CHAPTER II

AFFECTED ENVIRONMENT

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INTRODUCTION

The Central Valley Project (CVP) provides water service to much of California's Central Valley and affects major anadromous fisheries resources throughout most of California. Water storage and release patterns from 20 CVP reservoirs with a combined storage capacity of more than 11 million acre-feet, flow changes from eight powerplants, and more than 500 miles of major canals, tunnels, and aqueducts have adversely affected some species of fish and contributed to dramatic declines from historical levels in certain fish populations.

The CVPIA (Public Law 102-575, Title 34) amends the authorization of the U.S. Bureau of Reclamation's (Reclamation's) CVP to include fish and wildlife protection, restoration, and mitigation as project purposes having equal priority with other CVP purposes, such as navigation, flood control, irrigation, municipal water supply, power generation, and recreation. The CVPIA's emphasis on fisheries is embodied in four of its six primary purposes, as follows:

- protect, restore, and enhance fish, wildlife, and associated habitats in the Central Valley and Trinity River basins of California;
- address impacts of the CVP on fish, wildlife, and associated habitats;
- contribute to California's interim and long-term efforts to protect the San Francisco Bay/Sacramento-San Joaquin Delta (Delta) estuary; and
- achieve a reasonable balance among competing demands for CVP water, including the requirements of fish and wildlife, agricultural, municipal, and industrial and power contractors.

This chapter identifies the major factors affecting fisheries resources within the CVPIA study area. These factors include habitat modification and/or loss, species introductions, and overharvest. Activities that potentially significantly influence these factors include operation of federal, state, and other water supply facilities; agriculture; navigation; urban development; grazing; mining; and timber harvest.

Historical conditions and population trends for fisheries resources are presented for the period from 1850 to 1966. The goal of the CVPIA is to sustain natural populations of anadromous fish at double the average levels attained during 1967 to 1991. Accordingly, recent conditions for this period are presented for the following species and communities:

- chinook salmon
- steelhead trout
- green sturgeon and white sturgeon
- striped bass
- American shad
- delta and longfin smelt
- Sacramento splittail
- reservoir fisheries community
- bay fisheries communities
- estuarine invertebrate communities
- special-status aquatic species

Information on current instream flow and other fisheries-related regulatory requirements appears in the Surface Water Supplies and Facilities Operations Technical Appendix.

STUDY AREA

For most resources, the study area potentially affected by the CVPIA was divided into separate geographic regions. However, geographic groupings are not always practical in relation to anadromous species, which occur in several geographic areas during their life stages. Consequently, information in this technical appendix is presented primarily by species, not by geographic area. The following discussion of geographic regions is provided for consistency with other technical appendices and to give a brief synopsis of key features in each geographic region. The regions are consistent with regions used for other resources, with the following exceptions:

- San Francisco Bay Region was added to address fisheries habitats in the San Pablo, Central, and South bays, which are distinct from habitats in the Delta Region.

- North Coast and Central Coast regions have been modified to include not only the interior region but also the offshore chinook salmon commercial and sport fisheries.

Sacramento River Region

The Sacramento River is the largest river system in California, yielding 35 percent of the state's water supply. Most of the Sacramento River flow is controlled by Reclamation's Shasta Dam, which can store up to 4.5 million acre-feet. River flow is augmented in average water years by transferring up to 1 million acre-feet of Trinity River water through Clear Creek and Spring Creek tunnels to Keswick Reservoir. Reclamation's Shasta-Trinity Division of the CVP includes Shasta, Keswick, Trinity, Lewiston, and Whiskeytown dams; Red Bluff Diversion Dam (RBDD); and the Tehama-Colusa and Corning canals. (Reynolds et al., 1993.)

The Sacramento River supports a variety of anadromous species, including four races of chinook salmon, steelhead trout, white sturgeon and green sturgeon, striped bass, American shad, Pacific lamprey, and river lamprey. Fall-run chinook salmon is the predominant salmon run. Four distinct chinook salmon races exist in the Sacramento River: fall, late fall, winter, and spring runs. The fall-run provides the largest portion of the state's sport and commercial catch. More

than 90 percent of the Central Valley salmon population spawn in the Sacramento River system (Kjelson et al., 1982). The Sacramento River also supports one of the largest contiguous riverine and wetland ecosystems in the Central Valley. On June 16, 1993, the National Marine Fisheries Service (NMFS) designated critical habitat for Sacramento winter-run chinook salmon under the Endangered Species Act (ESA), which will include the Sacramento River from Keswick Dam (river mile [RM] 302) to the Golden Gate Bridge, and to the Bay Bridge in the south Bay. The winter-run chinook salmon is designated as an endangered species under the ESA. The Central Valley steelhead trout have been proposed for federal listing as endangered (61 FR 41541, August 9, 1996). The Feather and American river basins are major tributaries that support chinook salmon, steelhead trout, striped bass, and American shad populations. Several major east side tributaries, such as Battle, Deer, Mill, Butte, and Big Chico creeks, maintain small runs of spring-run chinook salmon, a declining fisheries resource. Three salmon and steelhead hatcheries (Coleman National Fish Hatchery, Feather River Fish Hatchery, and Nimbus Fish Hatchery) were constructed to mitigate the habitat lost from dam closure, providing major hatchery production of fall-run chinook salmon and steelhead trout.

