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PITTSBURG POWER PLANT
COOLING WATER INTAKE STRUCTURES
316(b) DEMONSTRATION

Prepared for

Pacific Gas and Electric Company
77 Beale Street
San Francisco, California 94106

Prepared by

Ecological Analysts, Inc.
2150 John Glenn Drive
Concord, California 94520

EA Report PGE60H1

August 1981

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PACIFIC GAS and ELECTRIC COMPANY

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EXECUTIVE SUMMARY

Introduction

This report documents the results of a study program conducted at Pacific Gas and Electric Company's Pittsburg Power Plant during 1978 and 1979, in accordance with Section 316(b) of the Clean Water Act, and Order 76-63 of the California Regional Water Quality Control Board, San Francisco Bay Region (Regional Board). The study program was designed to provide a sufficient basis for regulatory agencies to determine whether the existing cooling water intake structures or modifications of those structures reflect the best technology available for minimizing adverse environmental impact.

A key element in the 316(b) program was to determine the extent of the entrainment and impingement of aquatic organisms at the Pittsburg Power Plant and the associated environmental impact. Entrainment and impingement are the two major ways in which aquatic organisms are directly involved with the cooling water intake of a power plant. As cooling water is drawn through the plant, small organisms (those that can pass through the 3/8-in. mesh of the plant's intake screens) are carried along, or entrained, and subjected to the stresses of passage through the pumps and condensers. Organisms that are too large to pass through the mesh may be held against the screen by the water, or impinged, until they are removed by the moving screens and washed into a conveyance system for eventual return to the water source.

The numbers of aquatic organisms entrained and impinged were sampled weekly, and the potential survival of entrained and impinged organisms was examined. On the basis of these studies, the total annual cropping of organisms (the number not surviving entrainment or impingement) was estimated. These estimates were used, along with 20 years' worth of data about aquatic organisms and the water body, gathered by the California Department of Fish and Game, the State Department of Water Resources, and other investigators, to assess the impact of the intake systems of the Pittsburg Power Plant on the aquatic populations of the Sacramento - San Joaquin estuary. The assessed impact was considered with engineering and operating criteria to determine the best intake technology available for the Pittsburg Power Plant.

S.1 THE PITTSBURG POWER PLANT

The Pittsburg Power Plant is located on the south shore of eastern Suisun Bay just below the confluence of the Sacramento and San Joaquin rivers, adjacent to the city of Pittsburg, California, approximately 50 miles (80 km) east of San Francisco. The plant is situated on the large estuary formed by the delta of the Sacramento and San Joaquin rivers and San Francisco Bay (the Bay-Delta).

The plant comprises seven electrical generating units with a combined gross capacity of 2,060 MWe. Units 1-4, the oldest units, started commercial service in 1954, Units 5 and 6 were put into service in 1960 and 1961, respectively, and Unit 7 was put into service in 1972. All units except Unit 7 use once-through cooling, in which water is pumped through the plant to remove heat from the condensers and is then returned to the water body. Unit 7 is equipped with a closed-cycle cooling system, in which water is cooled in mechanical-draft cooling towers to dissipate heat from the condensers.

The Units 1-4 cooling water system includes a shoreline intake structure with bar racks to keep out large debris and traveling screens to keep out smaller debris and organisms. Each unit has two circulating water pumps, each supplying one-half of the unit's steam condenser through a separate pressure conduit. In typical operation, each unit's two main circulating water pumps are run simultaneously ("continuous normal" operation), resulting in a total cooling water flow rate of 394,000 gpm for the four units.

The cooling water system serving Units 5 and 6 includes a shoreline intake structure, adjacent to the Units 1-4 intake structure, consisting of bar racks, traveling screens, and circulating water pumps. Separate intake conduits conduct cooling water to the Unit 5 and Unit 6 condenser halves. The combined cooling water flow of Units 5 and 6 in normal operation is 321,000 gpm.

The makeup cooling water for Unit 7 needed to replace the water that is consumed by evaporation, or returned to the bay as blowdown during cooling tower operation, is obtained from behind the Units 1-4 intake structure with three 10,100 gpm pumps. In normal operation, two of the pumps are run simultaneously and furnish 20,200 gpm of water to the unit's cooling system.

S.2 ASSESSMENT OF ENTRAINMENT AND IMPINGEMENT EFFECTS

Criteria presented in the draft Federal 316(b) guidelines were used in this assessment of potential impact resulting from entrainment and impingement at the Pittsburg Power Plant. Environmental impact is considered to be adverse when the ecological function of the organisms of concern is impaired to a level which:

1. precludes maintenance of existing populations;
2. results in a reduction in optimum sustained yield to sport and/or commercial fisheries;
3. involves threatened or endangered species directly or indirectly; or
4. constitutes an unmitigable loss to the aquatic system.

The combined impact of entrainment and impingement at the Pittsburg Power Plant on selected fishes and invertebrates was considered in the impact assessment. The impact on the Bay-Delta populations of most species of fish and invertebrates was judged to be low: for almost all of these species, either the population was not declining or its involvement with the Pittsburg Power Plant was too small to make the plant a plausible agent of the decline. No threatened or endangered species are affected by operation of the plant. The information available indicates that the numbers of organisms entrained and impinged at the power plant will not result in an adverse environmental impact.

The potential impact of entrainment on the mysid shrimp population (*Neomysis mercedis*) was examined in detail, because the number entrained was relatively high and because of their importance in the Bay-Delta food web. Partly because of relatively high entrainment survival rates, the average daily proportion of the mysid population (2-17 mm in length) cropped as a result of entrainment between 1974 and 1979 was relatively small. It amounted to only 0.6 percent of the estimated total daily mortality rate for the mysid population in that period. It was concluded that entrainment mortality at the Pittsburg Power Plant does not adversely impact the Bay-Delta mysid population, and the observed level of entrainment cropping does not preclude maintenance of the existing population.

A more extensive impact assessment was required for striped bass, because the early life stages are entrained and impinged in relatively high numbers, because of their

recreational importance in the Bay-Delta sport fishery, and because the Bay-Delta striped bass population has experienced a general decline in abundance.

From a detailed examination of the effect of operation of the Pittsburg Power Plant cooling water systems on the numbers of newly produced striped bass each year (young-of-the-year), it was concluded that although entrainment and impingement mortality reduced the abundance of young-of-the-year striped bass, plant-induced cropping rates were low in comparison to the total mortality rates of these larval and juvenile striped bass in the Bay-Delta. Furthermore, when the effect of entrainment cropping at the Pittsburg Power Plant on the number of striped bass recruited into the sport fishery (those entering the fishery at age 3) was examined, there was no correlation between recruitment and estimates of percentage reduction due to entrainment. ("Percentage reduction" is the estimated proportion by which the number of fish produced in a given year is smaller than it otherwise would have been.)

The numbers of juvenile and subadult striped bass impinged in the study year were estimated to be the equivalent of about 16,000 3-year-old fish. Since the number of 3-year-old striped bass recruited into the Bay-Delta sport fishery typically ranges from about 800,000 to 1.3 million individuals per year, the estimated loss of 16,000 individuals does not represent a significant contribution to the recruitment potential of the population.

It was estimated that the overall average percentage reduction of the young-of-the-year striped bass population attributable to entrainment was 8 percent during the period 1961-1979. The impact of the plant on striped bass due to entrainment was found to depend, in part, on Delta outflow. Percentage reduction estimates ranged from a high of approximately 18 percent in drought years such as 1976 and 1977, when average May outflow was less than 5,000 cfs, to only 4 percent in wetter years, when average May outflow was more than 15,000 cfs.

An evaluation of the factors affecting the numbers of young striped bass produced each year led to the conclusion that such factors as toxicants, predation, entrainment and impingement, diversion losses, disease and parasitism are not the major factors in determining year class strength (the relative number of striped bass produced in a given year). Rather, it was concluded that striped bass year class strength is determined to a large degree by the carrying capacity (the number of individuals the habitat can support) of the Bay-Delta. Direct losses like those resulting

from power plant cooling system operation represent only a small contribution to the overall mortality rate. The statistical evidence and available biological information both support the hypothesis that year class strength is largely a function of freshwater outflow, since this determines the spatial and temporal distribution of larval striped bass with respect to the food supply needed for survival and growth.

There was no evidence in the available population data to suggest that a reduction in the historical levels of power plant mortality would directly increase the abundance of the adult striped bass stock. These conclusions are consistent with results of the analysis and discussion of factors affecting the year class strength of striped bass—that entrainment and impingement at the Pittsburg Power Plant contribute very little to the regulation and dynamics of the Bay-Delta striped bass population.

For these and other reasons explained in more detail in Chapter 5, it was concluded that the impact of the Pittsburg Power Plant should be regarded as low and that it is unlikely that the implementation of alternative intake technologies designed to further reduce entrainment or impingement mortality will result in a detectable increase in the abundance of those fish and invertebrates that inhabit the Bay-Delta.

S.3 INTAKE TECHNOLOGY EVALUATION

An evaluation system was developed to assess which alternative intake technologies would reduce biological losses and be feasible for application to the cooling water system of the Pittsburg Power Plant. Alternative intake technologies were evaluated on the basis of the following four criteria:

1. The alternative technology is available and proven (i.e., it has demonstrated operability and reliability at a cooling water intake having a similar size and environmental setting to that at the Pittsburg Power Plant site).
2. Implementation of the alternative technology will result in a reduction in the loss of aquatic organisms from the existing operating conditions.
3. Implementation of the alternative technology is feasible at the Pittsburg Power Plant, based on site-specific considerations of engineering, operation, and reliability.
4. The cost of the alternative technology is proportionate to the environmental benefits anticipated.

All intake technologies that were considered to be available and proven for application at the plant (Criterion 1) were given a biological evaluation (Criterion 2). Feasibility analyses (Criteria 3 and 4) were carried out for alternatives that would reduce biological losses. Conclusions were then made regarding the best intake technology available for the Units 1-7 cooling water systems at the Pittsburg Power Plant.

It is concluded that the existing cooling water systems and shoreline intakes reflect best technology available for minimizing adverse environmental impact, provided that present operating practices of (1) maintenance dredging to keep intake velocities low, (2) intermittent chlorination at low levels, and (3) reducing the operation of the circulating water pumps when a unit is out of service for an extended period of time, are continued.

Chapter 1

Introduction

1.1 Plant Location and Description

1.2 Description of 316(b) Studies

1.3 Organization of the Report

1.4 References

CHAPTER 1: INTRODUCTION

The following report documents the results of a demonstration program conducted at Pacific Gas and Electric Company's Pittsburg Power Plant in compliance with Section 316(b) of the Clean Water Act (Act). This report is submitted in accordance with the provisions of Order No. 76-63 of the California Regional Water Quality Control Board, San Francisco Bay Region (Regional Board), which granted National Pollutant Discharge Elimination System Permit No. CA 0004880 for the Pittsburg Power Plant.

Section 316(b) of the Act requires that "...the location, design, construction, and capacity of cooling water intake structures reflect the best technology available (BTA) for minimizing adverse environmental impact." Because no single intake design can be considered to be the "best technology available" for all sites, it is necessary to conduct a site-specific analysis of intake-related environmental impacts and a site-specific determination of the best technology available for minimizing those impacts. In this report, intake-related environmental impacts resulting from entrainment (the drawing of organisms into the cooling water system) and impingement (the retention of organisms on the intake screens) are discussed; intake technologies are evaluated according to operating, engineering, and biological criteria; and the "best technology available" for minimizing entrainment and impingement impacts is recommended for the cooling water intake structure of the Pittsburg Power Plant.

A companion report, a 316(a) demonstration for the Pittsburg Power Plant (Tetra Tech 1976) provides additional documentation and analysis of the effects of the cooling water systems of Units 1-6 on the ecological community in the receiving waters. In that report, an impact assessment is made of the entrainment mortality and the effects of the thermal plume on the aquatic community in the receiving waters. Information presented in the 316(a) demonstration has been integrated with the data and analyses developed in the 316(b) studies whenever appropriate, to help provide a broader basis for understanding the relationship between operation of the Pittsburg Power Plant cooling system and the aquatic community in the receiving waters.

1.1 PLANT LOCATION AND DESCRIPTION

The Pittsburg Power Plant is located on the south shore of eastern Suisun Bay just

below the confluence of the Sacramento and San Joaquin rivers (Figure 2-1). The plant site borders the city of Pittsburg, California, and is approximately 50 miles (80 km) east of San Francisco. Near the plant site, Suisun Bay is approximately 4,000 ft (1,220 m) wide and has a maximum depth of 70 ft (21 m) at mean lower low water (MLLW). The shoreline in the vicinity of the plant has been extensively altered by human activities. Marinas, wharves, and industrial activities now occupy much of the shore, particularly to the east of the power plant site.

The Pittsburg Power Plant comprises seven electrical generating units with a combined gross capacity of 2,060 MWe. Units 1-4, the oldest units at the plant, started commercial service in 1954. The units have gross generating capacities of 160 MWe, 170 MWe, 160 MWe, and 170 MWe, respectively. Units 5 and 6, each with a gross generating capacity of 330 MWe, were put into service in 1960 and 1961, respectively. Unit 7, the newest and largest unit at the plant (gross generating capacity 740 MWe), was put into service in 1972. All units except Unit 7 employ once-through cooling. Unit 7 is equipped with a closed-cycle cooling system in which mechanical draft cooling towers are used to dissipate heat from the steam condensers.

1.2 DESCRIPTION OF 316(b) STUDIES

1.2.1 Initiation of the 316(b) Studies

In October 1977, PGandE submitted to the San Francisco Regional Water Quality Control Board the Pittsburg Power Plant Cooling Water Intake Report and 316(b) Study Plan (the study plan) (PGandE 1977), which assessed the potential for adverse impact on various biotic groups of the aquatic community and presented a plan for a monitoring study. The study was designed to provide a basis for determining whether the existing cooling water intake structures represented the best technology available for minimizing adverse impact, or whether modifications were appropriate.

The study plan, based on state and federal 316(b) guidelines, was reviewed by several government agencies, including the Regional Board, the State Water Resources Control Board, the California Department of Fish and Game, and Region IX of the United States Environmental Protection Agency (U.S. EPA). The plan was then revised to incorporate agency comments and suggestions. The 316(b) field study program was initiated in March 1978.

1.2.2 Overview of the 316(b) Program

The objective of the field study program was to determine the extent of entrainment and impingement at the Pittsburg Power Plant and the environmental impact of the resultant cropping. The numbers of aquatic organisms entrained and impinged were sampled weekly, and the potential survival of entrained and impinged organisms was considered. Based on these studies, the total annual cropping of entrained and impinged organisms were estimated. These estimates formed the foundation for the assessment of the impact of the intake systems on the aquatic populations of the Sacramento - San Joaquin estuary. The assessed impact in turn was used in conjunction with engineering and operating criteria in determining the best intake technology available for the Pittsburg Power Plant.

1.2.3 Key Organisms Studied

The Pittsburg Power Plant 316(b) studies focused on the following aquatic organism groups, as specified in the study plan (PGandE 1977):

1. fish (all life stages) and
2. macroinvertebrates (e.g., decapods, amphipods, mysid shrimp).

These groups were selected because they may be entrained and impinged, and because aspects of their life history and distribution indicate that they are among the organisms most susceptible to cooling water intake system impact. These organism groups generally have longer generation times and lower reproductive potential than phytoplankton, zooplankton, and benthic organisms. Fish and macroinvertebrates were also selected because many of them are commercially or recreationally valuable or important in the food chain.

Organism groups not selected for study included:

1. phytoplankton,
2. zooplankton (small holoplanktonic and meroplanktonic invertebrates not included in the macroinvertebrates specified above), and
3. benthic invertebrates other than those included in the macroinvertebrates specified above (e.g., molluscs, annelids).

In accordance with an agreement reached with the Regional Board, based on U.S. EPA guidelines (1977), phytoplankton and zooplankton were excluded because of their wide geographic distributions, short generation times, and population regeneration potential and because the vigorous tidal currents continually replenish the phytoplankton and zooplankton populations near the plant, reducing the risk of localized population changes. On the same basis, benthic invertebrates were excluded because their bottom-dwelling habits minimize their involvement with the plant's cooling water intake system.

1.3 ORGANIZATION OF THE REPORT

This report, a 316(b) demonstration, is a summary and analysis of the data collected during the monitoring program at the Pittsburg Power Plant. All study elements specified in the study plan (PGandE 1977) are discussed in the order in which they appear in the study plan (Table 1-1).

The report includes an Executive Summary, Chapters 1 through 6, and Appendixes A through E. Appendix E is bound into the report in microfiche form; it has also been printed as a separate volume.

The design and operation of the Pittsburg Power Plant and its cooling water systems are described in Chapter 2, which may be considered a framework for assessing the impact of entrainment and impingement. Plant operating data specified in the study plan are presented in Section 2.1. The physical and chemical characteristics of the source water body in the vicinity of the plant are discussed in Sections 2.2.1-2.2.3 and in Appendix A. The aquatic biological environment of the source water body is described in Sections 2.2.4 and 2.2.5.

The experimental design, methods, and results of the entrainment and impingement abundance programs are presented in Chapters 3 and 4. The data are tabulated in Appendix E. Based on estimates of the total numbers of organisms entrained and impinged during the study and on estimates of the survival of those organisms, the total annual numbers of entrained and impinged organisms cropped are estimated.

The estimates of entrainment and impingement cropping developed in Chapters 3 and 4 are used in Chapter 5 to assess the impact of the combined cropping on selected

TABLE 1-1 ORGANIZATION OF THE REPORT—LOCATION OF STUDY ELEMENTS SPECIFIED IN STUDY PLAN

Study Element	Location in Report
1. Plant Design and Operation	Chapter 2 Appendix A
2. Entrainment Abundance and Survival Studies	Chapter 3 Appendix B
3. Impingement Abundance and Survival Studies	Chapter 4
4. Near-field Macrozooplankton Survey[a]	
5. Impact Assessment	Chapter 5 Appendix C
6. Best Technology Available Evaluation	Chapter 6 Appendix D
7. Data Base (microfiche or paper copy)	Appendix E

(a) Results of a near-field plankton survey conducted between May 1978 and April 1979 to determine spatial and temporal distributions of ichthyoplankton and macroinvertebrates in the vicinity of the Pittsburg Power Plant have been published separately (Ecological Analysts, Inc., 1981).

aquatic populations in the Sacramento - San Joaquin estuary. The mathematical methods used for impact assessment are presented in Appendix C.

The best technology available for minimizing adverse environmental impact for each of the intake systems is evaluated in Chapter 6. (The full range of possible intake technologies is presented in Appendix D.) Technologies that are considered to be proven and available are evaluated on the basis of their potential for reducing the biological losses associated with entrainment and impingement. A site-specific feasibility analysis is presented for the technologies that have the potential to reduce losses. Finally, based on the combination of biological, engineering, and cost considerations, the best technology available for each intake system is recommended.

1.4 REFERENCES

Ecological Analysts, Inc. (EA). 1981. Pittsburg Power Plant Units 1-7 ichthyoplankton and macrozooplankton survey. Prepared for Pacific Gas and Electric Co. EA, Concord, Calif.

Pacific Gas and Electric Company (PGandE). 1977. Pittsburg Power Plant cooling water intake report and 316(b) study plan. Rev. ed. Submitted to California Regional Water Quality Control Board, San Francisco Bay Region. PGandE, San Francisco.

Tetra Tech, Inc. 1976. 316(a) demonstration Pittsburg Power Plant. Prepared for Pacific Gas and Electric Company. Tetra Tech Rep. TC-638. Lafayette, Calif.

U. S. Environmental Protection Agency (U. S. EPA). 1977. Guidance for evaluating the adverse impact of cooling water intake structures on the aquatic environment: Section 316(b) P.L. 92-500. Draft. U.S. EPA, Office of Water Enforcement, Permits Division, Industrial Permits Branch, Washington, D.C.

Chapter 2

Description of the Pittsburg Power Plant And Characteristics of the Source Water Body

2.1 The Plant and its Cooling Systems

2.2 Physical, Chemical, and Biological Characteristics of the Source Water Body

2.3 References

CHAPTER 2: DESCRIPTION OF THE PITTSBURG POWER PLANT AND CHARACTERISTICS OF THE SOURCE WATER BODY

This chapter describes design and operational features of the Pittsburg Power Plant in the context of the aquatic environmental setting. Section 2.1 describes the plant and its cooling water systems, with emphasis on operational parameters outlined in the study plan (PGandE 1977) that are necessary for the calculation and interpretation of the entrainment and impingement estimates developed in Chapters 3 and 4 and the impact assessment in Chapter 5.

Section 2.2 characterizes physical, chemical, and biological aspects of the aquatic environment in which the plant operates, with emphasis on the parameters outlined in the study plan. The characterization of biological resources is used for impact assessment in Chapter 5, where species affected by the plant are discussed in more detail.

2.1 THE PLANT AND ITS COOLING SYSTEMS

This section describes the intake design and operational procedures for Units 1 through 4, Units 5 and 6, and Unit 7. An analysis of whether these systems represent the best technology available is given in Chapter 6.

2.1.1 Location

The Pittsburg Power Plant is located on the Sacramento - San Joaquin estuary at the nominal division point between the San Francisco Bay system on the west and the delta of the Sacramento and San Joaquin Rivers (the Delta) on the east. The plant is on the south shore of Suisun Bay near Pittsburg, California, approximately 50 miles (80 km) east of San Francisco, and is just west of the confluence of the Sacramento and San Joaquin rivers (Figure 2-1). The plant is situated in the industrial section of Pittsburg (Figure 2-2). The shoreline and bathymetry of Suisun Bay near the plant are shown in Figure 2-3.

2.1.2 Plant Description and Operation

The Pittsburg Power Plant consists of seven generating units with a combined gross

capacity of 2,060 MWe and a combined cooling water flow of 737,000 gpm (1,641 cfs; 48.5 m^3/sec). The configuration of the plant's cooling water system is shown in Figure 2-4.

Units 1-4, installed in 1954, share a common cooling water intake. They have the following design generating capacities and cooling water flows:

	Unit 1	Unit 2	Unit 3	Unit 4	Units 1-4
Capacity (MWe)	160	170	160	170	660
Flow (cfs)	220	220	220	220	880

Units 5 and 6, which began operation in 1960-1961, also share a single intake. Their generating capacities and cooling water flows are:

	Unit 5	Unit 6	Units 5 and 6
Capacity (MWe)	330	330	660
Flow (cfs)	358	358	716

These two groups of units will be treated in this report. Unit 7, a 740-MWe unit that began operation in 1972, uses 45 cfs of makeup water in its closed-cycle cooling system. Unit 7 draws its makeup water from the Units 1-4 intake structure after the water has passed through the traveling screens. Entrainment estimates were calculated separately for Unit 7 as the product of entrainment sampling densities and cooling water flows; impingement for the unit was included in the estimates for Units 1-4.

2.1.2.1 Units 1-4 Cooling Water System: Design and Operational Procedures

The cooling water system serving Units 1-4 is shown schematically as a portion of Figure 2-5. The intake, located on the shoreline, consists of bar racks and traveling screens. Circulating water pumps serving the individual units are located about 30 ft (10 m) behind the screen structure. Each unit is equipped with two circulating water pumps which discharge into separate pressure conduits, each supplying one-half of a unit's steam condenser. Cooling water from the condenser discharge conduits of Units 1-4 flows into a common rectangular conduit, separates into three parallel conduits at a gate well, and is discharged from a submerged outfall located

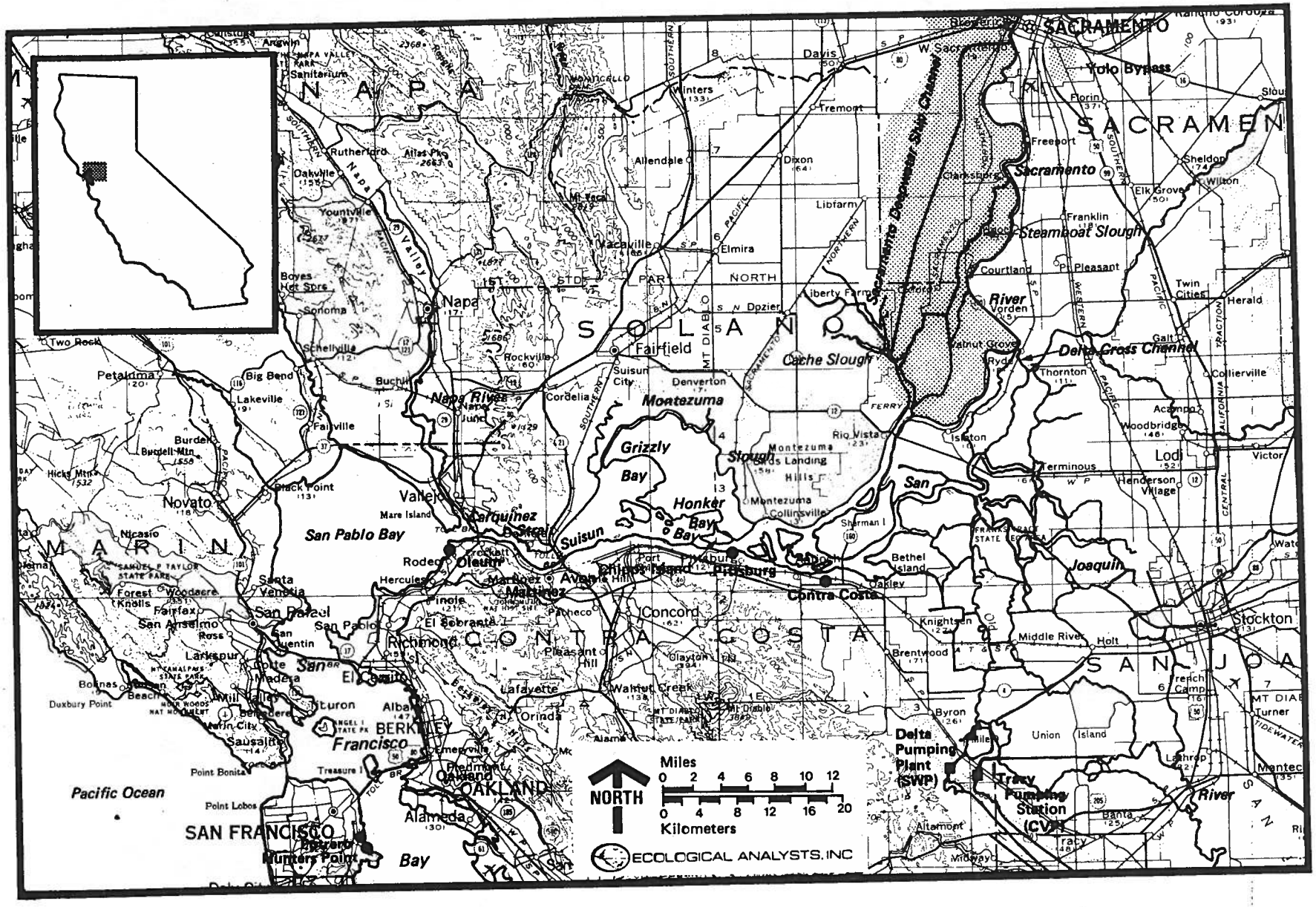

Figure 2-1. Northern San Francisco Bay and the Sacramento—San Joaquin Delta, with Pacific Gas and Electric Company Power Plants.

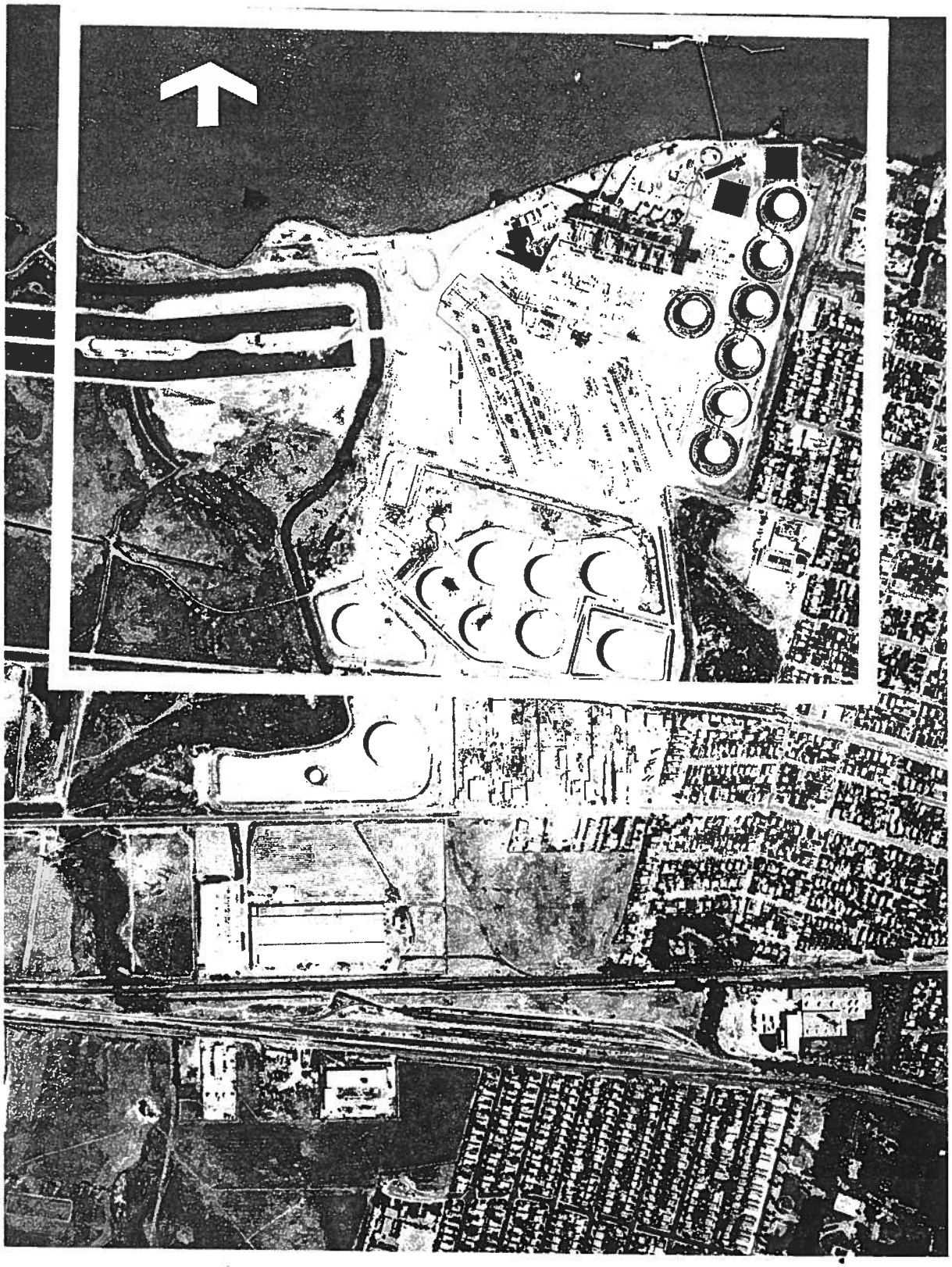

Figure 2-2. The Pittsburg Power Plant.

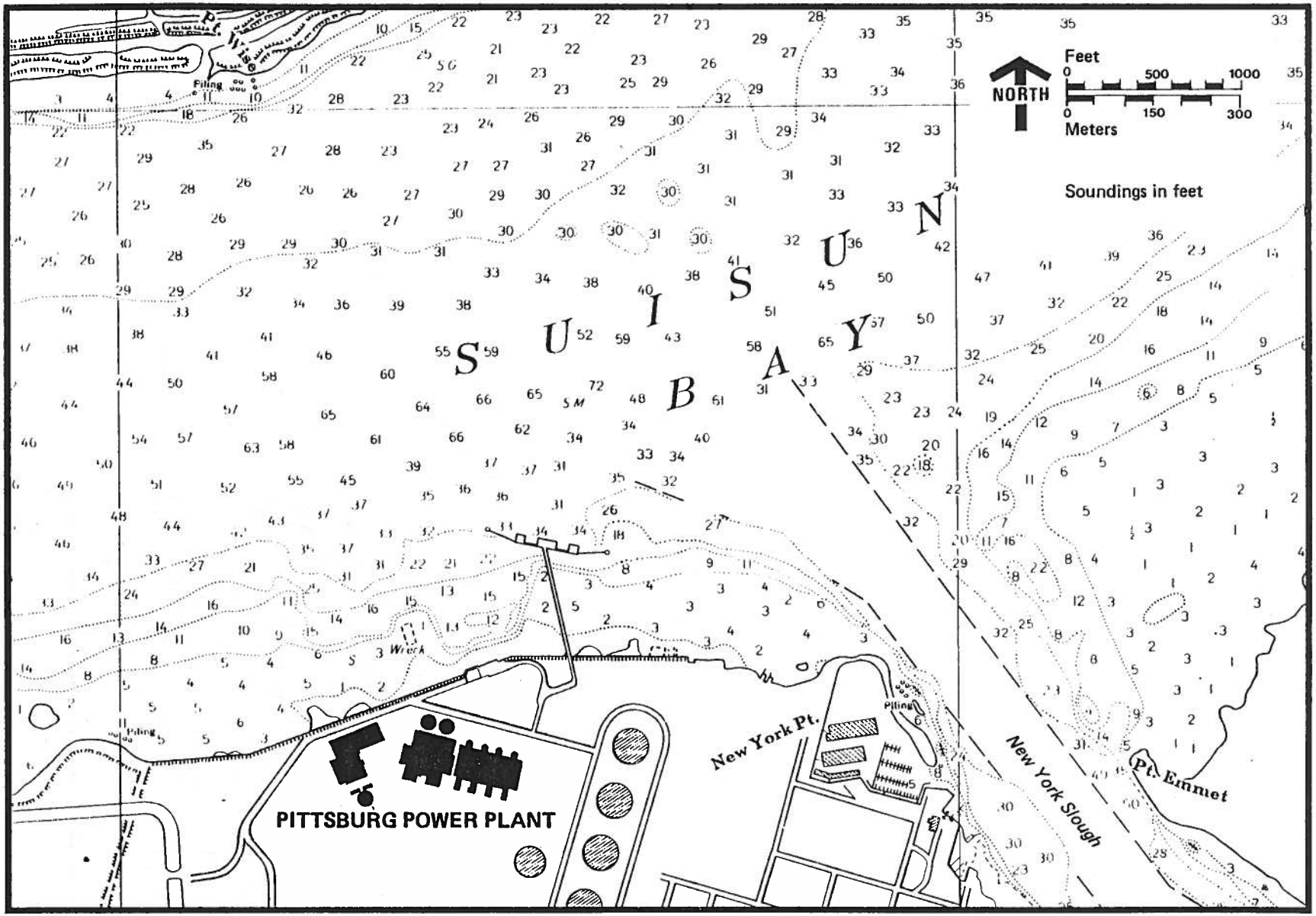

Figure 2-3. Shoreline and bathymetry of Suisun Bay near the Pittsburg Power Plant. (From NOAA 1977.)

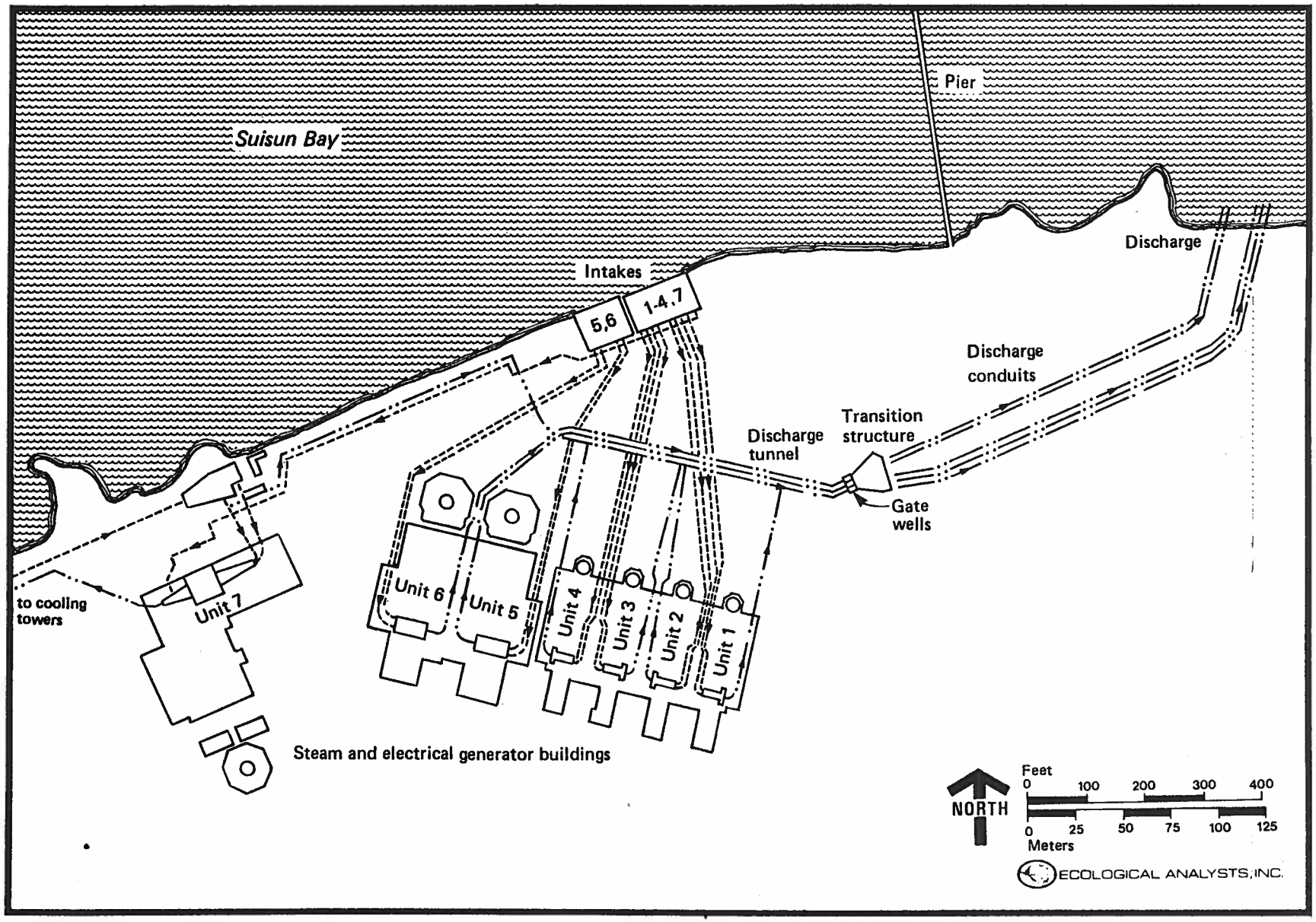

Figure 2-4. General configuration of the Pittsburg Power Plant Units 1-7 cooling water systems.

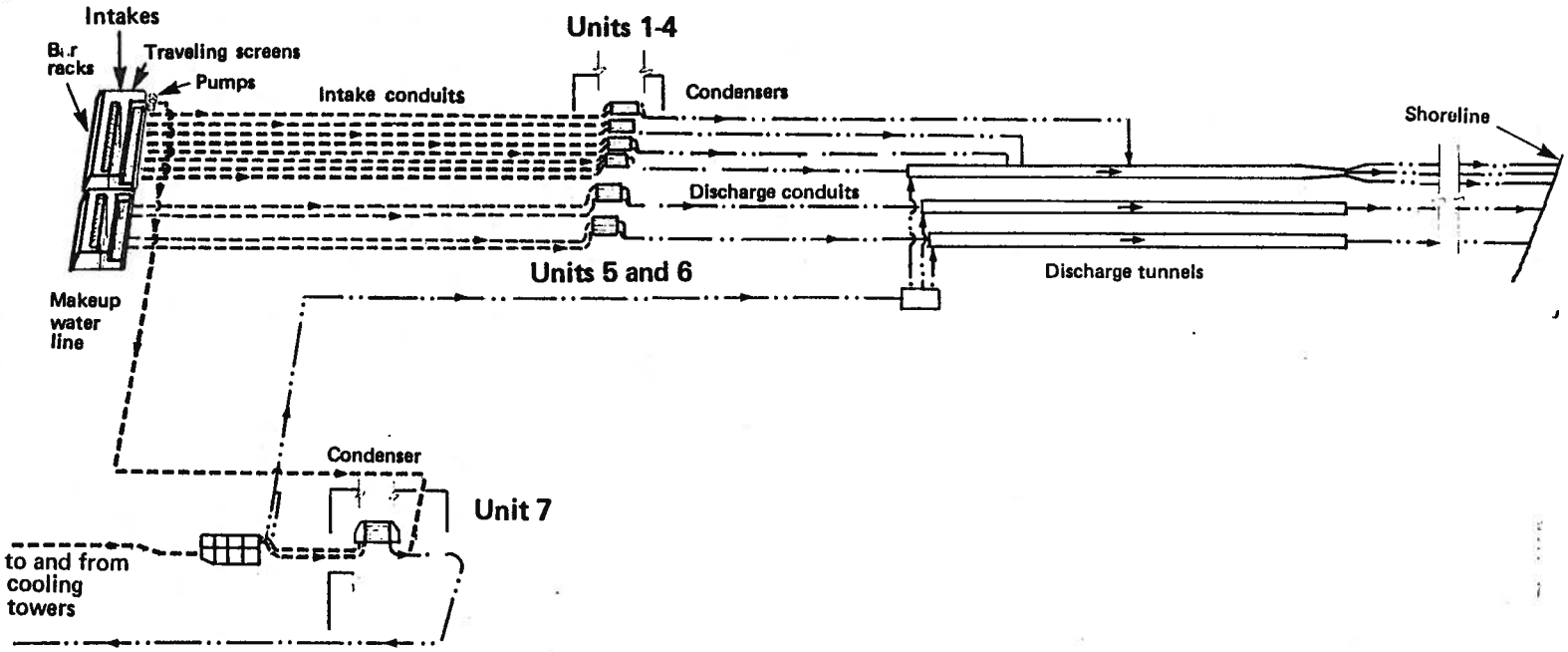

Figure 2-5. Schematic diagram of the Pittsburg Power Plant Units 1-7 cooling water systems.

approximately 30 ft (10 m) offshore at the northeast corner of the plant site. Specifications of the system are presented in Table 2-1.

Figure 2-6 shows the major features of the intake structure. Bar racks spaced 4 in. (10.2 cm) on center are located about 15 ft (4.6 m) in front of the vertical traveling screens, and prevent the entry of large objects into the cooling water system. The vertical traveling screens with a mesh size of 3/8 in. (0.9 cm) retain smaller objects. Debris, along with fish and invertebrates retained by the screens, is removed during screen rotation and washing which is initiated either by a timer, at approximately 4-hour intervals under normal operating conditions, or when the across-screen hydraulic differential exceeds a predetermined maximum. During screen washing, high-pressure (95 psi) spray nozzles wash debris and impinged organisms into a surrounding sluiceway which empties into a screenwash wet well which also receives the screenwash from the Units 5 and 6 traveling screens. The screenwash discharge is returned to the river by three large-diameter trash pumps of 2,800 gpm (6 cfs; 0.2 m^3/sec) capacity. These centrifugal, vertical open impeller pumps are activated in sequence, as the wet well fills with screenwash, by three pedestal float switches set at different heights; they run until the well is empty. The pumps discharge into 18-in. concrete pipe that empties into the Unit 5 discharge conduit.

During normal operation, each unit's two 49,300 gpm (110 cfs; 3.1 m^3/sec) circulating water pumps are run simultaneously and furnish 98,600 gpm (220 cfs; 6.2 m^3/sec) of cooling water to each of the four generating units. Single-pump operation occurs only during condenser maintenance inspections and pump outages. Under single-pump operation, electrical generation must be limited to less than 50 percent of a unit's maximum capacity. The combined cooling water flow of Units 1-4 in normal operation is 394,000 gpm (880 cfs; 25 m^3/sec).

In addition to the eight circulating water pumps, there are six 3,750-gpm (8.3 cfs, 0.25 m^3/sec) service water pumps which supply water to the Units 1-4 auxiliary cooling water heat exchangers from the Units 1-4 intake structure after the traveling screens. In normal operation, four of these six pumps are run to provide 15,000 gpm (33.4 cfs; 1.0 m^3/sec) of service water to the four generating units. This additional volume was not included in calculating entrainment estimates for the power plant, because operating records were not available. It constitutes less that 4 percent of the Units 1-4 cooling water flow.

TABLE 2-1 SPECIFICATIONS OF THE COOLING WATER SYSTEMS AT THE PITTSBURG POWER PLANT

Specification	Units 1-4 [a,b]	Units 5 and 6 [a,b]	Unit 7
Bar racks			
Number	8	6	(b)
Location	Shoreline	Shoreline	
Spacing O.C., in.(cm)	4 (10.2)	4.0 (10.2)	
Bar size, in.(cm)	3x0.4 (7.6x0.9)	3x0.4 (7.6x0.9)	
Traveling screens			
Location	Shoreline	Shoreline	(b)
Number	7	6	
Manufacturer	Link Belt	Link Belt	
Mesh size, in.(mm)	0.4 (9.5)	0.4 (9.5)	
Pumps			
Location	Onshore	Onshore	
Number per unit	2	2	3 [e]
Manufacturer	Foster Wheeler	Ingersoll-Rand	
Type	Mixed flow vertical single-stage	Mixed flow vertical single-stage	Mixed flow vertical single-stage
Capacity, each pump			
cfs	109.8	178.8	22.5
m^3/sec	3.11	5.06	0.64
gpm	49,300	80,250	10,100
Pressure conduits to condenser			
Number	2	2	NA
Diameter, ft(m)	4.5 (1.4)	5.5 (1.7)	
Length, ft(m)	600 (183) [c]	550 (168) [d]	

(a) Units 1-4 have identical specifications except as noted; so do Units 5 and 6.
(b) Units 1-4 and Unit 7 share a common intake structure; so do Units 5 and 6. Bar rack and traveling screen specifications are for the whole intake structure; other specifications are for individual units.
(c) Units 1 and 2. For Units 3 and 4 subtract 50 feet (15 m).
(d) Unit 5. For Unit 6, add 125 ft (38 m).
(e) Makeup water pumps for Unit 7 cooling water system.

TABLE 2-1 (CONT.)

Specification	Units 1-4	Units 5 and 6	Unit 7
Condenser			
Number of tubes	9,548	11,300	16,172
Tube material	aluminum brass	aluminum brass	copper/nickel
Tube O.D., in.(cm)	7/8(2.2)	1(2.5)	1-1/8(2.8)
Tube length, ft(m)	30.0(9.1)	44.0(13.4)	38.0(11.6)
Design delta-T, F(C)	15.0(8.3)	17.3(10)	na
Inlet pressure, psia	26.4	23.8	na
Pressure drop, psi	4.8	5.8	na
Discharge conduits			
Number	1(a)	1	(b)
Size, ft(m)		7.0x7.0(2.1x2.1)	
Length, ft(m)	1,360 (415)(c)	1,760 (540)	
Approximate travel time, sec			
Intake to pumps	55	60	
Pumps to condenser	97(d)	80(e)	
Through condenser	4.3	6.3	
Condenser to discharge	184(f)	323	
Total through plant	341(g)	470(e)	
Total heated	189(g)	330	
Total chlorinated	286(g)	410(e)	
Design water velocities, fps(cm/sec)			
Approach to bar racks	0.4 (12)	0.5 (15)	(h)
Through bar racks	0.5 (15)	0.6 (18)	(h)
Approach to screens	0.8 (24)	0.8 (24)	(h)
Through screens	2.0 (61)	1.5 (46)	(h)
Pumps to condenser	7.1 (216)	7.5 (229)	na
Through condenser	7.0 (213)	7.0 (213)	
Condenser to discharge	7.2 (219)	5.9 (195)	

(a) Common discharge from Units 1-4 to tranistion structure; separates into three conduits to the bay.
(b) Unit 7 utilizes the Units 1-4 discharge
(c) Unit 1. For Unit 2, add 150 ft (48m); for Unit 3, 155 ft (66 m); for Unit 4, 330 ft (116 m).
(d) Units 1 and 2. For Units 3 and 4, subtract 7 sec.
(e) Unit 5. For Unit 6 add 12 sec.
(f) Unit 1. For Unit 2, add 22 sec; for Unit 3, 30 sec: for Unit 4, 53 sec.
(g) Unit 1. For Unit 2, add 22 sec; for Unit 3, 23 sec; for Unit 4, 46 sec.
(h) Unit 7 shares a common intake structure with Units 1-4.

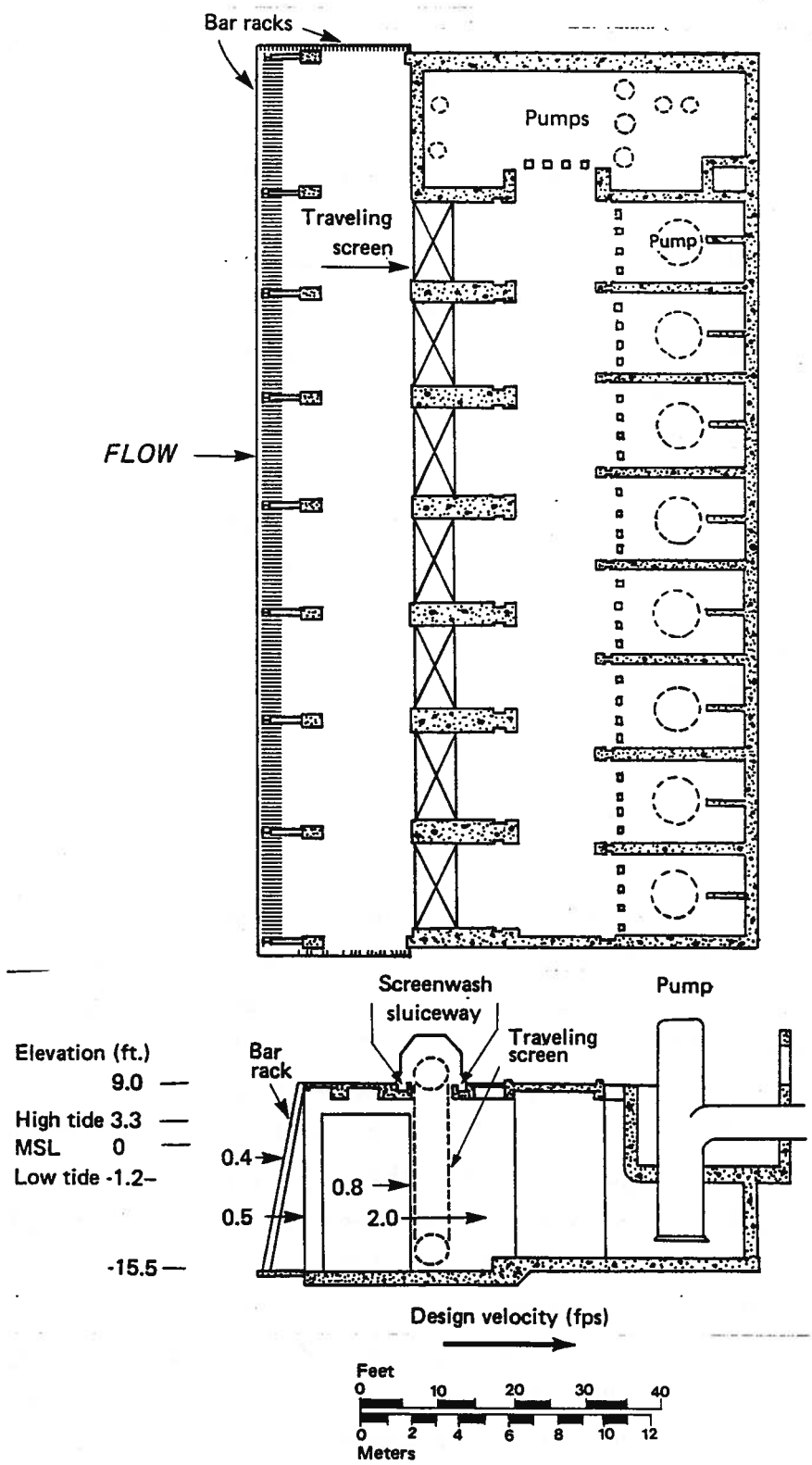

Figure 2-6. Plan and section schematic diagrams of the Pittsburg Power Plant Units 1-4 intake structure.

Cooling water is under pressure from the outlets of the circulating water pumps to the discharge. The pressure increases from atmospheric (14.7 psia) at the intake to 26.4 psia at the circulating water pump discharge. Pressure drops through the cooling system, with about a 5-psi drop across the condenser. Relative pressures do not change during various tidal stages. The design delta-T, the rise in water temperature across the condenser, is 15.6 F (8.7 C) in normal full-load operation.

Chlorine is injected into the cooling system just ahead of the circulating water pumps to prevent condenser biofouling. For Units 1-4, each intake tunnel has a chlorine injection diffuser located ahead of the circulating water pump. Injection into the eight tunnels is controlled by automatic timers that chlorinate tunnels 1-8 in sequence, so that only one tunnel at a time is chlorinated. Chlorination is done for 30 minutes on a frequency that varies with season and need (determined by inspection). The usual schedule is 1-3 times a week; the maximum is once a day. A chlorine residual of 0.2-0.5 mg/l is maintained at the condenser inlet. In accordance with NPDES permit requirements, chlorine residuals at the discharge have been reported to the state on a monthly basis since 1976. The usual analytical results show 0.0 chlorine residual, but several trace residuals have been reported.

2.1.2.2 Units 5 and 6 Cooling Water System: Design and Operational Procedures

The cooling water system serving Units 5 and 6 is shown schematically in Figure 2-5. The intake structure, adjacent to the Units 1-4 intake structure, also consists of bar racks, traveling screens, and circulating water pumps. Separate intake conduits conduct cooling water to the Unit 5 and Unit 6 condenser halves. The cooling water from the two condenser halves recombine at the condenser outlet; this flow remains separate from other unit discharges through transit and discharge into the bay from a submerged outfall adjacent to that of Units 1-4. Cooling system design specifications are presented in Table 2-1. Figure 2-7 shows the major features of the intake structure. Bar racks, spaced 4 in. (10.2 cm) on center, are located about 15 ft (4.6 m) in front of the vertical traveling screens. The traveling screens have 3/8-in. (0.9-cm) mesh. Debris, along with fish and invertebrates retained by the screens, is removed during screen rotation and washing which is initiated either by a timer, at approximately 4-hour intervals under normal operating conditions, or when the across-screen hydraulic differential exceeds a predetermined maximum. During screen washing, high-pressure (110 psi) spray nozzles wash debris and impinged

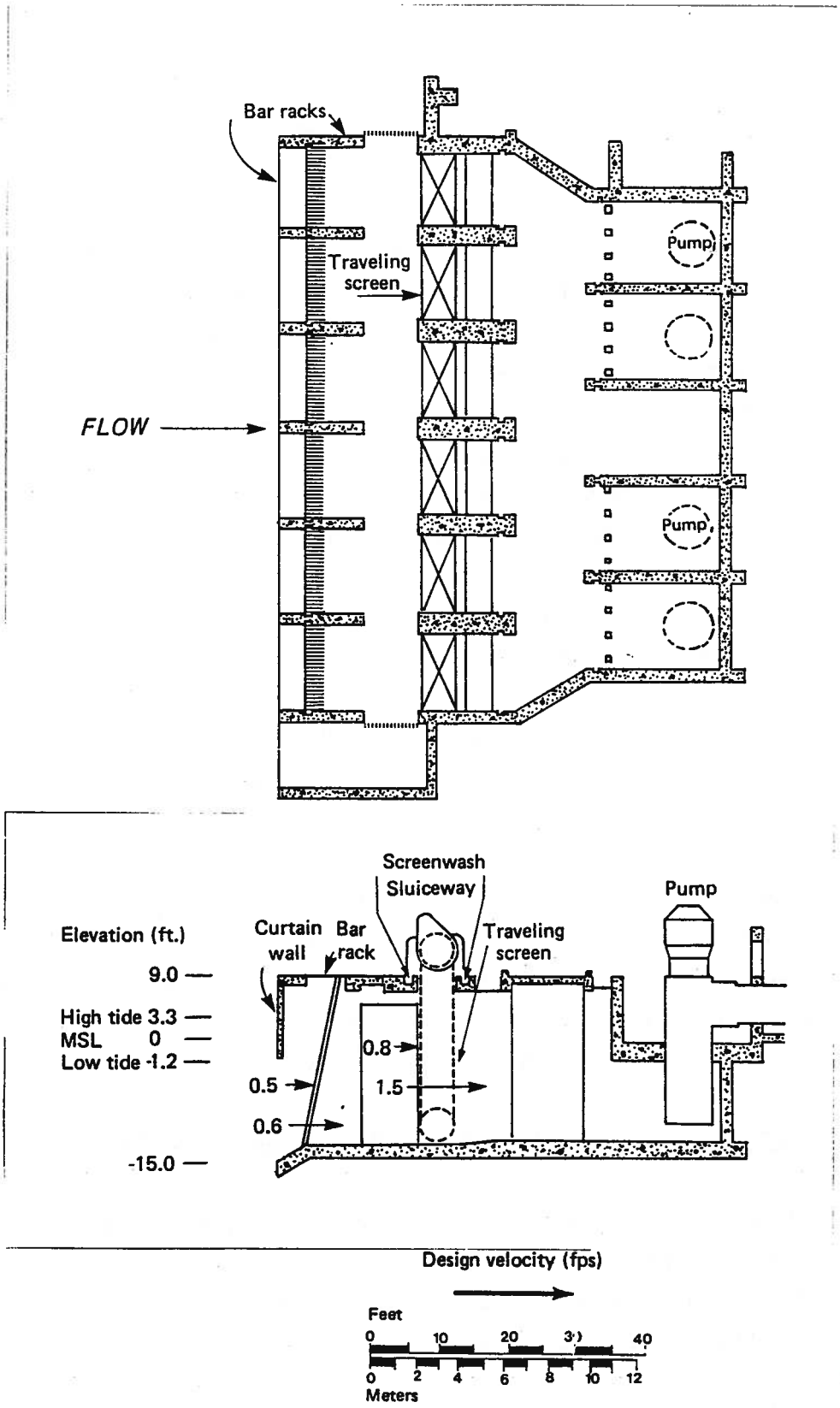

Figure 2-7. Plan and section schematic diagrams of the Pittsburg Power Plant Units 5 and 6 intake structure.

organisms into a surrounding sluiceway which empties into a common screenwash wet well that also receives the screenwash from Units 1-4. Large-diameter trash pumps periodically drain the wet well into the Unit 5 discharge conduit.

During normal operation, the four 80,250 gpm (178 cfs; 5.1 m^3/sec) circulating water pumps serving Units 5 and 6 furnish 321,000 gpm (716 cfs; 20.4 m^3/sec) of cooling water to the Units 5 and Unit 6 condensers. Single-pump operation at a unit occurs only during condenser maintenance inspections and pump outages. In single-pump operation, electrical generation from the unit is limited to less than 50 percent of its maximum capacity. A portion of the cooling water is drawn from the circulating water pump discharge ahead of the condensers, for use in closed-cycle auxiliary cooling water heat exchangers. This water is discharged to the individual unit circulating water discharge conduits after the condensers.

Cooling water is under pressure from the outlets of the circulating water pumps to the discharge structure. Pressure increases from atmospheric at the intake to 23.8 psia at the circulating water pump discharge. Pressure drops through the cooling system, with a 5.8-psi drop across the condenser. Relative pressures do not change during various tidal stages. The design delta-T is 17.3 F (9.6 C) under normal full-load operation.

Chlorine is injected into the cooling water system just ahead of the circulating water pumps to prevent condenser biofouling. For the four tunnels of Units 5 and 6, injection is controlled by automatic timers which chlorinate tunnels 9-12 in sequence following the chlorination of the Units 1-4 tunnels. Chlorination times, frequencies and residuals are the same as for Units 1-4 (Section 2.1.2.1).

2.1.2.3 Unit 7 Cooling Water System: Design and Operational Procedures

The cooling water system for Unit 7 is shown schematically in Figures 2-4 and 2-5. This closed-cycle cooling water system employs mechanical-draft, wet cooling towers to dissipate the heat transferred to the circulating water flow during transit through the condensers. In a closed-cycle cooling water system, "makeup" water is withdrawn from a source to replace that evaporated in the cooling towers or carried away in small droplets (drift) and to control the dissolved solids content of the cooling water. The portion returned to the water body with its higher concentration of minerals is

called blowdown. The ratio of water returned (blowdown) to water withdrawn (makeup) depends on a number of factors affecting the rate of evaporation (e.g., air temperature, humidity, power production).

Unit 7 is not equipped with a separate makeup water intake. Instead, makeup water is withdrawn from behind the Units 1-4 intake structure by three 10,100 gpm (22.5 cfs; 0.6 m^3/sec) makeup water pumps. In normal operation, two of the pumps are run simultaneously and furnish 45 cfs (1.2 m^3/sec) of makeup water to the cooling system. Maximum losses from drift and evaporation have been estimated by PGandE to be approximately 7,000 gpm (15.6 cfs; 0.44 m^3/sec); these decrease in proportion to reductions in unit output. Blowdown flow is discharged through a manifold system which distributes it into the discharge conduits of the once-through units (see Figure 2-5). Because the makeup water pumps are on continuously, the blowdown flow varies with unit operation, from a minimum of 29.4 cfs (0.8 m^3/sec) during periods of maximum evaporation to nearly the full makeup water flow when the generating load is low.

Chlorine is injected into the circulating water flow to prevent condenser biofouling. Unit 7 is chlorinated for 10 minutes twice a day to maintain a 0.5 mg/liter chlorine residual at the condenser inlet water box. No detectable chlorine residual remains in the blowdown flow.

2.1.3 Cooling Water System Operations

The plant design specifications and operational procedures discussed in Section 2.1.2 apply primarily to operation under full-load conditions: continuous normal circulating water pump operation and generating at full capacity. Because fossil-fuel generating units are usually operated to respond to seasonal and daily fluctuations in demand for electrical energy, the actual load is generally well below operating capacity when expressed as an average over a month or a year. One convenient statistic for description of plant operational status is the plant or unit's "capacity factor," defined as the ratio of actual gross generation to maximum possible generation over a specified interval, expressed as a percentage. A second convenient indicator is the operation of circulating water pumps (number of pumps x hours), which can be used to calculate the total volume of water pumped through the condenser cooling system during any specific period of time.

2.1.3.1 Monitoring Procedures

Electrical generation load and pump operation are monitored routinely during daily activities and recorded in operational logs. The logs were read and data compiled to prepare monthly averages of capacity factor and pumped water volume. Velocity measurements were made of water approaching the intake structures.

Intake and discharge water temperatures for Units 1-4 and Units 5 and 6 were continuously monitored from April 1978 through May 1979. Foxboro temperature sensors and recorders were located in:

1. the inlet pipe of the west condenser half of Unit 6,
2. the inlet pipe of the west condenser half of Unit 1,
3. the Units 1-4 discharge gate well,
4. the Unit 5 discharge gate well, and
5. the Unit 6 discharge gate well.

The Unit 5 intake temperatures were assumed to be identical to those of Unit 6. Temperatures were recorded continuously throughout the study and read from the chart recordings at 3-hour intervals to calculate average weekly values. The temperature recorders were calibrated using a two-point calibration at 32 F and 100 F (0 C and 37.8 C) with a thermometer traceable to a National Bureau of Standards reference (Appendix A).

Measurements of conductivity (salinity), intake water temperature, and dissolved oxygen (DO) were routinely made when biological samples were collected.

2.1.3.2 Study Year Operation

Electrical generation, circulating water pump operation, intake velocities, intake and discharge temperatures, and unit outages of the Pittsburg Power Plant are discussed for Units 1-4, Units 5 and 6, and Unit 7 in the following sections. Circulating water pump operation and the water volumes pumped are used in Chapters 3 and 4 to estimate annual entrainment and impingement abundances under actual operating conditions of each unit. Temperature measurements are related to entrainment survival estimates in Chapter 3. Intake water velocities are related to impingement

in Chapter 4. Water quality measurements made during the study period are discussed in Section 2.2.3.

2.1.3.2.1 Units 1-4 Operation

Units 1-4 are operated essentially as base load units, providing electrical power to meet normal daily demand. A summary of outages is given in Table 2-2. Outages are divided into three categories: scheduled, forced, and reserve shutdown. Scheduled outages are planned interruptions of unit operation required for inspections and preventive maintenance. Forced outages are interruptions in unit operation caused by unanticipated problems that occur during operation. Unscheduled outages generally last only a short time (seven days or less). Reserve shutdown outages occur during periods of low electrical demand when more efficient or economical generating units are capable of meeting the system demands. During these outages the units are kept fully available for operation to permit rapid response to peak energy demands.

A summary of the average monthly capacity factors (percentage of maximum generation) for the individual units and for their combined operation from March 1978 through December 1979 is given in Table 2-3. During this period, monthly generating capacity factors of Units 1-4 combined ranged from 9 to 82 percent and averaged 56 percent. Individual unit monthly capacity factors ranged from 0 to 92 percent for Unit 1, from 0 to 83 percent for Unit 2, from 0 to 93 percent for Unit 3, and from 0 to 92 percent for Unit 4, and average monthly capacity factors were 74, 64, 71, and 60 percent, respectively. Capacity factors were generally highest in the period from October to January and lowest in April and May.

The average monthly capacity factors presented in Table 2-3 reflect, in part, seasonal fluctuations in the demand for electrical power. In addition, the operation of Units 1-4 was regulated to meet hourly and daily requirements for electrical power. Hourly demand is generally greatest between 0700 and 1000 hours and between 1600 and 2000 hours, and least between 2000 and 0700 hours. Greatest daily demands for electrical power occur from Monday to Friday. Thus, actual hourly and daily operation of Units 1-4 fluctuates to a greater degree than is apparent from the average monthly capacity factor.

Table 2-2 TOTAL DURATIONS (hours) OF UNIT OUTAGES AT THE PITTSBURG POWER PLANT, 1978, 1979

Unit	Maintenance Outages		Reserve Shutdown
	Unscheduled	Scheduled	
	1978		
1	237	1,362	846
2	407	525	919
3	1,910	1,413	2,873
4	296	2,096	12
5	394	2,497	575
6	176	420	365
7	1,167	1,116	214
	1979		
1	108	405	50
2	71	877	56
3	162	787	42
4	397	1,345	0
5	1,463	2,078	0
6	1,156	1,744	0
7	2,645	1,759	0

Note: the annual summaries represent the period from January 1 to December 31.

Source: PGandE operating logs.

TABLE 2-3 MONTHLY CAPACITY FACTORS[a] (%) FOR UNITS 1-4 OF THE PITTSBURG POWER PLANT, MARCH 1978 - DECEMBER 1979

Month	Unit 1	Unit 2	Unit 3	Unit 4	Mean[b] Units 1-4
1978					
March	41.4	37.8	0	13.6	23.3
April	12.4	24.5	0	0	9.3
May	31.7	40.3	4.9	0	19.2
June	61.0	49.9	44.1	57.1	53.0
July	65.5	49.3	53.0	59.5	56.8
August	38.8	63.6	25.3	67.7	49.4
September	0	17.9	0	50.4	17.6
October	51.9	57.8	70.9	70.2	62.7
November	77.2	79.7	31.5	74.8	66.1
December	92.3	67.1	0	80.4	60.4
1979					
January	79.0	74.0	2.9	79.7	59.4
February	73.8	66.5	58.8	70.0	67.3
March	68.1	62.9	65.5	66.9	65.8
April	59.5	56.9	62.1	55.3	58.4
May	59.4	53.6	60.8	53.2	56.6
June	80.1	79.7	81.8	34.5	68.7
July	70.6	66.2	76.0	72.3	71.2
August	75.0	78.4	82.6	77.0	78.2
September	53.4	71.9	90.6	52.8	67.0
October	90.8	69.6	91.4	0	62.1
November	89.1	83.2	88.8	66.8	81.8
December	89.8	0	93.2	91.6	68.0

(a) Defined as the ratio of actual generation in the month to the maximum possible generation.
(b) Means are weighted by differing capacities: Units 1 and 3 are 160 MWe; Units 2 and 4 are 170 MWe.

In typical operation, each unit's two main circulating water pumps are run simultaneously ("continuous normal" operation), resulting in a cooling water flow rate of 394,000 gpm (880 cfs; 25 m^3/sec). The estimated total cooling water volumes pumped during each month from March 1978 through December 1979 and the percentages of continuous normal monthly cooling water flows that these volumes represent are presented in Table 2-4. The percentage ranged from 35.1 to 99.5. The average percentage of normal capacity for the study period was 80.

Water velocities approaching the intake forebays of Units 1-4 were measured at eight tide heights by PGandE personnel (PGandE 1981). Velocity profiles were based on measurements made in an array of sampling locations in each of the seven intake forebays, approximately 1.5 ft (0.5 m) in front of each forebay. Water velocities were not influenced substantially by variations in tidal stage. Water velocities ranged from 0.2 to 2.2 fps (6-67 cm/sec) and averaged 1.0 fps (30 cm/sec), higher than the design velocity (0.5 fps; 15 cm/sec). Sediment accumulations at the Units 1-4 intake forebays had reduced the cross-sectional areas by an average of 61 percent. This is considered to be the cause of the higher observed water velocities. Design velocities of water approaching the intake forebay structure, approaching the vertical traveling intake screens, and passing through the intake screen mesh are shown in Figure 2-6.

Water temperatures recorded every 3 hours from the intake and discharge of Units 1-4 were used to characterize the temperature regime of the units. Average weekly intake and discharge temperatures during the study period are shown in Figure 2-8. A frequency analysis of temperatures over the study period is presented in Table A-5; daily mean, minimum, and maximum temperatures are listed in Table E.3-1, Appendix E. Intake temperatures ranged from 45 to 85 F (7-29 C) during the study, with 59 percent of the readings below 68 F (20 C). Discharge temperatures at Units 1-4 ranged from 52 to 101 F (11-38 C), with 38 percent below 77 F (25 C) and 68 percent below 86 F (30 C). Under some circumstances water discharged from the plant is carried back to the cooling water intakes (recirculation). The extent of recirculation is determined primarily by tidal stage: because the plant intake structures are located downstream of the discharge, recirculation occurs primarily during ebb tides, when the thermal plume flows in a downstream direction and is held inshore by the tidal current. In a study of recirculation at the plant (PGandE 1970a), it was determined that the amount of recirculation was inversely related to tidal height and depended on the location of the intake. Recirculation at Units 1-4 was

TABLE 2-4 MONTHLY COOLING WATER VOLUMES AND PERCENTAGES OF NORMAL CAPACITY OF UNITS 1-4 OF THE PITTSBURG POWER PLANT, MARCH 1978 - DECEMBER 1979

Month	Volume										Percent Normal Capacity
	Unit 1		Unit 2		Unit 3		Unit 4		Units 1-4		
	$ft^3 \times 10^6$	$m^3 \times 10^6$	$ft^3 \times 10^6$	$m^3 \times 10^6$	$ft^3 \times 10^6$	$m^3 \times 10^6$	$ft^3 \times 10^6$	$m^3 \times 10^6$	$ft^3 \times 10^6$	$m^3 \times 10^6$	
1978											
March	523.4	14.8	527.3	14.9	0	0	152.2	4.3	1,202.8	34.1	51.1
April	388.6	11.0	409.5	11.6	0	0	0	0	798.1	22.6	35.1
May	495.3	14.0	569.2	16.1	49.0	1.4	136.8	3.9	1,250.3	35.4	53.1
June	536.0	15.8	543.9	15.4	364.1	10.3	563.3	16.0	2,007.2	56.9	88.2
July	583.4	16.5	494.9	14.0	346.3	9.8	584.2	16.6	2,008.8	56.9	85.4
August	466.0	13.2	568.8	16.1	225.7	6.4	581.1	16.5	1,841.6	52.2	78.3
September	151.4	4.3	235.6	6.7	0	0	558.5	15.8	945.5	26.8	41.5
October	435.2	12.3	563.7	16.0	500.8	14.2	579.1	16.4	2,078.8	58.9	88.4
November	518.2	14.7	565.3	16.0	209.1	5.9	522.2	14.8	1,814.7	51.4	79.7
December	550.2	15.6	517.8	14.7	0	0	528.1	15.0	1,596.1	45.2	70.1
1979											
January	576.3	16.3	575.5	16.3	71.2	2.0	586.6	16.6	1,809.6	51.3	76.9
February	531.3	15.1	523.4	14.8	521.4	14.8	514.7	14.6	2,090.6	59.2	98.4
March	585.0	16.6	584.2	16.6	583.4	16.6	587.0	16.6	2,339.7	66.3	99.5
April	513.5	14.5	501.6	14.2	540.0	15.3	494.9	14.0	2,049.9	58.1	90.0
May	588.2	16.7	561.3	15.9	588.2	16.7	576.7	16.3	2,314.4	65.6	98.4
June	547.1	15.5	569.2	16.1	543.5	15.4	366.8	10.9	2,026.6	57.4	89.0
July	548.3	15.5	588.2	16.7	581.9	16.5	575.9	16.3	2,294.2	65.0	97.5
August	519.8	14.7	564.5	16.0	567.6	16.1	530.9	15.0	2,182.7	61.8	92.8
September	394.1	11.2	557.3	15.8	563.3	16.0	350.6	9.9	1,865.3	52.8	81.9
October	585.8	16.6	494.9	14.0	586.6	16.6	5.5	0.2	1,672.8	47.4	71.1
November	564.5	16.0	566.8	16.1	558.9	15.8	509.1	14.4	2,199.3	62.3	96.6
December	583.4	16.5	3.2	0.1	586.6	16.6	585.0	16.6	1,758.2	49.8	74.7

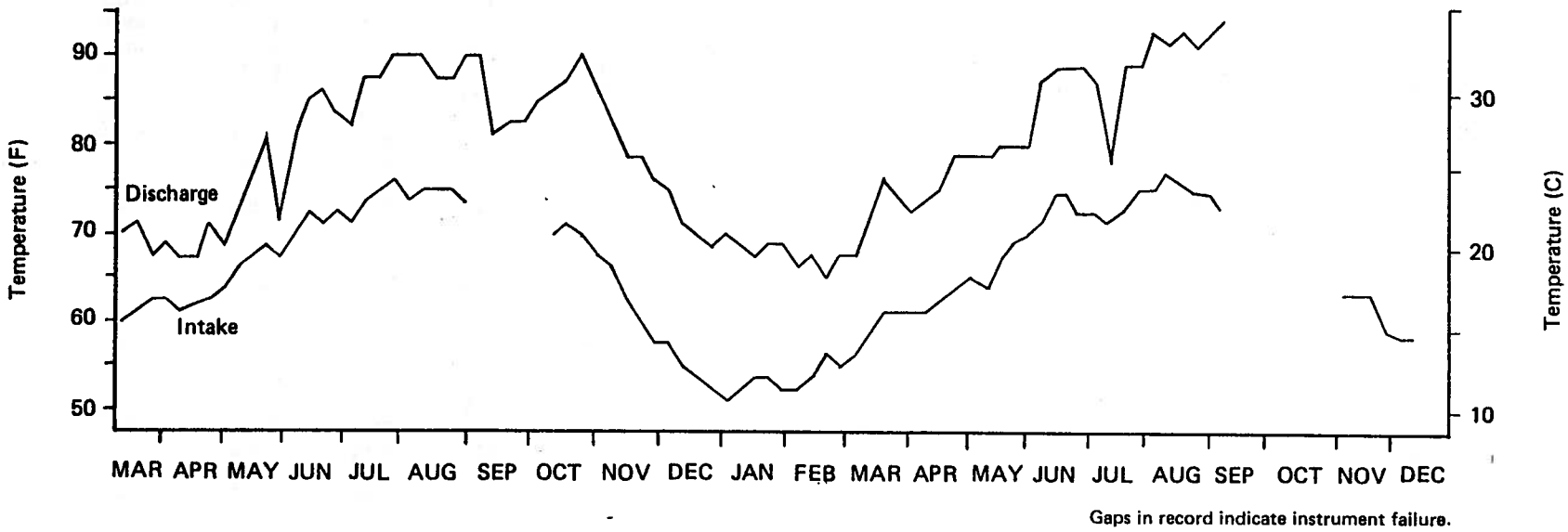

Figure 2-8. Average weekly intake and discharge temperature at the Pittsburg Power Plant Units 1-4, March 1978 – December 1979.

greater than at Units 5 and 6. Greatest recirculation (40 percent) was observed between the maximum ebb tide and the first hour following low tide; consequently, it is during this period that highest discharge temperatures occur. The degree of recirculation during other tidal stages was found to be low (PGandE 1970a).

An extensive discussion of the characteristics of the thermal discharge plume under various tidal conditions is presented in the 316(a) demonstration for the Pittsburg Power Plant (Tetra Tech 1976). Also presented in the 316(a) demonstration is a summary of the operation of the plant's cooling system, including intake and discharge temperatures and delta-Ts for the period 1970-1974. Tetra Tech reported intake temperatures up to 3.3 C (6 F) higher than ambient temperatures during periods of recirculation. Although not specifically designed to examine thermal recirculation, the continuous intake temperature monitoring (Appendix A, this study) showed a pattern in the relationship between increased intake water temperatures and tidal stage similar to that reported by Tetra Tech.

2.1.3.2.2 Units 5 and 6 Operation

Units 5 and 6 outages for the years 1978 and 1979 are summarized in Table 2-2. A summary of the average monthly capacity factors for Units 5 and 6 from March 1978 through December 1979 is given in Table 2-5. Monthly capacity factors ranged from 0 to 85 percent for Unit 5 and 0 to 88 percent for Unit 6. The average capacity factor for the study period was 40 percent for Unit 5 and 56 percent for Unit 6. Average monthly capacity factors did not vary greatly between months, and no seasonal pattern was apparent in Units 5 and 6 electrical generation. Units 5 and 6 are relatively efficient and are operated to meet the long-term, consistent demand (base load) for electrical power.

Four circulating water pumps with rated capacities of 80,200 gpm (178 cfs; 5.1 m^3/sec) each are installed in the Unit 5 and Unit 6 cooling systems. In typical operation, both pumps of each unit are used simultaneously, resulting in a cooling water flow rate of 321,000 gpm, (716 cfs; 20.4 m^3/sec). Estimated total cooling water volumes pumped during each month from March 1978 through December 1979 are summarized in Table 2-6. The percentage of continuous normal operation ranged from 33 to 99 percent, with an average of 73 percent for the study period.

TABLE 2-5 MONTHLY CAPACITY FACTORS[a] (%) FOR UNITS 5 AND 6 AND UNIT 7 OF THE PITTSBURG POWER PLANT, MARCH 1978 - DECEMBER 1979

Month	Unit 5	Unit 6	Mean Units 5 and 6[b]	Unit 7[b]
1978				
March	4.4	50.0	27.2	44.3
April	0	48.6	24.3	29.2
May	0	35.8	17.9	54.3
June	0	70.6	35.3	66.8
July	50.6	61.3	56.0	57.9
August	58.8	59.8	59.3	51.9
September	50.3	30.0	40.2	55.7
October	76.4	67.8	72.1	39.0
November	77.0	80.7	78.8	62.2
December	84.9	88.2	86.6	74.3
1979				
January	80.3	84.4	82.4	27.3
February	74.1	78.9	76.5	0
March	72.7	4.8	38.8	48.6
April	56.3	0	28.2	63.2
May	0	21.7	10.8	37.9
June	0	79.1	39.6	44.3
July	16.4	80.4	48.4	71.3
August	60.7	80.2	70.5	68.2
September	49.5	73.0	61.3	46.4
October	0	87.3	43.7	0
November	29.2	23.9	26.6	2.2
December	45.7	24.6	35.2	0

(a) Defined as the ratio of actual generation in the month to the maximum possible generation.
(b) Capacities of Units 5 and 6 are 330 MWe each; of Unit 7 740 MWe.

TABLE 2-6 MONTHLY COOLING WATER VOLUMES AND PERCENTAGES OF NORMAL CAPACITY FOR UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MARCH 1978 - DECEMBER 1979

Month	Volume: Unit 5 $ft^3 \times 10^6$	Volume: Unit 5 $m^3 \times 10^6$	Volume: Unit 6 $ft^3 \times 10^6$	Volume: Unit 6 $m^3 \times 10^6$	Volume: Units 5 and 6 $ft^3 \times 10^6$	Volume: Units 5 and 6 $m^3 \times 10^6$	Percent Normal Capacity
1978							
March	175.1	5.0	921.7	26.1	1,096.8	31.0	57.3
April	0	0	925.6	26.2	925.6	26.2	49.9
May	0	0	888.3	25.1	888.3	25.1	46.4
June	50.2	1.4	923.0	26.1	973.2	27.5	52.5
July	760.2	21.5	895.4	25.3	1,655.5	46.9	86.4
August	802.7	22.7	869.6	24.6	1,672.3	47.3	87.3
September	863.2	24.4	668.8	18.9	1,532.0	43.4	82.6
October	956.5	27.1	885.7	25.1	1,842.2	52.1	96.2
November	925.0	26.2	918.5	26.0	1,843.5	52.2	99.4
December	896.7	25.4	920.5	26.1	1,817.1	51.4	98.0
1979							
January	943.6	26.7	957.8	27.1	1,901.4	53.8	99.3
February	826.5	23.4	865.1	24.5	1,691.6	47.9	97.8
March	956.5	27.1	120.4	3.4	1,076.9	30.5	56.2
April	776.9	22.0	0	0	776.9	22.0	41.9
May	30.9	0.9	593.5	16.8	624.4	17.7	32.6
June	361.1	10.2	925.6	26.2	1,286.7	36.4	69.4
July	393.9	11.2	951.4	26.9	1,345.3	38.1	70.2
August	926.9	26.2	926.9	26.2	1,853.8	52.5	96.8
September	832.9	23.6	885.7	25.1	1,718.6	48.6	92.7
October	10.9	0.3	949.4	26.9	960.4	27.2	50.1
November	543.3	15.4	624.4	17.7	1,167.6	33.0	63.0
December	913.4	25.9	803.3	22.7	1,716.7	48.6	89.6

Water velocities approaching the intake forebays of Units 5 and 6 were measured at three tide heights by PGandE personnel (PGandE 1981). Velocity profiles were based on measurements made in an array of sampling locations in each of the six intake bays, approximately 1.5 ft (0.5 m) in front of each bar rack. Water velocities ranged from 0.1 to 1.3 fps (3-40 cm/sec) and averaged 0.7 fps (21 cm/sec). Water velocities were not influenced substantially by variations in tidal stage. The maximum average water velocity (0.7 fps; 21 cm/sec) is slightly greater than the design velocity (0.5 fps; 15.2 cm/sec). Sediment accumulations at the Units 5 and 6 intake forebays had reduced the cross-sectional area by an average of 26 percent. This is considered to be the cause of the higher observed water velocities. Design velocities of water approaching the intake forebay structure, approaching the vertical traveling intake screens, and passing through the intake screen mesh are shown in Figure 2-7.

Average weekly intake and discharge water temperatures during the study period are shown for Unit 5 and Units 6 in Figure 2-9. Discharge temperatures at Unit 5 were similar to those at Unit 6. Intake temperatures ranged from 45 to 82 F (7-28 C) during the study, with 93 percent of the readings below 77 F (25 C). Unit 5 discharge temperatures ranged from 46 to 100 F (8-38 C), with 47 percent below 77 (25 C) and 70 percent below 86 F (30 C) (Table A-6). Unit 6 discharge temperatures ranged from 51 to 103 F (11-39 C), with 41 percent below 77 C (25 C) and 62 percent below 86 F (30 C). Table A-5 presents a frequency analysis of intake and discharge temperatures over the study period; daily mean, minimum, and maximum temperatures are listed in Table E.3-1, Appendix E.

2.1.3.3 Unit 7 Operation

Outages at Unit 7 are summarized in Table 2-2 for the years 1978 and 1979. A summary of the average capacity factors for Unit 7 from March 1978 through December 1979 is given in Table 2-5. Monthly capacity factors ranged from 0 to 74 percent. Average capacity factor for the period was 49.7 percent.

Three makeup water pumps, each with a rated capacity of 10,100 gpm (22.5 cfs; 0.6 m^3/sec), are available for use in the Unit 7 cooling system. In typical operation, two of the three are used, resulting in a makeup water flow rate of 45 cfs (1.2 m^3/sec). Pump operation was not recorded during the study period. Instead, monthly makeup water flows were estimated on the basis of turbine operation.

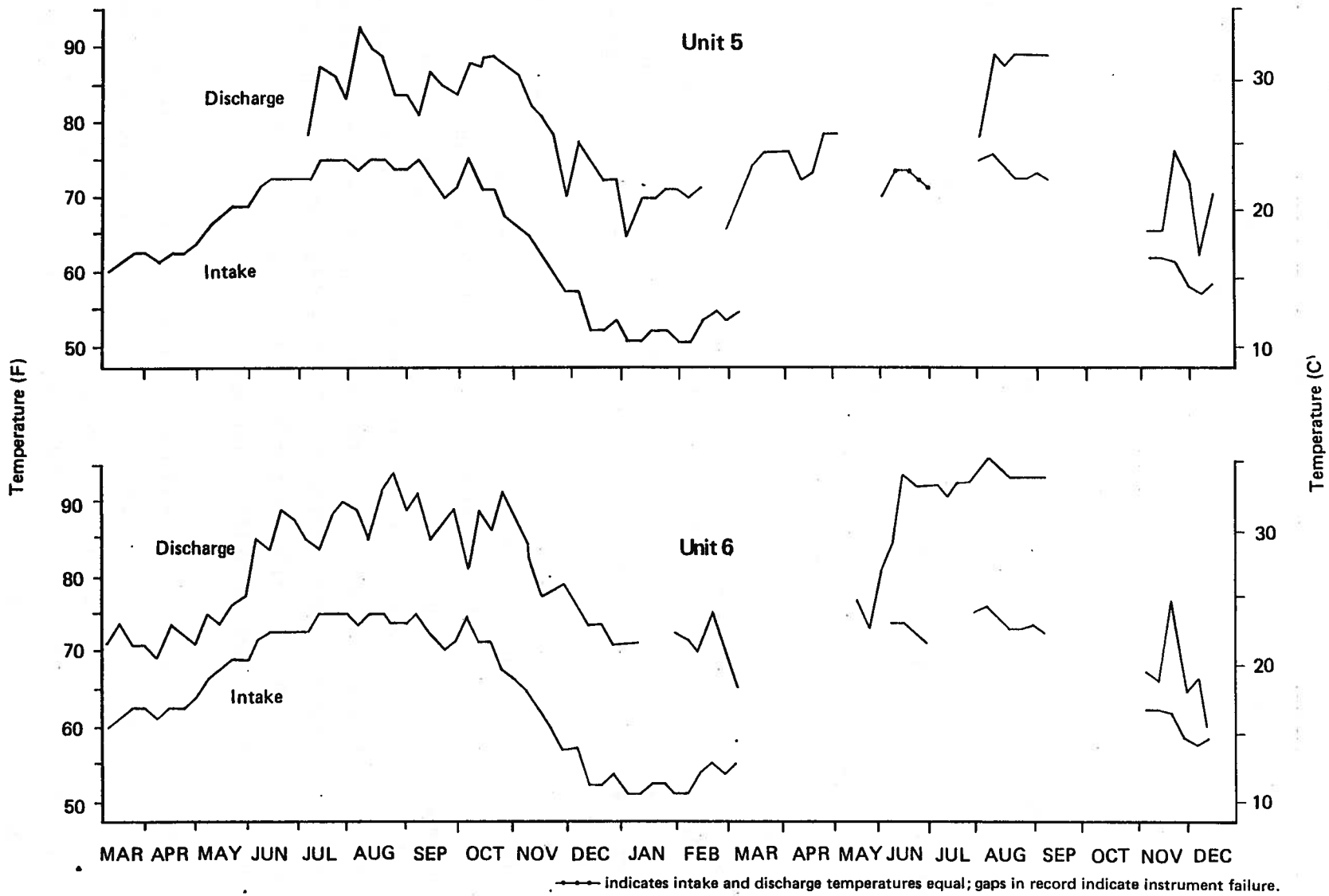

Figure 2-9. Average weekly intake and discharge temperatures at the Pittsburg Power Plant Units 5 and 6, March 1978–December 1979.

Makeup water pumps are generally used only when the Unit 7 turbine is in operation, so the number of hours of turbine operation in a given month is representative of the number of hours that two of the three makeup water pumps were operated in that month. Estimates of the total makeup water volumes pumped during each month from March 1978 through December 1979 are summarized in Table 2-7. The percentage of continuous normal operation ranged from 0 to 95 percent during the study period and averaged 62.5 percent.

Unit 7 makeup water pumps withdraw water from the Units 1-4 intake structure, so the Units 1-4 approach velocities include Unit 7 makeup water pump operation. Temperature was not measured at the Unit 7 intake and discharge, because no survival is assumed for organisms entrained through its cooling system. Temperature is generally used in determining entrainment survival rates, and where no survival is assumed, temperature data are not necessary.

2.1.4 Operational History and Projected Operation

Historically, during normal operation all units at the Pittsburg Power Plant seldom operated at full load. Over the 5-year period 1970-1974, for example, the average monthly capacity factor was approximately 40 percent for Units 1-4 and 50 percent for Units 5 and 6. For this same period, total generation for Units 1-6 equaled or exceeded 570 MWe (44 percent of capacity) only 50 percent of the time, and 1,000 MWe (78 percent of capacity) 10 percent of the time. The average monthly capacity factor for Units 1-6 over the 9-1/2-year period January 1967 - July 1976 was approximately 50 percent. For the 1-year period June 1976 - May 1977, the operating capacity for Units 1-6 averaged 70 percent; monthly operating capacity factors ranged from 52.8 to 87.6 percent. During the period March 1978 - December 1979, the mean monthly capacity factors averaged 65, 60, 60, and 62 percent for Units 1, 2, 3, and 4 (Table 2-3). Mean monthly capacity factors averaged 40, 59, and 41 percent for Units 5, 6, and 7 between March 1978 and December 1979 (Table 2-5).

Estimates of projected future operation of the Pittsburg Power Plant are presented in Table 2-8. These are based on a PGandE computer simulation model developed for long-term electric resource planning. The forecast incorporates expected population increases, electrical demand, and currently scheduled additions of new electrical generating capacity, including nuclear, hydroelectric, cogeneration, conventional fossil-fuel steam generation, geothermal, and wind facilities. The forecast also

TABLE 2-7 MONTHLY COOLING WATER VOLUMES AND PERCENTAGES OF NORMAL CAPACITY FOR UNIT 7 OF THE PITTSBURG POWER PLANT, MARCH 1978 - DECEMBER 1979

Month	Volume Unit 7		Percent Normal Capacity
	$ft^3 \times 10^6$	$m^3 \times 10^6$	
1978			
March	93.3	2.6	77.4
April	66.6	1.9	57.1
May	111.0	3.1	92.1
June	108.4	3.1	92.9
July	106.3	3.0	88.2
August	86.2	2.4	71.5
September	107.7	3.1	92.4
October	64.8	1.8	53.8
November	94.5	2.7	81.0
December	108.1	3.1	90.0
1979			
January	40.2	1.1	33.3
February	0	0	0
March	101.1	2.9	83.9
April	106.4	3.0	91.3
May	84.9	2.4	70.4
June	68.5	1.9	58.8
July	114.4	3.2	74.9
August	114.7	3.3	95.2
September	69.8	2.0	59.9
October	0	0	0
November	6.2	0.2	5.3
December	0	0	0

TABLE 2-8 PROJECTED[a] ANNUAL CAPACITY FACTORS FOR UNITS 1-7 OF THE PITTSBURG POWER PLANT, 1982-1990

Year	Unit 1	2	3	4	5	6	7
1982	0	18.9	17.9	19.9	47.1	42.9	18.1
1983	0	8.9	8.8	8.8	38.3	19.3	9.3
1984	0	8.7	4.2	8.7	18.0	12.5	3.9
1985	0	8.9	11.4	8.9	24.7	18.5	4.0
1986	0	11.1	11.8	11.7	27.5	30.9	10.1
1987	0	16.0	11.9	16.1	34.3	36.5	11.3
1988	0	17.8	14.0	18.4	55.6	37.7	10.7
1989	0	11.1	10.9	11.0	44.0	41.3	9.9
1990	0	24.1	18.2	15.7	45.6	41.9	11.3

(a) Based on average water year conditions and PGandE's 4 December 1980 electric resource plan.

Note: These capacity factors represent unit operating levels calculated from current load forecasts and existing and planned generating capacity. Any of these units may be operated at a higher or lower capacity factor, depending on load growth, other available resources, availability of energy from outside of the PGandE planning area, and the need to serve PGandE load and or the loads of PGandE's neighbors under conditions of major equipment outage. A projected capacity factor of 0 should therefore not be taken to mean that a unit has been retired from service.

includes electrical power purchases, consideration of generating reliability and reserve margin, fuel conservation, forced outages, and scheduled maintenance.

The forecast average annual capacity factors (Table 2-8) for the plant indicate that operation of all units is expected to decrease from present levels through the period 1982-1990. Projected future operation during the spring and early summer months (April-July), when biological production in the Bay-Delta is high, is discussed in Section 6.3.

2.2 PHYSICAL, CHEMICAL, AND BIOLOGICAL CHARACTERISTICS OF THE SOURCE WATER BODY

San Francisco Bay and the delta of the Sacramento and San Joaquin rivers (the Bay-Delta) form a complex of interconnected embayments, rivers, channels, sloughs and marshes. The Bay-Delta has been defined as the Delta, into which the Sacramento, San Joaquin, and smaller rivers flow, north San Francisco Bay, comprising Suisun Bay, Carquinez Strait, and San Pablo Bay, and the central and south portions of San Francisco Bay. The division between the central Bay and the south Bay is taken to be a line from Hunters Point to the south end of Alameda Island.

The Bay-Delta receives runoff from a 163,000-km^2 (64,000-mi^2) drainage basin which covers 40 percent of the land area of California (Conomos 1979). Ninety percent of the inflow to San Francisco Bay comes through the Delta, the rest from local drainages. The basin is drained in the north by the Sacramento River and its tributary streams, in the south by the San Joaquin River and its tributaries, and in the east by the Mokelumne, Cosumnes, and Calaveras rivers. The north, south, and east river systems contribute 80, 15, and 5 percent, respectively, of the inflow to the Delta (Conomos 1979). The Delta extends eastward from the western tip of Chipps Island to Hood on the Sacramento River, Stockton on the San Joaquin River, and Interstate Route 5 on its eastern border.

Delta inflow is highly seasonal, consisting of rain runoff during the fall and winter months and snowmelt runoff in spring and summer. The major rivers are dammed for flood control, water storage, and hydroelectric power. Water released from the dams reaches the Delta, where it is used to support local agriculture and diverted in large quantities for irrigation, industrial, and domestic uses.

The physical and chemical characteristics of the Bay-Delta range from marine near the Golden Gate to intermittently freshwater in the upper reaches of the north Bay and the Delta. Natural factors that affect the abundance and species composition of the aquatic community in the system include tidal flushing and current patterns, freshwater inflow, and fluctuations in salinity and temperature. At the Pittsburg Power Plant, on the south shore of eastern Suisun Bay just below the confluence of the Sacramento and San Joaquin rivers, Suisun Bay is approximately 4,000 ft (1,220 m) wide and has a maximum depth of 70 ft (21 m) at mean lower low water (MLLW). A cut of the San Joaquin River, New York Slough, enters Suisun Bay near the plant. The original character of the Delta in the vicinity of the plant has been modified by human activities: marinas, wharves, and factories now occupy much of the shore, particularly to the east of the plant.

The Delta was once a natural freshwater tidal marsh overlying a deep layer of peat, but shortly after 1850, the year California became a state, efforts were made to reclaim the Delta marshlands. Delta farms were formed by building levees to combat the tides and then draining the marshlands. The levees were gradually built higher and higher to withstand frequent floods. The peat-rich farmland inside the levees subsided, leaving some tracts as much as 6 m (20 ft) below the water level in the channels. Reclamation of the Delta was essentially complete by 1930.

Today's Delta consists of about 60 islands separated by 1,100 km (700 mi) of interconnected waterways (Figure 2-1). The Delta encompasses nearly 3,000 km^2 (1,150 mi^2), of which about seven percent remains as surface water (SWRCB 1978). Most of the Delta waterways are bordered by the steep banks of levees, but some are edged by narrow stretches of intertidal marsh. The waterways are shallow, few exceeding 30-40 ft in depth, and range in width from a few hundred feet to a mile (Kelley 1966). Deepwater ship channels are maintained for shipping, serving the inland ports of Stockton and Sacramento on the eastern and northern borders of the Delta. The Delta waterways are also extensively used for recreation (e.g., boating, fishing, waterskiing). Almost all the land in the central, northern, eastern, and southern Delta is committed to agriculture, but the western Delta is largely urbanized and industrialized. The south shore of the San Joaquin River in the western Delta is the site of the cities of Pittsburg and Antioch, as well as several industries, including chemical, steel, paper manufacturing, and electric power production.

The arrangement of the Delta waterways is complex (Figure 2-1). Near Sacramento, the man-made Sacramento Deepwater Ship Channel branches from the Sacramento River and extends southward 42 km (26 mi) before rejoining the Sacramento River near Rio Vista. Sutter and Steamboat sloughs split off the Sacramento River and form islands in the northwestern Delta, and then rejoin the Sacramento near Rio Vista. Below the upper end of these sloughs is the Delta Cross Channel, which connects the Sacramento and Mokelumne rivers via Snodgrass Slough. The Delta Cross Channel, along with Georgiana Slough, located about 0.8 km (0.5 mi) downstream, convey water to the interior Delta channels, from which it flows to the State Water Project (SWP) and Central Valley Project (CVP) export pumping facilities in the southwestern Delta. Further downstream on the Sacramento River, south of Rio Vista, Three Mile Slough connects the Sacramento River to the San Joaquin River.

The San Joaquin River, flowing through the Delta from Vernalis to its confluence with the Sacramento River near Collinsville, divides the central and southern portions of the Delta. Downstream of Vernalis, two major channels of the San Joaquin River, Old River and Middle River, branch from the main channel and convey water toward the western Delta during periods of high flow. Under regulated flow conditions, these channels, principally Old River, convey water toward the Delta (SWP) and Tracy (CVP) pumping plants.

In the western Delta, the main channels of the Sacramento and San Joaquin rivers join and form a single channel which enters Suisun Bay at Chipps Island. Montezuma Slough branches from the north side of the Sacramento River near Collinsville, loops around Suisun Bay, and joins the northwestern portion of Grizzly Bay, an extension of Suisun Bay.

During periods of high freshwater flow, floodwaters are diverted near Sacramento out of the main channel of the Sacramento River into the Yolo Bypass, a broad, leveed flood plain which extends southward, parallel to the Sacramento Deepwater ship Channel, toward Rio Vista. In winter, during sustained high flows (>50,000 cfs; 1,400 m^3/sec), the water flowing into the Delta through the Yolo Bypass is nearly equal to the flow in the main channels (Conomos 1979).

Suisun Bay

Water flowing out of the Delta toward the sea passes into Suisun Bay (Figure 2-1). Suisun Bay, extending from Chipps Island in the east to the Benicia Bridge in the west, is the smallest of the major bays of the San Francisco Bay complex, covering an area of about 100 km^2 (40 mi^2). Except for a shipping channel along its south shore, dredged to a depth of 9 m (30 ft), Suisun Bay is extremely shallow.

Over one-third of Suisun Bay is less than 2 m (6 ft) deep at mean lower low tide (MLLW) (Kelley 1966). North of the shipping channel lie two shallow extensions of Suisun Bay—Honker and Grizzly bays. The southern shore of Suisun Bay is partly urbanized and industrialized. Industries represented include chemical manufacturing and petroleum refining. The northern shore is less developed, consisting largely of an extensively managed wetland area, Suisun Marsh.

2.2.1 Hydrology

The surface and groundwater hydrology of the Bay-Delta is complex, involving a myriad of tributary inflows, unmeasurable outflows, dynamic flow distribution patterns, and variations in precipitation and stream flows (SWRCB 1978). Dams and canals constructed during the past 35 years have altered the natural hydrology of the Bay-Delta so that high winter and spring flows are reduced by upstream storage and low summer and fall flows are increased by storage releases. Seasonal variations and dry year - wet year fluctuations in flow have been considerably dampened. Saltwater intrusion into the Delta has become a major concern because of its effects on water quality and the fish and wildlife of the Bay-Delta.

Historical Perspective

Before the 1860s, saltwater intrusion into the Delta was a rare occurrence. However, from the 1860s onward, changes in the character of the Delta and the Central Valley combined to alter the natural hydrology of the Bay-Delta and consequently increase the frequency and extent of saltwater intrusion.

Factors increasing saltwater intrusion were levee construction and channelization, sedimentation, and export of water for irrigation. Levees were constructed along the

Sacramento River and its tributaries to control and channelize flood waters. The extensive tidal marshlands that had covered the Delta were converted to a maze of interconnected, leveed waterways surrounding reclaimed farmlands. Flood waters that had previously overflowed into the Delta marshlands and flood plains of the Central Valley began to run out into San Francisco Bay at a much faster rate than in pre-reclamation times, permitting greater saltwater intrusion after the flood waters had passed (Kahrl 1979). The vast amounts of hydraulic mining debris carried down the rivers from the Sierra Nevada gold mining country choked and narrowed the Delta waterways, and raised their beds by as much as 5 m (15 ft). Levees had to be built higher and higher, further channelizing the spring flood waters and increasing the potential for summer and fall salt intrusion. Around the turn of the century, large amounts of water began to be withdrawn from the Sacramento and San Joaquin rivers to irrigate the fertile plains of the Central Valley. Irrigation reduced summer Delta outflows and thereby further compounded the developing salt intrusion problem.

Significant salt intrusion occurred periodically until 1944, when the Shasta Dam on the upper Sacramento River was completed as part of the federal government's Central Valley Project (CVP) (Kahrl 1979). The resulting upstream storage permitted the reduction of winter and spring flows and the increasing of summer and fall flows, meeting the growing demands for fresh water during the summer months and at the same time reducing the problem of salt intrusion.

However, demands for fresh water continued to grow, and the opening in 1951 of the Delta Cross Channel near Walnut Grove and the CVP pumping plant in the southern Delta permitted the export of large quantities of water to the San Joaquin Valley and dramatically altered the flow patterns in the Delta waterways. Other upstream storage reservoirs were subsequently completed on the Trinity River and on tributaries of the Sacramento and San Joaquin rivers, under both the CVP and the State Water Project (SWP). In 1967, a SWP pumping plant in the southern Delta began exporting water to the San Joaquin Valley and southern California.

Present Delta Hydrology

The operation of the SWP and CVP to meet certain water quality objectives and export needs has resulted in higher Delta outflows during dry periods than would have otherwise occurred. Since the SWP and CVP began regulating flows, the salt front

(1,000 ppm chloride) has intruded into the Delta earlier in the year but has not extended as far upstream as under pre-project conditions (SWRCB 1978).

Flow conditions within the Delta have also been greatly influenced by operation of the CVP and SWP. With the increase in exports since 1951, and particularly since 1967, when the SWP export facilities began pumping, flow patterns in many of the Delta waterways have been altered. Flow reversals normally occur each year in Old River and Middle River, between the San Joaquin River in the central portion of the Delta and the export pumps near Tracy. The transfer of Sacramento River water into the central Delta via the Delta Cross Channel does not always meet export needs, resulting in flow reversal around the lower end of Sherman Island from the Sacramento River to the San Joaquin River during periods of low San Joaquin River flow, high exports, and high Delta consumptive uses. Such reverse flows can also occur with high Sacramento River flows, high exports, and low San Joaquin River flows. In addition, flow reversal in the main channel of the San Joaquin from Stockton south to the Old River bifurcation regularly occurs when export rates are more than five times as high as San Joaquin River inflows at Vernalis (SWRCB 1978).

The inflow of water to the Delta averages about 20 million acre feet (Maf) per year, but this varies substantially between dry and wet years. At the year 1980 level of development, about 6.1 Maf of the average inflow is exported by the CVP and SWP facilities, 1.6 Maf is consumed in the Delta (evaporation plus agricultural use), and the rest (12.3 Maf) flows out of the Delta into San Francisco Bay. At the year 2000 level of development, the volume exported is expected to increase to 7.6 Maf and Delta outflow will decrease to 10.5 Maf.

Delta outflow is not directly measurable because of the difficulties introduced by tidally reversing flows and the complex geometry of the channels. In lieu of actual measurements, the Delta Outflow Index (DOI) has been developed to estimate outflow. DOI equals the Delta inflow (from the the Sacramento and San Joaquin rivers) minus the water exported, used in the Delta, or lost by evaporation. The DOI is subject to substantial error, primarily during high-flow periods, because it excludes inputs from the Yolo Bypass and east-side streams.

Estimated Delta outflow at Chipps Island varies seasonally. During the dry summer months it generally ranges from 40 to 110 m^3/sec (1,500-4,000 cfs) but occasionally

falls below 28 m^3/sec (1,000 cfs) (DWR unpublished data). During the wet winter and spring months, Delta outflow usually ranges from 2,800 to 5,700 m^3/sec (100,000-200,000 cfs) and occasionally exceeds 10,600 m^3/sec (375,000 cfs) (DWR unpublished data). During dry summer months, exports and Delta consumptive uses exceed the Delta outflow, resulting in the alteration of flow direction in the Delta channels described above. The 10-year monthly averages and ranges are shown in Figure 2-10 for the period 1969-1978, along with the means and ranges of May flows for the period 1969-1979. The individual monthly means and ranges for the months when larval and juvenile fish are abundant, April-July, are listed for the same period in Table A-6, Appendix A.

Suisun Bay Hydrology

As an estuarine ecosystem the hydrology of Suisun Bay is of critical importance to the Bay-Delta biological environment, because the maximum extent of salt intrusion usually occurs in or near Suisun Bay during the biologically productive spring and early summer months. The position of the fresh water - salt water mixing zone and the characteristics of the mixing process affect the distribution and abundance of estuarine organisms in the Bay-Delta (Arthur and Ball 1979).

Fresh water flowing downstream from the Delta tends to flow over the denser, more saline water driven upstream by tidal currents, resulting in a two-layered flow. Although vertical mixing of fresh and salt water occurs, there are nevertheless two layers of opposing net flow separated by a plane of no net motion (Figure 2-11).

Suspended particulate matter is entrapped and accumulates in the upstream end of the fresh water - salt water mixing zone (Figure 2-11). It has been theorized that the causes of this entrapment are the increased flocculation, aggregation, and/or settling of suspended materials at conductivities above 1 mmho (0.6 ppt salinity). Also, materials are thought to be suspended by upward currents at the upstream end of the mixing zone and carried downstream in the seaward-flowing upper layer, where they settle into the lower layer and are subsequently transported upstream in the landward-flowing bottom currents.

The biological significance of this entrapment zone is that nutrients and many estuarine organisms also tend to concentrate there, potentially making it one of the

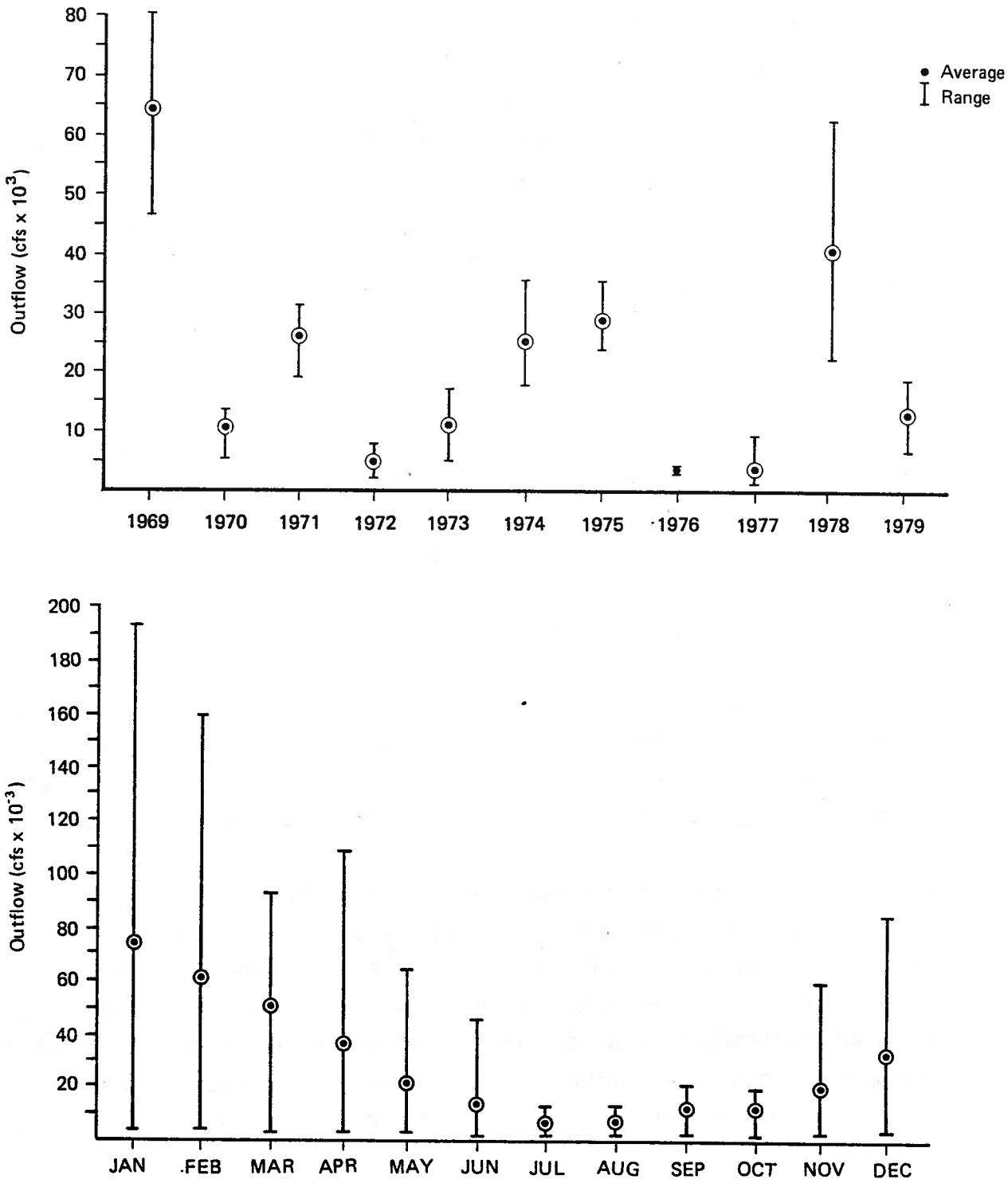

Figure 2-10. Averages and ranges of May Delta outflows (estimated) at Chipps Island,* 1969-1979, and 10-year (1969-1978) averages and ranges of monthly mean Delta outflows. (Data from Calif. Dep. Water Resour., unpublished.)

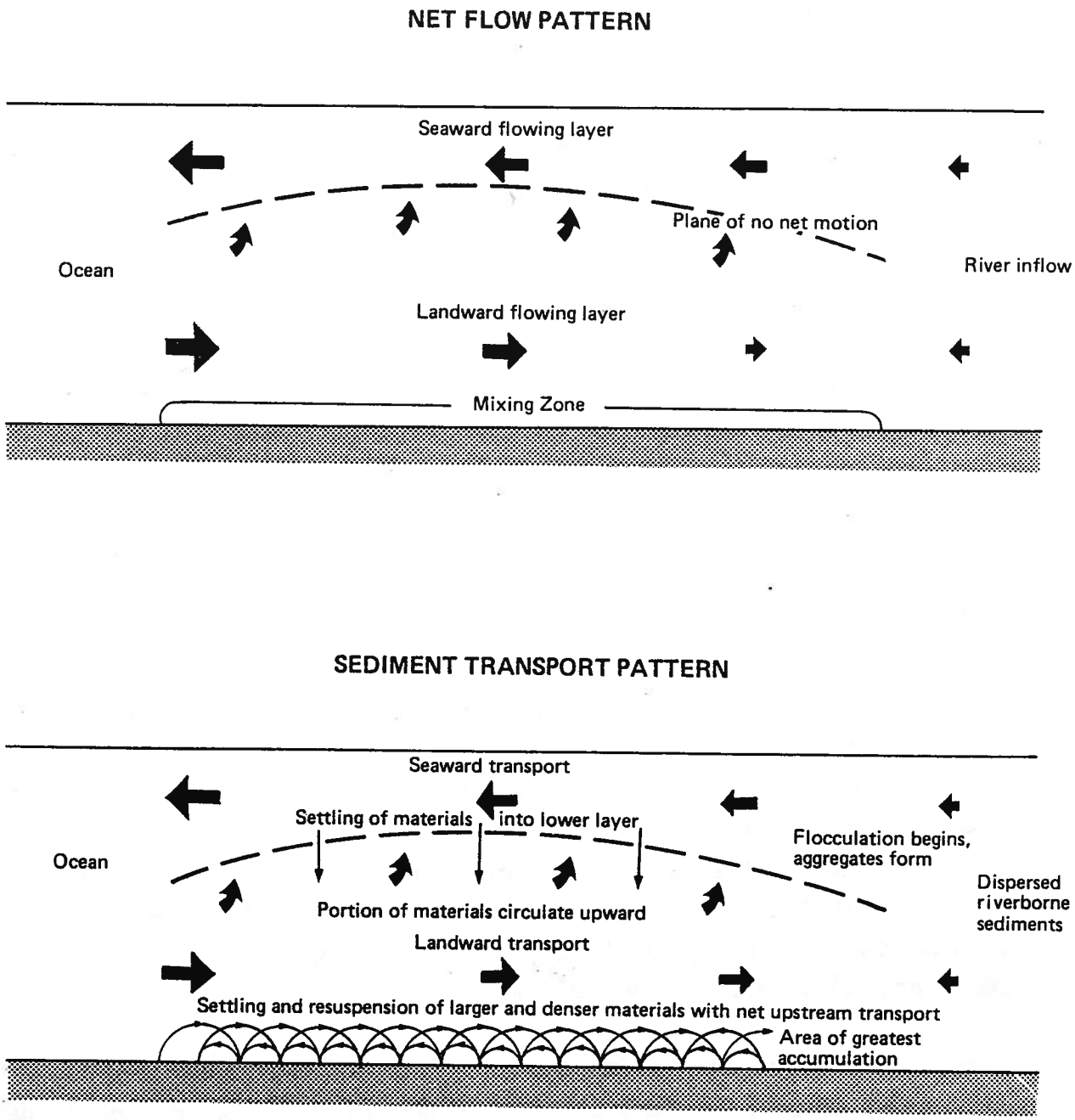

Figure 2-11. Theoretical patterns of net flow and transport of suspended materials in an estuary with two-layered flow and vertical mixing. (Adapted from CDF&G et al. 1975.)

most productive biological regions of the Bay-Delta. Phytoplankton, certain zooplankton (including the opossum shrimp Neomysis mercedis and the copepod Eurytemora affinis), and the larvae of the important game fish striped bass are among the organisms known to concentrate in the entrapment zone.

The position of the entrapment zone within the Bay-Delta has been shown to affect biological productivity (Arthur and Ball 1979). When the entrapment zone, demarcated by surface conductivities of 2-10 mmho (1-6 ppt salinity), occurs in shallow eastern Suisun Bay, the quantity of phytoplankton and certain other estuarine biota is enhanced. When the entrapment zone occurs upstream of Suisun Bay, on the other hand, biological productivity is low (Arthur and Ball 1979).

The mixing zone is driven seaward and compressed by high freshwater flows and driven landward and expanded by low freshwater flows (Figure 2-12). Thus, the position of the entrapment zone is largely a function of Delta outflow. At Delta outflows less than 290 m^3/sec (10,000 cfs), the entrapment zone is located well up into the Delta. Flows greater than 1,400 m^3/sec (50,000 cfs) drive the entrapment zone downstream into Carquinez Strait or San Pablo Bay. At extremely high flows ($\geq$5,600 m^3/sec; 200,000 cfs), the zone many reach central San Francisco Bay.

The hydrology of the Delta near the Pittsburg Power Plant is influenced by tidal action and freshwater flow. Tidal flow entering the Delta from Suisun Bay influences both the Sacramento and San Joaquin river systems. Tides are semidiurnal, with two flood and two ebb phases per 24.8-hour tidal day. Mean tidal range at Pittsburg is about 1 m (3.3 ft). The average tidal flow in front of the plant is approximately 170,000 cfs (4,800 m^3/sec) (PGandE 1970b). Average current velocities offshore from the site are near 1.2 knots (1.7 fps; 62 cm/sec) on flood tide and 1.3 knots (2.2 fps; 67 cm/sec) on ebb tide (PGandE 1973).

Current velocity patterns are influenced by tidal movement, Delta outflow, and wind. The effects of Delta outflow and wind are usually far outweighed by tidal movement. Near the Pittsburg Power Plant mean summer net flows are about 1/30 of the mean tidal flow (Chadwick and Stevens 1971). The effective volume of water that moves back and forth past the area depends on tidal conditions and freshwater outflow, and has been assumed to be equal to the tidal prism, i.e., the quantity of water passing the plant between successive tidal phases minus the Delta outflow, calculated to be approximately 3 billion ft^3 (85 million m^3) (Tetra Tech 1976).

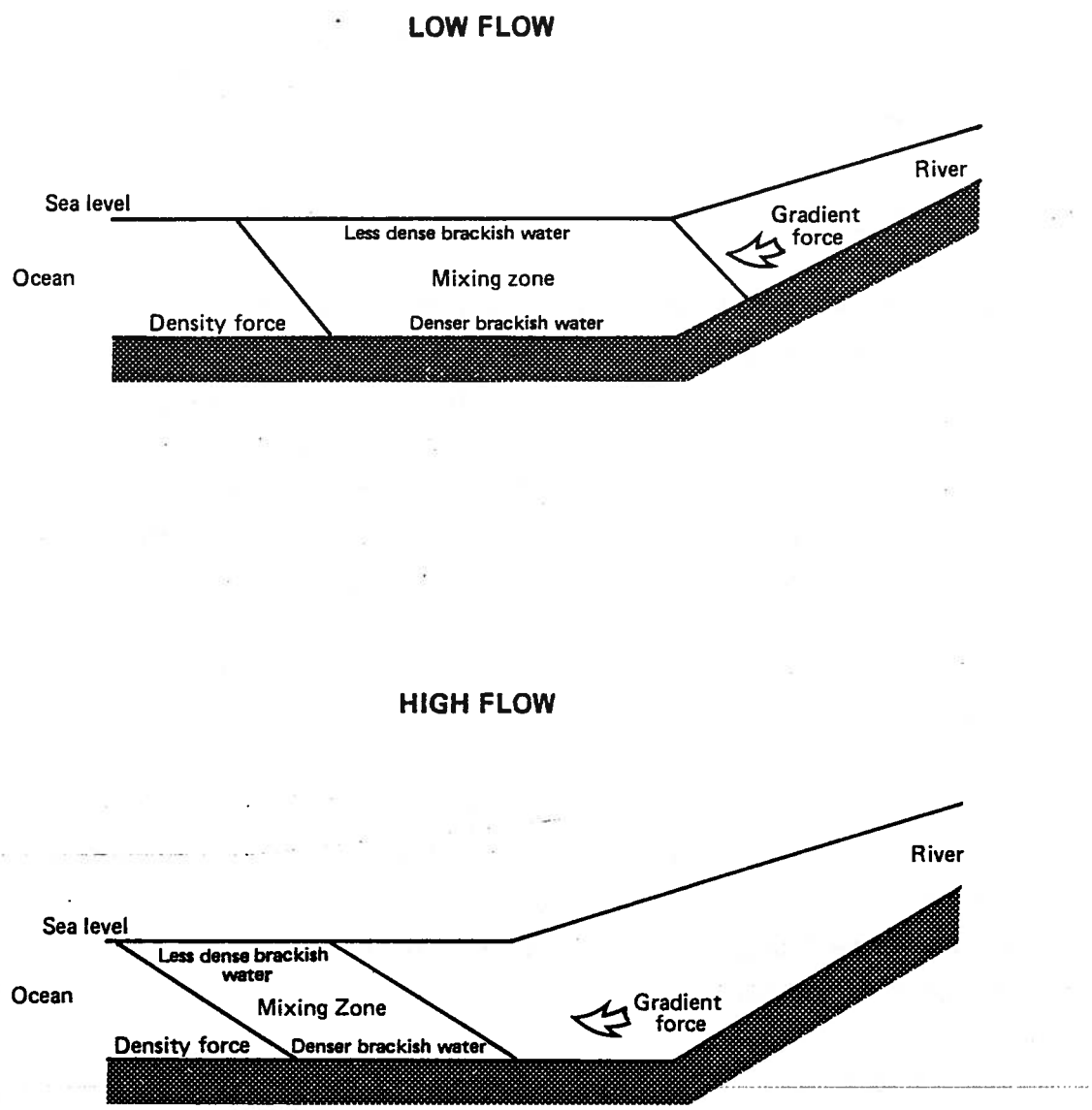

Figure 2-12. The effect of freshwater flow on the mixing of fresh and salt water in an estuary with two-layered flow and vertical mixing. (Adapted from CDF&G et al. 1975.)

These currents suspend peat and other detrital material from islands in the Delta, increasing the suspended solid content in the water body. Because of the relatively large particle size of the peat, it tends to clog the intake screening systems used at the cooling water intakes of power plants located in this area, causing operating and maintenence problems, primarily during the spring and fall.

2.2.2 Morphology

Eastern Suisun Bay in the vicinity of the Pittsburg Power Plant is generally shallow (<30 ft; 9 m) except for two deep channels which have been dredged for shipping access to the Delta and Central Valley. The main shipping channel for the Sacramento River lies near the south bank of the bay; the shipping channel serving the San Joaquin River branches off the main channel below Browns Island and passes through New York Slough before entering the main channel of the San Joaquin River. The lower Delta contains many islands composed of peat and silt deposits that accumulated in shallow marshes. In most instances the islands are surrounded by marshes or mudflats, except where they are exposed to strong river currents that scour the exposed banks or levees.

The shoreline next to the plant has been altered by human activities over the past fifty years. Industrial developments, including a steel mill, chemical manufacturers, and paper mills, dominate the shoreline from the city of Pittsburg to the Antioch Bridge. Several large marinas are located on the shore both east and west of the Pittsburg Power Plant.

The nearshore area at the Pittsburg Power Plant is characterized by pilings, riprap, and bulkheads. The nearshore bathymetry is moderately shallow (<30 ft; 9 m) except for the deepwater channel several hundred yards offshore (see Figure 2-3). Bottom substrates consist mainly of unconsolidated mixed and interbedded alluvial sediments, i.e., clays, silts, sands, and gravel (Steitz 1975). Maintenance of the channels for shipping traffic requires periodic dredging in the area. In addition, dredging of the approach channels to the power plant intakes has been necessary in the past. The substrate characteristics, sedimentation, and dredging activity all influence benthic and epibenthic invertebrates and plants in the area near the Pittsburg Power Plant.

2.2.3 Water Quality

Water quality affects not only the aquatic life of the Bay-Delta but also the agricultural, industrial, municipal, economic, recreational, and aesthetic uses of the water resource. It is likely that the water quality of San Francisco Bay began to deteriorate shortly after the area was settled in the mid-1800s: untreated domestic and industrial wastewater was routinely discharged into the Bay through the 1940s. About that time, shellfishing in the Bay was quarantined because of contamination by municipal and industrial sewage. Inflow of wastes resulted in low levels of dissolved oxygen in the south Bay, the Petaluma and Napa rivers flowing into San Pablo Bay, and many parts of the Delta (Kaiser Engineers 1969). During the 1950s sewage treatment facilities were constructed, resulting in some improvement in water quality.

The south Bay has the worst water quality in the Bay-Delta. Because there is little freshwater flow into the south Bay other than wastewater, except during high runoff periods, flushing is minimal and pollutants may be retained in the system for long periods. In some portions of the south Bay, low dissolved oxygen concentrations preclude survival of many fishes and invertebrates, and coliform bacteria counts often exceed standards for water contact recreation (SWRCB 1975). High concentrations of toxic heavy metals such as lead, copper, mercury, cadmium, zinc, and chromium characterize the bottom sediments, and pesticide and hexane extractables also are reported to be accumulating (SWRCB 1975).

Proceeding northward to the central Bay, the quality of the water, bottom sediment, and biotic conditions gradually improve, although water and sediment quality is poor around port facilities and points of major waste discharge (SWRCB 1975). Benthic animal populations are sparse in the vicinity of the Hunters Point and Oakland-Alameda Port facilities, which may be associated with a concentration of petroleum products and heavy metals (SWRCB 1975).

In the northern arm of the Bay, encompassing San Pablo Bay, Carquinez Strait, and Suisun Bay, water quality is superior to that in the south Bay. Despite large quantities of municipal and industrial wastes discharged into the northern reach, tidal flushing and freshwater flow prevent extensive environmental degradation. The northern reach is subject to considerable nutrient loading from Delta outflow and

local wastewater discharges, and during the summer months algal blooms are common in Suisun Bay, but nuisance conditions are rare.

In the Delta, the major water quality problem is salt intrusion. Salinity levels increase in the western and interior Delta during the summer and fall—and occasionally in winter and spring in dry years. Also, irrigation of the Delta islands and surrounding lands during the spring and summer causes the accumulation of salts that during fall and winter are leached into the Delta channels, adding to the salinity problem. Salt intrusion adversely affects the agricultural, industrial, and municipal water users, and can adversely affect the aquatic life in the western Delta, Suisun Bay, and Suisun Marsh.

Low dissolved oxygen concentrations in the San Joaquin River near Stockton in the fall and in the eastern Delta deadend sloughs during the summer and fall constitute another water quality problem in the Delta. This problem is attributed to biochemical oxygen demand (BOD) from municipal and industrial wastes and to excessive algal growths. The major sources of wastes are the Sacramento and Stockton metropolitan areas. A depletion of dissolved oxygen in the San Joaquin River below Stockton during the fall months was associated with the low net downstream flows in the river, high water temperatures, and the season when the canneries imposed a fairly heavy load on municipal waste disposal systems (Kaiser Engineers 1969). This has been alleviated by recent improvements in waste treatment facilities.

Water Temperature, Salinity, and Dissolved Oxygen Near the Pittsburg Power Plant

The aquatic environment near the Pittsburg Power Plant fluctuates between a typically freshwater environment in periods of high freshwater inflow and a brackish-water environment when freshwater inflow is low (<10,000 cfs; 280 m^3/sec). Seasonal changes in water temperature, salinity, and dissolved oxygen concentration affect species composition and abundance of the aquatic community in the area. Water quality in the vicinity of the plant, as in the Delta in general, is influenced primarily by freshwater inflow and tidal circulation.

Ambient water temperatures in the Delta near Pittsburg are lowest in December or January (7 C; 44 F) and highest in July or August (25 C; 75 F). In 1978 and 1979,

intake temperatures at Units 5 and 6 ranged from 6 C (43 F) to 28 C (83 F) (Figure 2-9). The highest temperatures were probably increased somewhat by recirculation.

The entrapment zone marks the upper boundary of salinity intrusion in the Bay-Delta estuary. Several models have been developed to predict the extent of salinity intrusion into the Delta over the range of expected Delta outflow rates. The level of saltwater intrusion is not substantial in the area until Delta outflow drops below 400 m^3/sec (14,000 cfs). In normal years, flows below this level can be expected in the period June-September. In dry years, however, flows may never exceed 400 m^3/sec: in 1976 and 1977 average monthly Delta outflow ranged from 62 to 260 m^3/sec (2,200-9,300 cfs), and conductivities at Department of Water Resources (DWR) Station D-10, located west of the confluence of the Sacramento and San Joaquin rivers between Chipps and Mallard islands, ranged from 80 to 21,700 µmho (DWR 1978, 1979), roughly equivalent to 0.1-12.6 ppt salinity. In 1978-1979, monthly mean conductivities at the Units 5 and 6 intake (Figure 2-13) ranged between 230 and 8,700 µmho (0.1-5 ppt); the extreme measurements recorded were 165 and 14,100 µmho (0.1-8.2 ppt) (Appendix A).

Dissolved oxygen concentrations near the power plant generally range from 7.8 to 11.2 ppm, between 82 and 113 percent of saturation (DWR 1976, 1977, 1978, 1979). In 1978-1979, monthly means at the Units 5 and 6 intake ranged between 7.6 and 9.3 mg/liter (Appendix A). The extreme measurements recorded were 6.3 mg/liter in January 1979 and 10.9 mg/liter in March and October 1979. This range of concentrations is above levels that would limit biological productivity or adversely affect the aquatic organisms that frequent the lower Delta.

2.2.4 Water Use in the Delta

The Sacramento-San Joaquin estuary serves as a water source for local agriculture, for industries and municipalities, and for the state and federal diversion facilities. The water entering the Bay-Delta is subject to considerable control by man. Water of the Sacramento and San Joaquin rivers and their tributaries, and even of some rivers, like the Trinity, that would ordinarily flow to the ocean, is regulated according to plan to provide hydroelectric power, water for irrigation and for cities, and maintenance flows of fresh water through the Bay-Delta.

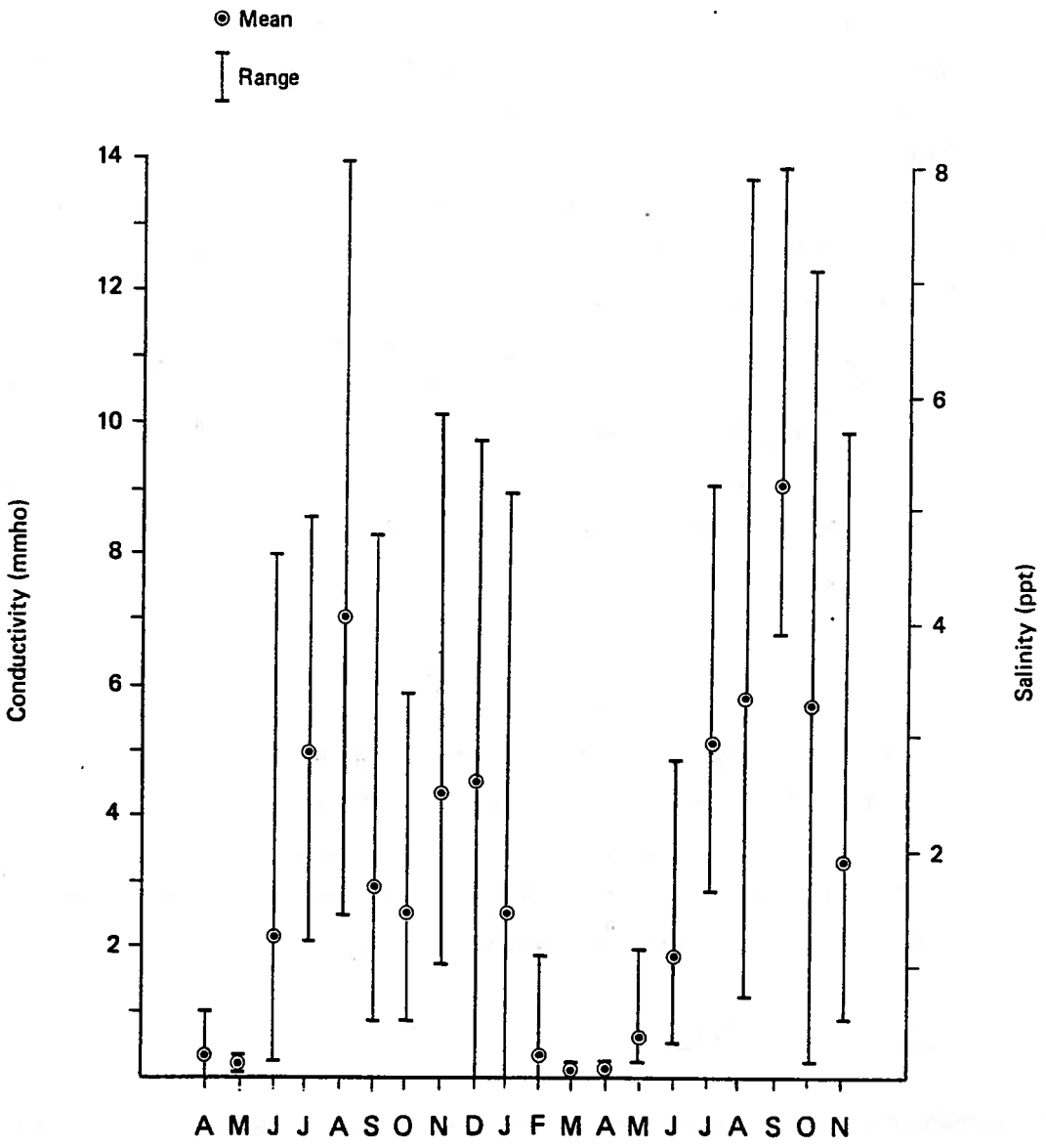

Figure 2-13. Monthly means and ranges of conductivity (salinity) at the Pittsburg Power Plant, April 1978—November 1979.

The mechanism for the control of these rivers is a pair of independent but coordinated and cooperative water systems: the federal Central Valley Project (CVP) and the California State Water Project (SWP). The balance between diversion of fresh water from the system and water storage and release from the reservoirs plays a critical part in the regulation and control of physical, chemical, and biological processes in the estuary. The release of stored water during the summer and fall dry seasons has considerably altered the freshwater flow and salinity regimes in the Delta. At the same time, diversions from the estuary of freshwater inflow have reduced the total freshwater input to the Bay-Delta and altered the patterns of flow and salinity there. Because the freshwater flow patterns are so important to the physical, chemical, and biological processes of the estuary, the ecology of the Bay-Delta is an important consideration in the regulation of the water projects by the State Water Resources Control Board (SWRCB).

The major beneficial uses to which Delta waters are put are power plants, state and federal water diversions, and local agricultural diversions. Water use includes both comsumptive and nonconsumptive use. Power plants and some industries use water for cooling purposes. Once-through cooling is a large-volume nonconsumptive use; closed-cycle cooling, which required smaller volumes of water to replenish evaporated water, is a consumptive use. Most of the major water diversions and local agricultural diversions are consumptive, although some of the locally diverted water is eventually returned as discharge or runoff.

The Delta, which comprises between 40,000 and 50,000 surface acres of water (Skinner 1972), provides municipal, domestic, and industrial water supplies and supports a variety of other beneficial uses. Delta channels, particularly the Sacramento and San Joaquin rivers and two deep-water channels, are used for commercial shipping. Over 700 mi of navigable, interconnected waterways and over 1,000 mi of shoreline provide a recreational resource for residents of large metropolitan areas such as the San Francisco Bay area, Sacramento, and Stockton. Water access to almost all areas is available to small craft, and the large fishery resource attracts many sport fishermen. The area also supports many species of waterfowl and wildlife. Much of the land area of the Delta is used for agriculture, and its waterways provide a readily available source of irrigation water (DWR 1974).

2.2.5 Aquatic Habitats

The area of the lower Delta from which the Pittsburg Power Plant's cooling water comes is characteristic of the estuarine transition zone that separates the upstream, freshwater Delta from the downstream, saltwater bays. The areas adjacent to the plant contain several types of aquatic habitat, including sloughs and cuts, shallow channel and shoal areas, and the main river channel. Together these habitats support a large and diverse biotic community which includes several commercially and recreationally important species of fish and waterfowl. The following sections describe these major habitats and the biological community of the Delta. Descriptions of the aquatic habitats and communities of the Delta, and in particular the area of the Pittsburg Power Plant, have been presented by Steitz (1975), PGandE (1973, 1977), and EA (1981a-h).

2.2.5.1 Sloughs and Cuts

There are many sloughs or cuts in the Pittsburg Power Plant study area. The major slough east of the power plant, New York Slough, is part of the shipping channel connecting eastern Suisun Bay and the San Joaquin River. Other, smaller sloughs and cuts are located on the western border of the site near Mallard Island.

Numerous man-made inlets were excavated for the mooring of grain boats, and have fallen into disuse in this century. Siltation has reduced the water depth in much of these areas. Extensive stands of emergent vegetation, particularly cattails and tule, border the cuts.

Common invertebrates that inhabit this area include amphipods, polychaetes, and small bivalves. Fish commonly found in this area include threadfin shad*, striped bass, splittail, tule perch, Sacramento squawfish, white catfish, yellowfin goby, carp, and Sacramento blackfish. In addition, the calm waters and shelter afforded by these cuts attract early life stages of many fish species.

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* Common names of fish used in this report follow Robins et al. (1980).

2.2.5.2 Shallow Channel and Shoal Areas

The area between the shore and the deepwater channel is characterized by water depths of less than 20 ft (6 m), a mud or mud-sand bottom, and reduced exposure to tidal and river currents. The inshore areas of the shoals are bordered by tule patches. Field studies conducted in the area showed that large numbers of small crustaceans, particularly the mysid Neomysis mercedis and amphipods of the genus Corophium, inhabit the area (EA 1981a, b, h) and are an important food supply for young-of-the-year striped bass and other small or young fish (PGandE unpublished c). These studies also found that the shallow river area in the vicinity of Pittsburg serves as a nursery area for striped bass and chinook salmon, as does much of the lower Delta and Suisun Bay.

Other fish found in this area include threespine stickleback, tule perch, Sacramento squawfish, gobies, inland silverside, starry flounder, splittail, carp, white catfish, and northern anchovy (EA 1981c). Northern anchovy are only found in the area when saltwater intrusion into the Delta extends up to Pittsburg.

2.2.5.3 Deep River Channels

The river channels are characterized by depths of more than 20 ft (6 m) and by strong tidal and river currents, 1.1-1.5 fps (30-46 cm/sec). These channels extend to within about 150 ft (50 m) of the shore near the Pittsburg Power Plant. The river bottom is generally composed of fine silts and sand. Invertebrates which inhabit this area include bottom-dwelling polychaetes, amphipods, and bivalves, and epibenthic shrimp, primarily Neomysis mercedis, Palaemon macrodactylus, and Crangon spp. The latter two taxa are harvested as part of a local bait fishery.

The open waters of the river serve as a migratory route for several species of anadromous fish that migrate to the freshwater reaches of the tributary rivers to spawn. These fish include striped bass, steelhead, chinook salmon, white and green sturgeon, and American shad. In addition, the river supports a large population of resident species, including Sacramento squawfish, white catfish, and splittail (EA 1981c).

2.2.6 The Biological Environment of the Bay-Delta

The biological environment of the Bay-Delta is a complex community of plants and animals inhabiting the saltwater, brackish-water, and freshwater areas of San Francisco Bay and the Sacramento - San Joaquin Delta. The main objective of this section is to review the interrelationships of Bay-Delta organisms with environmental factors affecting the distribution and abundance of particular species. This information is used in Chapter 5 in assessing the impact of the cooling water system of the Pittsburg Power Plant.

2.2.6.1 The Bay-Delta Aquatic Community

The Bay-Delta is a complex estuarine ecosystem, a transition zone between inland sources of fresh water and the salt water from the ocean. Along the salinity gradient extending from the Golden Gate up into the Delta, the organisms composing the aquatic community change dramatically, although the basic functional relationships among organisms remain similar throughout the system.

The Bay-Delta aquatic community is portrayed in Figure 2-14. The figure emphasizes the basic feeding relationships among community components and points out some of the more important species in the Bay-Delta. The primary energy input to the system is solar radiation, which is used, along with nutrients, by the primary producers (phytoplankton, vascular plants, and macroalgae) to convert inorganic carbon and nutrients into organic matter. The phytoplankton are preyed on by zooplankton (copepods, cladocerans, mysid shrimp) and some fishes, and the vascular plants and macroalgae are grazed on and also produce detritus which is decomposed by microbes and consumed by detritivores (polychaetes, amphipods, cladocerans, rotifers). These primary consumers are in turn preyed upon by secondary consumers, consisting mainly of a variety of invertebrates (cnidarians, polychaetes, gastropods, cyclopoid copepods, opossum shrimp, and decapods) and fishes (anchovies, herrings, silversides, smelts, croakers, flatfishes, gobies, sculpins, shad, minnows, juvenile salmon, and sunfishes). These are in turn consumed by top consumers, such as fish (striped bass, catfish, sturgeons, lampreys, sharks, and rays), marine mammals, birds, and man. The role of a species in the food web may be different at different life stages or it may utilize various levels of the food web simultaneously (omnivorous).

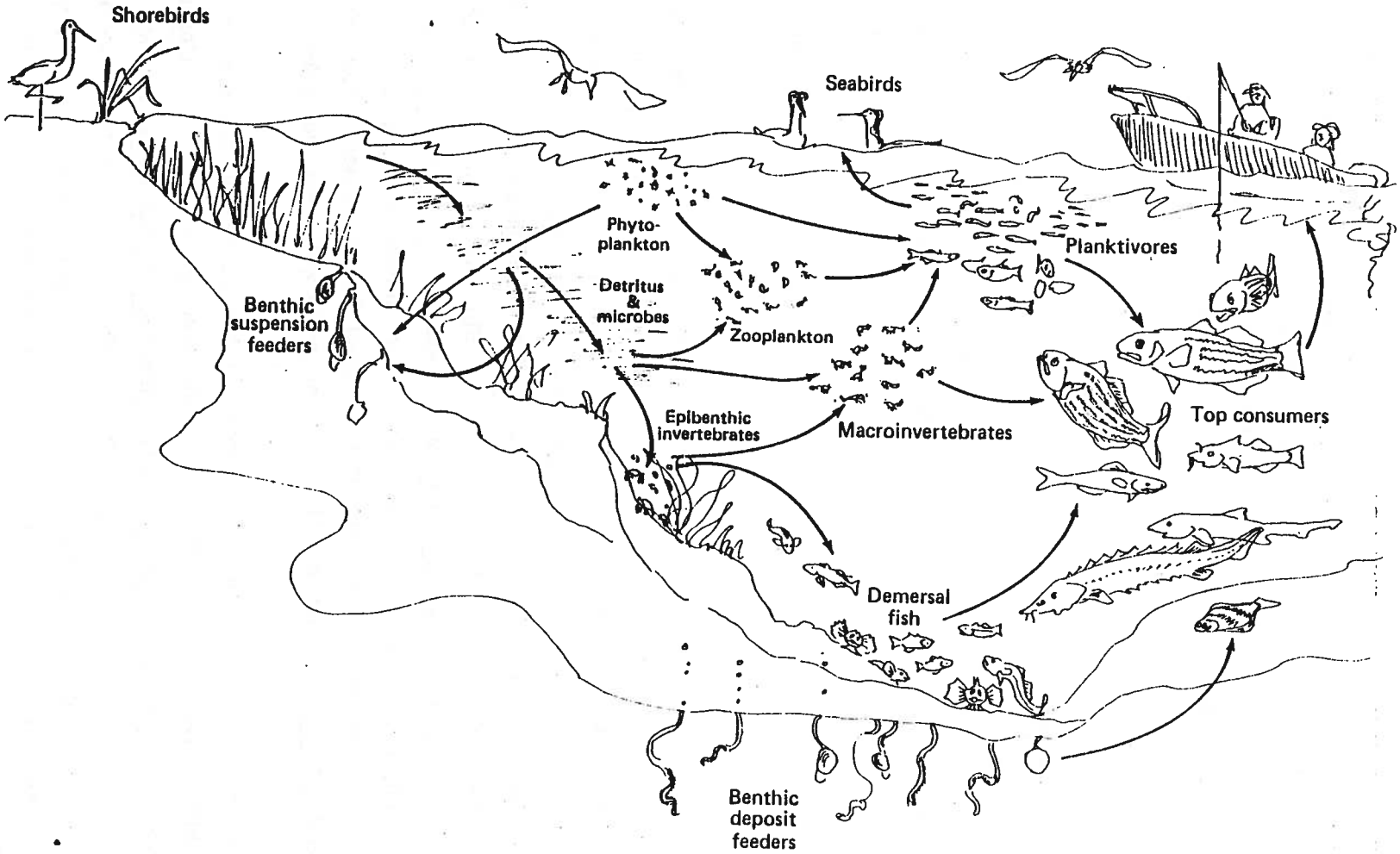

Figure 2-14. The Bay-Delta aquatic community and the primary feeding relationships among community components. (Adapted from CDF&G 1964.)

In the following sections the major components of the Bay-Delta aquatic community are discussed: (1) primary producers, (2) detritus and microbes, (3) primary consumers, (4) secondary consumers, and (5) top consumers. The importance of each component to the Bay-Delta food web is described; the major species of importance in the high-salinity, low-salinity, and freshwater portions of the Bay-Delta are identified; and the effects of physical and chemical factors (Sections 2.2.1-2.2.3) on the distribution and abundance of the major species are examined.

2.2.6.1.1 Primary Producers

Phytoplankton

Phytoplankton are of prime importance to the ecology of the Bay-Delta because of their position at the base of the food web. The seasonal abundance of copepods, cladocerans, and other pelagic herbivores closely follows that of phytoplankton in many parts of the Bay-Delta, and the success of the young of important fishes, such as striped bass, may largely depend on the quantity of phytoplankton available as a direct or indirect food source.

In the high-salinity areas of the Bay-Delta (i.e., the south and central Bay and San Pablo Bay), the dominant phytoplankton are diatoms and dinoflagellates. Diatom abundance is usually highest in spring when coastal upwelling first enriches the coastal and Bay waters with nutrients stored in the offshore depths, when incident solar radiation (insolation) is nearing its seasonal maximum, and when water temperatures are rising in the shallows. Blooms of diatoms of the genera Asterionella, Chaetoceros, Coscinodiscus, Skeletonema, Thalassiosira, and Nitschia are common. Later, in the summer, after diatom abundance has decreased substantially, blooms of coastal ocean (neritic) dinoflagellates, including Ceratium spp. and Gonyaulax spp. occur, often resulting in the well-known red tides.

In the low-salinity and freshwater areas of the Bay-Delta, diatoms are also the dominant phytoplankton. The most common genera include Coscinodiscus, Chaetoceros, Cyclotella, Melosira, Stephanodiscus, Skeletonema, and Microsiphona (Table 2-9) (Ball 1977). Green algae are also abundant, and during winter and spring may constitute as much as 60-70 percent of the phytoplankton populations of the Delta, especially in the dead-end sloughs of the eastern Delta. Cryptomonads, dinoflagellates, and flagellates may also be abundant in low-velocity waters (Ball and

TABLE 2-9 AREAS OF OCCURRENCE OF DOMINANT GENERA OF PHYTOPLANKTON DURING THE PERIOD SPRING THROUGH FALL, 1969-1974

Area	Genera
Northern Delta	Coscinodiscus, Cyclotella, Melosira
Western Delta	Coscinodiscus, Cyclotella, Melosira, Skeletonema, Stephanodiscus, Microsiphona
Suisun Bay	Chaetoceros, Coscinodiscus, Cyclotella, Melosira, Skeletonema, Stephanodiscus
San Pablo Bay	Chaetoceros, Coscinodiscus, Cyclotella, Skeletonema
Suisun Marsh	Coscinodiscus, Cyclotella, Stephanodiscus
Eastern Delta	Coscinodiscus, Cyclotella, Stephanodiscus, Melosira
Southern Delta	Coscinodiscus, Cyclotella, Stephanodiscus, Melosira

Source: Ball 1977.

Arthur 1979). Blue-green algae are generally not abundant, but may be common in the fresh, slowly flowing waters of the eastern Delta.

The phytoplankton standing crop, as measured by chlorophyll a concentration, varies widely in the low-salinity and freshwater areas of the Bay-Delta (Ball and Arthur 1979). Very low chlorophyll a concentrations occur in the Sacramento River water entering the Delta, whereas concentrations are very high in the San Joaquin River, whose outflow consists largely of agricultural and municipal waste water. The dead-end sloughs of the eastern Delta and the sloughs and backwaters of Suisun Marsh also have high chlorophyll a concentrations. In the western Delta and Suisun Bay, chlorophyll a concentrations are moderate, and in San Pablo Bay they are low. Maximum chlorophyll a concentrations in Suisun Bay are generally found in midsummer, as opposed to spring in the high-salinity areas of the Bay-Delta. During blooms, the highest concentrations in the main channel typically occur in the entrapment zone, where salinities range from about 1 to 6 ppt (Ball and Arthur 1979).

The position of the entrapment zone in the late spring and summer months appears to affect the standing crop of phytoplankton in Suisun Bay (Ball and Arthur 1979). During the 1976 and 1977 drought, Delta outflow was minimal and the entrapment zone was located upstream of eastern Suisun Bay during the spring and summer months. Phytoplankton abundance in the Suisun Bay area was very low. In other years, when the entrapment zone was located in eastern Suisun Bay, phytoplankton abundance was greater. The abundance of copepods and other estuarine biota that depend directly or indirectly on phytoplankton followed similar trends.

The physical and chemical factors associated with the timing, location, and magnitude of phytoplankton blooms include (1) insolation, (2) water temperature, (3) availability of nutrients, and (4) water residence time (i.e., time required for a parcel of water to move through a given area). Standing crops of phytoplankton are low during the winter because of low insolation and water temperatures, and short residence times (associated with high Delta outflows), although there is an ample supply of nutrients. During spring and summer, phytoplankton standing crops substantially increase in response to increasing insolation and temperature and lengthening residence times. Phytoplankton blooms would probably be more pronounced in much of the Bay-Delta except for the fact that nitrogen, a key nutrient, is depleted to limiting or near-limiting levels at many sites (Arthur and Ball

1979). Zooplankton predation has been shown to affect phytoplankton densities, but is not a principal factor controlling phytoplankton abundance.

Macroalgae

The macroalgae of the Bay-Delta are important primary producers, especially in the central Bay. In addition to providing habitat and food for various invertebrates, the macroalgae are an important source of detritus. Large quantities of macroalgae, primarily of the genera Bryopsis, Gracilaria, and Stenogramma, become detached from their substrates and decompose in the water body or drift ashore (Abbott and Hollenberg 1976).

The macroalgae are most abundant and diverse near the Golden Gate, where heavy wave action and rocky substrates support a large number of outer-coast species. Even the habitats near the Golden Gate are less exposed than those on the outer coast, eliminating such outer-coast species as Rhodoglossum affini, Macrocystis integrifolia (giant kelp), and Cystoceira osmundacea (Silva 1979). The rocky shores in the moderately protected western central Bay support about 90 species of macroalgae, but the fully protected habitats (mudflats, marshes, boat harbors) in the south Bay and the eastern central Bay support only about 30 species (Silva 1979). In San Pablo Bay and in areas upstream, the abundance and diversity of macroalgae decreases sharply. The major macroalga in brackish and fresh water areas is Cladophora spp., which often forms dense growths on rocky substrates and is considered a nuisance species.

The major physical and chemical factors affecting the distribution and abundance of macroalgae in the Bay-Delta are wave action, as discussed above, substrate, and salinity. Most macroalgae require stable substrates for growth, explaining the abundance of macroalgae in rocky areas and their paucity on muddy substrates. Most marine macroalgae cannot tolerate a sustained reduction of salinity below two-thirds normal sea water (Silva 1979), and since salinities in San Pablo Bay and the eastern part of the central Bay decline below that level in periods of high Delta outflow, it is not surprising that the diversity of macroalgae in those areas is substantially less than that near the Golden Gate.

Vascular Plants

Like phytoplankton and macroalgae, vascular plants are of prime importance to the ecology of the Bay-Delta because of their position at the base of the food web. Unlike phytoplankton and macroalgae, however, vascular plants are fed upon by few aquatic organisms. Rather, the input of energy from the vascular plants to the aquatic food web is mainly in the form of detritus. The aquatic vascular plants provide the main flow of detritus into the system. Other important functions of the vascular plants are to provide habitat for a variety of fishes, invertebrates, birds, and mammals; to contribute to the production of phytoplankton by providing a substrate for attached forms; to increase the dissolved oxygen concentrations in the water body during the highly productive spring and summer months; and to act as a biological filter, removing carbon monoxide from the air and a variety of pollutants from the water (Harvey et al. 1977).

Vascular plants are found in subtidal areas as well as in intertidal marshes. The only subtidal vascular plant in the high-salinity portions of the Bay-Delta is eelgrass (*Zostera marina*). Eelgrass grows in dense beds in shallow, sandy-bottom areas, mainly in the central Bay. The diversity of subtidal vascular plants increases in the Delta. The most abundant genera include *Ceratophyllum*, *Myriophyllum*, *Potamogeton*, *Polygonum*, *Elodea*, and *Ludvigia*. Also found in the Delta are the floating vascular plants of the genera *Azolla* and *Eichoria*. These subtidal and floating plants form dense growths along the edges of many of the sloughs and channels in the Delta, and even sometimes completely choke shallow sloughs. Many of these plants are also found in the brackish waters of Suisun Bay.

Marshlands are the major habitat for Bay-Delta vascular plants, but the marshlands have been extensively altered during the past 130 years. In 1850, natural tidal marshes covered an estimated 2,175 km^2 (840 mi^2) in the Bay-Delta. By 1975, sedimentation and land reclamation had reduced the total area of natural tidal marshes by 95 percent to 110 km^2 (42 mi^2) (Harvey et al. 1977). The greatest reduction occurred in the Delta, which in 1850 consisted of 1,375 km^2 (530 mi^2) of freshwater marshlands. Most of these marshlands were leveed for farming by 1930, and in 1970 only 20 km^2 (8 mi^2) remained as natural tidal marsh. In Suisun and San Pablo bays, much of the tidal marshlands were isolated by levees from daily inundation by the tides. Today there are large areas of nontidal marshlands along the northern borders of Suisun and San Pablo bays. The Suisun Marsh, north of Suisun

Bay, is largely a managed wetland which is flooded in the fall to attract overwintering waterfowl. Most of the tidal marshlands that once surrounded the central Bay and the south Bay were converted to salt-evaporation ponds or to sites for residential and industrial structures, transportation facilities, and garbage dumps.

The dominant vascular plants of the tidal marshland vary moving upstream along the Bay-Delta salinity gradient. In the high-salinity areas (the south Bay and the central Bay), the common pickleweed (Salicornia pacifica) and California cordgrass (Spartina foliosa) are the dominant plants (Atwater et al. 1979). The common pickleweed generally monopolizes the tidal marsh plains at the higher elevations. Other species grow in scattered patches next to sloughs, natural uplands, and levees. California cordgrass fringes the tidal marsh plains where they descend into mudflats.

The tidal marshes of San Pablo Bay, Carquinez Strait, and Suisun Bay provide a transition between the salt marshes of San Francisco Bay and the freshwater marshes of the Delta. The dominant plants in the transition zone include a mixture of saltwater species such as pickleweed and cordgrass, and freshwater species such as tule (Scirpus acutus, S. californicus, and S. olneyi), cattails, and common reeds. This zone also has some species not common in either the Bay or the Delta, such as the alkali bulrush (Scirpus robustus). The diked, managed marshes of Suisun Bay are dominated by alkali bulrush, although small patches of cattails, tules, and Olney's bulrush are frequent in low, undrained spots and in drainage canals. Other managed lands are dominated by the introduced brass buttons (Cotula coronopifolia).

Up in the Delta, freshwater marsh plants line some of the sloughs and colonize islands of sediment formed by shoaling. The dominant plants are the common tule (Scirpus acutus), Olney's bulrush (S. olneyi), cattails (Typha spp.), common reed (Phrangmites communis), and arroyo willow (Salix lasiolepis).

The major physical and chemical factors influencing tidal marsh vegetation are salinity and the frequency and duration of tidal flooding. The high soil salinity associated with saline tidal water discourages the growth of bulrushes, tules, cattails, and rushes (Juncus spp.) west of the Delta. Since the turn of the century, increased saltwater intrusion due to reduced Delta outflow has extended the range of the salt marsh species into San Pablo and Suisun bays at the expense of the native tules and bulrushes.

Other than salinity, the major factor affecting the distribution of marsh plants is the frequency and duration of tidal flooding. Sedimentation in the Bay-Delta has raised the elevation of the tidal mudflats, permitting marsh vegetation to extend onto the mudflats, and permitting lowland grasses, such as salt grass (Distichlus spicata), to extend into former marshlands. Marsh vegetation actually enhances sedimentation by slowing water velocities and trapping sediment.

Detritus and Microbes

Detritus, comprising all forms of organic matter from decomposing plants and animals, is a key component of the Bay-Delta food web, linking the primary producers to many of the upper trophic levels (Figure 2-14). Detritus represents a food store, trapped by the estuary's currents and sediments, that is available during the winter when there is little production of food by phytoplankton. In some estuaries, detritus and its associated microbes have been shown to be the major source of energy used by primary consumers, particularly suspension-feeding and deposit-feeding animals (Cooper 1974).

Sources of detritus in the Bay-Delta include upland vegetation, vascular plants in Bay-Delta marshlands, macroalgae and phytoplankton, dead aquatic animals, and wastewater discharges. Another significant source of detritus is peat—organic matter formed prior to reclamation by decomposing marsh vegetation in the Delta. Peat is a major soil component in the Delta, and is carried downstream in large quantities year-round, especially in the winter months.

Microbes associated with detritus include bacteria, fungi, and protozoa. These microbes decompose detritus, releasing proteins, oils, and other organic compounds that are used by other organisms. The microbes themselves are probably the primary energy source for many detritus-feeding organisms (Newell 1965; Odum 1971).

2.2.6.1.2 Primary Consumers

The primary consumers are at the center of the Bay-Delta food web, and are therefore important not only to the lower trophic levels upon which they feed (e.g., phytoplankton, decomposers), but also to the higher trophic levels for which they serve as prey (secondary consumers, top consumers). The primary consumers include

herbivores, which feed mainly on phytoplankton, and detritivores, which feed on decomposing organic matter (detritus) and microbes. Many primary consumers are both herbivores and detritivores, and many are also secondary consumers, feeding on other primary consumers as well as on phytoplankton and/or detritus. The primary consumers include both suspension and deposit feeders. The herbivores are primarily suspension feeders, filtering phytoplankton from the water. They include planktonic forms, such as calanoid copepods and cladocerans; benthic forms, such as most bivalves and some gastropods; and nektonic forms, such as northern anchovy, Sacramento blackfish, and at times topsmelt and jacksmelt. The detritivores are either suspension feeders, filtering particulate organic substances from the water, or deposit feeders, ingesting organic deposits in or on soft substrates. Suspension feeders include rotifers, cladocerans, and some amphipods. Deposit feeders include polychaetes, oligochaetes, some amphipods, some gastropods, and some bivalves. Deposit feeders are usually burrowing animals which ingest the soft substrate as they move about but digest only the detritus and the attached microbes.

The major taxa of primary consumers vary over the Bay-Delta salinity gradient. In the high-salinity portions of the system, the most important primary consumers include the calanoid copepods Acartia clausi, A. tonsa, and Paracalanus parvus; bivalves such as the bay mussel (Mytilus edulis), the Japanese littleneck clam (Tapes japonica), and the tiny gem clam (Gemma gemma); amphipods such as Ampelisca milleri, Jassa facalta, and Corophium insidiosum, and fishes, such as northern anchovy, topsmelt, and jacksmelt. Other primary consumers of the high-salinity zone are the polychaetes, gastropods, tunicates, hydroids, and sponges. In the low-salinity portions of the Bay-Delta, the most important primary consumers are the calanoid copepods Eurytemora affinis and Acartia clausi, the amphipods Corophium stimpsoni and C. spinicorne, and the opossum shrimp Neomysis mercedis, and they also include the clam Macoma balthica, the polychaetes Neanthes spp. and Boccardia spp., various oligochaetes, and the cladocerans Daphnia spp. and Diaphanosoma leuchtenbergianus. Finally, in the freshwater portions the most important primary consumers are the cladocerans Daphnia pulex and D. parvula, the calanoid copepods Diaptomus spp. and Limnocalanus macrurus, oligochaetes, amphipods, and the Sacramento blackfish. Rotifers, nematodes, and ostracods are important primary consumers throughout the Bay-Delta.

Salinity appears to be one of the major physical or chemical factors affecting the

distribution and abundance of primary consumers by the Bay-Delta, as evidenced by the changes in species composition along the Bay-Delta salinity gradient. For example, the distributions of Acartia clausi and Eurytemora affinis, the dominant copepods in the high-salinity and low-salinity regions, respectively, meet but do not significantly overlap in the salinity range of 6-9 ppt (conductivities of 10-15 mmho)(CDF&G et al. 1975). The dominant amphipods in the low-salinity and freshwater areas are Corophium stimpsoni and C. spinicorne, whereas C. insidiosum becomes dominant at higher salinities. The entrapment zone, appears to be the upstream limit of the distribution of many high-salinity species and the downstream limit of the distribution of many freshwater species.

The distribution and abundance of primary consumers is also related to the availability of food. Physical and chemical conditions that promote phytoplankton productivity (warm temperatures, high insolation, high nutrients, slow-moving waters) indirectly promote the productivity of the herbivores. Water body configuration and bathymetry also affect phytoplankton productivity, and, therefore, herbivore productivity. The shallow areas of Suisun Bay are highly productive, as are many of the shallow, slow-moving backwater areas of the Delta. Such waters are also optimum for the productivity of detritivores, especially of deposit feeders, which thrive on the fine-grained, organically rich sediments found in such areas.

The position of the entrapment zone in spring and summer is thought to have a powerful effect on the productivity of important estuarine primary consumers, such as Eurytemora affinis and Neomysis mercedis (Arthur and Ball 1979). When the entrapment zone is in the shallow eastern Suisun Bay, the abundance of these organisms is enhanced, probably because of increased productivity of the phytoplankton upon which they feed. But when the entrapment zone moves upstream in the channels of the Delta, as it did during the 1976-1977 drought years, productivity is reduced.

2.2.6.1.3 Secondary Consumers

The secondary consumers include both invertebrates and fish which are generally active predators but not necessarily entirely carnivorous. Because of their position near the top of the Bay-Delta food web, the secondary consumers are dependent on the productivity of all the trophic levels below them.

Invertebrates

The invertebrate secondary consumers have developed a variety of feeding strategies. There are planktonic, epibenthic, and benthic feeding types. The planktonic types are active carnivores, preying on herbivorous zooplankton as well as on other secondary consumers. Many species are omnivorous, selectively preying on phytoplankton as well. Planktonic secondary consumers include some rotifers, cnidarians, cyclopoid copepods, opossum shrimp, and decapod larvae.

The epibenthic types are also active predators. They remain on or just above the substrate and prey on benthic, epibenthic, and planktonic invertebrates, and, in some cases, fish. Most are omnivorous, feeding on phytoplankton and detritus as well. Epibenthic secondary consumers include shrimps, brachyuran crabs, and opossum shrimp.

The benthic secondary consumers are those that remain on, in, or attached to the substrate. These include barnacles, hydroids, and anemones, which feed on plankton that pass near their feeding structures, and polychaetes and predatory gastropods, which burrow through or crawl on the substrate, consuming benthic primary consumers, such as amphipods, polychaetes, and oligochaetes, as well as other secondary consumers.

The important invertebrate secondary consumers vary over the Bay-Delta salinity gradient. In the high-salinity areas, the important taxa include the hydrozoan medusae *Polyorchis* spp., the anemone *Metridium senile*, the barnacles *Balanus* spp., the blacktail shrimp *Crangon nigricauda* and *Cancer* spp. crabs, polychaetes, and gastropods. In low-salinity areas, the most important taxa include the hydrozoan medusae *Phiallela* spp., the colonial hydroid *Cordylophora lacustris*, the opossum shrimp *Neomysis mercedis*, the barnacle *Balanus improvisus*, the shrimps *Palaemon macrodactylus* and *Crangon franciscorum*, and predatory gastropods. Important taxa in the freshwater areas include the hydroids *Hydra* spp., the copepods *Cyclops* spp. and *Epischura nevadensis*, the cladoceran *Leptodora kindtii*, the rotifers *Asplanchna* spp., and the crayfishes *Procambarus clarkii* and *Pacifastacus leniusculus*.

Fishes

Like the invertebrates, the secondary consumer fishes include planktonic, epibenthic

(demersal), and benthic feeding types. Table 2-10 presents a listing of common names used for fish in this report and their scientific equivalent, following Robins et al. (1980).

Most Bay-Delta fishes have planktonic larval stages that feed on zooplankton and in some cases phytoplankton. Large pelagic fishes, such as striped bass, chinook salmon, and Sacramento squawfish, are planktivorous through the early juvenile life stage, whereupon they become more selective predators, feeding on large invertebrates and fish. Demersal fishes, such as sturgeons, flatfishes, gobies, sculpins, and croakers, are planktivorous as larvae, but begin to feed on epibenthic invertebrates as juveniles. Many fishes, including smelts, silversides, northern anchovy, and Pacific herring, are planktivorous throughout their lives.

Some Bay-Delta fishes do not rely on plankton as a major food source at any life stage. The live-bearing surfperches, for example, predominantly feed on epibenthic invertebrates, such as molluscs, crustaceans, and polychaetes, but are also planktivorous when young. Sharks and some skates and rays feed on benthic and epibenthic invertebrates by shoveling through the substrate, and also feed on fishes and large invertebrates in the water column. Many freshwater fishes prey primarily on benthic and drifting insect larvae and crustaceans, because zooplankton abundance is low in the swifter-flowing freshwater sloughs and rivers.

The most abundant secondary consumer fishes vary along the Bay-Delta salinity gradient. The most abundant taxa in the high-salinity areas include the schooling, pelagic forage fishes, such as northern anchovy, Pacific herring, topsmelt, jacksmelt, and true smelts (whitebait, surf smelt, and night smelt). Other groups include flatfishes, rockfishes, surfperches, gobies, and sharks. In the low-salinity areas, the most abundant taxa include the Delta smelt and longfin smelt, prickly sculpin and staghorn sculpin, threadfin shad, yellowfin goby, and starry flounder. Except for Delta smelt and threadfin shad, these fishes are also found in the high-salinity areas, but are less important there to the food web. The abundant freshwater taxa include the native minnows; the introduced catfishes (white catfish and channel catfish), bullheads, and sunfishes; and juvenile anadromous fishes, including striped bass, white sturgeon, American shad, and chinook salmon.

TABLE 2-10 SCIENTIFIC AND COMMON NAMES[a] OF FISHES REFERRED TO IN THIS REPORT

Family Genus Species	Common Name
Petromyzontidae	Lampreys
Lampetra tridentatus	Pacific lamprey
Lampetra ayresi	River lamprey
Charcharhinidae	Requiem sharks
Mustelus californicus	Grey smoothhound
Triakis semifasciata	Leopard shark
Squalidae	Dogfish sharks
Squalus acanthias	Spiny dogfish
Torpedinidae	Electric rays
Torpedo californica	Pacific electric ray
Acipenseridae	Sturgeons
Acipenser medirostris	Green sturgeon
Acipenser transmontanus	White sturgeon
Clupeidae	Herrings
*Alosa sapidissima**	American shad
Clupea harengus pallasi	Pacific herring
*Dorosoma petenense**	Threadfin shad
Engraulidae	Anchovies
Engraulis mordax	Northern anchovy
Salmonidae	Trouts
Oncorhynchus kisutch	Coho salmon
Oncorhynchus tshawytscha	Chinook salmon
Salmo gairdneri gairdneri	Steelhead
Osmeridae	Smelts
Allosmerus elongatus	Whitebait
Hypomesus pretiosus	Surf smelt
Hypomesus transpacificus	Delta smelt
Spirinchus starksi	Night smelt
Spirinchus thaleichthys	Longfin smelt
Cyprinidae	Minnows and carps
*Carassius auratus**	Goldfish
*Cyprinus carpio**	Carp
Lavinia exilicauda	Hitch

(a) Names follow Robins et. al., 1980.

Note: (*) indicates an introduced species.

TABLE 2-10 (CONT.)

Family Genus Species	Common Name
Cyprinidae (cont.)	Minnows and carps (cont.)
Mylopharodon conocephalus	Hardhead
*Notemigonus crysoleucas**	Golden shiner
Orthodon microlepidotus	Sacramento blackfish
*Pimephales promelas**	Fathead minnow
Pogonichthys macrolepidotus	Splittail
Ptychocheilus grandis	Sacramento squawfish
Catostomidae	Suckers
Catostomus occidentalis	Sacramento sucker
Ictaluridae	Freshwater catfishes
*Ictalurus catus**	White catfish
*Ictalurus punctatus**	Channel catfish
*Ictalurus melas**	Black bullhead
*Ictalurus nebulosus**	Brown bullhead
Batrachoididae	Toadfishes
Porichthys notatus	Plainfin midshipman
Gadidae	Codfishes
Microgadus proximus	Pacific tomcod
Cyprinodontidae	Killifishes
*Lucania parva**	Rainwater killifish
Atherinidae	Silversides
Atherinops affinis	Topsmelt
Atherinopsis californiensis	Jacksmelt
Menidia beryllina	Inland silverside
Gasterosteidae	Sticklebacks
Gasterosteus aculeatus	Threespine stickleback
Syngnathidae	Pipefishes and sea horses
Synagnathus griseolineatus	Bay pipefish
Percichthyidae	Temperate basses
*Morone saxatilis**	Striped bass
Centrarchidae	Sunfishes
Archoplites interruptus	Sacramento perch
*Lepomis gulosus**	Warmouth
*Micropterus salmoides**	Largemouth bass
*Pomoxis nigromaculatus**	Black crappie

TABLE 2-10 (CONT.)

Family Genus Species	Common Name
Percidae	Perches
*Percina macrolepida**	Bigscale logperch
Sciaenidae	Drums
Genyonemus lineatus	White croaker
Embiotocidae	Surfperches
Amphistichus rhodoterus	Redtail surfperch
Cymatogaster aggregata	Shiner perch
Embiotoca jacksoni	Black perch
Hyperprosopon argenteum	Walleye surfperch
Hysterocarpus traski	Tule perch
Micrometrus minimus	Dwarf perch
Phanerodon furcatus	White seaperch
Rhacochilus toxotes	Rubberlip seaperch
Rhacochilus vacca	Pile seaperch
Gobiidae	Gobies
*Acanthogobius flavimanus**	Yellowfin goby
Clevelandia ios	Arrow goby
Gillichthys mirabilis	Longjaw mudsucker
Eucyclogobius newberryi	Tidewater goby
Scorpaenidae	Scorpionfishes
Sebastes auriculatus	Brown rockfish
Hexagrammidae	Greenlings
Ophiodon elongatus	Lingcod
Cottidae	Sculpins
Cottus aleuticus	Coastrange sculpin
Cottus asper	Prickly sculpin
Leptocottus armatus	Staghorn sculpin
Scorpaenichthys marmoratus	Cabezon
Bothidae	Lefteye flounders
Citharichthys stigmaeus	Speckled sanddab
Pleuronectidae	Righteye flounders
Hypsopsetta guttulata	Diamond turbot
Parophrys vetulus	English sole
Platichthys stellatus	Starry flounder

Top Consumers

The top consumers, as their name implies, form the highest trophic level in the Bay-Delta food web. Their only predators are other top consumers and man. The top consumers are the furthest removed from primary production and those least immediately affected by variations in primary producer abundance. Nevertheless, they require high production by the intermediate trophic levels to maintain their numbers and biomass. The top consumers consist of fewer species and fewer individuals than the lower trophic levels. The top consumers of the Bay-Delta include some fishes, birds, marine mammals, and man.

The dominant top consumers vary along the Bay-Delta salinity gradient. In the high-salinity areas, the dominant top consumer fishes include the anadromous striped bass, white sturgeon, and Pacific lamprey, as well as sharks and rays, such as the spiny dogfish and the Pacific electric ray. Others of lesser importance include rockfishes, cabezon, lingcod, and flatfishes. The anadromous species live as adults in the coastal and high-salinity Bay waters, and they migrate seasonally into the fresh water to spawn. The striped bass is an open-water predator, the white sturgeon is primarily a bottom feeder, and the Pacific lamprey is a parasite that attaches itself to large fish and occasionally to marine mammals. The other species mentioned are ocean fishes that are generally common only in the central Bay. The top consumer birds of the high-salinity areas include shorebirds, wading birds, and sea birds, such as pelicans, terns, and cormorants. Many of the shorebirds and wading birds eat invertebrates and are best classified as secondary consumers. The dominant marine mammals of the high-salinity areas include the California sea lion (*Zalophus californianus*) and the harbor seal (*Phoca vetulina*). These are few in number, and respresent a minor component of the top consumers.

In the low-salinity areas, the striped bass is the dominant top consumer. The lack of other, competing top consumers in this highly productive area may partially explain the great success of this introduced species in the Bay-Delta.

In the freshwater areas, the dominant top consumers include striped bass and Sacramento squawfish. Other species include white catfish, yellowfin goby, Sacramento perch, sturgeon, and a number of introduced sunfishes.

The native squawfish and Sacramento perch and the introduced sunfishes are found mostly in weedy portions of sloughs and channels (Moyle 1976). White catfish are bottom feeders found in sloughs, and yellowfin gobies are bottom feeders found in sloughs, channels, and canals (Turner 1966b; Moyle 1976). The striped bass is the most voracious predator of the freshwater areas, and is abundant in the Delta during its late-spring spawning migration.

The ultimate top consumer is man. The Bay-Delta supports a number of important commercial and sport fisheries. In the high-salinity areas of the central Bay and San Pablo Bay, northern anchovy and Pacific herring support the most intensive commercial fisheries, but the bay shrimp (Crangon spp.), surfperches, and sharks are also sought by commercial fisherman. The major fishes sought by sport fisherman include striped bass (in the fall), white sturgeon, surfperches, jacksmelt, starry flounder, and sanddab. In the low-salinity and freshwater areas of Suisun Bay and the Delta, limited commercial fisheries exist for crayfish, staghorn sculpin, and longjaw mudsucker. Striped bass, American shad, salmon, and sturgeon are sought by sport fishermen in those areas.

2.2.7 Selected Species of the Bay-Delta Aquatic Community

The biological environment of the Bay-Delta was summarized in Section 2.2.6. The ways in which populations of aquatic organisms interact and respond to the physical and chemical environment was emphasized. Some of the more important species were mentioned but not considered in detail. The objective of Section 2.2.7 is to describe the life histories and seasonal and spatial distributions of selected species of the Bay-Delta, providing information necessary for evaluating the potential impact of the Pittsburg Power Plant cooling water intake system.

In this section, only those organism groups potentially involved with cooling water intakes are considered. Those groups include the microbial decomposers, phytoplankton, invertebrates with planktonic life stages, and fish. Groups not considered include the benthic invertebrates, birds, reptiles, amphibians, mammals, macroalgae, and vascular plants. On the basis of their abundance, wide distribution, and high reproduction potential, the microbial decomposers, phytoplankton, and many zooplankton (e.g., rotifers, mollusc larvae, polychaete larvae, barnacle larvae) were further excluded from consideration. Thus, only certain invertebrates and the fishes

are considered in this section. The organisms selected for discussion represent the community components subject to the greatest potential impact.

The species selected for discussion in this section are those that are important to man, important to the food web, and abundant in the Bay-Delta, and/or abundant in entrainment or impingement sampling (Chapters 3 and 4). Invertebrate species are described in Section 2.2.7.1 and fish species in Section 2.2.7.2.

The information presented in Section 2.2.7 includes what is known for each species about its distribution and abundance, position in the food web, reproduction, and development. Also presented is information concerning physical and biological factors regulating the population size and its location in the Bay-Delta. Some of the information presented has not been previously published. Unpublished data collected by the California Department of Fish and Game and the Department of Water Resources are presented in addition to and in support of data collected by EA in conjunction with 1978-1979 biological investigations of the Bay-Delta.

2.2.7.1 Invertebrates

2.2.7.1.1 *Crangon franciscorum*

Crangon franciscorum, the common bay shrimp of the Bay-Delta, is found in bays, estuaries, and occasionally on the open coast, ranging from southeastern Alaska to southern California. It is distributed in the Bay-Delta from the south Bay to the western Delta. *Crangon franciscorum* is found over a wide range of salinities but is not found in fresh water. Juveniles move into and reside in the less saline areas of the Bay-Delta (Israel 1936), accounting for the high abundance observed at the Pittsburg Power Plant (Chapter 4).

Spawning occurs in spring and summer after the adults have migrated back into the more saline areas of the Bay-Delta (Israel 1936). After spawning, the male dies and the female releases a brood of 2,000-5,000 eggs which is carried under the abdomen for about one month (Krygier and Horton 1975). After the eggs hatch and free-swimming larvae are released from the brood, most of the adult females die. The distribution of larvae in the Bay-Delta is not known. The larvae develop into adult shrimp over a period of 8-11 months (Krygier and Horton 1975). Adults and

juveniles remain near the bottom in muddy and sandy areas except at night, when they migrate upwards into the water column (Sitts and Knight 1979a).

The species is omnivorous, feeding principally on Neomysis mercedis, Corophium spp., Rhithropanopeus harrisii zoeae, and Palaemon macrodactylus larvae, as well as detritus, phytoplankton, larval fish, copepods, bivalves, and polychaetes (Siegfried et al. 1978; Sitts and Knight 1979b). Crangon franciscorum is in turn fed upon by many game fishes, such as green sturgeon, white sturgeon, American shad, chinook salmon, white catfish, striped bass, and staghorn sculpin (Ganssle 1966). It is also subject to a commercial bait fishery having a present annual yield of about 45,000 kg (100,000 lbs). Most of the fishery is located in San Pablo Bay, but there is some activity in the south Bay near Alviso.

2.2.7.1.2 Eurytemora affinis

Eurytemora affinis, a common calanoid copepod in many north temperate estuaries, occurs throughout much of the Bay-Delta, although it is most abundant in San Pablo Bay, Suisun Bay, and the western Delta. In EA's April-July 1978 plankton survey of the Bay-Delta, E. affinis was collected mainly in Suisun Bay and the lower San Joaquin River (EA 1981a). The species has been observed in greatest abundance in the entrapment zone (Arthur and Ball 1978), and since the location of the zone varies in response to Delta outflow (Section 2.2.1), the location of greatest abundance of E. affinis varies seasonally from San Pablo Bay, during periods of high outflow, to the western Delta, during periods of low outflow.

This species is most abundant at the middle and lower depths during the day and often at night, although it may be evenly distributed throughout the water column at night (Siegfried et al. 1978; Sitts and Knight 1979a). The greater abundance in the middle and lower depths helps to maintain it in and upstream of the entrapment zone.

Eurytemora affinis has been observed in densities of nearly 17,000/m^3 in Suisun Bay (Painter 1966). Caskey (1976) found that densities of E. affinis commonly averaged 300-3,000/m^3 in western Suisun Bay from 1972-74.

Reproduction occurs during the spring and summer months (Siegfried et al. 1978), and one or two broods per year may be produced. Development time from egg to adult varies from two to four months, and depends on temperature (Katona 1971).

Eurytemora affinis is a suspension feeder, consuming planktonic diatoms and flagellates (Katona 1971). It is in turn fed upon by larval and juvenile fishes, as well as predaceous crustaceans such as Neomysis mercedis, Crangon franciscorum, and Palaemon macrodactylus. Because of the numerical dominance of E. affinis in Suisun Bay and the western Delta, the species may be the most important copepod in the local food web.

2.2.7.1.3 Corophium stimpsoni

Corophium stimpsoni is a native, tube-dwelling amphipod that is widely distributed in the oligohaline (0.5-5 ppt) portions of the Bay-Delta (Hazel and Kelley 1966). Corophium stimpsoni has been observed in abundance from Suisun Bay to the lower Sacramento and San Joaquin rivers (EA 1981a). While found in the western Delta from mid-spring to late fall, C. stimpsoni is most abundant from July to September (EA 1981b).

Corophium stimpsoni is found in and above soft-bottom substrates, and has been observed in densities up to 14,500/m^2 (Siegfried at al. 1978). During the daytime, C. stimpsoni remains on or near the bottom, but it migrates off the bottom and into the water column at night (Hazel and Kelly 1966; Siegfried et al. 1978).

Corophium stimpsoni females produce one brood containing 3-54 eggs. Their progeny reproduce during the same year, yielding two generations per year The second generation overwinters and matures the following spring (Siegfried et al. 1978). Corophium stimpsoni is probably a suspension and selective deposit feeder. The species is eaten by a variety of fishes, and is a major forage species for game fish such as striped bass (Ganssle 1966; Hazel and Kelley 1966). The shrimps Palaemon macrodactylus and Crangon franciscorum also feed on C. stimpsoni in the Delta (Sitts and Knight 1979b).

2.2.7.1.4 Corophium spinicorne

Corophium spinicorne is a tube-dwelling amphipod common in Suisun Bay and the Delta (Aldrich 1961; Siegfried et al. 1978). Especially common on the surfaces of submerged logs and rocks, C. spinicorne has been reported to occur in densities exceeding 500/m^2 (Painter 1966). During 1978, the species was most abundant in

the lower Sacramento and San Joaquin rivers and in the Sacramento Deepwater Ship Channel, Cache Slough, and Steamboat Slough (EA 1981a). C. spinicorne was collected at the Pittsburg Power Plant from mid-spring to late fall, with maximum abundance occurring in late summer and early fall (Chapter 3).

2.2.7.1.5 Neomysis mercedis

Neomysis mercedis is the most abundant opossum shrimp in the Bay-Delta. The species is found in rivers and estuaries from Alaska to Point Conception, California, and in the Bay-Delta is distributed from south and central Bays to Sacramento and Knight's Landing on the Sacramento River and Stockton on the San Joaquin River (Tattersall 1932; Orsi and Knutson 1979). Although distributed over a wide range of salinities, they are less abundant in salinities greater than 7 ppt (Heubach 1969). Maximum abundance has been observed in the entrapment zone (Arthur and Ball 1978; Siegfried et al. 1979). Since the position of the entrapment zone is a function of Delta outflow (Section 2.2.1), the location of the maximum abundance of N. mercedis varies on a seasonal basis. During the 1978 and 1979 spring and summer plankton surveys of the Bay-Delta upstream of San Pablo Bay, the location of maximum abundance shifted upstream when the salinity in a given area began to exceed about 6 ppt (EA 1981a), consistent with the observations of Arthur and Ball (1978) and Siegfried et al. (1979).

The abundance of N. mercedis increases during the spring from low winter levels and remains high through mid-fall. Peaks in abundance may be observed during late spring and late summer, corresponding to the recruitment of new generations into the populations. Densities as great as 1,500-10,000/m^3 have been reported (Heubach 1972; Orsi and Knutson 1979).

They are found throughout the water column, but during the day they are more abundant near the bottom, and during the night nearer the middepth and surface (Heubach 1969; Siegfried et al. 1978; Sitts and Knight 1979a).

Neomysis mercedis reproduce throughout the year. In the western Delta, three generations per year may be produced: one in early spring, one in early summer, and one in late summer (Siegfried et al. 1979). The latter overwinters and produces the early spring generation of the next year (Siegfried et al. 1979). The number of eggs

produced per female increases with increasing size (Heubach 1969; Siegfried et al. 1979); estimates of fecundity range from 6 to 65 eggs. The eggs are held in a ventral brood pouch (hence the name opossum shrimp), and the developing embryos remain in the pouch for a period from four days to three months, depending on ambient temperature, judging from observation of other species of opossum shrimp at various latitudes (Tattersall and Tattersall 1951). The adults may die after the release of the brood (Heubach 1969), although some species of opossum shrimp are known to produce several broods per year (Tattersall and Tattersall 1951).

Juveniles approximately 2 mm in length are released from the brood pouch into open water. Individuals become sexually mature at 7 mm in warmer months and at 10 mm in cooler months (Heubach 1969). The growth period is 2-4 months in the spring and summer and 6-8 months in the fall and winter, judging from the amounts of time between peaks in the abundance of 2-3 mm organisms (CDF&G et al. 1976; Siegfried et al. 1979).

Neomysis mercedis plays a complex role in the Bay-Delta food web. Its diet consists mainly of phytoplankton (predominantly diatoms) and detritus, although copepods and other small zooplankton are also eaten, especially by larger individuals (Kost and Knight 1975; Wilson 1951). It is preyed on in turn by secondary consumer invertebrates, such as Crangon franciscorum and Palaemon macrodactylus; secondary consumer fishes, such as American shad, longfin smelt, and Delta smelt; and top consumer fishes, such as striped bass, white sturgeon, and white catfish (Heubach et al. 1963; Ganssle 1966; Turner 1966a,b; Stevens 1966a,b; Thomas 1967; Radtke 1966; Moyle 1976; Sitts and Knight 1979b). Thus, changes in the abundance of N. mercedis directly and indirectly affect a wide variety of organisms at all levels of the food web.

2.2.7.1.6 Palaemon macrodactylus

Palaemon macrodactylus is an introduced subtidal, non-burrowing shrimp common in many areas of the Bay-Delta. Though capable of swimming, P. macrodactylus is often found among masses of algae and among fouling organisms on pilings and other substrates. The species is most abundant in Suisun Bay and the western Delta, where salinities are low because of freshwater inflow.

Adults are present year-round from San Pablo Bay to the western Delta. Peak numbers collected at the Pittsburg Power Plant were observed during early winter (Chapter 4). Densities of adults and juveniles in Suisun Bay have been reported to average 15/100 m^3 and range up to 48/100 m^3 (Sitts and Knight 1979a).

Palaemon macrodactylus migrates vertically into the water column at night (Sitts and Knight 1979a). Adult and juvenile P. macrodactylus were observed in impingement samples predominantly at night (Chapter 4). Larvae and early juveniles were similarly observed predominantly at night in entrainment samples (Chapter 3), demonstrating that these life stages are also more active in the water column at night than during the day.

Palaemon macrodactylus spawns in Suisun Bay and the western Delta from mid-April to late October or November (Little 1969; Sitts and Knight 1979a). The females carry the eggs in a brood mass for a period of about a month, after which the eggs hatch into planktonic larvae. The larvae are abundant throughout the spawning season from San Pablo Bay to the western Delta. Peak densities were observed in Suisun Bay and the western Delta during the spring and summer (EA 1981f, h).

Palaemon macrodactylus is omnivorous. Its major prey include Neomysis mercedis and Corophium spp., but it also consumes organic detritus, P. macrodactylus larvae, Rhithropanopeus harrisii zoeae, copepods, polychaetes, small bivalves, and larval fish (Siegfried et al. 1979; Sitts and Knight 1979b). Predators of P. macrodactylus include sturgeon, white catfish (Ganssle 1966), and probably other game fish.

2.2.7.1.7 Rhithropanopeus harrisii

Rhithropanopeus harrisii is a small, introduced crab common in Suisun Bay and the western Delta. Native to East Coast estuaries, R. harrisii inhabits the intertidal and shallow subtidal zones of sloughs and bays with muddy banks (Smith and Carlton 1975), or on and under rocks as a member of the fouling community (Carlton 1978). The crabs were collected year-round, but in greatest numbers in September-October (Chapter 4). Competition with the shore crab Hemigrapsus oregonensis, rather than salinity tolerance, may limit the distribution of R. harrisii to the areas in and upstream of Carquinez Strait (Carlton 1978).

Rhithropanopeus harrisii spawn from late spring to fall. The larvae (zoeae) were observed in entrainment samples collected at the Pittsburg Power Plant throughout the spawning season, with maximum densities being observed during the summer months. In the 1978 plankton surveys of the Delta and Suisun Bay, the zoeae were first collected in mid-June, and were found in greatest abundance in Suisun Bay and as the summer progressed were present in the lower Sacramento and San Joaquin rivers (EA 1981a). The zoeae metamorphose into non-planktonic juveniles (megalopae) within several weeks.

Rhithropanopeus harrisii is omnivorous, probably feeding on algae and other aquatic vegetation as well as small benthic invertebrates, both living and dead. The crab is in turn preyed upon by white sturgeon, white catfish, and striped bass (Carlton 1978), and perhaps other fishes and birds.

2.2.7.2 Fishes

2.2.7.2.1 Northern Anchovy (Engraulis mordax)

The northern anchovy is a small (to 18 cm; 7 in.), slender schooling fish that is one of the most abundant fish species in coastal waters, particularly off Baja California and southern California (Frey 1971). Three distinct subpopulations are found off the California coast; the central and northern stocks overlap in the San Francisco Bay region. Most adult anchovy are found in offshore waters, but eggs, larvae, and juveniles are found in coastal waters and in embayments and estuaries, including the Bay-Delta.

Northern anchovy is the dominant fish species from the south Bay to Carquinez Strait throughout much of the year, and in Suisun Bay in midsummer. Abundance is greatest in the central Bay and San Pablo Bay during the midsummer months, according to the results of 12 years (1967-78) of midwater trawl surveys conducted by the California Department of Fish and Game (CDF&G unpublished).

Most anchovy mature at one year and spawn two or three times per year thereafter until they are up to five years old (Baxter 1967; Collins 1969). They spawn in coastal and Bay waters, mainly during the spring and fall months. Large females may produce 20,000-30,000 eggs per year (Baxter 1967).

Larval anchovy are abundant in San Francisco Bay throughout much of the year, and are found as far upstream as Suisun Bay during the summer months (EA 1981a, h); juveniles in schools are found throughout the higher-salinity areas of the south and central Bay and San Pablo Bay, and some juveniles penetrate into the upper estuary (CDF&G unpublished). However, the bulk of the juvenile population is found in coastal waters.

