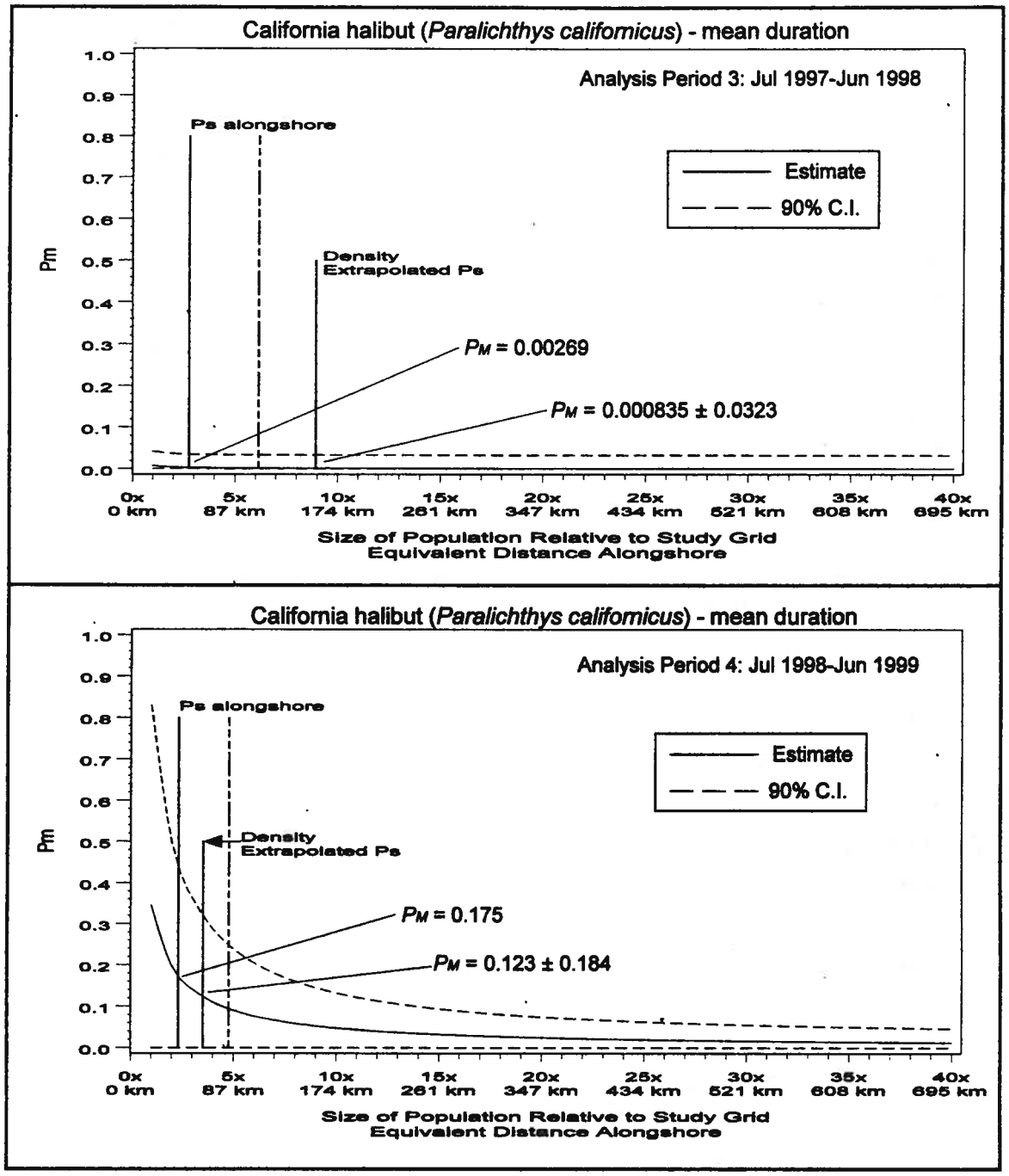


Figure 5.2.11-13a. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

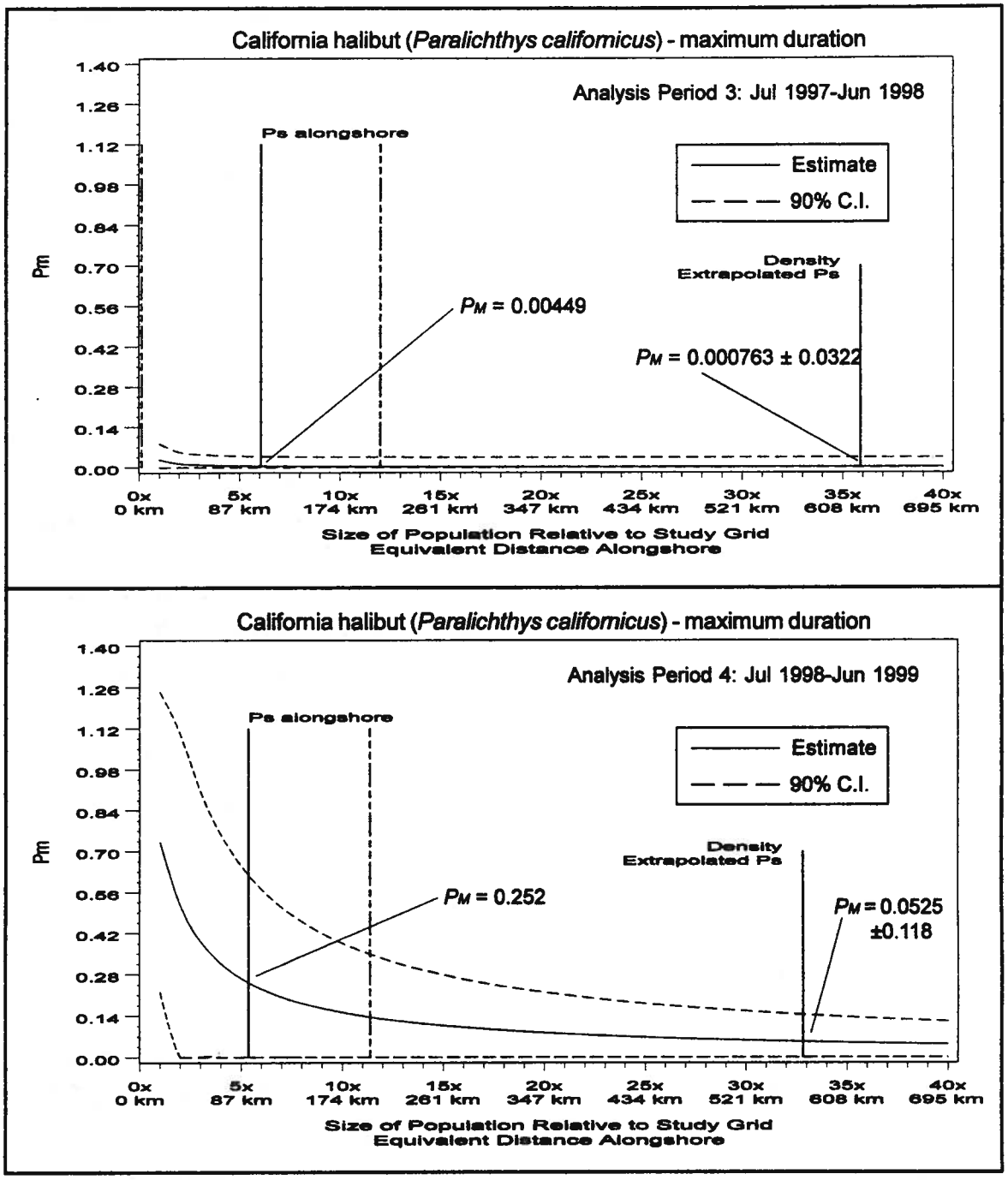


Figure 5.2.11-13b. Total annual entrainment mortality ($\widehat{P_M}$) and 90% confidence interval (C.I.) as a function of the reciprocal of the fraction of the population at risk and contained in the study grid ($\widehat{P_S}$) used to approximate the alongshore extent of the study grid population expanded on the basis of alongshore current movements and the offshore extrapolation of study grid abundance extended alongshore using both onshore and alongshore current movements with 90% C.I.'s indicated.

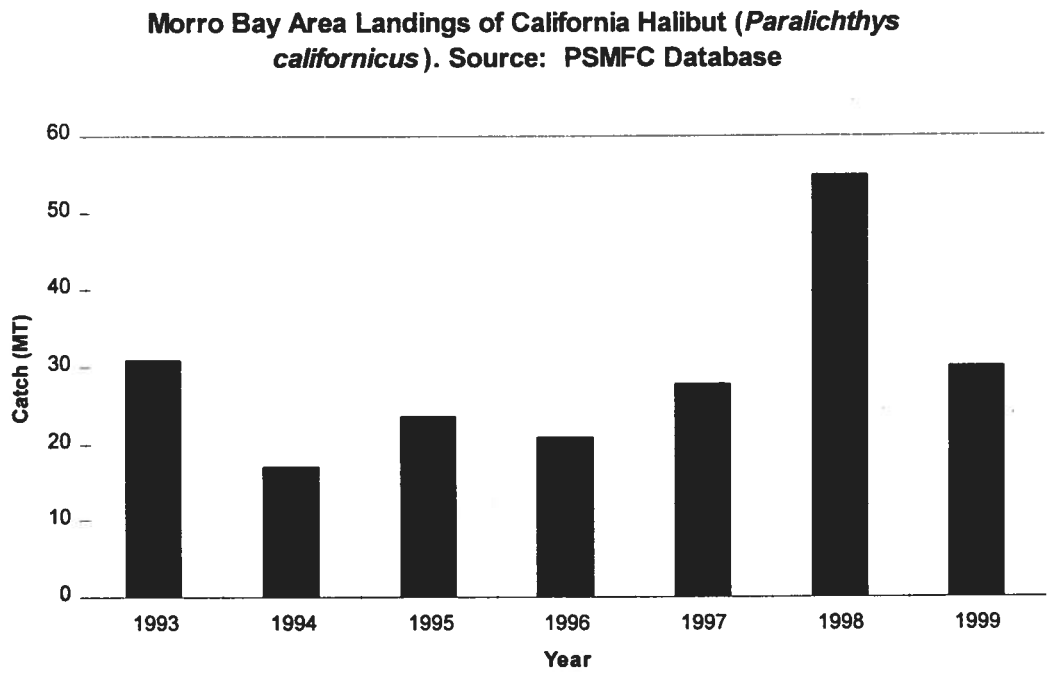


Figure 5.2.11-14. Commercial landings of California halibut (*Paralichthys californicus*) in the Morro Bay area between 1993 and 1999 (Source: PSMFC Database).

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6.0 EVALUATION OF ALTERNATIVE INTAKE TECHNOLOGIES

A number of cooling water technologies are explored which are currently in use elsewhere or proposed for use in power plant cooling water systems to minimize the loss of aquatic organisms due to entrainment and impingement. Section 316(b) of the Clean Water Act (CWA) states that "cooling water intake structures" are to "reflect the best technology available for minimizing adverse environmental impacts." As discussed in the Introduction (**Section 2**) of this report, USEPA has indicated that assessment of adverse environmental impacts (AEI) should be based on an evaluation of population-level effects. This report shows that Diablo Canyon Power Plant (DCPP) has not caused population-level effects and therefore, we concluded that DCPP has not caused AEI. However, we provide this section on alternative technologies in response to a requirement by the RWQCB. This section of the report includes a description of potential modifications to the present intake structure at DCPP. Additionally, for informational purposes, this section presents discussion of possible operational and technological alternatives to the current cooling water system that are not part of the intake structure although consideration of these alternatives is beyond the scope of the CWA.

This section evaluates the applicability of installing operational and technological alternatives for the existing DCPP cooling water system. For those applicable alternative technologies, we provide additional information relative to the potential cost of implementing the technology and the possible reduction of biological effects. These technologies include (1) cooling water system alternatives, (2) intake configuration alternatives, (3) behavioral and physical barriers, (4) fish collection and removal conveyance systems, and (5) intake maintenance and operational modifications.

6.1 Evaluation Criteria

A hierarchical evaluation system was used to assess which alternative intake technologies were initially evaluated on the basis of the following three criteria (**Figure 6-1**):

1. The alternative technology is available and proven (i.e., demonstrated operability and reliability at a cooling water intake similar in size and environment to the DCPP site).
2. Implementation of the alternative technology might result in a reduction in the loss of aquatic organisms compared to the present operating conditions.
3. Implementation of the alternative technology is applicable at the DCPP site, based on site-specific considerations of engineering, operations, and reliability.

For those alternatives that meet the three criteria, a detailed evaluation of applicability and general cost estimate is described in **Section 6-2**. All technologies considered and the application of the evaluation criteria are shown in **Table 6-1**.

A brief description and justification for the alternatives that do not meet Criteria 1, 2, and 3 are presented in the attached Appendices.

Appendix L– alternatives not meeting Criterion 1.

Appendix M–alternatives not meeting Criterion 2.

Appendix N–alternatives not meeting Criterion 3.

6.2 Applicability Analysis and Cost Estimates

This section discusses the applicability of each alternative intake technology that has met all three evaluation criteria. It is divided into two subsections: alternatives that reduce impingement and alternatives that reduce entrainment.

For those alternatives presented in previous reports, the cost figures presented here represent those original estimates (TERA 1982 and PG&E 1988a) in present day dollars as of December 1999. The annual cost and the life-of-plant cost figures presented in these earlier reports each consisted of two components: 1) capital, operation, and maintenance costs and 2) lost revenues. Other than considering the changing value of labor and material, the capital, operation, and maintenance costs of any of the alternatives was assumed not to have changed from previous estimates. To translate original cost estimates to present day values, we used representative indices from the Bureau of Labor Statistics for labor and material.

Due to significant changes in the price of electricity since the original estimates, the lost revenue component was re-forecast using the same assumed lost generation (MWe) resulting from load reduction (deratings) and lost production, but valued using a realistic plant capacity factor and the California Energy Commission's forecast of Power Exchange (PX) pricing for the price of electricity for the next 25 yr.

For comparative purposes, the cost of alternatives is expressed in net present value (NPV), which includes the total capital cost, the cost of the cumulative lost revenue, and increased maintenance costs over the life of the plant. Additionally, for the alternatives that reduce impingement, the cost per kg of fish is derived by dividing the total NPV by the cumulative weight (kg) of fish impinged for each option over the remaining licensed plant life, assumed to be 25 yr.

The cost estimations are considered to be accurate within an order of magnitude only. More accurate estimates would require detailed and plant-specific design of the various technologies. For example, neither TERA (1982) nor PG&E (1988a) considered the following costs, which would substantially add to the current estimates:

- Demolition and relocation of existing utilities, system components, plant facilities, buildings, etc.
- Security during construction and effect on permanent security boundary.
- Nuclear licensing (License Amendment Request to the NRC).
- Permitting (i.e. California Coastal Commission, RWQCB, etc.).
- Seismic design considerations.

Additionally, for cooling tower cost estimations, the following additional costs were not considered by TERA (1982):

- Tie-in with existing circulating water conduits.
- Change in pumping plant location, design, and configuration.
- New circulating water pumps (CWP) and system components.
- Changes in the turbine and condensate systems to accommodate higher cooling water temperatures.
- Salt water drift damage to existing plant facilities or structures and to electrical or mechanical equipment.

- Salt water drift damage to surrounding lands, terrestrial habitat and ecology, and agricultural productivity.

Therefore, the cost estimates provided in this report are considered approximate, likely to be conservative (low) and adequate as order of magnitude estimates.

6.2.1 Alternatives That Reduce Impingement Losses Only

The impingement rate at DCPP is low, based on 1985–86 impingement data and ongoing field observations (**Section 5.1.2**). In the 1985–86 study (Tenera 1998a) impingement rates were estimated at about 0.32 to 0.73 kg/day/unit. Divers observe fishes and invertebrates freely moving within the intake structure forebays and directly in front of the traveling screens (Tenera 1997a). Therefore, modifications to the intake structure can provide little improvement in reducing the number of impinged fish. The following discussion, however, provides information on possible alternative technologies to reduce impingement at DCPP, their estimated costs, and the comparison of costs to potential impingement reductions.

6.2.1.1 Increased Area Intake Structure

A detailed review of the intake velocities at DCPP (Wyman 1988), the literature on laboratory swimming performance of juvenile fish (reviewed in Tenera 1998a), and diver observations of impingement avoidance at the DCPP intake (Behrens and Larsson 1979) showed that fish impingement should be virtually independent of intake approach velocities of less than about 0.8 to 1.0 fps (present velocity at DCPP). Based on studies at Contra Costa Steam Plant (Kerr 1953) impingement is predicted to increase at intake approach velocities greater than 1.5 fps, particularly among juvenile fish that are less than approximately 80 mm in length. A reduction in approach velocities is not expected to substantially reduce fish impingement rates at DCPP. However, one approach to possibly reduce impingement would be to reduce the intake approach velocity. The approach velocity is inversely proportional to the cross-sectional area through which the water passes. Therefore, velocities entering the intake structure can be reduced by increasing the cross-sectional area exposed to the flow. Reduction in the intake approach velocity would require expansion of the intake structure including additional traveling screens to

increase the intake's cross sectional area. A design that halves the existing approach velocity was considered.