The following are several major problems in the Sacramento River that affect anadromous fish:

- RBDD impedes upstream migrating adults and downstream migrating juveniles;
- Glenn-Colusa Irrigation District (GCID) pumps divert up to 3,000 cubic feet per second (cfs) and approximately 1 million acre-feet of water annually through inadequate fish screens;
- Anderson-Cottonwood Irrigation District's (ACID's) seasonal flashboard dam in Redding diverts up to 400 cfs, and impedes upstream migrating adults and downstream migrating juveniles;
- access to historical spawning and rearing habitat is restricted;
- hundreds of small unscreened diversions entrain fish;
- bank protection projects reduce available remaining habitat;
- high water temperatures associated with reservoir storage decrease fish habitat;
- gravel mining operations and water storage facilities reduce recruitment of salmon spawning gravel;
- discharges of chemical waste from industrial, municipal, and agricultural sources decrease the quality of fish habitat; and
- chronic contamination from numerous and widespread sources (Reynolds et al., 1993).

Sacramento Valley CVP reservoirs, primarily Shasta and Folsom lakes, provide an important component of freshwater angling. These reservoirs have greatly increased warm-water game fish production (primarily centrarchids). Periodic stocking to meet sport fishing demands is usually

required to maintain some game fish populations because reproductive success is usually limited by water-level fluctuations and declining water levels during late spring.

San Joaquin River Region

The 250-mile-long San Joaquin Valley constitutes the southern portion of the Central Valley. On the arid west side of the basin, relatively small intermittent streams drain the Coast Ranges but rarely reach the San Joaquin River. On the east side, numerous streams and three major rivers drain the western Sierra Nevada and contribute flow to the San Joaquin River. The major east side tributaries, the Stanislaus, Tuolumne, and Merced rivers, all support fall-run chinook salmon spawning and rearing and have large impoundments. Friant Dam, operated by Reclamation for flood control, irrigation, and power generation, is the major storage reservoir on the mainstem San Joaquin River and stores up to 520,000 acre-feet of water in Millerton Lake. (Reynolds et al., 1993.)

Historically, the upper San Joaquin River provided habitat for the southernmost stocks of spring- and fall-run chinook salmon, and most likely steelhead. Spring-run chinook salmon runs were eliminated from the basin with the closure of Friant Dam in 1949, and fall-run chinook salmon and steelhead trout escapements have been extremely low in recent years. Other anadromous species rarely migrate up the San Joaquin River, but all anadromous species in the Central Valley could be found in the lower San Joaquin River near the Delta. The Merced River Fish Hatchery, the only hatchery in the region, produces a relatively small number of fall-run chinook salmon compared to production from Sacramento River fish hatcheries.

Friant Dam eliminated spring- and fall-run chinook salmon from the upper San Joaquin River and adversely affected anadromous fish runs in downstream tributaries by significantly reducing total basin outflow. Water quality severely altered by pesticides and salts from agricultural drainage has also adversely affected anadromous fisheries. Typical flow and water quality conditions in the Delta, particularly the south Delta, are also detrimental to the survival of San Joaquin salmon smolts because of low San Joaquin River inflow and high exports from Delta water diversions. Salmon straying into west side canals, numerous small- and medium-sized diversions, elevated water temperatures, channel dredging, waste discharges, and low dissolved oxygen are other factors adversely affecting anadromous fisheries resources. (Reynolds et al., 1993.)

Operations of CVP reservoirs, including Millerton Lake and New Melones, result in conditions limiting abundance of reservoir species similar to conditions described for Sacramento Valley reservoirs.

Sacramento-San Joaquin Delta Region

The Delta and Suisun Bay, on the western edge of the Delta, are located at the confluence of the Sacramento and San Joaquin rivers and represent the most important, complex, and controversial geographic area for both anadromous fisheries production and distribution of California water resources for numerous beneficial uses. Approximately 42 percent of the state's annual runoff flows through the Delta's maze of channels and sloughs surrounding 57 major reclaimed islands and nearly 800 unleveed islands (Water Education Foundation, 1992a). The Delta's channels are

used to transport water from upstream reservoirs to the south Delta, where federal and state facilities (Tracy Pumping Plant and Harvey O. Banks Delta Pumping Plant, respectively) pump water into CVP and State Water Project (SWP) canals, respectively.