Anchovy are planktivores at all life stages. Small zooplankton, particularly copepods, are the major food of larval anchovy. Juveniles consume copepods, small crustaceans, and other small zooplankton. Adults eat larger zooplankton and small fish, including their own eggs and larvae (Brewer 1978). Anchovy in turn are prey at every life stage to nearly all predatory fishes, birds, and mammals found in coastal and Bay waters (Fitch and Lavenberg 1975; NOAA 1978). Known predators in the Bay-Delta include chinook salmon, striped bass, surfperches, white croakers, and rockfishes.

Northern anchovy are subject to an intensive commercial fishery but no sport fishery. The commercial fishery, centered in southern California, annually harvests up to one-third of the total spawner biomass. Fish are processed into fish meal, fertilizers, fish bait, and other products not directly consumed by humans (NOAA 1978). San Francisco Bay region annual landings from 1972 to 1976 averaged 153,000 kg (700,000 lbs) (CDF&G, Marine Fish Landings 1974, 1975, 1976b, 1977, 1979). Up to 90 percent of that catch originates within the Bay (Smith and Kato 1979). The San Francisco Bay region catch has not exceeded 5 percent of the statewide landings since the fishery of anchovy for reduction to fish meal began in 1966.

2.2.7.2.2 Pacific Herring (*Clupea harengus pallasi*)

The Pacific herring is a small (to 25 cm; 10 in. in California waters), slender, schooling fish found in coastal waters from northern Baja California to Alaska. Pacific herring are seasonally abundant in San Francisco Bay, Tomales Bay, and Monterey Bay.

San Francisco Bay is the southern limit of substantial Pacific herring spawning (Alderdice and Velsen 1971). Mature at two years of age, Pacific herring enter San Francisco Bay to spawn beginning in November. Spawning continues through February in intertidal and subtidal areas from the Golden Gate to Berkeley and

Richmond, and during low rainfall years may extend upstream into San Pablo Bay and Carquinez Strait (Ganssle 1966). The adhesive eggs are deposited onto eelgrass, algae, intertidal rocks, pilings, or other forms of substrate. Each female produces 10,000-125,000 eggs (Fitch and Lavenberg 1975).

Pacific herring larvae are planktonic and are the most abundant fish larvae in the central Bay during the winter and early spring months (Eldridge 1977). The occurrence of larvae as far upstream as the Pittsburg Power Plant (EA 1981b) suggests that larvae may have moved upstream on flood tides from downstream spawning grounds in the central Bay and San Pablo Bay. Most herring larvae move out of the Bay-Delta soon after hatching, but some remain in the system until the late summer or early fall, when they move out to the ocean as juveniles (EA unpublished data).

Pacific herring of all ages feed on zooplankton (Fitch and Lavenberg 1975). Herring eggs are eaten by diving birds, gulls, crabs, and other fishes, which may result in up to 90 percent cropping of spawn in San Francisco Bay (Spratt 1976). Larvae, juveniles, and adults are preyed on by birds, marine mammals, fish, and man (Cleaver and Franett 1946; Fitch and Lavenberg 1975). Herring juveniles constituted a small component in the diet of sub-adult striped bass in the western Delta (Stevens 1966b).

Pacific herring have been fished commercially during several brief periods in the past, and presently support the second largest fishery (behind Dover sole) in the San Francisco Bay region (Hardwick 1973; CDF&G, Marine Fish Landings 1974, 1975, 1976b, 1977, 1979). Since the roe and bait fisheries began in 1974, the San Francisco Bay catch has averaged 1 million kg (2.3 million lb), or 61 percent of the statewide catch. The catch represents an average of 5 percent of the estimated total Bay spawner biomass: 23 million kg (51 million lb).

2.2.7.2.3 Threadfin Shad (Dorosoma petenense)

The threadfin shad is a small (to 10 cm; 4 in.), schooling fish recently introduced into California reservoirs as forage for game fishes (Kimsey 1954). It has since become well-established in the freshwater and oligohaline (0.5-5 ppt) portions of the Bay-Delta. During the late summer and early fall, threadfin shad was the most abundant fish collected by CDF&G midwater trawls in the Delta (CDF&G unpublished).

Most threadfin shad mature at age two and spawn from April through August. It may be that spawning is only successful in fresh water (Bryan and Sopher 1969). Females produce 900-21,000 adhesive eggs (Johnson 1971), which are attached to submerged vegetation, logs, or brush.

Larval threadfin shad are planktonic, and in 1978 and 1979 they were found in greatest abundance in the Delta during June (EA 1981a). Peak densities were observed in the main, Old River, and Middle River channels of the San Joaquin.

Juveniles congregate in small schools and inhabit inshore shallow waters where vegetation is dense. Juveniles are abundant in the Delta and Suisun Bay during the late summer and fall months (Turner 1966c; Ganssle 1966; EA 1981c).

Threadfin shad are planktivores at all life stages. Larvae eat phytoplankton, rotifers, and insect larvae. Juveniles and adults eat those organisms as well as planktonic crustaceans and small benthic organisms, such as worms and the clam Corbicula formosa (Hendricks 1961; Turner 1966c). Threadfin shad compete for food with other species, such as striped bass, minnows, suckers, and sunfishes, and may adversely affect their larval survival and growth (Moyle 1976). On the other hand, threadfin shad constitute a major item in the diets of juvenile, sub-adult, and adult striped bass and of black crappie (Stevens 1966b; Turner 1966a).

Threadfin shad support a small bait fishery in the Delta (140 kg; 305 lb in 1975), but are not sought by sport fishermen.

2.2.7.2.4 Delta Smelt (Hypomesus transpacificus)

The Delta smelt is a small (to 12 cm; 5 in.), schooling fish whose native range and present distribution is limited to the low-salinity waters of the Bay-Delta. The species is abundant much of the year from eastern Suisun Bay to the lower reaches of the Sacramento and San Joaquin rivers (CDF&G unpublished). Delta smelt are occasionally found west of Suisun Bay when surface salinities are less than 10 ppt (Ganssle 1966; Messersmith 1966).

Adults are believed to move upstream into the slow-moving waters of the deadend sloughs and channels of the Delta during the winter and spring (primarily February and March) to spawn (Moyle 1976). Spawning has been observed in April and May as

far upstream as Hood on the Sacramento River (Schaffter 1980). Mature at one year of age, Delta smelt females produce 1,400-2,800 adhesive eggs. Most adults die after spawning. Because of the limited native range, short life cycle, and relatively low fecundity, the population is subject to considerable year-to-year variability (Moyle 1976).

Delta smelt larvae are planktonic and are found in surface waters. Those spawned in upstream channels are rapidly transported to the western Delta and Suisun Bay, whereas those spawned in deadend sloughs may remain there until they move downstream as juveniles. Delta smelt and unidentified osmerid larvae were observed in 1978 in the north Delta waterways (Cache Slough, Steamboat Slough, Sacramento Deepwater Ship Channel), as well as in the western Delta, Suisun Bay, and Montezuma Slough from late April (the time of the first survey) to mid-June (EA 1981a).

Larval Delta smelt primarily eat planktonic copepods. Adults eat cladocerans, amphipods, opossum shrimp (Neomysis mercedis), other crustaceans, and insect larvae. Smelts are important forage fishes, but little is known regarding their predators. There are no sport or commercial fisheries for this species.

2.2.7.2.5 Longfin Smelt (Spirinchus thaleichthyes)

The longfin smelt is a small (to 14 cm; 6 in.), schooling fish common in estuarine waters from the Bay-Delta northward to Alaska (Miller and Lea 1972, 1976). The species is abundant much of the year in the Bay-Delta in mesohaline (5-18 ppt) waters from San Pablo Bay to the western Delta (Ganssle 1966; CDF&G unpublished). Moyle (1976) reported that the bulk of the population is found in salinities greater than 10 ppt.

Longfin smelt mature in their second year, spawn, and usually die after spawning. Mature fish move from San Pablo Bay into Suisun Bay and the western Delta during the fall and spawn from December to April in salinities less than 10 ppt (Radtke 1966). Spawning adults have been observed as far upstream as Hood on the Sacramento River (Schaffter 1980). Females produce from 5,000 to 24,000 adhesive eggs (Moyle 1976), which are deposited on vegetation, rocks, or other available substrate.

Planktonic larvae are abundant from San Pablo Bay to the western Delta during the winter and spring months. Peak densities were observed in April and May by Ganssle (1966), but during plankton sampling near the Pittsburg Power Plant in 1978-1979, peak densities of unidentified osmerid larvae (presumably longfin and Delta smelt) were observed from January through April (EA 1981h). Longfin smelt and unidentified osmerid larvae were observed in 1978 in the north Delta waterways, western Delta, Suisun Bay, and Montezuma Slough from late April (the time of the first survey) to mid-June (EA 1981a).

Larvae and small juveniles feed primarily on copepods and other crustaceans, but larger juveniles and adults primarily eat mysids. Although the longfin smelt has not been reported to be a major component of the diet of any Bay-Delta fishes, it is considered to be important in the estuarine food web and may compete with other species, such as striped bass, for N. mercedis. There is limited sport and no commercial fishery for this species.

2.2.7.2.6 Yellowfin Goby (Acanthogobius flavimanus)

The yellowfin goby is a recently introduced Asian estuarine fish found in benthic and epibenthic habitats throughout much of the Bay-Delta, from the south Bay to the upper reaches of the Delta (Brittan et al. 1970; Moyle 1976). The species is also found in Elkhorn Slough, Tomales Bay, and Los Angeles Harbor (Miller and Lea 1972, 1976; Haaker 1979; EA 1981i). It is the largest goby in California (up to 25 cm; 10 in.) and the population has undergone a rapid expansion in size and distribution since its accidental introduction (Brittan et al. 1970).

Yellowfin goby mature by the end of their first year and spawn in winter and early spring in the more saline ($\geq$18 ppt) areas of the Bay-Delta (EA unpublished data). Many yellowfin goby die after spawning, however, some individuals have been reported to spawn several times during a three to four year life span in Japan (Miyazaki 1940). The species is more fecund than the native gobies, each female producing as many as 18,000 eggs (Okada 1960; Brittan et al. 1963; EA unpublished data). The eggs are adhesive and are laid in burrows.

Yellowfin goby larvae leave the burrows and become planktonic. The young larvae cannot be distinguished from other goby larvae, and therefore it is not possible to

specify their distribution. Surveys made in Suisun Bay and the Delta from April to July 1978 indicated that yellowfin goby late larvae and early juveniles were abundant in the plankton from early May to mid-June (EA 1981a).

Yellowfin goby juveniles gradually settle onto the bottom, where they cling to substrates and aquatic plants. They are abundant throughout the Bay-Delta (EA unpublished data).

Larval goby feed on copepods, and the demersal juveniles and adults eat copepods, ostracods, amphipods, crabs, shrimp, and other fish (Haaker 1979; Moyle 1976; EA unpublished data). Predators of yellowfin goby are poorly known, although marine birds are reported to eat them (Brittan et al. 1970). No commercial fishery exists for the species, and few are taken by sport fishermen.

Yellowfin goby may threaten some of the native fishes of the Bay-Delta. The species is tolerant of wide fluctuations of temperature and salinity and, being very invasive into new areas, may be displacing and/or competing for food with the staghorn sculpin and the native tidewater goby (Brittan et al. 1970; Moyle 1976). The high fecundity of yellowfin goby may offer a competitive advantage over other, less-fecund native gobies.

2.2.7.2.7 Prickly Sculpin (*Cottus asper*)

The prickly sculpin is a small (to 17 cm; 7 in.) bottom fish common in inland freshwater habitats in the Delta and also present in the oligohaline (0.5-5 ppt) portions of Suisun Bay (Ganssle 1966; Miller and Lea 1972, 1976).

Prickly sculpin mature at 2-4 years of age and may spawn for several years thereafter. Adults move downstream into the tidal areas of the Delta to spawn from November to June. Most spawning occurs from March to May (Moyle 1976). Females produce from 300 to 11,000 adhesive eggs and deposit them in nests prepared by the males (Patten 1971; Moyle 1976). After the eggs are hatched, most adults return to upstream areas.

Newly-hatched larvae remain in the nest several days before becoming planktonic. Prickly sculpin larvae were collected throughout the Delta and Suisun Bay in the spring of 1978 and 1979 (EA 1981a).

Juveniles remain in the tidal portions of the Delta until the early summer of the year following their birth, when they ascend the tributaries of the Delta.

Young prickly sculpin (under 30 mm) feed primarily on planktonic crustaceans. Juveniles and adults eat aquatic insects, isopods, amphipods, molluscs, small fish, and their own larvae (Moyle 1976). They also eat salmon and trout eggs and larvae in streams, but are not considered to be a major salmonid predator. Salmon, trout, and other large fishes in turn eat prickly sculpin (Moyle 1976). There are no sport or commercial fisheries for this species.

2.2.7.2.8 Starry Flounder (Platichthys stellatus)

The starry flounder is a moderate-sized (up to 90 cm; 36 in.) flatfish that is common in the Bay-Delta and is unique among the flatfishes in that it is capable of existing in freshwater for a portion of its life. Starry flounder were collected from 1967 to 1977 by midwater trawl (an ineffective sampling gear for flatfish) mainly from San Pablo Bay to Suisun Bay and Montezuma Slough (CDF&G unpublished). The species was also collected in the 1978-1979 fishery survey at the Pittsburg Power Plant (EA 1981c). They are, however, also resident in coastal waters and in ocean-dominated bays.

Starry flounder mature at age two (males) or age three (females), and move inshore to spawn from November to February (Orcutt 1950) in coastal waters. No starry flounder eggs were collected in bay waters during plankton sampling in the western Delta (EA 1981a, b, h). Females are highly fecund, each freely broadcasting several million planktonic eggs (Orcutt 1950).

Starry flounder larvae remain planktonic, and are dispersed throughout the Bay-Delta. Larvae may use tidal currents to ascend the estuary from the spawning grounds. Larvae have been observed in Richardson Bay during March and August (Eldridge 1977), and in Suisun Bay, Montezuma Slough, and the lower Sacramento and San Joaquin rivers from April (the time of the first survey) to mid-July (EA 1981a).

Larvae metamorphose into juveniles at a length of about 11 mm (Orcutt 1950) and gradually settle on the bottom. Juveniles are abundant in the western Delta (Radtke 1966), and some have even appeared in San Luis Reservoir, which is filled by Delta water transported by the California Aqueduct (Moyle 1976).

Starry flounder feed on phytoplankton and planktonic crustaceans during the larval stage. Small juveniles eat amphipods, and larger flounders eat crabs, molluscs, and polychaetes (Orcutt 1950). In fresh water, insect larvae are also eaten (Moyle 1976).

The starry flounder supports a modest commercial fishery (Frey 1971), and is a popular fish taken by anglers in San Francisco Bay. Annual San Francisco Bay region commercial landings for the period 1972-76 averaged 109,000 kg (240,000 lb), or about one-third of the statewide total (CDF&G, Marine Fish Landings 1974, 1975, 1976b, 1977, 1979).

2.2.7.2.9 Striped Bass (Morone saxatilis)

The striped bass is a large (up to 1.25 m (49 in.); 41 kg (90 lb)), introduced fish abundant in the Bay-Delta and subject to the region's most intensive sport fishery. Native to the Atlantic and Gulf of Mexico coastal waters, estuaries, and rivers, striped bass quickly became established in the Bay-Delta after their introduction in 1879 and 1882. The Bay-Delta population remains the only major one on the Pacific Coast, and recent reductions in the striped bass population have prompted considerable interest in the environmental requirements of the species.

Adult striped bass use the Bay and coastal waters for feeding during the summer months, but in fall they move upstream, passing through Carquinez Strait in October and concentrating in Suisun Bay in November (Chadwick 1967). In spring, mature bass move upstream to spawn in the Sacramento River between Colusa and Sacramento and in the San Joaquin River between Antioch and Venice Island (Turner 1976). Spawning generally occurs from April through early June.

Males mature between age one and three and females usually between age four and six (Scofield 1931; Calhoun and Woodhull 1948). Both sexes commonly live and reproduce until they are 12-15 years old (females up to 30 years old), but older bass probably do not spawn every year (Merriman 1941; Moyle 1976). Spawning is apparently triggered as water temperatures approach 15 C (Farley 1966). San Joaquin River spawning generally precedes Sacramento River spawning because of warmer water temperatures in the San Joaquin.

Striped bass females, like most pelagic spawners, are prolific. Females have been

reported to contain 68,000 - 4.5 million eggs (Jackson and Tiller 1952). In the Bay-Delta, 5- to 9-year-old bass produce about 75 percent of the eggs each year (CDF&G unpublished). Eggs are planktonic and non-adhesive, and can be suspended by water currents greater than 30 cm/sec (1 fps). Suspension and initial exposure to fresh water are both essential for high egg survival (Albrecht 1964; Turner and Farley 1971). Once water hardening occurs, the eggs can tolerate salinities up to 10 ppt. Eggs hatch within 36-48 hours after fertilization when water temperatures are 17-18 C (62-65 F) (Mansueti 1958).

Because the eggs hatch quickly, they are not distributed far from the spawning areas. In the Sacramento River, most eggs hatch between Sacramento and Courtland (Turner 1976). In the San Joaquin River, where downstream flows are of lesser intensity and high tides and water exports cause reverse flows, eggs hatch mainly within the spawning area. Eggs tend to be concentrated near the bottom in midchannel (Farley 1966).

Larvae (<6 mm) are quickly carried out of the upper Sacramento River into the Delta. Most enter the lower Sacramento River, but may pass into the lower San Joaquin River via the Delta Cross-Channel and Georgiana Slough. Larvae become distributed throughout the western Delta and Suisun Bay above the entrapment zone. Larval densities decrease sharply at the upper end of the entrapment zone (known as the null zone), and few small larvae were collected below that point in the 1978 and 1979 river surveys (EA 1981a). Larger larvae (<16 mm) and small juveniles were collected as far downstream as the middle of the entrapment zone (EA 1981a). The location of peak abundance ranges from Carquinez Strait to the western Delta, depending on the location of the entrapment zone, which in turn is determined by the Delta outflow (Section 2.2.1). Small larvae (<7 mm) are distributed fairly evenly across the main channels of the western Delta (EA 1981a, b, d, f, h). Larger larvae and juveniles tend to concentrate near the shoreline (EA 1981a).

Young striped bass remain in and above the entrapment zone during their first summer, but migrate into San Pablo Bay and the central Bay in fall (Sasaki 1966b; Ganssle 1966). Some young bass remain in the Delta during the winter, and these are concentrated in the lower San Joaquin River.

Striped bass are pelagic predators which feed on a variety of organisms. Larval striped bass initially prey on copepods, such as Eurytemora affinis, Cyclops spp.,

Pseudodiaptomus euryhalinus, and Acartia clausi. Larger bass (>25 mm) feed primarily on Neomysis mercedis in the lower Sacramento and San Joaquin rivers, but also eat the amphipod Corophium spp. In the south Delta during the summer, copepods and cladocerans are the dominant food items (Thomas 1967). During the fall, N. mercedis become less important in the diet of the young bass in the lower San Joaquin River, and Corophium spp. become very important in the diet of young bass in the south Delta. Young bass in San Pablo Bay in the fall eat primarily N. mercedis, nereid worms, and amphipods. Young bass are generally not very selective, and they feed on the various zooplankton species in proportion to their abundance at any given time (Heubach et al. 1963).

Juvenile and adult bass are also opportunistic predators; they are piscivores, and the species eaten depends on what is abundant at the time. In spring and summer, northern anchovy is the most important prey for bass in the south and central Bay and in San Pablo and Suisun bays. Pacific herring is an important food item during the winter months when they enter the Bay to spawn. During the spring, salmon smolts are important in the diet of adult bass in the lower Sacramento River (Thomas 1967). The shrimp Crangon spp. and mysids are also important food items of yearling and older striped bass. Species of fish that are likely to prey on young bass in the Delta include channel catfish, largemouth bass, and warmouth (Turner 1966a, b; Sasaki 1966a). These species are found upstream of the center of young bass abundance. In Suisun Bay, young bass are preyed on by older striped bass and by Sacramento squawfish (EA unpublished data). Some invertebrates, such as N. mercedis, Palaemon macrodactylus, and certain copepods (e.g., Cyclops spp.), may also feed on larval striped bass.

Striped bass were initially the object of an intensive commercial fishery, which by 1903 harvested 900,000 kg (2 million lb) per year. The catch remained stable until 1915, but declined thereafter, in part because of changing regulations. The catch fluctuated between 200,000 and 450,000 kg (450,000-1 million lb) per year until 1935, when commercial fishing for striped bass in San Francisco Bay was banned (Skinner 1962; Stevens 1980).

The striped bass sport fishery is the most important in the Bay-Delta. Approximately two million angler days per year are spent fishing for the species (Turner 1972), and in recent years about 300,000 fish have been caught annually. In the early 1960s the annual catch was about 750,000 fish (D. Stevens, CDF&G, personal communication,

1980). The decline in the sport catch in recent years has been attributed to a decrease in bass abundance associated with low recruitment in most years since 1961 (Stevens 1977).

2.2.7.2.10 Chinook Salmon (Oncorhynchus tshawytscha)

Chinook (king) salmon, the largest Pacific salmon, typically reaches lengths of 70-80 cm (28-31 in.) and weights of 10-20 kg (22-50 lb) (Moyle 1976). Although chinook salmon are found as far south as Point Conception, the species spawns predominantly from the Bay-Delta northward to Alaska. The Bay-Delta population is the largest in California (Hoopaugh 1978).

Adult chinook salmon enter the Bay-Delta to spawn in three major runs: spring, fall, and winter. Consequently, large numbers of adults are present in the Bay-Delta for much of the year. In recent years the spring run has declined substantially. The majority of the salmon ascend the Sacramento River and its tributaries, but some (particularly fall-run fish) ascend the San Joaquin River system and the Mokelumne and Cosumnes rivers, and are propagated at hatcheries (Frey 1971; Hart 1973). Most salmon return to spawn in the same stream in which they were hatched; visual and olfactory cues permit generally accurate homing. Spawners are usually 4-5 years old, although many are three years old, and some immature males and mature fish up to eight years old also join the spawning runs (Frey 1971; Hart 1973). Chinook salmon spawn in cool (<14 C), gravel-bottom areas of streams and rivers. Females bury 2,000-14,000 eggs in loose gravel nests. Both females and males die after spawning (Moyle 1976).

Larvae emerge from the nests within about two weeks after hatching and begin to move downstream. A few larvae were collected in the western Delta during plankton and fishery sampling (EA 1981b-h), indicating that larvae are likely to remain in the upper tributaries adjacent to the spawning grounds for an additional one to two months before they begin to migrate to the sea.

Large numbers of small juveniles spend several months in the Delta before migrating out to the ocean (Moyle 1976). In the Delta, emigrant juveniles are most abundant in the Sacramento and Mokelumne rivers and in the north Delta channels. Juveniles are also found, in lower numbers, in the San Joaquin River and the south Delta channels, but many of these are believed to have originated in the Sacramento River and to

have been drawn into the south Delta by the reverse flows caused by the export of water at the south Delta pumping facilities (Sasaki 1966a). Thousands of juveniles are collected at the pumping facilities and subsequently released downstream in the Delta.

Most juveniles emigrate from the Delta from April to June, but some spring-run emigrants (smolts) leave as early as January and some yearling smolts leave in October and November (Jensen 1972; Fry 1973; SWRCB 1978). Peak numbers of smolts are observed in San Pablo and Suisun bays and Carquinez Strait during April-June, and lesser numbers in late fall (Ganssle 1966; Messersmith 1966).

Juvenile chinook are opportunistic feeders while in streams and in the Delta. Terrestrial and aquatic insects are the most important prey in streams. In the Delta, terrestrial insects, midge larvae, amphipods, and mysids are the most important prey (Sasaki 1966a). Adults caught in the ocean off San Francisco Bay were feeding on anchovy, rockfish, euphausids, Pacific herring, squid, other fishes, and crab megalopae (Merkel 1957). Predators of juveniles in the rivers and the Delta include striped bass, steelhead, American shad, and sturgeon (Stevens 1966b; D. Painter, CDF&G, personal communication, 1980). Stevens (1966b) reported that salmon constituted from 3 to 10 percent of the diet of subadult and adult striped bass in spring and summer. Predators of the adults include marine mammals such as seals, sea lions, and killer whales.

Chinook salmon supported a commercial fishery in the Bay-Delta for over one hundred years until 1957, when commercial fishing for anadromous fish in the Bay was banned. The commercial fishery now exists only outside of the Bay. The San Francisco Bay region catch in recent years (1972-1976) has ranged from 0.77-1.27 million kg (1.7-2.8 million lb) (CDF&G, Marine Fish Landings, 1974, 1975, 1976b, 1977, 1979).

Sport fishermen have been responsible for about 14 percent of the total salmon catch each year since 1950 (Frey 1971). The statewide ocean sport catch annually averages about 100,000 fish and the inland sport catch annually averages about 25,000 fish (Moyle 1976; SWRCB 1978). The estimated Bay-Delta spawning stock has averaged about 305,000 during the period 1966-1976, and has ranged from 181,000 in 1967 to 400,000 in 1971 (CDF&G Salmon Spawning Stock Estimates, various years).

Although the chinook salmon has been one of the most important commercial and sport fish in California, the population size of this native fish has been declining noticeably in the past hundred years, mainly because of the dams that block their upstream spawning migration routes (Moyle 1976). The artificial propagation program operated by governmental agencies has compensated for some of the losses, but the future of this anadromous fish is still uncertain.

2.2.7.2.11 American Shad (Alosa sapidissima)

The American shad is a moderate-sized (to 76 cm; 30 in.), introduced, anadromous fish that has become established in estuaries and rivers from the Bay-Delta northward to Alaska (Fry 1973; Leggett and Whitney 1971). Since the species was introduced in the 1870s, it has become one of the more abundant species in the Bay-Delta.

Adult shad enter the Bay from February to April and ascend the tributaries of the Delta to spawn from April to June. Major spawning grounds include the Sacramento River, from Rio Vista to Redding, and its tributaries. Very little spawning is believed to occur in the San Joaquin River system, based on fishing data collected by the California Department of Fish and Game (M. Meinz, CDF&G, personal communication, 1980). The mean age at first spawning is four years for males and five years for females. Shad have been known to spawn in as many as seven consecutive years, but repeat spawners make up only about 30 percent of the spawning population (M. Meinz, CDF&G, personal communication, 1980). Lifetime fecundity of American shad females has been estimated at 320,000-390,000 eggs (Leggett and Carscadden 1978). Shad eggs are freely broadcast; they slowly sink in the river while being carried downstream from the spawning areas, and they hatch in 4-6 days (Jones et al. 1978; SWRCB 1978).

American shad larvae are planktonic. Larvae and young juveniles are found in greatest abundance in the Sacramento River (from the Delta Cross Channel to a point 80 km, or 50 mi, above Chipps Island) and in the lower Mokelumne River (M. Meinz, CDF&G, personal communication, 1980). Larvae and young juveniles slowly move downstream into the northern Delta channels from May to July (EA 1981a).

Juvenile shad (80-180 mm; 3-7 in.) migrate into the Bay and eventually out to the

ocean from July to December. Juvenile shad were collected by midwater trawl in Suisun Bay and the Delta in July and August; in September and October, the numbers collected in San Pablo Bay, Carquinez Strait, and Suisun Bay increased, reflecting the downstream shad migration (CDF&G unpublished). By November and December, substantially fewer are collected in the Bay-Delta, confirming the late fall migration of shad out of the Bay to the ocean.

American shad are predominantly planktivores at all life stages. Larvae eat cladocerans (Maxfield 1953), and juveniles eat cladocerans, copepods, amphipods, Neomysis mercedis, and larval fish, including threadfin shad and salmon (D. Painter, CDF&G, personal communication, 1980). Adults eat the same organisms as the juveniles while they are present in the Bay-Delta (Ganssle 1966). Predators of shad eggs and larvae include larger American shad, catfish, minnows, suckers, and striped bass (Walburg and Nichols 1967; Kelley 1968). Juvenile shad are preyed on by striped bass (Stevens 1966b) and presumably other large fishes.

The American shad population increased rapidly following its introduction, and supported a commercial fishery that in some years landed over 0.45 million kg (1 million lb) until a 1957 ban on commercial fishing for anadromous fishes in San Francisco Bay (Skinner 1962). During the past 25 years, a sport fishery has developed, and present annual landings are from 110,000 to 160,000 fish. The sport fishery is concentrated in the Sacramento River and its tributaries, the American, Feather, and Yuba rivers. The adult population size in 1976 has been estimated at 1.48 million and in 1977 at 2.33 million (D. Painter, CDF&G, personal communication, 1980). Thus, the sport fishery crops only a small fraction of the adult stock.

2.2.7.2.12 White Sturgeon (Acipenser transmontanus)

White sturgeon is a native anadromous species which can grow to 2.8 m (9 ft) in length and weigh 210 kg (460 lb) and is the largest fish found in the fresh waters of California. White sturgeon are restricted to the coastal waters, estuaries, and rivers of the Pacific Coast from Ensenada, Mexico, to the Gulf of Alaska. Adult white sturgeon are found throughout the Bay-Delta, but are concentrated during most of the year in lower Suisun Bay and San Pablo Bay. Some adult Bay-Delta sturgeon make long ocean migrations (Miller 1972), but the extent and purpose of these migrations and the factors that initiate them are not known.

White sturgeon migrate into fresh water to spawn, and in California the only large spawning runs occur in the Sacramento and Feather rivers; there are lesser runs in the Russian, San Joaquin, Klamath, and Trinity rivers. Spawning migrations in the Sacramento River occur from February to May (Miller 1972). Age at maturity ranges from 11 to 34 years for females and from 11 to 22 years for males (Moyle 1976; Scott and Crossman 1973). White sturgeon do not spawn every year; intervals between spawning may be 4-11 years or more (Semakula and Larkin 1968). Fecundity is very high, estimates ranging from 100,000 eggs in 1.1-1.5 m fish to 4.7 million eggs in a 2.8-m fish. The adhesive eggs are spawned over gravel substrates from March to early May in the Sacramento River from Knights Landing to just above Colusa. Adults move downstream to the estuary when spawning is completed.

Larval sturgeon remain near the bottom and gradually move downstream into the Bay-Delta (Kohlhorst 1976). The rivers below the spawning area and the Delta are both important nursery areas, with the latter increasing in importance in years of high freshwater flow (Stevens and Miller 1970). Juvenile sturgeon primarily inhabit Suisun Bay and the western Delta, according to the CDF&G 12-year (1967-1978) midwater trawl study .

In the Bay-Delta juvenile sturgeon feed mostly on amphipods and on mysids. Their diet becomes more varied as they become larger, but still consists primarily of bottom-dwelling estuarine invertebrates, including clams, crabs, and shrimp. White sturgeon also eat other fish, especially herring, anchovy, striped bass, starry flounder, and smelt. They feed heavily on herring eggs when available (Moyle 1976). The fast growth and large size of sturgeon probably make them unimportant as a food resource to other fish species inhabiting the Bay-Delta.

White sturgeon have long been one of the most important fishery resources in the Bay-Delta. Commercial fishing in the late 1800s and early 1900s severely depleted the stocks of white sturgeon (Skinner 1962). This prompted a complete closure the fishery from 1917 to 1954, when sport fishing was again permitted. The Bay-Delta adult stock, which has fluctuated from approximately 20,000 to 100,000 in recent years, is being harvested at an annual rate of 6-7 percent, a rate which is thought to maintain the population adequately (Kohlhorst 1981).

2.2.8 Fishery Surveys in the Vicinity of the Pittsburg Power Plant

A number of field studies have been made of the larval, juvenile, and adult populations of fish near the Pittsburg Power Plant that can provide insight into the local aquatic community in eastern Suisun Bay and the western Delta. Among these are work by Gritz (1971), Steitz (1975), PGandE (1973), and EA (1981a-g). PGandE (unpublished c), and EA (1981h) have surveyed the southern shore of Suisun Bay about 2-1/2 mi (4 km) west of the Pittsburg Power Plant. This section will review the results of those studies to give something of an overview of the fish communities in the areas near the plant.

Plankton sampling at the Pittsburg Power Plant intake from May through October in the dry year 1976 (PGandE unpublished b) indicated that larval and early juvenile striped bass were present from late May through August, with peak abundances from the first week of June into the first week of July. Sizes ranged from a mean of 6-10 mm in late May and early June to 14-24 mm in early July.

Plankton surveys were conducted six times from May 1978 to April 1979 at three sampling stations in Suisun Bay near the plant (EA 1981b). The dominant taxa in the collections were striped bass, smelts, sculpins, and northern anchovy, which constituted 93 percent of the larval and juvenile fish collected. Larval and juvenile striped bass showed the highest densities of all taxa, and reached maximum densities in May. Striped bass yolk-sac larvae, larvae, and juveniles were collected in higher densities near the shoreline than in the main channel. Striped bass larvae and juveniles were generally collected in higher densities in samples 1 m above the bottom than in samples collected from the entire water column.

In the most recent fishery study, conducted near the Pittsburg Power Plant from August 1978 through July 1979, sampling was done with a variety of types of gear, both active and passive, in order to sample the varied habitats (EA 1981c). Beach seines, fyke nets, and electrofishing equipment were used in the shallows near the shore; electrofishing was especially effective in submerged vegetation. Gill nets, and an otter trawl sampled the demersal or more active species of fish in somewhat deeper waters. In 118 sampling efforts evenly distributed over the year, the 10 most abundant species collected were striped bass (45.3 percent of the whole year's catch), splittail (13.9 percent), tule perch (8.4 percent), threadfin shad (6.6 percent), yellowfin goby (5.7 percent), Sacramento squawfish (5.6 percent), inland silverside

(4.2 percent), Delta smelt (1.8 percent), carp (1.5 percent), and chinook salmon (1.2 percent). Of those species, two are anadromous and use the area as a nursery for young-of-the-year: striped bass, which use it extensively, and chinook salmon, which use it to a small extent. The rest of the abundant fishes are seasonal or permanent residents.

Striped bass, the most abundant species collected, were primarily young-of-the-year, although yearlings and adults were also collected. Striped bass were collected in both inshore and offshore sampling and were present throughout the year, suggesting that the area in the vicinity of the Pittsburg Power Plant serves as a nursery for juvenile bass. Yearling striped bass were present in collections throughout the year, indicating that a portion of the population resides in the area.

The species composition and general patterns of seasonal abundance for the species collected in the study (EA 1981c) are similar to those found in other fishery surveys conducted nearby during 1978-1979 (PGandE unpublished c; EA 1981e, g), and 1977-1978 (PGandE unpublished a). The findings of all these studies are consistent in suggesting that the area supports a diverse assemblage of introduced and native fish species. Furthermore, the survey results support the conclusion that the inshore habitat in the general area serves as an important nursery for chinook salmon, striped bass, and many non-game species.

The only endangered aquatic species whose distribution might include the study area is the thicktail chub. However, the last reported catch was from Steamboat Slough in the north Delta in 1957, and none has been caught since in the extensive fish collections made in the Delta or at the various water diversion pumping plants (CDF&G 1976a). Although not on Federal or State rare, threatened, or endangered species lists, the Sacramento perch, originally abundant in the Delta, has suffered a population decline since 1930, probably because of introduced competitors (Moyle 1976), and has not been collected in any of the more recent fish surveys.

Comparisons were made between the collection results from the most recent study (EA 1981c) and those from gill net and otter trawl collections made in the vicinity of the Pittsburg Power Plant during a one-year period in 1971 and 1972 (PGandE 1973). The dominant fish collected during the 1971-1972 survey were striped bass (78.7 percent of the total), white catfish (6.7 percent), carp (4 percent), and American shad (2 percent). A total of 17 species were collected (PGandE 1973). The gill net and

otter trawl results of the 1978-1979 fishery survey differed somewhat from those earlier collections: the dominant fish collected were striped bass (59.3 percent of the total), splittail (16.7 percent) and tule perch (7.8 percent). A total of 21 species were collected in the 1978-1979 surveys.

Although it is not directly comparable with results of the gill net and otter trawl collections, a creel census from the vicinity of the Pittsburg Power Plant (Steitz 1975) provides additional information on the composition of the fish populations in the area. The census, conducted between 28 June and 28 October 1974, indicated that striped bass represented 75 percent of the total sportfishing catch, followed in abundance by white catfish (15 percent), carp (6 percent), and channel catfish (1 percent). Other species caught included splittail, Sacramento squawfish, tule perch, black crappie, white sturgeon, and gobies.

A comparison of the results of the three fishery surveys from the vicinity of the Pittsburg Power Plant (based on the percentage of the total catch represented by selected species) is shown below:

	1971-1972 Gill net/ Otter trawl	1974 Creel census	1978-1979 Gill net/ Otter trawl
Striped bass	79	75	59
Splittail	1	2	17
Tule perch	1	0.3	8
White catfish	7	15	0.1
Carp	4	6	2
Channel catfish	0	1	0

Results of the 1971-1972 survey and the 1974 census are similar, except that the anglers reported a higher proportion of white catfish and carp. Comparing the 1971-1972 and 1978-1979 surveys, which used the same collection techniques, the relative abundance of striped bass and white catfish were lower for the 1978-1979 survey and the contribution of splittail and tule perch substantially higher.

An analysis of the data collected in the 1971-1972 survey suggested that, in general, striped bass, American shad, threadfin shad, and other fish species collected did not prefer warmer temperatures associated with the Pittsburg Power Plant thermal discharge (PGandE 1973). There was, however, some indication of increasing numbers of fish collected by otter trawl with increasing water temperatures.

Gritz (1971) examined the distribution of fish in relation to the thermal plume at the Pittsburg Power Plant and at control stations with ambient river temperatures. He found that seven species of fish, including striped bass, splittail, carp, and white catfish, were more abundant in the plume than out of it. The difference between catches in the plume and catches at ambient river temperatures were statistically significant for striped bass, carp, and splittail. Since no systematic sampling inside and outside of the thermal plume was conducted during the 1978-1979 survey (EA 1981c), no data are available from that study to confirm the observations made by Gritz.

The plankton and fishery surveys conducted in the vicinity of the Pittsburg Power Plant show that the area is populated by a diverse fish community which includes species typically found in freshwater and marine habitats. The area provides habitat for seasonal and resident populations, in addition to serving as a migratory pathway and nursery area for anadromous species.

2.3 REFERENCES

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Chapter 3

Entrainment

3.1 Entrainment Abundance

3.2 Entrainment Survival

3.3 Estimation of Total Entrainment Cropping at the Pittsburg Power Plant

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CHAPTER 3: ENTRAINMENT

Entrainment is one of two major ways in which aquatic organisms can be directly involved with the cooling water intakes of power plants; the other way is impingement, the subject of Chapter 4. Entrainment is the hydraulic capture and subsequent exposure of organisms to a cooling water system. The organisms involved are small, capable of passing through the 3/8-in (0.9-cm) mesh of the intake screens, and include phytoplankton, zooplankton, including early life stages of fish (ichthyoplankton), and macroinvertebrates. As these entrained organisms pass through the cooling system, they can be exposed to several types of stresses. These include mechanical stress from contact with the internal surfaces of pumps, pipes, and condenser tubes; pressure stress from exposure to relatively rapid pressure changes; shear stress from exposure to complex water flows; thermal stress from exposure to elevated temperatures in the condensers; and sometimes, chemical stress from exposure to biocides used in biofouling control. The environmental impact of a cooling water intake system is in part a function of the number of organisms that do not survive as a result of passage through the system.

A major objective of the 316(b) program for the Pittsburg Power Plant is to estimate the numbers and taxa of organisms lost during cooling system passage. The study plan (PGandE 1977) describes two studies required to achieve this objective. The first, entrainment abundance, was designed to estimate the total numbers and taxa of organisms entrained. The second, entrainment survival, was designed to estimate the proportion of entrained organisms not surviving passage through the cooling system. Combining the results of the two studies permits estimation of the total numbers of organisms cropped as a result of entrainment.

The Pittsburg Power Plant entrainment studies focused on planktonic early life stages of fish (ichthyoplankton) and on selected macroinvertebrate groups, especially amphipods, mysids, and decapods, as summarized in Chapter 1.

The results of the entrainment abundance and survival studies are presented in the following sections of this chapter. Entrainment abundance results are presented in Section 3.1, including estimated total numbers of ichthyoplankton and macroinvertebrates entrained at Units 1-7 during the study period. Section 3.1 also describes a study designed to determine the best location in the cooling system for

sampling entrainment abundance (i.e., the location from which the most representative estimates of total numbers entrained can be obtained). The entrainment survival study is described in Section 3.2. In Section 3.3, the total numbers of ichthyoplankton and macroinvertebrates cropped as a result of entrainment are estimated on the basis of the entrainment abundance and survival results. The data upon which Chapter 3 is based are presented in the data summary accompanying this report (Appendix E).

The estimates provided in this chapter of the total number of ichthyoplankton and macroinvertebrates cropped as a result of entrainment, along with similar estimates of impingement cropping (Chapter 4), form part of the foundation upon which the impact of cooling water withdrawal on the aquatic populations of the Sacramento - San Joaquin system is assessed in Chapter 5. These cropping estimates are also used in Chapter 6 in determining the best intake technology available for minimizing adverse impact.

3.1 ENTRAINMENT ABUNDANCE

The entrainment abundance study was designed to estimate the numbers of ichthyoplankton and macroinvertebrates entrained at Units 1-7. The study was also designed to determine (1) the species composition of the entrained biota, (2) the size (length) composition of the entrained ichthyoplankton, and (3) the seasonal and diel patterns of entrainment.

Total numbers entrained were estimated by sampling a portion of the cooling water flow once or twice a week for 16 months (March 1978 - July 1979), and then multiplying the densities of ichthyoplankton and macroinvertebrates observed by the volumes of cooling water passing through the plant.

Densities of entrained ichthyoplankton and macroinvertebrates observed at Unit 6 were used in estimating the numbers entrained at the other units. Because the two intake structures serving Units 1-7 are contiguous, no difference in entrainment between the two intakes was expected. Densities of entrained ichthyoplankton at Units 1-4, Unit 5, and Unit 6 were compared in a study described in detail in Section 3.1.2.2. Results of that study supported the hypothesis that densities at Unit 6 are representative of densities at the other units.

As specified in the study plan (PGandE 1977), entrainment abundance was sampled at

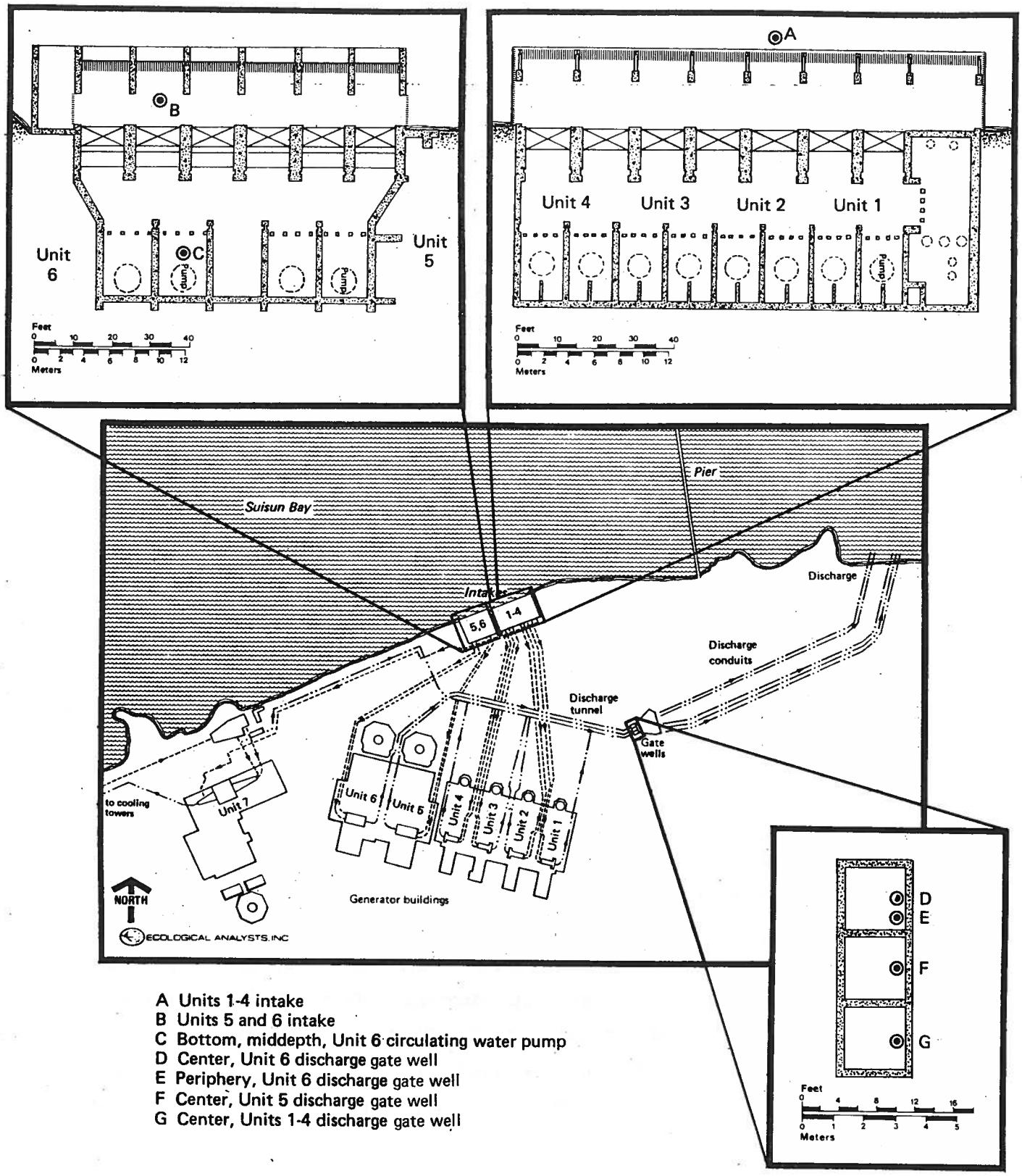
the discharge. Because of turbulence and through-plant mixing, organisms there were expected to be distributed more uniformly than in the intake. Samples collected at the discharge were expected to represent more accurately the true densities of entrained organisms than samples collected at the intake, provided that no losses of organisms were occurring during transit through the cooling system. The null hypothesis that organism densities were not different in the intake and discharge (i.e., that no through-plant losses of organisms were occurring) was tested. The results, presented in detail in Section 3.1.2.3, confirmed that no through-plant losses of organisms were occurring, and that, as expected, organisms were distributed more uniformly in the discharge than in the intake.

3.1.1 Methods

Entrainment abundance was sampled by filtering water pumped from the Unit 6 discharge gate well (D, Figure 3-1) during one 24-hour period per week from 23 March 1978 to 31 July 1979. Entrainment abundance estimates are divided into two periods, estimates for the first year of study are based on samples collected from 23 March 1978 to 12 March 1979, the second period includes samples from 19 March to 31 July 1979. During the peak entrainment season of striped bass (May-July), two 24-hour periods per week were sampled. Eight 3-hour samples were collected during each 24-hour sampling period through July 1978 and once a month thereafter; the rest of the samples were combined into 24-hour composites.

Major features of the sampling equipment are shown in Figure 3-2. Water samples were collected with a 4-in. (10.2-cm) diameter recessed-impeller pump. A 4-in. (10.2-cm) PVC intake line with a 3-in. (7.6-cm) reducer was directed into the cooling water flow. The velocity in the sampling inlet substantially exceeded cooling water flow velocity, preventing any back pressures around the inlet that might reduce organism collection efficiency. The pump discharged into a 4-in. (10.2-cm) diameter PVC line leading to the mouth of a 0.5-m (1.6-ft) plankton net with 505-μm mesh, suspended in a cylindrical polyethylene tank. The water level in the tank was maintained below the mouth of the net by an overflow prevention siphon and a drain rate control valve.

Sample volume and flow rate were measured with a Sparling Master-Flo inline flowmeter (factory calibrated at $\pm$ 2 percent accuracy) in the pump discharge line. The flow rate was maintained at about 0.9 m^3/min (0.5 cfs). At first, the pump was turned off for 10 minutes at the end of each 3-hour interval while the sample was



A Units 1-4 intake
B Units 5 and 6 intake
C Bottom, middepth, Unit 6 circulating water pump
D Center, Unit 6 discharge gate well
E Periphery, Unit 6 discharge gate well
F Center, Unit 5 discharge gate well
G Center, Units 1-4 discharge gate well

Figure 3-1. Sample locations for entrainment abundance, survival, and special studies at the Pittsburg Power Plant, 1978 - 1979.

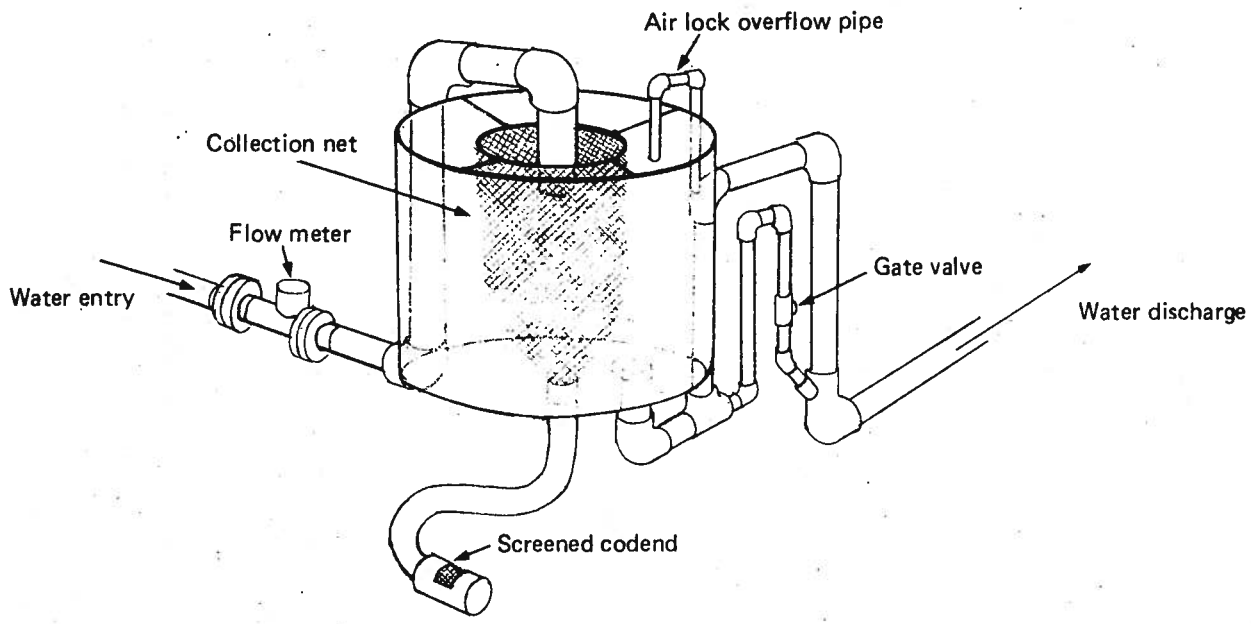


Figure 3-2. Entrainment abundance sampling equipment used at Units 1-4 and Units 5 and 6 of the Pittsburg Power Plant, March-July 1978.

collected from the net; soon, because of problems with net clogging, samples were taken and preserved at 90-minute or 1-hour intervals. Organisms were collected by draining the tank, rinsing the net from the outside with filtered water, concentrating the organisms in a screen-walled collection container, and preserving them in 10 percent buffered formalin.

The sampling equipment described above was replaced on 25 July 1978 with an automated entrainment abundance sampler. Samples were collected in the same manner as done using the manual sampler, except that the timing and sequencing of all operations was controlled by a microprocessor. Each 3-hour time block was divided into 30-minute periods; the sampling pump was operated for 27 minutes and then the plankton net was washed down and the sample concentrated into a holding container. The sample was chilled to 4 C (40 F) to minimize decomposition of the organisms collected. At the end of the 3-hour time block, 100 percent buffered formalin was injected into the container to yield a final formalin concentration of 10 percent. Then a turret moved a new container into place, and the next 3-hour collection began. Sample duration and cumulative sample volume were measured electronically and recorded.

After sample collection, samples were returned to the laboratory for processing. Ichthyoplankton and two macroinvertebrates, Neomysis mercedis and Palaemon macrodactylus, were sorted from all samples collected during the first year of the study (March 1978 - March 1979) and were identified and counted. Other macroinvertebrates collected in biweekly samples during the first year of the study were identified and counted, but were not sorted from the samples. From April through July 1979, striped bass were sorted from all samples and N. mercedis were sorted from biweekly samples only.

Because of the large numbers of macroinvertebrates collected, samples were split to 1/8 or 1/64 with a plankton splitter before macroinvertebrates were processed. Samples were split into equal 1/8 subsamples in one split or 1/64 subsamples in two consecutive splits. This is in contrast to the conventional Folsom-type splitter, where three consecutive splits are required to obtain a 1/8 sample and six consecutive splits are required to obtain a 1/64 subsample. Since any error in one split is compounded in all subsequent splits, the plankton splitter used in the study was more accurate than the Folsom type. The accuracy of the splitter was statistically tested using a chi-square goodness-of-fit test comparing the numbers of organisms found in 8 or 64

subsamples to the number expected to occur in each subsample when the number of organisms in the original sample was known. In eight replications of such a test, the resulting chi-square statistic was never significant ($\alpha = 0.05$). Thus, it was concluded that the splitter accurately partitioned samples into 1/8 or 1/64 subsamples.

Both fish larvae and macroinvertebrates were identified to the lowest taxonomic level practicable with available keys, and the life stage was noted. Fish eggs were counted, and were identified where possible. Fragments of organisms constituting less than 50 percent of total body length were not counted. Total lengths of fish larvae and juveniles were measured to the nearest millimeter. All sample processing operations were subject to a strict quality assurance program.

Densities, expressed as the number of organisms per cubic meter of cooling water, were computed for each taxon observed in each 3-hour or 24-hour sample. In the former case, the mean density for each 24-hour sampling day was computed as the simple average of the densities for the eight 3-hour samples. The 3-hour and 24-hour densities were used in studying the diel and seasonal patterns of entrainment, respectively. Length statistics (mean, standard deviation, and range) were also computed for the ichthyoplankton collected.

The total numbers of organisms in each ichthyoplankton and macroinvertebrate taxon entrained were estimated, following a stratified sampling design (Cochran 1977). The one-year period was divided into strata, two weeks in length for ichthyoplankton and four weeks for macroinvertebrates. For each stratum, the total number in each taxon entrained ($\hat{y}_{ijk}$) was estimated for each sampling period (3-hr or 24-hr) on at least two sampling days:

$$\hat{y}_{ijk} = V_{ijk} \frac{\hat{x}_{ijk}}{v_{ijk}} \qquad (3\text{-}1)$$

where

V_{ijk} = total cooling water flow (m^3) during period k

v_{ijk} = volume sampled in period k

$\hat{x}_{ijk}$ = estimated number of organisms in a given taxon sampled in period k.

When a sample was split, as in the case of macroinvertebrates, $\hat{x}_{ijk}$ was computed as

$$\hat{x}_{ijk} = \frac{z_{ijk}}{P_{ijk}} \tag{3-2}$$

where

z_{ijk} = number in a given taxon actually counted in sample k

P_{ijk} = the sample splitting proportion (1/8 or 1/64).

Having estimated y_{ijk} for each 3-hour or 24-hour sampling period in a given stratum, the mean number in a given taxon entrained per day during the stratum ($\overline{y}_i$) was estimated as

$$\overline{y}_i = \frac{1}{n_i} \sum_{j=1}^{n_i} \hat{y}_{ij} \tag{3-3}$$

where

$$\hat{y}_{ij} = \sum_{k=1}^{n_j} \hat{y}_{ijk}$$

n_i = number of sampling days in stratum i

n_j = number of periods sampled on sampling day j.

The total number of organisms of a given taxon entrained over the entire year (E) was then estimated by summing over all strata the product of the mean number entrained per day and the number of days the cooling system was in operation during each stratum:

$$\hat{E} = \sum_{i=1}^{L} N_i \overline{y}_i \tag{3-4}$$

where

N_i = number of days of cooling system operation in stratum i

L = number of strata in the one-year study period.

The variance of $\hat{E}$ includes three nested components associated with the three sampling stages:

1. variance related to sampling selected days during each stratum,
2. variance related to sampling a portion of the cooling water flow on a given day, and
3. variance related to processing a portion of a day's sample (applicable to macroinvertebrates only).

Working within the framework presented by Cochran (1977, pp. 285-289) and Yamane (1967, p. 300), the variance of $\hat{E}$ was computed as

$$\mathrm{Var}(\hat{E}) = \sum_{i=1}^{L} N_i \left[(N_i - n_i) \frac{s_i^2}{n_i} + \sum_{j=1}^{n_i} \mathrm{Var}(\hat{y}_{ij}) \right] \tag{3-5}$$

where

$$s_i^2 = \frac{1}{(n_i - 1)} \sum_{j=1}^{n_i} (\hat{y}_{ij} - \bar{y}_i)^2 \tag{3-6}$$

$$\mathrm{Var}(\hat{y}_{ij}) = \sum_{k=1}^{n_j} \frac{V_{ijk}}{n_i} \left[(V_{ijk} - v_{ijk}) \frac{\hat{x}_{ijk}}{v_{ijk}^2} + \frac{\mathrm{Var}(\hat{x}_{ijk})}{v_{ijk}} \right] \tag{3-7}$$

$$\mathrm{Var}(\hat{x}_{ijk}) = z_{ijk} \frac{1 - P_{ijk}}{P_{ijk}^2} . \tag{3-8}$$

The first component, s_i^2/n_i, is an unbiased estimate of day-to-day variability within strata (Cochran 1977). The second component, $\hat{x}_{ijk}/v_{ijk}^2$, an unbiased estimate of the volume-to-volume variability within a given 3-hour period, results from the assumption that the number of observed organisms in a given sample follows the Poisson distribution. The third component, $\mathrm{Var}(\hat{x}_{ijk})/v_{ijk})$, an unbiased estimate of the variability caused by splitting the volume sampled, results from the assumption that the number of organisms observed in a split subsample follows the binomial distribution.

The 95 percent confidence interval for the total number of a given taxon entrained over the entire year is $\hat{E} \pm 1.96$ S.E.($\hat{E}$), where S.E.($\hat{E}$) $= \sqrt{Var(E)}$.

The estimated total number in each taxon entrained during the study period, $\hat{E}$, and its standard error, S.E.($\hat{E}$), were computed for each cooling water system independently and for the combination of all three by summing the estimates for the individual cooling water systems.

The degree of recirculation of water, and presumably entrained organisms, between the discharge and intakes is substantial at the Pittsburg Power Plant (Section 2.1). No attempt was made in these studies to quantify the numbers of entrained organisms recirculated, although the effect of recirculation on entrainment survival was examined experimentally (Section 3.2). Although not quantified, recirculation of organisms would result in a potential overestimate of the number of organisms of each taxon entrained (E), and therefore entrainment estimates presented in this chapter are considered to be conservative.

Total numbers entrained, (E), were estimated both for actual cooling water flows during the study period and for the cooling water flows characteristic of continuous normal operation at Units 1-7.

The methodology just described was expanded for selected fish taxa to make it possible to estimate numbers of individuals entrained as a function of total body length. Equations 3-1 through 3-4 were applied separately to each length class ($\leq$5 mm, 6 mm, 7 mm, ..., 19 mm, $\geq$ 20 mm). The number of individuals in a 3-hour sample of a given length class (x_{ijk}) was computed by modifying Equation 3-2 as follows:

$$\hat{x}_{ijk} = \frac{z_{ijk}\,\hat{f}_{ijk}}{p_{ijk}} \qquad (3\text{-}9)$$

where

$\hat{f}_{ijk}$ = the observed proportion of individuals measured in the sample that were of the given length class.

Next, the total number in the given length class entrained during the study year (E) was estimated as before, using Equations 3-1, 3-3, and 3-4 unchanged.

In theory, the sum of the estimated numbers of each length class entrained should equal the total number entrained (E) estimated using Equation 3-4. In practice, however, the sum of the individual length classes was often somewhat less than the total E. This is because in certain 3-hour samples, one or more individuals of a given taxon were present but none were measured, usually because mutilation made the assignment of a length questionable. Such individuals were not included in the sum of the individual length classes, but were included in the total estimate of (E).

The methodology for computing $Var(\hat{E})$ was also modified. Equation 3-8 became

$$Var(\hat{x}_{ijk}) = \frac{z_{ijk} f_{ijk}}{P_{ijk}^{2}} \left[(1 - P_{ijk}) + \frac{(z_{ijk} - m_{ijk})(1 - \hat{f}_{ijk})}{P_{ijk}(m_{ijk} - 1)} \right] \qquad (3\text{-}10)$$

where

m_{ijk} = the number of measured individuals in a 3-hour sample.

Equations 3-5 and 3-6 were used unchanged.

3.1.2 Entrainment Abundance Support Studies

In this section three studies are described that support the entrainment abundance experimental design, as specified in the study plan (PGandE 1977). The first is an evaluation of the collection efficiency of the sampling gear. The second is a comparison of the densities of entrained organisms at Units 1-4, Unit 5, and Unit 6. The third is a mass balance study comparing the densities of entrained organisms between the intake and discharge of Unit 6.

3.1.2.1 Entrainment Abundance Gear Evaluation

The collection efficiency of the entrainment abundance sampling gear was documented in an experiment in which known numbers of stained organisms were directly released into the suction line of the sampling pump, and the numbers subsequently recovered in a 335- or 505- μm mesh net were recorded. After release, the pump was operated at 1,900 rpm, delivering 0.9 m^3/min (0.5 cfs) for test periods of 30, 60, 90, and 180 minutes. At the end of the test periods, the pump was

turned off, the tank drained, the net rinsed, and the sample collected. The number of organisms recovered was then determined.

Tests were conducted at the Contra Costa, Potrero, and Moss Landing power plants using brine shrimp and fish larvae. Recovery of brine shrimp was high (92-96 percent) for 30- and 60-minute tests with both 335- and 505-μm nets, but was low (43-85 percent) for 90- and 180-minute tests with the 505-μm net (Table 3-1). In the longer tests, a large proportion of the brine shrimp recovered were mutilated. Brine shrimp are very delicate, soft-bodied invertebrates, and are more susceptible to sampling damage than most of the macroinvertebrates entrained at the Pittsburg Power Plant. Additional tests conducted with larval fish at both mesh sizes resulted in 85-100 percent recovery (Table 3-1). It was concluded that the sampling gear collected larval fish and macroinvertebrates effectively, except for the most fragile forms, which may be mutilated by turbulence and abrasion in the collection net during long samples. In abundance sampling the numbers of mutilated organisms were minimized by the collection period to 60-90 minutes during sampling and to 30 minutes with the automated abundance sampler.

3.1.2.2 Inter-Discharge Density Comparison Study

The densities of fish larvae and juveniles entrained at Units 1-4, Unit 5, and Unit 6 were compared to determine whether or not densities observed at Unit 6 during routine entrainment abundance sampling were representative of densities at the other units. Cooling water samples were simultaneously collected from the Units 1-4, Unit 5, and Unit 6 discharge gate wells (D, F, G, Figure 3-1). Densities of larval and juvenile fish were computed for each sample, and the mean individual densities computed for the three sampling locations were compared.

Samples were collected by pumping water at 0.9 m^3/min (0.5 cfs) into plankton nets suspended in water-filled tanks, in the same manner as done during routine entrainment abundance sampling (Section 3.1.1). Samples were collected for periods of one hour. The volumes of water pumped during each sampling period were determined with inline flowmeters. At the end of each sampling period, organisms and debris retained in each net were concentrated into the codend of the net, placed in glass jars, and preserved in 10 percent buffered formalin.

TABLE 3-1 RESULTS OF COLLECTION EFFICIENCY TESTS OF ENTRAINMENT ABUNDANCE SAMPLING GEAR USED AT THE PITTSBURG POWER PLANT, 1979

Date	Test Number	Duration of Collection (min)	Mesh Size (μm)	Organisms	Number Released	Number Recovered	Condition [a]	Percentage Recovery
19 OCT	1	30	505	Live brine shrimp	100	95	Good	95
	2	60	505	Live brine shrimp	100	96	Good	96
25 OCT	3	30	505	Live brine shrimp	100	93	Good	93
	4	90	505	Live brine shrimp	100	85	Poor	85
	5	90	505	Live brine shrimp	100	64	Poor	64
	6	180	505	Live brine shrimp	100	84	Poor	84
	7	180	505	Live brine shrimp	100	43	Poor	43
1 NOV	8	180	505	Assorted larval fish	75	64	Good	85
21 NOV	9	60	335	Live brine shrimp	100	92	Good	92
	10	60	335	Assorted larval fish	100	100	Good	100
	11	60	335	Live brine shrimp	100	96	Good	96
	12	60	335	Assorted larval fish	100	98	Good	98
	13	30	335	Live brine shrimp	100	92	Good	92

(a) Good = organisms generally intact (not mutilated).
Poor = organisms generally not intact (mutilated).

Fish larvae and juveniles were sorted from all samples and counted. A density was computed for each sample by dividing the numbers counted by the sample volume. Mean densities were computed for each sampling location.

A preliminary sampling of seven trials was carried out in June 1978. Samples were collected only from the Units 1-4 and Unit 6 discharge gate wells. The mean density of larval and juvenile fish was $1.48/m^3$ at Units 1-4, but only $0.20/m^3$ at Unit 6 (Table 3-2). The large difference in densities was not expected, since the intakes for Units 1-4 and Units 5 and 6 are contiguous. The difference in densities between Units 1-4 and Unit 6 may have been due, at least in part, to the schooling of juvenile striped bass in the intake forebays.

Sampling was carried out at the gate wells of all three unit-group discharges in July 1978 and February 1979, with considerably different results (Table 3-2). In 11 trials in July, mean densities at Units 1-4, Unit 5, and Unit 6 were $0.13/m^3$, $0.16/m^3$, and $0.13/m^3$, respectively. The dominant species collected were striped bass, gobies, northern anchovy, and mosquitofish. There was no evidence of schooling by striped bass juveniles in the intake area. In 9 trials in February 1979, mean densities at Units 1-4, Unit 5, and Unit 6 were $0.36/m^3$, $0.37/m^3$ and $0.23/m^3$, respectively (Table 3-2). An analysis of variance of these complete series indicated no significant differences among intakes ($P > 0.5$).

Results of this study suggest that differences in densities of entrained fish larvae and juveniles occur among Units 1-4, 5, and 6, but that no consistent patterns occur over time. It was therefore concluded that densities measured at Unit 6 adequately represented densities at other units.

3.1.2.3 Intake-Discharge Mass Balance Study

The null hypothesis that organism densities in the intake and discharge are the same (i.e., that entrained organisms are not lost during cooling system passage) was tested in this study. Cooling water samples were simultaneously collected with manual entrainment sampling gear at Unit 6 from two intake and two discharge locations a total of 30 times. The densities of selected target organisms were determined from each sample, and the mean individual densities computed for the four sampling locations were compared statistically in a two-way analysis of variance.

TABLE 3-2 DENSITIES (no./m^3) OF LARVAL FISH COLLECTED IN THE INTER-DISCHARGE COMPARISON STUDY AT THE PITTSBURG POWER PLANT, 1978, 1979

Date	Time	Units 1-4	Unit 5	Unit 6
16 JUN 78	0920	0.77	—	0.09
	1030	0.69	—	0.07
	1140	1.35	—	0.03
	1245	2.57	—	0.27
	1350	2.49	—	0.56
	1500	1.35	—	0.24
	1610	1.12	—	0.14
Mean		1.48		0.20
10 JUL 78	1924	0.03	0.11	0.15
	2038	0.14	0.19	0.10
	2150	0.22	0.15	0.30
	2257	0.12	0.22	0.09
	2412	0.12	0.02	0.07
11 JUL 78	1817	0.07	0.07	0.02
	1931	0.04	0.13	0.04
	2042	0.14	0.28	0.11
	2153	0.11	0.41	0.41
	2304	0.19	0.11	0.15
12 JUL 78	0013	0.22	0.06	0.04
Mean		0.13	0.16	0.13
27 FEB 79	1740	0.24	0.15	0.33
	1850	0.15	0.19	0.09
	2000	0.15	0.13	0.09
28 FEB 79	1600	0.63	0.74	0.37
	1726	1.08	1.05	0.75
	1836	0.39	0.39	0.15
	1946	0.18	0.23	0.07
	2055	0.15	0.16	0.09
	2208	0.30	0.28	0.20
Mean		0.36	0.37	0.23

Note: (—) indicates no data collected.

Samples were collected at Unit 6, the site of routine entrainment abundance sampling (Section 3.1.1), by pumping water at 0.9 m^3/min (0.5 cfs) into plankton nets suspended in water-filled tanks, in the same manner as was done during routine entrainment abundance sampling (Section 3.1.1). Intake samples were collected from two points between the traveling screens and the inlet to circulating water pump 6A, 3 ft (0.9 m) and 9 ft (2.7 m) above the bottom (C, Figure 3-1). Discharge samples were collected from the center and the periphery of the Unit 6 discharge gate well (D, E, Figure 3-1). The intake sampling locations were selected on the basis of preliminary current velocity profiles of the intake structure cross-section. The two sampling locations were selected at points of similar flow velocity so that the densities observed at each location could be equally weighted in computing mean densities for the intake in the analysis of the experimental results.

Samples were collected for periods of one hour. The volumes of water pumped during each sampling period were determined with inline flowmeters. At the end of each sampling period, the organisms and debris retained in each net were concentrated into the codend of the net, washed into a glass jar, and preserved in 10 percent buffered formalin.

Fish larvae and juveniles were sorted from all samples and counted. Densities were computed by dividing the numbers counted per sample by the sample volume. Mean densities were computed for each sampling location by taking the simple average of the densities computed for the 30 trials.

The data were statistically analyzed in a two-way analysis of variance. By means of orthogonal contrasts, three null hypotheses were tested:

1. mean densities at the intake and discharge are not different;
2. mean densities at the intake middepth and bottom sampling location are not different;
3. mean densities at the discharge central and peripheral sampling location are not different.

Among the assumptions of the analysis of variance are (1) the assumption of homogeneity of variance (i.e., equal variance) among treatment groups, and (2) the assumptions of normality and independence of experimental observations. To meet the assumption of homogeneity of variance, the ichthyoplankton densities observed were transformed as follows:

$$Y = \log \frac{x + 1}{v} \qquad (3\text{-}11)$$

where

x = number collected in sample
v = volume of sample in cubic meters.

The assumption of normality was not a major concern, because the analysis of variance test comparing means is relatively insensitive to deviations from normality (Box 1953; Scheffe 1959). Because samples were independently collected, the experimental observations were assumed to be independent.

The overall mean density of larval and juvenile fish in 30 trials at the intake ($0.16/m^3$) was found not to be significantly different from the overall mean density at the discharge ($0.14/m^3$) (Table 3-3). The mean density at the intake middepth ($0.22/m^3$) was found to be significantly different ($\alpha = 0.05$) from the mean density at the intake bottom ($0.10/m^3$). The mean densities at the discharge center and periphery (both $0.14/m^3$) were not significantly different (Table 3-3).

On the basis of these results, it was concluded that:

1. there was no significant loss of organisms passing through the the cooling system, since overall intake and discharge mean densities were very similar;
2. the cooling water at the discharge was more thoroughly mixed than that at the intake: mean densities were very similar at the two discharge locations sampled, but were different at the two intake locations sampled; and
3. the discharge was the better location for sampling entrainment abundance, because samples collected there were likely to represent more accurately the actual densities of organisms entrained than were samples collected in the intake.

3.1.3 Results of the Entrainment Abundance Study

This section presents the results of the entrainment abundance study at Units 1-7 of

TABLE 3-3 DENSITIES (no./m^3) OF LARVAL FISH COLLECTED IN THE INTAKE-DISCHARGE MASS-BALANCE STUDY AT UNIT 6 OF THE PITTSBURG POWER PLANT, JULY 1978

Date	Time	Intake		Discharge	
		Middepth	Bottom	Center	Periphery
6 JUL 78	0000	0.17	0.03	0.07	0.11
	0017	0.15	0.02	0.24	0.14
	0244	0.36	0.09	0.05	0.35
	0400	0.78	0.31	0.72	0.50
	0520	0.47	0.18	0.29	0.36
	0635	0.16	0.42	0.18	0.27
	0080	0.29	0.12	0.33	0.18
	0915	0.10	0.06	0.04	0.02
	1630	0.11	0	0.02	0.06
	1740	0.17	0.07	0	0.13
	2318	0.15	0.07	0.10	0.15
7 JUL 78	1350	0.26	0.07	0.18	0.10
	1425	0.08	0.07	0.10	0.04
	1620	0.17	0.11	0.07	0.08
	1730	0.12	0.11	0.14	0
	1843	0.08	0.02	0.07	0.18
	2000	0.55	0.38	0.26	0.46
	2118	0.20	0	0.06	0.10
	2239	0.10	0.06	0.08	0.08
	2354	0.08	0.06	0.02	0.06
8 JUL 78	0245	0.26	0.09	0.07	0.17
	0638	0.06	0.09	0.15	0.19
	0757	0.54	0.17	0.45	0.22
	0913	0.27	0.17	0.11	0.17
	1040	0.17	0.02	0.08	0.02
	1155	0.19	0.02	0.04	0
	1310	0.06	0.04	0.02	0
	1425	0.27	0.12	0.08	0.05
	1543	0.22	0.07	0.15	0.07
	1655	0.08	0.04	0.07	0
Position Mean		0.22	0.10	0.14	0.14
Location Mean		0.16		0.14	

the Pittsburg Power Plant. It is divided into three parts: fish eggs (Section 3.1.3.1), fish larvae and juveniles (Section 3.1.3.2), and macroinvertebrates (Section 3.1.3.3). The numbers of ichthyoplankton and macroinvertebrates entrained were estimated on the basis of actual cooling water flows at Units 1-7 during the study period (Table 3-4). (Actual cooling water flows were calculated from pump capacities and hours of operation of each pump, taken from PGandE daily operating logs). Estimates were also computed on the assumption of continuous normal cooling water flows. The estimates based on continuous normal flows are approximately 30 percent greater than those based on actual flows at Units 1-4 and Units 5-6, and 10 percent greater at Unit 7.

3.1.3.1 Fish Eggs

An estimated 2.0 million fish eggs were entrained during the first 12 months of the study (March 1978 - March 1979); 55 percent were striped bass (Table 3-4). Other taxa were smelts (Osmeridae), northern anchovy, and minnows (Cyprinidae). Fish eggs were entrained from February to July, with greatest densities (up to $0.11/m^3$) in April 1978 (Figure 3-3). These were largely striped bass eggs, and were associated with the effects of high outflow in spring 1978 on striped bass spawning. Smaller peaks correspond to the spawning seasons of anchovy, American shad, and minows (June, July) and of smelts and sculpins (January, February). Fish eggs were entrained predominantly at night (Table 3-5).

3.1.3.2 Fish Larvae and Juveniles

An estimated 468 million fish larvae and juveniles were entrained at Units 1-7 during the first 12 months of the study (Table 3-4). The larva is the earliest life stage of a fish, very small, delicate, undeveloped, and essentially planktonic. As the development of fins is completed, it becomes a juvenile, an immature, small version of the adult. Many more larvae were collected than juveniles. Fish larvae and juveniles were entrained during the winter, spring, and summer, with greatest densities (up to $5.1/m^3$) in May (Figure 3-4).

Striped bass (*Morone saxatilis*), smelts (Osmeridae), prickly sculpin (*Cottus asper*), Pacific herring (*Clupea harengus pallasi*), gobies (Gobiidae), northern anchovy (*Engraulis mordax*) and threadfin shad (*Dorosoma petenense*) constituted 85 percent of the total fish larvae and juveniles entrained. Another 17 taxa constituted

TABLE 3-4 ESTIMATED[a] NUMBERS (millions) OF SELECTED ICHTHYOPLANKTON AND MACROINVERTEBRATES ENTRAINED AT THE PITTSBURG POWER PLANT UNDER ACTUAL PUMP OPERATION, MARCH 1978 - MARCH 1979

Taxon		Units 1-4		Units 5 and 6		Unit 7		Units 1-7	
Common Name	Scientific Name	Number Entrained	Standard Error	Number Entrained	Standard Error	Number Entrained	Standard Error	Total No. Entrained	Percentage Composition
Fish Larvae and Juveniles									
Striped bass	Morone saxatilis	177.6	21.66	96.70	13.22	9.01	1.27	283.29	60.5
Prickly sculpin[b]	Cottus asper	15.21	—	13.23	—	1.17	—	29.61	6.3
Threadfin shad	Dorosoma petenense	0.35	0.08	0.30	0.07	0.03	0.01	0.67	0.2
Smelts[c]	Osmeridae	26.75	—	22.42	—	1.14	—	50.29	10.7
Yellowfin goby	Acanthogobius flavimanus	3.55	0.35	2.30	0.23	0.21	0.02	6.07	1.3
Pacific herring	Clupea harengus pallasii	10.34	3.71	11.24	4.18	0.54	0.21	22.12	4.7
Northern anchovy	Engraulis mordax	2.74	0.95	2.22	0.77	0.11	0.04	5.08	1.1
Unidentified gobies	Gobiidae	2.37	0.58	1.60	0.39	0.11	0.02	4.07	0.9
Others		39.88	—	25.22	—	1.91	—	67.02	14.3
Total		278.79		175.23		14.23		468.22	
Fish Eggs									
Striped bass	Morone saxatilis	0.46	0.17	0.57	0.25	0.05	0.02	1.08	54.8
Smelts	Osmeridae	0.32	0.28	0.27	0.23	0.01	0.01	0.60	30.5
Minnows	Cyprinidae	0.02	0.01	0.01	0.01	*	*	0.03	1.5
Northern anchovy	Engraulis mordax	0.07	0.03	0.05	0.02	*	*	0.12	6.1
Unidentified	Osteichthyes	0.07	0.03	0.07	0.04	*	*	0.14	7.1
Total		0.94		0.97		0.06		1.97	
Macroinvertebrates									
Opossum shrimp	Neomysis mercedis	4,926.0	265.8	4,134.2	252.5	254.6	14.0	9,314.8	
Oriental shrimp	Palaemon macrodactylus	363.2	46.5	282.8	45.6	17.0	1.8	663.0	
Amphipod[d]	Gammaridae	467.0	143.5	374.0	122.9	19.4	5.4	860.4	
Amphipod	Corophium stimpsoni	425.9	140.2	342.7	121.2	17.6	5.3	786.2	
Pebble crab	Rhithropanopeus harrisii	242.3	55.1	217.2	55.8	11.2	2.8	470.7	
Total		6,424.4		5,350.9		319.8		12,095.1	

(a) Computed using Equation 3-5 and 3-6.
(b) All Cottus were determined to be C. asper after the data base was established.
(c) Smelts include Delta smelt, longfin smelt, and Osermidae
(d) Gammaridae includes all amphipods (see Appendix E).

Note: (*) indicates less than 10,000 individuals; (--) indicates that combined standard errors have not been calculated.

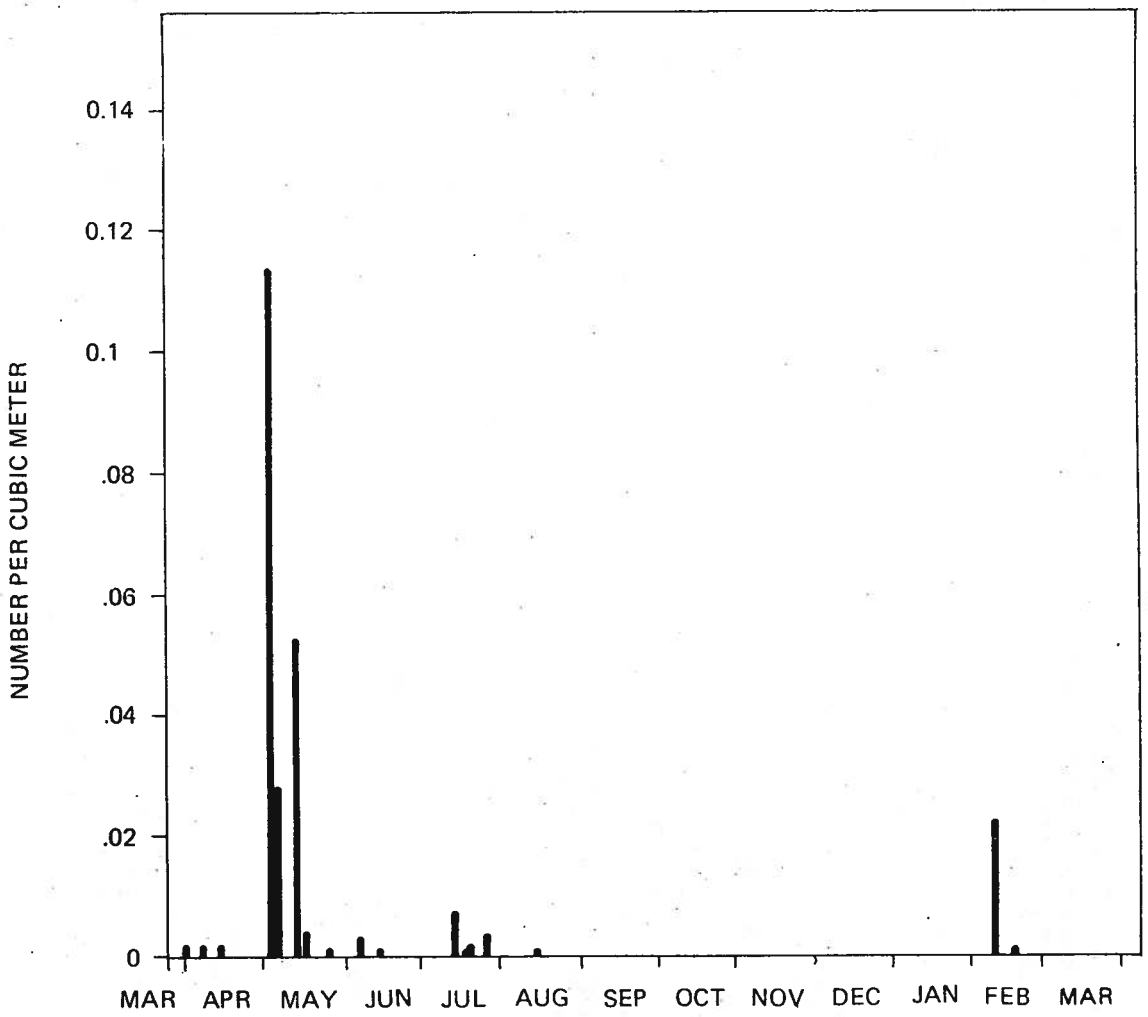


Figure 3-3. Mean densities of fish eggs entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.

TABLE 3-5 DIEL DISTRIBUTION (%) OF ENTRAINMENT OF SELECTED ICHTHYOPLANKTON AND MACROINVERTEBRATES AT UNITS 1-5 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Taxon		Sampling Period Start Time							
Common Name	Scientific Name	0600	0900	1200	1500	1800	2100	2400	0300
Fish Larvae and Juveniles									
Striped bass	*Morone saxatilis*	16	8	6	4	8	8	25	27
Sculpins	Cottidae	13	7	10	8	7	16	23	15
Smelts	Osmeridae	16	14	11	15	7	10	12	14
Gobies	Gobiidae	13	4	4	3	6	15	23	32
Pacific herring	*Clupea harengus pallasi*	13	17	18	9	19	9	15	0
Northern anchovy	*Engraulis mordax*	12	2	7	12	11	16	27	14
Threadfin shad	*Dorosoma petenense*	15	17	21	22	15	5	4	2
Fish Eggs									
Striped bass	*Morone saxatilis*	6	17	17	7	6	22	19	7
Northern anchovy	*Engraulis mordax*	0	12	0	0	14	0	11	64
Unidentified	Osteichthyes	0	0	10	20	11	58	0	0
Macroinvertebrates									
Opossum shrimp	*Neomysis mercedis*	13	6	4	6	9	13	24	24
Oriental shrimp	*Palaemon macrodactylus*	10	3	5	7	7	18	21	29
Amphipod	*Corophium stimpsoni*	2	1	1	1	1	10	11	74
Pebble crab	*Rhithropanopeus harrisii*	17	11	4	6	36	15	4	7

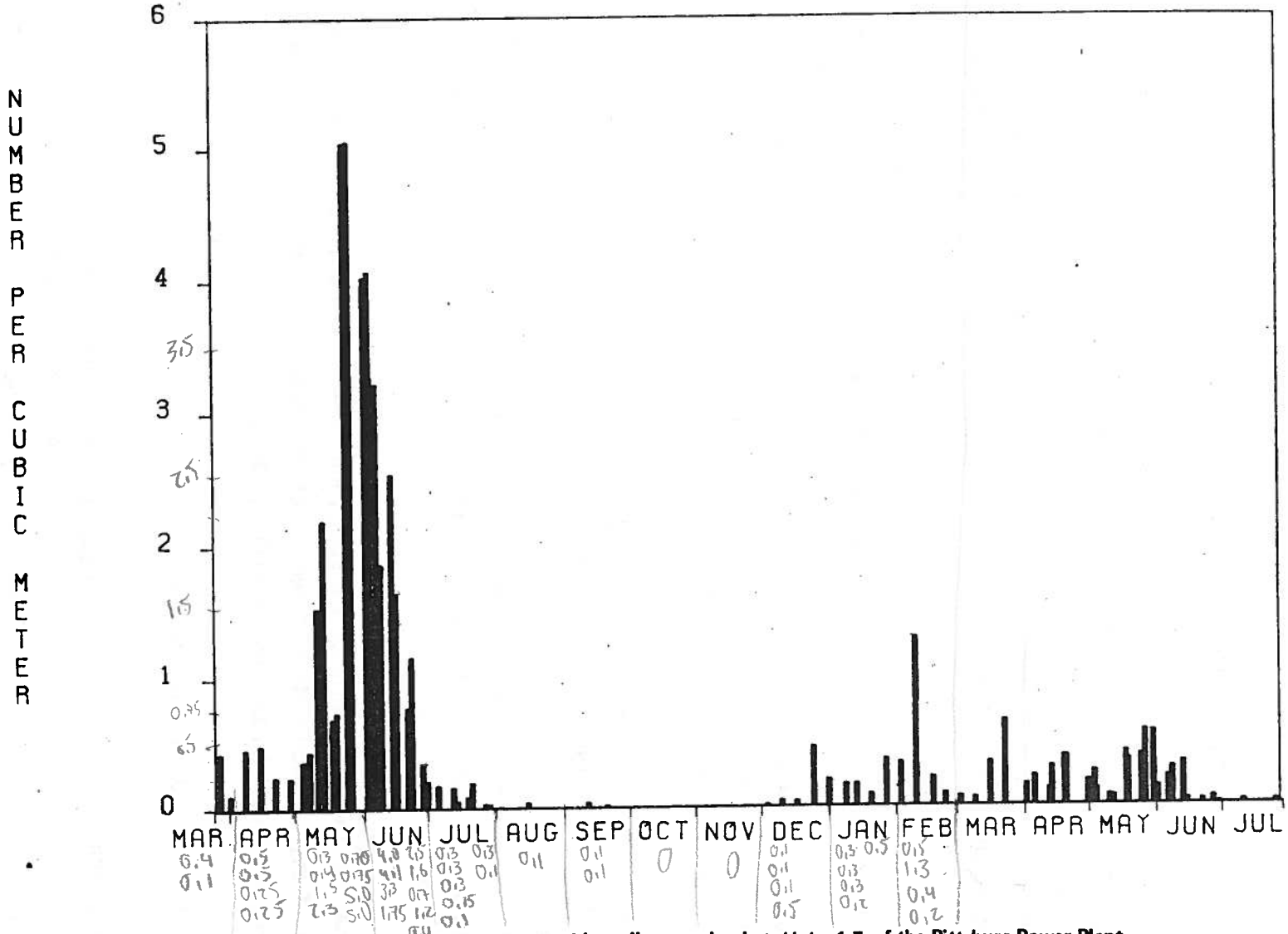


Figure 3-4. Mean densities of fish larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—July 1979.

approximately one percent of the total number (see Appendix E for a listing of taxa). The remainder were not identified below Class Osteichthyes; in many cases mutilation made identification questionable.

3.1.3.2.1 Striped Bass (Morone saxatilis)

Striped bass was the most abundantly entrained fish. An estimated 283 million were entrained at Units 1-7 during the first 12 months of the study, encompassing the 1978 spring spawning season, and 35 million during the last 4 months of the study, the 1979 spawning season. The smaller number entrained in 1979 resulted from a smaller larval striped bass population (Chapter 5) rather than a reduction in cooling water flow: cooling water flow was actually greater in spring 1979 than in spring 1978 (Section 2.1.3). The seasonal patterns of striped bass entrainment were similar in the two years (Figure 3-5). Striped bass were entrained from late March to September, with greatest densities (up to $4.7/m^3$) in May (Figure 3-5). Yolk-sac larvae (approximately 4-6 mm total length) were predominantly entrained from April to late June, larvae (7-16 mm) from April through late July, and juveniles ($\geq$17 mm) from May to mid-September (Table 3-6).

Larvae and juveniles ranged in size (total length) from 2 mm to 107 mm; mean length was 11.7 mm (Table 3-7). Mean length increased as the entrainment season progressed during both 1978 and 1979 (Figure 3-6). The numbers of striped bass entrained are presented as a function of length (1-mm increments) in Table 3-8. Additional length data for striped bass and other major taxa are presented in Appendix E.

During the first 12 months of the study, entrainment was monitored as a function of time of day. Approximately 67 percent of the striped bass larvae and juveniles were entrained at night (1800-0600 hours) (Table 3-5).

3.1.3.2.2 Smelts (Osmeridae)

Smelt, including the Delta smelt, Hypomesus transpacificus, and the longfin smelt, Spirinchus thaleichthys, were the second most abundantly entrained fish, with an estimated 50 million entrained at Units 1-7 during the first 12 months of the study (Table 3-4). Smelt were entrained from December through June, with a peak in January-February (Figure 3-7). The entrainment peak coincided with the spawning

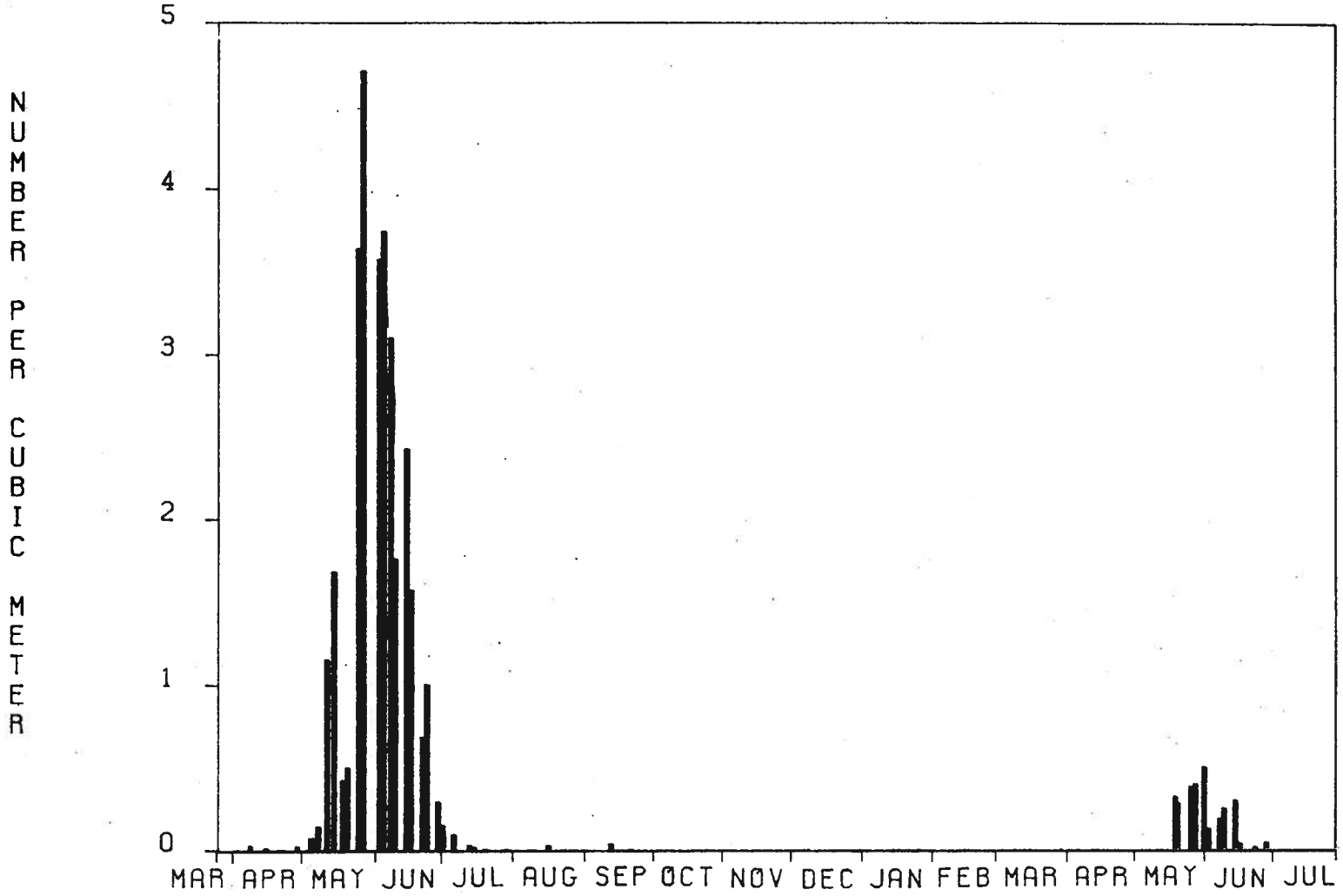


Figure 3-5. Mean densities of striped bass larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—July 1979.

TABLE 3-6 ESTIMATED[a] NUMBERS (millions) OF STRIPED BASS ENTRAINED, BY LIFE STAGE AND 14-DAY PERIOD, AT UNITS 1-7 OF THE PITTSBURG POWER PLANT UNDER ACTUAL PUMP OPERATION, 1978-1979

Period	Life Stage		
	Yolk-sac Larva	Larva	Juvenile
1978			
1-14 APR	0.01	0.48	0
14-28 APR	0.13	0.23	0
28 APR - 12 MAY	3.42	17.04	0.03
12-26 MAY	1.73	67.85	0.03
26 MAY - 9 JUN	0.76	117.88	7.06
9-23 JUN	0.07	38.64	17.68
23 JUN - 7 JUL	0	2.97	4.39
7-21 JUL	0	0.07	0.50
21 JUL - 4 AUG	0	0	0.17
4-25 AUG	0	0	0.28
8-22 SEP	1.07	0.68	0.09
29 DEC - 12 JAN 79	0	0	0.02
1979			
28 APR - 12 MAY	0.08	0.08	0
12-26 MAY	3.73	3.13	0
26 MAY - 9 JUN	4.06	12.43	0.75
9-23 JUN	0	5.89	4.08
23 JUN - 7 JUL	0.07	0.60	0.39
7 JUL - 4 AUG	0	0	0.14

(a) Computed using Equation 3-4.

TABLE 3-7 SUMMARY OF LENGTHS OF SELECTED ICHTHYOPLANKTON AND MACROINVERTEBRATES COLLECTED IN ENTRAINMENT SAMPLES AT UNITS 1-7 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Taxon		Number	Mean Length	Standard	Range (mm)	
Common Name	Scientific Name	Measured	(mm)	Deviation	Minimum	Maximum
Fish Larvae and Juveniles						
Striped bass	*Morone saxatilis*	4,934	11.7	7.5	2	107
Prickly sculpin[a]	*Cottus asper*	1,843	5.0	1.2	3	27
Staghorn sculpin	*Leptocottus armatus*	1	38.0			
Unidentified smelts	Osmeridae	564	6.7	3.2	3	32
Delta smelt	*Hypomesus transpacificus*	42	22.5	5.1	15	34
Longfin smelt	*Spirinchus thaleichthys*	6	40.0	17.3	24	68
Unidentified gobies	Gobiidae	353	14.1	5.9	3	29
Yellowfin goby	*Acanthogobius flavimanus*	688	22.2	6.6	14	67
Arrow goby	*Clevelandia ios*	2	32.5	7.8	27	38
Pacific herring	*Clupea harengus pallasi*	107	8.6	1.4	4	12
Northern anchovy	*Engraulis mordax*	270	8.0	4.1	3	37
Threadfin shad	*Dorosoma petenense*	90	6.0	12.6	3	99
American shad	*Alosa sapidissima*	19	13.9	14.1	4	66
King salmon	*Oncorhynchus tshawytscha*	2	40.0	1.4	39	41
Macroinvertebrates						
Opossum shrimp	*Neomysis mercedis*	234	4.2	1.8	2	11

(a) All *Cottus* spp. were found to be *C. asper*.

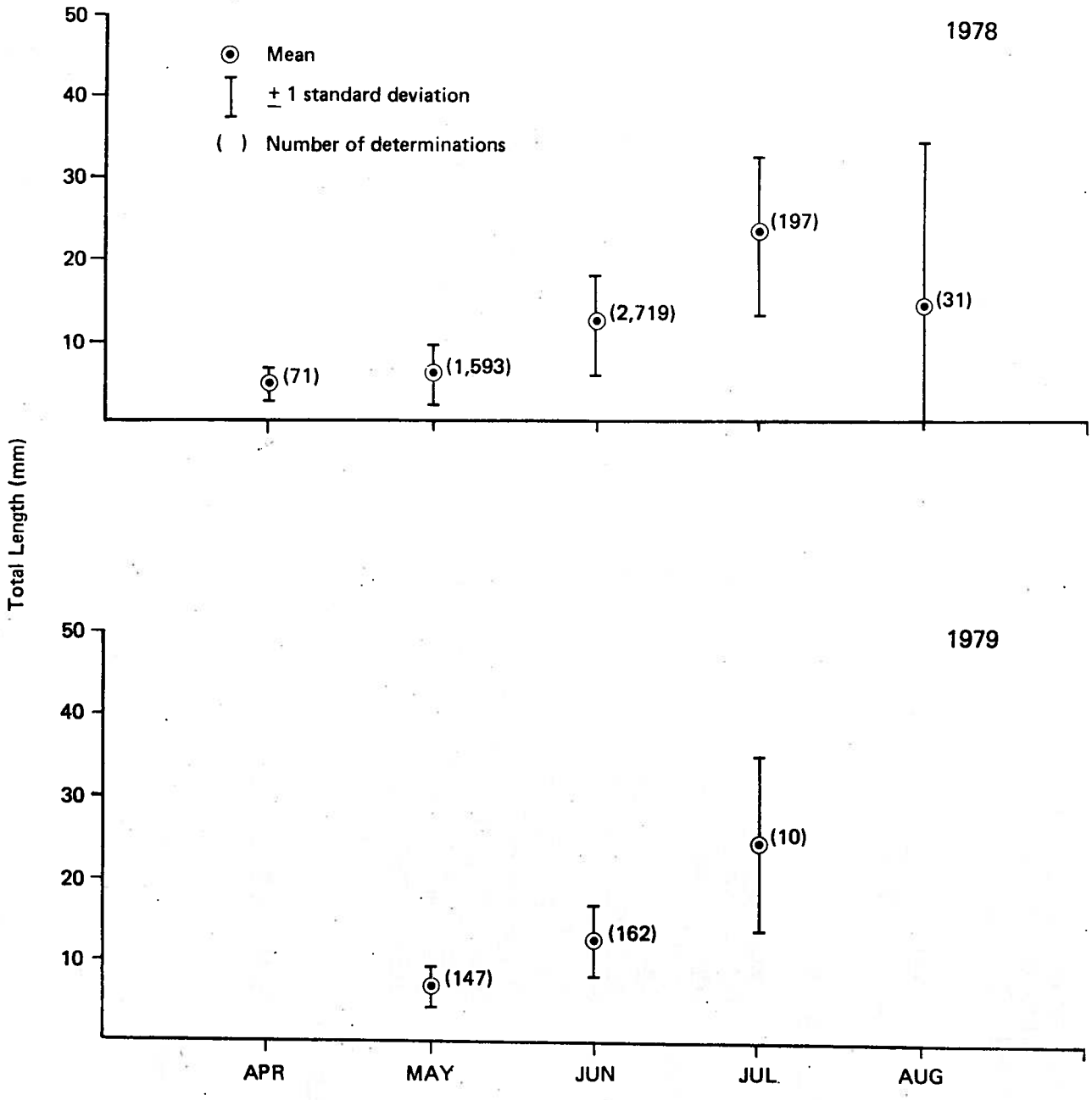


Figure 3-6. Mean lengths and standard deviations of measured striped bass collected in entrainment samples at Units 1-7 of the Pittsburg Power Plant, 1978, 1979.

TABLE 3-8 ESTIMATED[a] NUMBERS (millions) OF SELECTED ICHTHYOPLANKTON OF VARIOUS LENGTHS ENTRAINED AT UNITS 1-7 OF THE PITTSBURG POWER PLANT UNDER ACTUAL PUMP OPERATION, MARCH 1978 - MARCH 1979

Length Interval (mm)	Taxon						
	Striped Bass	Sculpin[b]	Smelts[c]	Gobies[d]	Pacific Herring	Northern Anchovy	Threadfin Shad
5	75.48	24.97	34.67	0.19	0.61	1.33	0.53
6	46.98	4.06	10.16	0.15	1.23	0.83	0
7	17.24	0.18	2.30	0.13	1.59	0.60	*
8	7.07	0.02	0.43	0.23	4.80	0.45	0.09
9	10.15	0.13	0.51	0.26	7.64	0.26	0.01
10	16.40	0.12	0.83	0.16	5.62	0.49	0
11	17.04	0.09	0.15	0.22	0.51	0.14	0
12	18.86	0.02	0.10	0.29	0.11	0.16	0
13	14.38	0.02	0.05	0.12	0	0.02	0
14	11.72	0	0.09	0.24	0	0.06	0
15	11.19	0	0.03	0.41	0	0.09	0
16	10.26	0	0.04	0.46	0	0.34	0
17	5.84	0	0.03	0.75	0	0.06	0
18	4.57	0	0.04	0.67	0	0.05	0
19	2.91	0	0.02	0.93	0	0.03	0
20	12.48	0.02	0.47	4.85	0	0.16	0.04

(a) Numbers of all length groups combined often do not exactly total numbers in Table 3-5, because some samples had unmeasured fish of a given taxon but no measured fish of that taxon. Such fish were not included in the length breakdown in this table.
(b) All *Cottus* spp. were found to be *C. asper* after the data base was established.
(c) Smelts includes Delta smelt, longfin smelt, and Osmeridae.
(d) Gobies includes yellowfin goby and Gobiidae.

Note: (*) indicated less than 10,000 individuals.

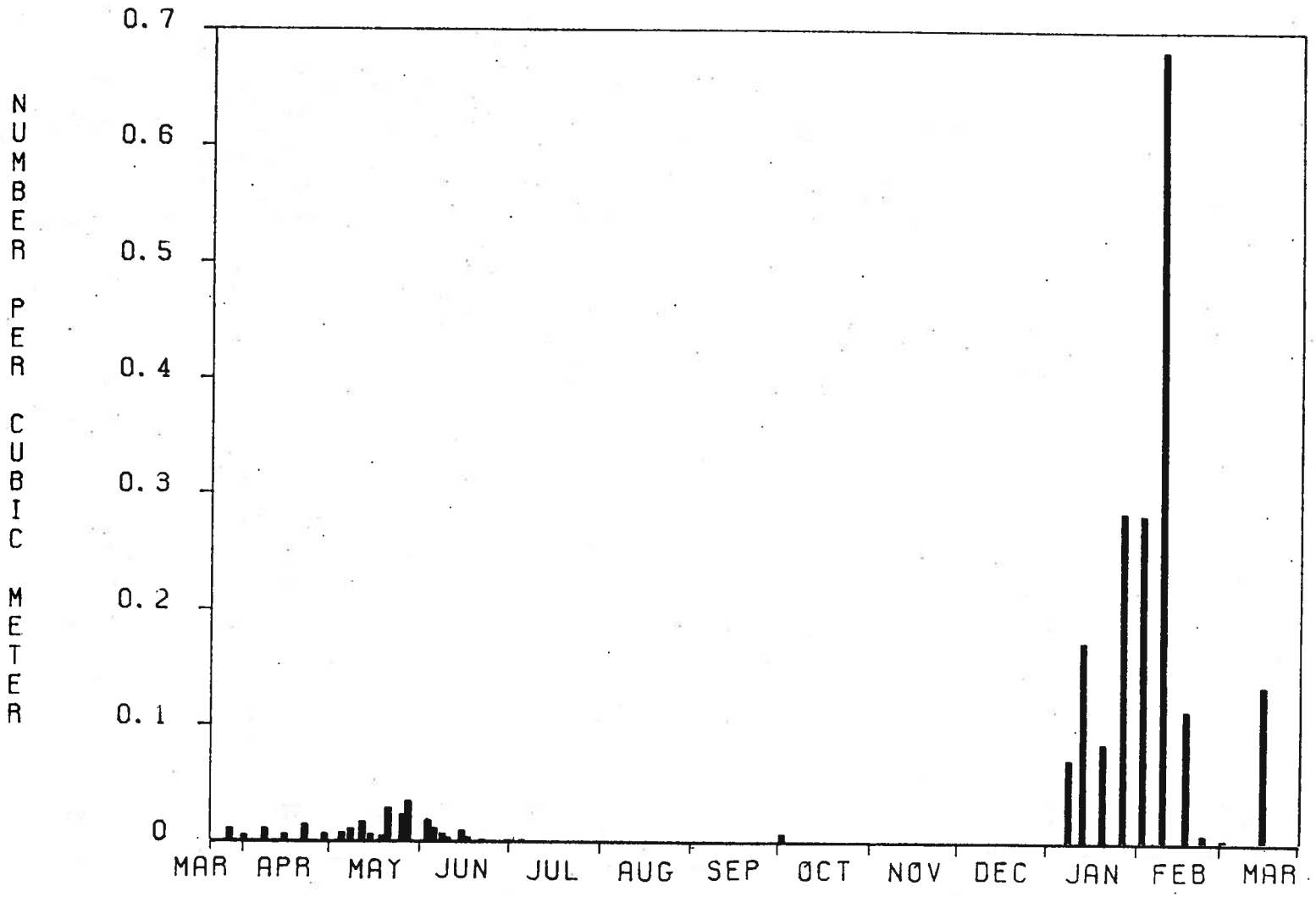


Figure 3-7. **Mean densities of smelt larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.**

season of both Delta smelt (Fry 1973; Moyle 1976) and longfin smelt (Radtke 1966; Moyle 1976).

Smelt ranged in size (total length) from 3 mm to 68 mm; mean length was 8.1 mm (Table 3-7). The numbers of smelt entrained as a function of length (in 1-mm increments) are presented in Table 3-8. Smelt larvae are pelagic and inhabit the upper zone of the water column (Erkkila et al. 1950; Radtke 1966), accounting for the nearly equal entrainment during the day and night (Table 3-5).

3.1.3.2.3 Prickly Sculpin (Cottus asper)

An estimated 30 million prickly sculpin larvae and juveniles were entrained at Units 1-7 during the first 12 months of the study (Table 3-4). Sculpin were entrained from February through May, with greatest densities (up to $0.41/m^3$) in March and April (Figure 3-8). The entrainment peak coincided with the spawning season for the prickly sculpin (Kresja 1967).

Prickly sculpin ranged in size (total length) from 3 mm to 27 mm; mean length was 5 mm (Table 3-7). The numbers of prickly sculpin entrained as a function of length (in 1-mm increments) are presented in Table 3-8. Approximately 61 percent of the prickly sculpin were entrained at night (Table 3-5).

3.1.3.2.4 Pacific Herring (Clupea harengus pallasi)

An estimated 22 million Pacific herring were entrained from November through February during the first 12 months of the study (Table 3-4). The greatest densities (up to $0.44/m^3$) were observed in December (Figure 3-9). The entrainment peak coincided with the spawning season of the Pacific herring in San Francisco Bay (Thomas and Dahlstrom 1979).

Pacific herring ranged in size (total length) from 4 mm to 12 mm; mean length was 8.6 mm (Table 3-7). The numbers of Pacific herring entrained as a function of length (in 1-mm increments) are presented in Table 3-8. Pacific herring were entrained equally during the day and night (Table 3-5).

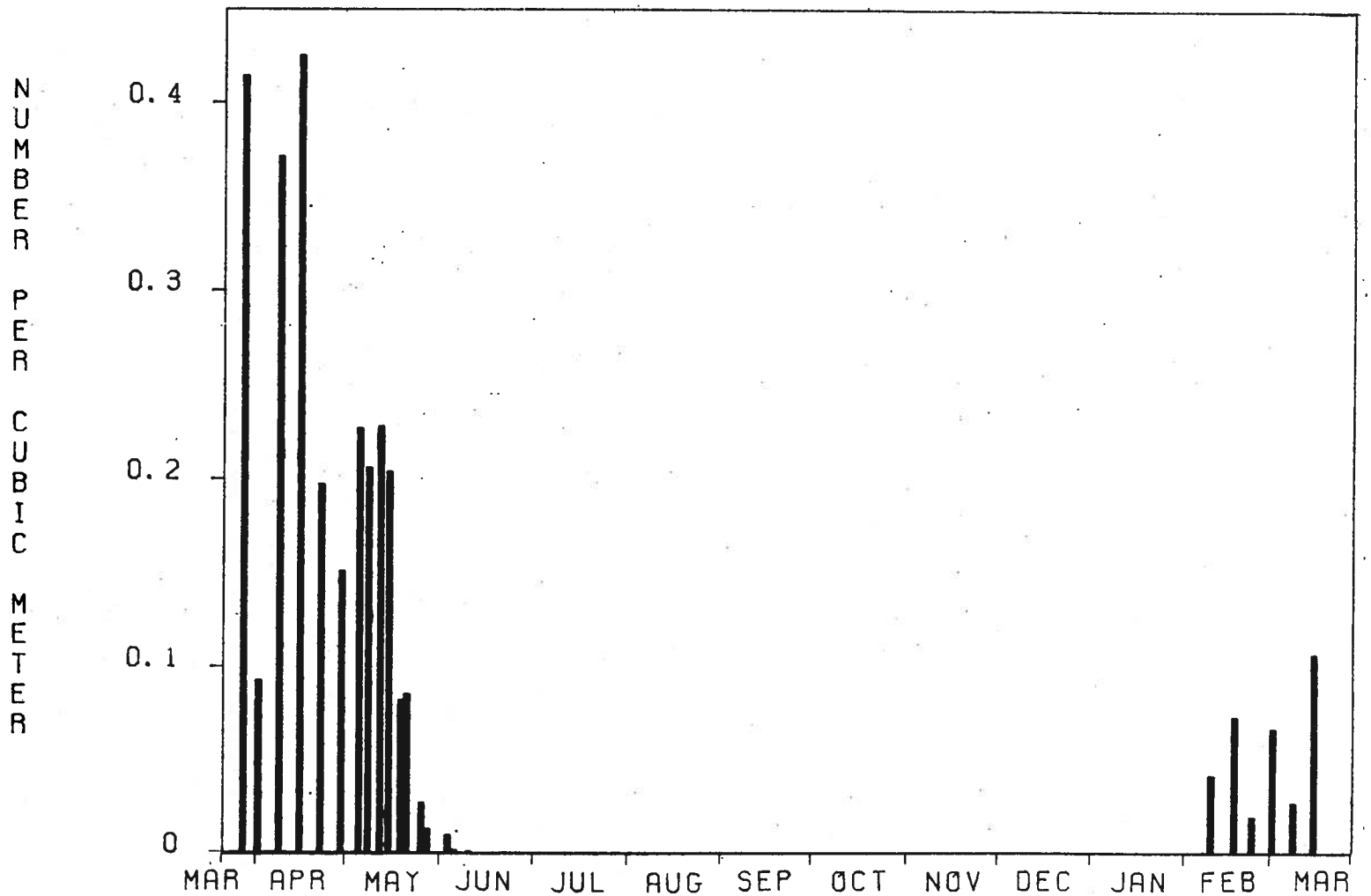


Figure 3-8. **Mean densities of sculpin larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.**

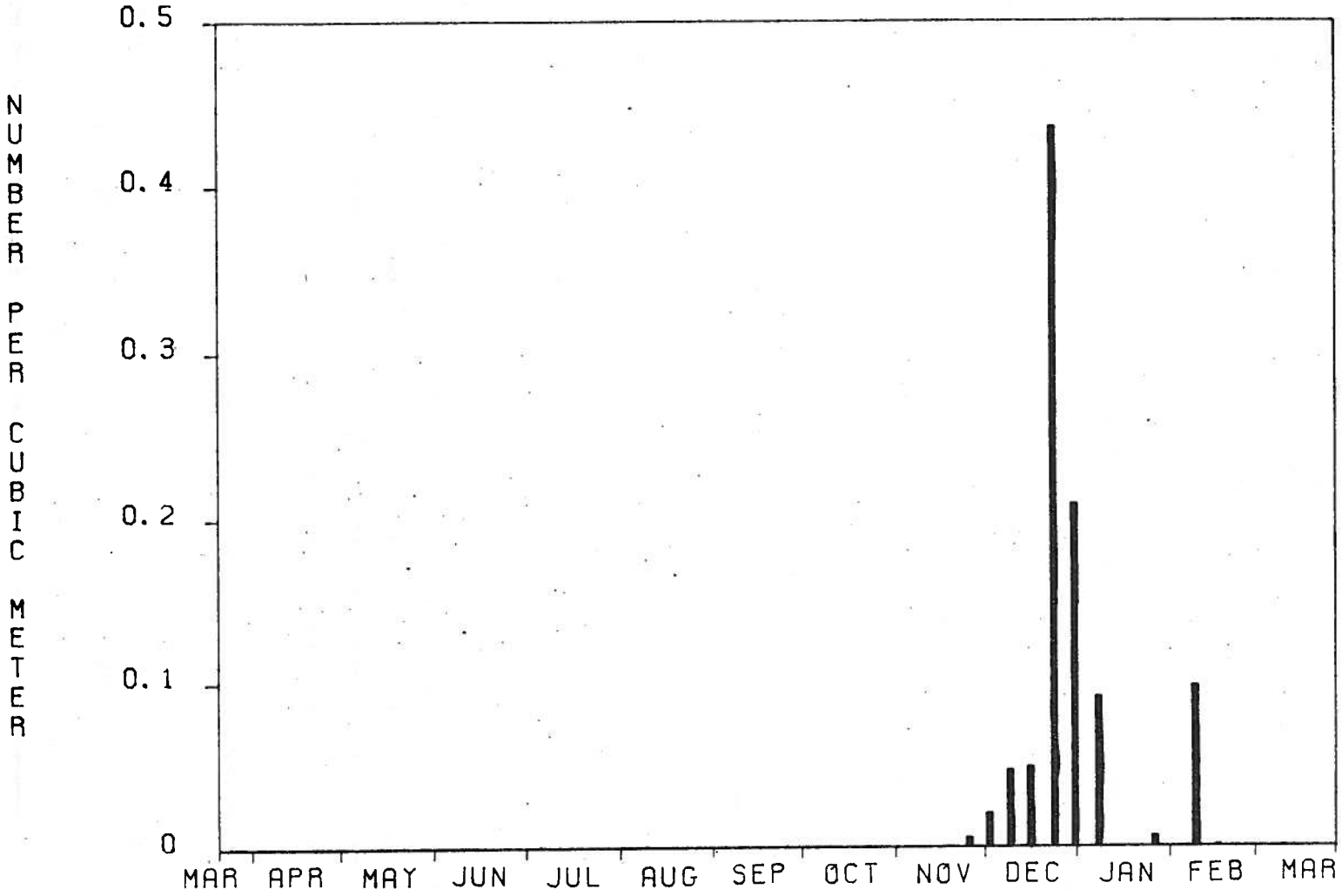


Figure 3-9. **Mean densities of Pacific herring larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.**

3.1.3.2.5 Gobies (Gobiidae)

An estimated 11 million gobies were entrained during the first 12 months of the study (Table 3-4). The major gobies entrained were the yellowfin goby, Acanthogobius flavimanus, the early juvenile stages of the bay goby, Lepidogobius lepidus, and the arrow goby, Clevelandia ios. Gobies were entrained from March through July, with greatest densities (up to $0.09/m^3$) in April and (up to $0.1/m^3$) in June (Figure 3-10). The period of goby entrainment coincides with or follows the spawning seasons of the yellowfin goby (January through May), the bay goby (November through June), and the arrow goby (December to August) (Brittan et al. 1970; Prasad 1959; Hart 1973).

Gobies ranged in size (total length) from 3 mm to 67 mm; mean length was 20 mm (Table 3-7). Total numbers of gobies entrained as a function of length (in 1-mm increments) are presented in Table 3-8. Approximately 76 percent of the gobies were entrained at night (Table 3-5).

3.1.3.2.6 Northern Anchovy (Engraulis mordax)

An estimated 5 million northern anchovy were entrained from June through August in the first 12 months of the study (Table 3-4), with the greatest densities (up to $0.17/m^3$) in July (Figure 3-11). Northern anchovy entrainment followed the winter-spring spawning season (Frey 1971).

Northern anchovy ranged in size (total length) from 3 mm to 37 mm; mean length was 8 mm (Table 3-7). The numbers of northern anchovy entrained as a function of length (in 1-mm increments) are presented in Table 3-8. Approximately 68 percent of the northern anchovy were entrained at night (Table 3-5).

3.1.3.2.7 Threadfin Shad (Dorosoma petenense)

An estimated 0.7 million threadfin shad were entrained during the first 12 months of the study (Table 3-4). Threadfin shad were entrained from late March to mid-December, with greatest densities (up to $0.19/m^3$) from March through May (Figure 3-12). Threadfin shad spawn once or perhaps twice from late spring to early fall (Kimsey 1958; Kilambi and Baglin 1969), accounting for the extended entrainment period.

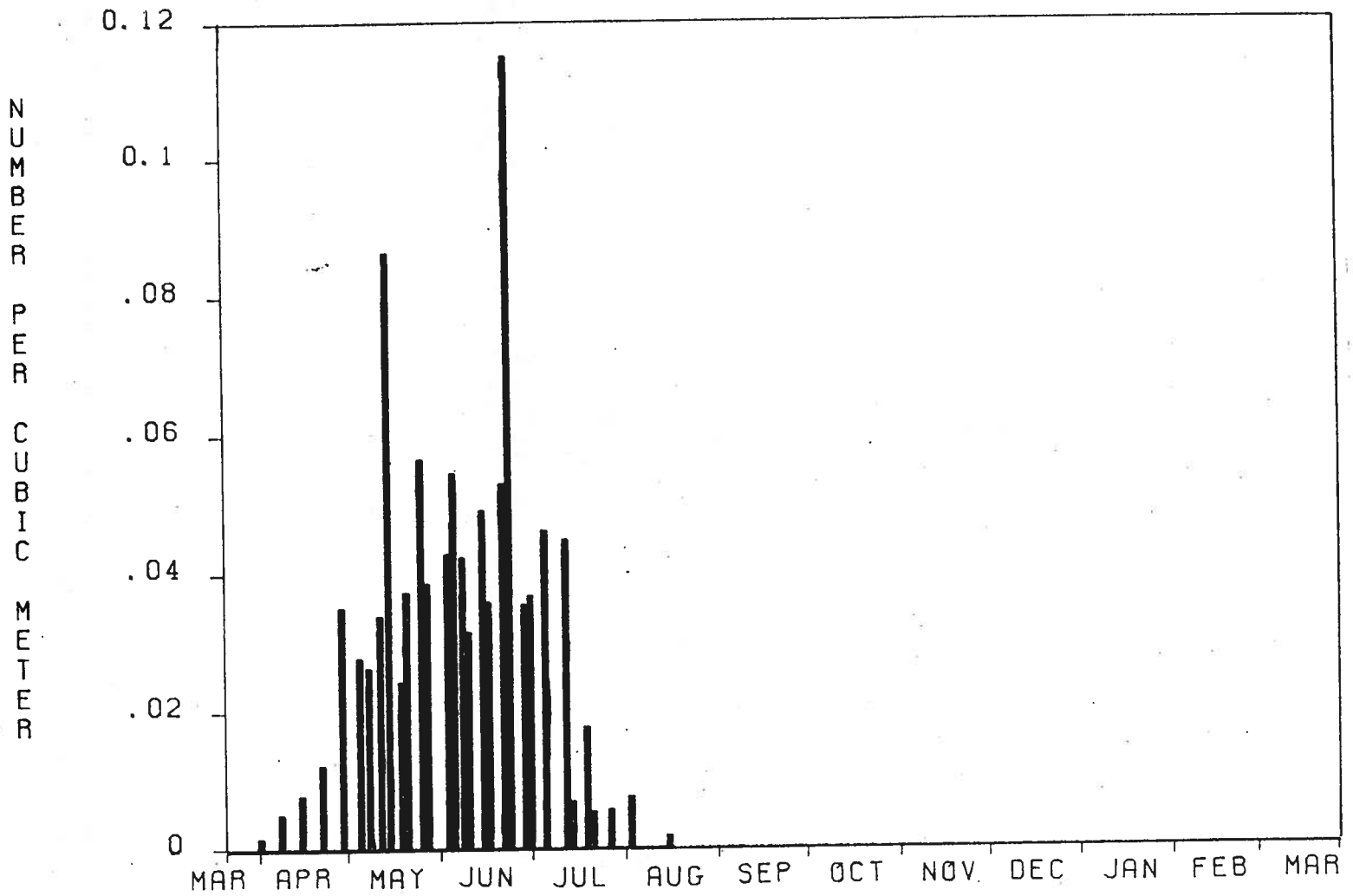


Figure 3-10. Mean densities of goby larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.

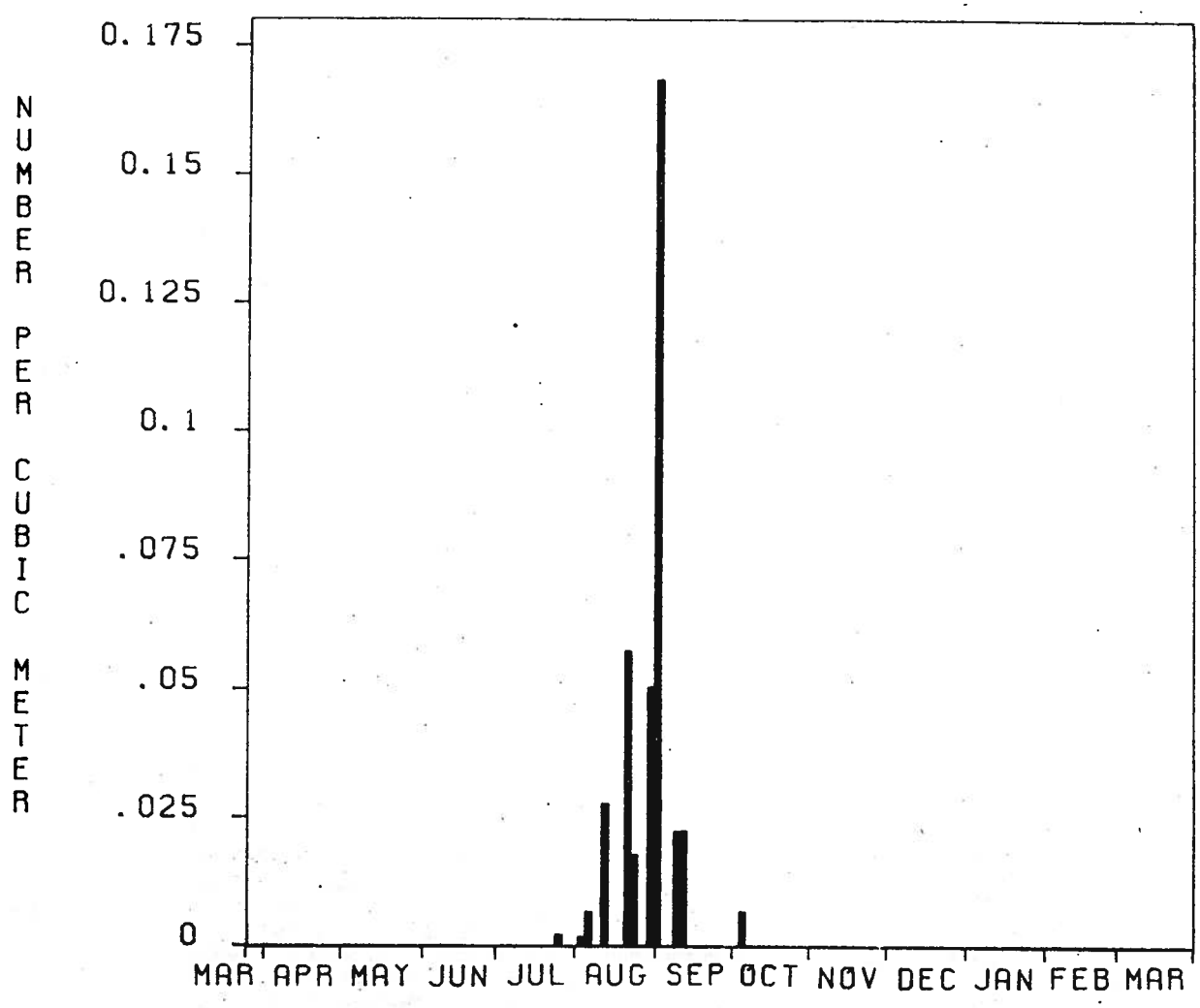


Figure 3-11. Mean densities of northern anchovy larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.

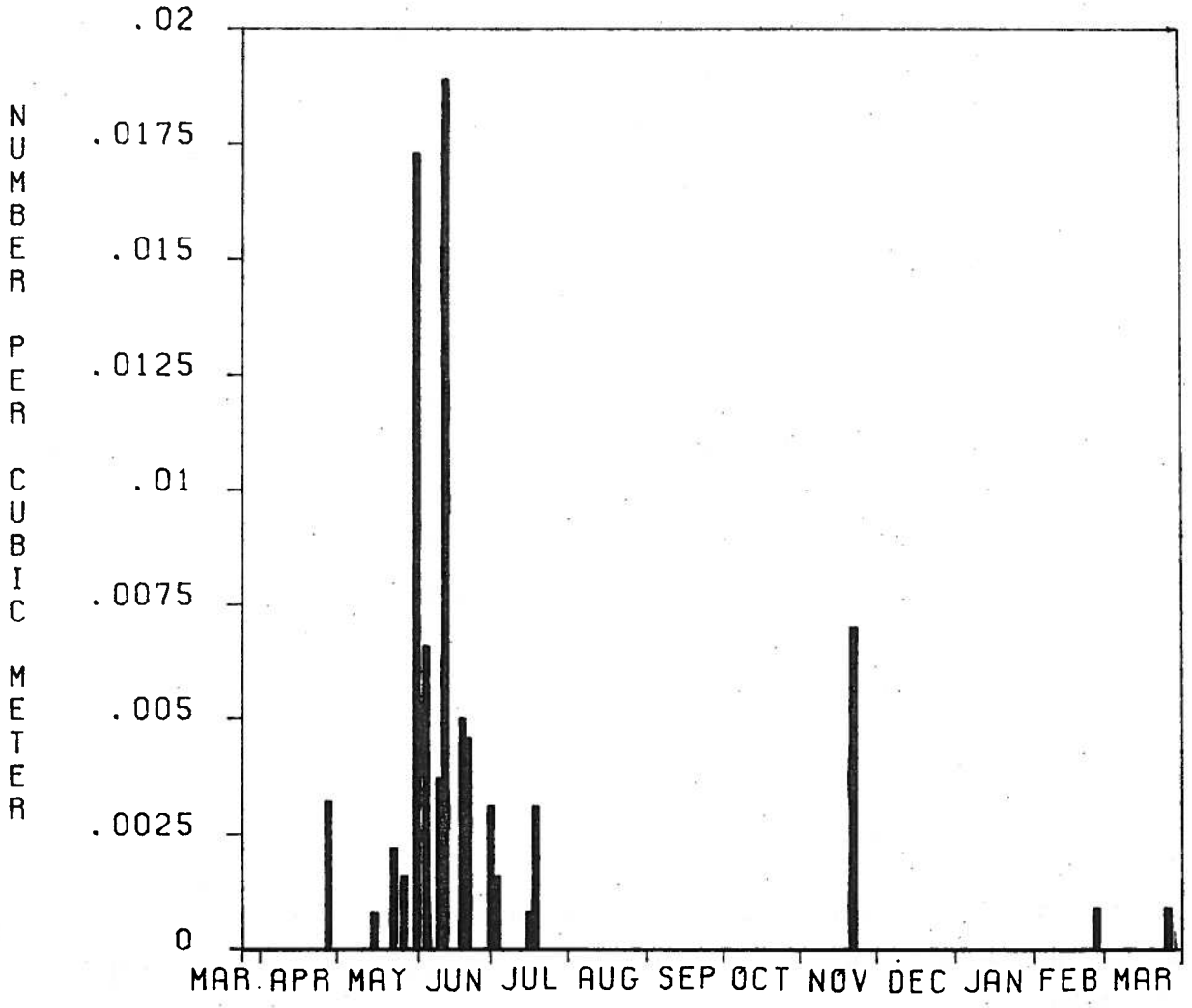


Figure 3-12. Mean densities of threadfin shad larvae and juveniles entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.

Threadfin shad ranged in size (total length) from 3 mm to 99 mm; mean length was 6 mm (Table 3-7). Total numbers of threadfin shad entrained as a function of length (in 1-mm increments) are presented in Table 3-8. Approximately 74 percent of the threadfin shad were entrained during the day (0600-1800 hours) (Table 3-5). Johnson (1970) reports that threadfin shad larvae and juveniles are demersal at night until they reach a length of 20-40 mm, accounting for the higher number entrained during the day.

3.1.3.2.8 Other Taxa

Other taxa entrained in relatively low abundance but having potential economic importance were chinook salmon, Oncorhynchus tshawytscha, and American shad, Alosa sapidissima. Twenty-one American shad collected, most between mid-May and mid-June 1978, represented an entrainment estimate of 170,000 for the first 12 months. They ranged in length from 4 to 66 mm, but mean length was 14 mm. An estimated 17,000 chinook salmon were entrained during the first 12 months of study. One was collected in March 1978 and one in March 1979; one was 39 and one 41 mm long, and both were probably hatchery-release fish. These taxa will be discussed further in Chapter 5.

3.1.3.3 Macroinvertebrates

The major macroinvertebrate taxa entrained from March 1978 to March 1979 were Neomysis mercedis, Corophium stimpsoni, Palaemon macrodactylus, and Rhithropanopeus harrisii.

3.1.3.3.1 Neomysis mercedis (Opossum Shrimp)

Neomysis mercedis was the most abundant macroinvertebrate entrained (Table 3-4). An estimated 9.3 billion were entrained during the first 12 months of the study (March 1978 - March 1979) and 5.9 billion from April to August 1979 (4.2 billion were entrained in the comparable period in 1978). The greater number entrained in 1979 resulted mainly from an increase in cooling water flow (Section 2.1.3): densities were comparable in the two years (Figure 3-13).

Neomysis mercedis were entrained throughout the year, with peak densities (up to $51.1/m^3$) from April to July (Figure 3-13). Reproduction accounts for the most

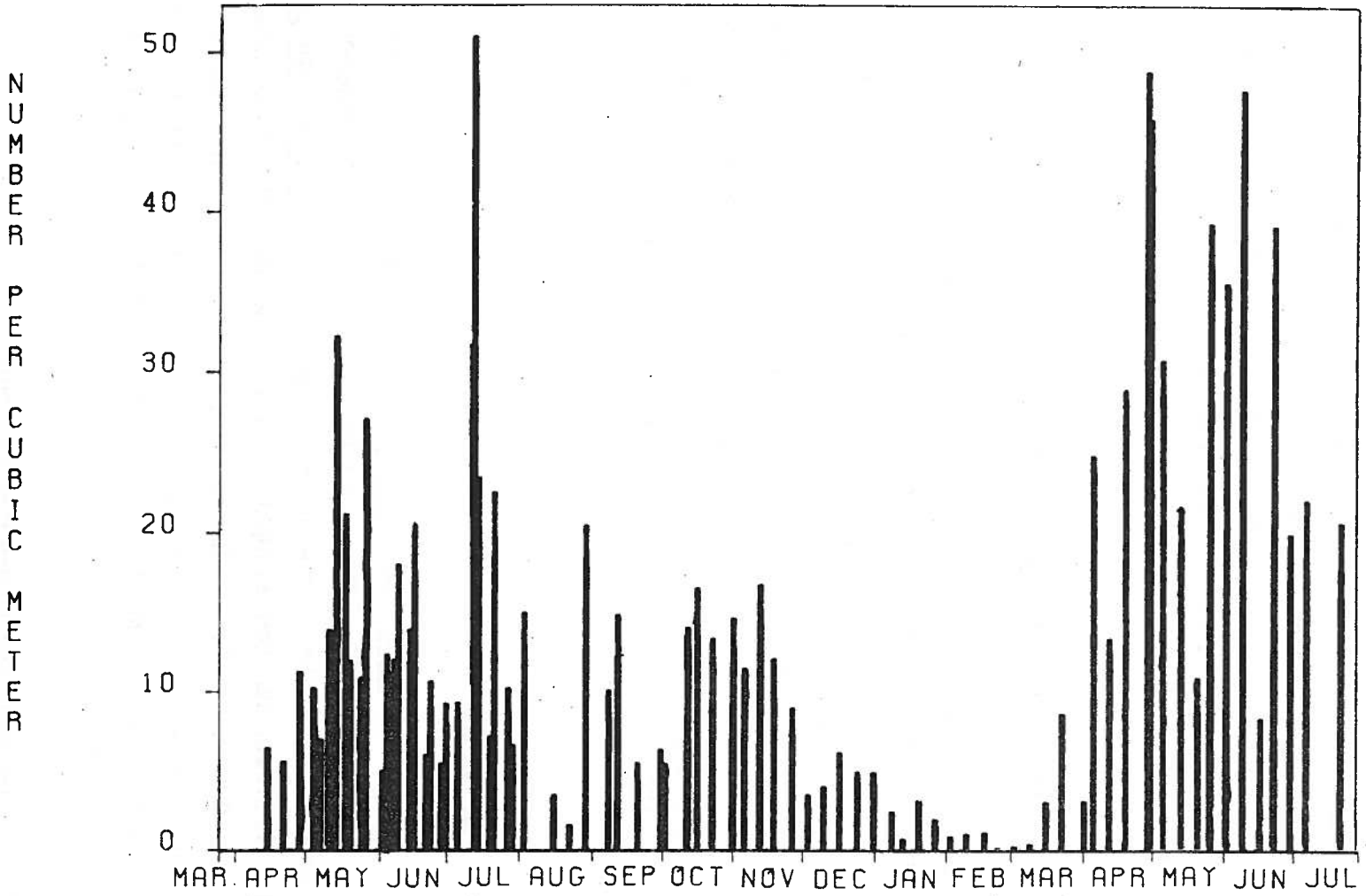


Figure 3-13. Mean densities of Neomysis mercedis entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—July 1979.

significant changes in N. mercedis seasonal abundance; mysid egg production and fecundity are highest in the spring and summer (Heubach 1972).

Neomysis mercedis ranged in size (tip of the rostrum to base of the telson) from 2 mm to 11 mm; the mean length was 4.2 mm (Table 3-7). They were entrained in greater numbers at night than during the day (Table 3-5). Neomysis mercedis are found on the bottom, offshore, and in deep water during the day, and in the water column and inshore at night (Heubach 1972), which accounts for the greater number entrained at night.

3.1.3.3.2 Palaemon macrodactylus (Oriental Shrimp)

An estimated 663 million Palaemon macrodactylus larvae and juveniles were entrained during the first 12 months of the study (Table 3-4). Densities were much lower than those of Neomysis mercedis. Although P. macrodactylus were entrained throughout the year, peak densities (up to $7.0/m^3$) were observed from June through August (Figure 3-14). This period corresponds with their peak spawning season and period of maximum larval abundance (Little 1969; Sitts and Knight 1979). Palaemon macrodactylus were entrained in greater numbers at night than during the day (Table 3-5). This species is known to migrate off the bottom and into the water column at night (Sitts and Knight 1979), accounting for the greater numbers entrained at night.

3.1.3.3.3 Corophium stimpsoni (Amphipod)

An estimated 786 million Corophium stimpsoni were entrained during the first 12 months of the study (Table 3-4). Corophium stimpsoni were entrained throughout the year, but predominantly in the summer months (Figure 3-15). Corophium stimpsoni were entrained in greater numbers at night than during the day (Table 3-5). Although C. stimpsoni is epibenthic, preferring fine sand bottoms in water depths greater than 3 m (10 ft) (Hazel and Kelley 1966), the species is known to migrate off the bottom and into the water column at night (Siegfried et al. 1978), explaining the higher numbers entrained at night.

3.1.3.3.4 Rhithropanopeus harrisii (Pebble Crab)

An estimated 471 million Rhithropanopeus harrisii larvae were entrained at Units 1-7

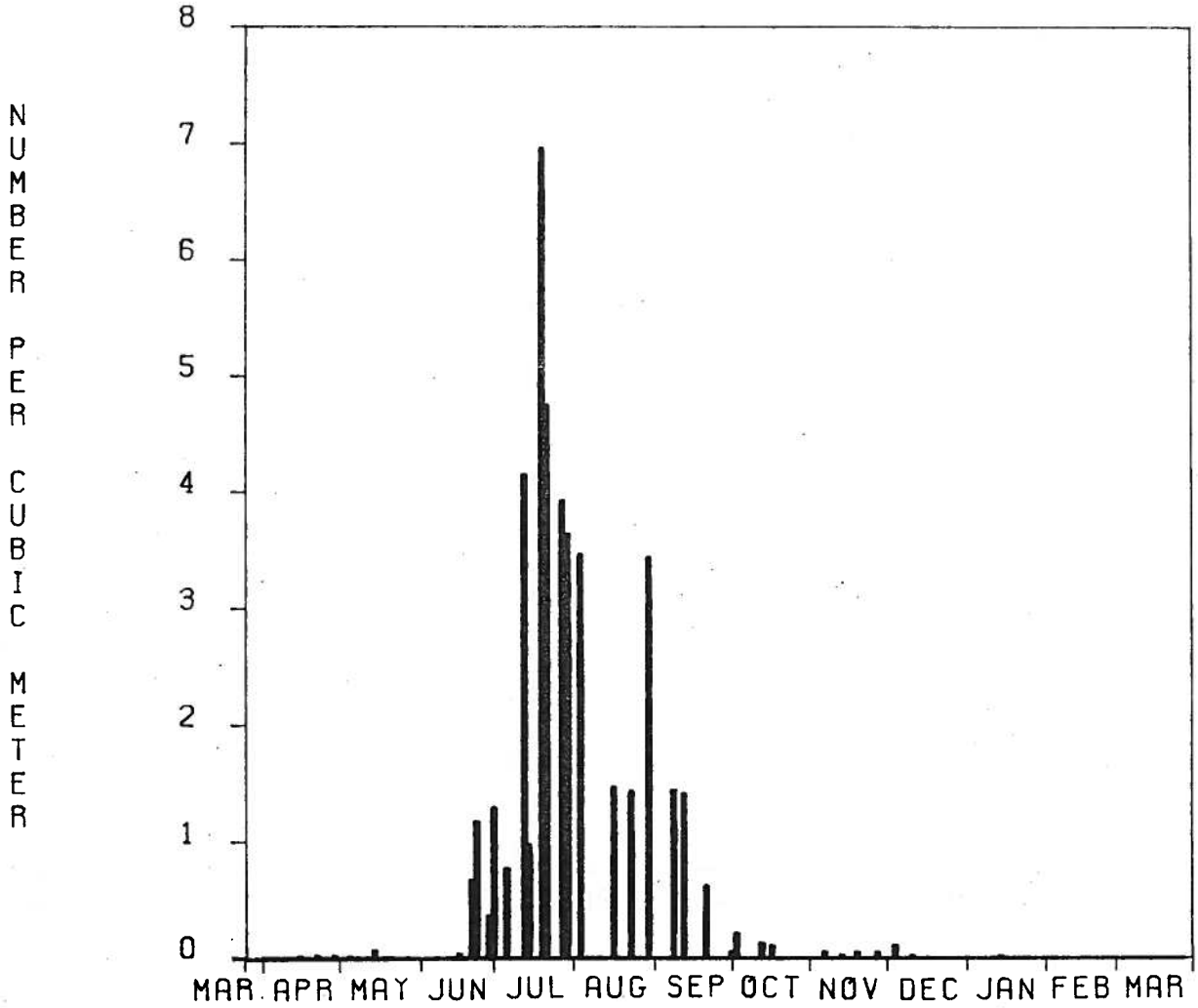


Figure 3-14. Mean densities of Palaemon macrodactylus entrained at Units 1-7 of the Pittsburg Power Plant, March 1978— March 1979.

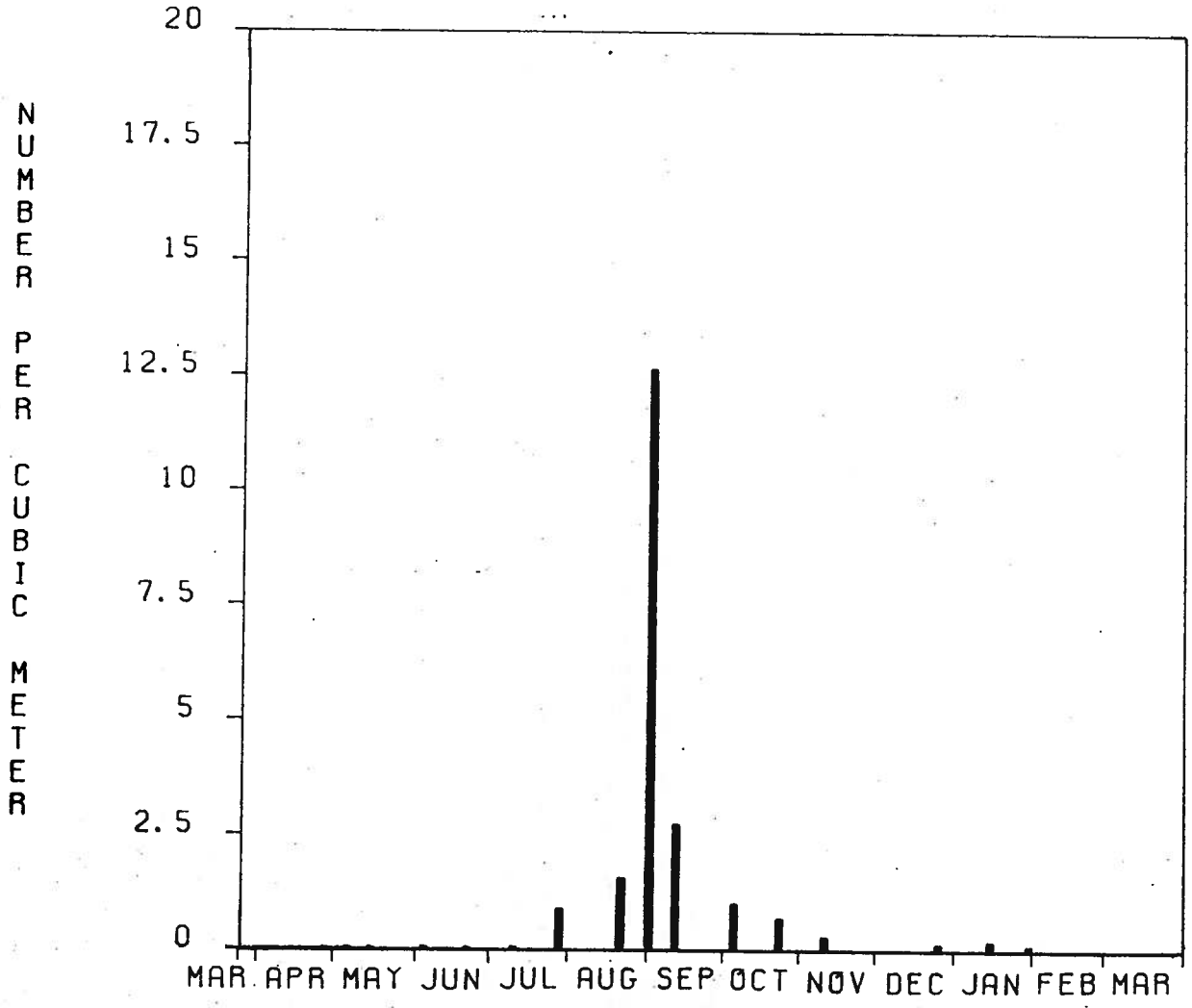


Figure 3-15. Mean densities of Corophium stimpsoni entrained at Units 1-7 of the Pittsburg Power Plant, March 1978—March 1979.

during the first 12 months of the study (Table 3-4). Peak larval densities (up to $4.7/m^3$) occurred from June to August (Figure 3-16). Rhithropanopeus harrisii were entrained predominantly at night (Table 3-5).

3.1.3.3.5 Other Taxa

Other species of invertebrates were entrained (see Appendix E for a complete listing). The more abundant taxa included Eurytemora affinis, Daphnia pulex, and Cyclopidae. Collection of these taxa was biased because the collection net (505-μm mesh) did not adequately retain the smaller or earlier life stages of these and other taxa.

3.2 ENTRAINMENT SURVIVAL

The survival of entrained organisms at the Pittsburg Power Plant was first examined by Kelley (1971), who reported that at discharge temperatures of 30 C (86 F) or lower, the mortality of entrained Neomysis awatschensis (= N. mercedis) was 6 percent or less. When discharge temperatures exceeded 32.2 C (90 F), mortality was approximately 65 percent. These findings are consistent with the thermal tolerance test results reported by Hair (1971): rapid temperature rises for exposure periods of 0-6 minutes caused little or no mortality of N. mercedis if the test temperature did not exceed 30.6 C (87 F). Entrainment mortality of zooplankton (predominantly copepods, polychaete larvae, and barnacle nauplii) at discharge temperatures below 30 C (86 F) was negligible; at 32.5 C (90.5 F) mortality increased to 17 percent (PGandE unpublished). Stevens and Finlayson (1978) reported that the entrainment mortality of striped bass (Morone saxatilis) larvae at the Pittsburg Power Plant was 14 percent when discharge temperatures were 31 C (88 F) or lower. Mortality increased to 50 percent at 33 C (91 F) and to 100 percent at 38 C (100 F). Results of these studies and of other laboratory and field entrainment survival studies (reviewed in Appendix B) indicate that at discharge temperatures of about 30 C (86 F) or lower, entrainment mortality of a variety of aquatic organisms at the Pittsburg Power Plant is low and independent of temperature, but that above 30 C entrainment mortality increases with discharge temperature.

The 316(b) study plan for the Pittsburg Power Plant (PGandE 1977) stated that additional entrainment survival studies would be conducted with larval fish and macroinvertebrates. The results of those studies are presented in this section. The

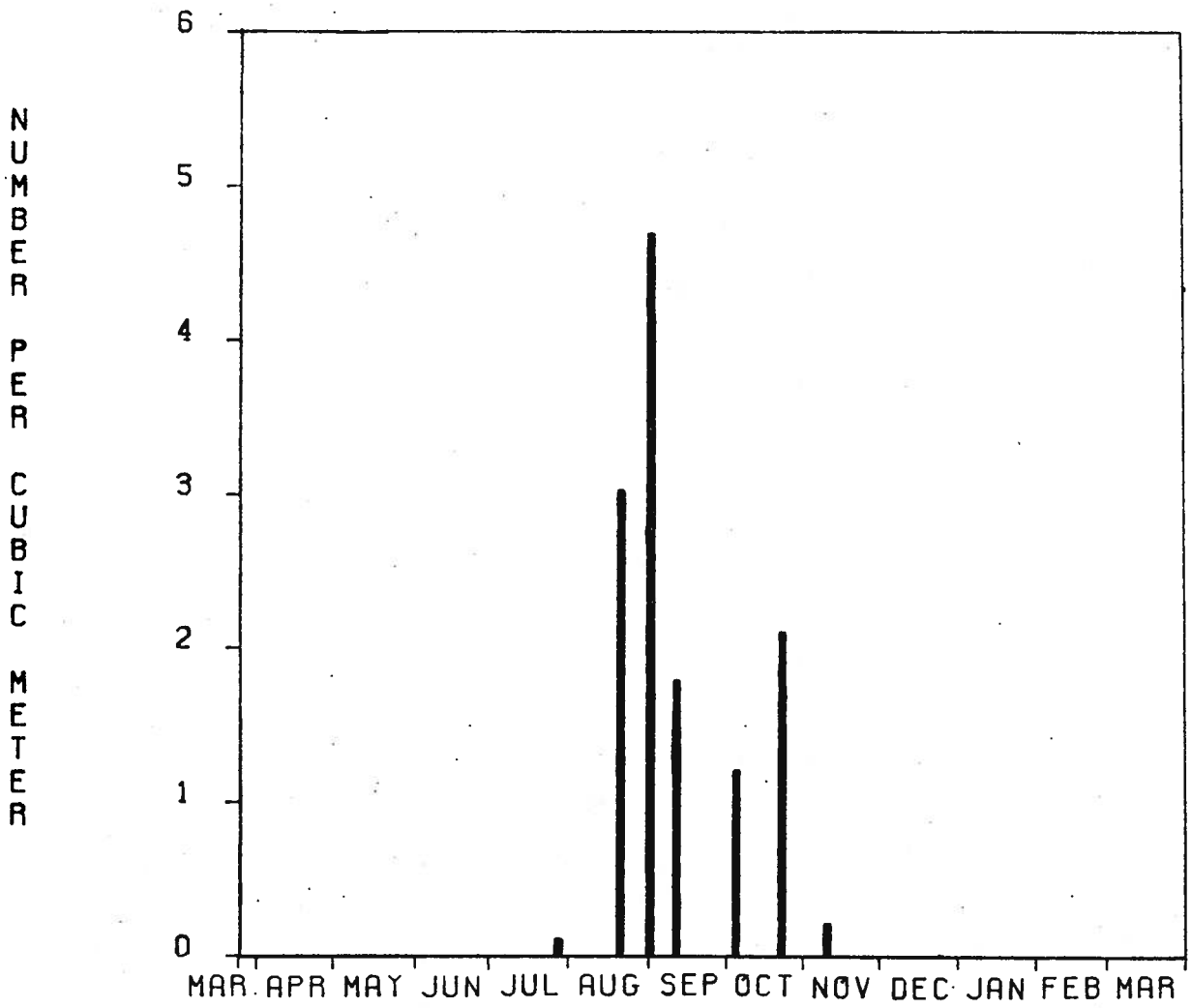


Figure 3-16. Mean densities of Rhithropanopeus harrisii entrained at Units 1-7 of the Pittsburg Power Plant, March 1978–March 1979.

overall objective of those studies was to determine the proportion of larval fish and macroinvertebrates not surviving entrainment, in order to permit estimation of total entrainment cropping.

Specific objectives of the studies were to determine the relationships between entrainment survival rates and:

1. discharge temperatures,
2. recirculation of discharged waters back into the intake, and
3. body size (length) of organism.

Because investigating the entrainment survival of all species of larval fish and macroinvertebrates was not feasible, three key taxa were selected, striped bass larvae, Neomysis mercedis, and gammaridean amphipods. Striped bass larvae were selected for study because:

1. they were the most abundantly entrained fish larvae at Pittsburg Power Plant, and
2. the species supports a valuable sport fishery.

Neomysis mercedis and gammaridean amphipods were selected because:

1. they were among the most abundantly entrained macroinvertebrates, and
2. they are an important food source for many fishes inhabiting Suisun Bay and the Delta.

The proportion of entrained individuals of each key taxon not surviving entrainment was estimated by determining the proportion dead in cooling water samples collected at the discharge relative to the proportion dead in control samples collected at the intake. Control samples were collected to account for mortality from sampling and other causes not related to entrainment. Organisms collected were held for 96 hours to allow for any delayed effects of entrainment.

Individuals of key taxa were collected over a wide range of discharge temperatures, so that the effects of discharge temperature on entrainment survival could be determined. Samples were collected during periods of recirculation of discharged

water back into the intake and in periods of no-recirculation, so that the effects of recirculation on entrainment survival could be determined. Total body lengths of striped bass larvae were measured so the relationship between size and survival could be determined.

3.2.1 Methods

3.2.1.1 Striped Bass

Entrainment survival of striped bass was sampled from May 1978 through July 1978 and from May 1979 through June 1979. Intake samples were collected at approximately middepth from one forebay of each of the two adjacent intake structures (A, B, Figure 3-1). Discharge samples were collected from the Units 1-4, Unit 5, and Unit 6 discharge gate wells, located approximately 800 ft (244 m) upstream of the discharge outfall (D, F, G, Figure 3-1).

Samples were collected by pumping water with two 4-in. (10.2-cm) recessed-impeller pumps through two 4-in. (10.2-cm) hoses, at both the intake and the discharge. The two pumps at each sampling station were operated at approximately 1,700 rpm and had a combined flow rate of about 1,325 liters/min (350 gpm). Both pumps at a sampling station discharged into a flow-reduction trough designed to minimize sampling mortality (Figure 3-17). The trough, known as a larval table, was modified from an original design described by McGroddy and Wyman (1977).

Nets of 505-μm mesh were put over the water entrances of the intake and discharge tables before the beginning of a sampling period to prevent any larval fish from entering a table as it filled. When the table was filled, the nets were removed and the 15-minute sampling period began. Organisms and debris pumped into the larval table were gently diverted by angled screens of 505-μm mesh into the back of the table where they were retained in a screened collection box (Figure 3-17). A flow of 75 liters/min (20 gpm) was maintained through the screens of the collection box to draw organisms and debris into the box.

Filtered water at ambient temperature was withdrawn from the intake area and injected into each sampling station's larval table just upstream of the collection box. The filtered water flow rate was 38-75 liters/min (10-20 gpm). This flow reduced the

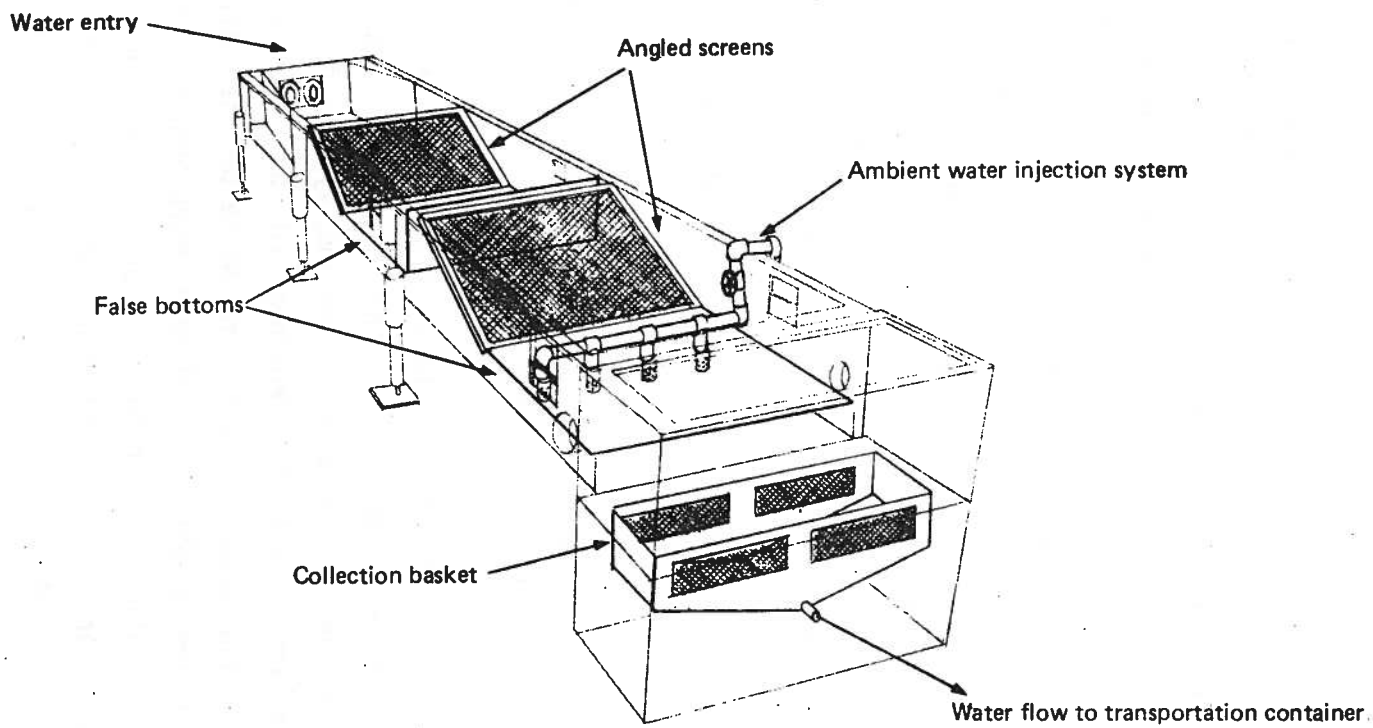


Figure 3-17. Larval table system used in entrainment survival studies at Units 5 and 6 of the Pittsburg Power Plant.

differential between discharge and ambient water temperatures in the collection box at the discharge sample station.

At the end of the 15-minute sampling period, the sampling pumps were turned off and the tables were drained through drain ports located beyond the angled screens and also through the screens of the collection box. The filtered ambient-temperature water continued to be pumped into the collection boxes as the tables drained. At the same time, the tables were thoroughly washed down with a gentle spray of filtered ambient-temperature water, and the remaining organisms and debris were concentrated in the collection boxes. When the tables were completely drained, the collection boxes were emptied into screen-topped containers designed for transporting the samples to the onsite laboratory for processing. The tables were then thoroughly rinsed again, and any organisms or debris remaining were collected in separate containers to quantify the sample collection efficiency (the numbers in the sample divided by the total number collected in the sample and the second washdown). Overall collection efficiency of all tables exceeded 99 percent. Organisms in the second washdown were not examined for viability.

Cooling system operating conditions (number of pumps operating and intake and discharge temperatures) were monitored during entrainment survival sampling. Sampling was scheduled to avoid periods of condenser chlorination; residual chlorine concentrations were not measured, but were considered to be zero. Temperatures were recorded in both the intake and the discharge larval tables.

Intake and discharge samples were sorted immediately in the onsite laboratory. Live and dead fish larvae were separated in the sample containers in an ambient-temperature water bath. Live larvae, those displaying body or operculer movement, were removed from the sample containers with large-bore pipettes and placed in 1-liter jars at a maximum density of five larvae per jar. After all live larvae were removed from the samples, the dead larvae were removed and preserved in 5 percent buffered formalin for later identification and counting. The jars containing live larvae were placed in the ambient-temperature water bath, aerated gently, and held for observation for 96 hours to allow detection of delayed effects. The larvae were not fed during the 96-hour observation period.

At elapsed times of 3, 6, 12, 24, 48, 72, and 96 hours after sample collection, the numbers of live larvae remaining in each jar were recorded. Dead larvae were

removed and preserved for later identification. At the end of the 96-hour observation period, the remaining live larvae were also preserved.

All fish larvae, including those held for 96 hours, were identified, to species when possible, and counted. Total lengths were measured (except for mutilated fish) so that the relationship between larval size and entrainment survival could be investigated.

3.2.1.2 Neomysis mercedis and Gammaridean Amphipods

Entrainment survival of N. mercedis and gammaridean amphipods was sampled from July 1978 through April 1979. Samples were collected from the Units 5 and 6 intake and discharge by pumping water with one 4-in. (10.2-cm) diameter recessed-impeller pump at a flow rate of approximately 660 liters/min (175 gpm). Instead of using larval tables to collect and concentrate the organisms, entrainment abundance tank-net systems (Figure 3-2), modified to enhance survival, were used.

Sampling procedures were similar to those used for striped bass. Just before each sampling period began, a net with 505-µm mesh was placed over the sampling pump discharge while the tank was filled with water. When the tank was full, the net was removed and the 15-minute sampling period began, during which time the water temperature in the tank was recorded and pump speed was maintained at 1,700 rpm. At the end of the sampling a drain hose was connected to a screened portable collection box. The 505-µm collection net was gently and thoroughly rinsed with ambient-temperature water as the tank was drained, washing all organisms into the collection box. The drain hose was then connected to a codend for a final rinse-down of the collection net to document sample collection efficiency. Overall collection efficiency of all samplers exceeded 98 percent.

The samples were immediately sorted in the onsite laboratory. Live and dead mysids ($\geq$7 mm in total length) and gammaridean amphipods ($\geq$5 mm in total length) were separated in the sample containers in an ambient-temperature water bath. Up to 50 live individuals of each taxon, those displaying periopod movement, were removed from the sample containers with large-bore pipettes and placed in 1-liter jars at a maximum density of five individuals per jar. Any additional live individuals and all dead individuals were placed in separate vials and preserved in 5 percent buffered formalin for later identification and counting. The jars containing live mysids and

gammaridean amphipods were placed in the ambient-temperature water bath, aerated gently, and held for observation for 96 hours. All mysids and gammaridean amphipods meeting the size criteria mentioned above were identified and counted in the laboratory. Lengths were not recorded.

3.2.1.3 Analytical Methods

An entrainment mortality probability model was used in analyzing the results of the study. The model was based on the following assumptions:

1. all individuals have an equal probability of dying from natural causes;
2. all individuals have an equal probability of dying from sampling;
3. all individuals have an equal probability of dying from entrainment; and
4. the probability of dying from natural causes and sampling is identical for all individuals collected from the intake and discharge.

The proportion of organisms not surviving entrainment was estimated, based on the above assumptions, as the relative difference between the observed proportions dead in the discharge and intake (Fleiss 1973):

$$\hat{p}_e = \frac{\hat{p}_D - \hat{p}_I}{1 - \hat{p}_I} \qquad (3\text{-}12)$$

where

$\hat{p}_e$ = estimated proportion not surviving entrainment
$\hat{p}_D$ = estimated proportion dead in discharge
$\hat{p}_I$ = estimated proportion dead in intake.

By Equation 3-12, $\hat{p}_e$ is a ratio estimate; its distribution is complex because both the numerator and denominator vary from sample to sample. The limiting distribution of the ratio estimate, as sample size becomes large, is normal (Cochran 1977). Thus, when sample size is large, 95 percent confidence intervals on $\hat{p}_e$ (the true but unknown mortality proportion due to entrainment) may be computed, given

an approximation of the standard error of the estimate $\hat{p}_e$. Such an approximation has been derived from the asymptotic properties of maximum likelihood estimates of Sheps (1959). An algebraic form of the approximation was presented by Fleiss (1973):

$$S.E.(\hat{p}_e) = \frac{1}{(1 - p_I)} \left[\frac{\hat{p}_D(1 - \hat{p}_D)}{\hat{N}_D} + (1 - \hat{p}_e)^2 \frac{\hat{p}_I(1 - \hat{p}_I)}{N_I} \right]^{1/2} \qquad (3\text{-}13)$$

where

N_D = total discharge sample size (live plus dead)
N_I = total intake sample size (live plus dead).

This approximation can also be derived using Cochran's (1977) approximation method for ratio estimates.

Because $\hat{p}_e$ is a ratio estimate, it is not unbiased (Cochran 1977). The nature and extent of the bias was investigated in simulation studies. Simulation results showed that for intake sample sizes of 200 and discharge sample sizes of 100, the bias in p_e was minimal, except when $\hat{p}_I$ was large. For $\hat{p}_I = 0.85$ and $\hat{p}_e = 0.15$, p_e was underestimated by 30 percent. When intake and discharge sample sizes were increased to 400 and 300, respectively, the bias became negligible. In all cases, the standard error approximation deviated from observed variability by less than 10 percent.

The accuracy of the estimate $\hat{p}_e$ depends on the extent to which the four assumptions stated previously are met. Preliminary analysis indicated that either or both of Assumptions 1 and 2 were not met for striped bass larvae: the proportion observed dead at the intake ($\hat{p}_I$) varied with body length, indicating that all individuals did not have an equal probability of dying from natural causes and/or an equal probability of dying from sampling. An estimate of p_e computed for striped bass would be biased if the length frequency distributions at the intake and discharge were not identical. To avoid this bias, p_e was estimated separately for each of several length groups of striped bass larvae. Preliminary analysis also indicated that Assumption 3 was not met for all taxa studied: the probability of dying from entrainment was not equal for all individuals, and in fact was a function of discharge temperature. Consequently, p_e was estimated separately for organisms collected in four ranges of discharge temperatures: less than (LT) 30 C, 30-31.9 C, 32-33.9 C, and greater than or equal to (GE) 34 C. Preliminary analysis indicated that entrainment

mortality was independent of discharge temperature when temperatures were less than 30 C or greater than 34 C.

Under certain tidal conditions, a portion of the water from the plant discharge is recirculated back into the intake and entrained organisms (both live and dead) are re-entrained. Re-entrained organisms were expected to have higher entrainment mortality than organisms entrained for the first time. Preliminary analysis indicated that the proportion dead at the discharge ($\hat{p}_D$) was often greater during periods of recirculation than during periods of no-recirculation. Consequently, to meet the requirements of Assumption 3, that all individuals have an equal probability of dying from entrainment, p_e was estimated separately for organisms collected during recirculation and no-recirculation.

In estimating the proportion not surviving entrainment during recirculation ($\hat{p}_{er}$), Equation 3-13 was modified as follows:

$$\hat{p}_{er} = \frac{\hat{p}_{Dr} - \hat{p}_{Inr}}{1 - \hat{p}_{Inr}} \qquad (3\text{-}14)$$

where

$\hat{p}_{Dr}$ = proportion dead at discharge during <u>recirculation</u>
$\hat{p}_{Inr}$ = proportion dead at intake during <u>no-recirculation</u>.

Equation 3-14 involves the proportion dead at the intake during no-recirculation ($\hat{p}_{Inr}$) because during recirculation dead organisms are re-entrained, increasing the proportion dead at the intake, and this would bias low the estimated proportion not surviving entrainment. Using $\hat{p}_{Inr}$ in Equation 3-14 eliminates this bias.

In summary, to meet the assumptions underlying the entrainment mortality probability model, entrainment mortality was estimated as a function of body length (striped bass only), discharge temperature, and recirculation status. Subdividing the organisms collected into length groups, temperature groups, and recirculation groups reduced the number of organisms (sample size) involved in each estimate of entrainment mortality. Since the precision of the estimates was directly related to sample size, a disadvantage of subdividing was the accompanying decrease in precision.

Consequently, in cases when $\hat{p}_e$ could be shown not to vary among groups (e.g., length groups, recirculation groups), the groups were pooled to increase sample size and, therefore, precision. Specific instances of pooling will be discussed in the next section.

Except where otherwise noted, estimates of entrainment mortality were based on the numbers of organisms alive and dead in the intake and discharge 12 hours after sampling. Preliminary analysis indicated that during the first 12 hours of the 96-hour observation period, instantaneous mortality rates of organisms collected from the discharge exceeded those of organisms collected from the intake, but beyond 12 hours, the intake and discharge rates were not significantly different. It was not possible to determine whether the difference in rates during the first 12 hours was the result of an entrainment-sampling synergism, whereby entrained organisms collected at the discharge were more susceptible to sampling stresses than organisms collected at the intake, or of an actual delayed effect of entrainment. To be conservative, the latter was assumed, and consequently the 12-hour data were used in estimating entrainment mortality. Estimates based on the 12-hour data were generally similar to or higher than the estimates based on initial observations of numbers live and dead. This approach to estimating entrainment mortality ensured that any delayed effects of entrainment were taken into account.

3.2.2 Results

Entrainment survival sampling was scheduled to coincide with the larval striped bass entrainment season; therefore, the predominant fish species collected was striped bass. Other fish species collected incidentally include northern anchovy, gobies, Delta smelt, and several others. A total of 778 fish larvae other than striped bass were collected, 382 from the intake and 396 from the discharge. The intake and discharge proportions dead 12 hours after sample collection were 0.85 and 0.96, respectively. The data were insufficient for estimating entrainment mortality, but mortality appeared to be less than 100 percent.

3.2.2.1 Striped Bass

Entrainment survival results for striped bass are reported in this section for Pittsburg Power Plant Units 5 and 6 and Units 1-4 in turn. Results for Units 5 and 6 are more extensive than for Units 1-4 and therefore are presented first.

3.2.2.1.1 Units 5 and 6

The numbers of live and dead striped bass larvae collected from the intake and discharge are presented in Table 3-9. Larvae collected at the Unit 5 and Unit 6 discharge gate wells were considered together in the following analysis. The numbers of live and dead larvae are presented in the table according to discharge temperature group, recirculation group, and length group. A total of 4,016 striped bass larvae were collected, 2,502 from the intake and 1,514 from the discharge. Of the total collected, 2,309 were collected during periods of no-recirculation and 1,707 during periods of recirculation. Of the 1,514 collected from the discharge, 682 were collected at temperatures less than 30 C, 491 at temperatures of 30-31.9 C, and 341 at temperatures of 32-33.9 C. None were collected at temperatures greater than or equal to (GE) 34 C. Forty percent of the larvae collected were less than or equal to 7 mm in length; approximately 15 percent were in each of the other four length groups: 8-11 mm, 12-15 mm, 16-19 mm, and GE 20 mm.

For the LT 30 C discharge temperature group, separate estimates of the proportion of each length group not surviving entrainment were obtained using Equation 3-12. These estimates and the corresponding estimated standard errors are reported in Table 3-10. Estimates of p_e were highly variable (0.14-0.79) and neither increased nor decreased with increasing length. Standard errors were large (0.08-0.18), reflecting the small sample sizes of most of the length groups.

For the LT 30 C recirculation group, estimates of p_{er} increased with length, ranging from 0.28 for the LE 7 mm group to 0.73 for the GE 20 mm group. Standard errors were large (0.13-0.18). Except for the 12-15 mm group, the estimated proportion not surviving entrainment was higher during recirculation than during no-recirculation.

To test whether or not entrainment mortality was greater during recirculation than during no-recirculation, a chi-square test was conducted comparing the numbers of live and dead larvae of each length group in the LT 30 C recirculation and no-recirculation groups to the numbers live and dead expected under the null hypothesis: $p_e = p_{er}$. In this case the null hypothesis could not be rejected, implying that p_e and p_{er} were not significantly different ($\alpha = 0.05$). However, the power of the test was low, and it is likely that entrainment mortality was higher during recirculation.

TABLE 3-9 NUMBERS OF STRIPED BASS LARVAE ALIVE AND DEAD 12 HOURS AFTER ENTRAINMENT SURVIVAL SAMPLING AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, 1978-1979

Sampling Location	Discharge Temp. (C)	Length (mm)	No-Recirculation		Recirculation		Total	
			Alive	Dead	Alive	Dead	Alive	Dead
Intake		≤7	158	363	110	295	268	658
		8-11	112	152	55	97	167	249
		12-15	111	72	108	90	219	162
		16-19	110	62	94	60	204	122
		≥20	232	80	99	42	331	122
		All	723	729	466	584	1,189	1,313
Discharge	<30	≤7	66	186	28	101	94	287
		8-11	14	29	11	30	25	59
		12-15	6	42	12	23	18	65
		16-19	22	26	12	8	34	34
		≥20	24	32	2	8	26	40
		All	132	315	65	170	197	485
	30-31.9	≤7	4	116	4	30	8	146
		8-11	7	63	3	14	10	77
		12-15	19	67	6	13	25	80
		16-19	18	48	4	10	22	58
		≥20	29	23	5	8	34	31
		All	77	317	22	75	99	392
	32-33.9	≤7	0	3	1	140	1	143
		8-11	0	1	9	34	9	35
		12-15	0	2	5	54	5	56
		16-19	1	3	8	47	9	50
		≥20	0	6	7	20	7	26
		All	1	15	30	295	31	310

TABLE 3-10 ESTIMATES OF THE PROPORTION NOT SURVIVING ENTRAINMENT ($\hat{p}_e$), ($\hat{p}_{er}$) OF STRIPED BASS LARVAE COLLECTED AT DISCHARGE TEMPERATURES LESS THAN 30 C AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, 1978-1979

	No-Recirculation		Recirculation	
Length	p_e	Standard Error	p_{er}	Standard Error
≤7	0.14	0.11	0.28	0.13
8-11	0.23	0.18	0.37	0.17
12-15	0.79	0.08	0.43	0.14
16-19	0.28	0.12	0.62	0.18
≥20	0.42	0.09	0.73	0.17

For the 30-31.9 C and 32-33.9 C groups, sample sizes were insufficient for computing separate estimates of p_e and p_{er}, so the data corresponding to recirculation and no-recirculation were pooled. In the 30-31.9 C case, the estimates of p_e decreased with increasing length, and associated standard errors were fairly small (0.06-0.09) (Table 3-11). Estimates of p_e decreased from 0.83 for the LE 7 mm group to 0.30 for the GE 20 mm group. In the 32-33.9 C group, estimates of p_e ranged between 0.52 for the 8-11 mm group and 0.98 for the LE 7 mm group (Table 3-11). Except for the 8-11 mm group, estimates of p_e decreased with increasing length. Standard errors ranged from 0.02 to 0.15.

The estimates of p_e for striped bass larvae reported in Tables 3-10 and 3-11 are highly variable, even within a given temperature group. The relationship of body length and entrainment mortality was not consistent. Variability in the data may have masked any actual relationship of length and mortality. Also, the process of subdividing the data into length groups, temperature groups, and recirculation groups, reduced sample sizes within specific groups and sacrificed precision .

Because of the lack of a consistent relationship between length and mortality, length groups were pooled and a single combined ratio estimate (Cochran 1977) of p_e was computed for each temperature group (and recirculation group in the LT 30 C case):

$$\hat{p}_e = \frac{\hat{p}_{D'} - \hat{p}_I}{1 - \hat{p}_I} \tag{3-15}$$

with an associated standard error

$$S.E.(\hat{p}_e) = \frac{1}{1 - \hat{p}_I} \left[\sum_{i=1}^{L} \left(\frac{N_{Ii}}{N_I} \right) \frac{\hat{p}_{Di}(1 - \hat{p}_{Di})}{N_{Di}} + (1 - p_e)^2 \sum_{i=1}^{L} \left(\frac{N_{Ii}}{N_I} \right) \frac{\hat{p}_{Ii}(1 - \hat{p}_{Ii})}{N_{Ii}} \right]^{1/2} \tag{3-16}$$

where

$$\hat{p}_{D'} = \sum_{i=1}^{L} \frac{N_{Ii}}{N_I} \hat{p}_{Di}$$

TABLE 3-11 ESTIMATES OF THE PROPORTION NOT SURVIVING ENTRAINMENT ($\hat{p}_e$) OF STRIPED BASS LARVAE COLLECTED AT DISCHARGE TEMPERATURES OF 30-31.9 C AND 32-33.9 C AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, 1978-1979

	Temperature (C)			
	30-31.9		32-33.9	
Length	p_e	Standard Error	p_e	Standard Error
≤7	0.83	0.06	0.98	0.02
8-11	0.73	0.08	0.52	0.15
12-15	0.61	0.07	0.87	0.06
16-19	0.57	0.08	0.76	0.07
≥20	0.30	0.09	0.72	0.10

$\hat{p}_{Di}$ = proportion of length class i dead in the discharge

$\hat{p}_{Ii}$ = proportion of length class i dead in the intake

N_i = discharge sample size (live + dead) of length class i

N_{Ii} = intake sample size (live + dead) of length class i

L = number of length classes.

Estimating p_e in this fashion for the five length groups pooled together offered several advantages. Pooling length groups resulted in increased sample size, and therefore reduced standard errors of $\hat{p}_e$ (increased precision) and reduced bias in $\hat{p}_e$ (increased accuracy). Use of Equation 3-15 permitted the pooling of length groups while taking into account differences in $\hat{p}_I$ among groups. If length groups had been ignored and p_e had been estimated using Equation 3-12 for all lengths combined, a significant bias could have resulted, especially if (1) $\hat{p}_I$ varied among length groups, and (2) the length frequency distributions in the intake and discharge samples were not identical.

Pooled estimates of p_e computed using Equation 3-15 for the LT 30 C no-recirculation and recirculation group, the 30-31.9 C group, and the 32-33.9 C group were 0.37, 0.43, 0.57, and 0.77, respectively. The approximate standard errors of $\hat{p}_e$ for the four groups were 0.05, 0.07, 0.04, and 0.04. The estimates of entrainment mortality clearly increased with temperature, and were greater during recirculation than no-recirculation.

For the LT 30 C group, a single estimate of entrainment mortality was needed for purposes of estimating entrainment losses (Section 3.3). Since recirculation has been observed to occur approximately half the time at the Pittsburg Power Plant intakes (Section 2.1.3.2), the no-recirculation and recirculation estimates of p_e were averaged to obtain a single estimate of 0.40. The standard error of the average was 0.06.

The basis for the selection of the 12-hour observation for estimating entrainment mortality is apparent in the plots of the proportions of initially live striped bass larvae remaining alive at various times within the 96-hour holding period. The discharge

proportions surviving, computed separately for the LT 30 C, 30-31.9 C, and 32-33.9 C groups (recirculation and no-recirculation combined) were plotted versus time, along with the intake proportions surviving for comparison (Figure 3-18).

The graphs for the three cases were very similar. In all cases, the discharge proportion surviving was less than the intake proportion surviving during the first 12 hours of the observation period. The differences between the intake and discharge proportions surviving were greatest in the LT 30 C case.

Instantaneous mortality rates (percent mortality per hour) were computed for the intake larvae and discharge larvae of the LT 30 C group (Figure 3-19). The instantaneous mortality rates were greatest during the first 12 hours of the holding period, and the discharge mortality rates exceeded those at the intake. The intake and discharge mortality rates at 24 and 48 hours were low and nearly identical. After 48 hours, both intake and discharge mortality rates increased, probably reflecting starvation of the larvae in the holding containers.

We attempted to fit a constant mortality rate model, the basic model used in death process analysis (Ricker 1975), to the intake and discharge data of Figure 3-19. By this model, the expected number of individuals alive at time t is

$$N_t = N_0 e^{-\lambda t} \tag{3-17}$$

where

N_t = expected number live at time t
N_0 = number live initially
λ = instantaneous mortality rate (proportion dying per hour).

Because the instantaneous mortality rate was observed to increase after 48 hours, the 72- and 96-hour data points were omitted from the analysis. The constant rate model did not fit the data, however, and consequently a more complex model was examined which permitted the postulating of several subgroups of larvae at the intake and discharge, each subgroup with its own constant mortality rate:

$$N_t = p_1 e^{-\lambda_1 t} + p_2 e^{-\lambda_2 t} + \ldots + p_n e^{-\lambda_n t} \tag{3-18}$$

where

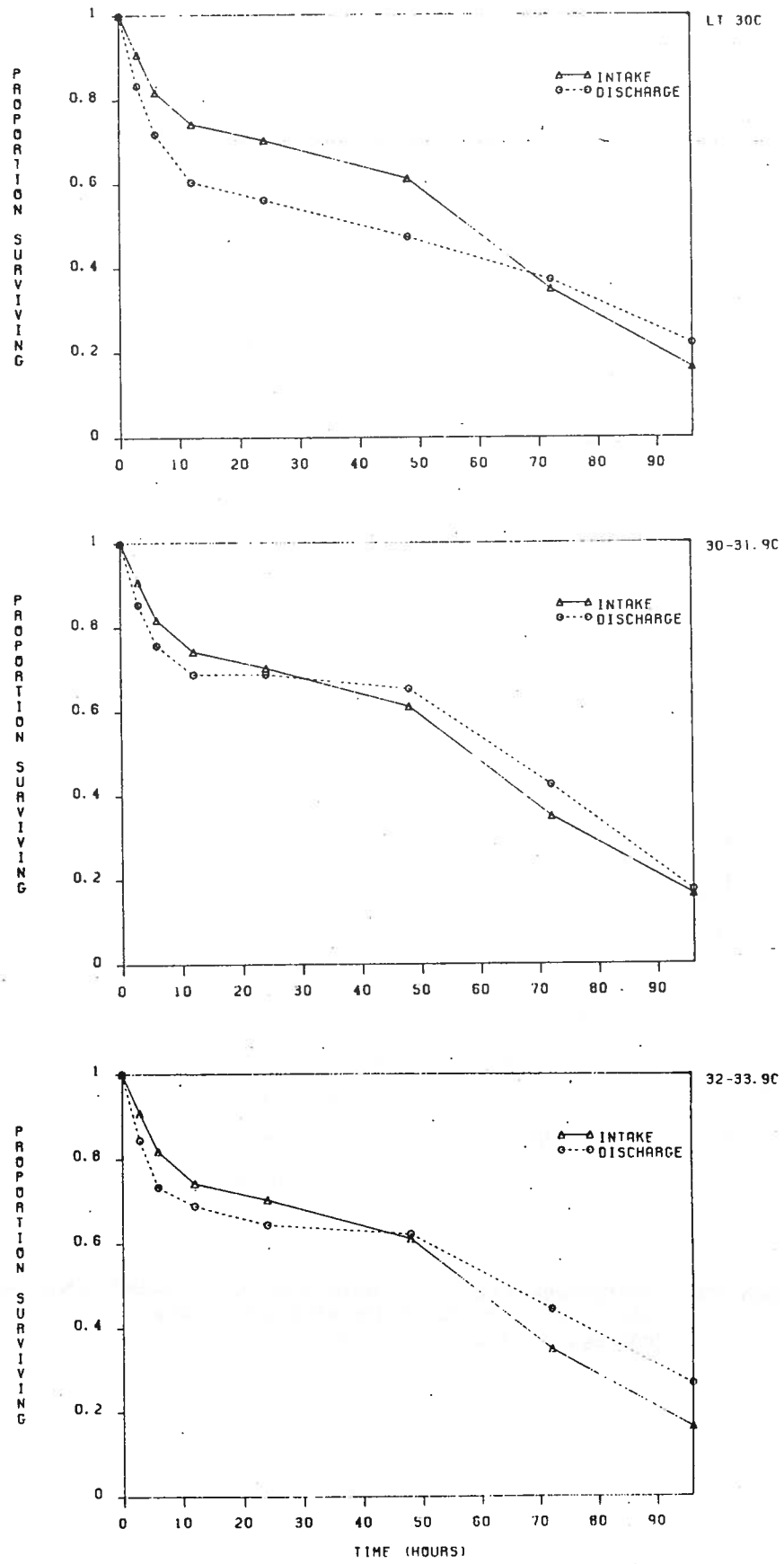


Figure 3-18. **Intake and discharge proportions of initially live striped bass surviving through 96 hours of observation at Units 5 and 6 of the Pittsburg Power Plant, 1978, 1979.**

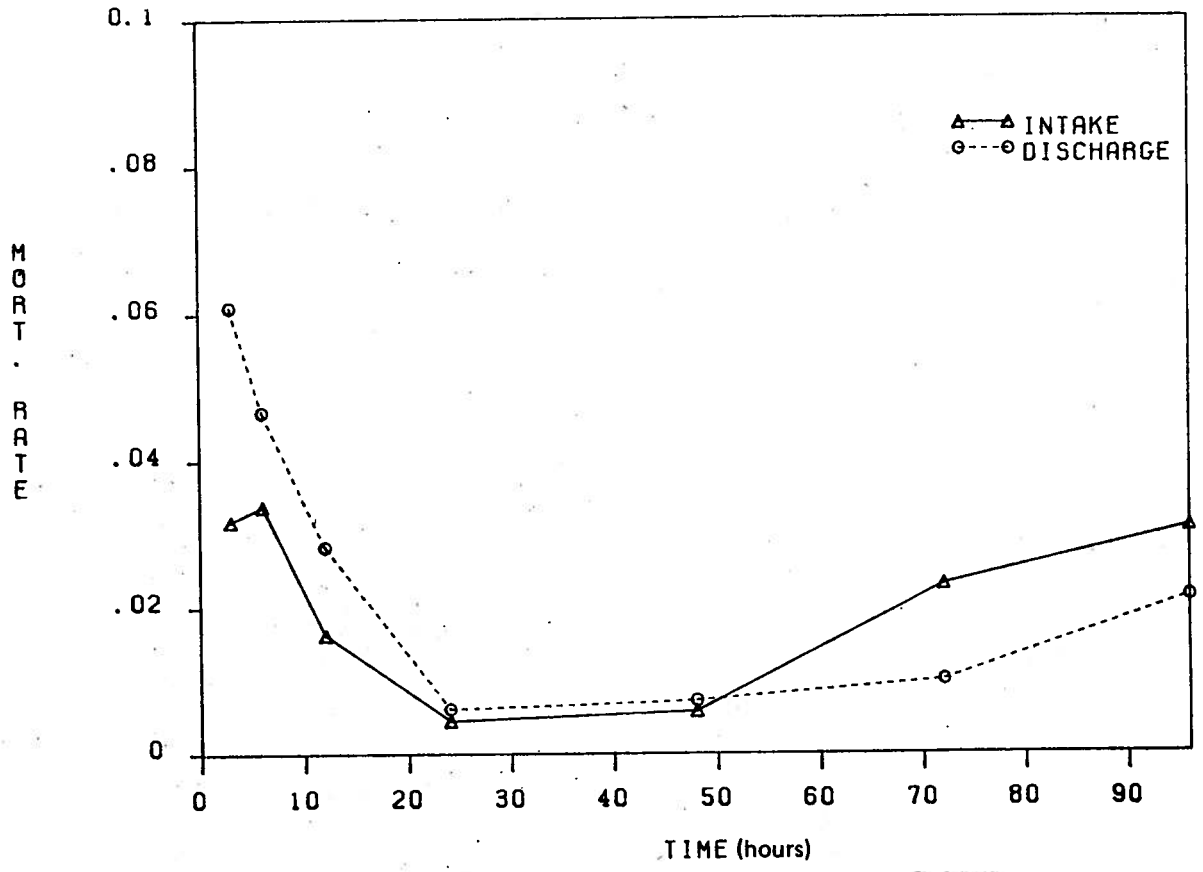


Figure 3-19. Instantaneous mortality rates through 96 hours for initially live striped bass collected at less than 30 C in the intake and discharge of Units 5 and 6 of the Pittsburg Power Plant, 1978, 1979.

$p_1, p_2, \ldots, p_n$ = proportions of the larvae in each subgroup ($p_1 + p_2 + \ldots + p_n = 1$)

$\lambda_1, \lambda_2, \ldots, \lambda_n$ = instantaneous mortality rate of each subgroup

n = number of subgroups.

The postulated subgroups may be viewed as strength groups, where the strongest group consists of those larvae not damaged by entrainment and/or sampling, and the weakest group consists of those larvae most severely damaged yet still living at the beginning of the holding period.

The above model was fitted to the observed results using maximum likelihood estimation procedures. In both the intake and discharge cases, the model fit best when two hypothetical strength groups, "weak" and "strong," were defined. Estimates of model parameters are presented in Table 3-12.

The estimates of the model parameters may provide insight into differences between the overall intake and discharge mortality curves that might be due to the fact that the postulated strong group made up 78 percent of the intake larvae but only 63 percent of the discharge larvae. The instantaneous mortality rates of the two strength groups were very different, both for the intake larvae and for the discharge larvae, but the instantaneous mortality rates of the intake and discharge strong groups were not significantly different ($\alpha = 0.05$), nor were the mortality rates of the intake and discharge weak groups. In the framework of the model, the observed declining mortality rate of larvae collected at both the intake and discharge throughout the first 48 hours of the holding period can be attributed to the rapid dying off of the weaker larvae, leaving a progressively greater proportion of stronger larvae that had a lower constant mortality rate.

Entrainment stresses may have resulted in the greater proportion of weak larvae which the model indicates were observed at the discharge. However, a difference in the magnitude of sampling stresses at the intake and discharge could also have led to the same result. If entrained larvae were more susceptible to sampling stresses, a greater proportion of the larvae sampled at the discharge would be expected to be weaker than those sampled at the intake.

Although it is impossible to determine whether the greater proportion of weak larvae which the model indicates were observed at the discharge was the result of an

TABLE 3-12 ESTIMATED[a] PARAMETERS OF A STRENGTH GROUP MODEL OF THE MORTALITY OF STRIPED BASS LARVAE AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, 1978-1979

Parameter[b]	Intake		Discharge LT 30 C	
	Value	Standard Error	Value	Standard Error
P_1	0.778	0.0181	0.625	0.038
P_2	0.222	0.0181	0.375	0.038
λ_1	0.005	0.0005	0.005	0.0009
λ_2	0.202	0.0217	0.236	0.0291

(a) Using Equation 3-18.
(b) p = proportion in strength group; λ = instantaneous mortality rate (proportion dying per hour).

entrainment-sampling synergism or was due to entrainment alone, we conservatively assumed the latter. Estimates of p_e were therefore computed based on the numbers alive and dead in the intake and discharge 12 hours after collection, because intake and discharge mortality rates were generally not different beyond 12 hours (Figure 3-18).