This design change assumes that reduced flow velocity translates to reduced impingement. A level of uncertainty exists around the use of a single parameter such as flow rate or velocity to accurately predict the potential for reducing impingement. Taft (1999) compares several sites with similar intake designs and flow characteristics that experience large differences in impingement. Higher rates of flow and intake velocity (through the traveling screens) are usually associated with higher impingement losses. Taft (1999) illustrates from the available data that neither velocity nor flow appear to be closely correlated to impingement rates. He concludes that impingement should be viewed as a site-specific event that is influenced by many parameters. It is the combinations of environmental conditions and species life stages that interact to influence impingement.

Plan and section views of the proposed modification to the intake structure are presented in **Figures 6-2** and **6-3**. As indicated in these figures, the modification would involve expanding the intake structure towards the ocean side with approximately 45° flare and deepening the existing intake invert slab elevation to (-) 46.0 feet.

Construction of the expanded intake structure would result in a major construction effort and require installing a cofferdam and dewatering the existing intake structure. Disruption associated with construction and dredging activities would contribute to localized impacts on kelp and benthic organisms inhabiting the Intake Cove.

During intake construction, the circulating water flow required for the operation of Units 1 and 2 would be disrupted, resulting in a complete loss of generating capability from both units for a period of approximately one year. Operating one unit while modifying the other unit would not change the cumulative revenue loss appreciably, because the construction duration for each unit would not change substantially. A major challenge to any construction project in front of the existing intake structure would be the potential for storm conditions interfering with work in the intake bays.

While the units are shut down, the auxiliary saltwater (ASW) system would have to remain operable to preserve the function of those systems necessary to maintain the plant in a safe shutdown mode and protect the nuclear fuel. Therefore, an extension of the ASW system would be required to provide continuous flow to the ASW pumps. Since the ASW system is a nuclear safety-related system, necessary safety-related design considerations such as seismic safety, control of heavy loads, and interaction between safety and non-safety related items would have to be evaluated as part of intake structure design and modification.

Prior to finalizing the design of the expanded intake structure, a detailed engineering evaluation would be necessary to ensure compatibility between hydraulic flow patterns, cooling water volumes, and pressure regimes associated with the expanded intake structure and the existing Units 1 and 2 condenser system. License amendments required to accommodate a modification such as this have not been reviewed. Additionally, other environmental and safety design considerations such as the effects of earthquakes, tsunamis, and probable maximum loads would require evaluation. These would result in additional costs that have not been factored into the following cost estimates.

The estimated capital cost for this project would be on the order of $51,000,000. In addition, the cumulative lost revenue and increased maintenance would be approximately $590,000,000 over the life of the plant. The implementation of this technology has a net present value (NPV) of (-) $275,000,000. This results in a cost of $61,700 to save each kilogram of impinged fish, assuming a 100% reduction in impingement over the remaining life of the plant.

6.2.1.2 Angled Screen Intake Structure

Angled traveling screen intakes are designed with conventional screens set at an angle to the incoming flow. A fish diversion system is installed at the downstream end of the intake such that incoming fish are directed along the face of the screens to the fish diversion path. Fish diversion systems include various designs of pivoting, fixed, or traveling screens, louvers, associated with traditional bar racks.

These intake designs have been utilized in both hydroelectric and thermal power station intakes to minimize impingement. At Brayton Point Station Unit 4 in Massachusetts, an 18 mo (October 1984 to March 1986) biological evaluation was conducted to determine the species, number, and initial and extended survival of fish diverted from the angled screen intake (Davis et al. 1988). The angled screen intake system had a high diversion capability and demonstrated effectiveness for mitigating fish impingement. Initial and extended survival varied by species; however, a certain group of numerically dominant taxa was classified by the authors as "fragile" (primarily, bay anchovy and Atlantic silverside). The fragile group had a calculated survival below 25 percent and a "hardy" group, dominated by winter flounder and northern pipefish, had survival values greater than 65 percent. The diversion efficiency for all species combined was 76 percent. Nine of the top 12 taxa collected had diversion efficiencies greater than 83 percent. The diversion flow collections resulted in an initial survival rate of 58 percent for all taxa (n= 28,186) combined. The initial survival rate ranged from 6 percent for bay anchovy to nearly 100 percent for American eel. Initial survival with the exclusion of bay anchovy was 83 percent. Extended survival for all fish (n=9,209) collected at the diversion flow was 63 percent. Extended survival trends were similar for the major species involved. Survival ranged from a low (bay anchovy) of 0 percent to a high over 99 percent (tautog wrasse).

Oswego Steam Station Unit 6 utilizes an angled screen diversion system similar to the system at Brayton Point Station (LMS 1992). Biological studies were conducted to investigate the effectiveness of the screens as systems. Alewife (herring) and rainbow smelt made up 90 percent of the collected species (from April 1981–March 1983). Diversion efficiency was 79 percent and 74 percent for alewife and rainbow smelt, respectively. The combined diversion efficiency for all the species collected was 78 percent, ranging from 53 percent for mottled sculpin to 95 percent for gizzard shad. Initial survival ranged from a low of 45 percent for rainbow smelt to a high of 87 percent for emerald shiner. A total of 34,294 individuals from the seven most frequently collected species were examined for initial survival, and 7,534 fish were observed for latent survival. The lowest latent survival rate was exhibited by alewife (22 percent) while the highest was mottled sculpin (94 percent). Overall, the angled screen system was effective in diverting

fish from the primary screenwell through the secondary screenwell back into the lake. The degree of effectiveness varied widely by species; size class or age, and condition of the population.

A full-scale angled screen test facility was constructed at the Danskammer Point Generating Station on the Hudson River in 1981 (LMS 1985). The angled screen facility was located in the cooling water intake canal and consisted of two 3 m (10 ft) wide vertical traveling screens set at a 25 degree angle to the approach flow. The effectiveness of the system was evaluated over a three year test period (LMS 1985). Diversion efficiency ranged from 95 to 100 percent, with a mean of 99 percent. Species included bay anchovy, blueback herring, white perch, spottail shiner, alewife, Atlantic tomcod, pumpkinseed (sunfish) and American shad. Overall, system efficiency (diversion efficiency times initial survival times latent [96 h] survival) ranged from 68 percent (alewife) to 99 percent (spottail shiner) with a mean of 84 percent (LMS 1985).

Angled traveling screen intake structures have been used in a marine environment and are well suited for reducing impingement. Wave energy and debris loading events that occur at DCPP were probably not experienced at facilities that have installed these screens. Angled screen intakes do not prevent entrainment of larval organisms.

One feature of angled screen intakes is that the velocities normal to the screens is quite low, on the order of 0.03 m/s (0.1 ft/sec) to 0.15 m/s (0.5 ft/sec) (EPRI 1999). Implementation of an angled screen intake at DCPP would require a major reconstruction of the intake to accommodate additional (at least twice the number currently installed) screens in the angled configuration. Although a specific design and cost estimate has not been prepared for this configuration, it is judged that the costs would be similar to the costs associated with the expanded area intake described in **Section 6.2.1.1**. Since this alternative addresses only impingement, the costs compared to the reported impingement rate would not make this option cost effective for implementation at DCPP.

6.2.1.3 Traveling Screen Operating Cycle Modifications and Fish Conveyance Systems

Operational modifications to the vertical traveling screens, such as the use of continuous screen rotation, screen baskets with "fish buckets", low-pressure spraywash, and fish return sluiceways, are alternatives that have been proposed by the industry to increase the biological effectiveness of conventional vertical traveling screens. Typically, all of these elements must be used together to effectively reduce impingement mortality. Costs for the combination of modifications are presented in the section on *Combinations of Vertical Traveling Screen Modifications* (below).

Continuous Traveling Screen Rotation

Several studies have been performed to determine the effectiveness of continuous screen rotation on impingement survival. Studies conducted at the Pittsburg Power Plant (PG&E 1992) show that increasing the screen rotation frequency from 3 h intervals to continuous rotation did not result in consistently improved impingement survival for invertebrates such as California bay shrimp, brackish-water crabs, and oriental shrimp.

In contrast, more (26–56% after 96 h) young-of-the-year (YOY) white perch (*Morone americana*) survived impingement on continuously operating traveling screens compared to those operated 2–4 h intermittently (19–32% survival after 96 h). Likewise, striped bass (*Morone saxatalis*) latent survival improved (32–62%) after impingement on continuously operating screens compared to bass impinged on intermittently operated screens (26% after 96 h; King et al. 1978). King et al. (1978) concluded continuous traveling screen operation allowed maximum initial and latent survival for white perch and striped bass YOY.

Continuous screen rotation did not consistently result in improved impingement survival of the marine organisms examined at the Moss Landing Power Plant (PG&E 1988a, Section 4.2). Among impinged fish, hardy species such as plainfin midshipman and crabs had a high rate of survival regardless of screen rotation frequency. Increasing screen rotation frequency at the Moss Landing Units 6 and 7 intake did, however, contribute to a substantial increase in impingement survival for both surfperch and rockfish, which together constitute about 11 percent (by weight) of the fish impinged at DCPP (PG&E 1988a, Section 4.2).

The studies referenced above show that impingement survival varies from site to site and is species specific. Testing of continuous screen rotation at DCPP would be required to determine if a reduction in impingement is achievable.

The current debris removal system at DCPP is designed to handle large quantities of kelp and other debris. The material washed from the screens is washed to a sluiceway and directed to a large sump. This material and wash water is pumped to a location approximately 244 m north of the intake structure at the shore end of the west breakwater. Grinders are installed at the entrance to the refuse sump to reduce the size of the kelp and algal debris to prevent clogging of the debris removal system. No impingement survival is expected with this system. This system is needed to support plant operation with the high debris loads. Increasing intake screen rotation offers the possibility of increasing survival of some impinged organisms only if this method is used in conjunction with other modifications (such as a gravity sluiceway and fish baskets to increase survival). These configurations are described in the next two sections.

Gravity Sluiceway Fish Return and Low Pressure Spraywash

There are two basic types of sluiceways for the return of impinged organisms and debris to the waterbody: one uses a pump to transport collected material away from the intake and one uses gravity flow. Based on the existing DCPP intake design, a gravity return sluiceway directing flow to the south end of the intake would be the least complex and least expensive option.

As described in the previous section, the current DCPP screenwash system is not designed to return impinged organisms to the receiving waterbody intact. The high debris loading experienced at DCPP requires that this system function in a way that makes impingement survival unlikely, and hence no survival of impinged organisms is expected.

To increase impingement survival at DCPP, a separate fish return trough, low pressure spray wash, and fish collection baskets as described in the next section would be required. Based on construction estimates, the implementation of a low pressure spray wash and fish return system would cost about of $12,000,000. NPV analysis for this modification in conjunction with replacing screen baskets is included in the next section.

Combinations of Vertical Traveling Screen Modifications

Several modifications to conventional vertical traveling screens have been studied in recent years in an attempt to reduce the mortality of impinged organisms by incorporating new design features that improve the survival of impinged organisms. Such state-of-the-art modifications act to enhance fish and invertebrate survival related to screen impingement and spraywash removal. Screens modified in this manner are commonly called "Ristroph Screens". These modifications include the following features:

1. Hydrodynamically improved, watertight fish collection buckets along the base of each screen panel to provide a holding area for organisms during screen rotation;
2. Smooth woven mesh (e.g. 1.6 mm by 12.7 mm rectangular mesh) installed on the screen baskets to minimize abrasion;
3. Lighter composite screen baskets which allow for increased rotational speed;
4. A second sluiceway/fish return system with combined low pressure spray wash to transport organisms removed from the screen by the low-pressure spraywash back to the receiving waterbody;
5. Improved screen-to-collection trough flap seal design; and
6. Modifications to traveling screen bearings and motors to permit continuous rotation and cleaning, minimizing the time an organism is impinged on the screen.

Studies of the biological effectiveness using these modified Ristoph screens was conducted at the Salem Generating Station on Delaware Bay in New Jersey (Ronafalvy et al. 1999; Heimbuch 1999). An initial evaluation was performed after six of the 12 existing traveling water screens at the cooling water intake structure had been replaced with the new, improved screens, allowing a side-by-side comparison of the effectiveness of the old and new screens. Tests were conducted on 19 separate dates between June 20 and August 24, 1996. Fish collected from the old and new screens were held separately for observation of 48 h survival. The only species occurring in sufficient numbers to provide a statistically valid data analysis was juvenile weakfish (*Cynoscion regalis*; n = 1082 for the old screens, n = 1559 for the new screens). Overall, statistical analyses demonstrated a 48 h survival rate (uncorrected for control mortality) of 57.8 percent with the old screens and 79.3 percent with the new screens.

A second series of impingement survival studies was conducted in 1997 and 1998 to provide estimates of impingement survival rates with all 12 of the modified screens installed on Salem Units 1 and 2 (EPRI 1999). White perch (*Morone americana*) impingement survival rate estimates ranged from 98 percent in December to 93 percent in April. Estimates for weakfish ranged from 88 percent in September to 18 percent in July. For bay anchovy (*Anchoa mitchilli*), survival estimates ranged from 72 percent in November to 20 percent in July. Atlantic croaker (*Micropogonias undulatus*) survival estimates ranged from 98 percent in November to 58 percent in April. The estimated survival for spot (*Leiostomus xanthurus*) was 93 percent in November (November was the only month in which a significant number of spot were collected). *Alosa* species (herrings) combined produced survival estimates that ranged from 82 percent in April to 78 percent in November.

Impingement mortality rates for the modified screens (1997 and 1998 studies) were compared to mortality rates for the original screens from the 1978 to 1982 studies. Based on the comparisons, intake modifications were effective in improving the rates of fish survival. Estimates of impingement mortality rates were lower for the modified screens than for corresponding estimates from the original screens for white perch, bay anchovy, Atlantic croaker, spot, and the *Alosa* species.

Based on impingement survival data collected at the Diablo Canyon and Moss Landing power plants and at other facilities, it was concluded that operation of modified intake screens in combination with fish return sluiceways could enhance impingement survival of many of the fish and macroinvertebrates impinged at DCPP, including skates and rays, rockfish, sculpin, plainfin midshipman, tubesnouts, rock crabs, and sea urchin (PG&E 1988a). On the basis of data collected in the impingement survival studies (PG&E 1988a), it was estimated that losses of impinged fish and selected macroinvertebrates may potentially be reduced by approximately 75 percent under conditions of intermittent rotation, assuming no incremental mortality associated with passage through the fish return system. Rotating intake screens continuously could reduce impingement losses of fish and selected macroinvertebrates, assuming no incremental mortality resulting from passage through the fish return system. In light of DCPP's low impingement rate,

however, this percentage reduction will not result in an appreciable biological benefit. The screen improvements are not expected to provide any reduction in entrainment.

Traveling screen modifications to reduce impingement mortality must be accompanied by a sluiceway designed to return organisms to the receiving waterbody. Most installations of modified traveling screens use a dual sluiceway return system: 1) a gravity sluiceway return system for impinged organisms removed from the screens by the low pressure spraywash and 2) another sluiceway for debris removed by the high-pressure spraywash. This type of system was selected for this evaluation.

The estimated capital cost for this project would be on the order of $13,000,000. In addition, the cumulative lost revenue and increased maintenance would be approximately $11,000,000 over the life of the plant. The implementation of this technology has a net present value (NPV) of (-) $11,000,000. This results in a cost of at least $2,200 to save each kilogram of impinged fish assuming a 100% reduction in impingement over the life of the power plant. The scope of implementing this alternative is expected to extend the duration of routine refueling outages, adding additional costs that have not been factored in to the estimates.

6.2.2 Alternatives That Reduce Entrainment Losses

The major focus of this study concerns losses to fish and invertebrate populations caused by withdrawal of larval stages. A majority of the young larvae are weak swimmers and cannot escape even very low entrainment velocities. In order to reduce entrainment, alternatives that use less cooling water are considered. The following discussion provides information on possible alternative technologies to reduce entrainment at DCPP and the estimated costs to install them.

6.2.2.1 Closed-Cycle Cooling Towers With Saltwater Makeup

Both mechanical and natural draft (hyperbolic) cooling towers using saltwater makeup were evaluated for applicability at DCPP (TERA 1982). Operational problems and environmental considerations (including air quality impacts from cooling tower drift and effects on vegetation from salt deposition) have limited the application of closed-cycle cooling where brackish water

or saltwater is used as a make-up source. The report concluded that no operating or proposed electric generating facilities in the United States use seawater in closed-cycle cooling systems. Since that report was written, natural draft cooling towers with saltwater makeup have been installed and operated at Crystal River units 4 and 5. Two cooling towers provide cooling for the two 750 MW coal-fired units. The towers are operated with a high blowdown rate (10%), such that the total saltwater demand from the ocean is reduced approximately 80 percent compared to a once-through system. Therefore, saltwater cooling towers, either with mechanical or natural draft, have been demonstrated on the scale required for a closed-loop system at DCPP. Based on the Crystal River experience, the use of closed-loop cooling towers with saltwater makeup at DCPP would reduce the cooling water flow rate by 80 percent (from 1.6 Mgpm to 0.32 Mgpm per unit), reducing entrainment by a similar percentage.

Conceptual designs and design parameters for both mechanical draft (**Figure 6-4**) and natural draft cooling towers (**Figure 6-5**) have adverse environmental impacts on air and water quality, land use, and aesthetics (TERA 1982).

- Air Quality — Would produce 37 h/yr of ground level fog, visible vapor plume and emit ca. 6,080–9,070 kg/d of salt drift.
- Water Quality — Would increase salinity of cooling water discharge by 1.5 times, worst case; effluent limitations could be exceeded if a treatment system is not installed.
- Land Use — Would require an additional 12 to 22 ha for cooling towers. Additionally, the terrestrial impact of salt drift would be significant downwind of the plant site, as well as to the plant facilities.
- Aesthetics — For a hyperbolic (natural draft) system with two 142 m diameter x 170 m high cooling towers, the vapor plumes could reach 1,000–2,000 m high. For a mechanical draft system with six 94 m diameter x 22 m high cooling towers, the vapor plumes could reach 100–200 m high.