Environmental conditions in the Delta depend primarily on the physical structure of Delta channels, inflow volume and source, Delta Cross Channel (DCC) operations, Delta exports and diversions, and tides. The CVP affects Delta conditions primarily through control of upstream storage and diversions, Delta exports and diversions, and DCC operations. These factors also determine outflow and the location of the entrapment zone, an area of high organic carbon that is critically important to a number of fish and invertebrate species, as well as to the overall ecology of the Delta and Suisun Bay. In addition to these physical factors, environmental conditions such as water temperature, predation, food production and availability, competition with introduced exotic fish and invertebrate species, and pollutant concentrations, all contribute to interactive, cumulative conditions that have substantial effects on Delta fish populations.

An estimated 25 percent of all warm-water and anadromous sport fishing and 80 percent of the state's commercial fishery depend on species that live in or migrate through the Delta. The Delta serves as a migration path for all Central Valley anadromous species returning to their natal rivers to spawn. Adult chinook salmon move through the Delta in most months of the year. Salmon and steelhead juveniles depend on the Delta as transient rearing habitat during migration through the system to the ocean and could remain for several months, feeding in marshes, tidal flats, and sloughs. All life stages of striped bass and American shad are found in the Delta; approximately 45 percent of striped bass spawn in the Delta, as do some American shad. Numerous resident native and introduced species live in the Delta year-round, including native delta smelt (a native species listed as threatened under the federal ESA) and Sacramento splittail (a native species proposed for listing as threatened under the federal ESA).

The Calaveras, Mokelumne, and Cosumnes (a tributary to the Mokelumne) rivers are minor tributaries to the Delta, supporting small populations of chinook salmon and steelhead. Chinook salmon are not consistently found in the Calaveras River. The Cosumnes River currently supports an average annual run of about 100 fall-run chinook salmon, and the Mokelumne River, including the Mokelumne River Fish Hatchery, supports an average run of less than 1,000 salmon. Water diversions, inadequate or nonexistent screening facilities, barriers to fish migration, and high water temperatures adversely affect salmon and steelhead production. (Reynolds et al., 1993.)

San Francisco Bay Region

The San Francisco Bay Region is predominantly developed for urban and industrial uses with limited anadromous fisheries. The area contains a number of small streams and reservoirs used primarily for domestic water supply. All anadromous species use these habitats, with the exception of some American shad and striped bass that complete their entire life cycles within the Delta and upstream. The four races of chinook salmon and steelhead trout migrate as adults from the Pacific Ocean, through the San Francisco Bay (Bay) and into their natal rivers, while salmon and steelhead smolts migrate downstream through the Bay on their way to the ocean. Most of the creeks and reservoirs are highly affected by the urban environment.

Remnant steelhead trout populations inhabit a few streams. Associated native nongame species include river lamprey, hitch, California roach, Sacramento sucker, threespine stickleback, and several sculpin species. Numerous introduced warm-water species, as well as seasonally introduced rainbow trout, can be found in the small reservoirs in this area.

Probably more than 200 fish species, mostly marine, exist in the Bay (Miller and Lea, 1972). The Bay is an important nursery area for marine and estuarine species, including bay shrimp, Dungeness crab, northern anchovy, Pacific herring, and English sole. The Bay provides a protective, highly productive habitat that enhances early survival and growth of these species. Predation and parasitism could be reduced because of increased turbidity, variable temperatures, and variable salinity. Nutrients from river inflow and other factors enhance food production and availability in the Bay.

Delta outflow is the principal variable affecting the Bay that could be changed by implementing the CVPIA. Delta outflow influences abundance and distribution of fish and invertebrates in the Bay through changes to salinity, currents, nutrient levels, and pollutant concentrations. The response of organisms to outflow depends on species and life stage. The variability in the response of organisms to variable volume outflow levels is important in the dynamics of the estuarine community. The effect of Delta outflow on Bay organisms is determined by its timing, magnitude, and duration. The cause-and-effect relationship between Delta outflow and organism abundance and distribution is complex and often dictated by a chain or web of events rather than by specific, direct effects. Although correlations between flows and organism abundance have been identified, the mechanisms of the relationships are largely unknown. Water residence time in the Bay determined by tides, local inflow, Delta outflow, and bathymetry also affect fish species abundance (Smith, 1987).

Tulare Lake Region

The Tulare Basin is located south of the San Joaquin River Basin and supports predominantly irrigated agriculture in the lower elevations. The Tulare Basin is normally considered a separate drainage basin, but during wet years it has historically contributed occasional flood overflows and subsurface flows to the San Joaquin River. Three rivers drain into the Tulare Basin: the Kings, Kaweah, and Tule. During the late spring, summer, and early fall irrigation seasons, however, all or nearly all of the entire flow from these rivers is diverted, primarily for agricultural irrigation.

Fish species in this region are similar to those of the San Joaquin River Region, except that no anadromous species are found in the Tulare Lake Region because there is rarely any connection to the San Joaquin River, and hence, the Pacific Ocean. Fish below the major reservoirs are primarily warm-water species and are only found seasonally in areas subject to dewatering. Most of the species are introduced. The reservoirs consist of a lower story rainbow and brown trout fishery, with an upper story warm-water fishery consisting primarily of introduced centrarchids and catfish.