3.2.2.1.2 Units 1-4

The numbers of live and dead striped bass larvae collected from the Unit 4 intake and discharge are presented in Table 3-13. Substantially fewer striped bass larvae were collected at the Units 1-4 discharge than at the Units 5 and 6 discharge, mainly because the sampling effort at Units 1-4 was less extensive. A total of 149 striped bass larvae were collected at the Units 1-4 discharge, 71 during no-recirculation and 78 during recirculation. Of the total collected at the discharge, 91 were collected at temperatures less than 30 C, 49 at temperatures of 30-31.9 C, 4 at temperatures of 32-33.9 C, and 5 at temperatures greater than or equal to 34 C.

Because of the small number of larvae collected at the Units 1-4 discharge, the no-recirculation and recirculation groups were pooled, as were all the length groups. The proportion not surviving entrainment ($\hat{p}_e$) and its standard error were estimated using Equations 3-12 and 3-13. For the 30-31.9 C temperature group, $\hat{p}_e$ and S.E.($\hat{p}_e$) were 0.69 and 0.08, respectively. For the 32-33.9 C and GE 34 C temperature groups, the number of larvae collected was insufficient for estimating p_e, but based on the results observed at Units 5 and 6, entrainment mortality was probably very high. For the LT 30 C group, the estimates of p_e and its standard error were 0.38 and 0.06, respectively.

In this case, p_e was estimated on the basis of the initial observation rather than the 12-hour observation, because, as is explained below, larvae collected at the discharge survived in higher proportions through the first 48 hours of the holding period than did larvae collected at the intake. The potential delayed effects of entrainment were examined in the same manner as described in Section 3.2.2.1.1 for Units 5 and 6. The proportions of initially live striped bass larvae remaining alive at various times within the 96-hour holding period were computed for larvae collected at the intake and at the discharge. Because of sample size limitation, only the LT 30 C group was considered, and larvae collected during no-recirculation and recirculation were pooled. The proportions surviving are plotted against time in Figure 3-20. As

TABLE 3-13 NUMBERS OF STRIPED BASS LARVAE INITIALLY ALIVE AND DEAD COLLECTED AT VARIOUS DISCHARGE TEMPERATURE RANGES DURING ENTRAINMENT SURVIVAL SAMPLING AT UNITS 1-4 OF THE PITTSBURG POWER PLANT, 1978-1979

Sampling Location	Discharge Temp. (C)	Length Interval (mm)	No-Recirculation		Recirculation		Total	
			Alive	Dead	Alive	Dead	Alive	Dead
Intake		≤7	298	223	207	198	505	421
		8-11	180	83	86	66	266	149
		12-15	138	44	131	67	269	111
		16-19	119	54	105	49	224	103
		≥20	240	73	103	38	343	111
		All	975	477	632	418	1,607	895
Discharge	<30	≤7	0	2	0	0	0	2
		8-11	4	16	0	4	4	20
		12-15	3	26	3	7	6	33
		16-19	22	10	4	7	26	17
		≥20	20	15	9	4	29	19
		All	49	69	16	22	65	91
	30-31.9	≤7	0	0	0	0	0	1
		8-11	0	0	0	8	0	8
		12-15	0	0	1	19	1	19
		16-19	0	0	3	15	3	15
		≥20	1	0	8	6	9	6
		All	1	0	12	49	13	49
	32-33.9	≤7	0	0	0	0	0	0
		8-11	0	0	0	0	0	0
		12-15	0	0	0	0	0	0
		16-19	0	1	0	0	0	1
		≥20	0	1	0	2	0	3
		All	0	2	0	2	0	4
	≥34	≤7	0	0	0	0	0	0
		8-11	0	0	0	0	0	0
		12-15	0	0	0	0	0	0
		16-19	0	0	0	0	0	0
		≥20	0	0	0	5	0	5
		All	0	0	0	5	0	5

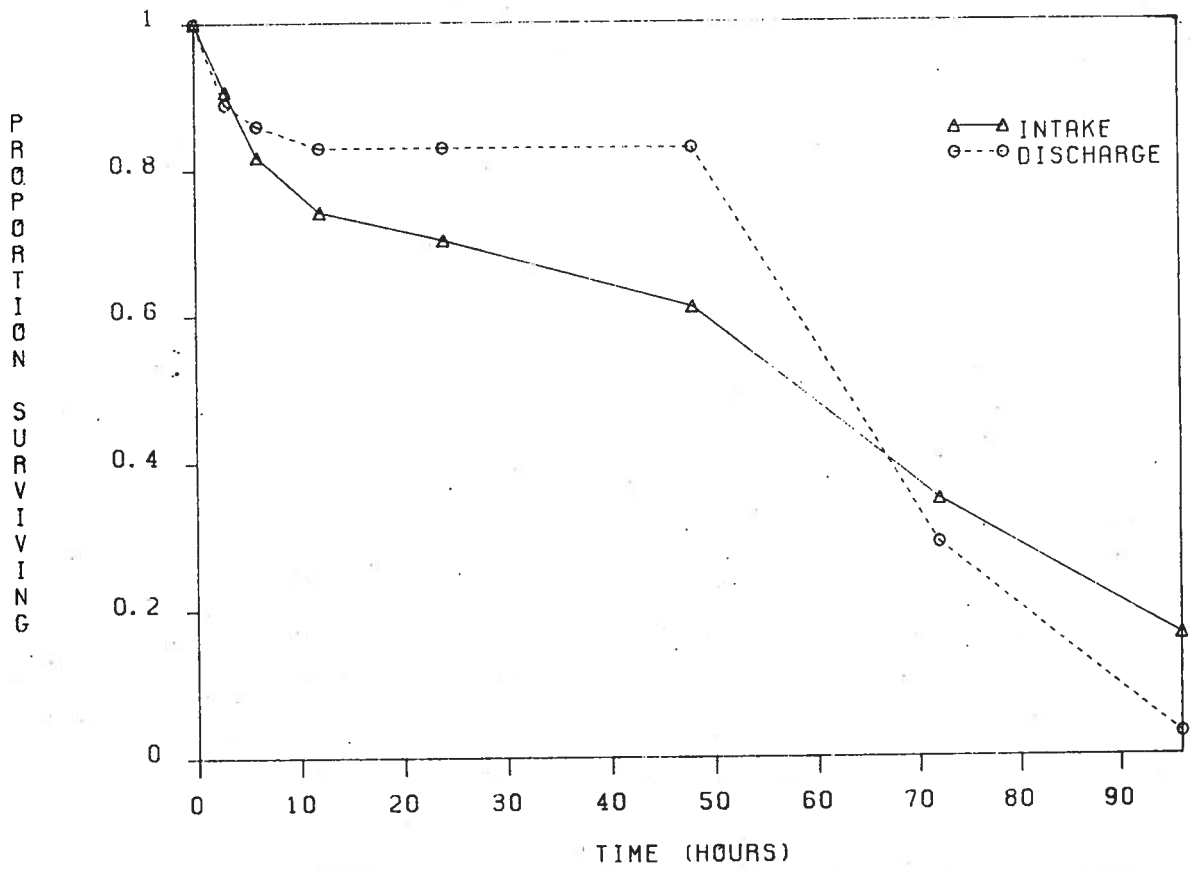


Figure 3-20. **Intake and discharge proportions of initially live striped bass collected at less than 30 C surviving through 96 hours of observation at Units 1-4 of the Pittsburg Power Plant, 1978, 1979.**

mentioned above, the discharge larvae survived in higher proportions than the intake larvae through the first 48 hours of the holding period. (As at Units 5 and 6, after 48 hours the mortality rates of both intake and discharge larvae increased, probably because of starvation).

The data presented in Figure 3-20 do not suggest any delayed effects of entrainment. However, in the framework of the strength group model discussed previously, the data do suggest the presence of weak and strong larvae in different proportions at the intake and discharge. Contrary to the case at Units 5 and 6, the proportion of postulated weak larvae was greater at the intake than at the discharge. A plausible explanation is that either the entrainment-sampling synergism or entrainment alone resulted in the death of a large proportion of the postulated weak larvae collected at the discharge, so that the proportion of the weak larvae was less at the discharge than at the intake. However, the mortality rate of the postulated strong larvae at the intake was greater than that of the strong larvae at the discharge, and therefore, delayed effects of entrainment seemed unlikely. It was concluded that the estimates of p_e based on the initial observation adequately accounted for all entrainment-related mortality at Units 1-4.

3.2.2.2 Neomysis mercedis and Gammaridean Amphipods

Entrainment survival results for Neomysis mercedis and gammaridean amphipods are reported in this section for Pittsburg Power Plant Units 5 and 6. Because of similarities in cooling system design and in observed survival of entrained striped bass (e.g., at LT 30, estimated mortalities were 38 and 40, respectively percent at Units 1-4 and Units 5 and 6), studies were not conducted for macroinvertebrates at Units 1-4.

The numbers of Neomysis mercedis and gammaridean amphipods alive and dead 12 hours after collection from the intake and discharge are presented in Table 3-14. A total of 5,767 N. mercedis were collected, 3,562 from the intake and 2,205 from the discharge. Of these, 3,156 were collected during periods of no-recirculation and 2,611 during recirculation. Of the 2,205 collected at the discharge, 667 were collected at temperatures less than 30C, 531 at 30-31.9 C, 404 at 32-33.9 C, and 603 at 34 C or more.

A total of 846 gammaridean amphipods were collected, 575 from the intake and 271 from the discharge (Table 3-14). Of the total, 615 were collected during periods of

TABLE 3-14 NUMBERS OF NEOMYSIS MERCEDIS AND GAMMARIDEAN AMPHIPODS ALIVE AND DEAD 12 HOURS AFTER ENTRAINMENT SURVIVAL SAMPLING AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, 1978-1979

Sampling Location	Discharge Temp. (C)	No-Recirculation		Recirculation		Total	
		Alive	Dead	Alive	Dead	Alive	Dead
		Neomysis mercedis					
Intake		1,228	463	1,295	576	2,523	1,039
Discharge	<30	204	146	220	97	424	243
	30-31.9	86	258	21	166	107	424
	32-33.9	0	312	0	92	0	404
	≥34	0	459	1	143	1	602
		Gammarideans					
Intake		364	85	100	26	464	111
Discharge	<30	50	14	28	5	78	19
	30-31.9	34	5	54	5	88	10
	32-33.9	7	14	1	2	8	16
	≥34	9	33	0	10	9	43

no-recirculation and 231 during recirculation. Of the 271 collected at the discharge, 97 were collected at temperatures less than 30 C, 98 at 30-31.9 C, 24 at 32-33.9 C, and 52 at 34 C or more. Most of the gammaridean amphipods were Corophium stimpsoni and C. spinicorne.

The proportion dead in intake samples collected during no-recirculation was relatively low for both taxa: 0.27 for N. mercedis and 0.19 for gammaridean amphipods. This indicates that sampling mortality was not excessive during the study. The intake proportion dead during recirculation was greater than during no-recirculation for N. mercedis (0.31) and for gammaridean amphipods (0.21).

Estimates of the proportion initially not surviving entrainment (p_e) and its standard error are presented in Table 3-15 for both taxa. The estimates were computed based on the proportion dead in the no-recirculation intake group and the combined recirculation and no-recirculation discharge groups. The discharge groups were combined because sample sizes were too small to permit separate estimates of p_e and p_{er} or the proportion dead during recirculation was less than that during no-recirculation, contrary to the trend that constituted the basis for the separate calculation.

Estimates of p_e for both taxa increased with increasing discharge temperature. For N. mercedis, $\hat{p}_e$ was 0.10 at discharge temperatures less than 30 C, 0.72 at 30-31.9 C, and 1.0 at 32 C and higher. For gammaridean amphipods, $\hat{p}_e$ was 0-0.04 at temperatures less than 32 C, 0.59 at 32-33.9 C, and 0.79 at 34 C or more.

The delayed effects of entrainment were examined for both taxa in the same way as in Section 3.2.2.1.1 for striped bass larvae at Units 5 and 6. The proportions of initially live N. mercedis and gammaridean amphipods remaining alive at various times during the 96-hour holding period are plotted against time in Figures 3-21 and 3-22. Survival curves for both the intake and discharge are shown on each figure for comparison. The discharge data correspond to the LT 30 C and 30-31.9 C groups, no-recirculation and recirculation combined.

The survival curves for N. mercedis and gammaridean amphipods are similar to those for striped bass larvae at Units 5 and 6, except that the proportion surviving at the intake did not decrease as rapidly during the first few hours of the holding period. The constant mortality rate model specified by Equation 3-18 was applied to the intake and

TABLE 3-15 ESTIMATES OF THE PROPORTION OF NEOMYSIS MERCEDIS AND GAMMARIDEAN AMPHIPODS NOT SURVIVING ENTRAINMENT (p_e) MEASURED 12 HOURS AFTER SAMPLING AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, 1978-1979

Discharge Temperature (C)	Neomysis mercedis		Gammaridean Amphipods	
	$\hat{p}_e$	Standard Error	$\hat{p}_e$	Standard Error
<30	0.10	0.03	0.04	0.05
30-31.9	0.72	0.02	0.00	0.04
32-33.9	1.00	0.00	0.59	0.12
≥34	1.00	0.00	0.79	0.07

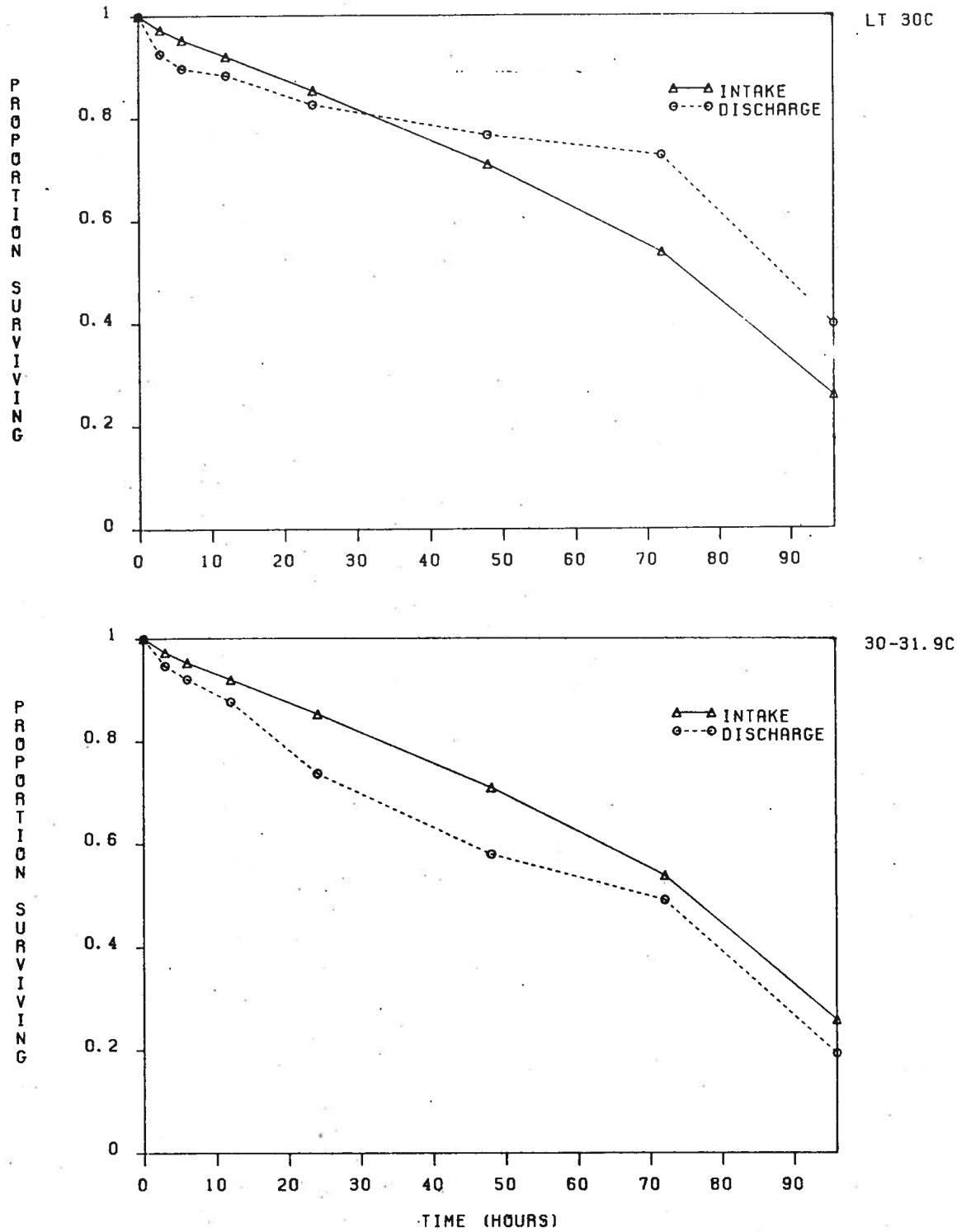


Figure 3-21. Intake and discharge proportions of initially live *Neomysis mercedis* surviving through 96 hours of observation at Units 5 and 6 of the Pittsburg Power Plant, 1978, 1979.

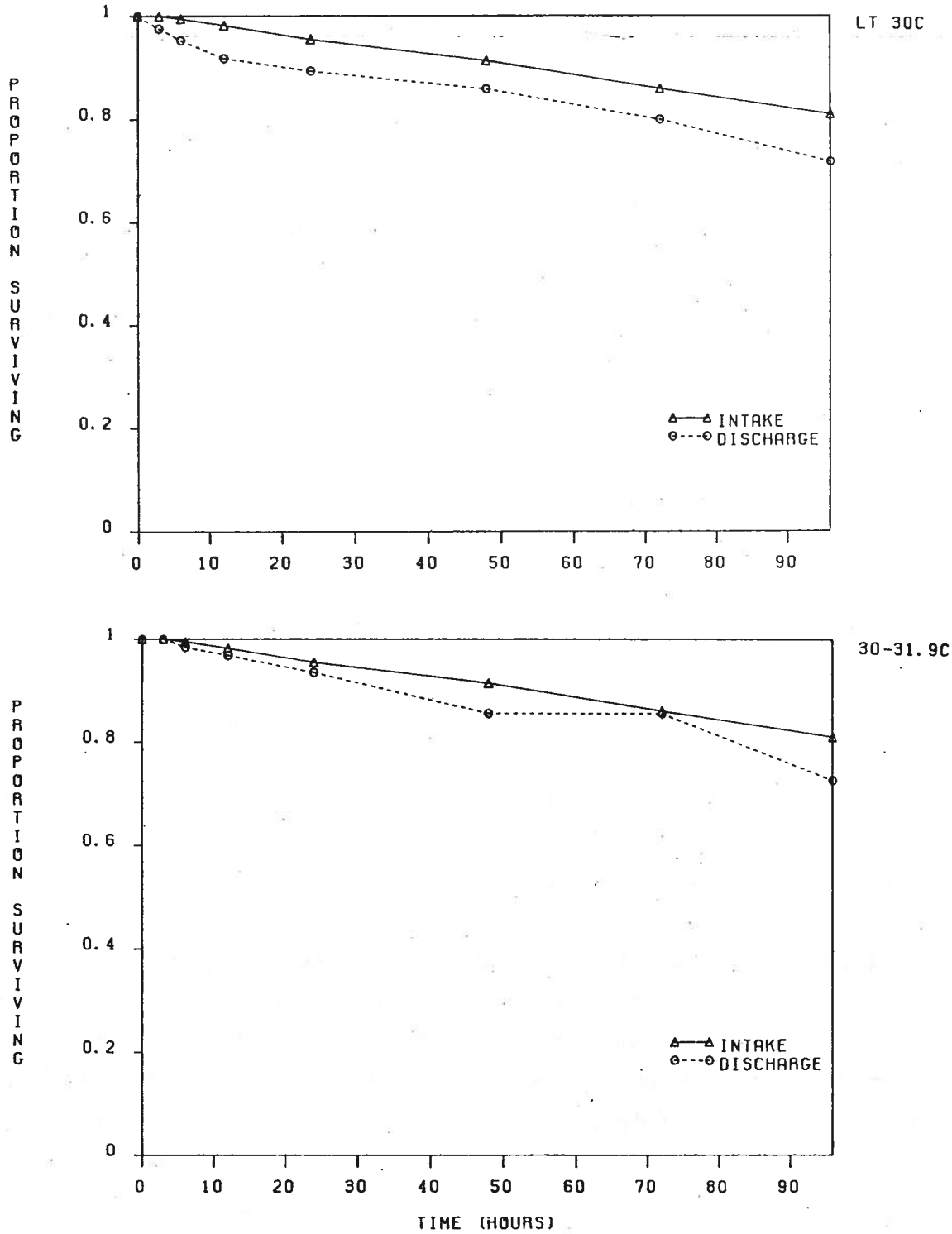


Figure 3-22. **Intake and discharge proportions of initially live Gammaridea surviving through 96 hours of observation at Units 5 and 6 of the Pittsburg Power Plant, 1978, 1979.**

discharge data for N. mercedis (LT 30 C) as was done for striped bass larvae. At the intake, the model fit well with a single strength group, but at the discharge two strength groups were required (Table 3-16). One strength group (designated strong) constituted 92 percent of the individuals and had a low constant mortality rate; the other (designated weak) had a very high mortality rate. Most of the individuals designated weak died within the first three hours of the holding period. The mortality rate at the discharge for the individuals designated strong was significantly lower ($\alpha < 0.05$) than the mortality rate for all individuals at the intake, implying a lack of any delayed effects of entrainment. In the framework of the strength group model, the appearance of a weak group at the discharge probably resulted from either the entrainment-sampling synergism, discussed in Section 3.2.2.1.1, or from entrainment alone.

3.2.3 Discussion and Conclusions

The entrainment mortality of striped bass larvae was estimated based on the results of intake and discharge sampling at the Pittsburg Power Plant. At Units 1-4, the estimated entrainment mortality of striped bass larvae was 38 percent at temperatures less than 30 C and 69 percent at 30-31.9 C. Data were insufficient for estimating mortality at temperatures of 32 C or more, and 100 percent mortality has been assumed. At Units 5 and 6, similar results were obtained: estimated entrainment mortality was 40 percent below 30 C, 57 percent at 30-31.9 C, and 77 percent at 32-33.9 C. Again, data were insufficient for estimating mortality at temperatures of 34 C or more, but mortality was probably nearly 100 percent.

The relationship between striped bass length and entrainment mortality and between recirculation and entrainment mortality was examined. Mortality varied among length groups but no consistent trends were observed. Mortality was greater during periods of recirculation than during periods of no-recirculation; it is assumed that this is because the probability of dying from entrainment was greater for re-entrained (i.e., recirculated) larvae.

The macroinvertebrates examined had lower entrainment mortality at temperatures less than 30 C than did striped bass larvae: 10 percent for N. mercedis and 4 percent for gammaridean amphipods. Mortality of N. mercedis increased sharply at temperatures above 30 C: 72 percent at 30-31.9 C and 100 percent at 32 C or more. The mortality of gammaridean amphipods remained negligible until temperatures

TABLE 3-16 ESTIMATED[a] PARAMETERS OF A STRENGTH GROUP MODEL OF THE MORTALITY OF NEOMYSIS MERCEDIS AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, 1978-1979

Parameter[b]	Intake		Discharge LT 30 C	
	Value	Standard Error	Value	Standard Error
P_1	1.000		0.917	0.0203
P_2	0		0.083	0.0203
λ_1	0.0071	0.0004	0.0038	0.0006
λ_2			0.5270	0.2031

(a) Using Equation 3-18.
(b) P = proportion of organisms in strength-group; λ = instantaneous mortality rate (proportion dying per hour).

exceeded 32 C, it was 59 percent at 32-33.9 C, and it was 79 percent at temperatures of 34 C or more.

Delayed effects of entrainment were investigated by holding initially live organisms for 96 hours. During the first 12 hours of the holding period, organisms collected at the discharge generally died at a faster rate than organisms collected at the intake. Application of a constant mortality rate model suggested that a greater proportion of the organisms collected at the discharge were weakened (i.e., had a higher mortality rate) than at the intake. The difference was attributed either to an entrainment-sampling synergism or to entrainment alone. To be conservative, the latter was assumed to be responsible for the difference, and thus the entrainment mortality estimates cited above were generally computed on the basis of the numbers alive and dead 12 hours after sampling.

The relatively low mortality observed in this study at discharge temperatures less than 30 C is consistent with that observed in entrainment survival studies conducted at other power plants in the United States. For example, estimates of entrainment mortality of striped bass larvae at five other plants with discharge temperatures less than 30 C ranged from 5 to 26 percent and averaged 20 percent (Appendix B). Estimates of entrainment mortality of gammaridean amphipods at five plants ranged from 0 to 27 percent, and averaged 7 percent. For mysid shrimp, estimates of entrainment mortality at three plants ranged from 0 to 8 percent, and averaged 5 percent (Appendix B).

The increased entrainment mortality observed in this study at discharge temperatures in excess of 30 C is also consistent with that observed in other studies, and is predictable from the results of laboratory thermal tolerance studies. EA reported that the mortality of striped bass larvae entrained at four Hudson River power plants increased abruptly at temperatures above about 32 C; the median lethal temperature for 30-minute exposure, 30-minute TL50, predicted from the results of laboratory studies was 32.5 C (Appendix B). Stevens and Finlayson (1978) noted a similar increase in striped bass entrainment mortality at temperatures above 31 C at the Pittsburg and Contra Costa power plants. EA 1978 reported that the mortality of the opossum shrimp Neomysis americana entrained at the Bowline Point Power Plant on the tidal Hudson River was nearly negligible until temperatures exceeded 32 C; the 30-minute TL50 for N. americana predicted from the results of laboratory studies was also 32 C. In the present study, entrainment mortality of N. mercedis increased from 10 percent

at temperatures less than 30 C to 72 percent at temperatures of 30-31.9 C; the 10-minute TL50 for N. mercedis predicted from the results of laboratory studies was between 30 and 31 C.

Travel times from the discharge gate wells to the end of the discharge conduits range from 2 to 2.5 minutes (Table 2-1). In the thermal plume, temperatures decrease to within 2 C (4 F) of ambient within 10 to 34 minutes, depending on tide (Section 2.1.3). Thus, entrained organisms not sampled remained in elevated temperatures for an additional 10-34 minutes beyond the point of sampling. However, sampled organisms were held in the larval table at discharge temperatures for up to 15 minutes during sampling plus an additional 10-30 minutes during the draining and rinsing of the larval tables. Thus, sampled organisms were held at elevated temperatures for longer periods of time than were entrained organisms not sampled. Consequently, entrainment mortality was not underestimated, but rather was probably overestimated.

Entrainment mortality might conceivably increase during periods of condenser chlorination. At the Pittsburg Power Plant the separate condenser halves of each unit are chlorinated at the circulating water pump for 30 minutes. During these periods, total residual chlorine concentrations (TRC) at the condenser inlet are in the range of 0.2-0.5 mg/liter (Section 2.1.2).

Chlorine concentrations in the cooling water flow decrease rapidly because of immediate dilution from the unchlorinated condenser half, followed almost immediately by further dilution from the unchlorinated units, and the high chlorine demand of the water, and no chlorine residual is detectable at the discharge (Section 2.1.2). The total period of exposure to chlorine concentrations is no more than 103 sec to the condenser outlet and 323 sec from condenser outlet to discharge.

The most sensitive marine and freshwater organisms have been estimated to be tolerant of 0.34 and 0.99 mg/liter TRC, respectively, when exposure duration is limited to 5 minutes (Appendix B). The time-concentration dosages encountered at the plant should not exceed threshold lethal levels for the organisms entrained there, and chlorination should have only a minimal effect on entrainment mortality.

Mortality of most fish larvae entrained at Pittsburg Power Plant is expected to be greater than or equal to that of striped bass. Striped bass and members of the family Percichthyidae were among the taxa with the lowest entrainment mortality estimates

reported in Appendix B. Families likely to have higher mortality include Clupeidae (herrings), Engraulidae (anchovies), Atherinidae (silversides), Gobiidae (gobies), and Catostomidae (suckers). Families likely to have entrainment mortality similar to that of striped bass include Cyprinidae (minnows), Percidae (perches), Sciaenidae (drums), and Pleuronectidae (right-eye flounders).

Mortality of most macroinvertebrates is expected to be as low as that of Neomysis mercedis and gammaridean amphipods. Low mortality of a number of macroinvertebrates has been observed at six power plants (Appendix B). In addition, Icanberry and Adams (1974) found low mortality of zooplankton at four coastal PGandE plants, and PGandE (1973) found low mortality of zooplankton at the Pittsburg Power Plant.

3.3 ESTIMATION OF TOTAL ENTRAINMENT CROPPING AT THE PITTSBURG POWER PLANT

Total entrainment cropping of ichthyoplankton and macroinvertebrates at the Pittsburg Power Plant was estimated by multiplying the estimates of total numbers entrained by the estimated proportions not surviving entrainment. For striped bass, Neomysis mercedis, and gammaridean amphipods (Corophium stimpsoni, C. spinicorne), the proportions not surviving entrainment at Units 1-4 and Units 5-6 were based on the results of Section 3.2. At Unit 7, 100 percent entrainment mortality was assumed, because the cooling water system is closed-cycle (Section 2.1). For other taxa, 100 percent entrainment mortality was assumed for all units because (1) data were insufficient for estimating entrainment mortality as a function of discharge temperature, and (2) the assumption would result in conservative estimates of total losses. Thus, for taxa other than striped bass, N. mercedis, and gammaridean amphipods, the estimates of total numbers entrained reported in Section 3.1 (Table 3-4) serve as estimates of total losses.

The remainder of the present section deals with total entrainment losses of striped bass, N. mercedis, and gammaridean amphipods only. The computational methods used in estimating total cropping are presented in Section 3.3.1; the estimates of total losses are presented in Section 3.3.2. The basic method used in calculating total entrainment cropping estimates and their variability is outlined in Table 3-17.

TABLE 3-17 PROCEDURE USED IN CALCULATING TOTAL ENTRAINMENT CROPPING ESTIMATES FOR STRIPED BASS AT THE PITTSBURG POWER PLANT

ESTIMATION OF ENTRAINMENT ABUNDANCE

Step 1. Estimation of the density of each taxon in each sample (either a 3-hour or 24-hour composit sample).

a. The density of each taxon from a sample (collected at the Unit 6 discharge gate well) was calculated by dividing the number in a given taxon actually counted in the sample by the volume of water sampled during the collection (the volume of water sampled was measured with an in-line flow meter). The estimated density of each taxon was expressed as the number of each taxon per cubic meter of water.

b. For macroinvertebrates, which were very numerous, the sample collected from the Unit 6 discharge was subsampled (split) in the laboratory and the mean density of each taxon was calculated by dividing the number in a given taxon actually counted in the subsample by the fraction of the total sample represented in the subsample (e.g., 1/8 or 1/64) and then dividing by volume of water sampled from the discharge (measured with the in-line flow meter).

c. For selected ichthyoplankton (fish) taxa the density of each length class (total length of a subsample of larval and juvenile fish was measured, in millimeters, in the laboratory) was calculated for each sample by multiplying the total number of a taxon actually counted in the sample by the fraction of individuals of the taxon measured in the sample that were of a given length class and then dividing by the volume of water sampled from the discharge.

Step 2. Estimation of the number of each taxon entrained through each cooling water system represented by each sample (either a 3-hour or 24-hour composite sample).

a. The estimated number of each taxon entrained was calculated by multiplying the density of the taxon calculated in Step 1 a and b (number per cubic meter of water) by the total cooling water flow through each cooling system (cubic meters; based on hourly circulating water pump operating records) and the rated capacity (cubic meters per second) of each pump during the same period of time as the sample used in calculating organism densities.

b. For selected ichthyoplankton taxa the number of individuals of each length class entrained was calculated by multiplying the density of individuals in a specific length class calculated in Step 1 c by the total cooling water flow through each cooling system during the same period of time as the sample used in calculating organism densities for each length class.

Step 3. Estimation of the mean number of each taxon entrained per day through each cooling water system during a sampling interval (a two-week interval for fish and a four-week period for macroinvertebrates).

a. The mean number in a given taxon entrained per day during a sampling interval was estimated by adding the estimated numbers entrained represented by each 3-hour sample (Step 2 a) during a sampling day to calculate the estimated total number entrained during the sampling day. The estimated daily totals entrained for each taxon for each sampling day during the sampling interval (two- or four-week period) were added together and then the total was divided by the number of sampling days included in the sampling interval for each cooling system separately.

b. Estimates of the mean number of each selected ichthyoplankton taxa within a specific size class entrained per day during a sampling interval were calculated using the method outlined in Step 3 a, but with the substitution of the length-specific entrainment estimates calculated in Step 2 b.

Step 4. Estimation of the total number of each taxon entrained during the study periods (March 1978 - March 1979 and April-July 1979).

a. The total estimated number of organisms for each taxon entrained was calculated by multiplying the daily mean number entrained during each sampling interval (Step 3 a) by the number of days each cooling system was in operation during the same sampling interval and then adding together all of the sampling interval estimates for the entire study period (e.g., March 1978 - March 1979) for each cooling system separately. The estimate for the power plant was calculated by adding together the estimates for the separate cooling systems.

b. The total estimated number of organisms for each length class from a selected ichthyoplankton taxon entrained was calculated using the method outlined in Step 4 a, but with the substitution of the length-specific entrainment estimates calculated in Step 3 b.

ESTIMATION OF ENTRAINMENT SURVIVAL

Step 1. Estimation of the proportion of organisms of selected taxon not surviving entrainment.

The proportion of organisms of selected taxa not surviving entrainment was calculated by subtracting the estimated proportion of organisms dead in the intake sample from the estimated proportion of organisms dead in corresponding (simultaneously collected) discharge sample and then dividing by the estimated proportion of the organisms alive in the intake sample. The estimated proportion of organisms not surviving entrainment was calculated separately for each pair of intake and discharge samples and the resulting estimates of entrainment mortality were then grouped with respect to discharge water temperatures measured at the time of sample collection and then an average entrainment mortality was calculated separately for each temperature group.

ESTIMATION OF TOTAL ENTRAINMENT CROPPING

Step 1. Estimates of total entrainment cropping were calculated for selected taxa by multiplying the estimated number entrained on each sampling day (Entrainment Abundance, Step 2 a) by the estimated average proportion not surviving entrainment on that sampling day based on temperature specific estimates of mortality (Entrainment Survival, Step 1) and discharge water temperatures on that sampling day (from water temperature monitoring records). The estimated number of each taxon cropped was then substituted for the estimated number entrained per day in Steps 4 a and b, to yield an estimate of the total numbers cropped for each taxon and for length classes of selected species.

Species for which entrainment survival data was not collected were assumed not to have survived entrainment and hence entrainment abundance estimates (Step 4) were used directly as estimates of entrainment cropping.

3.3.1 Methods

Total cropping of striped bass, N. mercedis, and gammaridean amphipods was estimated following the same strategy presented in Section 3.1.1. for estimating total numbers entrained (Equations 3-1 to 3-4). The estimated number entrained on sampling day j in stratum i ($\hat{y}_{ij}$) was multiplied by the estimated average proportion not surviving entrainment on that sampling day ($\hat{p}_e$) to yield the estimated number cropped on that sampling day ($\hat{y}_{ij}'$):

$$\hat{y}_{ij}' = (\hat{y}_{ij})(\hat{p}_e) \tag{3-19}$$

The estimated number cropped ($\hat{y}_{ij}'$) was then substituted for $\hat{y}_{ij}$ in Equation 3-3, and the total number cropped during the sampling year was computed using Equation 3-4.

The average proportion not surviving entrainment ($\hat{p}_e$) on each sampling day was estimated as

$$\hat{p}_e = p_1\hat{p}_{e_1} + p_2\hat{p}_{e_2} + p_3\hat{p}_{e_3} + p_4\hat{p}_{e_4} \tag{3-20}$$

where

p_1, p_2, p_3, p_4 = proportions of time on a given sampling day that discharge temperatures were less than 30 C, 30-31.9 C, 32-33.9 C, and greater than or equal to 34 C, respectively.

$\hat{p}_{e_1}, \hat{p}_{e_2}, \hat{p}_{e_3}, \hat{p}_{e_4}$ = entrainment mortality proportions for discharge temperatures less than 30 C, 30-31.9 C, 32-33.9 C and greater than or equal to 34 C, respectively.

The entrainment mortality proportions used were those derived in Section 3.2. The proportions p_1, p_2, p_3, and p_4 were derived from a continuous record of discharge temperatures measured at the Units 1-4, 5, and 6 discharge gate wells (Section 2.1.3.2).

For example, p_1 was computed as

$$p_1 = \frac{n_1}{N} \tag{3-21}$$

where

n_1 = number of temperatures on a given sampling day (read at 3 hour intervals) less than 30 C

N = total number of temperatures read on a given sampling day.

Thus, p_1 computed in this fashion was an estimate of the proportion of time on a given sampling day that discharge temperatures were less than 30 C. Equation 3-20 simply multiplies the proportion of time on a given sampling day that discharge temperatures were in each of the four possible ranges (e.g., p_1) by the corresponding entrainment mortality proportions (e.g., p_e1), and then sums these products to yield an estimate of the average mortality proportion for that day.

The variance of the total number cropped was estimated in the same manner as was the variance of the total number entrained (Section 3.1.1) (Equations 3-5 through 3-7). Equation 3-5 was modified by substituting for $Var(\hat{y}_{ij})$, the variance term $Var(\hat{y}_{ij}')$:

$$Var(\hat{y}_{ij}') = Var(\hat{y}_{ij})Var(\hat{p}_e) + \hat{y}_{ij}^{\,2}Var(\hat{p}_e) + \hat{p}_e^{\,2}\,Var(\hat{y}_{ij}). \tag{3-22}$$

$Var\,(\hat{p}_e)$ in Equation 3-22 is defined as follows:

$$Var(\hat{p}_e) = p_1^{\,2}Var(\hat{p}_{e_1}) + p_2^{\,2}Var(\hat{p}_{e_2}) + p_3^{\,2}Var(\hat{p}_{e_3}) + p_4^{\,2}Var(\hat{p}_{e_4}) \tag{3-23}$$

where, for simplicity, p_1, p_2, p_3, and p_4 are assumed to have been estimated without error. The variances of the entrainment mortality proportions (e.g., $Var(\hat{p}_{e_1})$) were derived in Section 3.2.

The above computational strategy yielded estimates of total entrainment cropping of striped bass, N. mercedis, and gammaridean amphipods. Comparing those estimates to the estimates of total numbers entrained, the average proportion not surviving entrainment over the sampling year was determined:

$$\overline{P_e} = \frac{E'}{E} \qquad (3\text{-}24)$$

where

$\overline{P_e}$ = average proportion not surviving entrainment

E' = estimated total number lost.

3.3.2 Results

Estimated entrainment cropping of striped bass at Units 1-7, calculated on the basis of actual pump operation, totaled 144 million during the first 12 months of the study and 23 million during the last 4 months (Table 3-18). The average proportion not surviving entrainment amounted to 0.51 during the former period and 0.65 during the latter. Estimates of entrainment cropping as a function of total body length are presented in Table 3-19. The average proportion not surviving entrainment ranged from 0.43 to 0.54 for striped bass entrained during the first 12 months, encompassing the 1978 spawning season, and from 0.53 to 0.77 during the last 4 months, encompassing the 1979 spawning season. The average proportion not surviving generally inceased with increasing length, because the larger fish were entrained later in the summer, when discharge temperatures were higher than in spring and early summer (Table 3-20).

Entrainment cropping of N. mercedis at Units 1-7 totaled 4.0 billion during the first 12 months of the study and 2.6 billion during the last 4 months (Table 3-18). The average proportion not surviving entrainment was 0.43 during the former period and 0.44 during the latter.

Entrainment cropping of gammaridean amphipods at Units 1-7 totaled 335 million during the first 12 months of the study (Table 3-18). The average proportion not surviving entrainment was 0.39.

3.4 ENTRAINMENT SUMMARY

Densities of entrained ichthyoplankton and macroinvertebrates were monitored at the Pittsburg Power Plant Units 1-7 for a period of one year (23 March 1978 - 12 March 1979). Densities of entrained striped bass and Neomysis mercedis were monitored for an additional period of four months (19 March - 31 July 1979). Cooling water was

TABLE 3-18 ESTIMATED ENTRAINMENT CROPPING (millions) OF STRIPED BASS, NEOMYSIS MERCEDIS, AND GAMMARIDEAN AMPHIPODS AT UNITS 1-7 OF THE PITTSBURG POWER PLANT UNDER ACTUAL PUMP OPERATION, 1978-1979, 1979

Taxon	Units 1-4		Units 5 and 6		Unit 7		Units 1-7		
	Number Cropped	Standard Error	Number Cropped	Standard Error	Number Cropped	Standard Error	(a) Number Entrained	Number Cropped	$\bar{p}_e$ (b)
March 1978 - March 1979									
Striped bass	87.5	8.3	47.1	5.5	9.0	1.3	283.3	143.6	0.51
Neomysis mercedis	2,085.9	210.7	1,685.9	170.5	254.6	14.0	9,314.8	4,026.4	0.43
Gammarideans	175.6	90.1	140.2	72.0	19.4	5.4	860.4	335.2	0.39
April 1979 - July 1979									
Striped bass	15.2	2.1	6.9	1.0	0.9	0.1	35.4	23.0	0.65
Neomysis mercedis	1,565.3	387.3	879.5	219.2	160.3	30.2	5,860.2	2,605.1	0.44
Gammarideans	—	—	—	—	—	—	—	—	—

(a) From Table 3-4.
(b) Overall average entrainment mortality proportion (Equation 3-24).

Note: (—) indicates organism not sampled.

TABLE 3-19 ESTIMATED ENTRAINMENT CROPPING (millions) OF STRIPED BASS, BY LENGTH, AT UNITS 1-7 OF THE PITTSBURG POWER PLANT UNDER ACTUAL PUMP OPERATION 1978-1979, 1979

Length (mm)	March 1978 - March 1979			April-July 1979		
	Number Entrained	Number Cropped	$\bar{p}_e$ (a)	Number Entrained	Number Cropped	$\bar{p}_e$
≤5	75.5	32.4	0.43	5.8	3.1	0.53
6	47.0	21.2	0.45	6.5	3.6	0.55
7	17.2	8.1	0.47	2.6	1.5	0.57
8	7.1	3.6	0.50	1.8	1.0	0.57
9	10.2	5.0	0.49	2.5	1.5	0.60
10	16.4	7.9	0.48	2.8	1.9	0.66
11	17.0	8.0	0.47	1.5	1.2	0.77
12	18.9	9.0	0.48	2.0	1.5	0.75
13	14.4	6.8	0.47	1.2	0.9	0.73
14	11.7	6.4	0.54	1.3	1.0	0.73
15	11.2	6.0	0.53	2.6	1.9	0.71
16	10.3	5.4	0.53	0.9	0.9	0.74
17	5.8	3.1	0.54	0.8	0.6	0.72
18	4.6	2.8	0.60	0.4	0.6	0.69
19	2.9	1.7	0.58	0.2	0.1	0.71
≥20	12.5	7.3	0.58	1.3	0.9	0.70

(a) Average proportion not surviving entrainment.

Note: The mortality proportion is largely a function of discharge temperatures; these numbers cropped depend on the discharge temperatures and length frequencies at the time of entrainment.

TABLE 3-20 ESTIMATED ENTRAINMENT CROPPING (millions) OF STRIPED BASS, BY LIFE STAGE AND PERIOD, AT UNITS 1-7 OF THE PITTSBURG POWER PLANT UNDER ACTUAL PUMP OPERATION, 1978-1979

Period	Life Stage		
	Yolk-sac Larva	Larva	Juvenile
1978			
1-14 APR	*	0.18	0
14-28 APR	0.05	0.09	0
28 APR - 12 MAY	1.27	6.36	0.01
12-26 MAY	0.07	27.4	0.01
26 MAY - 9 JUN	0.34	54.1	3.55
9-23 JUN	0.04	24.0	11.08
23 JUN - 7 JUL	0	1.49	2.17
7-21 JUL	0	0.04	0.33
21 JUL - 4 AUG	0	0	0.13
4-25 AUG	0.77	0	0.20
8-22 SEPT	0	0.34	0.03
29 DEC - 12 JAN 79	0	0	0.01
1979			
28 APR - 12 MAY	0.03	0.03	0
12-26 MAY	1.46	1.23	0
26 MAY - 9 JUN	2.57	7.88	0.48
9-23 JUN	0	4.66	2.93
23 JUN - 7 JUL	0.06	0.38	0.27
7 JUL - 4 AUG	0	0	0.10

sampled in the well-mixed discharge of Unit 6. Results of special studies indicated that no through-plant losses of organisms were occurring, making discharge sampling acceptable. Results of other special studies indicated that densities of organisms entrained at Unit 6 were representative of densities entrained at the other units.

The most abundant fish larvae and juveniles entrained were striped bass, smelts, prickly sculpin, Pacific herring, gobies, northern anchovy, and threadfin shad, in that order. An estimated 468 million fish larvae and juveniles were entrained at Units 1-7 during the first year of the study; of these, 283 million were striped bass. An estimated 35 million striped bass were entrained during the last four months of the study, encompassing the 1979 striped bass spawning season. During the first year of the study, an estimated 2.0 million fish eggs were entrained, 55 percent of which were of striped bass.

A variety of invertebrates were entrained, most of which were copepods (e.g., Eurytemora affinis), cladocerans (e.g., Daphnia spp.), and other small zooplankton. The major macroinvertebrates entrained during the first year of the study were Neomysis mercedis (9.3 billion), Corophium stimpsoni (786 million), Palaemon macrodactylus (663 million), and Rhithropanopeus harrisii (larvae) (471 million). During the last 4 months of the study, an estimated 5.9 billion N. mercedis were entrained; during the comparable period in the previous year 4.2 billion N. mercedis were entrained.

The proportions of striped bass larvae, N. mercedis, and gammaridean amphipods not surviving entrainment were estimated from the results of an entrainment survival study. The proportions not surviving entrainment were relatively low when discharge temperatures remained below 30 C. For striped bass, the proportion not surviving was 0.38 at Units 1-4 and 0.40 at Units 5 and 6. For N. mercedis and gammaridean amphipods, the proportions not surviving at Units 5 and 6 (the only units for which data are available) were 0.10 and 0.04, respectively. Mortality of striped bass and N. mercedis increased rapidly as temperatures increased above 30 C; mortality of gammaridean amphipods remained negligible until temperatures exceeded 32 C.

Total entrainment cropping of striped bass, Neomysis mercedis, and gammaridean amphipods were estimated from the estimated numbers entrained and the estimated proportions not surviving entrainment at Unit 1-4 and Units 5 and 6. At Unit 7, 100 percent entrainment mortality was assumed because of that unit's closed-cycle cooling

system. For other taxa, 100 percent mortality was conservatively assumed at all units, and therefore, for those taxa, estimated total cropping was equivalent to estimated total numbers entrained. For the three selected taxa, total cropping was computed by determining the average proportion not surviving entrainment on any given day, based on actual discharge temperatures, and multiplying that proportion by the estimated numbers entrained on that day. Of the estimated 283 million striped bass larvae and juveniles entrained during the first year of the study, an estimated 144 million did not survive. In the last 4 months of the study, an estimated 23 million of the estimated 35 million striped bass entrained did not survive. The average proportions not surviving were 0.51 and 0.65 in the 1978 and 1979 spawning seasons, respectively. Total losses of N. mercedis were estimated at 4 billion during the first year of the study and 2.6 billion during the last 4 months, and the average proportions not surviving were 0.43 and 0.44, respectively. Total cropping of gammaridean amphipods during the first year was estimated at 335 million, and the average proportion not surviving was 0.39. Gammarideans were not examined during the last 4 months of the study.

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Chapter 4

Impingement

4.1 Impingement Abundance

4.2 Impingement Survival

4.3 Estimation of Total Impingement Cropping at the Pittsburg Power Plant

4.4 Impingement Summary

4.5 References

CHAPTER 4: IMPINGEMENT

Impingement and entrainment (Chapter 3) are two principal ways in which cooling water systems can adversely affect aquatic organisms. Impingement occurs when an organism is held against the intake screens used to remove debris from the cooling water. The damage incurred depends on the species, life stage, and size of the organism; the intake water velocities; the duration of impingement; and the techniques of handling impinged organisms and returning them to the water body. Factors contributing to the abundance and survival of impinged organisms have been the subject of numerous reports and reviews (Jensen 1974, 1976, 1978; Grimes 1975; Hanson et al. 1977; Uziel 1978; Oak Ridge National Laboratory and Atomic Industrial Forum, Inc. 1979). In addition, considerable effort has been directed recently toward improving the design of cooling water intake systems to minimize the number of organisms impinged and to maximize the survival of organisms that are impinged.

The study plan for the Pittsburg Power Plant (PGandE 1977) described two studies designed to provide a quantitative basis for estimating the numbers of fish and macroinvertebrates cropped from local populations as a result of impingement. The first, impingement abundance, was designed to provide estimates of the total numbers of organisms impinged on the intake screens of Units 1-4 and of Units 5 and 6. Because Unit 7 withdraws makeup water from behind the Units 1-4 intake screens, the impingement estimates for Units 1-4 will include those for Unit 7. At continuous normal flow, Unit 7 flows add about 5 percent to those of Units 1-4. The second study, impingement survival, was designed to provide a basis for computing species-specific estimates of the proportion of organisms potentially surviving impingement. However, as a result of the screenwash sluiceway design at the Pittsburg Power Plant, which includes a wet well and a pump to transport impinged organisms and debris to the discharge, it has been assumed in the impact assessment that no organisms survive impingement.

The study design, methods, and results of the impingement abundance and survival studies are presented in this chapter. The impingement abundance study is described in Section 4.1, and the supporting data are tabulated in the data summary (Section E.2, Appendix E). The impingement survival study is described in Section 4.2. These data are used in Section 4.3 to estimate the total numbers of selected fish and macroinvertebrates cropped from local populations.

These estimates, along with similar estimates of entrainment cropping in Chapter 3, are used in Chapter 5 to assess the impact of cooling water system operations on the aquatic populations of the source water body. These results also provide site- and species-specific information necessary to evaluate the cooling water intakes in order to demonstrate the best intake technology available for the units of the Pittsburg Power Plant (Chapter 6).

4.1 IMPINGEMENT ABUNDANCE

The impingement abundance study was designed to provide a quantitative basis for estimating the numbers of each species of fish and macroinvertebrates impinged on the Units 1-4 and Units 5 and 6 intake screens. The study was designed to:

1. determine the species composition of the organisms impinged,
2. determine the lengths and weights of impinged organisms,
3. determine the diel and seasonal patterns of impingement,
4. examine the relationship between impingement and cooling water system operational parameters,
5. determine the sex ratio and degree of gonadal maturity for selected species, and
6. determine whether or not any significant impingement occurred on intake bar racks.

In the impingement abundance study, the numbers and sizes of fish and macroinvertebrates impinged on the Units 1-4 and Units 5 and 6 intake screens were determined from weekly sampling. Sample collections were periodically isolated into 3-hour time intervals within a 24-hour sampling period to determine if there was a diel pattern in impingement. The experimental design permitted detection of seasonal and diel patterns in the numbers, sizes, and species composition of impinged organisms and provided a framework for correlating these with varying plant operational parameters.

Impingement sampling extended over a 19-month period to include information on impingement of two cohorts of young-of-the-year striped bass. Sampling conducted between March 1978 and March 1979 is referred to as the 1978 sample period (annual estimates were computed for the period March 1978 - March 1979); sampling conducted between May and November 1979 is referred to as the 1979 sample period.

In addition to the routine impingement abundance collections, a series of experiments were carried out to document the collection and processing efficiency of the impingement sampling techniques. The methods and results of these studies are presented in Section 4.1.1.4.

4.1.1. Methods

4.1.1.1 Abundance Sample Collection

Impingement of fish and macroinvertebrates on vertical traveling intake screens was measured during one 24-hour period each week from 23 March 1978 through 9 April 1979 and from to 14 May to 27 November 1979. Impinged organisms were collected from the screenwash sluiceways of the Units 1-4 and Units 5 and 6 intakes. To facilitate comparisons, the two intakes were sampled simultaneously. During one sampling period in each month from August 1978 to November 1979, data were analyzed as eight discrete 3-hour samples to characterize the diel pattern in impingement abundance; in the rest of the study period the data were combined into 24-hour composites.

Before each sampling period all screens were rotated and washed to remove previously impinged organisms and debris. The screens then remained stationary for 2-3/4 hours and were then rotated for 15 minutes while impinged organisms were washed into a collection device. This cycle of stationary period and rotation was repeated through the 24-hour sampling period. Samples were collected from the screenwash sluiceways in containers lined with 1/4-in. (6.4-mm) steel or nylon mesh.

The fish and macroinvertebrates collected were sorted by hand from the detritus. Fish were identified to species, counted, and measured at the site. Fork length was measured for up to 50 individuals of each species in each 3-hour sample. With the exception of jellyfish, which were counted in the field, macroinvertebrates were preserved in 10 percent formalin and were counted, identified, and weighed later in the laboratory. A composite weight to the nearest 0.1 gram was recorded for each species.

Bar racks were inspected at intervals through the study to determine if impingement was occurring.

Gonads of fish of the most abundantly impinged species were periodically examined to assess spawning condition. Each fish examined was dissected and the condition of the gonads classified as undeveloped (undifferentiated tissue of testes or ovaries), developing (ovary integrity maintained, eggs compact and adhesive), mature (eggs and sperm uncompact and not adhesive), or spent (concave abdominal region and urogenital area swollen and red).

4.1.1.2 Subsampling Methods

Because of the considerable time and effort required to separate fish and macroinvertebrates from impingement samples when very large volumes of detritus were collected, impingement subsampling methods were employed. Two methods were used: volume subsampling and time subsampling.

Subsampling by volume consisted of collecting all material diverted by the traveling screens for a given 3-hour time block, and assigning the unsorted sample to as many five-gallon buckets as necessary. If any half-buckets were left over, the sample was distributed until there was a whole number of buckets collected. Each bucket was then assigned a number from 1 to X. Out of the total number of buckets collected, four were randomly selected for processing and the others discarded. Each bucket selected was then processed individually and the impinged organisms collected were segregated and recorded. This procedure was repeated for seven more 3-hour blocks until a 24-hour survey was completed.

In times of high debris loading, samples were collected at 1-hour instead of 3-hour intervals. Subsampling by time consisted of the random selection of two 1-hour collection periods in each of four 6-hour time blocks making up a survey. Each collection was then processed individually and assigned to the time block from which it was collected.

Sample processing was done as described in Section 4.1.1.1.

4.1.1.3 Impingement Estimation

The annual abundance estimate, I, is an estimate of the total numbers of organisms impinged on the intake screens during the period of study. The general equation for this estimate follows the two-stage sampling design. In the first stage, which was used

if time or volume subsampling was done, the observed organism subsample counts within a given sampling period were expanded to cover both the sampled and the unsampled time or volume. Sampling period estimates were then summed to derive a sample-day abundance estimate. In the second stage, the sample-day estimates in a given 2-week sampling period were extrapolated to cover both the sampled days and the unsampled days in the period. These sampling-period, or stratum, estimates were then summed to calculate the annual estimate.

The estimated variance of the annual estimate, Var(I), was calculated similarly. Subsampling variability was estimated and summed across a given sample day, and this is combined with the estimated variability associated with the day-to-day sampling. Variance estimates for the two-week period were then summed to derive the estimated variance of the annual estimate.

The general equation for the annual abundance estimate is

$$I = \sum_{i=1}^{L} N_i \bar{y}_i \qquad (4\text{-}1)$$

where

i = the ordinal number of a given two-week sampling period

L = the number of two-week intervals during the period of study

N_i = the total number of days the plant was in operation during interval i.

The estimated mean daily number of impinged organisms during interval i ($\bar{y}_i$) is calculated as

$$\bar{y}_i = \frac{1}{n_i} \sum_{j=1}^{n_i} y_{ij} \qquad (4\text{-}2)$$

where

j = the ordinal number of a given sampled day in interval i

n_i = the number of sampled days.

The estimated total impingement abundance on sampled day j of interval i, y_{ij}, is calculated as

$$y_{ij} = \sum_{k=1}^{T_{ij}} M_{ijk}\bar{y}_{ijk} \tag{4-3}$$

where

k	=	the ordinal number of a given time block on sampled day j
T_{ij}	=	the number of time blocks
M_{ijk}	=	the duration (minutes) of time block k.

The remaining term, the estimated impingement rate per minute during time block k, $\bar{y}_{ijk}$, is calculated as

$$\bar{y}_{ijk} = \frac{\sum_{l=1}^{t_{ijk}} y_{ijkl}}{\sum_{l=1}^{t_{ijk}} m_{ijkl}} \tag{4-4}$$

where

l	=	the ordinal number of a given time subsample in time block k
t_{ijk}	=	the number of time subsamples taken
m_{ijkl}	=	the duration (minutes) of time subsample l.

The estimated impingement abundance during time subsample l, y_{ijkl}, is calculated as

$$y_{ijkl} = B_{ijkl}\,\bar{y}_{ijkl} \tag{4-5}$$

where

B_{ijkl}	=	the number of 5-gallon buckets filled during time subsample l.

The estimated mean abundance per bucket, $\bar{y}_{ijkl}$, is calculated as

$$\bar{y}_{ijkl} = \frac{1}{b_{ijkl}} \sum_{m=1}^{b_{ijkl}} y_{ijklm} \qquad (4\text{-}6)$$

where

m	=	the ordinal number of a 5-gallon bucket subsampled in time subsample l
b_{ijkl}	=	the number of bucket subsamples
y_{ijklm}	=	the observed abundance in subsample bucket m.

The estimated variance of the abundance estimate, Var (I), is calculated as

$$Var(I) = \sum_{i=1}^{L} \frac{N_i}{n_i} (N_i - n_i)\, s_{1i}^{\,2} + \sum_{j=1}^{n_{ij}} Var(y_{ij}) \qquad (4\text{-}7)$$

where

$$s_{1i}^{\,2} = \frac{1}{(n_i - 1)} \sum_{j=1}^{n_i} (y_{ij} - \bar{y}_i)^2. \qquad (4\text{-}8)$$

$Var(y_{ij})$, the estimated variance of the abundance estimate for sample day j in interval i, is calculated as

$$Var(y_{ij}) = \sum_{k=1}^{T_{ij}} \frac{M_{ijk}}{\sum_{l=1}^{t_{ijk}} m_{ijkl}} \left[t_{ijk} \left(\frac{M_{ijk}}{\sum_{l=1}^{t_{ijk}} m_{ijkl}} - 1 \right) s_{2ijk}^{\,2} + \sum_{l=1}^{t_{ijk}} Var(y_{ijkl}) \right] \qquad (4\text{-}9)$$

where

$$s_{2ijk}^{\,2} = \frac{1}{(t_{ijk} - 1)} \sum_{l=1}^{t_{ijk}} m_{ijkl}^{\,2} \left(\frac{y_{ijkl}}{m_{ijkl}} - \bar{y}_{ijk} \right). \qquad (4\text{-}10)$$

The estimated variance of the abundance estimate for time subsample l, $Var(y_{ijkl})$, is calculated as

$$Var(y_{ijkl}) = \frac{B_{ijkl}(B_{ijkl} - b_{ijkl})}{b_{ijkl}} s_{3ijkl}^{\ 2} \tag{4-11}$$

where

$$s_{3ijkl}^{\ 2} = \frac{1}{(b_{ijkl} - 1)} \sum_{m=1}^{b_{ijkl}} (y_{ijklm} - \bar{y}_{ijkl})^2 . \tag{4-12}$$

The 95 percent confidence interval for the annual impingement abundance estimate can be calculated as $I \pm 1.96$ S.E.(I), where S.E., the standard error, is the square root of the variance. The assumption of normality that forms the basis for the 95 percent confidence interval is valid for species with relatively high impingement rates, but is not valid for species that are rarely impinged. A confidence interval extending below zero indicates that the assumption of normality may not be appropriate.

To estimate impingement under design (continuous normal) cooling system operation, a combined ratio estimate, I_R, was calculated on the assumption that an increase in the volume of cooling water pumped would lead to a proportional increase in the number of organisms impinged:

$$I_R = \frac{V'}{V} I \tag{4-13}$$

where

I = the previously described abundance estimate

V' = the total design plant pumping volume over the period of estimation.

V is an estimate, calculated as follows, of the total volume pumped over the year's total of sampling intervals:

$$V' = \sum_{i=1}^{L} N_i \bar{v}_i . \tag{4-14}$$

The term v_i, the average sample day pumping volume in interval i, is calculated as

$$\bar{v}_i = \frac{1}{n_i} \sum_{j=1}^{n_i} V_{ij} \tag{4-15}$$

where

V_{ij} = the plant pumping volume on sampled day j.

The variance of this combined ratio estimate is calculated as

$$Var(I_R) = Var(I) + \left[\frac{I}{V_i} \sum_{i=1}^{L} \frac{N_i(N_i - n_i)}{(n_i)} \frac{I}{V}(s_{Vi}{}^2 - 2C_i)\right]. \quad (4\text{-}16)$$

This incorporates the variance term $sV_i{}^2$:

$$s_{Vi}{}^2 = \frac{1}{N_i - 1} \sum_{z=1}^{N_i} (V_{iz} - \overline{V}_i)^2 \quad (4\text{-}17)$$

where

z = the ordinal number of any given day (not only a sampled day) in interval i

V_{iz} = the plant pumping volume on day z.

$\overline{V}_i$, the average daily pumping volume in interval i, is calculated as follows:

$$\overline{V}_i = \frac{1}{N_i} \sum_{z=1}^{N_i} V_{iz} \cdot \quad (4\text{-}18)$$

The term C_i, the covariance between plant volume and abundance in the estimate of variance, is calculated as

$$C_i = \frac{1}{(n_i - 1)} \sum_{j=1}^{n_i} (y_{ij} - \overline{y}_i)(v_{ij} - \overline{v}_i). \quad (4\text{-}19)$$

The standard error of the impingement combined ratio estimate is the square root of its variance, and the 95 percent confidence interval is $I_R \pm 1.96$ S.E.. (I_R). The assumption of normality that forms the basis for the 95 percent confidence interval is valid for species with relatively large impingement rates, but not for species that are rarely impinged. A confidence interval extending below zero indicates that the assumption of normality may not be appropriate.

4.1.1.4 Collection and Processing Efficiency

All sample collection and processing were subject to a strict quality assurance program. To determine the efficiency of impingement abundance sample collection techniques, various experiments were conducted. In the first test, marked dead fish and macroinvertebrates were released between the bar racks and the traveling screens of the Units 1-4 and Units 5 and 6 intake forebays. The released organisms were impinged and subsequently washed into the screenwash sluiceways and into the collection nets. Ninety-seven percent of 259 released organisms were recovered from Units 1-4 and 93 percent of 263 released organisms were recovered from the Units 5 and 6 intake. In the second test, marked dead fish and macroinvertebrates were released directly into the screenwash sluiceways during screen rotation and cleaning. There was 98 percent recovery of 351 organisms released at Units 1-4 and 97 percent recovery of 350 organisms released at Units 5 and 6. It was concluded from these studies that the small bias associated with the collection techniques was acceptable for the purpose of estimating the total numbers of fish and macroinvertebrates impinged.

Sample processing was checked to determine the number of fish and macroinvertebrates not separated from the detrital material collected in the screenwash. After routine sorting of the impingement sample, a second team resorted the detrital material. Eight hundred thirty impinged organisms were removed in the first sort and 39 in the second, representing a sorting efficiency of 96 percent.

The efficiency of taxonomic identification for fish and macroinvertebrates was routinely monitored as part of the quality assurance program. No impingement samples were found to be deficient.

4.1.2 General Characteristics of Impingement

Overall impingement estimates for the most abundantly impinged species are presented in Table 4-1 for the period March 1978 - March 1979 and in Table 4-2 for May-November 1979, calculated on the basis of actual cooling system operation. For clarity of exposition, the impingement estimates in the text have been rounded to three significant figures and the percentages to two significant figures. The estimated annual impingement of fish (Table 4-1) is 161,000 individuals at Units 1-4 and 220,000 individuals at Units 5 and 6 for the period March 1978 - March 1979. The impingement

TABLE 4-1 ESTIMATED NUMBERS AND WEIGHTS OF SELECTED FISHES AND MACROINVERTEBRATES IMPINGED AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Taxon		Units 1-4			Units 5 and 6			Total			
Common Name	Scientific Name	Number	Standard Error	Weight (kg)	Number	Standard Error	Weight (kg)	Number	Percent	Weight (kg)	Percent
Fish											
Striped bass	Morone saxatilis	48,057	9,687	334.6	63,242	17,047	403.7	111,299	29.2	738.3	33.7
Threadfin shad	Dorosoma petenense	42,400	9,200	242.8	46,465	11,124	249.4	88,865	23.3	492.2	22.5
Longfin smelt	Spirinchus thaleichthys	32,740	5,280	104.2	71,805	23,258	232.4	104,545	27.4	336.6	15.4
Yellowfin goby	Acanthogobius flavimanus	13,911	907	118.9	17,608	1,660	136.0	31,519	8.3	254.9	11.6
Starry flounder	Platichthys stellatus	5,684	1,032	14.1	6,172	907	7.7	11,856	3.1	21.8	1.0
Splittail	Pogonichthys macrolepidotus	5,573	2,330	42.3	1,025	155	15.9	6,598	1.7	58.2	2.7
Other fish		12,786	1,238	163.8	14,407	3,547	125.3	27,193	7.1	289.1	13.2
Total fish		161,151	18,663	1,020.7	220,364	53,558	1,170.4	381,515		2,191.1	
Macroinvertebrates											
Oriental shrimp	Palaemon macrodactylus	996,331	41,753	733.7	1,138,868	102,086	850.0	2,135,199	69.1	1,583.7	61.5
Bay shrimp	Crangon franciscorum	371,880	72,578	410.9	430,407	78,348	468.6	802,287	26.0	879.5	34.1
Pebble crab	Rhithropanopeus harrisii	49,211	6,162	44.0	76,872	31,398	69.4	126,083	4.1	113.4	4.4
Other macroinvertebrates		503	142	0.2	32	18	*	535	*	0.2	*
Jellyfish	Cnidaria	18,189	4,500		7614	2,000		25,803	1.0		
Total macroinvertebrates		1,436,115	89,775	1,188.8	1,653,793	134,918	1,388.0	3,089,908		2,576.8	

Note: There may be slight discrepancies in totals that are due to rounding.
(*) indicates a value less than 0.1.

TABLE 4-2 ESTIMATED NUMBERS AND WEIGHTS OF SELECTED FISHES AND MACROINVERTEBRATES IMPINGED AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MAY-NOVEMBER 1979

Taxon		Units 1-4			Units 5 and 6			Total			
Common Name	Scientific Name	Number	Standard Error	Weight (kg)	Number	Standard Error	Weight (kg)	Number	Percent	Weight (kg)	Percent
Fish											
Striped bass	Morone saxatilis	16,823	3,346	326.4	5,687	221	112.0	22,510	35.2	438.4	49.5
Threadfin shad	Dorosoma petenense	5,954	630	34.3	241	156	1.3	6,195	9.7	35.6	4.0
Starry flounder	Platichthys stellatus	5,514	870	7.4	4,098	418	6.4	9,612	15.0	13.8	1.5
Longfin smelt	Spirinchus thaleichthys	5,232	1,503	29.4	1,913	347	8.1	7,145	11.2	37.5	4.2
Splittail	Pogonichthys macrolepidotus	3,835	777	98.5	458	63	29.6	4,293	6.7	128.1	14.5
Yellowfin goby	Acanthogobius flavimanus	3,807	636	32.1	3,417	407	34.1	7,224	11.3	66.2	7.5
Other fish		5,160	709	118.9	1,887	128	48.2	7,074	11.1	167.1	18.9
Total fish		46,325	4,281	646.9	17,701	842	239.7	64,026		886.6	
Macroinvertebrates											
Bay shrimp	Crangon franciscorum	642,950	154,221	612.3	420,343	75,224	367.7	1,063,293	46.9	980.0	51.8
Oriental shrimp	Palaemon macrodactylus	492,603	37,685	399.6	506,217	33,525	369.1	998,802	44.0	768.7	40.7
Pebble crab	Rhithropanopeus harrisii	111,360	10,567	78.0	95,578	4,740	64.3	206,938	9.1	142.3	7.5
Total macroinvertebrates		1,246,913	156,692	1,089.3	1,022,138	90,770	801.1	2,269,051		1,890.4	

Note: There may be slight discrepancies in totals that are due to rounding.

estimate for May-November 1979 (Table 4-2) is 46,000 individuals at Units 1-4 and 17,700 individuals at Units 5 and 6.

The higher relative impingement at Units 1-4 in 1979 compared to that at Units 5 and 6 (46,000 vs. 17,700) and to the observed pattern in 1978, when impingement was greatest at Units 5 and 6, was primarily the result of prolonged scheduled and unscheduled maintenance outages of Units 5 and 6 in 1979 (Table 2-4). The impingement estimates, based on actual cooling system operation (Tables 4-1 and 4-2) reflect the differences in operation and maintenance outages of the units between 1978 and 1979.

The six most abundant species constituted 92 percent of the fish collected at both intakes over both study periods. These were striped bass (30 percent), longfin smelt (25 percent), threadfin shad (21 percent), yellowfin goby (9 percent), starry flounder (5 percent), and splittail (2 percent). The numbers and weights of impinged fish collected in 1978 are shown in Table 4-3.

In the 1978 study year, species composition at the two intakes was generally similar: 44 species of fish were collected at the Units 1-4 intake and 41 at Units 5 and 6. Six species found at Units 1-4 were not collected at Units 5 and 6, and three species at Units 5 and 6 were not collected at Units 1-4.

The estimated annual impingement of macroinvertebrates was 1.44 million at Units 1-4 and 1.65 million at Units 5 and 6 for the 1978 study year (Table 4-1). *Palaemon macrodactylus*, *Crangon franciscorum*, and *Rhithropanopeus harrisii* were the most commonly impinged macroinvertebrates in the 1978 study year, constituting 69, 26, and 4 percent of the total. Jellyfish (Cnidaria) and 4 other species constituted only 1 percent of the total. From May to November 1979, only *P. macrodactylus*, *C. franciscorum*, and *R. harrisii* were processed. The estimated impingement of these for the period May-November was 1.25 million at Units 1-4 and 1.02 million at Units 5 and 6 (Table 4-2).

The estimates of impinged fish and macroinvertebrates in Tables 4-1 and 4-2 were computed on the basis of actual days of cooling system operation at Units 1-4 and Units 5 and 6 during the study period. Estimates were also computed on the basis of maximum or design cooling system operation (Table E.2-1, Appendix E). The estimates based on design cooling system operation are 20-25 percent larger than those based on

TABLE 4-3 NUMBERS, WEIGHTS, AND PERCENTAGES OF SELECTED FISHES AND MACROINVERTEBRATES COLLECTED IN IMPINGEMENT SAMPLES AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Taxon		Units 1-4		Units 5 and 6		Total			
Common Name	Scientific Name	Number	Percent	Number	Percent	Number	Percent	Weight (kg)	Percent
Fish Larvae and Juveniles									
Striped bass	Morone saxatilis	6,468	28.4	7,782	30.8	14,250	29.7	89.0	31.5
Threadfin shad	Dorosoma petenense	6,187	27.3	6,894	27.3	13,081	27.2	63.5	22.5
Longfin smelt	Spirinchus thaleichthys	4,736	20.8	5,634	22.3	10,370	21.6	40.6	14.4
Yellowfin goby	Acanthogobius flavimanus	1,991	8.7	2,471	9.8	4,462	9.3	36.8	13.0
Starry flounder	Platichthys stellatus	733	3.2	860	3.4	1,593	3.3	2.9	1.0
Splittail	Pogonichthys macrolepidotus	752	3.3	143	0.6	895	1.9	7.8	2.8
Other fish		1,892	8.3	1,478	5.8	3,370	7.0	42.3	15.0
Total fish		22,759		25,262		48,021		282.9	
Macroinvertebrates									
Oriental shrimp	Palaemon macrodactylus	142,736	69.6	154,428	67.1	297,164		219.0	
Bay shrimp	Crangon franciscorum	52,347	25.5	63,878	27.8	116,225		127.3	
Pebble crab	Rhithropanopeus harrisii	7,622	3.7	10,849	4.7	18,471		16.8	
Other macroinvertebrates		69	0	4	0	73		0	
Subtotal macroinvertebrates		202,774		229,159		431,993		363.1	
Jellyfish	Cnidaria	7,622		1,012		8,634			
Total macroinvertebrates		210,396		230,171		440,627			

actual operation, because the plant was not operated at full capacity over the whole year.

To facilitate comparisons, the number of organisms impinged at each intake on a sampling day was divided by the cooling water volume and standardized to a number of organisms impinged per cubic hectometer, or million cubic meters, of cooling water (no./hm^3), called the impingement rate. In the 1978 study year, the mean impingement rate of fish on the Units 1-4 intake screens was 206/hm^3, compared to 345/hm^3 impinged at the Units 5 and 6 intake (Table 4-4). The mean impingement rates of impinged macroinvertebrates were roughly similar at the two intakes: 1,807/hm^3 at Units 1-4 and 2,577/hm^3 at Units 5 and 6. The higher rate of impingement at Units 5 and 6 for both fish and macroinvertebrates does not appear to be a function of intake water velocity: water velocity approaching the Units 1-4 intake averaged 1.0 fps (30 cm/sec), whereas average velocity at Units 5 and 6 was 0.7 fps (21 cm/sec) (Section 2.1.3.2). The mean impingement rates were not computed for all organisms for the 1979 study period, because of its short duration.

The size distribution for the most abundant fish species impinged are presented in Table 4-5. Impinged fish ranged in size from an 8-mm (0.3-in.) striped bass to a 620-mm (24.4-in.) striped bass, but most were between 30 and 150 mm long.

The impingement of fish was highest during the period October-February, with the highest peaks from late November to January (Figure 4-1). The highest day's collection, 4,042 fish (all units combined), occurred on 2 January 1979; threadfin shad accounted for 65 percent of the sample and striped bass for 18 percent. The lowest day's collection (all units combined) was on 18 April 1978 (eight individuals).

Only dead fish were observed or collected during periodic inspection of the bar racks. Identifiable fish collected included threadfin shad, sturgeon, and striped bass, but all were bloated or decomposed at the time of collection, and it is likely that they were dead fish that had been washed up on the bar racks. Fish which were dead (bloated or decomposed) collected in impingement samples were not included in the impingement abundance estimates.

Impingement of macroinvertebrates peaked in November in the 1978 study year and in August in 1979 (Figure 4-2). The highest sample, 31,056 (all units combined), occurred on 6 November 1978; Crangon franciscorum accounted for 61 percent of the sample.

TABLE 4-4 COMPARISON OF THE NUMBERS AND AVERAGE IMPINGEMENT RATES OF FISH AND MACROINVERTEBRATES IMPINGED AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

	Units 1-4	Units 5 and 6
Total Number		
Fish	161,151	220,364
Macroinvertebrates [a]	1,147,926	1,646,179
Daily Mean Number		
Fish	442	604
Macroinvertebrates	3,885	4,510
Impingement Rate [b]		
Fish	206	345
Macroinvertebrates	1,807	2,577

(a) Excluding Cnidaria.
(b) Average number of organisms per cubic hectometer (hm^3; 10^6 m^3) of cooling water flow.

Note: Average daily cooling water volume for Units 1-4 was 2.15 hm^3; for Units 5 and 6, 1.75 hm^3.

TABLE 4-5 SUMMARY OF THE LENGTHS OF SELECTED FISHES COLLECTED IN IMPINGEMENT SAMPLES AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979 AND MAY-NOVEMBER 1979

Taxon		Number Measured	Mean Length[a] (mm)	Standard Deviation	Size Range	
Common Name	Scientific Name				Minimum (mm)	Maximum (mm)
		1978				
Striped bass	Morone saxatilis	4,899	73.0	28.4	8	490
Longfin smelt	Spirinchus thaleichthys	4,740	72.2	8.4	24	120
Threadfin shad	Dorosoma petenense	3,379	75.0	13.7	35	135
Yellowfin goby	Acanthogobius flavimanus	3,000	103.4	29.1	18	225
Starry flounder	Platichthys stellatus	1,030	46.5	14.7	15	257
Splittail	Pogonichthys macrolepidotus	506	83.7	34.7	30	360
		1979				
Striped bass	Morone saxatilis	2,208	65.5	55.9	18	620
Longfin smelt	Spirinchus thaleichthys	768	75.0	25.8	20	128
Threadfin shad	Dorosoma petenense	370	74.7	11.2	45	137
Yellowfin goby	Acanthogobius flavimanus	886	90.6	36.8	24	225
Starry flounder	Platichthys stellatus	1,270	46.1	12.1	25	211
Splittail	Pogonichthys macrolepidotus	572	86.0	66.2	32	375

(a) Fork length of fish measured to 1 mm.

Note: A complete length summary for all impinged species is given in Appendix E.

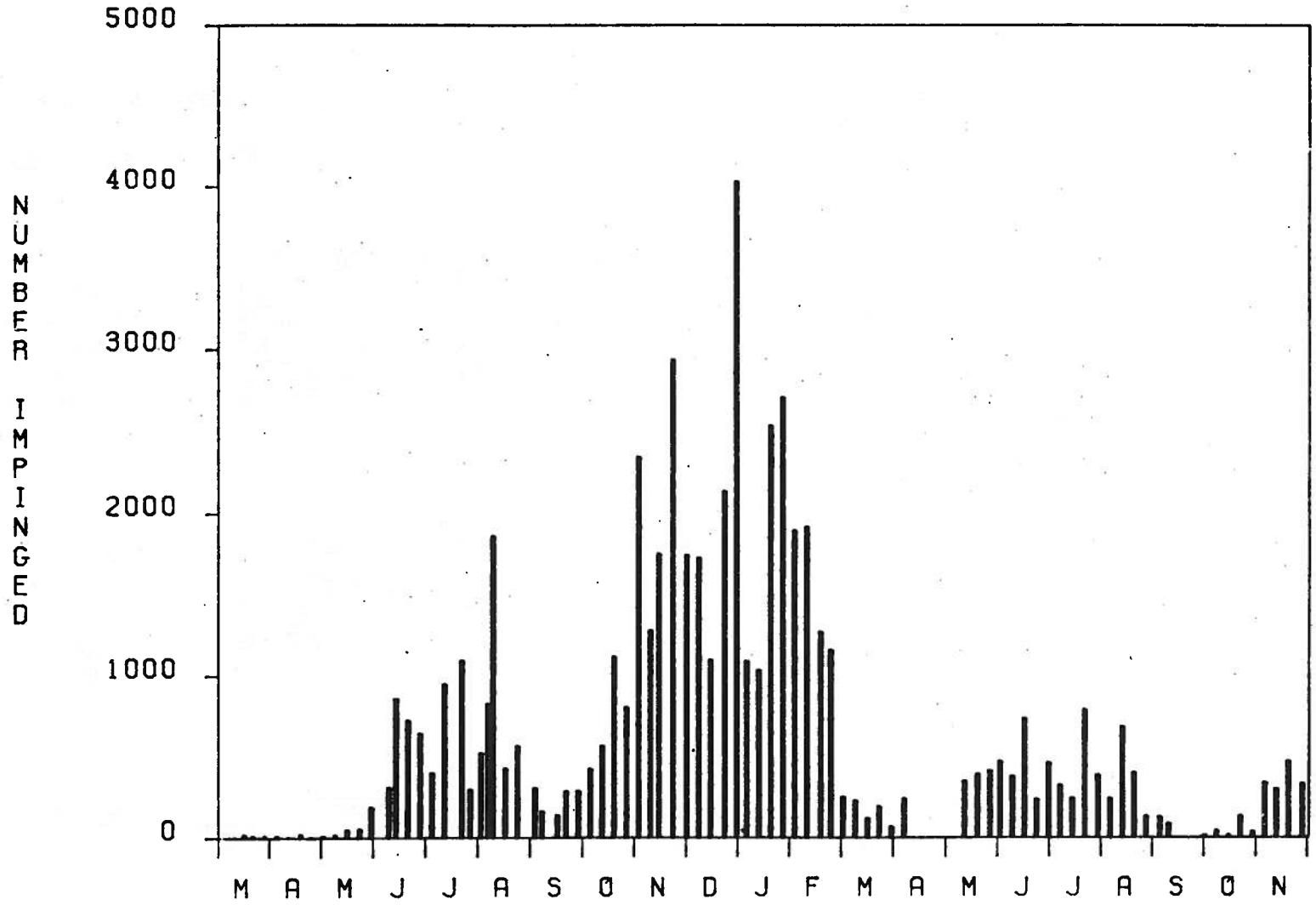


Figure 4-1. Numbers of impinged fish collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

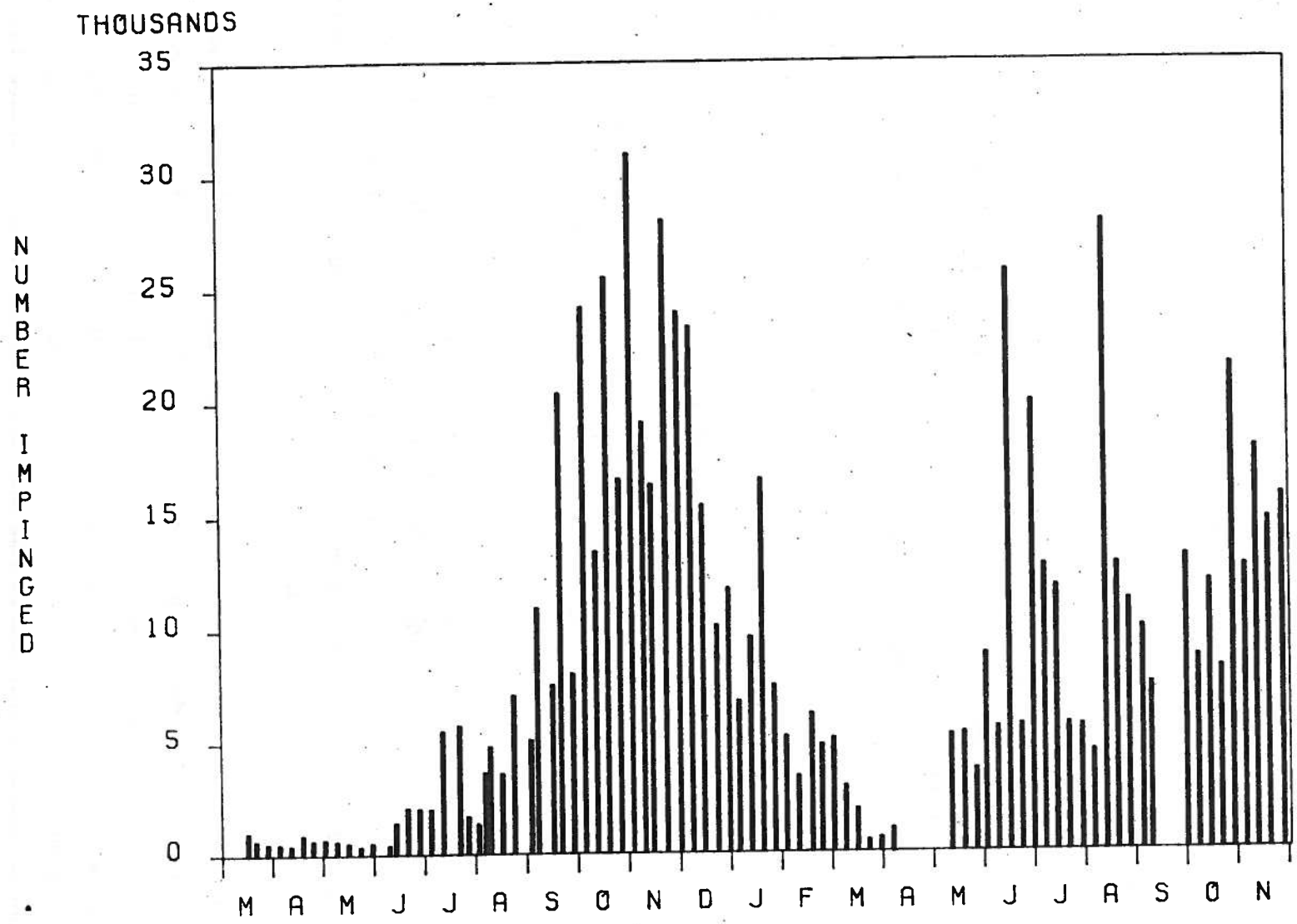


Figure 4-2. Numbers of impinged macroinvertebrates collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

The lowest, 375 (all units combined), occurred on 29 May 1978.

During diel sampling, more fish were impinged by night (defined as 1800-0600 hours), 62 percent, than by day (0600-1800 hours), 38 percent. The same was true for macroinvertebrates: 73 percent were impinged by night and 27 percent by day.

4.1.3 Results For Selected Taxa

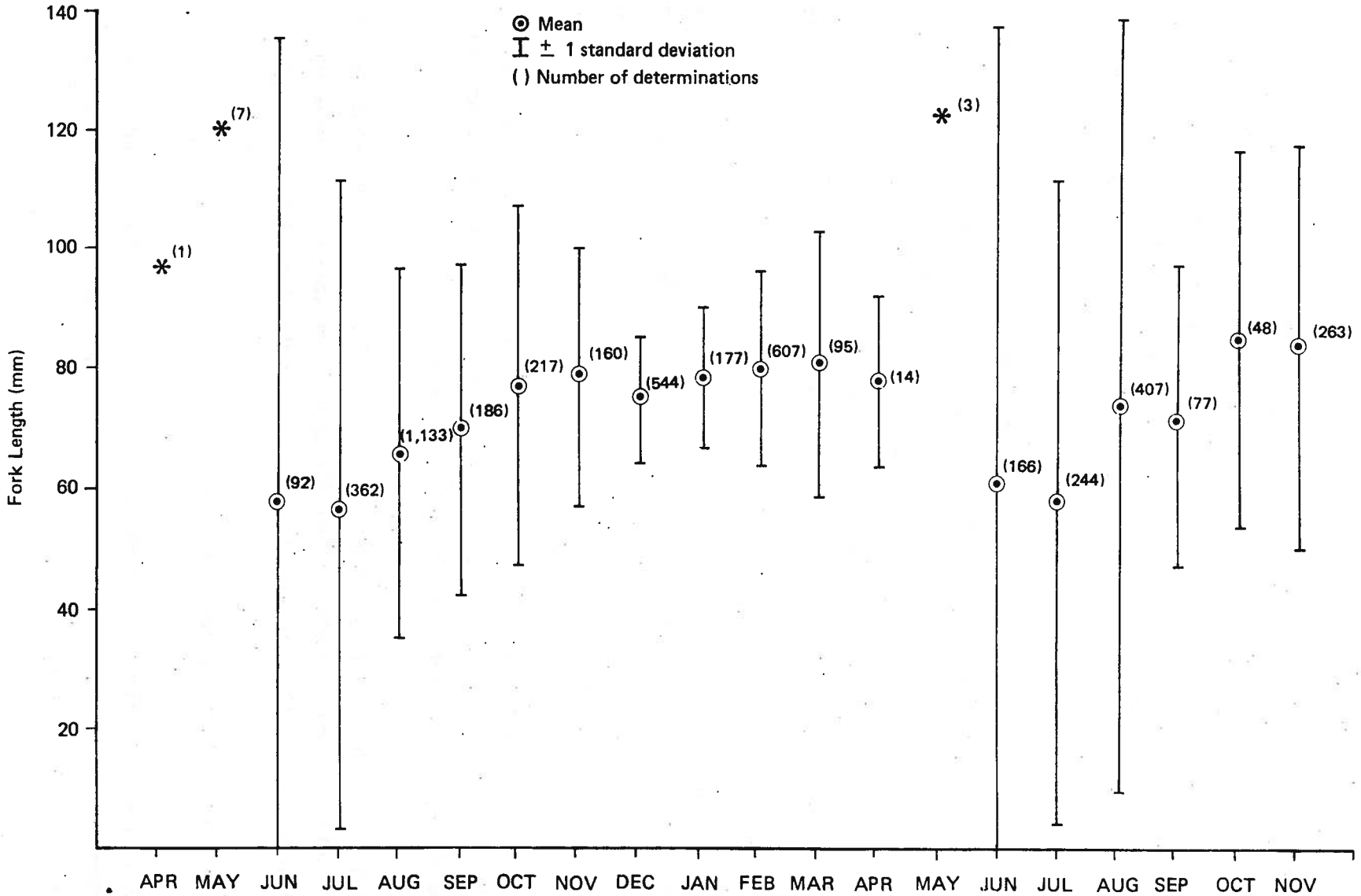
The four most abundantly impinged species of fish were striped bass, threadfin shad, longfin smelt, and yellowfin goby. Two estuarine shrimps, *Palaemon macrodactylus* and *Crangon franciscorum*, and the xanthid crab *Rhithropanopeus harrisii* were the three most abundantly impinged macroinvertebrates. The numbers reported in this section are impingement estimates based on actual cooling system operation during each year of study.

4.1.3.1 Striped Bass (Morone saxatilis)

Striped bass was the most abundantly impinged fish at both intakes. The 11,000 impinged in the 1978 study year constituted 29 percent of all fish impinged, and the 22,500 impinged in 1979 constituted 35 percent. In 1978, striped bass were impinged fairly equally at the two intake locations: 43 percent at Units 1-4 and 57 percent at Units 5 and 6.

The monthly size distributions of striped bass collected on the plant's intake screens is shown in Figure 4-3. The fish ranged in length from 8 to 620 mm (0.3-24.4 in.) (Table 4-5). The mean fork length computed for each month ranged from 57 to 122 mm (2.2-4.8 in.) over the whole study period. This indicates that primarily young-of-the-year fish were impinged.

The seasonal impingement of striped bass at the plant appears to be in part a function of the location of the interface between salt water and fresh water. This area, termed the "entrapment zone," has surface conductivities of 2-10 mmho (1-6 ppt salinity). Its location in the estuary depends on tidal flow and Delta outflow (Arthur and Ball 1979). The area is highly productive, containing high densities of phytoplankton, zooplankton, and young-of-the-year striped bass (Arthur and Ball 1979; Turner and Chadwick 1972). During the 1978 study year, the entrapment zone was adjacent to the Pittsburg Power



*indicates a sample too small for meaningful standard deviation.

Figure 4-3. Mean fork lengths and standard deviations of measured striped bass collected in impingement samples at Units 1-6 of the Pittsburg Power Plant, April 1978–November 1979.

Plant from summer to late fall, and this was the period of maximum impingement rates of striped bass, except for a short period in late January and February 1979 (Figure 4-4).

The apparent preference of young-of-the-year striped bass for the entrapment zone may be related to diet. In the Sacramento - San Joaquin estuary, mysid shrimp (Neomysis mercedis) were found in greatest densities in the entrapment zone, and other studies have indicated that they are one of the most important food items in the diet of young-of-the-year striped bass (Heubach et al. 1963; Stevens 1966; Orsi and Knutson 1979).

In January and February 1979, impingement of striped bass remained high, even though the entrapment zone had shifted downstream as a result of the increased winter Delta outflow. A comparison in January and February 1979 of impingement rates during recirculation of the plant's thermal plume with impingement rates during non-recirculation showed that a statistically significant greater number of striped bass were impinged during plume recirculation (ANOVA; $\alpha = 0.05$). This is consistent with recent research: thermal-preference investigations have indicated that when given a choice in a thermal gradient, young striped bass will consistently prefer temperatures higher than the ambient or acclimation temperature, from 5 to 25 C (Meldrim and Gift 1971; Texas Instruments 1976). In the Hudson River, young striped bass acclimated at temperatures ranging from 11.5 to 26 C were estimated to have a final preferred temperature of 27.8 C (EA unpublished data).

At the Pittsburg Power Plant, striped bass were considered to be numerous in the plume by Chadwick and Stevens (1971), and Gritz (1971). EA (1981) reported that striped bass were more abundant in the plume than outside of it in winter.

In summary, the impingement of striped bass at the Pittsburg Power Plant is (1) primarily limited to young-of-the-year fish, (2) highly correlated with the location of the entrapment zone, and (3) increased by recirculation of the thermal plume in winter.

4.1.3.2 Longfin Smelt (Spirinchus thaleichthys)

Longfin smelt was the second most abundantly impinged fish at Units 1-4 and Units 5 and 6 combined : 27 percent (105,000) of all fish impinged during the 1978 study year and 11 percent (7,150) in the 1979 study period.

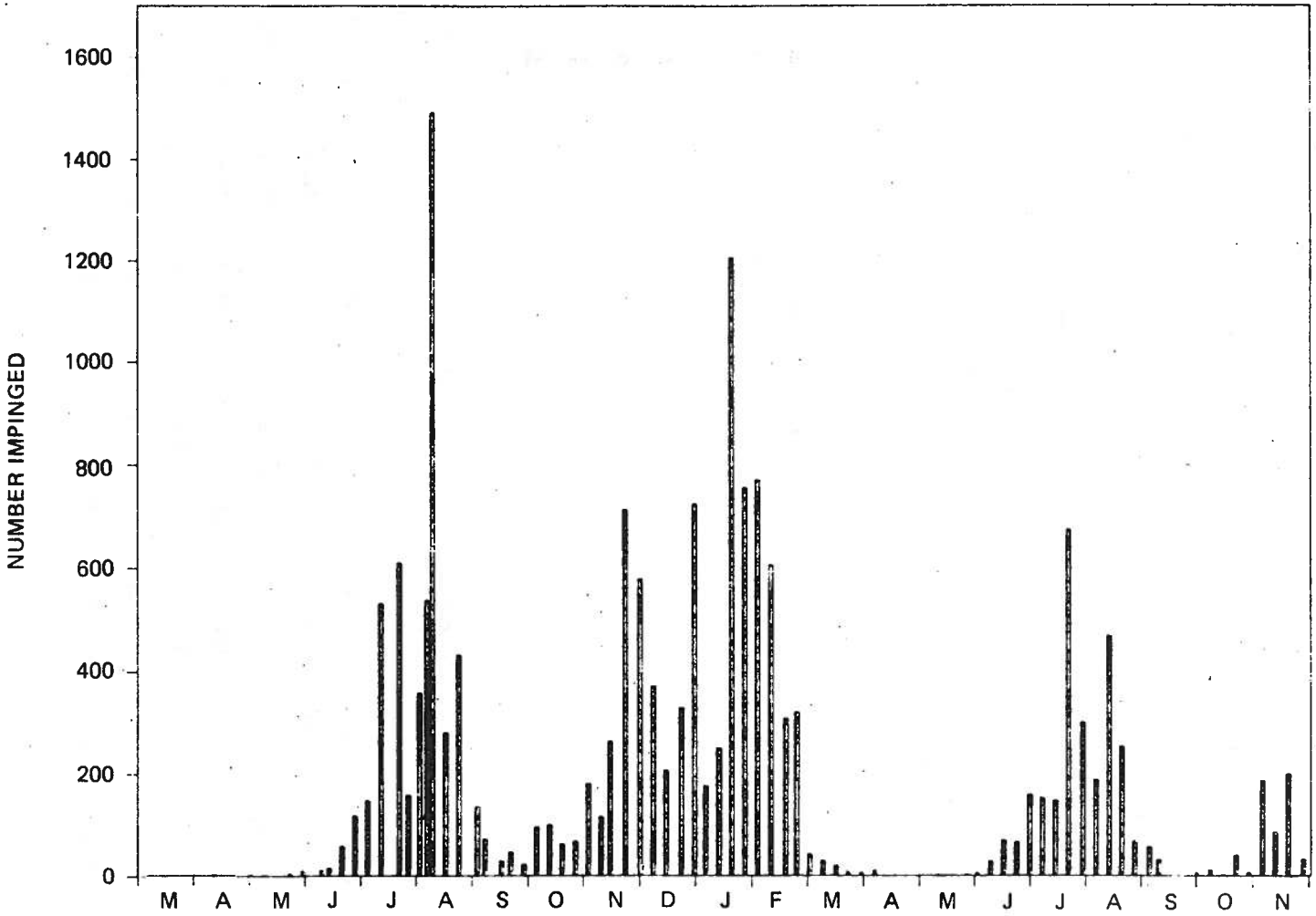


Figure 4-4. Numbers of impinged striped bass collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

The size distribution of longfin smelt collected each month on the Units 1-4 and Units 5 and 6 intake screens is shown in Figure 4-5. The mean fork length computed for each month ranged from 57 to 98 mm (2.2-3.9 in.). Longfin smelt were impinged in greatest abundance during the fall and winter (Figure 4-6), their period of greatest spawning activity (Moyle 1976). Eighty-three percent of the 106 adult longfin smelt examined between November and February had developing or mature gonads or had already spawned (Table 4-6); 60 percent were female, 23 percent were male, and the sex of 17 percent could not be determined.

Impingement of longfin smelt was higher at night (64 percent) than by day, with a peak in abundance from 2400 to 0300 hours.

4.1.3.3 Threadfin Shad (Dorosoma petenense)

Threadfin shad was the third most abundantly impinged fish at the Pittsburg Units 1-4 and Units 5 and 6 intake, representing 23 percent (89,000 individuals) of all fish impinged during the 1978 study year and 10 percent (6,200 individuals) during the 1979 study period.

Threadfin shad collected each month at the plant's intake screens ranged in length from 35 to 137 mm (1.4-5.4 in.) (Table 4-5). The mean fork length computed for each month ranged from 60 to 80 mm (2.4-3.1 in.).

Threadfin shad were impinged predominantly from October to March, with a peak in abundance occurring in late December and January (Figure 4-7). Ambient water temperatures played a significant role in determining the impingement rates of threadfin shad: as ambient water temperatures decreased during the winter, impingement rates increased. Loar et al. (1978) found that at 27 southeastern United States power plants, the highest impingement of threadfin shad occurred at temperatures below 10 C. Additional indications of sensitivity to cool water are reported by Turner (1966), who states that Sacramento - San Joaquin Delta populations of threadfin shad experience heavy die-offs every winter when water temperatures drop below 6-8 C.

More than 77 percent of the threadfin shad impinged were collected at night, with a peak in abundance just after sunset.

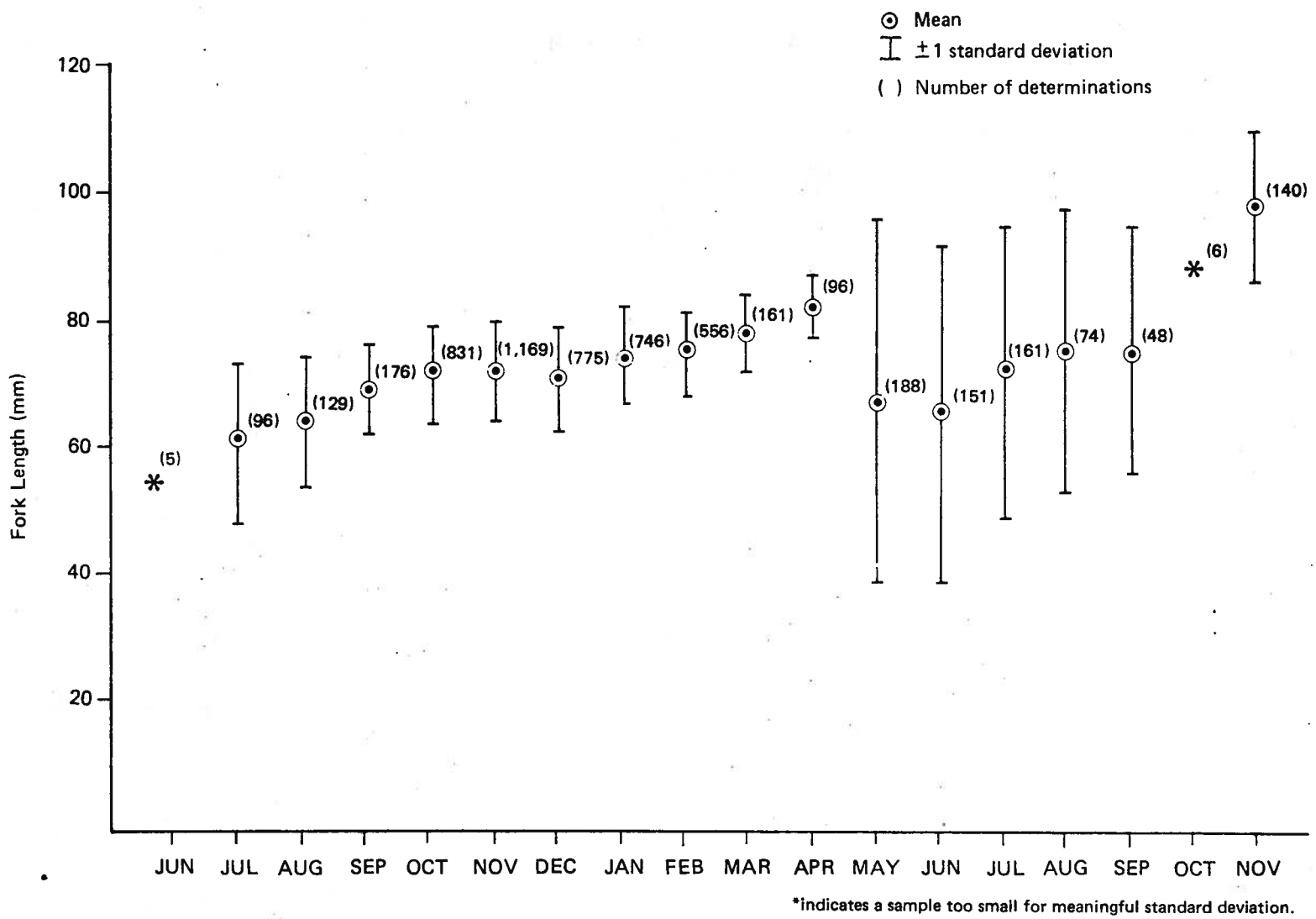


Figure 4-5. Mean fork lengths and standard deviations of longfin smelt collected in impingement samples at Units 1-6 of the Pittsburg Power Plant, June 1978—November 1979.

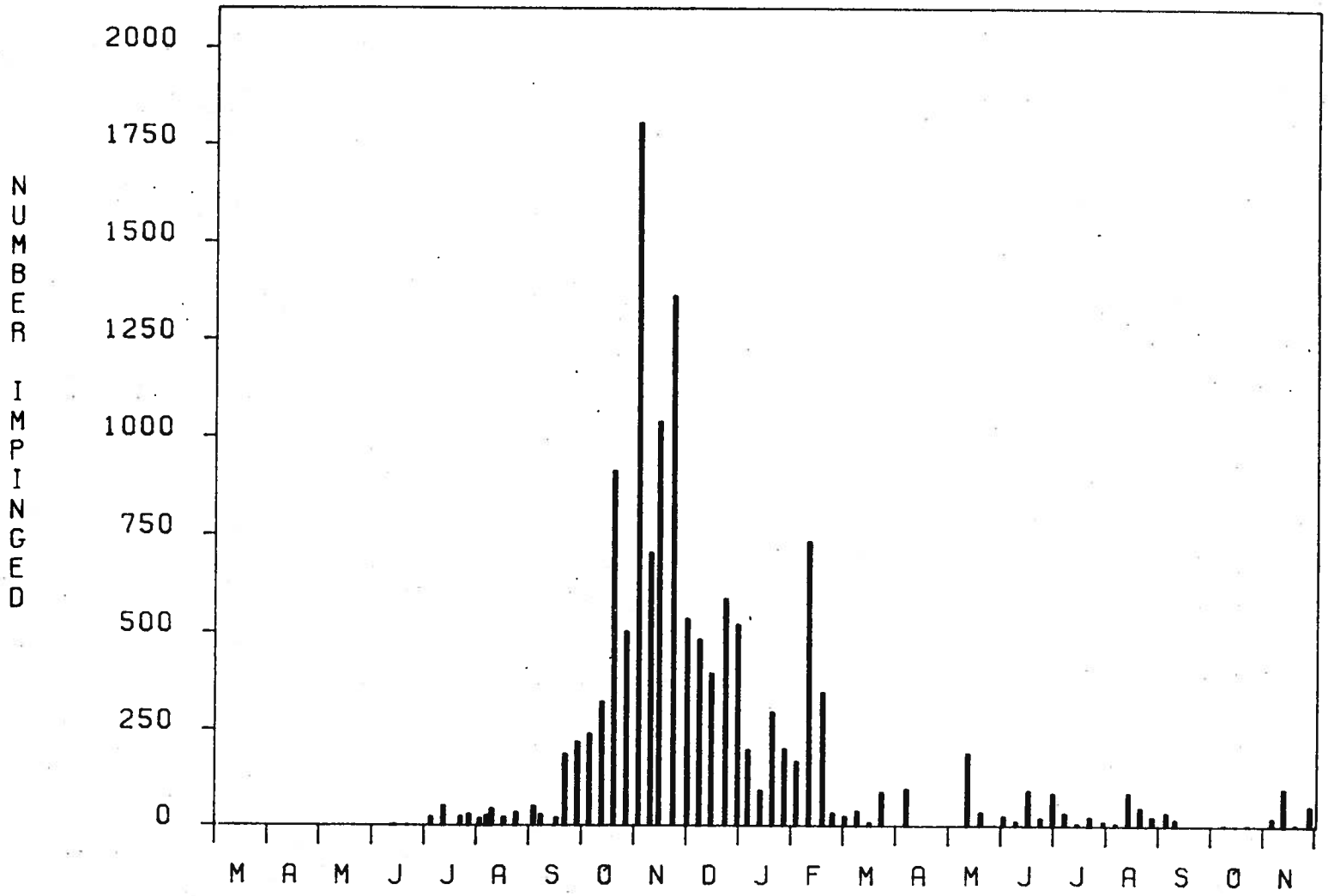


Figure 4-6. Numbers of impinged longfin smelt collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

TABLE 4-6 SEX AND GONADAL CONDITION OF A SUBSAMPLE OF LONGFIN SMELT AND YELLOWFIN GOBY IMPINGED AT THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

	Longfin Smelt (*Spirinchus thaleichthys*)	Yellowfin Goby (*Acanthogobius flavimanus*)
Period of observation	NOV-FEB	DEC-APR
Number examined	106	76
Sex		
Male	24	17
Female	64	51
Undetermined	18	8
Gonadal Condition		
Undetermined	18	8
Developing	62	59
Mature	15	9
Spent	11	0

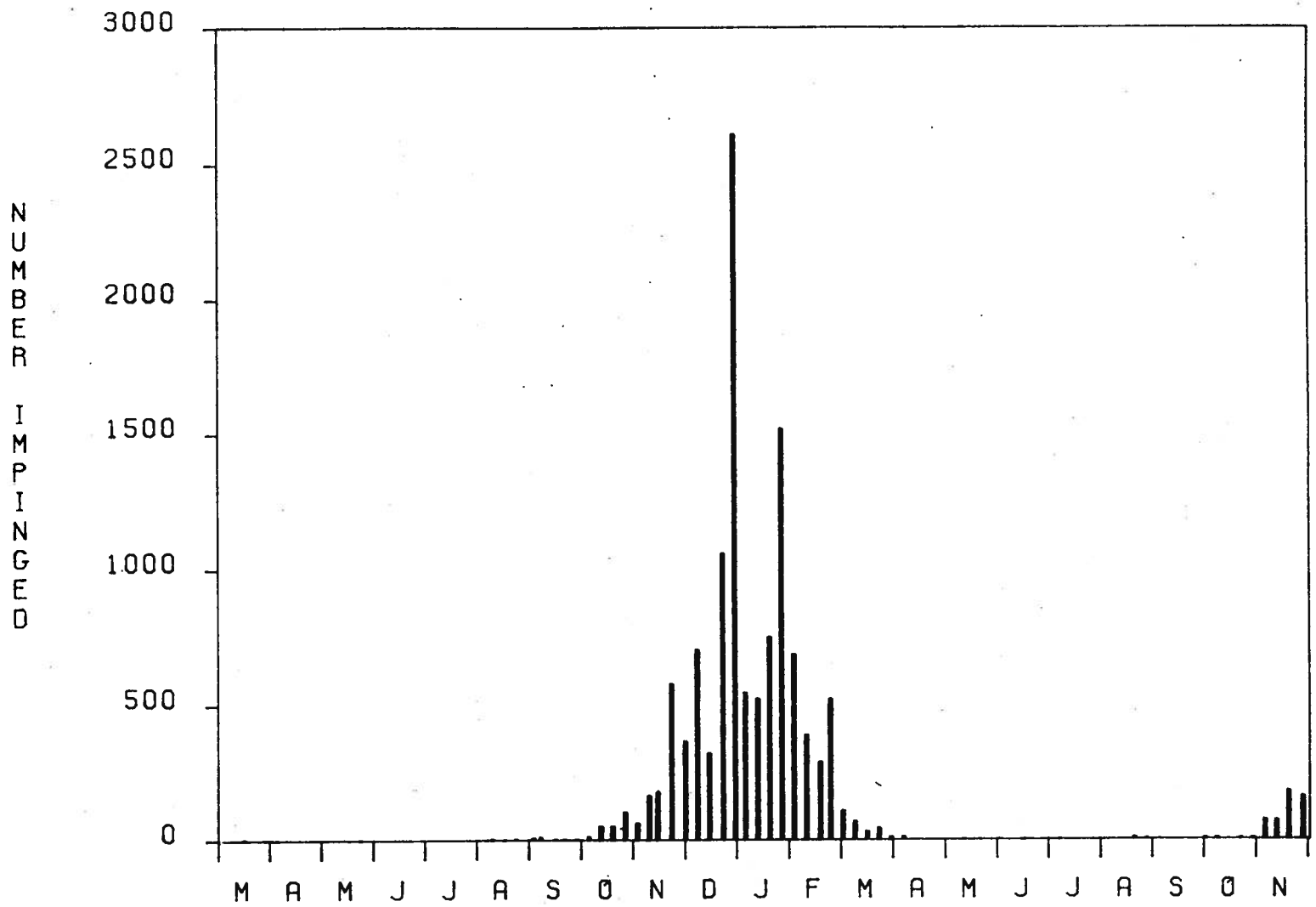


Figure 4-7. Numbers of impinged threadfin shad collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

4.1.3.4 Yellowfin Goby (Acanthogobius flavimanus)

Yellowfin goby was the fourth most abundantly impinged fish at Units 1-4 and Units 5 and 6, representing 8 percent (32,000 individuals) of all fish impinged during the 1978 study year and 11 percent (7,200 individuals) during the 1979 study period.

The size distribution of yellowfin goby is presented in Figure 4-8. The fish ranged in length from 18 to 225 mm (0.7-8.9 in.) (Table 4-5). The mean fork length computed for each month ranged from 67 to 130 mm (2.6-5.1 in.).

Yellowfin goby were impinged throughout the study period, with major peaks from June to August and from November to January (Figure 4-9). Impingement of yellowfin goby was greater at night (almost 74 percent), with a peak around midnight.

Seventy-six adult yellowfin goby were examined between December and April 1979. Sixty-seven percent were female, and 89 percent had either developing or mature gonads (Table 4-6).

4.1.3.5 Shrimps

Palaemon macrodactylus, the oriental shrimp, was the most abundantly impinged macroinvertebrate, representing 69 percent (2.1 million individuals) of all macroinvertebrates impinged during the 1978 study year and 47 percent (1.06 million) in the 1979 study period. They were impinged throughout the year, with a peak in abundance from October to January (Figure 4-10).

Crangon franciscorum, a bay shrimp, was the second most abundantly impinged macroinvertebrate, representing 26 percent (800,000 individuals) of the total macroinvertebrate catch for the 1978 study year and 46.2 percent (145,000) for the 1979 study period.

They were impinged predominantly from June to December, with peaks in abundance in June, August, and November (Figure 4-11). More than 80 percent of the C. franciscorum in diel samples were impinged between 2100 and 0600 hours and less than 3 percent between 0900 and 1800 hours. Previous investigations of this shrimp indicate that it typically remains near the bottom during the day and migrates upward into the

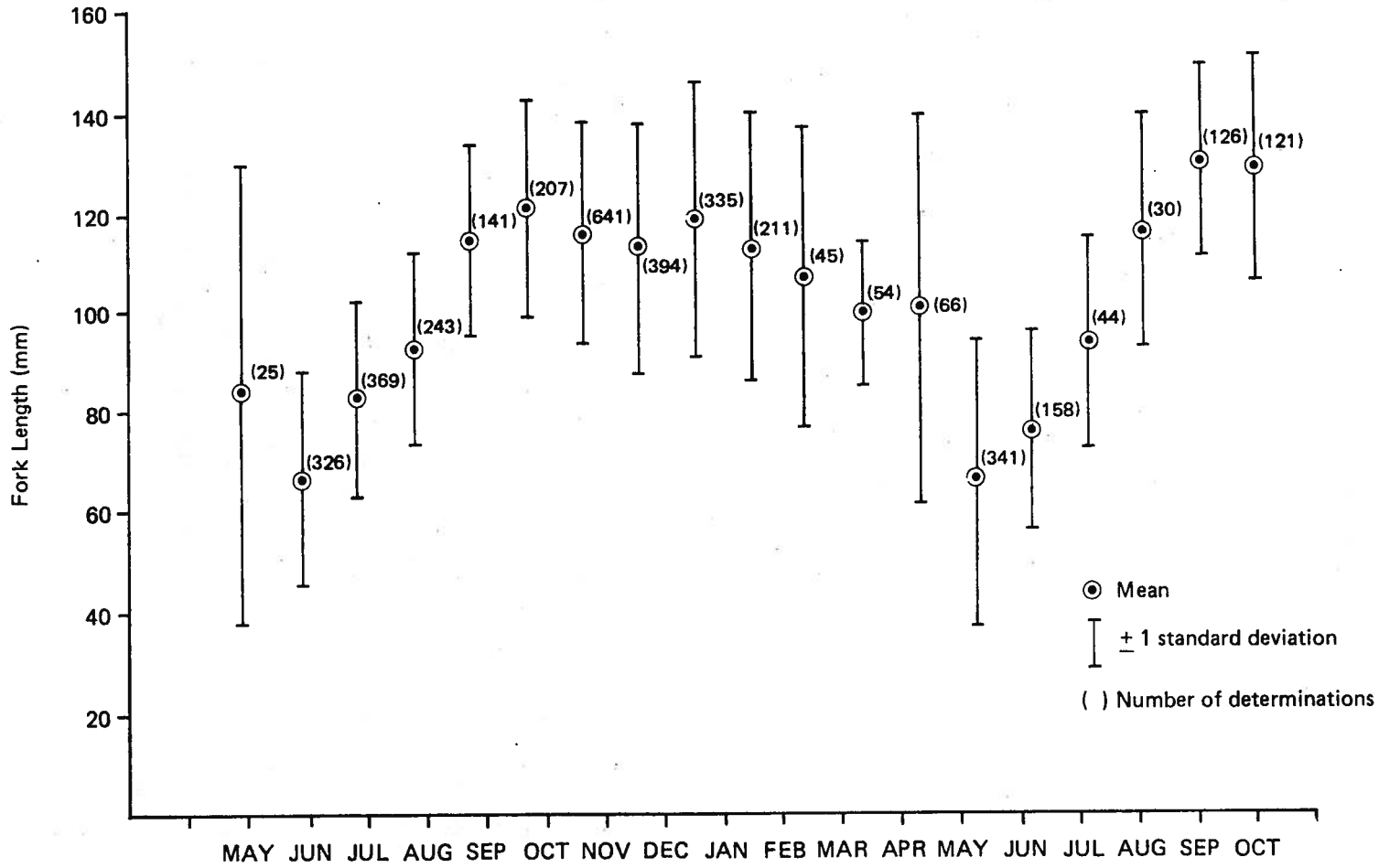


Figure 4-8. Mean fork lengths and standard deviations of measured yellowfin goby collected in impingement samples at Units 1-6 of the Pittsburg Power Plant, May 1978—October 1979.

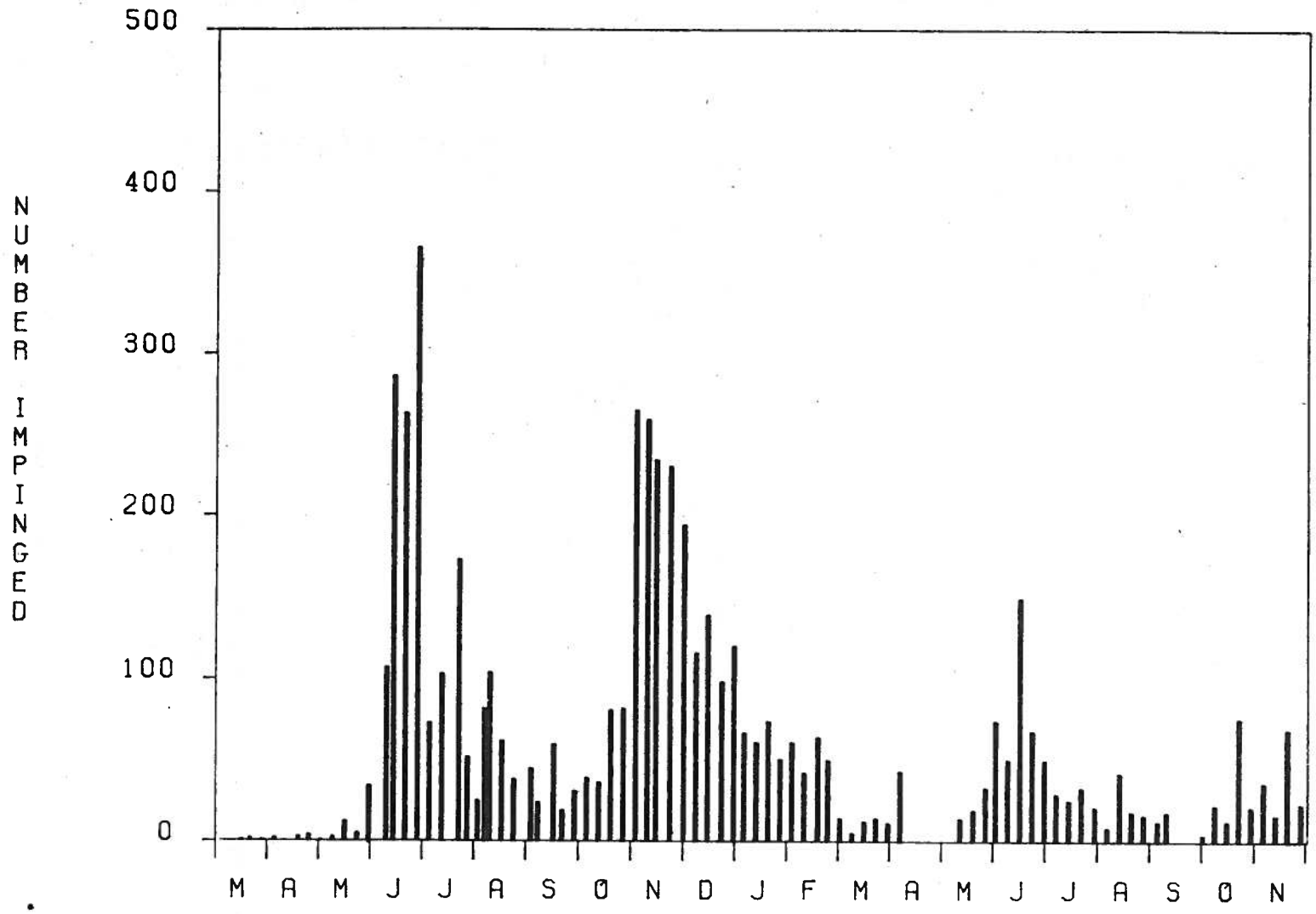


Figure 4-9. Numbers of impinged yellowfin goby collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

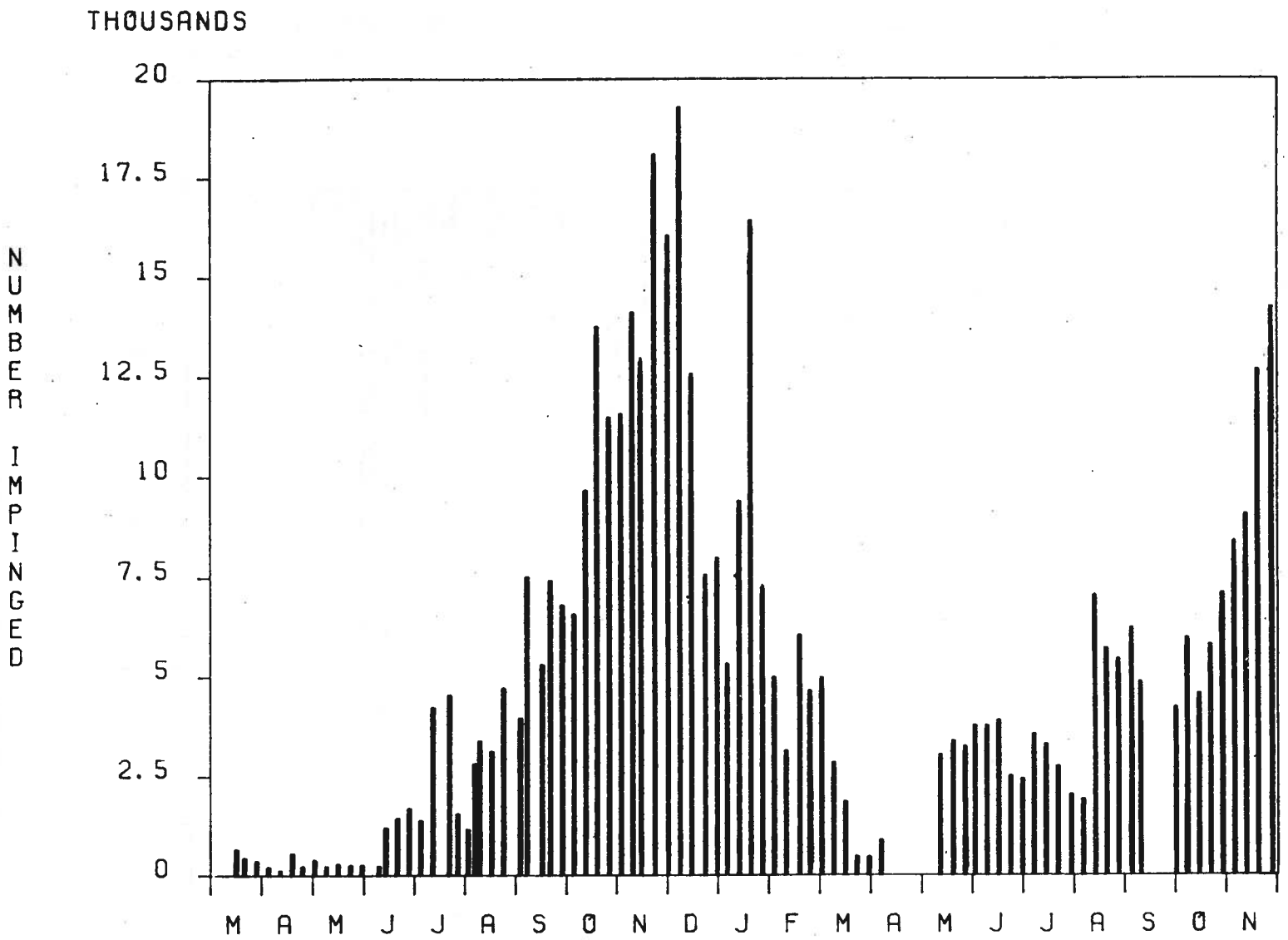


Figure 4-10. Numbers of impinged Palaemon macrodactylus collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

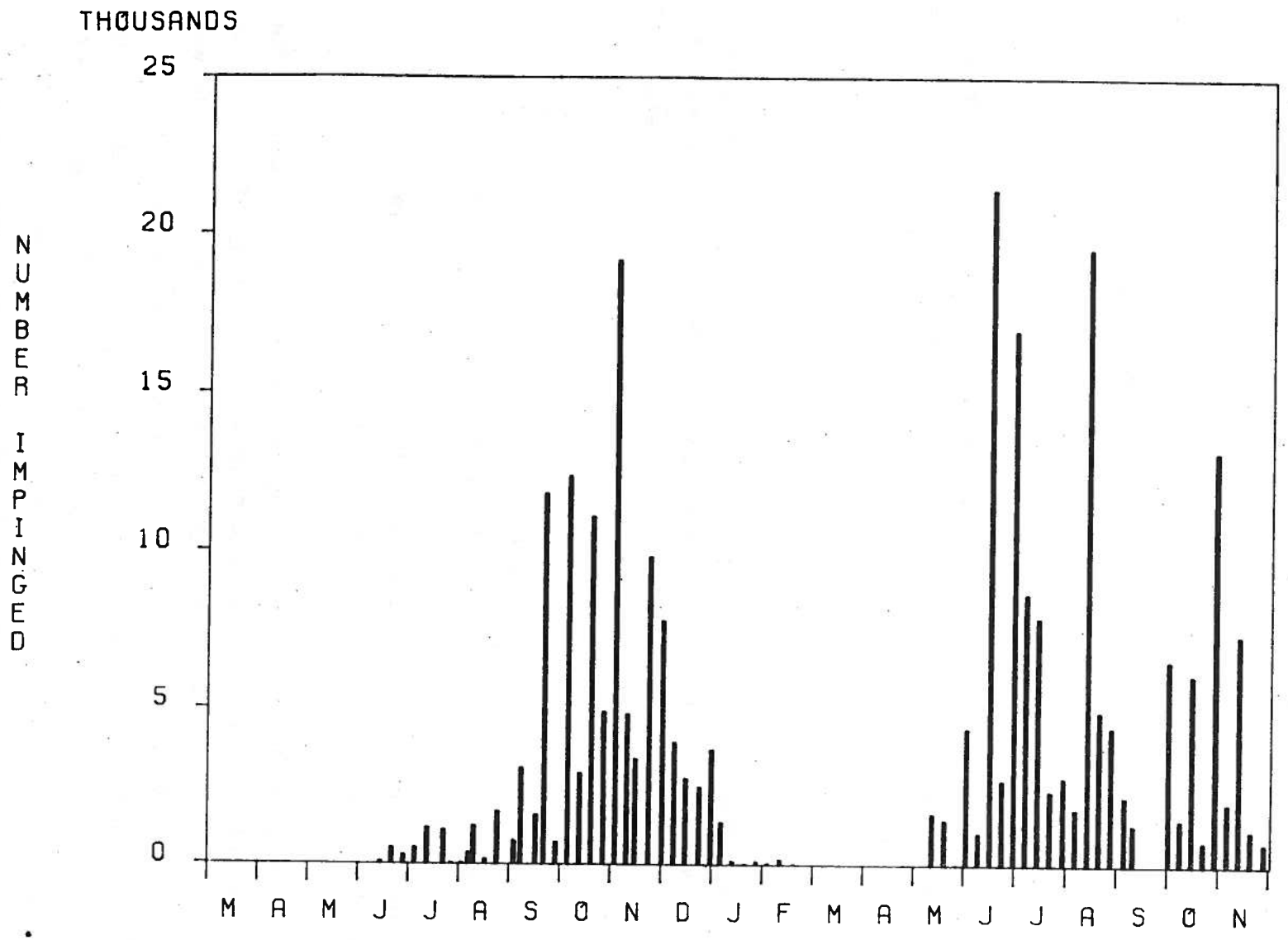


Figure 4-11. Numbers of impinged Crangon franciscorum collected at Units 1-6 of the Pittsburg Power Plant, March 1978–November 1979.

water column at night (Al-Adhub and Naylor 1975; Siegfried et al. 1978; Sitts and Knight 1979).

4.1.3.6 Crabs

Rhithropanopeus harrisii, the pebble crab, was the third most abundantly impinged macroinvertebrate, representing 4 percent (126,000 individuals) of the total macroinvertebrate catch for the 1978 study year and 9 percent (207,000) for the 1979 study period.

They were impinged year-round, with a peak in abundance in September and October in both study periods (Figure 4-12). Diel impingement of R. harrisii was fairly even: 46 percent by day and 54 percent at night.

4.2 IMPINGEMENT SURVIVAL

The Pittsburg Power Plant impingement survival study was designed to provide a quantitative basis for estimating the potential survival of fish and macroinvertebrates impinged on the intake screens of Units 1-4 and Units 5 and 6. Specific objectives of the study were:

1. to estimate the initial and latent (long-term) survival of fish and macroinvertebrates impinged on intake screens under routine operation, and
2. to examine the relationship between various screenwash operational cycles and initial and latent impingment survival.

Emphasis was given to those species impinged in greatest abundance. The impingement survival study was designed to estimate the species-specific proportions of fish and macroinvertebrates not surviving impingement. Impingement survival was determined for three modes of cooling water intake screen operation: continuous screen rotation, 1-hour intermittent rotation, and 3-hour intermittent rotation. The proportion of organisms not surviving impingement was estimated from the numbers of each species that were alive and dead immediately following removal from the intake screens and 96 hours after impingement. Control samples were collected to account for mortality from sampling and other causes not related to impingement. The

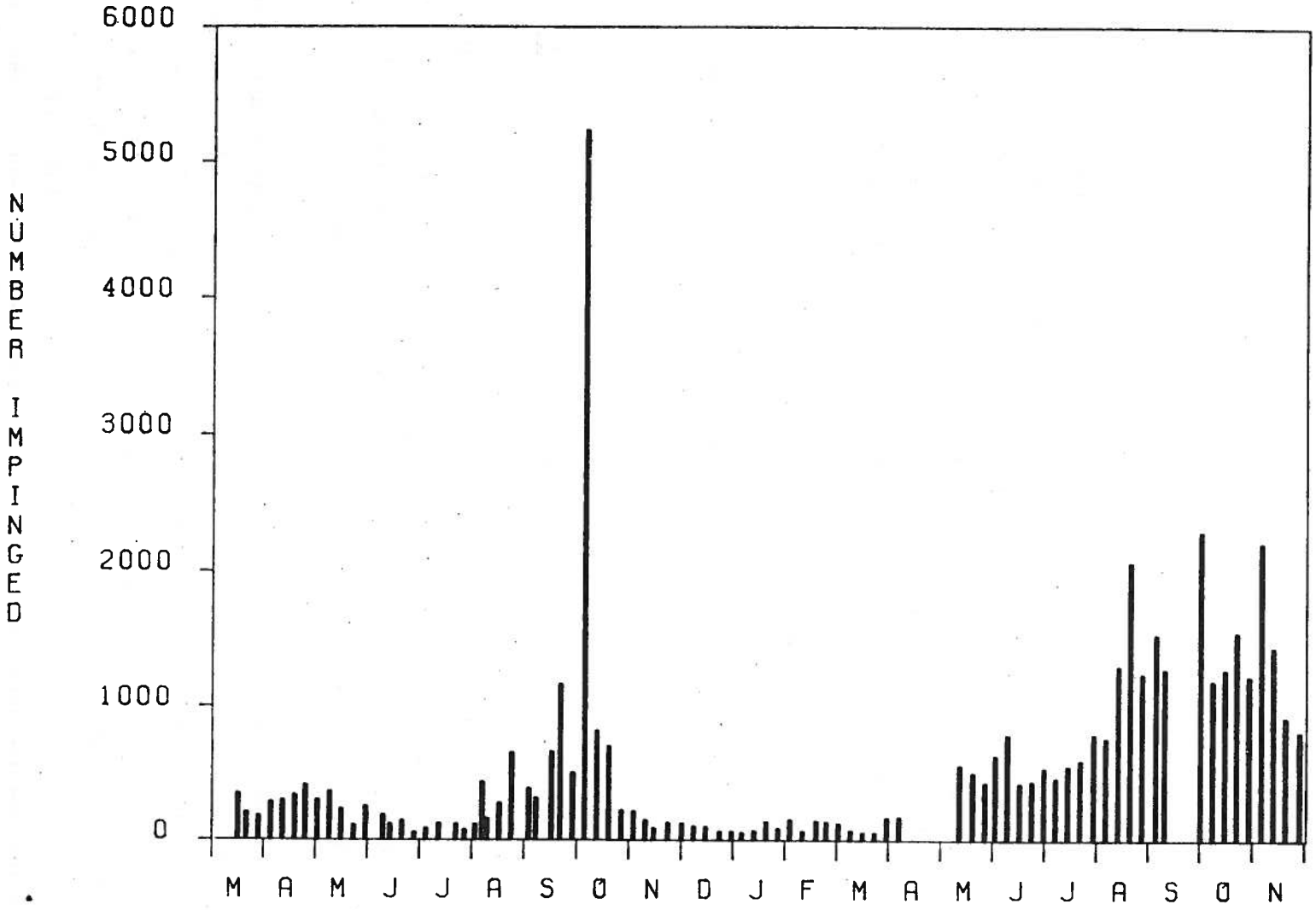


Figure 4-12. Numbers of impinged Rhithropanopeus harrisii collected at Units 1-6 of the Pittsburg Power Plant, March 1978—November 1979.

experimental design provided an opportunity to determine the relationship between the duration of impingement (i.e., intake screen rotation frequency) and the initial and latent survival of those species impinged in greatest abundance.

4.2.1 Methods and Materials

Fish and macroinvertebrates were collected in each of three intake screen operating modes:

1. a 3-hour intermittent screen rotation cycle in which the intake screens were stationary for 2-3/4 hours and were then rotated and cleaned for 15 minutes (an operating cycle typical of that used during impingement abundance sampling),
2. a 1-hour intermittent screen rotation cycle in which the intake screens were stationary for 45 minutes and were then rotated and cleaned for 15 minutes, and
3. a continuous screen rotation cycle.

The screen rotation cycle limited the amount of time an organism was impinged on the intake screen.

After screen rotation and washing, organisms and debris were collected from the screenwash sluiceway in nylon nets with 1/4-in (6.4-mm) mesh. Collection nets were held alternately in the sluiceway for 15 seconds at a time to reduce stress associated with collection. Organisms were placed in an ambient water bath and sorted from detrital material. During this sorting each organism was examined to determine whether it was alive or dead (initial observation). Organisms were considered dead if they showed no opercular movement or response to gentle probing.

Live animals were transferred to an ambient-temperature flow-through holding system. Fish were segregated according to size, life stage, and trophic level in separate 10-gallon (38-liter) tanks to reduce predation and cannibalism. Shrimp and crabs were held separate from all other organisms, since they are prey for many species of fish. The animals were observed periodically through a 96-hour period to quantify delayed mortality. Dead animals were removed from the holding system after each observation, identified to species, and preserved in formalin. At the end of the 96-hour observation all animals were identified, counted, and measured (fish to

fork length, crabs by carapace width, and shrimp from rostrum to telson). Live animals were released.

Control experiments were conducted to determine the mortality associated with the sluiceway collection, handling, and holding systems used. Fish were collected for the control experiments from the vicinity of the Pittsburg Power Plant. Beach seines were used most frequently, but otter trawls of about 2 minutes duration were also used; both methods captured fish in good condition. Fish were placed in 5-gallon (19-liter) buckets or in 50-gallon (200-liter) trash cans of aerated water with a salinity 2-4 ppt above ambient for transportation to the onsite holding system. Control fish were held for 2-4 days before testing to permit them to recover from the stress of collection and handling.

On the day of sampling, control fish were subjected to the experimental collecting and processing methods. The screenwash pumps were turned on, which filled the sluiceways with water, but the screens were not rotated. The control fish were then released into the sluiceway and subsequently collected. Control fish received the same treatment as experimental fish, except that they were not collected with, and hence not affected by, the detrital material usually associated with impingement collection.

The proportion of organisms not surviving impingement was calculated and corrected for control mortality by employing the following equation:

$$P_s = \frac{P_e - P_c}{1 - P_c} \qquad (4\text{-}20)$$

where

P_s = corrected proportion of organisms not surviving im- pingement

P_e = experimental proportion of organisms not surviving im- pingement

P_c = control proportion of organisms not surviving collection, processing, and holding.

4.2.2 Results

Survival estimates for impinged fish and macroinvertebrates, computed for three intake-screen rotation modes at Units 5 and 6 at a through-screen velocity of 1.5 fps (46 cm/sec), are summarized in Table 4-7. These results are considered to be representative of impingement survival at the Units 1-4 intake, which has similar through-screen velocities and design characteristics.

Initial survival of striped bass (predominantly young-of-the-year) was 54 percent after a 3-hour screen rotation cycle. In a 1-hour rotation cycle, initial survival increased to 64 percent, and it further increased to 81 percent for continuous intake-screen rotation. Initial survival of striped bass during continuous rotation was significantly higher statistically ($\alpha = 0.05$) than survival for either the 1-hour rotation cycle ($z = 2.40$) or the 3-hour cycle ($z = 3.67$). Initial survival for the 1-hour rotation cycle was not significantly different from that for the 3-hour cycle ($z = 0.68$).

Impinged striped bass had a high rate of mortality during the first 12 hours following impingement, reflected in the slopes of the curves in Figure 4-13, after which the mortality rate decreased substantially. Controls did not show this high early mortality rate, suggesting that the mortality during the first 12 hours is an impingement effect (e.g., mechanical abrasion, scale loss) and not an effect of the collection techniques used. Although initial and delayed impingement survival was different between the three screenwash modes tested (Figure 4-13) the survival curves were parallel, and the mortality rates were not significantly different ($\alpha = 0.05$) between the three modes.

Long-term (96-hour) survival after 3-hour, 1-hour, and continuous modes was 48 percent, 54 percent, and 68 percent, respectively. Decreasing the duration of impingement is expected to increase the survival of impinged striped bass.

Impingement survival of threadfin shad was characteristically low. Initial survival after 3-hour, 1-hour, and continuous intake-screen rotations was 36 percent, 31 percent, and 54 percent, respectively. Observations suggest that shad were sensitive to mechanical stresses of impingement: they typically exhibited a loss of equilibrium, erratic swimming behavior, and extensive scale loss. The initial mortality rate of threadfin shad was high, and none survived the 96-hour observation period. Alteration of screen operation to decrease the duration of impingement did not increase long-term survival.

TABLE 4-7 PERCENTAGE SURVIVAL OF FISH AND MACROINVERTEBRATES COLLECTED IN IMPINGEMENT SAMPLES AT UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT IN THREE SCREENWASH OPERATIONAL MODES, 1978

Taxon		Size Range (mm)	Screen Operational Mode: 3-Hour Intermittent			1-hour Intermittent			Continuous			Control		
				Survival (%)			Survival (%)			Survival (%)			Survival (%)	
Common Name	Scientific Name		No.	Initial	96-Hour	No.	Initial	96-Hour	No.	Initial	96-Hour	No.	Initial	96-Hour
Fish														
Striped bass	*Morone saxatilis*	26-602	116	54	48	77	64	54	69	81	68	110	100	87
Threadfin shad	*Dorosoma petenense*	55-125	22	36	0	13	31	0	13	54	0	10	100	20
Longfin smelt	*Spirinchus thaleichthys*	62-95	25	4	0	9	22	0	26	46	4	—	—	—
Delta smelt	*Hypomesus transpacificus*	46-78	17	0	0	7	0	0	2	0	0	—	—	—
Macroinvertebrates														
Oriental shrimp	*Palaemon macrodactylus*	22-57	696	72	53	384	80	63	426	75	58	—	—	—
Bay shrimp	*Crangon franciscorum*	26-67	33	67	45	42	79	60	23	65	30	—	—	—
Pebble crab	*Rhithropanopeus harrisii*	5-22	83	87	57	62	65	48	53	75	55	—	—	—

Note: Survival percentage for striped bass has been corrected to account for control mortality (Section 4.2.1); too few longfin smelt and Delta smelt were collected to permit control survival calculations.

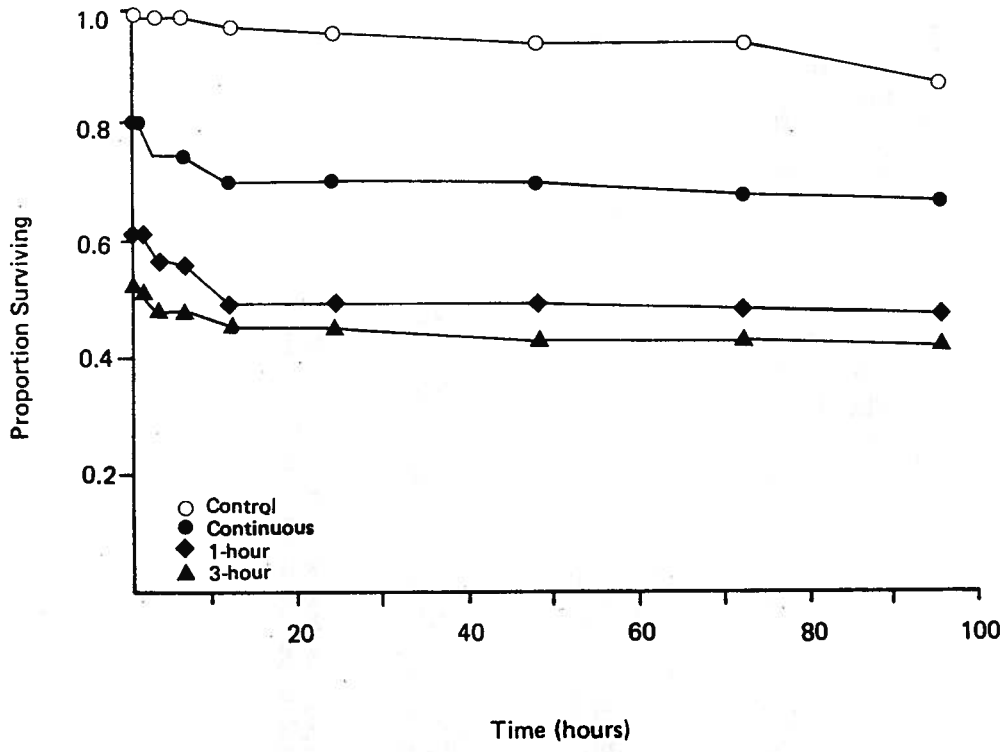


Figure 4-13. Proportions of striped bass surviving through 96 hours after impingement in various screen wash rotation modes at Units 1-6 of the Pittsburg Power Plant, 1978.

Longfin smelt and Delta smelt appear to be extremely sensitive to impingement stress. No initial survival was observed for impinged Delta smelt in any operational mode. Initial survival of longfin smelt increased from 4 percent to 22 percent to 46 percent as frequency of intake screen rotation increased from 3-hour to 1-hour to continuous, but long-term survival of longfin smelt was low for the three rotation modes tested: 0 percent for 3- and 1-hour screen rotation cycles and 4 percent for continuous rotation.

Impingement survival of Palaemon macrodactylus was characteristically high in all three screen rotation modes. Initial survival was 72 percent after 3-hour screen rotation, 80 percent after 1-hour rotation, and 75 percent after continuous screen rotation. No statistically significant differences were detected between the mortality rates of P. macrodactylus impinged in the three intake screen rotation modes (Figure 4-14).

Initial impingement survival of both Crangon franciscorum and Rhithropanopeus harrisii was high (more than 65 percent), and no relationship was apparent for either species between either initial or long-term survival and the three screen rotation modes (Table 4-7). Observations of impinged R. harrisii (initial survival more than 65 percent) suggest that their hard exoskeleton protects them against the stresses of impingement: those that did not survive had recently molted.

4.2.3 Discussion

Decreasing the duration of impingement by increasing the frequency of intake screenwash and rotation increased the survival of impinged striped bass. Long-term survival of threadfin shad, longfin smelt, and Delta smelt was low and appears not to have been enhanced substantially by increasing screen rotation frequency. Macroinvertebrates generally showed high impingement survival. Changes in screen operation that decreased the duration of impingement did not influence either the initial or long-term impingement survival of macroinvertebrates such as P. macrodactylus, C. franciscorum, and R. harrisii.

These initial and long-term survival observations reflect the survival of organisms removed from the intake screens. The estimates do not include effects associated with transit through the screenwash sluiceway or with the return of impinged organisms to the thermal discharge. In addition, infection resulting from abrasion or scale loss experienced during impingement were observed in the post-impingement observation

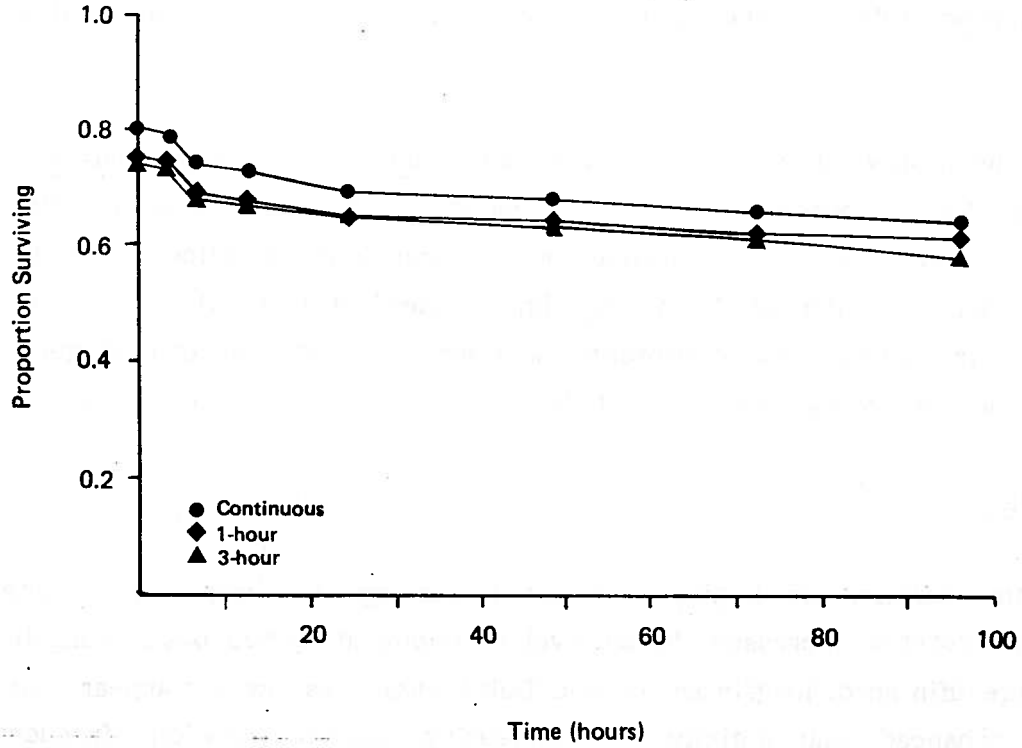


Figure 4-14. Proportions of Palaemon macrodactylus surviving through 96 hours after impingement in various screenwash rotation modes at the Pittsburg Power Plant, 1978.

period, but were not tested experimentally. Therefore, impingement survival study results represent the survival potential for organisms removed from the intake screens and not the survival of organisms returned to the receiving water body. Because of the screenwash sluiceway design at the Pittsburg Power Plant, which includes a wet well and pumps to transport impinged organisms and debris to the discharge, which impose mechanical stresses, it has been assumed in the impact assessment that no organisms survive impingement.

4.3 ESTIMATION OF TOTAL IMPINGEMENT CROPPING AT THE PITTSBURG POWER PLANT

Estimates of the numbers of selected fish and macroinvertebrates cropped by impingement at Units 1-4 and Units 5 and 6 of the Pittsburg Power Plant during the two study periods (March 1978 - March 1979 and May-November 1979) are summarized in Table 4-8. Estimated numbers cropped were calculated on the assumption of 100 percent impingement mortality, and also using the impingement survival rates presented in Section 4.2. The estimates were based on actual cooling water flow rates occurring during the study period. Total annual impingement cropping for fish was 382,000 individuals in 1978, assuming 100 percent impingement mortality. Based on impingement survival rates presented in Section 4.2, annual impingement cropping for fish was estimated to be 328,000 for the same period. The total number of macroinvertebrates cropped by impingement was estimated assuming 100 percent mortality and also assuming the survival rates presented in Section 4.2.

4.4 IMPINGEMENT SUMMARY

The abundance of impinged fish and macroinvertebrates was monitored at Units 1-4 and Units 5 and 6 of the Pittsburg Power Plant over a period of 20 months (March 1978 - November 1979). Impinged organisms were collected in the screenwash sluiceways of each intake. No mortality was attributable to the impingement of fish on the bar racks.

The most abundant fishes impinged were striped bass, threadfin shad, longfin smelt, yellowfin goby, starry flounder, and splittail, which accounted for 92 percent of the total catch at the two intakes. Total estimated impingement of fish under actual cooling system operation, for the year March 1978 - March 1979 was 161,000 at Units 1-4 and 220,000 at Units 5 and 6, a total of 381,000.

TABLE 4-8 ESTIMATED IMPINGEMENT CROPPING OF SELECTED FISHES AND MACROINVERTEBRATES AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT UNDER ACTUAL OPERATION, MARCH 1978 - MARCH 1979 AND MAY-NOVEMBER 1979

Taxon		Number (Assuming No Survival)		96-Hour Survival[a]	Number (Assuming Survival)[b]	
Common Name	Scientific Name	1978	1979	(%)	1978	1979
Fish						
Striped bass	*Morone saxatilis*	111,299	22,510	48	57,875	11,705
Threadfin shad	*Dorosoma petenense*	88,865	6,195	0	88,865	6,195
Longfin smelt	*Spirinchus thaleichthys*	104,545	9,612	0	104,545	9,612
Yellowfin goby	*Acanthogobius flavimanus*	31,519	7,145	0	31,519	7,145
Starry flounder	*Platichthys stellatus*	11,856	4,293	*	11,856	4,293
Splittail	*Pogonichthys macrolepidotus*	6,598	7,224	*	6,598	7,224
Other fish		27,193	7,074	*	27,193	7,074
Total fish		381,515	64,026		328,451	35,318
Macroinvertebrates						
Oriental shrimp	*Palaemon macrodactylus*	2,135,199	1,063,293	53	1,131,656	563,545
Bay shrimp	*Crangon franciscorum*	802,287	998,802	45	361,029	449,461
Pebble crabs	*Rhithropanopeus harrisii*	126,083	206,938	57	54,216	88,983
Other macroinvertebrates		535	0	*	535	0
Jellyfish	Cnidaria	25,803	0	*	25,803	0
Total macroinvertebrates		3,089,908	2,269,051		1,573,239	1,102,007

(a) Survival estimates are based on results of the 3-hour impingement survival studies.
(b) Because of the design and operation of the sluiceway - fish return, no survival is assumed in the impact assessment in Chapter 5.

Note: (*) indicates that survival data were unavailable; 0 percent survival was assumed.

The most abundant macroinvertebrates impinged were Palaemon macrodactylus, Crangon franciscorum, and Rhithropanopeus harrisii, accounting for 99 percent of the total catch at the two intakes. Total estimated impingement of macroinvertebrates under actual cooling system operation for the year March 1978 - March 1979 was 1.44 million individuals at Units 1-4 and 1.65 million individuals at Units 5 and 6, for a total of 3.09 million macroinvertebrates.

Results for selected species of fish indicate that the impingement of striped bass was (1) the highest of all species, (2) primarily limited to young-of-the-year, (3) highly correlated with the location of the entrapment zone, and (4) increased by recirculation of the thermal plume in winter.

The initial and latent survival of impinged organisms was examined at the Pittsburg Power Plant to estimate the species-specific potential for survival following impingement on the plant's intake screens. Initial and long-term survival of striped bass was relatively high (54 and 48 percent) in the 3-hour rotation mode and was found to increase further with decreasing duration of impingement. Impingement survival of threadfin shad, longfin smelt, and Delta smelt was low and was not enhanced by decreasing the duration of impingement. All three species of macroinvertebrates examined exhibited high survival following impingement, and their survival was not enhanced by alteration of the screenwash operating mode to decrease the duration of impingement.

Total impingement cropping of fish and macroinvertebrates was estimated on the basis of actual cooling water flows at Units 1-4 and Units 5 and 6. Total cropping was computed on the assumption of 100 percent impingement mortality and also by extrapolating species-specific results of the impingement survival study.

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Chapter 5

Impact Assessment

5.1 Approach and Decision Criteria for Impact Assessment

5.2 Estimated Entrainment and Impingement Cropping of Selected Invertebrates and Fish at the Pittsburg Power Plant

5.3 The Impact of Cooling Water Intake System Operation on Selected Invertebrates Other than *Neomysis mercedis*

5.4 The Impact of Cooling Water Intake System Operation on Selected Species of Fish Other than Striped Bass

5.5 Impact Assessment for Neomysis (*Neomysis mercedis*)

5.6 Impact Assessment for Striped Bass (*Morone saxatilis*)

5.7 Discussion

5.8 Summary and Conclusions

5.9 References

CHAPTER 5. IMPACT ASSESSMENT

This chapter assesses the impact of the combined entrainment and impingement cropping at the Pittsburg Power Plant on populations of fishes and macroinvertebrates of the source water body, the Sacramento and San Joaquin Delta. The impacts on individual species resulting from entrainment mortality (Chapter 3) and impingement mortality (Chapter 4) are evaluated by comparing the estimated numbers cropped with available information on the distribution, abundance, behavior, and life histories of the species. For some species, a quantitative approach to impact assessment was followed, whereby intake-related cropping was compared to measures of the water body population size to permit estimation of the proportion of the population cropped. For many species, however, lack of information concerning population size and the factors influencing population size has precluded a quantitative approach to impact assessment. In these cases an intuitive approach was followed, using information about general abundance, recent trends in abundance, distribution, and life history to estimate the impact of cooling water intake operation.

The Pittsburg Power Plant 316(b) impact assessment program has focused on two key species of the Bay-Delta: the striped bass (Morone saxatilis), a popular sport fish (discussed in Section 5.6); and the mysid shrimp, Neomysis mercedis, a major food source for young striped bass (discussed in Section 5.5). Other species of fish and macroinvertebrates potentially adversely affected by operation of the existing cooling water intakes are also considered in this chapter. Fish and macroinvertebrate species were selected for consideration if the potential for impact of cooling water intake systems at the Pittsburg Power Plant was considered to be high, based on life history and distribution information presented in Chapter 2 for each species.

Following the general approach outlined in the federal 316(b) guidelines (U.S. EPA 1977), the potential for impact on a given species was designated "high" if eastern San Pablo Bay, Suisun Bay, or the western Delta could be classified as a zone of high biological value for the species. Criteria used in determining whether or not the area was of high biological value were:

1. principal spawning (breeding) ground,
2. migratory pathway,
3. nursery or feeding area, and
4. large proportion of population present.

If any of the criteria were satisfied for a given species, the area was classified as having high biological value, and consequently the potential for impact was designated as high.

The criteria were applied to the 7 invertebrate and 21 fish species listed in Table 5-1 and the impact was assessed for those species for which the potential for impact was designated as high, based on distribution and life history information presented in Chapter 2. Species for which eastern Suisun Bay and the western Delta satisfied any of the four criteria for high biological value are noted in Table 5-1.

Organism Groups not Considered in Detail

Several organism groups are not considered in detail in the evaluation. These include (1) phytoplankton, (2) zooplankton (small completely or partially planktonic invertebrates), and (3) benthic invertebrates (molluscs, annelids, etc.).

In accordance with an agreement reached with the Regional Board, based on U.S. EPA guidelines (1977), phytoplankton and zooplankton were excluded because of their wide geographic distributions, short generation times, and population regeneration potential, and because the vigorous tidal currents continually replenish the phytoplankton and zooplankton populations near the plant, reducing the risk of localized population changes. On the same basis, benthic invertebrates were excluded because their bottom-dwelling habits minimize their involvement with the plant's cooling water intake systems.

Organization of the Chapter

The impact assessment for the Pittsburg Power Plant focuses on the data collected from March 1978 through March 1979 at the plant, because that represents one complete year. Additional data were collected from April to November 1979 (April-July 1979 for entrainment); these are presented for two key species in the Bay-Delta: the striped bass and the mysid shrimp *Neomysis mercedis*.

This Chapter consists of eight sections. In Section 5.1 the approach and decision criteria used in the impact assessment are discussed. Section 5.2 presents the estimated total numbers of individuals of the selected species entrained and impinged at the Pittsburg Power Plant during the period from March 1978 through March 1979.

TABLE 5-1 CRITERIA FOR DESIGNATION OF EASTERN SUISUN BAY AND THE WESTERN DELTA AS AN AREA OF HIGH BIOLOGICAL VALUE FOR SELECTED IMPORTANT SPECIES OF THE BAY-DELTA

Taxon	Principal Spawning Ground	Migratory Pathway	Nursery or Feeding Area	Large Portion of Population Present
Invertebrates				
Corophium insidiosum				
Crangon franciscorum			x	
Corophium stimpsoni	x		x	x
Corophium spinicorne	x		x	x
Neomysis mercedis	x		x	x
Palaemon macrodactylus	x		x	x
Rhithropanopeus harrisii	x		x	x
Fishes[a]				
Northern anchovy			x	
Pacific herring			x	
Jacksmelt				
Topsmelt				
White croaker				
Staghorn sculpin				
Gobies				
Surfperches				
Threadfin shad	x		x	x
Delta smelt	x		x	x
Longfin smelt	x		x	x
Yellowfin goby	x		x	x
Prickly sculpin	x		x	x
Starry flounder	x		x	x
White catfish				
Sacramento squawfish				
Splittail				
Striped bass		x	x	x
Chinook salmon		x		
American shad		x		
White sturgeon		x		

(a) Common names of fishes follow Robins et al. (1980).

In Section 5.3 the impact of cooling water intake operation on selected invertebrate species other than Neomysis mercedis is assessed, and in Section 5.4 the impact on selected fish species other than striped bass is assessed. The potential for impact on the N. mercedis population is discussed in Section 5.5, and in Section 5.6 the potential for impact on striped bass is assessed. Section 5.7 is a discussion of the impact assessment for the plant. Conclusions are presented in Section 5.8.

5.1 APPROACH AND DECISION CRITERIA FOR IMPACT ASSESSMENT

This impact assessment is the evaluation of the effects of entrainment and impingement cropping on populations of fishes and macroinvertebrates of the source water body, San Francisco Bay and the Sacramento-San Joaquin Delta (Bay-Delta). The impacts resulting from entrainment and impingement cropping are assessed on the basis of an evaluation of the estimated numbers cropped in the light of available information on species distribution, abundance, and life history. A cooling water intake can be judged to represent a minimization of environmental damage if the numbers of organisms entrained and impinged will not result in an adverse environmental impact (U.S. EPA 1977). Adverse environmental impacts occur when the ecological function of the organisms of concern are impaired to a level which:

1. precludes maintenance of existing populations;
2. results in a reduction in optimum sustained yield to sport and/or commercial fisheries;
3. involves threatened or endangered species directly or indirectly; or
4. constitutes an unmitigable loss to the aquatic system (U.S. EPA 1977).

The approach to impact assessment taken in this study program has been (1) to provide a qualitative evaluation of the probable net effects of entrainment and impingement mortality on populations of fishes and macroinvertebrates studied and, (2) to provide a quantitative estimate of the proportion of a population affected for fish species sustaining the largest numerical losses. The numbers of fish and macroinvertebrates entrained and impinged are discussed in relation to overall distribution and abundance of species in the Bay-Delta habitat, behavioral preferences, migration patterns, susceptibility to pump-induced flows, survival, and reproductive capabilities. The potential impact on populations of particular species or on the entire community has been assessed by estimating the proportion of the

population that is affected by the Pittsburg Power Plant and by evaluating the role of the species in the aquatic communities occurring in the Bay-Delta (Chapter 2).

Information is available from the California Department of Fish and Game (CDF&G) midwater trawl survey to examine trends in the relative abundance, expressed in terms of catch per unit effort (CPUE), for many of the selected fish species listed in Table 5-1. These data permit examination of year-to-year fluctuations in abundance and species composition for Bay-Delta fish populations from 1967 to 1978. To examine whether or not the population of a species of fish is impaired or reduced to the degree that long-term stability is affected, the midwater trawl data were tested for a significant decline in CPUE or a change in the relative composition of the fish community over the 12 years of monitoring. This approach assumes that if the compensatory reserves of a population are sufficient to nullify the effect of entrainment and impingement mortality and all other sources of mortality (both natural and man-induced), then no significant decline in the population would occur. The long-term trends in population abundance provide evidence useful in assessing whether or not the compensatory reserve of a population has been exceeded and therefore the potential for adverse impact resulting from entrainment and impingement cropping exists.

One of the most important single species in this impact assessment is a species of the order Mysidacea, Neomysis mercedis. Because of the frequency with which it is mentioned, it will be referred to as neomysis. The impact on neomysis resulting from entrainment cropping is assessed quantitatively in Section 5.5. Intake-related cropping is compared with estimates of population size to estimate the proportion of the population cropped. Factors regulating population size (e.g., birth and death rates, generation time, fecundity) are examined to determine the relative significance of power plant cropping as a factor affecting population size. The effect of reduced abundance of neomysis on the striped bass population and the effects of flow conditions on intake-related cropping are assessed qualitatively.

The potential impact on the striped bass population is assessed quantitatively: impingement and entrainment cropping is examined in perspective with factors affecting the regulation of striped bass population dynamics (Section 5.6). Entrainment cropping estimates are compared with estimates of population size to estimate the proportion of the young-of-the-year population cropped. The availability of long-term records on year-class strength and population characteristics of

striped bass in relationship to environmental conditions have made it possible to develop an adult stock model, which is used to examine the structure of the striped bass population with and without the influence of intake-related cropping, under present and future water management regimes. This population-modeling approach, based on quantitative relationships developed from data on the Bay-Delta population, is used to determine the relative significance of power plant cropping as a factor affecting population size.

A companion report, a 316(a) demonstration for the Pittsburg Power Plant (Tetra Tech 1976) provides additional documentation and analysis of the effects of Units 1-6 cooling system operation on the ecological community of the receiving waters. An impact assessment is made of entrainment mortality and the effects of the thermal plume on the aquatic community of the receiving waters. Information presented in the 316(a) demonstration has been integrated with the data and analyses developed in the 316(b) studies whenever appropriate, to help provide a broader basis for understanding the relationship between operation of the Pittsburg Power Plant cooling system and the aquatic community of the receiving waters.

5.2 ESTIMATED ENTRAINMENT AND IMPINGEMENT CROPPING OF SELECTED INVERTEBRATES AND FISH

Estimates of the number of invertebrates and fish entrained and impinged at the Pittsburg Power Plant from March 1978 through March 1979 are presented in Tables 5-2 and 5-3, respectively. Methods and results of the entrainment and impingement monitoring programs are discussed in detail in Chapters 3 and 4, and supporting data may be found in Appendix E.

At the Pittsburg Power Plant, entrainment survival studies were conducted with striped bass, neomysis, and gammaridean amphipods (Corophium stimpsoni and C. spinicorne), permitting the estimation of the proportions of entrained individuals not surviving entrainment. The overall proportions of striped bass, neomysis, and gammaridean amphipods not surviving entrainment during the one-year study were 0.51, 0.43, and 0.39, respectively (Section 3.2). Earlier studies conducted with zooplankton species such as Eurytemora affinis indicated that the proportion not surviving entrainment was nearly negligible except during periods of high (>33 C; 90 F) discharge temperatures (PGandE 1973; Appendix B).

TABLE 5-2 ESTIMATED CROPPING OF SELECTED INVERTEBRATES AT UNITS 1-7 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Taxon			
Common Name	Scientific Name	Entrainment[a]	Impingement[b]
Gammarid amphipod	*Corophium stimpsoni*	3.17×10^8	0
Gammarid amphipod	*Corophium spinicorne*	7.80×10^6	0
Oriental shrimp	*Palaemon macrodactylus*	6.63×10^8	2.14×10^6
Pebble crab	*Rhithropanopeus harrisii*	4.71×10^8	1.26×10^5
Bay shrimp	*Crangon franciscorum*	0	8.02×10^5
Opossum shrimp	*Neomysis mercedis*	4.03×10^9	0

(a) Estimates are based on actual cooling water flows for Units 1-7 and 100 percent entrainment mortality, except for *C. stimpsoni* and *C. spinicorne* (39 percent mortality) and *N. mercedis* (43 percent mortality).

(b) Estimates are based on actual cooling water flows for Units 1-7 and 100 percent impingement mortality.

TABLE 5-3 ESTIMATED CROPPING OF SELECTED FISHES AT UNITS 1-7 OF THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Taxon			
Common Name	Scientific Name	Entrainment[a]	Impingement[b]
Chinook salmon	*Oncorhynchus tshawytscha*	1.8×10^4	6.0×10^2
American shad	*Alosa sapidissima*	1.7×10^5	3.9×10^3
White sturgeon	*Acipenser transmontanus*	0	7
Northern anchovy	*Engraulis mordax*	5.1×10^6	1.3×10^3
Pacific herring	*Clupea harengus pallasi*	2.2×10^7	49
Threadfin shad	*Dorosoma petenense*	6.7×10^5	8.9×10^4
Delta smelt	*Hypomesus transpacificus*	5.0×10^7[c]	1.1×10^4
Longfin smelt	*Spirinchus thaleichthys*		1.0×10^5
Yellowfin goby	*Acanthogobius flavimanus*	6.1×10^6	3.2×10^4
Prickly sculpin	*Cottus asper*	3.0×10^7	4.0×10^2
Starry flounder	*Platichthys stellatus*	7.0×10^5	1.2×10^4
Striped bass	*Morone saxatilis*		
eggs		1.0×10^6	0
larvae-adult		1.4×10^8	1.1×10^5

(a) Estimates based on actual cooling water flows for Units 1-7 and 100 percent entrainment mortality, except striped bass (51 percent mortality).
(b) Estimates based on actual cooling water flows for Units 1-7 and 100 percent impingement mortality.
(c) Entrained smelts not identifiable to species.

Results of macroinvertebrate entrainment survival studies at power plants outside the PGandE system provide additional evidence that mortality of entrained macroinvertebrates is generally low (Appendix B). Cannon et al. (1978) reported that initial entrainment mortality of Gammarus spp. (gammaridean amphipods) at four power plants on the Hudson River ranged from 0 to 12 percent when discharge temperatures remained below 30 C (86 F). The authors concluded that for most macroinvertebrates, entrainment mortality was not observed except under extreme summer conditions (i.e., discharge temperatures $\geq$35 C; 95 F). Results of zooplankton entrainment survival studies conducted at the Potrero Power Plant in 1971-1972 (Icanberry and Adams 1974) also demonstrate that entrainment mortality is negligible for the smaller invertebrates. Observed initial mortality of total zooplankton was less than 2 percent. Given the low entrainment mortality observed for (1) zooplankton at the Potrero Power Plant, (2) gammarideans at the Pittsburg Power Plant, and (3) macroinvertebrates at four Hudson River power plants, the entrainment mortality of macroinvertebrates at the Bay-Delta power plants is likely to be low, probably less than 20 percent. A more detailed discussion of the survival of entrained fish and invertebrates is presented in Appendix B.

The total numbers of invertebrates and fish impinged at the Pittsburg Power Plant were estimated on the basis of densities of impinged organisms (numbers impinged per unit volume of cooling water) observed during weekly sampling and of actual cooling water flows during the study period. The estimated total numbers impinged at the plant during a 12-month period (March 1978 - March 1979) are presented in Tables 5-2 and 5-3. Impinged fish and invertebrates were assumed not to have survived, and the numbers reported represent estimates of the numbers cropped by impingement.

5.3 THE IMPACT OF COOLING WATER INTAKE SYSTEM OPERATION ON SELECTED INVERTEBRATES OTHER THAN NEOMYSIS MERCEDIS

This section assesses the impact of cooling water intake system operation on selected invertebrates. The focus of the section is on the combined entrainment and impingement impact of the Pittsburg Power Plant. The macroinvertebrates selected as key species were Corophium stimpsoni and C. spinicorne, Palaemon macrodactylus, Rhithropanopeus harrisii, and Crangon franciscorum.

5.3.1 Gammaridean Amphipods

The tube-dwelling amphipods Corophium stimpsoni and C. spinicorne are abundant and widely distributed from Suisun Bay to the lower Sacramento and San Joaquin rivers. Both species migrate off the bottom and into the water column at night (Siegfried et al. 1978), and they are entrained in large numbers. Entrainment survival, however, is high except at high discharge temperatures. At discharge temperatures below 30 C, entrainment mortality was found to be 4 percent at the Pittsburg Power Plant (Section B.3.1, Appendix B), and it remained low until temperatures exceeded 32 C. During the year of this study at the Pittsburg Power Plant, 39 percent of the gammaridean amphipods entrained were estimated to have been cropped (Section 3.2). Estimated entrainment cropping amounted to 317 million C. stimpsoni and 7.8 million C. spinicorne (Table 5-2).

No estimates of population size are available, but several observations of amphipod densities are available, from which the general abundance of the species can be ascertained. Corophium stimpsoni has been observed at densities of 4,000-5,000/m^2 in San Pablo Bay (Jones 1961), at 2,000/m^2 in Suisun Bay (Hazel and Kelley 1966), and 14,000/m^2 (Siegfried et al. 1978) in the lower Sacramento River. Corophium spinicorne has also been observed, in similarly high densities, in the lower Sacramento River: from 6,000/m^2 (Fisk and Doyle 1962) to 20,000/m^2 (Siegfried et al. 1978).

At an average density of 5,000/m^2, the number of Corophium spp. cropped during one year at the Pittsburg Power Plant would be equivalent to the number inhabiting 65,000 m^2 of benthic substrate, or about 0.2 percent of the bottom area of eastern Suisun Bay. Therefore, the numbers of amphipods cropped by entrainment are considered to be small compared to the potential sizes of the Bay-Delta populations.

Although the estimated numbers of these amphipods cropped by entrainment are high, the probability of direct impact on their populations and the indirect impact on the aquatic community is low. Any localized changes in the abundance of C. stimpsoni or C. spinicorne resulting from entrainment mortality are diminished by the rapid movement of water through the discharge area and mixing in the receiving waters. The short generation times of amphipods, and thus the large numbers of individuals produced, would compensate rapidly for any localized changes. In addition, since the plant cooling water flows represent only a small fraction of the water body's volume

and since the Corophium spp. populations are distributed through most of the Delta and Sacramento and San Joaquin rivers, as evidenced by plankton surveys conducted in 1978 and 1979 (EA 1981a), the potential for impact on the widely distributed Corophium spp. populations is minimal. Although they move into the water column for feeding, breeding, or dispersal, both species are either predominantly tube-dwelling or cling to substrates, this also reduces their susceptibility to entrainment. In addition, those individuals that are entrained into the once-through cooling systems of the Pittsburg Power Plant have a high probability of surviving entrainment and are returned to the receiving water body. Those amphipods that are cropped as a result of entrainment continue to provide a food resource for fish and invertebrates inhabiting the discharge and the receiving water body. It is therefore unlikely that the Pittsburg Power Plant has any detectable adverse impact on the populations of C. stimpsoni or C. spinicorne.

5.3.2 Palaemon macrodactylus

Palaemon macrodactylus is a nonburrowing, subtidal shrimp that was accidentally introduced into San Francisco Bay from Japan or Korea around 1954 (Newman 1963). Although not found along the outer coast, it is widely distributed in San Francisco Bay, extending up the estuary past Antioch (EA, unpublished data a). It is most abundant in the low-salinity areas of the San Francisco Bay-Delta system (Newman 1963; Ganssle 1966), where densities of juveniles and adults may exceed $10/m^3$ (Siegfried et al. 1978). The spawning season is from April to October (Little 1969). Fish are the most important natural predators of this shrimp. A commercial fishery existed in the 1950s (Newman 1963), but there is no fishery for this shrimp today.

Palaemon macrodactylus is the most abundantly entrained and impinged decapod at the Pittsburg Power Plant: an estimated 663 million larvae were entrained and 2.14 million juveniles and adults were impinged between March 1978 and March 1979 (Table 5-2).

Although no experimental data are available on the survival of P. macrodactylus entrained into once-through cooling systems, it is expected that survival will be similar to that for neomysis and amphipods, 85-95 percent at discharge temperatures below 30 C (86 F). Impingement survival was 53 percent in studies conducted at the plant.

The impact of entrainment and impingement cropping on the Bay-Delta population of P. macrodactylus is thought to be negligible, because of the rapidity with which the population has become established since its recent (1954) introduction from the Orient. By 1957 the shrimp had become abundant enough to be taken in the commercial catch from San Francisco Bay and San Pablo Bay. After that, the population virtually exploded (Newman 1963). The population spread from San Pablo Bay to Suisun Bay in 1959 and to the western Delta in 1960-1961. If the species were being adversely affected by entrainment and impingement cropping, it is unlikely that the population would have expanded so rapidly and maintained a high population.

The direct impact of the Pittsburg Power Plant on the P. macrodactylus populations and the indirect impact on the community are concluded to be minimal for the following reasons:

1. Palaemon macrodactylus are abundant and widely distributed in the San Francisco Bay and Delta system;
2. those entrained were predominantly larvae, and each adult female produces several thousand embryos that are widely distributed as plankton;
3. on the basis of survival rates of other invertebrate species (Chapter 3; Appendix B) survival of P. macrodactylus that are entrained is expected to be relatively high;
4. those shrimp that do not survive entrainment or impingement continue to provide a food source to invertebrates and fish;
5. entrainment and impingement cropping does not preclude maintenance of the existing population.

5.3.3 Rhithropanopeus harrisii

Rhithropanopeus harrisii is a small introduced crab common in Suisun Bay and the western Delta. Native to East Coast estuaries, R. harrisii inhabits the intertidal and shallow subtidal zones of sloughs and bays with muddy banks and bottom (Smith and Carlton 1975) or on and under rocks as a member of the fouling community (Carlton 1978).

Rhithropanopeus harrisii spawn from late spring to fall. The larvae (zoeae) were observed in entrainment samples collected at the Pittsburg Power Plant throughout

the spawning season, with maximum densities during the summer months. They were observed in impingement collections year-round at the plant, but in greatest numbers in September and October (see Figure 4-12).

The larvae are planktonic, and hence susceptible to entrainment, for only a few weeks before their metamorphosis to non-planktonic juveniles. The juveniles and adults are benthic and hence have a reduced susceptibility to cooling water intake structures.

During the entrainment and impingement studies at the Pittsburg Power Plant, an estimated 471 million R. harrisii larvae were entrained and an estimated 126,000 juveniles and adults were impinged (Table 5-2). The entrainment survival of R. harrisii is expected to be as high as that observed for other invertebrates, and the 96-hour impingement survival potential of R. harrisii is relatively high, 57 percent under standard operating conditions.

The direct impact of plant-related cropping on the Bay-Delta population is low. Most of the R. harrisii entrained were in the zoeal (early larval) stage. Although estimates of fecundity are not available for R. harrisii, it is expected that females produce thousands of zoeae per brood, of which only a very few survive to the adult stage. Those that do not survive entrainment or impingement continue to provide a food resource for other invertebrates and fish in the receiving water body. Entrainment and impingement cropping do not preclude maintenance of the existing population. It is concluded that operation of the Pittsburg Power Plant does not have an adverse impact on the population.

5.3.4 Crangon franciscorum

Crangon franciscorum, the common bay shrimp, is found in bays, estuaries, and occasionally on the open coast, ranging from southeastern Alaska to southern California. It is distributed in the Bay-Delta from the southern Bay to the western Delta. Crangon franciscorum is found over a wide range of salinities, but not in fresh water.

Spawning occurs in spring and summer after the adults have migrated back into the more saline areas of the Bay-Delta (Israel 1936). After spawning, the male dies and the female releases a brood of 2,000-5,000 eggs which is carried under the abdomen

for about one month (Krygier and Horton 1975). After the eggs hatch and free-swimming larvae are released from the brood, most of the adult females die. The distribution of larvae in the Bay-Delta is not known, but none were collected in entrainment samples at the Pittsburg Power Plant. The larvae develop into adult shrimp over a period of 8-11 months (Krygier and Horton 1975). Juveniles and adults remain near the bottom in muddy and sandy areas except at night, when they migrate upwards in the water column (Sitts and Knight 1979a).

Crangon franciscorum are not entrained at the Pittsburg Power Plant, as spawning occurs in the more saline areas of the Bay-Delta. Large numbers of adults are impinged: an estimated 802,000 during the one-year study, (Table 5-2). Juveniles move into and reside in the less saline areas of the Bay-Delta (Israel 1936), accounting for the high abundance observed at the plant.

Bay shrimp, especially C. franciscorum, have been fished commercially in the San Francisco Bay area since about 1869 (Frey 1971). Peak landings at the turn of the the century were about 2.9 million kg (6.5 million lb) (Skinner 1962), but the annual catch for 1973-1976 averaged only 34,500 kg (83,700 lb) (McAllister 1975, 1976; Pinkas 1977; W. Dahlstrom, CDF&G, personal communication, 1979). At present the fishery is limited by market demand rather than the size of the stock (Frey 1971; W. Dahlstrom, CDF&G, personal communication, 1979). Most of the shrimp landed since 1965 have been used as bait for striped bass and sturgeon sport fishing, but historically the shrimp were caught for human consumption. Although there is some activity in south San Francisco Bay near Alviso, most of the fishery is located in San Pablo Bay.

The annual impingement cropping of C. franciscorum at the Pittsburg Power Plant can be compared to the annual commercial catch, which presently averages about 45,000 kg (100,000 lb). Assuming an average weight of 0.5 g per adult (Sitts and Knight 1979a), the 802,000 individuals cropped annually by impingement would weigh 401 kg (884 lb), less than one percent of the annual commercial catch. Since the commercial fishery is limited by demand at present, it is likely that only a small portion of the population is caught annually, and therefore the impingement cropping constitutes only a very small fraction of the total population.

The direct impact of entrainment and impingement on the C. franciscorum population

and the indirect impact on the community are concluded to be low for the following reasons:

1. they are abundant and widely distributed throughout much of San Francisco Bay and the western Delta;
2. they spawn in more saline areas of the system and are not entrained at the Pittsburg Power Plant;
3. those shrimp that are cropped by impingement are returned to Suisun Bay and are available as food for the fish and invertebrates;
4. entrainment and impingement cropping does not preclude maintenance of the existing population.

5.4 THE IMPACT OF COOLING WATER INTAKE SYSTEM OPERATION ON SELECTED SPECIES OF FISH OTHER THAN STRIPED BASS

The major taxonomic groups of fish in the Bay-Delta are the anadromous fishes and those fishes which are either permanent or seasonal residents. The anadromous fishes include striped bass, sturgeon, American shad, salmon, and steelhead. Fishes inhabiting the Bay-Delta as permanent or seasonal residents include bait and forage fishes such as Pacific herring, northern anchovy, Delta smelt, longfin smelt, and threadfin shad and such species as yellowfin goby, prickly sculpin, and starry flounder.

Data from the 1967-1978 CDF&G midwater trawl surveys are used to determine if the Bay-Delta fish populations have declined significantly in the Bay-Delta. In these studies the relative abundances of fishes in the Bay-Delta were monitored monthly from August through March each year (see Stevens 1977a for details). Approximately 87 stations were sampled each month from nine strata, corresponding to major water bodies from the south Bay to the Delta. To provide a general indicator of population trends, a mean-catch-per-unit-effort (CPUE) was calculated for each year, combining all months. The primary index in the analysis was calculated as the mean CPUE for only those stations in Suisun Bay and the Delta to detect local trends in population abundance in the vicinity of the Pittsburg Power Plant.

The trend in abundance between 1967 and 1978 was examined for each species by fitting a least-squares linear regression to the annual CPUE for the entire Bay-Delta region or the Sacramento-San Joaquin Delta region and testing statistically the null hypothesis that the slope of each regression is equal to zero. If a population was

declining over the period 1967-1978, it is expected that the regression slopes would be negative and significantly different from zero. Regression slopes not significantly different from zero provide evidence that the population is not declining.

In addition to trends in abundance, the catch statistics are used to examine changes in the species composition between 1968 and 1978 (Table 5-4). Because of survey design and the selectivity of the midwater trawl, the data serve only as an indicator of changes in species composition; no data are available for either chinook salmon or prickly sculpin.

The midwater trawl data do not provide a preoperational-operational comparison, since by 1967 the populations had been subject to entrainment and impingement for 14 years at Units 1-4, for 7 years at Unit 5, and for 6 years at Unit 6. Nevertheless, if there have been any ongoing effects on the populations, they should be reflected in the 12 years of catches recorded.

In this section, the combined impact of the cooling water intake systems of the Pittsburg Power Plant on selected anadromous, seasonal and permanent resident fishes other than striped bass is assessed.

5.4.1 Anadromous Fishes

Anadromous fishes are considered by the SWRCB (1975) to be the most important fish in the Bay-Delta system. All of the anadromous fishes migrate into the freshwater portions of the Bay-Delta or the Sacramento and San Joaquin river systems to spawn, but spend part of their life cycles in the ocean and/or the marine and estuarine portions of the Bay-Delta. The anadromous fishes of the Bay-Delta discussed in this section include the chinook salmon, American shad, and white sturgeon. Striped bass are discussed in Section 5.6. Anadromous fishes not discussed in this section but of some importance in the Bay-Delta sport fishery include coho salmon, steelhead, and green sturgeon. These species are not considered in the impact assessment, because almost none were entrained or impinged at the Pittsburg Power Plant (Appendix E).

5.4.1.1 Chinook (King) Salmon (Oncorhynchus tshawytscha)

Chinook salmon is a well-known anadromous fish which migrates through the Bay and Delta to spawn in tributary rivers. The only significant involvement of these species

TABLE 5-4 ANNUAL MEAN CATCH-PER-UNIT-EFFORT (CPUE) AND PERCENTAGE OF THE CATCH FOR SELECTED FISH SPECIES COLLECTED IN THE CALIFORNIA DEPARTMENT OF FISH AND GAME MIDWATER TRAWL SURVEY FOR THE SACRAMENTO-SAN JOAQUIN DELTA REGION, 1968 AND 1978

Taxon		1968		1978		Change in Percentage
Common Name	Scientific Name	CPUE	%	CPUE	%	1968-1978
American shad	*Alosa sapidissima*	3.76	4.5	1.33	3.2	-1.3
White sturgeon	*Acipenser transmontanus*	1.13	1.4	1.00	2.4	+1.0
Northern anchovy	*Engraulis mordax*	27.61	33.3	0	0	-33.3
Pacific herring	*Clupea harengus pallasi*	1.13	1.4	5.00	12.1	+10.7
Threadfin shad	*Dorosoma petenense*	18.89	22.8	14.78	35.8	+13.0
Delta smelt	*Hypomesus transpacificus*	4.52	5.5	2.83	6.9	+1.4
Gobies	Gobiidae	1.00	1.2	1.26	3.1	+1.9
Starry flounder	*Platichthys stellatus*	1.55	1.9	2.33	5.6	+3.7
Striped bass	*Morone saxatilis*					
age 0		19.35	23.3	9.13	22.1	-1.2
age 1		2.47	3.0	1.87	4.5	+1.5
adults		1.50	1.8	1.78	4.3	+2.5

Note: No data are available for chinook salmon or prickly sculpin.

with the cooling water intakes occurs when the larvae and juveniles descend the tributaries and move through the estuary on their way to the ocean. Very few individuals are entrained at the Pittsburg Power Plant (the annual entrainment estimate is 18,100; Table 5-3), mainly because the larvae are found predominantly upstream of the power plant, and because the juveniles passing by the plant on their seaward migration are strong swimmers and can avoid entrainment. Juveniles are impinged, in relatively low numbers; an estimated 614 in the study year (Table 5-3).

The estimated number of juvenile chinook salmon (smolts) produced in hatcheries that passed through the Delta in 1978 is between 23 and 25 million (J. Robinson, CDF&G, personal communication, 1981). The contribution of natural spawning to the smolt population would substantially increase these estimates. The estimated entrainment and impingement cropping combined represents less than 0.08 percent of the estimated population of juvenile chinook salmon released into the Delta from hatcheries. The numbers of chinook salmon cropped by the Pittsburg Power Plant are small in comparison to the cropping of adults by sport and commercial fisheries, and to the estimated size of the adult population. The commercial catch of chinook salmon, from waters outside the Golden Gate, has ranged from 0.77 to 1.27 million kg (1.7-2.8 million lb) in recent years. The sport catch from ocean waters averages 100,000 fish annually, and that from the Bay-Delta averages 25,000 fish annually. The spawning stock in the Bay-Delta has been estimated at 181,000-400,000 fish in recent years. Since only a small proportion of the juveniles passing by the Pittsburg Power Plant return to the rivers as adults to spawn (R. Hallock, CDF&G, personal communication, 1980), the equivalent adult loss represented by the estimated 18,100 larvae and juveniles entrained and the 614 juveniles impinged is very small, and represents a negligible fraction of the sport and commercial catch.

Because the numbers of juvenile chinook salmon cropped at the Pittsburg Power Plant are very small in comparison with other known sources of cropping (e.g., natural mortality) and with the estimated size of the 1978 juvenile population, the impact of cooling water intake operation on the chinook salmon population is concluded to be negligible. Entrainment and impingement cropping does not preclude maintenance of the existing population or result in a detectable reduction in optimum sustained yield to the sport and commercial fisheries.

5.4.1.2 American shad (Alosa sapidissima)

American shad are anadromous and migrate through the Bay and Delta to spawn in the tributary rivers. The only significant involvement of this species with the cooling water intakes is when the larvae and juveniles pass through the Delta during their emigration to the ocean. An estimated 167,000 American shad were entrained and 3,900 impinged at the Pittsburg Power Plant during the one-year period March 1978 - March 1979 (Table 5-3). Survival of entrained American shad is expected to be relatively low (20-40 percent), on the basis of entrainment survival results for clupeids discussed in Section B.3.2.1 (Appendix B). No survival of American shad following impingement is expected, judging from survival data collected at the Pittsburg Power Plant for threadfin shad (Chapter 4).

General trends in American shad abundance, as reflected in CDF&G midwater trawl survey data, are shown in Figure 5-1 for the Sacramento-San Joaquin Delta. Although the data are highly variable during the late 1960s, there appears to be a general decline in the catch of American shad in the Delta. The slope of the linear regression, its standard error, and the associated T-value and probability are as follows:

	Slope	S. E.	T	P
Delta	-0.64	0.26	-2.48	0.03

The statistically significant negative slope suggests that the American shad population has declined during the period 1967-1978. The apparent decline, however, is due in part to high CPUE values for 1967-1969; the population trend appears to be relatively stable between 1970 and 1978 (Figure 5-1). The contribution of American shad to the fish community, expressed as a percentage of the midwater trawl catch, was similar in 1968 and 1978 (Table 5-4), despite the decline in CPUE's.

Female American shad release between 30,000 and 300,000 eggs during spawning, the number depending on the size of the female (Moyle 1976a). The present runs of American shad in the Sacramento River and tributaries have been estimated to contain between 750,000 and 4,000,000 3- to 6-lb. fish (Moyle 1976a) which spawn in the upper reaches of the system. No estimates of the actual size of the larval shad population are available for direct comparison with estimates of the numbers cropped by entrainment (J. Robinson, CDF&G, personal communication, 1981). However, at

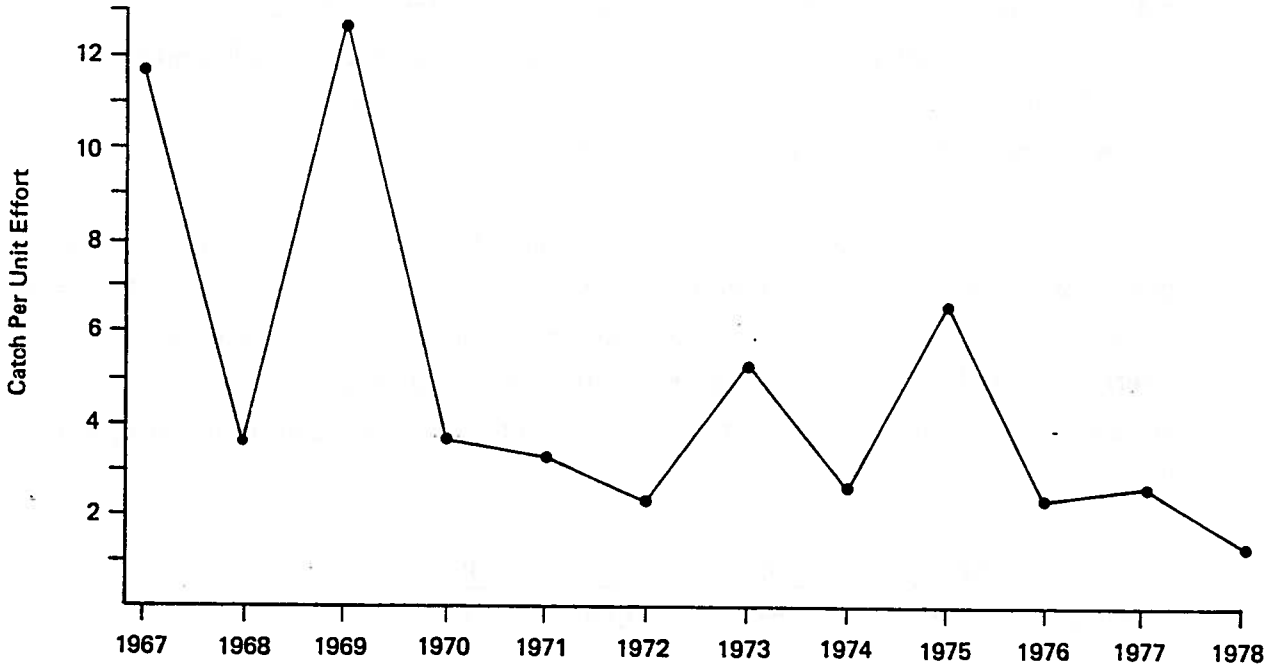


Figure 5-1. Annual average catch of American shad in midwater trawl surveys of the California Department of Fish and Game in the Sacramento—San Joaquin Delta, 1967-1978.

the present size of the adult spawning stock and their high fecundity, the cropping of an estimated 167,000 entrained larvae is likely to represent an insignificant fraction of the larval shad population in the Bay-Delta.