The estimated capital cost for the hyperbolic system alternative is on the order of $658,000,000. In addition, the cumulative lost revenue and increased maintenance would be approximately (-) $454,000,000 over the life of the plant. The NPV of this alternative's costs is (-) $503,000,000.

Actual project costs would be higher than the estimate as the use of a closed-loop cooling system would require major design changes to the DCPP turbine plant. The present circulating water pumps (CWP) are too large to supply a cooling tower system, so new cooling tower makeup pumps would be required. A new set of cooling tower supply pumps of similar capacity to the existing circulating water pumps would have to be installed in a new pumping facility. Supply and return water conduits would also have to be constructed. Since the DCPP main condenser and turbine cycle are designed to operate with cooling water temperatures between 11 and 14°C, extensive redesign and retrofitting of the condenser and other turbine systems would be required to allow reliable plant operation with closed-loop cooling with a supply temperature of 26 to 31°C. The costs of these modifications are not included in this estimate and would add substantially to the overall cost.

6.2.2.2 Closed-Cycle Cooling Towers with Freshwater Makeup

A closed-cycle cooling system using freshwater makeup is a well proven technology used at many different nuclear and fossil power plants on a scale similar to that required at DCPP. A conceptual design is outlined in this report.

A freshwater cooling tower system would require approximately a 132,500 m^3/d (92 m^3/min) makeup water supply. This corresponds to 43–49 million m^3 of freshwater per year. There is no supply of fresh water (including sanitary treatment plant effluent) within 40 km of DCPP available to supply cooling tower makeup (TERA 1982). A multistage flash distillation plant could supply the needed fresh water, as could a reverse osmosis system of similar capacity. Both flash distillation and reverse osmosis systems have been constructed on a scale needed to support fresh water cooling towers at DCPP.

Environmental impacts are similar to those described for the closed-loop cooling tower with saltwater makeup except that the use of fresh water would reduce the emission of particulates to approximately 1,540 kg/d.

The use of freshwater cooling towers would reduce the required saltwater flow for condenser cooling from ca. 9.5 million m^3/d (101 m^3/sec) to ca. 0.4 million m^3/d (5 m^3/sec). This would reduce intake flow by 95 percent, reducing entrainment by a similar amount.

The construction effort involved would be similar in cost to the saltwater systems described previously, except for the additional costs for a desalination plant of sufficient capacity to supply cooling tower makeup.

The estimated capital cost for this alternative is on the order of $1,174,000,000. In addition, the cumulative lost revenue and increased maintenance would be approximately $1,367,000,000 over the life of the plant. The net present value of this alternative's costs is (-) $1,072,000,000.

The use of saltwater or freshwater closed-loop cooling tower systems would reduce saltwater flow by 80 to 95%, reducing entrainment and impingement by a similar amount. Cooling tower technology is proven at many power plants, saltwater cooling tower operation has been demonstrated, and desalination plants of a size needed for freshwater cooling towers have been constructed. The implementation of a closed-loop system at DCPP would require a substantial design and construction effort. The costs of retrofitting a closed-loop system at DCPP (which range in net present value from (-) $503,000,000 to (-) $1,072,000,000 not including permitting challenges and extensive plant modifications to ensure reliable plant operation with cooling towers installed) would be difficult to justify, especially considering the uncertain value of the plant's electrical output as California deregulates electric generation. Although it would be possible to install a closed-loop cooling tower system, it is doubtful that an investment of the type described above would be viable in the current or future electrical generation market.

6.2.2.3 Fine Mesh Screens

Fine-mesh screening, frequently used in centerflow screens, has been investigated in laboratory studies to determine its potential to minimize entrainment at power plant intakes (Magliente et al. 1978). Application-specific studies are necessary to evaluate the survival of fish eggs and larvae impinged on fine-mesh screens.

Information from laboratory tests (Tomljanovich et al. 1978) shows that traveling screens equipped with 1.0 mm (0.04 in) screen mesh would substantially reduce entrainment of fish eggs and larvae at DCPP, and that entrainment of larval fish and macroinvertebrates could be virtually eliminated by use of 0.5 mm (0.02 in) intake screen mesh. Impingement survival for fish larvae, however, is species-specific: under laboratory conditions, the survival rates for larvae at 48 h after a 16 minute impingement on fine-mesh screens ranged from less than one percent for striped bass to 96 percent for bluegill and smallmouth bass (PG&E 1988a, Appendix D; Tomljanovich et al. 1978). The smaller intake screen mesh would increase impingement of larval and juvenile fish and invertebrates presently entrained at DCPP. The finer mesh screen would convert normally entrained organisms into impinged organisms.

In 1980, Tampa Electric Company (TECO) performed a pilot scale evaluation of a fine-mesh Ristroph screen in the intake canal to its Big Bend Station on Tampa Bay, Florida (Taft et al. 1981; Brueggemeyer et al. 1988). TECO agreed to evaluate the potential effectiveness of fine-mesh screens to reduce losses of the selected Representative Important Species: bay anchovy, black drum, silver perch, spotted seatrout, scaled sardine, tidewater silverside, stone crab, pink shrimp, American oyster, and blue crab. The screen was of the No-well design, a duo-flow screen design with the screen attached directly to the pump. Based on the positive results of the prototype testing, the regulatory agencies determined that Unit 4 could be constructed with a once-through condenser cooling system provided that fine-mesh screens were incorporated into the intake structures of both Units 3 and 4. Accordingly, six, 0.5 mm mesh No-well screens were installed at the station and studies of their biological effectiveness were conducted in 1985 (Brueggemeyer et al. 1988).

Initial and latent mortality varied by species and life stage. Collected invertebrates had mortality rates ranging from 10 to 35 percent. Engraulidae (primarily bay anchovy) had initial mortality rates ranging from 42 to 84 percent and latent mortality rates ranging from 32 to 35 percent. Bay anchovy, Atlantic tomcod, and Atlantic silverside eggs showed a total mortality of 72.4 percent (unadjusted for control). Yolk-sac larvae of mummichog, Atlantic silverside, Atlantic tomcod, white perch and winter flounder mortality ranged from 62 to 100 percent with the exception of

winter flounder, which had a projected mortality 11 to 62 percent. Assumed mortality for post-yolk-sac larvae ranged from 36 to 100 percent for all species in this life stage. The conclusion of the study was that survival rates were comparable to, and in some cases exceeded, those obtained during the prototype study. There was no significant difference in survival rates between the two sample locations.

At Brayton Point Station Unit 4, biological evaluations were conducted to determine the number, species, and initial and extended survival of fish impinged on the modified intake screens (Davis et al. 1988; LMS 1987). These fine-mesh, angled screens were installed at a new Unit 4 intake to divert larger, motile life stages and gently collect and recover early life stages. The lowest survival was calculated for bay anchovy and the highest was for tautog. Initial and extended survival varied by species; however, a certain group of numerically dominant taxa was classified by the authors as "fragile" (primarily, bay anchovy and Atlantic silverside). The fragile group had a calculated survival below 25 percent while a "hardy" group, dominated by winter flounder and northern pipefish, had survival values greater than 65 percent.

Retrofitting the existing DCPP through-flow screens with a fine mesh would be difficult due to the increased flow resistance increasing the potential for screen failure under high debris loading. The size of the existing DCPP circulating water pumps and the intake configuration would preclude the retrofitting of No-well screens. No-well screens would require a new, open intake structure, with new circulating water pumps.

As part of an evaluation of screening technologies for DCPP (PG&E 1996), retrofitting the DCPP intake with center flow screens with curved baskets and a fine mesh (2mm) was considered as a way to improve debris filtration. The curved baskets of the centerflow design would increase flow area and offset the flow restriction caused by the finer mesh. These screens could be equipped with finer mesh screens (1 mm) that would potentially allow screening of larval organisms. These screens could reduce entrainment but would require a site-specific evaluation to determine the following parameters:

1. The survivability of larval organisms washed from the screens.

2. The potential increase in impingement due to the screen basket shape and the flows associated with a centerflow design.
3. The ability of a fine mesh screen to function under high debris loading conditions experienced at DCPP.
4. The ability of the refuse handling system to process debris with minimal impact to screened organisms.

The estimated cost for these screens is estimated $7,000,000 (PG&E 1996). This does not include any modifications to the debris handling system or potential modifications to the intake structure to accommodate the new screens. The total cost would be approximately $10,000,000. The net present value of this modification is (-) $7,906,000.

Center flow screens have a much more complex distribution of flow velocities, especially when installed in a screen well type intake like that used at DCPP. The entrance area to the center flow screen is roughly 60% as wide as the existing traveling screens, so the velocity at the screen entrance would be about 67% greater (1.3 to 1.7 ft/sec). The velocities at the screens are about 70% compared to the current screen approach velocity. In between, the flow turns 90 degrees, resulting in a complicated flow pattern. These flow patterns could increase impingement of juveniles and adults. In addition, the baskets for the proposed center flow screen are convex in cross section, which could increase retention of any fish impinged. With the mixed survivability data for larval organisms impinged on fine mesh screens, any biological benefit (or cost if there is an increase in impingement) would have to be evaluated prior to determining the effectiveness of this alternative.

6.2.2.4 Single Circulating Water Pump Operation

A reduction in the number of operating CWP would reduce cooling water flows, proportionately reducing the number of organisms entrained at DCPP.

The Diablo Canyon Power Plant is designed and operated as a base-loaded plant with relatively constant electrical generation for extended periods of time. The operational characteristics of Units 1 and 2 limit the potential effectiveness of single pump operation as an alternative for reducing entrainment and impingement losses. This would require extended operation at 50%

power, which lowers plant efficiency, reduces revenue, and degrade critical plant equipment. An option to running both units at 50% power levels would be to run only one unit at 100% power.

The use of a single pump per unit at DCPP with the units operating at reduced load would also present certain technical operating difficulties. The motors that operate the CWP are large (13,000 hp) and undergo substantial stress and wear during the startup. In order to achieve appropriate performance of the unit at reduced load (e.g. at 50% power or less) or to increase load above 50% in response to increased demand, the second circulating water pump would be needed from time to time. Intermittent re-starts of the second pump would greatly increase the wear, reduce pump motor reliability, and increase the frequency of maintenance.

A second difficulty arises directly from single pump operations. With one circulating water pump operating, there is no backup on-line. Thus, if the operating pump were to fail, the unit would undergo a forced shutdown. Ordinarily, with two circulating pumps running, failure of one requires a curtailment to only 50% power. Although there is no increased risk of an accident affecting the health and safety of the public by operating with one circulating water pump, plant operational transients that force a unit shutdown are regarded as a negative indicator by the Nuclear Regulatory Commission.

Because of the loss of reliability, potential equipment degradation, and operational inefficiency of generating units that would accompany operating each unit with one circulating water pump in service, a more likely strategy would be to take the option of shutting down one unit and both of its pumps, if operation in this mode was expected to be for a long period.

The NPV of this option is extremely negative, based on loss of generation. The lost revenue and NPV for this option are, however, considered proprietary information.

This option has a negative contribution to the environment as a result of the need for the ISO to purchase replacement power for the 1,100 MW reduction in power production. It is assumed that the replacement power will be generated by gas-fired power plants constrained to the state of

California emission standards. The increased amount of emissions from a natural gas-fired power plant generating 1,100 MW would be as follows (tons/yr):

NO_X	CO	CO_2
5,500	3,700	5,900,000

NO_X and CO are EPA criteria pollutants regulated under Title V of the Clean Air Act. CO_2 is a "green house" gas and is of concern for global warming. Additional air toxics such as 1,3 butadiene, acroliene, and formaldehyde are generated in smaller quantities, but can represent a greater risk to the population and environment. These calculations are based upon emissions from a 750 MWe gas-fired power plant with moderate amounts of emissions controls. Emissions used were 95 ppm NO_X, 100 ppm CO, and 9.6% CO_2. Mass emissions were based upon operation of 24 h/d, 365 d/yr.

Because of the loss of generation that would accompany reduced circulating water pump operation, single circulating water pump operation or single unit operation is not considered an alternative that could reduce entrainment at DCPP in a cost effective manner.

6.2.2.5 Variable Speed Circulating Water Pumps

Installation of variable-speed motors on the circulating water pumps represents one approach to reducing cooling water flows to the minimum level necessary to maintain efficient operation of the unit at a specific generating load. For this analysis, the assumption is that circulating water flow would be reduced to that required for 100% power operation. The basis for this is that DCPP is designed to operate as a base loaded plant at 100% power and does not change power output on a daily or other scheduled basis.

The pumps currently in use are limited to no-flow or full-flow operation. Variable-frequency drives could be installed to allow these pumps to operate at lower speeds (and hence lower flows), thereby reducing the numbers of organisms entrained. Thus, the circulating water flow rate could be adjusted to provide only the flow needed for condenser cooling within operating limits imposed by ΔT and back pressure operating criteria. The magnitude of the resultant

reductions in entrainment losses would depend on the reduction in cooling water flow and the abundance of organisms at the times when the circulating water pumps were operated at reduced flow rates.

The intent of variable speed pumps is to reduce intake flow without impacting plant power operation (by adjusting circulating water flow to that needed to just allow operation within thermal limits) then the biological benefits of such a modification would be limited and based on the amount of flow reduction. DCPP was not designed for large extremes in cooling water ambient temperature, so there is little margin for reducing circulating flow. Based on calculations (VR Foster, PG&E, pers. comm. 1999), the maximum expected flow reduction would be 2–10% when ocean water temperatures are below the average (14.4° C). These calculations allowed for a temperature differential rise of 11.1° C (to allow for margin to the temperature limit) and did not consider the impact of condenser fouling. Further decreases in cooling flow would cause additional heating of the discharged circulating water.

Operation of the existing circulating water pumps is also limited by the pumps' ability to operate at reduced flows without cavitating. Hydraulic model test results for the existing circulating water pumps revealed that the available suction head in the present installation restricts operation at reduced speeds without subjecting the pumps to serious cavitation. There is data for cavitation performance of the circulating water pumps at 80%, 90%, and 100% of rated speed. The data demonstrates that the pumps will be more susceptible to cavitation as speed is reduced. Based on existing data, it is estimated that the greatest flow reduction would be on the order of 20 to 30% (corresponding to a speed reduction of 10%; PG&E Drawing DC663021-Sht. 29). The present circulating water pumps are not capable of reliable service at a significantly reduced speed. Since the circulating water pumps are embedded in the concrete intake structure, replacement of the circulating water pumps with pumps designed for variable speed operation would require the construction of a new intake structure similar in size to the existing structure. Variable speed circulating water pumps can allow substantial reductions in cases where power plants load follow (change power over the course of a day or week) and where plants are designed for large variations in heat sink temperatures. Since DCPP is a base loaded plant designed for a relatively

constant heat sink temperature, there is little flow reduction that can be expected (up to 10%). This limits the biological benefit of installing variable speed drives on the existing pumps. Since variable speed drives of a size needed are not normally produced, the costs for this modification are difficult to estimate. Based on input from one experienced vendor, the net present value of installing variable frequency power units would be approximately (-) $7,652,000.

Due to the low reduction in flow and relatively high (and uncertain) costs to install these drives, the minimal reduction in entrainment is not considered to provide a substantial biological benefit.

6.2.2.6 Refueling Outage Scheduling

Diablo Canyon Power Plant Units 1 and 2 are periodically removed from service for refueling and maintenance. The unit outages occur at an interval of approximately 18 mo. Future outages are expected to last approximately 30 d. The seasonal planned reduction alternative would involve the selective scheduling of unit outages to coincide with the peak periods of abundance for key larval taxa in the area. Ideally, refueling outages would be scheduled for specific times of the year when densities of entrained organisms are greatest.

Over the past 15 years the majority of the 18 refueling outages have occurred during the months of March–April (7 outages) and September–October (5 to 8 outages). Fall outages have occurred regularly during these months, but in recent years winter outages started earlier, running from February–March. Of the 14 target taxa including subgroups, the majority reproduce during the spring months of March–June (**Table 6-2**). Few of these taxa are vulnerable as larvae during fall. A shift in outage scheduling to the spring would lessen the susceptibility of these larvae to entrainment. Having both units refuel each spring is highly uneconomical. Extending each unit to a 24 mo cycle would allow for spring outages, but is not possible with current fuel design. During the normal operational cycle about one-third of the fuel is consumed over an 18–21 mo period before refueling is required. This limits the possibility of modifying outage schedules to a shorter duration. Without the ability to lengthen fuel cycles, outages could not be scheduled only

during the spring. Without this flexibility, shortening outage schedules to occur on 12 month intervals would become inefficient.

A second scenario of reduced operation could involve curtailments during periods of greater larval density. Monthly densities of fish larvae peak during the spring months from March through June, as depicted for 1997 and 1998 in (**Figure 6-6**). During the months March –May approximately 55% of the fish larvae for the nine months ending in September are at risk to entrainment. This nine month period was used because plankton data after September 1998 were analyzed only once-per-month with paired study grid surveys. During both years, larval densities peak in May each year when 23 and 31 percent of larvae were collected. These strategies would be protective of many species including KGB rockfish (*Sebastes* spp. V_De), northern anchovy, monkeyface eel, Pacific sardine, northern lampfish, smoothhead sculpin, white croaker, and California halibut. Spring time curtailments, however, would not be protective of other fishes including clinid kelpfish, blackeye goby, blue rockfish (*Sebastes* spp. V), snubnose sculpin, and some KGB rockfish (*Sebastes* spp. V_D) .