North Coast Region

The North Coast Region contains many of California's most undeveloped rivers, including the Klamath, the Smith, and the Eel. Other North Coast rivers, such as the Russian and the Gualala, are more developed for irrigation and domestic purposes. North Coast rivers account for more than 35 percent of the state's total runoff. The Klamath River is California's second largest river, with an average annual flow of 13 million acre-feet. Portions of the Klamath, Eel, Smith, Trinity, Van Duzen, Salmon, and Scott rivers are protected from further dams and diversions under federal and state wild and scenic rivers legislation.

Major North Coast Region rivers provide habitat for most of the anadromous species in the Central Valley. Fall-, late fall-, and spring-run chinook salmon are important fisheries resources, with the fall-run being the largest run of chinook salmon. Chinook salmon runs are present in the larger rivers. Coho salmon, federally listed as threathened (62 FR 24588, May 6, 1997), use the North Coast Region. Winter steelhead trout are present in all of the rivers with chinook salmon and, like coho salmon, are found in smaller coastal streams as well. Green sturgeon, white sturgeon, striped bass, American shad, and Pacific lamprey are all present primarily in the lower reaches of the larger rivers, but white sturgeon, striped bass, and American shad have only limited populations.

Major factors including dams, water diversion, poor land and forest practices, and road building adversely affect North Coast fisheries populations and habitats. The North Coast also supports significant offshore commercial and sport fisheries, primarily for chinook salmon, which affect this species' populations.

Central Coast Region

Principal rivers of the Central Coast Region include the Santa Ynez, Nacimiento, Salinas, Carmel, Big and Little Sur, and Santa Maria-Cuyama river systems. Numerous other small coastal streams also drain the west slope of the Coast Ranges. These rivers have flashy hydrology, typically low summer flows that are exacerbated by river diversions, and short-duration, high-peak flows during significant rainfall events.

The Central Coast Region maintains limited but important anadromous fish resources, primarily winter-run steelhead trout and coho salmon. Most of the large streams support steelhead trout, particularly during normal to wet water years, while coho salmon runs are limited now to only a few streams. Small lagoons at the mouths of these streams play a significant role as rearing habitat for steelhead trout and coho salmon (Shapovalov and Taft, 1954). The Central Coast's native fish fauna also include most of the common, strictly freshwater fishes of the Sacramento-San Joaquin River Basin. Habitat destruction from logging, urbanization, agricultural development, flood control, water diversions, and barriers has significantly reduced anadromous and native fish production and eliminated anadromous production from much of the historical habitat.

Like the North Coast Region, the Central Coast Region supports significant offshore commercial and sport fisheries, primarily for chinook salmon.

South Coast Region

The South Coast Region has been developed predominantly for urban and industrial uses with limited anadromous fisheries. Although half of California's population is within this area, the principal rivers provide less than 2 percent of the state's water supply. The Los Angeles and San Gabriel rivers are almost entirely confined to concrete channels built for flood control.

Remnant steelhead trout runs, in low numbers, are present in the Ventura, Santa Clara, Malibu, and Santa Ynez rivers and historically in the Los Angeles and Santa Ana rivers. Fisheries within the permanent, low-lying creeks can contain any of the four species of true freshwater fishes native to the Southern California area: arroyo chub, speckled dace, Santa Ana sucker, and unarmored threespine stickleback (a species federally listed as endangered that is a subspecies of threespine stickleback). Numerous introduced species are also found, including brown and rainbow trout in the upper watersheds. Reservoirs are generally stocked with rainbow trout and contain largemouth bass, smallmouth bass, other centrarchids, catfish, bullhead, carp, and numerous other species. Reservoir fish communities consist almost exclusively of introduced species.

AFFECTED FISH SPECIES OR COMMUNITIES

According to California Department of Fish and Game (DFG), only 38 percent of California's native freshwater species remain in a stable, healthy condition. Eight (or 7 percent) of California's 116 species and subspecies of native freshwater fish are extinct; 15 (or 13 percent) are listed as threatened or endangered; 24 (or 21 percent) qualify for listing; and 25 (or 21 percent) are either declining, exist in small isolated populations, or require special management to survive (Water Education Foundation, 1992b). Of particular concern are the state's declining anadromous fisheries that, because of their varied life history patterns, require a number of different habitats to spawn, incubate, rear, and migrate effectively. Significant impacts on habitats required for even one life stage can adversely affect species populations.

Human activities affecting fisheries and fisheries habitats in the Central Valley are numerous and inextricably woven together, requiring comprehensive, coordinated activities to restore major fisheries communities. These human effects have significantly modified conditions affecting fish abundance, distribution, and production throughout the Central Valley.

The CVPIA has the potential to affect two of the six major drainage systems in California described by Moyle (1976): (1) the Sacramento-San Joaquin River system, and (2) the south coastal drainages. In recent years, the fish fauna of each of these systems have been greatly augmented through the introduction of fish species from all over the world. The Sacramento-San Joaquin River system maintains 17 endemic species, 40 native or anadromous species, 15 native marine euryhaline species, and 36 introduced species. The south coastal drainages maintain 2 endemic species, 11 native resident or anadromous species, 11 native marine euryhaline species, and 30 introduced species. (Moyle, 1976.)