The numbers of American shad cropped by impingement at the plant are small in comparison to the cropping of adults by the sport fishery and to the estimated size of the adult population. The species supports a moderately large sport fishery with annual landings of 110,000-160,000 fish. The adult population size has been estimated at 1.48 million for 1976 and 2.33 million for 1977 (Section 2.2.7.2). If 10 percent of the estimated 3,900 juveniles impinged had instead matured into adults, the equivalent adult loss would have been 390 fish, representing less than 1 percent of the annual sport catch and less than 0.1 percent of the estimated adult population. The assumption of 10 percent survival from juvenile to adult is clearly conservative, because the adult annual survival rate has been estimated at 30 percent, and shad become adults at age 4 or 5. Thus, the equivalent adult loss of American shad represents a negligible fraction of the adult population.

Because the numbers of American shad cropped at the Pittsburg Power Plant are very small in comparison with natural mortality and with the estimated population size, it was concluded that the impact of cooling water intake operation is not a major factor influencing the population of American shad in the Bay-Delta. Entrainment and impingement cropping does not preclude maintenance of the existing population or result in a detectable reduction in optimum sustained yield to the sport fishery.

5.4.1.3 White Sturgeon (Acipenser transmontanus)

White sturgeon is a native anadromous species which spawns in the Sacramento, Feather, and San Joaquin rivers. Larval sturgeon remain in the tributaries and the freshwater areas of the Delta, and are not entrained to any measurable extent at the Pittsburg Power Plant (Table 5-3). Juveniles inhabit Suisun Bay and the western Delta, but generally avoid impingement; only seven juveniles were impinged at the Pittsburg Power Plant between March 1978 and March 1979.

General trends in white sturgeon abundance, based on CDF&G midwater trawl survey data (a relatively ineffective sampling gear for sturgeon), are shown in Figure 5-2. The CPUE's for the Delta reflect variability within a narrow range, and the regression does not indicate a decline:

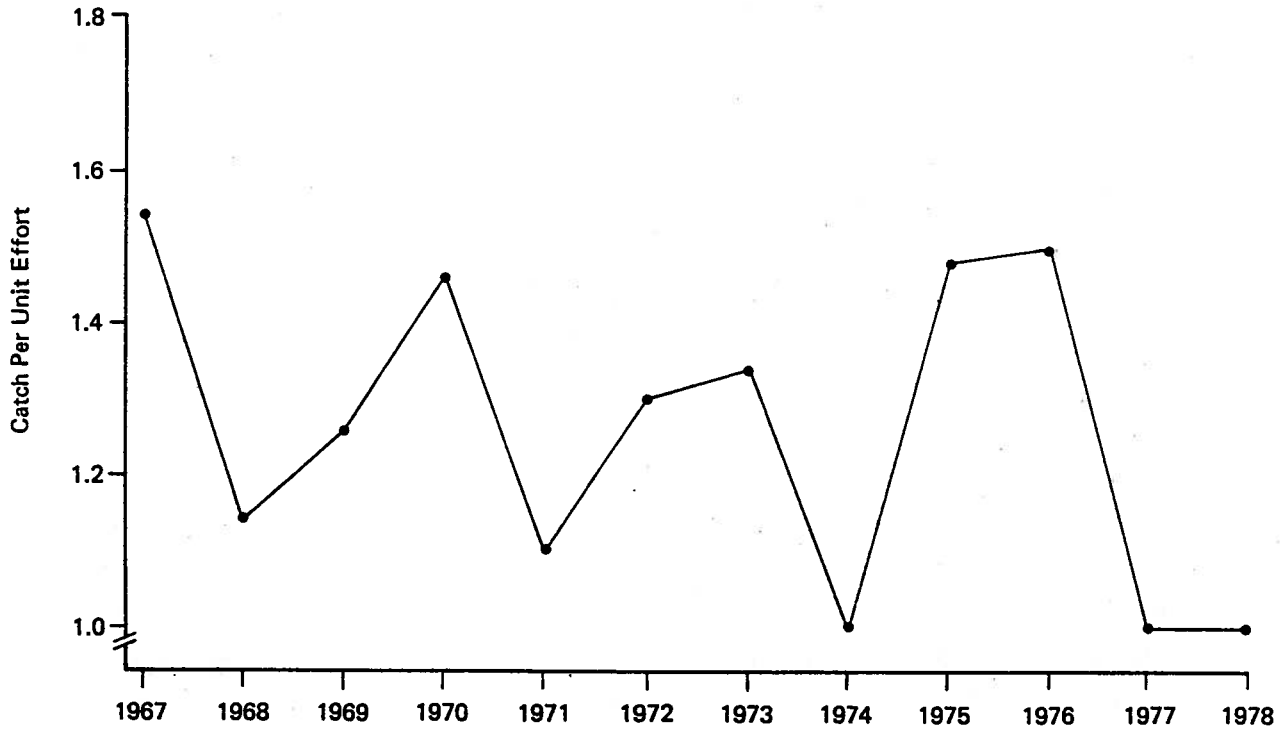


Figure 5-2. Annual average catch of white sturgeon in midwater trawl surveys of the California Department of Fish and Game in the Sacramento—San Joaquin Delta, 1967-1978.

	Slope	S. E.	T	P
Delta	-0.019	0.017	-1.12	0.29

The regression slope was not significantly different from zero, suggesting that the population is relatively stable. The contribution of white sturgeon to the fish community, expressed as a percentage of the midwater trawl catch, also remained stable (Table 5-4) between 1968 and 1978.

Since the Bay-Delta adult stock has fluctuated from approximately 20,000 to 100,000 fish in recent years, and is harvested by a modest sport fishery at an annual rate of 6 to 7 percent (Section 2.2.7.2), the small number of juveniles cropped at the Pittsburg Power Plant (seven individuals) is inconsequential to the adult stock. Thus, it was concluded that cooling water intake operation does not have an adverse impact on the white sturgeon population.

5.4.2 Seasonal and Permanent Residents

Fishes that are seasonal or permanent residents in the vicinity of the Pittsburg Power Plant and were selected for consideration include northern anchovy, Pacific herring, threadfin shad, Delta and longfin smelt, yellowfin goby, prickly sculpin, and starry flounder. A detailed discussion of the abundance, distribution, and life history characteristics of these species is presented in Section 2.2.7.2.

5.4.2.1 Northern Anchovy (Engraulis mordax)

The northern anchovy is a planktivorous, pelagic schooling species found in the Pacific Ocean from Baja California to Alaska (Miller and Lea 1972, 1976). Its population consists of many genetically separate subpopulations in this range (Frey 1971), including one in the area that encompasses San Francisco Bay.

The northern anchovy fishery has become highly intensive, and as a result of the dramatic decline of the sardine, it is the largest (in biomass) commercial fishery in California. The anchovy catch is canned, reduced to oil and meal, or sold as bait (Frey 1971). The San Francisco Bay region catch, ranging from 250 to nearly 520 tons (230-450 metric tons) annually between 1973 and 1976, is generally less than 5 percent of the statewide catch and less than 0.2 percent of the spawning biomass of

the subpopulation north of Point Conception, California. Most of this catch is taken off the coast in the Pacific Ocean and not within San Francisco Bay.

The central spawning stock is estimated at 860,000 tons (780,000 metric tons) for the 1979-1980 season; the northern subpopulation is a small fraction of this stock (J. Hunter, Natl. Mar. Fish. Serv., personal communication, 1979). The allowable catch for the 1979-1980 season was set at 120,000 tons (109,000 metric tons).

Most adult northern anchovy are found in offshore waters, but the essentially planktonic eggs, larvae, and juveniles are abundant in the Bay-Delta. An estimated 120,000 anchovy eggs and 5 million larvae and early juveniles were entrained at the Pittsburg Power Plant during the period March 1978 - March 1979; only 1,300 juveniles and adults were impinged (Table 5-3).

Studies conducted at the Potrero Power Plant on San Francisco Bay (PGandE 1980) suggest that survival of entrained northern anchovy larvae would be approximately 75 percent when discharge temperatures are less than 30 C (86 F) based on observations of Pacific herring. The potential for entrainment survival decreases as discharge temperatures increase above 30 C. Northern anchovy larvae were entrained in greatest abundance between June and August, during which time discharge temperatures at the Pittsburg Power Plant were below 30 C (86 F) 30 percent of the time at Units 1-4 and 20 percent of the time at Units 5 and 6. On the basis of these frequencies of occurrence of high discharge temperatures during the period of peak northern anchovy entrainment, survival of entrained northern anchovy is not expected to be high.

Impingement survival of northern anchovy was characteristically low at the Moss Landing Power Plant (cited in PGandE 1980). Initial and long-term (96-hour) survival of 2,643 northern anchovy were 16 and 1 percent, respectively after impingement on intake screens operated in a manner similar to those at the Pittsburg Power Plant. Survival was not improved by decreasing the duration of impingement.

Because of the frequency of discharge temperatures in excess of 30 C (86 F) during the period of peak northern anchovy entrainment and the low survival of impinged anchovy, it is assumed that entrained or impinged anchovy would not survive. Nevertheless, they are returned to the Bay-Delta, where they provide a food resource for other fish and invertebrates.

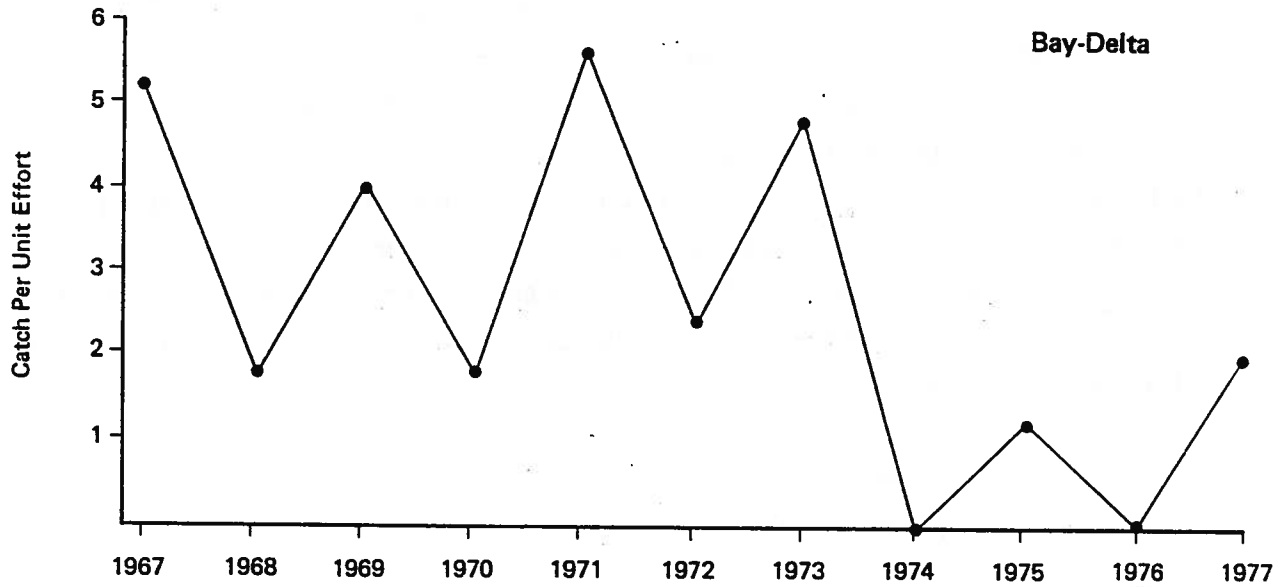
General trends in northern anchovy abundance, based on CDF&G midwater trawl survey data, are shown in Figure 5-3. Because of the wide distribution of northern anchovy in San Francisco Bay and the Delta, catch statistics are presented for the entire Bay-Delta and the Delta separately. Although the CPUE's are highly variable for the entire Bay-Delta, there is a general decline in catches over the 12-year survey period. The catch of northern anchovy from the Delta is considerably lower and less variable than that for the entire Bay-Delta. The abundance of northern anchovy in the Delta increased from a relatively low level in 1967 to a peak in 1969 that was followed by a steady decline, reaching a minimum in 1974, when no anchovy were collected in the Delta midwater trawl survey (Figure 5-3). The Delta catch increased to a second peak in 1977 and subsequently returned to zero in 1978. The slope of the linear regression was negative and significant for the Bay-Delta as a whole but not for the Delta:

	Slope	S. E.	T	P
Bay-Delta	-34.34	13.86	-2.48	0.03
Delta	- 1.77	1.51	-1.17	0.27

This suggests that the population as a whole declined from 1967 to 1978. The regression slope for the Delta catch was not significantly different from zero, suggesting that although the portion of the population using that part of the estuary has fluctuated during the survey period, there is no general decline in abundance in the Delta near the Pittsburg Power Plant.

The contribution of the northern anchovy to the fish community, expressed as a percentage of the midwater trawl collection, declined markedly between 1968 and 1978, both for the entire Bay-Delta (from 74 to 37 percent) and for the Delta portion (from 33 to 0 percent). There has been a decline in the relative abundance of northern anchovy in the fish community sampled by the midwater trawl.

Fishery surveys conducted in the vicinity of the Pittsburg Power Plant (EA 1981b) and in the lower Delta (EA 1981c; PGandE unpublished a) and the CDF&G midwater trawl surveys of the entire Bay-Delta indicate that most of the Bay-Delta northern anchovy population is located in the area from Carquinez Strait to central San Francisco Bay, and that abundances are low in Suisun Bay and the western Delta. This fact, a natural consequence of the low salinities in that part of the estuary (Section 2.2), limits the involvement of the northern anchovy population with the Pittsburg Power



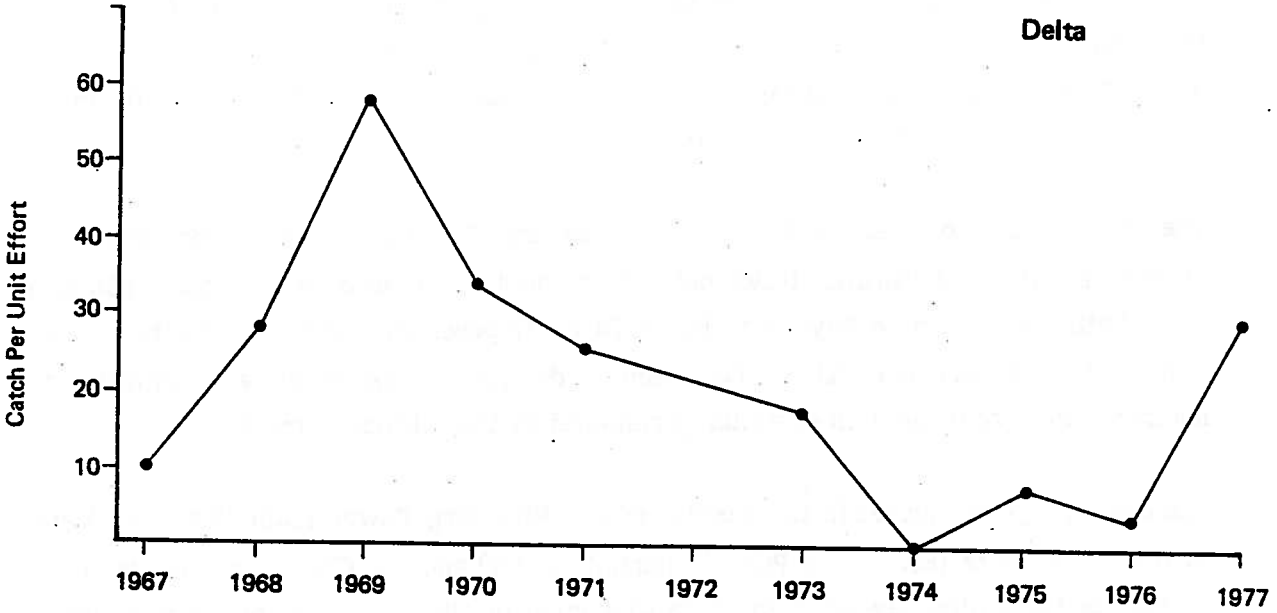


Figure 5-3. Annual average catch of northern anchovy in midwater trawl surveys of the California Department of Fish and Game in San Francisco Bay and the Sacramento—San Joaquin Delta, 1967-1977.

Plant and reduces the plant's potential for impact on the population.

Because of the general decline in the catch of northern anchovy in the Bay-Delta midwater trawl survey and the decline in the relative contribution of northern anchovy to the sampled fish community, the possible effect of entrainment and impingement cropping at the Pittsburg Power Plant on the population was estimated using a quantitative approach, described by Goodyear (1978), known as the equivalent adult model. This technique equates the numbers of early life stages cropped with the number of adults that would have been likely to mature from the young lost (see Section C.1, Appendix C). This method of analysis depends on the use of estimates of survival from the young stages to maturity. The estimate of equivalent adult losses can then be compared to either estimates of the total adult population or to catch statistics from a commerical or sport fishery.

The estimated equivalent adult loss was approximately 10,000 (Table 5-5; Appendix C). At an average weight of 0.0185 kg/fish, the equivalent adult loss amounts to 185 kg (407 lb) of fish. From 1972 to 1976 San Francisco Bay region annual landings averaged 153,000 kg (700,000 lb) (Section 2.2.7.2). Thus, in terms of relative cropping, the Pittsburg Power Plant equivalent adult loss is less than 0.06 percent of the annual cropping in the San Francisco Bay region by commercial fishermen.

An additional indication of the relative magnitude of cropping attributed to the Pittsburg Power Plant (equivalent adult loss approximately 407 lb) is given by a report of 18,000 lb of northern anchovy being caught in the San Francisco Bay bait fishery in just 40 minutes (S. F. Examiner & Chronicle, 1980)—after 12,000 lb were netted by the same bait fishermen in the morning. Although collections of this magnitude are not common in the San Francisco Bay bait fishery, they do provide a perspective on the cropping of the population by the Pittsburg Power Plant.

The immature life stages and adults of northern anchovy cropped by the plant are returned to the aquatic food chain, whereas the fish cropped by fishing are removed from the system.

Because of the widespread distribution and abundance of northern anchovy and the minor extent of the cropping occurring at the Pittsburg Power Plant, the impact of cooling water system operation on the Bay-Delta northern anchovy population is concluded to be negligible. The numbers cropped by entrainment and impingement do

TABLE 5-5 EQUIVALENT ADULT LOSSES[a] OF NORTHERN ANCHOVY AND PACIFIC HERRING ENTRAINED AND IMPINGED AT THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Source of Loss	Northern Anchovy		Pacific Herring	
	Number Lost	Equivalent Adult Loss	Number Lost	Equivalent Adult Loss
Entrainment				
eggs	120,000	24	0	0
larvae, juveniles	5,054,000	8,636	21,837,000	156,685
Impingement				
juveniles-adults	1,331	1,331	49	31
Total		9,991		156,716

(a) See Section C.1, Appendix C for details of the methodology needed in estimating equivalent adult losses.

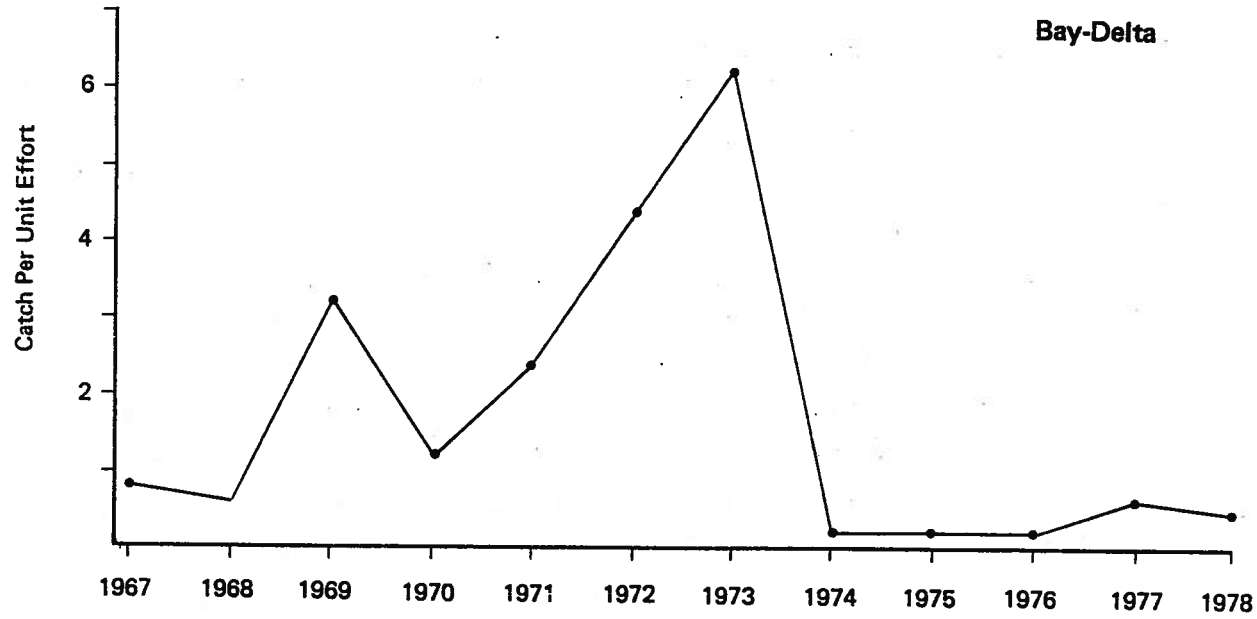
not represent an important source of mortality contributing to the apparent decline in the Bay-Delta northern anchovy population.

5.4.2.2 Pacific Herring (Clupea harengus pallasi)

The Pacific herring is a planktivorous, pelagic, schooling species found in the Pacific Ocean from Baja California to Alaska (Miller and Lea 1972, 1976). The Pacific Ocean spawning stock of Pacific herring has recently been estimated at 20,000 tons (18,000 metric tons) (Spratt 1976). Several spawning runs enter San Francisco Bay between December and March. The Bay spawning stock supports a considerable fishery, which varies from year to year depending on the demand. In recent years there has been a high demand from the Japanese market for roe. Recent catches in the San Francisco Bay region have ranged from 1,000 to 5,000 tons (900-4,500 metric tons) annually, most of it in San Francisco Bay, but up to 500 tons (450 metric tons) from Tomales Bay (Spratt 1976).

Pacific herring spawn in the central Bay and San Pablo Bay, during the winter months, and the larvae and juveniles are common in the Bay-Delta until midsummer, when most juveniles migrate to the ocean (Section 2.2.7.2). Eggs are demersal and adhesive and are not entrained. Larvae and juveniles are planktonic, and were entrained at the Pittsburg Power Plant in or shortly after the spawning season. An estimated 22 million were entrained between March 1978 and March 1979 (Table 5-3). Because juvenile and adult Pacific herring reside in high-salinity areas of San Francisco Bay, they have a low probability of being impinged: only 49 herring were estimated to have been impinged at the plant. Furthermore, none were collected in a one-year fishery survey in the area (EA 1981b). Any potential impact of operation of the plant on the Pacific herring population would be the result of entrainment of larval and early juvenile fish.

The general trend in Pacific herring abundance shown by CDF&G midwater trawl survey data for both the Bay-Delta and the Delta can be seen in Figure 5-4. The CPUE's for both the entire Bay-Delta and the Delta, variable between years and having no clear pattern, do not suggest that the populations are declining. The CPUE's for the Delta region are substantially lower than those for the entire Bay-Delta, which is in agreement with results of the Pittsburg Power Plant impingement monitoring program (Chapter 4) and fishery survey (EA 1981b). In both the Bay-Delta and Delta regions, however, there was a general increase in catch in



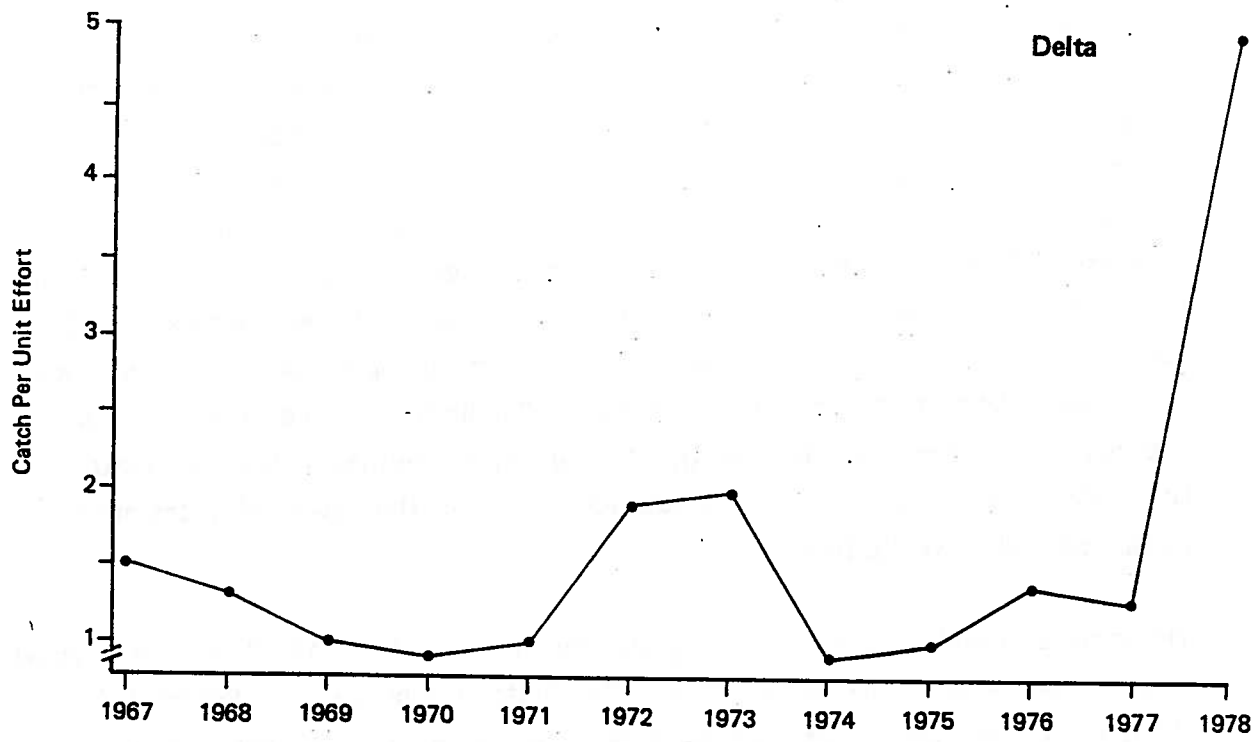


Figure 5-4. Annual average catch of Pacific herring in midwater trawl surveys of the California Department of Fish and Game in San Francisco Bay and the Sacramento—San Joaquin Delta, 1967-1978.

1972 and 1973 followed by a decline in 1974. The catch increased substantially in the Delta region in 1978, but no increase was evident for the Bay-Delta as a whole.

The linear regression slopes for both data sets are shown below:

	Slope	S. E.	T	P
Bay-Delta	-1.31	1.67	-0.78	0.45
Delta	0.15	0.10	1.55	0.16

The slopes were not significantly different from zero for either regression, suggesting that although the catches fluctuated between years the population is not declining. The contribution of Pacific herring to the fish community, expressed as a percentage of the midwater trawl catch, has increased in both the Bay-Delta and Delta regions between 1968 and 1978 (Table 5-4). There is no evidence from the general population trends of Pacific herring reflected in the CDF&G midwater trawl data to suggest that entrainment cropping at the Pittsburg Power Plant is contributing to a decline in the Bay-Delta herring population.

Because of the commercial importance of this species in San Francisco Bay, an equivalent adult model was used to assess the significance of cropping by the plant (Appendix C). Assuming 100 percent entrainment mortality, the Pacific herring equivalent adult loss was estimated at 157,000 fish (Table 5-5). At an average weight of 0.0708 kg/fish, this amounts to 11,100 kg (24,500 lb). In comparison, the San Francisco Bay commercial catch since 1974 has averaged 1 million kg (2.3 million lb), and the estimated total Bay spawning biomass is 23 million kg (51 million lb) (Section 2.2.7.2). Thus, in terms of relative cropping, the equivalent adult loss of Pacific herring at the Pittsburg Power Plant amounted to approximately 1 percent of the average annual cropping in San Francisco Bay by commercial fishermen and less than 0.05 percent of the estimated Bay spawner biomass. The commercial landing of Pacific herring from San Francisco Bay in 1978 was 4.5 million kg (10 million lb.) (W. Maxwell, CDF&G, personal communication, 1981). Equivalent adult loss of Pacific herring was approximately 0.2 percent of the 1978 commercial landing.

In entrainment survival studies carried out at the Potrero Power Plant in 1978-1979, the estimated proportion not surviving entrainment was 0.21 (i.e., 21 percent entrainment mortality) (PGandE 1980). Discharge temperatures were uniformly less than 30 C (86 F) at the time of herring entrainment, and it was concluded that

thermal stresses were not likely to be a significant cause of entrainment mortality. Most of the mortality was attributed to mechanical stresses. (A more detailed discussion of the survival of entrained organisms is presented in Appendix B.)

Pacific herring were entrained in greatest abundance at the Pittsburg Power Plant from November to February, during which time discharge temperatures were less than 30 C (86 F) 99 percent of the time at Units 1-4 and 98 percent of the time at Units 5 and 6. On the assumption that entrainment survival is as high at the Pittsburg Power Plant as at the Potrero Power Plant (80 percent), the equivalent adult loss was estimated at 31,300 fish, amounting to 2,200 kg (4,900 lb). This corresponds to 0.2 percent of the average annual San Francisco Bay commercial catch and less than 0.01 percent of the estimated Bay spawner biomass. It is only 0.05 percent of the 1978 San Francisco Bay commercial landings.

The immature life stages of Pacific herring cropped by the Pittsburg Power Plant are returned to the aquatic food chain.

Because of the abundance and wide distribution of Pacific herring and the minor extent of the cropping occurring at the Pittsburg Power Plant, the impact of cooling water system operation on the Bay-Delta Pacific herring population is concluded to be negligible. Entrainment and impingement cropping does not preclude maintenance of the existing population or result in a detectable reduction in yield to the commercial fishery.

5.4.2.3 Threadfin Shad (Dorosoma petenense)

The threadfin shad is a small (to 10 cm; 4 in.) schooling fish recently introduced into California reservoirs as forage for game fishes (Kimsey 1954). It has since become well-established in the freshwater and oligohaline (0.5-5 ppt salinity) portions of the Bay-Delta. During late summer and early fall, threadfin shad was the most abundant fish collected by CDF&G midwater trawls in the Delta (CDF&G unpublished data).

Entrainment of larval threadfin shad at the Pittsburg Power Plant was estimated to be 675,000 fish (Table 5-3). Although no quantitative data are available on the survival of entrained threadfin shad larvae, results of studies conducted on a variety of other species (Appendix B) suggest that survival of larval fish entrained into once-through cooling systems is generally high (approximately 75-80 percent survival)

when discharge temperatures are less than 30 C (86 F). At the Pittsburg Power Plant, threadfin shad were entrained in greatest abundance from March through May, when discharge temperatures were less than 30 C (86 F) 99 percent of the time. On the basis of these discharge temperatures, it is expected that a large proportion of the threadfin shad will survive entrainment at the Pittsburg Power Plant.

Impingement of threadfin shad occurred primarily from October to March, with a peak from late December through January that coincided with ambient water temperatures below 10 C (50 F). An indication of the sensitivity of threadfin shad to low water temperatures is reported by Turner (1966), who states that threadfin shad populations experience high natural mortality every winter when water temperatures decline below 6-8 C (43-46 F). Therefore, although the number of threadfin shad impinged at Pittsburg Power Plant is relatively high during the winter (Figure 4-7), it appears that the population is naturally stressed by low water temperatures and that many of the fish impinged at that time might have died as a result of cold temperatures even if they had not been impinged. The total estimated impingement of threadfin shad during the period March 1978 - March 1979 was 88,900 fish.

General trends in threadfin shad abundance, based on CDF&G midwater trawl survey data, are shown for the Delta region in Figure 5-5. The catch statistics for the Delta showed a general decline between 1967 and 1974, followed by an increase in abundance that peaked in 1977. The regression slope was not significantly different from zero:

	Slope	S. E.	T	P
Delta	-1.95	1.06	-1.84	0.10

These data suggest two important aspects of the threadfin shad population. First, although the population apparently declined between 1967 and 1974, it recovered quickly, suggesting a high degree of resiliency. Second, it is unlikely that if the decline had been due to entrainment and impingement cropping at the Pittsburg Power Plant, the population would have increased between 1974 and 1977 despite the continued operation of all units of the plant throughout this period (Section 2.1). Operation of the Pittsburg Power Plant increased slightly during the mid-1970s: Units 1-6 were in operation throughout the period 1968-1978, and Unit 7 began operation in 1972. These facts would suggest that operation of the Pittsburg Power

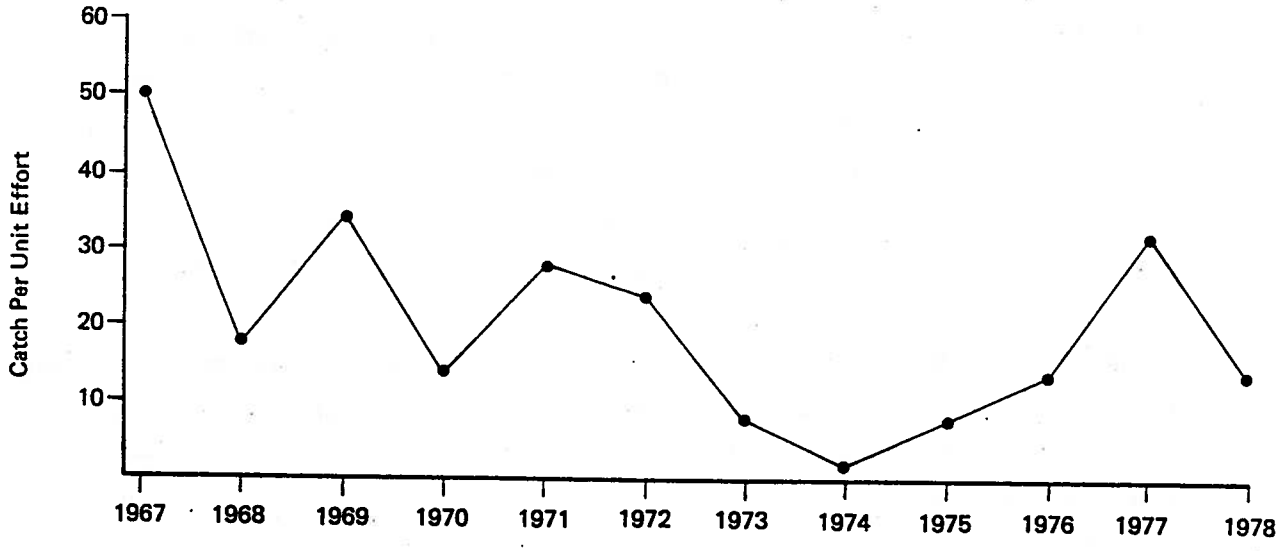


Figure 5-5. Annual average catch of threadfin shad in midwater trawl surveys of the California Department of Fish and Game in the Sacramento—San Joaquin Delta, 1967-1978.

Plant cooling system has not resulted in a significant decline in the threadfin shad population.

The percentage contribution of threadfin shad to the Delta fish community, increased from 23 percent in 1968 to 36 percent in 1978 (Table 5-4), again indicating that entrainment and impingement cropping at the Pittsburg Power Plant did not have a detrimental impact on the population.

Because the specific data are not available, it is not possible to assess the impact of cooling water intake system operation on this species quantitatively. However, several observations suggest that the cropping occurring at the Pittsburg Power Plant is not damaging the population to any measurable extent. The species was introduced into the Delta only about 25 years ago, and since then it has become well established, very abundant, and fairly widely distributed (Section 2.2.7.2). Power plant cropping occurred throughout this period. In the absence of population estimates, the CDF&G midwater trawl survey data for 1967-1978 (Figure 5-5) provide the best available indicator of the general abundance of threadfin shad and recent trends in its abundance. Catches in the vicinity of the Pittsburg Power Plant show no indication of declining abundance over the period 1967-1978, suggesting that the threadfin shad population has not been depressed, despite cropping by the Pittsburg Power Plant. It is concluded that operation of the Pittsburg Power Plant does not have an adverse impact on the population based on criteria presented in Section 5.1.

5.4.2.4 Delta Smelt (Hypomesus transpacificus) and Longfin Smelt (Spirinchus thaleichthys)

The Delta smelt and longfin smelt are abundant schooling fishes of the low-salinity and freshwater areas of the Bay-Delta. Although their ranges overlap in Suisun Bay and the western Delta, Delta smelt are most abundant from eastern Suisun Bay to the lower reaches of the Sacramento and San Joaquin rivers, and longfin smelt predominate from San Pablo Bay to the western Delta (Section 2.2.7.2). Both species spawn in the Delta, and produce adhesive eggs and planktonic larvae. The eggs, being adhesive, are not susceptible to entrainment, but more than 50 million larvae of the two species combined were estimated to have been entrained at the Pittsburg Power Plant in 1978-1979 (Table 5-3). Juveniles are impinged in large numbers also: impingement estimates were 105,000 longfin smelt and 11,000 Delta smelt during the one-year period (Table 5-3).

General trends in Delta smelt abundance, based on the midwater trawl data, are shown for the Delta in Figure 5-6. The catch decreased between 1967 and 1969, increased markedly in 1970, and then declined to a minimum for the survey period in 1978. Nevertheless, the linear regression slope was not significantly different from zero:

	Slope	S. E.	T	P
Delta	-0.12	0.17	-0.72	0.49

As with threadfin shad (Section 5.4.2.3), the quick recovery in 1970 suggests a high degree of resiliency in the population. Units 1-6 were in operation throughout the period 1967-1978, and it is unlikely that if entrainment and impingement cropping at the plant had a major impact on the Delta smelt it could have increased so markedly between 1969 and 1970 and maintained itself through 1973. The absence of a significant decline in CDF&G midwater trawl survey catches suggests a lack of adverse effects of cropping at the power plant. It was concluded that entrainment and impingement cropping are not precluding maintenance of the existing Delta smelt population.

In contrast to Delta smelt, catches of longfin smelt in the midwater trawl survey decreased considerably in the Delta between 1967 and 1978 (Figure 5-7). Catches of longfin smelt from Carquinez Strait to the Delta during the years 1974-1978 were approximately 90 percent less than catches in 1967-1969.

Because the relevant population data are not available, it is not possible to assess the impact of cooling water intake operation on the apparent decline in the longfin smelt population quantitatively. However, several important qualitative observations have been made which are useful in examining the potential impact of the Pittsburg Power Plant on the population. The primary involvement of smelt with the plant results from entrainment of larval and juvenile fish—the impingement of juveniles and adults being secondary. Smelt eggs are demersal and adhesive, so they are not susceptible to entrainment. Larvae and juveniles are planktonic and are entrained, with peak abundances occurring in January, February, and March (see Figure 3-7). Although no quantitative data are available on the survival of entrained smelt larvae at the Pittsburg Power Plant, results of studies conducted on a variety of other species (Appendix B) suggest that survival of larval fish entrained in once-through cooling water systems is generally high (75-80 percent) when discharge temperatures are less

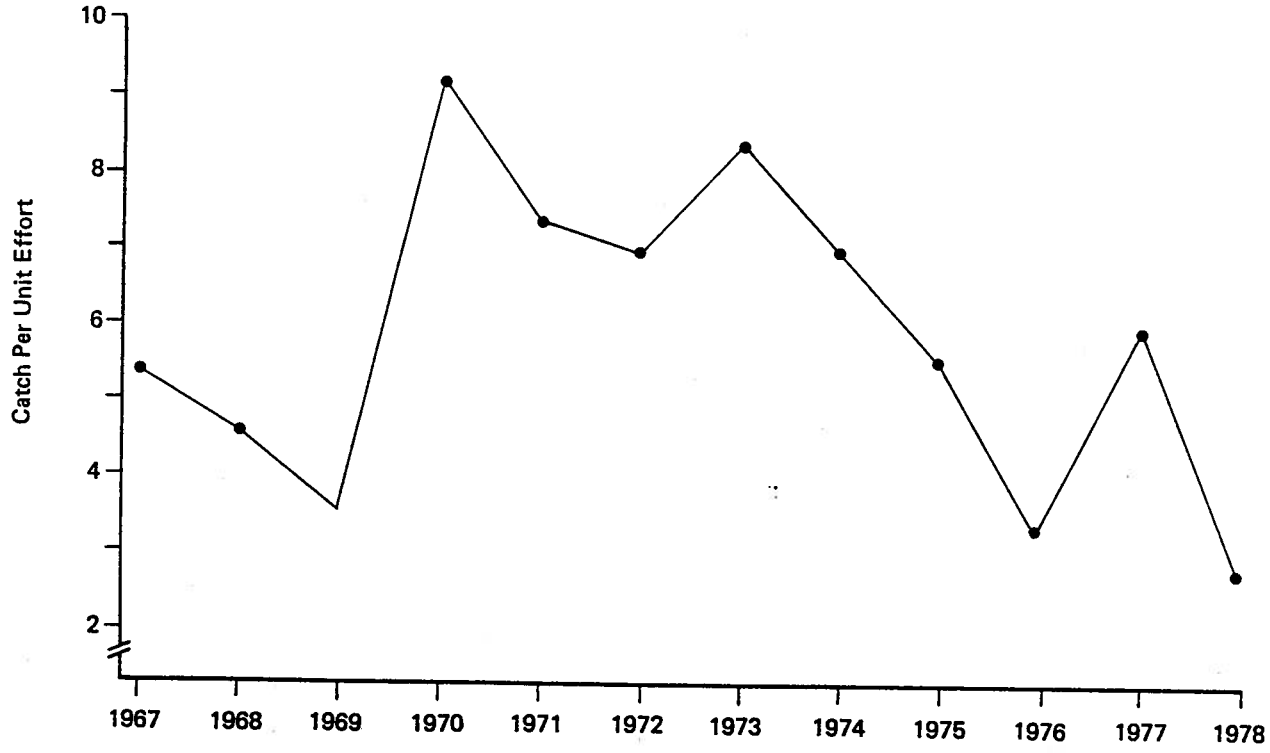


Figure 5-6. Annual average catch of Delta smelt in midwater trawl surveys of the California Department of Fish and Game in the Sacramento —San Joaquin Delta, 1967-1978.

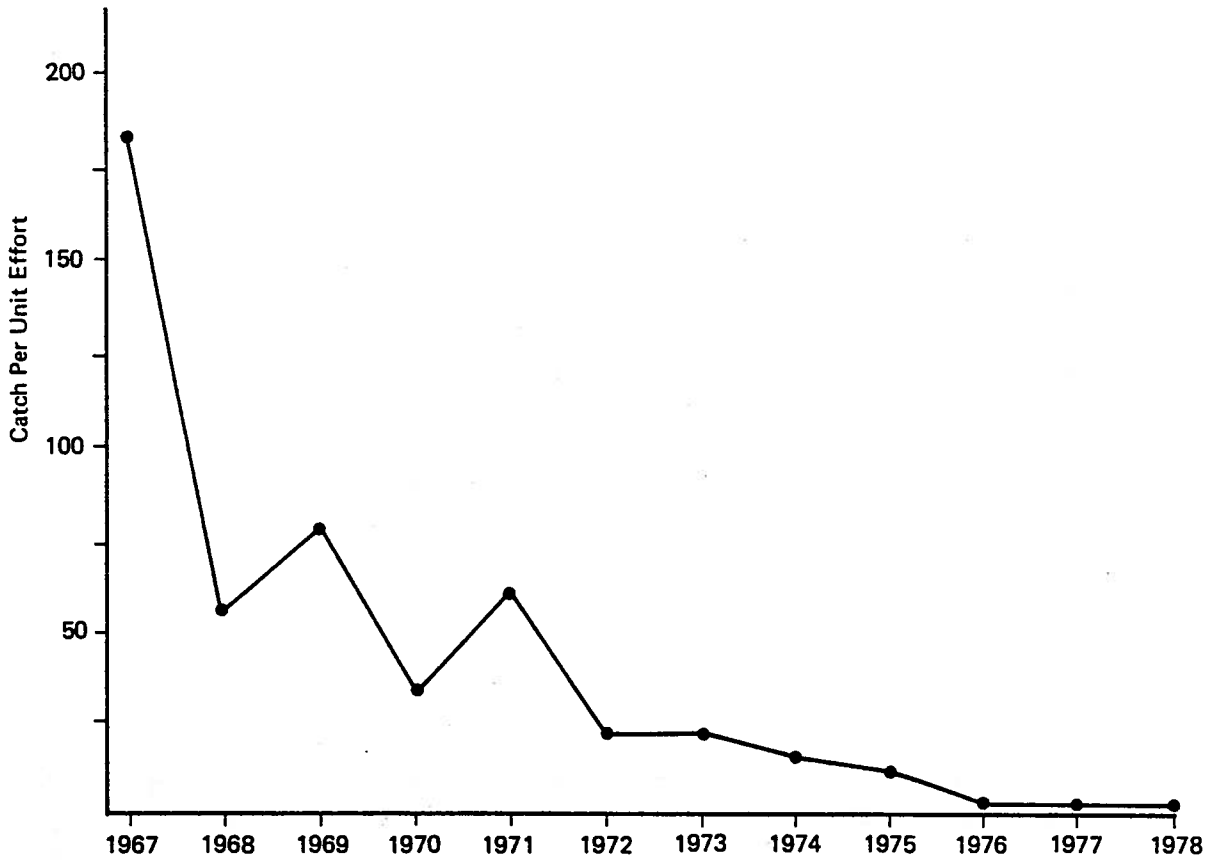


Figure 5-7. Annual average catch of longfin smelt in midwater trawl surveys of the California Department of Fish and Game in the Sacramento—San Joaquin Delta, 1967-1978.

than 30 C (86 F). During the period of greatest entrainment at the plant, discharge temperatures were less than 30 C (86 F) 100 percent of the time. It is therefore expected that the survival of entrained smelt is high at the Pittsburg Power Plant, which would substantially reduce entrainment cropping.

Although the catch statistics from the CDF&G midwater trawl surveys indicate that the relative abundance of longfin smelt in Suisun Bay and the western Delta has declined significantly since 1967, two additional important trends are evident in the midwater trawl survey data for the entire Bay-Delta. First, longfin smelt are widely distributed through the system, and a large proportion of the fish were collected in areas where salinities are more than 10 ppt. This apparent preference for higher-salinity waters is consistent with preferences reported by Moyle (1976a), and it contributes substantially to reduced susceptibility of longfin smelt to impingement at the Pittsburg Power Plant, where salinities are typically below 10 ppt. In addition, the CPUE's for longfin smelt declined not only near the plant (Figure 5-7), but also throughout the Bay-Delta (CDF&G unpublished data). The general decline in relative abundance reflected in the CPUE's for the entire Bay-Delta suggests a general change in the longfin smelt population abundance. Such a large-scale change in a widely distributed species is not likely to be the result of a localized point-source effect such as entrainment and impingement cropping at the Pittsburg Power Plant. Other factors which may be contributing to this change include increased predation on planktonic larvae and competition for food resources by introduced species. The introduction in the past several decades of fishes and macroinvertebrates like threadfin shad, yellowfin goby, inland silverside, and Palaemon macrodactylus may be related to the apparent decline in the longfin smelt population. Many of these introduced species, particularly yellowfin goby, prey on smelt larvae and early juveniles (and adults as well). These introduced species compete with longfin smelt for copepods, neomysis, and other crustaceans, and this may be contributing to a change in the dynamics of the Bay-Delta longfin smelt population. Populations of these recently introduced fish and macroinvertebrates have expanded rapidly, perhaps at the expense of native species like longfin smelt.

The Bay-Delta longfin smelt population is at the southern extreme of the geographic distribution of the species, and thus it can be speculated that slight changes in water temperature could have a major effect on population abundance. The change in Delta hydrology that has accompanied increased water exports and water management (Section 2.2.1) may have contributed to a general increase in ambient water

temperatures, for example, which might potentially result in a substantial decline in the population. Unfortunately, the necessary long-term data are unavailable to quantify the effect of changes in the Bay-Delta system on potentially sensitive species. For a widely distributed species such as longfin smelt, it is likely that a decline in the relative abundance of the entire population is the result of a large-scale change in the carrying capacity of the system (such as that which would be expected from increased competition, predation, or a change in environmental conditions), and not the result of direct cropping at any point source such as the Pittsburg Power Plant, agricultural diversions, or major export facilities.

Although the causes of the apparent decline in the longfin smelt population remain uncertain, it was concluded that cropping at the Pittsburg Power Plant resulting from entrainment and impingement is not likely to be a major factor adversely affecting the longfin smelt population. Because the effect of the plant is primarily the result of entrainment of larvae and early juveniles, which are abundant and widely distributed, and the survival of entrained larvae is expected to be high, the potential impact of cooling water intake system operation on the longfin smelt population is substantially reduced. Furthermore, the general preference of juvenile and adult longfin smelt for areas with salinities higher than those typically occurring at the Pittsburg Power Plant also contributes to reduced susceptibility to impingement.

5.4.2.5 Yellowfin Goby (Acanthogobius flavimanus)

The yellowfin goby is a recently introduced Asian estuarine fish found in benthic and epibenthic habitats throughout much of the Bay-Delta, from the south Bay to the upper reaches of the Delta (Brittan et al. 1970; Moyle 1976a). The species is also found in Elkhorn Slough, Tomales Bay, and Los Angeles Harbor (Miller and Lea 1972, 1976; Haaker 1979). It is the largest goby in California (up to 25 cm; 10 in.) and has rapidly expanded in population size and distribution since its accidental introduction (Brittan et al. 1970).

Yellowfin goby spawn in the high-salinity areas and lay adhesive eggs in burrows, and are therefore not susceptible to entrainment at the Pittsburg Power Plant. The larvae become planktonic, and migrate to areas of lower salinity. Entrainment estimates at the Pittsburg Power Plant were 6 million larvae for the study period (Table 5-3). Yellowfin goby juveniles are also impinged in fairly high numbers, an estimated 32,000 (Table 5-3).

Yellowfin goby, like other members of the family Gobiidae, are typically epibenthic, and hence their abundance and general population trends are not accurately reflected in midwater trawl collections. Moyle (1976a) notes that although no precise estimates of population abundance are available for yellowfin goby, the population has increased rapidly since it was first collected in 1963, and yellowfin goby are now one of the most abundant bottom fish in the Bay-Delta. Their relative abundance is reflected in fishery studies conducted near the Pittsburg Power Plant between August 1978 and July 1979 (EA 1981b): yellowfin goby were the fifth most abundant fish collected, constituting almost 6 percent of the total catch.

The rapid expansion of the population since its accidental introduction into the Bay-Delta from Japan suggests that the effects of cropping are inconsequential to the adult stock. Females of this species are more fecund than those of native gobies, producing as many as 18,000 eggs—about 10 times as many as other gobies (EA unpublished data a), and it tolerates wide fluctuations in temperature and salinity, giving it a competitive advantage over native species (Section 2.2.7.2). The direct impact of the Pittsburg Power Plant on the goby population and the ecological consequences of their entrainment and impingement was concluded to be minimal, because of their overall abundance, the rapid expansion of their population since their introduction, and their wide distribution in the Bay-Delta. Cooling water intake operation is therefore concluded not to have an adverse impact on the yellowfin goby population.

5.4.2.6 Prickly Sculpin (Cottus asper)

Prickly sculpin is a small (to 17 cm; 7 in.) bottom fish, common in inland freshwater habitats in the Delta and also present in the oligohaline (0.5-5 ppt) portions of Suisun Bay (Ganssle 1966; Miller and Lea 1972, 1976). Prickly sculpin mature at 2-4 years and may spawn for several years thereafter. Adults move downstream into the tidal areas of the Delta to spawn from November to June. Most spawning occurs from March to May (Turner 1966; Moyle 1976a). Females produce from 300 to 11,000 adhesive eggs and deposit them in nests prepared by the males (Patten 1971; Moyle 1976a), and so eggs are not susceptible to entrainment. After the eggs are hatched, most adults return to upstream areas.

Newly-hatched larvae remain in the nest several days before becoming planktonic. Larvae are abundant in the western Delta and Suisun Bay during winter and spring;

the highest densities (up to $0.4/m^3$) in entrainment samples at the Pittsburg Power Plant were observed from March to May. Prickly sculpin larvae were found throughout the Delta and Suisun Bay in spring (EA 1981a). Juveniles remain in the tidal portions of the Delta until early summer of the year following their birth, when they ascend the tributaries of the Delta.

An estimated 30 million planktonic larvae and early juveniles were entrained at the Pittsburg Power Plant during 1978-1979 (Table 5-3). Juveniles and adults reside on the bottom in the tidal portions of the Delta, and were not impinged to any great extent (Table 5-3).

Although no quantitative data are available on the survival of entrained prickly sculpin larvae, results of studies conducted on a variety of other species (Appendix B) show that survival of larval fish entrained into once-through cooling systems is generally high (approximately 75-80 percent survival) when discharge temperatures are less than 30 C (86 F). Prickly sculpin were entrained in greatest abundance at the plant from March through May, when discharge temperatures were less than 30 C (86 F) 99 percent of the time. It is therefore concluded that the survival of entrained prickly sculpin is high at the Pittsburg Power Plant, which would substantially reduce the numbers cropped by entrainment.

Population data for prickly sculpin are lacking, so the quantitative approach to impact assessment could not be used. Because juveniles and adults remain on the bottom, the CDF&G midwater trawl survey (1967-1978) provides no indication of recent trends in abundance. However, because the organisms cropped by the Pittsburg Power Plant are mainly larvae and because the larvae are very abundant and widely distributed in the Bay-Delta, the impact of cooling water intake system operation on the prickly sculpin population is concluded to be low. Entrainment and impingement cropping of prickly sculpin does not preclude maintenance of the existing population.

5.4.2.7 Starry Flounder (Platichthys stellatus)

The starry flounder is a flatfish of moderate size (up to 90 cm; 36 in.) that is common in the Bay-Delta. It is unique among the flatfishes in that it is capable of living in fresh water for a portion of its life. Starry flounder were collected in the northern Bay and the Delta from 1967 to 1978 by midwater trawl (an ineffective sampling gear

for flatfish, however), mainly from San Pablo Bay to Suisun Bay and Montezuma Slough (CDF&G unpublished data). They are also resident in coastal waters and in the predominantly marine bays.

Starry flounder mature at age two (males) or age three (females), and move inshore to spawn from November to February (Orcutt 1950) in coastal waters and perhaps in high-salinity waters of the Bay as well. No starry flounder eggs were identified collected during 1978-1979 in entrainment samples at the Pittsburg Power Plant. The females are highly fecund, each freely broadcasting several million planktonic eggs (Orcutt 1950).

Starry flounder larvae remain planktonic, and they are dispersed throughout the Bay-Delta. Larvae may use tidal currents to ascend the estuary from the spawning grounds. Larvae have been observed in Richardson Bay during March and August (Eldridge 1977) and in Suisun Bay, Montezuma Slough, and the lower Sacramento and San Joaquin rivers from April (the time of the first survey) to mid-July (EA 1981a).

Larvae metamorphose into juveniles at a length of about 11 mm (Orcutt 1950) and gradually settle on the bottom. Juveniles are abundant in the western Delta (Radtke 1966), and some have even appeared in the San Luis Reservoir, which is filled by Delta water transported by the California Aqueduct (Moyle 1976a).

Entrainment estimates for the planktonic larvae at the Pittsburg Power Plant in 1978-1979 were moderate (700,000). The demersal juveniles were impinged in fairly low numbers (12,000) during the same period (Table 5-3).

Although no quantitative data are available on the survival of entrained starry flounder larvae, results of studies conducted on a variety of other species (Appendix B) suggest that survival of larval fish entrained into once-through cooling systems is generally high (approximately 75-80 percent survival) when discharge temperatures are less than 30 C (86 F). Starry flounder were entrained in greatest abundance at the Pittsburg Power Plant from April through June, when discharge temperatures were less than 30 C (86 F) 65 percent of the time at Units 1-4 and 93 percent of the time at Units 5 and 6. On the basis of these discharge temperatures, it is expected that the survival of entrained starry flounder was moderately high at the Pittsburg Power Plant, which would reduce the numbers cropped by entrainment.

Considering the very high fecundity of the species (Section 2.2.7.2) and the broadcast spawning strategy, the natural mortality of the early life stages is likely to be very high. The 700,000 larvae cropped by entrainment probably represent an equivalent adult loss of just a few individuals.

On the conservative assumption that 25 percent of the juveniles impinged would have matured into adults at an average weight of 0.9 kg, the equivalent adult loss for the Pittsburg Power Plant would have amounted to 2,700 kg (5,950 lbs). By comparison, the commercial landings in the San Francisco Bay region in recent years have averaged 109,000 kg (240,000 lbs) (Section 2.2.7.2). Thus, the estimated impingement equivalent adult loss is approximately 2 percent of the annual cropping by commercial fisherman. A substantial sport fishery for starry flounder exists as well, decreasing the proportion of the total cropping of starry flounder attributable to impingement.

General trends in starry flounder abundance, based on CDF&G midwater trawl survey data (a relatively ineffective sampling gear for flounder) are shown in Figure 5-8 for the Delta region. Catches were generally stable between 1967 and 1973, with a marked increase to a peak in abundance in 1974 and a decline in 1975. No starry flounder were collected in the Delta midwater trawl survey in 1976. The catch began to increase again in 1977 and 1978. The catch data from the Delta provide no suggestion of a declining trend in starry flounder abundance. The linear regression slope shown below was not significantly different from zero:

	Slope	S. E.	T	P
Delta	0.14	0.20	0.70	0.50

The pattern of abundance in Figure 5-8 indicates that under certain circumstances the starry flounder population may undergo very rapid and substantial increases in abundance, a pattern typical of a species with high fecundity. If entrainment of larval starry flounder or impingement of juveniles were suppressing the population, it would be reasonable to assume that the increase in catch per unit effort during 1974 would have been the result of reduced cropping by the Pittsburg Power Plant in 1973-1974 (the year class that would have been collected in the 1974 midwater trawl survey). Average weekly power production for Pittsburg Power Plant Units 1-6 (net power production capacity 1,282 MWe) for 1972-1976 is shown below (PGandE unpublished b).

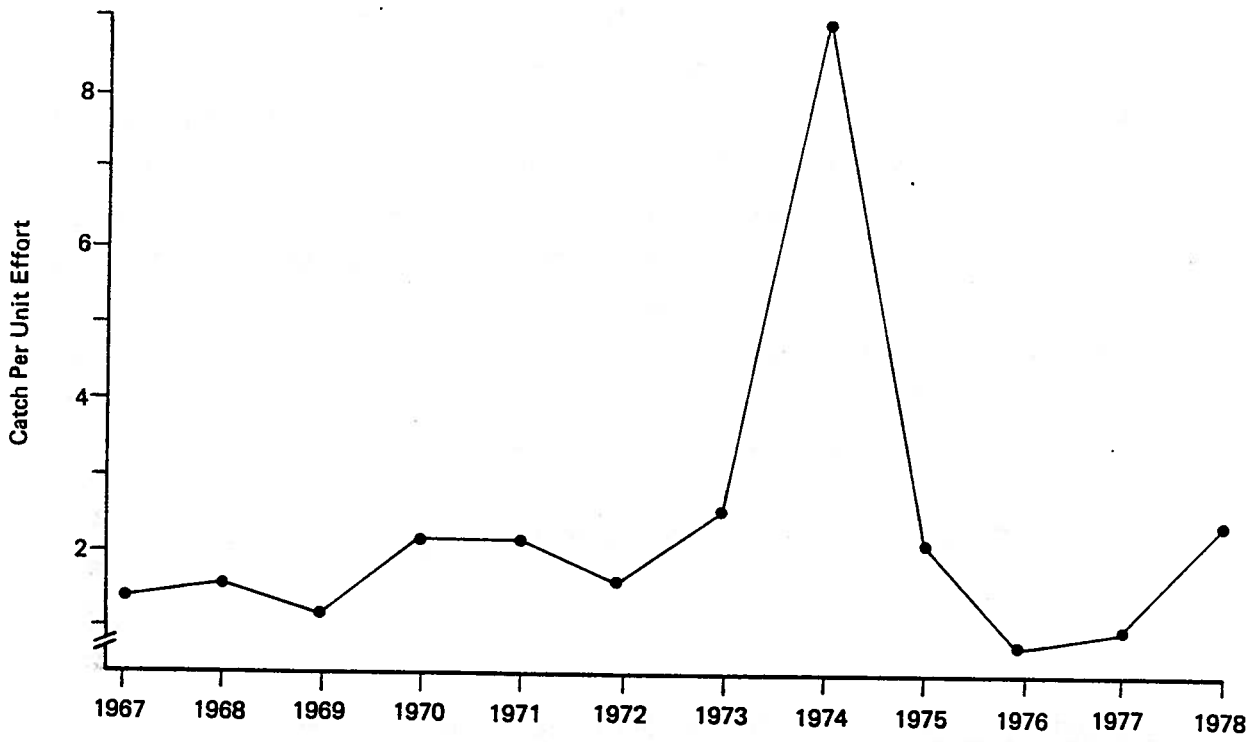
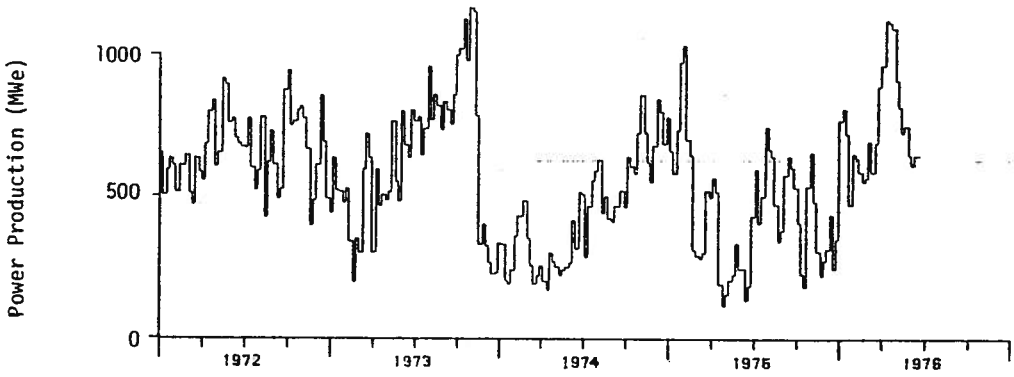


Figure 5-8. Annual average catch of starry flounder in midwater trawl surveys of the California Department of Fish and Game in the Sacramento—San Joaquin Delta, 1967 - 1978.



Power production, and thus the potential for entrainment and impingement, at the Pittsburg Power Plant was generally higher during 1973 than in 1974 or 1975, but a strong starry flounder year class was produced in 1973, and 1974 and 1975 did not contribute large year classes to the fish community sampled by the CDF&G midwater trawl. Therefore, the population trends reflected in the midwater trawl survey data for the Delta provide no evidence that entrainment or impingement cropping at the Pittsburg Power Plant is having an adverse impact on the starry flounder population in the Delta. Entrainment and impingement cropping does not preclude maintenance of the existing starry flounder population or result in a detectable reduction in optimum yield in the sport and commercial fishery.

5.5 IMPACT ASSESSMENT FOR NEOMYSIS (NEOMYSIS MERCEDIS)

5.5.1 Introduction

Neomysis mercedis, a species of Mysidacea (opossum shrimp), is an extremely abundant invertebrate in the Bay-Delta. Neomysis is a major food resource for many fish species in the Bay-Delta, including American shad, longfin and Delta smelt, white sturgeon, white catfish, and striped bass (Section 2.2.7.2). The species is found in rivers and estuaries from Alaska to Point Conception, California. In the Bay-Delta it is distributed from the south and central portions of San Francisco Bay to Sacramento and Knight's Landing on the Sacramento River and to Stockton on the San Joaquin River (Tattersall 1932; Orsi and Knutson 1979). Although found over a wide range of salinities, neomysis are less abundant in salinities greater than 7 ppt (Heubach 1969), and highest densities have been observed in the entrapment zone (Section 2.2.7.1), where salinities range from 1 to 6 ppt (Arthur and Ball 1978; Siegfried et al. 1979). Since the position of the entrapment zone is a function of Delta outflow, the location of maximum neomysis density varies seasonally with changing outflow. During spring and summer 1978 plankton surveys of the Bay-Delta

upstream of San Pablo Bay, maximum densities of neomysis were observed in western Suisun Bay from April to mid-June, in eastern Suisun Bay from mid-June to early July, and in the lower San Joaquin River in mid-July (EA 1981a). The location of maximum abundance shifted upstream when the salinity in a given area began to exceed about 6 ppt.

Because involvement of neomysis with the plant's cooling water system was relatively high (Table 5-2) and because of its importance in the Bay-Delta food web (Section 2.2.6.1), the significance of entrainment cropping requires further analysis to assess the potential for impact on the population.

The primary objective of the impact assessment was to determine the extent to which neomysis cropping due to entrainment at Pittsburg Power Plant controls the neomysis population in the Bay-Delta. Of particular interest is whether entrainment cropping changes the distribution, abundance or productivity of the neomysis population. These parameters reflect the status of the population over time, and they also reflect the supply of neomysis in various areas of the Bay-Delta and the rate at which the population grows and declines, both of which are fundamental to the provision of food resources for small fishes, including young striped bass.

A second objective was to determine whether neomysis entrainment cropping affects the production of young striped bass, and a third was to see whether entrainment impacts on neomysis will change as a result of Delta outflow regimes planned for the future. Neomysis is important because it is the major food resource of several small fishes. Changes in its abundance will change its availability to predatory fishes, and thereby may reduce the survival and production of those fishes. Since entrainment may change the availability of neomysis, it may indirectly reduce the production of young striped bass. Delta outflow changes the location of the entrapment zone, where neomysis density is greatest, and, depending on the level of Delta outflow, it may be located close to or far from the Pittsburg Power Plant. In addition, the abundance of neomysis has been positively associated with the level of Delta outflow (Orsi and Knutson 1979). Thus, since Delta outflow may determine both neomysis abundance and the proportion of the population susceptible to entrainment, planned changes in Delta outflow can change the level of entrainment impact on neomysis and its predators from what it was historically.

The approach used in the analysis included consideration of population characteristics, entrainment effects, effects of neomysis entrainment on striped bass, and effects of future flow conditions on neomysis entrainment. Changes in distribution patterns (abundances and densities) in various geographic areas of the Bay-Delta and over time (both annual and seasonal) are quantified, as are daily birth and death rates, including entrainment cropping rates.

This assessment passes through four successive interrelated sections, each yielding conclusions regarding power plant control over the neomysis population. The structure, distribution, abundance, and dynamics of the neomysis population is examined first, in Section 5.5.2. The dynamics of the population involves the processes which determine the changes in abundance—emigration, immigration, birth, and death—with rate estimates of the first three required to estimate death rates. Also, abundance estimates for Suisun Bay and the Delta will serve in estimating the proportion of the population cropped by entrainment, in Section 5.5.3. Additionally, death rate estimates will serve in Section 5.5.3 as a basis for estimating what proportion of the total population death rate is attributed to entrainment, and thereby providing data to evaluate whether entrainment controls the change in neomysis abundance.

Entrainment and its effects on neomysis abundance are addressed in Section 5.5.3. Estimates are made of the number of neomysis cropped per day and the proportion that this represents of the population dying per day. These estimates are the product of neomysis densities at the power plant intakes, percent mortality as determined by power plant operating conditions and entrainment mortality studies (Chapter 3; Section C.4, Appendix C), and estimates of concurrent abundance in Suisun Bay and the Delta. Daily entrainment mortality rates will be extrapolated to an annual period through estimates of percent reduction, the reduction in abundance that entrainment might have caused.

Once neomysis entrainment is evaluated, its effects on striped bass are considered in Section 5.5.4. The approach involves an evaluation of data on striped bass feeding habits and a comparison of an index of neomysis abundance during late spring and early summer with an index of young striped bass production (described in Section C.3, Appendix C). The index comparison permits a view of the degree of correlation between striped bass and neomysis abundance for various years. The predation rate on neomysis is also considered relative to the entrainment mortality rate.

The effects of future flow conditions are discussed in Section 5.5.5. Entrainment cropping estimates are compared with concurrent mean monthly Delta outflow levels to determine how entrainment cropping has changed in relationship to Delta outflow. The results of these comparisons are evaluated in regard to the changes in mean monthly Delta outflows between historic, 1985, and 1990 conditions.

Historical Review

The importance of neomysis stems from its role as a food resource for fish, especially striped bass, in Suisun Bay and the Delta. It is the main prey of young-of-the-year striped bass and it is important in the diet of juvenile striped bass as well (Heubach et al. 1963; Stevens 1966b; Thomas 1967; Stevens 1979; Smith and Kato 1979). Also, neomysis is an important prey of the larger invertebrates Crangon franciscorum and Palaemon macrodactylus (Sitts and Knight 1979b), which in turn are important prey of juvenile striped bass (Stevens 1966b). Further, smaller striped bass are important prey of larger striped bass (Stevens 1966b) Thus, neomysis is very important as food for the production of striped bass. Several other fish species in Suisun Bay and the Delta utilize neomysis as food (Kelly 1966; Stevens 1966a).

The neomysis population has been studied by the California Department of Fish and Game (CDF&G) since the early 1960s because of concern regarding the indirect effects that water projects may have on striped bass through effects on neomysis. In 1968, these studies took the form of a monitoring program that is ongoing. These studies have led to several conclusions (Orsi and Knutson 1979). First, although neomysis is found throughout San Francisco Bay and the Delta, it is most abundant in Suisun Bay and the western Delta from fresh water to the downstream end of the entrapment zone. Geographic variation in its abundance is controlled by tidal currents, estuarine circulation, and tidally influenced vertical migration by neomysis. In regard to temporal variations in abundance, seasonal variation is principally a function of changes in reproduction, while variation between years is apparently a function of habitat size and food supply. (Habitat size is greater when higher Delta outflows push the downstream end of the entrapment zone seaward.)

Concern over the impacts on neomysis of the Pittsburg Power Plant commenced in 1968 (Chadwick and Stevens 1971). The concern was based on the following factors: (1) the importance of neomysis as food for fish, (2) the belief that estuary temperatures were near or at levels which limited neomysis production, (3) the fact

that the plant was in the center of distribution for neomysis, and (4) that exposure to power-plant-induced temperature effects was continuous. The effects that caused concern were increases in water temperatures during passage through a power plant cooling water system, the localized plume of heated discharge water, and small but critical elevations in water temperature over large areas of the estuary. Following field and laboratory studies of potential impacts of power plant operation (Chadwick and Stevens 1971; Hair 1971), Chadwick (1974) concluded that although the necessary information was lacking to assess adequately the potential for impact, it was improbable that the Pittsburg and Contra Costa power plants threatened the neomysis population, even though they reduced neomysis abundance.

The effects of the Pittsburg Power Plant on the receiving waters were examined by Tetra Tech, Inc. (1976). Effects of exposure to chlorine and elevated temperatures during entrainment, effects of the thermal plume and effects of temperature elevation over thousands of acres of the estuary were considered, utilizing the 1971-1973 CDF&G neomysis survey data and power plant operation data. The overall conclusion was that although the power plant cropped billions of neomysis every year, primarily as a result of entrainment through the its cooling system, there was no trend in abundance from year to year, so the mortality apparently did not contribute to a decline in the population from one year to the next. Further, most dead neomysis still served as food for fish near the power plant cooling water discharges, as they normally would.

A number of aspects of power plant effects on neomysis had not been addressed up through 1977. Entrainment mortality had not been documented or quantified with in situ observations, nor had mechanical causes of mortality been considered. The total size range of neomysis individuals still needed to be considered in quantitative estimates. Further, since 1976 data had been collected from which total population birth and death rates could be computed for particular dates, so entrainment mortality needed to be compared with these rates rather than relying on an assumed constant death rate (which does not exist) over all time to determine whether entrainment controlled the population. Controls on the birth and death rates needed to be determined. Also, yet to be addressed were the potential cumulative effects of entrainment mortality, effects on entrainment of planned modifications of water flows into and out of the Delta, as well as effects of neomysis entrainment mortality on the production of young striped bass. Because of these weaknesses in understanding the extent of interaction between neomysis and power plant operation,

and because these considerations could be investigated with extensive new data, the impact of entrainment on neomysis is assessed in this chapter in a broader and more precise fashion than previously possible.

5.5.2 Population Dynamics and Distributional Characteristics

Population dynamics and distributional characteristics of the Bay-Delta neomysis population are examined in this section. Data collected in the neomysis surveys between 1968 and 1979 by CDF&G at sampling stations throughout the Bay-Delta (Figure 5-9) make it possible to describe trends in population abundance and to examine changes in abundance relative to power plant operation. This information provides a basis for analysis of neomysis population dynamics in Section 5.5.2.2 and analysis of entrainment cropping in Section 5.5.3.1.

The study area, or source water body, was defined as that area of the Bay-Delta extending from the east end of Carquinez Strait to Cache Slough, to the mouth of the Mokelumne River, to Stockton, and to Clifton Court Forebay. This area was selected because it provides a known habitat for neomysis, it has been consistently sampled, and it encompasses the area of greatest susceptibility to entrainment at the Pittsburg Power Plant. The area was divided into six reaches (Figure 5-9) for analysis. The water volume of each reach is listed in Table 5-6.

The analytical methods and data bases used in examining the population dynamics and distributional characteristics of neomysis are discussed in Section C.2, Appendix C.

5.5.2.1 Distributional Characteristics and Abundance

Seasonal Variation in Length

The length composition of the neomysis population undergoes seasonal changes, with smaller individuals present year-round and the largest individuals present only in cooler months. The percentage of the population each month (averaged over the years 1974-1979) in each of eight 2-mm length classes is shown in Table 5-7. The smallest individuals (2-3 mm in length) are present year-round, and along with the 4-5 mm individuals they make up from 48 to 80 percent of the population in every month. The 2-3 mm individuals are the most numerous component from February to

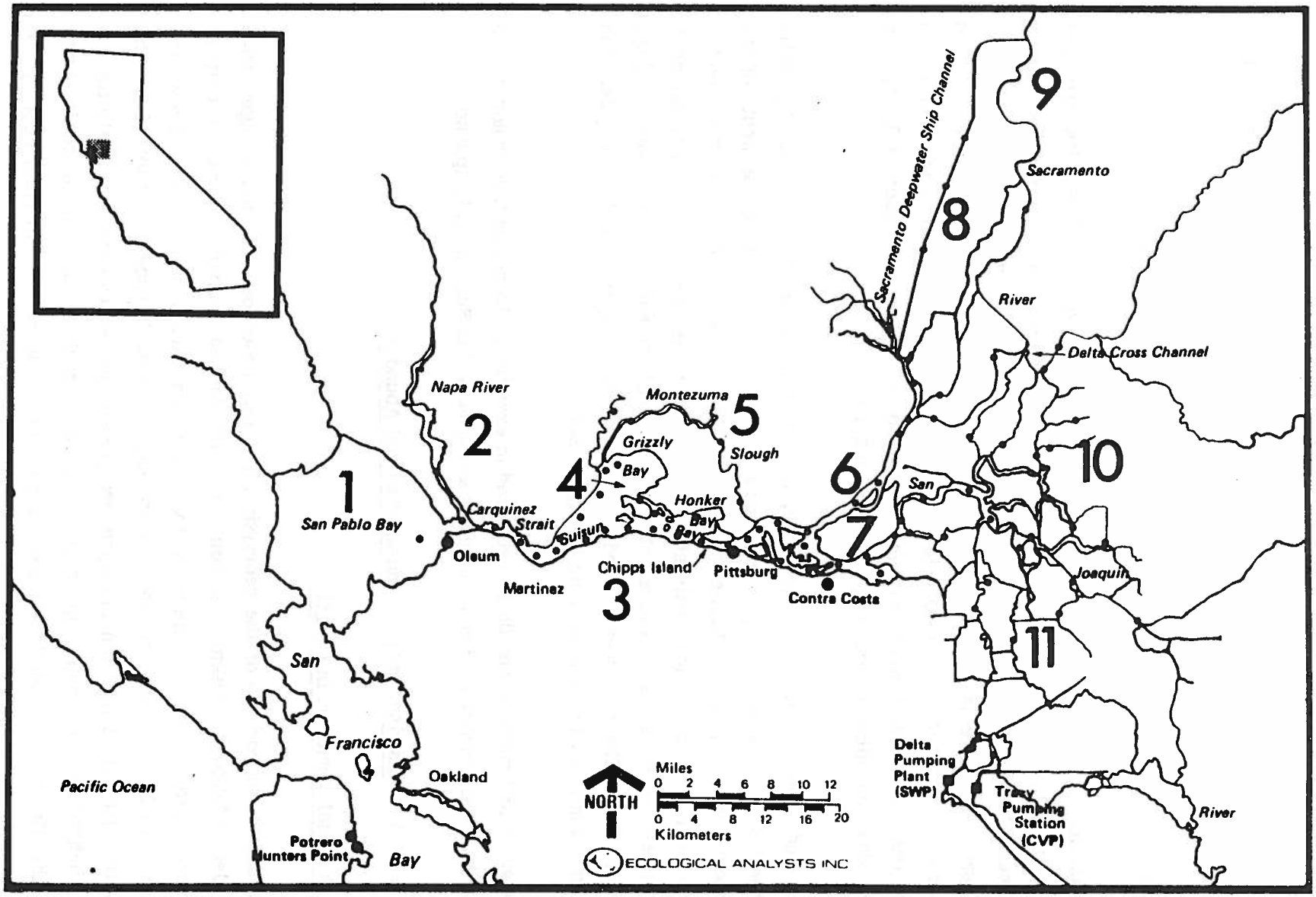


Figure 5-9. Northern San Francisco Bay and the Sacramento–San Joaquin Delta, with California Department of Fish and Game Zooplankton survey sampling stations and study-area strata. (Stations from CDF&G unpublished.)

TABLE 5-6 VOLUMES OF STRATA IN THE BAY-DELTA USED IN ESTIMATING ABUNDANCES OF NEOMYSIS MERCEDIS

Stratum		Volume	
Number	Name	($ft^3 \times 10^9$)	($m^3 \times 10^6$)
3[a]	Suisun Bay channel[b]	7.88	223
4	Suisun Bay shallows	12.05	341
5	Montezuma Slough	1.55	44
6	Lower Sacramento River	6.01	170
7	Lower San Joaquin River	7.75	219
8	Cache Slough, Steamboat Slough, Ship Channel	3.61	102
9	Upper Sacramento River	1.97	56
10	North Delta	2.18	62
11	South Delta	9.09	257
Total without 8, 9, and 10		44.33	1,254
Total with 8, 9, and 10		52.09	1,474

(a) Strata 1 and 2 were not part of this assessment.
(b) Does not include Carquinez Strait west of the Benicia Bridge.

TABLE 5-7 MONTHLY SIX-YEAR-MEAN SIZE FREQUENCIES OF NEOMYSIS MERCEDIS IN THE BAY-DELTA, 1974-1979

Month	Size Interval (mm)							
	2-3	4-5	6-7	8-9	10-11	12-13	14-15	16-17
JAN	14.3	48.8	23.3	6.2	3.3	3.3	0.9	0
FEB	29.4	18.7	12.6	0.4	7.8	16.0	6.0	0.1
MAR	52.3	26.7	5.8	3.1	2.5	5.5	3.7	0.3
APR	44.1	27.2	10.4	5.2	4.6	6.1	2.3	0.2
MAY	40.1	29.4	12.3	6.4	7.9	3.4	0.5	0
JUN	39.9	43.3	13.7	7.8	5.7	0.7	0	0
JUL	40.7	30.4	14.4	9.2	5.1	0.3	0	0
AUG	33.7	30.1	15.8	13.9	6.1	0.3	0	0
SEP	35.4	27.2	14.3	14.4	8.3	0.5	0	0
OCT	44.0	27.5	11.0	9.5	7.7	0.3	0	0
NOV	33.8	36.3	15.7	7.8	5.8	0.6	0	0
DEC	24.6	44.7	19.3	5.9	4.2	1.2	0.1	0

October; the 4-5 mm class are most numerous from November to January. The largest neomysis, 16-17 mm long, were collected only between February and May, and those 14-15 mm long were collected only between December and June. The disappearance and reappearance of all larger neomysis is a regular yearly event (Heubach 1969; Sitts and Knight 1979a), indicating that this annual cycle of length composition is a natural process within the population and not controlled by operation of the Pittsburg Power Plant.

Since neomysis rarely reach lengths greater than 16 mm and widths of 1-2 mm, few individuals are impinged on intake screens, which have 0.9-cm (3/8-in.) screen mesh. Entrainment, therefore, is the main source of neomysis involvement with the power plant cooling water systems.

Vertical Differences in Density

The neomysis population in the Bay-Delta undergoes regular nocturnal vertical migrations (Heubach 1969; Sitts and Knight 1979a). During the day, most individuals are near the bottom, and extremely few are near the surface. At night, they spread out through the entire water column. Since this phenomenon occurs among several planktonic invertebrate species worldwide, the vertical migration by neomysis is probably not controlled by power plant operation. Further, since neomysis seeks darker, deeper waters during the day, when power plant operation peaks, its susceptability to entrainment is reduced.

Density Variation Across Channels

Neomysis densities not only decrease from the bottom to the surface of a water column, they also decrease from midchannel areas to the shore. This difference has been observed in the vicinity of the Pittsburg Power Plant. EA measured neomysis entrainment densities at the plant in 1978 and 1979 (Chapter 3). The average density was $10.7/m^3$. Over the same period, CDF&G neomysis survey data from stations near the plant yielded a mean midchannel density of $71.5/m^3$. The differences between concurrent entrainment and midchannel density estimates were significant (t-test; $\alpha = 0.05$). The ratio of entrainment density to midchannel density estimates was 15 percent.

A similar conclusion regarding entrainment-to-midchannel density ratios may be drawn from an analysis of the results of sampling done by Tetra Tech, Inc. (PGandE unpublished b). Tetra Tech took pump samples from the intake of the Pittsburg Power Plant during the period May-October 1976 concurrently with net samples in midchannel. These resulted in an average entrainment density which was 39 percent of the average of midchannel density. In addition, the nets used to obtain the midchannel data were estimated to be 69 percent as efficient as the pumps. Applying a correction for net sampling efficiency, the entrainment density estimates are 27 percent of midchannel densities. Further, midchannel sampling was only in the upper two-thirds of the water column. The greatest concentrations of neomysis are near the bottom: the average concentration over six diel cycles in 1976 near the Pittsburg Power Plant for an entire 27-ft (8-m) water column was 137 percent of the average for the top and middle of the column (Sitts and Knight 1979a). Dividing the 27 percent power plant entrainment to midchannel density ratio by 1.37 to adjust for not sampling the entire midchannel water column yields a final estimate of 20 percent. This 20 percent estimate is close to the 15 percent estimate for the Pittsburg Power Plant that was based on 1978 and 1979 data from EA and CDF&G.

Density Variation Down Main Channels

Neomysis density in the main channel, between San Pablo Bay and Rio Vista on the Sacramento River, has been repeatedly observed to be greatest where salinities at 1 m depth are 1-6 ppt (2-10 mmho conductivity) (Figure 5-10) (Heubach 1969; CDF&G et al. 1975; Arthur and Ball 1979; Siegfried et al. 1979; Sitts and Knight 1979a). This 1-6 ppt salinity zone is commonly referred to as the entrapment zone (Section 2.2.1). Concentration of neomysis in the entrapment zone is probably a result of most neomysis being near the bottom, and the upstream net nontidal flow of water in the lower portion of the water column at salinities exceeding 6 ppt, and seaward net nontidal flow throughout the entire water column at salinities less than 1 ppt.

The location of the entrapment zone is a function of Delta outflow: increasing Delta outflows push the entrapment zone seaward (Figure 5-11) (Section 2.2; Appendix A). When Delta outflow is between 4,000 and 14,000 cfs, the entrapment zone is located adjacent to the Pittsburg Power Plant at some point in the tidal cycle.

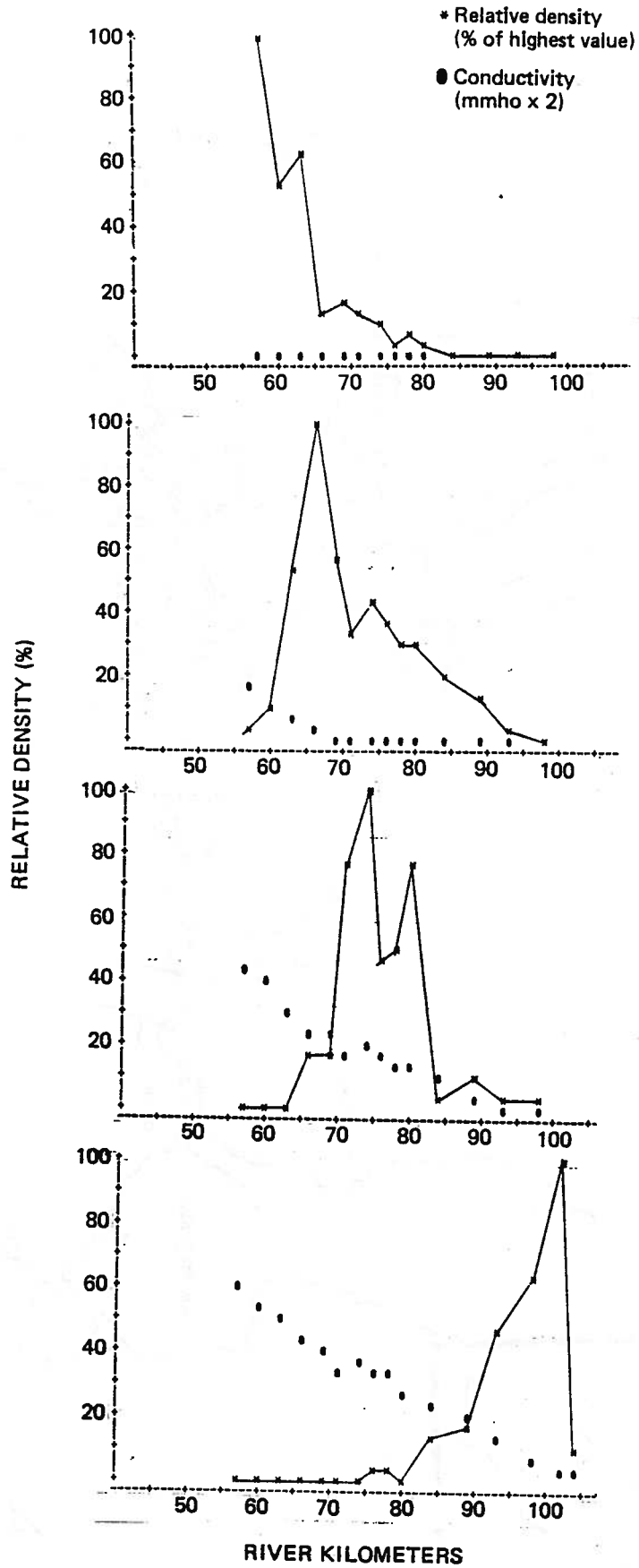
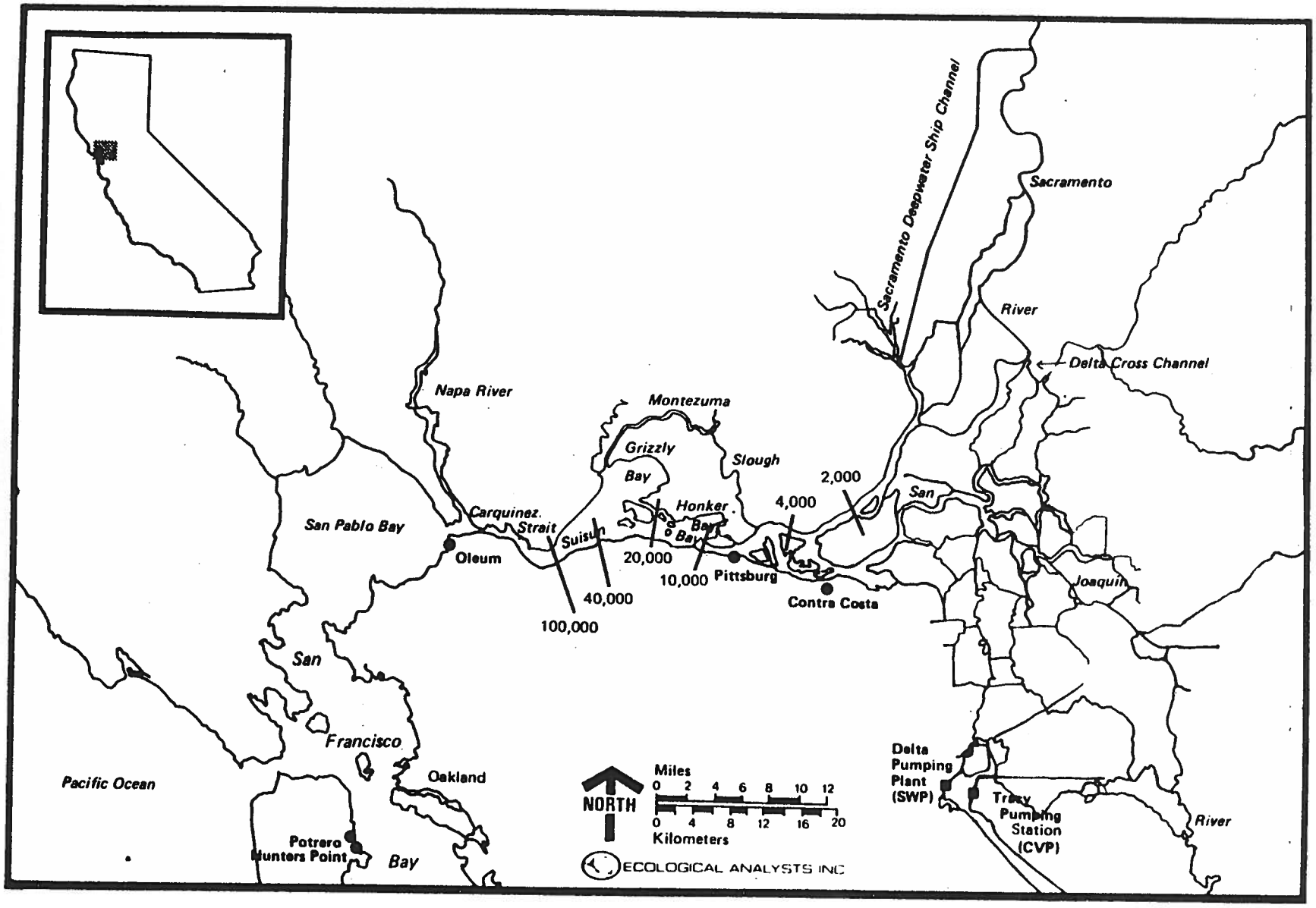


Figure 5-10. Relative densities of Neomysis mercedis at California Department of Fish and Game sampling stations in the main channel of Suisun Bay and the lower Sacramento River, with corresponding conductivity measurements.



Numbers indicate corresponding Delta outflows (cfs).

Figure 5-11. Position at high tide of the upper end of the region of 2-10 mmho surface conductivity at various Delta outflows predicted by the EA model.

The control of Delta outflow over the location of greatest neomysis densities in the main channel has two important ramifications in the assessment. One, it represents another attribute of the neomysis population that is not controlled by power plants. Two, it means that the number of neomysis entrained at any one time can be high or low depending on whether or not Delta outflow positions the entrapment zone opposite the power plant. This factor is taken into account in estimating entrainment mortality rates in Section 5.5.3, since the rates are for each CDF&G neomysis survey and therefore for particular Delta outflow conditions.

Geographical Variation in Density and Abundance

The population distribution changes geographically between months and between years. For instance, in 1974 monthly average densities relative to the main channel were nearly equal or greater in the shallows of Suisun Bay and in Montezuma Slough in April, May, August, and September (Figure 5-12). The same relationship held in 1975 from February through May and from August through November. Densities in 1977, the year of lowest outflows, were greatest in the lower Sacramento River, Cache Slough, and the upper Sacramento River (Figure 5-13). During 1978 and 1979, Montezuma Slough and the Suisun Bay shallows again had the greatest average neomysis densities in several months.

The abundance (total number of individuals) of neomysis is typically greatest in four reaches; Suisun Bay main channel, Suisun Bay shallows, the lower Sacramento River, and the lower San Joaquin River (Figure 5-12). Relative abundance in Montezuma Slough (except in February 1979) and the southern, eastern, and northern portions of the Delta were consistently lower than in the western Delta between 1974 and 1979. The level of Delta outflow is strongly associated with the location of maximum abundance. During periods of high outflow most neomysis are in the main channel and/or shallows of Suisun Bay, as they were in 1974 and 1975. Maximum abundance during the low outflow year of 1976 was farther upstream, in the lower Sacramento and lower San Joaquin rivers, where it remained until December 1977, when freshwater flow increased. In winter and spring 1978, Delta outflow was high, and most of the neomysis population was in Suisun Bay, predominantly in the shallows. Later in 1978, the population was widely distributed, only to return to a maximum abundance in Suisun Bay once high outflow returned early in 1979. Delta outflow was low again in the summer of 1979, and the area of maximum abundance again shifted upstream to the lower Sacramento River.

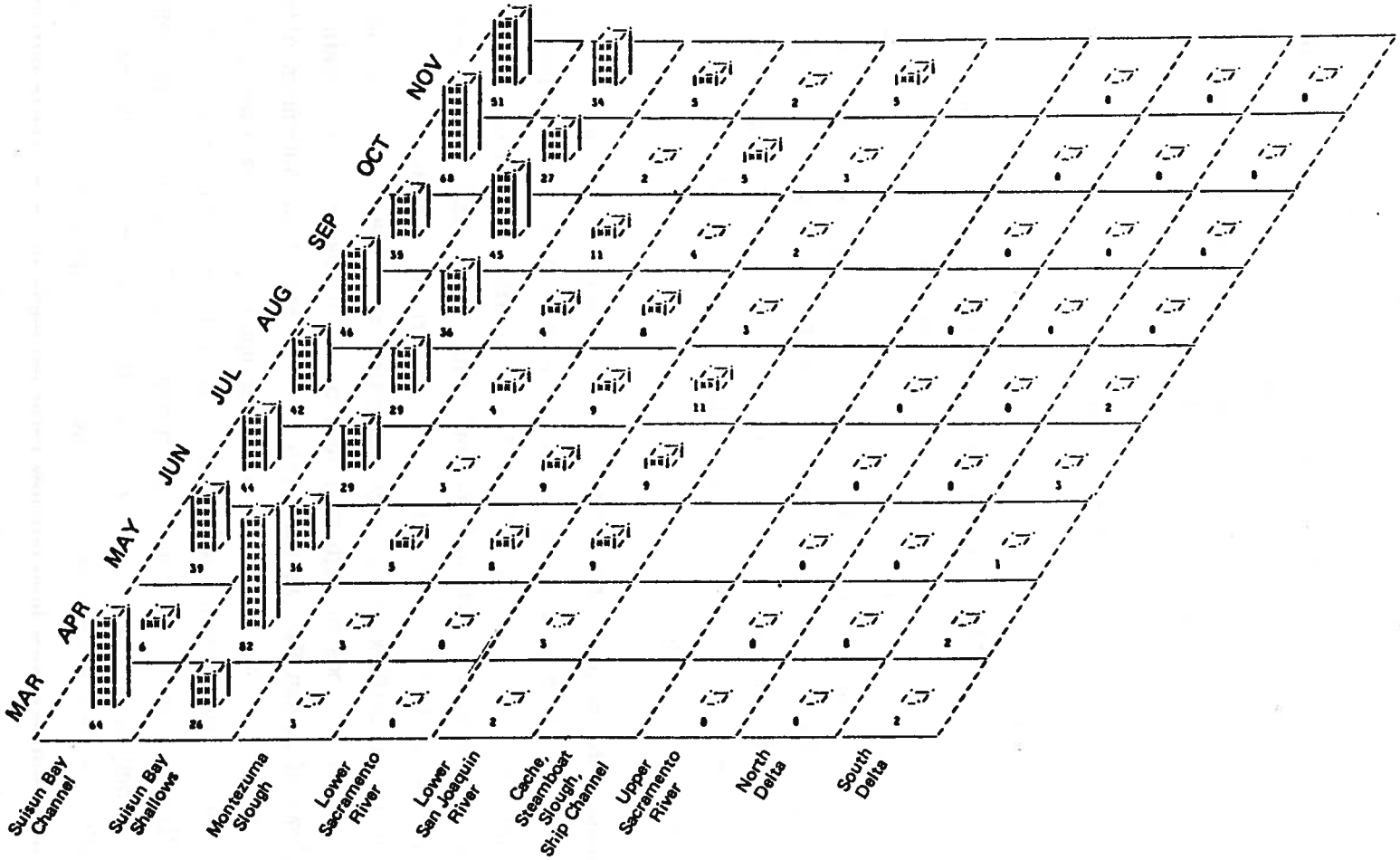


Figure 5-12. Percentages of the monthly study-area abundances of Neomysis mercedis found in various strata of the Bay-Delta in a year of high outflow, 1974.

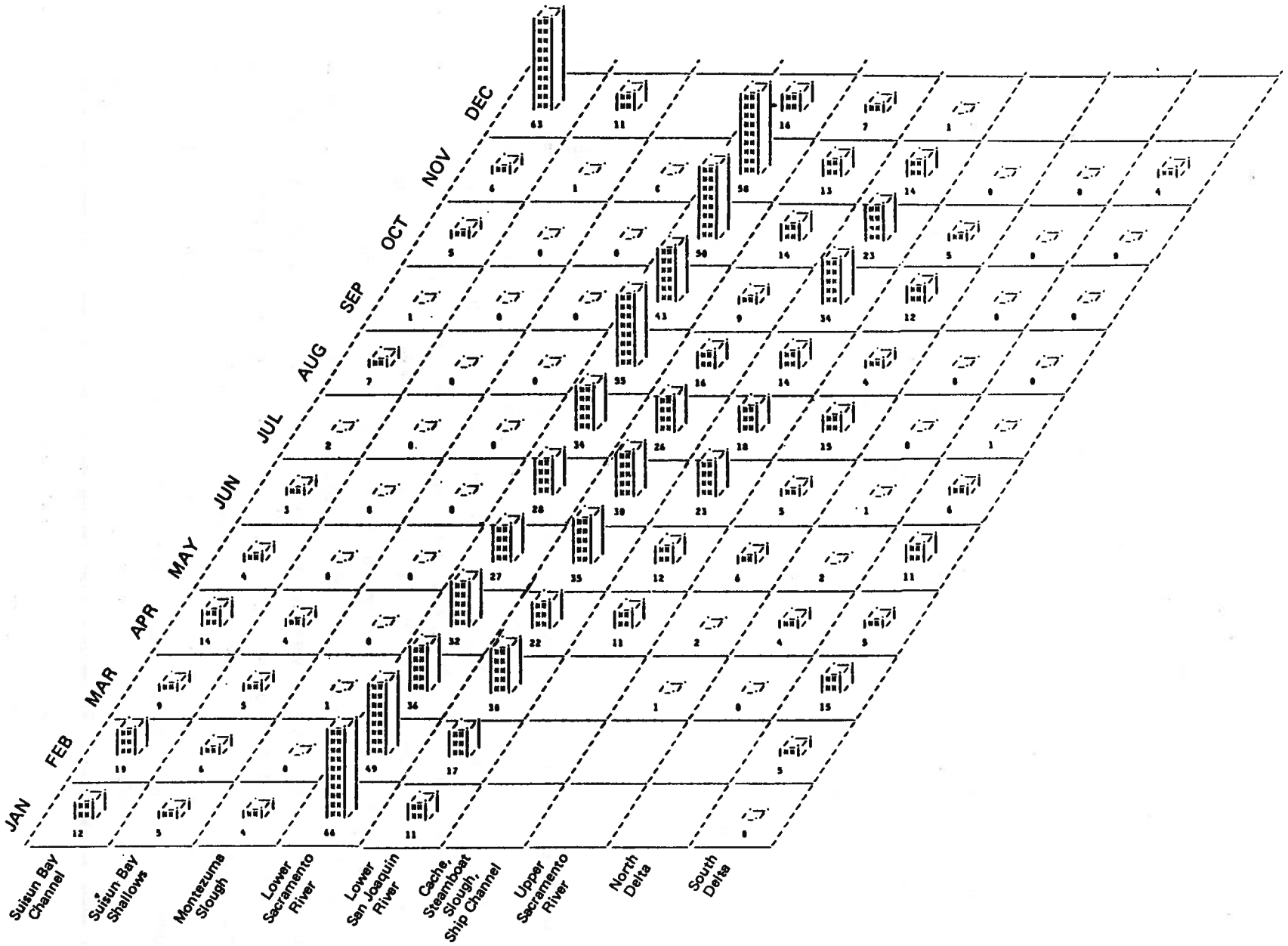


Figure 5-13. Percentages of the monthly study-area abundances of Neomysis mercedis found in various strata of the Bay-Delta in a year of low outflow, 1977.

Very high Delta outflows (50,000-110,000 cfs) have occurred concurrently with maximum abundance in Suisun Bay shallows. Examples occurred in April 1974, in February and March 1975, and January-April 1978. These large outflows leave the shallows much more populated than the adjacent main channel, probably because the shallows are sheltered from the seaward flow of the entrapment zone and freshwater flow into Carqiunez Strait and San Pablo Bay. During periods of high outflow, these protected shallows, which are more productive than the deeper, darker main channel, may be the area in which much of the spring population is produced.

It can be concluded from observations of geographic differences in density and abundance that changes in the distribution of neomysis are strongly associated with changes in Delta outflow. During periods of high Delta outflow most neomysis are in Suisun Bay, and as outflow decreases below 10,000 cfs the majority of the neomysis are found in the Delta, particularly the lower Sacramento and San Joaquin rivers. This shift seems largely out of control of the power plants, and strongly in the control of Delta outflow. Suisun Bay and much of the Delta and Montezuma Slough contain large percentages of the neomysis population that are not susceptible to entrainment.

Variation in Total Abundance

The neomysis population in the study area undergoes a regular annual cycle of low abundance in winter and maximum abundance in late spring or summer. This was observed from 1968 through 1979 for 4-17 mm individuals and from 1974 through 1979 for 2-17 mm individuals. (From 1968 through 1973, 2-3 mm individuals were not sampled efficiently, so they were deleted from analyses of these years.) For instance, abundances in 1978 for 2-17 mm neomysis (Figure 5-14) were estimated to fluctuate from 0.3 to 39 billion for January through April, 66 to 159 billion from May through July, and 53 to 15 billion from August through December. This pattern of abundance can appear in a variety of forms, from unimodal as in 1978 to bimodal as in 1973, or relatively low and constant as in 1977. In addition, the increase in abundance within a yearly cycle can be as much as a 500-fold increase, as in 1978, between January (0.3 billion) and late June (159 billion). The seasonal variation in abundance estimates over annual cycles is statistically significant, although variation within periods of a month or two usually is nonsignificant, as indicated by the overlap or lack of overlap of 95 percent confidence intervals. The range of the confidence interval about an abundance estimate reflects the density variation over areas of the Bay-Delta that was described in the previous subsections.

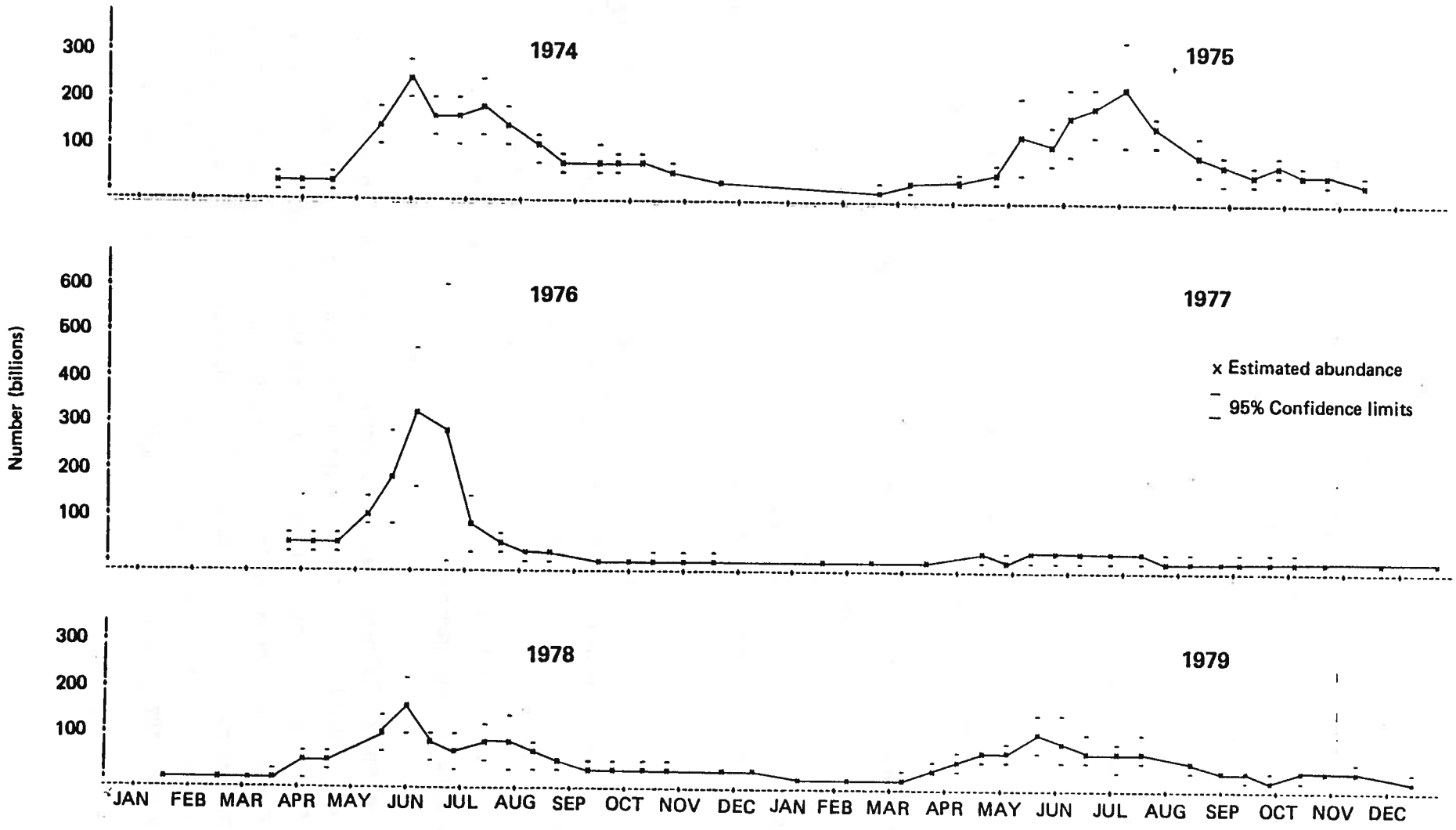


Figure 5-14. Estimated abundances of 2-17 mm Neomysis mercedis in the Bay-Delta study area, 1974-1979. (Data from CDF&G unpublished.)

Peaks in abundance estimates for each year have varied extensively over the period from 1968 to 1979. Peaks in annual abundance estimates for 4-17 mm neomysis (Figure 5-15) were greatest in 1973, 1974 and 1976, at 201, 174 and 157 billion, respectively. Years with the lowest peaks in abundance were 1970, 1977 and 1979, at 70, 12, and 66 billion, respectively. The greatest estimates for 2-17 mm neomysis over the period 1974-1979 (Figure 5-14) are for 1974 and 1976, at 236 and 312 billion, respectively, and least for 1977 and 1979, at 16 and 97 billion, respectively. In terms of rating the population's status by peak annual abundance, strong years were 1973, 1974, and 1976, and weak years were 1970, 1977, and 1979, especially 1977.

Neomysis abundance over the period 1974-1979 was never high when the bulk of the population was upstream of Suisun Bay, as in late 1976 and 1977. Apparently, this pattern of abundance is related to low Delta outflow levels and the hydrology of the entrapment zone. As Delta outflow declines, the entrapment zone moves upstream into the main channels of the rivers, and takes most neomysis with it. The river channels and the southern Delta cannot sustain high levels of neomysis abundance. This is probably because they constitute a deeper and darker habitat that produces less food for neomysis than Suisun Bay does, and a shorter and narrower entrapment zone that concentrates neomysis in a smaller area (Orsi and Knutson 1979).

The neomysis population does have the ability to recover following a period of very adverse conditions. This has been demonstrated over the period from 1976 to 1979. Abundance declined during the summer of 1976 and stayed at record minimums until 1978. During this period, record low Delta outflows occurred, which may have contributed to the low abundance. Following 1977, Delta outflow increased to very high levels and abundance increased.

Abundance estimates indicate that the presence of power plants operating in the Delta has not resulted in a continuous decline in the neomysis population, as would be expected if the power plants controlled the population. All of the units at the Pittsburg Power Plant except Unit 7 were already operating when the CDF&G neomysis surveys started in 1968. Since 1968, peak abundance estimates over each year fluctuated: from 1968 to 1973, they were moderate, followed by high peaks between 1973 and 1976, a minimum in 1977, and moderate levels in 1978 and 1979.

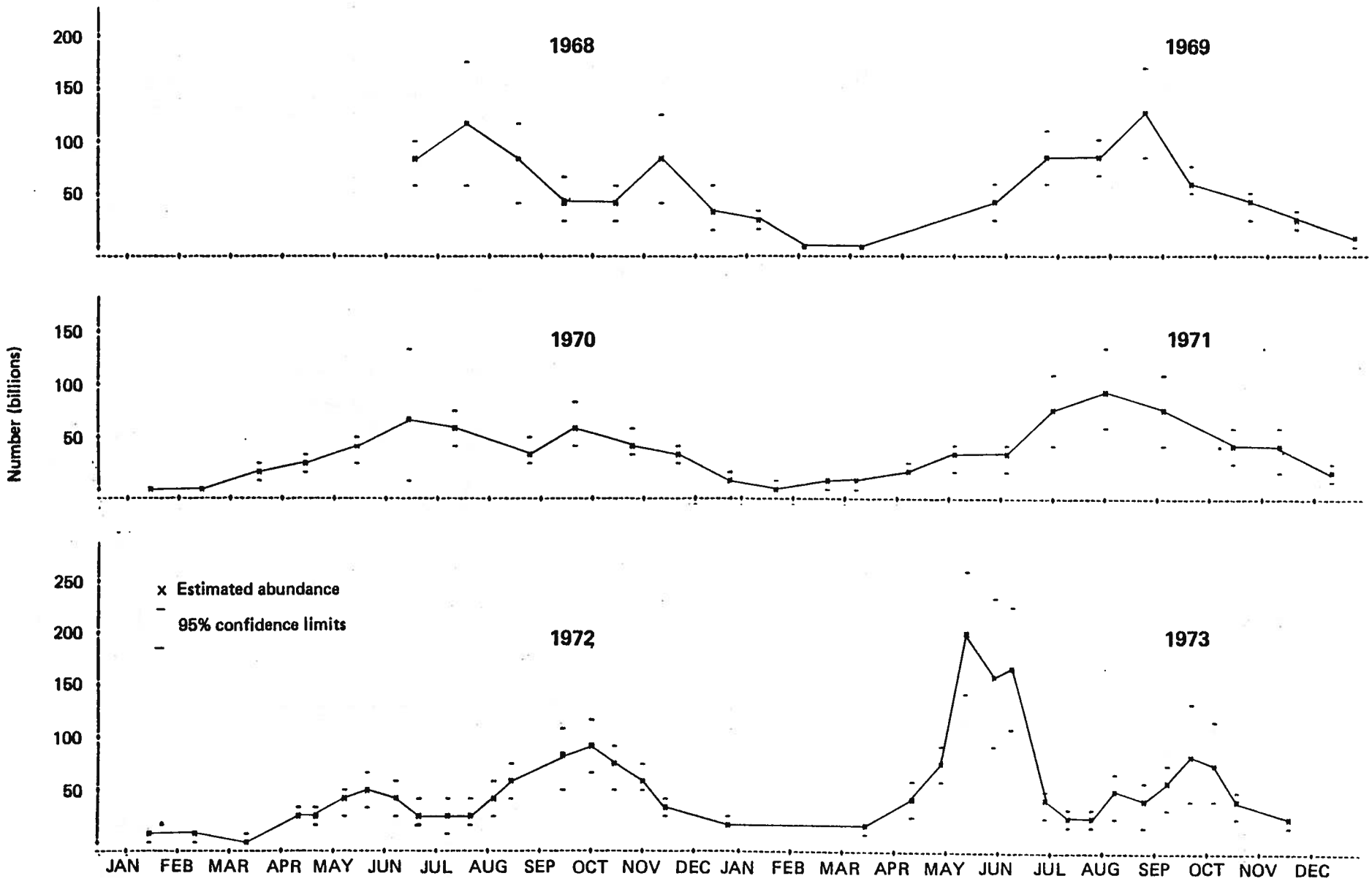


Figure 5-15. Estimated abundances of 4-17 mm Neomysis mercedis in the Bay-Delta study area, 1968-1979. (Data from CDF&G unpublished.)

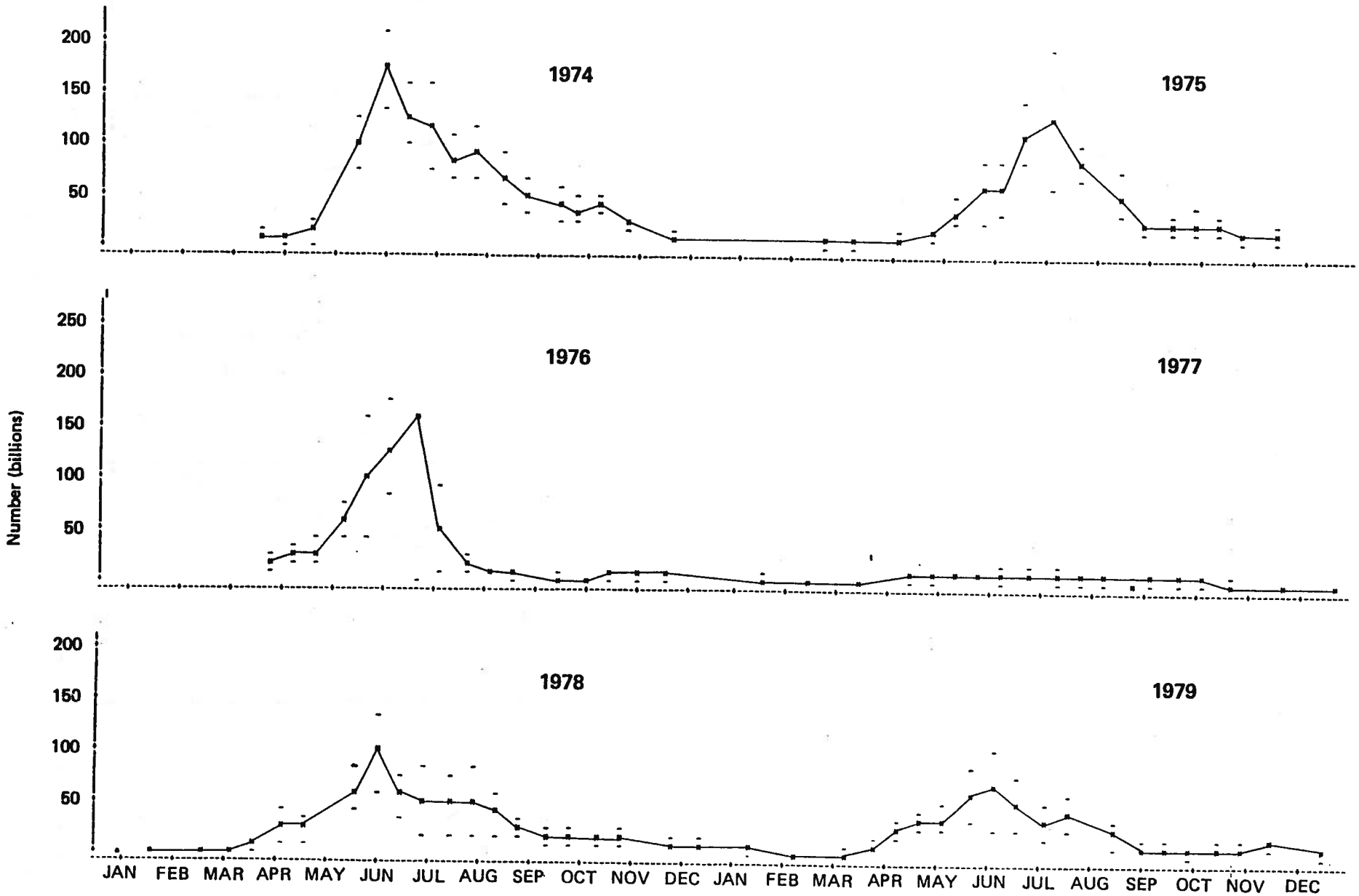


Figure 5-15 (cont.)

5.5.2.2 Processes Determining Neomysis Abundance

The abundance of neomysis in the study area is determined by four processes: immigration, emigration, birth, and death. In order to know the impact of entrainment mortality on the population, we must know the overall extent of mortality, birth rates and migration. The difference between birth rates and rates of change in abundance constitutes an estimate of the overall mortality rate, when migrations are insignificant. These overall mortality rate estimates can be compared with entrainment mortality rates to help evaluate the impact of the latter. Since neomysis are semiplanktonic, immigration and emigration can be estimated as a function of neomysis densities and net inflow and outflow of water at the boundaries of the study area. The rate at which abundance changes between surveys will be calculated, using abundance estimates from the previous section. Birth rates will be estimated from the numbers and sizes of gravid females and their associated fecundities, along with water temperatures and the resultant development times for embryos. A more detailed discussion of the methodology and data used in estimating migration and birth rates is presented in Section C.2, Appendix C.

Since information on neomysis reproduction is lacking for earlier years, rate estimates were made only for the period 1976-1979.

Immigration and Emigration

Immigration and emigration were considered to be significant processes affecting estimates of neomysis population dynamics when daily migration into or out of the study area involved more than 1 percent of the population. Surveys in which migration either into or out of the study area was considered to be significant were excluded from the calculation of population mortality rates.

Preliminary analysis of neomysis distribution data showed that except on one occasion, the only significant migration occurred under the Benicia Bridge. Migration through study area borders on Cache Slough on the Sacramento River, the San Joaquin and Mokelumne rivers, and Delta channels was insignificant except in December 1977.

Migration through Carquinez Strait (under the Benicia Bridge) was estimated using information on Delta outflow and changes in electrical conductivity measured at DWR

sampling stations 42 and 48. Based on these data and the estimated location of the entrapment zone (see Appendix A for details), periods of significant emigration and immigration through Carquinez Strait were determined. Migration was significant on 12 occasions between 1976 and 1979 (Table 5-8). Significant emigration occurred between the months of December and May and significant immigration occurred between April and June. Emigration and immigration change the abundance of neomysis in the study area and are a result of changes in Delta outflow. Power plant operaton has no control over such changes.

Birth Rates

Development time, from fertilization to liberation from the female's brood pouch as a juvenile, varies between mysid species and water temperatures. As with many mysid species, neomysis reproduce year-round, but the proportion of adult females carrying young (gravidity) and the number of young carried (fecundity) varies with season (Heubach 1969; Pezzack and Corey 1979). The equation used to calculate the daily birth rate, b, (Paloheimo 1974) is

$$b = \frac{\ln \quad (C_i + 1)/N_i}{D_i} \qquad (5\text{-}1)$$

where

i = the ordinal number of a survey

C_i = the number of eggs and embryos in the study area during survey i

N_i = the number of free-living individuals in the study area

D_i = development time (days).

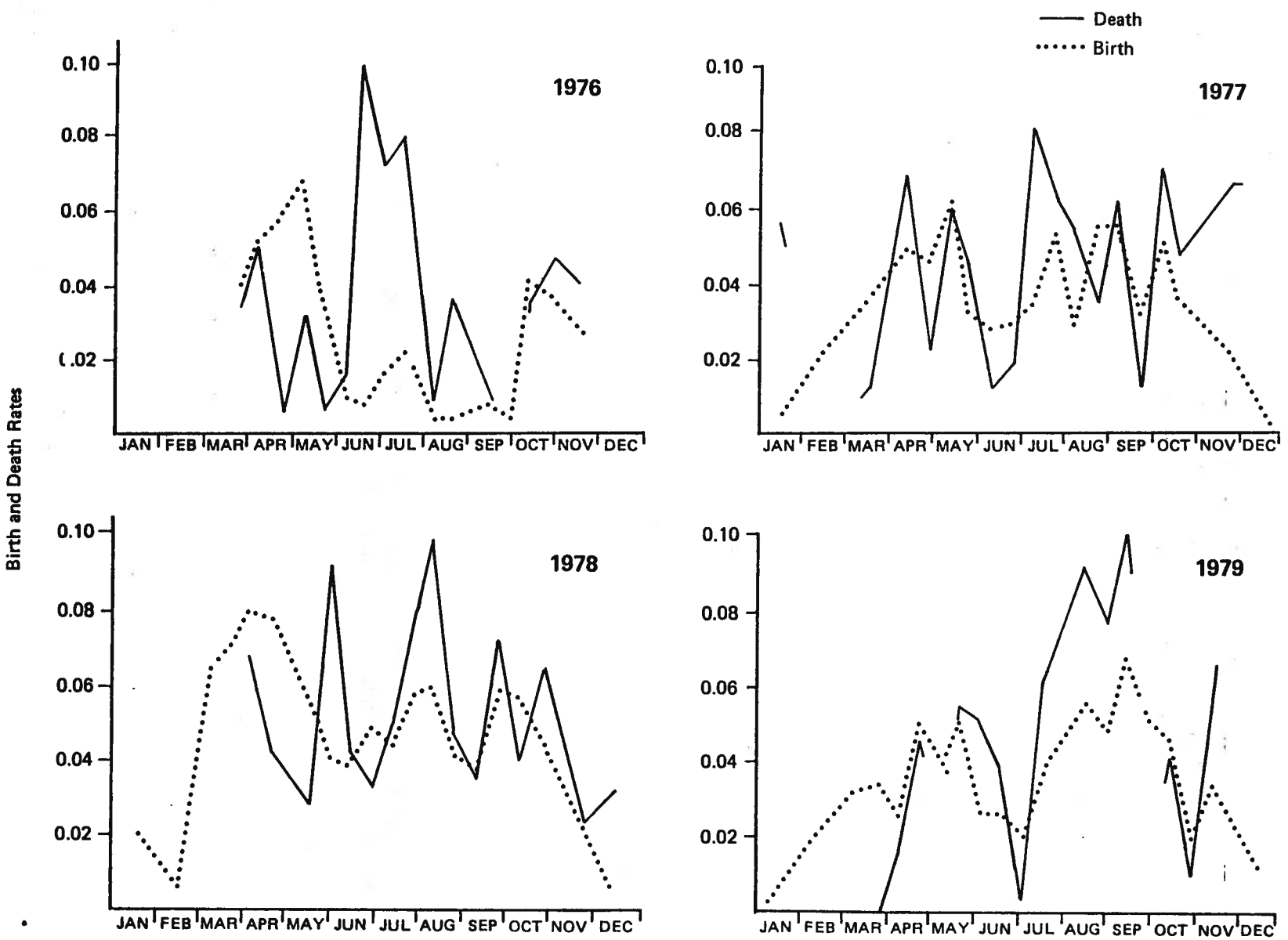
The methods used in calculating the number of eggs and embryos in the study area, the proportion of females gravid, fecundity, and development time are discussed in Section C.2, Appendix C.

Birth rate estimates range from 0 to 0.0795 over all four years, and within any year from 0.0085 to 0.0620 (Figure 5-16). Within this range, estimates of birth rates indicate that a general seasonal pattern exists. Birth rates in winter are low, a result of slower development and low fecundity. April and May peaks are a result of faster

TABLE 5-8 SURVEY DATES FOR WHICH *NEOMYSIS MERCEDIS* MIGRATION BETWEEN CARQUINEZ STRAIT AND SUISUN BAY WAS ESTIMATED TO BE SIGNIFICANT, 1976-1979

Date	Significant Immigration	Significant Emigration
20 December 1977		*
6 April 1978	*	
19 April 1978	*	
18 May 1978		*
31 May 1978	*	
14 June 1978	*	
8 January 1979		*
6 February 1979		*
6 April 1979	*	
23 April 1979		*
7 May 1979	*	
5 June 1979	*	

Note: Significant migration is defined as that involving more than 1 percent of the study area population.



Gaps mark periods when significant migration precluded estimation of death rates.

Figure 5-16. Birth and death rates of Neomysis mercedis in the Bay-Delta, 1976-1979.

development, maximum fecundity, and an abundance of large gravid females. In summer, birth rates are moderate, even though development is faster, because large females (>12mm) are absent and the fecundity for all sizes is low. Birth rates peak again in the fall when fecundity increases, development is still fast, and slightly larger adult females are present. These are followed by a return to low winter rates.

Birth rate estimates are used in the next section to determine mortality rate estimates for the neomysis population, which can then be compared with entrainment mortality rates.

Mortality Rates

The equation employed for estimating mortality rates (d) for neomysis when migration is insignificant is

$$d = b - r \tag{5-2}$$

which is derived from the more general equation

$$r = b - d + i - e \tag{5-3}$$

where

r = instantaneous rate of change
b = instantaneous birth rate
i = instantaneous immigration rate
e = instantaneous emigration rate.

The first step in estimating neomysis death rates in the study area is to calculate the instantaneous rate of change in the abundance of the study area population. This step assumes that abundance changes over time in an exponential pattern. The standard exponential function for change in abundance is

$$N_t = N_0 e^{rt} \tag{5-4}$$

where

N_t = the estimate of abundance at the end of some period
N_0 = the estimate of abundance at the beginning of the period
r = the daily rate of change in abundance
t = the duration of the period in days.

This can be solved for r:

$$r = \frac{\ln(N_t) - \ln(N_0)}{t}. \tag{5-5}$$

Since the end point of one period (survey interval) is the beginning point of the following one, r can be determined from each successive pair of surveys (i, i+1):

$$r_i = \frac{\ln(N_{i+1}) - \ln(N_i)}{T_i}. \tag{5-6}$$

where

N_i = the estimated population for survey i
N_{i+1} = the estimated population size for survey i+1
T_i = the number of days between survey i and survey i+1.

The variance of r_i can be approximated with a Taylor series:

$$Var(r_i) = d'Cd \tag{5-7}$$

where

$$d = \begin{bmatrix} -\frac{1}{N_i t_i} \\ \frac{1}{N_{i+1} t_i} \end{bmatrix}$$ = the vector of the first-order partial derivative of r_i with respect to N_i and N_{i+1}

$$C = \begin{bmatrix} Var(N_i) & 0 \\ 0 & Var(N_{i+1}) \end{bmatrix}$$ = the covariance matrix of the estimates N_i, N_{i+1}.

The estimates $\hat{N}_i$ and $\hat{N}_{i+1}$ and their variances, $Var(\hat{N}_i)$ and $Var(\hat{N}_{i+1})$ yield the following estimate of the variance of r_i:

$$Var(r_i) = \frac{1}{(T_i)^2} \left(\frac{Var(N_{i+1})}{(N_{i+1})^2} + \frac{Var(N_i)}{(N_i)^2} \right). \quad (5\text{-}8)$$

There are four requirements suggested by Winberg et al. (1971) for data employed in population dynamics analyses. The first requirement is that uniform techniques be employed over the same set of sampling sites. Very nearly the same set of sampling sites are covered, with several more additions than deletions over the years. Also, the same basic techiques have been used. The second requirement is that the duration of the sampling interval (time between surveys) be short relative to the duration of the immature stage. The period of the juvenile stage has been estimated to vary between 2 and 8 months (CDFG et al. 1976; Siegfried et al. 1979). Except for some winter periods, the sampling interval for the CDF&G neomysis survey data base is 0.5-1 month. Thus, in most cases the second requirement is met. The third requirement is that the sampling interval be several times shorter than the total life span. This interval was seldom as much as half of the juvenile phase, and thus constituted an even smaller proportion of the life span. During the summer the same cohorts were sampled at least five times. In winter, the sampling survey interval was monthly in all years except one, when it was 3 months. With a juvenile phase of eight months, and perhaps a month for development, a winter cohort sampled at intervals of 0.5, 1, and 3 months was sampled at least five times during its life span. The fourth requirement is that enough samples per survey be taken to allow sampling error determinations for estimates of standing crop and of the various rates. Several stations were sampled in each survey so variation in the standing crop estimates for neomysis could be calculated (Section 5.5.2.1).

There are two assumptions underlying the calculation of r, the rate of change, and the use of the corresponding equation to calculate mortality rate (Hall 1971): (1) that rates are continuous over time and (2) that birth and mortality rates follow the same pattern over the period of estimation.

The assumption that rates are continuous over time is necessary because these rates are supposed to be instantaneous. Since neomysis generation times are a matter of weeks and months, the use of a day as the instantaneous time unit has been accepted as sufficient to approach continuity (Hall 1971).

The second assumption, that birth and death rates follow the same pattern over sampling intervals is required for accepting a predicted abundance estimate N_x at a time t_x which is between times t_1 and t_2. Otherwise, the period from t_1 to t_x might be when, e.g., most of the births or deaths occur. If this should be the case, the predicted N_x would be inaccurate. For instance, if the birth rate estimate at time t_1, when the parameters that went into it were measured, would have been found to be higher if it had been calculated after t_x (but before t_2), then the death rate, which is the rate for the period t_1-t_2, would be too low, or even negative, since $d = b - r$.

The rate of change, r, was calculated for 2-17 mm neomysis for each CDF&G neomysis survey, except those excluded because of significant migration, for the period 1976-1979. The rates will be used in conjunction with the birth rates for the same period to calculate the overall mortality rates for 2-17 mm neomysis.

Mortality rate estimates over all four years ranged from 0 to 0.1065, that is, 0-11 percent of the population dying each day. The average overall for positive and migration-unaffected estimates (53 in all) is 0.0468. Thus, with an average 5 percent of the population dying per day, neomysis abundance still undergoes large changes within and between years. The mortality rate appears to be density-independent, since rates in the higher-abundance years 1976 and 1978 were similar to rates in the lower-abundance years 1977 and 1979. Also, mortality rates are erratic and lack general seasonal patterns, unlike birth rates, although annual peaks occurred in the period between June and September. Mortality rate estimates are similar in range (0 to 0.1065) to birth rate estimates (0 to 0.0795).

Processes Controlling Abundance Between 1976 and 1979

In this section the four processes affecting neomysis abundance are integrated to explain the large variation in abundance over the period 1976-1979 (Figure 5-15).

The results of the analysis demonstrate several points concerning controls on neomysis abundance. First, birth, death, immigration, and emigration have varying influences on the abundance of neomysis in the study area (Figure 5-16, Table 5-8). Birth and death were controlling influences in the low-flow years of 1976 and 1977, while immigration and emigration were important in winter and spring in 1978 and 1979. Second, changes in abundance alone are not accurate indicators of the extent of mortality:

increased abundance results if a high mortality rate is offset by a higher birth rate, as in May 1977, and decreased abundance can result from low mortality rates that coupled with even lower birth rates (June 1976). Abundance may change very little if birth and death are high but about equal, as they were in August and September 1979. Third, the annual pattern of each process varies separately in extent and timing. Migrations occur in wetter years in winter and spring, and depend on the extent of changes in Delta outflow. Birth rates can be high in spring and fall (1976), moderate and high alternately from spring through fall (1977), generally high from spring through fall (1978), or moderate from spring to mid-summer and high from mid-summer to fall (1979). Death rates, though similar in level over each year, oscillate erratically with no consistent seasonal trend. Not only is death only one of the four major processes which determine neomysis abundance in the study area, but also entrainment is only one component of the total mortality rate. Disease, predators, and other factors are also important, and high mortality rates occur in all seasons, not only in summer when entrainment mortality is greatest. Quantifying entrainment as part of the overall mortality rate and estimating the extent to which it affects abundance are the goals of the following section.

5.5.3 Impact

5.5.3.1 Entrainment Mortality Rates

In the previous section we have seen that the abundance of neomysis in the study area fluctuates widely under the influence of factors that cause varying rates of birth, immigration, emigration, and death. In the period 1976-1979 the range of estimated mortality rates—the daily proportions of the population dying—ranged from near zero to nearly 11 percent and averaged 5 percent. One component of this mortality is entrainment through the Pittsburg Power Plant. As stated initially, the main question is, to what extent does the proportion of the total (overall) neomysis mortality rate attributable to entrainment control neomysis abundance? In this section, two measures are developed which put entrainment cropping into perspective: (1) a daily entrainment mortality rate that can be compared with the overall mortality rate and (2) a measure of the compounded effect of daily entrainment mortality—annual percent reduction. Estimates were made for both 4-17 mm neomysis (1968-1979) and 2-17 mm neomysis (1974-1979). The latter estimates are directly comparable to overall death rates but do not cover the span of the former, which are used for a longer-term evaluation of entrainment.

The Delta and Tracy pumping plants were included in this assessment for the neomysis population in order to determine their relative influence on neomysis population abundance. The direct effects on the population by removal of individuals in the export water could be examined because (1) daily records of the volume of water exported through the pumping plants were available for the entire period encompassed by the neomysis surveys, and (2) neomysis density data collected from the vicinity of the pumping plants was available. The product of export flow and neomysis density provided estimates of daily cropping at the pumping plants which were comparable with entrainment cropping estimates made for the Pittsburg Power Plant.

Methods

The daily numbers of neomysis cropped by entrainment (k) were calculated as the product of the estimated densities of neomysis in entrained water multiplied by the volumes of water entrained and the proportion of entrained individuals cropped (Section C.4, Appendix C):

$$k = DRVM \qquad (5\text{-}9)$$

where

D = the average near-field density (no./m^3) of neomysis at CDF&G neomysis survey stations near the plant

R = the ratio of near-field densities to intake densities (R = 0.15 for the Pittsburg Power Plant)

V = volume (hm^3) of plant flow per day

M = the proportion of entrained neomysis that are cropped.

The variance of the cropping estimate k, Var(k), was calculated as

$$Var(k) = S.E.(D) \times (VM)^2 \qquad (5\text{-}10)$$

where

S.E.(D) = the standard error of D.

The CDF&G neomysis survey midchannel samples used to calculate D for the Pittsburg Power Plant were those collected at Stations 54, 56, 58, and 72 (Figure 5-9). The

density of neomysis decreases from midchannel to shore. Since the CDF&G neomysis survey takes midchannel samples, the ratio of intake densities to midchannel densities, R, derived in Section 5.5.2.1, was employed to calculate entrainment densities at the power plant at the time of each survey. The daily flow volumes, V, are monthly averages based on PGandE operating records.

The mortality factor, M, was 1 for the closed-cycle cooling system of the Pittsburg Power Plant Unit 7. At the once-through cooling water systems of the plant, M was taken to depend largely on mechanical damage and discharge temperature, with the latter depending on the intake water temperature and the delta-T imposed by the level of power production. Entrainment survival studies were carried out at the plant in 1978 and 1979, and a survival model incorporating four discharge temperature ranges, $\leq$30, 30-31.9, 32-33.9, and $\geq$34 C, was derived (Chapter 3). To use this model for years for which discharge temperatures were not available and to provide a consistent basis for analysis between years, the relationship between the monthly means of intake water temperature, generating load, cooling water flow, and discharge temperature was examined to derive predictive equations of monthly entrainment mortality proportions at the plant (the proportion of cooling water discharge flows in each of the four temperature ranges) (Section C.4, Appendix C). These two equations were used to calculate M, the mortality factor for the power plant for each month.

The proportion of the population cropped per day, P, was calculated as follows:

$$P = k/N \tag{5-11}$$

where

N = the estimate of neomysis abundance in the study area (Section C.2, Appendix C).

The variance of P, Var(P), was calculated as

$$Var(P) = \frac{Var(k) - 2\,Var(k)P + Var(N)P^2}{N^2} \tag{5-12}$$

where

Var(N) = the variance of the abundance estimate N.

Results and Discussion

The estimated numbers of 2-17 mm neomysis cropped by entrainment daily (k) at the Pittsburg Power Plant averaged 15 million, and ranged from 0 to 237 million (Table 5-9). Estimates of the proportion of the population cropped (P) averaged 0.0003 (0.03 percent), and ranged from 0 to 0.002 (0-0.2 percent).

The results for 4-17 mm neomysis for the period 1968-1979 are also summarized in Table 5-9. The 12-year mean daily number of 4-17 mm neomysis cropped (k) at the Pittsburg Power Plant during the period 1968-1979 is 10 million, with estimates ranging from 0 to 75 million. Estimates of the proportion of the population cropped (P) average 0.0003 (0.03 percent) and ranged from 0 to 0.0019 (0-0.19 percent).

In situ measurements by EA of entrainment cropping at the Pittsburg Power Plant for the period March 1978 - March 1979 resulted in an estimate of 4.0 billion neomysis cropped, which is greater than the 2.9 billion estimated from CDF&G data, as described above, but the two approaches result in similar estimates.

Figure 5-17 shows the abundance (N), the total overall mortality rate (d), and the daily instantaneous entrainment cropping rate (P) for each survey between 1976 and 1979. Throughout the period, P remains less than d in every survey. Those instances when the entrainment cropping for the Pittsburg Power Plant approaches d are infrequent and occur only when d is low. Such cases occurred in mid-May and early August 1976 and in early May and July 1979.

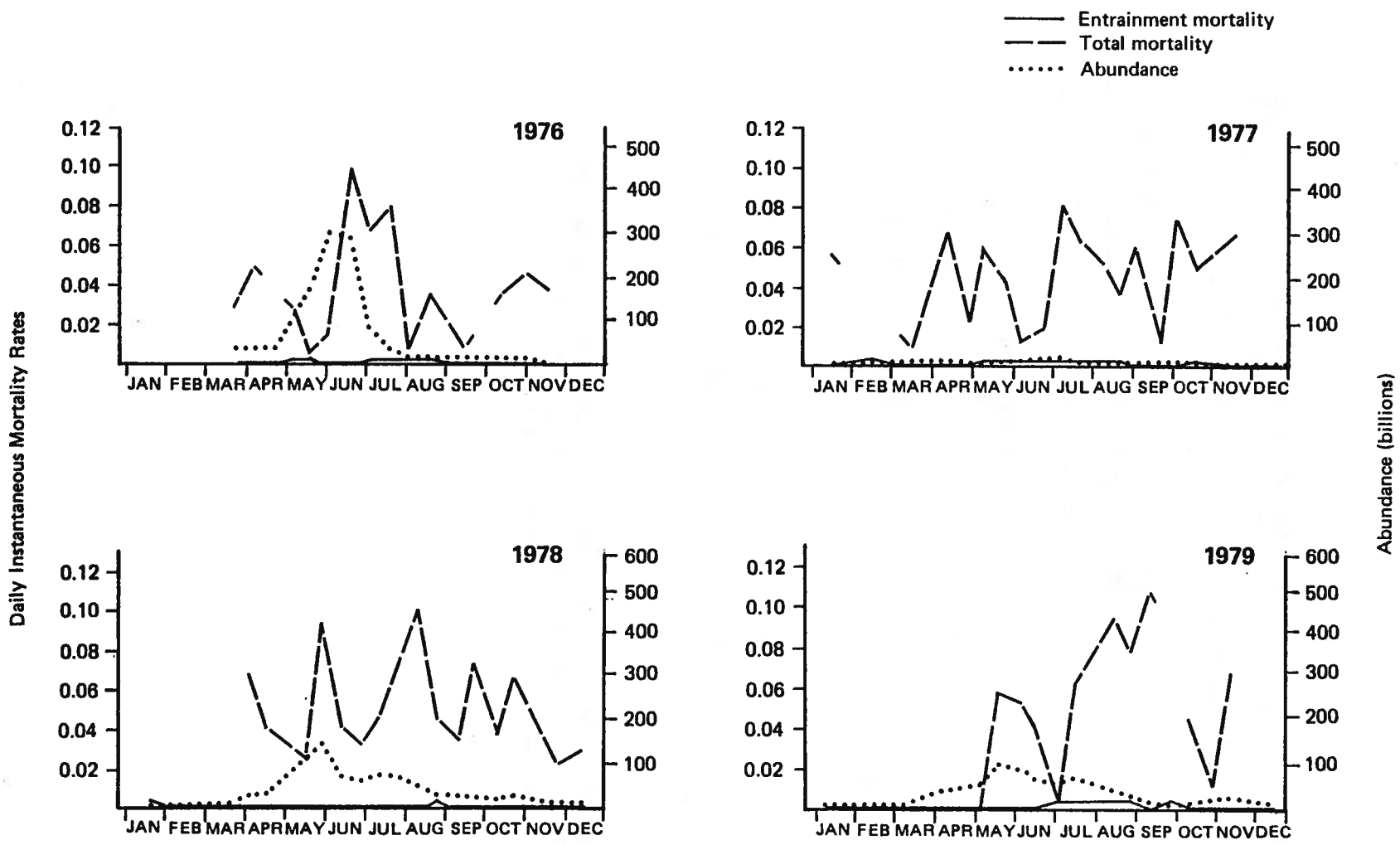
A comparison of the estimates of P for the Pittsburg Power Plant with estimates of neomysis abundance do not indicate a strong relationship between the two parameters. Values of P in spring were greater in 1976 than in 1978 and 1979, but abundance in spring was also greater in 1976 than in 1978 and 1979. These results indicate that higher entrainment cropping is correlated with more instead of fewer neomysis, the opposite of what might be expected if entrainment cropping controlled abundance.

There is little in a comparison of entrainment cropping and total mortality rates to indicate that entrainment at the plant controls neomysis abundance to any great extent. Mortality is only partially controlling abundance along with migration, and

TABLE 5-9 A SUMMARY OF ESTIMATED NUMBERS[a] OF *NEOMYSIS MERCEDIS* CROPPED DAILY DUE TO ENTRAINMENT, AND CORRESPONDING INSTANTANEOUS DAILY MORTALITY RATES

Site	Number Cropped per Day (millions)			Instantaneous (Daily) Mortality Rate		
	Mean	Min[b]	Max	Mean	Min	Max
		2-17 mm (1974-1979)				
Pittsburg 1-7	15	0	237	0.0003	0	0.002
Delta and Tracy pumping plants	18	0	320	0.0003	0	0.004
		4-17 mm (1968-1979)				
Pittsburg 1-7	10	0	75	0.0003	0	0.0019
Delta and Tracy pumping plants	38	0	512	0.0011	0	0.016

(a) Predicted on the basis of near-field densities and design cooling water flows; means are 6-year (2-17 mm) or 12-year (4-17 mm).
(b) Lowest (highest) estimate for any survey.



Gaps mark periods when significant migration precluded estimation of death rates.

Figure 5-17. Estimated total mortality and power plant entrainment mortality rates and overall abundances of *Neomysis mercedis* in the Bay-Delta study area, 1976-1979.

birth, as discussed in Section 5.5.2. Immigration and emigration occasionally reached levels higher than 1 percent per day, and births averaged 4 percent, about the same rate as overall deaths (5 percent). The 0.03 percent average proportion of the 2-17 mm population cropped (P) at the Pittsburg Power Plant is only 1/160 percent of the 4.67 percent average total mortality rate due to all causes, d, described in Section 5.5.2. Thus, it appears that entrainment has little control over changes in the abundance of neomysis.

The energy and nutrients in neomysis cropped at the Pittsburg Power Plant are not completely lost from the nutrient cycle and energy flow in the Bay-Delta food web (Section 2.2.6.1). Neomysis in the discharged cooling water are fed upon by fish and macroinvertebrates, and nutrients released by the decomposition of neomysis not surviving entrainment are utilized by algae, bacteria, and fungi, which are utilized as food by zooplankton.

5.5.3.3 Percentage Reduction

Percentage reduction is a theoretical estimate of the proportion by which population abundance is lower over some period than it would otherwise have been. For example, an estimate of annual percentage reduction, due to a given source of mortality or removal, of 20 percent means that the observed abundance at the end of a year is estimated to be 20 percent below what it would have been if the source had not been operating.

An important assumption underlying the interpretation of percentage reduction estimates is that there is no compensation, i.e., that eliminating a source of entrainment cropping does not change the rates of emigration, immigration, birth, and other sources of mortality. There is no evidence that the other mortality rates would remain constant when a source of mortality were removed, and there is reason to believe that in fact the rates would change. For instance, predation rates could increase because more prey would be available, or starvation might result when a decrease in entrainment mortality left too many individuals needing food. Thus, the assumption of no compensation represents an unlikely set of conditions, and results in a maximal estimate percentage reduction.

The annual period used in this analysis extends from January 1 through December 31 of each year from 1969 through 1979. This time period was chosen because neomysis

pass through regular yearly cycles of abundance and growth, which are indicated by minimum abundance in winter, the yearly cycle of growth-rate-determining temperatures, and the appearance in winter of individuals in the largest size classes. The methods used in calculating neomysis percentage reduction are described in Section C. 2, Appendix C.

Annual percentage reduction estimates at the Pittsburg Power Plant range from 2.7 to 17.0 percent (Table 5-10). Ranges are similar for the 4-17 mm and 2-17 mm neomysis size groups, being 2.7-16.0 percent and 2.8-17.0 percent, respectively.

Estimates of percentage reduction and the confidence interval of each estimate for 4-17 mm neomysis for the Pittsburg Power Plant are plotted in Figure 5-18. The pattern of estimates for the Pittsburg Power Plant over the period 1969-1979 is oscillatory, with a moderate peak in 1970-1973, a minimum in 1975, and a higher second peak in 1976-1979. Statistically significant changes between years were the 1969-1970 increase and the 1973-1974 decline and the 1975-1976 increase as indicated by the lack of overlap in confidence intervals.

Estimates of annual percentage reduction for the Delta (SWP) and Tracy (CVP) pumping plants combined range from 3.3 to 74.0 percent (Table 5-11). Estimates for 1974-1979 are greater for the 2-17 mm size group than the 4-17 mm size group. Percentage reduction estimates for 4-17 mm neomysis at the Delta and Tracy pumping plants were greatest over the period 1970-1972 (Figure 5-18). After 1972, a large, statistically signifcant decline to lower and gradually declining levels was observed.

The highest percentage reduction estimates at the Pittsburg Power Plant were those for the dry years of 1976 and 1979. These findings are consistent with low Delta outflow conditions: higher neomysis densities are found further up in the Delta, along with the entrapment zone, and abundance in the study area is low. The estimates for the Delta and Tracy pumping plants showed no obvious correlation with the dryness of the year.

It is unlikely that these estimated levels of reduction are ever reached in the Bay-Delta, because losses are probably compensated for in the remaining population. Since compensation is beyond accurate quantification, accurate estimation of the true level of percent reduction is presently impossible, and the question of the extent

TABLE 5-10 ESTIMATED PERCENTAGE REDUCTION OF THE 4-17 MM AND 2-17 MM *NEOMYSIS MERCEDIS* POPULATIONS DUE TO ENTRAINMENT CROPPING AT THE PITTSBURG POWER PLANT, 1969-1979

Year	4-17 mm		2-17 mm	
	Reduction	Standard Error	Reduction	Standard Error
1969	2.7	0.3		
1970	7.6	2.3		
1971	5.7	1.2		
1972	7.4	1.0		
1973	8.5	1.1		
1974	2.8	0.3	2.8	0.3
1975	2.8	0.3	2.8	0.3
1976	16.0	2.1	17.0	2.7
1977	9.1	1.7	7.2	1.4
1978	8.7	1.2	9.3	1.3
1979	14.9	1.9	15.2	2.3

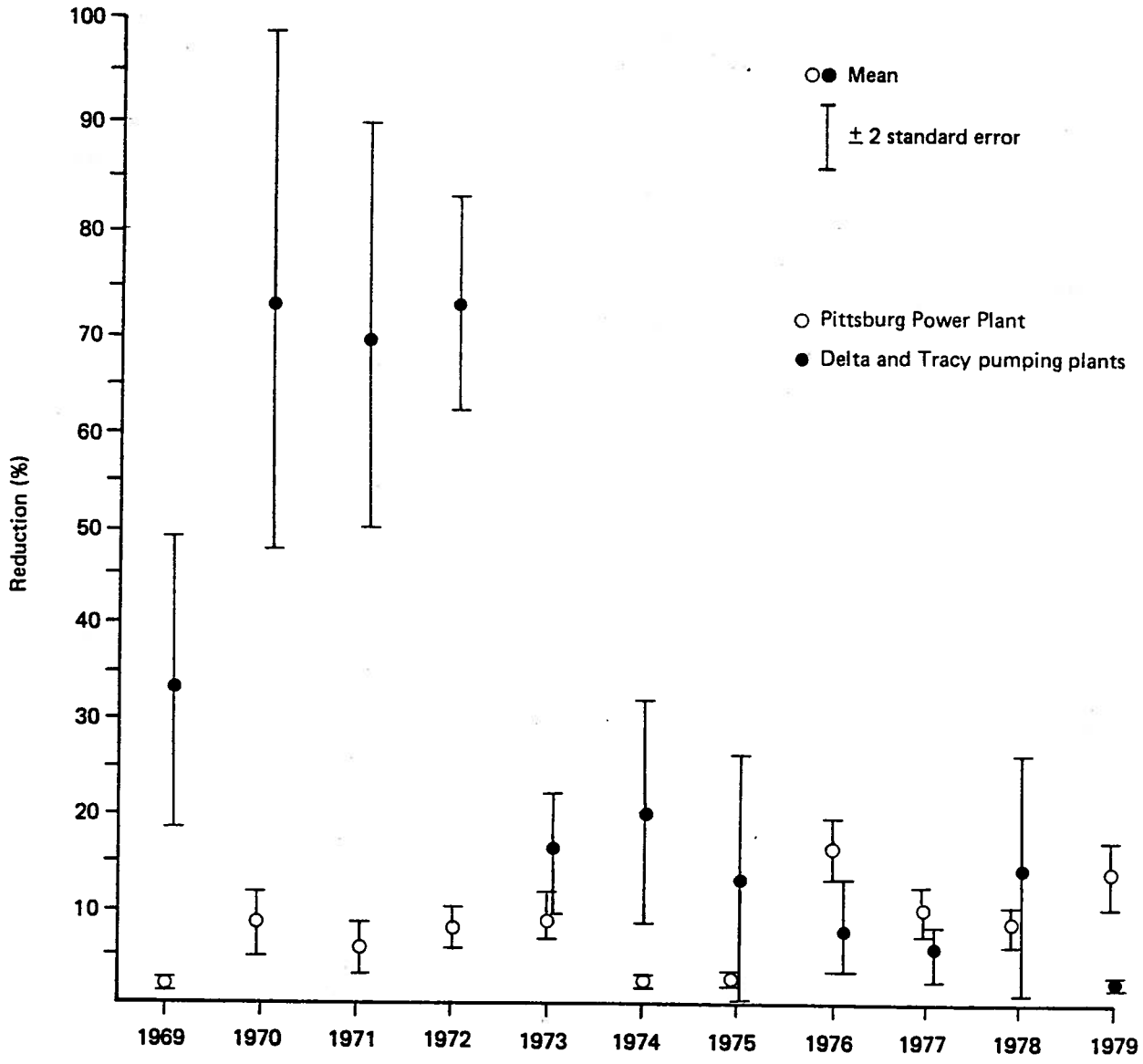


Figure 5-18. Percentage reduction of the Neomysis mercedis population of the study area due to entrainment at the Pittsburg Power Plant and at the Delta and Tracy pumping plants, 1969-1979.

TABLE 5-11 ESTIMATED PERCENTAGE REDUCTION OF THE 4-17 MM AND 2-17 MM *NEOMYSIS MERCEDIS* POPULATIONS DUE TO ENTRAINMENT CROPPING AT THE DELTA PUMPING PLANT (BYRON) AND THE TRACY PUMPING STATION, 1969-1979

Year	4-17 mm		2-17 mm	
	Reduction	Standard Error	Reduction	Standard Error
1969	33.6	8.1		
1970	74.0	13.1		
1972	74.0	5.6		
1973	15.8	3.9		
1974	20.4	6.3	41.0	13.6
1975	13.2	7.3	16.9	7.8
1976	8.8	2.5	9.4	2.6
1977	4.7	1.5	16.1	4.7
1978	14.6	7.0	17.5	8.9
1979	3.3	0.7	4.0	0.7

of reduction in abundance remains intuitive. Also, since compensation may occur on a daily basis and since entrainment accounts for only 0.6 percent of the overall mortality rate in the study area, compensation might take the form of slightly reduced natural mortality, so that the overall mortality rate might not change as a result of entrainment cropping.

5.5.4 Effects on Striped Bass

A reduction in neomysis abundance can only affect striped bass production if the level of neomysis abundance is limiting. If more neomysis are available than are required to meet the needs for survival and growth of young bass, then some reduction in neomysis abundance could occur with little or no effect on striped bass.

Neomysis constitute less than 20 percent of the striped bass diet until the bass reach 12 mm (Chadwick 1979; CDF&G unpublished data) (Figure 5-19). Judging from stomach content analysis, before striped bass are 12 mm long the abundance of other prey species, particularly copepods and cladocerans, dominates the effect that food has on striped bass survival (Figure 5-20). By 17.5 mm, striped bass are utilizing neomysis extensively, and they continue to do so for the next year (Chadwick 1979). During that period neomysis may be limiting.

5.5.4.1 The Abundance Correlation Approach

One approach to determining whether neomysis limit striped bass production is to compare abundance indices. A strong correlation between the two indices represents an association between two parameters and provides support for the hypothesis of limitation; although it does not prove a cause-and-effect relationship. The correlation approach was taken by Chadwick (1979), who suggested that food availability, i.e., average neomysis abundance over the 5-month period from June through October, possibly limited striped bass survival in the period 1976-1978. The reasoning was based on the observation that the abundance of neomysis was positively correlated with striped bass production.

Using the correlation approach and selecting 4-17 mm neomysis abundance averages for the 1.5-month period before the striped bass 35-39 mm index was determined (between mid-July and mid-August; see Section 5.6), neomysis abundance is indicated as not always limiting bass production (Figure 5-21). The index of striped bass year

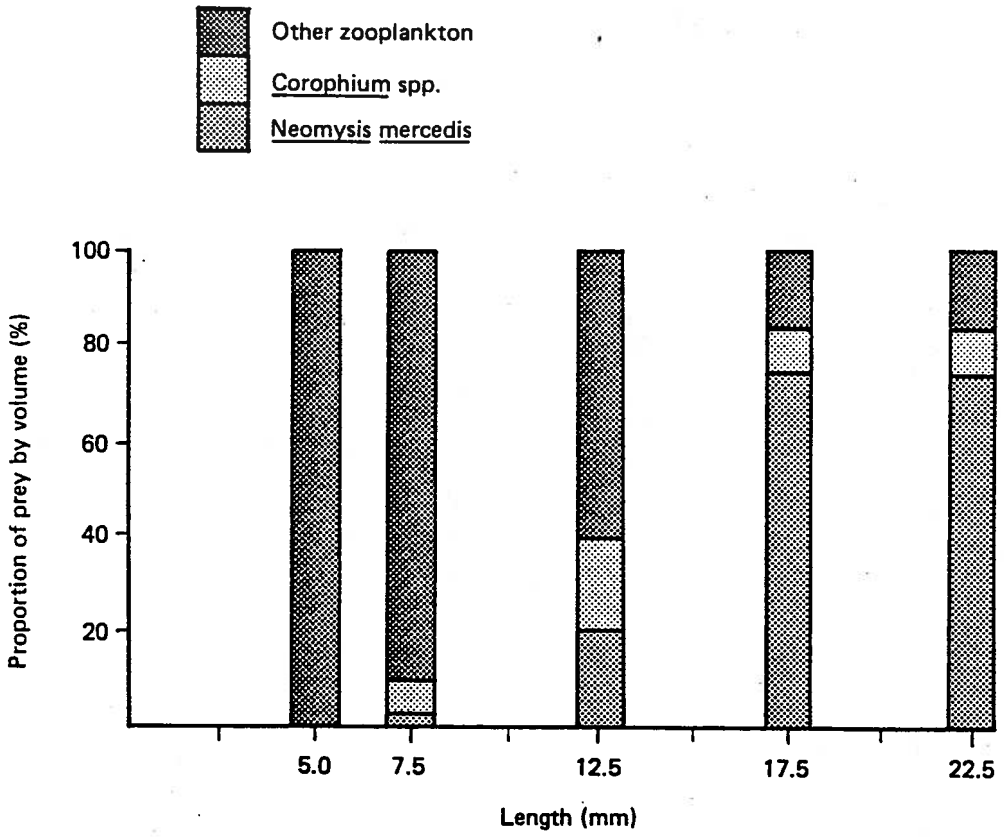


Figure 5-19. Proportions of various prey in the stomachs of young striped bass, April-June 1971. (Adapted from Chadwick 1979.)

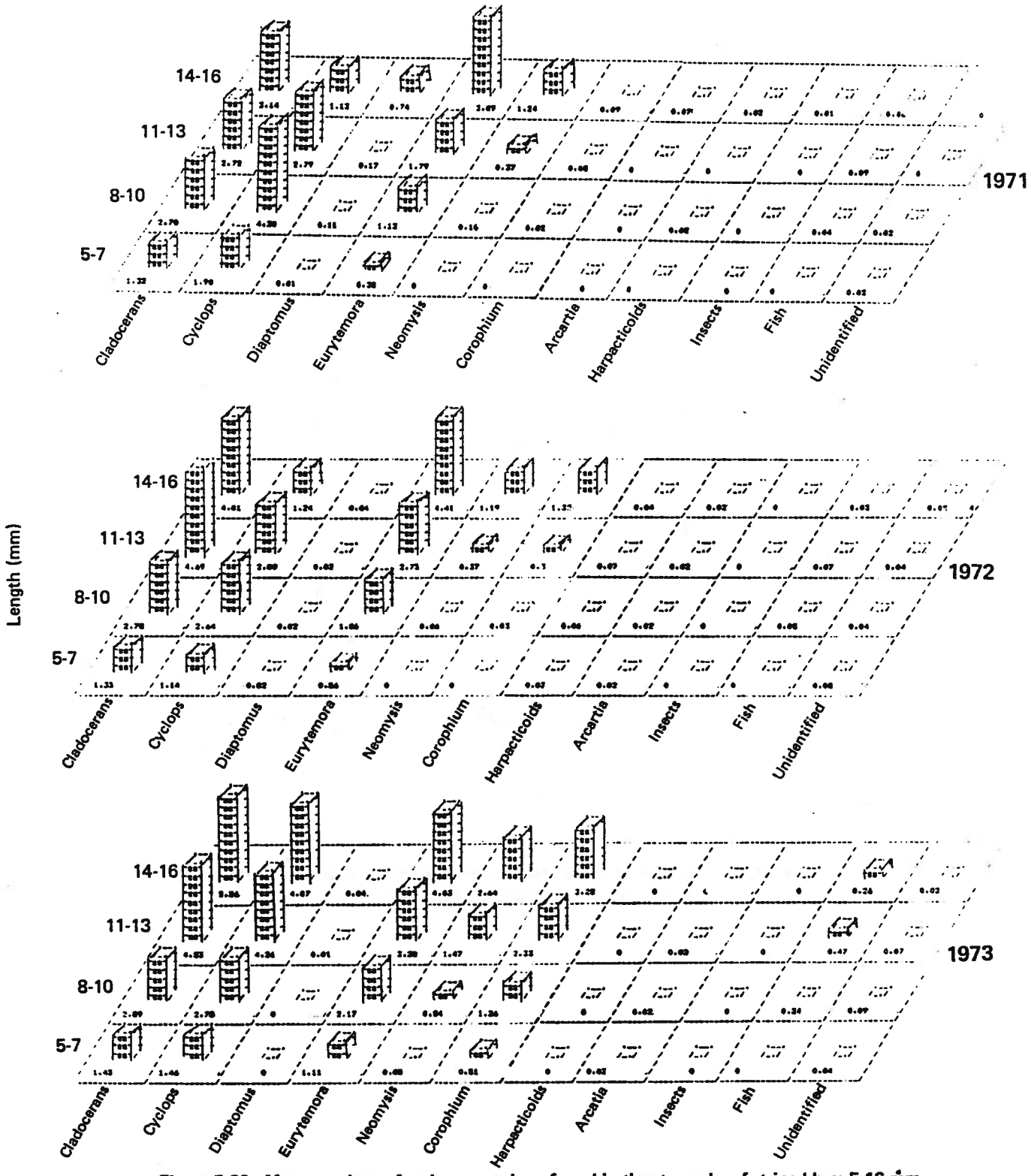


Figure 5-20. Mean numbers of various organisms found in the stomachs of striped bass 5-16 mm long, 1971-1973.

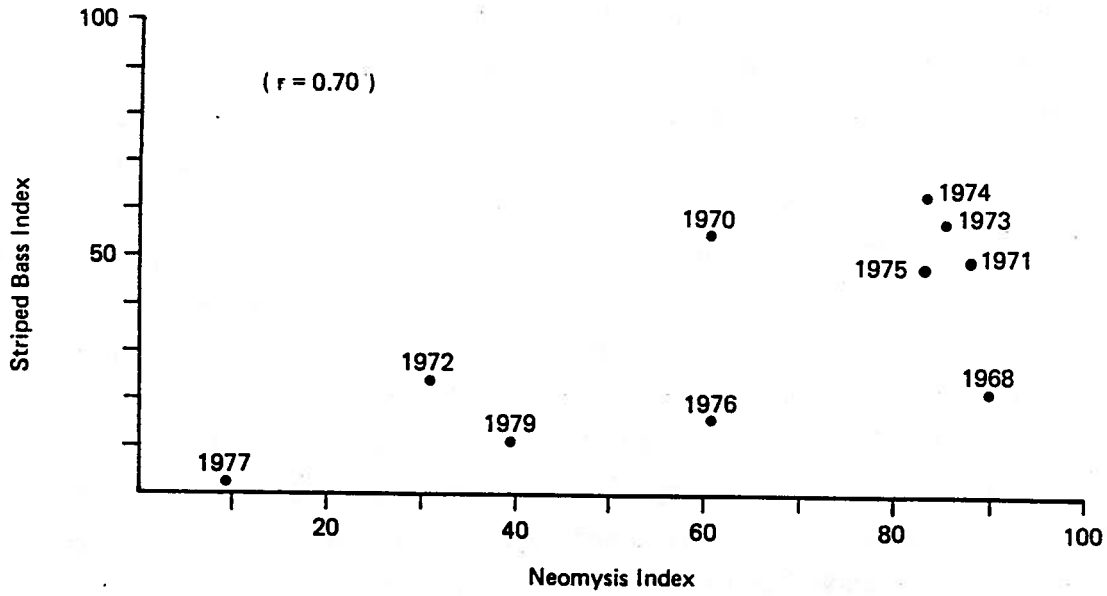


Figure 5-21. Correlation of the 38-mm striped bass index with an index of mean *Neomysis mercedis* abundance in the 1.5 months before the setting of the index, 1968-1979. (Striped bass index not available for 1969 or 1978.)

class strength was positively correlated with the index of neomysis abundance (r = 0.70). In 1968 and 1976, however, the neomysis abundance index was high but the striped bass index was low, which weakens support for the hypothesis that neomysis abundance is the limiting factor for striped bass in the Bay-Delta. Low springtime Delta outflows may have kept bass production low while low predation by bass, due to low bass production, permitted high neomysis abundance. Thus, any effect that entrainment at the Pittsburg Power Plant may have on neomysis abundance will not necessarily affect striped bass.

5.5.4.2 The Predation Rate Approach

A second approach to examining the effect of neomysis entrainment on striped bass production is extrapolation of predation and entrainment mortality rates to striped bass production. As indicated in Table 5-12, during the period of their greatest abundance, small (11-18 mm) striped bass prey on neomysis at rates greater than the neomysis entrainment mortality rates. For instance, the average daily instantaneous neomysis mortality rate due to predation by 11-18 mm bass is estimated to have been 0.015 in June of 1978, compared with average entrainment and total overall mortality rates of 0.00015 and 0.0385, respectively. The predation rate of striped bass on neomysis in June 1979 of 0.004 compares to entrainment mortality and total overall mortality rates of 0.0005 and 0.046, respectively. It can be estimated that daily entrainment cropping at the Pittsburg Power Plant amounted to 1 and 12 percent of the amount of neomysis cropping by 11-18-mm bass in June of 1978 and 1979, respectively. These estimates are based on the abundance of 11-18 mm striped bass and of 2-17 mm neomysis at the time and on an average of 3 neomysis consumed per bass per day for 11-13 mm bass and 6 for 14-16 mm bass. The consumption rate is based on an average of 1 and 2 neomysis for 11-13 mm and 14-16 mm bass, respectively (Figure 5-20), and an assumption that the average bass feeds three times daily, based on data in Eldridge et al. (1981).

Estimates of the percentage reduction in 35-39 mm striped bass can be obtained in order to assess possible effects of neomysis entrainment on striped bass production. It was assumed that bass production is directly proportional to predation on neomysis, that rates are constant, that no compensation occurs, and that it takes 60 days for a bass to grow from 11 mm to 38 mm. The estimate of potential percentage reduction in bass production due to neomysis entrainment, R, is determined as follows:

TABLE 5-12 ESTIMATED NUMBERS OF 2-17 MM *NEOMYSIS MERCEDIS* CONSUMED BY STRIPED BASS AND THE ASSOCIATED DAILY INSTANTANEOUS PREDATION RATES, ON SEVERAL DATES IN 1978 AND 1979

Date	Number Consumed (millions)[a] Striped Bass Size Class 11-14 mm	15-18 mm	11-18 mm	*Neomysis mercedis* Abundance (billions)	Daily Instantaneous Predation Rate
1978					
26 May	193	47	240	157	0.00153
7 June	1,080	1,000	2,080	114	0.01825
14 June	693	478	1,171	71	0.01649
21 June	274	381	655	68	0.00963
12 July	5	23	28	82	0.00034
1979					
16 May	6	0	6	97	0.00006
6 June	175	31	206	88	0.00234
20 June	149	216	365	64	0.00570
11 July	2	1	3	59	0.00005

(a) On the assumption that 11-14 mm striped bass eat an average of 3 *N. mercedis* per day, and 15-18 mm striped bass eat an average of 6, and using the estimated numbers of striped bass in the study area (Section 5.6).

$$R = 100(1 - Q^{60}) \tag{5-13}$$

where

Q = the theoretical proportion of potential striped bass production (which is proportional to predation on neomysis) that would occur without entrainment cropping.

Q may be calculated as follows:

$$Q = \frac{B}{B + BE} = \frac{1}{1 + E} \tag{5-14}$$

and

B = average daily instantaneous striped bass predation rate on neomysis in June

E = average daily instantaneous entrainment cropping rate for neomysis in June, for the Pittsburg Power Plant.

Assuming that the average value of E for June 1978 was 0.00015 and for June 1979 it was 0.0005, the resulting estimates of R are 0.6 and 3.0 percent, respectively. These estimates indicate that entrainment cropping of neomysis by the Pittsburg Power Plant reduced the production of young striped bass by 3.0 percent or less.

Since some compensation occurs, and since striped bass production probably is not proportional to neomysis abundance, these estimates are maximums, suggesting that entrainment of neomysis at the Pittsburg Power Plant probably has even less control over the production of 38-mm bass than the low amount indicated by the 0.6-3.0 percentage reduction estimates.

A result of this analysis of predation that is germane to the question of controls over neomysis abundance is that the 11-18 mm striped bass population appears to control neomysis abundance in some months. The estimated predation rates amounted to 39 and 9 percent of the average overall mortality rate, d, for June in 1978 June 1979, respectively. Since there were a variety of causes of death, including predation by striped bass, other fishes, and other invertebrates as well as

entrainment, disease, cannibalism, and senescence, one third of the deaths in June 1978 going to a subpopulation of one species (striped bass) may be the greatest proportion of deaths due to a single entity. Futhermore, the hypothetical percentage reduction in neomysis abundance after 30 days at the 1978 estimated predation rate of 0.015 is 36 percent, and at the 1979 predation rate of 0.004, the estimate is 11 percent. Thus, striped bass may control neomysis abundance, at least during June of some years, such as 1978, by dominating the overall death rate.

5.5.5 Future Conditions

Discussions of impacts on neomysis up to this point have dealt with historical conditions. However, conditions in the Bay-Delta, and in particular Delta outflow, will change over the next ten years. The future conditions of interest here are average mean monthly Delta outflows, which in the analysis are weighted by the frequency of each type of water year over the period 1921-1971, historically and as they would have been under the 1985 and 1990 water plans. Effects of future conditions on neomysis entrainment are assessed in this section by extrapolation of the association between observed levels of average daily instantaneous entrainment mortality rates, P, for 4-17 mm neomysis and concurrent mean monthly Delta outflows.

Outflow influences neomysis distribution by controlling the position of the entrapment zone, where the densities of neomysis are generally highest. The estimated Delta outflows that will place some portion of the entrapment zone off the Pittsburg Power Plant (calculated using the methods described in Appendix A) are 4,000-14,000 cfs.

Delta outflow (and thus the location of the entrapment zone), in conjunction with power plant operating conditions, influences entrainment cropping. The combined effect of these factors determines the entrainment mortality rate, which is plotted for 4-17 mm neomysis for all observations over 1968-1979 against Delta outflow, for the Pittsburg Power Plant and for the Delta and Tracy pumping plants, in Figure 5-22. Results for the Pittsburg Power Plant suggest that entrainment cropping does not increase until Delta outflow decreases below about 20,000 cfs and that it is greatest near outflows of 3,000-6,000 cfs. These results confirm the prediction of greater entrainment cropping at lower Delta outflows,

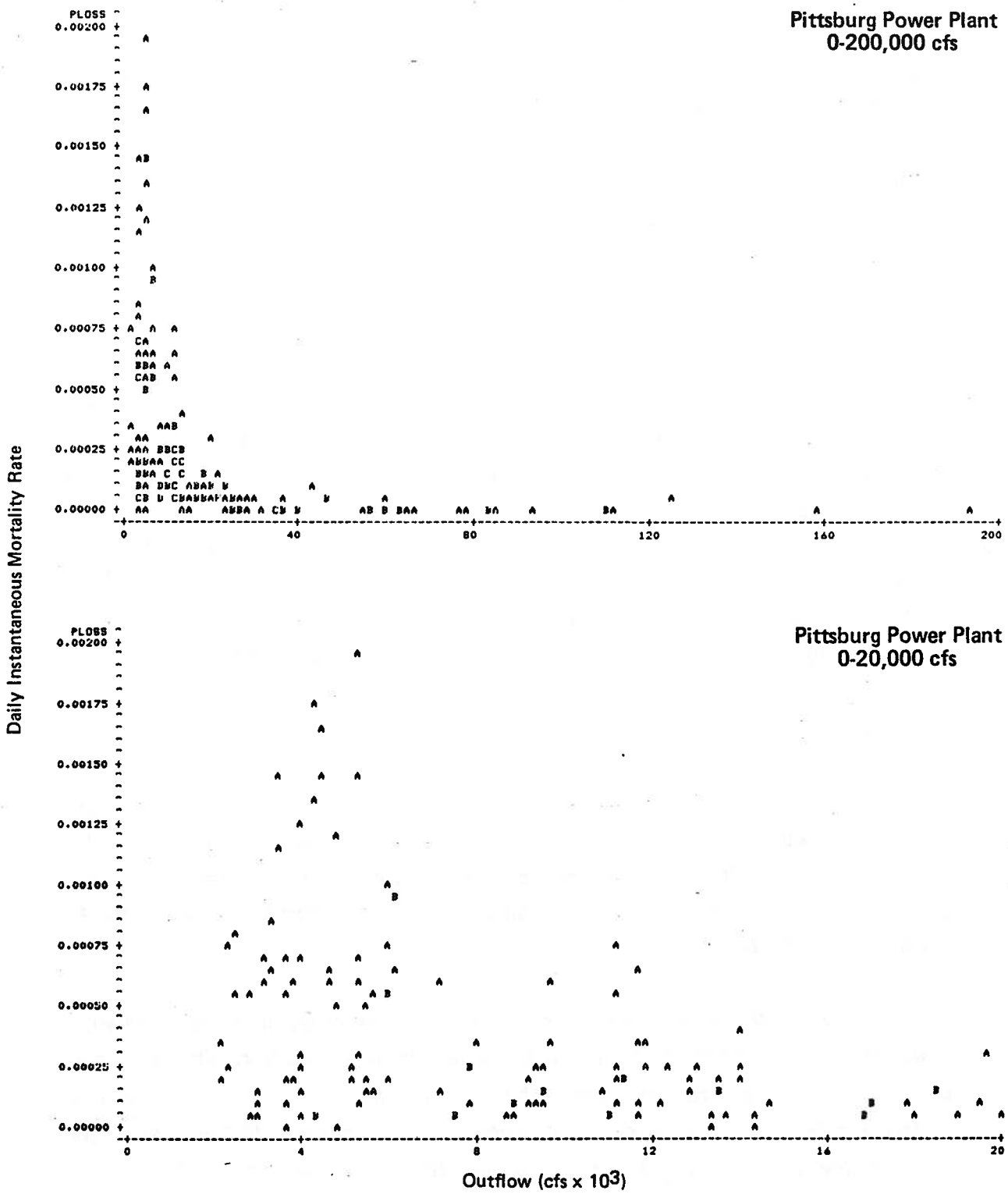


Figure 5-22. Delta outflow vs. entrainment mortality rates for Neomysis mercedis at the Pittsburg Power Plant and at the Delta and Tracy pumping plants, 1976 -1979.

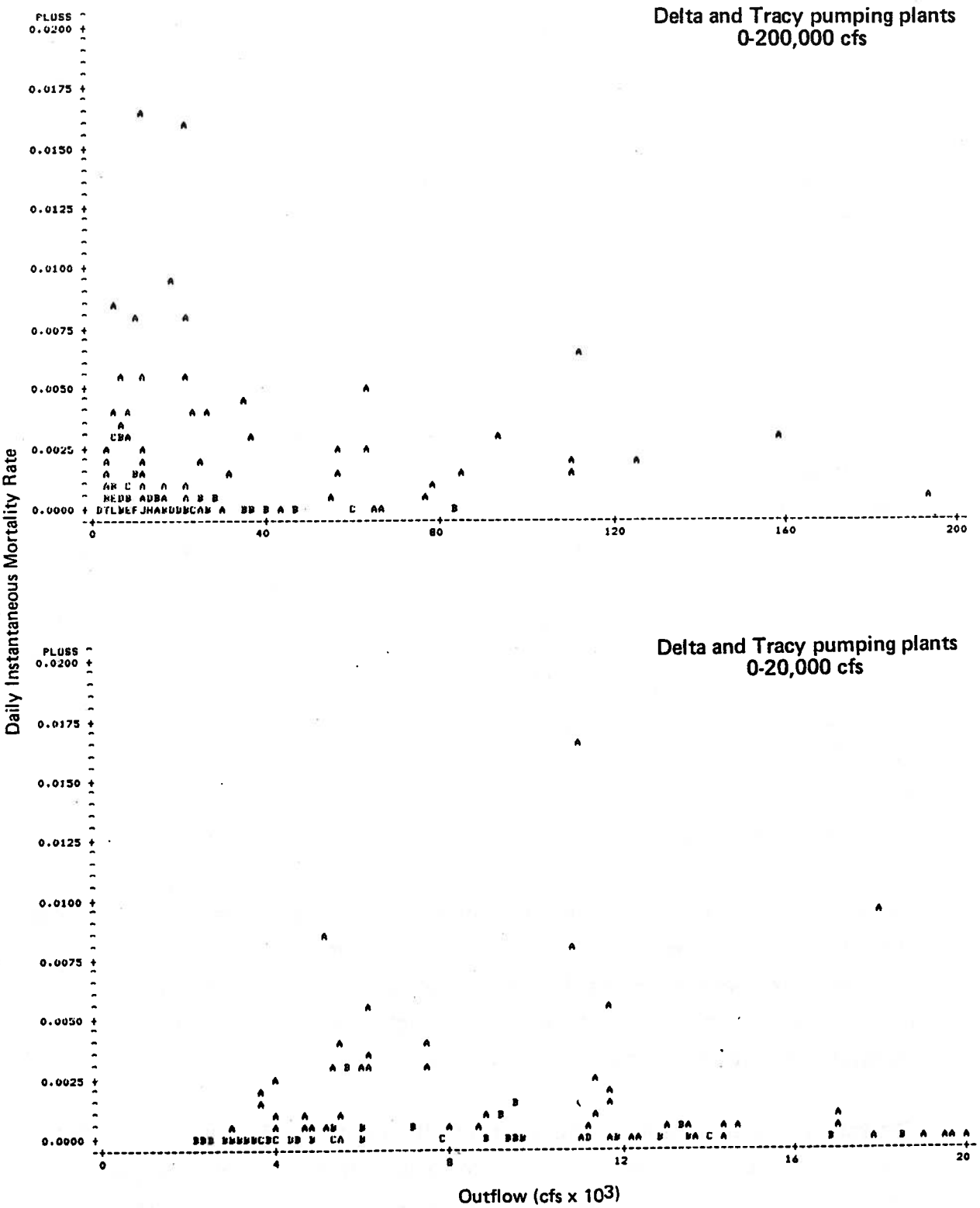


Figure 5-22. Cont.

which is based on the relationship of the entrapment zone location to Delta outflow. No obvious overall change in cropping with respect to Delta outflow is evident for the Delta and Tracy pumping plants.

Since only low values of P were observed when Delta outflows were >20,000 cfs, future mean monthly Delta outflows >20,000 cfs will, on the average, have very small effects on the magnitude of entrainment cropping. In general, months with outflow rates >20,000 cfs are January through April, with May and December on the borderline (Table 5-13).

Months with future mean Delta outflows <20,000 cfs are May through December. By determining the change in entrainment cropping over the anticipated change in average monthly Delta outflows (Table 5-13), monthly assessments are made for the Pittsburg Power Plant for 4-17 mm neomysis for each month over the period 1968-1979 (Figure 5-22). Little change in cropping is anticipated for May and December under the proposed 1985 and 1990 water plans. Cropping in June will increase under the future conditions. For July, average 1985 cropping levels will be lower than they were historically, but they will increase to a moderate level under the 1990 water plan as mean monthly Delta outflow changes from 5,100 to 8,000 cfs and finally to 6,500 cfs. Extrapolations for August indicate that entrainment cropping will decline from historical levels under the 1985 water plan and increase under the 1990 water plan. Lastly, September, October, and November trends are similar, each with moderate historical cropping levels which are expected to increase under the 1985 and 1990 water plans.

These extrapolations are based on the assumption that power plant operation and neomysis abundance remain unchanged on the average. Since a wide variety of Delta outflows were observed during the period from 1968 to 1979, the results reflect a wide range of Delta outflow effects on neomysis abundance, and therefore encompass future average flow conditions.

The values of entrainment mortality rates in the future should remain at observed levels ($\leq$0.2 percent), which appear to exert little control over neomysis abundance. Since they represent only 0.6 percent of the overall mortality rate, even for those months in which average Delta outflow changes may result in an increase cropping to the observed maximum, the mortality rate attributable to entrainment at the Pittsburg Power Plant should still remain a small part of the

TABLE 5-13 ESTIMATED MEAN MONTHLY DELTA OUTFLOW (cfs x 10^3) FOR THE PERIOD 1921-1971 AS IT WAS HISTORICALLY AND AS IT WOULD HAVE BEEN IF THE CONDITIONS DESCRIBED IN THE 1985 AND 1990 WATER PLANS HAD BEEN IN EFFECT

	Conditions		
Month	Historical	1985 Plan	1990 Plan
January	50	34	31
February	68	46	41
March	53	33	29
April	51	24	21
May	41	21	18
June	32	13	12
July	5.1	8.0	6.5
August	3.1	7.2	3.3
September	6.7	5.8	4.1
October	9.3	6.1	5.3
November	16	7.9	5.3
December	34	21	17

total mortality rate. Therefore, under future conditions entrainment cropping at the Pittsburg Power Plant should exert little control over neomysis abundance.

It appears that future changes in the cropping rate for neomysis at the Pittsburg Power Plant will not change the production of 38 mm striped bass. Certainly, neomysis is an important component of the diet of young striped bass, as mentioned initially. However, entrainment cropping appears to have little control over neomysis, and the extent to which neomysis abundance determines young striped bass production is uncertain. So, future entrainment cropping of neomysis should exert little, if any, control over young striped bass production.

5.6 IMPACT ASSESSMENT FOR STRIPED BASS (MORONE SAXATILIS)

5.6.1 Introduction

The striped bass is a large (up to 1.25 m, 49 in.; and 41 kg, 90 lbs.), introduced fish abundant in the Bay-Delta and subject to the region's most intensive sport fishery. Native to the Atlantic and Gulf of Mexico coastal waters, estuaries, and rivers, striped bass quickly became established in the Bay-Delta after their introduction in 1879 and 1882. The Bay-Delta population remains the only major one on the Pacific coast, and recent reductions have prompted considerable interest in the environmental requirements of the species.

Adult striped bass use the Bay and coastal waters for feeding during the summer months, but in fall they move upstream, passing through Carquinez Strait in October and concentrating in Suisun Bay in November (Chadwick 1967). In spring, mature bass move upstream to spawn in the Sacramento River between Colusa and Sacramento and in the San Joaquin River between Antioch and Venice Island (Turner 1976).

Males mature at age 1-3; females usually at age 4-6 (Scofield 1931; Calhoun and Woodhull 1948). Both sexes commonly live and reproduce until they are 12-15 years old (females up to 30 years old), but older bass probably do not spawn every year (Merriman 1941; Moyle 1976a). Spawning is apparently triggered as water temperatures approach 15 C (Farley 1966). San Joaquin River spawning generally precedes Sacramento River spawning, because the water warms up earlier in the former.

Striped bass females, like most pelagic spawners, are prolific. In the Bay-Delta, 5- to

9-year-old bass produce about 75 percent of the eggs each year (CDF&G unpublished data). Eggs are planktonic and non-adhesive, and can be suspended by water currents greater than 30 cm/sec (1 fps). Suspension and initial exposure to fresh water are both essential for high egg survival (Albrecht 1964; Turner and Farley 1971). Once water hardening occurs, the eggs can tolerate salinities up to 10 ppt. Eggs hatch in 36-48 hours after fertilization when water temperatures are 17-18 C (62-65 F) (Mansueti 1958).

Because the eggs hatch quickly, they are not distributed far from the spawning areas. In the Sacramento River, most eggs hatch between Sacramento and Courtland (Turner 1976). In the San Joaquin River, where downstream flows are of lesser intensity and high tides and water exports cause reverse flows, eggs hatch mainly within the spawning area. Eggs tend to be concentrated near the bottom in mid-channel (Farley 1966). Striped bass eggs were collected in very low densities near the Pittsburg Power Plant (EA 1981a, d, e), primarily because spawning and egg hatching occurs upstream of the site. On the basis of egg densities in entrainment samples, 1 million eggs were estimated to have been entrained at the Pittsburg Power Plant (Table 5-3) during the period March 1978 - March 1979. The number of striped bass eggs entrained is relatively small when compared to the fecundity of the female. Depending on size, age, and degree of sexual maturity, a female striped bass in the Bay-Delta produces between 11,000 and 2 million eggs (Moyle 1976a) with an overall average of approximately one milliion eggs per female per year. Thus, the number of eggs entrained at the Pittsburg Power Plant represents the production of a few females. Additional discussion of trends in egg production in the Bay-Delta striped bass population and of factors affecting the number of eggs in the system each year is presented in Section 5.7.2.

Larvae (<6 mm) are quickly carried out of the upper Sacramento River into the Delta. Most enter the lower Sacramento River, but some may pass into the lower San Joaquin River via the Delta Cross-Channel and Georgiana Slough. Larvae become distributed thoughout the western Delta and Suisun Bay above the entrapment zone (Section 2.2.3). Larval densities decrease sharply at the upper end of the entrapment zone (known as the null zone), and few small larvae are found below that point (EA 1981a). Larger larvae (<16 mm) and small juveniles are found as far downstream as the middle of the entrapment zone (EA 1981a). The location of peak abundance ranges from Carquinez Strait to the western Delta, depending on the location of the entrapment zone, which in turn is determined by Delta outflow.

The planktonic larvae and early juvenile stages of striped bass, although distributed throughout the Bay-Delta (EA 1981a), were present near the Pittsburg Power Plant and were susceptible to entrainment. The estimated number of striped bass entrained was 283 million from March 1978 to March 1979, of which 144 million were estimated not to have survived entrainment. A total of 35 million were estimated to have been entrained between April 1979 and July 1979, of which approximately 23 million were estimated not to have survived entrainment (Table 3-18).

Young striped bass remain in and above the entrapment zone during their first summer, but migrate into San Pablo Bay and central San Francisco Bay in the fall (Sasaki 1966a; Ganssle 1966). Some young bass remain in the Delta during the winter and are concentrated in the lower San Joaquin River. As a result of their residence in the vicinity of the power plant, young striped bass (primarily young-of-the-year) are susceptible to impingement. The estimated number of striped bass impinged at the Pittsburg Power Plant was approximately 111,300 (Table 5-3). Very few yearling, subadult, or adult striped bass are impinged at the plant (Chapter 4).

The impact of the Pittsburg Power Plant's cooling water system on the striped bass population at the Bay-Delta is a function of numbers entrained and impinged and the proportion surviving. The numbers entrained and impinged appear high, especially in summer and fall (Chapters 3 and 4). Because spawning occurs primarily upstream of the plant (EA 1981a), the entrainment of striped bass eggs is minimal. The lower Delta serves as a nursery area for larvae and early juveniles, which are thus abundant in the area of the plant in spring and summer. Peak entrainment of striped bass occurred between April and July, a time when discharge temperatures often approached or exceeded upper lethal tolerance levels (Section 3.2), resulting in reduced entrainment survival. In late fall, although a portion of the population overwinters in the lower Delta, young-of-the-year striped bass concentrate in San Pablo Bay, reducing the risk of impingement. Thus, the effect of the plant on the population is primarily a result of entrainment of larval and juvenile fish, and only secondarily a result of the impingement of young-of-the-year.