A dual unit curtailment for a three month period each spring would have a significantly negative financial impact on PG&E. The NPV of this option is extremely negative, based on loss of generation. The lost revenue and NPV for this option are, however, considered proprietary information. Replacement power obtained from gas-fired facilities would generate about one-half the EPA criteria pollutants and "green house" gas emissions calculated in **Section 6.2.2.4** each year under this scenario.

6.2.3 Conclusion

A hierarchical evaluation was used to assess which alternative intake technologies were technically feasible and could reduce impingement and entrainment at DCPP.

A reassessment of the 1985–86 316(b) Demonstration Study (Tenera 1998a) confirmed that estimated impingement at DCPP is very low (0.32 to 0.73 kg/d/unit). The current report found that, for any type of modification, very little improvement can be made to the present DCPP

intake configuration to reduce the number of impinged fish, without incurring substantial costs relative to the benefit achieved. Regardless, **Section 6-2** of this report studied the feasibility of three alternative technologies: an increased area intake, angled screen intake, and modified traveling screens with enhanced fish handling equipment.

Increasing the cross-sectional area of the intake structure would decrease the water flow rate. But, apparently healthy juvenile and adult fish have been observed living and swimming inside the intake structure at DCPP without being impinged. Also, implementing this alternative would require very substantial disruption of the intake cove as a cofferdam would be required to de-water the cove while dredging and associated modifications took place. The angled screen intake design is judged to have similar costs and benefits as the increased area intake, with the potential for increase fish recovery.

The cost of implementing either the increased area intake or the angled screen intake alternatives is estimated to have a net present value of approximately (-)$275,000,000. This equates to approximately $61,700 per kilogram of fish saved, assuming the modification could completely eliminate impingement.

Due to the high cost as well as the high impact to the Intake Cove during construction and negligible reduction in impingement, these alternative intake technologies are not considered cost effective for DCPP.

Applicability of the most current intake screen and fish conveyance technologies were evaluated. Proposed modifications to the existing configuration include the addition of angled intake screens, watertight fish collection buckets, low pressure spray wash, a gravity sluiceway to convey fish back to the ocean, and modifications to the screens to allow continuous rotation.

None of the organisms impinged on the screens at DCPP are assumed to survive because of the current debris removal configuration. Installation of a gravity sluiceway and low pressure spray wash could reduce the mortality rate of impinged organisms.

Although there is insufficient data to perform a thorough site-specific analysis, modified angled traveling screens with continuous rotation, low pressure spray wash, and watertight fish

collection buckets have the potential to reduce the mortality of impinged organisms at DCPP. Implementation of this modification has associated capital and maintenance costs resulting in a net present value (NPV) of (-)$11,000,000. The cost of saving impinged fish with this modification, assuming 100% survival, is on the order of $2,200 per kg of impinged fish assuming a 100% reduction in impingement. Site-specific testing of this proposed modification would be required to determine the actual biological benefit.

Because impingement at DCPP is already low, modification of the current traveling screens to further reduce impingement is not considered cost effective for DCPP.

For entrainment, five alternative technologies have been identified as technically feasible at DCPP. The alternatives are closed-cycle cooling towers, fine mesh screens, single circulating water pump operation, variable speed circulating water pumps, and changes to refueling outage scheduling.

Two types of closed-cycle cooling towers were evaluated. Both would reduce the cooling water requirement, possibly as much as 80–95%, and result in entrainment reductions. However, installation and operation of closed-cycle cooling towers have negative environmental impacts on air quality (salt drift), increased salinity of the cooling water discharge, terrestrial impacts of salt drift, and poor aesthetics as a consequence of high cooling towers and vapor plumes.

Additionally, closed-cycle cooling towers have high capital and maintenance costs. Both units would be de-rated to accommodate the new cooling medium. Order of magnitude estimated net present values (NPV) range from (-) $503,000,000 to (-) $1,072,000,000, plus costs for design, licensing, demolition of existing facilities, and environmental damage to the land. Because of the adverse terrestrial environmental impacts and high implementation costs associated with close-cycle cooling towers, they are not considered feasible at DCPP

Center flow screens with fine mesh (1 mm) baskets could be retrofitted to the existing DCPP intake structure to allow screening of larvae. Studies have shown wide ranging mortality rates for impinged larvae, ranging from 5% to near 100%, depending on species. A site-specific investigation would be needed to determine the mortality for larval organisms at DCPP. In addition, there is a potential that the center flow screens, with their higher inlet velocities,

complex flow patterns and curved baskets could increase impingement of juvenile and adult fishes. Both survivability of larvae and any changes to impingement would have to be evaluated prior to assessing a biological benefit for fine mesh baskets on center flow screens. The net present value of this modification is (-) $7,906,000, with an uncertain benefit.

Single circulating water pump operation was also evaluated as a method to reduce entrainment. Operating only one of the two CWP running per unit would reduce the cooling water requirement by about half, reducing entrainment by the same amount. Single circulating water pump operation would require that both units at DCPP be operated at 50% power or less or that one unit be shutdown. The loss of revenue associated with single circulating water pump operation does not make this a feasible alternative.

Variable speed drives for the circulating water pumps offer the possibility of reducing flow to the minimum required flowrate (optimizing generation and condensate depression) needed to support operation at a specific level of plant output. Since DCPP is a base loaded plant operating at 100% power most of the time, the potential flow reduction is limited to 2 to 10% of rated flow, depending on ocean temperatures. Additional flow reductions are limited by the capabilities of the pumps themselves, such that significant flow reductions (even with a reduction in power are limited to flow reduction of approximately 20 to 30%). The costs and limited potential flow reduction with this alternative does not make this a cost effective alternative.

The feasibility of scheduling refueling outages to coincide with periods of high larval densities of certain organisms was also evaluated. During refueling outages, the amount of cooling water required is reduced and strategic scheduling could reduce the amount of entrainment. It is possible that at least some of the refueling outages could be scheduled to coincide with periods of high density of certain larval fish or invertebrate taxa. Many fishes produce larvae to benefit from the increase productivity associated with spring and summer months. Conducting outages during the spring could reduce the impact on nearshore taxa 7–10% assuming two cooling water pumps are out of service for 30 d.

Longer curtailments of cooling water pump operation, exclusive of refueling outages were also evaluated. Water flow conditions would be severely reduced for one to three months under this

scenario. Some or all cooling water pumps would be shut down when a majority of the fish larvae are present in the plankton. The current data indicate a large percentage of the larval fishes are vulnerable to entrainment through the spring months March–June. Suspending pumping operations, except for vital ASW flows, for three months would protect about half of the fish larval density from risk of entrainment. The costs of these measures are extreme and many fishes that reproduce at other times or have long-lived larval stages would remain vulnerable to entrainment during the 9–11 months of operation.

Table 6-1. Listing of the alternative technologies evaluated and location of the evaluation in this report.

Category	Intake Technology	Meets Criteria 1	Meets Criteria 2	Meets Criteria 3	Evaluated in noted Section or Appendix
Cooling Water System	Once Through	N/A	N/A	N/A	in use at DCPP
	Closed-cycle cooling tower (saltwater)	●	●	●	Section 6.2.2.1
	Closed-cycle cooling tower (fresh water)	●	●	●	Section 6.2.2.2
	Closed-cycle Cooling Pond or Canal	●	●		App. N
	Cooling system component modification				App. L
Intake Configuration	Offshore Intake Location / Velocity Cap	●			App. M
	Alternate Onshore Intake Location	●			App. M
	Shoreline	N/A	N/A	N/A	in use at DCPP
	Recessed	●			App. M
	Increased Area Intake Structure	●	●	●	Section 6.2.1.1
	Angled Screen Intake Structure	●	●	●	Section 6.2.1.2
Behavioral Barriers	Light	●			App. M
	Sound	●			App. M
	Bubble screen	●			App. M
	Velocity gradient (water jet or other turbulence)				App. L
	Electrical barrier				App. L
	Louvers				App. L
	Chemicals barriers				App. L
	Magnetic field (barrier)				App. L
	Chain and cable barriers				App. L
Physical Barriers	Vertical traveling screen	N/A	N/A	N/A	in use at DCPP
	Drum screen	●			App. M
	Centerflow traveling screen	●			App. M
	Fine Mesh Screen	●	●	●	Section 6.2.2.3
	Media filter				App. L
	Stationary screen				App. L
	Horizontal traveling screen				App. L
	Inclined Plane screens				App. L
Fish Collection, Removal and Conveyance Systems	Combinations of vertical traveling screen modifications	●	●	●	Section 6.2.1.3
	Gravity sluiceway	●	●	●	Sections 6.2.1.2 and 6.2.1.3
Maintenance and Operational Modifications	Maintenance Dredging	●			App. M
	Single circulating water pump operation	●	●	●	Section 6.2.2.4.
	Variable Speed Circulating Water Pumps	●	●	●	Section 6.2.2.5
	Continuous screen rotation	●	●	●	Section 6.2.1.3
	Refueling outage scheduling	●	●	●	Section 6.2.2.6

N/A: Not applicable

Table 6-2. Estimated equivalent adults, months of peak density, and evaluation of susceptibility to entrainment for target crab and fish taxa collected in DCPP entrainment and study grid plankton samples.

Common Name	Months of peak density		Estimated Equivalent Adults	Reduced Entrainment
	Entrainment	Study Grid	(*FH* and *AEL*)	Feb-Mar & Sep-Oct
Cancer crabs				
Brown rock crab	Spring	May–Jun	9,100–23,400	maybe
Slender crab	Winter–Spring	May–Jun	895–5,460	maybe
Pacific sardine	Mar–May	Mar–May	3,170–8,460	Yes
Northern anchovy	Dec–May	Mar–May	16,100–120,000	Yes
Rockfishes				
KGB complex	Mar–Jul	May	497–1,120	Yes
Blue complex	Jan–Mar	Jan–Feb	18–353	Yes
Painted greenling	all year, esp. Mar–May	May–Jun	nd	Yes
Sculpins	Apr–Jul	Apr–Jul	nd	No
White croaker	Dec–Mar	Dec–May	5,110–15,000	Yes
Monkeyface prickleback	Mar–Jun	Apr–May	nd	Yes
Kelpfishes	Jan; Jun–Aug	Dec–Feb	nd	No
Blackeye goby	Mar–Nov	July–Sept	10,300–75,400	Yes
Flatfishes				
Sanddabs	Jul–Sept	July–Sept	92–2,370	Yes
California halibut	April–May	Apr–May; Jul	nd	No

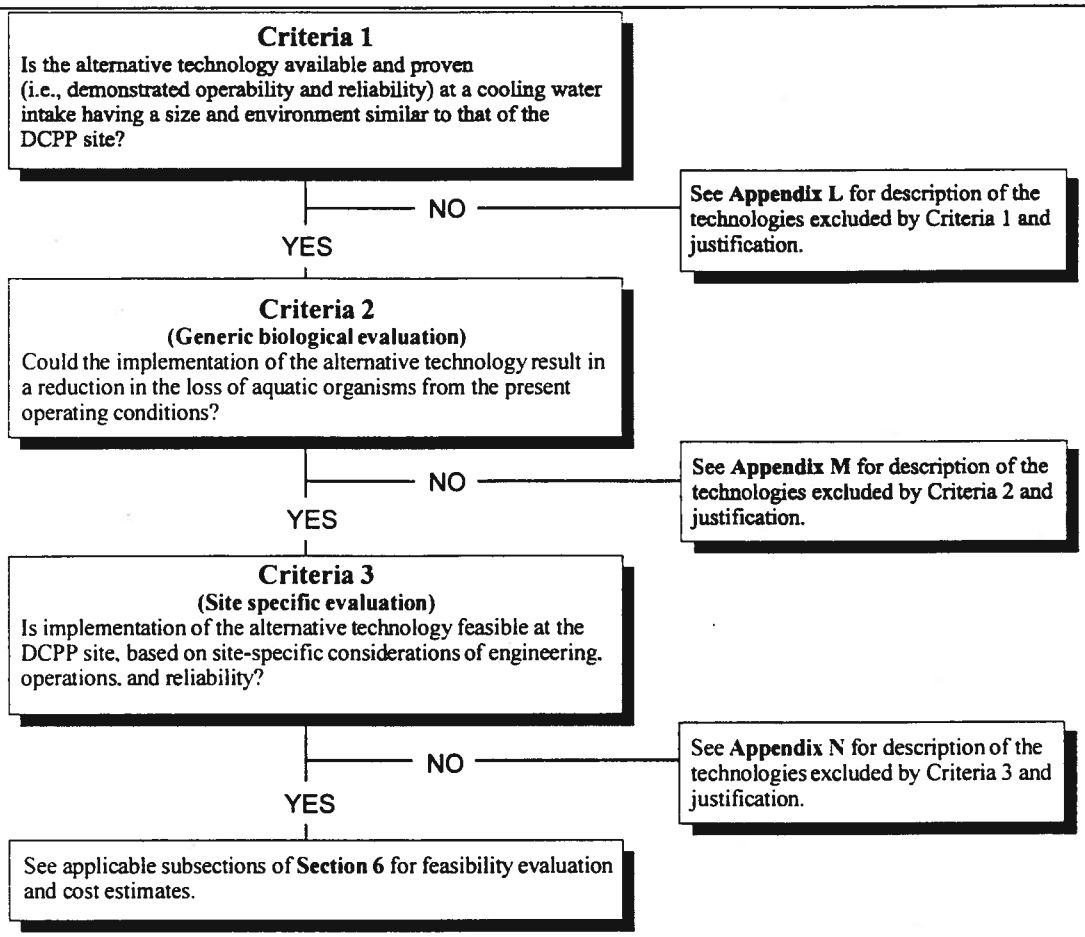

Figure 6-1. Alternative intake technologies evaluation flow chart.

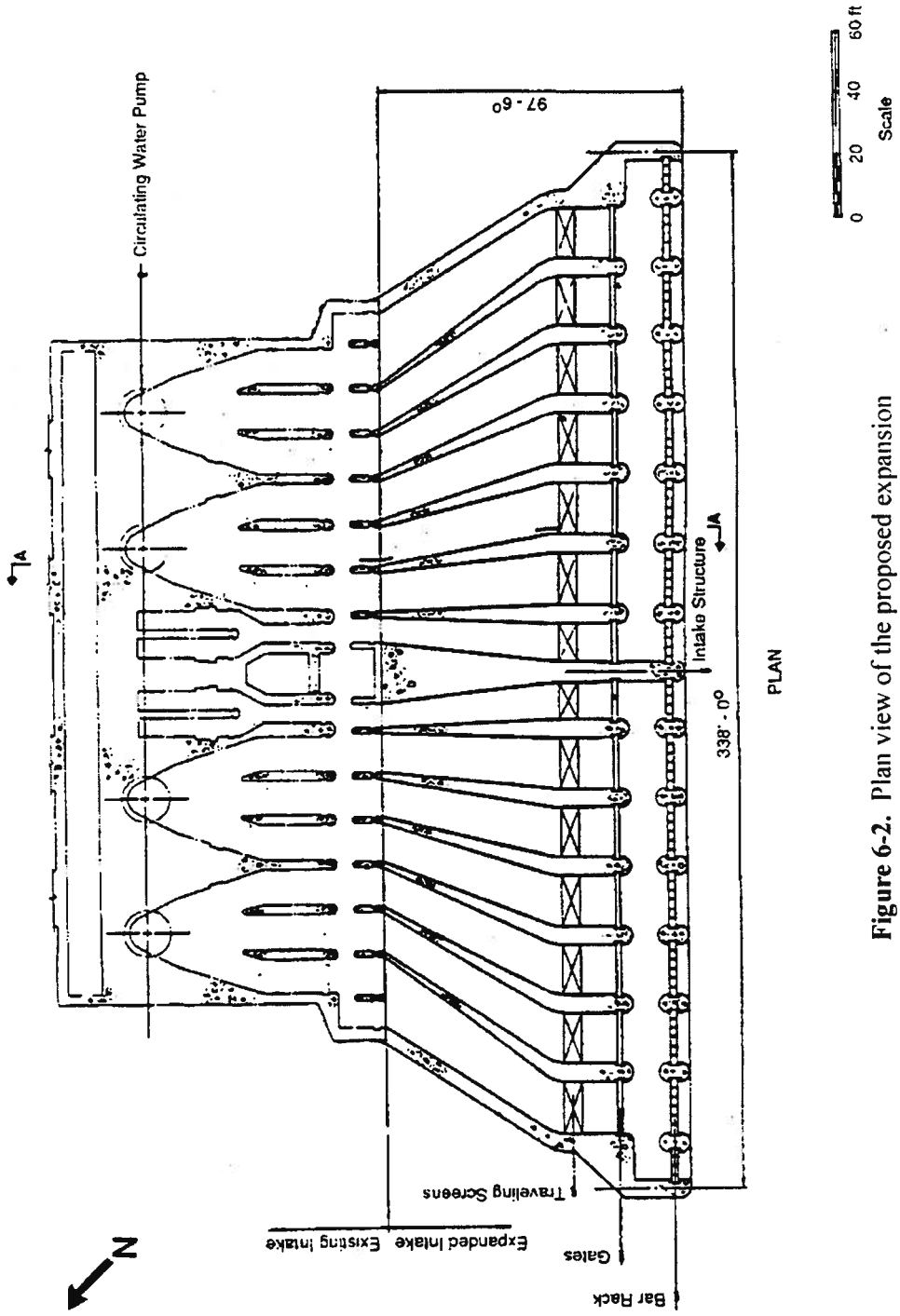

Figure 6-2. Plan view of the proposed expansion of the intakes at the Diablo Canyon Power Plant.