Species List for Geographic Regions

Table II-1 provides a species list adapted from Moyle (1976), with additional information from DFG (1987) and Urquhart (pers. comm.).

Table II-1 presents all inland fishes, anadromous fishes, and marine-euryhaline species. More than 200 species that exist in San Francisco Bay (Miller and Lea, 1972), most of which are marine species, are not presented. In addition, the Oregon Coast Region and marine species in the North Coast and Central coast regions were omitted from the table because these areas contain a large diversity of marine species, nearly all of which will not be affected by the implementation of the CVPIA.

Fish Species or Communities Included

The target species included in this technical appendix are primarily anadromous species, or threatened and endangered species. Target species include those that are economically important in California; are indicator species of habitat degradation common to other species, particularly native species; or have legal requirements for protection. Anadromous species in this appendix include fall, late-fall, winter and spring runs of chinook salmon, winter- and summer-run steelhead trout, white sturgeon and green sturgeon, striped bass, and American shad. Although striped bass and American shad are introduced species, both are abundant, contribute substantially to California's recreational fishery, and are specifically identified in the CVPIA.

Delta smelt, longfin smelt, and Sacramento splittail are target species because of their declining abundance in the Delta and status as threatened (delta smelt) and potentially threatened (Sacramento splittail) under the ESA. These species are native, and their abundance and distribution indicate the ecological health of the Sacramento-San Joaquin River system, the Delta, and the Bay.

Several species groups were included in this analysis, but at a community level, rather than at an individual species level. These communities include reservoir fisheries, Bay fisheries, estuarine invertebrates, and special-status aquatic species fishery resources. These communities warranted inclusion in this analysis because the CVPIA will likely affect them. In many cases, information is inadequate for an accurate discussion of impacts on individual species. Also, impact mechanisms potentially affecting any individual species within a community generally will act on all of the species, although not necessarily in the same manner.

Fish Species or Communities Excluded

Pacific lamprey and river lamprey are anadromous species, but they are not identified in the CVPIA.

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TABLE II-1

DISTRIBUTIONAL CHECKLIST OF THE INLAND FISHES OF CALIFORNIA BY MAJOR GEOGRAPHIC REGION

Family Species	Sacramento River	Sacramento-San Joaquin Delta	San Joaquin River	Tulare Lake	North Coast	Central Coast	San Francisco Bay	South Coast
Petromyzontidae								
Pacific lamprey	NA	NA	NA	--	NA	NA	NA	NA
River lamprey	NA	N	NA	--	NA	NA	NA	--
Pacific brook lamprey	N	--	N	N	NA	N	--	--
Pit-Klamath brook lamprey	--	--	--	--	N	--	--	--
Acipenseridae								
White sturgeon	NA	NA	NA	--	NA	--	NA	--
Green sturgeon	NA	NA	NA	--	NA	--	NA	--
Clupeidae								
Pacific herring	--	N	--	--	N	N	N	N
American shad	IA	IA	I	--	IA	--	IA	--
Threadfin shad	I	I	I	I	--	I	I	I
Osmeridae								
Eulachon	--	--	--	--	NA	--	NA	--
Delta smelt	--	N	--	--	--	--	N	--
Surf smelt	--	O	--	--	N	N	N	O
Longfin smelt	--	N	--	--	N	--	N	--
Wakasaki	I	I	I	--	--	--	--	--
Salmonidae								
Coho salmon	--	--	--	--	NA	NA	NA	--
Chinook salmon	NA	NA	NA	--	NA	NA	NA	--
Kokanee salmon	I	--	I	--	--	--	--	--
Brook trout	I	--	I	--	I	I	--	I
Cutthroat trout	--	--	--	--	N	--	--	--
Brown trout	I	--	I	I	I	I	--	I
Rainbow trout (resident and steelhead)	NA	NA	NA	N	NA	NA	NA	NA