General trends in the young-of-the-year and yearling striped bass abundance in the Delta, based on CDF&G midwater trawl survey data, are shown in Figure 5-23. The CPUE's for young-of-the-year indicate a decline in abundance between 1967 and 1978. The CPUE's for yearlings are variable within a relatively narrow range, with no

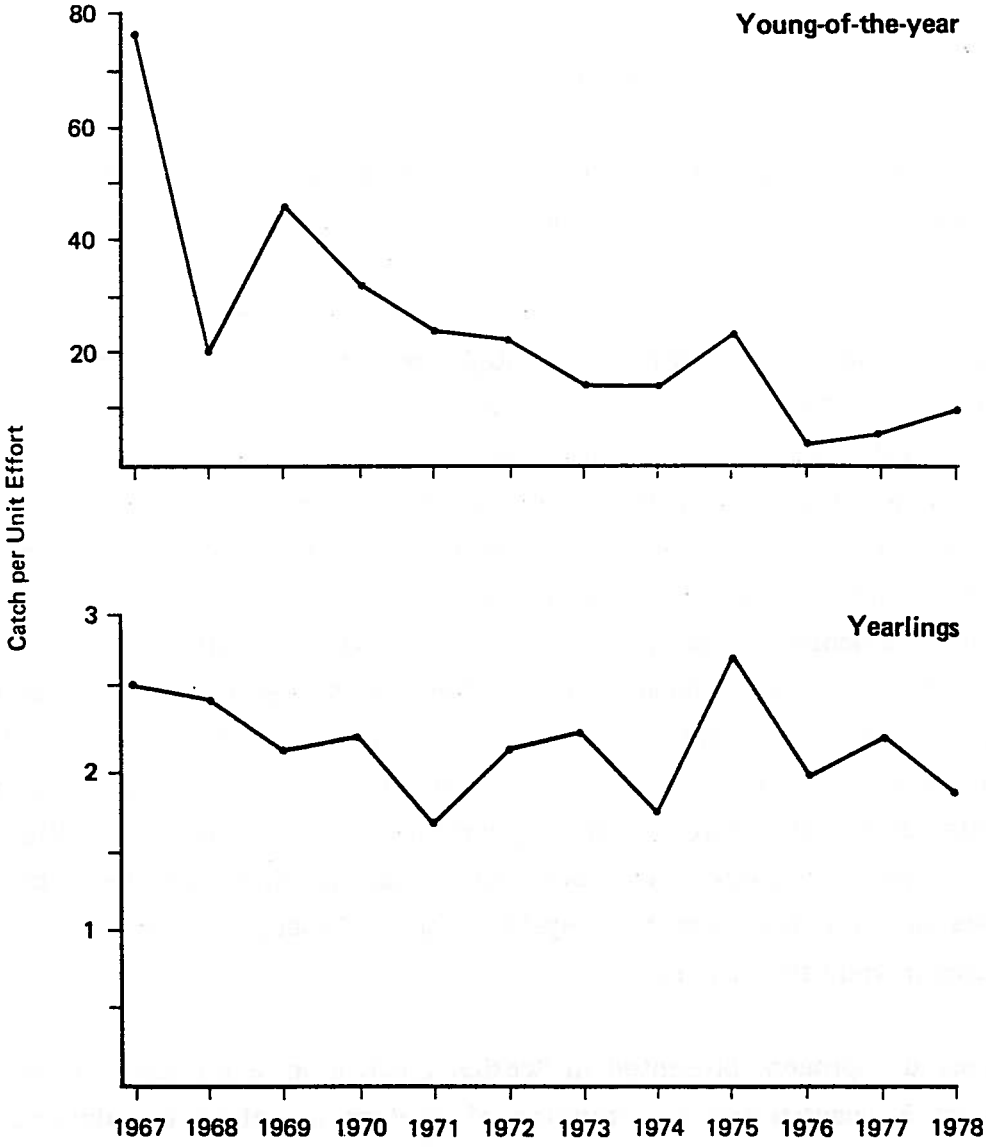


Figure 5-23. Annual average catch of young-of-the-year and yearling striped bass in the midwater trawl surveys of the California Department of Fish and Game in the Sacramento—San Joaquin Delta, 1967-1978.

clear trend in either increasing or decreasing catches. The linear regression slopes for both age classes are shown below:

	Slope	S.E.	T	P
Young-of-the-year	-4.30	1.13	-3.80	0.004
Yearlings	-0.003	0.03	-1.04	0.32

The significantly negative regression slope for young-of-the-year provides evidence of a decline in abundance, but this is not reflected in the regression slope for yearlings.

Because involvement of the early life stages with the plant's cooling water system is relatively high (Table 5-3) and the relative abundance of young-of-the-year striped bass is declining at a significant rate, the significance of entrainment and impingement cropping requires a more detailed analysis, which is presented in the following sections, to assess the potential for impact on the population. Two independent approaches are used to assess the significance of these losses. The first approach, presented in Section 5.6.2, is an equivalent adult model similar to that presented for northern anchovy and Pacific herring (Sections 5.4.2.1 and 5.4.2.2), which equates entrainment and impingement cropping (Chapters 3 and 4) with the number of adults that would have been likely to mature from the young lost, using estimates of natural survival from the young stages to maturity. The estimate can be compared with estimates of the total adult population size; it can also be compared with catch statistics from the Bay-Delta sport fishery to estimate the potential reduction in yield to the fishery.

The second approach, presented in Section 5.6.3, is an adult stock model, which is based on a quantitative investigation of factors affecting the dynamics of the Bay-Delta striped bass population. The adult stock model is used to estimate the structure and abundance of the striped bass population with and without the influence of intake-related losses, under present and future water management regimes. The population modeling approach is used to determine the relative significance of power plant cropping as a factor affecting the size of the Bay-Delta striped bass population. These models will be prefaced with a brief historical review of Bay-Delta striped bass studies which have led up to this impact assessment. The models draw on information from Section 2.2.7.2 on the life history and distributional characteristics of striped bass in the Bay-Delta.

Historical Review

The striped bass has demonstrated the capacity for adaptation and rapid population growth under a wide range of environmental conditions (Setzler et al. 1980). This capacity is best evidenced by the fact that the striped bass population of the Sacramento-San Joaquin river system originated with 135 yearlings 1.5-5 in. long released in Carquinez Strait in 1879 and 300 bass, 5-9 in. long, released in Suisun Bay in 1882 (Shebley 1917, cited in Sommani 1972). By 1915 the commercial catch of striped bass from the Bay-Delta had grown to 1.8 million pounds. Despite this capacity for rapid population growth, the strength of striped bass year classes is extremely variable. The recent decline in abundance during the past several decades has caused attention to be focused on the impact of water diversions for municipal and agricultural use on the striped bass population.

The striped bass population has been the subject of extensive study by the California Department of Fish and Game (CDF&G) since the late 1950s. Most of the research, centered on the impact of state and federal water projects on striped bass, indicated that environmental conditions have an important influence on spawning success and subsequent year class strength in the Bay-Delta. Earlier studies provided considerable knowledge of the life history, population dynamics, and basic environmental requirements of striped bass, but it was not until a large data base was established that specific patterns associating striped bass abundance with environmental factors emerged.

Turner and Chadwick (1972) reported high positive correlations between annual indices of young bass abundance (derived from summer tow-net survey data for 1959-1970) and mean daily June-July Delta outflow. The indices were also highly correlated with April and May outflows. In addition, the indices showed a high negative correlation with the percentage of Delta freshwater inflow diverted at the Delta and Tracy pumping plants in June and July. The correlation with outflow suggested that abundance was related to some environmental factor, such as food supply, while the correlation with percentage diverted suggested that the loss of young at diversions directly affects the number of young produced.

Chadwick et al. (1977) and CDF&G (1976) reported that indices for the years 1971-1976 were lower than would be expected from the relationship developed by

Turner and Chadwick (1972). They attributed the lower production of young to increases in the Delta water export rate, especially during May, from 1971 to 1976. The decline in production was apparent in the zone encompassing the Delta portion of the population. (The other portion of the population, located in Suisun Bay, remained as predicted from the earlier data.) The decline in the Delta was attributed to the direct loss of young at the diversions and/or a reduction in the food supply because of the higher flows in the Delta channels from the increased export rates at the Delta and Tracy pumping plants. The relationships developed, based on the 1959-1976 data, between young bass abundance, outflow, and rate of diversion at the pumping plants were used to evaluate the potential environmental impact of the Peripheral Canal (DWR 1976) and of the management practices impilied by the water quality standards prescribed in the Delta Water Plan (SWRCB 1978).

The abundance of young bass in 1977 and 1978 was lower than expected on the basis of the regression equations developed from the 1959-1976 data base. Chadwick (1979) reported that both the Suisun Bay and Delta indices were substantially lower than expected. The decline was attributed to unusually low levels of neomysis, a shrimp that is a major food item of the young bass (see Section 5.5), possibly brought on by the drought of 1976-1977.

Studies by CDF&G on neomysis since 1968 have indicated that its abundance is a function of habitat size and food supply, largely phytoplankton (Orsi and Knutson 1979); factors that are controlled by Delta outflow. Arthur and Ball (1979) suggested that the quantity of phytoplankton, zooplankton, and possibly striped bass are enhanced when the entrapment zone (the mixing zone between the freshwater and saltwater portions of the estuary, described in Section 2.2.3) is located in Suisun Bay, a situation that requires fairly high flows (110-700 m^3/sec; 4,000-25,000 cfs). At lower flows the entrapment zone was upstream in Delta channels, where there was a minimum of shoaling and less potential for production.

The effect of the power plant cooling water intakes on striped bass has been considered minimal by CDF&G (Chadwick and Stevens 1971; Chadwick et al. 1977). Stevens (1979) has said that "mortality caused by these plants is minimal compared to that caused by the water project and agriculture diversions and other environmental factors associated with the outflow." This conclusion was based on studies in the laboratory and at the power plants which indicated that survival of entrained striped bass was high. Another piece of evidence was that power plant effects were not

related to variations in the young abundance index during the period 1959-1976. Stevens (1980) has pointed out that the impact may be increasing, because increased power production causes more frequent occurrence of lethal temperatures in the water used for cooling (Finlayson and Stevens 1977).

Data from the CDF&G townet surveys were used by Tetra Tech, Inc. (PGandE unpublished b) in conjunction with thermal tolerance data for striped bass developed by Kelly and Chadwick (1971) to estimate the numbers of young-of-the-year striped bass cropped as a result of operation of the Pittsburg Power Plant cooling water systems. The computed numbers cropped and the specific mortality rates were compared to irrigation losses, export losses, and the specific rate of population decline. It was estimated that the Pittsburg Power Plant accounted for 3-6 percent of the observed total mortality rate. A regression model developed for the Bay-Delta striped bass population by Tetra Tech indicated that the recruitment of striped bass into the sport fishery was not related significantly to earlier young-of-the-year abundance. This would diminish the significance of young-of-the-year losses resulting from power plant operations on the adult population and the sport fishery.

As can be seen from this brief summary, much information has been gathered and much analysis has been done on the striped bass population of the Bay-Delta. It is this rich store of information that has made possible the equivalent adult analysis and the adult population model that are the subjects of the next two sections.

5.6.2 Equivalent Adult Model

The objective of the equivalent adult modeling approach to power plant impact assessment is to translate the number of young fish cropped by a plant into an equivalent number of adult fish, using the appropriate survival values for each length class or life stage. The estimate of the numbers of equivalent adults represented by fish that do not survive entrainment and impingement has two useful applications:

1. it can be used to put the cropping of the power plant into perspective with estimates of adult standing stock and
2. it can be compared to cropping associated with the sport fishery.

CDF&G has established an "index of recruitment," an estimate of the number of striped bass that have grown large enough to be kept by anglers, and are thus

"recruited" into the sport fishery. It is assumed, from long experience, that almost all of these will be fish in their fourth year of growth—three-year-old fish.

A modification of the equivalent adult model of Goodyear (Horst 1975; Goodyear 1978; Appendix C) was used, in which estimates of survival between the length groups entrained or impinged and three-year-old striped bass were multiplied by the length-group-specific level of entrainment and impingement cropping to yield the equivalent number of three-year-old striped bass lost, E:

$$E = \sum_{l=1}^{n} N_l S_{l,a} \qquad (5\text{-}15)$$

where

N_l = number of striped bass of length class l cropped by entrainment or impingement

$S_{l,a}$ = survival from the midpoint of length class l to three-year-old adult striped bass

n = number of length classes.

5.6.2.1 Estimation of Survival Rates

Survival rates for striped bass 7-49 mm in length, were calculated for nine length classes (7-10, 11-14, 15-19, 20-24, 25-29, 30-34, 35-39, 40-44, and 45-49 mm), using striped bass monitoring data collected by CDF&G and EA in the Bay-Delta between 1961 and 1979. The survival rate for each length class was calculated as 1 minus the instantaneous mortality rate.

Approximations for the instantaneous mortality rate for each length class were computed in the following manner:

Since

$$\frac{N_0^{(k+1)}}{N_0^{(k)}} = S^{(k)} = e^{-\lambda^{(k)} t^{(k)}}, \text{ then} \qquad (5\text{-}16)$$

$$\lambda^{(k)} = \frac{\ln N_0^{(k)} - \ln N_0^{(k+1)}}{t^{(k)}}. \tag{5-17}$$

Under the hypothesis of a constant mortality rate through time, leading to an exponentially declining population, N_0 for each length class is computed as

$$N_0 = \frac{A\lambda}{1 - e^{-\lambda t}} \tag{5-18}$$

An alternative hypothesis is that all mortality occurs at the end of the length class, resulting in a uniform rather than an exponential age distribution. Although this hypothesis is less realistic than the exponential age distribution hypothesis, it can be used to approximate the initial population size of each length class,

$$N_0'^{(k)} = \frac{A^{(k)}}{t^{(k)}} \tag{5-19}$$

where

$N_0'^{(k)}$ = approximation to initial population size of length class k.

When this is substituted in Equation 5-17, it yields

$$\lambda(k) = \frac{\ln \frac{A^{(k)}}{t^{(k)}} - \ln \frac{A^{(k+1)}}{t^{(k+1)}}}{t^{(k)}}. \tag{5-20}$$

These size-specific survival rates (Table 5-14) and survival rates for fish at age 1, 2, and 3 years were used to estimate the significance of losses in individuals of each size class on their potential contribution to the fishery population at age 3. For example, the survival of 40-mm striped bass was calculated as

$$S_{40,a} = S_{40,100}\, S_2\, S_3 \tag{5-21}$$

where

$S_{40,a}$ = survival from 40 mm to three-year-old adults

TABLE 5-14 SURVIVAL BY LENGTH CLASS FOR ALL YEARS OF USABLE SURVEYS

Year	Length Class								
	7-10 [a]	11-14	15-18 [b]	20-24	25-29	30-34	35-39	40-44	45-49
1961	—	—	—	—	—	0.874	0.700	0.544	0.371
1962	—	—	—	—	—	—	0.636	0.619	0.599
1963	—	—	—	—	—	—	—	—	—
1964	—	—	—	0.779	0.771	0.661	0.510	0.376	—
1965	—	—	—	—	—	0.891	0.662	0.666	—
1966	—	—	—	—	—	—	—	—	—
1967	—	—	—	—	—	—	—	—	—
1968	0.130	0.378	0.300	—	0.690	0.536	0.691	0.710	0.698
1969	—	—	—	—	—	—	—	—	—
1970	0.0846	0.456	0.486	—	—	—	0.828	0.629	0.503
1971	—	—	—	0.533	0.838	0.753	0.620	0.593	0.592
1972	0.0691	0.357	0.478	—	0.732	0.474	0.830	0.766	0.491
1973	—	—	—	—	—	—	0.608	0.542	0.469
1974	—	—	—	—	—	—	0.863	0.725	0.491
1975	0.0347	0.229	0.589	—	0.893	0.739	0.683	0.694	0.715
1977	—	—	—	—	—	—	0.847	0.716	0.503
1977	—	—	—	0.884	0.576	0.568	0.610	0.643	—
1978	0.376	0.490	0.658	—	—	—	—	—	—
1979	0.117	0.428	0.618	—	—	—	0.541	0.762	0.882
Mean	0.135	0.390	0.522	0.732	0.750	0.718	0.688	0.642	0.574
	6	6	6	3	6	9	14	14	11

(a) Survival through 5-6 mm was computed as 0.137 = 0.367 on the assumption of constant mortality from 5 to 10 mm.
(b) Survival for 19 mm was taken to be the fifth root of survival from 20-24 mm.

$S_{40,100}$ = survival from 40 mm to 100 mm, or one year

S_2 = survival from year one to year two

S_3 = survival from year two to year three

or, rewritten:

$$S_{40,100} = S_{40,a}/S_2S_3. \tag{5-22}$$

Survival from 40 mm to adulthood was computed to be 0.0256 using the formula

$$S_{40,a} = \frac{1}{r} \sum_{i=1}^{r} \frac{40\ mm_i}{3\ yr_{i+3}} \tag{5-23}$$

where

$40mm_i$ = 40mm striped bass index for year i

$3\ yr_{i+3}$ = three-year-old striped bass population estimate three years after year i

r = number of 40 mm-3 year old pairs.

Survival from year one to year two, S_2, and from year two to year three, S_3, were both set at 0.53 (Stevens 1977b). This produced a survival estimate from 40 to 100 mm of 0.091 (Equation 5-22).

If the mortality from 40 to 100 mm is considered to be a constant function of length, then

$$S_{40,100} = e^{-Z60} \tag{5-24}$$

where

Z = instantaneous mortality rate through length.

Solving for Z yields Z = -(ln 0.0911)/60 or 0.0399.

Survival from length l to 100 was then calculated as

$$S_{l,100} = e^{-0.0399(l-40)}, \tag{5-25}$$

and survival from length class 1 to adulthood was computed as

$$S_{1,a} = S_{1,100}S_2S_3. \qquad (5\text{-}26)$$

For impingement of length classes greater than 100 mm, survival to adulthood was set at 0.28, survival from year one to year three (0.53 x 0.53). This estimate is conservative, in that survival from length >100 mm to three years would be less than 0.28. Survival estimates were not calculated for striped bass eggs or larvae less than 7 mm in length because of insufficient field monitoring data for these early life stages.

5.6.2.2 Results and Discussion

The estimates of equivalent adult loss associated with entrainment and impingement at the Pittsburg Power Plant Units 1-7 from March 1978 - March 1979 are tabulated in Table 5-15. Equivalent adult loss is presented for specific length classes and for all classes combined. The estimated equivalent adult loss for the combined entrainment and impingement mortality occurring between March 1978 and March 1979 is 233,000. Of this total estimated equivalent adult loss, entrainment accounted for approximately 218,000 and impingement accounted for approximately 15,000.

Examination of the relative contribution of the various size classes to the estimated equivalent adult loss associated with entrainment mortality indicates that the largest contribution to the total loss (72 percent) was from fish 11-29 mm long. The small size classes (5-10 mm) represented the largest number entrained, but, their contributuion to the estimated equivalent adult loss was relatively small (4 percent) because of the extremely high natural mortality rates of these small size classes.

The largest proportion of the striped bass impinged at the Pittsburg Power Plant ranged in length from 51 to 100 mm, and this class also contributed the largest proportion (87 percent) to the estimated equivalent adult loss associated with impingement. Yearling striped bass (more than 100 mm long) represented a smaller proportion (11 percent) of the estimated equivalent adult loss associated with impingement.

The projection of equivalent adult losses (Table 5-15) confirms that entrainment mortality (representing 93 percent of the equivalent adult loss) is the major source of

TABLE 5-15 NUMBERS CROPPED, SURVIVAL, AND EQUIVALENT 3-YEAR-OLD LOSS ASSOCIATED WITH ENTRAINMENT AND IMPINGEMENT AT THE PITTSBURG POWER PLANT MARCH 1978 - MARCH 1979

Length Class (mm)	Number Lost (N)	Survival from Length Class to 3 Years $S_{1,a}$	Equivalent Adult Loss (E)
		Entrainment	
5-6	53,561,245	0.0000657	3,520
7-10	24,589,825	0.000179	4,404
11-14	30,177,827	0.00133	40,042
15-18	17,325,501	0.00340	58,941
19	1,680,005	0.00652	10,950
20-24	4,874,463	0.00693	33,799
25-29	1,333,433	0.00947	12,631
30-34	415,440	0.0126	5,247
35-39	318,060	0.0176	5,594
40-44	234,396	0.0256	6,000
45-49	109,919	0.0398	4,374
50	109,857	0.0694	7,624
52	6,899	0.0734	506
54	35,616	0.0776	2,763
56	35,616	0.0821	2,924
60	9,149	0.0917	838
61	35,616	0.0944	3,361
65	27,442	0.106	2,895
70	35,616	0.121	4,323
76	33,019	0.144	4,738
100	8,222	0.281	2,309
Total			217,791
		Impingement	
26-30	306	0.00947	2
31-40	2,141	0.0176	37
41-50	5,813	0.0398	231
51-75	61,389	0.0983	6,043
76-100	36,844	0.198	7,295
101-125	3,188	0.281	895
126-150	269	0.530	142
150	1,260	0.530	667
Total			15,312
Total Equivalent Adult Loss			233,103

potential impact of the Pittsburg Power Plant on the Bay-Delta striped bass population. Therefore, one of the primary objectives in developing a striped bass population model (Section 5.6.3) will be to examine the significance of entrainment cropping in terms of the abundance of young-of-the-year striped bass (year class strength) and the potential reduction in the adult striped bass population resulting from entrainment mortality.

The estimates of equivalent adult loss presented in Table 5-15 are based on the assumption that no density-dependent (compensatory) mechanisms are contributing to the survival of striped bass. There is no reason to believe, however, that density-dependent population regulation does not occur in the Bay-Delta striped bass population, and hence the estimates of equivalent adult loss are considered to be conservative. Both density-dependent and density-independent survival estimates will be considered in projections of entrainment mortality to the adult striped bass population in the model developed in Section 5.6.3.

In addition to providing a method for comparing the impacts of entrainment and impingement cropping, the estimated equivalent adult loss (Table 5-15) can be compared with estimates of recruitment of three-year-old striped bass into the sport fishery. Although no estimates are available for the abundance of three-year-old striped bass in 1981, reflecting recruitment of the 1978 year class for which the equivalent adult losses in Table 5-15 have been estimated, it has been estimated that the number of three-year-olds in the population averaged approximately 1.0 million between 1969 and 1976. Assuming that a similar level of recruitment (three-year-old abundance) is occurring in 1981, it can be estimated that entrainment and impingement at the Pittsburg Power Plant between March 1978 and March 1979 may have caused to a reduction in the 1981 three-year-old population of as much as 15-20 percent. This theoretical estimate of reduction in recruitment to the sport fishery is considered to be conservative, based as it is on the assumption that no compensatory survival is occurring.

The accuracy of the estimated equivalent adult loss is uncertain. A sensitivity analysis of the equivalent adult methodology indicated that the estimated loss is extremely sensitive to changes in age-specific survival rates. This finding has two important implications in regard to the degree of certainty placed in the equivalent adult estimates. First, estimates of the survival rate for early life stages were highly variable between years (Table 5-14) and this variability can have a significant effect

on the accuracy of equivalent adult projections. The variability and uncertainty in survival rates is further compounded by a lack of information on population dynamics of yearling and subadult striped bass. The high variability in age-specific survival rates between the larval and early juvenile stages and recruitment at age three is evidenced by the low coefficient of determination (r^2 = 0.36) for the relationship between an index of young-of-the-year abundance and the estimated recruitment three years later (Section 5.6.3.2.1). The second uncertainty is that the relative contribution of density-independent and density-dependent survival to the dynamics of the Bay-Delta striped bass population is unknown.

Projections of equivalent adult loss provide a useful tool in examining the relative contribution of entrainment and impingement mortality for various size classes and thereby identifying the areas of primary concern for further analysis in the impact assessment. It cannot be concluded with any certainty, however, that if the Pittsburg Power Plant had not been in operation between March 1978 and March 1979 an additional 233,000 three-year-old striped bass would have been recruited into the sport fishery in 1981. The relationship between recruitment and the numbers of young-of-the-year striped bass cropped as a result of operation of the power plant cooling water systems is discussed in Section 5.7.

5.6.3 Adult Population Model

Adult population models provide a method of analyzing the long-term impact of entrainment and impingement on fish populations, both within one generation and on subsequent generations. Such models can predict quantitatively the long-term impact of a reduction in numbers of early life stages on the total population. The long-term consequences of such a reduction may be a reduction of recruits into the spawning population and a corresponding reduction in spawning stock levels.

The sizes of fish populations are controlled by both density-independent and density-dependent (compensatory) mechanisms. In the striped bass population of the Bay-Delta, there is strong evidence that density-independent factors such as river outflow play a major role in regulating population size (Chadwick et al. 1977; Stevens 1977b). On the other hand, it is generally accepted that compensatory mechanisms resulting in increases or decreases in death and birth rates also act to regulate the population. Many examples of such forms of compensation are given by McFadden (1977). For these reasons, the most appropriate adult population model for a

species is one that makes provision for both the density-independent and the density-dependent mechanisms known to regulate that particular population.

One aspect of the analysis of impact is a consideration of the effect of cropping of early life stages (percentage reduction) on the adults of that year class. Beyond the life stage at which the strength of a year class is set, cropping will be reflected proportionately in the adults. Natural compensatory mechanisms can be expected to reduce the impact on adults of the cropping of early life stages (eggs, larvae, and juveniles) that are smaller than the size at which year class strength is set. Development of an adult model, then, requires a determination of the life stage at which year class strength is established. Long-term survey data collected by CDF&G have made it possible to develop an indicator of year class strength for the Bay-Delta striped bass population that can be used in the formulation of the adult stock model.

Determination of the influence of entrainment and impingement cropping on subsequent generations requires that the relationship between spawning stock and subsequent recruitment be addressed in development of the adult model. If recruitment is independent of stock size, the long-term population effect will be the same as the direct effect on adults of each year class. If spawning stock size and recruitment are not independent, the long-term effect on the population could be greater than the direct effect of the cropping, as a result of a cumulative reduction in spawning stock.

There have been few adult population models developed for particular fish species that incorporate age-specific mortality rates, the influence of environmental factors on year class strength, and a consideration of stock-recruitment relationships. Development of such adult models is generally precluded by their data requirements. Many of the elaborate population models that have been developed rely on hypothetical or weakly supported estimates of parameters. The availability of long-term records on year-class strength and the population characteristics of striped bass in relationship to environmental conditions have made it possible to develop a specific, data-based adult population model for the Bay-Delta. Using the available field monitoring data and the results of entrainment studies (Chapter 3), the model is used to examine the structure of the striped bass population both with and without the influence of intake-related cropping, under present and future water management regimes. The primary components in the Bay-Delta adult population model and their interrelationships are shown in Figure 5-24.

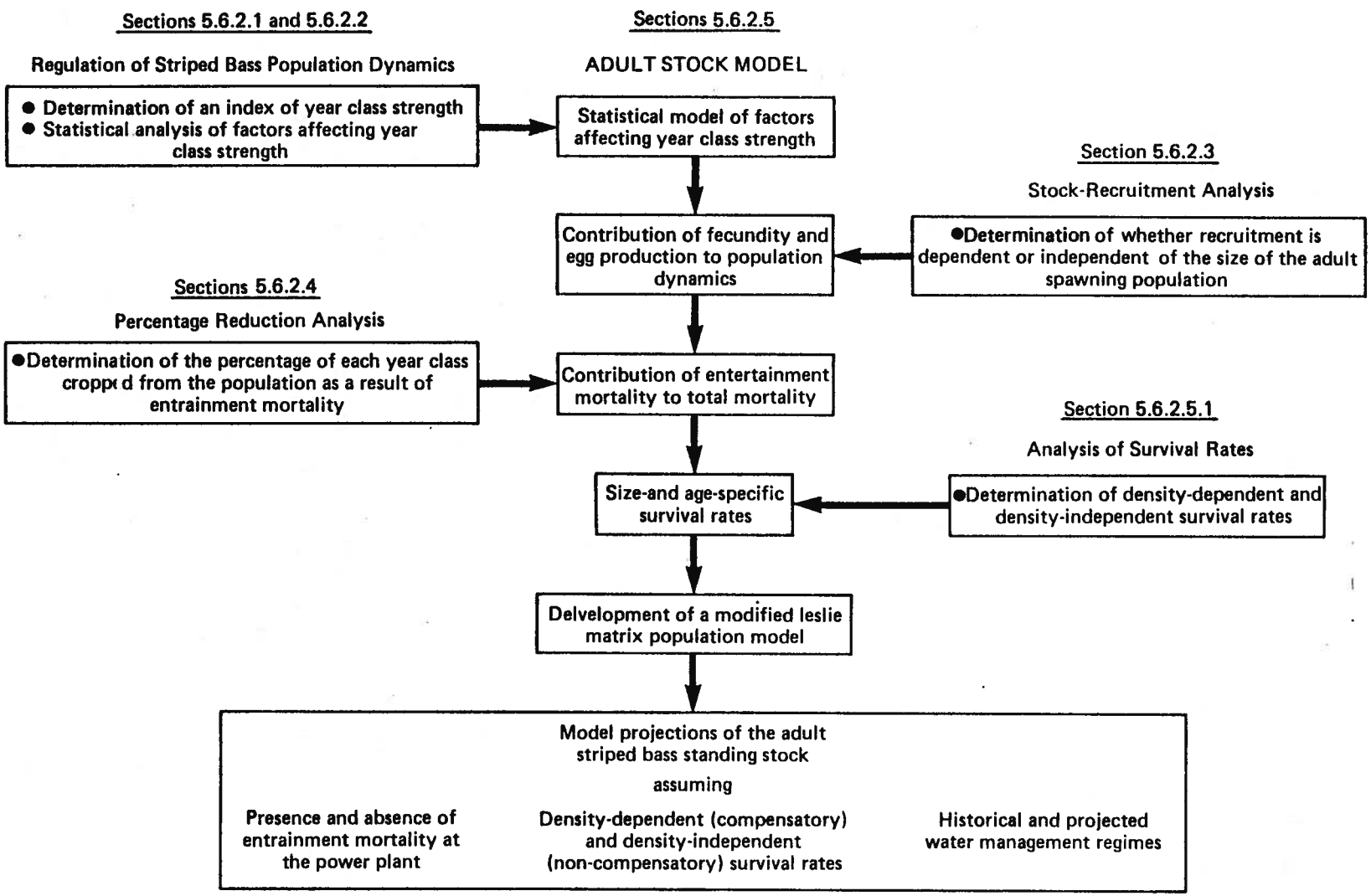


Figure 5-24. Relationships between the analyses contributing to an adult stock model for the striped bass population of the Bay-Delta.

5.6.3.1 Approach

The primary objective of this analysis was to determine the impact of operation of the Pittsburg Power Plant cooling water systems on the striped bass population. An ancillary objective was to determine the role that environmental factors played in affecting productivity in the estuary, so that the effect of the plant could be put in perspective with other biotic and abiotic factors influencing the striped bass population.

Complex and elaborate models of striped bass population dynamics have been developed in the past (Van Winkle et al. 1974; Christensen et al. 1977; Eraslan et al. 1976; McFadden and Lawler 1977), but they have been either hypothetical or based on weakly supported estimates of parameters, because of limited population data. In this study, a data-base approach was employed in developing a number of striped bass population models. These models were based largely on the extensive data bases collected by CDF&G over the past 20 years.

Section 5.6.3.2 summarizes the results of an investigation of the factors affecting striped bass year class strength in which the effects of the power plant, outflow, exports, and other environmental factors on the variability in young-of-the-year (YOY) population size are examined. In Section 5.6.3.3 the question of whether there is a relationship between stock size and recruitment is investigated. If recruitment is independent of stock size, the long-term effect of power plant cropping on young-of-the-year striped bass will be the same as the direct effect on adults of each year class. If, however, recruitment is dependent on stock size, the long-term effects of power plant cropping will be greater, because of the reduction in the number of adult spawners. In Section 5.6.3.4 the short-term impact of entrainment cropping of larval striped bass are determined and compared with estimates of the young-of-the-year population size to obtain a measure of the fraction of the population cropped. In Section 5.6.3.5 age-specific mortality rates are estimated and a modified Leslie-matrix adult model is developed.

The adult population model presented in Section 5.6.3.5 incorporates the information derived in the earlier sections to determine the effects of environmental factors and power plant operation on the adult striped bass population. With this adult model, the relative importance of power plant cropping and other environmental factors in

controlling the population dynamics of striped bass under future water management plans can be established.

5.6.3.2 Regulation of Striped Bass Population Dynamics

Through entrainment and, to a much lesser extent, impingement, striped bass are most vulnerable to the Pittsburg Power Plant when they are young (Chapters 3 and 4). In this section, the annual variability in young-of-the-year abundance and how it compares to the annual variability in power plant operation are examined. Since the present water quality standards and future water management plans in the Sacramento-San Joaquin estuary have been based in large part on the environmental requirements of striped bass (SWRCB 1978), the analysis was expanded to include a number of other environmental factors in addition to the power plant. The objective of this analysis is to identify those environmental factors (including power plants) that most affect or are correlated with the variability in the abundance of striped bass young-of-the-year.

The approach used was to develop a statistical regression model that related an index of striped bass abundance to a set of environmental variables. Using a variable-selection process, the relative importance of power plant operation and other environmental variables in controlling young-of-the-year abundance could be ascertained.

The approach, analytical methods, and data bases used in determining an index of striped bass abundance are discussed in Section C.3, Appendix C. The rationale for excluding data for some years and size classes from the analysis is also discussed in Section C.3.

5.6.3.2.1 Determination of an Index of Striped Bass Abundance

After exclusion of data for the years and size groups that did not meet specific data base requirements, the numbers of striped bass produced in the remaining years and size groups were calculated using the method described in Section C.3. As is evident in Table 5-16, information from a number of the larger size groups of the egg and larval survey and a number of the smaller size groups of the juvenile surveys was inadequate to make production estimates. The two larger size groups of striped bass (35-39 and 40-44 mm), collected during the summer juvenile surveys, contained

TABLE 5-16 ESTIMATED NUMBER OF STRIPED BASS PRODUCED IN EACH SIZE CLASS, 1961-1979

Year	Size Class							
	7-10	11-14	15-10	20-24	25-29	30-34	35-30	40-44
1961	—	—	—	*	56.5	46.6	31.6	16.5
1962	—	—	—	*	*	93.2	58.9	34.9
1963 (a)								
1964	—	—	162.9	136.7	108.4	72.4	28.4	17.8
1965	—	—	—	—	—	114.8	76.7	51.0
1966 (b)								
1967 (a)								
1968	16,560	2,040	*	91.2	69.0	28.7	21.6	16.2
1969 (a)								
1970	36,420	3,687	*	*	*	84.0	56.0	35.1
1971	30,413	*	309.9	139.7	118.8	97.5	49.3	29.2
1972	37,299	3,022	948.0	97.2	77.8	27.6	25.4	21.2
1973	35,630	2,845	744.0	*	*	*	59.3	
1974	*	*	*	*	*	77.7	67.1	48.5
1975	38,977	1,697	417.0	135.2	120.8	91.4	47.3	35.9
1976	*	*	*	*	*	21.0	17.8	12.8
1977	*	*	19.9	19.7	9.4	5.6	3.5	
1978 (a)								
1979	*	*	*	*	*	19.5	10.6	8.0

(a) Study excluded because null zone was in San Pablo Bay.
(b) No sampling done.

Note: (—) indicates that the size class was not sampled; (*), that no peak abundance was found for the size class.

Source of Data: CDF&G unpublished.

information for the largest number of years between 1961 and 1979. These results are not surprising, since (1) the sampling design and type of gear used by CDF&G was most appropriate for collecting striped bass in this size range and (2) the summer juvenile study was initiated 10 years earlier than the egg and larval surveys.

Year class strength is believed to be set by the time the population mean length reaches 38 mm (Chadwick 1964). Our analyses indicate (Table 5-17) that the number of striped bass produced in the 35-39 mm size group was significantly ($\alpha = 0.05$) correlated with the number produced in all size groups larger than 25 mm, including an index of yearling striped bass abundance based on CDF&G midwater trawl surveys (L. Miller, CDF&G, personal communication, 1979) and an index of three year old recruits ($\alpha = 0.10$) from the CDF&G mark-recapture studies (Stevens 1977b). Although the sample size was smaller, there appeared to be no statistically significant relationship between the number of 35-39 mm striped bass and those smaller than 25 mm. These results indicate that the year class strength is set sometime before the striped bass are recruited into the 35-39 mm size group. In fact, the data suggest that the year class is set early in the life history of the fish, possibly before they enter the 25-29 mm length class.

The number of striped bass produced in the 35-39 mm length class was selected as an index of year class strength because the year class is set before the population reaches this length and because this size group contains information for the largest number of years. The index of year class strength developed here has the following attributes: (1) it estimates the number of striped bass produced in the size most appropriate for the gear type and sampling design used by CDF&G, (2) it is largely unaffected by annual differences in the time of spawning and differential growth rates, because it incorporates data collected over a five-week period each year, and (3) it is applicable whenever spring flows are not so high as to transport young bass into San Pablo Bay. This index of striped bass year class strength will be regressed against a number of environmental variables in the next section in order to determine what factors affect or are correlated with the variability in striped bass year class strength.

5.6.3.2.2 Determination of Factors Affecting Year Class Strength

In this section we will investigate the environmental conditions that account for or explain the most variability in the year class strength index. This analysis requires

TABLE 5-17 CORRELATIONS BETWEEN THE NUMBER OF STRIPED BASS PRODUCED IN THE 35-39 MM SIZE GROUP AND THE NUMBER OF STRIPED BASS PRODUCED IN OTHER SIZE GROUPS, 1961-1979

Size Group (mm)	Number of Observations	Correlation Coefficient (R)	P
7-10	6	0.56	N.S.
11-14	5	0.34	N.S.
15-19	3	0.36	N.S.
20-24	4	0.95	0.01
25-29	7	0.79	0.05
30-34	9	0.92	0.01
35-39	—	—	—
40-44	14	0.96	0.01
60-100 (Index)	8	0.76	0.05
Age three (Index)	10	0.59	0.10

Note: N.S. indicates nonsignificance.

that a long-term data base of environmental factors that could potentially influence striped bass abundance be created and compiled. The index of year class strength is then regressed against the environmental factors, singly and in combination. A model relating striped bass abundance to environmental factors is developed using a variable-selection procedure. This analysis uses the technique of least-squares regression analysis.

Regression analysis may be broadly defined as the analysis of relationships among variables (Chatterjee and Price 1977). The relationships can be expressed in the form of an equation connecting the dependent variable (in this case, year class strength index—Y) to one or more independent variables (various environmental factors—X_1, X_2, ..., X_i). The relationship between Y and the X's can be formulated as a linear model:

$$Y_i = b_0 + b_1X_{1i} + b_2X_{2i} + \dots + b_p X_{pi} + \epsilon_i$$

where the constants b_0, b_1, b_2, ..., b_p are estimated through a least-squares procedure and the error term, ϵ_i, is assumed to be independent (uncorrelated) and distributed normally, with a mean of zero and constant variance (Steel and Torrie 1960). The purpose for which a regression equation is constructed (which in turn determines the criteria for selecting and evaluating the contribution of different variables) must be considered in the process of selecting variables. In this study we are attempting to build a statistical model that describes the relationship between environmental conditions and an index of striped bass year class strength.

According to Chatterjee and Price (1977), "In situations where description is the prime goal of model building, (the researcher) should choose the smallest number of independent variables that explain the most substantial part of the variation in the dependent variable."

In order to more easily understand the relationship between environmental conditions and striped bass year class strength, a regression model will be developed containing as few variables as possible. The model will be developed through a systematic investigation of all individual environmental conditions and combinations of environmental conditions for which a historical data base is available and which are thought to be capable of influencing striped bass year class strength. The investigation consisted of (1) the identification and collection of data on a number of

environmental factors that could potentially influence year class strength, (2) an evaluation of the "best" model based on a set of statistical selection criteria, and (3) a discussion of the biological rationale supporting the selected model.

Description of Data Bases

Based on the literature and results of previous analyses, a list of factors that could potentially influence striped bass year class strength was created and compiled for the period 1961-1979 (Table 5-18). Not all possible factors identified could be included in the analyses, since long-term records and monitoring results were not available for some. Delta outflow, Delta export, and Cross-Channel and Sacramento River flow rates were all obtained from California Department of Water Resources (DWR) records (unpublished data). The water temperature data were calculated from records obtained from the Pittsburg Power Plant. The solar radiation values for the region were obtained from the University of California at Davis (S. Hatfield, CDF&G, personal communication, 1980). A power plant operation index was calculated (Section C.4, Appendix C) using cooling water flow rates, intake, and discharge temperature regimes obtained from PGandE and entrainment survival results discussed in Chapter 3. The rationale for considering each factor is discussed below.

Turner and Chadwick (1972) reported high correlations between the CDF&G index of striped bass abundance and the Delta outflow index past Chipps Island in all months from April through July. They felt that higher outflows transported the young bass into more suitable regions of the estuary and provided a greater food supply. It was also felt that the higher flows transported bass eggs and young away from the state and federal water project diversions (Chadwick 1979; CDF&G 1976).

Since 1972, the effects of increased Delta export at the state (SWP; Delta) and federal (CVP; Tracy) water projects were thought to have reduced the CDF&G index of striped bass abundance. Chadwick (1979) believed that both the magnitude of freshwater export and the percentage of freshwater flow diverted at the pumping plants were important in controlling striped bass abundance. Chadwick et al. (1977) developed a statistical relationship between Delta outflow and diversions in May and June, and concluded that the exports were the major factor controlling the survival and hence year class strength of young bass in the Delta. According to Chadwick (1979), diversions act as a density-independent source of mortality, literally cropping a portion of the population.

TABLE 5-18 FACTORS POTENTIALLY INFLUENCING STRIPED BASS YEAR CLASS STRENGTH

Factor	Months	Transformation Used	Source of Data
Delta outflow past Chipps Island	April-July	Natural log	DWR
Exports at state (Delta) and federal (Tracy) water projects	May-July	Natural log	DWR
Delta Cross-Channel flow	April-July	Natural log	DWR
Percentage diverted at the Delta Cross-Channel	April-July	—	DWR
Percentage diverted at the export pumping plants	May-July	—	DWR
Temperature in Suisun Bay	April-July	—	PGandE
Solar radiation	April-July	—	U.C. Davis
Power plant operation index	May-July	—	PGandE

The magnitude of the Delta Cross Channel flow and the percentage of the Sacramento River flow diverted could affect the abundance of striped bass by diverting a portion of the young bass and eggs out of the Sacramento River and into the less productive regions of the interior Delta (Arthur and Ball 1979). The longer residence time of fish diverted into the interior Delta could adversely affect their chances of survival.

Turner (1976) found a significant correlation between water temperature and the time of striped bass spawning. He also pointed out that the time of spawning differed in various parts of the system: it was earlier in the San Joaquin River and later in the Sacramento River. Because the time of spawning was not available for all years since 1961 (the first year for which an index of striped bass year class strength was available), the water temperature in Suisun Bay (measured at the Pittsburg Power Plant cooling water intake) was chosen as a proxy variable (Mosteller and Tukey 1977) for the time of spawning.

There are no estimates of primary productivity in the Delta going back to 1961, but Ball and Arthur (1979) noted a relationship between primary productivity (chlorophyll a) and solar radiation, so average monthly solar radiation was used as a proxy variable. It was felt that, in conjunction with nutrient enrichment via Delta inflow, the solar radiation variable could be an indicator of the productivity of the estuary, and thus could affect the abundance of striped bass through the food chain (Section 2.2).

The effect of power plant cooling water withdrawal on the striped bass population was considered minimal by CDF&G (Chadwick and Stevens 1971; Chadwick et al. 1977), on the basis of studies in the laboratory and at the Pittsburg Power Plant which indicated high survival of entrained striped bass. Stevens (1980), however, expressed concern that the impact of the power plants is increasing with increased power production since 1976. To determine if power plant operation was related to the variability in year class strength, a power plant index (Section C.4, Appendix C) was developed that combined the monthly plant pumping volume, intake water temperature, and power production with the estimated striped bass entrainment survival rate. This index was a proxy variable designed to estimate the numbers of striped bass cropped by entrainment at the Pittsburg Power Plant.

In addition to the variables listed, the natural logarithms of the outflow, export, and Cross-Channel flow variables were also examined. Although a large number of other variables could conceivably affect striped bass abundance (e.g., neomysis abundance, irrigation withdrawals, toxic substances, predators, competitors) the lack of historical information, either through direct measurement or by proxy, prevented their inclusion in a regression model.

Analytical Methods

An important goal in regression analysis is to arrive at an adequate description of the observed phenomenon in terms of as few meaningful variables as possible. This economy has two advantages: (1) it enables the researcher to isolate important variables, and (2) it provides a simpler description of the process under study (Chatterjee and Price 1977). Because of the relatively small number of observations (years) available, this investigation was limited to just one- and two- variable models. The procedure used in selecting factors involved the fitting of all possible one- and two-variable regression equations to various subsets of the data base. The first step in the analysis was to create a number of data subsets. Mosteller and Tukey (1977) recommend that one begin an exploratory analysis with a subsample of the entire data set and then work up to successively larger bodies of data. As Anscombe is quoted by Chatterjee and Price (1977) concerning data subsetting, "We are usually happier about asserting a regression relation if the relation is still appropriate after a few observations have been deleted, that is we are happier if the regression relation seems to permeate all the observations and does not derive largely from one or two." Using Mosteller and Tukey's suggestion, the data base was divided into 10 subsets of increasing size from 4 observations (1961-1965) to 14 observations (1961-1979). For each subset, the best models were determined by sequentially applying four evaluation criteria:

1. the model must possess a high coefficient of determination, r^2,
2. the partial correlation coefficients must be significantly ($\alpha = 0.05$) different from zero,
3. the residuals from the model must not be correlated, and
4. the independent variables retained in the model must not be correlated among themselves.

This method of data subsetting and evaluation will produce the maximum amount of information available concerning the nature of the relationship between the index of year class strength and the set of environmental factors.

The measure most widely used for evaluating the goodness of fit in a regression model is the square of the multiple correlation coefficient, the coefficient of determination, r^2. The r^2 value is defined as the proportion of the total variability (sum of squares) in the dependent variable that is explained or attributable to the independent variables (Steel and Torrie 1960). Values of r^2 range between zero and one, with high values indicating a good fit of the model to the data. For each of the 10 data subsets, we examined all one- and two-variable models and retained for further analysis those models that produced the highest r^2 (Criterion 1).

The partial correlation coefficients, b_i, of the models retained by Criterion 1 were then examined for statistical significance, and only those models whose coefficients for the environmental factors were significantly different from zero ($\alpha = 0.05$) were retained (Criterion 2). These exclusions eliminated extraneous variables and kept the models relatively simple.

Although a large r^2 and statistically significant coefficients are desirable features of a model, they alone do not insure that the data have been fitted well (Chatterjee and Price 1977). The third and fourth selection criteria, absence of correlated residuals and absence of multicollinearity, were included to ensure that the model assumptions were met.

The presence of correlated residuals (autocorrelation) has several detrimental effects on the usefulness of regression models, including (1) unreliable least-squares estimates, (2) understated estimates of regression coefficient standard errors, and (3) invalidation of tests of significance (Chatterjee and Price 1977). For each model considered, the Durbin-Watson test statistic for autocorrelation (SAS 1979) was computed and tested for significance (Criterion 3).

Finally, the interpretability of a multiple regression model depends on the assumption that the independent variables are not strongly interrelated (multicollinearity). The difficulty associated with multicollinear data arises from measuring essentially the same quantity under different names and then trying to use the several measures to

get a regression relationship (Mosteller and Tukey 1977). Multicollinearity can destroy the precision of estimation and simplicity of interpretation of the coefficients. If multicollinearity was detected in the "best" two-variable model produced for each subset, that model was deleted and the next-best two-variable model was considered (Criterion 4).

Applying these selection criteria to the 10 subsets of the data, the best regression model relating the index of year class strength with environmental factors in each subset was determined. The factors that emerged consistently in the "best" model for each subsetted data base were selected as being overall the most important factors (among those examined) related to the index of striped bass year class strength.

Results

Highly significant ($P < 0.01$) correlations were found between the index of striped bass year class strength and certain environmental factors (Table 5-19): outflow past Chipps Island in April, May, and June; Delta Cross Channel flow in April; the percentage of Sacramento River flow diverted at the Cross Channel during April, May, and June; and power plant operation in July. It is important to note here that Delta outflow in July and exports in any month were not significantly correlated with the index of year class strength. CDF&G had believed that these variables were important in controlling the abundance of young-of-the-year striped bass (Chadwick 1979; Stevens 1977b, 1980). Our results indicate that although export flows at the Delta (SWP) and Tracy (CVP) pumping plants may crop (entrain) a large number of young striped bass, the variability in the magnitude of the export flow is not related to the observed variability in the index of striped bass year class strength. Our results also indicate that water temperature and solar radiation are not, by themselves, significantly correlated with striped bass year class strength.

One of the difficulties in interpreting the results of a regression analysis is that a number of the variables highly correlated with the index of year class strength may also be highly correlated with each other. For instance, outflow in April, May, and June is significantly correlated ($P < 0.01$) with the percentage of water diverted from the Sacramento River at the Cross Channel in the same months. Power plant operation in June and July is significantly correlated ($P < 0.05$) with outflow in June and July. This is because during periods of low outflow the power plant has to

TABLE 5-19 ENVIRONMENTAL FACTORS FOUND TO BE HIGHLY CORRELATED ($P < 0.01$) WITH STRIPED BASS YEAR CLASS STRENGTH

Factor	Correlation Coefficient
Log April outflow	0.82
Log April Cross-Channel flow	0.79
Log May outflow	0.78
Log June outflow	0.77
Percentage diverted at Cross-Channel in May	-0.78
April Cross-Channel flow	0.75
May outflow	0.74
July power plant operation	-0.74
April outflow	0.68
Percentage diverted at Cross-Channel in April	-0.67
Percentage diverted at Cross-Channel in June	-0.71

produce more electricity than in high outflow periods in order to make up for the decreased availability of hydroelectric power. We also found that Cross Channel flow in April is significantly correlated ($P < 0.05$) with the outflow past Chipps Island in April.

The correlation among independent variables often makes it difficult to interpret cause-and-effect relationships. For example, there is a positive, significant relationship between Cross Channel flow in April and the index of year class strength. This significant correlation would imply that high Cross Channel flows result in high striped bass abundance. But this does not make biological sense, since higher Cross Channel flows mean more bass would be diverted into the less productive interior Delta (Arthur and Ball 1979), which should result in higher natural mortality and a decrease in abundance. Cross Channel flow appears to be important in explaining year class strength variability simply because it is highly correlated with another variable which will be shown to be very important in controlling striped bass abundance, outflow.

The technique of data subsetting and examining those variables most highly correlated to year class strength in each subset was undertaken in an attempt to identify the controlling variables, the variables that consistently emerge as being the most important in explaining the observed variability in the index of year class strength. For each of the 10 data subsets, the 5 environmental factors exhibiting the highest correlation coefficient were determined (Table 5-20).

In every data subset, outflow (or its natural logarithm) during April, May, or June was either the first or the second factor most highly correlated with the index of year class strength. Although outflow in April and June appear to be important, the natural log of May outflow was selected seven out of 10 times as being the factor in the one-variable model that explained the most variability in the index of year class strength (Table 5-21). Although there was usually a slight increase in r^2 values for the two-variable models, no consistent combination of variables emerged in the various data subsets. In a number of the data subsets there was no two-variable model that met all four evaluation criteria. Again, the problem of multicollinearity was encountered in a number of the two-variable models. The lack of a consistent two-variable model indicates that the single-variable model, specifically log May outflow, generally explains as much of the variability in the index of year class strength as possible using the available data bases. The addition of a second variable

TABLE 5-20 THE FIVE VARIABLES MOST HIGHLY CORRELATED WITH STRIPED BASS YEAR CLASS STRENGTH IN EACH OF 10 DATA BASE SUBSETS (IN ORDER OF DECREASING CORRELATION)

Years	Variables 1	2	3	4	5
1961-1968	Log May outflow	June outflow	May outflow	Log June outflow	June temperature
1961-1970	June temperature	Log June outflow	Log May outflow	June outflow	April temperature
1961-1971	Log May outflow	June temperature	Percentage diverted at exports in June	May outflow	July temperature
1961-1972	Log May outflow	May outflow	Percentage diverted at exports in June	April outflow	Log April outflow
1961-1973	Log May outflow	Log April outflow	April outflow	May outflow	Percentage diverted at Cross-Channel in May
1961-1974	Log May outflow	May outflow	Log April outflow	Log June outflow	Percentage diverted at Cross-Channel in May
1961-1975	Log April outflow	Log May outflow	May outflow	Percentage diverted at Cross-Channel in May	Percentage diverted at Cross-Channel in April
1961-1976	Log May outflow	Log April outflow	May outflow	Percentage diverted at Cross-Channel in May	Percentage diverted at exports in May
1961-1977	Log May outflow	Log April outflow	Percentage diverted at Cross-Channel in May	Percentage diverted at Cross-Channel in April	May outflow
1961-1979	Log April outflow	Log Cross-Channel flow April	Log May outflow	Percentage diverted at Cross-Channel in May	Log June outflow

TABLE 5-21 THE BEST ONE-VARIABLE MODELS RELATING STRIPED BASS YEAR CLASS STRENGTH TO ENVIRONMENTAL FACTORS

Subset	Number of Observations	Best One-Variable Model	Coefficient of Determination (r^2)
1961-1968	5	Log May outflow	0.99
1961-1970	6	June temperature	0.90
1961-1971	7	Log May outflow	0.72
1961-1972	8	Log May ouflow	0.76
1961-1973	9	Log May outflow	0.69
1961-1974	10	Log May outflow	0.73
1961-1975	11	Log April outflow	0.62
1961-1976	12	Log May outflow	0.70
1961-1977	13	Log May outflow	0.75
1961-1979	14	Log April outflow	0.67

did not contribute significantly to our understanding of factors affecting striped bass year class strength.

Discussion

Studies by CDF&G and EA have shown that young striped bass are concentrated in the vicinity of the entrapment zone, the location of which is controlled by outflow. At outflows ranging from 15,000 to 40,000 cfs the entrapment zone is located in the highly productive shoal areas of east and west Suisun Bay. Arthur and Ball (1979) have suggested that the quantity of phytoplankton and zooplankton is enhanced when the entrapment zone is located in upper Suisun Bay. High outflows also tend to distribute the larvae more widely in the system, which reduces crowding and intraspecific competition for food. The high outflows increase the rate of downstream dispersion of the larvae so that they reach the productive entrapment zone sooner in their life history, giving them a greater chance for survival.

When outflow is low, the entrapment zone (and with it the eggs and larvae of striped bass) is located in the channels of the western Delta, which have a minimum of shoaling and low primary and secondary productivity. Low flows also increase crowding and food competition and also the time it takes for the larvae to reach the entrapment zone. In addition, low flows tend to concentrate the bass near the Pittsburg Power Plant, which results in increased cropping due to entrainment, further decreasing abundance.

In general, our conclusion agrees with the conclusion reached by CDF&G that Delta outflow is the major factor controlling the abundance of striped bass young-of-the-year. According to Chadwick et al. (1977), the direct effect of flow on the spatial distribution of young bass in the estuary is a major mechanism of the flow-survival relationship.

Turner and Chadwick (1972), Stevens (1977b), and Chadwick (1979) cite a number of reasons why outflow might affect striped bass survival. According to Stevens (1977b), the striped bass population is at present density-limited in a restricted environment, the carrying capacity of the estuary being controlled by late spring and early summer flows. Chadwick (1979) states that the most probable mechanism relating survival to flow is the availability of food, through a combination of the effect of flow on bass distribution, detritus and nutrient input, and spawning time.

concluded that a reduction in outflow would delay the transport of young bass from the spawning areas to the downstream nursery area, resulting in higher mortality rates. One area where our findings conflict with those of CDF&G is in which month outflow is more important in controlling abundance: CDF&G has long held that June and July were the most important months, whereas our findings indicate that flow in May is most important. Furthermore, results of our analysis suggest that diversion losses at the state and federal export facilities (Delta and Tracy pumping plants) is not a significant factor in determining the striped bass year class strenth. The contribution of the export facilities was not identified as being significant in any of the one- or two-variable models generated (Tables 5-20 and 5-21).

Striped bass spawning in the Sacramento - San Joaquin system normally peaks in May (Turner 1976). The period soon after hatching and absorption of the yolk sac, considered by Ricker (1971) to be the critical period for survival of the young for many fish species, occurs for striped bass in most years during May. Outflow during May apparently has a great influence on the conditions in the estuary that affect the availability of food (notably copepods and cladocerans) during this critical period. In order to survive once the yolk sac is absorbed, the larvae must encounter adequate food quickly. If they are overcrowded or concentrated in areas of low productivity in May, when this critical period occurs, poor survival would be expected, but if they are widely dispersed in highly productive areas, we should expect high survival and a strong year class. Stevens (1977b) and Hassler (1958, cited in Setzler et al. 1980) report that the survival of young bass in the Bay-Delta and the Roanoke River, North Carolina, is affected by flow during or shortly after spawning.

The results of this analysis indicate that Delta outflow in spring, and especially in the month of May, is the single most important factor affecting striped bass year class strength in the Bay-Delta. As shown in Table 5-17, the index of three-year-old fish recruited into the striped bass fishery was significantly correlated with the year-class strength index. In the following section we will examine the relationship between recruitment, on the one hand, and adult spawning stock and/or May outflow three years earlier, on the other.

5.6.3.3 The Stock-Recruitment Relationship

In assessing the long-term impact of water use on the striped bass population in the Bay-Delta, it is important to examine the relationship between the number of fish in

the adult spawning stock in a given year and the number of fish recruited into the fishery some time later—the stock-recruitment relationship. The two extremes of the stock-recruitment relationship are complete density-dependence, a condition in which the number of recruits depends only on the number of adults three years earlier, and density-independence, in which the number of fish recruited into the population as three-year-olds is governed by environmental factors and is completely independent of the size of the parent stock. To the extent that recruitment of striped bass is density-dependent, the long-term effect of cropping on the population will be cumulative, and the long-term population effect can be greater than the direct effect on one year class, because a reduction in the population of spawning adults will result in a subsequent reduction in recruitment. If, on the other hand, environmental factors have the greatest effect on recruitment, management of the population requires knowledge of the environmental factor(s) involved.

Because striped bass populations are subject to rather large year-to-year variations in size, it is possible to use regression techniques to examine the stock-recruitment relationship. The California Department of Fish and Game (CDF&G) has established and maintained long-term records of the estimated numbers of three-year old and adult striped bass in the Bay-Delta. The absolute accuracy of these records is not known, but as they are established consistently from year to year, they can serve as an index for correlation and population trends.

For reasons discussed in the previous section, it is thought likely that the flow of freshwater through the Delta has a considerable effect on year class strength (Section 5.6.3.2) and thus on subsequent recruitment of the year class. If differences in year class strength at three years correlated more highly with the freshwater flow three years earlier than with the size of the spawning stock, that could be taken as evidence that recruitment depends more on this environmental factor than on the size of the spawning stock; it would indicate that recruitment is primarily density-independent and that long-term power plant effects would be minimal.

The approach used was to choose an environemental variable and to examine the correlation between it and the recruitment index. As a first step, the same correlations were made between recruitment and mean outflow, and between recruitment and log mean outflow for the months of spawning and larval abundance as were made for young-of-the-year three years earlier. For outflow, the correlation coefficients and their probabilities were:

	APR	MAY	JUN	JUL	APR-MAY	MAY-JUN	JUN-JUL
r	0.59	0.78	0.60	0.29	0.71	0.72	0.54
P	0.0077	0.0001	0.0064	0.23	0.0007	0.0006	0.017

For log outflow, the correlation coefficients and their probabilities were:

	APR	MAY	JUN	JUL	APR-MAY	MAY-JUN	JUN-JUL
r	0.65	0.84	0.65	0.35	0.75	0.79	0.61
P	0.0024	0.0001	0.0026	0.145	0.0002	0.0001	0.0059

As for young-of-the-year, the highest correlation coefficients between outflow and the recruitment index were found using May outflow ($r = 0.78$; $P = 0.0001$) and log May outflow ($r = 0.84$; $P = 0.0001$). On the basis of these results, May outflow was selected as the environmental variable for inclusion in the stock-recruitment analysis.

5.6.3.3.1 Description of Data Bases

Three data bases were used in this analysis: the CDF&G recruitment indices for the years 1958-1974, the corresponding adult population abundance indices three years earlier (1955-1971), and mean Delta outflow past Chipps Island in May for the years 1955-1971.

Indices of the adult population and number of striped bass recruited into the sport fishery for the years 1955-1974 were calculated by CDF&G (Stevens 1977b; D. Stevens, CDF&G, personal communication, 1980). The California Department of Fish and Game requires licensed party-boat captains to register the striped bass catch taken on each expedition. The total catch reported is compiled for a year that runs from 1 May through 30 April. It is assumed that the party-boat catch will constitute a consistent fraction of the total striped bass catch from year to year. In addition, CDF&G uses a tagging program to estimate the size of the adult striped bass population and its survival rates, by means of mark-recapture techniques. The adult striped bass population index (P) used in this analysis was calculated for each year, using the equation

$$P_t = \frac{k_t}{u_t} \qquad (5\text{-}27)$$

where

P_t = the adult population index for year t

k_t = the reported catch of legal (>40.6 cm length) bass from party-boat records in year t

u_t = an annual estimate of the fraction of the total population harvested in the fishery, as determined by results of an annual mark-recapture study.

An index of recruit abundance for each year was calculated by CDF&G using the equation

$$R_t = P_t - (P_{t-1} S_{t-1}) \qquad (5\text{-}28)$$

where

R_t = the index of recruitment (assumed to be an index of three-year old bass abundance) for the year t, assuming that all recruitment occurred instantaneously on 1 May

P_t = the population index for year t

P_{t-1} = the population index in year t-1

S_{t-1} = the estimated survival rate in year t - 1, computed from the results of tagging studies.

Stevens (1977a,b) discusses several potential biases in the indices of population abundance and recruitment:

1. the fraction of the total sport fishery catch each year by party boats is not constant, as reflected by angler surveys which suggest that the fraction may vary from 10 to 17 percent;
2. the fraction of the party-boat catch reported is not constant, but varies annually and between regions of the Bay-Delta;
3. recruitment into the fishery is not instantaneous on 1 May, but occurs throughout the year; and
4. recruitment is not restricted to three-year-old fish, but may also involve a number of two-year-old fish.

Since the absolute values of the size of the recruitment and of the adult populations are uncertain, there can be no quantitative estimate of the effect of these biases on the accuracy or reliability of the indices, but they are calculated consistently from year to year, and may thus constitute a basis for examining correlations. A constant, 60,000, was added to each recruitment index to remove negative values.

Delta outflow data are the calculated mean outflows (cfs) past Chipps Island during May, the Delta Outflow Index of the California Department of Water Resources (DWR).

The recruitment index for each year from 1958 to 1974 and the corresponding population abundance index (assumed to be proportional to spawning stock) and May flow are presented in Table 5-22. The population and May flow rates are for a time three years earlier than the recruitment index, to reflect the spawning stock and flow rates for the year in which each cohort recruited into the fishery was spawned.

5.6.3.3.2 Analytical Methods

Examination of the relationship between spawning stock size, May outflow, and recruitment was done using least-squares regression techniques. Regression models used in the analyses were adult-stock-based models developed by Ricker (1975) and by Beverton and Holt (Ricker 1975). In addition to these two models which contained only a stock component, three models with only an environmental component and two with both stock and environmental components were examined.

The basic Ricker stock-recruitment model was used in the form

$$R = aS\ e^{-bS} \qquad (5\text{-}29)$$

where

R = the recruitment index

S = the population abundance index which is assumed to represent the spawning stock

a,b = regression coefficients.

The Beverton-Holt stock-recruitment model took the form

TABLE 5-22 STRIPED BASS RECRUITMENT INDICES, 1958-1974, AND CORRESPONDING POPULATION INDICES AND MAY DELTA OUTFLOW, 1955-1971

Year	Recruitment Index	Population Index[a]	May Flow[a] (cfs)
1958	250,850	139,160	79,105
1959	134,130	198,860	7,384
1960	126,670	218,630	12,328
1961	66,190	322,250	8,580
1962	117,300	287,460	18,173
1963	224,360	236,840	53,123
1964	53,280	126,220	9,783
1965	152,310	127,610	32,370
1966	46,140	247,940	9,835
1967	110,560	148,970	74,550
1968	81,410	188,700	6,737
1969	202,790	115,780	64,284
1970	80,280	125,470	10,761
1971	185,810	86,020	26,405
1972	1,380	202,580	5,140
1973	73,910	146,500	11,699
1974	71,265	213,420	25,544

(a) These values are for three years earlier than those listed, which are the years of recruitment for three-year-olds.

Source: Stevens 1977b, 1980; DWR unpublished data.

$$R = \frac{S}{(aS + b)} \quad . \tag{5-30}$$

Three models were examined for the relationship between recruitment (R) and a function of May outflow (f). A simple linear equation was used in the form

$$R = a + bf. \tag{5-31}$$

The basic linear model was also used with the logarithm of May outflow, in the form

$$R = a + b \log f. \tag{5-32}$$

and in an exponential form:

$$R = a \cdot e^{-bf} . \tag{5-33}$$

Two models which included both stock and environmental factors were examined, a modification of the simple linear equation (5-31):

$$R = a + b(\log f) + cS \tag{5-34}$$

and a multiplicative model:

$$R = a + bfS. \tag{5-35}$$

5.6.3.3.3 Results

The results of the stock-recruitment analysis are displayed in Table 5-23. They indicate (1) that there is no significant relationship between stock size and recruitment and (2) that there is a significant ($\alpha = 0.05$) positive relationship between May outflow and recruitment three years later (Figure 5-25). The coefficient of determination (r^2) for the regression models that contained only a stock component were considerably lower (r^2 = 0.20 and 0.02) than the coefficients for the models that contained only the May flow variable (r^2 = 0.54, 0.61, and 0.24). Regression models that included consideration of both the population index and May outflow exhibited similar results to the models using only May outflow.

TABLE 5-23 SUMMARY OF STOCK-RECRUITMENT REGRESSION ANALYSIS

Equation Number	Model	Result	Coefficient of Determination (r^2)
5-29	$R = aSe^{-bS}$	$2.5\ Se^{-0.000009S}$	0.20
5-30	$R = S/(aS + b)$	$S/0.0012S - 11.8$	0.02
5-31	$R = a + bf$	$62,740 + 2.0F$	0.54
5-32	$R = a + b(\log f)$	$-467,421 + 59,519(\ln f)$	0.61
5-33	$R = ae^{-bf}$	$45,706e^{0.00002f}$	0.24
5-34	$R = a + b(\log f) + cS$	$-447,195 + 58,344(\ln f) - 0.05S$	0.61
5-35	$R = a + bfS$	$141,657 - 0.014fS$	0.02

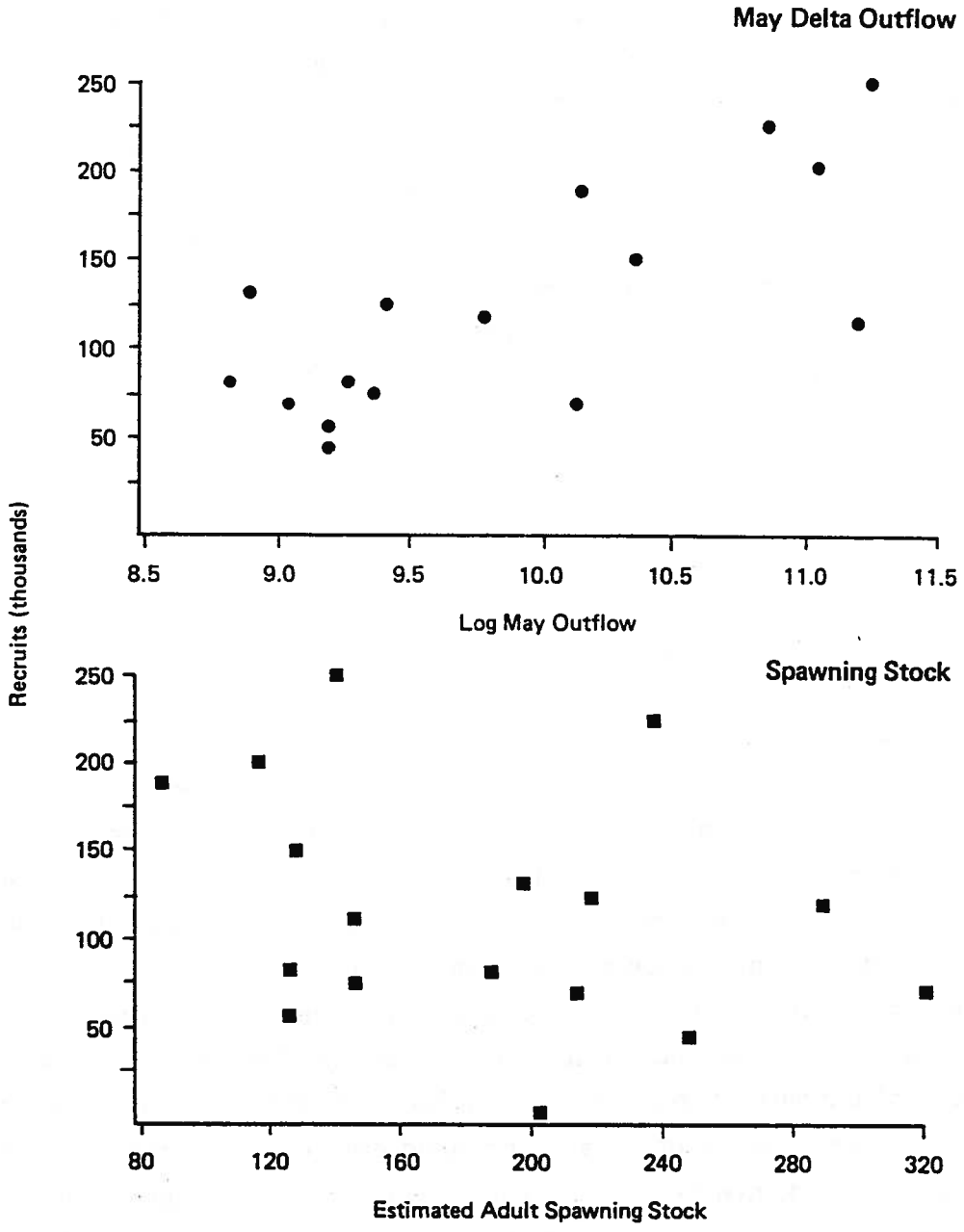


Figure 5-25. Correlation of the recruitment of three-year-old striped bass with May Delta outflow and adult stock three years earlier, 1958-1974.

The regression coefficients of both the Ricker model (Equation 5-29) and the Beverton-Holt model (Equation 5-30) were not significantly different from zero, indicating that the variability in the recruitment index is not strongly correlated with the variability in the stock size index. All three models that related the recruitment index to various functions of May outflow had regression coefficients significantly different from zero. As with the young-of-the-year analysis (Section 5.6.3.2), the model that included only the logarithm of May outflow explained the greatest amount of variability in the recruitment index ($r^2 = 0.61$). In the model that combined flow and stock size additively (Equation 5-34), the coefficient for outflow was highly significant ($P < 0.01$), whereas the coefficient for stock size was nonsignificant. There was no significant relationship between the recruitment index and a multiplicative combination of flow and stock size. These results lend further support to the conclusion that density-independent factors influence the reproductive success (recruitment) of striped bass in the Bay-Delta. At current population levels, the size of the adult spawning stock does not strongly influence striped bass recruitment into the sport fishery, whereas May outflow during larval development showed a much stronger relationship to recruitment.

5.6.3.3.4 Discussion

Results of the stock-recruitment analysis provide evidence that a density-independent factor, May outflow during the period of early development, accounts for a substantial fraction (up to 61 percent in these analyses) of the variability found in subsequent recruitment of the cohort into the sport fishery. The relationship between adult stock and recruitment appears to be weak when compared to the influence of the environmental condition. As discussed in Section 5.6.3.2, May outflow could influence spawning success of striped bass by affecting the distribution of striped bass eggs and larvae in the Delta and by changing the size and carrying capacity of the nursery area (Turner and Chadwick 1972). A reduction in the size of the nursery area could contribute to increased predation on larval bass and even to cannibalism, in addition to increased competition for food. Reduced flow through the Delta might reduce the influx of nutrients and detritus into the Delta system, an influx which would otherwise stimulate phytoplankton growth and hence increased production of the zooplankton that is important for the survival of larval and juvenile bass.

The conclusion that the size of the index of adult stock is not strongly related to the recruitment index at current population levels is consistent with the observed population growth of the Bay-Delta stock from an original planting of fewer than 500 yearlings. The fact that a small adult stock is capable of producing a large population is in part due to the high fecundity of striped bass (Section 2.7). CDF&G (B. Collins, CDF&G, personal communication, 1980) estimates that the number of eggs per female ranges from approximately 100,000 for three-year-old bass to 2.5 million for 14-year-olds (Moyle 1976a, gives a range of fecundities from 11,000 to 2 million). This high reproductive potential is a mechanism through which recruitment could be independent of adult stock size (i.e., a large spawning stock would not be required to produce a strong year class).

There are a number of limitations inherent in the available data base. Indices of recruitment and the adult population derived from commercial party-boat catch records (Stevens 1977a) are imprecise (Section 5.6.3.3.1). An important limitation is the lack of knowledge about what constitutes an effective spawning population in the Bay-Delta: existing data are insufficient to quantify the number of eggs produced or the age and sex structure of the spawning population over a sufficient number of years to be useful as indices of the adult spawning stock. Efforts are being made by CDF&G to collect quantitative data which will provide greater confidence in examining the relationship between stock size, environmental factors, and recruitment for the striped bass population in the Delta.

On the basis of available population data there appears to be no obvious dependency between the adult stock of striped bass in the Bay-Delta and their reproductive success as measured by recruitment into the sport fishery. At present population levels, a density-independent factor, Delta outflow, shows dominant influence on year-class strength and also on subsequent recruitment. These findings indicate that the apparent decline in the population abundance of striped bass in the Delta is not the result of failure in adult spawning, but may be the result of the influence of density-independent environmental factors contributing to reduced year class strength and poor recruitment.

The adult striped bass population model will be developed in Section 5.6.3.5 on the assumption that spawning success and recruitment are independent of the size of the adult stock at present population levels. Obviously, for any population there is a critical threshold level below which adult stock size would seriously limit the

reproduction of the population. The critical threshold level for the striped bass population of the Bay-Delta is unknown, but it appears to be below the levels reached in recent years.

5.6.3.4 Short-term Impact—Percentage Reduction

Estimates of the absolute numbers of striped bass entrained and impinged (Table 5-3; Chapters 3 and 4), while useful in a comparative sense, do not in themselves provide enough information to assess power plant impact on a broad scale. Additional information is necessary to transform the level of impact assessment from considerations of direct cropping to considerations of short-term population effects. The approach used to assess impact on the population level of the affected life stage was the percentage reduction approach, which uses an estimate of abundance of the affected population to arrive at an index of relative loss that is comparable from year to year. It has been applied to estimates of entrainment cropping at the Pittsburg Power Plant for years of adequate population data, using historical data bases of field abundance to arrive at population size estimates and entrainment rates.

Power plant entrainment sampling is not a requirement of this methodology. From the historical estimates of power plant involvement two analyses are possible:

1. power plant involvement for a number of years can be compared, and
2. future power plant involvement can be projected on the basis of similarities in hydraulic conditions of the Delta between past and future years.

5.6.3.4.1 Approach

Predicting percentage reduction is a frequent goal in the assessment of power plant impact, and a number of methods have been developed for this purpose. These methodologies can be grouped into two categories, hydrodynamic models and empirical models.

In the hydrodynamic model approach, the distribution of the early life stages is predicted through the use of mathematical models that simulate the dynamics of the water body. This is achieved by combining the output of a hydrodynamic model,

generally consisting of physical characteristics of the water body (tides, freshwater flows, etc.), with hypothesized responses of the affected life stages of organisms to these characteristics. Beginning with the egg stage, these models predict the development into the succeeding life stages and their distributions. From the predicted distributions of the different life stages and a knowledge of power plant location within the water body, entrainment rates are estimated and estimates of percentage reduction are ultimately predicted.

These models are often mathematically and intuitively complex, requiring formulation of several critical assumptions, application of substantial data sets, and the estimation of model parameters that represent poorly understood biological phenomena (Swartzman 1979). Wallace (1975) found model hypotheses defining ichthyoplankton distributions (vertical and horizontal) and various physical characteristics to be in poor agreement with reliable field data. Because of the complexities of such models and inherent difficulties in their implementation, the hydrodynamic model approach was not selected for computing percentage reduction of the striped bass in the Bay-Delta.

In the empirical model approach, the distribution of the early life stages is derived directly from extensive field surveys rather than predicted through simulation models (Boreman et al. 1978; Goodyear 1977; Texas Instruments, Inc. 1975). The input to empirical models consists of (1) the densities of organisms within various spatially distinct strata of the water body, (2) the volumes of those strata, (3) the densities of organisms near the intake structures of the power plant, and (4) power plant pumping volumes. The advantage of models of this type is that they are simpler than hydrodynamic models, relying on fewer assumptions and requiring smaller data bases.

The model chosen to predict percentage reduction of striped bass in the Sacramento - San Joaquin Delta is an empirical model. It combines ideas originally developed by Polgar (1977), Ricker (1975), and Hackney and Webb (1978). The model was designed to predict percentage reduction for any year for which a comprehensive survey of the field abundance of larval and juvenile striped bass was available. The basic procedure used in calculating percentage reduction estimates is outlined in Table 5-24.

The population of entrained organisms was divided into length classes rather than life stages or ages, because the major data bases that have been used classify striped bass

TABLE 5-24 PROCEDURE USED IN ESTIMATING PERCENTAGE REDUCTION OF STRIPED BASS YOUNG-OF-THE-YEAR CROPPED BY ENTRAINMENT AT THE PITTSBURG POWER PLANT

Step	Computation	Input
1	Compute numbers of larvae at each sampling location for each time period and for each length group.	Field estimates of larval densities separated into 4-mm length groups for different times of the year and at different sampling locations of the Delta. Water volumes of portions of the Delta that correspond to the different sampling locations.
2	Sum abundance estimates over all sampling locations to arrive at total abundance of larval length groups for each time period of the year.	
3	Compute duration of each length group from differences in timing of total abundance between successive length groups.	
4	Sum total abundance estimates over time to arrive at a yearly total abundance estimate for each length group.	
5	Compute total mortality rate for each length group from reductions in total yearly abundance of successive length groups.	

TABLE 5-24 (CONT.)

Step	Computation	Input
6	Compute monthly number of larvae cropped by the power plant for each length group.	Field estimates of larval densities occurring in the vicinity of the power plant, at different times of the year. Power plant monthly pumping volumes. Survival estimates of larvae passing through the cooling system of the power plant.
7	Sum estimates of numbers cropped over time to arrive at yearly cropping estimates for each length group.	
8	Partition the total mortality rates into mortality due to power plants and mortality due to other sources by comparing estimates of total yearly abundance and total yearly cropping for each length group.	
9	Compute percentage reduction for each length group and cumulative percentage reduction.	

on the basis of length. The total population of entrained organisms was defined to include all larval and juvenile striped bass up to a length of 50 mm. This range was determined by examining estimates, based on plant sampling, of the numbers of striped bass entrained and impinged at the Pittsburg Power Plant for 1978. The estimates of entrainment (Table 5-25) show a general decline in the number of organisms removed from the water body as length increases up to 50 mm, suggesting that entrainment is not operating on striped bass more than 50 mm long. These data do not provide conclusive proof of the hypothesized range of entrained lengths, because the decline in absolute number entrained can also be explained in part by the natural reduction in the number of organisms present in the field through successive length classes. However, additional evidence suggesting 50 mm as the end point of entrainment is provided by the impingement data (Table 5-26). Assuming that juvenile striped bass involved with the plant are either entrained or impinged, the sharp rise in impingement at the 50-75 mm length class provides further evidence that striped bass are not being entrained above 50 mm to a large extent.

5.6.3.4.2 Data Bases Used

To predict the percentage reduction of striped bass less than 50 mm in length, three field abundance data bases were used:

1. The egg and larval survey of CDF&G. This survey was conducted in the spring and early summer months from 1967 to 1977 (except 1974). Each survey consisted of a series of one-day sampling periods every other day at 33 locations throughout the Delta (except in 1969, when the time between sampling periods averaged six days). The purpose of this survey was to estimate the abundance of larval striped bass for each year. The portion of the data base used in the percentage reduction model consisted of the densities of striped bass measured at each station during each sampling period, segregated by 1-mm increments.
2. The Ecological Analysts plankton survey (EA 1981a). This survey was conducted in the spring and early summer months of 1978 and 1979. It consisted of sampling periods of approximately four days separated by three-day intervals, with samples collected at randomly selected stations within designated, spatially distinct reaches of the Delta. The

TABLE 5-25 ESTIMATED NUMBER OF STRIPED BASS LARVAE ENTRAINED, BY LENGTH CLASS, AT THE PITTSBURG POWER PLANT, 1978 - 1979

Length Class (mm)	Number
≤ 5	75,500,000
6	47,000,000
7	17,200,000
8	7,100,000
9	10,100,000
10	16,400,000
11	17,000,000
12	18,900,000
13	14,400,000
14	12,400,000
15	11,200,000
16	10,300,000
17	5,840,000
18	4,570,000
19	2,910,000
20	3,020,000
21	1,980,000
22	1,750,000
23	895,000
24	601,000
25	562,000
26	800,000
27	415,000
28	180,000
29	374,000
30	267,000
31	143,000
32	147,000
33	106,000
34	70,300
35	114,000
36	76,300
37	38,700
38	54,000
39	160,000
40	24,200
41	30,400
42	34,100
43	9,900
44	—
45	11,500
46	50,200
47	9,800
48	25,700
49	9,400
≥ 50	515,000

TABLE 5-26 ESTIMATED NUMBERS OF STRIPED BASS IMPINGED, BY LENGTH CLASS, AT THE PITTSBURG POWER PLANTS, 1978, 1979

Length Class (mm)	1978	1979
25-30	306	333
30-40	2,141	2,216
40-50	5,813	4,818
50-75	61,389	10,122
75-100	36,844	3,246
100-125	3,188	695
125-150	269	186
> 150	1,260	706

purpose of this survey was to estimate the abundance of larval striped bass for the years 1978 and 1979. The data used consisted of the densities of striped bass collected at each station, segregated by 1-mm increments.

3. The summer townet survey of CDF&G (Stevens 1977a). This survey was conducted during the summer months from 1961 to 1979 (except 1966). Each survey consisted of a series of one-day sampling periods every two weeks, with 30 stations sampled. The number of sampling periods per survey varied from three to five. The purpose of this survey was to estimate the striped bass population size when the average length of individiduals in the population was 38 mm (1.5 in). The portion of the data base used in the percentage reduction model consisted of densities of striped bass at each station during each sampling period, segregated by length increments of 2.5 mm (0.1 in).

The egg and larval and plankton surveys were used to predict percentage reduction for 7-18-mm striped bass; the summer townet survey was used for 20-50-mm striped bass. These length categories were established from the following considerations:

1. Striped bass less than 7 mm long were not adequately surveyed, because spawning generally occurred in reaches of the Sacramento River upstream from the area covered by the plankton surveys. Thus, by the time larval striped bass spawned in those reaches entered the survey area, the majority had developed beyond the early life stages. This hypothesis was supported by the catch curves of the 1978 and 1979 surveys (EA unpublished data b) showing striped bass <7 mm captured in lower densities than those >7 mm.
2. The plankton surveys used nets of 505-µm and 930-µm mesh, which do not reliably sample striped bass more than 26 mm long (L. W. Miller 1977).
3. For most years the egg and larval surveys ended before length classes greater than 18 mm reached their peak field abundances. Thus, accurate abundance estimates were not possible for the larger length classes.
4. The gear used in the summer townet survey efficiently sampled striped bass between 17 and 50 mm long (Chadwick 1964).

5. The townet survey was extensive enough during most years for the peaks in abundance to be measured for nearly all length classes between 20 and 50 mm.

One of the major assumptions of any percentage reduction model is that mortality rates are constant within a given population. Since mortality rates are known to differ among different size classes of larval and early juvenile fish, more accurate estimates can be made by grouping fish by size if data are available to establish mortality rates on this basis. For this reason, the population of entrainable lengths (7-50 mm) was divided into length classes.

For the 7-18 mm striped bass covered by the egg and larval plankton survey, 4-mm length classes were selected on the basis of an analysis of the 1978 and 1979 plankton survey data for fish up to 26 mm long. The surveys of these two years were chosen because more complete peaks in abundance were observed for the longer length classes. The length data were arranged in four separate patterns (2-, 3-, 4-, or 5-mm classes), and approximations of the instantaneous total mortality rate were computed for each pattern (Table 5-27). (See Appendix C for a discussion of the method used to compute the approximate mortality rates.) Four-mm classes were chosen for subsequent analyses, because they did not result in negative mortality rates and because they included a break between the 14- and 15-mm classes which indicated a definite change in mortality rates corresponding to the transition from larval to juvenile striped bass.

For the 20-50 mm striped bass covered by the summer townet survey, 5-mm length classes were selected, because lengths were recorded in tenths of inches in that survey and 5-mm groupings consistently enclosed pairs of 0.1-in. increments. For example, the 30-34 mm group enclosed the 1.2- and 1.3-in. length classes; the 35-39 mm group enclosed the 1.4- and 1.5-in. classes.

5.6.3.4.3 Model Description

Percentage reduction for each length class was computed by dividing an estimate of the number of individuals surviving the length class by an estimate of the number of individuals that would have survived the length class in the absence of power plant cropping, expressed as

TABLE 5-27 MORTALITY RATES COMPUTED FOR FOUR LENGTH CLASS GROUPINGS OF STRIPED BASS COLLECTED IN PLANKTON SURVEYS, 1978, 1979

Length Class Increment	Length Classes (mm)																			
	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26
1978																					
2 mm		0.131		0.105		0.076		0.165		0.049		0.068		0.150		0.094		0.077		0.116	
3 mm	0.213			0.090			0.153			0.058			0.136			0.089			0.152		
4 mm		0.135				0.131				0.088				0.118				0.146			
5 mm			0.123					0.117					0.096				0.162				
1979																					
2 mm		0.141		0.304		0.075		0.201		0.027		0.059		0.170		0.022		-0.041		0.025	
3 mm	0.277			0.249			0.133			0.036			0.144			-0.020			0.071		
4 mm		0.236				0.138				0.076				0.058				0.013			
5 mm			0.216					0.091					0.071				0.026				

$$P = \left(1 - \frac{\text{Numbers surviving with the power plant}}{\text{Numbers surviving without the power plant}}\right) \times 100$$

or,

$$P = \left(1 - \frac{N_0 \; S_n \; S_p}{N_0 \; S_n}\right) \times 100 \qquad (5\text{-}36)$$

where

P = percentage reduction

N_0 = initial population size of the length class

S_n = probability of survival due to natural mortality

S_p = probability of survival due to power plant effects.

Equation 5-36 can be written in the simple form

$$P = (1 - S_p) \times 100 \, . \qquad (5\text{-}37)$$

Percentage reductions through the entire range of affected length classes were computed as

$$P_c = \left(1 - \prod_{k=1}^{l} S_p^{(k)}\right) \times 100 \qquad (5\text{-}38)$$

where

P_c = cumulative percentage reduction

l = total number of affected length classes

$S_p^{(k)}$ = probability of survival for length class k due to the power plant.

Standing Stock Estimates

In each year, a certain (very large) number of fish are born to the population. These fish, born at various times but usually only at a restricted number of places, begin to die as soon as they are born, and so the number passing out of any length class of the new population is smaller than that which went into it. The exponential decline with time ($N_t = N_0 e^{-\lambda t}$), described for a hypothetical year class consisting of a

single cohort born at a particular time and developing in unison is shown in Figure 5-26. Two curves are shown: the decline is steeper and the resulting number of fish at any time is fewer if an additional source of mortality is affecting the young of the population ($N_t = N_0 e^{-(n+p)t}$). The values of the two curves and the difference between them are used to calculate the percentage reduction imposed on the population by the additional source of mortality.

The exponential decline can be reasonably approximated by estimates of the decline in the standing stock across a limited number of length intervals (Figure 5-26). These curves, however, are still hypothetical, based on a single-cohort population. To estimate these hypothetical curves from the field abundances encountered in a water body, use is made of the equivalence of the areas under two curves, the cumulative duration-abundance curve found in field sampling for length-class-k and the theoretical length-class-k duration-abundance curve (Figure 5-26).

The methods used to derive the model and to calculate the corresponding mortality rates are described in Section C.3, Appendix C. The estimation of the percentage reduction for each length class requires comparable estimates of the abundances (standing stocks) of a given length class and of the number cropped as a result of entrainment. These two parameters, estimated for specific size classes through time, are the primary inputs to the percentage reduction analysis.

The length-specific field abundance data gathered in the three Delta surveys were used to estimate the value for the area under the field abundance curve, $A^{(k)}$. To compute $A^{(k)}$ the following data were required:

1. the mean density of organisms in length class k within each spatially distinct stratum,
2. the volume of each stratum,
3. the duration of each survey sampling period, and
4. the time between each pair of sampling periods.

For both the egg and larval survey and the summer townet survey, the duration of sampling periods and the time between them were consistent from year to year. In each case the sampling period consisted of one day. The surveys differed in the time between sampling periods: 2-10 days for the egg and larval survey and two weeks for the summer townet survey.

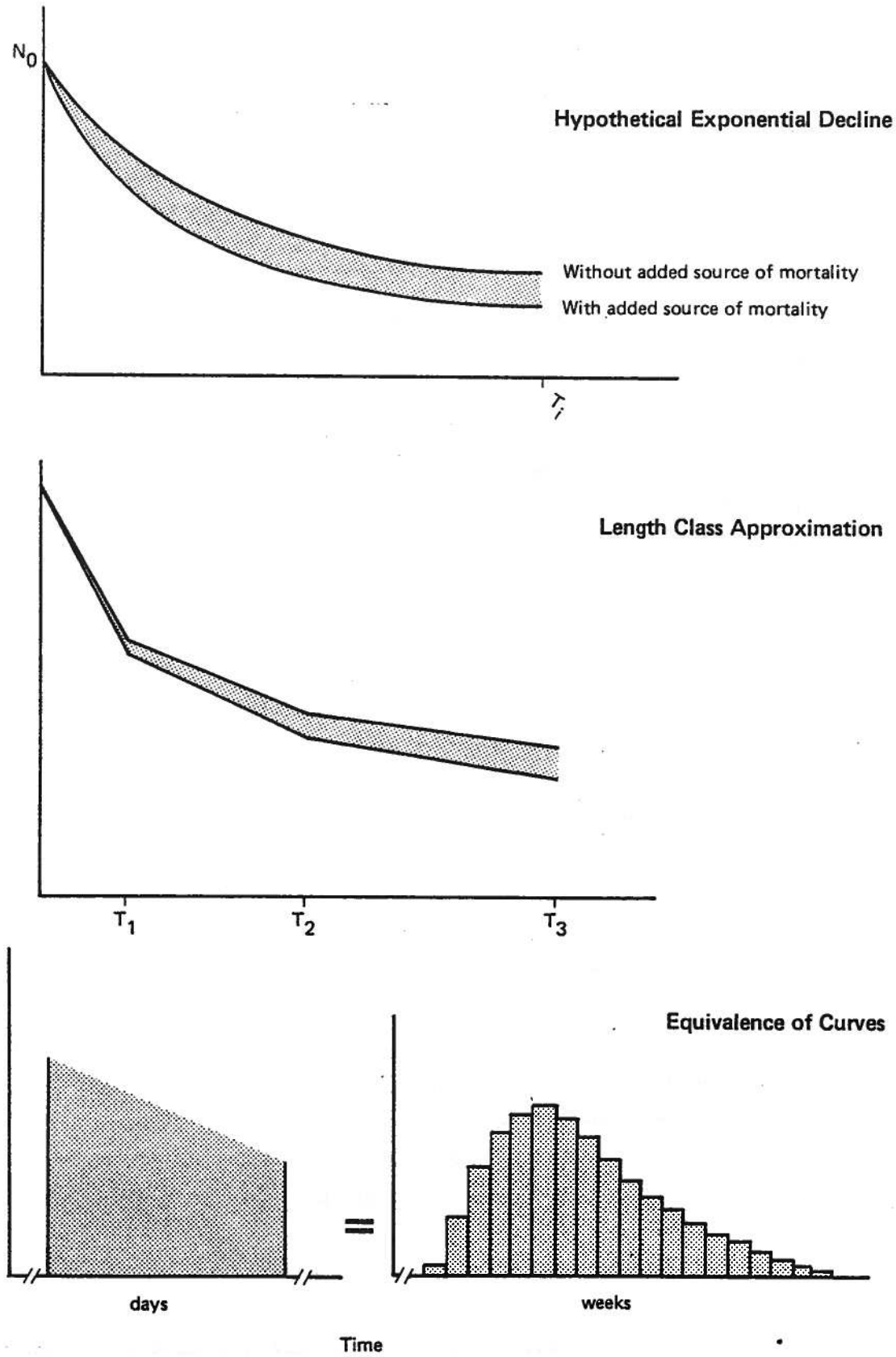


Figure 5-26. Hypothetical curves showing the exponential decline in abundance of a single-cohort population of young fish and the equivalence of a single-cohort period of abundance to cumulative abundances in a multiple-cohort population.

The estimate of $A^{(k)}$ for each survey was computed as

$$A^{(k)} = d \sum_{i=1}^{S} \sum_{j=1}^{t} D_{ij}^{(k)} V_j \qquad (5\text{-}39)$$

where

$D_{ij}^{(k)}$ = mean density of length class k in stratum j in sampling period i
V_j = volume of stratum j
S = number of sampling periods
t = number of strata
d = number of days between sampling periods.

Both the summer townet survey and the egg and larval survey sampled a consistent set of stations each year (Appendix C). Stratum designations and associated volumes in billions of cubic feet were developed for the egg and larval survey:

Designation	Stations	Volume
Main channel and bays	1-17	27.8
Lower Sacramento River	19-32	6.0
Lower San Joaquin River	33-53	7.0
Delta region	55-61	11.3

and for the summer townet survey:

Designation	Stations	Volume
Main channel	405,411,501,504,508,513	14.2
Upper bays	602,418,519	12.1
Montezuma Slough	606,609,610	1.6
Lower Sacramento River	704,706,707,711	6.0
Lower San Joaquin River	801,520,804,809,812,815	7.8
South Delta region	902,906,914,910,912,915,918	9.1
North Delta region	919	2.2

Since the duration of sampling periods and the time between samples varied in each year of the 1978 and 1979 plankton surveys, a different method for estimating $A^{(k)}$ was employed. Rather than applying a constant value for the time between each sampling period, each sampling period was assigned a value for d that was determined by the duration of the sampling period and the time between it and the previous and succeeding sampling period. The formula used to compute $A^{(k)}$ was

$$A^{(k)} = \sum_{i=1}^{S} d_i \sum_{j=1}^{t} D_{ij}^{(k)} V_j \qquad (5\text{-}40)$$

where

d_i = number of days applied to the field abundance measured during sampling period i.

The values assigned to d_i for the 1978 and 1979 plankton surveys are listed in Table 5-28.

The two sampling periods before that of 18-20 May of the 1978 survey were not used in estimating $A^{(k)}$ because they covered only part of the Sacramento - San Joaquin Delta. The exclusion of these sampling periods does not greatly affect the estimate of percentage reduction for 1978, because at the time of the 18-20 May sampling period the abundance of the shortest length class (7-10 mm) was low, indicating that only a small portion of the population had not been sampled.

Entrainment Cropping Estimates

The computation of power plant cropping for each length class required the following data:

1. the mean number of organisms of each length class subject to entrainment during each sampling period,
2. power plant pumping volumes during the sampling period,
3. duration of the sampling period,
4. entrainment mortality estimates, and
5. the time period between each pair of sampling periods.

The density of organisms subject to entrainment was calculated from the CDF&G egg and larval survey data using the densities measured at the three egg-and-larval sampling stations closest to the power plant, 11, 13, and 15.

The number of striped bass cropped at the power plant for each CDF&G egg and larval survey was calculated as

TABLE 5-28 SAMPLING PERIODS AND NUMBER OF DAYS ASSOCIATED WITH EACH SAMPLING PERIOD FOR THE 1978 AND 1979 PLANKTON SURVEYS

Sampling Period	d_i (days)
1978	
18-20 May	10
25-27 May	10
6-9 June	9
12-16 June	8
19-24 June	8
26 June - 1 July	8
6-8 July	6
10-14 July	7
1979	
18-20 April	6
24-28 April	6
1-4 June	7
7-10 June	7
15-18 June	8
22-25 June	7
29 June - 1 July	6
5-8 July	6
10-15 July	7
18-22 July	6
23-29 July	8
3-6 August	8
10-13 August	7

$$c = \sum_{l=1}^{m} T_l S_l \frac{\sum_{j=1}^{r_l} D_{jl} P_{jl}}{r_l} \qquad (5\text{-}41)$$

where

$D_{jl}^{(k)}$ = average density of striped bass of length class k measured during sampling period j of month l

P_{jl} = volume of water pumped on the day of sampling

r_l = number of sampling periods in month l

T_l = number of days in month l covered by the survey

S_l = entrainment mortality for month l

m = number of months covered by the survey.

The estimates of numbers cropped for the Pittsburg Power Plant were applied to the percentage reduction model.

The density of organisms in the 20-50 mm range subject to entrainment was calculated from the CDF&G townet survey data at sampling station 520, the sampling station closest to the plant. The number of striped bass cropped at the power plant for each survey was calculated as

$$c = \sum_{j=1}^{r} D_j^{(k)} P_j M_j T \qquad (5\text{-}42)$$

where

$D_j^{(k)}$ = density of striped bass of length class k measured during sampling period j

P_j = volume of water pumped on the day of sampling period j

r = number of sampling periods

M_j = entrainment mortality on the day of sampling period j

T = number of days betweeen sampling periods.

The sampling stations in the 1978 and 1979 plankton surveys (EA 1981a) were randomly selected within designated river strata, so the densities of striped bass

considered vulnerable to entrainment for a given sampling period were determined from the group of sampling stations that fell within a defined region near the power plant. A map of the regions used for the power plant is presented in Figure 5-27. The actual portions of the river strata used for the plant were all of the New York Slough stratum and the Suisun Bay stratum between Kilometers 13 and 19.

The number of striped bass cropped was calculated from the data collected at sampling stations within these regions as

$$c = \sum_{l=1}^{m} \frac{\sum_{j=1}^{r_l} \left(\sum_{i=1}^{n} N_{ijl} \Big/ \sum_{i=1}^{n} V_{ijl} P_{jl} \right)}{r_l} T_l M_l \qquad (5\text{-}43)$$

where

$N_{ijl}^{(k)}$ = number striped bass of length class k collected at station i during sampling period j of month l

V_{ijl} = volume of water sampled at station i during sampling period j of month l

P_{ij} = volume of water pumped on the day corresponding to the midpoint of sampling period j

n = number of stations falling within plant vicinity boundaries

r_l = number of sampling periods in month l

T_l = number of days in month l covered by the survey

m = number of months covered by the survey

M_l = estimated entrainment mortality for month l.

5.6.3.4.4 Results and Discussion

Percentage reduction was calculated for each length class and year of available data for the Pittsburg Power Plant (Table 5-29). Cumulative percentage reduction was also calculated for the 7-18, 20-50, and 7-50 mm size ranges for the power plant. When estimates of percentage reduction were not attainable for all length classes within a particular size range, an average value for the attainable estimates in the range was determined and applied to Equation 5-38.

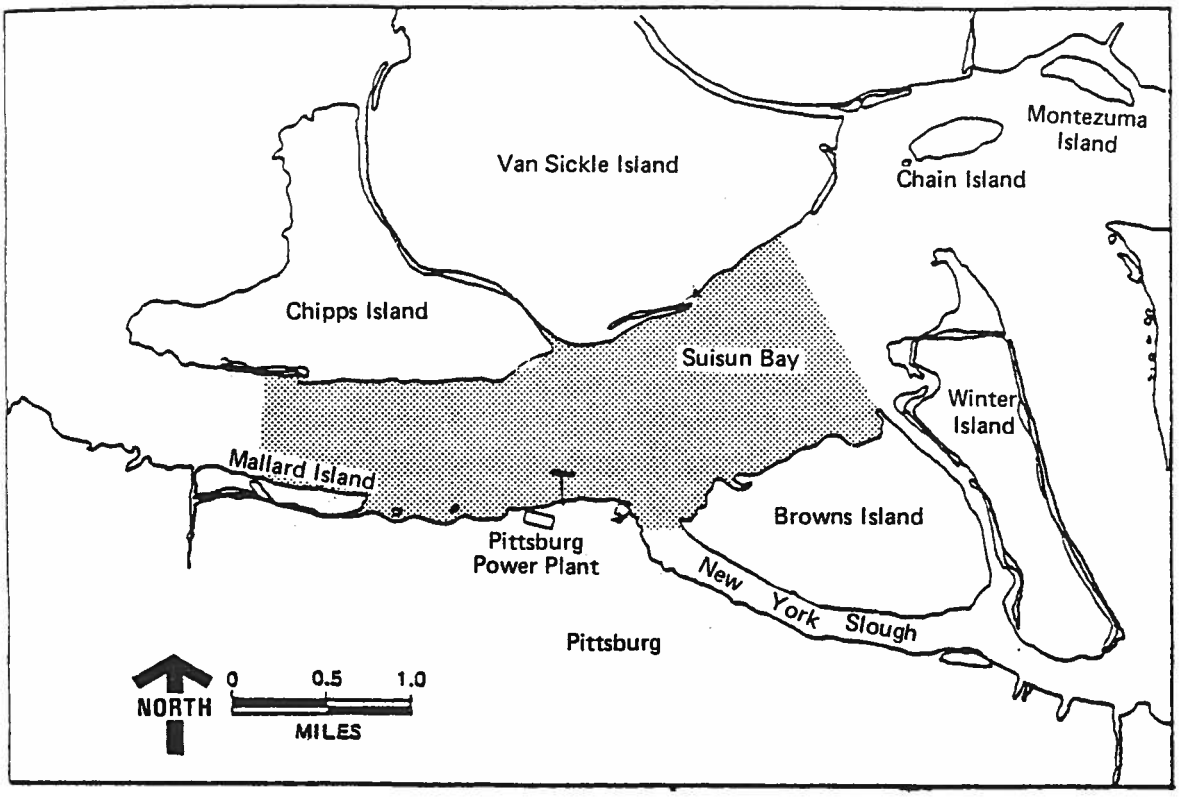


Figure 5-27. Reaches of Suisun Bay from which samples were used to estimate entrainment densities at the Pittsburg Power Plant, 1978, 1979.

TABLE 5-29 ESTIMATED PERCENTAGE REDUCTION IN YOUNG-OF-THE-YEAR STRIPED BASS RESULTING FROM ENTRAINMENT AT THE PITTSBURG POWER PLANT, 1961-1979

Year	Size Interval (mm) 7-10	11-14	15-18	Cum. I	Size Interval (mm) 20-24	25-29	30-34	35-39	40-44	45-49	Cum. II	Cum. III
1961							1.04	0.826	2.23	2.14	9.00	13.19
1962								0.746	0.520	0.502	3.48	5.18
1964					0.484	0.351	0.371	0.265	0.259		2.06	3.07
1965							0.690	0.670	0.599		3.85	5.73
1968	0.239	0.337	0.554	1.13		1.33	1.18	1.28	1.19	1.35	7.36	10.83
1970	0.327	0.501	0.554	1.38			0.379	0.320	0.407	0.410	2.25	3.60
1971					0.533	0.838	0.753	0.620	0.593	0.592	3.50	5.20
1972	0.277	0.538	0.431	1.24		0.887	0.869	0.703	0.591	0.698	4.41	5.60
1973								0.503	0.716	0.824	4.02	5.96
1974								0.280	0.275	0.259	1.62	2.42
1975	0.420	0.472	0.441	1.33		0.822	0.645	0.578	0.430	0.578	3.61	4.89
1976								2.96	2.68	2.46	15.15	21.83
1977					0.559	0.909	1.30	3.23	2.24		9.49	13.89
1978	0.664	0.323	0.231	1.21								3.59
1979	2.02	2.89	3.07	7.78				1.34	1.03	1.80	8.05	15.21

Note: Cum. I = 7-18 mm; Cum. II = 20-49 mm; Cum. III = 7-49 mm.

Percentage reduction was not calculated for each length class for all years, because of various forms of data inadequacy in the field survey data bases. Six criteria were used to eliminate data of various length classes from consideration:

1. A survey was not conducted which covered the length class.
2. A peak in population abundance for the length class was not measured (because the survey began after the length class had reached its peak or ended before the peak abundance occurred).
3. The survey did not cover the entire distributional range of the length class. (This occurred exclusively in the summer townet survey, when available evidence indicated that a large proportion of the population of juvenile striped bass were residing outside the bounds of the survey, in San Pablo Bay.)
4. Abundance data were available for only one or two consecutive length classes. (A minimum of three consecutive length class abundance values were necessary for the application of the model.)
5. The field abundance measured for the length class represented an increase rather than a decrease in abundance from the previous length class. (Application of such data to the model would result in predictions of negative mortality rates.)
6. The length class field abundance data represented unrealistic population dynamics. (This criterion applied only to the 1973 egg and larval surveys, where sharp rises and declines in field abundance clearly indicated errors in the data.)

The values presented in Table 5-29 represent the best estimates of percentage reduction in the striped bass population resulting from entrainment at the Pittsburg Power Plant. Proper interpretation of these values, however, requires a clear understanding of the limitations of both the model and its input data.

One such limitation of the model is the assumption that the striped bass within the 7-50 mm size range are not able to compensate for power plant losses. This is a weak assumption, in that compensation of density-dependent mortality is generally thought to occur in fish populations, as indicated by the results of numerous studies summarized by McFadden (1977). Further evidence supporting compensation comes from Cushing (1974), who developed population dynamics models that incorporate density-dependent larval mortality using data from populations of cod, plaice, and

herring. If compensatory mechanisms are operating in the larval and juvenile striped bass population of the Sacramento - San Joaquin Delta, the predictions of percentage reduction computed with this model overestimate the loss.

Another point that should be considered in interpreting the results of the model is the reliability of using water body samples to estimate densities of striped bass entrained at power plants. As discussed in Section 5.6.3.4.2, midchannel samples collected in the CDF&G egg and larval and summer townet surveys were used to estimate entrainment densities for the power plant. To investigate the reliability of these midchannel density estimates, EA conducted a special study in 1979, collecting samples at stations located near the intake structure of the power plant and at the corresponding midchannel stations. These paired stations, part of the plankton survey of 1979, were sampled during each sampling period. The data were grouped by length class (7-10, 11-14, and 15-18 mm), and statistically analyzed for differences in larval density between midchannel and intake stations. The statistical test was a randomized block design, and variations in densities due to differences between sampling periods were removed from the comparison of midchannel vs. intake stations. There were no significant differences between intake and midchannel stations for any length class ($\alpha = 0.05$).

For the length classes covered by the summer townet survey, a comparison of field densities and power plant entrainment densities is possible for the years 1978 and 1979 (Table 5-30). In both years the densities measured in midchannel are greater than the densities measured at the power plant by pump sampling. Since estimates of density measured at the plants are a reliable indicator of entrainment, using the densities measured in the field from the summer townet sampling stations to predict entrainment leads to overestimation of percentage reduction.

With respect to the field stations of the plankton survey used to estimate entrainment densities of larval striped bass in 1978 and 1979, a third comparison is possible. Table 5-30 lists the estimates of larval entrainment for these years derived from both field and plant sampling for the Pittsburg Power Plant. These values show that for the 7-10 mm length class, estimates derived from field sampling were consistently higher than estimates derived from pump sampling at the plant. For the longer length classes, on the other hand, plant- and field-derived estimates were similar, except for those for 1978. On the assumption that the pump samples provide reliable estimates

TABLE 5-30 NUMBERS (millions) OF STRIPED BASS LARVAE ENTRAINED AT THE PITTSBURG POWER PLANT, ESTIMATED BY MEANS OF ENTRAINMENT PUMP SAMPLING (PUMP) AND OF MAIN CHANNEL DENSITY - COOLING WATER VOLUME CALCULATIONS (NET), 1978-1979

Length Class (mm)	1978		1979	
	Net	Pump	Net	Pump
7-10	70,400	50,800	50,100	9,000
11-14	13,300	62,700	11,600	5,900
15-18	5,100	31,900	6,200	5,400

of the true entrainment densities, it can be concluded that entrainment is overestimated for the years 1978 and 1979 for the 7-10 mm length class.

Apart from the question of the reliability of using field sampling stations to estimate entrainment is the question of the level of confidence in the field abundance and entrainment estimates derived from the same stations. The fewer the sampling stations and sampling periods in a survey, the lower the confidence in the estimates of the field abundance and entrainment. The CDF&G egg and larval surveys and the EA plankton surveys had enough sampling stations and sampling periods to establish a reasonable degree of confidence in the estimates of both field abundance and entrainment, although confidence limits were not calculated for these estimates. The summer townet surveys, however, while possessing an adequate number of sampling stations, were characterized by only 3 to 5 sampling periods separated by 2-week intervals. This feature of the summer townet survey lowers the level of confidence in the estimates of field abundance and entrainment, since the estimation procedure is based, in part, on the assumption that densities measured during the sampling periods were constant throughout the two weeks separating one period from another.

From the above considerations, certain conclusions can be drawn from the results of the percentage reduction model. For all years examined, short-term entrainment impact on young-of-the-year striped bass at the Pittsburg Power Plant is relatively low, ranging from 2.5 to 22 percent (mean value 8 percent). Estimates of percentage reduction were generally highest in those years with below-average Delta outflows. The critical low-flow years 1976 and 1977 show a comparatively more severe impact. The higher level of impact predicted for these years is theorized to be the result of the movement of the entrapment zone in spring and summer into the area where the plant is located. Since larval and juvenile striped bass are found in highest density within this zone, the numbers removed from the population by the operation of the power plant during those two years constituted a larger fraction of the population than those removed in years of greater outflow, when the entrapment zone lies downriver from the confluence.

Annual percentage reductions of young-of-the-year striped bass for the Pittsburg Power Plant combined were estimated to be 18 percent during years of critically low Delta outflows in May (<5,000 cfs), 8.2 percent for May flows between 5,000 and 15,000 cfs, and 4.2 percent for years with May outflow greater than 15,000 cfs. The

question of whether the level of percentage reduction predicted has an impact on the adult population of striped bass is examined in the next section.

5.6.3.5 Adult Population Model Development and Projections

Although there are some very slight losses of adult striped bass at the power plant intake screens, the principal concern is the effect of the power plant's reducing recruitment of young bass to the adult population. This is because the principal value of the fishery is in the fish three years old and older.

Up to this point the impact assessment has concentrated on the population dynamics of striped bass young-of-the-year and immediate impact. In the preceding sections we have developed a measure of young-of-the-year abundance, identified the most important environmental factor affecting young-of-the-year variability, estimated the percentage of the young-of-the-year population loss at the power plant, and examined the relationship between young-of-the-year abundance and stock size. In this section we will continue the investigation by calculating age-specific survival from young-of-the-year to three-year old, 3- to 4-, 4- to 5-, and 5- to 6-year-old fish. This information will be combined with our previous findings to construct an adult population model that will assess the effect of environmental factors and of young-of-the-year losses at the power plant on the adult population of striped bass in the Bay-Delta.

Before any investigation into adult modeling can begin, however, the question of compensatory survival, that is, whether or not the survival between age groups is a function of population size, must be addressed. McFadden (1977) suggests that investigators include compensation in their adult modeling attempts. Although it is widely believed that compensatory mechanisms are operating in fish populations, the lack of adequate data and the difficulty encountered in measuring the magnitude of the phenomenon has forced many researchers to ignore its existence. Although the striped bass age-specific data base is relatively small, we will attempt to identify and quantify any apparent compensatory survival.

As in the previous sections, we will use a data-base approach for the adult population modeling. Since no obvious dependency was found between the adult stock of striped bass and recruitment into the sport fishery (Section 5.6.3.3), and since at present population levels Delta outflow appears to be the major factor affecting year class

strength (Section 5.6.3.2), we have used a modified Leslie matrix model in our investigation. The modified model contains the age-specific mortality rates (calculated in this section) normally found in the original model (Leslie 1945); it also includes environmental factors, in a design similar to those used by Sommani (1972) and Kohlenstein (1980). Our model does not, however, include any estimates of age-specific fecundity, because it was found (Section 5.6.3.3) that variability in egg production (stock size) did not appear to affect recruitment. Annual young-of-the-year abundance was calculated from Delta outflow (based on a relationship developed in Section 5.6.3.2) for a 50-year period under each of the three water management plans (historic, 1985 and 1990). Young-of-the-year losses at the power plant were assumed to crop a differential percentage (18 percent for May flows <5,000 cfs; 8.2 percent for May flows between 5,000 and 15,000 cfs; and 4.2 percent for May flows >15,000 cfs) of the young-of-the-year population, as discussed in Section 5.6.3.4. Using the estimated young-of-the-year abundance and age-specific survival, the Leslie matrix model was run under the three water management plans assuming (1) that the power plant was in operation and (2) that the power plant was not in operation. The differences in adult population levels resulting from the two assumptions were then compared to determine the effect of power plant operation on the adult striped bass population.

5.6.3.5.1 Age-Specific Survival Rates

In calculating age-specific survival, only two data sets are required, one containing the year class strength measures developed in Section 5.6.3.2, and another containing striped bass age-specific abundance estimates for the years 1969-1976 (D. Stevens personal communication, 1979) (Table 5-31). In the years when no survey was conducted (1966), or when spring flows were too high (1967 and 1969), an estimate of year class strength was made using the regression model developed in Section 5.6.3.2.

The survival from young-of-the-year to three-year-old was calculated by dividing the number of striped bass produced in the 35-39 mm size group by the number of age-three adults, three years later. The survival rates between the other age groups were calculated in a similar manner. In an attempt to identify compensation, the survival rate between young-of-the-year and three years was plotted against young-of-the-year abundance. Similar plots were made between survival and abundance for the older age classes of fish.

TABLE 5-31 ESTIMATED AGE-SPECIFIC ABUNDANCES OF ADULT STRIPED BASS IN THE BAY-DELTA, 1969-1976

Year	Age Group 3	4	5	6	7+
1969	1,113,000	412,000	269,000	170,000	252,000
1970	1,275,000	484,000	201,000	129,000	258,000
1971	840,000	602,000	224,000	118,000	225,000
1972	1,260,000	522,000	407,000	124,000	205,000
1973	766,000	481,000	235,000	177,000	374,000
1974	975,000	339,000	273,000	136,000	259,000
1975	1,024,000	644,000	266,000	161,000	337,000
1976	873,000	484,000	186,000	131,000	260,000

There was no significant difference in the average survival rates between 3- and 4- (P = 0.50), 4- and 5- (P = 0.53), and 5- and 6-year old fish (P = 0.53). There was also no apparent relationship between the survival of fish from 3 to 4 years and the abundance of three-year-olds, and although there appeared to be a slightly stronger relationship between survival and abundance for the 4- and 5-year-old fish (Figure 5-28), the large amount of variability in the data made interpretation difficult. It was decided that the mean age-specific survival rates of 3-, 4-, and 5-year old fish would be used in the modified Leslie matrix model.

The calculated average survival from young-of-the-year to three years using both measured and estimated young-of-the-year abundance was 0.023; using only measured abundance, it was 0.026. In addition, there was a highly significant ($P < 0.01$) negative correlation between survival from young-of-the-year to three years and the abundance of young-of-the-year (Figure 5-29). Using a variable-selection technique similar to the type used in Section 5.6.3.2, an exponential (natural logarithm) curve was fitted to the data which resulted a coefficient of determination, r^2, of 0.87. Although the sample size is small, these results indicate that differential survival between young-of-the-year and three years continues to occur after the year class is set. It appears that in the Sacramento - San Joaquin striped bass population, year classes with high young-of-the-year abundance tend to have lower survival to three years than year classes with low young-of-the-year abundance: in other words, there appears to be compensatory survival. This compensatory survival, if it exists, reduces the high variability in young-of-the-year abundance (a coefficient of variation (CV) of 48 percent), resulting in less variability among the three-year-olds (CV = 18 percent). Although a high (or low) young-of-the-year abundance will generally result in a correspondingly high (or low) abundance of three-year-olds, this abundance will not be as high (or low) as would be expected without compensatory survival. Thus, the three-year-old abundance estimates are a reflection of the striped bass year class strength three years earlier and the compensatory survival demonstrated here. Based on these results, it was decided to run the modified Leslie matrix model assuming two alternatives: (1) constant survival of 0.026 from young-of-the-year to three years, and (2) differential survival from young-of-the-year to three years based on the compensatory relationship developed in this section.

5.6.3.5.2 Model Calibration

Using the modified Leslie matrix, two adult models were developed, one assuming

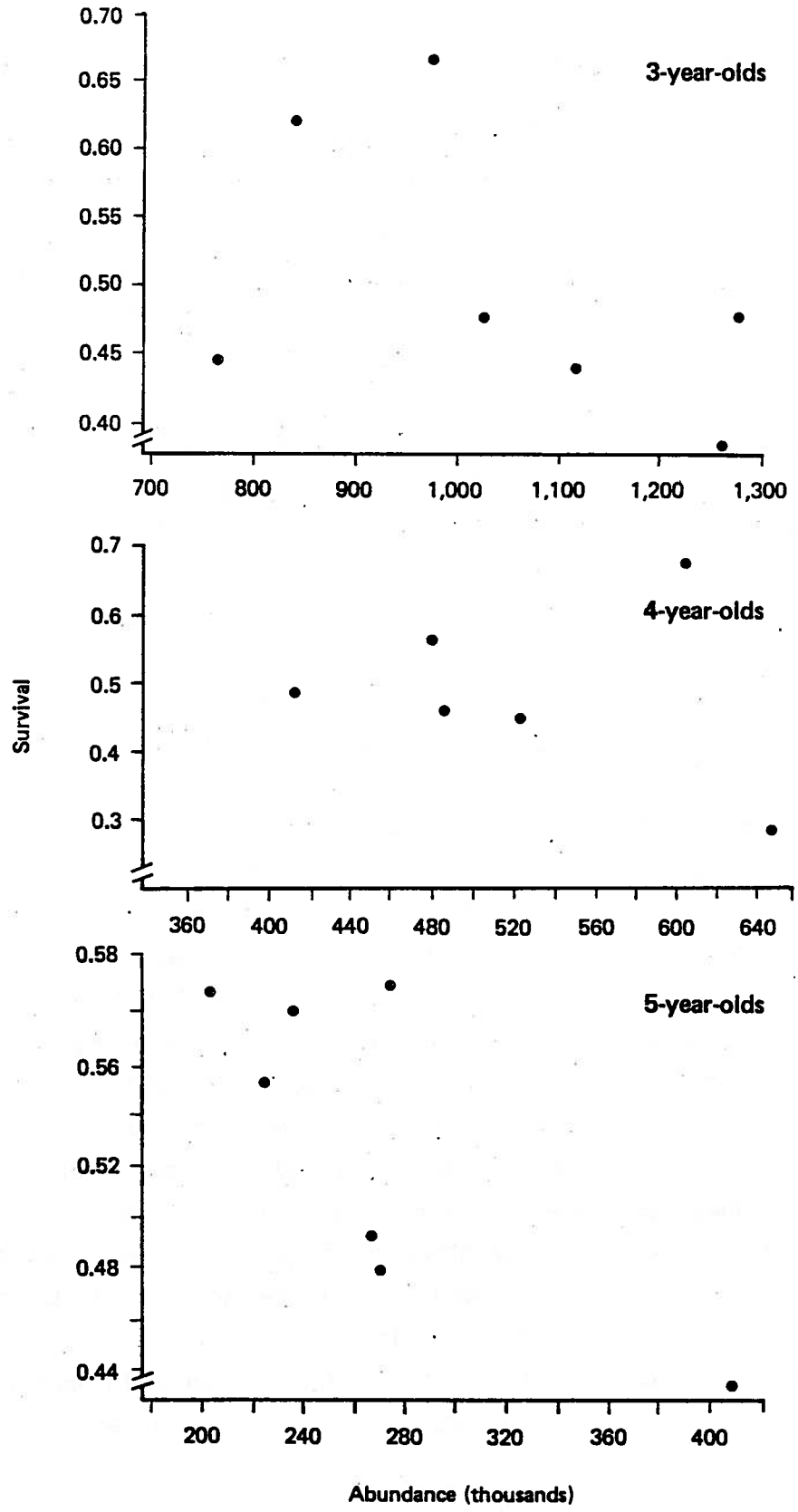


Figure 5-28. Abundance vs. survival for three–, four–, and five-year-old striped bass.

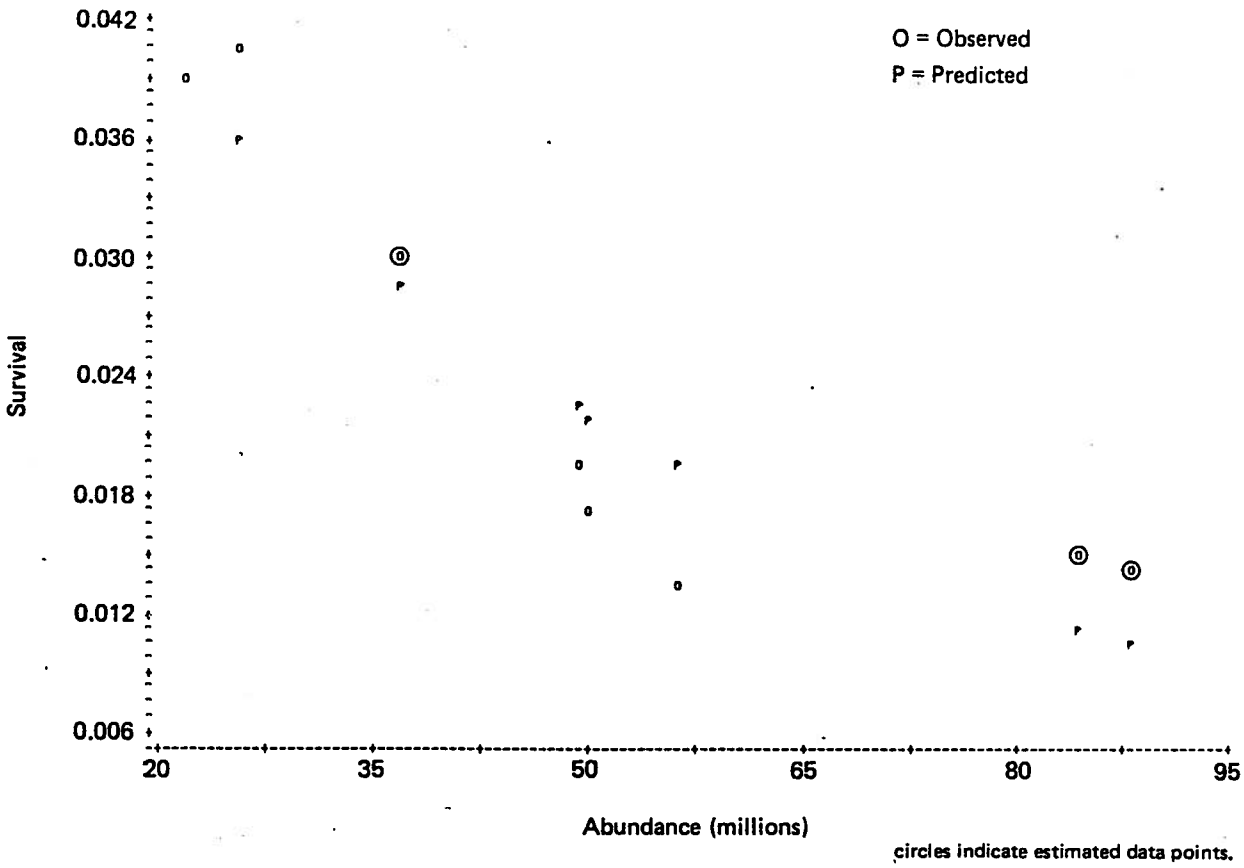


Figure 5-29. Correlation of estimated striped bass young-of-the-year abundance with survival from young-of-the-year to three years, 1969-1976.

constant survival from young-of-the-year to three years, the other assuming compensatory survival of the type just described. Both models assumed that survival was constant from three to four, from four to five, and from five to six years, and that young-of-the-year abundance was not a function of stock size but rather depended on the magnitude of May outflow, as described in Section 5.6.3.2. Estimates of May outflow for the period 1922-1971 were obtained from the Department of Water Resources (unpublished data). The Leslie matrix model produced yearly population estimates of adult striped bass three years old and older. In addition, the total number of legal size striped bass (calculated by adding one-half of the abundance of three-year-old to the abundance of fish four years old and older) was determined for each year.

5.6.3.5.3 Results and Discussion

On the assumption of noncompensatory survival, the estimated abundance (1928-1971) of legal size striped bass ranged between 1.8 and 3.8 million and averaged 2.8 million (Figure 5-30). On the assumption of compensatory survival between young-of-the-year and three years, the estimated 50 year abundance of legal size striped bass ranged between 1.4 million and 1.8 million and averaged 1.7 million (Figure 5-30). Under the compensatory survival model the range in abundance is only 400,000, and the mean abundance is 1.1 million, less than the estimate based on the noncompensatory survival model. In order to select the more appropriate of the two models, these calculated estimates need to be compared with actual field estimates of the historical population size.

After the initial introduction of striped bass into the estuary in the 1880s, the population was abundant from 1900 to 1940 and has generally declined since (Stevens 1980). In the early part of the century, the population sustained a large commercial and sport fishery. In 1935 the commercial fishery was closed because of competition with the sport fishery (Chadwick et al. 1977). Since 1936 CDF&G has monitored the sport fishery catch in order to evaluate the status of the population. Of the two principal monitoring methods used, postcard surveys and party boat catch records, the party boat records are considered to be the best long-term data available on striped bass abundance (Stevens 1977). The catch and the number of anglers on each trip has been reported by party boat operators since 1938 (Figure 5-31). These records indicate that during the 1940s and early 1950s the sport catch declined steadily, which brought about more stringent regulations from 1954 to 1957

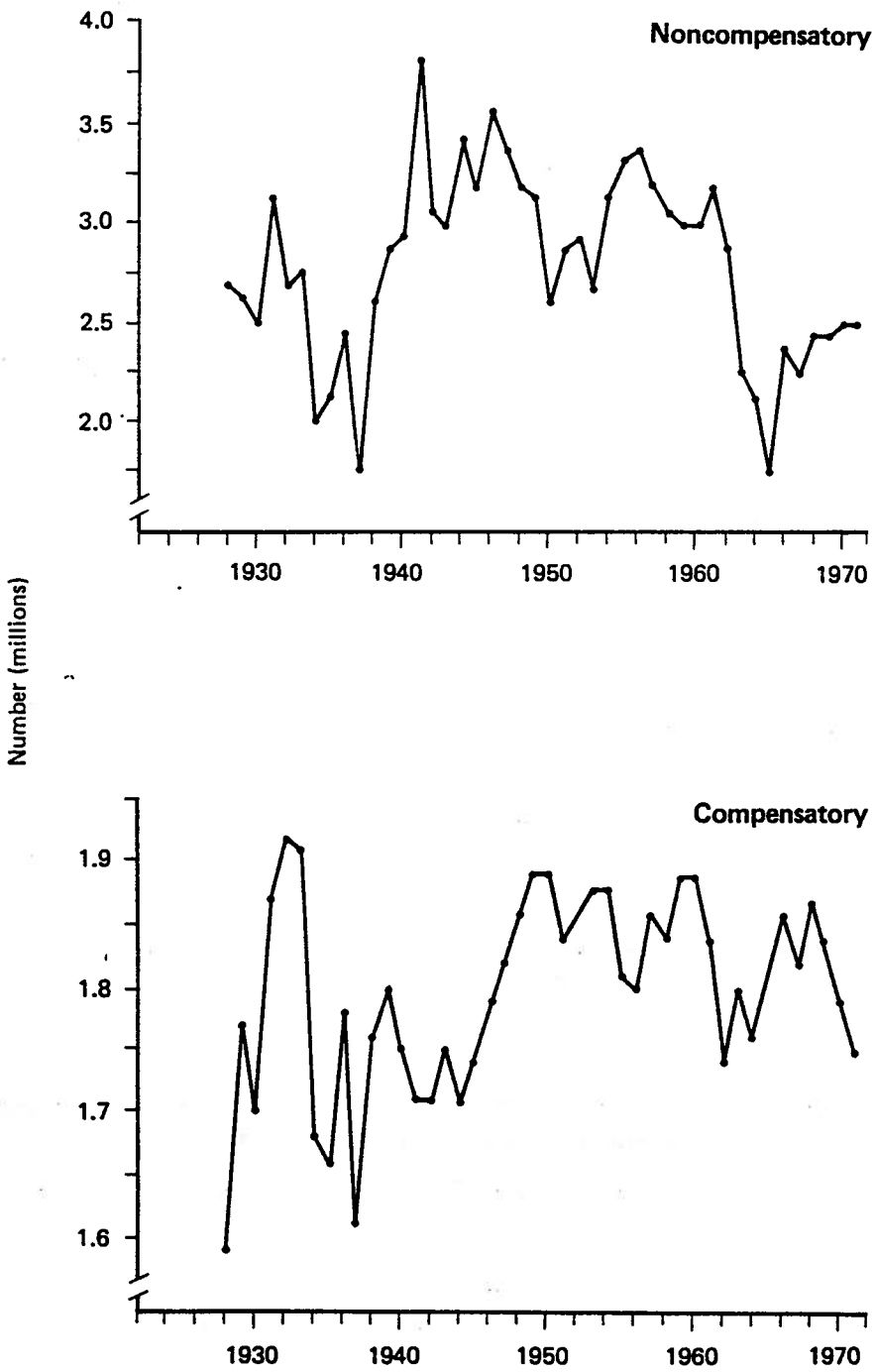


Figure 5-30. Estimated numbers of legal-size striped bass in the Bay-Delta predicted by compensatory and noncompensatory survival models, 1928-1971.

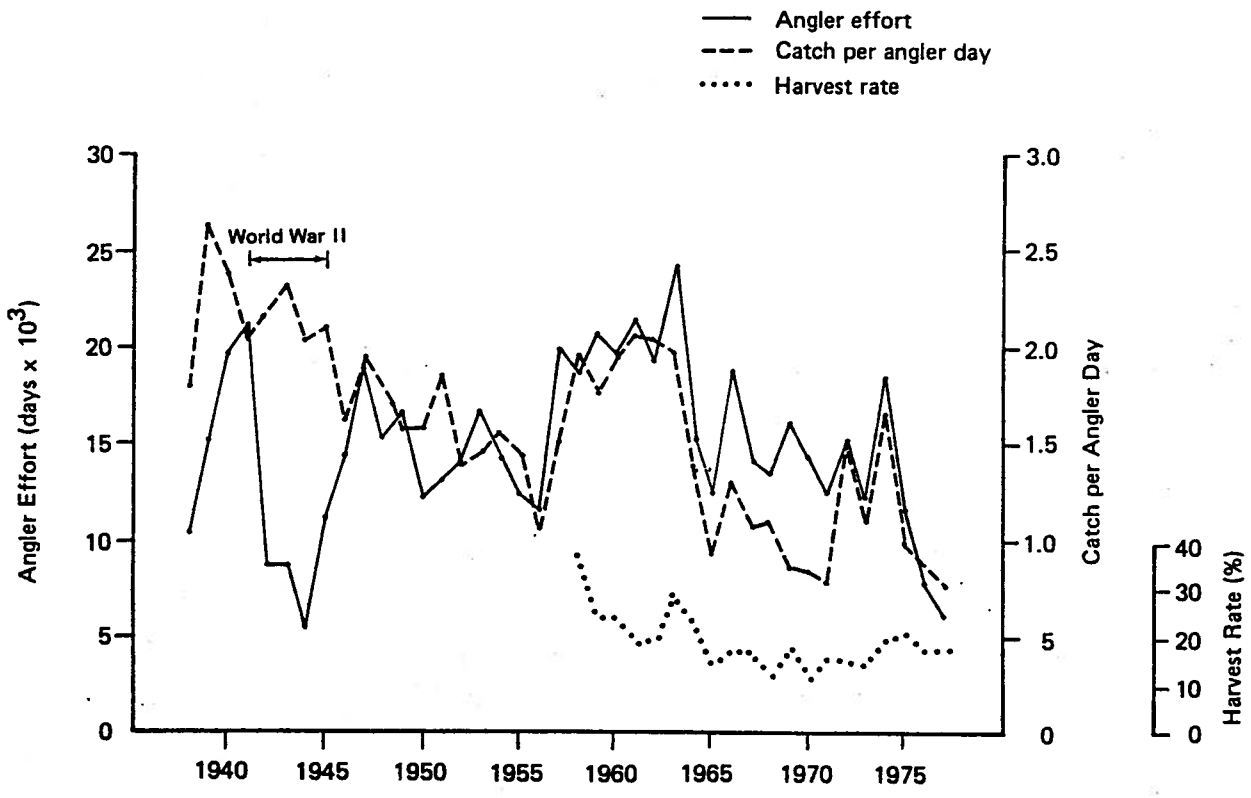


Figure 5-31. Trends in angler success, angler effort, and harvest rate in the striped bass party-boat fishery of the Bay-Delta, 1938-1977.

(Chadwick 1962). An increase in angler catch occurred during the late 1950s and early 1960s because of an increase in bass abundance and improved fishing techniques (Stevens 1980). Following this, the angler catch declined in the late 1960s and early 1970s.

Before comparing actual field abundance estimates to model prediction estimates, it should be remembered that the indices in Figure 5-31 are estimates of two-year-old bass abundance for 1938-1954 and estimates of three-year-old bass abundance for 1955-1977, and not estimates of total legal-size striped bass as developed in the Leslie matrix models. In spite of this, the noncompensatory model predictions agree very well with the historic trends. The populations reached a peak abundance in 1941; declined through the 1940s and into the early 1950s, began to rise again in 1951; peaked in 1956 and 1961; declined to an all-time low in 1965, and then began to increase again from 1966 to 1971. The compensatory-model predictions vary considerably less, and they do not agree as well with the historical trends: the population size increases from the 1940s to the mid 1950s; there is a slight decline in the population size in the late 1950s and early 1960s; the population size increases from 1962 to 1968 and then decreases from 1969 to 1971 (Figure 5-30). Although the noncompensatory model appears to fit the historical trends better than the compensatory model, the range and mean population abundance estimates of the compensatory model agree much better with actual population estimates. The noncompensatory average of the historical population abundance is 2.8 million, whereas the compensatory average is 1.7 million. The only actual population size estimate available is for the period 1969-1976 (D. Stevens, CDF&G personal communication, 1979), and it ranges from 1.5 million to 1.9 million with an average of 1.7, very close to the average population size estimated from the compensatory model. This close agreement is not surprising, however, since the compensatory model was generated using the same data base. But it should also be remembered that the noncompensatory model was also generated using the same data base. The question that remains to be answered is whether or not the population abundance really varied as much as the noncompensatory model predicts (between 1.8 and 3.8 million) or as little as the compensatory model predicts (between 1.4 and 1.8 million).

Unfortunately, there is no known data base available that can answer this question, and thus no clear-cut answer to the question of compensatory survival in the adult model. CDF&G has also reported conflicting views on the existence of compensation. Until the 1970s they believed that the adult stock of striped bass was

largely controlled by density-dependent (compensatory) processes (Chadwick 1979). The extreme amount of cannibalism evident in the population (Stevens 1966b) led CDF&G to believe that the population size was largely self-controlled. Upon detailed study and analysis, however, they changed their opinion. The decline in the sport catch since the early 1960s was believed to be due to a decline in the adult stock that was associated with low recruitment (Stevens 1977b). The low recruitment was attributed to a deterioration in the environment (CDF&G 1976), possibly caused by pollution, diversions at the Delta and Tracy pumping plants, and power plant entrainment (Stevens 1980). On the basis of our findings and those of CDF&G, we have decided to use both the constant survival and the compensatory models in our investigation of the effects of environmental factors and power plant cropping on the adult striped bass population.

5.6.3.5.4 Projected Population Levels

The modified Leslie matrix model was run twelve times: using historical, 1985, and 1990 water management plans, with and without a differential power plant percentage reduction, and assuming both constant survival and compensatory survival from young-of-the-year to three years. The total number of fish three years old and older plus the total number of legal size striped bass was determined for each year under each of the 12 combinations of conditions.

The average number, in millions, of legal size striped bass predicted by the model for each water plan with and without the Pittsburg Power Plant, assuming no compensatory survival from young-of-the-year to three years is

	Historical	1985	1990
With the power plant	2.83	2.00	1.90
Without the power plant	2.97	2.13	2.03

The noncompensatory model predicts that there would have been an average abundance of 2.8 million bass under historic flow conditions, between 1922 and 1971, assuming the power plant was operating. Under the 1985 and 1990 water plans the average historic abundance would decrease 800,000 and 900,000, respectively, to levels of 2.0 and 1.9 million bass. The reduction in outflow associated with the 1985 and 1990 water plans would decrease the adult population size 29 and 33 percent,

respectively, below historic levels of use, assuming that there is no compensatory survival. If the numbers lost at the power plant are added back into the model, the average abundance of striped bass under historic conditions would increase by approximately 140,000 (a 5 percent increase over projected levels with the power plant operating). Under both 1985 and 1990 water plans, the striped bass abundance would increase approximately 125,000 over the projected levels with the plant operating. As discussed in Section 5.6.3.4, the average young-of-the-year percent reduction is probably an overestimate of power plant effects that is due to the observed differences in near-field vs. actual entrainment densities and the inclusion of the unusually high percentage reductions of 1976 and 1977 in the overall average. Under the assumption of noncompensatory survival, the effect of the water management plan is between three and four times as high as the effect of the power plant on the adult striped bass population size. These results imply that other environmental factors are more important in controlling adult striped bass abundance than power plant operation.

The average number of legal-size striped bass, in millions, predicted by the model for each water plan, with and without the power plant, assuming compensatory survival from young-of-the-year to three years, is as follows:

	Historical	1985	1990
With the power plant	1.71	1.68	1.67
Without the power plant	1.69	1.70	1.69

Under the assumption of compensatory survival, the range in average abundance (1.67-1.71 million) under the 6 combinations of water plan and power plant operation is considerably less than the range in abundance under the noncompensatory assumption (1.9-2.97 million). Under historic conditions the predicted average population size is 1.71 million, compared to 1.68 million and 1.67 million under the 1985 and 1990 water management plans, a difference of less than 200,000.

When the effect of power plant operation is removed from the model, the average population size under the historic condition is actually slightly less (1.69 million vs. 1.71 million) than would be expected if the power plant was operating. Under the 1985 and 1990 water plans, the effect of power plant operation on the adult population is negligible, about 20,000 or 1 percent. Under the assumption of compensatory survival, the observed decrease in young-of-the-year abundance due to

lower flows under the 1985 and 1990 water plans and the decrease in young-of-the-year abundance due to the power plant are not translated into significant decreases in the adult population size. We would conclude that under the assumption of compensatory survival from young-of-the-year to three years, there is no significant adverse effect on the adult population size of young-of-the-year losses due to power plant operation or young-of-the-year reduction due to decreased flow rates under the 1985 and 1990 water plans.

Although only one assumption, the existence or nonexistence of compensatory survival from young-of-the-year to three years, separates the two models, the conclusions to be drawn from the two are radically different. One model predicts impacts (a reduction in abundance) of both water plans and the power plant, the other model predicts essentially no impact. The relationship between survival from young-of-the-year to three years is discussed in more detail below.

As stated earlier, the relative variability in young-of-the-year abundance (CV = 48%) is higher than the relative variability in three year old abundance (CV = 18%). The only way in which this difference in variablity could occur is through some differential survival mechanisms from young-of-the-year to three years. The shape of the survival vs. abundance curve could take many forms (linear, logarithmic, or step) depending on the biological mechanisms involved. The survival vs. young-of-the-year abundance curve in Figure 5-29 is one possible representation of the mechanisms, based on existing data. The main interpretational problem with Figure 5-29, however, is that the curve may simply be a reflection of the relationship between a number and its inverse (recall that survival was defined as the product of the number of three year olds and the inverse of the number of young-of-the-year striped bass three years earlier). In 5 out of 10 simulations with 8 random pairs of numbers (X and Y), the correlation coefficient between Y/X and the natural logarithm of X was significantly ($P < 0.05$) different from zero. Although this does not prove that the compensatory survival model developed in Section 5.6.3.5.1 is invalid, it indicates that the form of the differential survival curve from young-of-the-year to three years could possibly take other shapes.

Of the 16 data points that went into the construction of the survival vs. abundance relationship (eight year-class-strength estimates and eight three-year-old estimates), 3 of the data points, circled in Figure 5-29, were estimated from the relationship

between outflow and young-of-the-year abundance developed in Section 5.6.3.2. The first circled data point was estimated from a year (1966) with a moderate outflow that fell within the range of outflows used in the construction of the regression model. In other words, we did not need to extrapolate beyond our data to obtain that estimate of year class strength. The estimated value falls very close to the value predicted by the relationship, i.e., it was not an outlier.

The other two estimated data points are at the lower end of the survival-vs.-abundance curve. These points are the young-of-the-year estimates from the very high outflow years 1967 and 1969. The actual year class strengths cannot be estimated precisely, because the high outflows in those two years (>40,000 cfs) fell outside of the range of outflows used in the construction of the model. We believe that our estimates are close to the true values of year class strength, which would not radically change the survival estimate. We can conclude, then, that the weakness, if there is one, is not in the estimation of young-of-the-year abundance. This leaves the estimate of three-year-old abundance, obtained from CDF&G, to be examined. Since we were unable to obtain the original data that went into constructing the estimates of three-year-old abundance, we have had to rely on CDF&G calculations. The question of compensatory survival from young-of-the-year to three years hinges on the accuracy of the estimates of three-year-old abundance. We would recommend that the methods used to calculate the three-year-old abundance be reviewed and that data collection be continued to obtain additional points to define the survival-vs.-abundance curve.

Through our investigation we have come to this conclusion: the impact of the water management plan and the power plant is probably not as severe as the noncompensatory model predicts, nor are the effects as benign as the compensatory model predicts, but rather the impact falls somewhere in between these two extremes. We further believe that the effect on the young-of-the-year population of reduced flows associated with future water management plans has a greater potential for adversely affecting the adult striped bass population size than does the effect of young-of-the-year cropping by the Pittsburg Power Plant. The absolute magnitude of the impact could not be ascertained, however, due to inconclusive evidence on compensatory survival.

5.7 DISCUSSION

Adverse environmental impacts, as defined by U.S. EPA (1977), occur when the ecological function of the organisms of concern are impaired to a level which (1) precludes maintenance of existing populations, (2) results in a reduction in optimum sustained yield to a sport and/or commercial fishery, (3) involves threatened or endangered species directly or indirectly, or (4) constitutes an unmitigable loss to the aquatic system. Information available for a majority of the species discussed in the previous sections indicates that the numbers of organisms entrained and impinged at the Pittsburg Power Plant will not result in an adverse environmental impact. This is concluded because, for many of the species entrained or impinged in relatively high numbers, their populations are generally considered to be stable or increasing (e.g., threadfin shad, yellowfin goby, Palaemon macrodactylus, Crangon franciscorum).

The striped bass population of the Bay-Delta, however, has experienced a general decline, as evidenced by a decline in year class strength (Figure 5-23) and a declining catch-per-unit-effort in the sport fishery (Figure 5-31). The evidence that recruitment into the sport fishery is a function of the strength of the juvenile year class (Stevens 1980) suggests that the declining year class strength may be contributing to the reduction in the adult population available to the fishery. For this reason, the specific objective of this discussion is to examine the relative contribution of operation of the Pittsburg Power Plant cooling systems to the survival of young striped bass and to the decline in year class strength. Operation of the Pittsburg Power Plant cooling systems could affect the striped bass population either directly, as a result of entrainment or impingement, or indirectly, by affecting the dynamics of the trophic system which supports the striped bass population. A conceptual model was developed to provide a framework for understanding the mechanisms which affect striped bass year class strength and the interactions between striped bass and abiotic and biotic factors which influence their survival and subsequent population abundance. Because of the complexity of interactions between striped bass, other species in the community, and abiotic factors, and because of the dynamic and variable nature of the Bay-Delta, the approach has necessarily been simplified.

In Section 5.6, as in analyses carried out by CDF&G (Chadwick et al. 1977; Stevens 1980), multiple linear regression analyses were used to identify the factors most closely related to the variability in the index of striped bass year class strength.

This approach has several problems. First, any important environmental factor for which the available data base is insufficient cannot be identified and will not be reflected in the results. Second, unless one or two environmental factors have a consistent and dominant effect on the abundance of striped bass, variation in other biotic and abiotic factors within and between years will confound the analysis. In addition to this, multiple regression analyses show only the degree of association between parameters, and do not necessarily reflect a causal mechanism. These problems become most apparent when relationships defined by regression analysis suddenly change, as has happened recently with the Bay-Delta striped bass population.

Nevertheless, any conceptual model of striped bass population regulation must take into consideration the results of previous regression analyses. The results of the statistical analyses presented in Section 5.6 and the information on the trophic structure and life histories of prominent species of fish and invertebrates described in Section 2.2 provide the basis for much of the following discussion. The key question to be addressed is, "What do we know about the biology of striped bass, its ecological interactions, and its involvement with the Pittsburg Power Plant that will put the latter into perspective with those other factors that are contributing to the apparent decline in year class strength?" Because of the complexity and number of factors potentially contributing to striped bass population regulation, it was necessary to examine systematically the relevant information on the direct and indirect effects of cooling system operation on each life stage, along with factors independent of power plant operation, in order to evaluate the potential impact of the Pittsburg Power Plant in perspective with those factors that have a dominant effect on the abundance of striped bass in the system.

A wide range of factors potentially affecting population abundance and regulation of striped bass were considered in the Pittsburg Power Plant impact assessment. A number of factors that would influence the abundance, growth, or survival of each of the major life stages of striped bass were identified (Table 5-32), and available data and information from the literature were compiled to help in identifying and selecting possible factors and mechanisms contributing to the decline in the striped bass population. Factors influencing the adult stock, egg production, survival to the 38-mm stage (production of striped bass 35-39 mm in length serves as the index of year class strength), and survival to three years were considered. Information regarding each of these factors and its potential for population regulation and contribution to the decline in year class strength is summarized below.

TABLE 5-32 FACTORS INFLUENCING THE ABUNDANCE, SURVIVAL, AND GROWTH OF STRIPED BASS IN THE BAY-DELTA

Factors Contributing to Higher Abundance, Survival, and Growth	Life Stage	Factors Contributing to Lower Abundance, Survival, and Growth
Food availability Recruitment	Spawning adults	Fishing Pressure Predation Impingement at power plants Loss at water diversions Parasitism Disease Toxic substances Senescence
Availability of spawning habitat Fecundity	Eggs	Toxic substances Entrainment at power plants Loss at water diversions
Food availability Habitat availability	Young-of-the-year (35-39 mm)	Predation Entrainment at power plants Loss at water diversions Parasitism Disease Toxic substances Genetic fitness
Food availability	Recruits (3-year olds)	Fishing pressure Predation Impingement at power plants Loss at Water Diversions Parasitism Disease Toxic substances

5.7.1 Adult Stock

The adult standing stock in the Bay-Delta, measured by CDF&G using mark-recapture techniques, shows a fluctuation in abundance between 1.5 and 1.9 million bass during the period from 1969 to 1976 (Figure 5-32). The estimated population declined markedly in 1977, and then increased somewhat in 1978 and 1979. Stevens (1977b) describes the methods and biases in the abundance estimates based on mark-recapture data. Although the absolute accuracy of these population estimates is unknown, they are the best estimates available of general population trends.

5.7.1.1 Possible Factors

Food Availability Adult striped bass prey on species such as northern anchovy, threadfin shad, Pacific herring, shrimp (*Crangon* spp. and *Palaemon* sp.), and neomysis (Section 2.2.7.2). Entrainment and impingement cropping of prey species such as Pacific herring and northern anchovy were considered to be small in comparison to the population abundance, and they would not represent a significant depletion in the prey available to the adult striped bass population. Estimates of relative abundance of the forage fish, based on CDF&G midwater trawl survey data for the period 1967-1978, provide no evidence to suggest that the striped bass decline in 1977 could be attributed to reduced food availability.

Recruitment Evidence is presented in Section 5.6.3.2.1 which supports the relationship between production of striped bass greater than approximately 25 mm long and subsequent recruitment of the year class. Therefore the estimated numbers cropped by entrainment or impingement may be reflected in reduced recruitment. Information on the age structure of the Bay-Delta adult population (Table 5-33) indicates that the three-year-old fish are the major portion of the adult standing stock. It is reasonable to assume that a reduction in the adult population in 1977 would have resulted if a high proportion of the larval or juvenile population in 1974 had been lost as a result of operation of the power plant's cooling water systems. Estimated percentage reduction for 1974 was 2.4 percent for the Pittsburg Power Plant (Table 5-29). The estimate for 1974 was the lowest of all those available for the period 1961-1979. In addition to being a year with a low percentage reduction at the plant, 1974 was a year when the year class strength was high (Table 5-17), and would have been expected to result in strong recruitment in 1977 and 1978.

Figure 5-32. Estimated abundances of adult striped bass in the Bay-Delta, 1969-1979. (CDF&G unpublished)

TABLE 5-33 PERCENTAGE OF THE TOTAL ESTIMATED ADULT POPULATION OF STRIPED BASS IN THE BAY-DELTA REPRESENTED BY EACH AGE CLASS, 1969-1976

Year	Age 3	4	5	6	7+
1969	50.2	18.6	12.1	7.7	11.4
1970	54.3	20.6	8.6	5.5	11.0
1971	41.8	30.0	11.1	5.9	11.2
1972	50.0	20.7	16.2	4.9	8.1
1973	37.7	23.7	11.6	8.7	18.4
1974	49.2	17.1	13.8	6.9	13.1
1975	42.1	26.5	10.9	6.6	13.9
1976	45.1	25.0	9.6	6.8	13.4

Source: D. Stevens, CDF&G, personal communication, 1979.

Fishing Pressure Although there has been a general decrease in fishing pressure over time, probably the response of anglers to a generally decreasing catch per unit effort (Figure 5-31), the harvest rate for the population has remained between 15 and 20 percent since 1965. Stevens (1980) concluded that the striped bass harvest by the sport fishery could be increased without significantly increasing total mortality—the natural mortality rate is approximately 30 percent (about twice the fraction currently being harvested). This suggests that the population is being harvested at less than the optimum rate. In addition, it would be reasonable to expect a shift in the age structure of the population if fishing pressure were resulting in a decline in abundance—but no change in age structure is apparent (Table 5-33).

Predation Because of their relatively large size, adult striped bass have few natural predators. There is no information available to suggest that increased predation is responsible for the decrease in bass abundance.

Impingement at Power Plants Observations made at power plants indicate that very few adult striped bass are lost to the population as a result of water use.

Parasitism and Disease High levels of parasitism, lesions, and skeletal deformities were observed in bass collected in the Bay-Delta. Lesions initiated by heavy infestations of cestode larvae (*Lacistorhynchus tenuis*) and secondary bacterial infections were reported to have caused considerable damage to tissues and organs, particularly the gonads and liver (Whipple 1980). Large die-offs of adult striped bass occur each year in the Bay-Delta. The contribution of these die-offs to the declining population, and the role of parasites and disease in the die-offs, possibly interacting with chronic levels of pollutants in the fish, are unknown.

Toxic Substances The presence of chronic levels of heavy metals (e.g., zinc, copper, iron) and of monocyclic aromatic and methyl cyclohexane petroleum hydrocarbons in striped bass from the Bay-Delta has received attention recently (Whipple 1980; SWRCB 1980). In addition, cooperative striped bass studies (Toxic Substances Control Program) are ongoing to examine the accumulation of DDT, other pesticides, and PCB's in striped bass. Preliminary results suggest that the accumulation of toxic substances is particularly detrimental to the liver and gonads. The implications of the accumulation of toxic substances in the gonads on egg production and egg viability are uncertain.

It is encouraging that concentrations of DDT and its metabolites in the flesh of Bay-Delta striped bass are generally decreasing:

	1964	1965	1967	1969	1970	1971	1972	1974	1975	1976
Concentration (ppm)	0.62	0.54	1.62	1.72	1.69	1.45	1.80	0.59	0.21	0.12
Sample size	7	6	2	29	49	18	22	8	13	9

The effects on striped bass of chronic concentrations of toxic substances in the striped bass themselves and in their prey is unknown.

Senescence Striped bass in the Bay-Delta have life spans exceeding seven years, and females generally live longer than males. The natural mortality rate (due to unexplained causes) in the adult population is estimated to be approximately 30 percent (Stevens 1980). The contribution of senescence to this natural mortality rate is unknown, but there is no reason to believe that it is increasing or that this would be a determining factor in the declining abundance of striped bass.

5.7.1.2 Conclusions

Two principal conclusions can be drawn with respect to the population characteristics of adult striped bass. First, there is a long-term trend of declining abundance in the Bay-Delta, as evidenced by declining CPUE's in the sport fishery (Figure 5-31) and population estimates (Figure 5-32). Second, the primary mechanism contributing to the apparent decline in adult bass in 1977 remains unknown. The significant relationship between year class strength and subsequent recruitment into the fishery suggests that a more detailed understanding of the factors contributing to survival of larvae and juveniles is required. Trends in recruitment alone, however, do not provide an explanation of recent population trends. Examination of parasitism, disease, and toxic substances appears promising.

The stock-recruitment analysis presented in Section 5.6.3.3 indicated that there is no obvious dependency between the adult stock of striped bass and reproductive success as measured by recruitment into the sport fishery. An adult population model based on the stock-recruitment analysis, percentage reduction estimates for entrainment cropping, and regression analysis of factors contributing to year class strength (Section 5.6.3.5) indicated that operation of the Pittsburg Power Plant cooling

systems would contribute approximately 8 percent to the decrease in the adult striped bass population, assuming no compensatory survival. This reduction in standing stock would not be compounded from one generation to the next, and therefore, although it would reduce the size of the population in any given year, it would not result in a continuing decline of the population. Since there is no evidence that entrainment or impingement of prey species is having an indirect effect on the availability of food for striped bass, the decline in the population would not appear to be due to a reduced carrying capacity of the system for adults.

5.7.2 Egg Production

A decline in the numbers of adult striped bass spawning would be expected to result in a decline in egg production. Two approaches have been used to determine whether or not egg production in the Bay-Delta is declining. The first consists of direct sampling with plankton nets to capture eggs in the rivers and then estimating egg production in a manner similar to that described for larvae in Section 5.6.3.4. Because striped bass eggs are not distributed uniformly throughout the water column (densities are greater near the bottom), and because they are present at a given sampling station for only a short time, sampling bias is a significant problem in the estimation of egg production. The second approach relies on monitoring the abundance of each age class of adults in the spawning areas and using age-specific fecundity estimates to estimate egg production. Sampling bias is also a problem with this approach. Despite the uncertainty associated with egg production estimates, there appears to be a general decline in the Bay-Delta (Figure 5-33). The factors contributing to the apparent decline in egg production—and, more importantly, the significance of the decline with regard to subsequent year class strengh—are the subject of considerable controversy.

5.7.2.1 Factors in Egg Production

Availability of Spawning Habitat Striped bass spawn primarily in the lower San Joaquin River and adjacent sloughs from Antioch to Venice Island and in the reach of the Sacramento River between Sacramento and Grimes (CDF&G et al. 1973). Although many of the spawning areas in the two rivers have been altered by dredging, channelization, and land reclamation since the early 1900s, there has been no substantial reduction in the availability of spawning habitat since 1960 that could account for the apparent decline in egg production.

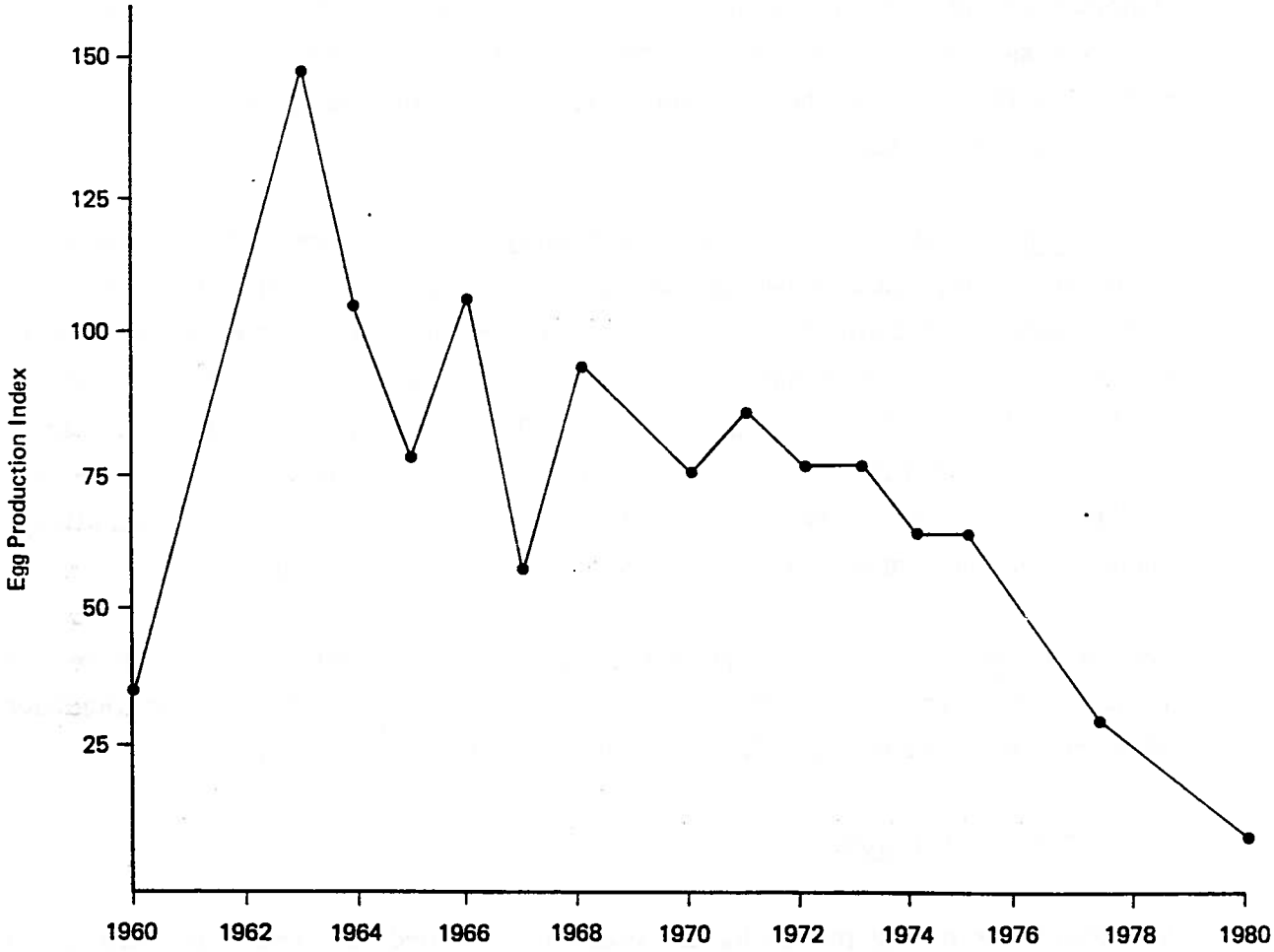


Figure 5-33. An estimate of striped bass egg production in the Bay-Delta, 1960-1980. (From CDF&G unpublished)

Fecundity No information is available to determine whether or not age-specific striped bass fecundity has decreased.

Toxic Substances Preliminary data on the presence of heavy metals and other toxic substances in striped bass gonadal tissues (Whipple 1980) suggests that the reproductive potential of the population could have been adversely affected. The role of toxic substances in determining egg and sperm production and viability is under investigation by the National Marine Fisheries Services as part of the cooperative striped bass study.

Entrainment at Power Plants The susceptibility of striped bass eggs to entrainment is low, primarily because spawning and egg hatching occurs upstream of the Pittsburg Power Plant (EA 1981a). An estimated 1 million striped bass eggs were entrained at the Pittsburg Power Plant during 1978 (Section 3.1). Female striped bass produce an estimated 11,000 to 2 million eggs per year, depending on age, and thus the number of eggs entrained at the Pittsburg Power Plant represents the production of a few females. This level of mortality would not be expected to result in significant reductions of the number of eggs produced in the Bay-Delta system.

Loss at the Water Diversions Striped bass eggs are undoubtedly lost from the system through entrainment at irrigation diversions and at the Delta water export facilities (SWP and CVP, see Section 2.2). The magnitude of these losses is unknown.

5.7.2.2 Conclusions

The exact role of egg production in determining striped bass year class strength is unknown. Chadwick (1974) noted that in years with high spring outflow, most of the eggs and larvae produced in the early and middle portions of the spawning season do not contribute to the larval population and year class strength, because they are transported into San Pablo Bay, where survival is thought to be low. The size distribution in midsummer of the young-of-the-year population indicated that these striped bass were produced from a small, late-spawning portion of the population (Chadwick 1974). A similar conclusion was drawn for the Potomac River population (Ulanowicz and Polgar 1980). In years of high outflow, year class strength is typically higher than average, despite the loss of a substantial proportion of the eggs produced. The ultimate success of spawning in the striped bass population does not appear to depend strongly on the absolute number of eggs spawned.

The stock-recruitment analysis presented in Section 5.6.3.3 led to the conclusion that recruitment into the fishery was largely independent of the size of the adult population (and thus presumably egg production) three years earlier. It was concluded from the stock-recruitment data that large spawning populations, and hence high egg production, do not ensure subsequent large year classes, and that conversely, small spawning populations can give rise to strong year classes. Variations in year class strength appear to be more a function of egg viability and differential larval mortality than of the absolute numbers of eggs produced. This conclusion is supported by the observation that in the Potomac River, the year of greatest recorded spawning (egg production) produced the poorest year class of striped bass observed (Setzler et al. 1980).

Prelimary findings indicate that the accumulation of aromatic hydrocarbons and heavy metals, interacting with parasitism and other environmental factors which result in physiological stress on adult striped bass, may reduce both fecundity and egg viability (Whipple 1980). Genetic differences within the adult population may further contribute to reduced spawning of viable eggs (Whipple 1980), with genetically separate subpopulations which spawn with differential success in wet or dry years. The information available is insufficient to determine the significance of these mechanisms in regulating the striped bass population. It has, however, been reported that in the striped bass population of the Roanoke River, changes in egg viability have paralleled a decline in an index of young-of-the-year abundance. A similar phenomenon may be occurring in the Bay-Delta.

The magnitude of egg loss due to entrainment at agricultural withdrawals and the Delta and Tracy pumping plants is unknown. Entrainment of striped bass eggs at the Pittsburg Power Plant does not appear to represent a major contribution to the overall mortality of striped bass eggs. Although no estimates of total mortality of striped bass eggs are available for the Bay-Delta, mortality in other systems typically exceeds 95 percent (Setzler et al. 1980). The relatively low numbers of eggs entrained, in comparison to fecundity and total egg production and the extremely high total mortality of eggs in the system, suggests that power plant entrainment does not contribute substantially to a reduction in subsequent year class strength.

5.7.3 Young-of-the-year

The index of abundance for striped bass young-of-the-year, those 35-39 mm long, is

shown in Figure 5-34 for the period 1961-1980. The index of abundance illustrates two significant features in the Bay-Delta population: (1) there has been considerable variation in year class strength between years throughout the period for which population surveys have been conducted, and (2) the 1974 year class was relatively strong but was followed by a marked decline in year class strength for all succeeding years. The 1977 year class had the poorest production of juvenile striped bass in the period of record. Furthermore, the 1979 and 1980 year classes were considerably lower than was expected from the relationship between the index of abundance and freshwater outflow during May developed in Section 5.6.3.2.2. For the years before 1976 there was a statistically significant relationship between the index of abundance and outflow rates during the late spring and early summer in our analysis (Section 5.6.3.2.2) and in those of CDF&G (Stevens 1980). In more recent years the abundance-outflow regression models have not provided accurate predictions of year class strength—the predicted year class strength has been consistently higher (by as much as 400 percent) than observed population levels (Figure 5-34). Neither the factors contributing to the marked decline in the index of abundance, nor the reasons that the abundance-outflow relationship no longer holds are currently understood.

The approach we have taken in this discussion is to develop a conceptual model of the relationship between ouflow and year class strength which would be consistent both with results of the earlier regression models and with other information about striped bass in the Bay-Delta. This conceptual model provides a framework for expanding the discussion to include the influence of some of the other factors noted in Table 5-32. We can then examine those factors which may have changed in the mid-1970s and the potential consequences of those changes on striped bass year class strength.

5.7.3.1 A Conceptual Model of the Relationship Between Outflow and Striped Bass Year Class Strength

A number of mechanisms have been hypothesized as the basis of the relationship between spring or summer outflow rates and year class strength as measured through the index of abundance (Turner and Chadwick 1972; CDF&G 1976). After a detailed examination of the available data on the distribution of larval and juvenile bass within the Bay-Delta, size-specific mortality rates, growth rates, water quality characteristics, and the physiological responses and requirements of young-of-the-year striped bass, we have developed the conceptual model of the relationship between outflow and striped bass year class strength shown in Figure

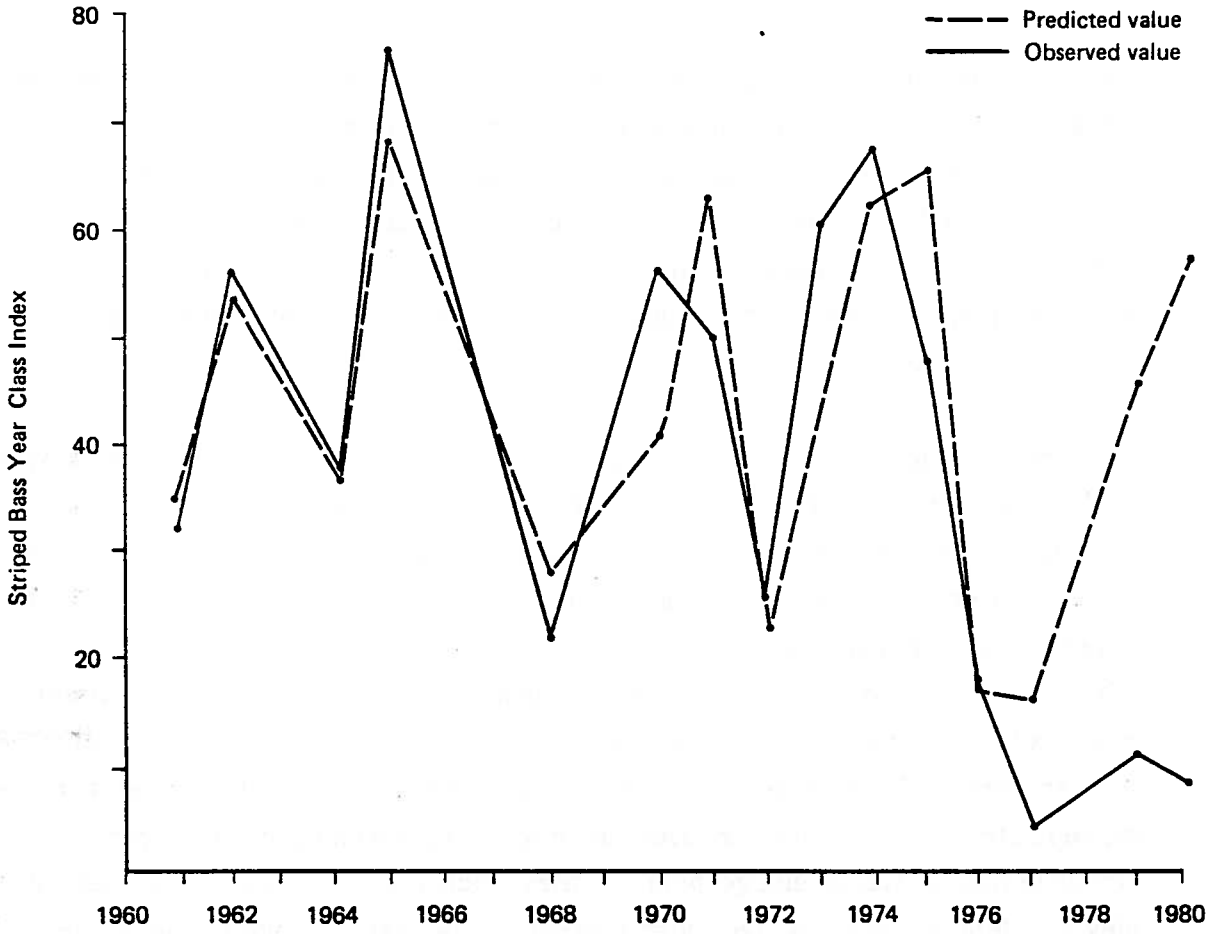


Figure 5-34. Observed index of striped bass year class strength and the index predicted by a function of Delta outflow in May, 1961-1980.

5-35. The model includes consideration of the relationship between outflow and larval striped bass metabolic requirements for food that are modulated by water temperature. The model also includes consideration of the age-specific distribution of the larvae in the Bay-Delta with respect to availability of food resources in sufficient density to meet their metabolic requirements for growth and survival. The evidence and logic used in developing the model are summarized below.

The magnitude of freshwater flows passing through the Bay-Delta system during the spring and early summer has a considerable influence on water temperatures in the system. An analysis of surface water temperature in Suisun Bay during May and freshwater outflow during May indicated a statistically significant ($P = 0.03$) inverse relationship between water temperature and the logarithm of outflow. A similar relationship was found in April, June, and July. Therefore, at higher outflow rates, water temperatures are lower.

Water temperatures recorded in April, May, and June in Suisun Bay (CDF&G Station 54) between 1969 and 1979 are shown in Figure 5-36. Water temperature on any given date varied by approximately 10 F (5.5 C) in the course of the 11-year monitoring period. As noted earlier, a primary factor contributing to the variability in water temperature was the rate of outflow. A secondary factor is the Pittsburg Power Plant thermal discharge. The thermal discharge from the plant was estimated to potentially increase water temperatures from 0.5 to 1 F over an area of several thousand acres (Tetra Tech 1976). The thermal plume (within the 4 F isotherm) was concentrated near the surface over an area of approximately 100 acres. Vertical stratification of the discharge plume (Tetra Tech 1976) would serve to restrict the elevated temperatures to the upper portion of the water column. Larval and early juvenile striped bass are distributed throughout the water column (EA 1981a, b, e), considerably reducing their exposure to elevated water temperature associated with the thermal discharge.

The metabolic rate of juvenile striped bass weighing 1-3 g is temperature-dependent, with energy consumption increasing as temperature increases (Klyashtorin and Yarzhombek 1975). Kruger and Brocksen (1978) reported a similar temperature dependence for both the standard metabolic rate and the "scope for activity" (the difference between the standard metabolic rate and the active metabolic rate) for juvenile striped bass 10-17.5 cm long. The temperature dependency in metabolic rate observed for striped bass is consistent with that of other fish species (Webb 1975).

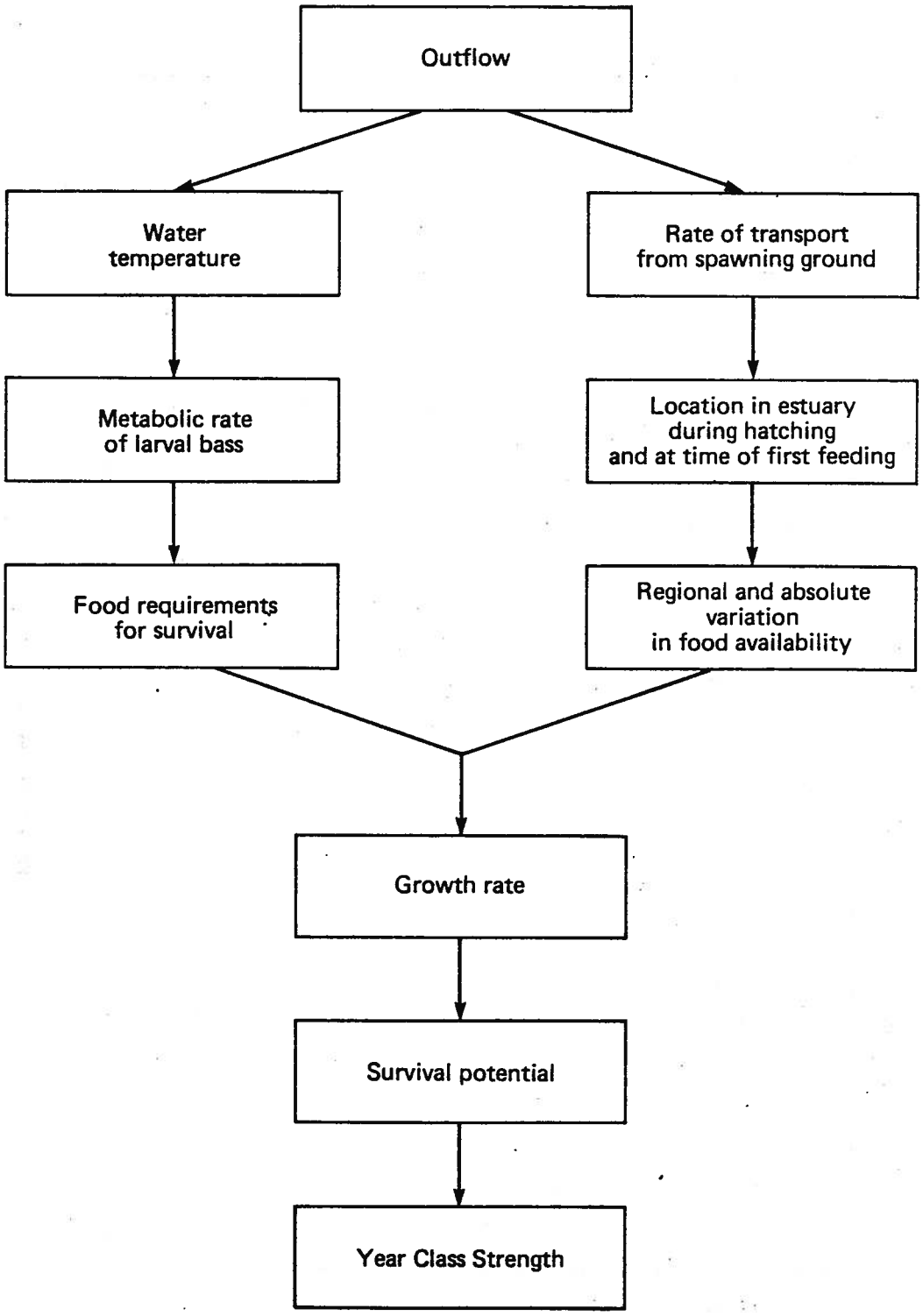


Figure 5-35. A model of factors in the relationship between Delta outflow and striped bass year class strength in the Bay-Delta.

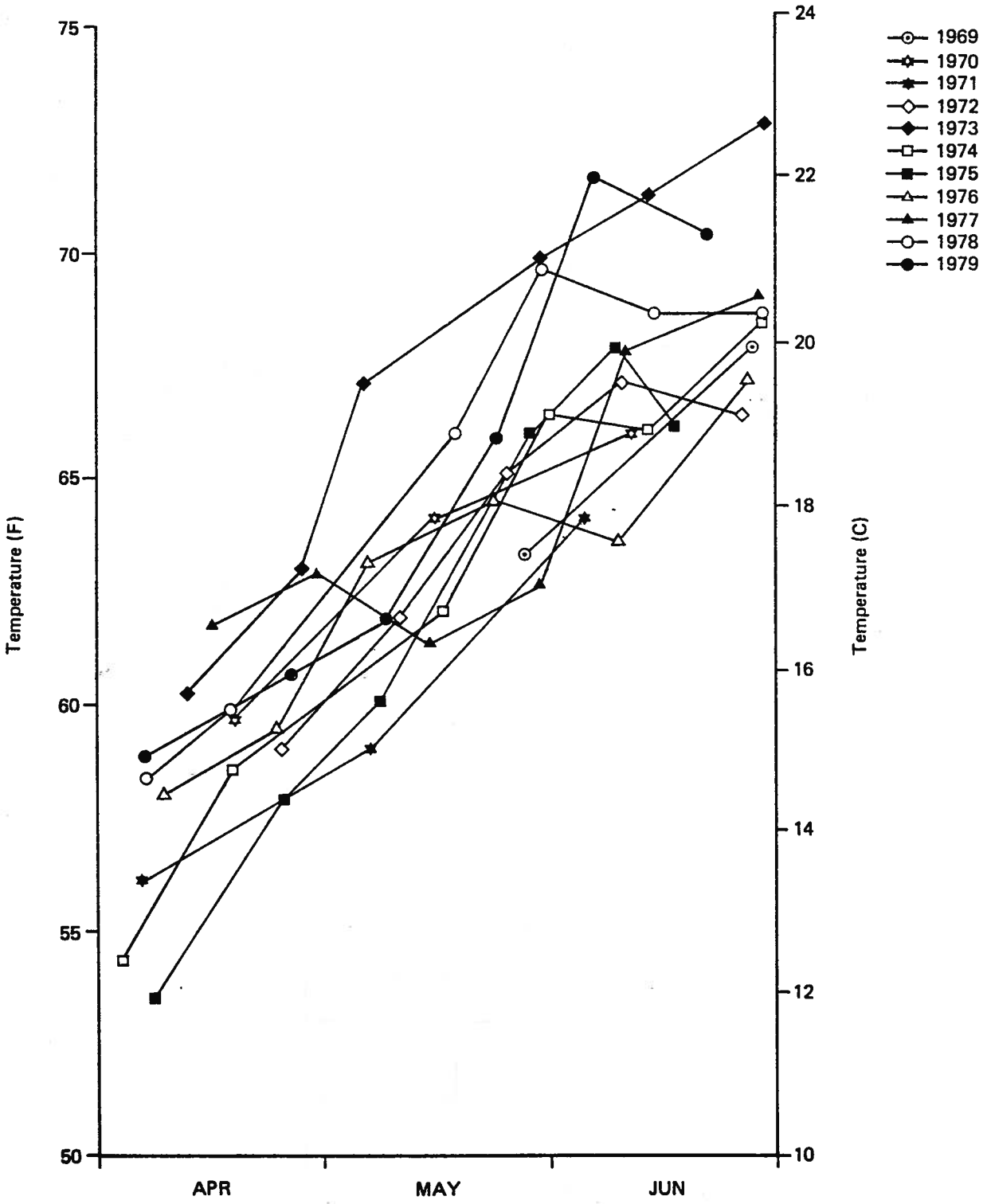


Figure 5-36. Water temperatures measured on sampling days in April, May, and June by the California Department of Fish and Game in Suisun Bay (Station 54), 1969-1979.

Although no metabolic studies are available for larval striped bass (6-10 mm long) there is a sufficient physiological basis to assume that metabolic rate, and hence the energy requirements of larval bass, would increase with increasing water temperature.

The food requirement for a growing larval striped bass may be expressed as

$$\frac{dw}{dt} = R - Q \qquad (5\text{-}44)$$

which states that growth (dw/dt) is a function of the assimilated ration in the diet (R) and the total metabolic rate (Q). For the larval fish to simply survive (but not grow), the assimilated ration in the diet must equal the total metabolic rate. Since the metabolic rate increases with increasing water temperature (and decreasing outflow), the number of prey items consumed and assimilated must also increase to sustain the fish.

The total metabolic costs (Q) can be expressed as

$$Q = Q_s + Q_a + Q_f + Q_c + \ldots + Q_x \qquad (5\text{-}45)$$

where

Q_s = standard metabolic rate
Q_a = energy cost of spontaneous activity
Q_f = energy cost of foraging activity
Q_c = energy cost to utilize food (SDE)
Q_x = other (minor) energy costs.

At the time of hatching, striped bass larvae obtain nourishment from a yolk sac which contains a large oil globule. The rate of absorption of the yolk sac is temperature-dependent, and a temperature difference of 1-3 C (3-5 F) has a significant effect on the rate of absorption (Figure 5-37). After the yolk sac has been absorbed, the larvae must start feeding on zooplankton at a rate sufficient to meet their metabolic demand or they will die. Therefore, food availability during the initiation of feeding is a critical factor in the survival of larval striped bass.

The relationship expressed in Equation 5-45 implies not only that the metabolic costs are temperature-dependent but also that they increase as a function of the increasing cost of foraging activity, which is dependent on the density of acceptable prey

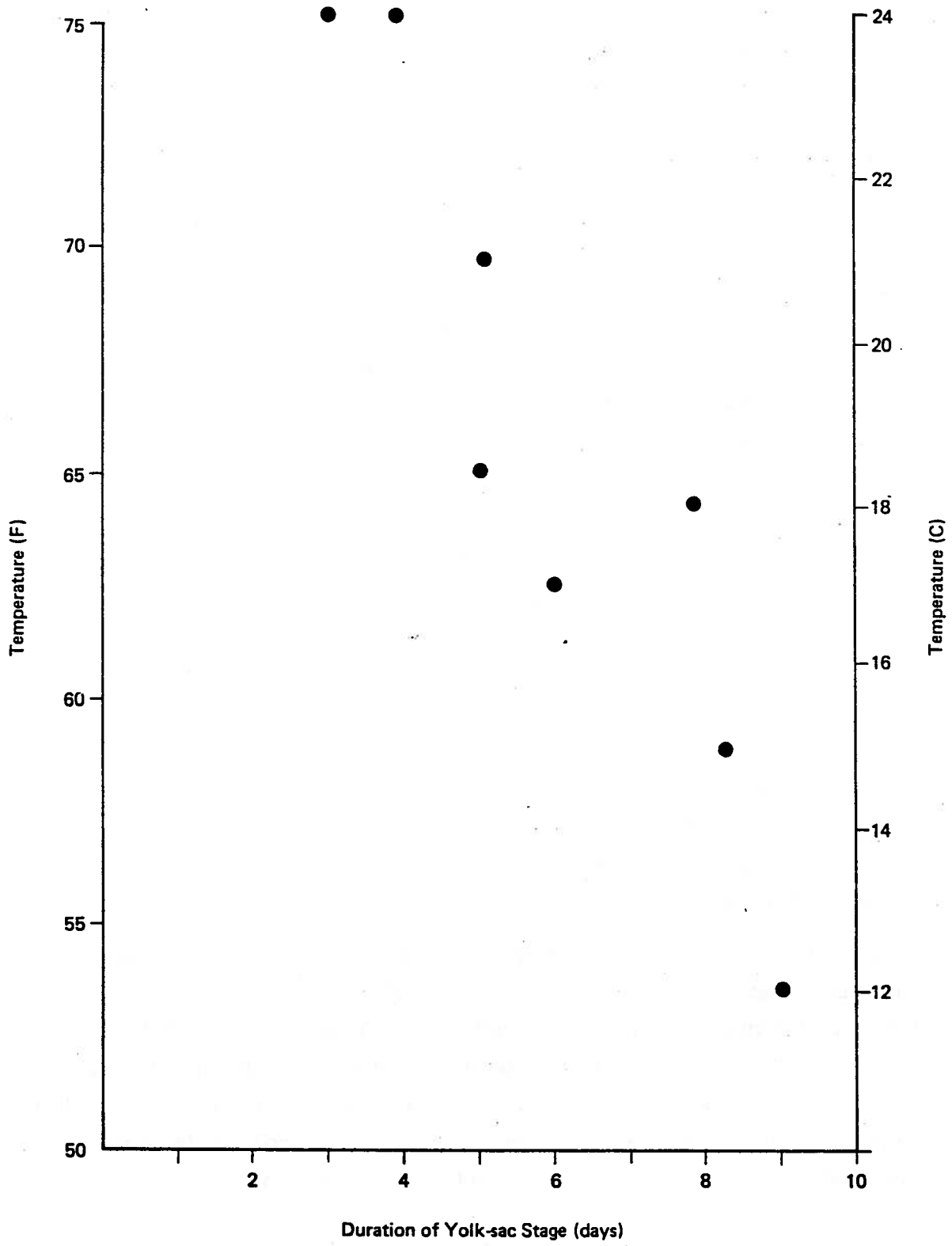


Figure 5-37. Duration of yolk-sac stage (period of absorption of the yolk-sac) in striped bass at various water temperatures. (Data from Setzler et al. 1980.)

(zooplankton) in the immediate vicinity of the larva.

Lasker (1975) identified four factors important for successful feeding and subsequent growth and survival of larval northern anchovy which are equally appropriate for larval striped bass, since both species rely on zooplankton as a food source:

1. particle size at first feeding,
2. the nutritive value of food organisms,
3. the number of particles per unit volume in the larva's environment,
4. (related to 3) the frequency of feeding strikes, and the consequent degree of success in capturing food.

Studies of prey size preferences and prey concentrations required to initiate feeding and sustain larval striped bass indicate that small zooplankton (e.g., nauplii and copepodite stages of copepods and cladocerans) in a size range from 0.15 to 0.5 mm are the primary prey of early larval striped bass (Doroshev 1970; CDF&G unpublished data). P. Miller (1977), examining the feeding behavior and requirements of larval striped bass, estimated that a minimum prey concentration of 1,800 Artemia spp. nauplii per liter was required to establish first-feeding in the laboratory. CDF&G et al. (1973) conducted laboratory studies on the effects of food concentration on the survival of larval striped bass 5-10 mm long and concluded that survival was reduced significantly when the prey concentration was less than 7.5 small zooplankton per liter. Doroshev (1970) utilized an initial prey concentration of 15,000 Cyclops nauplii and copepodites per liter in the successful initiation of first feeding of striped bass 9 days after hatching. Eldridge et al. (1977) reported that survival of larval striped bass was directly proportional to prey densities in laboratory studies. The relationship between percent survival of larval striped bass and the density of zooplankton nauplii (no./liter) is shown below:

Survival (%)	28	52	62
Density	2,000	4,000	6,200

Eldridge (cited in Whipple 1980) estimated the energetic demands of larval striped bass and concluded that average zooplankton densities in the nursery areas are inadequate to meet the metobolic demands of larvae. On the basis of a similar conclusion from laboratory feeding studies, P. Miller (1977) speculated on the importance of the patchy distribution of zooplankton in striped bass nursery areas:

those larvae that by chance locate and remain in patches of suitable-sized zooplankton in densities in excess of 1,000/liter are able to feed at a rate sufficient to meet metabolic demands for growth and survival.

First-feeding, the critical transition from an endogenous food source (the oil globule) to an exogenous one, may represent the maximum metabolic demand on the larvae and require maximum prey concentrations, since prey capture efficiency increases with increasing larval length and swimming performance.

The metabolic demand, and hence the food requirement, of larval striped bass is high, and varies in part as a function of water temperature, which in turn is influenced directly by outflow rates. The survival and growth of larval bass depends in large part on their encountering prey organisms of appropriate size in sufficiently high density. As outflow during spring and early summer decreases and water temperature increases, the nutritional demands to meet metabolic requirements may be exceeded for a substantial portion of the population which encounters inadequate prey densities, resulting in high mortality. In years of high outflow, water temperatures are typically lower, reducing larval metabolic demands and food requirements. Under these conditions, lower zooplankton densities would be required to support the larval population, and increased survival and consequently a stronger year class would result.

Outflow is important not only in its indirect influence on metabolic demands and food requirements of larval striped bass but also in its effect on the distribution of larval bass in the Bay-Delta, which has an impact on their available food supply. Outflow rates directly affect current velocities in the Sacramento and San Joaquin rivers and thus the rate of downstream transport of eggs from the spawning grounds. Furthermore, the rate of egg development varies directly (and thus the duration before hatching inversely) with water temperature, which is influenced by outflow: the time required for hatching varies from 29 hours at 72 F to 80 hours at 52 F (Setzler et al. 1980). The location of the larvae in the estuary at the time of hatching and the time of first feeding depend on the spawning location, the rate of downstream transport, and the water temperature as it affects development time.

Zooplankton that would provide an acceptable food source for larval striped bass are not distributed uniformly within the Bay-Delta, and thus some areas have a substantially higher probability of successfully supporting the nutritional demands of larval bass than others. Suisun Bay and the lower Sacramento and San Joaquin rivers,

for example, typically have high primary productivity and higher zooplankton densities, capable of supporting larval striped bass, than do the interior Delta channels, the middle and upper reaches of the Sacramento River, and San Pablo Bay (Arthur and Ball 1979, EA 1981a).

In years with low spring outflows and the associated warmer water temperatures, the rate of downstream transport would be slower and rate of development higher. This would result in many of the eggs hatching in upstream areas that have zooplankton densities too low to meet the higher metabolic demands of the larvae. Without a sufficient food supply the population would have high mortality rates and a reduced year class strength. In years when higher spring outflow rates result in lower water temperatures and higher rates of downstream transport, the eggs hatch and the larvae are situated in areas of higher zooplankton densities, and larval survival is high, resulting in a stronger year class.

The relationships between zooplankton densities, growth rates, and estimates of striped bass production were investigated using available survey data from the period 1961-1979. The number of observations is insufficient to permit solid conclusions, but the information suggests certain basic trends between striped bass growth and survival and zooplankton prey densities. There is a positive relationship ($P = 0.08$) between larval striped bass growth rates in the Bay-Delta and densities of copepodite zooplankton in the region of highest larval bass densities, and a similar positive relationship ($P = 0.10$) between larval growth rates and densities of adult copepods in the region of highest bass densities. Furthermore, estimates of striped bass production for 30-34 mm and 35-39 mm size classes showed significant positive correlations with an index of neomysis abundance ($P = 0.04$, $P = 0.03$). These data provide further support of the relationship between growth and production of larval and juvenile striped bass and the availability of a prey resource: increased prey availability was reflected in increased growth and production, which contribute to increased year class strength.