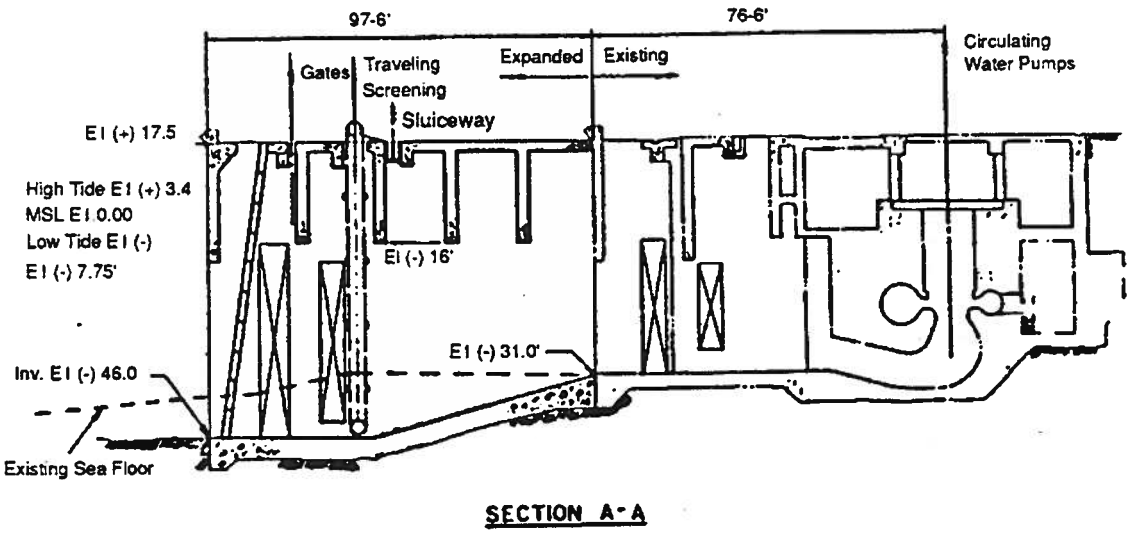

Figure 6-3. Sectional view of the proposed expansion of the intake at the Diablo Canyon Power Plant.

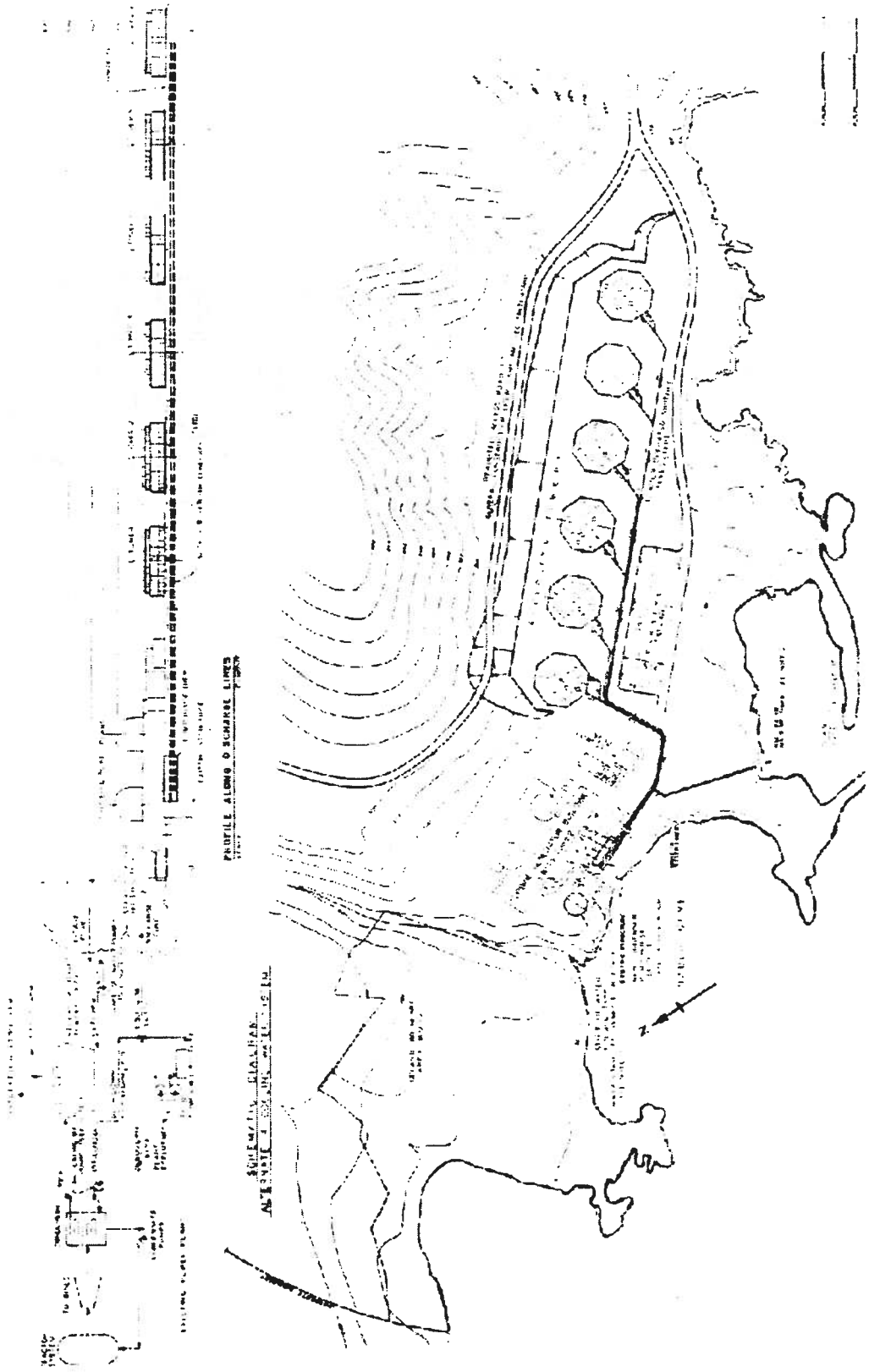

Figure 6-4. Schematic drawing of potential siting of mechanical draft cooling towers at the Diablo Canyon Power Plant site.

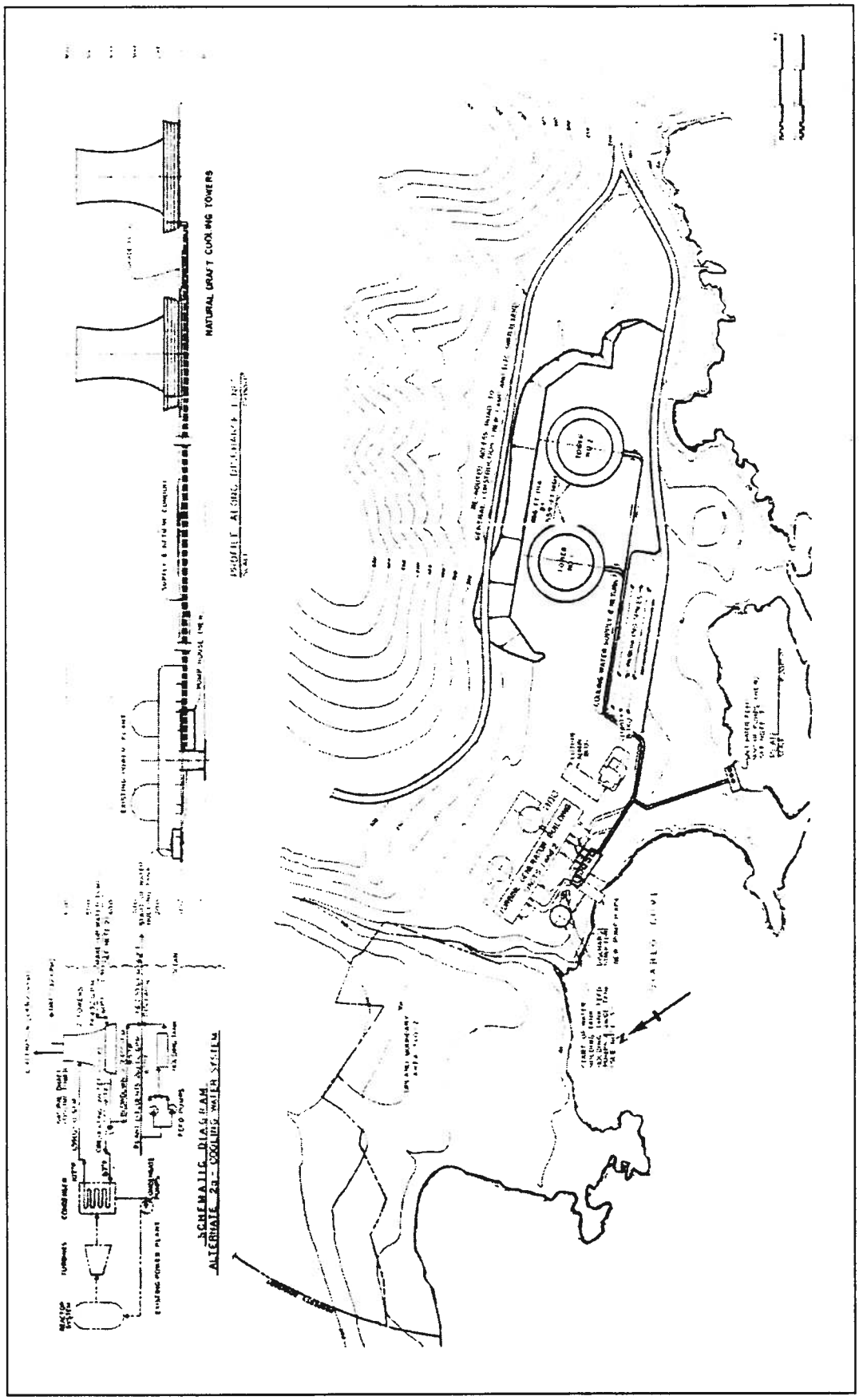

Figure 6-5. Schematic drawing of potential siting of natural draft cooling towers at the Diablo Canyon Power Plant site.

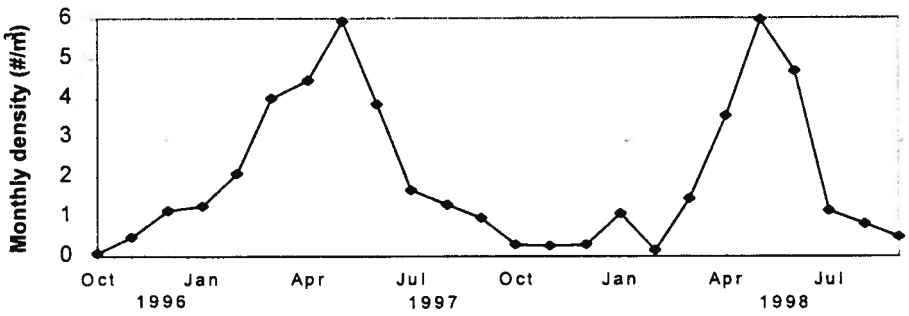

Figure 6-6. Monthly density of all fish larvae collected in weekly entrainment surveys conducted from Oct 1996 to Sep 1998.

7.0 Discussion

The purpose of this study was to assess the effects of the Diablo Canyon Power Plant (DCPP) cooling water intake structure on populations of fishes and invertebrates. Effects from the plant's cooling water intake structure can result from impingement of organisms on the intake traveling screens or entrainment into the plant's cooling water intake structure. This study focused on entrainment effects, because an evaluation by the Regional Board's Entrainment Technical Workgroup (ETWG) determined that impingement effects were not significant (Tenera 1998b). Although many marine organisms have planktonic forms that could be entrained by the power plant, the ETWG focused on the larvae of fishes, crabs, and sea urchins. Although hundreds of different taxa were entrained, the ETWG chose 16 target taxa for this assessment that included 14 fishes and two species of crabs. Their decision was based on criteria that included the abundance of a taxon, the availability of suitable life-history information to meet assessment model requirements, and criteria outlined in USEPA Draft Guidelines (USEPA 1977) and modified by the ETWG (Tenera 1997b). The ETWG determined that several assessment approaches would be applied to the data for each taxon where possible. Where we were able to complete two or more approaches, we compared results to provide a basis for determining the accuracy of our assessment for an individual taxon.

7.1 Assessment Approaches

Three assessment approaches developed by PG&E, its consultants, and other members of the ETWG were used to estimate entrainment effects at DCPP. Estimates of entrainment loss, and the demographic data reported in the scientific literature, were used to parameterize two demographic approaches for estimating entrainment effects: Fecundity Hindcasting (*FH*) and Adult Equivalent Loss (*AEL*). These methods are similar except that *FH* uses the number of larvae entrained to hindcast the number of eggs, and the number of eggs is then used to estimate the number of female adults that would have produced them. The *AEL* method forecasts the number of adults that would have survived from the estimated number of entrained larvae, as if they had survived entrainment, using knowledge of natural mortality rates. It has been assumed throughout this

study that 100% mortality occurs for all entrained larvae. A third method, the Empirical Transport Model (*ETM*), estimates an average annual larval mortality due to entrainment $\widehat{P_{M}}$ per individual taxon, using estimates of proportional entrainment (*PE*) that compare the number of larvae entrained in one day to the number of available larvae in the source water body. Larval mortality is calculated after *PE* is weighted by the estimated fraction of the total population affected and compounded by the time larvae are susceptible to entrainment.

The three approaches combine demographic information with environmental setting to model entrainment effects and then, where possible, compares these results to corollary data. The corollary data used in this report included fisheries information, data from Intake Cove plankton tows, and subtidal observations of juvenile and adult fishes. The integration of growth, reproduction, and mortality parameters in the three models are used in obtaining estimates of adult losses and entrainment mortality. These estimates help us better understand population measures such as length-frequency distributions and indices of year-class strength. Much of the effort in compiling these models is similar to the process undertaken by fishery scientists to conduct stock assessments used to gauge the health of fished populations.

Stock assessments are tools fishery scientists use to determine the health of a fish population. These assessments combine available biological and demographic data with information on fishing effort and landings to assess trends of fish abundance. These trends can be assessed from sources such as long running observational data sets (e.g., subtidal fish observations, Tenera 1997c), annual fishery catch statistics, and data on fishing effort. Fishery scientists divide the amount of catch by the amount of time spent fishing to create an index of abundance called 'catch-per-unit-effort.'

Catch data alone are an imperfect estimator of abundance since they are an indirect measure of the available fish population and may also reflect biases toward larger, more easily harvested, or more valuable fishes. Therefore, fishery scientists must combine biological data describing the population under study with catch data to understand the trends they observe. The types of biological data that are useful to have are: the age structure of the population, age at first entry into the fishery (recruitment), age at first reproduction, estimates of longevity, and perhaps most

importantly, estimates of total mortality. Total mortality is the combination of mortality due to both natural and anthropogenic causes like fishing. Understanding the effects of entrainment on larval fish populations requires similar steps as those taken to assess the health of adult populations.

Similar to the process fishery scientists use for assessing fish populations, the assessment models for DCPP required several decisions regarding the assumptions, level of complexity, data inputs, and application. Issues considered included the following:

- compensation;
- entrained eggs;
- calculation of larval duration; and
- application of model results in an assessment of habitat damage.

In addition, all assessments were based on the assumption of 100% mortality of all entrained organisms.

7.1.1 Compensation

Saila et al. (1987) define compensation as "the ability of a fish population to offset, in whole or part, reduction in numbers caused by impacts from natural and/or man-induced stresses, which includes fishing." Postulated compensatory mechanisms include growth, competition, predation, cannibalism, size at maturity, fecundity, disease, and reproduction. Ricker (1975) reports that compensatory mortality, as a function of population size, reduces the intrinsic rate of population increase. Nisbet et al. (1996) say that compensation fails to prevent declines in adult stocks in response to added mortality of larvae by power plants, except when compensation occurs in the late immature stages, after entrainment but before maturation. Density-dependence is not confined to acting through mortality; growth and fecundity may also be density-dependent. Therefore, the processes that regulate populations are complex. Some entrainment studies have assumed that compensation is not acting between entrainment and the time when adult recruitment would have taken place, and further, that this specific assumption resulted in conservative estimates of projected adult losses (Saila et al. 1997). Others, such as Parker and DeMartini (1989), did not include compensatory mortality in estimates of equivalent adult losses

because of a lack of consensus on how to include it in the models and, more importantly, uncertainty about how compensation would operate on the populations under study. The uncertainty arises because the effect of compensation on the ultimate number of adults is directly related to the vital processes (fecundity, somatic growth, mortality) and affected life stages. In particular, Nisbet et al. (1996) showed that neglecting compensation does not always lead to conservative long-term estimates of equivalent adult losses. Density-dependent recruitment compensation is also hard to distinguish from competing climatic effects. Confounding of environmental and density-dependent effects is a major problem that can result in spurious environmental effects being mistaken as density-dependent effects(Clark et al. 1999). The sampling plan and modeling evaluation for these entrainment studies at DCPP (Tenera 1998a) considered that population regulation could include both positive and negative compensatory mechanisms. We also concluded that it would be beyond the scope of the current study to include population regulation in the models used to predict adult losses.