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TABLE II-1. CONTINUED

Family Species	Sacramento River	Sacramento-San Joaquin Delta	San Joaquin River	Tulare Lake	North Coast	Central Coast	San Francisco Bay	South Coast
Cyprinidae								
Carp	I	I	I	I	I	I	I	I
Goldfish	I	I	I	I	--	I	I	I
Tench	--	--	--	--	--	I	--	--
Golden shiner	I	I	I	I	I	I	--	I
Sacramento blackfish	N	N	N	N	I	N	--	I
Hardhead	N	N	N	N	N	N	--	--
Hitch	N	N	N	N	N	N	N	--
Sacramento squawfish	N	N	N	N	N	N	--	--
Tui chub	NE	NE	NE	NE?	--	--	--	--
Arroyo chub	--	--	--	--	--	--	--	N
Thicktail chub	NE	NE	NE	NE	--	--	--	--
Sacramento splittail	N	N	N	NR	--	--	--	--
California roach	N	N	N	N	N	N	N	I
Speckled dace	N	N	N	N	N	N	--	N
Lahontan redside	I	I	I	I	--	--	--	--
Red shiner	I	I	I	I?	--	--	I	--
Fathead minnow	I	I	I	I	--	I?	--	--
Catostomidae								
Bigmouth buffalo	--	--	--	--	--	--	--	I
Santa Ana sucker	--	--	--	--	--	--	--	N
Owens sucker	--	--	--	--	--	--	--	I?
Sacramento sucker	N	N	N	N	N	N	N	--
Ictaluridae								
Blue catfish	--	I	--	--	--	--	--	I
Channel catfish	I	I	I	I	--	--	--	I
White catfish	I	I	I	I	I	I	--	I
Yellow bullhead	I	I	I	I	--	--	--	I
Brown bullhead	I	I	I	I	I	I	--	I
Black bullhead	I	I	I	I	--	I	I?	I?
Cobitidae								
Chinese weatherfish	--	--	--	--	--	--	--	I

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C-081342

TABLE II-1. CONTINUED

Family Species	Sacramento River	Sacramento-San Joaquin Delta	San Joaquin River	Tulare Lake	North Coast	Central Coast	San Francisco Bay	South Coast
Cyprinodontidae								
Rainwater killifish	--	I	--	--	--	--	--	I
Argentine pearlfish	--	--	--	--	--	--	--	I
California killifish	--	--	--	--	O	O	--	N
Poeciliidae								
Mosquitofish	I	I	I	I	I	I	I	I
Guppy	--	--	--	--	--	--	--	I?
Green swordtail	--	--	--	--	--	--	--	I?
Atherinidae								
Topsmelt	--	N	--	--	N	O?	N	O
Inland silverside	I	I	I	I	I	I	I	--
Gasterosteidae								
Threespine stickleback	N	N	N	N	N	N	N	N
Syngnathidae								
Bay pipefish	--	O	--	--	N	O	N	--
Percichthyidae								
Striped bass	I	I	I	I	I	I	IA	--
White bass	--	--	--	I	--	--	--	I
Centrarchidae								
Sacramento perch	--	NR	NR	NR	I?	NE?	--	--
Black crappie	I	I	I	I	I	I	--	I
White crappie	I	I	I	I	I	--	--	I
Warmouth	I	I	I	I	--	I?	--	--
Green sunfish	I	I	I	I	I	I	I	I
Bluegill	I	I	I	I	I	I	I	I
Pumpkinseed	--	I	--	--	--	--	--	I
Redear sunfish	I	I	I	I	--	I	--	I
Largemouth bass	I	I	I	I	I	I	I	I
Spotted bass	I	I	--	--	--	--	--	I?

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TABLE II-1. CONTINUED

Family Species	Sacramento River	Sacramento-San Joaquin Delta	San Joaquin River	Tulare Lake	North Coast	Central Coast	San Francisco Bay	South Coast
Smallmouth bass	I	I	I	I	I	I	--	I
Redeye bass	--	--	--	--	--	--	--	I
Percidae								
Yellow perch	IE	IE	--	--	I	--	--	--
Bigscale logperch	I	I	I	I	--	--	--	--
Embiotocidae								
Shiner perch	--	O	--	--	N	N	O	O
Tule perch	N	N	N	N	N	NE	--	--
Mugilidae								
Striped mullet	--	O	--	--	--	--	O	O
Gobiidae								
Tidewater goby	--	N	--	--	N	N	N	N
Yellowfin goby	--	I	--	--	I	I	I	--
Longjaw mudsucker	--	N	--	--	--	N	N	O
Arrow goby	--	O	--	--	O	O	N	O
Chameleon/shimafin goby	--	I	--	--	--	--	I	I
Pholidae								
Penpoint gunnel	--	O	--	--	O	O	O	--
Saddleback gunnel	--	O	--	--	O	O	N	--
Cottidae								
Sharpnose sculpin	--	--	O?	--	O	O?	--	--
Staghorn sculpin	--	N	--	--	N	N	N	N
Coastrange sculpin	N	--	--	--	N	N	N	N
Prickly sculpin	N	N	N	N	N	N	N	N
Riffle sculpin	N	N	N	N	N	N	N	N
Pleuronectidae								
Starry flounder	--	N	--	--	N	N	N	N
Number of native spp.	22	29	22	16	30	25	28	14
Number of introduced spp.	31	32	29	25	19	20	14	28
Total number of spp.	53	61	51	41	49	45	42	42

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TABLE II-1. CONTINUED

Family Species	Sacramento River	Sacramento-San Joaquin Delta	San Joaquin River	Tulare Lake	North Coast	Central Coast	San Francisco Bay	South Coast

NOTES:
Species whose occurrence was uncertain were not included in totals. Occasional marine visitors are included in totals.