The mechanism postulated for the relationship between outflow in spring and early summer and striped bass year class strength, is one in which outflow acts on water temperature to influence the metabolic rate and food requirements for survival and growth of larval and juvenile striped bass, and also influences their location in the estuary with respect to the availability of an adequate prey resource when the eggs hatch and the larvae must begin foraging to survive. Using this conceptual model of

the mechanism for the relationship between outflow and striped bass year class strength as a focal point, we can examine the contribution of other factors listed in Table 5-32 to the regulation of the striped bass population. Possible changes within the Bay-Delta system which occurred during the mid-1970s and which may have contributed to the recent decline in year class strength will be discussed in the next section.

5.7.3.2 Factors Affecting Abundance and Survival

A number of factors, acting individually and in combination (Table 5-32), affect the abundance, survival, and growth of young-of-the-year striped bass in the Bay-Delta. The conceptual model developed around the relationship between outflow and striped bass year class strength has been expanded (Figure 5-38) to include additional factors and interactions which might contribute to year class strength through their direct or indirect effect on larval striped bass survival rates. A significant change in the balance of these interactions would be likely to result in a change in the previous relationship established between outflow and year class strength.

Food availability Food availability, which includes prey size, type of prey and its nutritional value, and prey density, is a critical factor in the survival of larval striped bass. As noted in the previous section, the balance between metabolic demands and the consequent food requirements of larval bass and the availability of prey to meet these demands has a direct affect on larval survival and year class strength. Food availability is a primary factor regulating the carrying capacity of the Bay-Delta for larval and juvenile striped bass. A reduction in food availability during the early life stages of striped bass will result in a reduction in carrying capacity and consequently a reduction in year class strength.

The production of herbivorous and omnivorous zooplankton (e.g., copepods and cladocerans, particularly nauplii and copepodites), which serve as the primary food resource during the period when striped bass larvae are switching from an endogenous to an exogenous nutrient source, is supported to a large degree by the primary trophic level. The primary zooplankton prey species of interest in the striped bass food web forage on heterotrophic bacteria, diatoms, and small algae. CDF&G et al. (1975) reported that the densities of rotifers and crustaceans (largely cladocerans and copepods) had high positive correlations with concentrations of chlorophyll a, which is

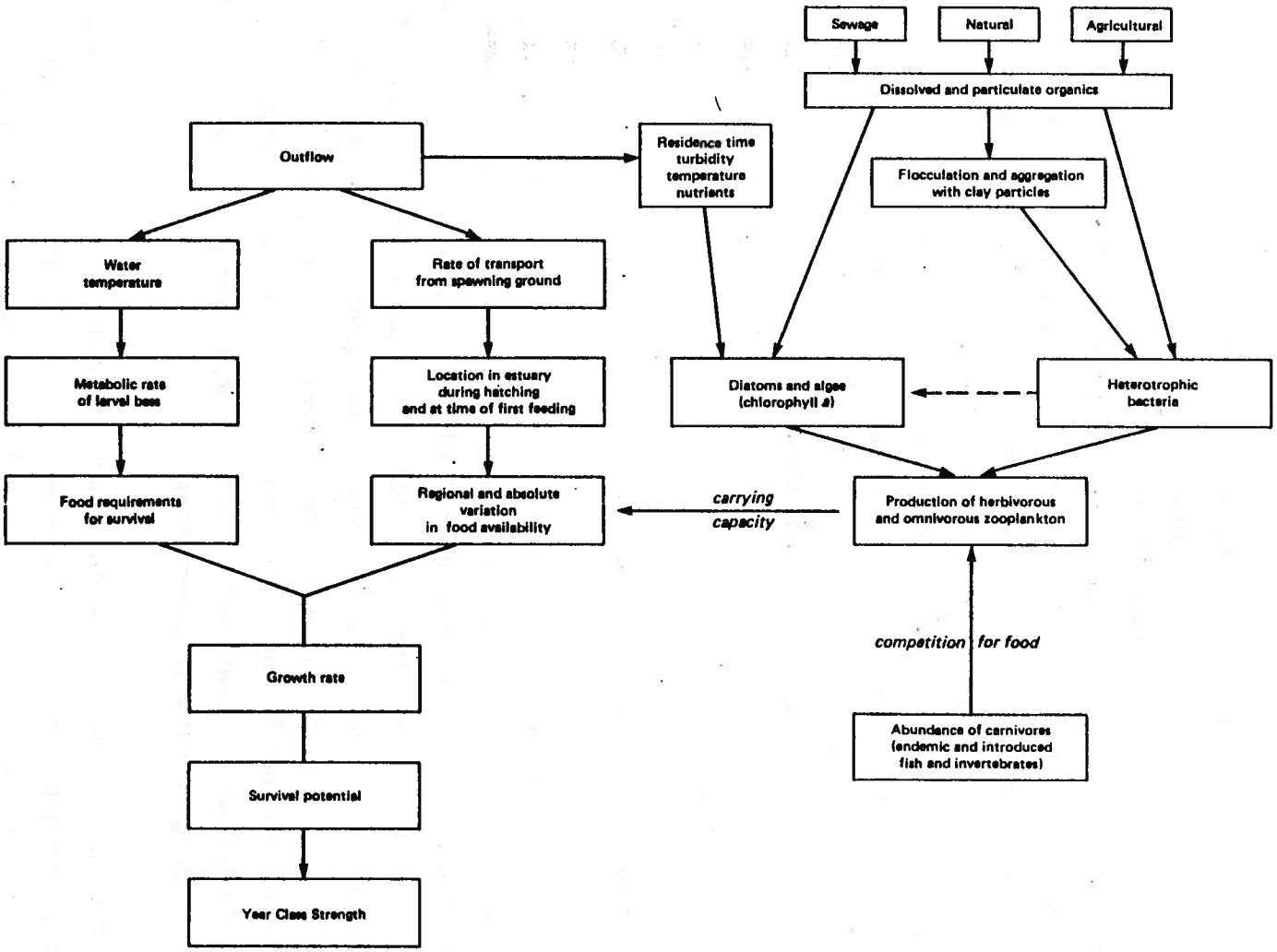


Figure 5-38. A model of the relationship between Delta outflow, the carrying capacity of the system for striped bass, and striped bass year class strength in the Bay-Delta.

an indicator of phytoplankton abundance. Since these zooplankton are known to feed on phytoplankton, the correlation may reflect the effect of variations in phytoplankton availability on zooplankton production.

Zooplankton densities increased as phytoplankton densities (chlorophyll a) increased. It is interesting to note that chlorophyll a concentrations measured in western Suisun Bay (Station D6) showed a general decline from 1969 to 1977, with a slight increase in 1978 and 1979 (Figure 5-39). It can be speculated, on the basis of these data, that zooplankton densities may have declined with declining phytoplankton abundance, resulting in a reduced carrying capacity for larval and juvenile striped bass. For most species of microzooplankton, no quantitative population estimates have been developed in the Bay-Delta that could be used to address the question of declining carrying capacity. However, an index of abundance for neomysis for the period 1968-1979 (Figure 5-40) although variable, suggests that the neomysis population is declining. Neomysis, like many other zooplankton in the Bay-Delta, feed on diatoms and on detritus and its associated bacteria (Section 2.2). These observations provide further support for the speculation that the reduction observed in striped bass year class strength reflects a general reduction in carrying capacity as a result of changes in trophic dynamics within the Bay-Delta ecosystem.

One possible mechanism which would contribute to a reduction in primary productivity of the system would be a reduction in inorganic nutrient loading. To examine this possible mechanism, nitrate and nitrite concentrations were compiled for the months March, April, and May during the period 1970-1978 for the Sacramento River (Station D24), the San Joaquin River (Station P12), and western Suisun Bay (Station D6). There was no apparent decreasing trend in nitrate and nitrite concentrations for any of the months at any of the sampling stations. It appears from these data that the decline in primary productivity is not in response to decreasing inorganic nutrient concentrations.

Another possible mechanism which would contribute to a reduction in primary production (particularly diatoms), and also to a reduction in heterotrophic bacteria, would be a reduction in dissolved and particulate organics coming into the Bay-Delta system. A reduction in the loading of organic material to the system would result in a reduction in energy and material resources (substrate) and a consequent reduction in the production of heterotrophic bacteria. These heterotrophic bacteria are essential cyclers of organic substrates, nitrogen, and phosphorus (Goldman 1974). The

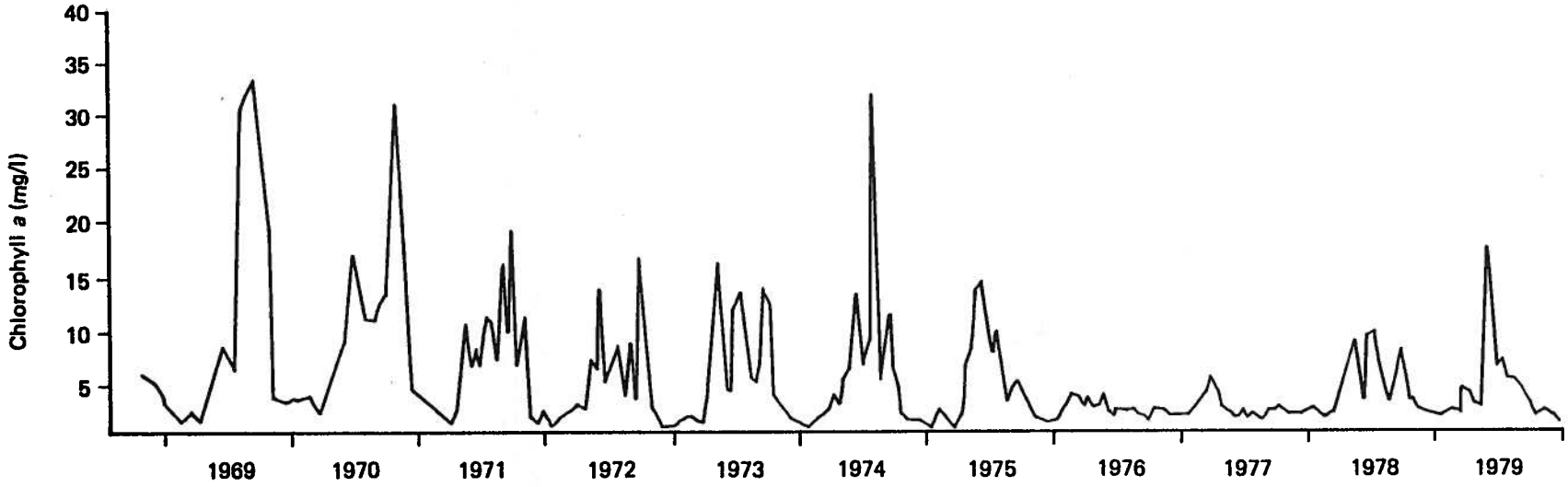


Figure 5-39. Chlorophyll *a* concentrations measured in western Suisun Bay (Station D6) by the California Department of Water Resources, 1968-1979.

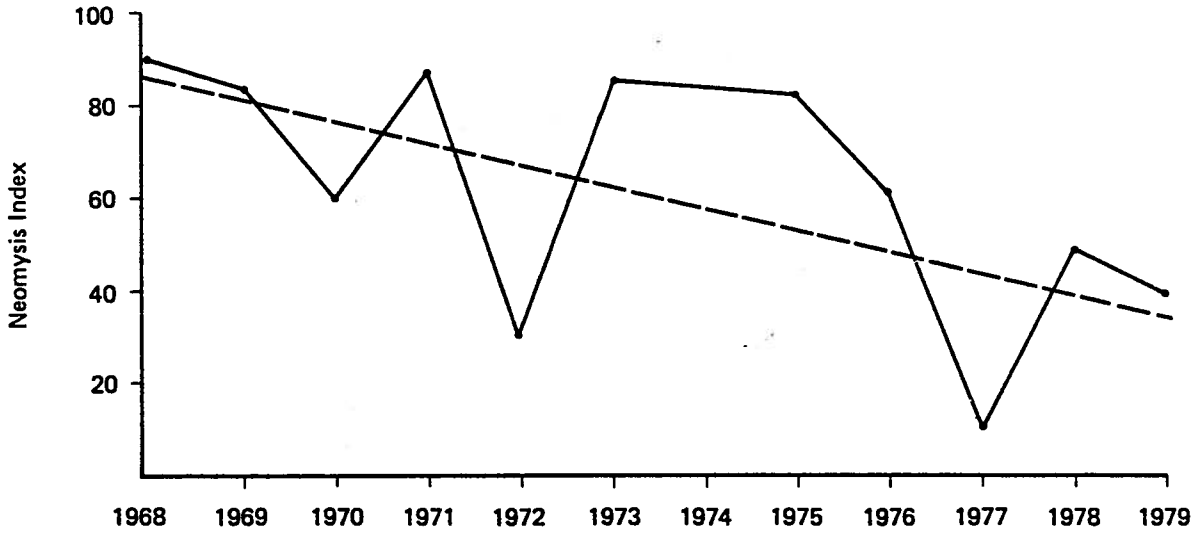


Figure 5-40. An index of the abundance of Neomysis mercedis in the 1.5 months preceding the time when the mean length of striped bass young is 38mm, 1968-1979.

bacteria, in turn, through death and autolysis, release nutrients that support primary production. Knight et al. (1980) reported a high positive correlation between the biomass of heterotrophic bacteria and phytoplankton biomass in the Bay-Delta. Heterotrophic bacteria are also a major food source, used directly by many omnivorous filter-feeding microzooplankton. A reduction in dissolved and particulate organic matter would be expected to result in a reduction in the biomass of heterotrophic bacteria, and through that a reduction in microzooplankton production and hence carrying capacity for larval and juvenile striped bass. A reduction in organic input may also result in a reduction in primary productivity, either directly through a reduction in nitrates, phosphates, and other inorganic plant nutrients made available through the decomposition of the organic matter itself or indirectly through microbial pathways.

Dissolved and particulate organics enter the Bay-Delta as allochthonous detrital material, agricultural waste, and domestic sewage. There is no clear evidence that organic loading from either natural detritus or agricultural material has changed substantially during the past decade. One of the major sources of organic material entering the Bay-Delta, however, is the discharge of domestic sewage. In the first half of the 1970s a major reduction in the organic load entering the system as sewage occurred with the implementation of secondary sewage treatment. The biochemical oxygen demand (BOD) of sewage effluent is a direct indicator of the dissolved and particulate organic concentration. As an example, the BOD, expressed as thousands of pounds per day, in the effluent entering the Sacramento River from the Sacramento Main Sewage Treatment Plant is shown in Figure 5-41 for the period 1970-1979. Typical organic loading (BOD) in spring 1970 and 1971 ranged from 64,000 to 88,000 lb/day. After initiation of secondary sewage treatment in 1971, organic loading decreased to about 20,000 lb/day. Similar reductions in organic loading to the Bay-Delta were achieved in the first half of the decade through the installation of secondary sewage effluent treatment at all major facilities discharging into the system. This substantial reduction in energy input available to support microbial decomposers and primary production would contribute to a marked reduction in production of microzooplankton and would be reflected in reduced carrying capacity and year class strength of striped bass. A sudden and substantial reduction in carrying capacity of this type would result in a striped bass year class much smaller than that predicted on the basis of the outflow - year class strength relationship that was developed from data collected when carrying capacity was substantially higher.

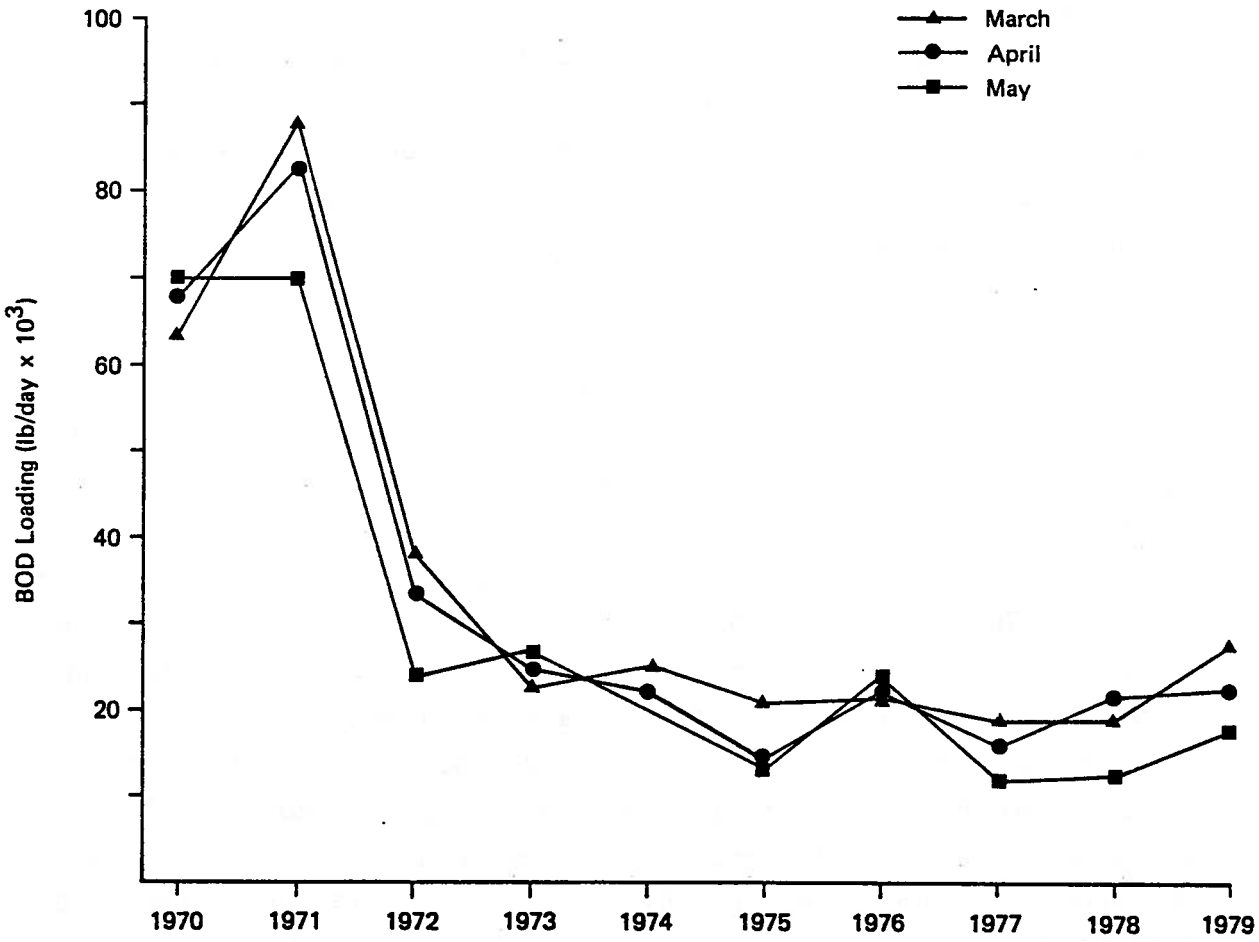


Figure 5-41. Monthly mean biochemical oxygen demand loading of the Sacramento Main Sewage Treatment Plant in March, April, and May, 1970-1979.

Competition Another major factor which would influence food availability for larval striped bass is interspecific competition for zooplankton. Interspecific competition for food would only be an important factor if food availability were a limiting factor in the determination of larval survival. Two major changes have occurred recently in the Bay-Delta that would support the hypothesis that food availability is now limiting striped bass year class strength: the reduction in organic material entering the system as sewage effluent, resulting in reduced zooplankton production, and a rapid increase in abundance of at least four species recently introduced into the Bay-Delta which also forage on zooplankton and therefore compete with larval and juvenile striped bass. Although there is little information on population abundance available for recently introduced species, the rapid growth of their populations is apparent in the relationship between the dates of their first appearance in the system (Moyle 1976b; Newman 1963) and their percentages of total biomass in impingement samples (Chapter 4) from the Pittsburg Power Plant:

	First Appearance	Percentage (Biomass)
Threadfin shad	1953	22.5
Yellowfin goby	1963	11.6
Inland silverside	1967	<1
Palaemon macrodactylus	1954	61.5 (invertebrates)

Moyle (1976b) discusses briefly the impact of competition on fish populations resulting from the introduction and rapid population expansion of a species. Ecosystem alteration by introduced fish species such as shad, which can drastically alter the composition and characteristics of the zooplankton community (Brooks and Dodson 1965), has been documented. Alteration of the zooplankton communities in reservoirs as a result of the introduction of threadfin shad has often reduced the growth and survival of other fish species which forage on zooplankton (von Geldern and Mitchill 1975, cited in Moyle 1976b). Alterations in the zooplankton community of Clear Lake caused by the recent introduction of inland silverside may have had an adverse impact on native cyprinids which also depend on zooplankton (Moyle 1976b). There are no data on changes in the zooplankton community caused by introduced species in the Bay-Delta, but it is likely that an alteration in the zooplankton community (either a reduction in density or a shift in species composition) could have an adverse affect on striped bass carrying capacity.

Catch data for threadfin shad, in the form of CPUE's for the CDF&G midwater trawl surveys from 1967 to 1978, can be used to examine potential competitive interactions

with striped bass. Threadfin shad forage on zooplankton during all life stages, and they have been reported in greatest abundance in those areas of the Bay-Delta where crustacean zooplankton densities are highest (Turner 1966). Threadfin shad are extremely effective foragers, with no apparent prey size selection. The index of neomysis abundance will be used as an indicator of competition, since both striped bass and threadfin shad feed on neomysis, and since it is the only zooplankton species in the Bay-Delta for which population trends have been quantified (Section 5.5).

As noted in Section 5.5, there is a significant ($P = 0.03$) positive relationship between the index of neomysis abundance and the index of striped bass year class strength measured as production of 35-39 mm bass. Although neomysis abundance as a food source may not be limiting for striped bass abundance in all years, neomysis are the principal component in the diet of juvenile striped bass (Figure 5-19), and the positive relationship between the two indices of abundance suggests that a reduction in neomysis abundance may result in a reduction in juvenile striped bass abundance and hence year class strength.

Threadfin shad are known to forage on neomysis in the Bay-Delta to support their large population biomass, and they may be competing with juvenile (and presumably larval) striped bass for zooplankton. If foraging by threadfin shad were reducing the abundance of neomysis, a negative relationship would be expected between the index of neomysis abundance and the abundance of threadfin shad in Suisun Bay and the lower Delta, as reflected in their CPUE's in the CDF&G midwater trawl surveys.

Two separate trends appear in the association between neomysis and threadfin shad. The first trend (A, Figure 5-42) indicates neomysis abundance was high and had no apparent association with catches of threadfin shad: there was no reduction in the index of neomysis abundance attributable to foraging by threadfin shad. It should be noted that data represented in this trend (1968, 1969, 1971) reflect conditions prior to the reduction in organic loading resulting from secondary sewage treatment. One speculation that can be made is that during the period of high organic loading the neomysis population was highly productive as a result of an abundant food supply (bacteria, organic detritus, and algae) which contributed to high birth rates and low death rates (Section 5.5). During this period of high productivity, predation by fish, for example striped bass or threadfin shad, would have little or no effect on the population abundance of neomysis. The high birth rate and short generation time of

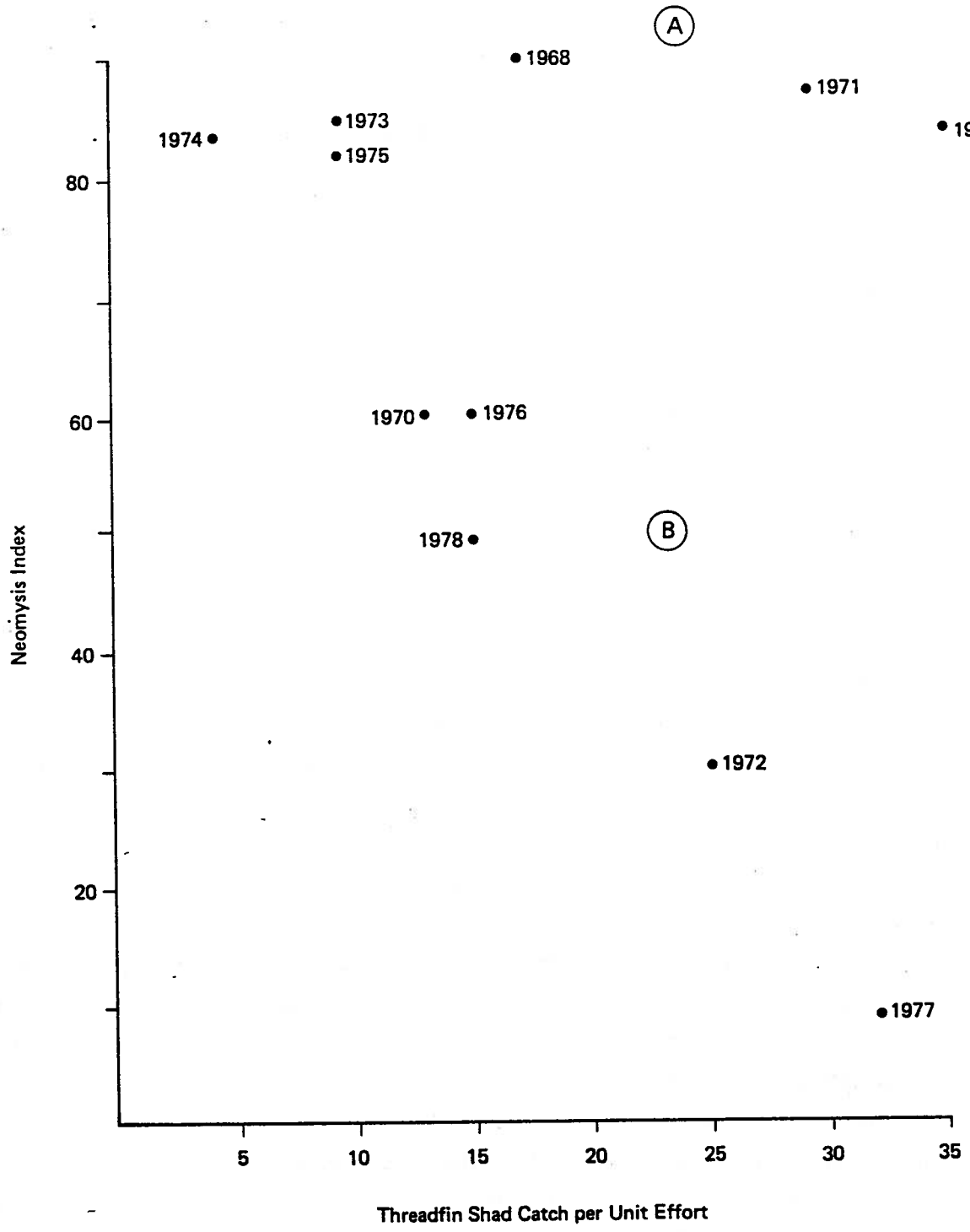


Figure 5-42. Abundance of Neomysis mercedis (index of neomysis abundance) vs. abundance of threadfin shad (CPUE, CDF&G midwater trawl), in the Bay-Delta.

neomysis would compensate rapidly in the presence of an abundant food supply for losses resulting from predation.

The second trend (B, Figure 5-42) reflects a negative association between the index of neomysis abundance and threadfin shad catches: there is a reduction in the index of neomysis abundance which may be attributable to foraging by threadfin shad. The data represented in this trend reflect conditions in the mid- to late-1970s, when the system's carrying capacity for neomysis may have been reduced to a level where the neomysis population was unable to compensate for high predation losses. During this period, foraging by threadfin shad would have a direct negative effect on the availability of neomysis (and possibly most other zooplankton species) as a food resource for striped bass. Further evidence suggesting that competition from threadfin shad is having a negative effect on striped bass year class strength, a negative relationship between the index of striped bass year class strength and threadfin shad catches, is shown in Figure 5-43. Presumably, when threadfin shad are abundant in the Bay-Delta they effectively reduce the abundance of zooplankton such as neomysis with a consequent reduction in the carrying capacity and year class strength of striped bass.

An interesting note on the potential competitive interaction between striped bass and threadfin shad is that, in addition to spring outflow rates, colder than normal winter water temperatures have been asssociated with strong striped bass year class strength. Findings summarized by Setzler et al. (1980) for data developed from East-Coast systems (Merriman 1941; Koo 1970; Heinle et al. 1976; Boynton et al. 1977) indicate that strong year classes of striped bass were associated with lower-than-normal winter temperatures. A mechanism that could be hypothesized for this relationship is that in the systems examined (e.g., Chesapeake Bay, the Potomac River), shad are present in large numbers and are competing for a limited zooplankton resource with larval and juvenile striped bass. Threadfin shad, however, are sensitive to low water temperatures and experience high die-offs as a result of cold winter water temperatures (Chapter 4; Turner 1966). The association between strong striped bass year classes and unusually cold winter water temperatures may be due to a large die-off of shad during the winter, resulting in lower competitive pressure for zooplankton in the spring.

Other recently introduced species which prey on zooplankton and thus may also compete with larval and juvenile striped bass include yellowfin goby, inland

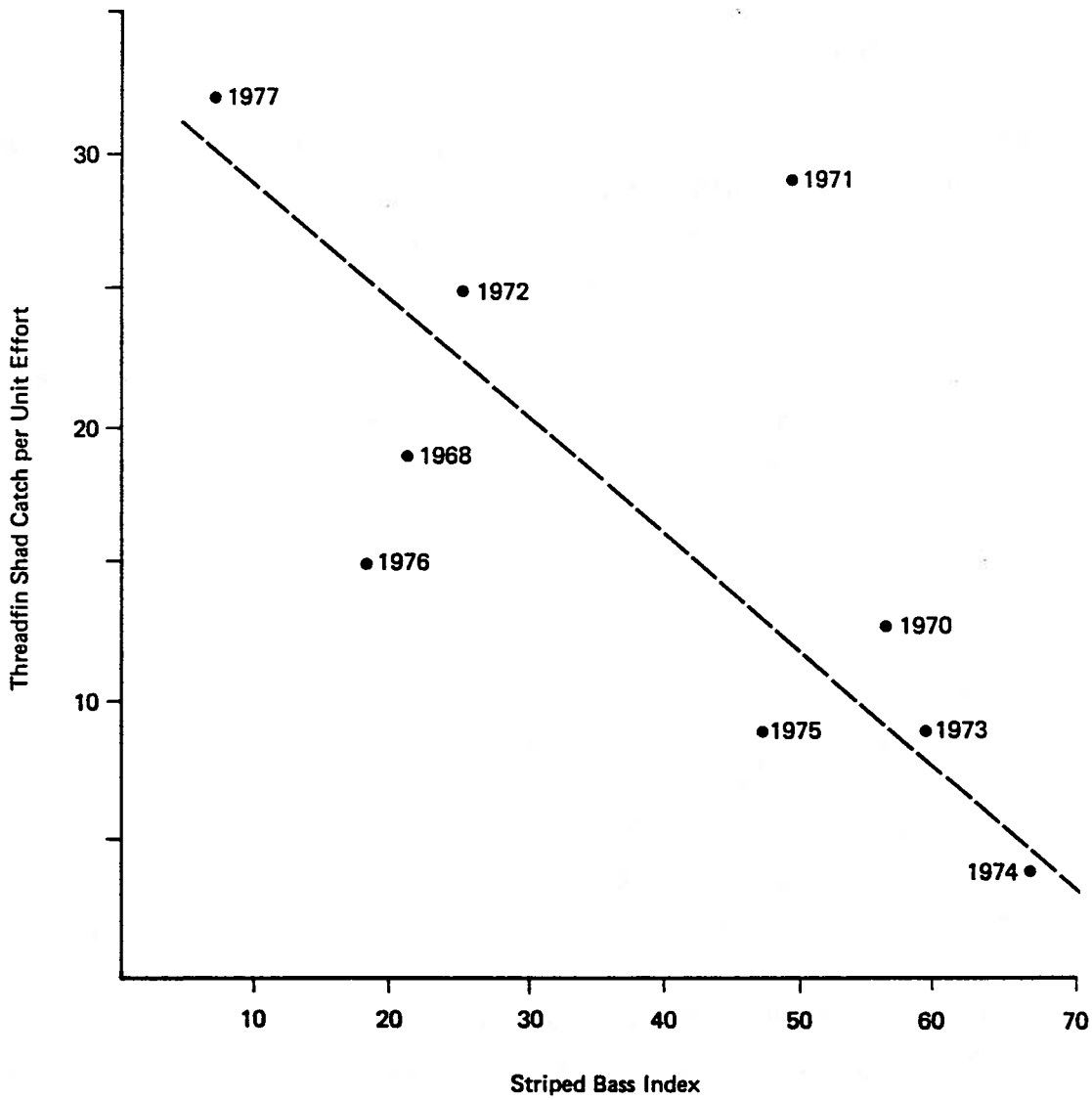


Figure 5-43. Abundance of threadfin shad (CPUE, CDF&G midwater trawl) vs. abundance of striped bass (index of year class strength) in the Bay-Delta.

silverside, and Palaemon macrodactylus. The abundance of these species has increased rapidly and is still increasing in the Bay-Delta. This rapid increase in abundance may be contributing to a reduction in the systems carrying capacity for striped bass. There is little or no population information for many of these recently introduced species, so their potential for impact on the striped bass population through competitive interaction is only speculative.

Other factors which reduce the availability of zooplankton in the Bay-Delta include entrainment at the Pittsburg Power Plant, loss at water diversions and agricultural withdrawals, and the toxicity of chemical pollutants. The cropping of zooplankton as a result of entrainment at the Pittsburg Power Plant is discussed in Chapter 3 and in Sections 5.3 and 5.5. The cropping of neomysis and the potential consequences for the neomysis population are discussed in detail in Section 5.5. Entrainment cropping was not considered to be a major factor contributing to population regulation for zooplankton. In general, survival of entrained invertebrates is high (90-95 percent survival for small zooplankton such as copepods, and 65-70 percent survival for neomysis; Section 3.2; Appendix B). Furthermore, those invertebrates which do not survive entrainment are returned to the receiving water body and remain part of the trophic system. An examination of the proportion of total mortality of neomysis represented by entrainment cropping (Section 5.5) indicated that on the average, entrainment mortality at the Pittsburg Power Plant accounted for a very small fraction (0.6 percent) of the total daily mortality rate. Therefore, although entrainment does contribute to a reduction in standing stock of zooplankton, it does not appear to be a dominant factor in the regulation of their populations or a major contributor in the apparent decline in the index of neomysis abundance.

The loss of zooplankton at the major water diversions and at agricultural withdrawals throughout the Delta has not been quantified. The effect of these losses on the carrying capacity of the system for striped bass may be relatively high. These diversions withdraw substantial volumes of water, and thus zooplankton, and the organisms entrained are removed completely from the Bay-Delta trophic system.

The effects of toxicants on zooplankton in the Bay-Delta has not been extensively studied. Heavy metals, pesticides, petroleum, and chlorinated hydrocarbons have been shown in laboratory studies to be toxic to crustaceans. The direct toxic effect of these pollutants at concentrations found in the Bay-Delta on the production of

zooplankton is not known, nor is their indirect effect acting through the trophic system, for example on microbial decomposers.

The final factor that would potentially affect food availability and the carrying capacity of the system for striped bass was density-dependent intraspecific competition. The synchrony of striped bass spawning and limited distribution within the nursery area (EA 1981a) suggest that the resulting relatively high densities of striped bass larvae could possibly lead to strong intraspecific competition for a limited food resource. To examine this mechanism, the index of striped bass production was regressed against the estimated size-specific survival rate for various size classes of larvae and juveniles. It was expected that if intraspecific competition was an important factor in population regulation, a negative relationship would exist: high survival at low production levels and low survival in years of high production. The results presented below do not show any relationship between estimates of production and estimates of size-specific survival:

Size class(mm)	Observations	Slope	S.E.	P
7-10	2	—		
11-14	2	—		
15-17	3	-0.09	0.06	0.38
18-22	2	—		
23-27	5	0.004	0.002	0.15
28-32	6	0.002	0.0015	0.20
33-37	13	0.002	0.001	0.90
38-40	10	-0.002	0.002	0.46

The significance of these results is in question, however, because of biases associated with estimating survival (or mortality) rates for larval fish. The basic sampling design used in the larval fish surveys may be inadequate to precisely define size-specific larval survival rates. The spatially and temporally patchy distribution of larval striped bass and the very high mortality rates of early life stages contribute to low sampling precision. In addition, the loss of small larvae in plankton net collections and the avoidance of collection nets by the larger larvae bias survey results. Furthermore, establishing reliable survival rates is difficult for adjacent size classes with different variances and distributional patterns such as those typical of larval and juvenile striped bass. Thus, the importance of density-dependent intraspecific competition in determining larval survival and influencing year class strength remains uncertain.

Habitat Availability Habitat availability for larval and juvenile striped bass depends in part on suitable water temperatures, salinities, and dissolved oxygen concentrations to support growth and survival. The availability of suitable habitat is also interrelated with food availability, as previously discussed. These factors are influenced by the hydrologic conditions (which are dominated by the interaction between freshwater outflow and saltwater intrusion; Section 2.2) and the morphology of the Bay-Delta.

Water temperature and dissolved oxygen concentrations in the Bay-Delta system are within the range of requirements for growth and survival. Davies (1970) reported the temperature range for larval survival as 59-72 F (15-22 C). Temperatures measured in Suisun Bay (Figure 5-36) are within the reported tolerance range for the period from mid-May through June when larval bass are present in the system. Based on a summary of available literature (Setzler et al. 1980) it was concluded that dissolved oxygen concentrations in excess of approximately 5 ppm are sufficient to support larval and juvenile striped bass. A review of water quality data collected throughout the Bay-Delta (data from the USBR-DWR Coordinated Monitoring Program) indicated that dissolved oxygen concentrations are consistently in excess of the minimum required. No information was found to suggest that either water temperature or dissolved oxygen concentration have limited habitat availability for young-of-the-year striped bass.

Survival and growth of young striped bass have been demonstrated in laboratory studies to be influenced significantly by salinity as well as food availability. Optimal survival of striped bass between 1 and 20 days after hatching occurred within a salinity range from 3.4 to 6.8 ppt, with lower salinities optimal for younger larvae (Lal et al. 1977). Survival and growth rates of larval striped bass have been consistently shown to be higher in low-salinity water than in fresh water (see Setzler et al. 1980 for a review). These data are consistent with the notion that the primary habitat for young striped bass is within the entrapment zone, where salinities range from approximately 1 to 6 ppt (Section 2.2). In addition, the entrapment zone has been shown to have higher primary productivity and zooplankton densities, and hence higher food availability for young bass, than other areas of the Bay-Delta (Arthur and Ball 1979). The availability of habitat with the greatest potential for supporting striped bass survival therefore depends on the configuration, location, and productivity of the entrapment zone.

In years when spring and early summer outflow rates are moderate (5,000-20,000 cfs), the entrapment zone is located within Suisun Bay, where the water is generally shallow and zooplankton densities are high. In years of low spring and early summer outflow (less than 5,000 cfs) the entrapment zone is located further upstream in the Delta and the lower Sacramento and San Joaquin rivers. As a result of channelization of these areas, the water is deeper and current velocities are typically greater, and productivity is generally lower than in Suisun Bay. The availability of optimum habitat is considered to be higher in years in which outflow results in location of the entrapment zone in Suisun Bay and lower when the entrapment zone is located in the interior Delta and river channels.

The role of habitat availability, as reflected in both optimal salinity and food availability, in determining striped bass year class strength is included in the basic outflow mechanism presented in Figure 5-35. Variations in salinity would have a direct effect on the energy requirements of larval striped bass for osmoregulation, which contributes to an increased metabolic rate. Variations in productivity within the entrapment zone, a function of the effect of outflow on the location of the entrapment zone in relation to basic morphological and hydrological characteristics of the system, would have a direct effect on regional and absolute variation in food availability for larval and juvenile bass. Therefore, changes in availability of optimal habitat do occur as a function of changes in outflow rates, and this would have an effect on year class strength.

An important long-term trend in reducing available habitat suitable as a striped bass nursery area in the Delta has resulted from levee construction and channelization. The extensive tidal marshlands that once covered the Delta and provided a nursery habitat for young striped bass have gradually been converted to a maze of interconnected, leveed waterways surrounding reclaimed farmlands. The long-term reduction in available habitat as a nursery area in the Bay Delta is one factor which has potentially contributed to the general decline in the striped bass population since the early 1900s.

Predation Direct evidence of the effect of predation and cannibalism on striped bass year class strength is lacking. Several species of fish and invertebrates, including yellowfin goby, Sacramento squawfish, white catfish, prickly sculpin, *Palaemon macrodactylus*, *Crangon franciscorum*, and neomysis, are potential predators on larval and juvenile striped bass that occur in the Bay-Delta in relatively

high abundance. The frequency of occurrence of striped bass larvae in the diet of these species and the magnitude of the effect of predation on year class strength are unknown.

Sommani (1972), examining the stock-recruitment relationship for the Bay-Delta striped bass population, concluded that cannibalism was an important factor affecting year class strength and subsequent recruitment of the year class. Because a portion of the yearling striped bass population resides in the area with the young-of-the-year population (EA 1981b, c, f) and are known to prey on juveniles (Stevens 1966b; Manooch 1973), the potential affect of cannibalism was investigated. It was expected that if cannibalism was an important factor affecting year class strength, there should be a significant negative relationship between the abundance of young-of-the-year striped bass of a specific size and an index of yearling abundance (CPUE in the CDF&G midwater trawl surveys; Figure 5-23). The results of the analysis, for both the 30-34 mm and 35-39 mm size classes:

Size class	N (years)	Slope	S.E.	Probability
30-34	8	-10.0	45.3	0.83
35-39	9	-12.4	26.2	0.65

indicate that cannibalism is not a major factor contributing to the regulation of striped bass year class strength.

Entrainment at the Power Plant The contribution of entrainment mortality at the Pittsburg Power Plant to the total estimated mortality rate for 35-39 mm striped bass is shown in Figure 5-44 for the years 1961-1979. Estimated mortality rates were calculated using the survey data and methodologies described in Section 5.6. Two features are important to note in the comparison of mortality rates. First, with the exception of the drought years 1976 and 1977, the estimated mortality rate associated with entrainment has been consistently low, and has shown no apparent tendency to increase. Second, the proportion of the total mortality rate attributable to entrainment at the power plant is small. These data suggest that the direct cropping of young-of-the-year striped bass as a result of entrainment is not a major factor regulating year class strength, and in the absence of an increasing trend in entrainment mortality, this does not appear to be a major factor contributing to the apparent decline in year class strength. The pattern in mortality rates was similar for all size classes for which mortality estimates are available.

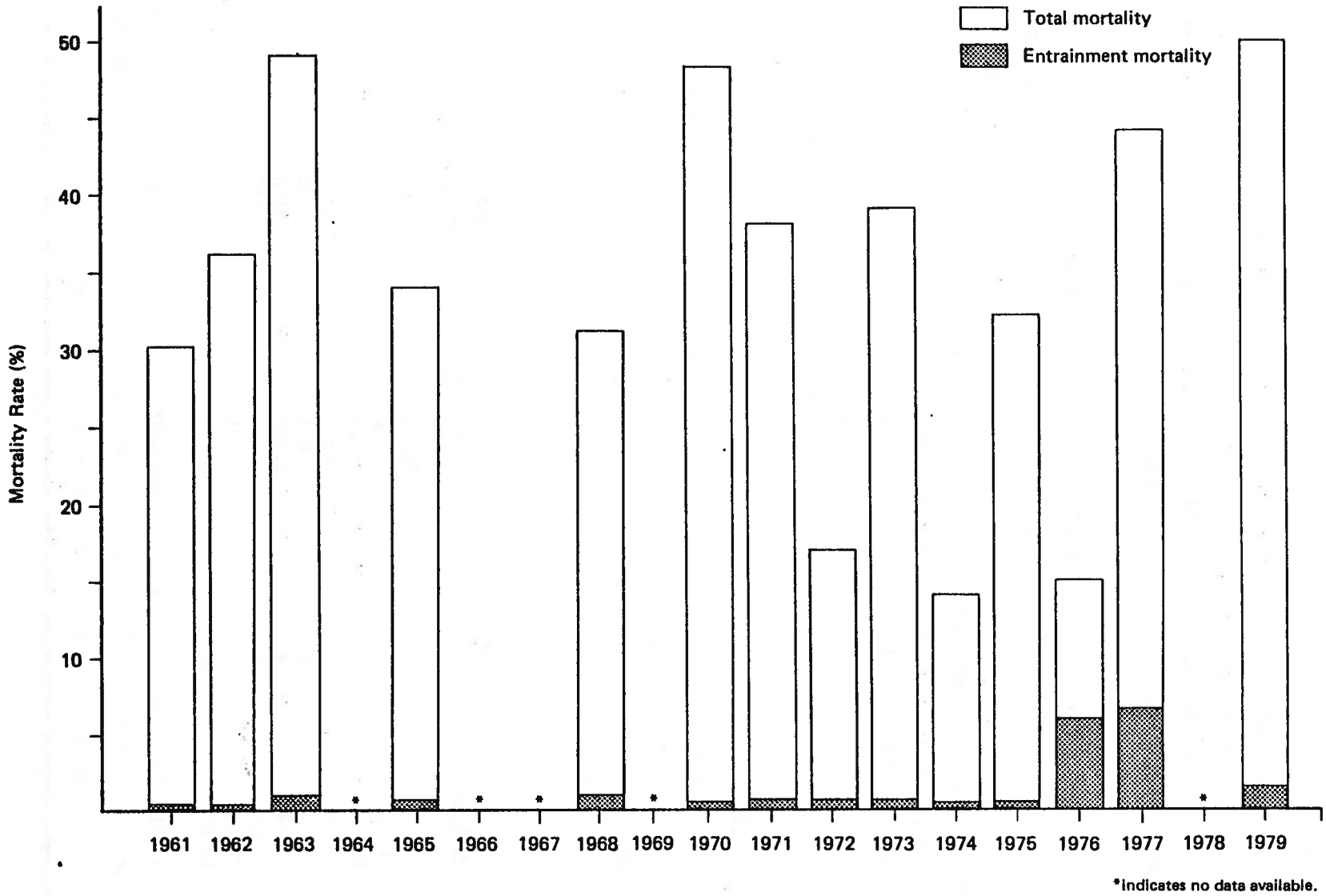


Figure 5-44. Estimated entrainment mortality and total mortality of the 35-39 mm size class of striped bass in the Bay-Delta, 1961-1979.

The substantially higher mortality rates attributed to entrainment in 1976 and 1977 compared with other years were the result of drought conditions. The drought conditions during these two years resulted in abnormally low outflow rates during the spring and summer, and hence the entrapment zone and distribution centers of the young-of-the-year striped bass population were located further east in Suisun Bay and the lower Delta, and therefore striped bass were more susceptible to entrainment at the cooling water intakes of the Pittsburg Power Plant than in other years. In addition to the upstream shift in the population distribution, there was less water available for hydroelectric generation, and the warm spring and summer temperatures increased the demand for electrical power, which was met in part by increased power production at the Pittsburg Power Plant. Increased electrical demand resulted in increased loads and higher discharge temperatures, which resulted directly in higher mortality rates for entrained larval and juvenile striped bass. Entrainment mortality rates for 35-39 mm fish increased from 0.009 (0.9 percent) in 1975 to 0.062 and 0.077 in 1976 and 1977. The entrainment mortality rate declined in 1979 to 0.021 as outflow increased. The average mortality rate for 35-39 mm striped bass attributable to entrainment for the period 1961-1979 was 0.019 (1.9 percent; S.D. = 0.022; n = 14) compared with an average size-specific total mortality rate of 0.312 (31.2 percent; S.D. = 0.114; n = 14) for the same period. On the basis of these comparisons and of similar data for various size classes summarized below, it was concluded that entrainment mortality at the Pittsburg Power Plant is a small part of the total mortality rate of the larval and juvenile striped bass population.

Size (mm)	Number of Years	Mean Entrainment Mortality Rate	Mean Total Mortality Rate
9-10	6	0.012	0.865
11-14	6	0.016	0.620
15-19	6	0.022	0.480
20-24	3	0.029	0.268
25-29	6	0.021	0.250
30-34	8	0.020	0.316
35-39	14	0.019	0.312
40-44	14	0.021	0.358
45-49	11	0.016	0.426

Despite the small contribution of entrainment mortality to the total mortality rate, the cropping of young-of-the-year striped bass as a result of entrainment does effect population abundance directly. The relationship between involvement of the young-of-the-year population with the power plant and subsequent recruitment of the

year class into the sport fishery will be discussed in Section 5.7.4.1.

Loss at Water Diversions An analysis of factors affecting or correlated with the variability in the index of striped bass year class strength (Section 5.6.3.2.2) included consideration of water exports at the Delta and Tracy pumping plants (SWP and CVP). Water exports at these two major diversions were not significantly correlated with the index of year class strength in any of the analyses. Our results indicate that although export flows at both pumping facilities do remove a large number of young striped bass from the system, the variability in the magnitude of export flow is not related to the observed variability in the index of striped bass year class strength.

Information on the spatial distribution of larval and juvenile striped bass in the Bay-Delta in spring and early summer 1978 and 1979 (EA 1981a) indicates that the young-of-the-year populations were distributed throughout a large area of the Bay-Delta system. The distribution of the striped bass population was not centered in the vicinity of the pumping facilities, which are located on or adjacent to Clifton Court Forebay on the Old River channel of the San Joaquin River (Figure 2-1). The wide geographic distribution of the population suggests that only a small fraction of the year class would be susceptible to the direct effect, entrainment in the export flow, of the pumping facilities. Therefore, although the number of striped bass diverted is relatively large, the proportion of the total population affected appears to be small and not a dominant factor in the regulation of year class strength.

Although CDF&G has concluded that water export and the associated entrainment of young striped bass at the Delta and Tracy pumping plants are important in controlling the abundance of young-of-the-year striped bass (Chadwick 1979; Stevens 1977b, 1980), results of our analyses provide no direct evidence to support this conclusion.

The magnitude of the loss of young striped bass resulting from entrainment at the numerous agricultural diversions located throughout the Sacramento - San Joaquin Delta and river systems is unknown. Although these diversions are not directly quantified, there is no clear evidence that the number of agricultural diversions or the volume of water (and presumably the numbers of striped bass) diverted have increased at a rate substantial enough to be a dominant factor in the recent decline in striped bass year class strength.

Parasitism and Disease There is no evidence suggesting that disease or parasitism is an important factor controlling year class strength through increased mortality of larval and juvenile striped bass. Disease and parasitic infestation in early life stages (eggs and larvae) constitute an important potential source of mortality in hatchery and pond culture (Amlacher 1970), but this has not been well-documented in natural populations. Most of the information available on disease among striped bass in natural systems is from examination of yearling and subadult fish (Setzler et al. 1980).

It can be speculated that disease might be an important factor in the mortality of larval striped bass stressed by a lack of food, which would increase their susceptibility to infection. Disease transmission would be facilitated by the relatively high densities of striped bass during early life stages, but this mechanism has not been documented as an important factor contributing to the establishment of a year class or the general decline in year class strength.

Toxic Substances Data on acute toxicity of 61 pesticides, heavy metals, and pharmaceutical drugs on striped bass eggs, larvae, and juveniles has been compiled by Bonn et al. (1976). Research currently underway (Whipple 1980) will provide additional information on chronic and delayed effects of exposure of early life stages to very low levels (ppb) of toxic substances such as petrochemicals (benzene). Preliminary results suggest that striped bass larvae accumulate benzene and its metabolites through uptake from the water and food until death occurs (Whipple 1980). Yolk-sac larvae and larvae appear to be particularly susceptible to toxic substances, many of which are lipid-soluble. The susceptibility of the early life stages is due in part to their high lipid content, high metabolic rates, and high surface-area-to-volume ratio. The exact role of toxic substances in affecting larval and juvenile mortality rates and subsequent year class strength is unknown.

Genetic Fitness Whipple (1981) speculated that there may be genetic polymorphism within the Bay-Delta striped bass population, with certain genotypes adapted to conditions occurring during years with high spring outflow and other genotypes adapted for low outflow conditions. The role of genetic variability in affecting year class strength through differential fitness has not been determined.

5.7.3.3 Conclusions

It is apparent that a number of factors, acting independently and in combination,

affect the numbers of young-of-the-year striped bass produced. Figure 5-45 presents a simplified overview of the relationship between striped bass year class strength and many of the factors discussed. Although a number of these factors contribute directly to a reduction in the abundance of young-of-the-year striped bass (toxicants, predation, power plant cropping, diversion cropping, disease and parasitism), these do not appear to be major factors in determining year class strength. The available statistical evidence and biological information support the hypothesis that year class strength is a function of freshwater outflow acting throught metabolic pathways and the spatial and temporal distribution of larval striped bass with respect to the availability of suitable prey, modulated by factors which affect the availability of zooplankton, the food source necessary for growth and survival of larval and juvenile striped bass. The available information suggests that the decline in striped bass year class strength may be attributed in large part to a decline in the carrying capacity of the system for striped bass. We have speculated that the declining carrying capacity is a result of a decline in zooplankton abundance, which may be attributed to a general reduction in available nutrients in the form of heterotrophic bacteria, diatoms, and algae, combined with increased competition for zooplankton from introduced species, such as threadfin shad, which are presently abundant in the Bay-Delta. Entrainment of zooplankton at the Pittsburg Power Plant does not appear to be a major indirect factor in the decline in year class strength through a reduction in carrying capacity, because the entrainment survival of invertebrates is high. The indirect effects on carrying capacity of toxicants and of the cropping of zooplankton at water diversions remain uncertain. Cropping at diversions does not appear to be a significant factor contributing to an apparant decline in the abundance of neomysis.

It appears that a complex assemblage of biotic and abiotic factors combine to determine striped bass year class strength. Interactions among these factors influence the metabolism of larval bass, their distribution, and the carrying capacity of the system for them. Striped bass year class strength is determined to a large degree by the carrying capacity of the system, with direct cropping such as those resulting from power plant cooling system operation representing a small contribution to the overall mortality rate. It is also apparent that despite a large body of data on the distribution and abundance of young striped bass, and the recent compilation of an extensive data base on involvement with power plant cooling systems (entrainment, impingement, and thermal effects), there is a lack of information on factors such as trophic dynamics and the rate of interspecific competition for food,

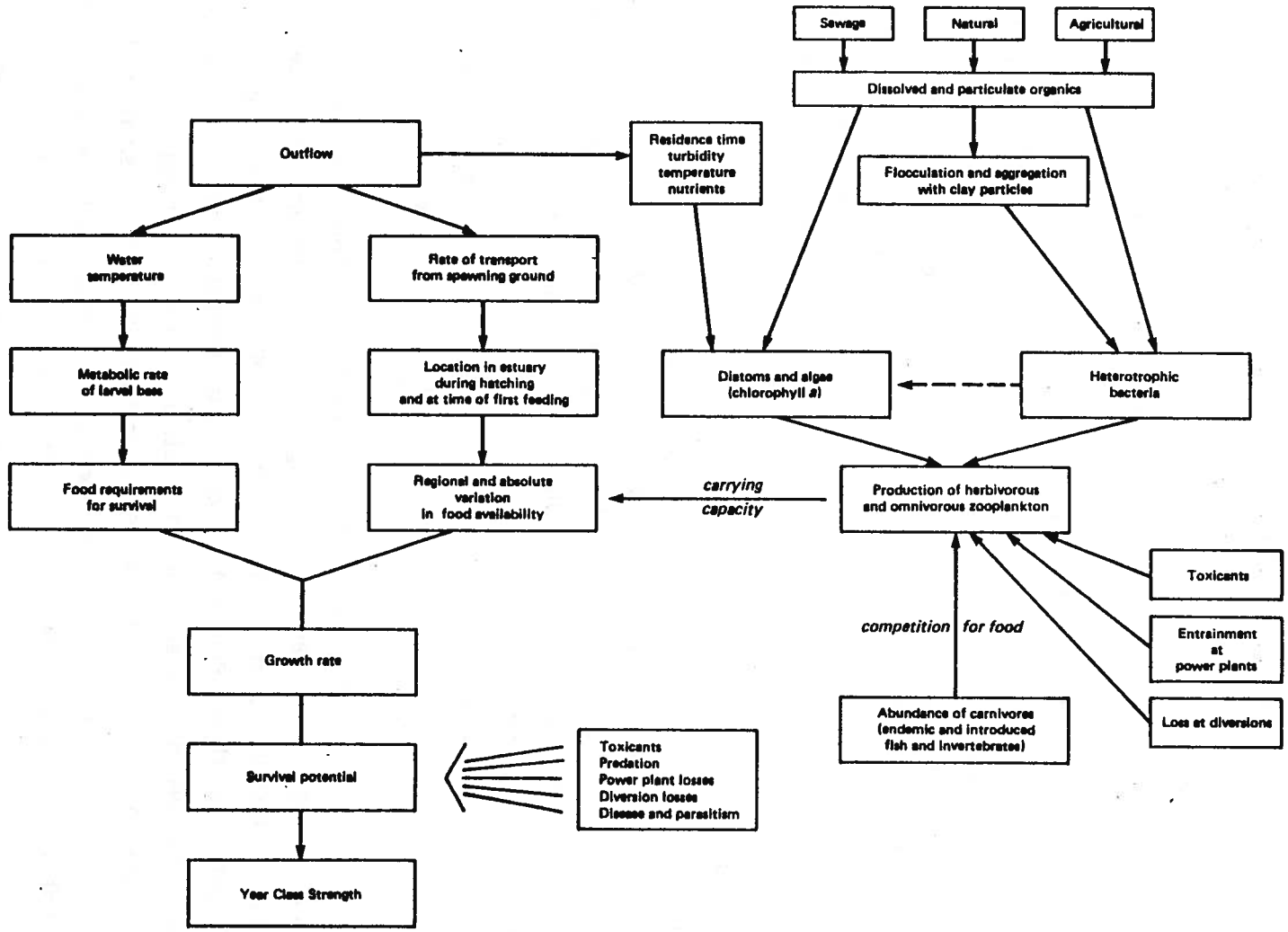


Figure 5-45. A model of the relationship between Delta outflow, system carrying capacity, sources of mortality, and striped bass year class strength in the Bay-Delta.

especially with introduced species, which determine the carrying capacity of the system for striped bass.

Compounding the complexity of interactions among species and trophic levels is the high variability in abiotic factors such as freshwater outflow, both between years and within years. Daily variation in outflow during the spring and early summer months may be an important consideration which is obscured by the use of mean monthly flows. Variability in outflow between months may also be a regulating factor in the survival of young striped bass.

The effect of reductions in young-of-the-year abundance on subsequent recruitment into the sport fishery, in addition to other factors affecting subadult striped bass, are discussed in the next section.

5.7.4 Recruits

There is a significant positive relationship (Section 5.6.3.2.1) between the abundance of young-of-the-year striped bass and subsequent recruitment into the population available for harvest in the Bay-Delta sport fishery. Therefore, factors contributing directly or indirectly to an increase or a decrease in the abundance of young-of-the-year fish may be reflected subsequently in the abundance of three-year-old bass entering the fishery. This relationship, however, is modified by biotic and abiotic factors which influence the growth and survival of the potential recruits.

5.7.4.1 Factors Affecting Recruit Abundance

Year Class Strength The primary mechanism which has been implicated in the general decline in recruitment (based on the decline in sport fishery CPUEs, Figure 5-31) to the sport fishery has been a decline in year class strength. Factors which may be contributing to a trend in declining year class strength have been discussed in the previous section. The primary objective of this discussion is to examine the potential effects of power plant cooling system operation on the number of striped bass recruited into the sport fishery as a result of the entrainment or impingement of young-of-the-year bass. The approach used was to examine the trends in the number of 3-year-old striped bass (larger than 16 inches) on May 1 (Figure 5-46) and estimates of the involvement of young-of-the-year striped bass with the power plant

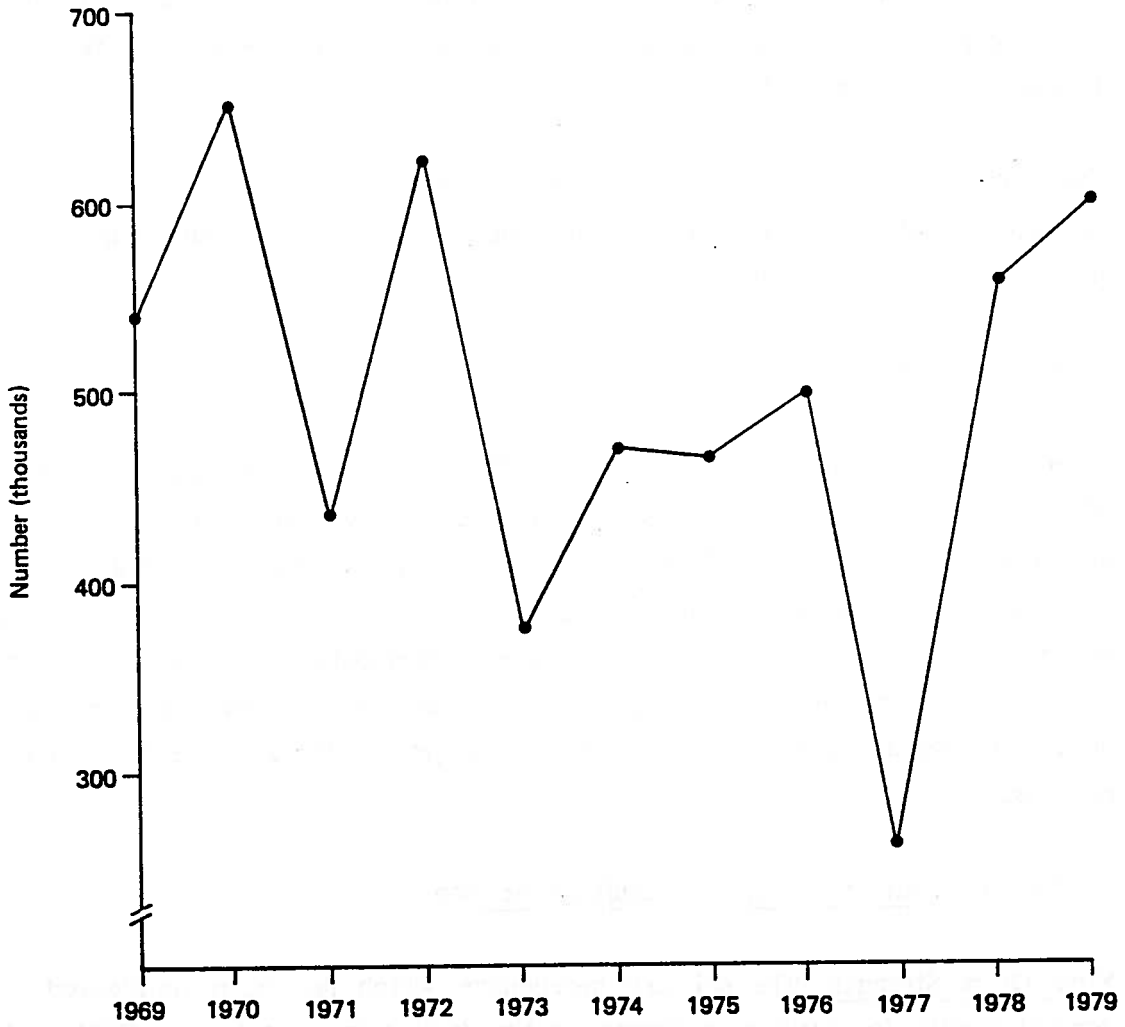


Figure 5-46. Estimated numbers of three-year-old striped bass (those over 16 in. long) in the Bay-Delta, 1969-1979. (From CDF&G unpublished.)

cooling systems three years earlier. It was hypothesized that if mortality resulting from cooling system operation was having a significant effect on the population, years in which power plant involvement was high should result in subsequent low recruitment and years characterized by low young-of-the-year involvement should be reflected in high recruitment. Percentage reduction estimates (Table 5-29) were used as the indicator of striped bass involvement with the Pittsburg Power Plant.

A comparison between the index of recruitment and the corresponding estimated percentage reduction in young-of-the-year striped bass resulting from entrainment at the Pittsburg Power Plant three years earlier is shown below for years characterized by low and high recruitment:

	Recruitment	Reduction (%)
1971	440	10.8
1973	390	3.6
1976	500	6.0
1977	280	2.4
1978	550	4.9
1979	600	21.8

There does not appear to be any relationship between estimates of percentage reduction attributable to entrainment and subsequent recruitment. The year with the lowest recruitment index (1977) corresponds to the year in which estimated percentage reduction was lowest (1974). One of the years with the highest recruitment (1979) corresponds to the year in which the estimated reduction in the young-of-the-year population attributable to entrainment at the Pittsburg Power Plant was greatest (1976). These data provide no evidence to suggest that the magnitude of entrainment cropping at the Pittsburg Power Plant is a dominant factor in determining whether recruitment will be high or low.

It is also interesting to note that, at least in some years, a high year class strength index is not necessarily reflected in subsequent high recruitment. For example, the 1974 year class strength index was one of the highest during the period of record (Figure 5-34), and yet the index of recruitment into the sport fishery in 1977 was the lowest observed between 1969 and 1979 (Figure 5-46). In contrast, the year class strength index for 1976 was abnormally low but recruitment in 1979 was high. These apparent discrepancies in the relationship between indices of year class strength and recruitment suggest that the actual relationship is much more complex than it originally appeared. A regression analysis of the relationship between the index of

year class strength (35-39 mm production) and the recruitment index indicated that although the relationship was significant ($P < 0.10$), variation in year class strength accounted for only 35 percent of the variability in the recruitment index. Factors such as density-dependent mortality from young-of-the-year to 3-year-olds may also be contributing to the relationship between year class strength and recruitment.

The implication of the high variability in the relationship between the year class strength and recruitment is that efforts to further reduce the relatively small contribution of power-plant-related mortality to total mortality through the implementation of alternative cooling system technology are not likely to contribute to a detectable improvement in recruitment to the sport fishery. There is no evidence in the available population data to suggest that a reduction in the historical levels of power plant mortality would have a direct effect on an increase in abundance in the adult stock. This conclusion is consistent with results of the analysis and discussion of factors affecting the year class strength of striped bass—power plant involvement contributes very little to the regulation and dynamics of the Bay-Delta striped bass population.

This analysis does not imply, however, that recruitment is independent of year class strength. Therefore, those factors such as freshwater outflow and striped bass carrying capacity discussed in the previous section which have a major effect on the regulation of year class strength will affect subsequent recruitment to a detectable degree. This conclusion agrees with results of the stock-recruitment analysis presented in Section 5.6.3.3, in which recruitment was found to be significantly related to spring outflow rates three years earlier, and this is entirely consistent with the results of the CDF&G analysis (Stevens 1980).

Food Availability Yearling and subadult striped bass prey on a variety of fish and invertebrates. Northern anchovy, Pacific herring, yellowfin goby, smelt, threadfin shad, *Palaemon macrodactylus*, *Crangon franciscorum*, and neomysis are all known prey of striped bass. On the basis of the information presented in Sections 5.3, 5.4, and 5.5, there is no evidence to suggest that entrainment or impingement cropping of these species at the Pittsburg Power Plant have reduced their abundance to a level that would represent a significant depletion of the prey available to striped bass.

Fishing Pressure Although it is not legal to harvest striped bass in the Bay-Delta less than 16 inches in length, these smaller fish are caught in the sport fishery and

are subject to unknown levels of hooking mortality and illegal possession. This is not expected to represent a major mortality source for subadult striped bass.

Predation There are few predators in the Bay-Delta that are of sufficient size to prey on subadult striped bass effectively.

Impingement at Power Plants Yearling striped bass are impinged in low numbers at the Pittsburg Power Plant, primarily during the summer and fall. It was estimated that in 1978 and 1979 approximateley 4,700 yearling and subadult striped bass more than 100 mm long were impinged (Table 5-15). Assuming density-independent uniform survival rates for each age class, the equivalent number of striped bass which potentially would have been recruited was estimated to be 1,700 individuals. The number of 3-year-old striped bass recruited into the Bay-Delta sport fishery typically ranges from approximately 800,000 to 1.3 million per year (Table 5-31). Therefore the equivalent adult loss of 1,700 individuals attributed to impingement of yearlings and subadults, or more generally, impingement of all sizes of striped bass at the Pittsburg Power Plant (a total estimated equivalent adult loss of approximately 15,300 individuals between March 1978 and March 1979; Table 5-15) does not represent a significant contribution to the recruitment potential of the population.

The number of yearling striped bass impinged, although low, was higher than originally expected. It was hypothesized that some biotic or abiotic factor was contributing to reduced fitness, reduced swimming performance capability, or physiological stress in a portion of the yearling striped bass population and resulting in an increased susceptibility to impingement. One possible mechanism which would contribute to increased susceptibility to impingement, parasitic infestation, was examined, and it is described in the section on disease and parasitism.

Loss at Water Diversions The cropping of yearling and subadult striped bass from the system at agricultural withdrawals and water export facilities has not been extensively studied. The majority of agricultural withdrawals, although unscreened, are relatively small in size and are located in the Delta and river systems which are outside of the more saline region of the Bay-Delta where population abundance is greatest. The estimated number of yearling striped bass involved with the Delta Pumping Plant between 1968 and 1977 ranged from 41,500 to 408,000 and averaged 297,000 annually. Diversion efficiency studies conducted at the facility suggest that

the majority of the yearling striped bass entering the facility would be diverted by the guiding louvers and salvaged from the system to be returned to the Bay-Delta. The combined magnitude of the numbers cropped at all diversions, however, is unknown.

Disease and Parasitism A number of diseases and parasites are known to attack yearling and subadults striped bass (Setzler et al. 1980); infestation of cestode larvae (*Lacistorhynchus tenuis*) has received the greatest attention with regard to the Bay-Delta population (Whipple 1980). A synergism between exposure to toxic substances, disease, and parasite infestation has also be speculated (Whipple 1980). The effect of these factors on the the growth and survival of yearling and subadult striped bass is unknown. Furthermore, the historical importance of disease and parasite infestation in striped bass die-offs and in the long term trends in population abundance is uncertain.

During 1978 and 1979 a study of parasitic infestation in yearling striped bass (approximately 100 mm in length) was conducted (EA unpublished data a) to compare the degree of infestation in fish impinged on cooling water intake screens at the power plant and that in fish collected from the source water body with standard fishery sampling gear (EA 1981b, c). A total of 389 yearlings from impingement collections at the Pittsburg Power Plant and 62 yearlings from fishery collections were examined for both internal and external parasite infestation. The most common parasites observed were trichodina, helminth, and cestode larvae. A chi-square test for significance in the number of infested and noninfested fish in impingement and fishery samples indicated that the proportion of impinged fish having parasitic infestation was significantly higher ($P = 0.01$) than that observed from the fishery collections.

Although the exact reason for the significantly higher frequency of parasitized fish in the impingement collections is unknown, it can be speculated that the survival potential of the infested fish would be lower than those fish not parasitized. There is a high probability of increased natural mortality for those yearling striped bass susceptible to impingement (and presumably to many other sources of mortality such as increased predation and reduced foraging success) that is due to the influence of parasitic infestation. Therefore, those yearling striped bass which are impinged may be characterized by reduced fitness and hence contribute less to the recruited population than would otherwise be expected. The significance of differential

survival potential within the yearling and subadult populations on the population dynamics and recruitment potential of a year class is unknown.

Toxic substances Preliminary findings in laboratory studies indicate that the response of striped bass to low levels of petrochemicals varies with life stage (Whipple 1980). Yearlings and subadults which are nonreproductive were less sensitive to toxic substances than other life stages. The importance of toxic substances such as pesticides (DDT, dieldrin, lindane), heavy metals, and monocyclic aromatic hydrocarbons in the growth and survival of striped bass, and the possible synergisms with parasite infestation, are continuing to be studied.

5.7.4.2 Conclusions

Entrainment A detailed examination of entrainment mortality and striped bass population dynamics leads to the conclusion that there is no clear relationship between estimates of percentage reduction attributable to entrainment at the Pittsburg Power Plant and subsequent recruitment. Entrainment cropping at the plant is not a dominant factor in determining whether subsequent recruitment will be high or low. There is no evidence based on population dynamics suggesting that changes in intake technologies at the plant could result in a detectable change in the sport fishery. It appears that only those factors, such as freshwater outflow and carrying capacity, which have the dominant effect on regulation of year class strength are likely to affect subsequent recruitment to a detectable extent.

Impingement The direct effect of the Pittsburg Power Plant on the population, acting through impingement of yearling and subadult striped bass, does not appear to be significant. The number of yearling striped bass impinged is low, representing an estimated equivalent adult loss of 1,700 individuals. In addition, the yearling striped bass that were impinged were found to have a significantly higher rate of parasitic infestation than fish collected from the source water body. This parasitic infestation could contribute not only to an increased susceptibility to impingement, but also to a reduced potential for survival in the Bay-Delta, futher reducing the direct effect attributable to impingement cropping on the potential recruitment of striped bass into the sport fishery.

Other Factors The role of factors such as parasitic infestation and its potential synergism with low concentrations of toxic substances on the survival of striped bass is only speculative. As noted earlier, these factors do suggest the possibility of differential survival within a cohort (age class) or within subpopulations in the Bay-Delta. The importance of these types of factors in understanding the effect of power plant operation on the striped bass population and in the more general objective of resource management, is evidenced by the fact that only 35 percent of the variability in the index of recruitment was explained by variability in the index of year class strength. Because of the complexity of interactions involved in regulating the striped bass population, it cannot be confidently asserted that an increase in young-of-the-year abundance, for example as a result of a reduction in entrainment cropping at the Pittsburg Power Plant, will be reflected in an increase in recruitment into the sport fishery. This does not imply that factors contributing to the mortality of striped bass at any age are not important, but rather that the relative magnitude of the effect on the population associated with each mortality source needs to be considered in perspective with the factors that have a dominant effect on the abundance of the population in the system.

5.8 Summary and Conclusions

Available information for a majority of the species considered in the impact assessment indicates that the numbers of organisms entrained and impinged at the Pittsburg Power Plant will not result in an adverse environmental impact. Rather, for many of the species entrained or impinged in relatively high numbers, their populations are generally considered to be stable (goby, *Palaemon macrodactylus*, *Crangon franciscorum*). The striped bass population of the Bay-Delta, however, has experienced a general decline in abundance, and therefore one of the primary objectives of the impact assessment analysis has been to determine what the relative contribution of operation of the Pittsburg Power Plant cooling water systems might be among the factors affecting the survival of young striped bass and the decline in year class strength. Operation of the Pittsburg Power Plant cooling systems could affect the striped bass population either directly, as a result of entrainment or impingement, or indirectly, by affecting the dynamics of the trophic system which supports the striped bass population. Available data and information from the literature were compiled to help in identifying and selecting possible factors and mechanisms contributing to the decline in the striped bass population. Factors

influencing the adult stock, egg production, young-of-the-year survival, and survival to three years were considered in the impact analysis.

A stock-recruitment analysis (Section 5.6.3.3) indicated that there is no obvious dependency between the adult stock of striped bass and reproductive success as measured by recruitment into the sport fishery. An adult population model based on the stock-recruitment analysis, percentage reduction estimates for entrainment cropping, and regression analysis of factors contributing to year class strength (Section 5.6.3.5) indicated that operation of the cooling systems of the Pittsburg Power Plant would contribute approximately 8 percent to a decrease in the adult striped bass population, assuming no compensatory survival. This potential reduction in standing stock would not be compounded from one generation to the next, and therefore, although it would reduce the size of the population in any given year it would not result in a continuing decline of the population. Since there is no evidence that entrainment or impingement of prey species is having an indirect effect on the food availability of striped bass, the decline in the population would not appear to be due to carrying capacity of the system for adults.

The susceptibility of striped bass eggs to entrainment is low, primarily because spawning and hatching occur upstream of the Pittsburg Power Plant. An estimated 1 million striped bass eggs were entrained at the plant during 1978 (Section 3.1). Female striped bass produce an estimated 11,000 - 2 million eggs per year, depending on age (an overall average of approximately 1 million eggs per female per year), and thus the number of eggs entrained at the Pittsburg Power Plant represents the production of a few females. This level of mortality would not be expected to result in significant depletion of the eggs produced in the Bay-Delta system.

A number of factors, acting independently and in combination, affect the numbers of young-of-the-year striped bass produced. Although some of the factors contribute directly to a reduction in the abundance of young-of-the-year striped bass (toxicants, predation, cropping at the power plant, cropping at diversions, disease and parasitism), these do not appear to be major factors in determining year class strength. The striped bass year class strength is determined to a large degree by the carrying capacity of the system, and direct losses like those resulting from power plant cooling system operation, represent a small contribution to the overall mortality rate. The available statistical evidence and biological information support the hypothesis that year class strength is a function of freshwater outflow acting

through metabolic pathways and the spatial and temporal distribution of larval striped bass with respect to the availability of suitable prey.

This proposed outflow - year class mechanism is modulated by factors which affect the availability of zooplankton, the food source necessary for growth and survival of larval and juvenile striped bass. The available information suggests that the decline in striped bass year class strength may be attributed in large part to a decline in the carrying capacity of the system for striped bass. We have speculated that the declining carrying capacity is a result of declining zooplankton abundance, which may well be due to a general reduction in available nutrients in the form of heterotrophic bacteria, diatoms, and algae, as well as increased competition for zooplankton from introduced species such as threadfin shad which are presently abundant in the Bay-Delta. Entrainment of zooplankton at the Pittsburg Power Plant does not appear to be a major causative factor in the reduction in carrying capacity, because the entrainment survival of invertebrates is high. Examination of the proportion of total mortality of neomysis represented by entrainment mortality at the Pittsburg Power Plant indicated that the average entrainment mortality rate was a very small fraction (0.6 percent) of the total daily mortality rate. Therefore, although entrainment does contribute to a reduction in standing stock of zooplankton, it is not a dominant factor in the regulation of their populations.

The potential effects of power plant cooling system operation on the number of striped bass recruited into the sport fishery which results from the entrainment or impingement of young-of-the-year bass or impingement of subadult striped bass were also examined. There is no relationship between estimates of recruitment and percentage reduction attributable to entrainment. And in fact, the year with the lowest recruitment index (1977) corresponds to the year in which estimated percentage reduction was lowest (1974), and one of the years with the highest recruitment (1979) corresponds to the year in which the estimated percentage reduction was highest (1976). These data provide no evidence to suggest that the magnitude of entrainment cropping at the Pittsburg Power Plant is a dominant factor in determining whether recruitment of a year class into the sport fishery will be high or low.

A regression analysis of the relationship between the index of year class strength and the recruitment index indicated that although the correlation was positive ($P < 0.10$),

variation in year class strength accounted for only 35 percent of the variability in the recruitment index. Factors such as density-dependent mortality between young-of-the-year and three years may also be contributing to the relationship between year class strength and recruitment. The implication of the high variability in the year class strength - recruitment relationship is that efforts made to reduce further the relatively small contribution of power-plant-related mortality to the total mortality of young-of-the-year striped bass through implementation of alternative cooling system technology is not likely to make a detectable improvement in recruitment to the sport fishery. There is no evidence in the available population data to suggest that a reduction in the historical levels of power plant mortality would result in an increase in abundance in the adult stock. This conclusion is consistent with results of the analysis and discussion of factors affecting the year class strength of striped bass — power plant involvement contributes very little to the regulation and dynamics of the Bay-Delta striped bass population.

Striped bass are impinged, in low numbers, at the Pittsburg Power Plant, primarily in summer and fall. It is estimated that approximately 4,700 yearling and subadult striped bass were impinged (Chapter 4). Assuming density-independent uniform survival rates for each age class, the equivalent number of striped bass which potentially would have been recruited was estimated at 1,700. The number of 3-year-old striped bass recruited into the Bay-Delta sport fishery typically ranges from approximately 800,000 to 1.3 million individuals per year. Therefore the equivalent adult loss of 1,700 individuals attributed to impingement of yearlings and subadults, or even the impingement of all sizes of striped bass at the Pittsburg Power Plant (total estimated equivalent adult loss of approximately 15,300 individuals; March 1978 - March 1979) does not represent a significant contribution to the recruitment potential of the population.

Available information suggests the possibility of differential survival within a cohort (age class) or between subpopulations in the Bay-Delta. The importance of examining the contribution of a variety of biotic and abiotic factors in the regulation of striped bass population dynamics in an effort to understand the effect of power plant operation on the striped bass population, as in the more general objective of resource management, is evidenced by the fact that only 35 percent of the variability in the index of recruitment was explained by variability in the index of year class strength. Because of the complexity of the interactions of factors regulating the striped bass population, it cannot confidently be asserted that an increase in young-of-the-year

abundance, for example as a result of a reduction in entrainment or impingement mortality at the Pittsburg Power Plant brought about through implementation of alternative intake technologies, will be reflected in an increase in recruitment into the sport fishery. This does not imply that factors contributing to the mortality of striped bass at any age are not important, but rather that the relative magnitude of the effect on the population associated with each mortality source needs to be considered in perspective with the factors that have a dominant effect on the abundance of the population in the system.

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Chapter 6

Evaluation of Alternative Intake Technologies

6.1 Demonstrated Operation Criterion

6.2 Biological Benefit Criterion

6.3 Feasibility Analysis

6.4 Summary and Conclusions

6.5 References

CHAPTER 6: EVALUATION OF ALTERNATIVE INTAKE TECHNOLOGIES

The purpose of this chapter is to evaluate alternative intake technologies for the Pittsburg Power Plant Units 1-7 cooling water structures. A hierarchical evaluation system was developed to assess which alternative intake technologies would reduce biological losses and be feasible for application to the cooling water system of the Pittsburg Power Plant. Alternative intake technologies were evaluated on the basis of the following four criteria:

1. the alternative technology is available and proven (i.e., it has demonstrated operability and reliability at a cooling water intake having a similar size and environmental setting to that at the Pittsburg Power Plant site;
2. implementation of the alternative technology will result in a reduction in the loss of aquatic organisms from the existing operating conditions described in Chapter 2;
3. implementation of the alternative technology is feasible at the Pittsburg Power Plant, based on site-specific considerations of engineering, operations, and reliability; and
4. the total economic cost of the alternative technology is proportionate to the environmental benefits anticipated.

These criteria were applied to all alternatives considered for application at the Pittsburg Power Plant. All intake technologies that were considered to be available and proved for application at the plant (Criterion 1) were given a biological evaluation (Criterion 2). Feasibility analyses (Criteria 3 and 4) were carried out for alternatives that would reduce biological losses. The process is illustrated in Figure 6-1. The chapter ends with a discussion of and judgment as to the best intake technology available for the Units 1-7 cooling water systems at the Pittsburg Power Plant.

Intake technologies are reviewed in Appendix D. Evaluation of whether or not an intake technology is available and will minimize impacts requires site-specific analyses which are presented in this chapter. The existing design and operation of the cooling water systems of Units 1-7 are described in Chapter 2, along with a discussion of the physical and biological characteristics of the source water body.

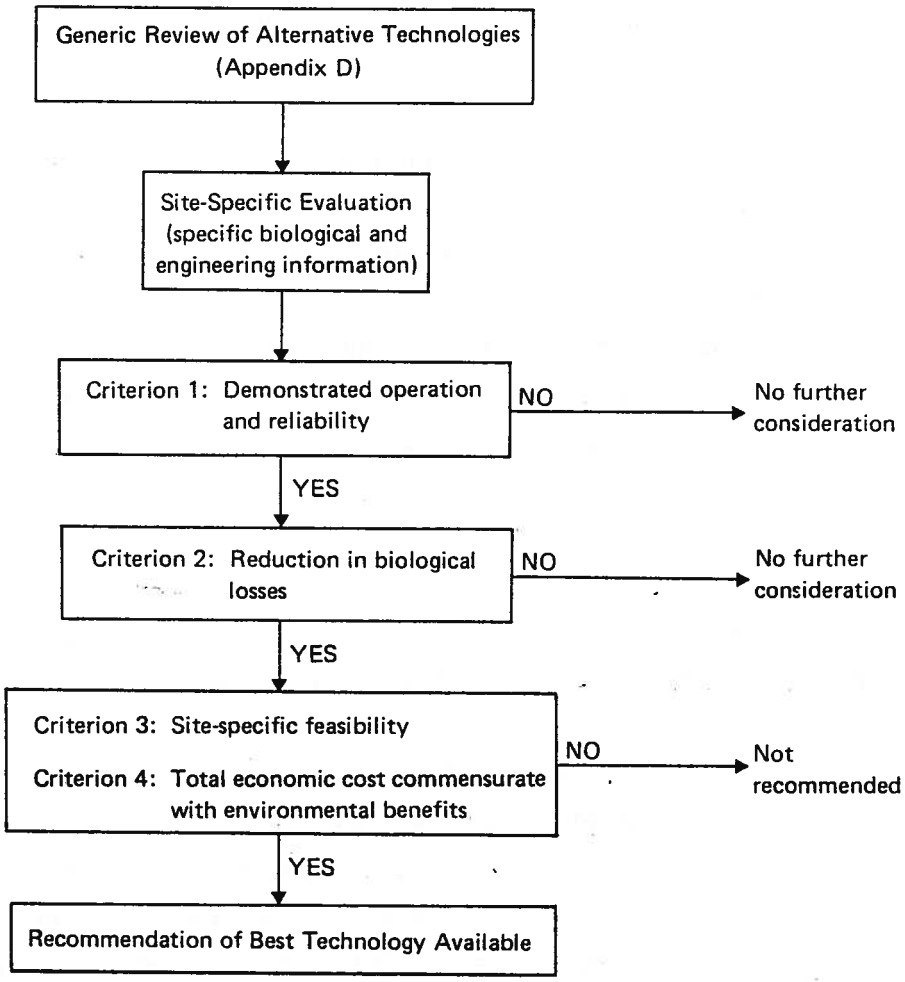

Figure 6-1. Schematic representation of the procedure used to evaluate alternative intake technologies for the Pittsburg Power Plant.

Chapters 3 and 4 present estimates and interpretations of entrainment and impingement necessary to assess the magnitude and significance of biological losses that are discussed in Chapter 5. Data from Chapters 2, 3, 4, and 5 provide the site-specific framework for evaluating the engineering feasibility and potential biological effectiveness of each intake technology considered.

6.1 DEMONSTRATED OPERATION CRITERION

Intake technologies that were determined to be proven and available for consideration at the Pittsburg Power Plant (Table 6-1) include offshore, onshore, and recessed intake locations and configurations, once-through and closed-cycle cooling water systems, light, sound, bubble screens, and velocity caps. Drum screens, center-flow screens, and vertical traveling screens were also determined to be appropriate for further consideration. Fish return systems, including sluiceways, fish pump systems, and vertical traveling screen modifications, were considered to be proven and available technologies, as were operational modifications such as curtailment and circulating water pump volume reduction. A brief discussion of the alternatives presented in Appendix D that failed to satisfy the first evaluation criterion, and hence are not considered further in the analysis, is presented in Section 6.1.1 and 6.1.2.

6.1.1 Behavioral Barriers

Velocity gradients, magnetic and chemical barriers, and chain and cable barriers have not been used on power plant intake structures, and are considered to be in an experimental stage of development. Electric barriers have been used with limited success in fresh water, but because of low electrical resistance, no application of electric fish barriers has been made in salt or brackish waters. Louvers have been used effectively at several large agricultural water diversions, but the application of louvers to power plant cooling water intakes has not been attempted. Louvers do not provide a positive barrier either to entrained organisms or to debris that could block the condenser tube system and lead to reduced operating reliability and increased maintenance. Velocity gradients, electric barriers, louvers, magnetic barriers, chemicals, and chain and cable barriers are not considered to be available and proven technologies for the Pittsburg Power Plant.

TABLE 6-1 OPERATIONAL FEASIBILITY OF INTAKE TECHNOLOGIES AND OPERATIONAL ALTERNATIVES CONSIDERED FOR THE PITTSBURG POWER PLANT

Intake Technologies	Operation Demonstrated	Operation Not Demonstrated
Cooling water system	Once-through Closed-cycle	
Intake location	Offshore Onshore	
Intake configuration	Shoreline Recessed	
Behavioral barrier	Light Sound Bubble screen Velocity cap	Velocity gradient Electrical barrier Louvers Chemicals Magnetic field Chains and cables
Physical barrier	Drum screen Centerflow screen Vertical traveling screen	Media filter Porous dike Radial well Stationary screen Horizontal traveling screen Angled screen - louver Caisson
Fish collection, removal, and conveyance systems	Vertical traveling screen modifications Gravity sluiceway Fish pump	
Operational Alternatives		
Maintenance and operational modifications	Dredging Circulating water pump volume reduction Curtailment Alternative biofouling control Cooling system modifications Temperature regulation	

6.1.2 Physical Barriers

Media filters such as rapid sand filters, porous dikes, and radial well intakes have had limited application in providing power plant cooling water. Prototype tests have been conducted that have identified debris accumulation and sedimentation as two major constraints in the application of media filters. In the absence of demonstrated performance capabilities and operational reliability in power plant water cooling systems, media filters are not considered to be an available technology for the Pittsburg Power Plant.

Stationary screens, such as perforated plate and pipe systems and cylindrical wedge-wire screens, were also eliminated from consideration. Fixed-screen intake systems do not have either demonstrated performance capability or operational reliability in cooling system applications where detrital loads are high. Accumulation of debris has been identified as a factor that would substantially decrease the operational reliability of fixed-screen intake structures sited in the Sacramento - San Joaquin Delta.

Horizontal traveling screens were not considered to be an available technology for the Pittsburg Power Plant because of a lack of demonstrated operational reliability (Farr and Prentice 1974).

An angled screen - louver intake structure and a caisson intake design were not considered to be available technologies for the Pittsburg Power Plant because of a lack of demonstrated performance. One U.S. utility has an angled traveling screen - louver intake structure under construction, but no data are currently available for either the operating reliability or the biological effectiveness. A caisson intake design has been proposed for use at the marine site of a liquified natural gas terminal (California Public Utilities Commission 1978), but construction has not yet begun.

6.2 BIOLOGICAL BENEFIT CRITERION

Each technology that satisfied the demonstrated operation criterion (Table 6-1) was evaluated to determine whether or not it would reduce the entrainment and impingement cropping reported in Chapters 3 and 4. Results of the evaluation are summarized in the following sections.

6.2.1 Cooling Water Systems

Two alternative cooling water system designs for the existing plant were considered—once-through and closed-cycle cooling. Closed-cycle cooling requires lower volumes of cooling water from the source water body and has therefore been suggested as a means of reducing impingement and entrainment rates. Use of closed-cycle cooling, however, results in 100 percent mortality of entrained organisms, in contrast to relatively low mortality rates reported for many species entrained into once-through cooling systems (Chapter 3; Appendix B). Impingement of fish and invertebrates would be lower for a closed-cycle cooling system, because once-through cooling requires higher water flows. Annual estimates of the number of larval striped bass and neomysis removed from the source water body by entrainment into a once-through cooling system and a closed-cycle cooling system were used for evaluating the biological benefit of the two alternative cooling water systems. Design criteria for the once-through cooling system include a flow rate of 24.9 m^3/sec (402,000 gpm; 878 cfs) at Units 1-4 and 20.2 m^3/sec (321,000 gpm; 715 cfs) at Units 5 and 6. Survival of entrained striped bass and neomysis passing through the once-through cooling systems was assumed to be 49 percent and 57 percent respectively, based on results of the studies conducted at the plant that are described in Chapter 3. Design criteria for a closed-cycle cooling system included flow rates of 1.14 m^3/sec (18,100 gpm; 40.3 cfs) at Units 1-4 and 1.11 m^3/sec (17,600 gpm; 39.2 cfs) at Units 5 and 6. It was assumed that there would be no survival of entrained organisms. Results of the cooling system comparison (Table 6-2) confirm the conclusion that use of a closed-cycle cooling system would be biologically advantageous at the Pittsburg Power Plant Units 1-6.

The closed-cycle cooling system in use at Unit 7 represents an acceptable technology, since implementation of a once - through cooling system will not result in a reduction in the loss of aquatic organisms. Therefore, no further consideration is given to alternative cooling water systems for Unit 7.

6.2.2 Intake Location

Alternative intake locations for the Pittsburg Power Plant include submerged offshore and shoreline locations. The existing shoreline intake location and configuration is the standard technology against which each alternative is compared.

TABLE 6-2 COMPARISON[a] OF THE ESTIMATED NUMBERS OF STRIPED BASS AND *NEOMYSIS MERCEDIS* CROPPED AS A RESULT OF ENTRAINMENT AT THE PITTSBURG POWER PLANT UNDER DESIGN CONDITIONS FOR ONCE-THROUGH AND CLOSED-CYCLE COOLING SYSTEMS

Model Parameter	Units 1-4		Units 5 and 6	
	Once-through	Closed-cycle	Once-through	Closed-cycle
Cooling water flow rate (m^3/sec)	24.86	1.14	20.24	1.11
Proportion of organisms not surviving				
Striped bass larvae	0.51	1.0	0.51	1.0
Neomysis mercedis	0.43	1.0	0.43	1.0
Estimated number (millions) cropped[b]				
Striped bass larvae	119.56	10.76	63.78	5.59
Neomysis mercedis	2,796.31	289.2	2,299.24	238.70

(a) Unit 7, being closed-cycle design, was not included in the comparison.
(b) Estimates are based on organism densities in cooling water samples collected from March 1978 to March 1979 and thus constitute an annual estimate; see Chapter 3 for details.

The effectiveness of an offshore intake in reducing entrainment depends, to a large degree, on the existence of vertical stratification of entrainable organisms in the water column at the point of water withdrawal. In such a system, a reduction in entrainment is achieved by locating the submerged intake at a depth where the density of entrainable organisms is lower. Plankton surveys conducted near the Pittsburg Power Plant (PGandE unpublished b; EA 1981a) indicate that striped bass larvae are distributed in approximately equal density near the shoreline and the existing intake structure and in the main channel where an offshore intake structure would be located. Statistical analyses of striped bass densities at onshore and offshore sampling locations indicated that differences were not significant (mean density $0.08/m^3$ at the intake and $0.07/m^3$ in the channel; PGandE unpublished b). Using data on the distribution of striped bass throughout the Delta, EA (1981a) concluded that differences in the densities of 7-10 and 11-14 mm larvae were not significantly different between midchannel and onshore (shoal) sampling stations. Densities of larger larvae and juveniles were, however, higher near the shore (EA 1981a). Neomysis densities were substantially lower at onshore sampling stations than at midchannel stations (PGandE unpublished b; EA 1981a). At the midchannel sampling station, both striped bass and neomysis were collected in generally higher densities near the bottom (PGandE unpublished b; EA 1981a) where they would be susceptible to entrainment at a submerged offshore intake. There is no evidence from the distributional information available from plankton surveys conducted near the Pittsburg Power Plant (EA 1981a, b, c) that entrainment of striped bass would be reduced by use of an offshore intake structure located near midchannel, and the entrainment of neomysis would be increased by an offshore intake structure. Thus, a submerged intake system could not be expected to result in reduced entrainment rates at the Pittsburg Power Plant.

For reducing the number of impinged organisms, the effectiveness of a submerged offshore intake depends on locating the intake in an area where such impingeable organisms are not abundant. Many of the dominant groups of fish and invertebrates (e.g., young-of-the-year and yearling striped bass, yellowfin goby, catfish) are typically found in higher abundance in association with the offshore bottom habitat in the vicinity of the plant site (EA 1981d), and many of the typically pelagic fish species, such as threadfin shad, are commonly found in large schools which move through the water column, often concentrating near the bottom substrates during the daytime (EA unpublished data). In addition, submerged offshore intakes generally

have high approach velocities and utilize conduits within which fish can become entrapped. Thus, a submerged offshore intake system would probably result in higher rates of impingement than those at the existing shoreline intake. Therefore, an offshore intake location is not considered to be an acceptable alternative to the existing shoreline intake location of Pittsburg Power Plant.

The general similarity of the shorezone habitat throughout the area surrounding the Pittsburg Power Plant site suggests that entrainment and impingement would not be substantially different between available shoreline locations. The pattern of along-shore currents and mixing in the area supports the conclusion that the densities of organisms are similar throughout the local shore zone. Fishery collections indicate that the abundance and species composition of impingeable fishes are similar throughout the local shore zone surrounding the Pittsburg Power Plant (EA 1981d). No reasonable alternative shoreline intake location has been identified that would be advantageous for reducing entrainment or impingement cropping.

6.2.3 Intake Configuration

Two intake configurations considered were an approach channel (recessed) intake screen configuration and a shoreline intake screen configuration. The recessed screen consists of an intake conduit or channel leading from the point of water withdrawal to intake screens located inland. Fish can become trapped within the confined intake conduit, where approach velocities are high, and may become stressed or fatigued and eventually impinged. The shoreline intake configuration puts the intake screens on or near the shoreline and offers the possibility of greater behavioral avoidance and reduced entrapment of organisms.

Approach channel (recessed) intake configurations are used to provide a protected area for intake screens, to separate the intake and discharge locations in order to minimize thermal recirculation, or for other engineering concerns. Because of fish entrapment within an approach channel intake, these should usually be avoided whenever an alternative intake configuration can be used, and they are not considered to be an acceptable alternative for use at the Pittsburg Power Plant.

The existing shoreline intake structure, incorporating fish escape channels and the low approach velocity to the bar racks (design velocity 0.5 fps; 15 cm/sec) constitute

the most effective intake configuration available for minimizing the impingement of fish and invertebrates.

The shoreline intake configuration promotes greater behavioral avoidance and reduced entrapment of impingeable fish than is generally possible with recessed intakes. The number of larval fish and invertebrates entrained, on the other hand, is independent of intake configuration, because they are planktonic.

6.2.4 Behavioral Barriers

Light, sound, and air bubbles were evaluated to determine their potential effectiveness for reducing biological losses at the Pittsburg Power Plant. The velocity cap was not considered, since its applicability is restricted to offshore intakes, which were not considered for use at the plant (Section 6.2.2).

Light, sound, or air bubble screens have not proven successful in reducing the numbers of fish impinged at power plant intakes (Appendix D). In addition, such barriers are not expected to reduce the numbers of entrained organisms or the impingement rates of macroinvertebrates. Behavioral barriers are not considered to represent an effective alternative for reducing entrainment or impingement at the Pittsburg Power Plant.

6.2.5 Physical Barriers

The applicability of drum screens, centerflow screens, and vertical traveling screens for reducing biological losses associated with entrainment and impingement at the Pittsburg Power Plant is evaluated in the following sections.

6.2.5.1 Drum Screens

Drums screens are not considered to be biologically effective, based on the limited data available (Eicher 1974), and are not expected to reduce the numbers of organisms entrained or impinged at the Pittsburg Power Plant cooling water intake structure. There is no information available to suggest that survival of organisms impinged on drum screens would be substantially different from impingement survival on conventional vertical traveling screens. In the absence of any predicted biological

advantages, drum screens are not considered to be an acceptable alternative intake technology applicable to the Pittsburg Power Plant.

6.2.5.2 Centerflow Traveling Screens

The centerflow traveling screen represents a relatively new intake screening technology in the United States, although it has been utilized extensively in European industrial and electric generating facilities for almost three decades. Centerflow screens (Figure D-8, Appendix D) are oriented parallel to the approaching water flow, and water enters through the center of the screen and exits through both vertical faces. Screen panels on centerflow screens are attached to a continuous drive chain and, unlike conventional vertical traveling screens, each screen panel forms a concave basket, increasing available screening surface area. Screen mesh sizes range from 0.02 to 0.4 in. (0.5-9.5 mm), with many units having 0.04- to 0.1-in. (1.0- to 2.5-mm) mesh. As a result of potentially rapid debris accumulation on the small mesh commonly used in centerflow screens, they are designed to operate with continuous screen rotation.

The hydraulic flow patterns associated with centerflow screens are more complex than those associated with vertical traveling screens. The water flows into the screen chamber through a keyhole-shaped opening in the front wall of the intake screen structure and turns 90 degrees to complete its transit to the circulating water pump. Effective screening occurs on both the ascending and descending sides of centerflow screens, increasing the available screen surface area. As a result, the average through-screen velocity of the centerflow screen is less than the approach velocity to the screen. However, through-screen velocities are highly variable across the screen face because of flow patterns and screen orientation, and velocities tend to increase with distance downstream from the chamber entrance. The turbulent effect of the flow may negate potential biological benefits from reduced average through-screen velocities. The high velocities at the entrance to the screen may also serve as a behavioral barrier, preventing the escape of fish entrapped within the screen structure.

Centerflow traveling screens are not expected to offer any major biological advantages over conventional or modified vertical traveling screens (Cada et al. 1979). The number of organisms impinged is not expected to be lower on centerflow

screens with 3/8-in. (0.9-cm) mesh, and significantly increased impingement survival has not been satisfactorily demonstrated for these systems.

Irregular velocity patterns, turbulence, and attendant head losses are inherent with the waterflow patterns for the centerflow screen. The highest water velocities occur in the entrance area of centerflow screens, and these could contribute to fish entrapment. Within the screen chamber, velocity varies irregularly as it changes direction, resulting in nonuniform flow through the screening surface.

Several alternative design configurations for the centerflow screen concept have been proposed in an attempt to minimize the probability of organisms becoming entrapped within the screen structure (see Appendix D). In addition to the single-entrance/dual-exit design, there is a double-entrance/single-exit design in which water enters the screen from two sides and passes out through one end. This design can be located either within a screenwell or supported from a platform with no confining concrete housing. A double-entrance/double-exit screen design has also been proposed. To date, biological and engineering data are not available to compare the relative effectiveness of these alternative configurations.

Fine-mesh centerflow screens are being investigated in laboratory studies to determine their potential effectiveness for protecting larval fish at power plant intakes (Magliente et al. 1978). Field and laboratory studies are required to evaluate the initial and long-term survival of eggs and larval fish impinged on fine-mesh centerflow screens. Insufficient data prevent a detailed comparison of the potential survival of egg and larval fish impinged on modified vertical traveling screens (fine-mesh screen material, fish buckets, low-pressure spraywash, continuous rotation) and centerflow screens (fine-mesh screen material, continuous rotation). To date, no studies have been conducted of long-term survival of fish impinged on centerflow screens operated in a power plant cooling water intake.

Information available from laboratory tests indicates that centerflow traveling screens equipped with 1.0-mm (0.04-in.) mesh would minimize the entrainment of fish eggs and larval fish at the Pittsburg Power Plant and that entrainment of larval fish can be virtually eliminated by use of 0.5-mm (0.02-in.) intake screen mesh (Tomljanovich et al. 1978). However, survival of larval fish impinged on fine-mesh intake screens may be no higher, or even lower, than the survival of larval fish

passing through a cooling water system (Cada et al. 1979). In addition, survival of larval fish impinged on a fine-mesh intake screen that is operated in an intermittent screenwash mode or is subject to high detrital loads (such as those that frequently occur in the Delta) may be extremely low (Tomljanovich et al. 1977).

No specific information concerning impingement survival of such ichthyoplankton as smelt, prickly sculpin, Pacific herring, northern anchovy, threadfin shad, or striped bass impinged on fine-mesh centerflow screens is available for comparison with the survival of entrained larvae reported in Section 3.2. Impingement survival of smelt, herring, and anchovy is expected to be similar to that of Clupeiformes impinged on fine-mesh vertical traveling screens (<1 percent) (Table 6-3). Survival of impinged threadfin shad larvae under laboratory conditions has been estimated to be 20 percent (Table 6-4), but this estimate does not include exposure to debris during impingement or stress related to transit through a fish return system. Even so, survival of larval striped bass 48 hours after impingement for 16 minutes was less than 1 percent (Table 6-4).

The most comprehensive studies to date on the survival of larval striped bass impinged on fine-mesh screens have been conducted by the Alden Research Laboratory and Stone & Webster Engineering Corporation (1980). A series of laboratory flume studies were performed using a total of 1,288 yolk-sac larvae (5.4-6.3 mm) and 10,086 larvae (6.4-17 mm). The larval bass were impinged on fine-mesh (0.35- and 0.5-mm) polyester screens at approach velocities ranging from 0.5 to 3.0 fps (15-90 cm/sec) for durations ranging from 2 to 16 minutes. Impingement mortality 96 hours after testing ranged from 51.2 to 100 percent for yolk-sac larvae (Table 6-5). Mortality increased both with increasing approach velocity and with increasing impingement duration. Although fewer test conditions were examined with larval bass, it appears from Table 6-5 that larger larvae had lower mortality rates than smaller larvae.

Impingement mortality of prolarval striped bass (yolk-sac larvae) (Table 6-5) was in excess of 90 percent following impingement for eight minutes at all velocities. These results reflect only mortality associated with impingement on fine-mesh screens under controlled laboratory conditions. Larval striped bass impinged on fine-mesh centerflow screens at the Pittsburg Power Plant would be subjected to debris loading during impingement, turbulence and irregular water flow patterns approaching and

TABLE 6-3 INITIAL AND 96-HOUR SURVIVAL OF LARVAL AND JUVENILE CLUPEIFORMES IMPINGED ON A CONTINUOUSLY ROTATING FINE-MESH VERTICAL TRAVELING SCREEN AT THE INDIAN POINT POWER PLANT

Taxon		Length (mm)	Mean Number Collected	Impingement Survival (%)	
Common Name	Scientific Name			Initial	96-hour[a]
Blueback herring	*Alosa aestivalis*	48	234	77	1
American Shad	*Alosa sapidissima*	55	23	87	0
Alewife	*Alosa pseudoharengus*	60	7	86	0
	Alosa spp.	8	62	0	0
Bay anchovy	*Anchoa mitchilli*	18	65	0	0

(a) 96-hour survival (%) =

$$\frac{\text{Number live + stunned at the end of 96 hours}}{\text{Initial number live + stunned tested for latent effects}} \times 100.$$

Source: Ecological Analysts 1979.

TABLE 6-4 MEAN SURVIVAL (%) OF LARVAL FISH IMPINGED ON A SIMULATED FINE-MESH CENTERFLOW SCREEN UNDER LABORATORY CONDITIONS

Taxon		Impingement Survival[a]		Control Survival
Common Name	Scientific Name	Initial	48-hour	48-hour
Jewelfish	*Hemichromis bimaculatus*	43	30	100
Threadfin shad	*Dorosoma petenense*	55	20	98
Golden shiner/ Fathead minnow	*Notemigonus crysoleucas*/ *Pimephales promelas*	89	79	99
White sucker	*Catostomus commersoni*	74	36	100
Channel catfish	*Ictalurus punctatus*	97	88	99.5
Striped bass	*Morone saxatilis*	7	1	44
Bluegill	*Lepomis macrochirus*	96	96	99
Smallmouth bass	*Micropterus dolomieui*	100	96	100
Largemouth bass	*Micropterus salmoides*	70	52	93
Walleye	*Stizostedion vitreum*	9	2	71

(a) Duration of impingement 16 minutes; screen mesh sizes 0.5, 1.0, 1.3, 1.8 and 2.5 mm combined; test velocities 15, 31, and 46 cm/sec combined.

Source: Tomljanovich et al. 1978.

TABLE 6-5 SUMMARY OF 96-HOUR MEAN MORTALITY (%) FOR STRIPED BASS IMPINGED ON 0.35- TO 0.5-mm SCREEN UNDER LABORATORY CONDITIONS

Life Stage	Velocity[a] (fps)	Duration of Impingement (min.) 2	4	8	16
Prolarvae[b]	0.5	74.2	51.2	91.8	84.6
	1.0	63.4	62.9	90.7	96.0
	1.5	70.7	92.0	98.7	100.0
	2.0	97.3	100.0	100.0	100.0
	3.0	—	—	—	—
Postlarvae[b]	0.5	—	—	—	43.9
	1.0	—	—	—	58.9
	1.5	—	—	—	97.6
	2.0	—	—	—	—
	3.0	18.4	49.1	—	—

(a) Average approach velocity to the screen.
(b) A total of 1,288 prolarvae (length 5.4-6.3 mm) and 10,086 postlarvae (6.4-17.1 mm) were tested.

Note: (—) indicates no test.

Source: Alden Research Laboratories and Stone & Webster Engineering Corporation 1980.

passing through the intake screens, exposure to a pressure spraywash, and abrasion during transit through a fish return sluiceway. Based on the high mortality of larvae impinged under laboratory conditions and the severity of additional stress (especially high debris loading by peat and filamentous algae common at the Pittsburg Power Plant cooling water intakes) expected to occur under actual operating conditions in the Delta, the survival of impinged larval striped bass is expected to be negligible. In comparison, the 96-hour survival rates for striped bass larvae following entrainment through the once-through cooling systems at the Pittsburg Power Plant were 49 percent and 45 percent in 1978 and 1979. Similar results were obtained for entrained neomysis: 57 percent entrainment survival in 1978 and 56 percent in 1979. The proportion of gammaridean amphipods surviving entrainment was 61 percent. In the absence of a demonstrated potential for long-term survival for impinged striped bass larvae and other ichthyoplankton and invertebrates in excess of the survival of entrained larvae (Section 3.2), fine-mesh centerflow screens are not considered to represent an acceptable alternative intake technology for use at the Pittsburg Power Plant.

6.2.5.3 Vertical Traveling Screens

Conventional vertical traveling screens with 3/8-in. (0.9-cm) mesh are presently in operation at the Pittsburg Power Plant (Section 2.1) and are the standard technology against which each alternative screening technology considered in this analysis is compared.

6.2.6 Fish Collection, Removal, and Conveyance Systems

6.2.6.1 Vertical Traveling Screen Fish Buckets and Accessories

Several modifications to conventional vertical traveling screens have been considered in recent years in an attempt to increase their biological effectiveness. Such modifications to the intake screens typically referred to as Ristroph modified screens (Figure 6-2); have been aimed at reducing the mortality of impinged organisms through the following features:

1. watertight fish collection buckets along the base of each screen panel to prevent repeated impingement of organisms and to

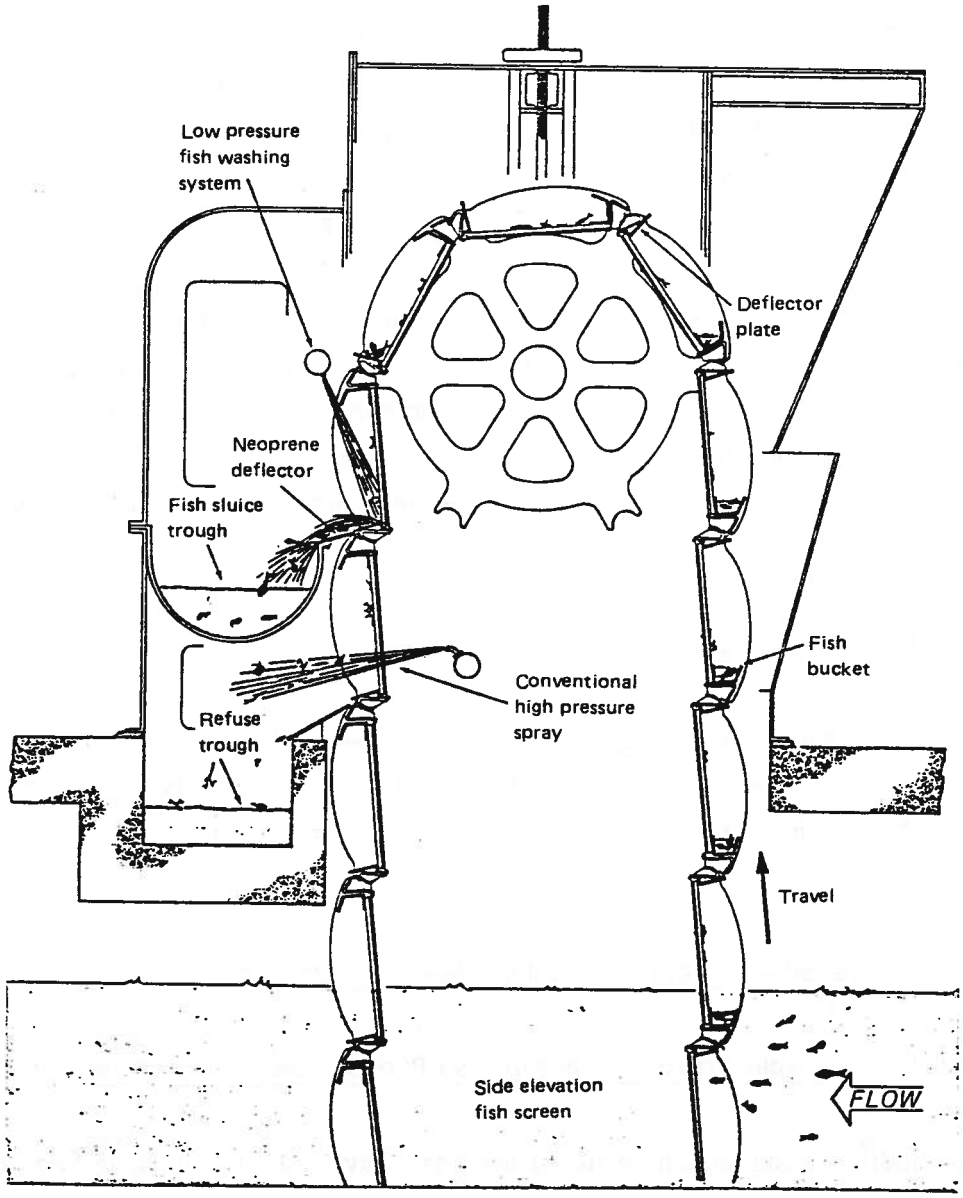

Figure 6-2. Ristroph modified traveling screen with fish buckets, a modified spray water system, and a sluiceway for fish. (Courtesy of Envirex.)

provide a holding area for organisms during screen rotation,

2. a low-pressure wash system to remove impinged organisms from the screen, reducing the stress and abrasion that results from exposure to the high-pressure spraywash required for the removal of debris,
3. a second sluiceway - fish return system to transport organisms removed from the screen by the low-pressure spraywash back to the receiving water body, and
4. heavy-duty bearings and motors to permit continuous rotation and cleaning, minimizing the time an organism is retained on the screen.

Limited information is available to assess the effectiveness of modified vertical screens for improving impingement survival for those species of fish expected to be impinged in greatest abundance at the Pittsburg Power Plant (Chapter 4). Information is available on the effectiveness of various screen rotation frequencies from studies conducted at the plant (Section 4.2) and information is also available for chinook salmon impingement survival from the Columbia River (Page et al. 1976, 1978) and for striped bass from the Hudson River (EA 1979; Texas Instruments, Inc. 1977). Data from these and other studies are used on a generic basis to provide additional information useful in examining the potential effectiveness of modified vertical screens at the Pittsburg Power Plant.

Certain operational modifications to vertical traveling screens, such as continuous screen rotation and reduced spraywash pressures, have been identified as alternatives that may increase their biological effectiveness. Continuous screen rotation at the Pittsburg Power Plant improved impingement survival of juvenile striped bass, but does not appear to have improved long-term survival for either threadfin shad or longfin smelt (Table 4-6). Increasing the screen rotation frequency from a 3-hour interval to continuous rotation did not result in consistently improved impingement survival for invertebrates such as Palaemon macrodactylus, Crangon franciscorum, or Rhithropanopeus harrisii (Table 4-6).

King et al. (1978) compared the effects of different screen rotation frequencies and screenwash pressures on the survival of fish impinged on conventional vertical traveling screens at three power plants on the Hudson River, with results similar to

those at the Pittsburg Power Plant. Continuous screen rotation resulted in better impingement survival for young-of-the-year white perch than screenwash at either 2- or 4-hour intervals. Low-pressure screenwash sprays (50 psi) resulted in better initial and long-term (48-hour) survival of impinged white perch on several occasions, but the results of screenwash pressure studies were generally inconclusive.

The effectiveness of a low-pressure spraywash system in removing impinged fish from a modified vertical traveling screen was examined by Public Service Electric and Gas Company (1977, in Cannon et al. 1979) on modified screens installed at the Salem Nuclear Generating Station. The number of impinged fish washed into a fish return sluiceway by a low-pressure spraywash (15 psi) was compared to the number washed into a debris sluiceway by a high-pressure spraywash (100 psi). A total of 3,770 fish were collected, of which 23.3 percent were removed from modified traveling screens by the low-pressure spraywash. These results indicate that low-pressure spraywash systems may be ineffective in removing impinged fish from an intake screen. Additional development and evaluation of low-pressure spraywash systems would be required for intake screens at the Pittsburg Power Plant, which are subject to high detrital loading.

Impingement survival of juvenile chinook salmon on modified vertical traveling screens located on the Columbia River was reported by Page et al. (1976, 1978). Initial survival of impinged chinook salmon was 95.6 and 97.2 percent, respectively, in 1976 and 1977. Long-term (96-hour) survival of impinged chinook salmon was 94.8 percent. Juvenile chinook salmon, although present in the vicinity of the Pittsburg Power Plant, are not impinged in high numbers (Section 4.1).

At the Indian Point Generating Station on the Hudson River, Texas Instruments Incorporated (1977) examined the survival of young-of-the-year and yearling fish impinged on a modified vertical traveling screen (low-pressure spraywash, dual screenwash sluiceways, fish buckets) equipped with 0.1-in. (2.5-mm) nylon screen mesh. The vertical traveling screen was rotated continuously during the impingement survival studies. Most of the 3,619 fish collected were bay anchovy, blueback herring, and white perch. Mean initial survival of all species was 41 percent, and mean survival after 84 hours was 20 percent for all species. A summary of results for impingement survival of each species is presented in Table 6-6. Species-specific survival patterns are apparent, with high long-term survival for species such as white catfish, white

TABLE 6-6 INITIAL AND LONG-TERM IMPINGEMENT SURVIVAL OF YOUNG-OF-THE-YEAR AND YEARLING FISH IMPINGED ON A CONTINUOUSLY ROTATING FINE-MESH (2.5-mm) MODIFIED VERTICAL TRAVELING SCREEN

Taxon		Age Class[a]	Number Collected	Survival (%)	
Common name	Scientific Name			Initial	84-hr
Striped bass	*Morone saxatilis*	1	13	85	75
White perch	*Morone americana*	1	223	45	26
White perch	*Morone americana*	2	37	41	18
Atlantic tomcod	*Microgadus tomcod*	1	78	92	58
Blueback herring	*Alosa aestivalis*	1	509	93	19
Blueback herring	*Alosa aestivalis*	2	1	100	0
Bay anchovy	*Anchoa mitchilli*	1	2,415	25	1
Bay anchovy	*Anchoa mitchilli*	2	65	55	13
Alewife	*Alosa pseudoharengus*	1	107	69	6
American shad	*Alosa sapidissima*	1	4	100	0
Hogchoker	*Trinectes maculatus*	1	4	100	100
American eel	*Anguilla rostrata*	2	4	100	100
Banded killifish	*Fundulus diaphanus*	1	1	100	ND[b]
White catfish	*Ictalurus catus*	1	27	100	100
Spottail shiner	*Notropis hudsonius*	1	2	100	100
Sea lamprey	*Petromyzon marinus*	1	1	100	100
Largemouth bass	*Micropterus salmoides*	1	1	100	100
Yellow perch	*Perca flavescens*	1	1	100	100
Weakfish	*Cynoscion regalis*	1	3	67	67
Bluefish	*Pomatomus saltatrix*	1	4	100	0
Rainbow smelt	*Osmerus mordax*	1	20	10	0
Rainbow smelt	*Osmerus mordax*	2	4	25	0
Menhaden	*Brevoortia* spp.	1	1	100	ND
Gizzard shad	*Dorosoma cepedianus*	2	5	100	0
Unid. clupeids	Clupeidae	1	80	51	ND
Centrarchids	Centrarchidae	1	5	100	100
Northern pipefish	*Syngnathus fuscus*	1	1	0	ND

(a) Age class: 1 = young-of-the-year; 2 = yearling or older.
(b) ND = no data.

Source: Texas Instruments, Inc. 1977.

perch, and Atlantic tomcod and low survival for species such as bay anchovy and rainbow smelt.

Based on results of the impingement monitoring program and actual plant operation it was estimated that approximately 111,000 striped bass were impinged on the Pittsburg Power Plant intake screens between March 1978 and March 1979 (Table 4-1). Impingement survival studies indicate a potential reduction in the cropping of impinged striped bass, with long-term survival after impingement of 48 percent, 54 percent, and 68 percent for 3-hour, 1-hour, and continuous screen rotation (Table 4-6). Assuming that a sluiceway return system was in use which would effectively return impinged striped bass to the receiving water body with a minimum of stress, the number of impinged striped bass cropped could be reduced from 111,000 assuming no impingement survival to 58,000 assuming survival based on 3-hour intermittent screen rotation and to 36,000 with continuous screen rotation. On the basis of an estimated 75 percent survival of striped bass impinged on a continuously rotating modified intake screen (fish buckets, low-pressure spraywash, dual screenwash sluiceway) (Table 6-6), the cropping of impinged striped bass at the Pittsburg Power Plant from March 1978 to March 1979 could be reduced to approximately 28,000 fish. It appears from these estimates that although a substantial reduction in the cropping of impinged striped bass could be expected with the implementation of a continuously rotating modified screen, the greatest contribution to a reduction in impingement mortality can come from decreasing the duration of impingement by increasing screen rotation frequency. An effective fish return system, an integral part of a system for reducing mortality of impinged organisms, will be discussed in Section 6.2.6.2.

Vertical traveling screens have recently been modified to include fine-mesh screen material. Vertical traveling screens equipped with 0.5- to 1.0-mm (0.02- to 0.04-in.) screen mesh would substantially reduce, if not eliminate, the entrainment of most ichthyoplankton and macroinvertebrates at the Pittsburg Power Plant. As discussed in Section 6.2.5.2, the survival of larval striped bass impinged on fine-mesh intake screens that are subjected to the high detrital loading typical of the Delta is expected to be negligible. Data available from studies conducted by Tomljanovich et al. (1978), Ecological Analysts (1979), and Alden Research Laboratory and Stone & Webster Engineering Corporation (1980) indicate that impingement survival of larval striped bass increases as fish length increases. However, the information is insufficient to predict at what size class impingement survival on intake screens at the Pittsburg Power Plant might start to increase.

Long-term survival of striped bass entrained into the cooling systems of Units 1-7 was estimated to be 49 percent and 35 percent in the 1978 and 1979 spawning seasons, respectively (Section 3.2). In contrast, long-term survival of striped bass larvae following impingement on fine-mesh screens for 16 minutes under laboratory conditions was less than 1 percent (Tomljanovich et al 1978). Survival of impinged striped bass in extensive laboratory studies conducted by the Alden Research Laboratory and Stone & Webster Engineering Corporation (1980) ranged from 0 to 49 percent. Survival of entrained invertebrates was also high (long-term survival of neomysis was 57 percent and that of gammaridean amphipods 61 percent; Section 3.2), although no comparative data are available for impinged invertebrates. In the absence of a demonstrated potential for long-term survival of impinged striped bass, smelt, shad, herring, and anchovy larvae and other ichthyoplankton, it cannot be asserted that the use of fine-mesh vertical traveling screens would reduce the combined impact of entrainment and impingement at the Pittsburg Power Plant.

In summary, modified vertical traveling screens, including fish buckets, a low-pressure wash system, a fish return system, and provisions for continuous rotation, do represent an alternative technology with the potential for increasing impingement survival of juvenile and yearling striped bass and salmon. The modified screens would not, however, reduce losses for such species as shad, smelt, shrimp, and crabs. Fine-mesh vertical traveling screens are not considered to be effective for reducing the loss of larval fish.

6.2.6.2 Fish Return Conveyance Systems

There are two basic types of sluiceways for the return of impinged organisms and debris to the water body: one using a trash pump to transport material away from the intake and one using gravity flow. The pump-augmented return has the advantage of minimizing recirculation and reimpingement of debris on intake screens, but often results in mechanical abrasion and high mortality of organisms. The sluiceway return system in use at the Pittsburg Power Plant is pump-augmented, and it returns impinged material to the thermal discharge conduit.

The gravity sluiceway alternative was examined. On the basis of existing information, no quantitative estimate can be developed for the reduction in impingement cropping (Section 4.2) that might be achieved by a gravity-sluiceway return system.

Impingement survival of hardy species, such as striped bass, chinook salmon, catfish, Palaemon macrodactylus, Crangon franciscorum, and Rhithropanopeus harrisii, is expected to be moderately high, and thus a gravity-sluiceway return system would be effective in reducing the loss of these species. Impingement survival of fragile species such as threadfin shad, longfin smelt, and Delta smelt is low (Section 4.2), and a gravity-sluiceway return system would not contribute substantially to improved survival for these species.

As noted in Section 6.2.6.1, there is a potential for substantially reducing the cropping of impinged striped bass by taking advantage of impingement survival under the existing mode of 3-hour intermittent screen operation (48 percent survival) or under alternative operational modes (i.e. 1-hour intermittent or continuous rotation) during periods of peak striped bass impingement. For these reasons a gravity sluiceway return system will by given further consideration in Section 6.3.2.

The use of fish pumps as an alternative fish return conveyance system is not considered likely to reduce impingement cropping further at a shoreline intake structure (Section 6.2.3). The low design approach water velocity (0.5 fps; 15 cm/sec) and the presence of fish escape channels within the shoreline intake structure (Section 2.1) are expected to minimize fish entrapment. For these reasons the application of a fish pump return system at the Pittsburg Power Plant is not considered further.

6.2.7 Intake Maintenance and Operational Modifications

Operational modifications and maintenance which may reduce entrainment and impingement cropping include (1) maintenance dredging of the cooling water intakes, (2) circulating water pump volume reduction, (3) seasonal curtailment of cooling system operation, (4) use of alternative biofouling schemes, (5) cooling system structural modifications, and (6) through-plant temperature regulation.

6.2.7.1 Maintenance Dredging

Sediment accumulation within a cooling water intake structure may reduce the open area of the intake, resulting in increased water velocities, and these will, in many cases, result in increased rates of impingement.

Bathymetric measurement made in the intake structures of the Pittsburg Power Plant (PGandE 1981) indicated that there was an appreciable accumulation of sediment. In the Units 1-4 intake structure it had reduced the available cross-sectional area of the intake by an average of 61 percent. In the intake servicing Units 5 and 6, sediment accumulation had reduced the intake area by 26 percent at the east end of the intake structure but very little at the west end. Sediment accumulation within the intake structures had resulted in increased intake water velocities (Section 2.1; PGandE 1981). The reduction in the numbers of organisms impinged that would result from removing accumulated sediment and reducing intake velocities cannot be estimated on the basis of available data, but there is little doubt that maintenance dredging of the intakes reduces approach velocities and reduces the numbers of impinged organisms.

6.2.7.2 Circulating Water Pump Volume Reduction

It has been asked whether seasonal reduction in the number of circulating water pumps in operation would be an alternative operational strategy for reducing cooling water volumes and intake approach velocities, and hence reducing the numbers of organisms entrained and impinged. Increased temperature differentials through the condenser system (delta-T) and higher condenser back-pressures resulting in reduced turbine cycle thermal efficiency are to be expected when cooling water flow rates are reduced during generation. Although a reduction in cooling water volume would be expected to result in a decrease in the number of organisms entrained, the associated increase in delta-T might substantially reduce the survival of the organisms that are entrained.

The effectiveness of reducing entrainment cropping by reducing the volume of cooling water withdrawn has not been extensively examined. Quantitative information is needed on the relative contribution of mechanical stresses for each species and life stage being considered before any benefits from reduced cooling water volume can be estimated. Too little information is available to optimize the balance between cooling water volume and delta-T for most power plants and fish species. In the absence of a demonstrated potential for reducing the cropping of entrained organisms as a result of seasonal (long-term) circulating water pump regulation, this operational strategy is not considered to be an acceptable alternative for use at the Pittsburg Power Plant.

Short-term regulation of circulating water pump volume would be most effective biologically if cooling water volumes were reduced when the unit was out of service

for fuel conservation or maintenance. This operational alternative would result in reduced numbers of organisms being entrained while maintaining high survival of those which were entrained. When a unit is taken out of service at the Pittsburg Power Plant, either as a forced outage or as a fuel conservation outage, the circulating water pumps are shut down if the outage is of sufficient length and if reserve capacity is available to meet expected energy demands. Thus, short-term regulation of circulating water pump operation is already in effect at the Pittsburg Power Plant. Entrainment estimates under actual flows (Table 3-4) are approximately 30 percent less than those based on design (continuous normal) flows at Units 1-4 and Units 5 and 6, and 10 percent less at Unit 7.

The number of fish impinged is expected to be reduced by reducing circulating water pump operation. Impingement estimates computed on the basis of actual cooling system operation were 20 to 25 percent less than impingement estimates based on the assumption of continuous cooling system operation (Section 4.1.2).

Despite the lack of quantitative data, it is concluded that short-term reductions (hours or days) in the volume of cooling water coinciding with reduced generation are likely to reduce entrainment and impingement cropping.

6.2.7.3 Seasonal Curtailment

Seasonal curtailment of cooling system operation would reduce the numbers of organisms cropped as a result of entrainment and impingement. The amount of the reduction would depend on the length of time the cooling system is out of operation and the density of organisms during the period of curtailment. Based on entrainment and impingement estimates for the Pittsburg Power Plant (Chapters 3 and 4) two peak periods of abundance, May-June (ichthyoplankton entrainment) and July-August (striped bass impingement), were selected as possible periods for curtailment. The estimated reduction in the numbers of striped bass and neomysis entrained and striped bass impinged as a result of seasonal curtailment is described in Table 6-7. Seasonal curtailment of cooling system operation, if permissible, would result in a reduction in the numbers of organisms entrained and impinged and is therefore considered for evaluation as an alternative technology for cost, engineering, and operating review at the Pittsburg Power Plant.

TABLE 6-7 ESTIMATED REDUCTION (%) IN THE NUMBERS OF SELECTED ORGANISMS ENTRAINED OR IMPINGED ANNUALLY AT THE PITTSBURG POWER PLANT AS A RESULT OF SEASONAL CURTAILMENT IN COOLING SYSTEM OPERATION

Taxon		Period of Curtailment		
Common Name	Scientific Name	May-June	July-August	Combined May-June July-August
	Entrainment			
Striped bass	*Morone saxatilis*			
eggs		99	0	99
larvae		98	1	99
Neomysis	*Neomysis mercedis*	16	38	54
	Impingement			
Striped bass	*Morone saxatilis*	1	20	21

6.2.7.4 Alternative Biofouling Schemes

The biofouling control procedure currently used at the Pittsburg Power Plant includes intermittent chlorination (Chapter 2). The procedure is considered to be the minimum required for the control of bacterial and algal accumulation within the cooling system. The mortality of nontarget entrained organisms as a result of the present antibiofouling program is unknown. On the basis of the periods of exposure to chlorine encountered and the typical concentration maintained at the condenser inlet (0.2-0.5 mg/liter), current chlorination practices at the Pittsburg Power Plant should have a minimal effect on mortality of the organisms entrained at the plant. During most of the year each unit is chlorinated for approximately 30 minutes once a week, with chlorination frequency increasing to 2-3 times a week during periods of heavy algal accumulation (Section 2.1.2). In addition, each portion of the system is chlorinated in sequence, so dilution by cooling water from units not being chlorinated shortens exposure durations or reduces chlorine concentrations and thus reduces entrainment mortality. Intermittent chlorination for a short period each week, at the low dosage rates and the amount of dilution used, represents an acceptable biofouling procedure at the Pittsburg Power Plant.

6.2.7.5 Cooling System Modifications

Structural modification of cooling system components to reduce the mortality of entrained organisms is not effective at the present time. Insufficient quantitative data are available to isolate specific sources of mortality within cooling water systems. Design parameters for specifying pressure regimes, circulating water pump design and operation, tolerable shear stresses, and cooling system designs for minimizing mechanical abrasion have not been developed.

6.2.7.6 Discharge Temperature Regulation

Exposure to discharge temperatures below 30 C (86 F) for short periods during cooling system transit are generally considered to be nonlethal to most entrained organisms (Section 3.2; Appendix B). Discharge temperatures at the Pittsburg Power Plant were below 30 C (86 F) 67 percent of the time at Units 1-4, 70 percent of the time at Unit 5, and 63 percent of the time at Unit 6. During periods of peak striped bass entrainment (Figure 3-5), discharge temperatures exceeded 30 C less than 10 percent of the time

during late spring (April-May), but the frequency of higher discharge temperatures increased in early summer, with discharge temperatures exceeding 30 C (86 F) approximately 55 percent of the time in June and July. Therefore, thermal stresses are not expected to be a significant cause of mortality to entrained fish or invertebrates during most of the year. The higher discharge temperatures that occur in summer (primarily June and July) are expected to contribute to increased mortality of entrained larval fish such as striped bass. Discharge temperature regulation is expected to result in a reduction of the number of organisms cropped by entrainment through the Pittsburg Power Plant Units 1-6 cooling systems during June and July and is therefore considered in the feasibility analysis presented in Section 6.3. Since no survival is expected for organisms entrained through the closed-cycle cooling system of Unit 7, discharge temperature regulation would have no biological benefit at this unit.

6.2.8 Conclusions: Biological Effects

On the basis of the biological evaluation (Section 6.2), the following conclusions were reached:

1. the existing shoreline intake location and configuration is acceptable, and entrainment and impingement cropping would not be reduced by use of either an offshore or a recessed intake structure,
2. behavioral barriers are not expected to reduce the numbers of organisms exposed to either entrainment or impingement,
3. entrainment and impingement cropping would not be reduced by use of drum screens or centerflow traveling screens,
4. a screen mesh size of 3/8 in. (0.9 cm) is acceptable, since survival of ichthyoplankton (fish eggs and larvae) impinged on fine-mesh screens would not exceed the observed survival of organisms entrained through the Pittsburg Power Plant cooling system, and
5. alternative biofouling schemes and cooling system structural modifications are not expected to reduce the mortality of entrained organisms.

The following alternative intake technologies if implemented would reduce

entrainment and/or impingement cropping at the Pittsburg Power Plant and were therefore selected for a feasibility analysis:

1. installation of closed-cycle cooling at Units 1-6,
2. installation of modified vertical traveling screens and fish return systems,
3. dredging the intake area to reduce intake velocities,
4. short-term reduction in circulating water pump operation,
5. seasonal curtailment of cooling system operation, and
6. discharge temperature regulation during summer months.

6.3 FEASIBILITY ANALYSIS

Each alternative technology that satisfied the biological reduction criterion (Section 6.2) and differed in design or operation from that presently in use at the Pittsburg Power Plant is evaluated with regard to engineering feasibility, operation, and reliability. In addition, the total economic cost in 1980 dollars associated with each alternative is estimated. Cost estimates reflect direct capital costs and indirect costs (e.g., the loss of generating capacity) whenever possible.

6.3.1 Installation of Closed-cycle Cooling at Units 1-6

Closed-cycle cooling (cooling towers) would decrease the volume of water withdrawn from the source water body and thus reduce the number of organisms entrained (Section 6.2.1). The feasibility of retrofitting a power plant having a once-through cooling system with a closed-cycle system, however, requires extensive investigation. Since the cooling system is a major and integral part of the initial plant design, a proposal to change it requires examination of overlapping operational, environmental, and economic considerations. Operational considerations include an evaluation of thermal performance, operating requirements, electrical conductor contamination by deposition of cooling tower drift, plant reliability, and derating (loss of generating capacity). Although not discussed in this evaluation, environmental factors to be considered prior to installation of a closed-cycle cooling system center on air quality effects of cooling tower drift, terrestrial effects from salt deposition, land use requirements, and aesthetics. Economic factors include an evaluation of the capital costs for cooling system conversion, plant performance penalties, lower efficiency and loss of capacity, and increased operation and maintenance costs.

The capital costs used in computing the costs for retrofitting the Pittsburg Power Plant cooling system are based on price information provided by various cooling system equipment manufacturers. Costs of installation and maintenance and the evaluation of operating performance for the retrofit system are based, in large part, on information and experience gained in operation of the mechanical-draft cooling tower system of Unit 7 of the plant.

The closed-cycle cooling system design for Units 1-6 discussed below includes mechanical-draft cooling towers arranged in two independent systems—one system for Units 1-4 and a second system for Units 5 and 6. Although similar in design, the two cooling systems would permit independent operation of the two sets of units. Cooling water makeup flow rates for the closed-cycle cooling systems would be 1.14 m^3/sec (18,100 gpm; 40.3 cfs) for Units 1-4 and 1.11 m^3/sec (17,600 gpm; 39.2 cfs) for Units 5 and 6. The closed-cycle cooling systems are designed for operation with a concentration factor of 1.5 to prevent scaling in the cooling system.

Retrofitting a closed-cycle cooling system would require construction of four banks of mechanical draft cooling towers, two servicing Units 1-4 and two servicing Units 5 and 6. The cooling tower banks would be located adjacent to the Unit 7 cooling towers and catchment basin. The subsurface soil conditions at the site are expected to require that all structures associated with the cooling system modification have pilings as foundation support. Installation of the closed-cycle cooling systems would also require relocation of a 24-in. (60-cm) gas main now at the site. A new pressure conduit made of prestressed concrete and supported by pedestals or pile-founded bents at each joint would be required for transporting cooling water from the condenser system to the cooling towers. A detailed engineering evaluation would also be necessary, before final specification of the closed-cycle cooling system design criteria, to ensure compatibility between the hydraulic flow patterns, cooling water volumes, and pressure regimes of the new cooling systems and those of the existing condenser systems.

Table 6-8 summarizes the estimated direct costs for constructing a closed-cycle cooling system for Units 1-6. Direct capital cost of the closed-cycle cooling system was estimated to be $62.7 million (1980 dollars). The primary components contributing to the total capital cost were circulating water conduits, including cooling tower make-up and blowdown lines ($36.2 million); the mechanical-draft cooling towers and

TABE 6-8 ESTIMATED DIRECT COSTS[a] FOR CONSTRUCTING A CLOSED-CYCLE COOLING SYSTEM FOR UNITS 1-6 OF THE PITTSBURG POWER PLANT

Equipment	Installed Cost ($)
Mechanical draft cooling towers and basins	16,666,500
Circulating water pumps, including associated piping and valves	5,377,100
Pump house station	965,400
Circulating water conduits, including cooling tower makeup and blowdown lines	36,162,300
Forebay flumes	1,473,600
System electrical and control, including transmission	2,026,000
Collection pond	--
Total installed 1980 capital cost	62,670,900

(a) In 1980 dollars.

basins ($16.7 million); and the circulating water pumps, piping, and control valves ($5.4 million).

Table 6-9 indicates the effects that a mechanical-draft cooling tower system would have on plant performance compared to a once-through cooling system. Plant performance was calculated over a historical range of wet-bulb temperatures based on nine years of weather data collected at the Pittsburg Power Plant. The range of wet-bulb temperatures were weighted according to their frequency of occurrence, and a summer peak wet-bulb temperature of 70 F (21 C) (3.3 percent occurrence) was selected as the design condition to establish the plant's maximum loss in generating capacity. The mean annual change in plant performance due to the addition of the closed-cycle cooling tower system was calculated on the basis of a 60 F (16 C) annual average wet-bulb temperature (44.3 percent occurrence), as indicated in Table 6-10.

The loss of generating capacity at the Pittsburg Power Plant Units 1-6 that would result from both unit outage during cooling tower installation and reduced thermal performance during operation have a cost associated with replacement energy and capacity. However, no quantitative estimate of the cost of lost generation has been developed, because of the uncertainty of the time frame when cooling tower installation would occur and of the expected power demands and reserve capacity available at that time. In addition, fluctuating fuel costs, which are a major factor in the cost of replacement energy, make accurate projections of future energy costs impossible.

Table 6-11 includes indirect costs of $156,680,000 and an estimated maintenance cost of $150,000 per year in addition to the direct capital costs listed in Table 6-8. Indirect costs include the costs of engineering design, interest carrying costs, contingency costs, general overhead, and administrative costs. Estimated maintenance costs of the closed-cycle cooling system are approximately $125,000 per year for cooling tower maintenance and $25,000 per year for maintenance of the cooling tower pumps. These costs include required inspection, repair, and replacement of mechanical parts, refilling of cooling towers, and cleaning of basins.

The installation of a mechanical-draft closed-cycle cooling system at Pittsburg Power Plant Units 1-4 and Units 5 and 6 is possible. The loss of generating capacity, however, both during installation of the modified cooling system and during operation,

TABLE 6-9 EFFECTS OF MECHANICAL-DRAFT WET COOLING TOWERS ON SYSTEM PARAMETERS AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT AT MAXIMUM EXPECTED COOLING WATER TEMPERATURES AND MAXIMUM LOAD

Parameter	Units 1-4		Units 5 and 6	
	Once-through	Closed cycle	Once-through	Closed cycle
Expected wet-bulb temperature (F)	—	70	—	70
Cooling water temperature at condensers (F) [a]	74	81	74	81
Temperature rise across condensers (F)	14.98	15.39	18.26	18.62
Turbine back pressure (psi)	1.00	1.24	1.04	1.26
Gross generation (kW)	654,800	642,200	664,600	658,700
Reduction in gross generation (kW)	—	12,600	—	5,900
Increase in auxiliary power for cooling towers (kW)	—	8,200	—	7,300
Loss in available generation (kW)	—	20,800	—	13,200

(a) Highest water temperature expected for once-through; includes expected approach to wet-bulb temperature (11 F) for closed-cycle.

Note: (—) indicates value not relevant or zero.

TABLE 6-10 EFFECTS OF MECHNICAL-DRAFT WET COOLING TOWERS ON SYSTEM PARAMETERS AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT AT THE EXPECTED ANNUAL AVERAGE TEMPERATURE AND MAXIMUM LOAD

Parameter	Units 1-4		Units 5 and 6	
	Once-through	Closed cycle	Once-through	Closed cycle
Annual average wet bulb temperature (F)	--	50	—	50
Average cooling water temperature at condensers (F) [a]	60	67.78	60	66.69
Temperature rise across condensers (F)	15.00	15.05	18.25	18.34
Turbine back pressure (psi)	0.63	0.85	0.74	0.87
Gross generation (kW)	666,200	660,000	673,200	670,200
Net generation (kW)	638,500	624,100	654,200	643,800
Net heat rate (Btu/kWh)	9,520	9,740	9,430	9,580
Change in heat rate (Btu/kWh)	—	220	—	152

(a) Annual mean for once-through; includes expected approach to wet-bulb temperature for closed-cycle.

Note: (—) indicates value not relevant or zero.

TABLE 6-11 PERFORMANCE PENALTIES AND ESTIMATED COSTS[a] FOR CLOSED-CYCLE CONDENSER COOLING AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT

Parameter	Units 1-4	Units 5 and 6
Maximum expected capacity loss (kW)	20,800	13,200
Increase in unit heat rate (Btu/kWh)[b]	220	152
Installed capital cost	$62,670,000	
Indirect costs[c]	$156,680,000	
Total	$219,350,000	
Annual maintenance cost	$150,000	

(a) In 1980 dollars.
(b) Mean annual change in unit heat rate due to addition of closed-cycle cooling towers; each unit.
(c) Does not include the energy replacement cost for unit outages during installation or for reduced generating capacity during operation.

because of reduced unit generating efficiency, would reduce the reliability of electrical service during periods of peak demand. The estimated direct and indirect costs of modifying the existing once-through cooling systems at the Pittsburg Power Plant to include closed-cycle cooling is estimated to be $219,350,000 (Table 6-11). In addition, an annual cost of approximately $150,000 is anticipated for additional maintenance of the closed-cycle cooling system. Energy and capacity replacement costs associated with reduced generation for the duration of commercial operation of the units and replacement energy and capacity during cooling tower installation have not been included in the cost estimates.

6.3.2 Modified Vertical Traveling Screen and Fish Return System

The mortality rate of fish impinged on intake screens can, for some species, be reduced through structural modifications to conventional vertical traveling screens and a change in intake screen operation from intermittent to continuous rotation (see Section 6.2.6.1). The structural modifications include installation of a watertight fish collection basket along the base of each screen panel, neoprene deflectors and seals between fish baskets and the screen, both low-pressure and high-pressure wash systems, and a fish return sluiceway. In addition, a number of components, such as foot shaft bushings, main carrier chains, boot plates, main frame roller guides, head shaft roller bearings, and reducer drives must be upgraded for screens to operate continuously with acceptable performance. A differential control and two-speed motor are also recommended, so that when the screen is operated continuously it rotates at slow speeds, and as fish and/or debris loads increase, the screen rotation rate can be automatically increased. In general, 3/8-in. (0.9-cm) screen mesh is used on modified vertical traveling screens.

Screens modified to reduce impingement mortality need to accompanied by a sluiceway (Section 6.2.6.2) designed to return impinged organisms to the receiving water body. Most installations of modified traveling screens use a dual sluiceway return system: a gravity-sluiceway return system for impinged organisms removed from the screens by the low-pressure spraywash and another sluiceway for debris removed by the high-pressure spraywash (Figure 6-2). Basic criteria for the design of fish return conveyance systems are discussed in Appendix D (Section D.5.3.2).

Both conventional and modified vertical traveling screens are available commercially

from several intake screen manufacturers. The installation procedures are similar for both conventional and modified screens. At the Pittsburg Power Plant, an existing installation, the additional capital costs of a modified screen are only those associated with the structural modifications to the existing conventional screens and the costs of the additional fish return sluiceway. A judgement would have to be made as to whether the condition of the existing screens warrants such modification or whether new screens which had already been modified should be purchased. In either case, replacement could be scheduled to be done during routine maintenance of the present screens to avoid an additional unit outage.

Modification of the existing conventional intake screens for fish handling (fish baskets, deflectors, dual spray wash system, etc.) would have a capital cost of approximately $21,500 per screen. The estimated cost for upgrading motor drive components and switching gear for continuous screen rotation would further increase the cost per screen by approximately $18,000. The total estimated capital cost of modifying the thirteen vertical traveling screens at the Pittsburg Power Plant is $722,000 (1980 dollars), compared to an estimated capital cost of $1.2 million for purchase of new modified vertical traveling screens (Table 6-12). There are additional capital costs associated with the incorporation of a dual-spraywash system consisting of a low-pressure spray to remove impinged fish and macroinvertebrates, followed by the conventional high-pressure spray for debris. Installation of the dual-spray system would require modification and reconstruction of a portion of the existing intake to include necessary spraywash pumps, valves, piping, and electrical controls. A gravity sluiceway collector for the low-pressure spraywash system to return impinged organisms to the water body would also need to be designed and constructed. The additional cost of the dual-wash system and a gravity sluiceway would be about $300,000. The total estimated capital cost of modifications to the intake screens at the Pittsburg Power Plant is about $1.2 million - $1.5 million (1980 dollars). Additional indirect costs associated with engineering design, specific engineering feasibility studies, accounting, administrative, and interest costs have not been included in the estimate.

The operating and maintenance (O&M) costs of the modified vertical traveling screens depend primarily on the frequency of screen rotation. Therefore, O&M costs of a modified screen operated periodically in a continuous rotation mode will be higher than costs of the existing conventional screens operated intermittently. The increased costs of maintenance, parts replacement, and associated increases in operation and

TABLE 6-12 SUMMARY OF ESTIMATED COSTS[a] OF CONVERTING CONVENTIONAL VERTICAL TRAVELING INTAKE SCREEN SYSTEMS FOR THE PITTSBURG POWER PLANT TO INCLUDE FISH HANDLING FACILITIES

	Modification of Existing Screens	Installation of New Screens
Base price per screen	—	$48,000
Fish buckets, deflectors, dual spraywash system	$21,500	$21,500
Upgrading for continuous screen rotation	$18,000	$18,000
Installation labor	$16,000	$8,000
Cost per screen	$55,500	$95,500
Total estimated capital cost for replacement of the 13 existing screens	$721,500	$1,241,500

(a) Average manufacturer's suggested price in 1980 dollars.

maintenance labor for modified traveling screens are not available. PEDCo. Environmental, Inc. (1977, cited in Cada et al. 1979) estimated that the additional costs of auxiliary power to operate modified screens and screenwash pumps continuously would be approximately $18,000 (1976 dollars) per year per screen. Although not quantified, the total O&M costs of modified vertical traveling screens used at the Pittsburg Power Plant will exceed substantially those of a conventional vertical traveling screen system.

Modifications designed to reduce impingement mortality like those specified above have been retrofitted to existing intake structures and are presently in operation at the Surry Power Station, Salem Generating Station, and Hanford Generating Project. No design features of the Pittsburg Power Plant have been identified that would preclude application of this alternative screening system. These modifications would not alter the intake flow significantly, so no significant effect on the operation of the circulating-water system would be expected. Because conventional intake screens with such modifications do not have an extensive operating history, predictions of system reliability cannot be made with certainty. One primary concern is the increased maintenance schedule required for continuous operation over extended periods and the effect of high detrital loads experienced at the Pittsburg site on screen reliability. The degree of clogging by impinged debris (primarily peat and filamentous algae) and the way in which it would affect system reliability cannot be determined from available data for modified screens operated in the Bay-Delta environment. Although modifications to the existing intake screens at the Pittsburg Power Plant appear to be possible, some system development after making these modifications would be required to ensure reliable performance.

6.3.3 Dredging the Intake Area

Sediment deposition during normal operation of cooling water systems at the Pittsburg Power Plant in the area of the intake bar racks decreased the open area of the intake and resulted in increased intake approach velocities (Section 2.1). Dredging the intake area is expected to result in a reduction in the numbers of fish impinged and will also increase the operational efficiency of the circulating water pumps.

PGandE contracted for the dredging and removal of accumulated sediment from the Pittsburg Power Plant intake. Dredging was completed in December 1980 at a total

cost of approximately $200,000, and has reduced intake approach velocities (PGandE unpublished a).

6.3.4 Reduction in Circulating Water Pump Operation

A reduction in the number of circulating water pumps in operation is an operational strategy for reducing cooling water volume and intake approach velocities, and hence the rates of entrainment and impingement (Apprendix D). The normal circulating water pump operation at the Pittsburg Power Plant (Section 2.1.2) is the minimum level necessary to sustain full unit generation. Reduction in circulating water pump operation from two pumps per unit to one would be expected to result in a 50 percent reduction in load capacity. There would be increased temperature differentials through the condenser system (delta-T), and thus reduced turbine cycle thermal efficiency. These operational constraints severely limit the performance efficiency of a generating unit. Reducing circulating water pump operation would reduce approach velocity and would be expected to result in a reduced impingement rate, but the degree of reduction cannot be quantified. A reduction in cooling water volume of 50 percent, as a result of operating one pump per unit, is expected to reduce the number of entrained organisms by 50 percent, but the associated increase in delta-T might reduce the overall survival of those organisms that were entrained. Because of the generating penalty, loss of reliability, and operational inefficiency of generating units that would accompany reduced circulating water pump operation during periods of high load, further investigation of reduced pump operation as an approach to reducing biological losses is not warranted.

Reducing the number of circulating water pumps in operation on a unit when generating loads are low or the unit is out of service for fuel conservation or maintenance may not only reduce biological losses, but also reduce operating costs. However, removing one of the usual two operating pumps from service on a unit would restrict its availability for load increase to approximately one-half load until the other pump was returned to service. The additional starts and stops on the pumps would increase the maintenance required and increase the likelihood of pump motor failure in equipment not designed for either frequent or numerous starts. To permit rapid response to energy demands, less efficient units may be operated at low or minimum loads but kept capable of an immediate increase to full load. To meet this requirement, both circulating water pumps are normally kept in service. If the circulating water pumps were taken out of service, the condenser and conduits would

drain. The time required to refill the cooling system would not permit fulfillment of the requirements of reserve operation.

Consistent with system load demand less efficient units are taken out of service (generation) for fuel conservation outages. In this case the circulating water pumps may or may not be left in service dependent on the length of the fuel conservation outage. Normally, if an extended outage is expected the circulating water pumps are shut down to conserve energy. For anticipated short outages of a day or so the circulating water pumps are usually not taken out of service. A fuel conservation outage is intended to be an outage in which the unit is kept fully capable of returning to service on short notice. If maintenance is to be performed on a unit during a fuel conservation outage which in any way jeopardizes the units return to service or its capability, the outage reverts to a maintenance outage.

During periods when a unit is removed from service for either fuel conservation or maintenance, one or more circulating water pumps can be shut down, depending on the nature and expected length of the outage. Reduced circulating water pump operation during periods when units are out of service for extended periods is standard operating procedure at the Pittsburg Power Plant.

6.3.5 Curtailment of Cooling System Operation

Seasonal curtailment of cooling system operation at the Pittsburg Power Plant is not feasible because of the high demand for electrical power and the uncertainty of availability of electrical power from other sources to replace it. The present operation of Units 1-4 is to meet peak load demand, and Units 5, 6, and 7 are used primarily to meet base load demand. Setting aside the question of a replacement source of energy and capacity to meet system demands, the estimated replacement cost of energy alone resulting from curtailment of operation of Units 1-4, Units 5 and 6, and Unit 7 during May and June and during July and August is summarized in Table 6-13, based on operating data and costs of replacement power in 1979 (expressed as 1980 dollars). However, no quantitative estimate of the projected future cost of replacement energy has been developed, because of the uncertainty of the expected power demands and reserve capacity available at that time. In addition, fluctuating fuel costs, which are a major factor in the cost of replacement energy, make accurate projections of future energy costs impossible. If curtailment could be implemented

TABLE 6-13 ESTIMATED ANNUAL COST (1980 dollars)[a] OF REPLACEMENT ENERGY DURING TWO PERIODS OF CURTAILMENT OF PLANT OPERATION AT THE PITTSBURG POWER PLANT

Period of Curtailment	Cost of Energy Replacement ($/MWh)	Hours/ Month	Unit 1 Capacity Factor (%)	Unit 1 Energy Replacement Cost ($)	Unit 2 Capacity Factor (%)	Unit 2 Energy Replacement Cost ($)	Unit 3 Capacity Factor (%)	Unit 3 Energy Replacement Cost ($)	Unit 4 Capacity Factor (%)	Unit 4 Energy Replacement Cost ($)
May	57	744	59.4	4,031,000	53.6	3,864,000	60.8	4,125,000	53.2	3,835,000
June	57	720	80.1	5,260,000	79.7	5,561,000	81.8	5,371,000	34.5	2,407,000
Total				9,291,000		9,425,000		9,496,000		6,242,000
July	57	744	70.6	4,790,000	66.2	4,773,000	76.0	5,157,000	72.3	5,212,000
August	57	744	75.0	5,089,000	78.4	5,652,000	82.6	5,605,000	77.0	5,551,000
Total				9,879,000		10,425,000		10,762,000		10,763,000

Period of Curtailment	Cost of Energy Replacement ($/MWh)	Hours/ Month	Unit 5 Capacity Factor (%)	Unit 5 Energy Replacement Cost ($)	Unit 6 Capacity Factor (%)	Unit 6 Energy Replacement Cost ($)	Unit 7 Capacity Factor (%)	Unit 7 Energy Replacement Cost ($)
May	57	744	0	0	21.7	3,037,000	37.9	11,894,000
June	57	720	0	0	79.1	10,713,000	44.3	13,454,000
Total				0		13,750,000		25,348,000
July	57	744	16.4	2,295,000	80.4	11,252,000	71.3	22,375,000
August	57	744	60.7	8,495,000	80.2	11,224,000	68.2	21,402,000
Total				10,790,000		22,476,000		43,777,000

(a) Energy replacement cost is based on actual annual average power pool replacement costs (1980 dollars) and does not necessarily reflect present costs or the costs of a guaranteed energy replacement capacity through construction of new facilities. Cost estimates do not include potential savings associated with reduced operating and maintenance expenses.

Note: Monthly capacity factors are based on 1979 data summarized in Tables 2-2 and 2-5.

during these two periods there would be reduced biological losses resulting from both entrainment and impingement. This curtailment strategy, however, is infeasible because it would also severely reduce electrical generating capacity during these two periods of high electrical demand.

An alternative curtailment strategy would be to schedule maintenance outages to coincide with periods of greatest biological loss. Maintenance outages for unit overhauls are scheduled on the basis of 50,000 hours of operation—approximately five years between scheduled unit overhauls. A major overhaul would require an outage of approximately eight weeks for each unit. Scheduled maintenance outages for overhauls are determined on a yearly basis and require coordination between projected system generating requirements and manpower availability. Because of the infrequency of major scheduled maintenance outages and the uncertainties of maintenance scheduling, curtailment based on maintenance outages is an ineffective method of reducing biological losses at the Pittsburg Power Plant.

Daily curtailment of cooling system operation (e.g., at night or when load is low) was also considered as an alternative approach for reducing biological loss. Even though the units at the Pittsburg Power Plant may operate at reduced load at times, these units are used for base load demand and, as such, must be available for increases in load consistent with system demands. Daily curtailment of cooling system operation is therefore considered to be impractical for the Pittsburg Power Plant, based on the need for rapid response to electrical demands within the system and the limited reserve generating capacity available.

The various strategies for curtailment of cooling system operation would result in a reduction of both entrainment and impingement cropping in an amount that would depend on the abundance of organisms present during the period of curtailment and the duration of the outage. However, curtailment of operation at the Pittsburg Power Plant beyond what occurs normally is not acceptable, because it removes the generating capacity of the plant from reliable service at the very time of year that it is most needed to serve loads. The availability of replacement power is uncertain, and if available, it would be very costly. Consideration of curtailment of power generation as a method of reducing entrainment and impingement cropping at the Pittsburg Power Plant is not a feasible alternative.

6.3.6 Discharge Temperature Regulation

The temperature of the cooling water increases rapidly as it passes through the condenser tubes, and then remains constant or declines slightly until the water is discharged back into Suisun Bay. Reducing thermal stresses on entrained organisms by maintaining discharge water temperatures below 30 C (86 F) has been suggested as a means of increasing the survival of entrained ichthyoplankton and invertebrates (Section 6.2.6.6). At the Pittsburg Power Plant discharge temperatures exceeded 30 C approximately 33 percent of the time at Units 1-4, 30 percent of the time at Unit 5, and 37 percent of the time at Unit 6. Discharge temperatures in excess of 30 C occurred primarily between June and November (Figures 2-8 and 2-9), with peak temperatures in July, August, and September.

At reduced generating loads the cooling water flow usually remains the same, resulting in lower temperature increases. Temperature increases (delta-Ts) through the condensers as a function of unit load were estimated for Units 1 and 6, using power generation logs and condenser heat transfer tests (Tetra Tech 1976). At Unit 1, delta-Ts were 5-5.6 C (9-10 F) at 50 percent capacity and about 10 C (17.5 F) at maximum capacity (Figure 6-3). Estimated delta-Ts at Unit 6 were approximately 4 C (7.5 F) at 36 percent, 9 C (16 F) at 70 percent, and 13-14 C (23-25 F) at maximum capacity. Delta-Ts estimated for Unit 1 are representative of those for Units 2, 3, and 4, and those for Unit 6 are representative of Unit 5.

Discharge temperature regulation during the period April-July, the period of peak entrainment of larval and juvenile fish and in particular striped bass (Figures 3-4 and 3-5), would undoubtedly reduce the numbers of organisms lost to entrainment. There is conflict, however, in that the period of peak entrainment coincides not only with the period of highest intake water temperatures (Figures 2-8 and 2-9), but also with high electrical demand.

On the basis of intake water temperatures measured during 1978 and 1979 and the relationship between load and delta-T, it was estimated that in maintaining discharge temperatures less than 30 C during June and July, Units 1-4 would be limited to a capacity factor of 65 percent and Units 5 and 6 to a capacity factor of 45 percent. Average monthly capacity factors for Units 1-4 ranged from 44 to 66 percent during June and July 1978 and from 35 to 82 percent during June and July 1979 (Table 2-2).

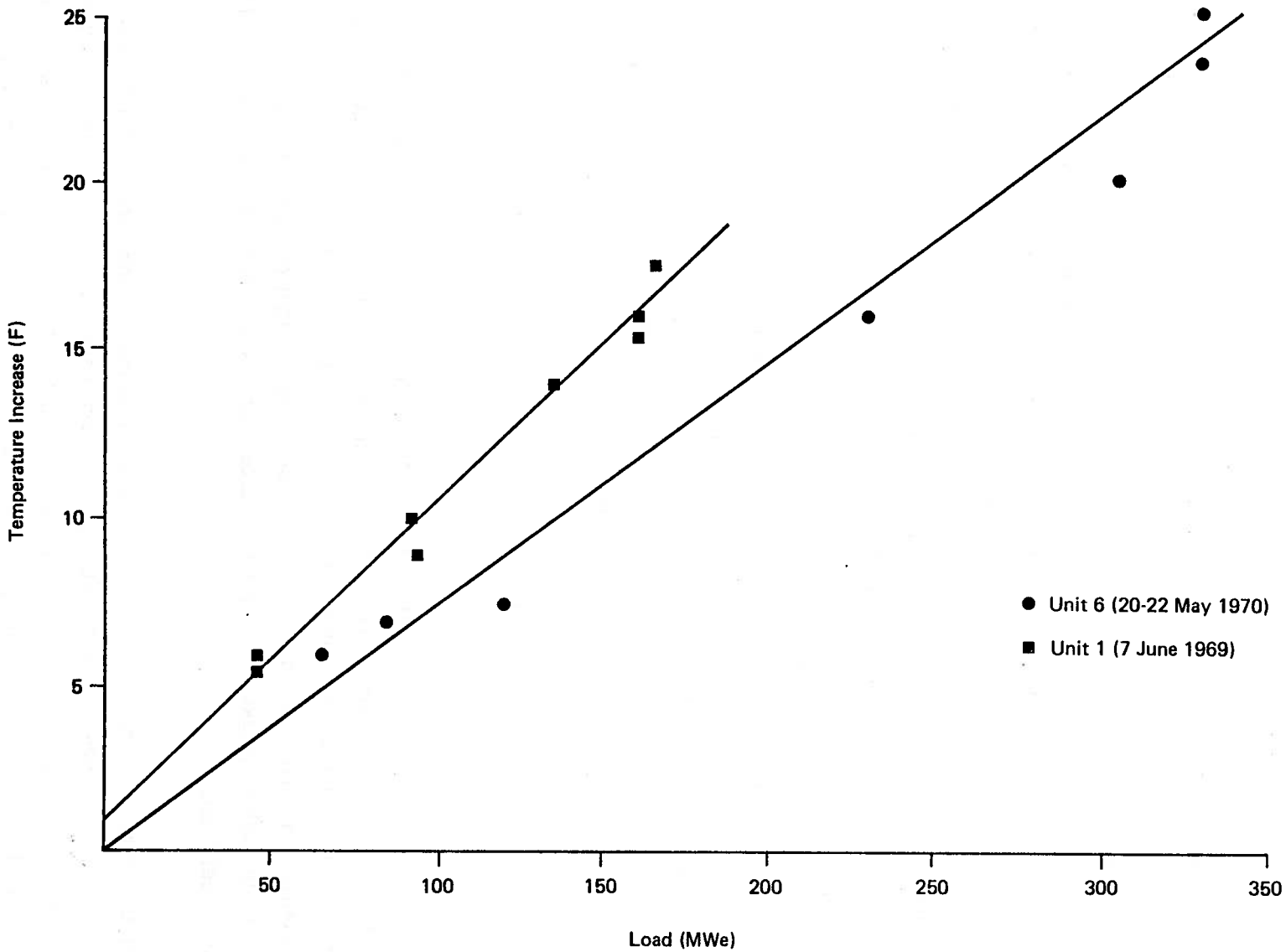

Figure 6-3. Temperature increase (ΔT) as a function of unit load, Pittsburg Power Plant. (From PGandE 1973.)

Monthly capacity factors for Units 5 and 6 ranged from 0 to 71 percent in June and July 1978 and from 0 to 80 percent in June and July 1979. On the basis of forecast average monthly capacity factors appearing in PGandE's 4 December 1980 Electric Resource Plan, which assume average water-year conditions, monthly mean unit-group (Units 1-4, Units 5 and 6) capacity factors will be significantly lower (primarily because of operation of the Diablo Canyon Power Plant) than at present starting in 1982. Units 1-4 will operate individually at 25 percent or less in June and 30 percent or less in July, and Units 5 and 6 at 45 percent or less (sometimes considerably less) in June and 55 percent or less in July. The forecast unit capacity factors are considerably lower, almost all zero, in May (Table 6-14). Thus, projected unit operation on a monthly basis is approaching levels where the frequency of discharge temperatures exceeding 30 C (86 F) will decrease substantially during the spring and early summer. This general trend in decreased unit generation at the Pittsburg Power Plant will contribute to reduced entrainment and impingement cropping under conditions of projected future operation.

6.4 SUMMARY AND CONCLUSIONS

The biological benefits, costs, and engineering constraints of the alternative intake technologies considered in Section 6.3 are summarized below. On the basis of this information, a conclusion is drawn as to the best technology available for the Pittsburg Power Plant.

The combined impact of entrainment and impingement at the Pittsburg Power Plant cooling water systems on selected anadromous, seasonal, and permanent resident fishes and invertebrates was considered in Chapter 5. The impact of the cooling water systems of the power plant on populations of fish and invertebrates inhabiting the Sacramento - San Joaquin Bay-Delta system was judged to be low for most species. Information available for a majority of the species considered indicates that the numbers of organisms entrained and impinged at the power plant will not result in an adverse environmental impact. The primary species considered in the impact assessment is the striped bass, partly because the early life stages are entrained and impinged in relatively high numbers and partly because the Bay-Delta striped bass population has experienced a general decline in abundance. A detailed examination of the relative contribution of operation of the Pittsburg Power Plant cooling water systems to the year class strength of striped bass and to a potential reduction in the

TABLE 6-14 FORECAST[a] AVERAGE MONTHLY CAPACITY FACTORS FOR UNITS 1-7 OF THE PITTSBURG POWER PLANT, 1982-1990

Month	Unit 1	2	3	4	5	6	7
1982							
APR	0	0	0	0	0	0	0
MAY	0	0	0	0	49.9	0	0
JUN	0	22.1	22.4	21.9	42.9	44.2	20.4
JUL	0	26.1	26.1	25.9	50.9	51.8	25.8
1983							
APR	0	0	0	0	0	0	0
MAY	0	0	0	0	0	0	0
JUN	0	0	0	0	39.6	0	0
JUL	0	26.1	26.2	26.0	31.1	27.9	24.2
1984							
APR	0	0	0	0	0	0	0
MAY	0	0	0	0	0	0	0
JUN	0	0	0	0	0	0	0
JUL	0	25.1	25.2	25.0	27.7	30.7	23.2
1985							
APR	0	0	0	0	0	0	0
MAY	0	0	0	0	0	0	0
JUN	0	0	0	0	24.0	0	0
JUL	0	26.0	26.2	25.9	30.5	31.7	24.0
1986							
APR	0	0	0	0	0	0	0
MAY	0	0	0	0	0	0	0
JUN	0	0	0	0	25.1	42.2	0
JUL	0	27.5	27.7	27.5	45.3	52.0	25.4
1987							
APR	0	0	0	0	0	0	0
MAY	0	0	0	0	0	0	0
JUN	0	26.1	0	0	25.9	43.6	0
JUL	0	28.6	28.8	28.6	49.4	53.5	26.4
1988							
APR	0	0	0	0	55.4	0	0
MAY	0	0	0	0	46.7	0	0
JUN	0	25.0	25.1	24.9	28.2	41.3	23.2
JUL	0	29.0	29.2	29.0	53.4	53.7	26.8
1989							
APR	0	0	0	0	0	0	0
MAY	0	0	0	0	0	0	0
JUN	0	24.8	23.2	24.7	24.8	40.4	23.0
JUL	0	28.7	29.0	28.7	51.5	53.5	26.5
1990							
APR	0	0	0	0	56.4	0	0
MAY	0	0	0	0	30.1	0	0
JUN	0	24.8	24.9	24.6	24.6	36.3	22.9
JUL	0	29.0	29.3	29.0	51.4	53.4	26.7

(a) On the basis of average water year conditions and PGandE's 4 December 1980 Electrical Resource Plan.

Note: A projected capacity factor of 0 should not be taken to mean that a unit has been retired from service (see Section 2.1.4).

number of bass recruited into the sport fishery was undertaken. It was concluded that although entrainment and impingement mortality contributed directly to a reduction in the abundance of young-of-the-year striped bass, the contribution is small in comparison to the total mortality rates of larval and juvenile striped bass, and entrainment and impingement mortality are not major factors determining year class strength. Furthermore, there was no apparent relationship between the magnitude of entrainment mortality and subsequent recruitment: the effect of operation of the Pittsburg Power Plant cooling water system is not a dominant factor in determining whether recruitment of a striped bass year class into the sport fishery will be high or low.

An examination of the relationship between the striped bass index of year class strength and subsequent recruitment into the sport fishery indicated that variation in year class strength accounted for only 35 percent of the variability in estimates of the number of striped bass entering the fishery at age three. It was concluded, on the basis of the high variability in the young-of-the-year - recruitment relationship, that efforts made to further reduce the relatively small contribution of power-plant-related mortality to the total mortality of young-of-the-year striped bass through implementation of alternative cooling system technology are not likely to contribute to a detectable improvement to the sport fishery. Nor was there any evidence in the available population data to suggest that a reduction in the historical levels of power plant mortality would directly increase abundance in the adult striped bass stock. For these reasons, it was concluded that the impact of the Pittsburg Power Plant is low and that there is no certainty that implementation of alternative intake technologies designed to further reduce entrainment or impingement mortality will be reflected in a benefit through increased population abundance for those fish and invertebrate species inhabiting the Bay-Delta. The discussion and evaluation presented here of alternative intake technologies is based in part on this conclusion.

6.4.1 Discussion

Alternative intake technologies considered for the Pittsburg Power Plant included a closed-cycle cooling system at Units 1-6, modified vertical traveling screens and fish return systems, dredging the intake area to reduce intake velocities, short-term reductions in circulating water pump operations, seasonal curtailment of cooling system operation, and discharge temperature regulation during summer months. Each

of these alternatives is expected to reduce the cropping of organisms as a result of entrainment and/or impingement.

Replacement of the existing once-through cooling systems of Units 1-6 with closed-cycle cooling systems would result in a substantial reduction in cooling water flow and therefore a reduced rate of entrainment and impingement. The reduction in entrainment mortality of striped bass larvae is expected to be approximately 91 percent, with similar reductions in entrainment mortality expected for other fish and invertebrate species. The potential reduction in impingement mortality associated with closed-cycle cooling cannot be quantified, but the reduction in cooling water flow would result in reduced intake velocities and flows and hence a reduction in impingement. The total estimated capital cost (not including the costs of replacement power associated with the unit outage or long-term unit derating) was approximately $219 million (1980 dollars). The costs associated with implementation of closed-cycle cooling at Units 1-6 are not considered to be proportionate to the benefits resulting from a reduction in entrainment and impingement mortality because (based on the foregoing report) it is unlikely that the expected reduction in striped bass mortality would contribute to a detectable increase in the adult striped bass stock available to the sport fishery. On the basis of the high costs and uncertainty of any benefit, it was concluded that modification of the existing cooling water systems to include closed-cycle cooling at Units 1-6 does not reflect the best available technology for Unit 1-6 at the Pittsburg Power Plant.

Installation of modified vertical traveling screens and fish return system could reduce impingement mortality of young-of-the-year and yearling striped bass from approximately 111,000 to 28,000 fish. No reduction in impingement mortality, however, is expected for species such as threadfin shad, longfin smelt, *Palaemon macrodactylus*, *Crangon franciscorum*, or *Rhithropanopeus harrisii*. The estimated direct capital cost of modifying the intake of the Pittsburg Power Plant to include fish handling facilities ranged from approximately $1.2 million to $1.5 million (1980 dollars). Additional indirect costs associated with increased operational and maintenance costs and engineering studies to insure reliable intake screen performance under the high detrital loading at the plant were not quantified because of a lack of experience with modified intake screens under such conditions. Although the benefits of improved impingement survival on a subsequent increase in adult population abundance are uncertain, modification of the intake screens would

contribute to a reduction in the numbers of young-of-the-year and yearling striped bass cropped. The benefits of modified traveling screens and a fish return system are considered to be sufficient to warrant further engineering evaluation of their performance and reliability at the Pittsburg Power Plant. Because of present uncertainty regarding the operational characteristics of modified screens and potential problems with separation of debris and organisms using dual spraywash systems, and the associated concern of debris recirculation, it is not recommended that the Pittsburg Power Plant intake be modified at this time.

Dredging the intake area to remove accumulated sediment has been completed. Dredging of the intake area has resulted in a reduction in intake approach velocities (PGandE, unpublished a) and is expected to result in a reduced impingement rate for fish.

Reducing circulating water pump operation when conditions permit while maintaining reliability of reserve generating capacity is a standard operating practice at the Pittsburg Power Plant. This has been identified as a biologically effective method of reducing the cropping of organisms resulting from entrainment and impingement. Reduced circulating water pump operation whenever practical reduces the cost and energy required to operate the cooling system as well as contributing to reduced entrainment and impingement rates. As the projected operation of many of the units at the Pittsburg Power Plant continues to decline, the frequency and duration of circulating water pump operation are expected to be reduced.

Although curtailment of cooling water system operation would result in a reduction of both entrainment and impingement cropping (dependent on the abundance of organisms present during the period of curtailment and the duration of the outage), the high cost of replacement power—if available—and reductions in the reliability of electrical service and possible impairment of ability to serve load during the period of curtailment make this an unacceptable option. Therefore, consideration of curtailment of power generation as a method of reducing entrainment and impingement cropping at the Pittsburg Power Plant is not a feasible alternative.

Discharge temperature regulation during the summer months is expected to result in improved survival of larval fish and invertebrates entrained into the cooling systems. Entrainment survival for most species is relatively high when discharge temperatures

are below 30 C (86 F), but it decreases as discharge temperatures increase above 30 C (86 F). Discharge temperatures are a function of intake water temperature and electrical generating load on a unit (assuming constant cooling water flow). On the basis of the 4 December 1980 forecast generation of the Pittsburg Power Plant (Table 6-14), it is expected that operation of the units will decline to levels where the extent of discharge water temperatures exceeding 30 C (86 F) will be low. Based on the declining projected generation of the units (and hence reduced discharge temperatures), we conclude that no further investigation of discharge temperature regulation as an approach to reducing entrainment mortality at Units 1-6 is warranted.

6.4.2 Conclusions

The judgment of the best technology available for each cooling system at the Pittsburg Power Plant is based on a consideration of the impact of entrainment and impingement on the local aquatic community, biological effectiveness of various technologies for further reducing entrainment and impingement cropping, engineering feasibility, and cost-effectiveness, as outlined in the guidance manual (U.S. EPA 1977). Results of the analysis are summarized in Table 6-15.

The conclusion drawn from the preceding sections and Table 6-15 is that the existing cooling water system and shoreline intake location and configuration reflect the best technology available, provided the present operating practices of (1) maintenance dredging to keep intake velocities low, (2) intermittent chlorination at low levels, and (3) reducing the operation of the circulating water pumps when a unit is out of service, are continued.

6.5 REFERENCES

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Cada, G. F., J. B. Cannon, and D. W. Lee. 1979. Alternative screening devices at Indian Point Units 2 and 3, Bowline Point, and Roseton generating stations. Testimony prepared for Hudson River NPDES case before the U.S. Environmental Protection Agency, Region II. EPA Docket No. C/II-WP-77-01.

TABLE 6-15 SUMMARY OF THE EVALUATION OF INTAKE TECHNOLOGIES AND OPERATIONAL ALTERNATIVES CONSIDERED FOR THE PITTSBURG POWER PLANT

	Demonstrated (Meets Criterion 1)	Biological Benefit (Meets Criterion 2)	Recommended (Meets Criteria 3 and 4)
Intake Technologies			
Cooling Water Systems			
Closed-cycle			
Units 1-6	Yes	Yes	No
Unit 7[a]	Yes	Yes	Yes
Once-through			
Units 1-6[a]	Yes	—	Yes
Unit 7	Yes	No	—
Intake Locations			
Offshore	Yes	No	—
Onshore[a]	Yes	—	Yes
Intake Configurations			
Shoreline[a]	Yes	—	Yes
Recessed	Yes	No	—
Behavioral Barriers			
Velocity gradient	No	—	—
Electrical barrier	No	—	—
Louvers	No	—	—
Chemicals	No	—	—
Magnetic field	No	—	—
Chains and cables	No	—	—
Light	Yes	No	—
Sound	Yes	No	—
Bubble screen	Yes	No	—
Velocity cap	Yes	No	—
Physical Barriers			
Media filter	No	—	—
Porous dike	No	—	—
Radial well	No	—	—
Stationary screen	No	—	—
Horizontal traveling screen	No	—	—
Angled screen - louver	No	—	—
Caisson	No	—	—
Drum screen	Yes	No	—
Centerflow screen	Yes	No	—
Vertical traveling screen[a]	Yes	—	Yes
Fish Collection, Removal, and Conveyance Systems			
Vertical traveling screen modifications	Yes	Yes	No
Gravity sluiceway	Yes	Yes	No
Fish pumps	Yes	No	—
Operational Alternatives			
Maintenance			
Dredging	Yes	Yes	Yes
Operational Modifications			
Circulating water pump volume reduction[a]	Yes	Yes	Yes
Curtailment	Yes	Yes	No
Alternative biofouling control	Yes	No	—
Cooling system modifications	Yes	No	—
Temperature regulation	Yes	Yes	No

(a) Existing structure or operation.

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PITTSBURG POWER PLANT
COOLING WATER INTAKE STRUCTURES
316(b) DEMONSTRATION

APPENDIXES

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Appendix A

Water Quality, Saltwater Intrusion Model, and Intake Velocity Studies

A.1 Water Quality Data

A.2 Saltwater Intrusion Model

A.3 Intake Velocity Study

A.4 References

APPENDIX A: WATER QUALITY, SALTWATER INTRUSION MODEL, AND INTAKE VELOCITY STUDIES

A.1 WATER QUALITY DATA

Water temperature, conductivity, and dissolved oxygen concentration were measured during each impingement and entrainment survey. Intake and discharge temperatures of the Units 1-4 and Units 5 and 6 cooling water flows were continuously recorded. The data provided background information on the temporal and seasonal changes in physical and chemical characteristics of the cooling water.

A.1.1 Methods

Martek Mark V water quality analyzers were used to collect temperature, conductivity, and dissolved oxygen data during all biological surveys of the Pittsburg Power Plant cooling water system between March 1978 and December 1979. These instruments provided data of sufficient accuracy for the purposes of the studies. The manufacturers' specifications for each parameter are listed in Table A-1. Backup systems available in the event of unit failures consisted of calibrated mercury thermometers, fixing reagents for preserving samples for subsequent laboratory determination of dissolved oxygen, and portable Yellow Springs Instruments conductivity meters.

Use of the water quality analyzers is based on a standard operating procedure and operator training program. Each analyzer was calibrated and maintained on a weekly or biweekly basis. The calibration procedures for each parameter are described in Section A.1.3.

Water temperatures were measured and continuously recorded at the intake and discharge of both Units 1-4 and Units 5 and 6 using Foxboro Model 12R temperature recorders. These units have a temperature range of 30-110 F (-1 C - 43 C), and are equipped with 7-day clock-driven recording charts.

A.1.2 Sampling Locations

Water quality measurements were made with Martek water quality analyzers in the Units 5 and 6 intake forebays between the bar racks and the curtain wall,

TABLE A-1 MANUFACTURERS' ACCURACY SPECIFICATIONS FOR WATER QUALITY INSTRUMENTS USED AT THE PITTSBURG POWER PLANT, 1978-1979

Instrument	Parameter	Probe Type	Accuracy
Martek Mark V	Temperature	Thermolinear array mounted in stainless steel tube	±0.1 C
	Conductivity	Platinized nickel electrode; automatic temperature compensation of 25 C	±1.0 mmho
	Dissolved Oxygen	Polarographic gold/silver electrode system; automatic temperature compensation	±0.5 mg/liter
Foxboro Model 12 R	Temperature	Temperature sensitive bulb connected by capillary tube to measuring element inside recorder housing	±0.5% of full scale

approximately 1.6 ft (0.5 m) below the surface. Access to the cooling water flow at both intake structures was severely restricted. Martek sampling at the Units 1-4 intake was not feasible, so the Units 5 and 6 sampling station was used for all water quality data collections at the plant. Because the two intakes were adjacent, the differences in the three parameters are not expected to be significant.

Foxboro temperature sensors were located as follows:

1. the inlet pipe of the west condenser half of Unit 1,
2. the Units 1-4 discharge gate well,
3. the inlet pipe of the west condenser half of Unit 6,
4. the Unit 5 discharge gate well, and
5. the Unit 6 discharge gate well.

Placement of the intake temperature sensors in an area where the cooling water flow was well mixed was not possible because of siting and installation difficulties. The intake monitoring stations that were selected measured temperatures in an area where some temperature stratification of the cooling water flow may occur. The degree of stratification was not expected to be great (range less than 2 F; 1.1 C) and thus was considered unimportant for the purpose of these studies.

A.1.3 Instrument Calibration and Correction Factors

A.1.3.1 Martek Calibration

Water quality analyzers were calibrated on a weekly or biweekly basis, depending on the frequency of use and the stability of the individual unit. The calibration procedures for each parameter are briefly described below. In addition, a formula used to convert conductivity to salinity is presented.

Temperature

A two-point temperature calibration procedure was employed to ensure accuracy of the sensor. The sensor was placed in an ice bath and the temperature readout was adjusted to correspond to the actual temperature, as measured with a thermometer traceable to an NBS standard (± 0.1 C). The sensor was then placed in a water bath

maintained at approximately 40 C (104 F) and further adjustments were made to the readout circuit if the unit's output did not agree with the actual temperature of the water bath. Calibration adjustments were made at both temperatures until all instrument readings agreed with the standard without further adjustment.

Conductivity

Calibration of the unit consisted of first electronically zeroing the system with the sensor out of water, then immersing the sensor in a solution of known conductivity (approximately 35 mmho at 25 C) and adjusting the readout to agree with the standard. The conductivity of the standard solution was measured with a YSI Model 31 conductivity bridge (accuracy ± 1 percent of reading, resolution better than 0.5 percent) and was occasionally checked by volumetric silver-nitrate titration (Strickland and Parsons 1972).

Conductivity is automatically compensated for temperature to 25 C, so direct comparison of readings taken at different temperatures is possible without manipulation. In many cases it may be desirable to report salinity instead of conductivity. Salinity is related to conductivity by the following equation:

$$S(ppt) = \frac{0.4738}{C_{25}^{-1.07} - 0.00075} + 0.03 \qquad \text{(A-1)}$$

where

C_{25} = specific conductance at 25 C (mmho).

Dissolved Oxygen

Calibration of the dissolved oxygen system consisted of electronically zeroing the internal readout circuit and then immersing the sensor in fresh water with a dissolved oxygen concentration determined by a Winkler titration (APHA et al. 1976). The water quality analyzer was adjusted to bring the unit into agreement with the standard.

A.1.3.2 Foxboro Calibration

The Foxboro recorders were calibrated prior to installation and were recalibrated at 2-month intervals for the duration of the study period. Calibration consisted of

removing the sensor bulb from the cooling water flow and comparing its reading to the reading of an NBS traceable thermometer (± 0.1 C) in an icewater bath (temperature approximately 32 F; 0 C) and in a warm water bath (temperature approximately 110 F; 41 C). Adjustments were made if the recorder reading differed from the actual temperature by more than 1.0 F (0.5 C).

A.1.4 Verification of Data Analysis

A.1.4.1 Verification of Martek Data

Data collected with the water quality analyzers were evaluated on the basis of calibration results obtained upon return of the units to the laboratory after a period of use in the field. A certain amount of drift is inherent in these instruments, and it is primarily on the basis of the amount of relative variation between calibrations that data are rejected. Instrument error was calculated from the results of calibration as follows:

$$\text{Error (percent)} = \frac{V_i - V_s}{V_s} \times 100 \qquad \text{(A-4)}$$

where

V_i = instrument reading in a standard solution
V_s = actual value of the standard.

The relative variation (RV) between calibrations was calculated as follows:

$$RV = \text{Error at } T_1 - \text{Error at } T_2 \qquad \text{(A-5)}$$

where

Error at T_2 = calibration error after field sampling
Error at T_1 = calibration error immediately following the previous calibration (assumed to be zero).

Calibration errors for temperature, conductivity, and dissolved oxygen in excess of plus or minus 2.5, 5.0, and 10.0 percent, respectively, were interpreted as instrument

failures for the period preceding recalibration, and data for parameters whose calibration error exceeded these limits were eliminated from the data file. In general, the calibration errors were much lower than the rejection limits for all three parameters. Relative variations of water quality analyzers are presented in Figure A-1 for all sampling days in which acceptable data for each parameter were obtained.

A.1.4.2 Verification of Foxboro Data

The temperature response of a Foxboro recorder was compared to a precision thermometer over the full temperature range of the unit. Foxboro response was linear over the full range with no deviation in the middle of the range more than 0.3 F (0.2 C) (Table A-2), so no adjustments were required to establish a temperature response curve.

Through periodic calibration, the Foxboro recorders used at the Pittsburg Power Plant were maintained at an accuracy within 1 F (0.5 C) during the study period, so compensation of recorded temperatures was unnecessary for the purposes of the study.

A.1.5 Data Summary

The monthly means and ranges of conductivity and dissolved oxygen measurements made along with the biological sampling are presented in Tables A-3 and A-4.

Foxboro records were read at 3-hour intervals and the temperatures entered in the computer. The data were used to determine the times that discharge temperatures were in certain ranges, as described in Chapters 3 and 5 and Appendix C. Table A-5 is a frequency analysis of intake and discharge temperatures over the period March 1978 - November 1979. The daily means and ranges of intake and discharge temperatures for the study period are listed in Table E.3-1, Appendix E.

A.2 SALTWATER INTRUSION MODEL

A.2.1 Introduction

The entrapment zone, generally defined as the portion of the estuary at the interface

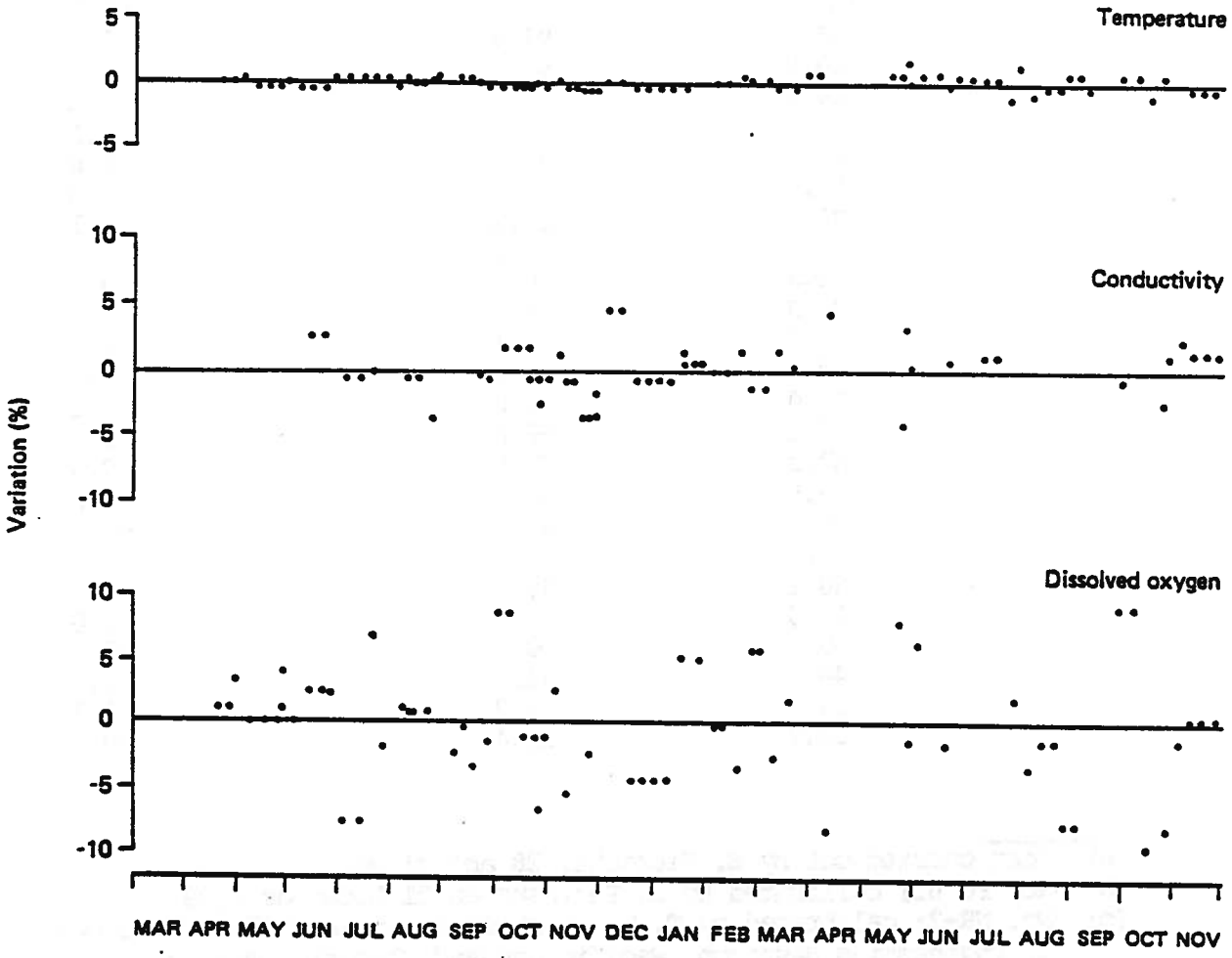


Figure A-1. Relative variation between calibrations of water quality analyzers used at the Pittsburg Power Plant, 1978-1979 .