Compensation (density-dependence) is also an important component of the fishery management concept of surplus production (Ricker 1975; **Figure 7-1**). This theoretical relationship describes the relationship between adult abundance and the number of progeny they produce. In this construct, there is an equilibrium population level at which there are just sufficient adults in the population to replace themselves. Production above and beyond this level is termed "surplus production" that can be harvested without depleting the population. Entrainment mortality is analogous to harvesting mortality, except that it removes younger animals from the population. It appears that for many of the target taxa examined at DCPP, the additional mortality on early life stages added by entrainment in the cooling water intake system could be attributed to surplus production since there are no concurrent decreases in adult or larval abundance measured at other locations.

7.1.2 Eggs

Planktonic eggs are entrained along with larvae at DCPP. Early in the process of developing the present study, the ETWG decided not to enumerate fish eggs in the plankton samples collected. The reasons for this decision were that, while there are some descriptions of marine fish eggs,

their classification is both difficult and time consuming, and ultimately, the initial list of possible target taxa did not include many fishes with pelagic eggs that would be subject to entrainment. Egg losses have been quantified in other 316(b) entrainment studies (e.g., Murdoch et al. 1989; Bailey et al. 1998; Cherry and Currie 1998; Jacobson et al. 1998; Richkus 1998). Lacking the egg entrainment data, one might assume that the numbers of adult equivalent losses resulting from egg entrainment would be equal to those estimated from larvae. However, the actual situation may be more complex; for example, there are differences in dispersion patterns between eggs and larvae, larval behavior may affect entrainment, etc. (Lasker 1981).

Populations of the five target taxa with pelagic eggs (Pacific sardine, northern anchovy, white croaker, sanddabs, and California halibut) are not adversely affected by the DCPP intake indicated by generally low to moderate estimates of annual entrainment mortality ($\widehat{P_{AI}}$), $\widehat{FH}$, or $\widehat{AEL}$. Furthermore, these taxa are protected against localized effects by having widespread populations or spawning centers distant from DCPP. In the case of California halibut, with a relatively high $\widehat{P_{AI}}$, but low approximated $\widehat{FH}$, very few individuals are entrained at DCPP so there is no adverse impact on their population. All other target taxa either brood their eggs attached to their bodies (*Cancer* spp. crabs), extrude live larvae (rockfishes), or have demersal adhesive eggs (greenlings, sculpins, pricklebacks, clinid kelpfishes, and gobies). Therefore, egg losses are not assumed to be an important factor in estimating entrainment effects.

7.1.3 Larval Duration

Larval lengths at entrainment were used to estimate age at entrainment from growth rates reported or derived from current scientific literature. This method has many assumptions with implications for the results of the present study. Primary among these is the assumption that the point estimates reported in the literature, often from areas geographically removed from the central coast of California, are representative of growth rates for larvae in the area around DCPP. Dividing all larval lengths by the same growth rate to obtain estimated ages assumes that growth is constant over the durations estimated. Variation in growth rates, more probable for larger individuals, could lead to miscalculations of larval age. Examination of length-frequency histograms for the various taxa treated in this study and comparison of these with reported hatch

lengths leads to the conclusion that there is much wider variation in hatch lengths of these fishes than is presently reflected in the literature. This follows from the observation that many of the larvae collected and measured in this study were smaller than the reported hatch lengths. In many cases, the average lengths of larvae entrained were less than reported hatch lengths. This may be due in part to larval shrinkage resulting from preservation (Theilacker 1980). The approximately-normal distributions observed for very small length larvae could also result from natural random variation in hatch lengths and could explain the lengths being less than those reported for hatching (Matarese et al. 1989; Moser 1996). If the actual larval ages at entrainment were younger than our estimates the areas containing the affected population would be smaller than those estimated using the present methods. If the affected populations are actually smaller than those used in the *ETM*, then $\widehat{P_M}$ could be underestimated.

7.1.4 Application to Habitat

The conservation of fish habitat has been recognized by scientists and managers as an essential issue in fisheries management as a result of measures included in the reauthorization of the Magnuson-Stevens Fishery Conservation and Management Act (Benaka 1999). These measures emphasize the need to preserve essential fish habitat and integrate ecosystem-based concepts into fisheries management. The ability to quantify the success of management practices for preserving essential habitat is potentially easier than quantifying fish populations that are mobile and difficult to sample. It is recognized that effective management of essential fish habitat requires knowledge of species-specific habitat requirements, descriptions of the physical environment and available habitats, and estimates of fish populations in the different habitats (Cobb et al. 1999). The type of information required to convert $\widehat{P_M}$ into something more tangible that is representative of the effective area or habitat affected is not available for our nearshore target taxa. In addition, while loss of habitat may be logically converted into fish population losses, the conversion of entrainment losses into habitat is less reasonable because there is no specific shoreline area associated with, or degraded by, entrainment. The conversion of entrainment losses into habitat losses also ignores the fact that entrainment only affects one life stage, and recruitment and migration from unaffected portions of the population will continue to fill

available habitat niches. Continuing long-term shifts in oceanographic conditions affecting fish habitat quality and productivity also confounds attempts to convert entrainment losses into habitat.

7.2 Assessment of Entrainment Effects

The following criteria were developed by PG&E, its consultants, and other members of the ETWG for evaluating adverse environmental impacts (AEI) at DCPP. These criteria are specific to the marine environment around Diablo Canyon, in most cases are unique to marine aquatic organisms, and are directly applicable to the present 316(b) entrainment study:

- environmental trends (climatological or oceanographic);
- abundance trends (e.g., Intake Cove plankton tows, subtidal fish observations, fishery catch data);
- life history strategies (e.g., longevity and fecundity);
- population distribution; and
- magnitude of effects.

These criteria are discussed in the sections that follow. The criteria were considered on a taxon-specific basis when trying to determine the extent of entrainment effects on the local environment and on the population at large. This provides a basis for our conclusions in **Section 7.3** for assessing AEI using USEPA guidelines to determine the "*relative biological value of the source water body zone of influence for selected species and determining the potential for damage by the intake structure*" (USEPA 1977). The USEPA (1977) also stated that the biological value of a given area to a particular species be based on "*principal spawning (breeding) ground, migratory pathways, nursery or feeding areas, numbers of individuals present, and other functions critical during the life history.*"

7.2.1 Environmental Trends

Changes in recruitment and adult abundance appear to be periodically accentuated by El Niño events observed in the DCPP receiving water monitoring program (RWMP) long-term subtidal juvenile and adult fish observation data. A decline in abundance in the subtidal fish observation data following the 1991–1993 El Niño event is common to many of the target taxa. In 1992–

1993, nearly all taxa experienced a sharp decline in abundance followed by relatively constant but lowered abundance to the present time. The extended El Niño event between 1991 and 1993 had important implications for many of the fish taxa found here (Yoklavich et al. 1996). Warm water events can have very large detrimental effects on rockfish year-class strength since these are typically species with cold water affinities (Karpov et al. 1995; Lenarz et al. 1995; Ralston and Howard 1995; Lea et al. 1999; Love et al. 1999). Abnormally high water temperatures and suppression of annual upwelling along the California coast have been linked to large-scale reductions in the primary prey of juvenile and adult rockfishes (Mullin and Conversi 1989). Reductions in food sources can negatively affect the somatic and reproductive condition of these life stages (Lenarz and Wyllie Echeverria 1986; VenTresca et al. 1995).

Another strong El Niño event occurred along the central California coast and eastern Pacific Coast during the fall, winter, and spring seasons of 1997–98 (Lynn et al. 1998). This may explain the differences in entrainment estimates between the years 1996–97 and 1997–98 for some of the taxa (**Table 7-1**). Changes in distribution and dispersal patterns of pelagic larvae are expected during anomalous warm oceanographic events (Bailey and Incze 1985; Brodeur et al. 1985). For example, estimates of entrainment, *FH* and *AEL* for both Pacific sardine and northern anchovy increased markedly between the first and second year of the study. The spawning biomass for both of these species is typically centered south of Point Conception (Hart 1973; Butler et al. 1987; PFMC 1998; Hill et al. in press), and it was likely displaced northward with the north-flowing El Niño currents present in the second year. White croaker $\widehat{FH}$ and $\widehat{AEL}$ also appeared to increase slightly, which may also be due to additional transport northward during the El Niño year. Other species appeared to be somewhat less affected by El Niño-related transport since $\widehat{FH}$ and $\widehat{AEL}$ remained relatively constant or decreased only slightly between years (e.g., KGB rockfishes, blue rockfish complex, blackeye goby, and sanddabs). These fishes probably have more localized spawning populations that were less affected by the El Niño event.

Coastal marine ecosystems in the Southern California Bight have experienced reduced productivity associated with a climate regime shift towards warmer water conditions that began during the 1976–77 El Niño (Holbrook et al. 1997). These changes have led to large impacts on

population abundances and trophic structure in nearshore benthic communities within the Bight. In general, they showed that the diversity and abundance of fishes with more northern biogeographic distributions have declined in the Southern California Bight since the advent of this warmer water regime. Furthermore, these assemblages have shown a shift in the dominant species away from more northern distributed taxa and toward more southerly distributed species that are presumably more tolerant of warm water. Similar trends were also observed further north at Santa Cruz Island where populations of surfperches, the standing stock of their crustacean prey, and the biomass of understory macroalgae all declined by ca. 80%. Holbrook et al. (1997) hypothesize that fish abundance declined as a result of declining recruitment that became insufficient to compensate for losses of older age classes. This can be termed "recruitment overfishing" defined by Gulland (1983) as limiting the amount of larvae a population produces by taking primarily older, more fecund adults and leaving young, less fecund individuals. These recruitment levels fell more than one order of magnitude over two decades and were correlated among years with broad indicators of Bight-wide productivity of the coastal marine ecosystem; namely the biomass of zooplankton in the California Current (Holbrook et al. 1997).

7.2.2 Abundance Trends

Data collected for this 316(b) study can be compared with other corollary and long-term data sets to give broader temporal context to the 2–3 year study at DCPP. Several of the target taxa are nearshore species that are not taken by commercial or recreational fisheries. For these taxa, data from long-term studies at DCPP on larval, juvenile, and adult fish abundance were used to examine changes in abundance that may be correlated with entrainment effects.

Declines in adult abundances of clinid kelpfishes and snubnose sculpin over the last several years were detected in DCPP subtidal fish observation data in the control area (Tenera 1997c). This decrease in adult abundance also appears to be correlated with a decline in their larval abundance apparent from Intake Cove plankton tows. These taxa also had some of the highest estimates of total annual entrainment among the target taxa. Smoothhead sculpin, while having relatively high total annual entrainment, did not show similar declines in adult or larval abundance. These and

other nearshore taxa appear to demonstrate similar variation to that observed in previous years from subtidal fish observations and Intake Cove plankton tows.

Intake Cove plankton tows were collected at DCPP starting in 1990. Although sampling occurred throughout the year, larval density estimates were calculated from only the seven months of the year when larval fishes were most abundant (December–June). These data were used to adjust entrainment estimates to a long-term average density in the Cove. This procedure could introduce a bias in the adjusted estimate for fishes whose larvae are only abundant in the water column during the other five months of the year (e.g., sanddabs), and for some taxa an adjustment was not calculated. Two taxa, blackeye goby and clinid kelpfishes, were very abundant during the five months of Intake Cove samples that were not processed. They were adjusted to the long-term average because larvae for these taxa were present in the sorted samples.

Trends in adult abundance of the commercially important blue rockfish complex and cabezon also showed declines in our long-term fish observation data. The early and mid-1990's saw the advent of live-fish fishing along the central coast of California (Bloeser 1999) that could be related to these declines. Live-fish fishing has been identified as a cause of "growth overfishing" (Gulland 1983) and sharp declines in adult abundance (Starr et al. 1998). Landings at Morro Bay area ports do not indicate long-term declines in the catch of adult cabezon, although a 26% reduction in landings in 1999 compared to 1998 suggests that the 'live-fish' fishery cannot be sustained at high exploitation rates for long. Fishery harvest data do not indicate any long-term declining trends for other commercially or recreationally important species also entrained at DCPP. Any potential effects of power plant larval entrainment on catches of harvested species may be masked by changes in exploitation rates and the lack of credible catch-per-unit-effort data needed to examine long-term trends. Clearly, other longer-term declines associated with a climate regime shift that began in the late 1970's (Holbrook et al. 1997) must also be considered.

7.2.3 Life History Strategies

Life history strategies played an important role in the three assessment approaches we used to assess the magnitude of entrainment effects. Fecundity hindcasting (*FH*) estimates of adult losses relied on the number of eggs and fish or crab larvae produced during their life. Pelagic taxa like

northern anchovy typically produce numerous eggs that hatch relatively under-developed larvae. This strategy hedges against high egg and larval mortalities by reducing the amount of energy placed into any one progeny and assures that at least a few will survive to adulthood. Another common strategy found among the target fish taxa is producing relatively fewer, demersal, adhesive eggs. More energy is expended into fewer eggs that hatch more developed larvae than the taxa with pelagic eggs. This strategy is advantageous when combined with the demersal eggs since this latter adaptation reduces otherwise high egg mortalities. Thus, egg and larval survival rates up to the point of entrainment are directly linked to the reproductive strategy of the particular species being assessed and have marked effects on the demographic approaches used for estimating entrainment effects. The *AEL* approach depends on survival estimates from the time of entrainment to recruitment of adults into the fishery. Estimates of survivorship are undocumented for many of the taxa considered in this report and therefore they had to be estimated or substituted from other related taxa. The level of uncertainty associated with these survivorship estimates is typically unknown and can contribute to overall uncertainty in the estimates from these approaches. This also shows the importance of larval duration in the formulation of the approaches. Differences in life histories and adult distributions were accounted for in the formulation of the *ETM* among taxa. For example, pelagic species that released planktonic eggs are treated differently from nearshore species with demersal egg masses by estimating the population of inference using offshore extrapolation and alongshore extension of study grid abundance.

The availability of life history information allowed multiple assessment approaches to be used on several of the target taxa, but in the absence of independent population estimates for these taxa, our ability to assess effects was limited. For example, demographic data on egg and larval mortality for blackeye goby allowed the application of multiple assessment approaches, but in the absence of a population assessment or any fishery data for this species we could not provide any context for the estimates. The utility of any modeling technique to yield relative assessments of population-level effects depends on data describing the size of the adult fish populations and the ability to convert *FH*, *AEL*, and *ETM* estimates into comparable units of fishes in the wild.

For taxa where all three assessment approaches were completed, comparison among the results illustrate the usefulness of the multiple assessment approach. While the range and variance of life history parameters and population estimates are sources of uncertainty in our estimates, we can determine our areas of uncertainty through model comparison. Where both *FH* and *AEL* estimates exist, assuming a 50:50 sex ratio and aligning the ages of recruitment and maturation, the models can be compared directly using the relationship $AEL \equiv 2FH$ presented in previous assessment reports (Tenera 1999a, b). This conversion requires that ages of *AEL* and *FH* individuals are aligned and equal. The results from the demographic approaches are in similar units of either adult females (*FH*) or adults (*AEL*).

The *FH* and *AEL* model estimates for some of the target taxa were not in particularly close agreement (**Table 7-1b**). The age-aligned estimate for 2*FH* was over 4 times greater than the initial *AEL* estimate for Pacific sardine. In contrast, the *FH* estimates for sanddabs were low relative to *AEL*. When *FH* estimates are aligned for age the estimates remain low, relative to *AEL*. Sensitivity analyses were conducted to help determine the parameter estimates contributing the greatest influence on the *FH* and *AEL* estimates. The uncertainty associated with many of the parameters resulted in the use of a coefficient of variation of 100% to estimate their variance, in accordance with ETWG recommendations. This reduced the usefulness of the sensitivity analyses in interpreting model results.

Results for target taxa that were in close agreement with the relationship $2FH \equiv AEL$ (subject to age alignment) provide some assurance that the parameters used in the models were representative for the central California populations we sampled. They also increase confidence that the assessments of effects on these populations are reasonably accurate. Two examples were northern anchovy and the blue rockfish complex. In the blue rockfish complex, *FH* and *AEL* estimates were used to extrapolate population level effects using local fishery landings (**Section 5.2.4**). Using an estimate of the weight per adult blue rockfish in the fishery of 0.5 kg, an estimated *AEL* of 343 adults represented 172 kg of 3 yr old recruits, ca. 1% of the catch landed in the Morro Bay area in 1999, and valued at approximately $740. Blue rockfish P_{AI} ranged from 0.004–0.02, bracketing the *AEL*-based percentage of catch. It was also relatively linear over an

extrapolated area of 220–1,100 km^2 (smaller mortality associated with larger areas). Despite the limitations and assumptions of the models, the results of the assessment for the blue rockfish complex provide assurance that the effects on this taxon are low.

Model assumptions also need to be considered when interpreting the large *FH* estimate for brown rock crabs (**Table 7-1a**). The estimated number of reproductive adults using the *FH* model was extrapolated to known fishery-sized animals using estimates of adult mortality because females are reproductive before they enter the fishery. This reduced the *FH* estimate by an order of magnitude. Therefore, because of the additional assumptions required for the *FH* estimate, the estimate of P_{Af} was used in extrapolating fishery effects. If all larval stages in the population are equally entrainable, then the estimate of P_{Af} that was based on megalops would be representative of all larval stages. This assumption was made when $\widehat{P_{Af}}$ was applied to the catch.