SOURCES:
Adapted from Moyle, 1976; Urquhart, pers. comm.; and DFG, 1987.

LEGEND:

N = native.	E = extinct.
I = introduced.	R = rare.
O = occasional marine visitor.	? = occurrence uncertain.
A = anadromous.	-- = does not occur.

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STUDY PERIOD

The study period for fisheries broadly covers the years from 1850 through October 1992, when the CVPIA was signed into law. The general periods of 1850-1966 and 1967-1991 are discussed.

The historical period of 1850-1940, discussed in the Historical Perspective section, is important because existing habitat conditions in numerous waterways have evolved from actions taken as early as the 1850s (e.g., hydraulic mining and flood control development), and historical population trends indicate conditions prior to development of CVP facilities.

Although generally unavailable, some quantitative data are available for the historical period of 1941-1966.

CVP conditions during 1967-1991 are the focus of the Affected Environment and Environmental Consequences chapters because the CVPIA goal is to double 1967-1991 natural anadromous fish production. In addition, the most quantitative fish population data are available for this time period. The DFG report covering Central Valley anadromous sport fish annual run size, harvest estimates, and population trends for 1967-1991 (Mills and Fisher, 1993) provided much of the abundance data for the 1967-1991 period.

DATA SOURCES

The overriding objective for data collection was to amass sufficiently detailed information about historical, existing, and estimated future conditions so that it will be possible to identify significant relationships and trends for key fisheries resources and to estimate potential impacts under a range of alternatives evaluated in the PEIS. Data were collected on statewide, regional, and subregional levels. For fisheries, the smallest unit of geographic measurement at the subregional level was an individual watershed. The smallest unit for any species was an individual life stage. Nearly all data were obtained from existing and published compilation reports, technical reports, management plans, or personal communications.

COORDINATION WITH OTHER ISSUE AREAS

Fisheries data collection and methodologies have been closely integrated and coordinated with the surface water and habitat water quality resource areas. In nearly all cases, the potential effects on fisheries resources exhibit themselves through changes in surface water variables (e.g., river flow, reservoir surface area and drawdown, and diversion entrainment) or habitat water quality variables (e.g., water temperature, salinity, or toxic metal concentrations). Consequently, information collected and presented for these two resource areas has been coordinated to ensure that these data are appropriate for input into the fisheries analyses.

The fisheries resource area itself will provide input for the fish, wildlife, and recreation economics issue area. In this context, the fisheries data collection efforts have been conducted at a level of detail sufficient to satisfy the input needs for determining the fish, wildlife, and recreation economics associated with each PEIS alternative.

DATA COLLECTION AND COMPILATION

Time Frame

Data collected for fisheries depended on availability and accessibility. Available and accessible data include published information or data that could be readily provided in print or electronically within a given period of time. For fisheries, data were considered available for the Affected Environment and Environmental Consequences chapters if they had been published by January 15, 1997.

Sources

Available reports covering existing conditions were reviewed and summarized in the description of the affected environment. These reports, mostly presenting recent fisheries studies in the Central Valley, were used to the fullest extent.

Additional sources of information proposed by Reclamation, the U.S. Fish and Wildlife Service (Service), NMFS, and DFG were compiled and used in the development of the fisheries section of the PEIS. Specific data sources are cited in subsequent sections.

Compilation Methodology

Data were collected for each target species and community within several geographic regions, as appropriate: Sacramento River (including significant west side and east side tributaries); Feather and Yuba rivers; American River; San Joaquin River (including Stanislaus, Merced, and Tuolumne rivers); the Delta and Suisun Bay; San Francisco Bay; Central, North, and South coasts; and reservoirs (primarily Whiskeytown Lake, Shasta Lake, Black Butte Lake, Lake Oroville, Folsom Lake, San Luis, New Melones, and Millerton Lake). In many cases, particularly for anadromous species, species life history stages correspond to several geographic areas. The fisheries analyses were developed by species or community, rather than strictly by geographic area.

A standardized data collection format was used for time sequence data, such as hydrology, operations, fish population abundance and distribution, and habitat conditions. Standardizing the format enabled use of data bases for all species and life stages. No format was developed for other types of data, most of which are contained in existing reports that were summarized.

Limitations

Although detailed species information was collected, information on factors related to the effects of CVP operations and other limiting factors was not available for many species. In some cases, information known for one species or race was assumed to apply to other species or races. Cases where insufficient information was available to precisely define a condition or effect were identified. When important species responses to historical conditions and factors were unknown, likely species responses were assumed and verified or modified based on agency input.

COORDINATION ACTIVITIES WITH AGENCIES

Coordination activities were conducted throughout the development of this technical appendix. Primary agencies involved in coordination included Reclamation, the Service, NMFS, and DFG. These agencies reviewed the preliminary outline for this Fisheries Technical Appendix; reviewed and agreed on primary data sources; and reviewed draft report sections for level of detail, accuracy, assumptions, and appropriateness.