TABLE A-2 COMPARISON[a] OF TEMPERATURE MEASUREMENTS OBTAINED WITH A FOXBORO RECORDER CALIBRATED AT 32.2 F AND 106.0 F AND A CALIBRATED PRECISION THERMOMETER

Temperature Recorded (F)		
Foxboro Recorder[b]	Thermometer[c]	Difference (F)
106.0	106.0	0.0
101.0	101.2	-0.2
96.6	96.6	0.0
93.0	92.8	+0.2
89.5	89.4	+0.1
86.6	86.6	0.0
84.0	84.0	0.0
81.5	81.5	0.0
79.3	39.2	+0.1
77.5	77.2	+0.3
76.3	76.2	+0.1
75.0	75.0	0.0
73.5	73.4	+0.1
72.7	72.5	+0.2
72.0	72.0	0.0
70.0	69.8	+0.2
67.5	67.5	0.0
66.5	66.5	0.0
65.3	65.2	+0.1
63.6	63.6	0.0
60.2	60.2	0.0
55.5	55.5	0.0
50.3	50.2	+0.1
44.6	44.6	0.0
39.3	39.2	+0.1
33.2	33.4	-0.2

(a) Test carried out by E. Petroske, 28 and 31 December 1978.
(b) No. FO 01; calibrated by E. Petroske on 21 December 1979.
(c) No. NR-2; calibrated by T. W. Holmbeck in August 1979 at the Department of Engineering Research, Pacific Gas and Electric Company.

TABLE A-3 MONTHLY MEANS AND RANGES OF CONDUCTIVITY MEASUREMENTS TAKEN AT THE UNITS 5 AND 6 INTAKE STRUCTURE OF THE PITTSBURG POWER PLANT, 1978-1979

Month	Mean	Range (mho) Minimum	Maximum	Number of Observations
1978				
March	0	0	0	0
April	326	0	1,052	19
May	183	164	269	62
June	2,184	250	7,991	60
July	5,030	2,112	8,623	64
August	7,058	2,480	14,100	80
September	2,971	909	8,336	73
October	2,599	928	5,892	92
November	4,436	1,815	10,200	91
December	4,609	—	9,768	69
1979				
January	2,610	145	9,092	79
February	438	184	1,915	64
March	229	184	350	64
April	256	235	307	41
May	740	370	2,100	24
June	1,978	663	4,890	32
July	5,256	2,962	9,157	60
August	5,957	1,362	13,800	61
September	8,676	6,431	14,000	23
October	5,836	281	12,440	64
November	3,469	1,104	9,994	60

Note: Means are weighted by the monthly number of observations.

TABLE A-4 MONTHLY MEANS AND RANGES OF DISSOLVED OXYGEN MEASUREMENTS TAKEN AT THE UNITS 5 AND 6 INTAKE STRUCTURE OF THE PITTSBURG POWER PLANT, 1978-1979

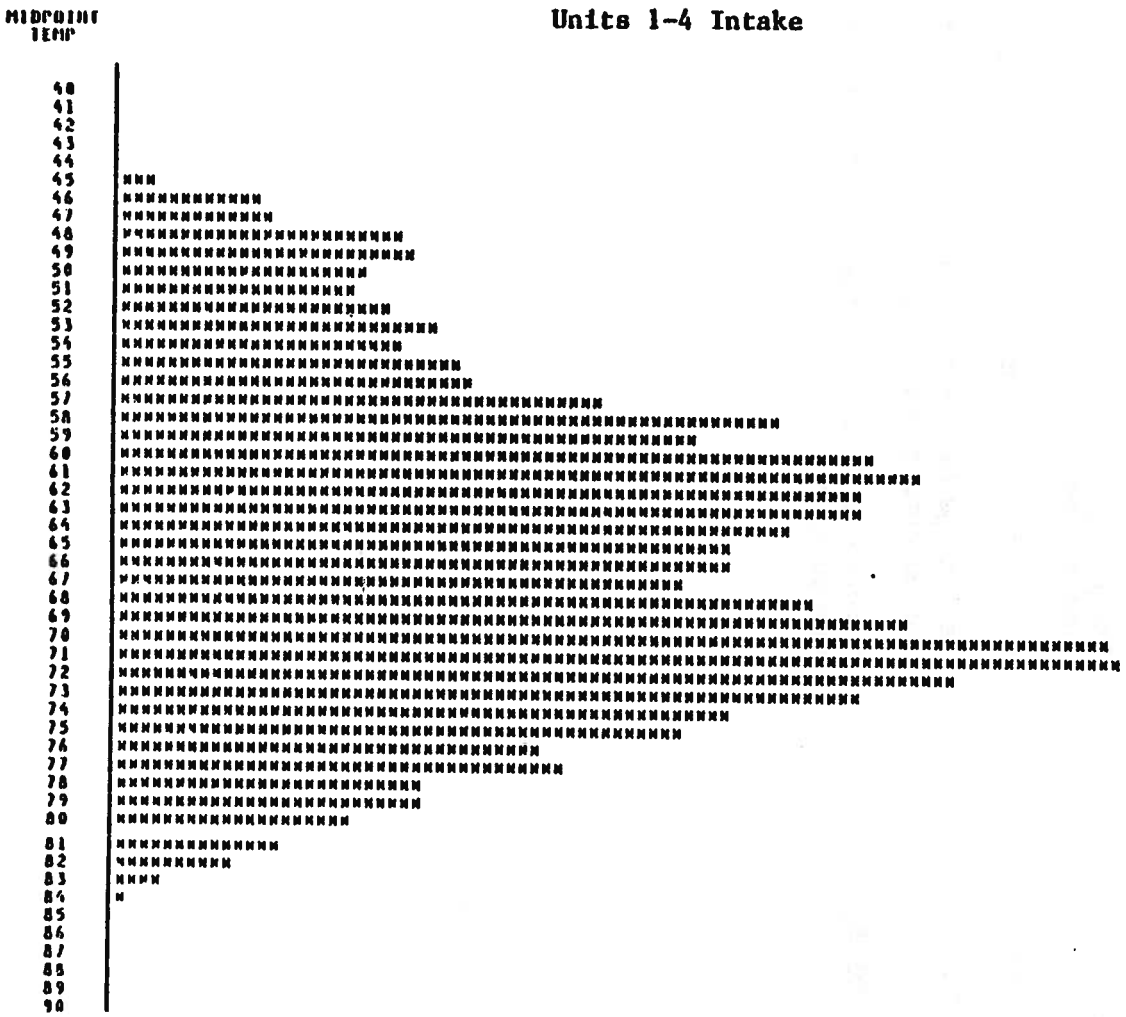
Month	Mean (mg/l)	Range (mg/l) Minimum	Range (mg/l) Maximum	Number of Observations
1978				
March	0	0	0	0
April	0	0	0	0
May	9.1	7.5	10.0	61
June	9.3	8.6	10.1	44
July	8.4	7.0	9.5	30
August	8.8	4.8	9.9	60
September	8.1	7.6	8.9	77
October	8.4	7.8	9.2	98
November	9.4	8.2	10.6	91
December	8.8	7.4	10.0	53
1979				
January	8.8	6.3	10.4	81
February	8.7	8.3	10.5	64
March	8.9	7.6	10.9	62
April	7.9	7.7	8.5	17
May	7.6	6.7	9.3	24
June	8.1	7.8	8.6	16
July	8.3	7.7	9.0	16
August	8.0	7.0	9.3	62
September	8.3	7.4	10.5	24
October	8.9	7.2	10.9	64
November	8.7	7.4	9.8	64
Summary	8.7	4.8	10.9	1,008

Note: Means are weighted by the number of monthly observations.

TABLE A-5 SUMMARY OF THE INTAKE AND DISCHARGE TEMPERATURES OF THE PITTSBURG POWER PLANT UNITS 1-4 AND UNITS 5 AND 6, MARCH 1978 - NOVEMBER 1979

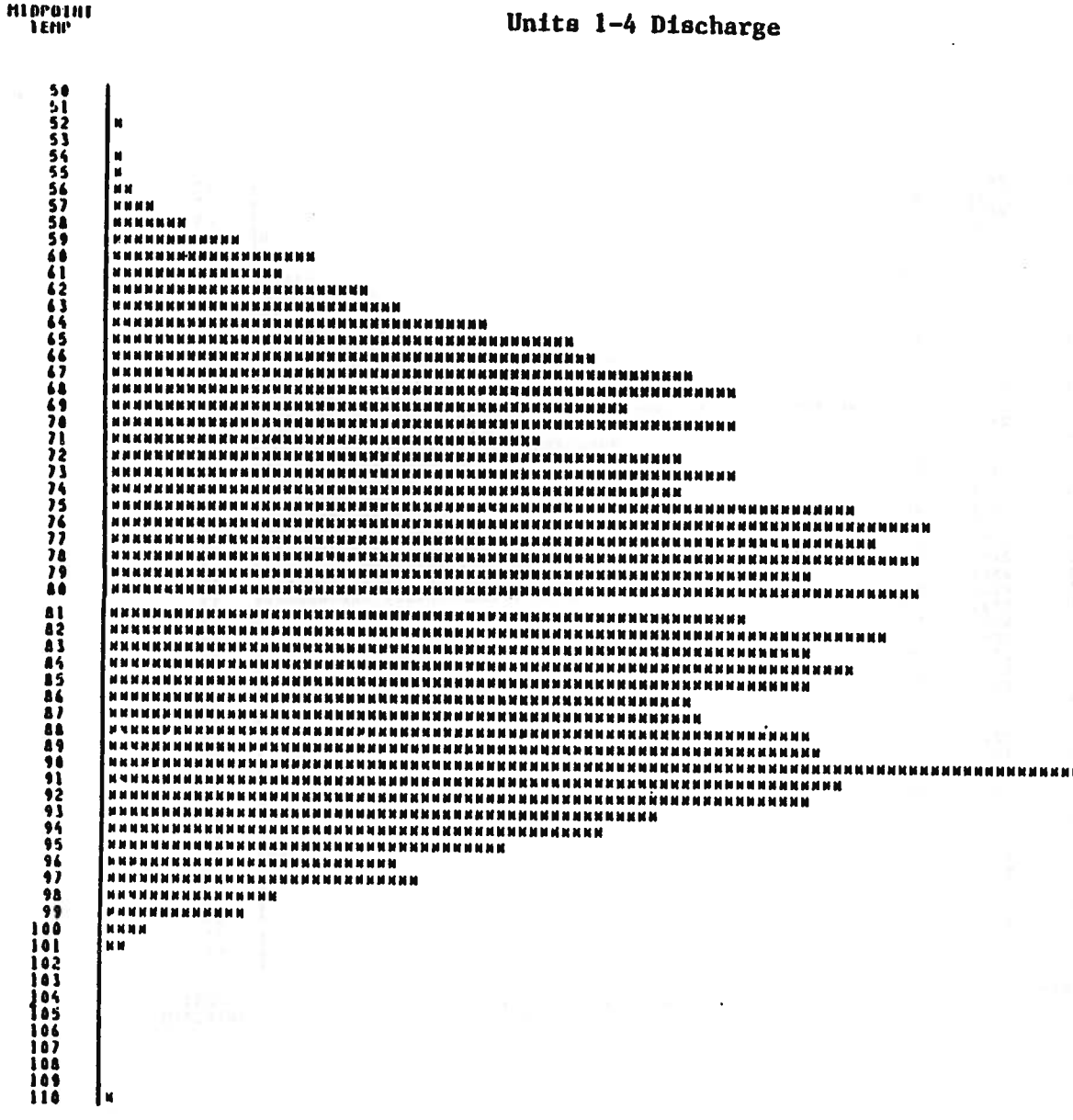
Computer Code	Definition
LOC	PBIU01A1 = Units 1-4 intakes PBCD01 = Units 1-4 discharge PBIU06A1 = Units 5 and 6 intakes PBDU05 = Units 5 discharge PBDU06 = Units 6 discharge
MIDPOINT TEMP	Observed temperature (F)
FREQ	Frequency of observations recorded for each temperature
CUM. FREQ	Cumulative frequency of temperature observations
PERCENT	Percentage of observations represented by each temperature based on frequency of occurrence
CUM. PERCENT	Cumulative percentage represented by each temperature

Units 1-4 Intake



MIDPOINT TEMP	FREQ	CUM. FREQ	PERCENT	CUM. PERCENT
40				
41	0	0	0.00	0.00
42	0	0	0.00	0.00
43	0	0	0.00	0.00
44	0	0	0.00	0.00
45	0	0	0.00	0.00
46	8	8	0.20	0.20
47	30	38	0.76	0.97
48	32	70	0.82	1.78
49	60	130	1.53	3.31
50	62	192	1.58	4.89
51	52	244	1.32	6.21
52	49	293	1.25	7.46
53	58	351	1.48	8.94
54	68	419	1.73	10.67
55	60	479	1.53	12.20
56	73	552	1.86	14.06
57	76	628	1.94	16.00
58	103	731	2.62	18.62
59	141	872	3.59	22.21
60	122	994	3.11	25.32
61	160	1154	4.08	29.39
62	171	1325	4.36	33.75
63	158	1483	4.02	37.77
64	158	1641	4.02	41.80
65	143	1784	3.64	45.44
66	131	1915	3.34	48.78
67	130	2045	3.31	52.09
68	119	2164	3.03	55.12
69	148	2312	3.77	58.89
70	167	2479	4.25	63.14
71	211	2690	5.37	68.52
72	212	2902	5.40	73.92
73	178	3080	4.53	78.45
74	158	3238	4.02	82.48
75	130	3368	3.31	85.79
76	120	3488	3.06	88.84
77	90	3578	2.29	91.14
78	94	3672	2.39	93.53
79	64	3736	1.63	95.16
80	65	3801	1.66	96.82
81	50	3851	1.27	98.09
82	35	3886	0.89	98.98
83	24	3910	0.61	99.59
84	10	3920	0.25	99.85
85	3	3923	0.08	99.92
86	1	3924	0.03	99.95
87	0	3924	0.00	99.95
88	1	3925	0.03	99.97
89	1	3926	0.03	100.00
90	0	3926	0.00	100.00

Units 1-4 Discharge

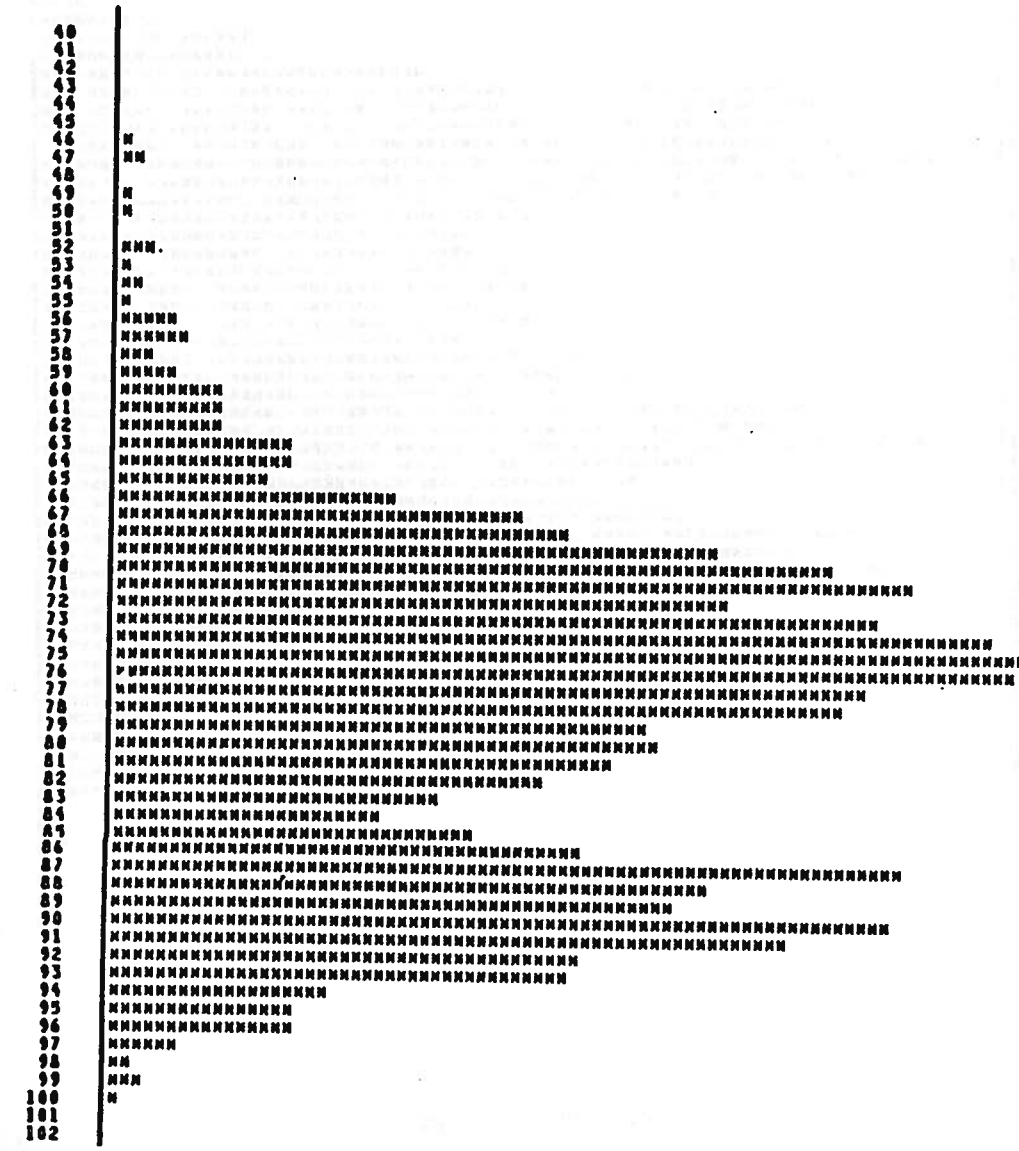


MIDPOINT TEMP	FREQ	CUM. FREQ	PERCENT	CUM. PERCENT
50	0	0	0.00	0.0
51	0	0	0.00	0.0
52	1	1	0.02	0.0
53	0	1	0.00	0.0
54	1	2	0.02	0.0
55	2	4	0.05	0.1
56	3	7	0.07	0.1
57	8	15	0.19	0.3
58	14	29	0.33	0.7
59	24	53	0.57	1.3
60	38	91	0.91	2.1
61	32	123	0.76	2.9
62	47	170	1.12	4.0
63	54	224	1.29	5.3
64	69	293	1.64	6.9
65	85	378	2.02	9.0
66	90	468	2.14	11.1
67	107	575	2.55	13.7
68	115	690	2.74	16.4
69	96	786	2.29	18.7
70	115	901	2.74	21.4
71	79	980	1.88	23.3
72	106	1086	2.53	25.8
73	116	1202	2.76	28.6
74	105	1307	2.50	31.1
75	138	1445	3.29	34.4
76	151	1596	3.60	38.0
77	141	1737	3.36	41.
78	150	1887	3.57	44.9
79	130	2017	3.10	48.0
80	150	2167	3.57	51.6
81	118	2285	2.81	54.4
82	144	2429	3.43	57.8
83	130	2559	3.10	60.9
84	137	2696	3.26	64.2
85	130	2826	3.10	67.
86	107	2933	2.55	69.7
87	110	3043	2.62	72.4
88	130	3173	3.10	75.5
89	131	3304	3.12	78.6
90	180	3484	4.29	82.9
91	135	3619	3.22	86.1
92	129	3748	3.07	89.2
93	102	3850	2.43	91.6
94	91	3941	2.17	93.8
95	74	4015	1.76	95.6
96	54	4069	1.29	96.9
97	58	4127	1.38	98.3
98	32	4159	0.76	99.
99	26	4185	0.62	99.7
100	8	4193	0.19	99.
101	3	4196	0.07	99.9
102	0	4196	0.00	99.9
103	0	4196	0.00	99.9
104	0	4196	0.00	99.9
105	0	4196	0.00	99.9
106	0	4196	0.00	99.9
107	0	4196	0.00	99.9
108	0	4196	0.00	99.9
109	0	4196	0.00	99.9
110	2	4198	0.05	100.0

Unit 6 Intake

MIDPOINT TEMP	FREQ	CUM. FREQ	PERCENT	CUM. PERCENT
40	0	0	0.00	0.00
41	0	0	0.00	0.00
42	0	0	0.00	0.00
43	0	0	0.00	0.00
44	0	0	0.00	0.00
45	11	11	0.30	0.30
46	26	37	0.71	1.00
47	23	60	0.62	1.63
48	55	115	1.49	3.12
49	66	181	1.79	4.91
50	37	218	1.00	5.91
51	40	258	1.09	7.00
52	71	329	1.93	8.93
53	79	408	2.14	11.07
54	89	497	2.41	13.48
55	104	601	2.82	16.30
56	97	698	2.63	18.94
57	89	787	2.41	21.35
58	70	857	1.90	23.25
59	91	948	2.47	25.72
60	112	1060	3.04	28.76
61	145	1205	3.93	32.69
62	144	1349	3.91	36.60
63	123	1472	3.34	39.93
64	119	1591	3.23	43.16
65	98	1689	2.66	45.82
66	92	1781	2.50	48.32
67	66	1847	1.79	50.11
68	72	1919	1.95	52.06
69	105	2024	2.85	54.91
70	167	2191	4.53	59.44
71	217	2408	5.89	65.33
72	287	2695	7.79	73.11
73	238	2933	6.46	79.57
74	198	3131	5.37	84.94
75	179	3310	4.86	89.80
76	126	3436	3.42	93.22
77	105	3541	2.85	96.07
78	58	3599	1.57	97.64
79	54	3653	1.47	99.10
80	24	3677	0.65	99.76
81	4	3681	0.11	99.86
82	4	3685	0.11	99.97
83	1	3686	0.03	100.00
84	0	3686	0.00	100.00
85	0	3686	0.00	100.00

Unit 5 Discharge



MIDPOINT TEMP	FREQ	CUM. FREQ	PERCENT	CUM. PERCENT
40	0	0	0.00	0.00
41	0	0	0.00	0.00
42	0	0	0.00	0.00
43	0	0	0.00	0.00
44	0	0	0.00	0.00
45	0	0	0.00	0.00
46	1	1	0.03	0.03
47	4	5	0.13	0.16
48	0	5	0.00	0.16
49	1	6	0.03	0.19
50	1	7	0.03	0.22
51	0	7	0.00	0.22
52	6	13	0.19	0.41
53	2	15	0.06	0.47
54	4	19	0.13	0.60
55	1	20	0.03	0.63
56	10	30	0.32	0.95
57	11	41	0.35	1.29
58	6	47	0.19	1.48
59	9	56	0.28	1.77
60	18	74	0.57	2.34
61	18	92	0.57	2.90
62	18	110	0.57	3.47
63	29	139	0.92	4.39
64	29	168	0.92	5.30
65	25	193	0.79	6.09
66	47	240	1.48	7.57
67	70	310	2.21	9.78
68	78	388	2.46	12.24
69	103	491	3.25	15.49
70	124	615	3.91	19.41
71	138	753	4.35	23.76
72	105	858	3.31	27.07
73	132	990	4.17	31.24
74	152	1142	4.80	36.04
75	157	1299	4.95	40.99
76	155	1454	4.89	45.88
77	129	1583	4.07	49.95
78	125	1708	3.94	53.90
79	92	1800	2.90	56.80
80	93	1893	2.93	59.73
81	86	1979	2.71	62.45
82	74	2053	2.34	64.78
83	55	2108	1.74	66.52
84	45	2153	1.42	67.94
85	62	2215	1.96	69.90
86	82	2297	2.59	72.48
87	137	2434	4.32	76.81
88	104	2538	3.28	80.09
89	97	2635	3.06	83.15
90	135	2770	4.26	87.41
91	117	2887	3.69	91.10
92	82	2969	2.59	93.69
93	79	3048	2.49	96.18
94	38	3086	1.20	97.38
95	31	3117	0.98	98.36
96	31	3148	0.98	99.34
97	11	3159	0.35	99.68
98	4	3163	0.13	99.81
99	5	3168	0.16	99.97
100	1	3169	0.03	100.00
101	0	3169	0.00	100.00
102				

Unit 6 Discharge

MIDPOINT TEMP	FREQ	CUM. FREQ	PERCENT	CUM. PERCENT
40	0	0	0.00	0.00
41	0	0	0.00	0.00
42	0	0	0.00	0.00
43	0	0	0.00	0.00
44	0	0	0.00	0.00
45	0	0	0.00	0.00
46	0	0	0.00	0.00
47	0	0	0.00	0.00
48	0	0	0.00	0.00
49	0	0	0.00	0.00
50	0	0	0.00	0.00
51	2	2	0.05	0.05
52	0	2	0.00	0.05
53	3	5	0.08	0.13
54	3	8	0.08	0.21
55	4	12	0.11	0.32
56	13	25	0.35	0.67
57	16	41	0.43	1.10
58	15	56	0.40	1.50
59	13	69	0.35	1.85
60	19	88	0.51	2.36
61	21	109	0.56	2.93
62	28	137	0.75	3.68
63	28	165	0.75	4.43
64	40	205	1.07	5.51
65	39	244	1.05	6.55
66	51	295	1.37	7.92
67	93	388	2.50	10.42
68	152	540	4.08	14.50
69	129	669	3.46	17.97
70	141	810	3.79	21.76
71	111	921	2.98	24.74
72	97	1018	2.61	27.34
73	101	1119	2.71	30.06
74	117	1236	3.14	33.20
75	161	1397	4.32	37.52
76	131	1528	3.52	41.04
77	132	1660	3.55	44.59
78	87	1747	2.34	46.92
79	102	1849	2.74	49.66
80	92	1941	2.47	52.14
81	77	2018	2.07	54.20
82	85	2103	2.28	56.49
83	75	2178	2.01	58.50
84	83	2261	2.23	60.73
85	79	2340	2.12	62.85
86	76	2416	2.04	64.89
87	79	2495	2.12	67.02
88	83	2578	2.23	69.25
89	115	2693	3.09	72.33
90	147	2840	3.95	76.28
91	156	2996	4.19	80.47
92	147	3143	3.95	84.42
93	132	3275	3.55	87.97
94	140	3415	3.76	91.73
95	134	3549	3.60	95.33
96	66	3615	1.77	97.10
97	39	3654	1.05	98.15
98	37	3691	0.99	99.14
99	20	3711	0.54	99.68
100	8	3719	0.21	99.89
101	2	3721	0.05	99.95
102	1	3722	0.03	99.97
103	1	3723	0.03	100.00
104	0	3723	0.00	100.00
105	0	3723	0.00	100.00

between salt water and fresh, where electrical conductivities are between 2 and 10 millimhos per centimeter (mmho), about 1-6 ppt salinity, supports a highly productive biotic community. Movement of the entrapment zone up and down the estuary appears to be controlled primarily by the amount of freshwater inflow (Arthur and Ball 1978). A mathematical relationship (model) between inflow and the location of the entrapment zone is established for use in evaluating the effects of varying freshwater inflow on biological productivity. The model can also be used to estimate total dissolved solids (TDS) concentrations, which influence the operation of closed-cycle cooling systems.

The model was developed through a multiple linear regression analysis in which freshwater inflow, tidal stage, and tidal amplitude were evaluated for their influence on saltwater intrusion into the Delta. Electrical conductivity was used to indicate salinity. The conductivity data used to develop the model were obtained from Delta water quality surveys conducted from 1975 to 1978 by the California Department of Water Resources (DWR 1976, 1977, 1978, 1979). The Delta outflow index, an estimate of outflow at Chipps Island prepared by DWR, was used for days for which conductivity data were available.

Tidal amplitudes were excluded from the final version of the equation, because their incorporation did not improve the fit of the regression. Tide stage, on the other hand, was influential: the tide moves the entrapment zone a considerable distance up and down the estuary on each cycle. However, since high tide data predominated in the DWR data base, only data taken near high tide were used in developing the model. Thus, by restricting the model to high tide data, tidal stage was eliminated as a variable, at the cost of restricting the model to use at high tide.

The model is presented below in two forms. The first is used to predict the location of a particular conductivity level, given a Delta outflow; the second, to predict the conductivity at a given location at a given Delta outflow:

$$M_c = 69.94 - 7.52 \ln Q - 1.46\, C_{25} \qquad \text{(A-6)}$$

and

$$C_{25} = \frac{(69.94 - M_c) - 7.52 \ln Q}{1.46} \qquad \text{(A-7)}$$

where

M_c = location at high tide of a given conductivity, in miles from the Golden Gate Bridge

Q = Delta outflow index (cfs x 10^3)

C_{25} = conductivity (corrected to 25 C).

The mean, minimum, and maximum Delta outflow rates (Q) in the months of April through July, the period of greatest involvement of aquatic organisms with the plant, are shown for the period 1969-1979 in Table A-6.

Table A-7 presents several landmarks and their corresponding mile point (MP) designations. Figure A-2 is a graph of the predicted location of the entrapment zone, the area between surface conductivities (at 1 m depth) of 10 and 2 mmho (6 and 1 ppt salinity), over a wide range of Delta outflows. Figure A-3 is a plot of conductivity vs. Delta outflow in the vicinity of Pittsburg (MP 46.6). These figures are provided to indicate possible uses of the model in its two forms.

A.2.2 Verification

The predictive power of the model was verified by comparing predicted conductivities with observed values from a data set not used in developing the model itself. The data set is a series of high-tide conductivity values reported by Arthur and Ball (1978) between 1973 and 1977. Excellent agreement was obtained in comparisons between actual and predicted conductivites. Figure A-4 is a plot of the actual location of conductivities of 2 and 10 mmho, from the data reported by Arthur and Ball, vs. the location as predicted by the model. A linear regression of the observed vs. predcited locations results in a coefficient of determination (r^2) of 0.913, indicating good agreement of the model with observed conductivities.

Figure A-5 is a graph of total dissolved solids concentrations (TDS) measured at Collinsville in 1976, the average TDS predicted by a model developed by Hydroscience, Inc. (Hydroscience 1978), and the high tide surface TDS predicted by

TABLE A-6 DELTA OUTFLOW (cfs) INDEX FOR THE MONTHS OF APRIL, MAY, JUNE, AND JULY, 1969-1979

Year	Value	April	May	June	July
1969	Mean	69,400	64,600	46,600	13,100
	Minimum	60,300	46,500	22,300	8,930
	Maximum	86,700	80,800	67,100	20,500
1970	Mean	11,000	10,800	6,210	5,260
	Minimum	6,780	6,170	3,890	3,620
	Maximum	20,100	13,900	9,810	7,080
1971	Mean	37,000	26,400	21,200	11,700
	Minimum	19,700	19,500	15,400	8,100
	Maximum	63,700	31,900	31,100	17,400
1972	Mean	7,540	5,140	5,000 (a)	6,210
	Minimum	2,640	2,700	2,230	4,230
	Maximum	16,000	8,590	11,100	9,930
1973	Mean	22,200	11,700	7,210	4,600
	Minimum	11,800	5,390	3,990	2,260
	Maximum	36,600	17,400	15,300	5,780
1974	Mean	110,000	25,500	16,900	9,360
	Minimum	34,800	18,100	5,310	4,830
	Maximum	260,000	35,800	26,000	18,200
1975	Mean	34,500	28,800	22,500	11,100
	Minimum	21,300	23,900	10,900	8,180
	Maximum	67,800	35,600	31,500	13,800
1976	Mean	8,840	4,060	3,920	4,350
	Minimum	3,990	3,250	3,070	3,840
	Maximum	23,700	5,030	4,620	5,200
1977	Mean	3,080	4,030	2,510	3,170
	Minimum	1,720	1,340	1,650	1,690
	Maximum	3,800	9,750	3,470	4,550
1978	Mean	60,400	40,800	9,100	3,990
	Minimum	43,700	22,600	3,740	1,440
	Maximum	68,200	63,000	19,400	4,780
1979	Mean	14,300	13,400	5,360	5,400
	Minimum	7,210	7,160	3,800	3,210
	Maximum	37,300	19,000	10,800	12,200

(a) Two anomalous values: -37,433 and -34,260 were replaced by 3,743 and 8,000 to get a provisional mean of approximately 5,000.

TABLE A-7 DISTANCES OF CERTAIN LANDMARKS FROM THE GOLDEN GATE

Landmark	Distance (miles)	Distance (kilometers)
Golden Gate Bridge	0	0
Lone Tree Point	24.8	39.9
Carquinez Bridge (I-80)	27.9	44.9
Point Carquinez	31.1	50.0
Bullshead Point (east of Benicia Bridge)	34.3	55.1
Roe Island (west end)	38.3	61.6
Middle Point	40.8	65.6
Simmons Point (Chipps Island)	44.2	71.1
New York Point (Pittsburg Power Plant)	46.6	75.0
Point Sacramento	49.2	79.2
Emmaton	54.8	88.2
Rio Vista Bridge	61.4	98.8
Antioch Point	51.6	83.0
West Island (Contra Costa Power Plant)	54.8	88.2
Antioch Bridge	55.6	89.4
Jersey Point	58.3	93.8

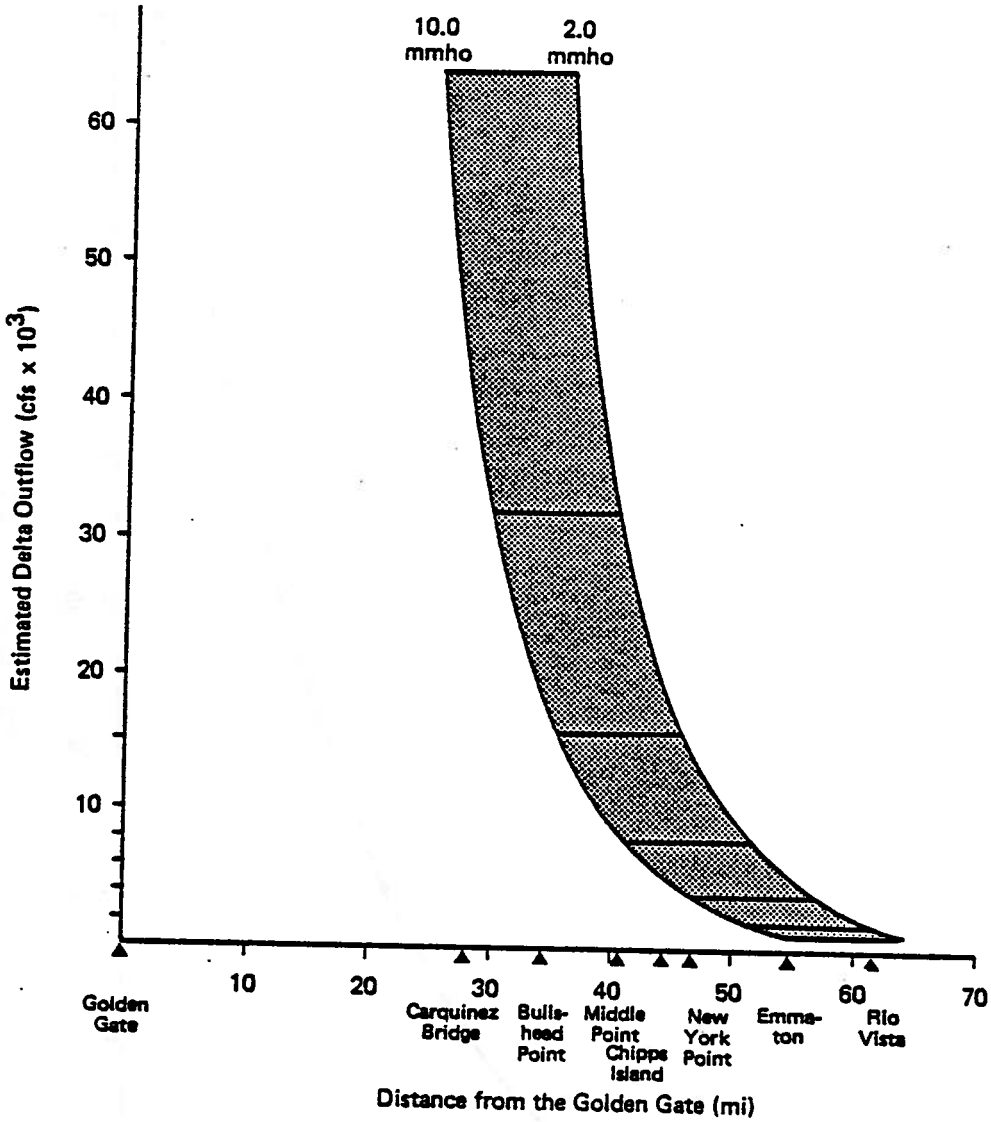


Figure A-2. Location of the entrapment zone (region of 2-10 mmho conductivity surface conductivity at high tide) at various Delta outflows predicted by EA model.

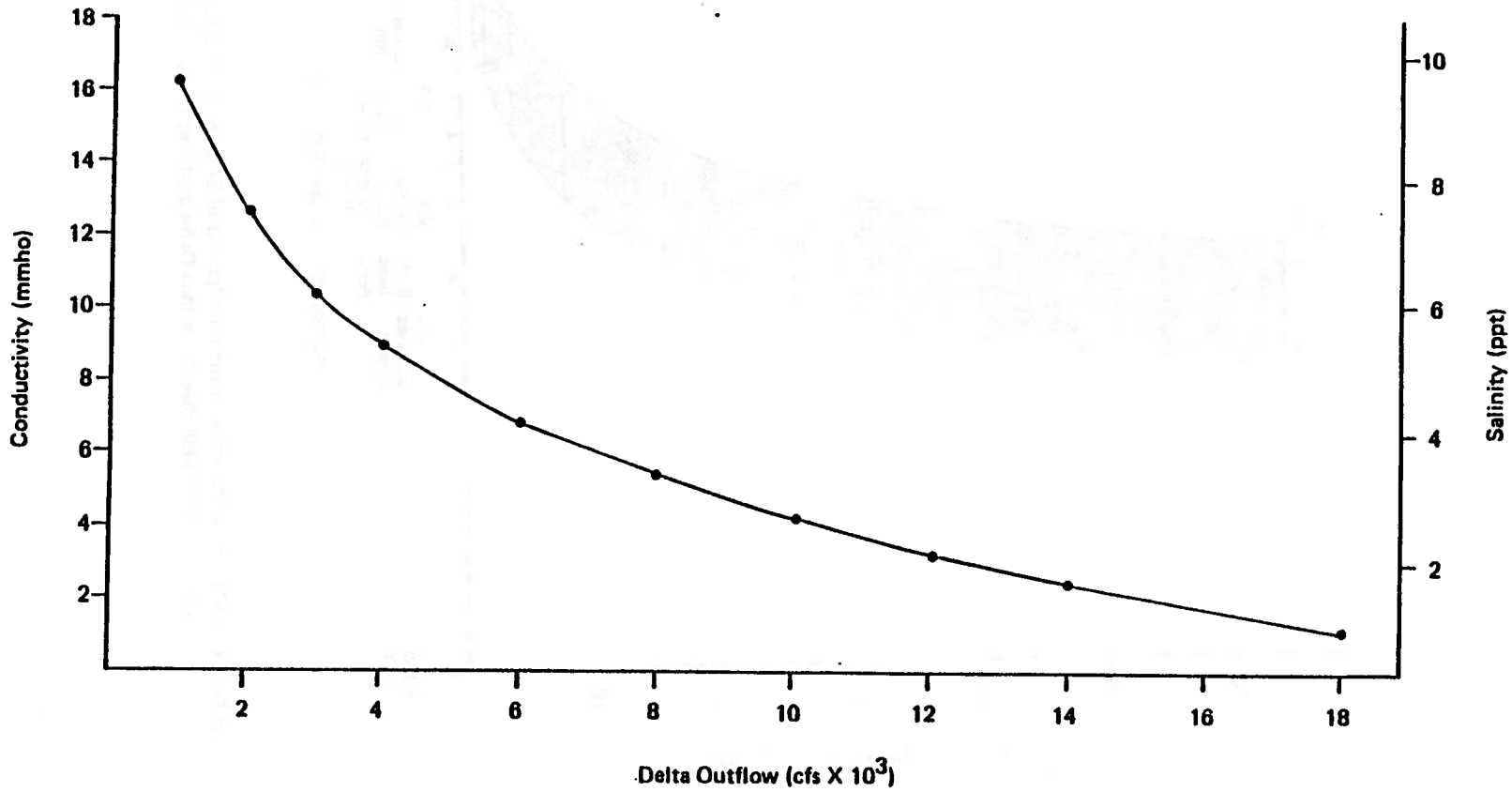


Figure A-3. Conductivity (salinity) at high tide of the water of the Sacramento River near the Pittsburg Power Plant predicted by a Delta outflow conductivity model.

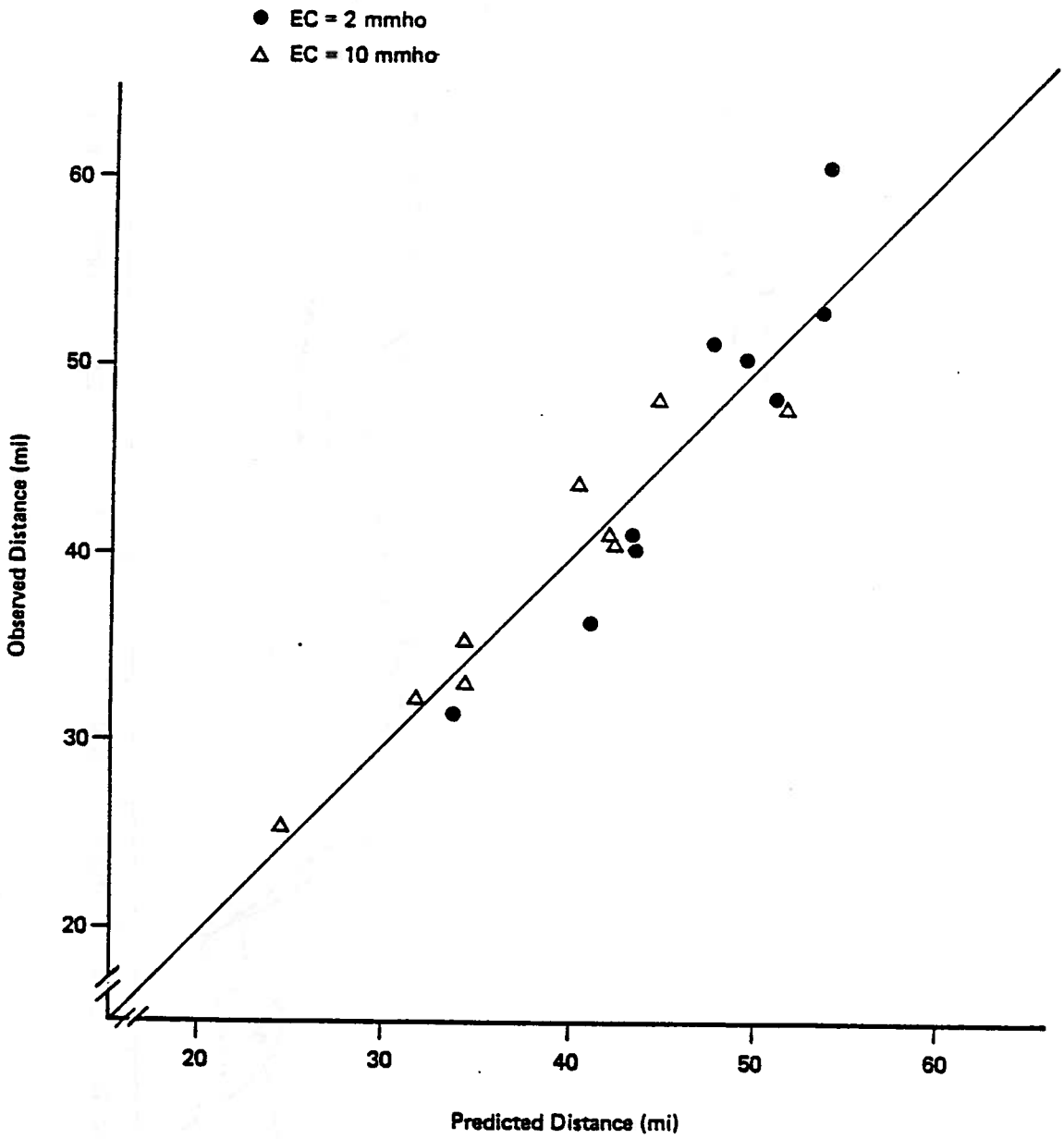


Figure A-4. Observed vs. predicted location (distance from the Golden Gate) of conductivities in the Bay-Delta. (Data from Arthur and Ball 1978.)

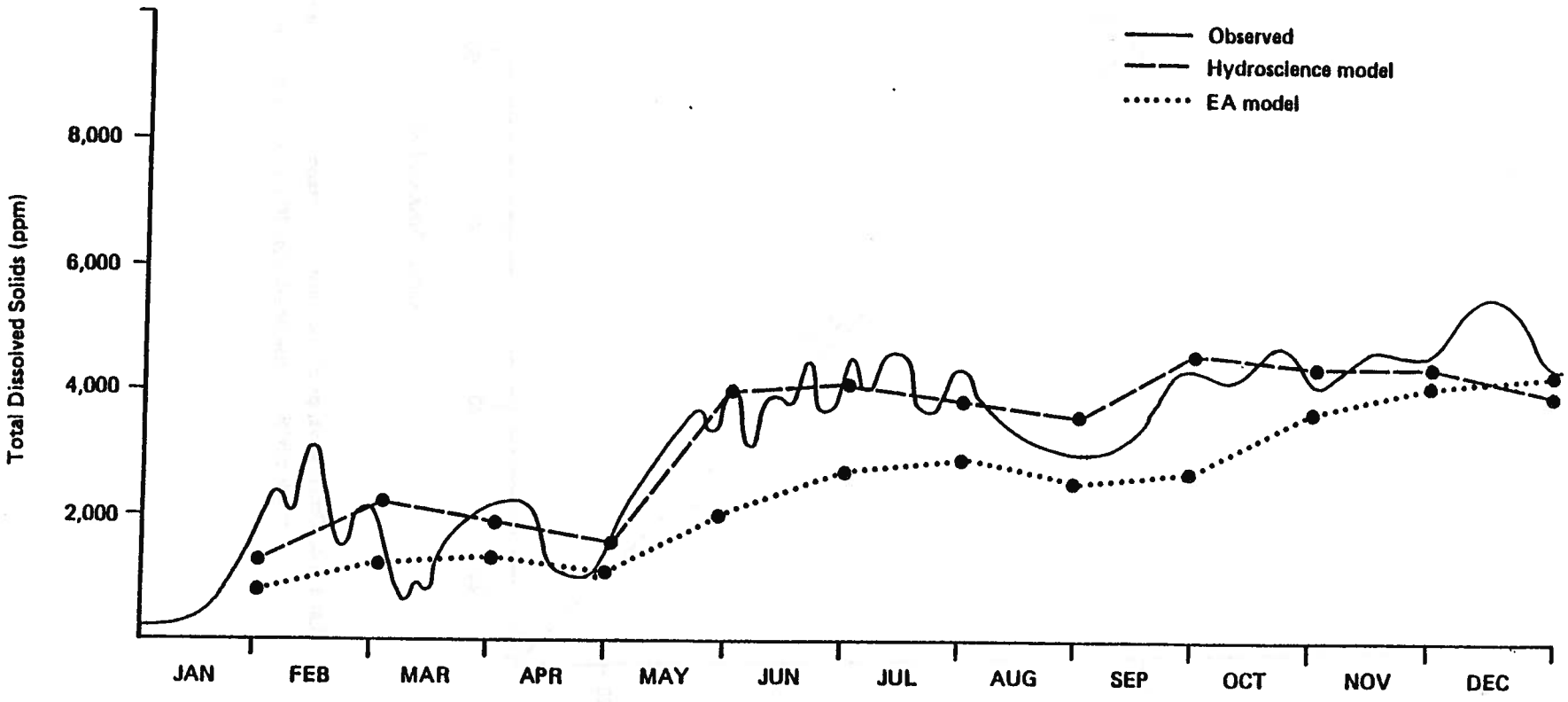


Figure A-5. Observed and predicted concentrations of total dissolved solids in the Sacramento River near Collinsville, 1976. (Data from DWR 1977, Hydroscience 1978.)

the EA model. The predicted values were based on average monthly Delta outflows. Both models follow the general pattern of TDS observed at Collinsville. The major difference between the models is that the EA model predicts TDS at high tide, when the concentration at a given distance up the estuary is usually at its peak, whereas the Hydroscience model predicts average TDS, which will generally be lower. The conductivities predicted by the EA model were converted to TDS with the function $TDS = 0.58\ C_{25}$. This conversion factor was developed from DWR data collected near Collinsville in 1977 and 1978 (DWR 1978, 1979), and will provide a useful estimate of TDS, generally accurate to within ± 7 percent.

A.2.3 Limitations of The Model

The model will provide a reasonable estimate of salinities in the north Bay and the Delta, with certain limitations:

1. Conductivities predicted or used in this model are high tide, surface values.
2. The model is intended for use in conductivities below 15 mmho. Error becomes large above this level.
3. The mile points predicted by the model are generally accurate to within two or three miles.
4. The model is less accurate in predicting salinities in the Delta above the confluence of the two rivers when lower San Joaquin River flows exceed 1,000 cfs or are lower than -1,000 cfs. Estimates may be off by as much as 3.5-7.0 mmho (2,000-4,000 ppm TDS) under those conditions.
5. The model is most reliable under steady state outflow conditions, when Delta outflow has been relatively constant for at least four days before the day for which estimates are being made.

A.3 INTAKE VELOCITY STUDY

An intake velocity study was conducted at the Pittsburg Power Plant by the PGandE's Department of Engineering Research (DER) in support of the biological field surveys conducted at the plant between March 1978 and December 1979. Results of this study are reported in a technical report published by DER (PGandE 1981).

A.4 REFERENCES

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Appendix B

Review of Laboratory and Field Entrainment Survival Studies

B.1 Introduction

B.2 Review of Laboratory Studies Pertinent to Entrainment Survival

B.3 Review of Field Studies Pertinent to Entrainment Survival

B.4 Summary and Conclusions

B.5 References

APPENDIX B: REVIEW OF LABORATORY AND FIELD ENTRAINMENT SURVIVAL STUDIES

B.1 INTRODUCTION

Steam electric generating stations drawing cooling water from a water body entrain a variety of aquatic organisms into the cooling water flow. Plankton and weak-swimming nekton capable of passing through intake screens are carried along with the cooling water through the plant and are subjected to a variety of thermal, physical, and chemical (biocidal) stresses. Entrained organisms are exposed to velocity shear forces, buffeting, and abrasion; to pressure increases in the circulating water pumps; and to rapid temperature increases and pressure decreases in the condensers. They may also be exposed to chlorine during cooling system passage, if this biocide is used to prevent fouling of condenser tube surfaces. In once-through cooling systems, entrained organisms are returned to the source water body, where they are subjected to rapidly decreasing temperatures.

Entrainment survival at power plants was initially thought to be low or negligible, based on the results of several early field studies (Marcy 1971, 1973; Carpenter et al. 1974). Consequently, fine-mesh screening was proposed as a mechanism for preventing entrainment of the larger plankton and weak-swimming nekton. However, recent studies have shown entrainment survival of a variety of aquatic organisms to be high under many plant operating conditions, suggesting that exclusion by means of fine-mesh screening may not be the best means of minimizing impact on aquatic organisms.

The purpose of this review of the results of laboratory and field studies pertinent to entrainment survival is to form a basis for estimating survival at power plants for which no site-specific data exist and to support the results of limited site-specific survival studies. The review is applicable to power plants with once-through cooling systems (entrainment survival at power plants with closed-cycle cooling is assumed to be negligible).

The review is intended to demonstrate that high entrainment survival of a variety of aquatic organisms is likely, and to specify the plant operating conditions under which high survival can be expected.

B.2 REVIEW OF LABORATORY STUDIES PERTINENT TO ENTRAINMENT SURVIVAL

Physical, thermal, and chemical (biocidal) stresses potentially influence entrainment survival. The effects of these stresses have been isolated in laboratory studies.

B.2.1 Physical Stresses

Physical stresses include mechanical buffeting, abrasion, and collision; velocity shear forces; and changes in hydrostatic pressure. Entrained organisms are subjected to physical stresses throughout the cooling system, although such stresses may be greatest in the circulating water pumps and condensers.

The separate effects of most physical stresses have not been isolated in the laboratory. The effects of the combined physical stresses associated with passage of entrained organisms through actual or simulated condenser tubes have been quantified by several investigators (Section B.2.1.1), and the effects of rapid changes in hydrostatic pressure and velocity shear forces have been specifically studied (Sections B.2.1.2 and B.2.1.3, respectively).

B.2.1.1 Condenser Passage Experiments

Condenser passage experiments provide data on the effects of physical stresses on entrained organisms, although they do not allow distinguishing the effects of the various individual physical stresses.

Kerr (1953) passed 100 live young chinook salmon and striped bass through a condenser tube of PGandE's Contra Costa Power Plant operating at full load. Young salmon (34-61 mm in length) withstood condenser passage and an 8.9 C (16 F) temperature rise of 3-1/2 to 5 minutes with no fatalities after a 10-day observation period. Young striped bass (21-46 mm in length) suffered only 6 percent mortality after a 5-day observation period.

Coutant and Kedl (1975) passed young striped bass and larvae of five other species of fish through an experimental condenser tube designed to simulate the physical forces, such as pressure (reductions of one-half atmosphere from acclimation pressure),

turbulence, and shear, associated with actual power plant condensers. Mortalities of organisms passed through the condenser simulator at velocities of 7-19 fps (213-579 cm/sec) were no greater than mortalities of controls. When temperature stress was added (6-minute exposures), resulting mortality was comparable to that observed in laboratory thermal tolerance studies (6-minute exposures), indicating the independence of physical and thermal stresses. The authors concluded that physical damage to larvae of most species due to passage through a typical power plant condenser tube is minimal (<5 percent mortality).

Another condenser simulator study was conducted at New York University (NYU 1979). The investigators concluded that "the mechanical effects of condenser tube passage alone had little or no effect on survival of striped bass yolk-sac larvae and larvae...." In addition, the study showed that "the combined effects of increased temperature, condenser tube passage, and biocide (chlorine) application was essentially additive for striped bass...."

B.2.1.2 Changes in Hydrostatic Pressure

Changes in hydrostatic pressure within the cooling water system may stress entrained organisms. Pressure increases occur at the circulating water pumps, and pressure decreases occur across the condenser tubes. Increases and decreases are usually on the order of one atmosphere (14.7 psi) or less.

Sudden increases in pressure of up to 250 psi above atmospheric pressure for 15 minutes had no significant effect on the development and survival of eggs and larvae of striped bass (determined 24 hours after exposure) (NYU 1974a). Striped bass juveniles tolerated pressure increases of up to 495 psi above atmospheric pressure for 12 minutes (NYU 1975). Such pressure increases are far greater than those found in once-through cooling systems.

Results of NYU (1974a, 1975) studies indicated that striped bass eggs, larvae, and juveniles were more sensitive to decreases than increases in pressure. Sudden pressure drops, either from atmospheric pressure to subatmospheric pressure or from greater-than-atmospheric pressure to atmospheric pressure, often resulted in significantly lower survival of test organisms than of control organisms. Pressure decreases tested ranged from 8.7 to 30 psi. Survival of striped bass eggs and larvae

was reduced by 0-13 percent after exposure to pressure decreases of up to 13 psi from atmospheric pressure (survival measured 24 hours after exposure). Survival of juveniles was reduced by 71 percent after a 13 psi decrease from atmospheric pressure. Pressure decreases from 30 psi above atmospheric pressure back to atmospheric pressure reduced survival of striped bass larvae by 33-88 percent, but had no effect on juveniles.

The macroinvertebrates Gammarus spp. and Neomysis americana were also tested at NYU (1975), and were found to be tolerant of both decreases and increases in pressure. Neither taxon showed significantly reduced survival after exposure to a pressure decrease from an acclimation pressure of 44.7 psia to 1.5 psia for 15 seconds followed by pressure increases to 35-510 psia for 12 minutes. Survival, measured 24 hours after exposure, was reduced by only 5-9 percent. Survival of Gammarus spp. exposed to a pressure decrease from 44.7 psia to 1.5 psia for 15 seconds was reduced by 2 percent.

The NYU (1974a, 1975) pressure studies indicate that pressure decreases are more likely than pressure increases to affect entrained organisms adversely. Pressure decreases in excess of one atmosphere (about 15 psi) may be expected to cause substantial entrainment mortality. Pressure decreases are less tolerable to organisms than increases because of a tendency for gas cavities, such as vacuoles or gas bladders, to explode and for dissolved gases in blood or other body fluids to come out of solution, forming bubbles which can cause physical trauma (gas-bubble disease).

B.2.1.3 Velocity Shear Forces

Shear forces develop when spatial differences in velocity exist in a moving fluid. The greatest shear forces occur close to solid surfaces, such as pipe walls, pump impellers, traveling screens, and water boxes (Marcy et al. 1978). Shear stress has two major components, rotation and deformation.

The only known laboratory studies on the effects of shear stresses on early life stages of fish are those reported by Morgan (1974) and Morgan et al. (1973, 1976). Eggs and larvae of striped bass and white perch were exposed to velocity fields in a chamber formed by three concentric cylinders, the middle one of which was rotated to produce the experimental forces. According to Marcy et al. (1978), Morgan's data suggest

that shear stress alone would be enough to cause up to 100 percent mortality of entrained eggs and larvae. Such conclusions are inconsistent with actual entrainment survival data obtained at numerous power plants across the nation (Secion B.3), and with the results of the condenser simulator studies (Section B.2.1.1). A detailed analysis of Morgan's experiments (EA 1979a) indicates that the effect of velocity forces in these experiments cannot be related mathematically to average boundary shear, and that the experimental results cannot be extrapolated to other flow field situations, such as those occurring in once-through cooling systems.

B.2.2 Thermal Stresses

Entrained organisms are subjected to a rapid rise in temperature as they pass through the condensers. The duration of exposure to elevated temperatures is short, usually on the order of 1-10 minutes, followed by a gradual return to ambient temperature as the warmwater discharge is mixed with the waters of the receiving water body.

The thermal stresses experienced during cooling system passage, and subsequently in the discharge waters, depend upon the temperature rise above ambient (delta-T), the absolute value of the maximum temperature, and the duration of exposure. With brief exposures such as those typical of power plant passage, the maximum temperature and the maximum delta-T tolerated may be higher than those tolerated during extended exposures.

Specialized terminology used in the following sections is defined in this paragraph. A safe temperature is a temperature reported to cause no significant mortality upon exposure for a specified period of time. A TL50 is a temperature leading to mortality of 50 percent of the test organisms upon exposure for a specified period of time. A lethal temperature is a temperature reported to cause mortality of 100 percent of the test organisms upon exposure for a specified period of time. A threshold lethal temperature is a maximum safe temperature, above which significant mortality may be expected. Threshold lethal temperatures can be approximated by subtracting 2 C from TL50s (U.S. EPA 1976).

B.2.2.1 Fish Eggs

Investigations of short-term thermal tolerance have been made for the eggs of eight

species of fish representing five families (Table B-1). All of these families are represented in San Francisco Bay or in the Sacramento and San Joaquin rivers.

Safe temperatures reported for short exposures (5-60 minutes) of fish eggs range from 19 to 35 C (Table B-1). Most of the maximal safe temperatures reported that were less than 30 C were the highest temperatures tested in the experiments. Consequently, threshold lethal temperatures are probably higher than the safe temperatures reported in these cases. This assertion is supported by the observation that threshold lethal temperatures computed by subtracting 2 C from the TL50s reported in Table B-1 exceed 30 C in the majority of the cases. Notable exceptions, were alewife blastulae acclimated to 12-13 C, whose 30-minute TL50 was 26-27 C; northern anchovy blastulae acclimated to 16 C, whose 60-minute TL50 was 30-31 C; and Atlantic tomcod embryos acclimated to 2-3 C, whose 10- to 60-minute TL50s ranged from 20 to 31 C. All three of these species spawn in relatively cold water, especially Atlantic tomcod, a cold-water fish of the north Atlantic coast of North America that spawns in midwinter at temperatures below 5 C.

The data presented in Table B-1 support the general assertion that short exposures to elevated temperatures below 30 C will not harm the eggs of most species of fish. The eggs of some species will tolerate short exposures to temperatures of up to 30 C, but few if any species can be expected to tolerate exposure to temperatures over 35 C.

B.2.2.2 Fish Larvae and Juveniles

Investigations of short-term thermal tolerance have also been made for larvae and juveniles of 20 species of fish (Table B-2). These species represent 13 families, 12 of which are represented in San Francisco Bay or the Sacramento and San Joaquin rivers.

Fish larvae and juveniles are capable of tolerating short exposures (5-60 minutes) to temperatures 10-20 C above ambient during most seasons and 7-10 C above the midsummer maximum ambient temperatures. All of the reported safe temperatures less than 25 C were obtained in experiments in which acclimation temperatures were very low (0-10 C); at higher acclimation temperatures, the safe temperatures would probably be higher. Lethal threshold temperatures, computed by subtracting 2 C from the TL50s reported in Table B-2 for 14 species, ranged from 28 to 38 C, with

TABLE B-1 SUMMARY OF SHORT-EXPOSURE THERMAL TOLERANCE DATA FOR FISH EGGS

Taxon: Common Name	Taxon: Family / Species	Life Stages	Acclimation Temperature (C)	Exposure Duration (min)	Thermal Tolerance Limits: Safe	Thermal Tolerance Limits: TL50	Thermal Tolerance Limits: Lethal	Reference
	Clupeidae							
Alewife	*Alosa pseudoharengus*	Blastula	12-13	30		26-27		EA 1978b
		Tail-bud embryo	12-13	30		32-33		EA 1978b
		Tail-free embryo	12-13	30		32-34		EA 1978b
		4-8 cell	12	(a)	19 (b)			Schubel 1974
		Early blastula	13	(a)	23 (b)			Schubel 1974
		Late gastrula	13-14	(a)	23-24 (b)			Schubel 1974
		Late embryo	15	(a)	25 (b)			Schubel 1974
Blueback herring	*Alosa aestivalis*	Late blastula/early gastrula	18	(a)	28 (b)			Schubel 1974
		Early embryo	15-18	(a)	28 (b)			Schubel 1974
		Late embryo	18	(a)	28 (b)			Schubel 1974
		Early to late embryo	18-21	(a)	28-31 (c,d)			Schubel and Koo 1976
American Shad	*Alosa sapidissima*	Late cleavage	17	variable	27 (b)			Schubel 1974
		Early embryo	17	variable	27 (b)			Schubel 1974
		Mid to late embryo	17	variable	27 (b)			Schubel 1974
		Early to mid embryo	20-21	variable	30-31 (c,d)			Schubel and Koo 1976
	Engraulidae							
Northern Anchovy	*Engraulis mordax*	Blastodisc	16	5		33		Brewer 1976
		Blastopore closure	16	5		36		Brewer 1976
		Blastodisc	16	60		30		Brewer 1976
		Blastopore closure	16	60		31		Brewer 1976
	Percichthyidae							
Striped Bass	*Morone saxatilis*	Gastrula	16-18	30		32		EA 1978b
		Tail-bud embryo	16	30		34		EA 1978b
		Tail-free embryo	19	30		35		EA 1978b
		Late embryo	17	30		35		EA 1978b
		Early to late embryo	15-18	(a)	25-28 (b)			Schubel 1974
		Early to mid embryo	17-20	(a)	32-35 (b)			Schubel and Koo 1976
White perch	*Morone americana*	Early gastrula to early embryo	14-15	(a)	25-25 (b)			Schubel 1974
	Cyprinidae							
Carp	*Cyprinis carpio*	Egg (various stages)	25	10			40	Frank 1974
		Egg (various stages)	25	10	35			Frank 1974
	Gadidae							
Atlantic tomcod	*Microgadus tomcod*	Tail-bud embryo	2-3	10-60		20-27		EA 1978b
		Eyed-up embryo	2-3	10-60		24-31		EA 1978b

(a) 4- to 60-minute exposure followed by gradual cooling to within 1 C of acclimation temperature.
(b) Highest temperature tested resulted in mortality not significantly different from control mortality.
(c) Temperature 5 C higher significantly increased mortality.
(d) In one experiment, a brief (5 min) exposure to 29 C (8 C delta-T) resulted in increased mortality.

TABLE B-2 SUMMARY OF SHORT EXPOSURE THERMAL TOLERANCE DATA FOR FISH LARVAE

Taxon: Common Name	Taxon: Family / Species	Life Stage (size in mm)	Acclimation Temperature (C)	Exposure Duration (min)	Thermal Tolerance Limits: Safe	Thermal Tolerance Limits: TL50	Thermal Tolerance Limits: Lethal	Reference
	Clupeidae							
Alewife	*Alosa pseudoharengus*	Yolk-sac larva	14-24	10-60		32-37		EA 1978b
		Larva	22	30		31		EA 1978b
Blueback herring	*Alosa aestivalis*	Juvenile (34)	25	60		34		EA 1978b
		Yolk-sac larva	20-21	(a)			27-41 (b)	Schubel et al. 1977
American shad	*Alosa sapidissima*	Juvenile (66)	24	60		33		EA 1978b
		Yolk-sac larva	21	(a)			28-41 (b)	Schubel et al. 1977
Atlantic menhaden	*Brevoortia tyrannus*	Larva	10-15	0-40	22-25			Hoss et al. 1974
	Engraulidae							
Northern anchovy	*Engraulis mordax*	Yolk-sac larva	16	5		37		Brewer 1976
		Yolk-sac larva	16	60		32		Brewer 1976
Bay anchovy	*Anchoa mitchilli*	Juvenile (45)	24	60		33		EA 1978b
	Cyprinidae							
Goldfish	*Carassius auratus*	Yolk-sac larva	20	10-30		38-39		EA 1978b
		Larva	23	10-30		37-38		EA 1978b
Spottail shiner	*Notropis hudsonius*	Early juvenile (13-36)	23-26	5-30		36-38		EA 1978b
		Juvenile (20-65)	9-26	60		31-36		EA 1978b
	Ictaluridae							
White catfish	*Ictalurus catus*	Larva	25	10-30		36-37		EA 1978b
		Early juvenile	24-25	10-60		36-38		EA 1978b
		Juvenile (73-77)	8-17	60		31		EA 1978b
Brown bullhead	*Ictalurus nebulosus*	Larva	25	10-60		36-38		EA 1978b
		Early juvenile	26	10-30		36-38		EA 1978b
		Juvenile (44)	24	60		36		EA 1978b

(a) 4- to 60-minute exposure followed by gradual cooling to within 1 C of acclimation temperature.
(b) Highest temperature tested resulted in mortality not significantly different from control mortality.
(c) Temperature 5 C higher significantly increased mortality.
(d) In one experiment, a brief (5 min) exposure to 29 C (8 C delta-T) resulted in increased mortality.

TABLE B-2 (CONT.)

Taxon: Common Name	Taxon: Family Species	Life Stages	Acclimation Temperature (C)	Exposure Duration (min)	Thermal Tolerance Limits: Safe	Thermal Tolerance Limits: TL50	Thermal Tolerance Limits: Lethal	Reference
	Gadidae							
Atlantic tomcod	*Microgadus tomcod*	Yolk-sac larva	2-5	10-60		25-27		EA 1978b
		Larva	3-8	10-60		23-28		EA 1978b
	Cyprinodontidae							
Striped killifish	*Fundulus majalis*	Juvenile	22	10		39-40		Hoss et al. 1971
	Atherinidae							
Atlantic silverside	*Menidia menidia*	Larva	17-20	13	31-34			Austin et al 1975
	Percichthyidae							
Striped bass	*Morone saxatilis*	Yolk-sac larva	15-19	10-30		32-36		EA 1978b
		Larva	20-24	10-60		33-36		EA 1978b
		Early juvenile	18-24	10-60		32-37		EA 1978b
		Yolk-sac larva	19-21	(a)	29 (b)			Schubel et al. 1977
		Larva (5-13)	17-21	(c)	32			Kelly and Chadwick 1971
		Early juvenile (20-38)	16-22	(c)	32			Kelly and Chadwick 1971
		Larva	22	30	29			Coutant and Kedl 1975
White perch	*Morone americana*	Yolk-sac larva	15-22	10-60		31-38		EA 1978b
		Early juvenile	27	10-30		37		EA 1978b
Yellow perch	Percidae							
	Perca flavescens	Yolk-sac larva	15	10-30		30-32		EA 1978b
		Larva	15	10-30		32-35		EA 1978b
	Sparidae							
Pinfish	*Lagodon rhomboides*	Larva	15-20	0-40	27-32			Hoss et al. 1974
		Larva	5-10	0-40	20-25			Hoss et al. 1974
	Sciaenidae							
Spot	*Leiostomus xanthurus*	Larva	5-15	0-40	17-27			Hoss et al. 1974
	Pleuronectidae							
Winter flounder	*Pseudopleuronectes americanus*	Larva	0-12	13	14-28 (d)			Valenti 1974 cited in Schubel et al. 1978
	Bothidae							
Gulf, summer, southern flounder	*Paralichthys* spp.	Larva	5-10	0-40	23-28			Hoss et al. 1974
		Larva	15	0-40	30			Hoss et al. 1974

one exception: the lethal threshold for the cold-water Atlantic tomcod was less than 28 C. However, Atlantic tomcod larvae were capable of tolerating temperature increases (delta-T) of up to 21 C over acclimation temperatures of 3-8 C.

The data presented in Table B-2 support the assertion that, in general, larvae and juveniles of most fish species are not harmed by short exposures to elevated temperatures remaining below 30 C. Threshold lethal temperatures for larvae and juveniles acclimated to temperatures above 10 C exceed 30 C in most cases, although temperatures above 35 C are likely to be lethal to most species.

B.2.2.3 Macroinvertebrates

The short-term thermal tolerances of 13 species of macroinvertebrates have been investigated (Table B-3). Most of the species examined are similar to or closely related to species found in San Francisco Bay or the Sacramento and San Joaquin rivers. At least two of the species, *Neomysis mercedis* and *Rhithropanopeus harrisii*, are common in the San Francisco Bay system.

Safe temperatures reported for short exposures (5-60 minutes) of macroinvertebrates range from 10 to 39 C (Table B-3). Every safe temperature reported as less than 26 C was the highest temperature tested (at a given acclimation temperature). As noted for fish eggs (Section B.2.2.1), threshold lethal temperatures are probably higher than the reported safe temperatures in these cases. Again, this assertion is supported by the observation that threshold lethal temperatures, computed by subtracting 2 C from the TL50s reported in Table B-3, exceed 30 C in most cases.

The more thermally sensitive species tested include the mysid shrimps *Neomysis mercedis* and *N. americana* and the true shrimp *Crangon septemspinosa*. Lethal thresholds for these species range from approximately 28 to 32 C, depending on duration of exposure and acclimation temperature. *Neomysis mercedis* appears to be the most sensitive of the three species: when acclimated to 17-21 C it will not tolerate 10-minute exposures to temperatures in excess of 28-29 C (EA unpublished data).

The data presented in Table B-3 support the assertion that many macroinvertebrates can tolerate short exposure to elevated temperatures in excess of 30 C, while some,

TABLE B-3 SUMMARY OF SHORT EXPOSURE THERMAL TOLERANCE DATA FOR MACROINVERTEBRATES

Group/Species	Acclimation Temperature (C)	Exposure Duration (min)	Thermal Tolerance Limits Safe	Thermal Tolerance Limits TL50	Reference
Amphipoda (amphipods)					
Gammarus spp.	3	30		28	Lauer et al. 1974
	11	5-60		30-33	Lauer et al. 1974
	20	5-60		34-36	Lauer et al. 1974
	26	5-60		37-39	Lauer et al. 1974
Gammarus spp.	26	5-60	34		Ginn et al. 1976
	27	5	38		Ginn et al. 1976
Gammarus spp.	5-25	4(a)	10-30		Burton et al. 1976
Gammarus limnaeus	1-3	60	26		Krog 1954
	15-22	60	30-32		Krog 1954
Gammarus tigrinis	25	5-60	33-35		Krog 1954
Gammarus daiberi	25	5-60	33-35		Krog 1954
Gammarus spp.	25	10-60		35-39	EA 1978b
Monoculodes edwardsi	26	5-30		36-37	Lauer et al. 1974
Mysidacea (opossum shrimp)					
Neomysis americana	24-26	5-30		33-34	Lauer et al. 1974
Neomysis americana	24	10-30		31-34	EA 1978b
Neomysis americana	5	4(a)	10(b)		Burton et al. 1976
Neomysis awatschensis(c)	14-22	2-6(a)	30-31(d)		Hair 1971
Neomysis mercedis	17-21	10		30-31	EA (unpublished data)
Caridea (shrimps)					
Crangon septemspinosa	16-24	10-60		31-33	EA 1978b
Palaemonetes spp.	5-30	4(a)	10-35(b)		Burton et al. 1976
Brachyura (crabs)					
Callinectes sapidus	15-30	4(a)	20-35(b)		Burton et al. 1976
Rhithropanopeus harrisii	15-30	4(a)	20-35(b)		Burton et al. 1976
Insecta (insects)					
Hydropsyche sp. (caddisfly)	7-24	1-30	28(e)		Sherberger et al. 1977
Isonychia sp. (mayfly)	4-24	1-40	27(e)		Sherberger et al. 1977
Chaoborus sp. (midge)	25	30	39		NYU 1974b
Chaoborus sp. (midge)	19-26	10-60		39-43	EA 1978b

(a) Constant temperature exposure was followed by gradual cooling to acclimation temperature over 16 minutes.
(b) Highest temperature tested resulted in no mortality.
(c) N. awatschensis has been determined to be N. mercedis.
(d) Reproduction (release of young) was inhibited at temperatures above 28 C.
(e) Some exposures to temperatures greater than the reported safe level were tolerated.

such as N. mercedis, cannot tolerate elevated temperatures in excess of approximately 28-29 C. In general, threshold lethal temperatures for macroinvertebrates are higher than those for fish eggs and larvae.

B.2.3 Chemical (Biocidal) Stresses

Entrained organisms may be exposed to chlorine when it is intermittently injected into the cooling water flow as a routine maintenance procedure to control fouling organisms. Average concentrations at condenser inlets are usually less than 0.5 mg/liter total residual chlorine. Organisms entrained during chlorination are exposed to chlorine at varying concentrations for periods of up to a few minutes.

The effects on freshwater and marine organisms of short exposures to chlorine have been extensively reviewed (Brungs 1973; Mattice and Zittel 1976; Morgan and Carpenter 1978; Envircsphere Company 1978, 1979). The extent of this variability has made it difficult to establish chlorine criteria to protect aquatic life.

Mattice and Zittel (1976) were the first to review the literature and organize the chlorine toxicity data to provide models for estimating exposure-dependent threshold lethal concentrations for the most sensitive marine and freshwater organisms.

Envirosphere Company (1978, 1979) subsequently refined and updated the Mattice and Zittel models. The effects of specific power plant chlorine exposures can be predicted using the Envirosphere Company models. For example, for an exposure of 5 minutes the predicted threshold lethal concentration is 0.34 mg/liter total residual chlorine (TRC) for marine organisms and 0.99 mg/liter TRC for freshwater organisms. At 60 minutes, the predicted threshold lethal concentrations are 0.14 mg/liter TRC for marine organisms and 0.25 mg/liter for freshwater organisms. If chlorine exposure for a given duration exceeds the corresponding lethal threshold concentration, mortality of entrained organisms can be expected. The duration of the resulting entrainment mortality will depend on the amount by which the exposure exceeds the lethal threshold.

B.3 REVIEW OF FIELD STUDIES PERTINENT TO ENTRAINMENT SURVIVAL

Discussed in this section are pertinent results of entrainment survival studies

conducted at PGandE power plants and at plants outside of the PGandE system. These studies share a common experimental design. Cooling water samples are withdrawn from the discharge of a power plant and from a suitable control area, usually the plant intake. The control samples account for sampling mortality and any other mortality unrelated to entrainment. Numbers of live and dead organisms in the discharge and control samples are determined immediately. Live organisms may be held for up to 96 hours to allow detection of possible delayed entrainment effects. Entrainment mortality is estimated by comparing the proportions dead in the discharge and control samples, initially or at the end of the delayed-effects observation period.

Two methods of comparing the proportions dead in the discharge and control samples have evolved. By the first method, entrainment mortality is computed by subtracting the control proportion dead from the discharge proportion dead. This method results in the underestimation of entrainment mortality, especially when control mortality is high. By the second method, entrainment mortality is computed as follows:

$$p_e = \frac{p_D - p_I}{1 - p_I} \qquad \text{(B-1)}$$

where

p_e = proportion not surviving entrainment
p_D = proportion dead in discharge
p_I = proportion dead in intake.

This method provides more accurate estimates of entrainment mortality (or survival) than the first method, because entrainment mortality is not underestimated when control mortality is high.

In the following review of entrainment survival studies involving ichthyoplankton and macroinvertebrates, any entrainment mortality estimates originally computed using the first method have been recomputed using the second method. This was done so that all mortality estimates could be compared on a common basis.

To further facilitate comparison of studies, entrainment mortality estimates presented are based on discharge temperatures less than 30 C whenever possible. Thermal stress was thereby excluded as a factor affecting the entrainment mortality estimates. The mortality estimates presented largely represent mortality due to physical stresses alone.

B.3.1 Entrainment Survival Studies at PGandE Plants

Entrainment survival studies were conducted during 1978 and 1979 at the Potrero, Pittsburg, and Contra Costa power plants. The Potrero Power Plant study focused on Pacific herring larvae (PGandE 1980). The Pittsburg and Contra Costa power plant studies focused on striped bass larvae, gammaridean amphipods, and *Neomysis mercedis*.

All of the 1978-1979 PGandE studies shared the common experimental design discussed at the beginning of this section: control samples were collected from the plant intakes; both control and discharge samples were collected using high-capacity pumps; and live fish larvae were held for 96 hours to allow detection of any delayed effects of entrainment.

At the Potrero Power Plant, control sample mortality of Pacific herring larvae decreased with increasing body length. However, entrainment mortality did not vary with body length, so a single estimate of the proportion not surviving entrainment was computed. That estimate was 0.21 (i.e., 21 percent mortality). Ninety-five percent confidence limits on the true percentage not surviving were 5 and 37 percent. No delayed effects of entrainment were detected. Survival of larvae collected from the intake and discharge did not differ significantly over the 96-hour observation period. Discharge temperatures during the study did not exceed 20 C.

Results of the 1978-1979 entrainment survival studies at the Pittsburg and Contra Costa power plants indicate that at discharge temperatures below 30 C mortality is low (Table B-4). Mortality of striped bass larvae and juveniles at temperatures less than 30 C ranged from 19 to 40 percent at Pittsburg Power Plant Units 1-4 and Units 5 and 6 and Contra Costa Power Plant Units 1-5 and Units 6 and 7. Mortality for the opossum shrimp, *Neomysis mercedis* ranged from 10 to 28 percent, and that of gammaridean amphipods ranged only from 4 to 5 percent (Table B-4). Mortality of all

TABLE B-4 SUMMARY OF ENTRAINMENT MORTALITY ESTIMATES FOR STRIPED BASS LARVAE, *NEOMYSIS MERCEDIS*, AND GAMMARIDEAN AMPHIPODS AT THE PITTSBURG AND CONTRA COSTA POWER PLANTS

Taxon	Discharge Temperature (C)	Mortality (%)			
		Pittsburg		Contra Costa	
		Units 1-4	Units 5, 6	Units 1-5	Units 6, 7
Striped bass	30	38	40	19	21
	30-31.9	69	57	65	58
	32-33.9	100(a)	77	79	75
	34	100(a)	—	95	95
Neomysis mercedis	30	—	10	—	28
	30-31.9	—	72	—	55
	32-33.9	—	100	—	100
	34	—	100	—	100
Gammaridean Amphipods	30	—	4	—	5
	30-31.9	—	0	—	12
	32-33.9	—	59	—	21
	34	—	79	—	84

(a) Low sample size—estimate imprecise.

Note: (—) indicates no data available.

taxa increased at higher discharge temperatures. N. mercedis appeared to be the most sensitive to elevated temperatures of the three taxa.

Stevens and Finlayson (1978) studied entrainment mortality of striped bass larvae at the Pittsburg and Contra Costa power plants in 1976, and estimated survival to be 14 percent when discharge temperatures remained at or below 31 C. The low mortality observed by Stevens and Finlayson (1978) is consistent with that observed by Ecological Analysts.

The above studies indicate that at low discharge temperatures (<30 C) entrainment mortality at the PGandE power plants investigated is low (4-40 percent) Mortality has been estimated to be low even for Pacific herring, a delicate clupeid species.

Clupeids have been shown to be among the taxa most sensitive to entrainment stresses (see Section B.3.2.1). Consequently, the mortality of the species not tested is likely to be as low or lower than that of Pacific herring.

B.3.2 Entrainment Survival at Plants Outside the PGandE System

Entrainment survival studies have been conducted at a variety of freshwater, estuarine, and marine power plants across the United States. EA (1979a) critically reviewed and summarized survival studies dealing with ichthyoplankton. The results of that review are presented here along with a summary of survival studies dealing with macroinvertebrates.

B.3.2.1 Ichthyoplankton

Ichthyoplankton entrainment survival studies have been conducted with 21 taxa at 11 different power plants, six on freshwater systems, three on brackish-water systems, and two on marine (saltwater) systems, summarized in Table B-5. The entrainment mortality estimates in Table B-5 represent, for the most part, mortality due to physical entrainment stresses. As indicated in the table, in some cases it was not possible to distinguish whether or not thermal stresses were present. The entrainment mortality estimates in Table B-5 pertain to fish larvae and juveniles only; no mortality data pertaining to fish eggs are available.

TABLE B-5 ICHTHYOPLANKTON ENTRAINMENT MORTALITY ESTIMATES FOR POWER PLANTS OUTSIDE OF THE PSandE SYSTEM

Power Plant	Water Body	Salinity	Taxon: Common Name	Taxon: Family Species	Life Stage (Length mm)	Percentage Mortality[a]	Reference
Connecticut Yankee	Connecticut River	Fresh	Herrings	Clupeidae	(15) (20)	71 87	Marcy 1976
Bowline Point	Hudson River	Brackish	Striped bass	*Morone saxatilis*	Larva Juvenile	12 10	EA 1978a
			White perch	*Morone americana*	Larva Juvenile	18 20	EA 1978a
			Herrings	Clupeidae	Larva	23	EA 1978a
			Atlantic tomcod	*Microgadus tomcod*	Yolk-sac larva Larva	8 46	EA 1978a
			Silversides	*Menidia* spp.	Larva	57[b]	EA 1978a
Roseton	Hudson River	Fresh	Striped bass	*Morone saxatilis*	Larva Juvenile	40 0	EA 1978g
			White perch	*Morone americana*	Larva	49	EA 1978g
			Herrings	Clupeidae	Larva	71	EA 1978g
			Atlantic tomcod	Microgadus tomcod	Yolk-sac larva	69[c]	EA 1978g
			Tessellated darter	*Etheostoma olmstedi*	Larva	2[b]	EA 1976b, 1978f
			Minnows	Cyprinidae	Larva	11	EA 1976b, 1978f
Lovett	Hudson	Brackish	Striped bass	*Morone saxatilis*	Larva	25	EA 1978d
			White perch	*Morone americana*	Larva	26	EA 1977, 1978d
			Herrings	Clupeidae	Larva	43	EA 1978d
Danskammer Point	Hudson River	Fresh	Striped bass	*Morone saxatilis*	Larva	5	EA 1976a
			White perch	*Morone americana*	Larva	0	EA 1976a
			Herring	Clupeidae	Larva	44	EA 1976a
			Minnows	Cyprinidae	Larva	48	EA 1976a
Indian Point	Hudson River	Brackish	Striped bass	*Morone saxatilis*	Yolk-sac larva Larva	37 15	EA 1978c
			White perch	*Morone americana*	Larva	23	EA 1978c
				Morone spp.	Larva	14[d]	Lauer et al. 1974
			Herrings	Clupeidae	Larva	60	EA 1978c

(a) Mortality in absence of thermal effects. Values represent annual estimates, or means of annual estimates, for those studies conducted for more than one year.
(b) Some thermal effects may be included.
(c) Represents weighted average of 22% for 2 circulating pumps operating and 77% for 3 circulating pumps operating.
(d) Mortality recomputed using Equation B-1 in text.

Source: EA 1979a.

TABLE B-5 (CONT.)

Power Plant	Water Body	Salinity	Taxon: Common Name	Taxon: Family Species	Life Stage (Length mm)	Percentage Mortality[a]	Reference
Port Jefferson	Long Island Sound	Salt	Winter flounder	Pseudopleuronectes americanus	Larva	0	EA 1978e
			Sand lances	Ammodytes spp.	Larva	73	EA 1978e
			American eel	Anguilla rostrata	Juvenile	0	EA 1978e
Brunswick	Cape Fear	Salt	Bay anchovy	Anchoa mitchilli	—	73[b,d]	Copeland et al. 1975
			Spot	Leiostomus xanthurus	—	25[b,d]	Copeland et al. 1975
			Atlantic croaker	Micropogon undulatus	—	36[b,d]	Copeland et al. 1975
			Gobies	Gobionellus spp.	—	40[b,d]	Copeland et al. 1975
			Gobies	Gobiosoma spp.	—	65[b,d]	Copeland et al. 1975
			Silversides	Atherinidae	—	73[b,d]	Copeland et al. 1975
Quad Cities	Mississippi River	Fresh	Freshwater drum	Aplodinotus grunniens	Yolk-sac larva to juvenile	38	Bastaino et al. 1978
			Minnows	Cyprinidae	Yolk-sac larva to juvenile	49	Bastaino et al. 1978
			Carp	Cyprinus carpio	Larva	21	Bastaino et al. 1978
			Suckers	Catostomidae	Larva	40	Bastaino et al. 1978
Monticello	Mississippi River	Fresh	Fathead minnow	Pimephales promelas	(36–60)	22	Knutson et al. 1976
Ginna	Lake Ontario	Fresh	Alewife	Alosa pseudoharengus	(≥ 15)	77	EA 1979b

The entrainment mortality estimates in Table B-5 range from a low value of 0 percent for striped bass juveniles at Roseton Power Plant (EA 1978g), white perch larvae at Danskammer Point Power Plant (EA 1976a), and winter flounder larvae and American eel juveniles at Port Jefferson Power Plant (EA 1978e) to a high value of 87 percent estimated for clupeids (<20 mm) entrained at the Connecticut Yankee Power Plant (Marcy 1976). Nearly three-quarters of the mortality estimates in Table B-5 are less than 50 percent.

Table B-6 summarizes entrainment mortality data by taxonomic category. Of the 13 families represented in this listing, Clupeidae*, Engraulidae, Atherinidae, and Ammodytidae appear relatively sensitive to entrainment (57-87 percent mortality). Entrainment mortality estimates of the Gadidae, Gobiidae, and Catostomidae were intermediate (40-65 percent), and those for Anguillidae, Cyprinidae, Percichthyidae, Percidae, Sciaenidae, and Pleuronectidae were low (0-38 percent).

The results of field studies of entrainment survival reported in Tables B-5 and B-6 demonstrate that mortality under certain plant operating conditions for hardy species may be as low as 0 percent. Mortality has been shown to vary widely among species.

B.3.2.2 Macroinvertebrates

Macroinvertebrate entrainment survival studies conducted with 10 taxa at six different power plants, three on freshwater systems and three on brackish water systems, are summarized in Table B-7. The entrainment mortality estimates in Table B-7 represent, for the most part, mortality due to physical entrainment stresses, although, as noted in the table, in some cases thermal effects may have been involved.

The entrainment mortality estimates presented in Table B-7 range from 0 percent for *Gammarus daiberi* entrained at Bowline Point, *Neomysis americana* entrained at Lovett, and *Chaoborus punctipennis* entrained at Danskammer Point, three plants located on the Hudson River, to 27 percent for *Monoculodes edwardsi* entrained at Lovett Power Plant. Almost three-quarters of the mortality estimates reported were

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* Clupeids entrained in a freshwater environment appear to be more vulnerable to entrainment damage than those in brackish water. The median clupeid mortality at fresh- and brackish-water sites was 74 and 48 percent, respectively. The range of mortality values was 44-79 percent at freshwater sites and 23-60 percent at brackish-water sites.

TABLE B-6 TAXONOMIC SUMMARY OF ICTHYOPLANKTON ENTRAINMENT MORTALITY ESTIMATES FOR POWER PLANTS OUTSIDE OF THE PGandE SYSTEM

Taxon				Percentage Mortality[b]	
Order	Common Name	Family Species	N[a]	Median	Range
Anguilliformes	Freshwater eels	Anguillidae	1	0	—
	American eel	*Anguilla rostrata*	1	0	—
Clupeiformes	Herrings	Clupeidae	7	60	23-79
	Alewife	*Alosa pseudoharengus*	1	77	—
	Anchovies	Engraulidae	1	73	—
	Bay anchovy	*Anchoa mitchilli*	1	73	—
Cyprinidotiformes	Silversides	Atherinidae	2	65	57-73
		Menidia spp.	1	57	—
Cypriniformes	Minnows	Cyprinidae	4	22	11-49
	Fathead minnow	*Pimephales promelas*	1	22	—
	Carp	*Cyprinus carpio*	1	22	—
	Suckers	Catostomidae	1	40	—
Gadiformes	Codfishes	Gadidae	2	48	27-69
	Atlantic tomcod	*Microgadus tomcod*	2	48	27-69

(a) Number of power plants for which estimates have been made.
(b) Average of all life stages.

Source: EA 1979a.

TABLE B-6 (CONT.)

Taxon				Percentage Mortality[b]	
Order	Common Name	Family Species	N[a]	Median	Range
Perciformes	Sand lances	Ammodytidae	1	73	—
		Ammodytes spp.	1	73	—
	Gobies	Gobiidae	1	52	40-65
		Gobionellus spp.	1	40	—
		Gobiosoma spp.	1	65	—
	Temperate basses	Percichthyidae	5	18	0-49
	White perch	*Morone americana*	5	23	0-49
	Striped bass	*Morone saxatilis*	5	20	5-26
		Morone spp.	1	14	—
	Perches	Percidae	1	2	—
	Tessellated darter	*Etheostoma olmstedi*	1	2	—
	Drums	Sciaenidae	2	36	25-38
	Freshwater drum	*Aplodinotus grunniens*	1	38	—
	Spot	*Leiostomus xanthurus*	1	25	—
	Atlantic croaker	*Micropogon undulatus*	1	36	—
Pleuronectiformes	Flounders	Pleuronectidae	1	0	—
	Winter flounder	*Pseudopleuronectes americanus*	1	0	—

TABLE B-7 MACROINVERTEBRATE ENTRAINMENT MORTALITY ESTIMATES FOR POWER PLANTS OUTSIDE OF THE PGandE SYSTEM

Power Plant	Water Body	Salinity	Taxon	Percentage Mortality[a]	Reference
Fort Calhoun	Missouri River	Fresh	Total drifting macroinvertebrates	12[b,c]	Carter 1978
			Ephemeroptera	8[b,c]	Carter 1978
			Hydropsychidae	8[b,c]	Carter 1978
			Other Trichoptera	8[b,c]	Carter 1978
			Chironomidae	16[b,c]	Carter 1978
			Other Diptera	13[b,c]	Carter 1978
			Other organisms	12[b,c]	Carter 1978
Roseton	Hudson River	Fresh	*Gammarus daiberi*	4	Cannon et al. 1978
			Chaoborus punctipennis	4	Cannon et al. 1978
Bowline Point	Hudson River	Brackish	*Gammarus daiberi*	0	Cannon et al. 1978
			Monoculodes edwardsi	8	Cannon et al. 1978
			Chaoborus punctipennis	2[c]	Cannon et al. 1978
			Neomysis americana	6	Cannon et al. 1978
Danskammer Point	Hudson River	Fresh	*Gammarus daiberi*	12	Cannon et al. 1978
			Chaoborus punctipennis	0	Cannon et al. 1978
Lovett	Hudson River	Brackish	*Gammarus daiberi*	4	Cannon et al. 1978
			Monoculodes edwardsi	27	Cannon et al. 1978
			Chaoborus punctipennis	1	Cannon et al. 1978
			Neomysis americana	0	Cannon et al. 1978
Indian Point	Hudson River	Brackish	*Gammarus* spp.	1[b]	Lauer et al. 1974
			Monoculodes edwardsi	2[b]	Lauer et al. 1974
			Neomysis americana	8[b]	Lauer et al. 1974

(a) Mortality in absence of thermal effects. Values represent annual estimates, or means of annual estimates for those studies conducted for more than one year.
(b) Entrainment mortality recomputed using Equation B-1 in text.
(c) Some thermal effects may be included.

less than 10 percent. These data indicate that entrainment mortality is low for a variety of macroinvertebrates, and is generally lower than that for larval fish (Section B.3.2.1).

B.4 SUMMARY AND CONCLUSIONS

Entrainment survival is influenced by physical, thermal, and chemical (biocidal) stresses potentially incurred by entrained organisms during passage through cooling water systems. The effects of some of these different stresses have been isolated in laboratory studies.

Entrained organisms appear to be tolerant of the physical stresses associated with condenser passage. Pressure increases at the circulating water pumps are not expected to cause significant entrainment mortality, but large pressure decreases (more than about one atmosphere) across the condensers may damage entrained organisms.

Discharge temperatures greater than 30 C generally reduce entrainment survival, although thermal tolerance varies widely among species, and among life stages within species. Some species acclimated to temperatures of 26 C or less are capable of tolerating short exposures (5-60 minutes) to elevated temperatures as high as 40 C.

Entrainment survival may be reduced during condenser chlorination. Studies have shown that the maximum chlorine concentration tolerated by a given species or life stage is a function of the exposure duration. The maximum concentrations tolerated by the most sensitive marine and freshwater species are 0.34 and 0.99 mg/liter total residual chlorine (TRC) respectively, when exposure duration is five minutes. At a duration of 60 minutes, considerably longer than the usual duration in once-through cooling water systems, the maximum concentrations tolerated by the most sensitive marine and freshwater organisms are 0.14 and 0.25 mg/liter TRC, respectively. Chlorine tolerance varies widely among species, and therefore the concentrations tolerated by many species exceed the levels reported for the most sensitive species.

Field studies of entrainment survival demonstrate that mortality of hardy species under certain plant operating conditions may be as low as 0 percent. Mortality has been shown to vary widely among species. The families of fish whose larvae are most

sensitive include Clupeidae, Engraulidae, Atherinidae, and Ammodytidae. Mortality of entrained macroinvertebrates is as low as or lower than that of fish larvae. No data are available on the mortality of entrained fish eggs.

Entainment mortality of fish larvae and macroinvertebrates at PGandE power plants has been shown to be low when discharge temperatures remain below 30 C. Mortality of striped bass larvae at the Pittsburg and Contra Costa power plants ranged from 19 to 40 percent. Mortality of the opossum shrimp Neomysis mercedis ranged from 10 to 28 percent, and that of gammaridean amphipods ranged only from 4 to 5 percent. Mortality of Pacific herring, a sensitive clupeid species, averaged 21 percent at the Potrero Power Plant, considerably lower than values observed for clupeids at plants outside the PGandE system.

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Appendix C

Impact Assessment Methods

C.1 Equivalent Adult Estimates for Pacific Herring and Northern Anchovy

C.2 *Neomysis mercedis* Population Dynamics and Entrainment Mortality

C.3 Development of an Index of Striped Bass Year Class Strength and Population Dynamics Used in Estimating Percentage Reduction

C.4 Estimation of Historical Entrainment Mortality Proportions and Cooling Water Flows

C.5 References

APPENDIX C: IMPACT ASSESSMENT METHODS

This appendix presents additional discussion of specific analytical methods utilized in the assessment of entrainment and impingement impact summarized in Chapter 5. Section C.1 describes the methodology used in computing equivalent adult estimates for Pacific herring and northern anchovy. Section C.2 describes the approach and analyses used in examining the population dynamics and entrainment cropping of neomysis. Section C.3 describes the approach used in developing an index of striped bass abundance (year class strength) and in examining the population dynamics of young-of-the-year striped bass that have been used in estimating the percentage reduction resulting from entrainment mortality. Section C.4 derives the equations for calculating the historical entrainment mortality proportions of neomysis and striped bass.

C.1 EQUIVALENT ADULT ESTIMATES FOR PACIFIC HERRING AND NORTHERN ANCHOVY

The objective of the equivalent adult approach to power plant impact assessment is to translate the number of young fish cropped by a plant into an equivalent number of adult fish, using the appropriate survival values. The estimate of equivalent adults represented by fish that do not survive entrainment and impingement has two useful applications:

1. the estimated equivalent adults can be used to put the cropping of the power plant into perspective with estimates of adult standing stock, if these are available; and
2. for populations subject to a fishery, the estimated adult equivalents can be compared to cropping associated with the fishery.

The equivalent adult method has been applied to Pacific herring and northern anchovy because data are sufficient to do so, there is a fishery for each of these species in the San Francisco Bay region, and cropping estimates are high enough to warrant further examination of impact.

This section presents the methodology employed to compute equivalent adult losses

for Pacific herring and northern anchovy at the Pittsburg Power Plant. A model developed by Goodyear (1978) for computing equivalent adult loss associated with larval entrainment, and also an earlier, simpler model (Horst 1975) from which the former model was derived are briefly described and provide the foundation for the estimation of equivalent adult losses. The derivation of each term needed for making the calculations is explained, using information obtained during the monitoring study or from the literature for both Pacific herring and northern anchovy. Techniques are also presented for computing equivalent adult losses associated with the entrainment of eggs and the impingement of juveniles.

In the model of Horst (1975), the number of equivalent adults lost, N_a, is given by

$$N_a = SN \tag{C-1}$$

where

S = probability of survival from larval stage to adulthood
N = number of larvae that do not survive.

By treating all larvae as a single class, Horst gives each entrained organism equal weight in computing equivalent adults. This can impart considerable bias into the predictions if survival rates vary among members of the entrained age groups. In fact, young larvae, with low natural survival rates, contribute fewer equivalent adults than an equal number of entrained older larvae, which have higher survival rates. To avoid this bias, Goodyear (1978) divides larvae into separate length classes, computes equivalent adults for each length class, and sums to get a more accurate estimate of equivalent adult loss associated with larval entrainment:

$$N_a = \sum_{k=1}^{n} F_k N_k S_k \tag{C-2}$$

where

k = an ordinal number assigned to each length class
n = number of length classes considered
F_k = entrainment mortality for length class k

N_k = number of organisms entrained of length class k
S_k = survival from length class k to adulthood.

The Goodyear model was applied to Pacific herring and northern anchovy length classes at 1-mm increments beginning with the length at hatch for each species. The application of these figures is presented in Tables C-1 and C-2. The entrainment mortality estimate, F_k, was set at 1.00 for all calculations. The other major terms of the equation, N_k and S_k, are derived in Sections C.1.1 and C.1.2.

C.1.1 Estimation of the Number of Organisms Entrained of Length Class k (N_k)

The numbers of Pacific herring and northern anchovy entrained as a function of length were estimated using methods described in Chapter 3. The estimated numbers entrained are reported in Tables C-3 and C-4.

C.1.2 Estimation of Survival from Length Class k to Adulthood (S_k)

Since in most cases the data needed to compute S_k directly are unavailable, it must be estimated indirectly:

$$S_k = \frac{S_T}{S_{e,k}} \qquad \text{(C-3)}$$

where

S_T = survival from egg to adulthood
$S_{e,k}$ = survival from egg to length class k.

The estimation of S_T used stems directly from the population theory for determining replacement level reproductive success (Leslie 1945; Vaughan and Saila 1976). For a population to remain at equilibrium with a sex ratio of 1:1, each female must produce enough eggs through her lifetime that two individuals will survive to maturity to replace her and one adult male. The survival from egg to adulthood needed to achieve this equilibrium is twice the reciprocal of average lifetime fecundity,

$$S_T = \frac{2}{F_a} \qquad \text{(C-4)}$$

TABLE C-1 FACTORS USED TO COMPUTE EQUIVALENT ADULT (TWO-YEAR-OLD) PACIFIC HERRING CORRESPONDING TO CROPPING OF VARIOUS LARVAL LENGTH CLASSES AT THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Length Class (mm) (k)	Numbers Entrained (thousands) (N_k)	Survival From Egg to Length Class L ($S_{e,k}$)	Survival From Length Class L to Adulthood (S_k)	Equivalent Adults[a] ($N_k F_k S_k$)
6	1,842	0.1000000	0.0003275	603
7	1,574	0.0325256	0.0010069	1,585
8	4,745	0.0122941	0.0026638	12,640
9	7,519	0.0052119	0.0062835	47,246
10	5,535	0.0024189	0.0135392	74,939
11	505	0.0012079	0.0271131	13,692
12	117	0.0006408	0.0511101	5,980
Total	21,837			156,685

(a) F_k, entrainment mortality, was set at 1.00 for all length classes.

TABLE C-2 FACTORS USED TO COMPUTE EQUIVALENT ADULT (ONE-YEAR-OLD) NORTHERN ANCHOVY CORRESPONDING TO CROPPING OF VARIOUS LARVAL LENGTH CLASSES AT THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Length Class (mm) (k)	Numbers Entrained (thousands) (N_k)	Survival From Egg to Length Class L ($S_{e,k}$)	Survival From Length Class L to Adulthood (S_k)	Equivalent Adults[a] ($N_k F_k S_k$)
5	1,324	0.2866200	0.0006851	907
6	832	0.2483792	0.0007906	658
7	602	0.2152407	0.0009123	549
8	451	0.1865235	0.0010528	475
9	257	0.1616377	0.0012149	312
10	492	0.1400721	0.0014019	690
11	144	0.1213838	0.0016177	233
12	160	0.1051888	0.0018668	299
13	15	0.0911546	0.0021542	32
14	62	0.0789928	0.0024859	154
15	85	0.0684537	0.0028686	244
16	341	0.0593206	0.0033103	1,129
17	62	0.0514061	0.0038199	237
18	46	0.0445476	0.0044080	203
19	29	0.0386040	0.0050867	148
20	32	0.0334535	0.0058699	188
21	0	0.0289902	0.0067736	0
22	0	0.0251223	0.0078164	0
23	0	0.0217705	0.0090199	0
24	0	0.0188659	0.0104086	0
25	120	0.0163488	0.0120111	1,441
26	0	0.0141676	0.0138603	0
27	0	0.0122774	0.0159942	0
28	0	0.0106393	0.0184567	0
29	0	0.0092198	0.0212983	0
30	0	0.0079897	0.0245774	0
31	0	0.0069237	0.0283614	0
32	0	0.0060000	0.0327279	0
33	0	0.0051995	0.0377668	0
34	0	0.0045058	0.0435811	0
35	0	0.0039046	0.0502912	0
36	0	0.0033837	0.0580340	0
37	11	0.0029322	0.0669690	737
Total	5,065			8,636

(a) F_k, entrainment mortality, was set at 1.00 for all length classes.

TABLE C-3 ESTIMATED NUMBERS, N_k, (thousands) OF PACIFIC HERRING ENTRAINED AT THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Length Class (k, mm)	Unit-Group 1-4	5 and 6	7	Total
5	324	279	13	616
6	655	549	22	1,226
7	803	749	21	1,574
8	2,199	2,482	64	4,745
9	3,496	3,950	73	7,519
10	2,583	2,901	52	5,536
11	230	269	6	505
12	55	59	3	117
Total				21,837

TABLE C-4 ESTIMATED NUMBERS, N_k (thousands) OF NORTHERN ANCHOVY ENTRAINED AT THE PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Length Class (k, mm)	Unit-Group 1-4	5 and 6	7	Total
5	681.8	621.9	30.5	1,334.2
6	441.3	372.2	18.4	831.8
7	328.9	260.5	12.9	602.3
8	234.5	206.6	10.2	451.3
9	136.6	114.6	5.7	256.9
10	276.6	204.7	10.9	492.2
11	80.5	60.6	3.2	144.3
12	88.5	67.9	3.5	159.9
13	7.9	6.4	0.3	14.6
14	34.9	25.7	1.3	61.9
15	45.5	37.4	1.9	84.8
16	204.2	129.1	7.8	341.2
17	33.3	27.1	1.3	61.7
18	27.7	17.1	1.1	45.9
19	15.8	12.8	0.6	29.2
20	20.0	11.3	0.8	32.1
.				
25	75.0	42.5	2.9	120.4
.				
.				
37	5.8	4.7	0.2	10.7
Total				5,065.3

where

F_a = average lifetime fecundity of an adult female.

Substituting Equation C-4 into Equation C-3 yields

$$S_k = \frac{2}{S_{e,k}F_a} . \qquad \text{(C-5)}$$

C.1.2.1 Estimation of Lifetime Fecundity (F_a)

Average lifetime fecundity is a function of age of recruitment, frequency of spawning, fecundity of age class, and survival within each age class. Defining the beginning of the adult stage as the initial age of spawning, average lifetime fecundity was computed as

$$F_a = \sum_{j=r}^{m} P_j E_j S_j \qquad \text{(C-6)}$$

where

j = an ordinal number assigned to each age (year) class
r = age of recruitment into the adult population
m = number of age classes in the population
P_j = fraction of females that are mature in age class j
E_j = average fecundity of mature females in age class j
S_j = survival from recruitment to age class j.

Pacific Herring

The estimate of F_a for Pacific herring, 61,070, is listed in Table C-5 along with values for the terms of Equation C-6. The age of recruitment into the adult population, r, and the number of age classes, m, were set at two and eight respectively (Spratt 1976). Since all females of age class two and greater were considered to be mature (Rabin 1976), the value of P_j was set at 1.00 for all age classes.

TABLE C-5 FACTORS USED TO ESTIMATE AVERAGE LIFETIME FECUNDITY OF PACIFIC HERRING

Age Class j	Standard Length SL_j (mm)	Eggs per Female E_j	Proportion of Females Mature P_j	Survival from Age 2 to j S_j	Relative Contribution $E_jP_jS_j$
2	159	15,040	1.00	1.0000	15,040
3	176	23,480	1.00	0.6225	14,616
4	186	28,450	1.00	0.3875	11,024
5	197	33,910	1.00	0.2412	8,180
6	204	37,380	1.00	0.1502	5,615
7	214	42,350	1.00	0.0935	3,960
8	220	45,330	1.00	0.0582	2,635
Average lifetime fecundity, F_a					61,070

The average fecundity of females in each age class, E_j, was computed from the linear relationship between standard length and fecundity taken from Rabin and Barnhart (1977):

$$E_j = -63{,}920.9 + 496.6(SL_j) \qquad \text{(C-7)}$$

where

SL_j = average standard length (mm) of age class j.

The values for average standard length were taken from Spratt (1976).

Survival from recruitment to age class j, S_j, was computed from the equation

$$S_j = e^{-Z(j - r)} \qquad \text{(C-8)}$$

where

r = age of recruitment into the adult population
Z = yearly instantaneous mortality rate.

Z was calculated from the age composition of fish in the commercial catch, following MacCall (1974):

$$Z = -\ln \frac{T}{n + T - 1} - \frac{(n - 1)(n - 2)}{n(T + 1)(n + T - 1)} \qquad \text{(C-9)}$$

where

$$n = \sum_{k=0}^{m-r} N_k$$

$$T = \sum_{k=0}^{m-r} kN_k$$

and

k = coded age: $k = 0$ for the youngest age group fully vulnerable to the sampling, $k = 1, 2, \ldots, m-r$ for successively older age groups, assuming that all older ages are completely vulnerable

N_k = number of fish of coded age k found in the catch.

The age composition data used were those from the 1976-1977 San Francisco Bay round haul catch, considered by Spratt (CDF&G, 1979, personal communication) to be the only data set representative of an entire spawning season. Data from other years were rejected either because sampling occurred late in the spawning season or because the fishery was closed after only one or two spawning runs in these years. The round haul catch was selected over the only other form of catch, gill netting, because the latter tends to be selective for larger fish (Spratt 1976), which would inject a negative bias into the estimate of Z. Instantaneous annual adult mortality, Z, for Pacific herring was computed to be 0.474; the resulting values of S_j are found in Table C-5.

Northern Anchovy

The estimate of average lifetime fecundity, F_a, for northern anchovy is listed in Table C-6, along with values for the terms of Equation C-6. The age of recruitment into the adult population, r, was set at one, because females have been shown to reach sexual maturity at that age (Clark and Phillips 1952; Brewer 1978).

The number of age classes, m, was set at six (Baxter 1967). Values for P_j, the fraction of females that are mature for each class, were taken from Clark and Phillips (1952).

Northern anchovy are capable of spawning more than once a year (Baxter 1967), but the number of spawning attempts is not known. The number is here taken to be two, based on Frey (1971) and on evidence from seasonal abundances of entrained eggs collected at the Potrero Power Plant (PGandE 1980). This estimate was used in computing E_j, the average fecundity of northern anchovy in age class j:

$$E_j = 2W_jF \qquad \text{(C-10)}$$

where

TABLE C-6 FACTORS USED TO ESTIMATE AVERAGE LIFETIME FECUNDITY OF NORTHERN ANCHOVY

Age Class j	Standard Length SL_j (mm)	Weight W_j (g)	Eggs per Female E_j	Proportion of Females Mature P_j	Survival from Age 2 to j S_j	Relative Contribution $E_jP_jS_j$
1	121	18.5	21,225	0.167	1.0000	3,545
2	133	24.0	27,510	0.292	0.3362	2,701
3	140	27.6	31,665	0.567	0.1130	2,030
4	146	31.0	35,530	0.918	0.0380	1,240
5	152	34.6	39,680	0.975	0.0128	494
6	154	35.8	41,125	0.990	0.0043	175
Average Lifetime Fecundity, F_a						10,185

W_j = weight (g) per female in age class j

F = a constant, 574 (eggs/g) (MacGregor 1968)

2 = number of spawns per female per year.

Weight per female was computed from the following power function (Collins 1969):

$$W_j = 0.00003582\,(SL_j)^{2.74285}. \qquad \text{(C-11)}$$

Values for SL_j were taken from Collins (1969).

Values for S_j were computed from Equation C-8, with the yearly instantaneous mortality, Z, set at 1.09 (MacCall 1974). Average lifetime fecundity (F_a) for northern anchovy was computed to be 10,185 (Table C-6).

C.1.2.2 Estimation of Survival from Egg to Larval Length Class k ($S_{e,k}$)

Survival from egg to larval length class k, $S_{e,k}$, was estimated with the general formula

$$S_{e,k} = H \frac{N_k}{N_0} \qquad \text{(C-12)}$$

where

H = the fraction of eggs that hatch

N_k = the number of larvae of length class k

N_0 = the number of larvae that have hatched.

Pacific Herring

Egg viability, H, is highly variable in Pacific herring. Under certain laboratory situations, hatching success reaches nearly 100 percent (Alderdice and Velsen 1971; Taylor 1971), but Taylor has found that fewer than 20 percent of all eggs that are in masses more than ten layers thick will hatch. Predation by birds, fish, and shellfish can cause over 90 percent loss of herring eggs (Cleaver and Franett 1946; Outram 1958), and Spratt (1976) made spawning biomass estimates for San Francisco Bay that assumed 95 percent predation in one week. When spawning is heavy, predation is

moderate (Spratt 1976), but hatching success is low (Taylor 1971). Pacific herring spawning was heavy in San Francisco Bay during the winter of 1978-1979 (D. H. Thomas, CDF&G, 1979, personal communication). Consequently, the fraction of eggs that hatched, H, was set at 0.1.

A model devised by Hackney and Webb (1978) for computing the number of hatchlings that survive to length k has been used to compute the ratio N_k/N_0:

$$\frac{N_k}{N_0} = L_0{}^b L_k{}^{-b} \qquad \text{(C-13)}$$

where

L_k = total length (mm) of fish in length class k
L_0 = total length of hatchling, set at 6.0 mm
b = a rate coefficient describing a decrease in numbers through length classes.

This equation may be manipulated and log-transformed to yield

$$\ln N_k = \ln N_0 + b(\ln L_0) - b(\ln L_k). \qquad \text{(C-14)}$$

Combining constants produces

$$\ln N_k = a - b(\ln L_k), \qquad \text{(C-15)}$$

where

$$a = \ln N_0 + b(\ln L_0).$$

The rate coefficient, b, then becomes the negative slope of a straight line relationship between natural log abundance and natural log length. Figure C-1 depicts this relationship for Pacific herring, based on entrainment length frequency data collected during the winter of 1978-1979 at the Potrero Power Plant. A least-squares curve-fitting technique applied to these data (omitting the five longest length classes) yielded estimates of a = 31.638 and b = 7.246 (r^2 = 0.959; $p < 0.0001$).

If Equation C-13 is substituted in Equation C-12, it yields

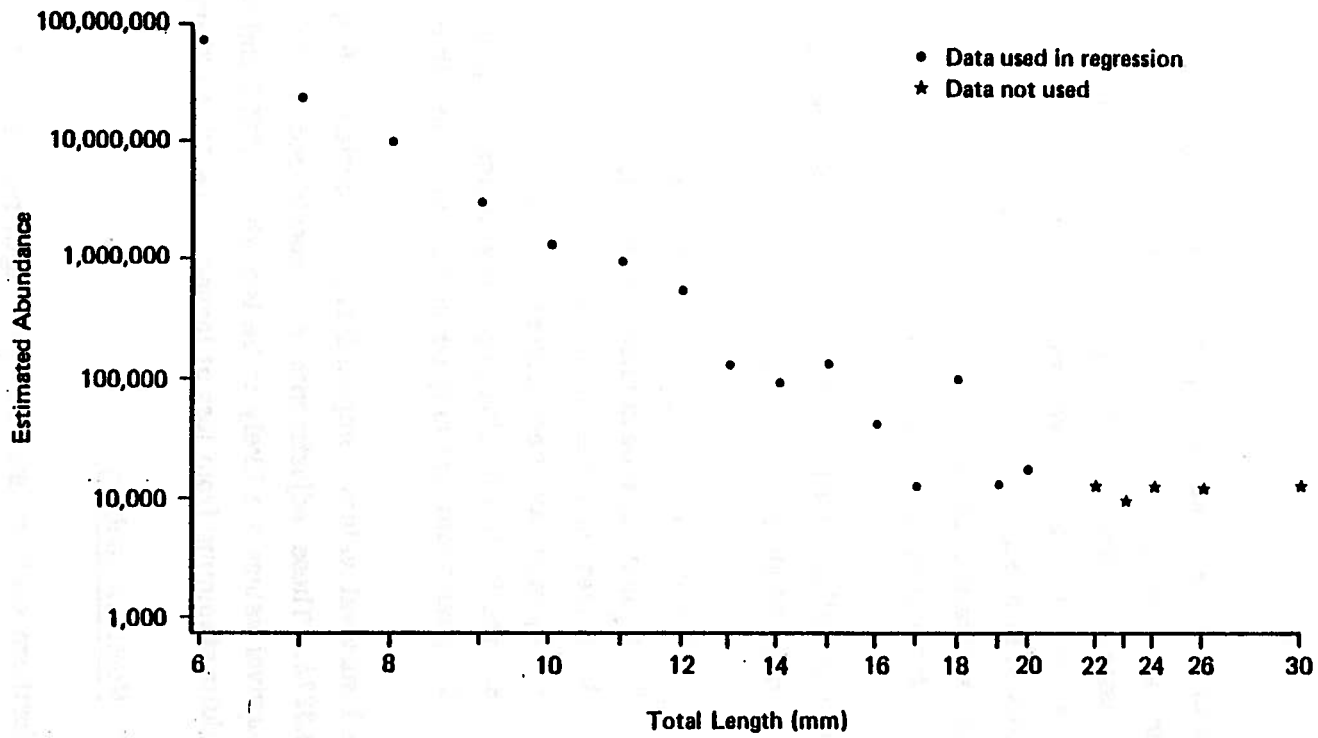


Figure C-1. Length frequency of Pacific herring entrained at the Potrero Power Plant from October 1978 to March 1979 used to compute larval mortality rate.

$$S_{e,k} = HL_0{}^{b} L_k{}^{-b}. \qquad \text{(C-16)}$$

From Equation C-16, estimates of $S_{e,k}$ were computed for each length class from 6 to 15 mm. These values were then applied to Equation C-5, along with F_a, equal to 61,070 (Table C-5), to produce an estimate of S_k for each length class in the range 6-15 mm.

Using the same equations, estimates of S_k for the 16- to 30-mm length classes proved to be unrealistically high, reaching values greater than 1.00 for the larger length classes. Unrealistically high survival estimates were defined as values which were greater than 0.3875, two-year adult survival. An upper limit of two-year adult survival was used because survival in early life stages is generally lower than adult survival for an equivalent period, and the time between the stage of larval entrainment and adulthood is approximately two years.

The unrealistically high estimates may have resulted from a number of causes. Some of the more likely are the following:

1. b may not be constant for all length classes,
2. F_a may have been underestimated,
3. H may have been underestimated,
4. L_0 may have been underestimated,
5. the Pacific herring population is not in equilibrium, and
6. gear avoidance may occur for longer length classes.

All survival values computed from Equation C-5 greater than 0.3875 were set at 0.3875. These adjustments are considered to be conservative, in that the true survival values are likely to be less than 0.3875 and would result in fewer equivalent adults stemming from loss of larvae in the 16- to 30-mm length classes.

Northern anchovy

Northern anchovy entrainment length frequency data collected at the Potrero Power Plant from March 1978 to March 1979 (PGandE 1980) were used to compute values for N_k/N_0 for use in Equation C-12.

The model used, from Goodyear (1978), was

$$\frac{N_k}{N_0} = e^{-d(L_k - L_0)} \qquad \text{(C-17)}$$

where

d = mortality rate coefficient

L_k = total length in millimeters

L_0 = total length at hatching, set at 2.5 mm (Baxter 1967).

No single cohort could be distinguished from the data, because of the continual spawning exhibited by northern anchovy (Baxter 1967). All length classes were used in computing d, the mortality rate coefficient, except for those less than 10 mm and greater than 50 mm, which were excluded because too few fish of these sizes were found in the samples.

This model, which assumes mortality to be a constant function of length, gave the best least-squares fit to the data, indicated by linear relationship between length and natural log of abundance (Figure C-2). The mortality rate coefficient, d, was computed at 0.1432 ($r^2 = 0.831$, $P < 0.0001$).

Although data from the 3-10 mm length classes were not used to estimate the larval mortality rate coefficient, the value of d was applied to these length classes in Equation C-17. This decision is supported by data presented by Smith (1972), which indicate that mortality rates of larvae less than 10 mm long are equal to those of larvae 10-16 mm long.

Substituting Equation C-17 into Equation C-12 yields

$$S_{e,k} = He^{-d(L_k - L_0)}. \qquad \text{(C-18)}$$

The fraction of eggs that hatch was computed from data provided by Lasker and Smith (1977) as the mean number of 2.5-mm larvae divided by the mean number of eggs captured in California Cooperative Oceanic Fisheries Investigations surveys from 1955 through 1960 (Lasker and Smith 1977). In this manner H was computed to be 0.41.

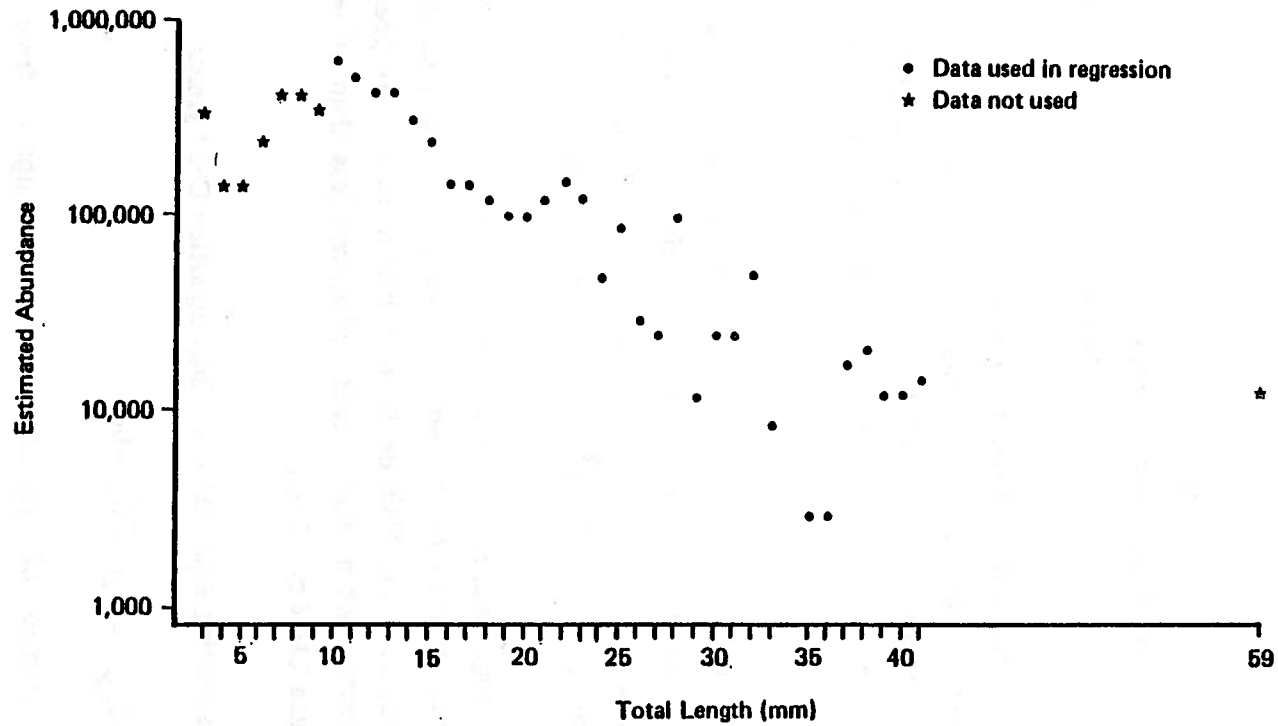


Figure C-2. Length frequency of northern anchovy entrained at the Potrero Power Plant from March 1978 to March 1979 used to compute larval mortality rate.

With H set at 0.41 and d equal to 0.1432, survival between egg and length class k, $S_{e,k}$, was found, and used in Equation C-5 with F_a equal to 10,185 (see Table C-6) to produce an estimate of S_k for each length class. The only length class for which Equation C-5 produced an unrealistically high value for S_k was the 59-mm class. Applying the same logic used in the case of Pacific herring, S_k for this length class was adjusted to 0.3362, the one-year survival for adult northern anchovy. These values, and the corresponding equivalent adult loss estimates, are presented in Table C-2.

C.1.3 Estimate of Total Equivalent Adults

The equivalent adults methodology discussed so far has focused on the cropping of the larvae of Pacific herring and northern anchovy. An estimate of total equivalent adults must also include those associated with the cropping of eggs and juveniles. Total equivalent adults, N_T, was computed with the following formula:

$$N_T = N_a + \frac{2N_e}{F_a} + N_I S_{j,a} \qquad \text{(C-19)}$$

where

N_a = equivalent adults associated with larval (yolk-sac larvae and larvae) entrainment

N_e = total number of entrained eggs

F_a = average lifetime fecundity

N_I = total number of impinged juveniles

$S_{j,a}$ = survival from juvenile to adult stage.

To facilitate computations, all impinged juvenile anchovy were regarded as one-year-old fish. This caused each impinged juvenile to translate as an adult, since individuals of this species enter the adult population at age one. Since mortality rates for juvenile Pacific herring are not available, it was assumed that survival in the juvenile stage was equal to that of the adult stage, 0.6225, and survival from the juvenile to adult stage, $S_{j,a}$, was set at this figure. Both of these decisions are conservative: the effect of treating impinged juveniles as one-year-old fish and assigning adult survival rates to them is to overestimate the equivalent adult loss

associated with impinged juveniles. The total equivalent adult estimates are discussed in Section 5.4.2.

C.2. NEOMYSIS MERCEDIS POPULATION DYNAMICS AND ENTRAINMENT MORTALITY

This section describes the methods used to estimate population dynamics and entrainment loss parameters for Neomysis mercedis (referred to as neomysis) in the study area.

The Neomysis mercedis survey of CDF&G, the source of data for this analysis, will be described briefly, followed by derivation of the equations used in computing migration, birth rates, and entrainment losses.

C.2.1 Neomysis Survey Methods

Data from the 185 CDF&G neomysis surveys over the years 1968-1979 were employed. In that time CDF&G took samples at stations from San Pablo Bay to near Sacramento and Stockton in the Delta. The sampling stations are indicated in Figure 5-9. The part of that area reaching from the east end of Carquinez Strait to Cache Slough, to the mouth of the Mokulumne River, to Stockton, and to Clifton Court Forebay has been chosen as the study area, or source water body. This is because most of the Bay-Delta neomysis population may be found there, it has been consistently sampled, and it includes the Pittsburg Power Plant. The study area has been divided into six strata (Figure 5-9), to permit analyses of abundance within it. These strata and their volumes (estimated with data from Water Resources Engineers 1975) are listed in Table C-7. (The numbering of the strata is a consequence of stratum designations used in striped bass and ichthyoplankton surveys. For clarity, in equations we will treat them as being numbered 1-6.) The estimated water volume of the study area is 44.3 billion cubic feet, just over a million acre-feet (1.2 billion m^3).

For purposes of comparison of abundances among strata, in some months the abundance of densities of neomysis in Strata 8, 9, and 10, the Delta west of the Sacramento River, the upper Sacramento River, and the east Delta, are shown. These strata and their volumes are also shown in Figure 5-9 and in Table C-7.

TABLE C-7 VOLUMES OF STRATA IN THE BAY-DELTA USED IN ESTIMATING ABUNDANCES OF NEOMYSIS MERCEDIS

Stratum Number	Stratum Name	Volume ($ft^3 \times 10^9$)	Volume ($m^3 \times 10^6$)
3	Suisun Bay channel[a]	7.88	223
4	Suisun Bay shallows	12.05	341
5	Montezuma Slough	1.55	44
6	Lower Sacramento River	6.01	170
7	Lower San Joaquin River	7.75	219
8	Cache Slough, Steamboat Slough, Ship Channel	3.61	102
9	Upper Sacramento River	1.97	56
10	North Delta	2.18	62
11	South Delta	9.09	257
Total without 8, 9 and 10		44.33	1,254
Total with 8, 9 and 10		52.09	1,474

(a) East of the Benicia Bridge; does not include Carquinez Strait.

Note: Strata 1 and 2, San Pablo Bay and the Napa River, were not considered in this analysis.

Surveys were conducted at intervals ranging from two weeks to four months. A survey, consisting of sampling once at each of several stations, usually took 4-8 days to complete. Surveys were monthly from June 1968 through February 1972. From then through 1976, the pattern of surveys was one a month in March and November, two a month from April through October, and none in December, January, and February (except for a February survey in 1975). In 1977, single December, January, and February surveys were added, and in 1978 a second March survey was added.

Samples were taken in midchannel and near high tide, by means of 10-minute oblique tows from the bottom to the surface, at varying numbers of the stations. Tows were made against the current with nets that had, variously, mouth openings of 0.1-0.7 m^2 and lengths of 1-1.5 m. Mesh size was 0.9-1.0 mm through 1973; after that it was changed to 505 µm to sample mysids 2-3 mm long more efficiently (Miller 1977). The surface water temperature, electrical conductivity, and chlorophyll a (starting in 1976) were also measured.

Samples with more than 400 neomysis were subsampled: integral sixteenths were counted until at least 200 organisms were counted. The first 100 counted were measured for length. Rounding was such that, for example, those 1.0-1.9mm long (later 1.1-2.0 mm) were categorized as being in the 2-mm length interval. Starting in 1976, measures of life stage, sex, percentage of adult females gravid, and fecundity (described in Section C.2.2.2) were made. Densities were calculated for each sample, using readings from a flowmeter mounted in the mouth of each net.

The sample densities (d_{ij}) were averaged to give a mean stratum density $\bar{d}_i$:

$$\bar{d}_i = \frac{1}{J_i} \sum_{j=1}^{J_i} d_{ij} \qquad \text{(C-20)}$$

where

i = the ordinal of a stratum
j = the ordinal of a station within a stratum
J_i = the number of stations sampled in stratum i.

These stratum densities were volume-weighted to estimate abundance within a stratum, and summed across the six strata to give an estimate, N, of the abundance in the study area for a given survey:

$$N = \sum_{i=1}^{6} V_i \bar{d}_i \qquad \text{(C-21)}$$

where

V_i = the volume of a stratum.

The variance of N was estimated as

$$Var(N) = \sum_{i=1}^{6} V_i^{\,2} \frac{1}{(J_1 - 1)} \sum_{j=1}^{J_i} (d_{ij} - \bar{d}_i)^2 . \qquad \text{(C-22)}$$

The 95 percent confidence interval of the estimate N is N $\pm$ 1.96 Var (N). N was calculated both for the population as a whole and for each length class.

C.2.2 Population Dynamics

Examination of the population dynamics of neomysis involves consideration of migration into and out of the study area, birth rates, and death rates. A brief description of the approach and methodology used to examine each of these aspects of neomysis population dynamics follows.

C.2.2.1 Migration

The migration rate for each survey will be calculated first. Because of the uncertainty of migration estimates, because migration only occasionally affected the study-area population, and because daily death rates were almost always above 1 percent, only migration estimates greater than 1 percent will be categorized as significant. Surveys in the significant-migration category were excluded from the calculation of death rates.

Preliminary analyses led to the conclusion that except on one occasion, the only significant migration occurred under the Benicia Bridge. Abundance is usually very low near and upstream of the borders of the study area in strata 8, 9, and 10 (see Figures 5-12 and 5-13). Periods of very low flow result in more individuals near the upstream borders. At such times, migration may occur there, but changes in freshwater inflow would be small and slow, so migration would be low. Of course, when low flow periods were terminated by large freshwater inflows, immigration to the study area might be important, as a large proportion of the population would be involved. Overall, such periods were rare, so the migration at upstream study area borders are assumed to be insignificant except in December 1977.

The criterion for significance of 1 percent was established with hypothetical conditions. The parameter used to estimate changes in net flow, and thus migration, was electrical conductivity (millimhos per centimeter; mmho) of the water at CDF&G Station 42, about 2.5 km (1.5 mi) upstream of Benicia Bridge. Following the salt-front model developed by EA (Appendix A), it was assumed that a change in conductivity (ΔEC) of 1 mmho at Station 42 represented a movement of the entrapment zone of 1.6 km. The Suisun Bay channel (Stratum 3) upstream of the bridge was divided into five reaches, each which was assumed to have a homogeneous cross-section and thus a consistent volume per channel kilometer. The combined volume of a reach of the channel and its adjacent shoals was taken to be three times that of the channel alone. The reaches and their assigned volumes per kilometer (hm^3/km) are as follows:

Reach (km)	0-4.8	4.8-9.6	9.6-14.8	14.8-20.9	20.9-24.5
Volume/km	16.7	20.7	21.2	27.5	25.2

Using the assumptions of a nominal inter-survey interval of 15 days and of even densities between the channel and the shallows, Strata 3 and 4, when the entrapment zone was near the Benicia Bridge boundary, the following equation was used to estimate the daily percentage of the study area population that migrated (M):

$$M(\%) = \frac{100 \times P \times 3 \sum_{i=1}^{4} (V_i \times 1.6\Delta EC)}{564 \times 15} \text{ or,} \qquad \text{(C-23)}$$

$$M(\%) = 0.0364 \times P \sum_{i=1}^{4} (V_i \times 1.6\Delta EC)$$

where

P	=	proportion of the neomysis population in Strata 3 and 4
3	=	ratio of total volume to main-channel volume
V_i	=	volume (hm^3) per kilometer of cumulative stretch i of the main channel
1.6	=	distance (km) the salt front moved for each 1-mmho change in EC
ΔEC	=	change in electrical conductivity (mmho) at Station 42, or in its absence, Station 48
564	=	total volume (hm^3) of Strata 3 and 4
15	=	survey interval (days).

A set of hypothetical M's (%) were calculated for ΔEC's of 1, 3, and 10 mmho and P's of 0.1, 0.5, and 0.9:

EC	P = 0.1	P = 0.5	P = 0.9
1	0.1	0.5	0.9
3	0.3	1.4	2.6
10	1.2	5.8	10.4

On the basis of a minimum significant migration rate of 1 percent, a $\Delta EC > 1$ mmho was required for significant migration when ≤ 90 percent of the study area population was in Strata 3 and 4, the most frequent case in surveys over 1974-1979.

Two alternative sets of criteria were posited for significant migration, since neomysis are not evenly distributed within Strata 3 and 4 as in the hypothetical case above. Individuals are strongly concentrated in the region with EC's of 2-10 mmho, the entrapment zone, so the extent of migration also depends on the location of the zone relative to Benicia Bridge. The first criterion was applied when the entrapment zone was at Station 42 and the second when it was just up the estuary (≤ 8 km) from Station 42. First, EC at Station 42 is ≤ 10 mmho for either survey and ΔEC is >1 mmho. This not being the case, EC had to be between 10 and 15 mmho, inclusively, and ΔEC had to be >2 mmho. In the second case, ΔEC, and therefore movement of water, was greater, but density at the border of the study area was expected to be less than a few kilometers upstream, since an EC of, say, 12 mmho indicated that the

seaward end (10 mmho) of the entrapment zone was 3.2 km (12 - 10 x 1.6) up the estuary from the Benicia Bridge.

By these criteria, migration was significant on 12 occasions between 1976 and 1979 (Table 5-9). Significant emigrations occurred between the months of December and May, and significant immigrations occurred between April and June.

C.2.2.2 Birth

Mysids, the group of invertebrates to which neomysis belongs, are called opossum shrimp because the embryos develop in a ventral brood pouch on the female parent. The embryos lie closely packed, and ambient water is circulated over them by movement of brood pouch plates. Development time, from fertilization to liberation from the pouch as a juvenile, varies between species and also, within species, with temperature. Some mysid species reproduce year-round, and season influences both gravidity, the proportion of adult females carrying young, and fecundity, the number of young carried (Heubach 1969; Pezzack and Corey 1979). The equation used to calculate the daily birth rate, b, from Paloheimo (1974), is

$$b = \frac{\ln(C_i + 1 / N_i)}{D_i} \qquad \text{(C-24)}$$

where

i = the ordinal number of a survey
C_i = the number of eggs and embryos in the study area in survey i
N_i = the number of free-living individuals in the study area
D_i = the development time (days).

The Number of Eggs and Embryos in the Study Area

The number of eggs and embryos in the study area at the time of each survey, C_i, was calculated as

$$C_i = \sum_{j=7}^{17} G_{ij}F_{ij} \qquad \text{(C-25)}$$

where

j = the ordinal of a length class at a 1-mm increment
G_{ij} = the number of neomysis gravid of length j in the study area at survey i
F_{ij} = the mean fecundity of neomysis of length j.

Proportion Gravid

The calculation of G_{ij}, the length-specific number of neomysis gravid involved the CDF&G data for each sample. The data included the proportion of adults in the population, the proportion of adult females gravid, and overall and length-specific densities. Using the proportion of adults, the proportion of juveniles was calculated, then multiplied by the overall density for a sample to yield the density of juveniles in the sample. From this number were subtracted, length class by length class, starting with the smallest, the number of neomysis in each length class. Once the juveniles did not account for all the numbers in a length class, the rest of that length class and of all longer length classes were regarded as adults. The proportion of adult females in each sample was multiplied by each length-specific adult density, and this product by the stratum volume. These stratum abundances were summed to get study-area abundances of adult females in each length class.

Since only the overall proportion of females gravid was reported, a means of assigning this proportion to the various length intervals was devised. Heubach (1969) studied the proportion gravid in the study area from March 1965 to September 1966. He reported the proportion gravid for length classes at 2-mm increments (which were apportioned by regression into 1-mm-increment length classes for this analysis). Heubach found that the proportion gravid varied with length and that the maximum proportion shifted from longer to shorter lengths between winter and summer. Also, the smallest length of gravid females shifts similarly, being approximately 11 mm in winter and 7 mm in summer.

An equation predicting the proportion of adult females gravid was obtained on the assumption that the proportion gravid was a joint function of water temperature at the time of a survey, T; water temperature a month earlier, M; and the length of the individual, L. A logistic regression (Cox 1970), based on 13 variables: L, T, M, LT, LM, MT, LMT, L^2, L^3, T^2, T^3, M^2, and M^3, was used with the BMDP

nonlinear regression program P3R (Dixon and Brown 1977) and with Heubach's data. Twelve variables were included in the final equation. Their coefficients and standard errors are in Table C-8. Since the sample size is very large, the adequacy of the model's fit can best be determined by inspecting graphs and summary statistics (Cox 1970).

The curves resulting from application of this predictive equation to a set of relevant lengths and the mean T and M for each month of Heubach's study have been superimposed on Heubach's observed proportion-gravid values in Figure C-3. The correlation between observed and predicted proportions was 0.941 (r^2 = 0.885). The similarities between the seven pairs of corresponding months in the two years are further evidence that the model should perform well as a predictive tool.

The equation was used with the T and M and length-specific densities reported for the CDF&G surveys for 1976-1979 to apportion the reported overall proportions gravid among the specific length classes. These proportions were multiplied by the numbers of adult females in each length class to get G_{ij}, the estimated number of gravid females in each length class in the study area for a given survey. G_{ij} was then summed for all lengths, and the sum divided by the total number of gravid females of all lengths estimated from the survey data to yield a correction factor C. G_{ij} values for a survey were then multiplied by C to yield a final, corrected G_{ij} for the study area for a given survey, the first term of Equation C-25.

Fecundity

Neomysis fecundity increases with length and varies over the year (Heubach 1969). Mean fecundity estimates from the CDF&G neomysis fecundity data for the period 1976-1979 repeat the length relationship observed by Heubach (1969). Up to 20 gravid females with intact marsupia from the sample at each station were inspected by CDF&G personnel for fecundity—the number of eggs and embryos in the marsupium. The fecundities were reported by 1-mm length class. In this analysis, the length-specific product of the station mean fecundity and the number inspected was summed across stations, and this was divided by the total number inspected for the study area to give a mean study-area fecundity for each length class. Mean fecundity for 1976-1979 increased with length from 9 to 17, 13 to 39, 16 to 65, 26 to 59, and 43 to 75 eggs and embryos, respectively for 8-, 10-, 12-, 14-, and 16-mm females. Also,

TABLE C-8 COMPONENTS OF A MODEL USED TO PREDICT THE PROPORTION OF ADULT FEMALE NEOMYSIS MERCEDIS GRAVID

Parameter[a]	Coefficient Estimate	Standard Error
Intercept	-66.238	4.525
L	12.772	1.154
T	-3.823	0.458
M	3.019	0.314
LT	0.1068	0.0133
TM	0.1172	0.00832
LTM	-0.00826	0.000469
L^2	-0.88	0.114
L^3	0.0203	0.0037
T^2	0.1811	0.0257
T^3	-0.00422	0.000543
M^2	-0.19	0.0218
M^3	0.00317	0.000416

(a) L = length of organism; T = temperature at time of survey; M = temperature 1 month before survey.

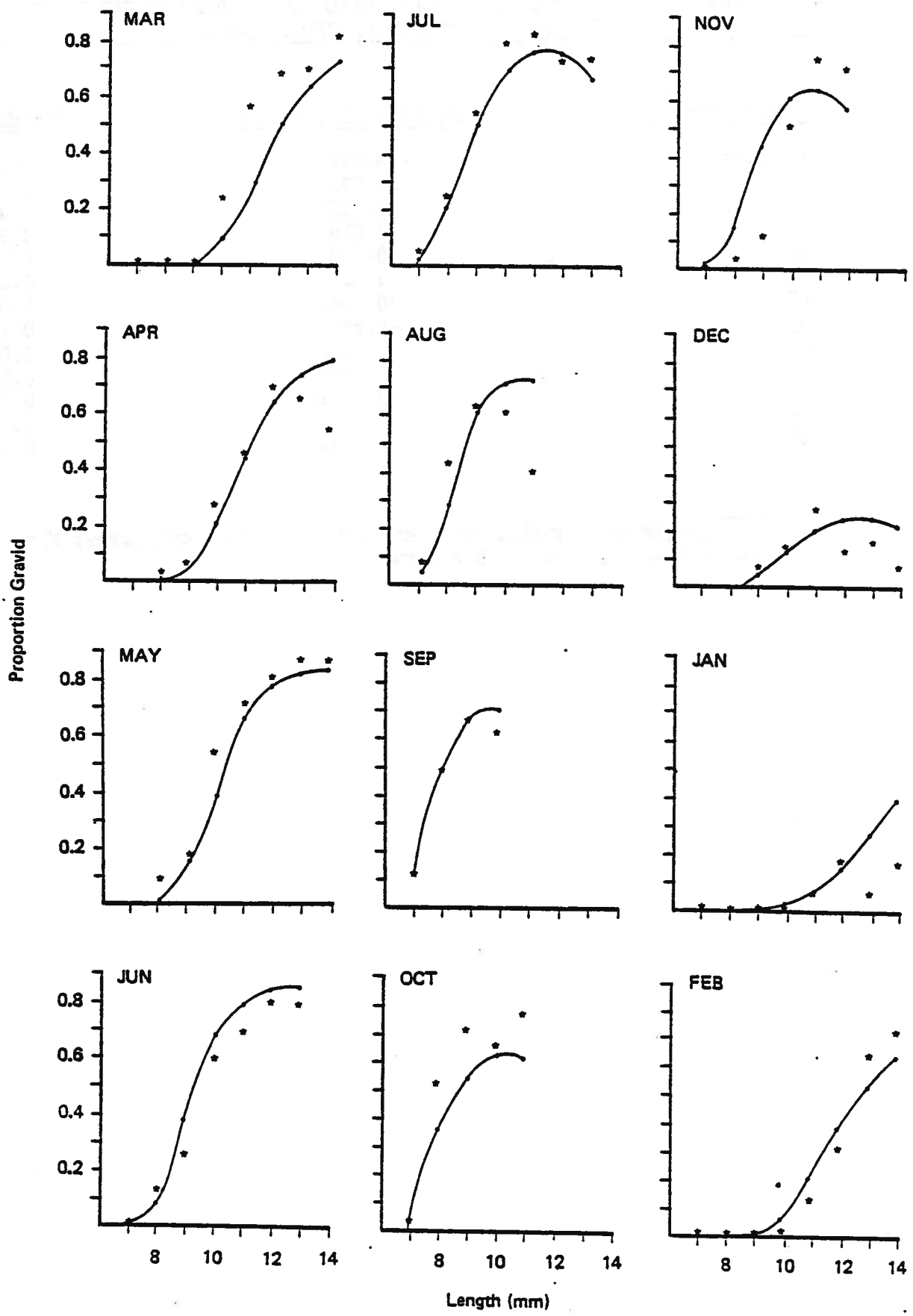
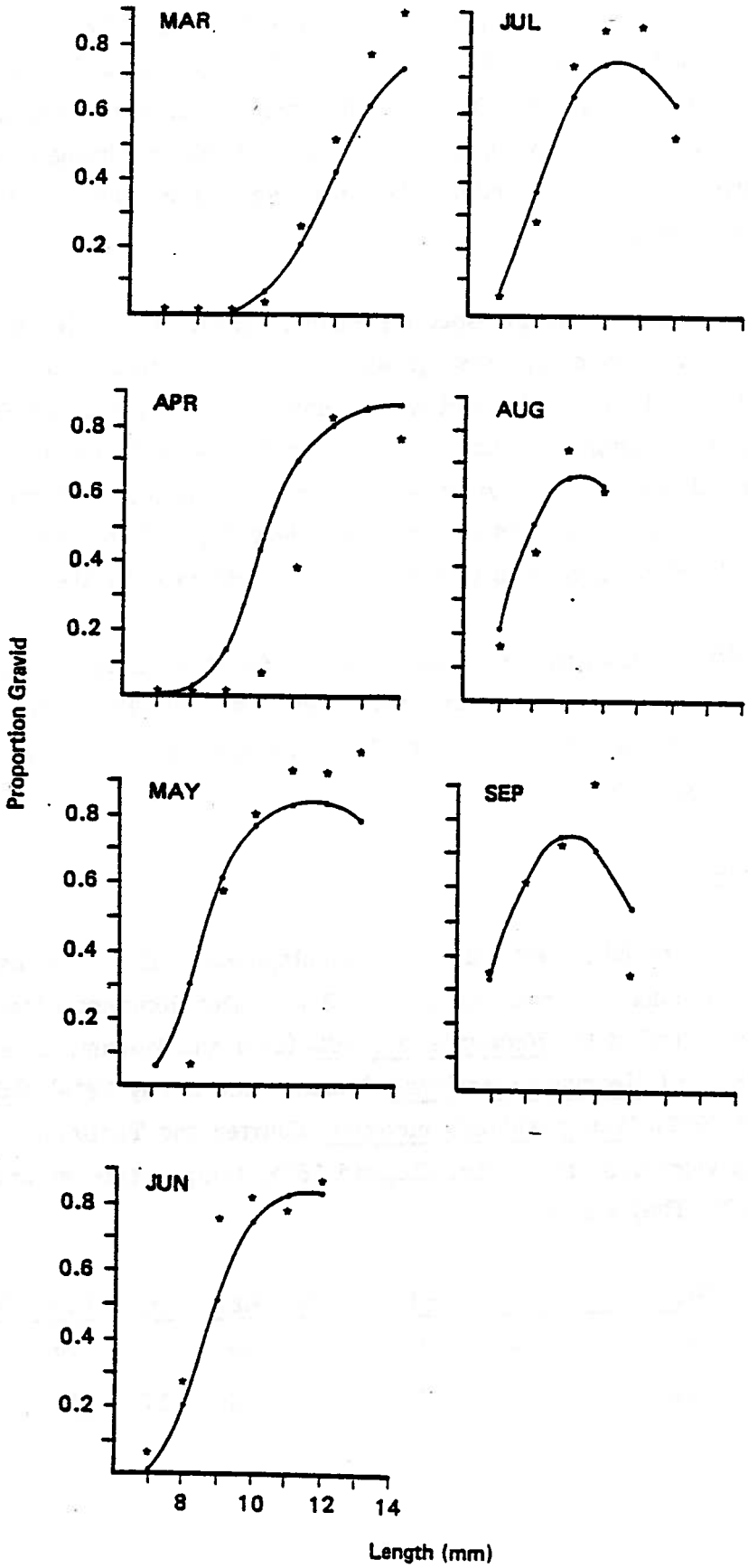


Figure C-3. Measured and predicted proportions of neomysis gravid for various water temperatures and body lengths, March 1965-September 1966. (Data from Heubach 1969.)


MAR
JUL
APR
AUG
MAY
SEP
JUN
0.8
0.6
0.4
0.2
Proportion Gravid
8
10
12
14
Length (mm)

certain lengths of females are fecund only at certain times of the year. Generally, longer lengths of females were fecund in cooler months, while shorter females are fecund in warmer months. Over 1976-1979, 8-mm individuals were fecund from May through December. Fecund 12-mm females were present in every month, whereas 16-mm gravid females were observed only from January through May. The warm-season restriction on fecund individuals more than 12 mm long is due to absence of individuals of these lengths.

There is a seasonal pattern of length-specific mean fecundities that is similar for all lengths (Figure C-4). Fecundity was greatest in the periods March-May and October-December, and lower in the periods December-February and May-September. Yearly differences for a length are slight, indicating that variation over lengths and seasons is largely independent of environmental variations between years. Fecundity differences between lengths, the absence of larger lengths, and the regular seasonal patterns of fecundity all contribute to variations in the birth rate for the population.

These length-specific fecundities, F_{ij}, were used in Equation C-25 to calculate C_i, the number eggs and embryos in the study area for each survey date i. With the term C_i determined thus, and N_i as in Equation C-21, the term yet to be calculated in Equation C-24 is D_i, development time.

Development Time

Since there are no published estimates of development times for neomysis, a regression equation was derived from the data available. Development times at known temperatures were compiled for Neomysis mercedis (Orsi and Mecum, CDF&G, 1979, personal communication), Neomysis americana (Pezzack and Corey 1979), Schistomysis spiritus (Mauchline 1967), Metamysidopsis elongata (Clutter and Theilacker 1971), and Praunus inermis (development time from Blegvad 1922; temperature estimated from Sverdrup et al. 1942). They were:

	N.a.	P.i.	S.s.	P.i.	N.m.	N.a.	P.i.	N.m.	M.e.
Temperature (C)	10	12.5	12.5	13	15	16	17	18	18
Development time (days)	24	29	21	25	22	13	17	15	10

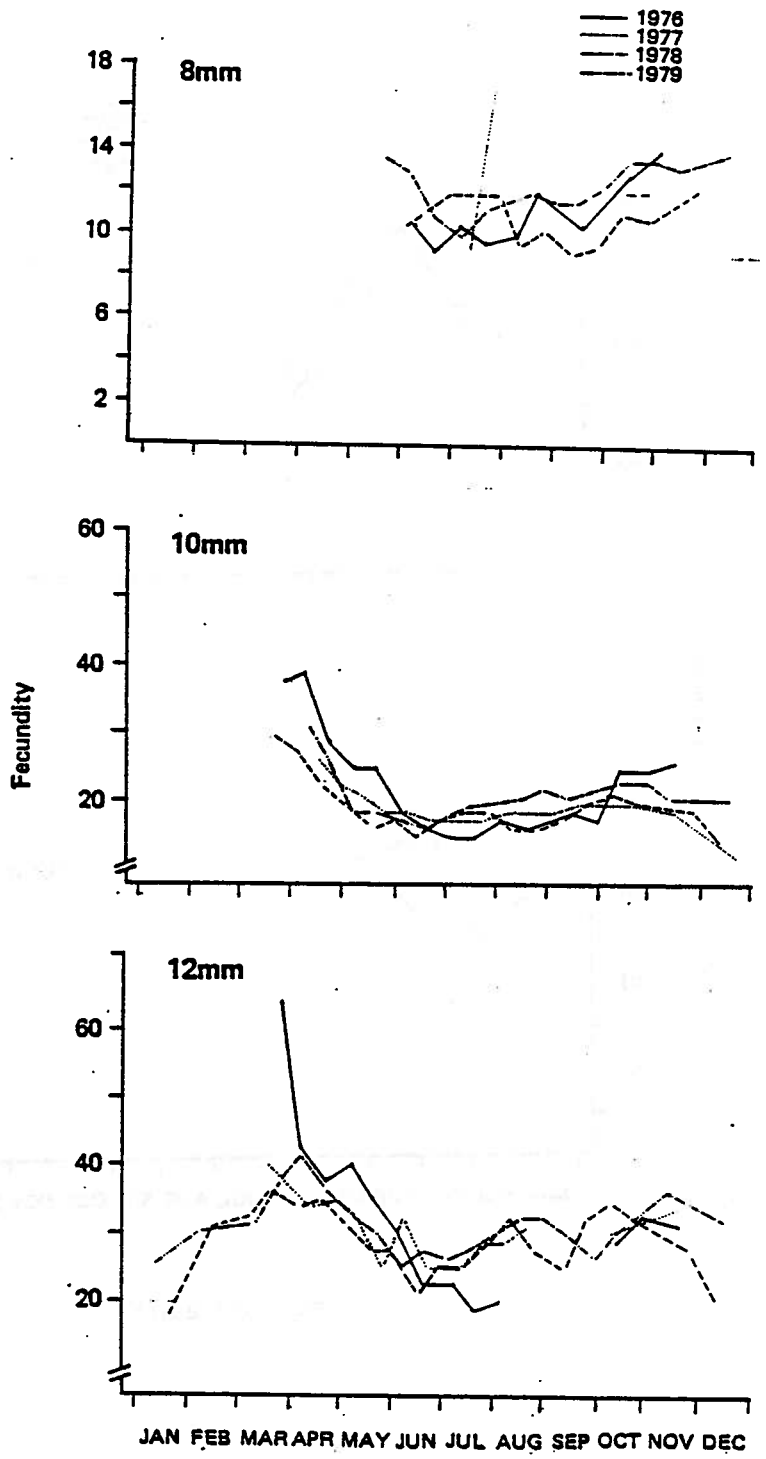


Figure C-4. Length-specific monthly mean fecundities of neomysis in the Bay-Delta study area, 1976-1979.

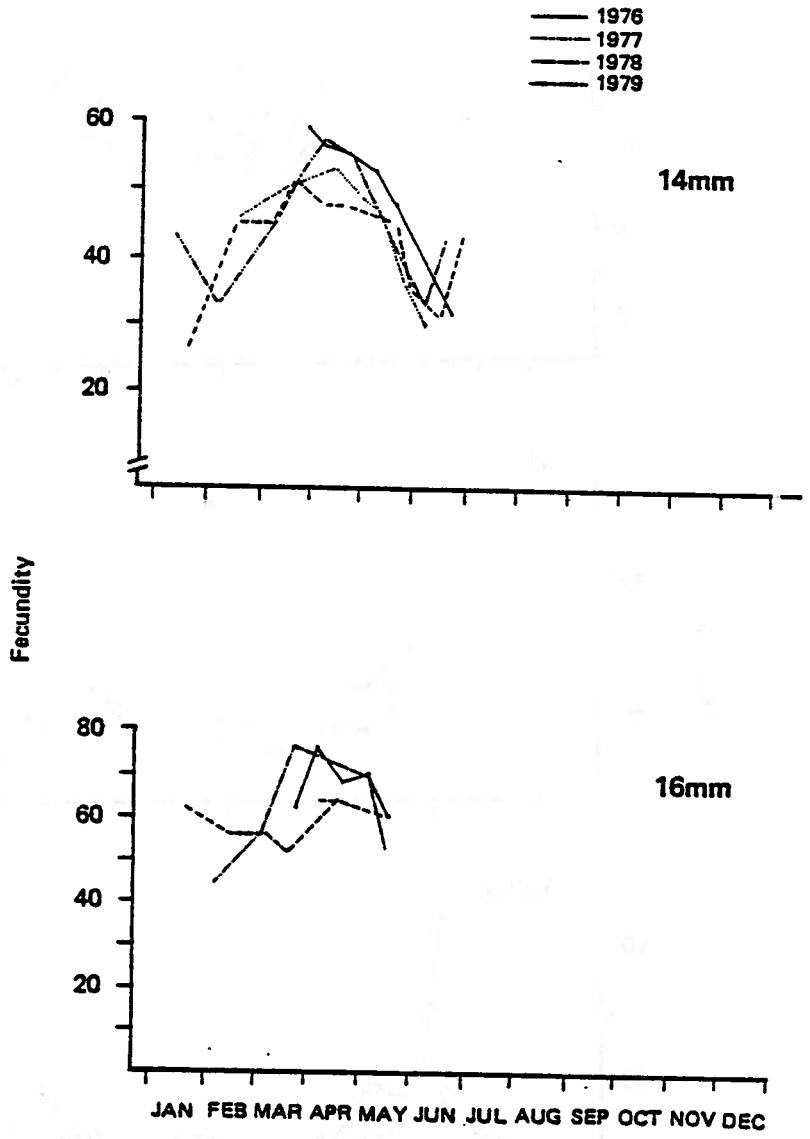


Figure C-4 (cont.)

Only data with temperatures in the 8-25 C range were included in the analysis, as this is the typical range to which neomysis is exposed during a year, and at lower temperatures species reproduce only at one period in the year or once every two years. The equation that results from regression analysis is

$$D = 676.3\,(TEMP)^{-1.3468}. \qquad \text{(C-26)}$$

With the number of samples equal to 9, r^2 (the coefficient of determination) was 0.61.

The development time, D_i, that resulted from applying Equation C-26 to the water temperature at the time of each survey was used in Equation C-24 to calculate the birth rate between each pair of surveys. The results are described in Section 5.5.2.2

C.2.3 Entrainment Mortality

The abundance of organisms in a population at a given time N_t is a function of a previous abundance, N_0, and the rate of change in abundance:

$$N_t = N_0 e^{rt} \qquad \text{(C-26)}$$

Setting migration aside, the rate of change, r, is the difference between the birth rate and the death rate of the population: $r = b - d$. The overall death rate, (d), includes among its various components P, the daily instantaneous entrainment mortality rate.

In the absence of entrainment mortality, the hypothetical rate of change, r', would be higher: $r' = r + P$, and so N_p, the abundance that would have been found in the absence of P, is correspondingly higher:

$$N_P = N_0\, e^{(r+P)t} \qquad \text{(C-27)}$$

Percentage reduction, R, the percentage difference between abundance with entrainment mortality and abundance without it, may be expressed as

$$R = 100(1 - N_t/N_p), \text{ or} \qquad \text{(C-28)}$$

$$R = 100(1 - N_0e^{rt}/N_0e^{(r+P)t})$$

$$= 100(1 - N_0e^{rt}/N_0e^{rt}e^{Pt})$$

$$= 100(1 - 1/e^{Pt})$$

$$= 100(1 - e^{-Pt}).$$

For a series of periods in a year, the cumulative product of P and t, called C, is calculated as

$$C = \sum_{i=1}^{Y} P_i t_i \qquad \text{(C-29)}$$

where

P_i = the daily instantaneous entrainment mortality rate for period i (= P in Equation 5-10)

t_i = the number of days in period i

Y = the number of periods in a year.

In these terms, R is calculated for the year as

$$R = 100(1 - e^{-C}). \qquad \text{(C-30)}$$

Thus, the various entrainment mortality rates calculated, and their associated periods, can be used directly in the calculation of percent reduction.

The variance of R is the following:

$$Var(R) = 100e^{-C^2} \sum_{i=1}^{Y} Var(P_i)t_i^{\,2} \qquad \text{(C-31)}$$

where

$Var(P_i)$ = the variance of P, the daily instantaneous entrainment mortality estimate, calculated as in Section 5.5.3.1.

C.3 DEVELOPMENT OF AN INDEX OF STRIPED BASS YEAR CLASS STRENGTH AND POPULATION DYNAMICS USED IN ESTIMATING PERCENTAGE REDUCTION

Development of the adult striped bass population model described in Section 5.6.2 required the determination of an index of young-of-the-year striped bass abundance for use in regression modeling and an estimate of the percentage of the population lost as a result of entrainment mortality each year. The approach and methodology used to develop an index of abundance for striped bass and various aspects of the population dynamics used in estimating percentage reduction are described in this section.

C.3.1 Determination of an Index of Striped Bass Abundance

An unbiased measure or index of striped bass abundance must be developed before any regression modeling of factors affecting year class strength can be attempted. This index should reflect the abundance of the entire striped bass population at some point in time after the year class is set, and should be calculated in a consistent manner across all years, using a long-term data base. The approach used in our development is divided into 4 sections:

1. a definition of striped bass year class strength,
2. a description of the data base,
3. analytical methods, and
4. data exclusions.

C.3.1.1 Definition Of Striped Bass Year Class Strength

Ricker (1975) has defined year class strength as the relative abundance of successive year classes. Tesch (in Ricker 1971) defines year class as the number of organisms produced in a given year. For the purposes of this report, we have defined striped bass year class strength as the abundance of organisms in some index size-group produced in a given year relative to other years.

Chadwick et al. (1977) have said that most of the annual variability in the mortality rates of striped bass occurs during the first two months of life. According to Ricker (1971), the survival rates of later life history stages of fish do not generally cause

fluctuations in year class strength. That is, once the year class is set, the mortality rates of the older life stages are relatively constant from year to year. This implies that if one can determine at what size the year class is set, a determination of the relative abundance of any larger size group can be used as an unbiased measure of year class strength.

Chadwick et al. (1977) concluded that most of the variability in survival of striped bass in the Sacramento - San Joaquin Delta occurs before the population mean length reaches 38 mm. Chadwick (1964) reports a direct relationship between the relative abundance of striped bass young-of-the-year (measured at 38 mm) and subsequent year class recruitment. It appears from the literature (Polgar 1977) that the strength of the striped bass year class is relatively independent of egg production and is set some time before the mean length of the population reaches 38 mm. This implies that before an index of abundance can be derived, the time when the year class is set must be determined. Thus, one of the first steps in the analysis is to determine at what minimum size the year class is set and then to estimate the number of striped bass produced in some size group larger than the size at which year class was set.

Polgar (1977) presents a method of estimating the number of organisms produced in a particular size class which requires only data on the numbers and sizes of individuals in a population at a series of points in time. Data of this type have been collected and tabulated by the CDF&G for the past 20 years; they are described in the next section.

C.3.1.2 Description of Data Bases

CDF&G has conducted three types of annual studies of young-of-the-year striped bass: a spring egg and larval study, a summer juvenile study, and a fall and winter yearling study.

The spring egg and larval study was designed to monitor spawning and to measure the densities of eggs and larvae at various stages, to permit the estimation of mortality rates (Stevens 1977). A complete survey of approximately 30 stations (Figure C-5) was usually conducted every two days from April through June (Table C-9). A single 10-minute depth-integrated oblique tow was taken at each station with either a 930-μm or a 505-μm mesh net. The water volume sampled was determined by

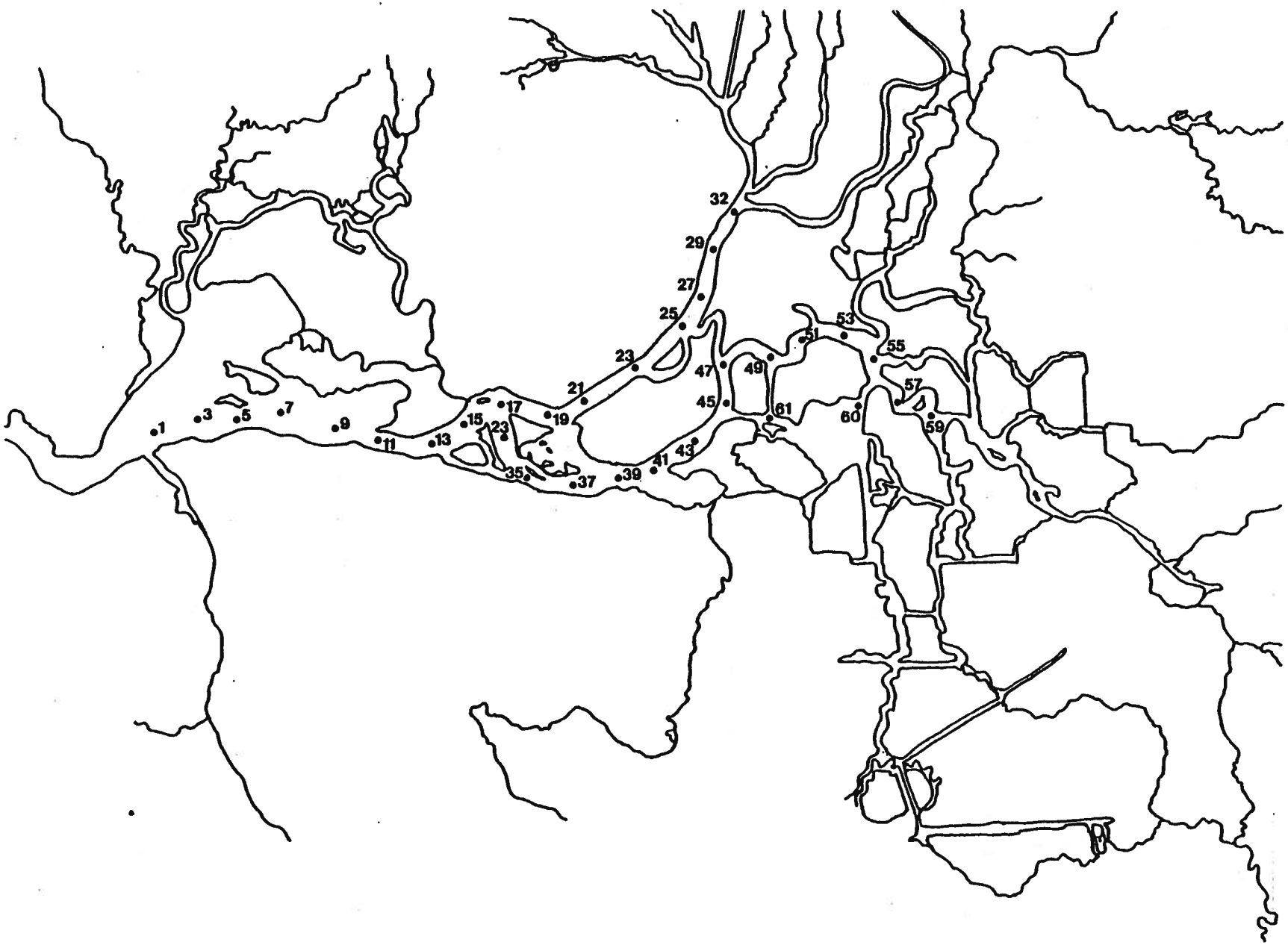


Figure C-5. Sampling stations used by the California Department of Fish and Game for striped bass egg and larval surveys in the Bay-Delta, 1961-1977.

TABLE C-9 DESIGN OF THE CALIFORNIA DEPARTMENT OF FISH AND GAME STRIPED BASS EGG AND LARVAL STUDY, 1967-1977

Year	Julian Date Study Began	Julian Date Study Ended	Number of Days Between Surveys	Stations Sampled	Size of Mesh (m)
1967	132	178	2-8	1-61	930
1968	94	180	2	1-61	930
1969	107	191	7	1-61	930
1970	111	181	2	1-61	930
1971	144	196	2	1-61	930
1972	118	194	2	1-94	930
1973	121	199	2	1-84	930
1974					
1975	121	199	2	1-84	505
1976	102	148	2-10	13-7 33-61	505
1977	101	159	2	11-82	505

Note: No survey was made in 1974.

Source: CDF&G unpublished.

flowmeters mounted on the net frames (Stevens 1977). All striped bass collected were counted and measured. CDF&G compiled the density of striped bass of each 1-mm size interval for each station in each survey (CDF&G unpublished data).

The summer juvenile study was designed to measure the relative abundance of striped bass between 17 and 50 mm, fork length (Chadwick 1964). Although surveys have been made with various tow nets since 1947, a standard survey design was not adopted until 1962 (Stevens 1977). The annual studies consisted of 3-5 surveys covering 30 stations from the Benicia Bridge east through the Delta (Figure C-6). The surveys were conducted at 2-week intervals, starting in mid-June and continuing until the mean length of the striped bass collected exceeded 38 mm, which generally occurred in late July or early August (Table C-10). During each survey, three 10-minute depth-integrated oblique tows were taken at each station using a net described by Miller (1977). CDF&G assumed that all tows strained equal volumes of water. CDF&G compiled the total number of striped bass in each 0.1-in. (1961-1970) or 1.0-mm (1971-present) length group collected for each tow (CDF&G unpublished data).

The fall and winter yearling midwater trawl study was designed to estimate the total abundance of striped bass between 60 and 140 mm fork length. The primary objective of the survey was to determine if the mortality rate during the fall could vary enough between years to be a major factor causing variation in the abundance of adult bass (Stevens 1977). Approximately 87 stations scattered from south San Francisco Bay through the Delta were sampled once a month from August or September to March (Table C-11). One 12-minute depth-integrated oblique tow was made at each station with gear described by Stevens (1977). As in the summer juvenile study, all tows were assumed to strain equal volumes of water. Although length measurements were made, CDF&G reported only the total number of striped bass collected at each station in each survey. Upon request, the mean length of striped bass collected in each monthly survey was obtained (L. W. Miller, CDF&G, 1979, personal communication).

C.3.1.3 Analytical Methods

The first objective of this analysis was to determine the size at which the year class was set; the second, to develop an unbiased measure of striped bass year class

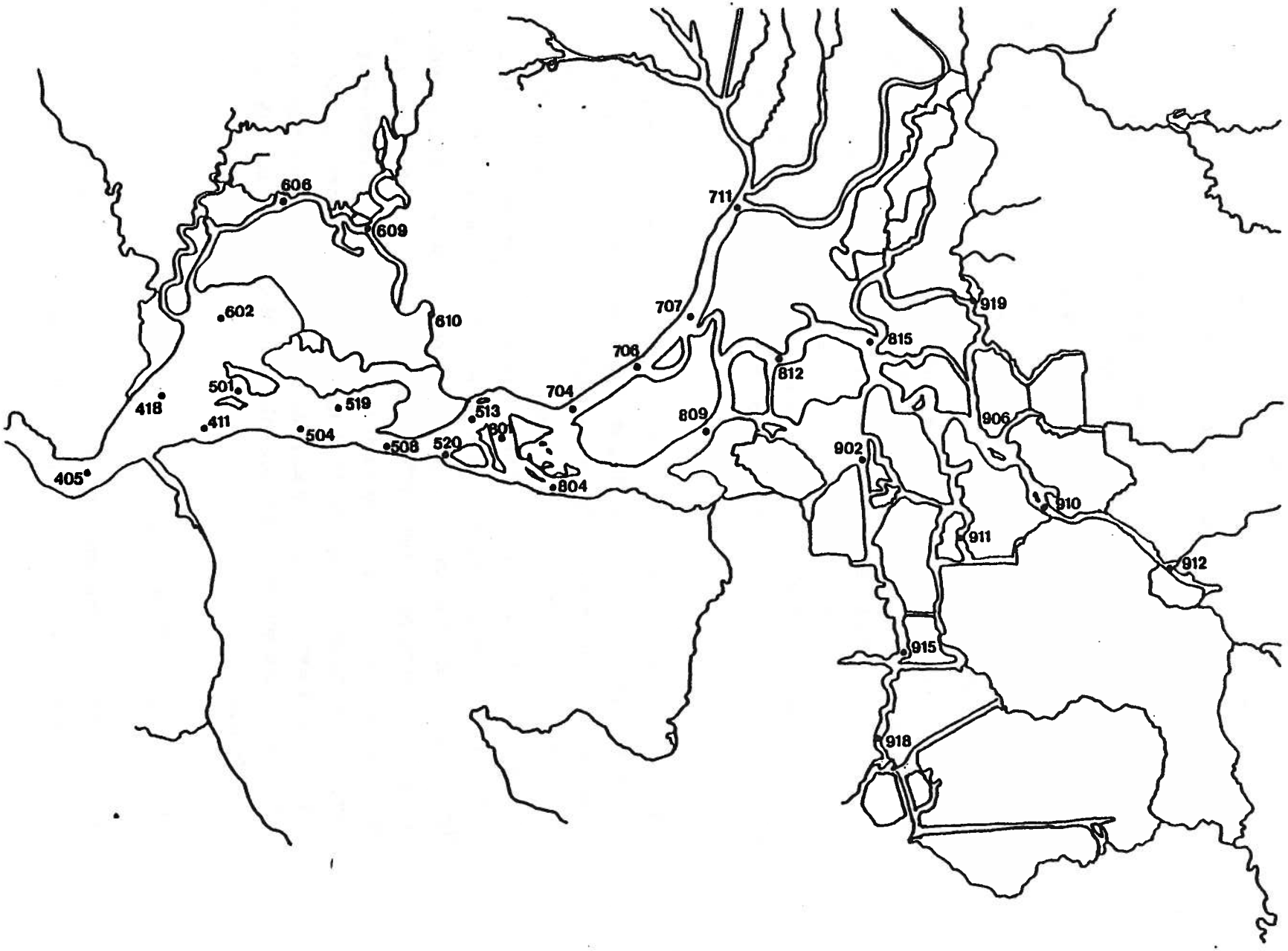


Figure C-8. Sampling stations used by the [illegible]ia Department of Fish and Game to collect 17-50mm juvenile [illegible]au bass in the Bay-Delta, 1961-1978.

TABLE C-10 DESIGN OF THE CALIFORNIA DEPARTMENT OF FISH AND GAME STRIPED BASS JUVENILE STUDY, 1961-1979

Year	Julian Date Study Began	Julian Date Study Began	Number of Surveys
1961	159	215	5
1962	180	221	4
1963	168	224	5
1964	170	212	4
1965	175	218	4
1966			
1967	181	239	5
1968	171	213	4
1969	189	231	4
1970	178	220	4
1971	182	238	5
1972	172	215	4
1973	176	204	3
1974	180	209	3
1975	184	226	4
1976	186	214	3
1977	177	219	4
1978	180	208	3
1979	184	213	3

Note: No study was made in 1966.

Source: CDF&G unpublished.

TABLE C-11 DESIGN OF THE CALIFORNIA DEPARTMENT OF FISH AND GAME STRIPED BASS YEARLING STUDY, 1967-1979

Year	Months of Study	Number of Surveys
1967	September-March	7
1968	August-March	8
1969	August-October, December-January, March	6
1970	August-March	8
1971	September-March	7
1972	August-January, March	7
1973	September-January	7
1974		
1975	September-December	4
1976	October, November, Janauary	3
1977	September-January, March	6
1978	September-December, February	5
1979		

Note: No study was made in 1974 and 1979.

Source: CDF&G unpublished.

strength. To this end, we applied Polgar's (1977) methods to the CDF&G data bases in order to estimate the number of striped bass in various size groups produced each year.

The frequency of sampling varied among the CDF&G studies: every two days for the egg and larval surveys, every two weeks for the juvenile surveys, and once a month for the yearling surveys. The difference in the frequency of sampling, however, does not seriously affect the methods used to estimate production. For each survey, the number or density of striped bass at each station for various size groups was reported. In studies where flow measurements were not taken, a constant volume of water, 1,128 m^3 (40,000 ft^3) was assumed to have been strained by the nets. For this analysis, the Sacramento - San Joaquin estuary was partitioned into a number of strata (Table C-12), selected on the basis of biological and hydrological similarity and ease of boundary definition. Each CDF&G station was assigned to one of the strata.

An instantaneous stratum density was calculated for each survey for each size group by averaging the densities for all stations within a stratum. These densities were then multiplied by the estimated volume in each stratum (Table C-12) to obtain the an estimate of the total abundance of striped bass in that stratum for that survey. Polgar (1977) refers to these abundance values as "snapshot" representations of the fish standing crop in any one stratum at the time of measurement. The stratum abundance estimates are then summed to obtain an estimate for each survey, i, of the total number of striped bass for each size group, A_i, in the Sacramento - San Joaquin estuary. It should be noted here that this estimate, A_i, assumes that the stations sampled cover the entire distribution of the population. If, as will be pointed out later, the sampling design misses a portion of the population, the calculated abundance estimates, A_i, will be biased an unknown quantity low.

As Hackney and Webb (1978) point out, over time the total number of fish in a particular size group, A_i, first increases, then peaks, and then decreases (Figure C-7). In a multiple-cohort population like that of striped bass, not all fish are recruited into a particular size group simultaneously, since they were spawned at different times and grow at different rates. As individuals grow into a size class, their absolute numbers increase, as indicated by the ascending limb of the abundance vs. time distribution. Once most of the fish have entered the size class, the peak of the distribution, their absolute numbers begin to decrease as they either die or grow

TABLE C-12 STRATUM DESIGNATIONS AND VOLUMES AND CORRESPONDING CALIFORNIA DEPARTMENT OF FISH AND GAME SAMPLING STATIONS

Stratum	Volume ($ft^3 \times 10^9$)	Sampling Stations	
		Egg and Larval Survey	Juvenile Survey
South Delta	9.09	57-61	902-918
North Delta	2.18		919
Lower San Joaquin River	7.75	23, 35-55	520, 801-815
Lower Sacramento River	6.01	17-29	704-711
Upper Sacramento River	1.97	32, 84	
Montezuma Slough	1.55		606-610
Suisun Bay main channel[a]	14.16	1-15	405, 411, 501-513
Grizzly and Honker bays	12.05		418, 602, 519
San Pablo Bay	51.28		—

(a) Includes Carquinez Strait.

Note: (—) indicates no samples taken.

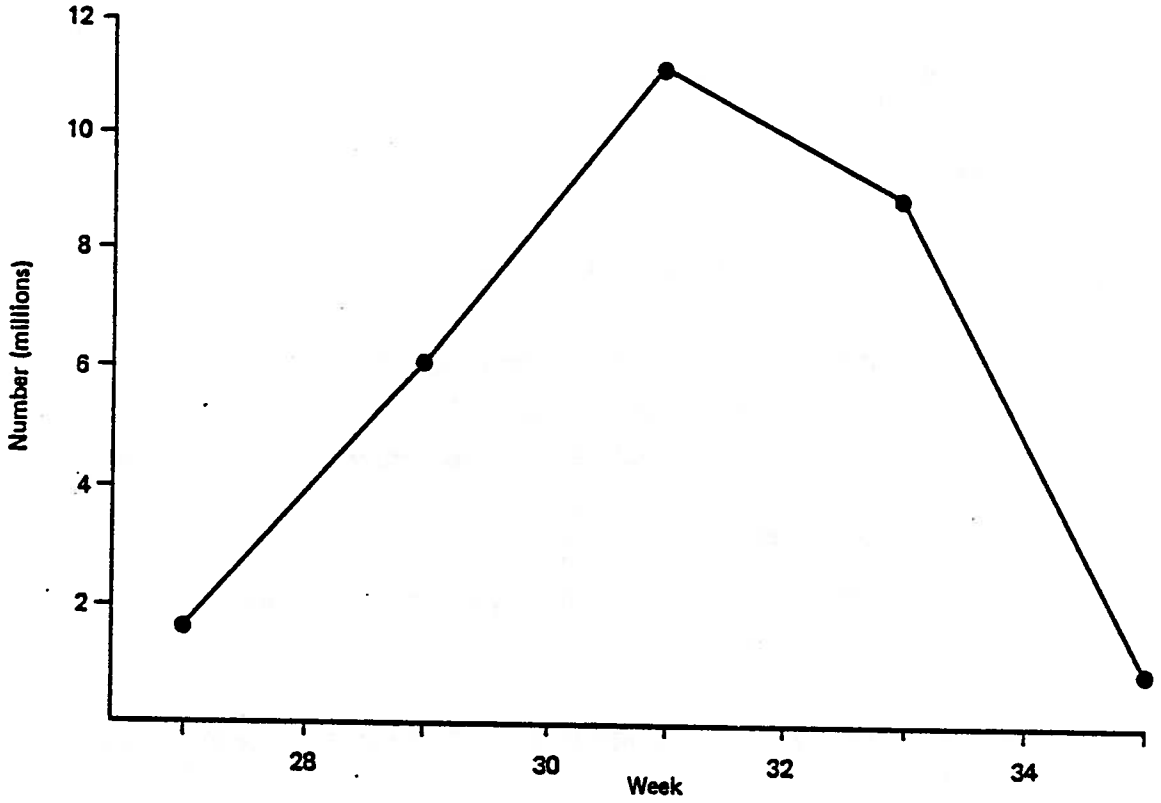


Figure C-7. Estimated numbers of 35-39mm striped bass in the Bay-Delta, 1971.

out of the size class, as indicated by the descending limb of the distribution. In the field this results in a large collection of individual cohorts separated in time.

According to Polgar (1977), the collective abundance produced by these individual cohorts is equivalent to the abundance that would be observed if all the cohorts were combined into a single group present in the system only for the duration of the length class. This assumption, illustrated in Figure C-8, can be expressed mathematically as

$$tA = \int_0^D N_0 e^{-\lambda t}\, dt \qquad \text{(C-33)}$$

where

t = time between sampling periods
D = duration of the length class
N_0 = initial number of fish in the length class
λ = instantaneous total mortality for the length class
A = the total number of fish in a year class passing through a given length class; $\sum_{i=1}^{J} A_i$
A_i = abundance of fish in the length class in survey i
J = number of surveys.

Integrating the right side of Equation C-33 and solving for N_0 yields

$$N_0 = \frac{tA}{1-e^{D}}. \qquad \text{(C-34)}$$

If this expression for N_0 is inserted into the equation

$$N_s = N_0 e^{-\lambda D} \qquad \text{(C-35)}$$

where

N_s = number of individuals surviving or produced from the length class,

the following estimate of N_s can be made:

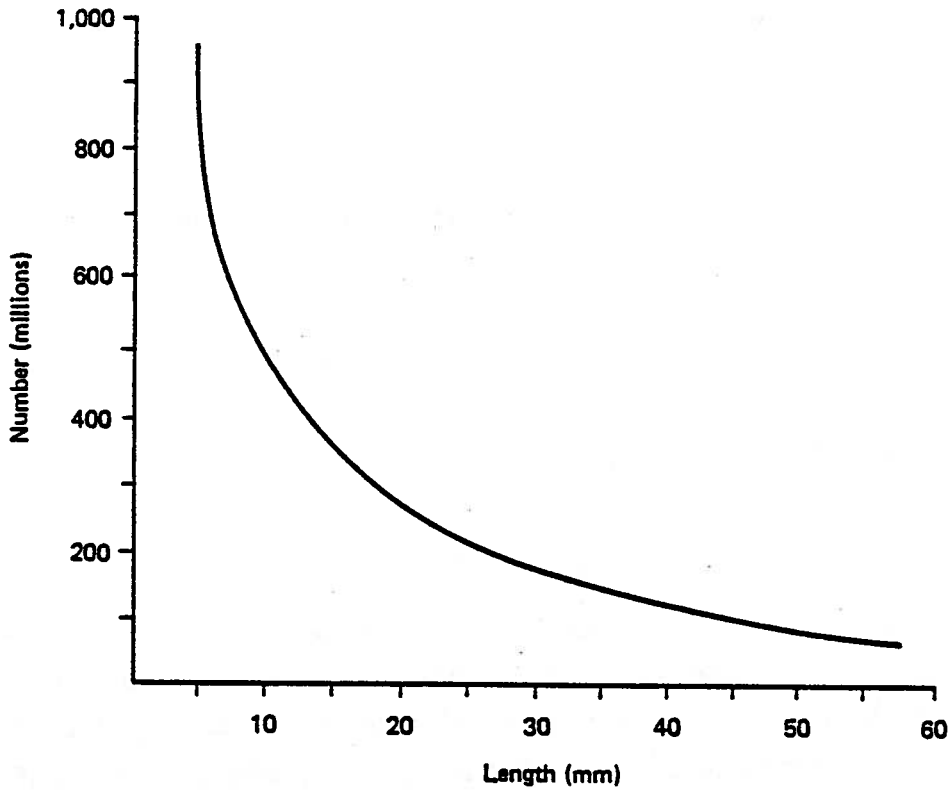


Figure C-8. Hypothetical curve of abundance vs. length for striped bass larvae and juveniles.

$$N_s = \frac{tA\ e^{-\lambda D}}{1-e^{\lambda D}} \qquad \text{(C-36)}$$

In order to compute N_s, the only parameters that remain to be estimated are the instantaneous rate of mortality, λ, and the duration of the length class, D. The instantaneous mortality rate between successive length classes, λ, can be computed using the following equation:

$$N_s = N_0^{(k+1)} \qquad \text{(C-37)}$$

where

$N_0^{(k+1)}$ = the initial number of organisms in the succeeding length class, k_a.

A substitution in Equations C-34 and C-36 then gives us

$$\frac{A\ \lambda e^{-\lambda D}}{1-e^{\lambda D}} = \frac{A_{k+1}\lambda_{k+1}}{1-e^{\left(\lambda_{k+1} D_{k+1}\right)}}, \qquad \text{(C-38)}$$

which permits the computation of λ through an iterative process. Knowing the instantaneous mortality rate, λ, and the estimated duration of the length class, D (see Section C.3.2.1 for discussion), the number of organisms in a particular size group, k, produced, $N_s^{(k)}$, can then be computed using Equation C-36.

This method of analysis was applied to the CDF&G egg and larval and juvenile survey data. It could not be applied to the yearling survey data, because the size-specific abundance estimates required by this method were not available from CDF&G. After consideration of potential biases in the data set (Section C.3.1.4), production estimates were made for all applicable size groups. These size-specific production estimates were examined to determine when the year class was set. A decision was then made to as to what size group to use for the index of striped bass year class (Section 5.6.2.2.1).

C.3.1.4 Data Exclusions

Although the methods just described will generate estimates of the number of striped

bass produced in various size groups, these estimates are not necessarily unbiased. Upon examination of the data collected by CDF&G, it was concluded that unbiased estimates of numbers produced could not be made for all size groups in all years. Data exclusions were required, either because the annual sampling survey did not cover the entire distribution of striped bass or because the surveys used to estimate the total abundance, A_i, did not include the peak recruitment into the size class. The years and size groups excluded from further analysis are discussed in this section.

The first and most important data base requirement for estimating year class strength is that the sampling design cover the entire striped bass distribution. If striped bass are distributed in regions outside of the sampling area, the abundance estimates, and in turn the production estimates, are biased an unknown amount low. Weekly surveys conducted by EA in 1978 and 1979 (EA 1981) indicate that the distribution of the early life stages of striped bass in the estuary is a function of the location of the entrapment zone (2-10 mmho) which is in turn determined by the magnitutude of freshwater outflow. Striped bass begin spawning in late April and early May, and the peak of spawning generally occurs in mid-May (Turner 1976). The eggs generally hatch before they reach the entrapment zone. The planktonic yolk-sac larvae (3-6 mm) generally do not penetrate the entrapment zone and are rarely found in conductivities greater than 1-2 mmho. Larval bass (6-16 mm) may penetrate further into the entrapment zone (up to 5 mmho), but peak densities usually occur at 0.5-1 mmho. The densities of juvenile bass (>16 mm) peak in the 1-6 mmho range.

In summary, the yolk-sac and larval bass are generally located in conductivities less than 2 mmho (the null zone), whereas the juvenile striped bass can be found at conductivities up to 5 and 6 mmho (the upper end of the entrapment zone). As discussed in Appendix A, the locations of the entrapment and null zones are directly related logarithmically to outflow past Chipps Island (see Figure A-2). It follows that the location of young striped bass in the estuary should be related to the magnitude of the outflow. Figure C-9 shows the distribution of 7-10 mm striped bass in May in years of relatively low and relatively high outflow. In the lower-flow years, the center of the striped bass distribution is located in the lower reaches of both the Sacramento and San Joaquin rivers. In the higher-flow years, the center of the striped bass distribution was located further downstream, in east and west Suisun Bay. In the very high outflow years of 1967 and 1969, it appears that a part of the striped bass distribution extended into San Pablo Bay, an area not covered by the CDF&G surveys. The occurrence of striped bass in San Pablo Bay during high outflow

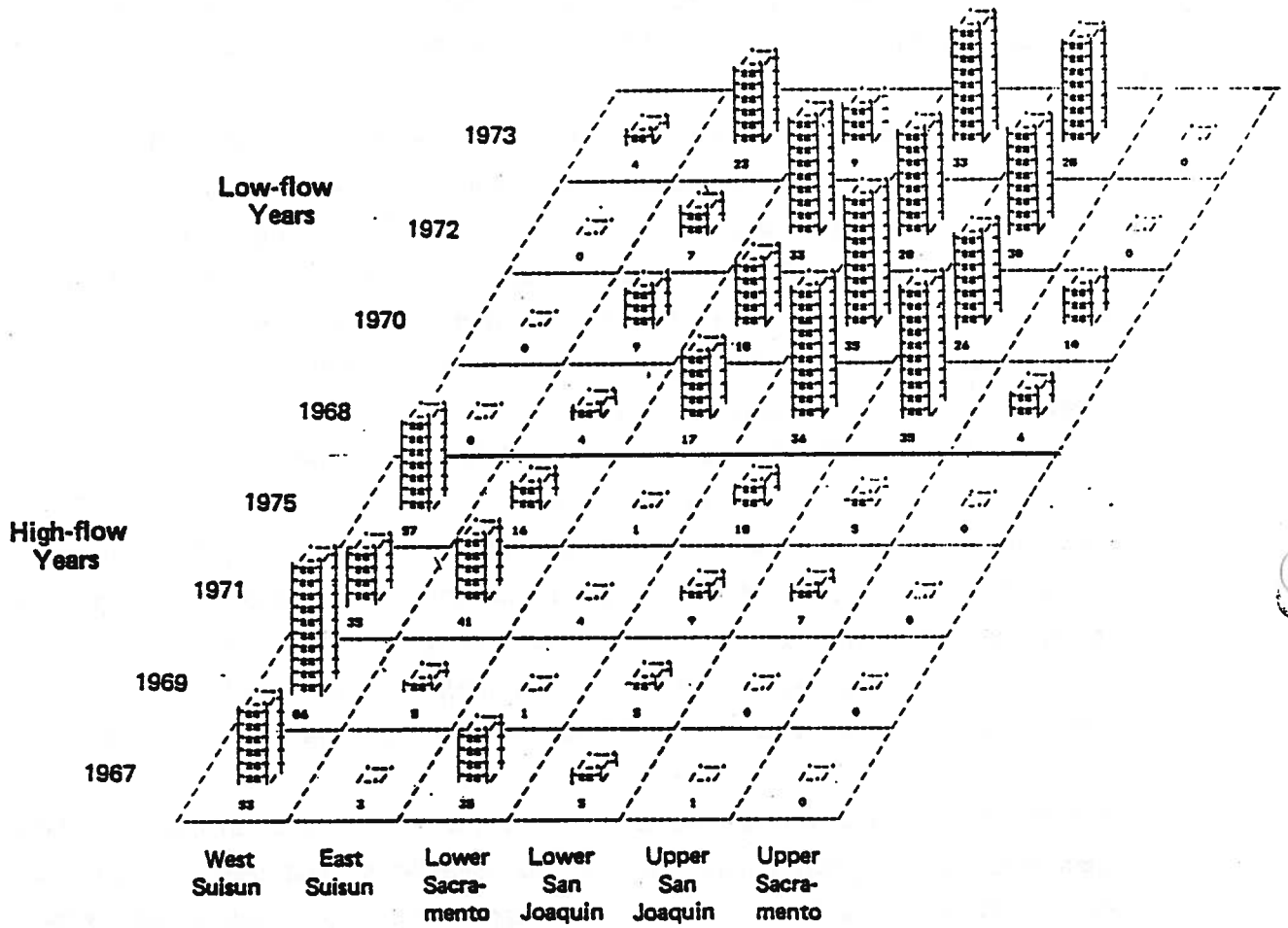


Figure C-9. Relative abundances (%) of 7-10mm striped bass in various strata of the Bay-Delta during May in years of high and low Delta outflow.

years is well documented in the literature. Calhoun (1952) theorized that in high-flow years the extensive shoals in northern San Pablo Bay could trap the young bass that pass through Carquinez Strait and that they might remain there after growing large enough to control their own location.