For target taxa without available life history information, measurements of larvae from the entrainment and study grid provided some insight into their larval life history and ecology. The length ranges for most of the entrained target larval fish taxa indicate that their exposure to entrainment occurs over a relatively short time period during their development. The length-frequencies for several of the target taxa, including KGB and blue rockfish complex larvae, larval painted greenling, cabezon, and blackeye goby, appeared normally distributed. Average lengths were small demonstrating that they were exposed to entrainment for a brief period during their larval development. This supposition is supported by another characteristic common to these taxa; very few flexion or post-flexion larvae were entrained. Flexion and post-flexion stage larvae have more developed swimming abilities than younger individuals and could be avoiding the nets and potentially entrainment. The lack of these later developmental stages in our samples may also indicate that these taxa demonstrate larval behavior that removes them from risk of entrainment as they develop (e.g., settlement to benthic habitats or migration into deeper areas away from the intake). In contrast, the length-frequency distributions for the two target taxa with evidence of localized entrainment effects, clinid kelpfishes and snubnose sculpin, were skewed and included larger individuals than other taxa. Thus, it appears that these two taxa are exposed to entrainment over longer periods of development that include flexion and post-flexion stage larvae. An

extended duration of exposure to entrainment may partially explain the apparent declines in adult and larval abundance for these two taxa. Paradoxically, the extended exposure to entrainment that apparently results in localized effects on these taxa may also insure that localized effects do not become greater population-level effects. Extended planktonic duration will allow for larval transport and dispersal to occur over a much larger area.

7.2.4 Population Distribution

The assessment considered taxa that occurred in the entrainment samples at DCPP. Their abundance depended on temporal and spatial distribution of adults as well as larval life stages. The taxa can be described as pelagic (*Cancer* crabs, Pacific sardine, and northern anchovy), subtidal (rockfishes, painted greenling, snubnose sculpin, cabezon, white croaker, blackeye goby, sanddabs, and California halibut), and nearshore subtidal and intertidal (smoothhead sculpin, monkeyface prickleback, and clinid kelpfishes). These different spatial distributions contributed to the ETWG's decision to use two strategies in applying the *ETM*. Nearshore species population size was assessed by alongshore extension of study grid abundance only; otherwise, an offshore extrapolation of study grid abundance was conducted and then extended alongshore. These expansions of the study grid abundance were based on measures of current speed and direction at the DCPP current meter station and constrained by larval ages at entrainment.

The *ETM* approach applied to all target taxa required an estimate of a population that was defined by extrapolating larval density over an area delimited by estimated larval duration combined with current speed and direction in the study period. Estimates of total annual entrainment mortality ($\widehat{P_M}$) rely on the estimates of P_S (# of larvae at risk in the study grid/# of larvae in the population of inference). The results for target fish taxa in this 316(b) study indicate that while some entrainment effects remain localized, other effects can be spread over wide areas of inference (**Table 7-2**).

When estimated larval durations were relatively short and the subsequent area containing the population of inference remained relatively small, the extent of extrapolation was also small, based on the current meter data. Under these conditions, $\widehat{P_S}$ was likely a more precise estimate, potentially less susceptible to model bias, resulting in a more accurate estimate of entrainment

effects, $\widehat{P_M}$. There is less confidence in our estimates of $\widehat{P_S}$ where the extrapolated areas were substantially larger than the study grid. The variance component, resulting from density estimates projected offshore well beyond the zone that the measurements represented, was not included in our variance estimates of $\widehat{P_M}$ (**Appendix F**). Thus, the estimated reference population of fishes with short larval duration of exposure (e.g., cabezon and blackeye gobies) are potentially more precise than those with a long time of susceptibility to entrainment (e.g., crabs and Pacific sardine). However, the situation for nearshore species is more complex. For example, those with longer times of susceptibility to entrainment (durations) may disperse into the study grid area, becoming more available to plankton sampling gear, resulting in better estimates of $\widehat{P_M}$.

It is unlikely that areas on the order of hundreds to thousands of times greater than the study grid are representative of the actual area affected by entrainment. Therefore, when $\widehat{P_M}$ was applied to a stock assessment or fishery (e.g., for northern anchovy), it was based on a proportion of areas (area of extrapolation / stock assessment area). In some cases, this proportion was more approximate and, for brown rock crab, relied upon an estimate of the fishery grounds.

Target taxa that are not commercially or recreationally harvested are primarily small, nearshore fishes. As might be expected due to the shallow water, shoreline location of the DCPP intake structure, several of these taxa are entrained in relatively high numbers, resulting in large P_M estimates (**Table 7-2**). For example, adult kelpfish occur in rocky, nearshore habitat and also had one of the highest P_M estimates (ca. 30–40%). If we use only the alongshore estimate of P_S for calculating $\widehat{P_M}$ to compare results, those taxa with nearshore distributions, including smoothhead sculpin, snubnose sculpin, and monkeyface prickleback, had relatively large P_M estimates. Their greatest larval abundance occurred in the study grid cells closest to shore. This distribution is consistent with their adult distributions. Confidence in estimates of entrainment mortality for nearshore and intertidal fishes is supported when noting that $\widehat{P_M}$ for these taxa does not vary between extremes of larval ages at entrainment when extrapolating larval abundance alongshore.

Although entrainment sampling occurred at the shoreline intake where these nearshore taxa are in greatest abundance, the study grid sampling did not adequately sample other nearshore areas. Therefore, modifications were made to the calculations of *PE* to account for nearshore volumes

that were not sampled. While the addition of these inshore volumes may have overestimated $\widehat{P_M}$ for some of these taxa, estimates for blackeye goby and California halibut, species that had broader adult distributions, were also high (July 1998–June 1999 period only: **Table 7-2**). Blackeye goby larvae ($\widehat{P_M} \approx 20\%$) were distributed throughout the study grid and occur as adults from the intertidal out to recorded depths of over 100 m (Eschmeyer et al. 1983). The relatively high $\widehat{P_M}$ for blackeye goby may be due, in part, to the high density of adults directly in front of the intake structure. Entrainment of larvae hatched from demersal eggs within Intake Cove may explain our estimate of larval duration (6 d) that is considerably shorter than literature values (75 d). Therefore, while it is likely that some bias is introduced into the estimates of P_M due to the nearshore areas not sampled by the study grid surveys, the bias is probably species dependent and may not be related solely to adult distributions.

In contrast to species with nearshore distributions, estimates of P_M for species with more widespread, pelagic larvae (e.g., Pacific sardine and northern anchovy) were low, as expected by their long larval durations and large areas estimated by alongshore and offshore extrapolation. An assumption of the *ETM* is that larval distributions in the nearshore study grid are representative of densities in the area used for calculating P_S. While this assumption may be reasonable for values of P_S that represent relatively small areas of coastline, density in the grid is probably not representative of offshore areas used in calculating P_S for pelagic and subtidal species. As a result, applying $\widehat{P_M}$ to stock assessments or fisheries data may not accurately represent entrainment effects because density in the nearshore study grid is probably not representative of average density in the areas used for estimating standing stock biomass. However, the extrapolation of the larval density trends observed in the study grid out to waters further offshore was an attempt to account for this inaccuracy. When $\widehat{P_M}$ was applied to the fishery stock assessment by proportional area, in the case of northern anchovy, the biomass was similar to that determined using the two other methods of estimating adult losses, *AEL* and *FH*. The close correspondence in results for the different models suggests that the methodology was appropriate, and gives us confidence in the accuracy of the results.

7.2.5 Magnitude of Effects

Results from the present study indicate that effects on commercially and recreationally harvested species with subtidal or pelagic distributions are minimal. For cases where we were able to apply all three assessment approaches, the effects detected were relatively small, appeared to be localized, and thus could not affect the overall adult populations. These conclusions are supported by results for taxa such as KGB rockfishes where the estimated low levels of entrainment effects were corroborated by the corollary data that showed no consequent adult abundance or larval production decreases. While the alignment of results among assessment methods for other commercially or recreationally important taxa was less compelling, it nevertheless did not indicate the potential for any significant long-term population-level effects.

There was very little available information describing the demography of target taxa that were not commercially or recreationally important and this limited the applicable assessment approaches to the *ETM*. These were typically taxa distributed nearshore as adults, rendering their larvae highly susceptible to entrainment. In fact, these taxa were included in the targeted list by the ETWG because of their high abundance in entrainment samples. The *ETM* approach showed that entrainment could remove 10–40% of the annual larval production of several of these nearshore and intertidal taxa. In some of these cases, corollary data indicate that entrainment is probably only removing surplus production since there are no apparent long-term declines in local adult populations. For those nearshore taxa that do not display decreasing adult abundance, it is apparent from their length-frequency distributions that they were exposed to entrainment mortality for a brief portion of their early development. In contrast, for kelpfishes and snubnose sculpin, whose adult abundance do appear to be decreasing (with corresponding declines in long-term plankton tow index), the length-frequency distributions indicate a longer duration of exposure to entrainment. The combination of length-frequency analyses with the *ETM* estimates offer support to the conclusion that some nearshore taxa are affected differentially by entrainment. The increased duration of exposure to entrainment for kelpfishes and snubnose sculpin may also insure that these effects remain localized since extended planktonic duration provides additional opportunity for larval transport and dispersal.

7.2 Summary Assessment

In summary, populations of fishes and crabs do not appear to be adversely affected by the DCPP cooling water intake structure. Some are commercially important taxa with pelagic eggs and widespread populations (e.g., northern anchovy, Pacific sardine, white croaker, sanddabs, and California halibut). Their assessments resulted in either low estimated larval mortalities or small numbers of adult losses to their populations. Other widespread species also had low numbers of estimated adult equivalent losses to their populations and low estimated larval mortality, with populations that are distributed well beyond the zone of influence of DCPP. These included commercial species (rockfishes, crab, and cabezon). Five taxa (smoothhead sculpin, snubnose sculpin, monkeyface prickleback, clinid kelpfishes, and blackeye goby) had larval mortalities that exceeded 10% in reference to that part of the population living in an area 2–8 times the study grid area. For most of these (except kelpfishes and snubnose sculpin), subtidal observations of adults and long-term plankton sampling of larvae showed no decline in numbers over recent years in the vicinity of DCPP. Effects on kelpfishes and snubnose sculpin appear to be localized and the area affected is a small part of their total geographic distribution.

The models used for entrainment assessment considered functions critical to the life history of the 16 target taxa. These models were applied both at the point of entrainment for estimating the numbers of individuals entrained and also in the adjacent offshore areas for estimating the population of inference. We also considered migratory, pelagic species and other widespread taxa differently than nearshore taxa in estimating the population of inference. The area around DCPP encompasses nursery and feeding areas for many species including the 16 target taxa. These areas also extend well outside the DCPP zone of influence and in the cases of northern anchovy and Pacific sardine the center of the spawning biomass is located south of Point Conception. Measurements of larvae indicate that most of the target taxa are exposed to entrainment for a relatively short period of time during their larval development and thus were produced locally. These results indicate that entrainment effects appear to be limited to localized effects on nearshore species. Therefore, the potential for damage due to entrainment on the biological value of the source water body is low.

7.3 Best Technology Available

Although no adverse environmental impact was determined, a generic review of alternative cooling water technologies applicable to DCPP was evaluated in **Section 6**. A hierarchical set of three criteria were applied to the alternatives that were considered:

A review of alternative cooling water technologies applicable to DCPP was evaluated in **Section 6,** as required by the RWQCB. A hierarchical set of three criteria were applied to the alternatives and considered:

- available technology with demonstrated operability and reliability at intakes similar to DCPP;
- a reduction in the loss of aquatic organisms; and
- applicability at DCPP, based on engineering, operations, and reliability considerations.

The technology assessments examined modifications that may affect impingement and entrainment although impingement rates are very low at DCPP. Site-specific studies determined impingement rates were low ranging between 0.3 and 0.7 kg per day per unit, which is equivalent to 360 kg/yr for both units. Compared to other west coast generating stations (Tenera 1998a) that impinge between 2,800 to 24,100 kg/yr, impingement at DCPP does not represent an adverse environmental impact. Consultation with industry authorities and published engineering evaluations of demonstrated technologies applicable to reducing entrainment, revealed few intake modifications that might reduce the density of fish larvae at risk to entrainment in a cost effective manner. Although several facilities have reduced entrainment by modifying intake screen systems, the harsh coastal environment at DCPP precludes application of these strategies without extensive site-specific testing. Reduction of operation to periods when fish densities are lower was considered not viable in the current power exchange pricing environment. Replacement of the heat exchange processes from open-cycle seawater cooling systems to closed-cycle cooling tower strategies was determined to be economically unfeasible and added substantial environmental risk to terrestrial communities.

As discussed in the other sections of this report, DCPP is not causing an adverse environmental impact (AEI) and therefore no consideration of alternative cooling water intake structure

technologies for minimizing AEI is necessary. However, this assessment of alternative technologies is included in response to a requirement by the RWQCB. The conclusion based on this assessment is that the existing cooling water intake structure is the best technology available for DCPP.

Table 7-1a. Estimates of fecundity hindcast adult females ($\widehat{FH}$) and projected equivalent adults ($\widehat{AEL}$) in numbers of individuals for Analysis Periods 1 (December 1996–November 1997) and 3 (July 1997–June 1998) for the two target crab taxa collected and analyzed at DCPP with their potential relative value.

Taxon	Analysis Period	$\widehat{FH}$	$\widehat{AEL}$	Interpreted Annual Value (applied modeling approach)		
Brown rock crab	1.	117,000 (11,700) [a]	234,000 (23,400) [b]	$1,090 [c] ($\widehat{FH}$)	$2,170 [c] ($\widehat{AEL}$)	$1,760 (ETM)
	3.	91,000 (9,100) [a]	182,000 (18,200) [b]	$845 [c] ($\widehat{FH}$)	$1,690 [c] ($\widehat{AEL}$)	$2,040 (ETM)
Slender crab	1.	27,300 (2,730) [a]	54,600 (5,460) [b]	no fishery value		
	3.	8,950 (895) [a]	17,900 (1,790) [b]	no fishery value		

[a] Estimate extrapolated to fishery-sized female crabs.

[b] Estimate extrapolated to fishery-sized crabs, both male and female.

[c] Value based on assumption of 10% vulnerability to fishery (from mark-recapture experiments and fishery recapture data: see Section 5.2.1.1).

Table 7-1b. Estimates of adjusted annual entrainment ($\hat{E}_{Adj-T}$), fecundity hindcast adult females ($\widehat{FH}$), and projected equivalent adults ($\widehat{AEL}$) in numbers of individuals for Analysis Periods 1 (October 1996–September 1997), 2 (October 1997–September 1998), and 3 (July 1997–June 1998) for the 14 target fish taxa collected and analyzed at DCPP with their potential relative value. When more than one valuation procedure was used, the largest value is listed. The total number of target taxa impinged during April 1985–March 1986 are included for comparison.

Taxon	Analysis Period	$\hat{E}_{Adj-T}$	$\widehat{FH}$	$\widehat{AEL}$	Total Impinged 1985-86	Interpreted Annual Value (applicable valuation method)
Pacific sardine	1.	8.470.000	3.170	2.630	0	$710 (*ETM* modeling applied to reduction and live bait fisheries)
	2.	22.600.000	8.460	7.000		
	3.	22.600.000	8.460	7.000		
Northern anchovy	1.	136.000.000	16.100	43.200	1	$576 (*AEL and ETM* modeling applied to non-reduction wetfish fishery)
	2.	376.000.000	44.700	120.000		
	3.	377.000.000	44.700	120.000		
KGB rockfish complex	1.	275.000.000	617	1.120	12	$20.000 (*ETM* modeling applied to commercial live-fish fishery)
	2.	222.000.000	497	905		
	3.	222.000.000	497	906		
Blue rockfish complex	1.	84.040.000	43	353	8	$1.230 (*ETM* modeling applied to commercial fishery)
	2.	33.800.000	18	164		
	3.	33.900.000	20	142		
Painted greenling	1.	24.200.000	—	—	1	forage species (no economic valuation)
	2.	9.610.000				
	3.	12.100.000				
Smoothhead sculpin	1.	57.700.000	—	—	10	forage species (no economic valuation)
	2.	115.000.000				
	3.	129.000.000				
Snubnose sculpin	1.	110.000.000	—	—	9	forage species (no economic valuation)
	2.	83.500.000				
	3.	105.000.000				
Cabezon	1.	51.900.000	—	—	1	$9.000 (*ETM* modeling applied to commercial live-fish fishery)
	2.	36.300.000				
	3.	36.300.000				
White croaker	1.	305.000.000	5.110	14.700†	0	$872 (*AEL* modeling applied to commercial fishery:)
	2.	440.000.000	7.380	21.300†		
	3.	447.000.000	7.500	21.600†		
Monkeyface prickleback	1.	83.100.000	—	—	0	no demographic or economic data were available for this species to estimate adult losses or fishery valuation
	2.	61.500.000				
	3.	60.200.000				
Clinid kelpfishes	1.	181.000.000	—	—	17	forage species (no economic valuation)
	2.	308.000.000				
	3.	458.000.000				
Blackeye goby	1.	128.000.000	12.000	75.200	0	forage species (no economic valuation)
	2.	109.000.000	10.300	64.100		
	3.	128.000.000	12.100	75.400		
Sanddabs*	1.	7.160.000	426	2.370	4	$82 (*ETM* modeling applied to commercial fishery)
	2.	1.540.000	92	511		
	3.	6.610.000	393	2.190		
California halibut*	1.	8.260.000	—	—	0	$10.060 (*ETM* modeling applied to commercial fishery)
	2.	15.700.000				
	3.	15.500.000				

— = not estimated
* = entrainment estimates were not adjusted to the long-term (1990–98) Intake Cove plankton tow mean
† = 2*FH* aligned to fishery-age adult equivalents

Table 7-2a. Total annual entrainment mortality ($\widehat{P_M}$) and zone of extrapolation ($1/\widehat{P_S}$) as a multiple of study grid areas (using alongshore currents) or of study grid population (using alongshore and offshore extrapolation) for larvae of the two target crab taxa from Analysis Periods 3 (July 1997–June 1998) and 4 (July 1998–June 1999) estimated by using 2 methods of calculating $\widehat{P_S}$ using maximum larval duration estimates through the megalops stage at DCPP and water current measurements.

		$\widehat{P_S}$ calculated using alongshore currents		$\widehat{P_S}$ calculated using alongshore currents and extrapolating densities offshore	
Taxon	**Analysis Period**	$\widehat{P_M}$	$1/\widehat{P_S}$	$\widehat{P_M}$	$1/\widehat{P_S}$
brown rock crab	3.	0.00563	8.3	0.0000186	2.590.7
	4.	0.00652	9.3	0.000146	417.6
slender crab	3.	0.0844	8.1	0.0107	236.9
	4.	0.0113	7.9	0.000784	114.2

Table 7-2b. Total annual entrainment mortality ($\widehat{P_M}$) and zone of extrapolation ($1/\widehat{P_S}$) as a multiple of study grid areas (using alongshore currents) or of study grid population (using alongshore and offshore extrapolation) for larvae of the 14 target fish taxa from Analysis Periods 3 (July 1997–June 1998) and 4 (July 1998–June 1999) estimated by using 2 methods of calculating $\widehat{P_S}$ using both mean and maximum larval duration estimates derived from lengths of entrained larvae at DCPP and water current measurements.

		$\widehat{P_S}$ calculated using alongshore currents				$\widehat{P_S}$ calculated using alongshore currents and extrapolating densities offshore			
		based on mean duration		based on max. duration		based on mean duration		based on max. duration	
Taxon	**Analysis Period**	$\widehat{P_M}$	$1/\widehat{P_S}$	$\widehat{P_M}$	$1/\widehat{P_S}$	$\widehat{P_M}$	$1/\widehat{P_S}$	$\widehat{P_M}$	$1/\widehat{P_S}$
Pacific	3.	–	–	–	–	0.000284	47.3	0.0000669	1.078.1
sardine	4.	–	–	–	–	*		*	
northern	3.	–	–	–	–	0.000575	16.5	0.0000786	683.8
anchovy	4.	–	–	–	–	0.00205	7.9	0.000201	454.5
KGB	3.	0.0365	2.8	0.0532	5.1	0.0146	7.2	0.00962	29.5
rockfishes	4.	0.0380	2.5	0.0376	6.7	0.0218	4.4	0.00476	53.9
blue rockfish	3.	0.00338	2.6	0.00468	4.1	0.000955	9.3	0.000456	42.1
complex	4.	0.0471	2.0	0.0411	5.0	0.0211	4.6	0.00356	60.0
painted	3.	0.0490	3.6	0.0671	6.2	0.00877	20.7	0.00910	47.7
greenling	4.	0.0319	3.6	0.0381	7.3	0.0114	10.3	0.00444	63.9
smoothhead	3.	0.105	2.8	0.153	7.3	–	–	–	–
sculpin	4.	0.146	3.0	0.198	8.2	–	–	–	–
snubnose	3.	0.102	2.0	0.144	4.3	0.0361	7.0	0.0231	39.3
sculpin	4.	0.202	1.5	0.239	4.2	0.1206	2.6	0.0210	55.8
cabezon	3.	0.0243	2.5	0.0337	3.5	0.00683	9.1	0.00569	21.8
	4.	0.0140	1.4	0.0161	2.5	0.00835	2.4	0.00919	4.4
white	3.	0.0190	2.8	0.0297	5.5	0.00255	21.2	0.00133	127.5
croaker	4.	0.0366	1.9	0.0550	3.9	0.0211	3.4	0.00704	32.3
monkeyface	3.	0.162	3.0	0.232	6.9	–	–	–	–
prickleback	4.	0.108	2.4	0.113	8.0	–	–	–	–
clinid	3.	0.318	3.1	0.410	7.3	–	–	–	–
kelpfishes	4.	0.294	2.7	0.395	6.2	–	–	–	–
blackeye	3.	0.190	1.3	0.228	2.4	0.131	2.0	0.0787	8.6
goby	4.	0.174	1.2	0.220	1.8	0.163	1.3	0.179	2.5
sanddabs	3.	0.0145	2.4	0.0157	3.2	0.00493	11.7	0.00413	22.4
	4.	0.0663	1.8	0.0537	3.5	0.0459	2.7	0.0106	18.5
California	3.	0.00269	2.8	0.00449	6.1	0.000835	8.9	0.000763	35.9
halibut	4.	0.175	2.3	0.252	5.4	0.123	3.5	0.0525	32.8

'–'no calculations of this class made for this taxon

* No Pacific sardine larvae were collected in paired entrainment and study grid samples for this period.

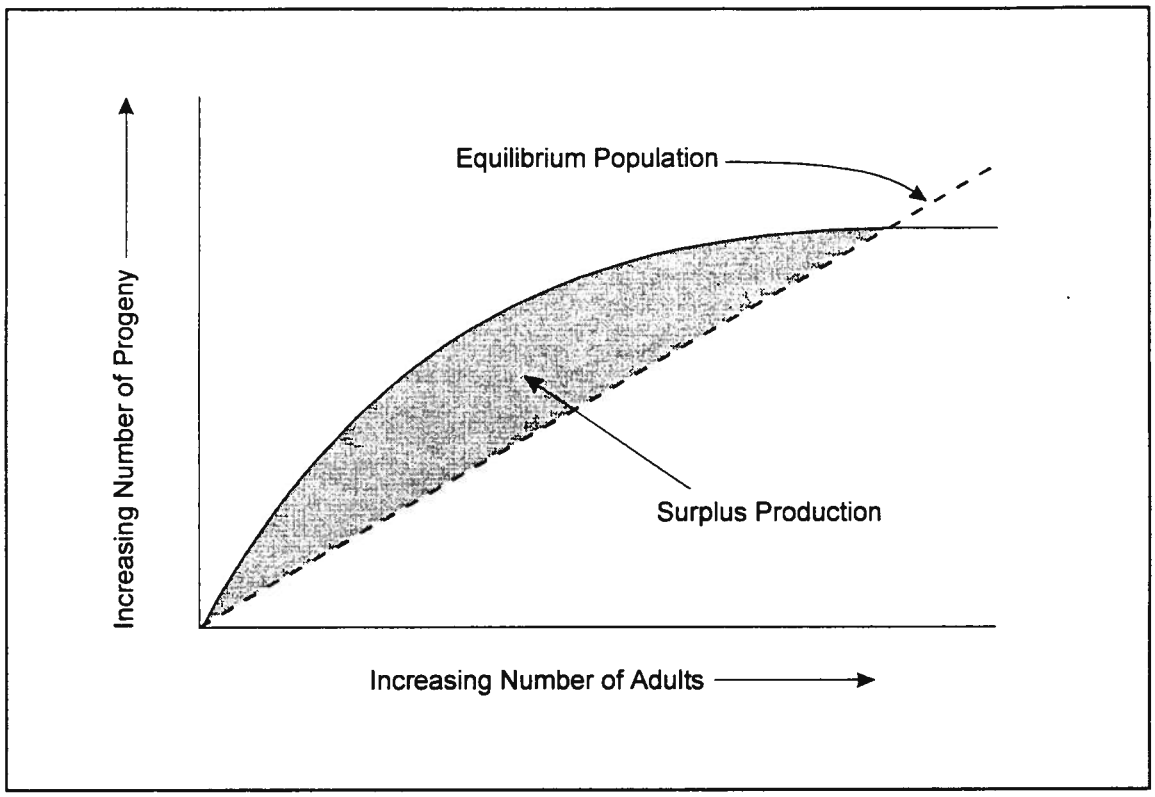

Figure 7-1. Surplus production model.

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8.0 Conclusion

Section 316(b) of the Federal Water Pollution Control Act Amendments of 1972 (Clean Water Act) requires that the Diablo Canyon Power Plant (DCPP) cooling water intake structure represents the best technology available (BTA) to minimize adverse environmental impacts (AEI). As discussed in the Introduction of this report, the United States Environmental Protection Agency (USEPA) has indicated that assessment of AEI should be based on an evaluation of population-level effects. The results of the assessments on the 16 target taxa selected for this DCPP 316(b) study by the Entrainment Technical Work Group (ETWG) show no population level effects due to entrainment and impingement. Consequently, we conclude that the DCPP intake structure is not causing AEI.

no!

Since DCPP is not causing population-level effects, no consideration of alternative cooling water intake structure technologies for minimizing AEI is necessary. However, this report does include an assessment of alternative technologies in response to a requirement by the Central Coast RWQCB. The conclusion of this alternative technologies assessment is that the existing cooling water intake structure is the best technology available for DCPP.

Summary of Effects

- Populations of the five target taxa with pelagic eggs (Pacific sardine, northern anchovy, white croaker, sanddabs, and California halibut) are not adversely impacted by the DCPP intake indicated by generally low to moderate estimates of $\widehat{P_{AI}}$, $\widehat{FH}$, or $\widehat{AEL}$. Furthermore, these taxa are protected against localized effects by having widespread populations or spawning centers distant from DCPP. In the case of California halibut, with a relatively high $\widehat{P_{AI}}$, but low approximated $\widehat{FH}$, very few individuals are entrained at DCPP so there is no adverse impact on their population.

but what about cumulative effects of multiple plants?

- Both groups of rockfishes (KGB and blue rockfish complexes) had low numbers of estimated adult equivalent losses to their populations. Cabezon had low estimated larval mortality (<1%), based on extrapolations using the alongshore and onshore currents. This study also estimated that the larval mortality of painted greenling was generally below 1%, using similar extrapolations. In addition, brown rock crab and

slender crab had low estimated larval mortality. All of these species have populations that are distributed well beyond the zone of influence of DCPP.

- The remaining taxa (smoothhead sculpin, snubnose sculpin, monkeyface prickleback, clinid kelpfishes, and blackeye goby) had larval mortalities that exceeded 10% in reference to that part of the population living in an area 2–8 times the study grid area. However, the area affected is only part of these species' geographic range. Therefore, these effects appear to localized and do not constitute population-level effects.

- Of the five taxa with relatively high estimated larval mortalities, only two had corroborating evidence that local adult and juvenile abundance could be affected by the DCPP cooling water intake system: clinid kelpfishes and snubnose sculpin. Both subtidal observations of adults and long term plankton sampling of larvae showed declining numbers over recent years in the vicinity of DCPP.

It is important to note that the corroborating evidence noted above for clinid kelpfishes and snubnose sculpin is based on limited sampling at one local subtidal sampling transect four times per year (as visibility permits) and weekly plankton tows at the mouth of the Intake Cove. Additionally, there is some evidence in southern California (Holbrook et al. 1996) that the effects of a regime shift in the California Current towards warmer water temperatures combined with successive El Niño events have continued to cause general population declines in more northerly distributed species (e.g., rockfishes). It should also be noted that the *ETM* methodology was originally developed and adapted for species with larvae dispersed throughout the study grid area and not for nearshore species. Clinid kelpfishes and snubnose sculpin are nearshore fishes and thus were undersampled by the study grid methodology. Both kelpfishes and snubnose sculpin (as well as the other nearshore species) are found in these inshore areas, and thus their larval mortality estimates could be overestimated.

The effects of the cooling water intake structure estimated by the *AEL* and *FH* models did not include the very small losses due to impingement. When added to the *AEL* and *FH* estimates, these losses do not change the overall conclusion that entrainment effects of the plant's cooling water intake system do not constitute AEI for populations of the 16 target taxa for which assessments were made.

Using criteria supplied by the USEPA (1977) for determining the extent of AEI and supplemented by criteria specific to the present 316(b) study at DCPP, estimates of entrainment effects on the populations of the 16 target taxa presented in this report do not constitute AEI. For two of the nearshore taxa assessed (clinid kelpfishes and snubnose sculpin) there is some evidence of local decreases in both larval and adult abundance which may be attributable to entrainment of early life stages in the DCPP cooling water intake system. However, climatological and oceanographic conditions (e.g., regime shifts, El Niño events) may have contributed to these local declines to an unknown extent and should be considered.

Alternative Technologies

Alternatives to the current cooling water system evaluated for DCPP included seasonal refueling outages; reduction in annual operation; modifications to the traveling screens; sluiceways and their operation; expansion of the existing shoreline intake structure and installation of modified screening; and reductions in cooling water flow rates. In addition, consideration was given to transferring the ocean heat sink, open-cycle cooling process to land-based closed-cycle cooling towers. Each alternative is expected to offer some potential for reducing the losses of organisms resulting from entrainment or impingement.

The judgment of the best technology available for the Diablo Canyon Power Plant is based on a consideration of the level of impact caused by entrainment and impingement on the marine community, demonstrated operation and reliability of various alternative technologies, and the biological effectiveness of these technologies for further reducing entrainment and impingement, engineering and operational feasibility, and cost effectiveness of the alternative technologies.

Based on the evaluation of alternative intake technologies for DCPP it was concluded that:

- There is no reasonable alternative intake location that would reduce entrainment and impingement losses.
- Behavioral barriers would not reduce the numbers of fish exposed to impingement or entrainment.
- Media filters, barrier nets, and Gunderboom were determined to be not applicable at the DCPP site due to the large flow volumes, high debris loading, and potential for rough ocean conditions.

- Inclined screens, drum screens, and centerflow screens with fine mesh, would not significantly reduce impingement losses and have not been proven to substantially reduce entrainment losses.
- Modifications to the intake screens and the addition of a fish return system could reduce the losses due to impingement. Because of the low numbers of fish impinged on existing intake screens and the relatively high costs involved, installation of modified intake screens is disproportionate to the benefit.
- Expansion of the intake structure to achieve a lower approach velocity or to provide an angled screen intake could reduce impingement. The cost of modification, however, is disproportionate to the benefits of reduced impingement.
- Variable speed pump drives could be provided to allow reductions in circulating water flow to minimums required for plant operation. The design margins of the condenser system at DCPP could allow a 2 to 10 % reduction in circulating flow when ambient seawater temperatures are below 14.4 °C without exceeding discharge temperature limits. The high cost, uncertain reliability, and low biological benefit make this option impractical.
- Seasonal patterns were detected for entrained larval fishes and some benefit has occurred when cooling water pumps are shut down for refueling and maintenance outages in spring and fall. Fewer larval fishes would be susceptible to entrainment if all outages were scheduled for the spring. However, scheduling outages to coincide with the seasonal distribution in abundance of a particular fish or group of fishes is not economically feasible, due to the characteristics of the nuclear fuel.
- Consideration was given to curtailing power plant operation during spring months when density of fish larvae was highest. In the current pricing structure practice for electrical production, the losses associated with nine month operation windows make this scenario not economically feasible. Replacement power from gas-fired facilities would also have a negative contribution to air quality because of the release of emissions that are not generated by nuclear facilities.
- Closed-cycle cooling towers were evaluated to reduce entrainment and impingement losses. Cooling towers using saltwater would impact the land and biota with their

large "foot-print," corrosive fogs, vapor plumes, and salt drift. There is no resource large enough to provide freshwater makeup for cooling towers, and the addition of a seawater desalination plant of sufficient capacity would add considerably to project costs. The costs to retrofit closed-loop cooling towers to DCPP, which would require substantial construction, refitting, and redesign of the plant, would not be viable in the current or future power generation market.

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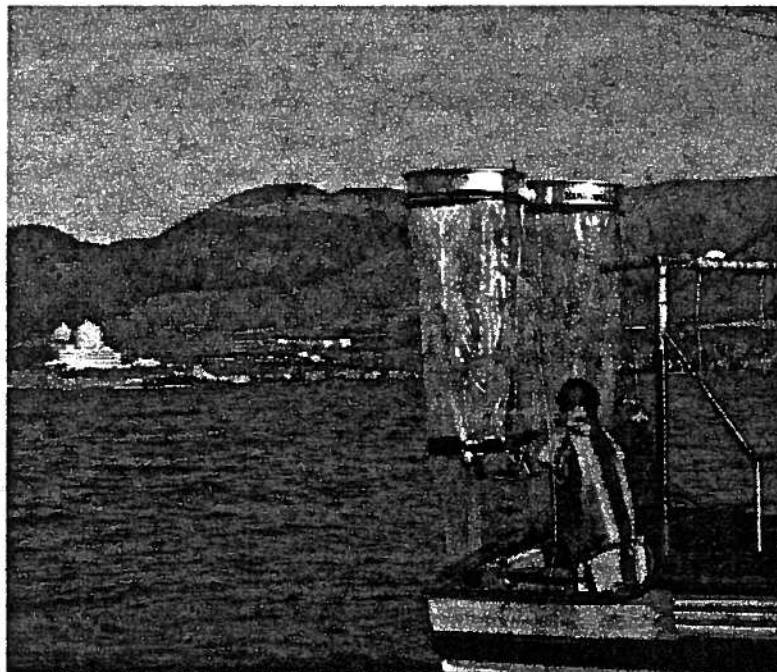
Diablo Canyon Power Plant

316(b) Demonstration Report

APPENDICES



Plankton sampling offshore of DCPP

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APPENDICES