Calhoun also points out that no bass fry were ever taken in deep waters in San Pablo Bay, whereas bass fry were readily taken in the shallow areas. Although townet sampling in 1958 (May outflow >79,000 cfs) collected only a few bass in San Pablo Bay (Chadwick 1964), beach seining during the same year caught more than 50 young-of-the-year per haul in San Pablo Bay at Rodeo. Chadwick points out that appreciable concentrations of young striped bass occurred in the Napa River area in 1958, but felt that this may have been an unusual occurrence, since virtually no bass were caught there in 1957, 1959, 1960, and 1961 (all years with relatively low May outflows: 33,000, 7,000, 12,000, and 8,000 cfs, respectively). Chadwick goes on to state that the omission of San Pablo Bay, the Napa River, and the western part of Carquinez Strait may have seriously biased his survey results. Chadwick found striped bass in San Pablo Bay and the Napa River when May outflows were 79,000 cfs but not when May flows were less than 33,000 cfs. In late summer and early fall 1967 (May outflow >75,000 cfs), Wooster (1971) reports collections of over 100 young-of-the-year striped bass per seine haul in San Pablo Bay, and an additional 300 striped bass were collected in central San Francisco Bay in late summer 1967. In their report to the State Water Resources Control Board, CDF&G (1976) pointed out that sampling in 1967 and 1974 established that high flows do wash larval bass to San Pablo Bay and San Francisco Bay and that many of those bass survive and live along the shoreline, where they are not vulnerable to the tow net. The report goes on to say that the survival limit demonstrated by the CDF&G young-of-the-year index is artificial because the survey does not sample the area west of Carquinez Strait adequately. According to Stevens (1977), when spring and summer flows were low or moderate, young bass abundance was indexed effectively by the CDF&G summer townet survey, but when high outflows transported bass to San Pablo Bay, the survey underestimated bass abundance through inadequate sampling in San Pablo Bay. Stevens goes on to say that although the catches of bass in San Pablo Bay are relatively small, significant numbers could inhabit that bay because of its large volume. In 1978 (May outflow >40,000 cfs), striped bass entrainment densities at the Oleum Power Plant on San Pablo Bay ranged between 1 and 2 fish per cubic meter in June and July (PGandE unpublished).

This review strongly suggests that during years of high spring outflow, striped bass are transported into San Pablo Bay and that these bass survive and live in the shoal areas, where they are not vulnerable to CDF&G townet surveys. Although the data are limited, studies by CDF&G in 1958 and by EA in 1978 indicate that when young bass are transported into San Pablo Bay, they remain there after the null zone has retreated back up into the delta. In June and July 1958, more than 650 bass were collected by CDF&G in the Napa River area (Chadwick 1964), when the null zone was estimated to be 20 miles further upstream. In 1978, striped bass were entrained throughout the months of June and July at the Oleum Power Plant, at a time when the null zone was located near Chipps Island.

On the basis of the salt front model described in Appendix A and the assumption that the null zone is located approximately 10 miles further downstream at low tide than at high tide, the number of days the null zone washed into San Pablo Bay during April, May, June, and July for the years 1961-1979 is presented in Table C-13. In 1963, 1967, and 1969, the null zone was located in San Pablo Bay throughout the entire months of April and May and for a part of June (and into July in 1967). In 1978, the null zone was in San Pablo Bay during all of April and the first two weeks in May, and in 1965 the null zone was there in April and the first week of May. The longer the null zone is located in San Pablo Bay, the greater the opportunity for striped bass to be transported there and remain, resulting in a greater bias in the standing crop estimates. For this reason we have chosen to eliminate the years 1963, 1967, 1969, and 1978 from the analysis, since an unknown proportion of the striped bass population, residing in San Pablo Bay, was not represented in CDF&G sampling surveys. Although some striped bass were presumably transported into San Pablo Bay in 1965, we believe that the biases in standing crop estimates are relatively low and that that year should still be included in further analyses.

In summary, when striped bass are transported into San Pablo Bay by high spring flows and remain there after the null zone has moved back upstream, an unbiased estimate of year class strength cannot be made from the available data.

The second data base requirement for estimating year class strength is that the surveys used to estimate the total striped bass abundance, A_i, be of sufficient length to include the peak of recruitment into the size class. As discussed earlier, in a multiple-cohort population not all fish are recruited into a particular size group

TABLE C-13 ESTIMATED NUMBER OF DAYS THE NULL ZONE WASHED INTO SAN PABLO BAY IN VARIOUS MONTHS, 1961-1979

Year	Month			
	April	May	June	July
1961	0	0	0	0
1962	0	0	0	0
1963	30	31	3	0
1964	0	0	0	0
1965	21	4	0	0
1966	0	0	0	0
1967	30	31	30	6
1968	0	0	0	0
1969	30	31	20	0
1970	0	0	0	0
1971	10	0	0	0
1972	0	0	0	0
1973	0	0	0	0
1974	27	0	0	0
1975	10	0	0	0
1976	0	0	0	0
1977	0	0	0	0
1978	30	13	0	0
1979	0	0	0	0

simultaneously. In order to avoid biases associated with differences in time of spawning and in growth rates and to be assured of completely representing the size class, the ascending limb, the peak, and the descending limb of the total abundance vs. time distribution (Figure C-7) must be present in the data. Furthermore, in order to make unbiased comparisons across years, the same size groups and number of surveys must be used each year. In examining the abundance vs. time distributions, it was found that surveys capturing the peak recruitment and covering a 5-week period were of sufficient length to represent the recruitment into the size class adequately. If the peak in recruitment was not evident in the data, the number of striped bass produced in that size class could not be calculated.

In summary, data were excluded from further analysis if (1) the sampling design used in that year did not adequately cover the entire striped bass distribution or (2) the peak of recruitment into the size class was not evident in the data. These exclusions were required in order to be assured of obtaining a consistent, unbiased estimate of striped bass year class strength.

C.3.2 Striped Bass Population Dynamics Used in Estimating Percentage Reduction

Estimates of percentage reduction from various length classes resulting from entrainment mortality were computed by dividing an estimate of the number of individuals surviving the length class by an estimate of the number of individuals that would have survived that length class in the absence of entrainment mortality, expressed mathematically as

$$P = \left(1 - \frac{N_0 S_n S_p}{N_0 S_n}\right) \times 100 \qquad \text{(C-39)}$$

where

P = percentage reduction
N_0 = initial population size of the length class
S_n = probability of survival due to natural mortality
S_p = probability of survival due to power plant entrainment.

To compute power plant survival, S_p, for all length classes, a number of

computational steps were taken. Prior to these steps, two preliminary assumptions were examined upon which the percentage reduction model was based. The first of these was a hypothesis defining the functional relationship between numbers of individuals in each length class and time:

$$N_t = N_0 e^{-\lambda t} \qquad \text{(C-40)}$$

where

N_t = population size at time t
N_0 = initial population size
λ = instantaneous total mortality rate
t = arbitrary time unit.

The exponential decline in a population represented by this assumption is commonly used in describing population dynamics of both vertebrate (Hutchinson 1978) and invertebrate populations (Edmondson 1977). It has been applied in other empirical models used in computing percentage reduction (Boreman et al. 1978; Texas Instruments 1975) of fish.

The second major assumption of the model was that power-plant-related and natural mortality act as competing sources of mortality in removing organisms from the populations of each length class. This is expressed mathematically as

$$\lambda = p + n \qquad \text{(C-41)}$$

where

p = instantaneous mortality rate associated with power plants
n = instantaneous mortality rate associated with all other sources of mortality.

This assumption implies that both sources of mortality affect the population simultaneously throughout the duration of the length class. It is equivalent to one frequently used in fisheries management, where fishing mortality and natural mortality are considered to be competing (Ricker 1975). The assumption further implies that the two sources of mortality are independent forces, so that removal of

one source does not influence the effect of the other on the population.

Substituting Equation C-41 into Equation C-40 yields

$$N_t = N_0 e^{-(p+n)t} . \qquad \text{(C-42)}$$

Removing natural mortality from Equation C-42 produces

$$N_t = N_0 e^{-pt} \qquad \text{(C-43)}$$

where

N_t = population size at time t in the absence of natural mortality.

With t set at the duration of the length class, it follows that

$$\frac{N_t}{N_0} = S_p \qquad \text{(C-44)}$$

and that therefore,

$$S_p = e^{-pt} . \qquad \text{(C-45)}$$

Thus, to estimate power plant survival, S_p, (and ultimately percentage reduction) for each length class, the total instantaneous mortality rate was first determined, from which the instantaneous power plant mortality was then computed.

C.3.2.1 Estimation Of Total Instantaneous Mortality Rate

The model presented in Equation C-40 represents the dynamics of a single-cohort population where all members of the length class begin the stage at the same time. This model would accurately represent the striped bass of the Sacramento - San Joaquin system if all spawning occurred on the same day and development up to and through the length class was uniform among all members of this single-cohort population. Since striped bass spawn over an extended period of time in the Sacramento and San Joaquin rivers, and development times most likely vary between cohorts, the initial and final population sizes of Equation C-40 cannot be measured

directly from field observations. Therefore, an indirect method of estimating these variables, originally proposed by Polgar (1977) was used, in which the area under the hypothetical single-cohort curve was equated to the area under the observed field abundance curve, written as

$$A = \int_0^t N_0 e^{-\lambda t} \, dt \qquad \text{(C-46)}$$

where

- A = area under the field abundance curve of a population, expressed in organism-days.
- N = initial population size
- λ = total instantaneous mortality
- t = duration of length class, in days.

This can be expressed geometrically as in Figure C-10.

Integrating the right side of Equation C-46 and solving for N_0 gives

$$N_0 = \frac{A\lambda}{1 - e^{-\lambda t}} \qquad \text{(C-47)}$$

Substituting the above expression for N_0 into Equation C-40 yields

$$N_t = \frac{A\lambda e^{-\lambda t}}{1 - e^{-\lambda t}} \qquad \text{(C-48)}$$

Since the end point of a given length class is the beginning point of the following length class, $N_t^{(k)} = N_0^{(k+1)}$, it follows for a succession of length classes (k), each seen in Equations C-46, C-47, and C-48 as a separate population, that

$$\frac{A^{(k)} \lambda^{(k)} e^{-\lambda^{(k)} t^{(k)}}}{1 - e^{-\lambda^{(k)} t^{(k)}}} = \frac{A^{(k+1)} \lambda^{(k+1)}}{1 - e^{-\lambda^{(k+1)} t^{(k+1)}}} \qquad \text{(C-49)}$$

For n successive length classes there are n-1 equations in the form of Equation C-49.

Solving for the instantaneous mortality rate of each length class requires (1) estimation of the duration of each length class, and (2) estimation of a

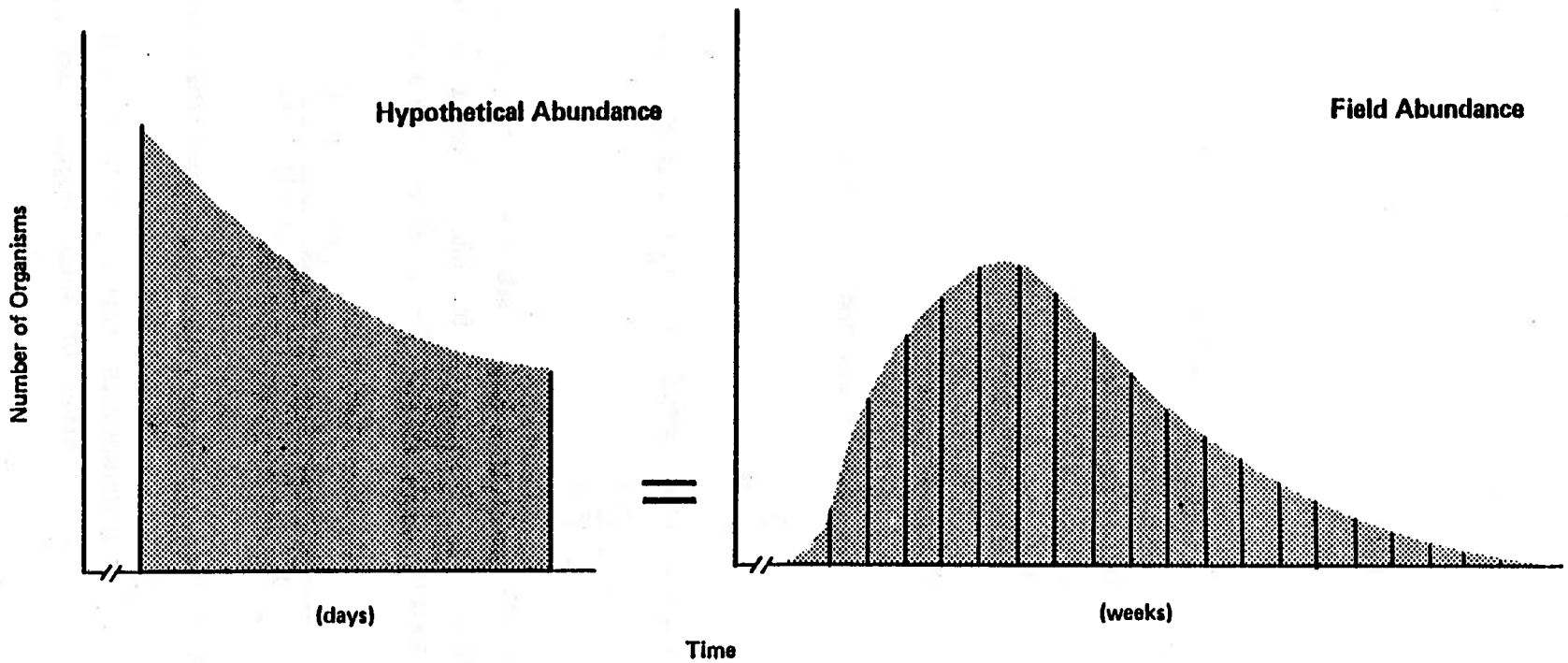


Figure C-10. Equivalence of the hypothetical, exponentially declining numbers of organisms in a length interval with the numbers collected in a field survey season.

proportionality constant defining the relationship between $\lambda^{(k)}$ and $\lambda^{(k+1)}$:

$$\lambda^{(k+1)} = r\lambda^{(k)}. \tag{C-50}$$

Once the values for these parameters have been determined, solving for the instantaneous mortality rates requires an iterative procedure, since there is no closed-form solution to Equation C-49. Such a procedure was developed for this purpose in the form of a BASIC program that estimated the value of $\lambda^{(k)}$.

The above method was used to compute estimates of the instantaneous mortality rate for all length classes except the last. The method is not applicable to the last length class because a proportionality constant between the mortality rates of the last and next-to-last length classes cannot be estimated without field abundance estimates for a length class beyond the last length class. The instantaneous mortality rate for the last length class was computed using Equation C-49 with a known value for $\lambda^{(k)}$, computed from the method involving proportionality constants.

Estimation Of Length-Class-Specific Durations $t^{(k)}$

The procedure used to estimate the duration of each length class (referred to as D in Section C. 3.1.3) for the egg and larval plankton surveys follows a method developed by Hackney and Webb (1978). The first step of this procedure was to compute an "average days of existence" for each length class, $\bar{t}^{(k)}$, defined as

$$\bar{t}^{(k)} = \frac{\sum_{i=1}^{s} A_i^{(k)} J_i}{\sum_{i=1}^{s} A_i^{(k)}} \tag{C-51}$$

where

i	=	the ordinal of a given sampling period
s	=	the number of sampling periods
$A_i^{(k)}$	=	abundance of length class k in sampling period i
J_i	=	julian number associated with sampling period i.

Values computed for $t^{(k)}$ were then regressed against the natural logarithm of length at the midpoint of each length class, ln L, using the model

$$\ln L = \ln a + b\bar{t} \qquad \text{(C-52)}$$

where

a = constant (intercept of y axis)

b = constant describing the rate of linear change between the natural logarithm of length and average days of existence

$\bar{t}$ = average days of existence.

Duration for a given length class was then computed by taking the difference between the estimated average day of existence betweeen the two endpoints of the length class as

$$t^{(k)} = \bar{t}(k2) - \bar{t}(k1) = \frac{\ln L2 - \ln L1}{b} \qquad \text{(C-53)}$$

where

$t^{(k)}$ = duration of length class k

$\bar{t}(k1)$ = estimated average day of existence for the upper endpoint of length class k1

$\bar{t}(k2)$ = estimated average day of existence for the upper endpoint of length class k2

L_1 = length of lower endpoint of length class k

L_2 = length of upper endpoint of length class k.

The duration of the length class, $t^{(k)}$, represents the average number of days that individuals of length class k were present in the system. If, for each length class, growth and mortality rates are constant throughout the system and between cohorts, the observed difference between average days of existence for the endpoints of a length class accurately reflect the duration of the length class. However, growth rates most likely do vary between cohorts and between spatially segregated subpopulations of the same cohort, because of variations in temperature and other physical parameters that

affect development. Under these circumstances, length class durations derived from Equation C-53 represent average values.

The above procedure for computing the duration of each length class was not used for the summer townet survey. When the data of the summer townet survey were used in the computation (Equation C-53), five 5-mm length class durations ranging from 1.8 to 3.5 days were predicted, values that represent unrealistic development rates. This is most likely due to the low level of resolution in predicting the average day of existence for each length class, an artifact of the small number of sampling periods characteristic of the summer townet survey. The number of sampling periods of the summer townet survey ranged from three to five, as opposed to the CDF&G egg and larval survey (up to 43 surveys) and 8 and 13 for the two EA 1978 and 1979 plankton surveys. The duration for each length class between 20 and 50 mm was set at a more realistic value of five days, on the basis of an analysis of the change in the average length of striped bass during summer townet survey sampling period.

Determination Of Proportionality Constants

The second step in preparing to calculate instantaneous mortality rates, calculating the proportionality constants that define the relationship between mortality rates of successive length classes (Equation C-50), was taken by calculating the ratio of computed approximations of the mortality rates:

$$r = \frac{\lambda'(k+1)}{\lambda'(k)} \qquad \text{(C-54)}$$

where

$\lambda'(k+1)$ = approximation to the instantaneous mortality rate of stage k+1
$\lambda'(k)$ = approximation to the instantaneous mortality rate of stage k
r = proportionality constant.

The approximations for the instantaneous mortality rate for each length class were computed in the following manner:

Since

$$\frac{N_0^{(k+1)}}{N_0^{(k)}} = S^{(k)} = e^{-\lambda^{(k)} t^{(k)}}, \tag{C-55}$$

$$\lambda^{(k)} = \frac{\ln N_0^{(k)} - \ln N_0^{(k+1)}}{t^{(k)}}. \tag{C-56}$$

Under the hypothesis of a constant mortality rate through time, leading to an exponentially declining population (Equation C-40), N_0 for each length class is computed as in Equation C-47. An alternative to this hypothesis is that all mortality occurs at the end of the length class, resulting in a uniform rather than an exponential age distribution. Although this hypothesis is less realistic than the exponential age distribution hypothesis, it can be used to calculate the approximate initial population size of each length class, $N_{0'}^{(k)}$:

$$N_{0'}^{(k)} = \frac{A^{(k)}}{t^{(k)}}. \tag{C-57}$$

When this is substituted in Equation C-56, it yields

$$\lambda^{(k)} = \frac{\ln \frac{A^{(k)}}{t^{(k)}} - \ln \frac{A^{(k+1)}}{t^{(k+1)}}}{t^{(k)}}. \tag{C-58}$$

C.3.2.2 Estimating Power Plant Mortality

Once values for the total instantaneous mortality rate have been estimated for each length class, the next step is to compute the instantaneous power plant mortality rate, $p^{(k)}$. For the sake of clarity, the superscript (k) will be dropped from the following development. In computing the instantaneous power plant mortality rate, a new variable was introduced, the expectation of power plant mortality, v, defined as the fraction of the length class population removed by the power plants, and computed as

$$v = (p/\lambda)m \qquad \text{(C-59)}$$

where

v = expectation of power plant mortality
m = finite mortality = $1 - e^{-\lambda t}$.

The expectation of power plant mortality defined in Equation C-59 is analogous to the exploitation rate used in fisheries management (Ricker 1975). The value of v was computed from the power plant cropping estimates as

$$v = c/N_0 \qquad \text{(C-60)}$$

where

c = number of organisms cropped as a result of entrainment at the power plant.

The variable c represents the total number of striped bass of a given length class that were cropped by the power plants during the field survey period. The methods used to estimate power plant cropping for the different surveys are discussed in Section 5.6.2.4.3

Solving for p in Equation C-59 yields

$$p = v\lambda/t. \qquad \text{(C-61)}$$

The value of p for each length class was then used to compute power plant survival as

$$Sp = e^{-pt} \qquad \text{(C-62)}$$

and applied to Equation C-39 to compute percentage reduction.

C.4 ESTIMATION OF HISTORICAL ENTRAINMENT MORTALITY PROPORTIONS AND COOLING WATER FLOWS

Sections 5.5 and 5.6 present the numbers of neomysis and striped bass cropped by

entrainment at the Pittsburg Power Plant during the period 1961-1979. The numbers cropped were estimated on the basis of (1) the densities of neomysis and striped bass in the vicinity of the intake structures, (2) the volumes of cooling water withdrawn from the source water body, and (3) the estimated proportions of entrained neomysis and striped bass not surviving. The means used to estimate the latter two items are the subject of this section.

The approach followed in estimating historical entrainment mortality proportions is summarized in Figure C-11. The figure shows the major steps in the approach. The upper half of the diagram shows the steps involved in estimating the discharge temperatures during the period 1961-1979. The lower half shows the steps involved in estimating entrainment mortality proportions on the basis of estimated discharge temperatures. The approach would have been more straightforward if discharge temperatures had been available for the entire historical period. Since they were not, a method had to be developed that would estimate discharge temperatures from intake temperatures, which were available, and plant generating data.

The rest of this section describes each step in the approach shown in Figure C-11. In Section C.4.1, monthly capacity factors (proportion of maximum possible output actually generated during each month) were calculated for each group of units sharing a common discharge at the Pittsburg Power Plant: Units 1-4, Unit 5, and Unit 6. The computed capacity factors of the unit groups for 1978 and 1979 were used as the independent variable in a simple linear regression model, developed in Sections C.4.2 and C.4.3 on the basis of intake temperatures, capacity factors, and temperatures discharge, which predicted the unit-group median monthly cooling water delta-T (temperature rise) during the period March 1978 - October 1979, the period for which discharge temperatures were available. The regression model was then used with monthly capacity factors in Section C.4.4 to estimate median monthly delta-Ts for each unit-group over the period 1961-1979.

The historical median delta-Ts were added to historical cooling water intake temperatures in Section C.4.5 to yield median monthly discharge temperatures. Next, the distribution of discharge temperatures around the median was estimated in Section C.4.6 on the basis of distributions observed during the period March 1978 - October 1979. The estimated distribution was used in Section C.4.7 to estimate the proportions of time during each month in the historical period that discharge

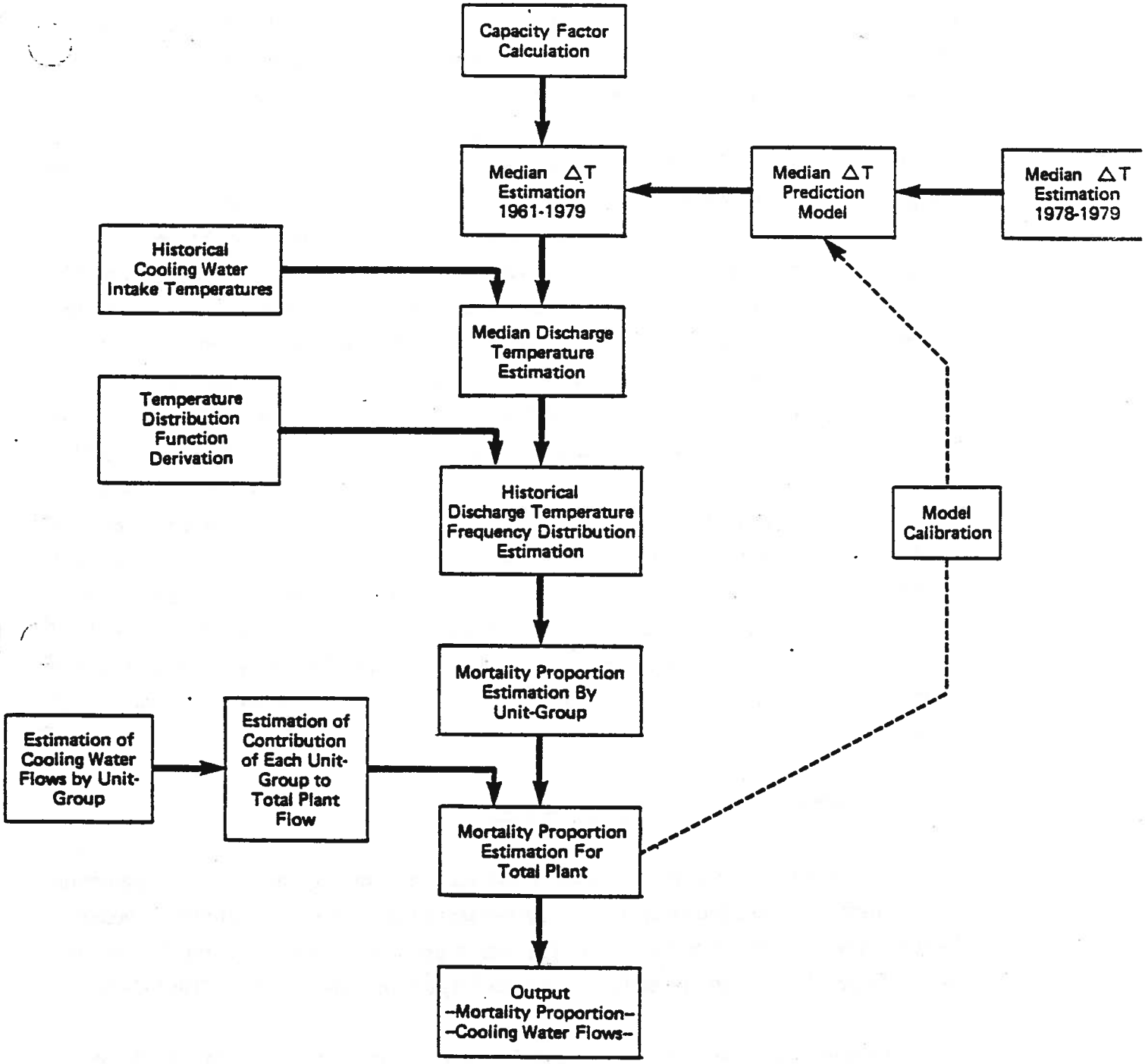


Figure C–11. Flow chart of approach used in estimating historical entrainment mortality proportions and cooling water flows at the Pittsburg Power Plant, 1961–1979.

temperatures were in each of four temperature ranges: <30 C, 30-31.9 C, 32-33.9 C, and ≥34 C. These temperature ranges are the ones for which entrainment mortality factors were estimated for N. mercedis and striped bass (Chapter 3).

The proportions of time that discharge temperatures were in each of the four temperature ranges were multiplied by the respective entrainment mortality factors, and the products summed in Section C.4.8 to estimate the monthly entrainment mortality proportions for each unit-group. Next, in order to estimate monthly entrainment mortality proportions for each plant (unit-groups combined), the contribution of each unit-group to the total monthly plant cooling water flow was estimated in Sections C.4.9 and C.4.10. Then, in Section C.4.11, the total monthly plant mortality proportions were computed by summing the products of the proportion of the total flow atttributed to each unit-group and the respective unit-group mortality proportions. The monthly mortality proportions so computed were compared in Section C.4.12 to mortality proportions estimated based on discharge temperatures actually measured during 1978 and 1979. The median delta-T regression model of Section C.4.3 was adjusted until the best agreement possible was reached between the two sets of entrainment mortality proportions. The final outputs, the monthly plant entrainment mortality proportions and cooling water volumes, have been used in Section 5.5 and 5.6 to estimate entrainment cropping of neomysis and striped.

C.4.1 Monthly Capacity Factor Calculation

The monthly capacity factor of a generating unit or group of units is the proportion of the maximum possible output actually generated during a given month. Capacity factors were calculated on the basis of monthly gross generation and turbine service hour data provided by the PGandE Operations Department for the years 1961-1979.

The gross generating capacity of Units 1-4 of the Pittsburg Power Plant is 660 MWe; of Unit 5, 330 MWe; and of Unit 6, 330 MWe.

Capacity factors (CF) were calculated as

$$CF = \frac{\text{gross generation}}{(\text{gross capacity}) \times (\text{hours of operation})} \qquad \text{(C-63)}$$

where

gross generation = total electrical output during the month (MWh)
gross capacity = total capacity in MWe.

For each unit whose cooling water discharge is separate from that of other units (Unit 5, Unit 6), the monthly total number of hours of turbine operation was used in Equation C-63. For units sharing a common discharge (Units 1-4), the total number of hours in the month was used in Equation C-63. In the former case, capacity factors were based on the level of operation during periods when the units were actually generating electricity; in the latter case they were based on the level of operation during the entire month (including outages). This method of computing capacity factors was found to best reflect the actual thermal output of the various cooling water systems.

Capacity factors were calculated for each unit-group for the months of April through July for the period 1961-1967 and for all months from January 1968 through December 1979. Capacity factors were not calculated for Unit 7 at the Pittsburg Power Plant because entrainment mortality proportions were not to be estimated for this unit. Since its cooling water system is closed-cycle, 100 percent entrainment mortality is assumed.

C.4.2 Median Delta-T Estimation (1978-1979)

Monthly median delta-Ts were estimated for all unit-groups at the Pittsburg Power Plant, except Unit 7, for the period March 1978 - October 1979. Delta-T estimates were based on intake and discharge temperatures continuously recorded during the study period. The difference between the intake and discharge temperature at each unit-group was caculated at 3-hour intervals, and monthly cumulative frequency distributions of delta-T's were generated. The 50th percentile or median of the cumulative frequency distribution was determined for each month.

C.4.3 Median Delta-T Prediction Model

The relationship between capacity factor (Section C.4.1) and median delta-T (Section C.4.2) was investigated using linear regression techniques. Capacity factor and delta-T data for the period March 1978 - October 1979 were used in the model (Table

C-14). A separate regression line was derived for each unit-group at each plant. Analysis of covariance techniques were then employed to determine whether or not a single line was adequate for all unit-groups combined.

The regression model used was

$$T = a + bCF + \epsilon \qquad \text{(C-64)}$$

where

a,b = regression coefficents
CF = capacity factor (percentage)
ϵ = random error.

Computed values of the regression coefficients for the individual unit-groups and the three unit-groups combined are:

	1-4	5	6	1-6
a	5.21	1.88	0.28	4.51
b	0.16	0.20	0.23	0.17

The slopes of the individual regression lines were found not to be significantly different ($\alpha = 0.05$), so the data for the three unit-groups were combined, and the single line that resulted had an r^2 of 0.79.

C.4.4 Historical Median Delta-T Estimation

Median delta-T's were estimated for each unit-group on a monthly basis during the period 1961-1979 using the delta-T regression model (Section C.4.3) and historical capacity factors (Section C.4.1).

C.4.5 Median Discharge Temperature Estimation

Historical median monthly discharge temperatures were estimated by adding the predicted median monthly delta-T's (Section C.4.4) to monthly average intake temperature provided by the PGandE Operations Department. Intake temperatures were measured at the inlets to the steam condensers of the Pittsburg Power Plant.

TABLE C-14 MONTHLY CAPACITY FACTORS AND MEDIAN DELTA-T FOR THE PITTSBURG POWER PLANT, MARCH 1978 - OCTOBER 1979

Month	Units 1-4		Unit 5		Unit 6	
	Capacity Factor	Median Temperature (C)	Capacity Factor	Median Temperature (C)	Capacity Factor	Median Temperature (C)
1978						
MAR	23.3	9	—	—	53.3	8
APR	9.3	5	—	—	48.6	7
MAY	19.3	7	—	—	46.3	7
JUN	53.0	12	—	—	70.7	16
JUL	56.7	13	60.4	15	67.9	14
AUG	49.4	15	74.1	15	70.6	18
SEP	17.6	11	56.8	14	55.6	13
OCT	62.8	18	76.4	19	77.7	19
NOV	61.6	18	80.1	18	85.0	19
DEC	60.3	17	88.1	20	88.2	20
1979						
JAN	59.4	16	82.6	19	84.4	20
FAB	67.3	13	79.9	18	79.5	19
MAR	65.8	14	72.7	15	—	—
APR	58.3	12	73.0	14	—	—
MAY	56.6	12	—	—	54.8	7
JUN	68.7	17	—	—	85.0	19
JUL	71.2	14	55.6	12	80.4	19
AUG	78.2	19	60.7	15	80.2	19
SEP	—	—	—	—	—	—
OCT	—	—	—	—	—	—

The measured temperature took into account any rise in intake temperature due to thermal recirculation.

C.4.6 Delta-T Distribution Function

Accurately estimating entrainment mortality proportions based on discharge temperature requires a knowledge of the distribution of discharge temperatures in a given month—mortality estimates based on the median discharge temperature alone would be biased low. The monthly distributions of delta-T's during the period March 1978 - October 1979 were examined in an attempt to specify a distribution function that would approximate the observed discharge temperature distributions at each unit-group.

The observed discharge temperature distributions were generally negatively skewed; that is, the lower tails were usually extended further from the median than the upper tails. An example, based on two sampling dates at the Contra Costa Power Plant, is shown in Figure C-12. The distribution of observed discharge temperatures could be approximated by a normal distribution whose mean was defined as the median of the observed distribution and whose standard deviation was defined, after some trial and error, as 2.5. The fit of the normal approximation at the higher delta-T's was good in most cases. Any lack of fit at the lower delta-T's was of minor consequence, because entrainment mortality was generally independent of temperature at low delta-T's. Therefore, despite the observed skewness, the normal approximation was used in predicting the distribution of delta-T's at each unit-group during the historical period.

C.4.7 Historical Discharge Temperature Frequency Distribution

The normal approximation to the delta-T frequency distribution was used in predicting delta-T's during the historical period. The predicted delta-T's were added to the mean monthly intake temperatures (Section C.4.5) to yield the distribution of discharge temperatures at each unit-group.

Next, the proportion of time during each month that discharge temperatures were in each of four temperature ranges (<30 C, 30-31.9 C, 32-33.9 C, and $\geq$34C) estimated from the predicted distribution of discharge temperatures. These four ranges are the ones for which entrainment mortality proportions were estimated for neomysis and striped bass on the basis of 1978-1979 entrainment survival sampling at the Pittsburg

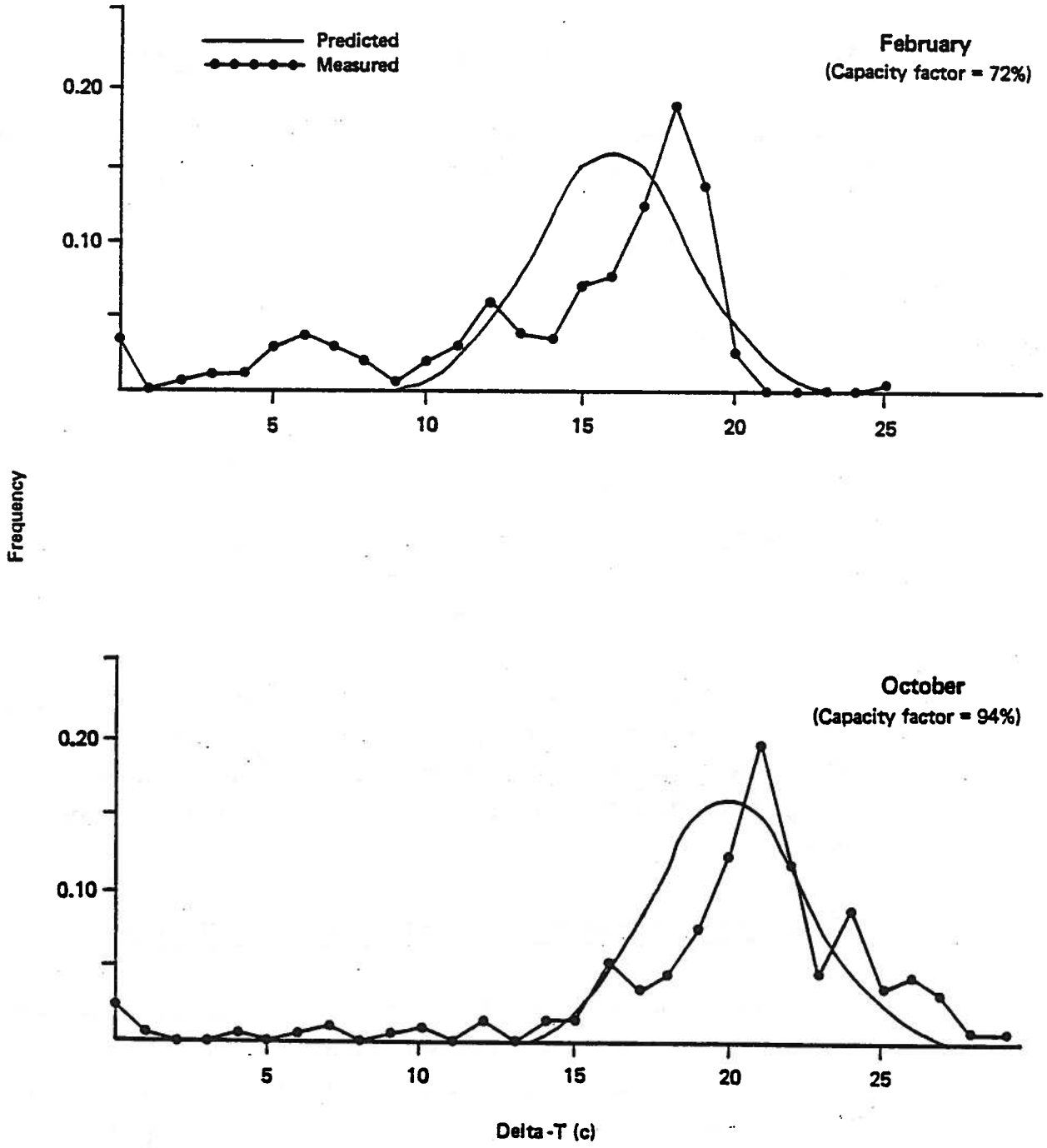


Figure C-12. Measured and predicted frequency analysis of delta-T at the Contra Costa Power Plant, 1979.

Power Plant. The time proportion was estimated for each discharge temperature range by computing the area under the normal curve corresponding to that range. The four proportions, so computed, were designated p_1, p_2, p_3, and p_4.

C.4.8 Historical Entrainment Mortality Factor Estimation By Unit-Group

The average monthly entrainment mortality factor (the proportion not surviving entrainment) for each taxon at each unit-group, P_e, was estimated as

$$P_e = p_1 P_{e1} + p_2 P_{e2} + p_3 P_{e3} + p_4 P_{e4} \qquad \text{(C-65)}$$

where

p_1, p_2, p_3, p_4 = proportions of time during the month that discharge temperatures were <30 C, 30-31.9 C, 32-33.9 C, and ≥34 C, respectively

$P_{e1}, P_{e2}, P_{e3}, P_{e4}$ = entrainment mortality proportions corresponding to discharge temperatures in the four ranges mentioned above.

The proportions p_1, p_2, p_3, and p_4 were derived in Section C.4.7, and the entrainment mortality proportions P_{e1}, P_{e2}, P_{e3}, and P_{e4} were estimated based on the results of 1978-1979 entrainment survival studies conducted at the Pittsburg Power Plant (Table C-15). (The entrainment mortality factor for Unit 7 was assumed to be 1.)

C.4.9 Estimation Of Historical Cooling Water Flows For Each Unit-Group

In Section C.4.11, monthly average entrainment mortality factors for each unit-group will be used to estimate total plant average monthly mortality factors. The latter are calculated as the weighted average of the individual mortality factors for each unit-group. This section and Section C.4.10 derive weighting factors to be used in Section C.4.11. The weighting factors are based on the contribution of each unit-group to the total plant monthly cooling water flow.

TABLE C-15 ENTRAINMENT MORTALITY PROPORTIONS FOR NEOMYSIS MERCEDIS AND STRIPED BASS IN VARIOUS DISCHARGE TEMPERATURE RANGES, PITTSBURG POWER PLANT, MARCH 1978 - MARCH 1979

Organism	Discharge Temperature	Proportion Not Surviving	
		Units 1-4	Units 5-6
Neomysis mercedis	30	—	0.10
	30-31.9	—	0.72
	32-33.9	—	1.00
	34	—	1.00
Striped bass	30	0.38	0.40
	30-31.9	0.69	0.57
	32-33.9	1.00	0.77
	34	1.00	(a)

(a) No data available. The proportion not surviving was assumed to be 1.00.

Note: (—) indicates no data available. Data from other unit groups at the plant were used in the analysis.

Monthly cooling water flows are derived in this section for each unit-group during the historical period. Because no records of cooling water flow were available, monthly turbine operating hours were used in place of monthly circulating water pump operating hours. Pumps are generally turned on approximately 12 hours before turbine generators begin operating, and are turned off approximately 12 hours after turbine generators stop operating; therefore, the numbers of turbine generator and circulating water pump operating hours are nearly equivalent.

Monthly cooling water flows were computed as

$$\text{Vol} = \frac{\text{Hours}}{\text{Number}}(3{,}600)(\text{PC}) \qquad \text{(C-66)}$$

where

Vol	=	monthly unit-group cooling water flow
Hours	=	total combined turbine service hours of all units in unit-group
Number	=	number of units in unit-group
PC	=	pump capacity (m^3/sec) of each unit-group
3,600	=	number of seconds per hour.

Pump capacities were 24.9 m^3/sec for Units 1-4, 10.1 m^3/sec for Unit 5, and 10.1 m^3/sec for Unit 6.

C.4.10 Calculation Of Total Plant Monthly And Daily Average Cooling Water Flows And Proportions Of Flow Carried By Each Unit-Group

Total plant monthly cooling water flow (Vol) was computed as

$$\text{Vol} = \sum_{i=1}^{n} \text{Vol}_i \qquad \text{(C-67)}$$

where

Vol_i = monthly cooling water flow for unit-group i (= "Vol" of Section C.4.9)

n = number of unit-groups.

Daily average cooling water flow (V_d) was computed as

$$V_d = \frac{Vol}{d} \tag{C-68}$$

where

d = number of days in month.

The proportion of flow carried by the ith unit-group (P_{V_i}) was computed as

$$P_{V_i} = \frac{Vol_i}{Vol} \tag{C-69}$$

C.4.11 Total Plant Monthly Entrainment Mortality Factors

Total plant monthly entrainment mortality factors were estimated for neomysis and striped bass for the historical period based on the proportions of the total cooling water flow carried by each unit-group (Section C.4.10) and the unit-group monthly entrainment mortality factors (Section C.4.8):

$$P_e = \sum_{i=1}^{n} P_{V_i} P_{e_i} \tag{C-70}$$

where

P_e = total plant monthly entrainment mortality factor
P_{e_i} = unit-group monthly entrainment mortality factor.

C.4.12 Calibration of the Model

The estimated total plant monthly entrainment mortality factors computed in Section C.4.11 for striped bass were compared to similar factors computed independently on the basis of discharge temperatures observed during the period April-July in 1978 and 1979 (Chapter 3). The observed mortality factors were computed by dividing the

estimated number of striped bass cropped at each unit-group by the estimated number entrained during several two-week periods. The periods were grouped by month, and the average monthly mortality factor for each unit-group was computed. The computed factors for all unit-groups at a given plant were then averaged in the same way as in Section C.4.11. The resulting total plant monthly entrainment mortality factors for 1978 ("observed"), and the corresponding factors predicted in the present study, were

	APR	MAY	JUN	JUL	AUG
Observed	0.39	0.41	0.56	0.60	0.76
Predicted	0.39	0.39	0.67	0.87	0.86

The values for 1979 were

	APR	MAY	JUN	JUL
Observed	—	0.43	0.72	0.69
Predicted	0.39	0.39	0.68	0.82

The predicted mortality factors for striped bass generally exceeded the observed factors during the summer months of both 1978 and 1979, but the difference was usually reasonably small.

The estimated monthly entrainment mortality factors for neomysis and striped bass and the estimated daily average cooling water flows for the Pittsburg and Contra Costa power plants for the period 1961-1979 are used in the impact assessment analysis for neomysis and striped bass presented in Sections 5.5 and 5.6.

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Appendix D

Review of Technologies for Thermal Power Plant Cooling Water Systems

D.1 Summary

D.2 Cooling Water Systems

D.3 Intake Location in the Source Water Body

D.4 Intake Structure Design Parameters

D.5 Intake Guidance and Diversion Technologies

D.6 Intake Maintenance and Operation

D.7 References

APPENDIX D: REVIEW OF TECHNOLOGIES FOR THERMAL POWER PLANT COOLING WATER SYSTEMS

As a consequence of enactment of Section 316(b) of the Clean Water Act (PL 92-500 and 95-217), there are a number of cooling water system intake technologies that are either currently in use or have been proposed for use in power plant cooling water systems in an effort to minimize the loss of aquatic organisms. These technologies can be generally categorized as involving (1) cooling system alternatives, (2) intake location in the source water body, (3) intake structure design parameters, (4) intake guidance and diversion technologies, or (5) intake maintenance and operation.

The loss of aquatic organisms during the operation of a cooling water intake structure results from entrainment, entrapment, and impingement. Entrainment is the pumping of organisms into and through a condenser cooling water system. Entrapment results from the physical blocking of larger entrained organisms by a barrier, generally some type of screen located within the intake structure. Impingement occurs when aquatic organisms are caught, trapped, or held against the screening device of a cooling water intake as a result of cooling water withdrawal from the source water body through the screening system (Ohio EPA 1978). One way to minimize the impact on aquatic organisms is to reduce the probability of their encountering the intake and thereby reduce the number of organisms entrained or impinged. A second approach is to reduce the mortality rate among those organisms which are entrained or impinged. The biological and engineering considerations involved in selecting and implementing alternative technologies to achieve these goals are discussed in this review.

The objectives and scope of this review are to provide a comprehensive and systematic listing of technologies for thermal power plant cooling water systems, and to discuss the strengths and weaknesses of each alternative.

An attempt has been made to identify intake designs that appear to be biologically effective in reducing organism losses while incorporating both demonstrated engineering feasibility and direct applicability to power plant intake systems. Intake designs that have not proven to be effective are also identified so that unsuccessful alternatives can be eliminated from the detailed site-specific intake evaluations. Information has been synthesized from published literature and reports, and from

discussions with plant operators, design engineers, utility and regulatory personnel, and equipment vendors.

D.1 SUMMARY

An overview of the recent information available on intake technologies and their effectiveness in minimizing the loss of aquatic organisms resulting from entrainment, entrapment, and impingement is presented. Factors contributing to the design of intake structures which reduce the number and mortality of impinged organisms are examined with emphasis on cooling water system designs, intake location in the source water body, and intake structure design parameters. Consideration is also given to the use of guidance, diversion, and fish return systems as well as operational strategies for reducing the biological impact of cooling water withdrawals.

D.1.1 Cooling Water Systems

Two alternative cooling systems are currently employed in thermal power plants: once-through and closed-cycle cooling. Closed-cycle cooling requires lower volumes of cooling water from the source water body and has therefore been suggested as a means of reducing impingement and entrainment rates. Use of closed-cycle cooling, however, results in 100 percent mortality of entrained organisms, in contrast to low mortality rates reported for many species entrained into once-through cooling systems. The effectiveness of closed-cycle cooling as an alternative to once-through cooling for reducing biological losses must be determined on a site-specific basis.

The operational efficiency of closed-cycle cooling towers is in part dependent on the water quality (e.g., sediment load and salinity) and may not be practical for use at many sites. Once-through cooling systems are limited to sites where a reliable water supply of sufficient volume is available.

D.1.2 Intake Location in the Source Water Body

Rivers, lakes, reservoirs, groundwater, estuaries, and oceans are used to supply cooling water to thermal power plants. The location of a cooling water intake structure within a water body may be along the shoreline, submerged offshore, or recessed from the shoreline. In order to minimize biological impact, the primary

factor in siting an intake structure is to avoid areas of high fish and invertebrate population densities, such as spawning and nursery areas and feeding grounds.

D.1.3 Intake Structure Design Parameters

Principal intake design parameters which influence biological effectiveness are the intake velocity; intake screen design, mesh size, and screening material; intake configuration and location; and the design of a fish return system. Intake velocity design criteria include consideration of approach velocity and through-screen velocity. Swimming performance capabilities of fish provide a biological basis for establishing approach velocity criteria. Survival of impinged organisms is one factor to be considered in establishing through-screen velocity criteria. Engineering and biological considerations for the design of intake screens and selection of screen mesh sizes and materials are discussed. Specific criteria for intake screen design must be developed on a site-specific basis. The configuration of the intake structure should be designed to minimize the entrapment of organisms. Provisions for returning entrapped or impinged organisms to the source water body are required to reduce biological losses associated with the intake structure.

D.1.4 Intake Guidance and Diversion Technologies

The applicability and effectiveness of various behavioral and physical barrier systems for use in reducing entrainment and impingement at cooling water intakes are discussed. Behavioral barriers are intended to cause a guidance or avoidance response to divert organisms into bypass channels. To date, behavioral barriers have not proven to be an effective solution to the problem of minimizing the impingement of fish or macroinvertebrates. Sound, lights, and bubble curtains sometimes actually increase fish attraction to an intake structure. Electrical barriers cannot be considered for application in marine or estuarine sites. Velocity gradients and water-jet deflectors cause sediment deposition or erosion, and also may trap organisms within intake structures. Louvers and velocity caps rely on specific behavioral responses and swimming performance capability to be effective, and hence are of little or no value in protecting planktonic fish eggs and larvae. Because of variability in response patterns between species and age classes of fish, behavioral barriers are considered to be appropriate only in cases where a single species of fish of relatively uniform size and age is of concern.

Physical barriers are designed to block the passage of entrained or entrapped organisms. Some physical barriers rely on impinging the organisms, which are then mechanically transferred to a bypass or return channel. Some physical barriers are designed to act as both a physical and behavioral barrier to exclude organisms from a cooling system. Recently, several physical barrier systems have been modified to include fine-mesh screens in an attempt to minimize the number of fish eggs and larval fish entrained into cooling water systems. The engineering reliability and operation as well as the biological effectiveness of various physical barrier systems are discussed. The application of alternative physical barriers for protecting larval as well as juvenile and adult fishes is considered. The demonstrated performance and operating experience with physical barrier systems used in association with once-through and closed-cycle cooling systems are considered.

The application and effectiveness of various fish conveyance and removal systems, including screenwash sluiceways, fish pump systems, fish basket collectors, and fish bypass - escape channels at intake structures are discussed. In specific applications, all of the fish conveyance and removal systems considered have proven to be effective. Factors such as intake structure configuration and design, the size and species of fish, and the behavioral response of the fish to various conditions encountered within an intake structure all affect the design and efficiency of a fish return system.

D.1.5 Intake Maintenance and Operation

Dredging to remove accumulated sediment from an intake is an effective means of ensuring that design velocities are maintained and uniformity of velocity gradients is maximized.

Reducing cooling water volume by limiting the operation of circulating water pumps reduces the rates of entrainment and impingement. Reduced cooling system operation results in losses in generation capacity and thermal transfer efficiency of the cooling system, and increased temperature differential through the condenser system (delta-T). Although a reduction in cooling water volume reduces the number of organisms entrained, the increased delta-T may substantially reduce the survival of entrained organisms.

Cooling system curtailment is an effective approach for eliminating entrainment and impingement. Cooling system curtailment, however, also precludes electrical generation. Curtailment for short time periods on a seasonal basis may be effective in reducing biological losses, although the costs of replacement power and reduced generation reliability throughout the service area must be considered.

Strategies for intermittent biofouling control using biocides such as chlorine are briefly discussed.

It was concluded, based on available data, that insufficient understanding of the role of shear forces, pressure regimes, and mechanical abrasion exists for entrained organisms to permit recommendations of structural modifications to cooling system components (e.g., circulating water pumps) to reduce entrainment stresses.

Regulation of thermal elevations (delta-T) was examined. It was concluded that site- and species-specific information on the relative importance of thermal stress and mechanical stress in influencing survival of entrained organisms is needed to establish an effective management strategy. The process of determining the optimal cooling system operating strategy is briefly discussed.

D.2 COOLING WATER SYSTEMS

There are two basic power plant cooling system designs—once-through (open-cycle) and closed-cycle cooling. In once-through systems, water is pumped directly from the source water body to the condenser and is then discharged into the receiving water body. In closed-cycle cooling systems, the cooling water is recirculated between the condenser and a cooling tower, or cooling pond. In an evaporative closed-cycle cooling system, water is withdrawn from the source water body to provide makeup water during operation. The selection and design of a thermal power plant cooling system is dependent upon a number of factors, including water availability, engineering constraints, and regulatory limitations.

D.2.1 Open-Cycle or Once-Through Cooling

Once-through cooling may be considered to be feasible at power plant sites if a reliable water supply of sufficient volume is available. Variations in water quality

and suspended sediments do not generally influence the acceptability of a power plant site for use with a once-through cooling system.

D.2.2 Closed-Cycle Cooling

Closed-cycle cooling (cooling towers) decreases the volume of water withdrawn from a source water body and thus reduces the numbers of phytoplankton, zooplankton, and ichthyoplankton entrained. However, for organisms that are entrained, 100 percent mortality results from repeated exposure to increased temperatures, biocides, and mechanical abrasion. Therefore, the effectiveness of closed-cycle cooling in reducing entrainment and impingement losses as an alternative to once-through cooling must be determined on a site-specific basis.

Evaporative closed-cycle systems have had widespread freshwater application in the United States where water availability has been limited.

A survey of operating experience with saltwater closed-cycled cooling systems prepared for The Utility Water Act Group (Stone & Webster Engineering Corporation 1978) concluded that there are no known operating or proposed electric generating facilities in the United States which use seawater (salinity 26-36 ppt) makeup in closed-cycle cooling systems. The only power generating facility in the world which uses seawater in cooling towers is the Fleetwood Power Station in England, a small peaking plant (three 32-MWe units) with a low capacity factor (Stone & Webster Engineering Corporation 1978). The units are cooled by two natural draft cooling towers with a circulating water flow rate of 67 cfs (1.9 m^3/sec) per unit. Salinity of the cooling water makeup at the Fleetwood Power Station typically ranges from 26 to 27 ppt. The circulating water is normally concentrated 1.2-1.5 times. Operational problems associated with corrosion and material deterioration have been reported.

There are approximately 25 power plants in the United States, ranging in capacity from 10 to 1,510 MWe and with circulating water rates of 4,900-540,000 gpm (11-1200 cfs; 0.3-34 m^3/sec), that use brackish water (0.5-26 ppt) makeup in evaporative closed-cycle cooling systems. A survey of operating experience at these plants indicated several areas where operational difficulties occurred or could be attributed to the use of brackish water: (1) degradation of materials of the cooling

tower, spray modules, or other plant structures, (2) switchyard and/or transmission line arcing, and (3) off-site property effects. The magnitude of these problems varied widely, from corrosion of handrails to forced outages of the units. Corrections ranged from replacement of the complete cooling system to conversion to freshwater makeup. In addition, although the difficulty could not be attributed solely to the use of brackish water, nonattainment of design thermal performance was reported in a number of cases.

In addition, a number of other aspects must be considered in the early stages of power plant planning. These include air quality impacts from cooling tower drift, terrestrial effects of salt deposition, land use requirements, aesthetic considerations, and economic factors. Stone & Webster's review (1978) showed that no seawater closed-cycle unit has ever operated anywhere in the world for units of the size presently being constructed or planned by the utility industry (i.e., 700 MWe fossil or 1,150 MWe nuclear).

Dry closed-water-loop cooling systems are attractive as water-saving devices. They are technically feasible, but are costly and generally offer poorer plant performance than evaporative systems. The California Energy Commission reports that, "effective heat transfer in dry cooling towers requires significantly more heat exchanger surface than in wet cooling systems (closed-cycle). Construction costs are therefore much higher. Although dry cooling systems use essentially no water, they work less effectively the higher the air temperature" (Stamets 1977).

A combination of wet and dry closed-cycle cooling is technologically feasible. Public Service Company of New Mexico faced a limited water allotment and has installed a wet/dry cooling tower of new design on its 500 MWe San Juan Unit 3. The plant is soon due to become operational. Cooling will be 70 percent dry (air) and 30 percent wet, resulting in roughly a 1,200 gpm (2.7 cfs; 0.075 m^3/sec) maximum makeup rate compared to a wet tower rate of 4,400 gpm (9.8 cfs; 0.28 m^3/sec) (Rittenhouse 1979). The value of the savings in water must be considered in relation to the relatively high capital costs of the wet/dry cooling tower as opposed to wet cooling towers.

D.2.3 Cooling System Retrofits

The feasibility of retrofitting the once-through cooling system at an existing power

plant with a closed-cycle system would require extensive investigation. Since the cooling system is a major and integral part of the initial plant design, its proposed change would require careful examination of overlapping operational, environmental, and economic considerations, and could be more complex than the design of a new plant.

Operational considerations would include an evaluation of thermal performance, operating requirements, service life of system components, electrical conductor contamination by cooling tower deposition, and plant reliability.

Environmental considerations with closed-cycle cooling would center on air quality effects of cooling tower drift, terrestrial effects from salt deposition, land use requirements, and aesthetics.

Economic factors would include an evaluation of the capital costs for cooling system conversion, plant performance penalties, increased operation and maintenance costs, higher site-related costs, and costs associated with the effect on system reliability of reduced generating capacity of the modified unit(s).

D.3 INTAKE LOCATION IN THE SOURCE WATER BODY

Rivers, lakes, reservoirs, groundwater, estuaries, and oceans are used to supply cooling water to thermal power plants in the United States. Each of these source water bodies has characteristics which affect the location, design, and operation of intake structures. Site selection involves consideration of the potential environmental impact and the location of suitable water bodies to provide cooling water. Biological information on potential impacts provides one input into a multidisciplinary site evaluation and choice of intake location.

Selecting the location of a cooling water intake structure to avoid areas of high fish and invertebrate population densities is desirable, and involves consideration of both the type of water body (i.e., lake, river, estuary, open ocean) and the location of the intake structure within the water body (onshore or offshore). The most favorable intake locations are those in areas considered to be of low biological value (U.S. EPA 1977). Few, if any intake locations are completely free of entrainable and impingeable organisms, but those near migratory routes, spawning areas, nursery

areas, or feeding grounds should be avoided whenever possible.

The location of a cooling water intake in a water body may assume one of three basic orientations with respect to the shoreline: (1) a shoreline intake structure where the intake circulating pump well and screens are combined into a single structure, (2) an offshore intake where the inlet and screenwell-pump well are separated by some distance and connected through an enclosed conduit, and (3) an approach channel intake (recessed intake) where cooling water is diverted from the source water body to the intake screens by a conveyance channel. All three basic intake locations are currently used in the United States.

D.3.1 Shoreline Intake Structures

Shoreline intake structures similar to that shown in Figure D-1 are commonly used to supply cooling water to power plants located on lakes, rivers, estuaries, and bays. Locating a shoreline intake structure on a water body requires consideration of the vertical and horizontal distributions of aquatic organisms to avoid siting the intake in an area of high biological productivity. Consideration of the hydrology of the area is important to minimize recirculation and allow for variations in water level. Problems of sediment deposition and debris accumulation along shorelines also have to be considered.

General recommendations for the location of shoreline intake structures may be summarized as follows:

1. avoid locating the intake in highly productive or high population density areas for fish and shellfish, or where there is susceptibility to debris or sediment accumulation; and
2. locate the intake in an area that will minimize the recirculation of cooling water, entrained organisms, and debris.

D.3.2 Offshore Intake Structures

Offshore intake structures similar to that shown in Figure D-2 are primarily used to supply cooling water to power plants located on large lakes and on beach zones along

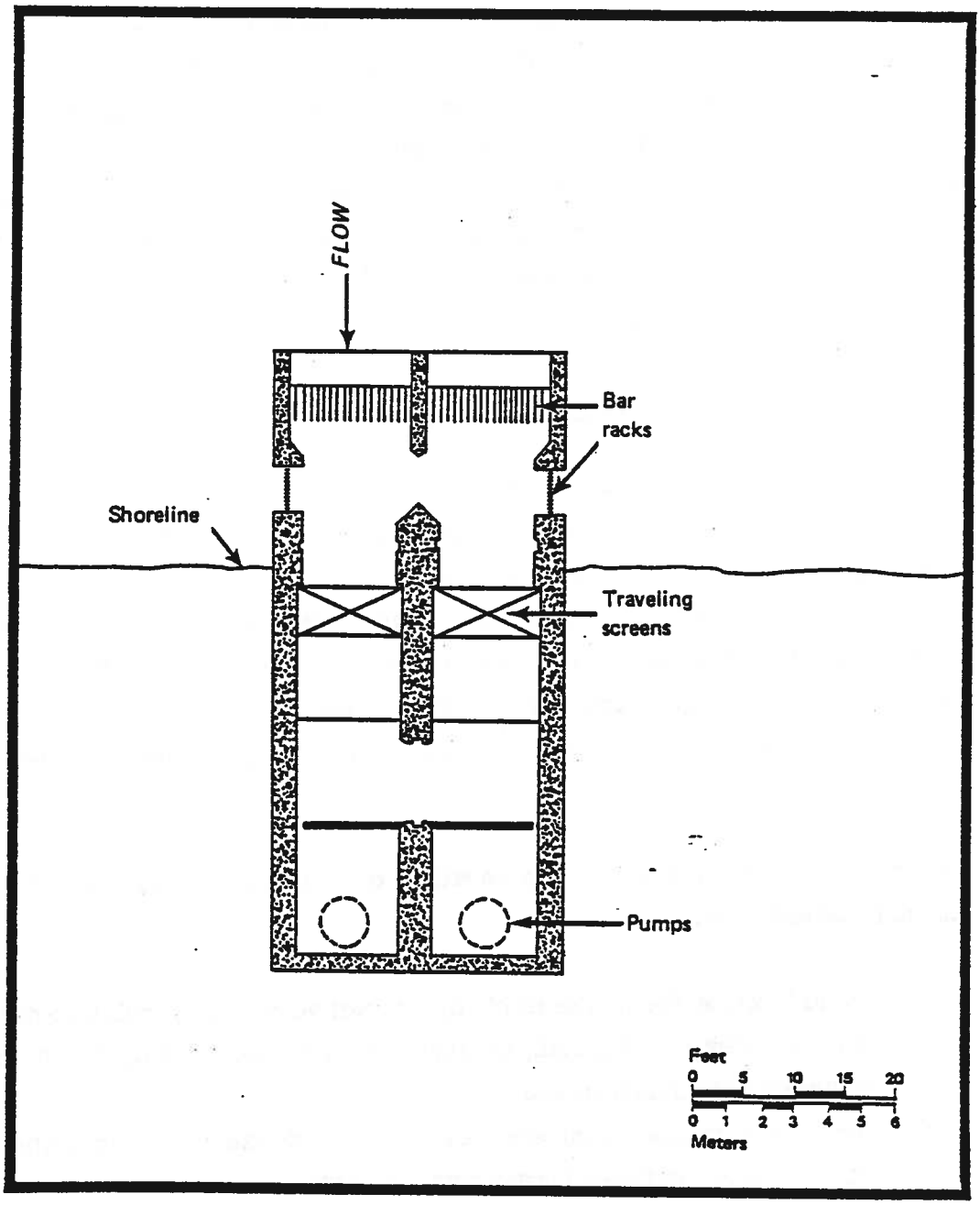

Figure D-1. Typical shoreline intake configuration.

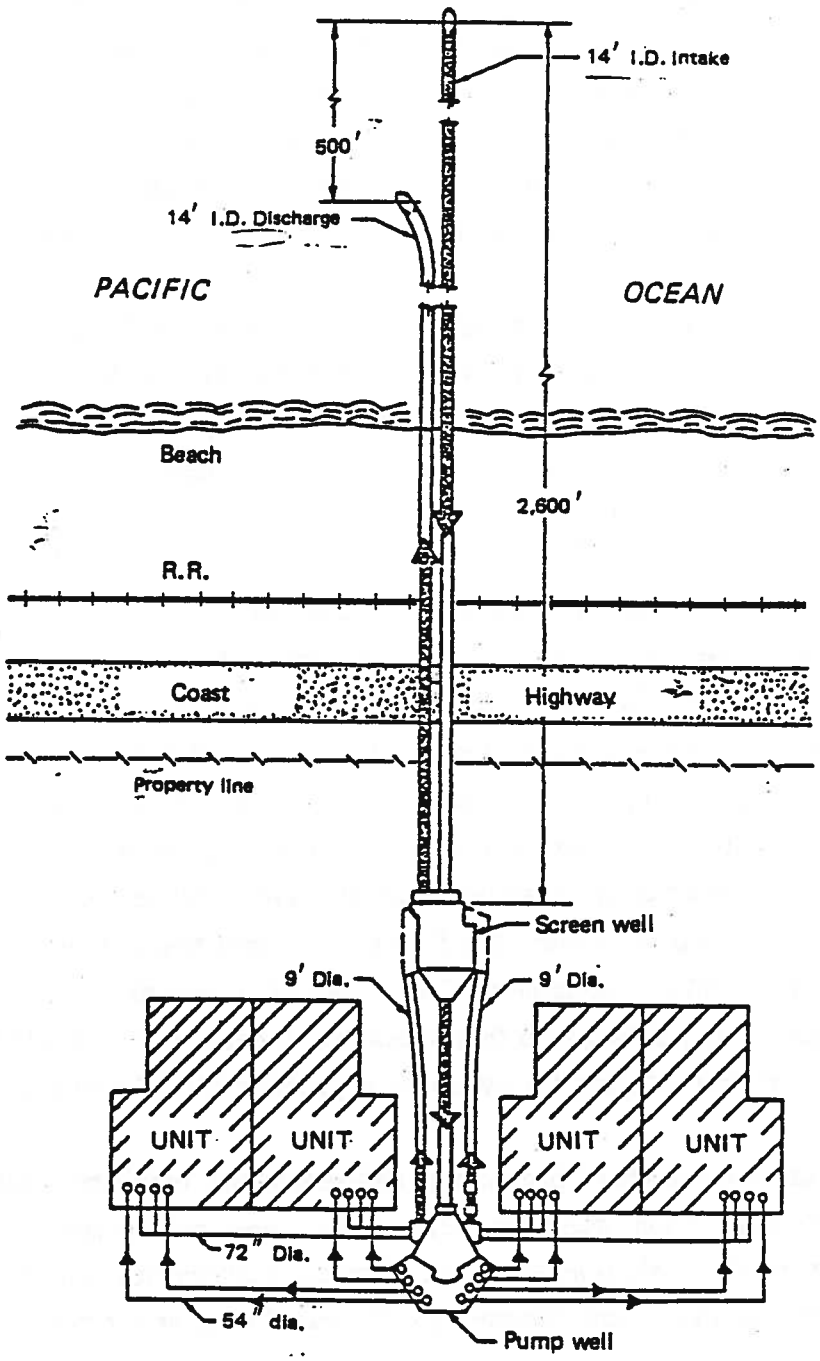

Figure D-2. Plan view of an offshore intake. (From Sonnichsen 1975.)

the coasts of oceans. Offshore intakes are used primarily to allow the selective removal of cooler subsurface water in water bodies that are vertically stratified, to isolate the cooling water intake from the discharge to minimize thermal recirculation, to avoid disruption of lateral drift of beach sand, and to avoid shallow nearshore areas where the topography prohibits use of a shoreline intake structure. Offshore intake structures provide a greater degree of flexibility in selecting an intake location than is available for a shoreline intake, but the cost of installing an offshore intake is substantially higher than that for a comparable shoreline structure.

Locating an offshore intake structure within a water body requires consideration of the spatial distribution of aquatic organisms, vertical stratification of water temperature, and the hydrology of the area. Problems associated with sedimentation, biofouling control, fish removal and return, and obstruction of navigable waters must be addressed in the design and location of offshore intakes.

The arrangement of offshore intake structures includes an offshore point of water withdrawal, an enclosed conduit used to convey the cooling water to the power plant, and an onshore screenwell. This design results in a zone of fish entrapment between the offshore entrance to the intake structure and the intake screens. Attempts to alleviate the entrapment of fish within the intake have focused on preventing the entry of fish into the intake and on developing removal systems to return entrapped fish to the water body. Methods used to prevent fish entry into the intake include use of a velocity cap to reorient the flow pattern into the intake entrance from a vertical intake flow into a horizontal flow. Use of a velocity cap on the offshore intake structure of the El Segundo Power Station, located in the Pacific Ocean, reduced the tonnage of fish entering the intake by approximately 95 percent (Weight 1958).

Alternative methods of preventing fish entry into offshore intake structures include use of infiltration galleries, stationary fine-mesh screens, or traveling screens (caisson intake design) at the point of water withdrawal. These alternatives, and also fish removal and return systems, are discussed in later sections of the review.

General recommendations for the location of offshore intake structures are summarized as follows:

1. avoid locating the intake in areas which are highly productive or have high population densities of fish and shellfish;

2. locate the intake in an area which will minimize the recirculation of cooling water, entrained organisms, and debris;
3. base design intake velocities on the swimming performance capabilties and behavioral response of resident and migratory fishes as well as engineering constraints;
4. include provisions for the escape or removal of organisms from the intake structure; and
5. locate the intake where it will not impede navigation.

D.3.3 Approach Channel Intake Structures

Approach channel intake structures (recessed intakes) are used to provide a protected area for intake screens, to separate the intake and discharge locations to minimize thermal recirculation, or for other engineering reasons.

Such a system generally consists of an approach channel conveying water some distance from the shoreline to an intake structure, in either an enclosed conduit or an open canal. Fish can become trapped in the confined conduit or canal and may become stressed or fatigued, and eventually impinged.

Open approach channels generally have lower water velocities than do enclosed intake conduits. The open channel configuration also offers the possibility of greater behavioral avoidance and reduced entrapment of organisms. In general, the design of intake structures which utilize confined or enclosed conduits should be avoided in proposed installations whenever an alternative intake configuration can be used.

D.4 INTAKE STRUCTURE DESIGN PARAMETERS

The principal design parameters of an intake structure that influence organisms losses are (1) intake velocities, (2) intake screen design, mesh size, and screen material, (3) intake configuration, and (4) fish return system. A brief discussion of intake velocity and intake screen design follows. A discussion of intake configuration is presented in Section D.3. Fish return systems are described in Section D.5.3.

D.4.1 Intake Velocities

Intake velocities at two locations have received the widest biological attention: the approach velocity of water entering the intake (generally in front of the bar rack structure), and the through-screen velocity. To reduce impingement losses, the approach velocities should be based, in part, on the swimming performance of fish species present in the vicinity of the intake. Through-screen velocity should be based, in part, on the relationship between through-screen velocity and survival of impinged organisms.

D.4.1.1 Intake Approach Velocity

The development and evaluation of design criteria for intake structures based on swimming performance requires consideration of several factors. First, those species of fish to be protected are generally not the same from one geographical region to another. Second, certain species of fish are apparently more susceptible to entrainment and subsequent impingement than others. In addition, the susceptibility of a species to entrainment, entrapment, or impingement varies for each life stage. Third, key environmental variables such as water temperature, salinity, illumination, and dissolved oxygen concentration may alter the ability of a species to avoid entrapment and impingement. Fourth, the presence of a predator species may alter the behavioral pattern of individual fish or schools of fish and consequently increase their susceptibility to entrapment (Landry and Strawn 1974).

Extensive studies of fish swimming performance capabilities have been done on a variety of species by a number of researchers. It is difficult, however, to correlate their observations and establish a realistic approach velocity design criterion for swimming ability, because of variations, inconsistencies, or omissions in reporting data, pretest conditions, experimental design, physiological conditions, and environmental considerations.

Approach velocity design criteria typically range from 0.2 to 0.8 fps (6–24 cm/sec) for shoreline intake structures and 1.5 to 2.0 fps (46–61 cm/sec) for offshore velocity cap intake structures. An intake structure design should allow for an essentially uniform velocity profile across the entrance to the intake. In addition to biological

considerations, the approach velocity for any intake structure is dependent on engineering constraints associated with the intake design and configuration, hydrology, and water quality of the source water body—all of which must be determined on a site-specific basis.

Dredging to remove sediment deposits from an existing intake structure will result in reduced approach velocities. Routine maintenance of the bar racks and intake entrance area to remove accumulated debris and obstructions is important to assure that intake operation is within the range of design parameters established for the specific intake structure.

D.4.1.2 Through-Screen Velocity

The probability of impingement mortality increases with increasing impingement duration and with higher water velocity, which hinder respiration by water pressure, a factor especially important for previously fatigued fish. Abrasion by the screen surface also results in higher impingement mortality associated with higher through-screen water velocities.

Biological studies on the survival of impinged fish have been conducted by Bibko et al. (1974), Prentice and Ossiander (1974), Landry and Strawn (1974), King et al. (1978), and many others (see Oak Ridge National Laboratory and Atomic Industrial Forum, Inc. 1979 for an extensive review). Findings of these studies indicate that, in addition to being affected by through-screen velocity, impingement survival is a function of the species, size, and life stage of impinged organisms; the season; spraywash pressure; and the duration of impingement. Hemorrhaging, descaling, and mechanical abrasion are important factors contributing to impingement mortality. As a result of the number of interacting factors which influence impingement mortality, it is difficult to isolate the contribution of through-screen velocity.

Through-screen velocities at intakes of U.S. utilities typically range from 0.5 to 1.5 fps (15-46 cm/sec) for conventional vertical traveling screen intake structures. As the available open area for water to pass through the intake screen mesh decreases, the through-screen velocity increases. Establishing appropriate through-screen velocity and impingement duration relationships is essential if a fine-mesh intake screen is being considered to remove fish eggs and larval fish from the cooling water

(Tomljanovich et al. 1978). Data necessary to establish through-screen velocities based on impingement survival are presently not available for most species of fish. Data which are available on the relationship between impingement survival, duration of impingement, and through-screen velocity are, in many cases, inconclusive and do not provide a reliable basis for establishing biological design criteria for through-screen velocity for a broad variety of fish species and life stages.

D.4.2 Intake Screen Design, Mesh Size, and Screen Material

A variety of intake screen designs are commercially available for use in power plant cooling water intake structures (see Section D.5.2 for a review of intake screening devices). The major engineering and biological design requirements for an intake screen include:

1. the intake screens must be of sufficient size to accommodate the coolant flow while maintaining through-screen velocities within the design parameter limits;
2. the intake screen must efficiently remove debris which would otherwise clog the condenser system;
3. the intake screen must provide sustained reliable operation to ensure an adequate supply of cooling water; and
4. the intake screen should be designed to minimize mechanical effects resulting in injury or death of organisms from abrasion or suffocation during impingement on screen surfaces.

Design parameters for an intake screen to be utilized at a specific site require knowledge of the power plant cooling system design, characteristics of the source water body (including sediment and debris loads), and the species and size of organisms to be protected. As a result of the variety of conditions which influence intake screen design and the variety of intake structures available, no single intake screen design can be identified that would be the best available design for all sites.

Intake screen mesh sizing has historically been based on a design criterion which specifies that screen mesh openings should be approximately 50 percent of the condenser tube diameter. As a result of this criterion, the majority of power plant cooling systems, which are designed with 7/8-in. (22.2-mm) diameter condenser tubes,

have 3/8-in. (9.5-mm) square woven-wire mesh intake screens. Recently, increased emphasis has been placed on determining screen mesh sizes based on the size of organisms to be eliminated from entrainment through the cooling water system. The primary criterion in selecting an intake screen mesh size for biological reasons is to determine if the particular organisms and life stage in question is better able to withstand impingement stresses than the stresses associated with entrainment through the cooling system.

A summary of the retention of various species of fish larvae by screen meshes of various sizes is presented in Figure D-3. Screen mesh with 0.02-in. (0.5-mm) openings is available commercially and will retain the majority of egg and larval fishes present in most source water bodies. However, before selecting a specific mesh size for excluding organisms from a cooling system, it is important to consider the relative survival of impinged versus entrained individuals. Tomljanovich et al. (1978) present a detailed discussion of experimental studies conducted under laboratory conditions to determine the feasibility of protecting larval fish at water intakes using fine-mesh screening. Additional considerations in selecting a screen mesh size include debris loading and screen clogging, available open area and through-screen velocity, and screen reliability under site-specific operating conditions.

The variety of screen mesh configurations currently available for use in cooling water intakes includes square mesh woven-wire, horizontal slot configurations using woven wire, and horizontal slot configurations using a welded wedge-wire design. Each of these mesh configurations is available in a variety of mesh sizes. Hydrodynamic and biological investigations are ongoing (e.g., Cook 1978; Heuer and Tomljanovich 1978; Walker and Kamata 1979) to determine the advantages and disadvantages of various intake mesh configurations. Insufficient data are currently available to conclude that one intake screen mesh configuration significantly reduces biological losses, debris loading, and hydraulic losses in comparison to other available configurations.

Intake screens are available in a variety of materials, including polyester, nylon, Monel, and stainless steel. The selection of a screening material is dependent on the mesh size and configuration selected and on the desired durability of the screen. Polyester and nylon screen is used in applications where a fine-mesh screen (0.02-0.06 in.; 0.05-1.5 mm) is desired and when screen mesh is expected to be altered seasonally. Stainless steel is used in applications where durability over a long period

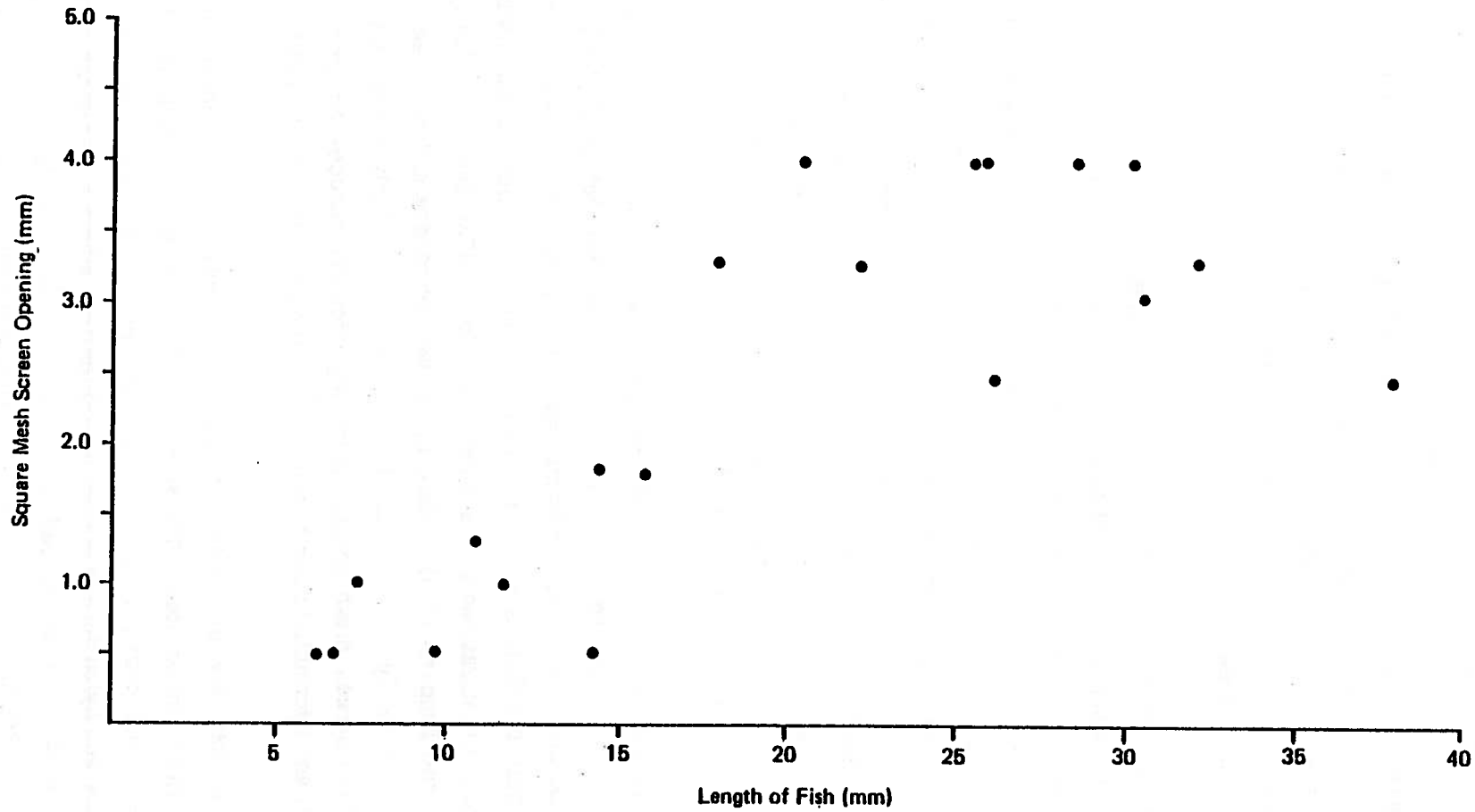

Figure D-3. Relationship between square mesh screen opening size required for 90 percent retention and length, for 12 species of fish. (Data from Erkkila et al. 1950; Kerr 1953; Clay 1961; Prentice and Ossiander 1974; Fisher 1978; Tomljanovich et al. 1978.)

of time in a corrosive environment such as salt water is needed. Monel is used extensively in both fresh- and saltwater environments, but requires greater maintenance and more frequent replacement than stainless steel. In addition to durability, capital cost and replacement cost are considered in the selection of screening material.

No documentation exists that the type of screen material influences the survival of impinged organisms.

D.4.3 Other Intake Design Parameters

In the early 1950s PGandE and Bechtel Corporation undertook a study in cooperation with the California Department of Fish and Game to minimize intake entrapment and impingement losses (Kerr 1953). As direct outgrowth of the study, PGandE established design criteria that have since been incorporated in all subsequent intake structures.

A typical shoreline intake configuration can be seen in Figure D-1. Basically, the structure meets a number of hydraulic and mechanical criteria in addition to reducing entrapment and impingement. Fish protection criteria identify preferences for location in the water body (previously discussed in Section D.3) and provide for lateral fish escapes between the bar racks and traveling screens. Low approach velocities, shore cooling water system transit times, and minimization of turbulence are specified.

D.5 INTAKE GUIDANCE AND DIVERSION TECHNOLOGIES

D.5.1 Behavioral Barriers

Various types of stimuli, such as light, sound, velocity gradients, electric shock, and chemicals, have been tested in an attempt to frighten or deter fish from entering cooling water intakes. Louvers and velocity caps have been used to provide stimuli for guiding fish away from a water intake. Because these behavioral barriers rely on the ability of an organism to respond to a stimulus, they are of little or no value in protecting planktonic fish eggs and larvae. In addition, the behavioral response of fish to artificial stimuli varies among species of fish and also within a species

because of differences in age or physiological state (Sonnichsen 1975). Because of the variability in responses to a stimulus, behavioral barriers are considered to be most appropriate in those cases where a single species of fish of relatively uniform size and age is of concern.

The primary application of most behavioral barriers has been in addition to, or in combination with, conventional intake systems (physical barriers) on either an intermittent or continuous basis. Many behavioral barriers have not been subjected to rigorous testing under field conditions, so conclusions as to their effectiveness have been drawn, where possible, from laboratory studies. Qualitative conclusions are presented in Table D-1.

D.5.1.1 Light

Intense illumination was characterized by Bibko et al. (1974) as a passive deterrent for striped bass, only temporarily deterring fish passage under experimental conditions. Fields (1966) reported that the avoidance response of salmonids to artificial light varied, depending on the light adaptation of the species, water clarity, and flow conditions. Constant light was reported by Fields to be more effective than interrupted or flashing light for guiding young salmon. Adaptation to light intensity substantially decreases the guidance efficiency of light barriers.

Available data on the effectiveness of light as a behavioral barrier are contradictory. In some tests, fish are attracted to light, while in other tests fish are repelled by it. In addition, light would not provide an effective barrier during the daytime or in highly turbid water. Thus, light barriers are not considered to be an effective method of reducing the numbers of organisms entrained or impinged at an intake structure.

D.5.1.2 Sound

Sonic techniques for repelling fish from industrial water intake structures have been reported by Moore and Newman (1956), Burner and Moore (1967), VanDerwalker (1964), and Trefethen (1968). Moulton and Backus (1955) reviewed the literature regarding the guidance efficiency of sonic barriers.

TABLE D-1 SUMMARY OF EXPERIENCE WITH BEHAVIORAL BARRIERS FOR REDUCING ORGANISM LOSSES AT COOLING WATER INTAKES

Technology	Reduces Larval Entrainment	Reduces Impingement	Reduces Entrapment	Operationally Reliable	Comments
Light	○	?	○	●	Attracts some species
Sound	○	○	○	●	After initial avoidance, fish acclimate to sound
Velocity gradient	○	○	○	○	Requires relatively high velocities, stable water level
Bubble screen	○	?	○	●	Attracts some species at night
Electrical barrier	○	?	○	?	Applicable to fresh water only; can induce fish mortality
Chemicals	○	○	○	○	Discharge of chemicals into receiving waters; undemonstrated
Magnetic barrier	○	○	○	○	Undemonstrated
Cables and chains	○	○	○	○	Undemonstrated
Louvers	○	◒	○	◒	Lack automatic debris-cleaning capability
Velocity cap	○	●	●	●	Applicable to offshore intakes only

Legend

● Demonstrated effective

◒ Demonstrated somewhat effective

? Data inconclusive, conflicting or variable

○ Demonstrated ineffective or no data

In general, maximum avoidance response has been observed for low-frequency, high intensity sound, but variation in hearing ability among species and high-level background noise leads to poor repeatability of the behavioral response of fish to sonic barriers. Burner and Moore (1962) reported that the behavioral response to sound frequency or intensity was insufficient to be effective in guiding young salmonids to safe passage around dams and diversions. The Virginia Electric and Power Company (Vepco) attempted to use relatively high-intensity, multifrequency sound to repel fish from power plant intakes (J. White, Vepco, 1976, personal communication). It was concluded that although sound was partially effective, sound alone was inadequate for repelling fish from the cooling water intake, because of the diversity of species, the diversity of sizes if the fish encountered, and the diversity of behavioral patterns.

An underwater pneumatic impact device (popper) was tested as a sonic barrier at a power plant intake structure by Schuler and Larson (1975). It was concluded that the effective range of sonic barriers is limited and that fish quickly become accustomed to a sound stimulus and no longer show an avoidance response.

Thus, although fish are initially frightened by sound barriers, tests conducted under laboratory conditions and at power plant intakes indicate that they rapidly become accustomed to sound stimuli, rendering the barrier ineffective. On the basis of negative results from field and laboratory studies, no further testing of sonic barriers at power plants intakes is warranted.

D.5.1.3 Velocity Gradients

Kerr (1953), Clay (1961), Bates (1964) Bates and VanDerwalker (1964), Niggol (1964), and Prentice (1966) discussed flow acceleration or deceleration barriers for guiding or deflecting fish. Flow acceleration barriers produce an increase in the approach velocity over a relatively short distance by use of wedges in approach channels. Bates and VanDerwalker (1964) reported a 70 percent diversion efficiency for an experimental water-jet deflector at an approach velocity of 2.5 fps (76 cm/sec). High diversion efficiencies (81 percent) have been reported by Prentice and Ossiander (1974) for vertical flow accelerators oriented at a 20° deflection angle for channel velocities ranging from 1.2 to 2.4 fps (37-73 cm/sec). Horizontal flow accelerators

had an average deflection efficiency of 56 percent for channel velocities from 1.5 to 2.3 fps (46-70 cm/sec) (Prentice and Ossiander 1974). No difference in diversion efficiency was observed for tests conducted during the day and night. Blinded fish, tested by Gerold and Niggol (1964), were guided by flow acceleration barriers with diversion efficiencies comparable to those for normal fish.

No application of velocity gradients for diverting fish from power plant intake structures has been tested. The use of velocity gradients within an intake structure requires a confined approach channel to the intake structure and relatively high approach velocities. This combination is expected to result in a condition which would entrain and entrap fish and would require effective guidance of diverted organisms into a bypass system. Further, this combination may also lead to the establishment of a resident predator population. Velocity gradients are not a reliable technique for reducing entrainment or impingement at power plant intakes.

D.5.1.4 Air-Bubble Screens

Air-bubble screens have generally been reported to be unsuccessful at consistently diverting fish consistently (Brett and MacKinnon 1953; Fields 1966; Mayo 1974). However, several cases have been reported where partial success was observed. Bibko et al. (1974) reported that striped bass would not actively pass through an air-bubble screen at 40 or 52 F (4.5 or 11.1 C), but were found to drift passively through the screens when the water temperature was 33 F (0.8 C). Striped bass were found to pass through an air-bubble screen at all test temperatures if openings 2 in. (5.2 cm) or greater were allowed in the screen. Gizzard shad would not pass through an air-bubble screen at water temperatures of 52 F (11.1 C), but were not deterred at 33 F (0.8 C) (Bibko et al. 1974). Bates and VanDerwalker (1964), studying juvenile migrant salmon, reported that air-bubble screens produced diversion efficiencies up to 95 percent during daylight, but declined to 28 percent at night and were not improved by artificial lighting. Alevras (1974) observed that an air-bubble screen at the Indian Point Power Station on the Hudson River did not appear to repel fish during the daytime, and his preliminary data indicated that the air-bubble screen may attract fish to the intake at night.

Because of the variability in response among species, poor effectiveness at night and in turbid water, and lack of success in field applications, air-bubble screens are not

an effective method of reducing either entrainment or impingement.

D.5.1.5 Electric Barriers

Electric barriers have been used to divert fish from several small power plants, dams, irrigation canals, and municipal water supply systems with variable success. Holmes (1948) discussed the history, developmental problems, and practical applications of electrical techniques for fish diversion. Applegate et al. (1954) reviewed the literature related to the use of electric fish screens.

Pugh (1962), Pugh et al. (1964), and Elliott (1970) reported that pulsed current was most effective in terms of guidance, diversion, and power requirements. The behavioral reaction of fish to an electric field are: (1) avoidance, (2) electrotaxis, and (3) electronarcosis leading to paralysis and eventual death (Applegate et al. 1954; Elliott 1970; Maxwell 1973). In general, the required current density and resulting behavioral response varies among species and sizes of fish (Pugh 1962; Pugh et al. 1964).

Trefethen (1955) reported 68 percent guidance efficiency of electric barriers in large scale laboratory experiments using fingerling salmon. Efficiency decreases as water velocity exceeds 0.5 fps (15 cm/sec) (Pugh 1962; Pugh et al. 1964). Maxwell (1973) reported variable success of electric barriers used at small freshwater intakes, usually with resident rather than migrating fish species. Because of low electrical resistance, no application of electric fish barriers has been made in salt or brackish waters.

Electric barriers are most effective at intake structures where the impingement of only a few species of relatively uniform size is of concern. At intakes where a variety of species and sizes of fish are encountered, or at intakes sited in marine or estuarine waters, electric barriers are not effective in minimizing entrainment (U.S. EPA 1976).

D.5.1.6 Chemicals

A barrier consisting of a fish-repelling chemical, discharged near the entrance of an intake structure, may be used to discourage the entry of fish into the intake.

Fish-repelling chemicals are available, but no large-scale testing has been conducted. Chemical barriers would be of little practical value because of the large volumes of cooling water involved and the adverse effect of the discharge of fish-repelling chemicals on nontarget organisms inhabiting the receiving water body.

D.5.1.7 Magnetic Barriers

Magnetic fields have been examined as a potential method of guiding juvenile salmonids past water diversions (Cannon et al. 1979). Salmonids did not respond to changes in magnetic fields under experimental conditions. No tests have been conducted to examine the effectiveness of magnetic barriers at power plant intake structures. In the absence of any information suggesting the applicability of magnetic fields for reducing entrainment or impingement, further consideration of magnetic barriers is not warranted.

D.5.1.8 Chains and Cables

Chains and cables hung vertically at the entrance to a water intake structure or water diversion have been examined as a potential behavioral barrier. Results of experiments conducted with juvenile salmonids by Brett and Alderdice (1958) suggest that chain and cable barriers are not effective in consistently diverting fish. In the absence of demonstrated effectiveness, chain and cable barriers are not considered to be a reliable technique for reducing the entrainment or impingement of fish at water intakes.

D.5.1.9 Louvers

The application of fixed louvered deflectors for diverting juvenile migrant salmonids and striped bass has been discussed by Bates and Vinsonhaler (1957), Bates and Jewett (1961), and Skinner (1974). Diversion of downstream migrants of anadromous species with louver deflectors has had variable success. Diversion efficiency is influenced by the approach velocities and the species and sizes of fish. Louvers have proven to be a relatively well-developed engineering and biological design concept, requiring minimum handling of fish. They do not, however, divert fish eggs and larvae, are vulnerable to biofouling accumulations, and are not self-cleaning.

Skinner (1974) discussed the diversion efficiency of the Delta Fish Protective Facility near Byron, California, which diverts water from the Sacramento - San Joaquin river system primarily for agricultural usage. The facility is a fixed-louver design oriented 15° to the flow, with primary and secondary bypass facilities. Present diversion capacity of the facility is approximately 6,000 cfs (170 m^3/sec). Louver efficiency for white catfish was found to be directly related to fish length. Primary diversion efficiency ranged from 4 percent for white catfish 0.4-0.5 in. long to 68 percent for those 3.0-3.9 in. (10-12.5 mm and 75-100 mm) long. Diversion efficiency for salmonids was related to fish length and water velocity. Velocity was found to influence diversion efficiency of smaller fish greatly, but as fish increased in size and swimming capabilities, velocity became less critical. Diversion efficiency for salmonids ranged from 65 to 84 percent in 1970 and from 84 to 90 percent in 1971 for salmon 2.0-5.9 in. (50-150 mm) long. Differences in diversion efficiency between the two years was thought to be related to the operator's ability to attain and maintain test parameters. Based on observations of 1.3 million striped bass 0.2-4.9 in. (5-125 mm) in length, there appeared to be an inverse relationship between diversion efficiency and channel approach water velocity. Lower approach velocities were most favorable for striped bass less than 1.2 in. (30 mm) long. The diversion efficiency of the louver system for striped bass was 69 percent.

Schuler (1974), using an experimental approach, observed the behavioral reaction of marine fish to fixed louvers oriented at various angles to the flow, to various louver vane spacings, and to various approach velocities. For marine fish, highest guidance efficiency was observed for louvers oriented 20° and 30° to the flow with vanes 1 in. (2.5 cm) apart. Maximum diversion efficiency occurred at a mean approach velocity of 2 fps (61 cm/sec). Schuler (1974) observed for marine species that the guidance component extended 5.9-7.9 in. (15-20 cm) from the louver face and that test species guided better along an array of louvers than along screens.

Louvers have been used effectively for diverting migrating anadromous fishes from several large agricultural water diversions, but the application of louvers to power plant cooling water intakes has not been attempted. Louvers do not provide a positive barrier for eliminating either entrained planktonic organisms or debris from a cooling water flow, and entrainment could result in blockage of the condenser tube system and lead to reduced operating reliability and increased maintenance of the cooling water system.

One U. S. utility has recently begun construction of a combined louver - traveling screen intake for application at a power plant cooling system intake. The structure is designed to provide the guidance characteristics of louvers and the positive physical barrier of a screen. No conclusions can be drawn regarding the effectiveness of the louver - traveling screen system in diverting fish from a power plant intake until the intake structure is operational and field studies can be completed.

D.5.1.10 Velocity Cap

During the early 1950s, operation of offshore intakes would occasionally result in large schools of fish becoming entrapped within the intake conduit. The entrance of these offshore intake conduits were unscreened and oriented vertically in the water column to minimize sediment deposition within the intake structure. The large numbers of fish entrapped within the intake would eventually become impinged on the intake screens and would, on occasion, force shutdown of the power plant cooling system. Based on results of experimental testing, it was determined that fish sense and react to vertical flow fields much more slowly than to horizontal flow fields. As a result of these observations, a velocity cap (Figure D-4) was placed over the entrance to the offshore intake to reorient the flow patterns from a vertical to a horizontal flow field. The quantity of fish removed from one intake structure decreased from 272 tons the year before installation of the velocity cap to 15 tons the year after installation, a reduction of nearly 95 percent (Weight 1958).

Velocity cap intake structures are presently in use in offshore intakes in the Pacific Ocean off southern California and in Lake Michigan. A velocity cap functions as a behavioral barrier and is used in combination with onshore intake screens, which function as a positive physical barrier for those fish that do enter the intake. A velocity cap does act as an effective behavioral barrier for reducing the numbers of impinged fish when used with an offshore intake structure. Velocity caps are not expected to reduce the numbers of planktonic organisms entrained.

D.5.2 Physical Barriers

Historically, intake screens and other physical barriers used in power plant intake structures have been designed to remove entrained debris from the cooling water supply prior to passage through the condenser cooling system. During recent years,

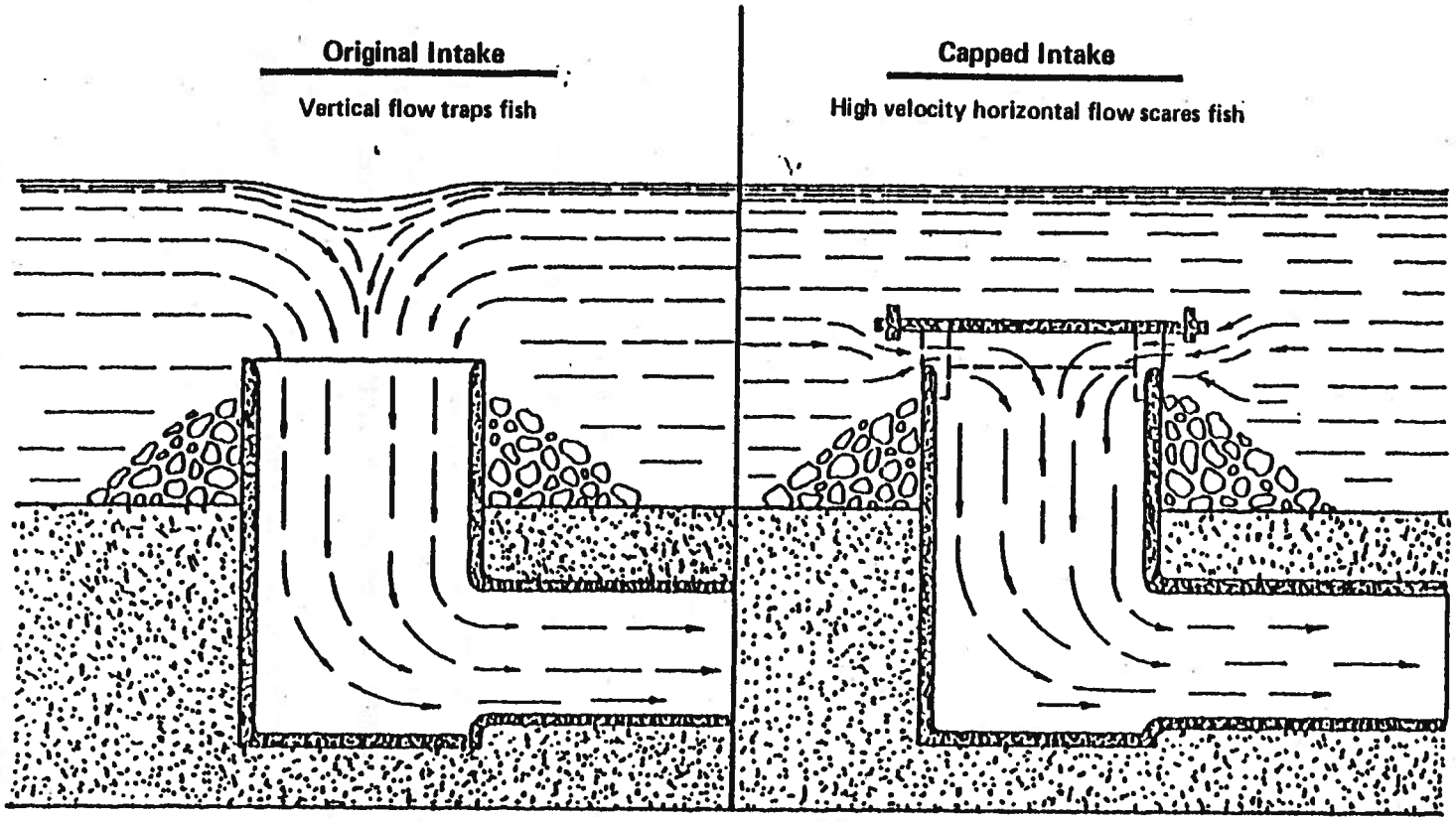

Figure D-4. Schematic representation of an offshore intake with velocity cap. (From Weight 1958.)

extensive effort has been devoted to designing and modifying physical barriers to provide exclusion of entrained fish and invertebrates from the water supply while minimizing mortality due to impingement. In addition to excluding debris and organisms from a cooling water supply, an acceptable physical barrier must be mechanically dependable and operate as designed under a variety of conditions.

Physical barriers are designed to block the passage of entrained or entrapped organisms on the basis of the size of organisms encountered at specific intake systems. Some physical barriers rely on impinging organisms for brief periods while they are mechanically transferred to a bypass channel. Recently, many physical barrier systems (e.g., vertical traveling screens and centerflow screens) have been modified on an experimental basis to include fine-mesh (0.02- to 0.1-in.; 0.5- to 2.5-mm) screen material, in an attempt to minimize the numbers of eggs and larval fishes entrained into cooling water systems.

In the sections which follow, biological and engineering considerations associated with the use of various physical barriers at power plant intakes are reviewed. Discussions are based on a review of laboratory studies, prototype field studies, and actual operating experience at water intake and diversion structures. Alternative designs discussed include various media filters and infiltration systems, various fixed screen concepts, and various traveling screen designs. Qualitative conclusions are presented in Table D-2.

D.5.2.1 Rapid Sand Filters

The high-capacity rapid sand filter (infiltration gallery) has the potential to prevent entrainment, entrapment, and impingement of all species and life stages of fish in power plant cooling systems. Free flow across the filter surface and low approach velocity combine to eliminate potential fish kills and the need for handling and disposal of debris. These filter systems, however, are plagued by operational problems, primarily those associated with clogging of the filter media. As a result of these operational problems, filters have had limited use at power plant intake structures.

The Montour Stream Electric Generating Plant, located on the Susquehanna River, Pennsylvania, utilized a filter bed designed to provide makeup cooling water at a rate

TABLE D-2 SUMMARY OF EXPERIENCE WITH PHYSICAL BARRIERS FOR REDUCING ORGANISM LOSSES AT COOLING WATER INTAKES

Technology	Applicable to Once-Through	Applicable to Closed-Cycle	Reduces Entrainment	Reduces Impingement	Operationally Reliable	Comments
Rapid sand filter	○	●	●	●	○	Suitability limited to extremely low flow rates
Porous dike	Ex	Ex	Ex	Ex	Ex	Three-year field training program underway, 1979-1981
Radial well collector	○	●	●	●	○	Suitable for low flows; depends on geology, groundwater levels, and water quality
Stationary screen, perforated pipe, and plate	○	●	Ex	Ex	○	Possibly suitable where debris loading is light and biofouling minimal
Drum screen	○	○	○	?	●	Vertical type ineffective for subadult fish
Horizontal traveling screen	○	○	○	?	○	Mechanical performance under sediment load and varying water conditions unsatisfactory
Rotating disc screen	○	○	○	○	○	Limited non-power-plant application
Centerflow traveling screen	●	●	○	?	?	Requires increased screen-to-pump distance; irregular flow patterns
Vertical traveling screen	●	●	○	○	●	Impingement survival depends on design and operation parameters

Legend

● Demonstrated

? Inconclusive, conflicting or variable data

Ex Undergoing experimental studies, developmental stage

○ Demonstrated ineffective or no data

of approximately 33,000 gpm (73 cfs; 2 m^3/sec). Filter operation was considered to be unacceptable as a result of the clogging of the filter media by algae and fine sediments. Despite several modifications to the original design, the operational reliability of the filter was considered unacceptable, and its use was discontinued. The filter bed intake was replaced by a perforated pipe intake design.

The feasibility of using a filtration bed as a cooling water intake structure in the marine environment was examined by Stober et al. (1974) and Strandberg (1974). It was concluded that a filter intake sytem is an effective means of excluding pelagic eggs and larval fish from power plant cooling systems. However, the present technology is not considered to be sufficiently developed to allow reliable design and operation of filter intake systems.

Low infiltration rates of media filters require large surface areas, approximately 10 acres (24.7 hectare) for every 1,000 gpm (2 cfs; 0.07 m^3/sec) of cooling water, which makes their application at many sites infeasible even when used to supply makeup water for closed-cycle cooling systems.

Operational problems and poor reliability of filter intake systems, as a result of clogging, siltation, and biofouling, make further consideration of the concept for use at power plant intakes unwarranted.

D.5.2.2 Porous Dike

A porous dike or leaky levee uses a filter medium of stone and gravel to exclude organisms from a cooling water system. Porous dikes are in the developmental stage and have not been used to date. The porous-dike concept is applicable for both shoreline and offshore intakes, may provide a behavioral and physical barrier for larval and juvenile-adult fishes, and eliminates the necessity of a fish return system. Operational reliability under actual field conditions and maintenance associated with biofouling, clogging, and sediment accumulation have not been determined.

Preliminary criteria for the design and use of a porous dike as an intake structure include the following:

1. the dike must be stable and resistant to erosion by water currents and wave action;
2. the size and composition of the medium must provide structural integrity, acceptable water flow rates, and exclusion of organisms;
3. the hydraulic efficiency must be high and biofouling and sediment accumulation low; and
4. the dike must not require excessive maintenance.

Hydraulic investigations of flow-through porous-dike structures have been conducted under laboratory conditions (Roberge 1978) and provide the basic foundation for the design of large-scale porous dikes. It was concluded from an engineering evaluation of porous-dike intake structures (Roberge 1978), that the concept is feasible, but long-term prototype testing under field conditions is required to develop a more complete understanding of the engineering feasibility and performance of a porous-dike intake.

The primary biological advantage of a porous dike is its effectiveness in excluding larval and juvenile fishes from the cooling water system. Schrader and Ketschke (1978) reviewed the biological aspects of porous-dike intake structures and concluded that a porous dike may act as both a behavioral and a physical barrier for larval and adult fishes. Available information, however, is insufficient to determine its exclusion efficiency for various sizes and species of fish as a function of medium size and approach velocity; nor have the exclusion efficiency and survival of entrained fish eggs been determined for porous-dike intake structures.

A 3-year field testing program to determine the operation and maintenance, engineering performance, and biological effectiveness of a large-scale prototype porous dike is currently being considered. Testing would be conducted at the Brayton Point Steam Electric Station located on an estuarine portion of Narragansett Bay, Massachusetts.

No conclusions regarding the engineering feasibility or biological effectiveness of porous-dike intake structures can be made until data from the field testing program become available.

D.5.2.3 Radial Well Collectors

The radial well is a subterranean intake system that withdraws water from an aquifer or from an overlying water source such as a river or lake. Radial wells have been used to supply water for domestic and industrial usage and are planned to be used to supply cooling tower makeup water at several steam electric power plants. The maximum capacity of a typical radial well collector is approximately 55,500 gpm (124 cfs; 3.5 m^3/sec), which is insufficient to meet the demands of a once-through cooling system. Limitations in aquifer size, sediment permeability, and groundwater hydrology make it technically and economically impractical to build radial wells big enough to provide adequate water volume for once-through cooling.

Radial well collectors rely on induced infiltration through unconsolidated alluvial deposits; they furnish water that is typically low in turbidity and concentrations of suspended solids. In addition, because of the extremely large surface area associated with radial well collectors, the water velocity at the water body/sediment interface is extremely low, less than 0.0003 fps (0.01 cm/sec) (Mikels and Bennett 1978). The low velocities and positive physical barrier created by the substrate result in complete exclusion of egg and larval fish and microzooplankton from the water being withdrawn. Thus, the radial well functions as an effective intake device for minimizing or eliminating the biological impact associated with either entrainment or impingement.

Engineering and operational considerations include the suitability of geological structures and groundwater hydrology of a particular site, sediment deposition, presence of iron bacteria, biological fouling, and accumulation of debris. Before a radial well collector can be considered acceptable for a specific site, the potential for depleting groundwater storage reserves, particularly during periods of drought, and the operational reliability of the intake system through the life of the power plant must be evaluated.

The radial well collector provides a biologically effective intake structure which eliminates entrainment and impingement. The use of radial wells, however, is limited to specific sites, because of substrate permeability and aquifer characteristics, and is limited to meeting small water demands comparable to the makeup water requirements for closed-cycle cooling. The limited capacity and siting constraints significantly limit the applicability of radial well collectors at power plant intakes.

D.5.2.4 Stationary Mesh Screens, Perforated Plates, and Pipes

Stationary screens have long been used in irrigation canals and industrial water intake systems. Debris accumulations on screen surfaces reduce water flow volume and the efficiency of these screen installations. Blockage of a stationary intake screen by impinged debris results in unacceptable operational reliability of stationary screens for use in once-through cooling water intakes.

Perforated plates and pipes provide a positive barrier for adult and subadult fish and have the potential for the protection of eggs, larvae, and juvenile fish (Maxwell 1973; Richards 1978). Perforated intakes, however, are not self-cleaning, and thus require backwashing and mechanical removal of debris; and biofouling and debris accumulations reduce reliability. Perforated pipe intake structures are being used to supply makeup water at several power plants, but perforated pipe has not been used for once-through cooling systems.

Stationary cylindrical screens are presently being considered for use in power plant cooling water intakes. Currently, one power plant, Big Sandy Generating Station in Louisa, Kentucky, uses four cylindrical screens, each 2 ft (0.6 m) in diameter and 9.8 ft (3 m) long, to provide cooling tower makeup water at a rate of 22.2 cfs (0.63 m^3/sec). The application of stationary cylindrical screens for once-through cooling system intakes is being evaluated (Key and Miller 1978; Cannon et al. 1979). Wedge-wire cylindrical screens are presently in the developmental stage and are not considered to be available technologies for application to once-through cooling water systems (Cada et al. 1979).

The design of stationary, submerged, cylindrical screens typically includes the use of a wedge-wire material. Wedge-wire screen is manufactured by welding wire of triangular cross-sections to a support structure (Figure D-5). Wedge-wire screen is generally constructed with slot widths ranging fron 0.01 to 0.1 in. (0.25-2.5 mm) and offers 20-35 percent open area for water passage. This design results in a smooth screen surface and screen openings that widen inwards, facilitating screen cleaning (backwashing) and increased hydraulic efficiency (Cook 1978). The cylindrical screen configuration, which promotes radial flow into the screen, and the use of wedge-wire

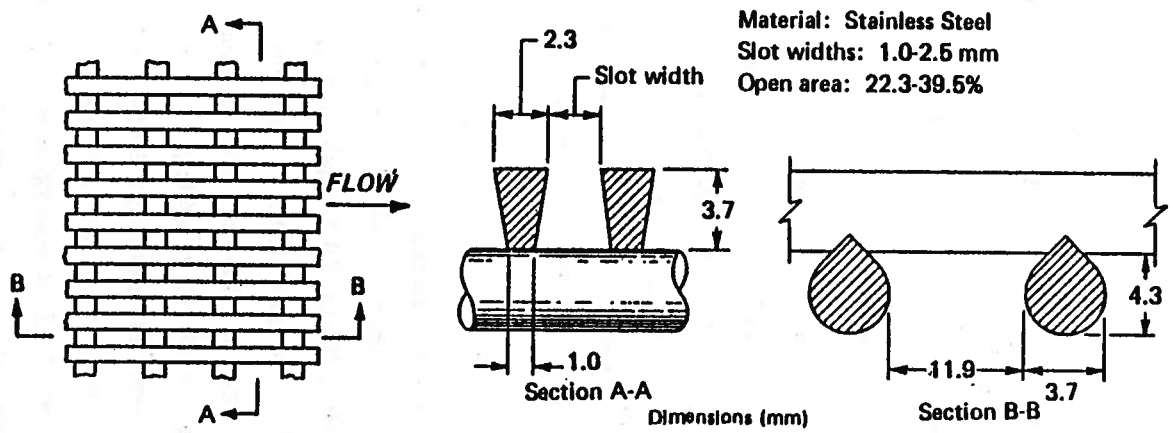

Figure D-5. Characteristics of flat wedge-wire screen panels used in the TVA hydraulic test. (From Cada et al. 1979.)

screen material are expected to reduce screen clogging from impinged organisms and debris.

A typical intake system with stationary, completely submerged, cylindrical screens is shown in Figure D-6. The cylindrical screens can be used individually, in pairs, or in a manifold design to provide the necessary cooling water volume. The fish protection potential of a cylindrical wedge-wire screen depends on the location and orientation of the screen with respect to ambient water currents, on the size of the screen openings, and on the through-screen velocity. The absence of a confining screenwell which may entrap organisms and the design of screens with through-screen velocities less than 0.5 fps (15 cm/sec) serve to reduce the numbers of organisms entrained and impinged.

Biological data on the potential effectiveness of cylindrical screens using wedge-wire screening to minimize entrainment and impingement are limited to prototype flume experiments conducted by Hanson et al. (1978). The entrainment and impingement of fish of various sizes (eggs, larvae, and subadults) on screen cylinders 1 ft (0.3 m) in diameter and 2 ft (0.6 m) long, made up of wedge-wire screen with slot widths ranging from 0.01 to 0.04 in. (0.25-1.0 mm) was tested.

Through-screen velocity was 0.5 fps (15 cm/sec) and channel velocity ranged from zero to 2.6 fps (79 cm/sec). Results of these experiments indicate that (1) impingement of fish larger than 0.6-in. (15 mm) in fork length was virtually eliminated on screens with 0.02-in. (0.5-mm) slot width; (2) fish 0.5-0.7 in. (12-17 mm) in fork length which were impinged on the screen showed excellent ability to escape impingement; (3) less than 0.1 percent of the striped bass eggs tested were entrained through screens with slot widths of 0.01-0.02 in. (0.25-0.5 mm); (4) mortality of striped bass eggs impinged on 0.02-in. (0.5-mm) wedge-wire screen for 0.5-2 minutes ranged from 0 to 11.9 percent and averaged 1.4 percent for all tests. These results suggest that cylindrical wedge-wire screen offer a potentially effective means of reducing impingement and entrainment losses at power plant cooling water intakes. A cylindrical wedge-wire screens 3.3 ft (1 m) in diameter and 9.8 ft (3 m) long with slot widths of 0.04 in. (1 mm) is currently being field tested (Ecological Analysts 1979) to evaluate the potential effectiveness of this screen concept for reducing fish entrainment under operating conditions.

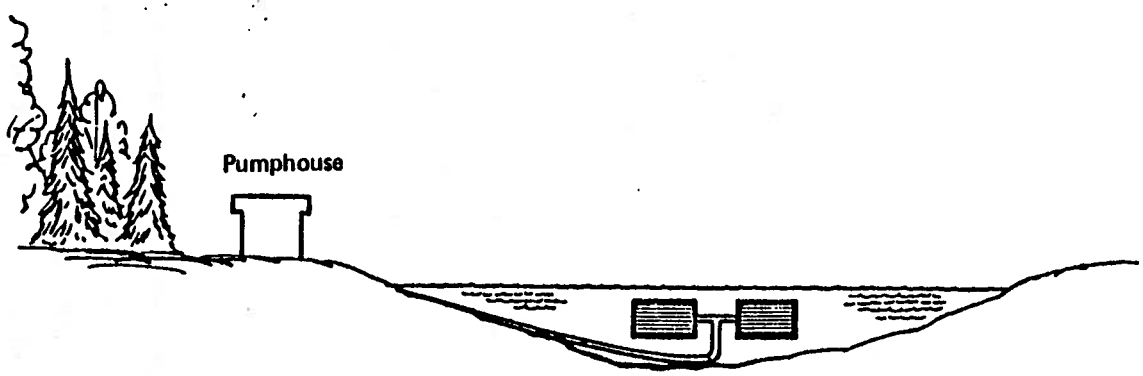

Figure D-6. A design concept for cylindrical wedge-wire screens at power plant cooling water intakes. (From Cada et al. 1979.)

Primary engineering concerns for stationary cylindrical wedge-wire screens focus on maintenance and operational reliability of the intake, which are influenced by the extent and frequency of debris clogging and biofouling accumulation on the screen surface. Each cylindrical screen must be equipped for either air-burst or hydraulic backwashing to remove accumulated debris. An ambient water velocity past the screen greater than the through-screen velocity promotes self-cleaning. The significance of a tangential velocity component across the screen to minimize clogging is discussed by Hanson et al. (1978); the problem of removing biofouling organisms from screens in brackish water or marine environments is described by Key and Miller (1978). The effects of clogging and biofouling on the operational reliability of stationary cylindrical screens must be extensively investigated, especially for intakes sited in the marine environment or in areas of heavy detrital loading, before they can be used in once-through power plant cooling water intakes.

The design and operation of stationary screening systems have been established in many small industrial applications. Stationary screening systems such as the wedge-wire cylindrical screens have only recently been applied to power plant intakes. Field and laboratory testing is underway to establish the biological effectiveness of stationary screens and to refine engineering design criteria for screen location, orientation, through-screen velocity, and mesh opening size. Operational reliability, screen clogging and biofouling, and maintenance requirements for stationary screens sited in fresh and salt water need to be determined. The operational reliability of stationary screen intakes for once-through cooling systems has not been demonstrated.

D.5.2.5 Drum Screens

Vertical drum screens have been used for fish diversion in irrigation canals, and in British steam electric stations for the protection of salmonids, with variable success (Eicher 1974). Vertical drum screens having diameters of approximately 9.8 ft (3 m) are commonly aligned in rows leading to a bypass channel. Screen rotation carries impinged fish toward the bypass and provides self-cleaning of debris. Vertical drum screens provide a positive barrier for adult and subadult fish, but do not appear to be capable of protecting eggs and larval fish. Fluctuations in water depth do not affect the performance of vertical drum screens.

Horizontal drum screens have been used for fish diversion in irrigation facilities, power plants, and hydroelectric projects, as discussed by Maxwell (1973), Eicher (1974), and Mayo (1974). These well-developed and proven screens provide a positive barrier for adult and subadult fish, but do not protect fish eggs or larvae. The screen surface is cleaned through screen rotation, but screen rotation is detrimental to impinged fish if no bypass facility is provided. Diversion efficiency of horizontal drum screens is sensitive to fluctuating water levels.

Drum screens, although acceptable for use in intake structures of once-through power plant cooling systems, offer no biological or engineering advantages over conventional vertical traveling screens. Drum screens have not been utilized extensively in electric generating stations in the United States, partly because of problems associated with the carry-over of debris into the condensers, which reduces system reliability.

D.5.2.6 Horizontal Traveling Screens

The horizontal traveling screen (Figure D-7) developed by Bates (1969) has been proposed for fish diversion in canals, hydroelectric dams, irrigation intakes, and power plant intakes. Major attributes of the horizontal traveling screen concept include (1) formation of a complete physical barrier, (2) high diversion efficiency of juvenile migrant fish, and (3) release of impinged fish into a bypass without passing the air-water interface. Operation of horizontal traveling screens is not greatly affected by fluctuations in water depth. The screen is self-cleaning, minimizing head loss, and is capable of operating under higher approach velocities than is normally possible with other types of screens. Research and development of the horizontal traveling screen has been reported by Bates (1969), Bates et al. (1970), Farr and Prentice (1974), and Prentice and Ossiander (1974).

Prentice and Ossiander (1974), studying 2.8-in. (70-mm) fingerling chinook salmon (Oncorhynchus tshawytscha), observed 98 percent diversion efficiency of a traveling screen aligned at an angle of 30° to a flow of 1.5 fps (46 cm/sec) under lighted conditions, and a 91 percent diversion efficiency under dark conditions. Diversion efficiency for 6.7-in. (170-mm) salmonids tested under similar conditions was 99.6 and 99.8 percent during day and night tests, respectively. No screen impingement was observed during these tests, and 48-hour post-test survival exceeded 97 percent for all experiments.

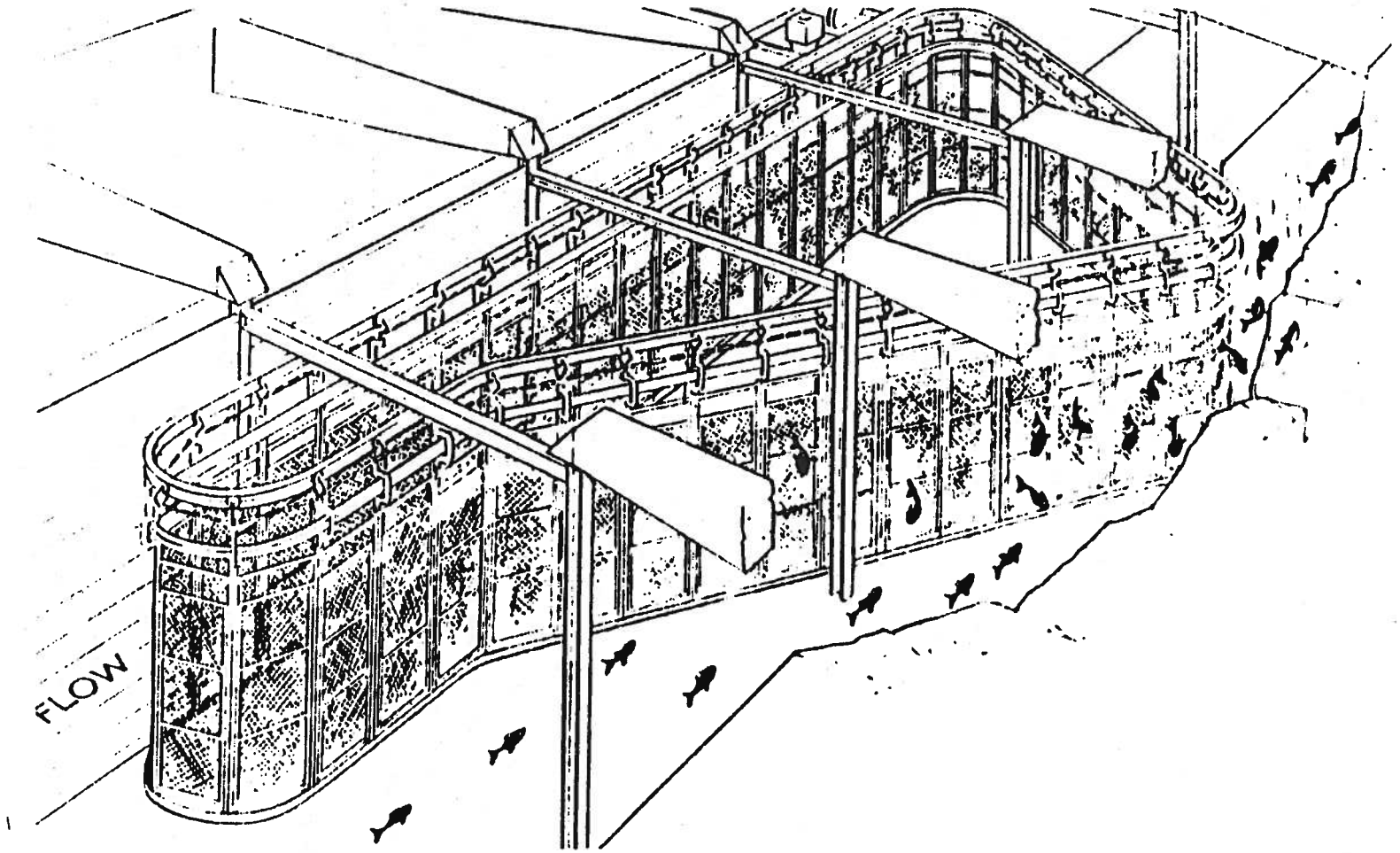

Figure D-7. Horizontal traveling screen. (Courtesy of Envirex.)

Mechanical performance of horizontal traveling screens is not, at present, acceptable for continuous operation at a power plant intake. Prototype tests designed to examine mechanical operation and to assess performance limitations are discussed by Farr and Prentice (1974). Solutions to mechanical problems, resulting from suspended sediment and sediment bedload, have yet to be perfected.

Horizontal traveling screens appear to be effective in minimizing fish impingement but are not expected to reduce the number of organisms entrained into a cooling system. The lack of reliable operation of horizontal traveling screens, however, precludes any further consideration for their use in cooling water intake structures.

D.5.2.7 Rotating Disc Screens

Rotating disc screens have been used in both small irrigation diversions and municipal water supplies. No application of rotating disc screens in power plant intake structures has been reported. Rotating disc screens offer no biological or engineering advantage over conventional vertical traveling screens for use in cooling water intakes.

D.5.2.8 Centerflow Traveling Screens (Single-Entrance, Dual-Exit Traveling Screens)

The centerflow traveling screen represents a relatively new intake screening technology in the United States, although it has been utilized extensively in European industrial and electric generating facilities for almost three decades. Centerflow screens (Figure D-8) are oriented parallel to the approaching waterflow, and water enters through the center of the screen and exits through both vertical faces. Screen panels on centerflow screens are attached to a continuous drive chain and unlike conventional vertical traveling screens, each screen panel forms a concave basket, increasing available screening surface area. Screen mesh sizes range from 0.02 to 0.4 in. (0.5-9.5 mm), with many units having 0.04- to 0.1-in. (1.0- to 2.5-mm) mesh. Because debris can accumulate rapidly on the small mesh commonly used in centerflow screens, they are designed to operate with continuous screen rotation.

The hydraulic flow patterns associated with centerflow screens are more complex than those associated with vertical traveling screens. Water flows into the screen

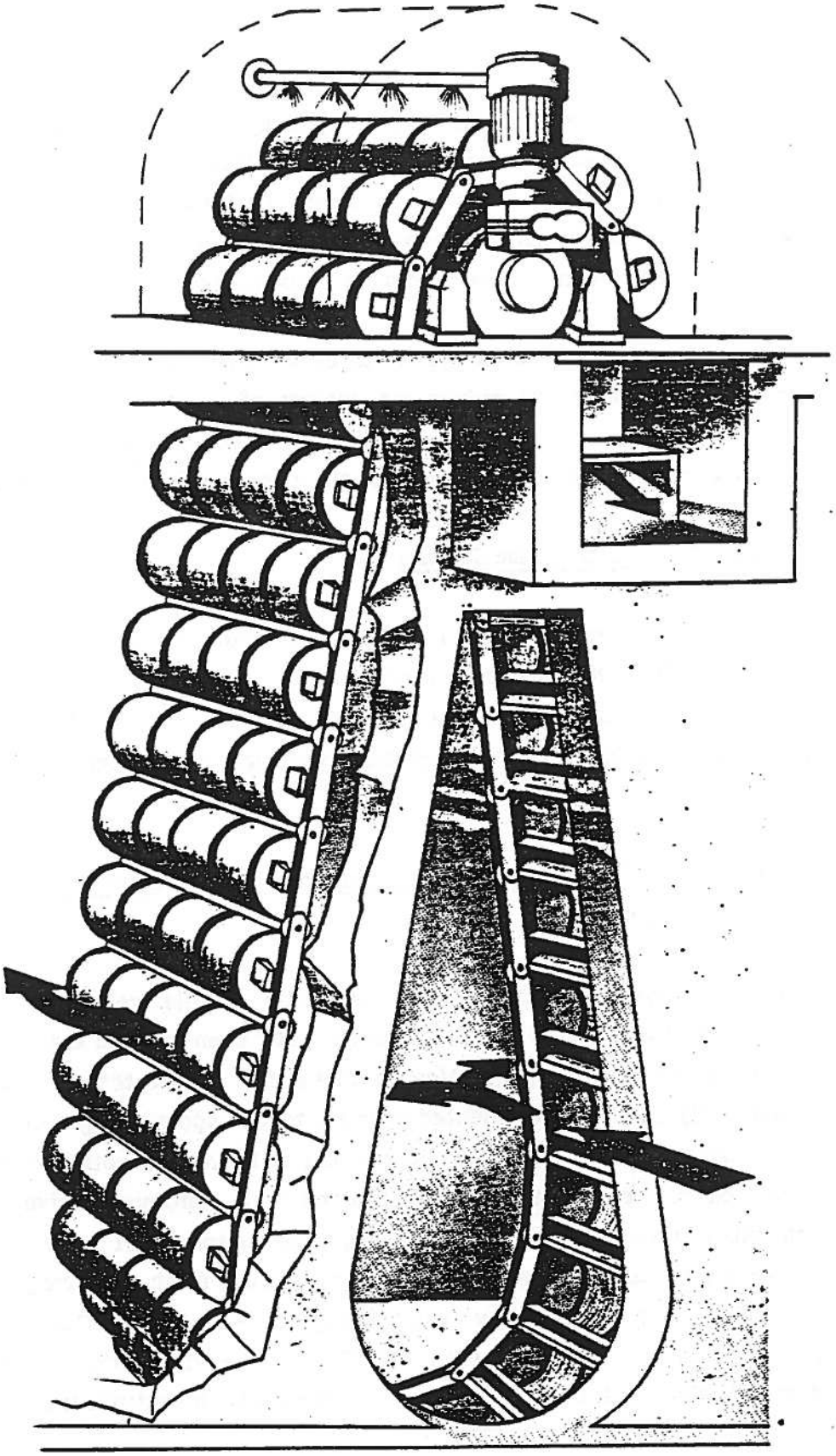

Figure D-8. Center-flow traveling screen. (Courtesy of the Passavant Corporation.)

chamber through a keyhole-shaped opening in the front wall of the intake screen structure, turns 90° to flow through the screen mesh, and turns 90° to complete its transit to the circulating water pump. Effective screening occurs on both the ascending and descending sides of centerflow screens, increasing available screen surface area. As a result, the average through-screen velocity of a centerflow screen is less than the approach velocity to the screen. However, through-screen velocities are highly variable across the screen face because of flow patterns and screen orientation, and velocities tend to increase with distance downstream from the chamber entrance. The turbulence of the flow may negate potential benefits from reduced average through-screen velocities. High velocities at the entrance to the screen may also serve as a behavioral barrier, preventing the escape of fish entrapped within the screen structure.

Screenwash mechanisms are similar for both centerflow and vertical traveling screens. Spraywash pressures generally range from 50 to 100 psi. Debris and impinged organisms dislodged from the screens fall directly into a single sluiceway for return to the source water body. Fish-protection systems for centerflow screens, although not currently commercially available, may involve such modifications as watertight holding troughs at the base of each screen panel; a low pressure (10-20 psi) spraywash, and an additional sluiceway to return impinged organisms to the source water body.

Murray and Jinnette (1978) examined the initial survival of organisms impinged on continuously rotating centerflow screens located at the Barney M. Davis Steam Power Plant, Corpus Christi, Texas. Initial impingement survival for 15 species of invertebrates and 37 species of fish was estimated to be 86 percent (10,406 individuals alive; 1,654 individuals dead) during the 12-month study. Impingement survival was found to vary among species and among months of the year. Lowest initial mortality for the dominant species by month occurred in February, when menhaden (mean standard length = 0.8 in.; 21 mm) experienced a 5 percent initial impingement mortality. Highest initial mortality for a dominant species by month occurred in June, when bay anchovy (mean standard length = 1.0 in.; 25 mm) experienced a 98 percent initial impingement mortality. No studies were performed to examine long-term survival of organisms impinged on centerflow screens.

Fine-mesh centerflow screens are being investigated in laboratory studies to

determine their potential effectiveness for protecting larval fish at power plant intakes (Magliente et al. 1978). Field and laboratory studies are required to evaluate the initial and long-term survival of eggs and larval fish impinged on fine-mesh centerflow screens. Insufficient data preclude a detailed comparison of the potential survival of egg and larval fish impinged on modified vertical traveling screens (fine-mesh screen material, fish buckets, low-pressure spraywash, continuous rotation) and centerflow screens (fine-mesh screen material, continuous rotation).

Centerflow screens require increased screen-to-pump distance over that required for conventional vertical traveling screens. This may make a screen retrofit of an existing intake infeasible because of space limitations. Irregular velocity patterns, turbulence, and attendant head losses are inherent with the waterflow patterns for the centerflow screen. Highest water velocities occur in the entrance area of centerflow screens and could contribute to fish entrapment. Within the screen chamber, velocity varies irregularly as it changes direction, resulting in nonuniform flow through the screening surface. Use of average approach and through-screen velocities for centerflow screens is thus an inappropriate and misleading design parameter. The turbulence associated with centerflow screen systems was examined by Alden Research Laboratory (1974), who found that circulating water pump performance was adversely affected by flow irregularities if screen-to-pump distances were two short or baffling was inadequate.

Operating and maintenance experience with centerflow screens in the United States is limited to one installation, the Barney M. Davis Steam Power Plant, where centerflow screens have been in use since 1974. Originally a 0.02-in (0.5-mm) polyester mesh screening material was used on these screens. However, the polyester screen material fatigued and subsequently failed during operation and was replaced with a nylon mesh material with 0.04-in. (1.0-mm) openings which is giving satisfactory performance. Other than operational problems encountered during start-up of the centerflow screens, routine maintenance has been described as nominal (P. Benes, Passavant Corp., 1978, personal communication). Details of operating and maintenance experience for centerflow screens outside the United States are unavailable.

Several alternative design configurations for the centerflow screen concept have been proposed in an attempt to minimize the probability of organisms becoming

entrapped within the screen structure. In addition to the single-entrance/dual-exit design shown in Figure D-8, there is a double-entry/single-exit design in which water enters the screen from two sides and passes out through one end. This design can be located either within a screenwell or supported from a platform with no confining concrete housing (Figure D-9). A double-entry/double-exit screen design has also been proposed. To date, biological and engineering data are not available to compare the relative effectiveness of these alternative screen configurations.

It can be concluded from available information that centerflow screens represent an acceptable intake screening technology for proposed units based on a consideration of engineering performance, operational reliability, and maintenance. Retrofit installation of centerflow screens into an existing intake structure would require extensive modification of the intake, and might adversely affect the performance and reliability of circulating water pumps. Centerflow screens do not offer any major biological advantages over conventional vertical traveling screens. Significantly increased impingement survival has not been satisfactorily demonstrated for these systems.

D.5.2.9 Vertical Traveling Screens

The conventional vertical traveling screen (VTS), the most commonly used intake screening device at power plants in the United States, is commercially available from a number of manufacturers in a wide range of lengths, widths, and materials. It consists of a series of screen panels attached to a chain drive, mounted vertically in a concrete structure which normally functions as the screen well and circulating pump well (Figure D-10). A design recently proposed by Western LNG Associates (California Public Utilities Commission 1978) would put the vertical traveling screens, but not the circulating water pumps, in an offshore intake structure in the form of a caisson. Although in recent years modifications have been made to enhance the fish protection capabilities of the screens, their primary function is to prevent clogging of the condenser tubes. The screens are designed to remove particles whose diameters are more than one-half that of a condenser tube. In most situations, 3/8-in. (9.5-mm) mesh is used. The screens rotate intermittently, either at regular intervals of several hours or when debris buildup causes a predetermined pressure differential across the screen; most screens are not equipped to rotate continuously for extended periods of time. During rotation, debris is removed by a

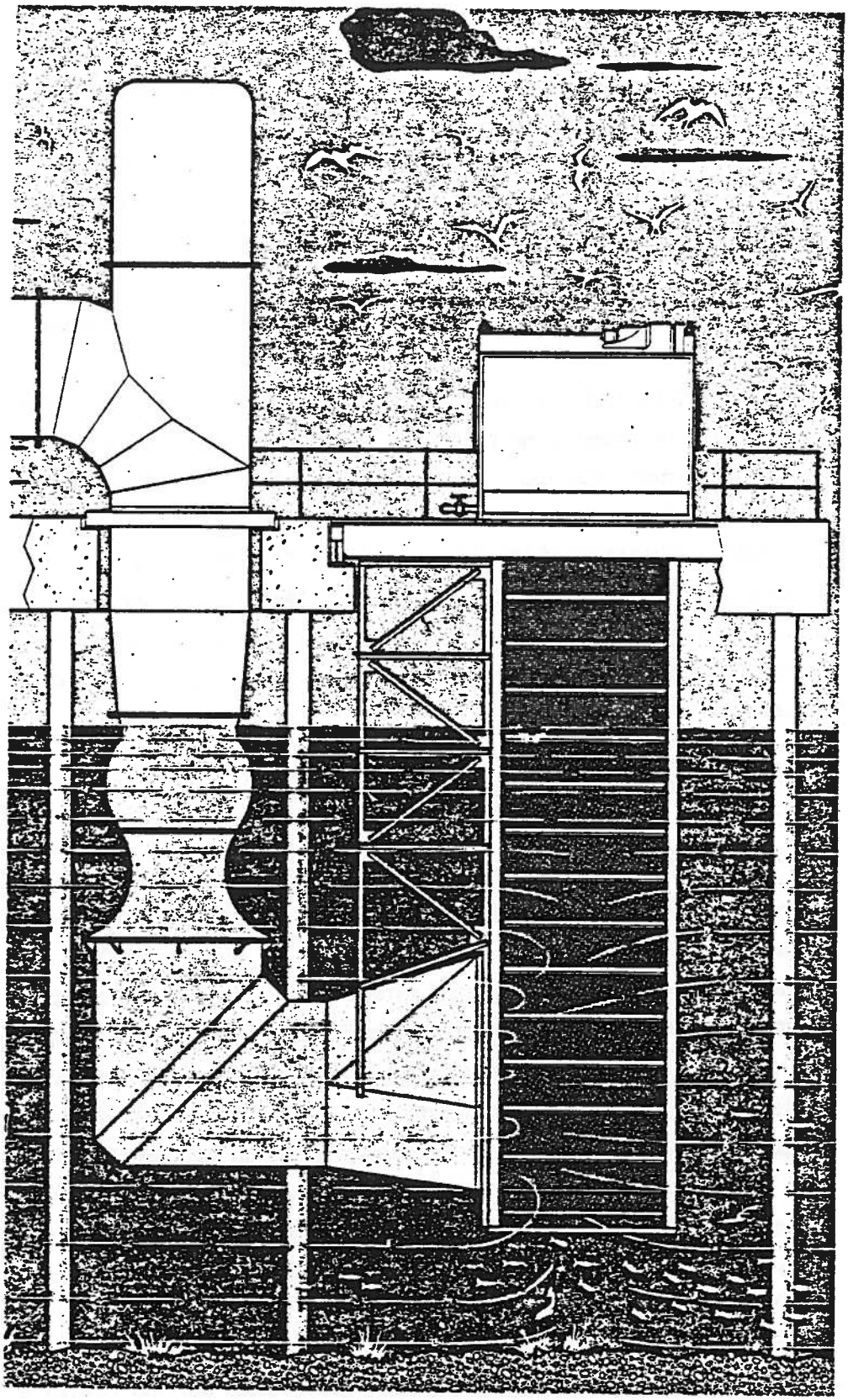

Figure D-9. Double-entry/single-exit screen. (Courtesy of FMC Corporation.)

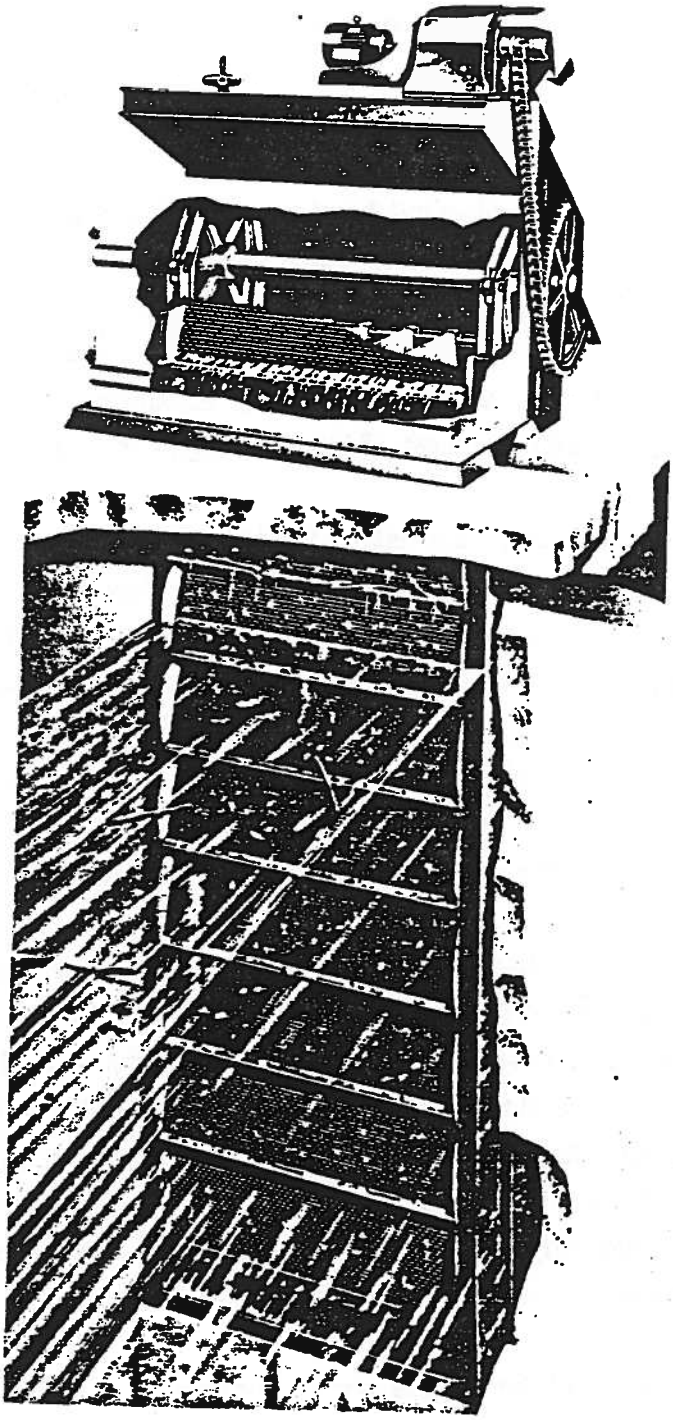

Figure D-10. Conventional vertical traveling screen. (Courtesy of Envirex.)

high-pressure (50-100 psi) spraywash and either returned to the source water body or transferred to trash bins for disposal.

D.5.2.10 Other Physical Barriers

A wide variety of physical barriers, in addition to those previously discussed, either have been or are currently being examined for their application to power plant intakes. Many of these alternative designs originated as intake structures for industrial or municipal water supplies, at irrigation water diversions, or in association with hydroelectric dams. There is an expanding effort to identify, design, and test the potential effectiveness of intake design concepts under laboratory conditions.

The most promising alternative intake design concept is the use of vertical traveling screens oriented at an angle to the intake flow and leading to a bypass. This design concept has been tested under laboratory conditions (Taft and Mussalli 1978) and was 100 percent effective in diverting alewife into a bypass. The angled-screen intake design is currently being incorporated into the intake structure design of several proposed and existing power plants. Field verification of the diversion efficiency of an angled-screen intake under actual operating conditions is not yet available. Engineering performance and operational reliability under field conditions is unknown.

Attempts have been made to exclude fish from the area around an intake by encircling the intake area with netting. The netting is believed to form both a physical and behavioral barrier for fish. Results of field testing conducted to date have been inconclusive. Problems associated with debris accumulation on the net, the maintenance of the integrity of the net barrier, especially in areas where bottom topography is irregular or following storms, and the hazard to navigation have not been adequately resolved. Net barriers are applicable only in lacustrine environments, where currents are minimal.

Variations of intake designs previously discussed include the use of offshore louvers, a caisson offshore intake structure which is surrounded by vertical traveling screens, inclined traveling screens (conveyor screen), and "hump-back" screens. Sonnichsen (1975) discusses the development and use of many of these alternative designs. Few of these alternatives are generally applicable to power plant intake screening, but may have site-specific applicability depending on the biological characteristics and engineering requirements.

D.5.3 Fish Collection, Removal, and Conveyance Systems

Three basic approaches exist for designing fish return systems for use in association with a cooling water intake structure. The first approach relies on excluding organisms from the cooling system by use of an intake structure that minimizes or eliminates impingement and entrapment (e.g., radial wells, sand filters, porous dikes, submerged stationary cylindrical screens, perforated pipes), and therefore requires no fish return system. The second approach, use of fish return-conveyance systems, relies on impinging organisms for brief periods until they are mechanically transferred to a screenwash sluiceway for return to the receiving water body. The third approach, fish removal systems, relies on concentrating entrapped organisms within an intake structure and removing them by use of pumps, elevator baskets, or bypass channels.

The following sections review the design considerations, effectiveness, and application of fish return - conveyance systems and fish removal systems at power plant intakes. Intake structures which rely on exclusion of organisms rather than use of return system were discussed in Section D.5.2. Qualitative conclusions are presented in Table D-3.

D.5.3.1 Vertical Traveling Screen Fish Buckets and Accessories

Impingement survival of aquatic organisms depends upon a number of interrelated factors: the species, life stage, size, impingement time, stress, fatigue, hydraulic parameters, and debris loading.

Impingement survival for some species has been improved by the incorporation of watertight collection buckets along the base of each screen panel to prevent repeated impingement of organisms and to provide a holding area for organisms during screen rotation (Figure D-11). In conjunction with this addition, a low-pressure wash system has been installed to remove impinged organisms from the screen and reduce the stress and abrasion from exposure to the high-pressure debris removal spraywash. A separate fish-return sluiceway is used to transport organisms back to the receiving water body. Continuous screen rotation has also been used to reduce stress on impinged organisms.

TABLE D-3 SUMMARY OF EXPERIENCE WITH FISH COLLECTION, REMOVAL, AND CONVEYANCE SYSTEMS FOR REDUCING ORGANISM LOSSES AT COOLING WATER INTAKES

Technology	Enhances Impingement Survival	Reduces Impingement	Reduces Entrainment	Reduces Entrapment	Operationally Reliable	Comments
Vertical traveling screen, fish buckets, and accessories	●	○	○	○	●	Some larvae are subject to increased mortality through physical handling; debris-fish-larvae separation undeveloped
Fish pump	○	●	○	●	●	
Fish bypasses or escapes	○	●	○	●	●	Lateral fish escapes are an outgrowth of the Kerr (1953) studies
Fish basket collector	○	○	○	Ex	Ex	Possibly applicable when fish pumps are not
Sluiceway and pipeline return	?	○	○	○	?	Design parameters difficult to identify for biological effectiveness

Legend

● Demonstrated

? Inconclusive, conflicting, or variable data

Ex Undergoing experimental studies, developmental stage

○ Demonstrated ineffective or no data

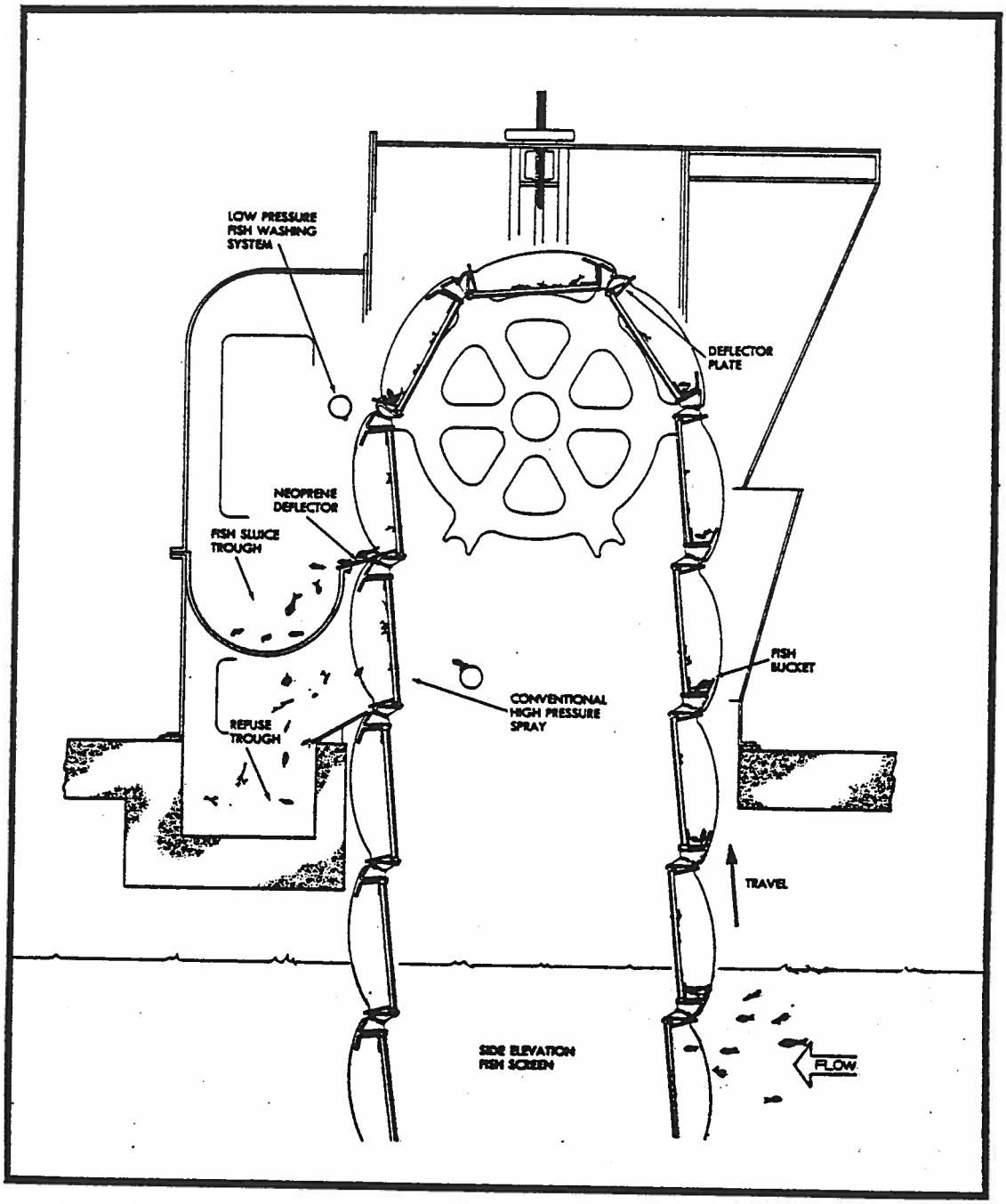

Figure D-11. Ristroph modified traveling screen with fish buckets, a modified spray system, and a sluiceway for fish. (Courtesy of Envirex.)

The modification of an existing vertical traveling screen to incorporate some of these features involves consideration of the specific intake structure and its performance. Potential problems include (1) alteration of the hydraulic characteristics of the intake structure which would affect the performance and reliability of circulating water pumps; (2) requirements for additional screen area to maintain acceptable approach velocities while providing adequate cooling water volumes; and (3) increased maintenance resulting from, e.g., screen clogging and reduced reliability of drive motors from continuous operation.

A dual spraywash system would require separate sluiceways for fish and debris, additional pumps, control mechanisms, piping, with an associated increase in maintenance costs. Installation of either watertight buckets on each screen panel or fine-mesh screen material would necessitate reinforcement of structural supports to maintain the structural integrity of the screen under increased static and dynamic loads. In the absence of extensive operating information for the modified screens, especially for those equipped with fine mesh, predictions of operation and maintenance problems that might influence reliability cannot be made with certainty.

King et al. (1978) compared the effects of different screen rotation frequencies and screenwash pressures on the survival of fish impinged on conventional vertical traveling screens at three power plants on the Hudson River (Table D-4). Continuous screen rotation resulted in better impingement survival for young-of-the-year white perch than screenwash at either 2- or 4-hour intervals (Figure D-12). Low-pressure screenwash sprays (50 psi) resulted in better initial and long-term (84-hour) survival of impinged white perch on several occasions, but the results of screenwash pressure studies were generally inconclusive.

White and Brehmer (1976) evaluated the initial survival of fish impinged on a modified vertical traveling screen similar to the one shown in Figure D-11, which has been in operation since 1974 at the Surry Power Station, James River, Virginia. Average initial survival for 58 species of impinged fish was 93.3 percent. Survival of impinged fish beyond the 15-minute holding and observation period was not determined.

The effectiveness of a low-pressure spraywash system in removing impinged fish from a modified vertical traveling screen was examined by Public Service Electric and Gas

TABLE D-4 IMPINGEMENT SURVIVAL OF JUVENILE WHITE PERCH UNDER VARIOUS CONVENTIONAL VERTICAL TRAVELING SCREEN OPERATIONAL MODES AND SCREENWASH PRESSURES AT THREE HUDSON RIVER POWER PLANTS DURING NOVEMBER AND DECEMBER 1976

Power Plant	Screen Operational Mode	Washwater Pressure (psi)	Survival (%)	
			Initial	Long-Term[a]
Roseton	Continuous	50	98	44
	Continuous	100	79	6
	2-hr intermittent	50	96	8
	2-hr intermittent	100	92	5
	4-hr intermittent	50	96	17
		100	71	0
Danskammer Point	Continuous	55-65	82	29
	2-hr intermittent	55-65	62	15
Bowline Point	Continuous	10-20	98	56
	Continuous	30-50	96	55
	2-hr intermittent	10-20	90	26
	2-hr intermittent	30-50	90	32
	4-hr intermittent	10-20	75	19
	4-hr intermittent	30-50	69	19

(a) Long-term survival at Roseton and Danskammer Point determined 84 hours after impingement; at Bowline POint, 96 hours after impingement.

Source: King et al. 1978

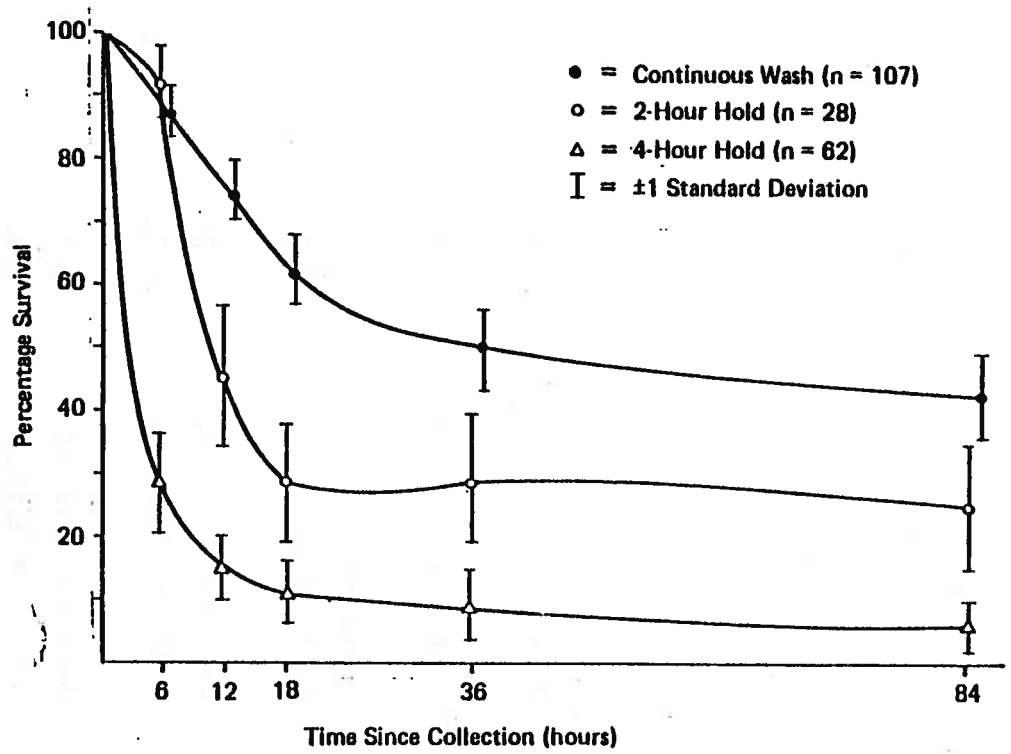

Figure D-12. Probability of surviving impingement for young-of-the-year white perch and adult Atlantic tomcod collected at the Danskammer Point Plant. (From King et al. 1978.)

Company (1977, cited in Cannon et al. 1979) on modified screens installed at the Salem Nuclear Generating Station. The number of impinged fish washed into a fish return sluiceway by a low-pressure spraywash (15 psi) and those washed into a debris sluiceway by a high-pressure spraywash (100 psi) was determined. A total of 3,770 fish were collected, of which only 23.3 percent were removed from the modified screens by the low-pressure spraywash. These results indicate that low-pressure spraywash systems may be ineffective in removing impinged fish from an intake screen. Additional development and evaluation of low-pressure spraywash systems is required, especially when intake screens are subjected to high detrital loading or when large fish are expected to be impinged.

Impingement survival of juvenile chinook salmon on modified vertical traveling screens located on the Columbia River was reported by Page et al. (1976, 1978). Initial survival was 95.6 and 97.2 percent, respectively, in 1976 and 1977. Long-term (96-hour) survival of impinged chinook salmon in 1976 was 94.8 percent; long-term survival was not reported for 1977. In addition, Page et al. estimated initial impingement survival of yellow perch fry to be approximately 92 percent (1978). Additional information on the initial and long-term survival of a variety of species and sizes of fish, impinged on modified vertical traveling screens under a variety of environmental conditions, is needed before the general effectiveness of intake screen modifications designed to reduce impingement mortality can be assessed.

Ecological Analysts (1979) has investigated the use of fine-mesh screen material (0.1-in.; 2.5-mm nylon mesh) in a vertical traveling screen tested under field conditions, in an effort to reduce the number of larval fish entrained into a power plant cooling system. It was concluded that early life stages of striped bass were not able to withstand the stresses imposed by impingement and collection. For larval striped bass (0.6 in.; 15-mm mean length) 96-hour survival was estimated to be 60 percent. For juvenile striped bass (0.8 in, 19 mm or greater in length) survival is expected to exceed 75 percent.

The survival of entrained striped bass passing through the Indian Point Power Plant cooling system was compared with survival of striped bass impinged on the 0.1-in. (2.5 mm) screen at the Indian Point cooling water intake (Ecological Analysts 1979). It was concluded, based on the low retention and survival of eggs and smaller larvae impinged on the fine-mesh traveling screen, that fine-mesh screens would not be

effective for reducing entrainment losses for these life stages. In addition, since the survival of larger larvae and juveniles following entrainment was comparable to that following impingement on the fine-mesh screens, selectively diverting these fish at the intake would not appreciably affect their survival.

Texas Instruments Inc. (1977) examined the survival of young-of-the-year and yearling fish impinged on a modified vertical traveling screen (low-pressure spraywash, dual screenwash sluiceways, fish buckets) equipped with 0.1-in. (2.5-mm) nylon screen mesh. The vertical traveling screen was rotated continuously during the impingement survival studies. Most of the 3,619 fish collected were bay anchovy, blueback herring, and white perch. Mean initial survival of all species was 41 percent, and mean survival of each species is presented in Table D-5. Species-specific survival patterns are apparent, with high long-term survival for species such as white catfish, white perch, and Atlantic tomcod and low survival for species such as bay anchovy and rainbow smelt.

Tomljanovich et al. (1978) conducted a series of laboratory studies designed to examine factors influencing the retention and postimpingement survival of larval fish impinged on fine-mesh screen material. Four square-mesh screen sizes ranging from 0.02 to 0.1 in. (0.5-2.5 mm) were tested; only the 0.02-in. (0.5-mm) screen provided 100 percent retention of the larval fish tested. Post-impingement survival of larval fish was shown to be inversely related to the duration of impingement and dependent on the species and size of fish tested and, to a lesser extent, on water velocity. Survival of larval fish 48 hours after impingement for 16 minutes ranged from less than 1 percent for striped bass to 96 percent for bluegill and smallmouth bass.

The results of studies available to date suggest that the entrainment of larval fish through power plant cooling water systems can be virtually eliminated by use of 0.02-in. (0.5-mm) intake screen mesh. But survival of larval fish passing through a cooling water system may be relatively high (Cada et al. 1979), and survival on a fine-mesh screen that is operated in an intermittent screenwash mode or is subject to high detrital loads may be extremely low (Tomljanovich et al. 1977).

It can be concluded from available information that vertical traveling screens are an acceptable intake screening technology. The vertical traveling screen has demonstrated performance capability and an extensive and impressive history of

TABLE D-5 INITIAL AND LONG-TERM IMPINGEMENT SURVIVAL OF YOUNG-OF-THE-YEAR AND YEARLING FISH IMPINGED ON A CONTINUOUSLY ROTATING FINE-MESH (2.5-mm) MODIFIED VERTICAL TRAVELING SCREEN

Taxon		Age Class[a]	Number Collected	Survival (%)	
Common Name	Scientific Name			Initial	84-hour
Striped bass	*Morone saxatilis*	1	13	85	75
White perch	*Morone americana*	1	223	45	26
White perch	*Morone americana*	2	37	41	18
Atlantic tomcod	*Microgadus tomcod*	1	78	92	58
Atlantic tomcod	*Microgadus tomcod*	2	2	100	100
Blueback herring	*Alosa aestivalis*	1	509	93	19
Blueback herring	*Alosa aestivalis*	2	1	100	0
Bay anchovy	*Anchoa mitchilli*	1	2,415	25	1
Bay anchovy	*Anchoa mitchilli*	2	65	55	13
Alewife	*Alosa pseudoharengus*	1	107	69	6
American shad	*Alosa sapidissima*	1	4	100	0
Hogchoker	*Trinectes maculatus*	1	4	100	100
American eel	*Anguilla rostrata*	2	4	100	100
Banded killifish	*Fundulus diaphanus*	1	1	100	—
White catfish	*Ictalurus catus*	1	27	100	100
Spottail shiner	*Notropis hudsonius*	1	2	100	100
Sea lamprey	*Petromyzon marinus*	1	1	100	100
Largemouth bass	*Micropterus salmoides*	1	1	100	100
Yellow perch	*Perca flavescens*	1	1	100	100
Weakfish	*Cynoscion regalis*	1	3	67	67
Bluefish	*Pomatomus saltatrix*	1	4	100	0
Rainbow smelt	*Osmerus mordax*	1	20	10	0
Rainbow smelt	*Osmerus mordax*	2	4	25	0
Menhaden	*Brevoortia* spp.	1	1	100	—
Gizzard shad	*Dorosoma cepedianus*	2	5	100	0
Unid. clupeids	*Clupeidae*	1	80	51	—
Centrarchids	*Centrarchidae*	1	5	100	100
Northern pipefish	*Syngnathus fuscus*	1	1	0	—

(a) Age class: 1 = young-of-the-year; 2 = yearling or older.

Note: (—) indicates no data available.

Source: Data from Texas Instruments Incorporated 1977.

operational reliability. Two major classes of modifications have been proposed for vertical traveling screens: (1) "fish buckets," a system of troughs and low-pressure sprays to minimize damage to impinged fish, and (2) fine-mesh screens, which impinge ichthyoplankton instead of entraining them. The merits of these modifications depend on the specific biological and engineering conditions at the particular plant where they are to be installed.

D.5.3.2 Fish Return Conveyance Systems

A fish return conveyance system, which is generally used in association with vertical traveling intake screens, is simply an open or enclosed trough which returns organisms and debris removed from the intake screens to the receiving water body. The sluiceway troughs are typically made of concrete or steel and utilize either gravity-induced flow or a pumping system. The design of conveyance systems has not been standardized or extensively studied, so they vary widely from one power plant intake to the next. The limited data available on the relationship between conveyance system design and survival of organisms returned to the receiving water body make it difficult to identify and quantify the effects of various design parameters on the biological effectiveness of conveyance systems. The difficulties in extrapolating generalizations from one site to another at the present time are particularly important in considering the design of a fish return conveyance system that would be used with a fine-mesh intake screen for removing egg and larval fish from the intake.

Cada et al. (1979) proposed a set of recommendations concerning fish return conveyance sytems which can be expected to contribute to the survival of fish being returned to the water body. These recommendations include:

1. maintain sufficient water levels and velocities to expedite return;
2. avoid hydraulic turbulence in the conveyance wherever possible;
3. prevent ice formation in the conveyance in cold weather;
4. maintain smooth surfaces in the conveyance;
5. enclose the conveyance to minimize avian predation;
6. consider multiple release location to minimize predation at the outfall;
7. prevent re-entrapment;
8. avoid thermal shock; and
9. separate debris sluiceways from fish sluiceways.

Because sluiceway and pipeline returns are usually part of a physical barrier technology and an intake design technology, their implementation must be considered on a site-specific basis.

D.5.3.3 Fish Pump Systems

Fish pump removal systems have been developed in response to entrapment problems in which fish are known to be concentrated in a confined area of the intake structure and are susceptible to removal using a pump system. The first application of a fish pump removal system at a power plant intake occurred at the Contra Costa Power Plant (Kerr 1953). Prior to installing the fish pump system, the behavioral reponses of fish to various pump collector (intake) designs were tested in a number of various velocity gradients. Based on observations and tests, a zone of fish concentration between the vertical traveling intake screen and a curtain wall was located within the intake structure. The placement of a large intake collector (similar in design to the inlet of a vacuum cleaner) in the zone of fish concentration, in combination with low suction velocities, resulted in high diversion efficiency and high survival of all species removed from the intake structure. Kerr (1953) estimated the survival of fish following passage through the fish pump system to be greater than 98 percent. Recent studies of the Contra Costa fish pump system (PGandE 1981) indicate that the number of fish impinged on the power plant intake screens is reduced by approximately 80-98 percent by the fish pump removal system. Following passage through the fish pump, initial and latent (96-hour) survival of juvenile striped bass were approximately 100 percent and 85-95 percent, respectively (PGandE 1981). Presently available data indicate that the fish pump systems at the Contra Costa Power Plant effectively reduce the numbers of fish impinged while maintaining high survival of those fish returned to the receiving water body. Some problems have been encountered with maintenance and with loss of pump prime at very low tides.

Other fish pump installations have had mixed success. Studies of the Monroe Nuclear Generating Station estimate removal efficiencies of 52-99 percent and gizzard shad survival of 10-99 percent (Cannon et al. 1979). Southern California Edison reports a lack of success at its Huntington Beach and El Segundo generating stations (Stipanov 1976).

Mussalli and Taft (1977, cited in Cannon et al. 1979) have investigated the use of jet

pumps as an alternative to either single-port or dual-port centrifugal pumps for use in fish removal system. Based on results of laboratory testing, a peripheral-type jet pump fish return system has been designed for inclusion into the intake structures of the Nine Mile Point Nuclear Station Unit 2 and Oswego Steam Station Unit 6, both located on Lake Ontario.

Available data indicate that fish pump return systems may substantially reduce the entrapment and impingement of fish at many cooling water intake structures, but data are insufficient to permit evaluation of the conflicting nature of many of the findings. Additional research is needed to identify the specific behavioral responses of fish, which can be used to concentrate organisms within a confined area of an intake structure, and the design paramenters for a fish pump system which will minimize fish mortality during removal from an intake.

D.5.3.4 Fish Basket Collectors

Fish basket collectors were developed for use in intake structures where fish pump return systems had not been effective. Fish basket collectors are designed to allow fish to congregate in an area directly over the collection basket, which can then be raised to capture and return the entrapped fish to the receiving water body.

Testing of fish basket collectors has been conducted at the Alden Laboratory (Stone & Webster Engineering Corporation 1975) and at the Redondo Beach test facility of the Southern California Electric Company (Schuler and Larson 1975). In both series of tests, a fish basket collector provided acceptable removal of fish. A full-scale fish basket collector has been developed for use at the San Onofre Nuclear Generating Station. To date, however, no operating experience is available on the effectiveness of fish basket collectors at power plant intakes.

Fish basket collectors are not expected to be effective for all species and sizes of fish. In addition, the hydraulic conditions within an intake structure, debris loading, and the configuration and design of an intake structure are expected to influence the effectiveness of fish basket collectors for removing entrapped fish. Fish basket collector systems are not generally applicable to shoreline intake structures.

D.5.3.5 Fish Bypasses and Escape Channels

Two intake structure configurations have been developed which provide bypasses or escape channels to minimize entrapment within the intake structure. A shoreline intake structure has been designed, and is currently in use at several power plants, including the Pittsburg and Contra Costa power plants, in which the vertical traveling intake screens are located flush with the shoreline. In this shoreline intake configuration, concrete dividing walls between forebays have been eliminated and bar racks have been installed as a cage-type structure on both the front and lateral sides of the intake structure. The configuration allows a lateral flow of water across the face of the intake screens tending to guide entrapped fish away from the intake screens and out of the intake structure.

A second proposed intake structure configuration utilized vertical traveling screens oriented at an angle to the inflow of water (Section D.5.2.10). The angled intake screens are intended to divert entrapped fish laterally across the face of the screen and into a bypass channel. This approach combines the advantages of intake screening with the diversion capability typical of louver intake structures.

Results of prototype testing (Taft and Mussalli 1978) indicate that high diversion efficiencies and survival rates for fish diverted by angled screens are possible. At present, however, no full-scale tests have been conducted to evaluate the efficiency of angled intake screens under field operating conditions.

One aspect of the use of angled screens to divert fish into a bypass which requires detailed consideration is the hydraulic flow pattern associated with the bypass. Concentration of fish in a bypass channel requires a nonturbulent flow in order to guide fish into the bypass. Backflows, eddies, and rapid changes in velocity of the water approaching a bypass will stimulate avoidance responses in fish, which interfere with their transit past the intake screens and into the bypass. A gradual acceleration of flow in the direction of the bypass provides the optimal hydraulic conditions for fish diversion (Taft et al. 1976).

The shoreline intake structure with fish escape channels has demonstrated performance at power plant intakes. The angled-screen diversion appears promising based on prototype evaluations; however, full-scale operation under field conditions is

needed to confirm the effectiveness of diverting entrapped fish. The use of fine-mesh screen material on an angled-screen diversion may be effective in diverting larval fish, but no data are currently available. Angled-screen diversions are applicable for use in either offshore intake structures with a fish return system, or in shoreline intake structures.

D.6 INTAKE MAINTENANCE AND OPERATION

Operational modifications of power plant cooling systems to reduce entrainment and impingement losses focus on reducing the exposure of organisms to physical, chemical, and thermal stresses.

Physical stresses result when an organism is impinged on intake screens or when an entrained organism is exposed to shear and pressure forces or mechanical abrasion during cooling system transit. Physical stresses can be reduced by maintenance dredging in front of the intake to keep approach velocities low, or by reducing the volume of cooling water used. Cooling water volume reduction can generally be achieved by either limiting the operation of circulating water pumps or by complete curtailment of cooling system operation and thus, also, electrical generation. At the present time, insufficient data are available to specify design parameters for cooling system components such as circulating water pumps which can assure minimization of the physical stress on entrained organisms.

Chemical stresses on entrained organisms result primarily from exposure to biocides used in cooling systems to control biological fouling. Operational modifications to reduce chemical stresses on entrained organisms focus primarily on minimizing the frequency, duration, and concentration of chemicals used for biofouling control.

Thermal stresses on entrained organisms result from exposure to elevated temperatures in the condensers and cooling water discharge. Operational modifications to minimize thermal stresses on entrained organisms focus on minimizing the temperature elevation (delta-T) and time of exposure during cooling system transit.

Qualitative conclusions are presented in Table D-6.

TABLE D-6 SUMMARY OF EXPERIENCE WITH MAINTENANCE AND OPERATIONAL MEASURES FOR REDUCING ORGANISM LOSSES AT COOLING WATER INTAKES

Technology	Reduces Physical Stress	Reduces Chemical Stress	Reduces Thermal Stress	Operationally Reliable	Requires Outage Time to Implement	Comments
Dredging in front of intake	●	○	○	●	NO	
Reducing circulating water flows	●	○	⊗	○	YES	Increases delta-T
Curtailing plant operation	●	●	●	⊗	YES	Produces no electricity
Establishing biofouling procedures	○	●	○	●	NO	
Modifying circulating water system	●	○	○	○	YES	No mechanical data
Lowering the temperature differential	○	○	●	○	YES	Retrofit of major plant systems (Section C.2.3)

Legend

● Demonstrated or expected improvement

○ No change or unknown

⊗ Demonstrated or expected deterioration

D.6.1 Dredging

Sediment transport, sediment deposition, and littoral drift are frequently encountered in the vicinity of an intake structure. Such deposition increases approach velocities. Dredging probably represents the simplest and most cost-effective means of minimizing physical stresses on aquatic organisms by ensuring the design velocities are maintained and uniformity of velocity gradients is maximized.

D.6.2 Reducing Circulating Water Flows

Reduction in the number of circulating water pumps in operation has been suggested as an operational strategy for reducng cooling water volume and intake approach velocities and hence reducing the rate of entrainment and impingement. A reduction in cooling water flow rate during full-load generation is expected to result in losses in both generating capacity and thermal transfer efficiency, with more significant losses at higher ambient inlet water temperatures. Changes in condenser backpressure resulting in reduced turbine cycle thermal efficiency, and increased temperature differentials through the condenser system (delta-T) are expected when cooling water flow rates are reduced during generation. Although a reduction in cooling water volume is expected to result in a decrease in the number of entrained organisms, the associated increase in delta-T may substantially reduce the survival of those organisms which are entrained.

The biological effectiveness of reducing entrainment and impingement losses by reducing the volume of cooling water withdrawn has not been extensively examined. Quantitative information is needed on the relative contribution of physical stresses and thermal stresses for each species and life stage being considered before the benefits of reduced losses resulting from reduced cooling water volume can be determined. Insufficient data are available to optimize the balance between cooling water volume and delta-T for most power plants and fish species.

The regulation of cooling system volume and delta-T becomes increasingly complex when a diverse array of species and life stages, each with individual and unique tolerances to physical and thermal stresses, are present at an intake simultaneously. In addition, during those periods when the delta-T and absolute discharge water temperature exceed the upper lethal limit for thermal tolerance of a species, it may

be desirable to increase the volume of cooling water and thus decrease the discharge water temperature below a lethal limit. The alternative approach of reducing cooling water volume to a minimum and increasing the delta-T above the thermal tolerance of a species may result in an absolute increase in the total number lost to the population.

The biological complexities of cooling system operational modifications are compounded by engineering constraints. Many circulating water pumps do not have the capability to be throttled or regulated. In addition, reducing the volume of cooling water reduces the thermal efficiency of the cooling system, resulting in decreased generating capacity of the unit. Therefore, considerations of reduced generation in perspective with electrical demand are needed on a site-specific basis for a specific period of time.

Eliminating the use of circulating water pumps when loads are low may not only reduce biological losses, but may also reduce the cost required to operate one or more circulating water pumps.

D.6.3 Curtailing Plant Operation

Entainment and impingement losses of aquatic organisms at a power plant can be entirely eliminated by shutting down a particular unit or units. Cooling system curtailment, however, also results in curtailment of electrical generation, and hence involves implications of cost and of reduced system-wide capability for meeting electrical demand.

Cooling system curtailment may be feasible if the seasonal period of maximum abundance of a critical species is short and coincides with a period during the year when electrical generating capacity exceeds demand. The cost implications from loss of generation, electrical transmission from alternative sources, and availability of alternative sources of electricity (e.g., hydroelectric), must all be evaluated on a site-specific basis. Coordinating plant activities, such as scheduled outages for maintenance or refueling, with periods of high organism densities could also be considered.

Coordinating cooling system curtailment with periods of highest abundance of a

critical species requires an ongoing biological monitoring program at each site to identify the period of maximum abundance. Coordinating scheduled outages with the temporal distribution of a particular species is also difficult in that outages require precise scheduling, generally months in advance, while the temporal pattern of species abundance may not be as predictable as necessary.

The feasibility of utilizing cooling system curtailment to minimize biological losses must be examined on a site-specific and species-specific basis.

D.6.4 Biofouling Procedures

Schubel and Marcy (1978) have made the following biofouling recommendations for minimizing mortality associated with chemical stresses (most commonly from chlorine) during entrainment:

1. chlorinate intermittently;
2. reduce the concentration of free chlorine to the lowest possible level;
3. minimize transit time through the cooling system and thus exposure to the biocide; and
4. maximize the rate of dilution of the discharge water.

The use of alternative biocidal agents does not appear to be an effctive method of minimizing mortality of nontarget entrained organisms. Mechanical fouling control techniques have not proven to be completely effective in controlling fouling accumulations in power plant cooling systems. Testing of alternative fouling control techniques such as the use of "no foul" is currently under way.

D.6.5 Modifying Cooling Water System Components

Structural modification of cooling system components in an attempt to reduce the mortality of entrained organisms is not effective at the present time. Insufficient quantitative data are available to isolate specific sources of mortality within a cooling water system. Design parameters for specifying pressure regimes, circulating water pump design and operation (e.g., rpm, impeller design), tolerable shear stresses, and cooling system designs for minimizing mechanical abrasion have not been developed.

Structural modifications have focused on the design and operation of intake structures, with particular emphasis on intake screening systems, which will reduce stresses imposed on impinged organisms (Section D.5.2). To date, greater effort has been devoted to developing methods for excluding fish eggs and larvae from cooling systems than on developing specific criteria for structural modifications which would reduce stresses imposed on entrained organisms.

D.6.6 Regulation of Thermal Elevations (delta-T)

To minimize mortality of entrained organisms resulting from thermal stress, Schubel and Marcy (1978) recommend that a once-through cooling system be operated in such a way that:

1. the minimum delta-T is used that is technologically practicable, based on cooling system design and load;
2. transit time through the cooling system is minimized; and
3. the cooling water is discharged in a way that will promote rapid mixing and dilution.

Regulation of thermal elevations requires the integration of information on plant generating loads and delta-T, and species- and size-specific thermal tolerance data for evaluating the effectiveness of various cooling system operating modes for reducing entrainment losses.

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Appendix E

Data Summary

E.1 Entrainment Abundance

E.2 Impingement Abundance

E.3 Cooling Water Temperatures

E.1 ENTRAINMENT ABUNDANCE DATA SUMMARY

Table E.1-1 Numbers and Percentage Composition of Entrained Organisms Collected at the Pittsburg Power Plant, March 1978 - July 1979.

Table E.1-2 Estimated Total Annual Entrainment of Ichthyoplankton and Macrozooplankton at the Pittsburg Power Plant under Actual Cooling System Operation and Under the Assumption of Continuous Normal Operation, March 1978 - March 1979, April-July 1979.

Table E.1-3 Summary of the Length Statistics for Selected Species of Fish Collected in Entrainment Samples at the Pittsburg Power Plant, March 1978 - July 1979.

Table E.1-4 Daily Entrainment Abundances of Organisms Collected at the Pittsburg Power Plant, March 1978 - July 1979.

TABLE E.1-1 NUMBERS AND PERCENTAGE COMPOSITION OF ENTRAINED ORGANISMS COLLECTED AT THE PITTSBURG POWER PLANT, MARCH 1978 - JULY 1979

Computer Code	Definition
GROUP = FISH	Data summary for fish collected in entrainment samples
GROUP = INVERTS	Data summary for macroinvertebrates collected in entrainment samples
STATION = UNITS 5-6	Samples collected at the Units 5 and 6 discharge gate well
NAME	Common name or lowest taxonomic level identifiable if fish; scientific name or lowest taxonomic level identifiable if invertebrate. Organisms identifiable as a specific group but not distinguishable to a lower taxonomic level are identified as a type (A, B, etc.) within a taxon
NUMBER	Total number of each taxon collected during the entrainment abundance sampling
PERCENTAGE	Percentage composition of each taxon based on numerical abundance

TABLE E.1-2 ESTIMATED TOTAL ANNUAL ENTRAINMENT OF ICHTHYOPLANKTON AND MACROZOOPLANKTON AT THE PITTSBURG POWER PLANT COOLING SYSTEM OPERATION AND UNDER THE ASSUMPTION OF CONTINUOUS NORMAL OPERATION, MARCH 1978 - MARCH 1979, ACTUAL APRIL-JULY 1979.

Computer Code	Definition
STATION = PBCD01	Estimate for Units 1-4 cooling water system
STATION = PBDU06	Estimate for Unit 5 and 6 cooling water system
STATION = PBDU07	Estimates for Unit 7 cooling water system
E	Estimated number of each taxon entrained over the period indicated, based on actual cooling water flows.
DESIGN	Estimated number of each taxon entrained over the period indicated; based on continuous normal cooling water flows.
LIFE STG	Organism life stage. E = eggs; Z = non-eggs, including yolk-sac larval, larval, and juvenile life stages of fish and all life stages of macroinvertebrates.
SE (SED)	Standard error associated with the entrainment estimate

TABLE E.1-3 SUMMARY OF THE LENGTH STATISTICS FOR SELECTED SPECIES OF FISH COLLECTED IN ENTRAINMENT SAMPLES AT THE PITTSBURG POWER PLANT, MARCH 1978 - JULY 1979

Computer Code	Definition
STATION = UNITS 5-6	Samples collected at the Unit 6 discharge gate well
GROUP = FISH	Data summary for fish collected in entrainment samples
NAME	Species name or lowest identifiable taxon
N	Number of measurements
MEAN	Mean length (mm) of the fish measured
STD	Standard deviation around the mean (mm) of the fish measured
MIN	Length of shortest fish measured (mm)
MAX	Length of longest fish measured (mm)

TABLE E.1-4 DAILY ENTRAINMENT ABUNDANCES OF ORGANISMS COLLECTED AT THE PITTSBURG POWER PLANT, MARCH 1978 - JULY 1979

Computer Code	Definition
GROUP = FISH	Summary table for fish
GROUP = INVERTS	Summary table for macroinvertebrates
DATE = 78-03-21	Year-month-day of sample
NAME	Species name or lowest identifiable taxonomic level
LIFE STAGE	Organism life stage
NUMBER	Total number of organisms collected during the 24-hour sampling period
DENSITY	Mean number of organisms collected per thousand cubic meters of cooling water sampled

E.2 IMPINGEMENT ABUNDANCE DATA SUMMARY

Table E.2-1 Numbers, Percentage Composition, and Weights of Impinged Organisms Collected at Units 1-4 and Units 5 and 6 of the Pittsburg Power Plant, March 1978 - November 1979.

Table E.2-2 Estimated Total Impingement of Fish and Macroinvertebrates at the Pittsburg Power Plant Under Actual Cooling System Operation Under the Assumption of Continuous Normal Operation, March 1978 - March 1979, May-November 1979.

Table E.2-3 Summary and Monthly Length Statistics for Fish Collected in Impingement Samples at the Pittsburg Power Plant, March 1978 - November 1979.

Table E.2-4 Daily Impingement Abundance of Organisms Collected at the Pittsburg Power Plant, March 1978 - November 1979.

TABLE E.2-1 NUMBERS, PERCENTAGE COMPOSITION, AND WEIGHTS OF IMPINGED ORGANISMS COLLECTED AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MARCH 1978 - NOVEMBER 1979

Computer Code	Definition
STATION: CP01	Units 1-4 intake screens
STATION: CP02	Units 5 and 6 intake screens
TYPE: INVERTS	Impingement summary for invertebrates
TYPE: FISH	Impingement summary for fish
TAXON	Common name if fish species, scientific name if fish genus or invertebrates
FRAGMENT	Indicates that fragments of organisms were collected, fragments were not counted and were not used in establishing percentage composition
NUMBER	Total number of each species collected in all the 24-hour impingement samples in a study period
CUM PER	Cumulative percentage composition of a given species collected plus all more abundant species, by intake, based on numbers
PERCENT	Percentage composition of species collected, by intake, based on numbers
WEIGHT	Composite weight of each species

TABLE E.2-2 ESTIMATED TOTAL IMPINGEMENT OF FISH AND MACROINVERTEBRATES AT THE PITTSBURG POWER PLANT UNDER ACTUAL COOLING SYSTEM OPERATION AND UNDER THE ASSUMPTION OF CONTINUOUS NORMAL OPERATION, MARCH 1978 - 1979, MAY-NOVEMBER 1979

Computer Code	Definition
NAME	Common name if fish; scientific name if invertebrate
STATION = PBCP01	Data summary for impingement at the Units 1-4 intake screens
STATION = PBCP02	Data summary for impingement at the Units 5 and 6 intake screens
I	Total estimated impingement for period indicated under actual flows
IR	Total estimated impingement for period indicated under the assumption of continuous normal flows
DESIGN (1979)	Total estimated impingement for period indicated under the assumption of continuous normal flows
SE	Standard error associated with the estimated impingement under actual flows
SER	Standard error associated with impingement under the assumption of continuous normal flows

TABLE E.2-3 SUMMARY AND MONTHLY LENGTH STATISTICS FOR FISH COLLECTED IN IMPINGEMENT SAMPLES AT THE PITTSBURG POWER PLANT, MARCH 1978 - NOVEMBER 1979

Computer Code	Definition
STATION = ALL	Combined Units 1-4 and Units 5 and 6 intake screens
NAME	Species name; common name if fish species, scientific name if fish genus
YEAR	Year of sample (In the summary tables, 1978 = March 1978 - March 1979; 1979 = May-November 1979)
MONTH	Month of sample: 01 = January, etc.
N	Number of measurements
MEAN	Mean length (mm) of the organisms measured over the month
STD	Standard deviation around mean (mm) of organisms measured during the month
MIN	Length (mm) of shortest organism measured
MAX	Length (mm) of largest organism measured

TABLE E.2-4 DAILY IMPINGEMENT ABUNDANCE OF ORGANISMS COLLECTED AT THE PITTSBURG POWER PLANT, MARCH 1978 - NOVEMBER 1979

Computer Code	Definition
STATION: CP01	Units 1-4 intake screens
STATION: CP02	Units 5 and 6 intake screens
DATE: 78-05-10	Year-month-day of sample
TYPE: FISH	Summary for impinged fish
TYPE: INVERTS	Summary for impinged macroinvertebrates
TAXON	Species name; common name if fish, scientific name if invertebrate
FRAGMENT	Represents organism fragment. Fragments are not counted and not used in establishing percentage composition
NUMBER	Total number of organisms collected during 24-hour sampling period
PERCENT	Percentage of daily catch that the organism represents, based on numbers
CUM PER	Cumulative percentage of catch, based on numbers
WEIGHT	Total weight (g) of organisms collected over 24-hour period

E.3 COOLING WATER SYSTEM TEMPERATURE DATA SUMMARY

Table E.3-1 Daily Mean, Minimum, and Maximum Intake and Discharge Temperatures at Units 1-4 and Units 5 and 6 of the Pittsburg Power Plant, March 1978 - November 1979.

TABLE E.3-1 DAILY MEAN, MINIMUM, AND MAXIMUM INTAKE AND DISCHARGE TEMPERATURE AT UNITS 1-4 AND UNITS 5 AND 6 OF THE PITTSBURG POWER PLANT, MARCH 1978 - NOVEMBER 1979

Computer Code	Definition
STR CODE = STR NO =	IU 01 = Units 1-4 intake IU 06 = Unit 6 intake CD 01 = Units 1-4 common discharge DU 05 = Unit 5 discharge DU 06 = Unit 6 discharge
DATE	24-hour day of temperature recording, comprising eight readings at 3-hour intervals
MEAN	Mean of eight readings
MIN	Lowest temperature reading
MAX	Highest temperature reading
STDERR	Standard deviation of day's mean temperature
MISSING	Number of observations missing
N	Number of temperature readings in day