HISTORICAL PERSPECTIVE

Historical fishery resources within the CVPIA study area were quite different from the fish resources present today. Many native species have declined in abundance and distribution, and several introduced species have become well established (see Table II-1). Habitat modification, species introductions, and overfishing are the major factors affecting fisheries resources within the CVPIA study area. These and other human activities adversely affecting Central Valley fisheries resources are presented in Table II-2. A timeline of major historical activities affecting fisheries and fish habitats is presented in Table II-3.

Major human-induced modifications of fish habitats began during the first major settlement of California that followed the 1849 gold rush. Dredging and hydraulic mining for gold producedenormous quantities of sediments that were deposited downstream, altering fish habitats in streams, rivers, the Delta, and San Francisco Bay. In the late 1800s and early 1900s, agricultural development and flood control led to construction of 1,000 miles of levees that destroyed much of the original marshland and riparian fish habitat. Agricultural, grazing, mining, logging, navigation, and urban development activities degraded water quality in almost all fish habitats. Debris control, flood control, and water supply projects developed by federal, state, local, and private entities included dams that blocked most major rivers in the Central Valley. Water storage and diversions greatly altered natural river and Delta flow patterns, and fish entrainment at water diversions affected unknown numbers of young fish every year.

The commercial fishery typically landed millions of pounds of Central Valley salmon per year throughout the late 1800s, hitting its peak in 1882, but lingered on until 1919 when the industry was abolished by legislative action (Skinner, 1962). Exotic species were accidentally or intentionally introduced in the Sacramento-San Joaquin River system. These species adversely affected native species through predation, competition for habitat and food, and hybridization.

HISTORICAL FACTORS AFFECTING ABUNDANCE

1850-1939

Habitat Modifications. Hydraulic mining in the Sierra foothills between 1852 and 1884 washed almost 1 billion cubic yards of sediment into the rivers draining the Gold Country (California State Lands Commission, 1991). Sand and cobbles filled pools and riffles, and finer materials smothered spawning gravels and marshland habitats. The sediments raised riverbeds and clogged channels and sloughs, changing hydraulic characteristics of the river systems and

TABLE II-2

HUMAN ACTIVITIES ADVERSELY AFFECTING CENTRAL VALLEY FISHERIES RESOURCES

Activity	Affected Area		
	Sacramento and San Joaquin River Basins	Delta and Bay	Pacific Ocean
Hydraulic mining	X	X	
Upstream storage and controlled releases	X	X	
CVP export - direct effects	X	X	
CVP export - indirect effects	X	X	
SWP export - direct effects	X	X	
SWP export - indirect effects	X	X	
Agricultural diversions	X	X	
Municipal diversions	X	X	
Industrial diversions	X	X	
Channel blockage (barriers)	X	X	
Channel dredging	X	X	
Gravel extraction	X		
Agricultural drainage	X	X	
Municipal waste discharges	X	X	X
Industrial chemical waste discharges	X	X	X
Industrial thermal waste discharges	X	X	X
Power diversions and release rates	X	X	
Upstream erosion	X	X	
Nonpoint-source pollution	X	X	X
Flow impedance (e.g., bridges)	X	X	
Wake erosion	X	X	
Riparian vegetation removal	X	X	
Levee development	X	X	
Use of riprap	X	X	
Floodflow dispersion (distributaries)	X		
Upstream flow additions	X	X	
Delta island reclamation	X	X	
Slough conversion to channels	X	X	
Upstream drainage and waste discharge	X	X	
Upstream nonpoint-source pollution	X	X	
Fill and reclamation	X	X	
Island flooding (levee breaks)		X	
Species introductions	X	X	X
Fish hatchery production	X	X	X
Sport fishing harvest	X	X	X
Commercial fishing harvest	X	X	X
Poaching (illegal harvest)	X	X	X

TABLE II-3

TIMELINE OF MAJOR HISTORICAL ACTIVITIES AFFECTING FISH HABITATS IN THE CENTRAL VALLEY REGION

Year	Event
1849	Gold rush began
1849	First major levee constructed in Delta on Grand Island
1850	Congress adopted Arkansas Swamp Act to sell floodplain land to developers who would construct levees and drainage systems
1852	Wheaton Mining Dam constructed at La Grange on the Tuolumne River
1852	Hydraulic mining activities began
1854	Large irrigation facilities constructed to divert Merced River water
1859-1865	Large irrigation facilities constructed to divert Tule River water
1860	California Legislature authorized formation of levees and reclamation districts
1868	California Legislature adopted Green Act that allowed formation of reclamation districts with taxing authority
1870	State Fish Commission created to enforce catch restrictions and require fish ladders for all physical obstructions
1870s-1880s	Diversions and canals constructed to convey water from Kings, San Joaquin, Kern, and Merced rivers
1871	Mendota Dam (Weir) constructed
1880	California Legislature approved Drainage Act to provide flood control in Central Valley
1880s	Tulare Lake water quality was extremely poor due to return flows and could not be used for irrigation or potable water supplies
1880s	Large salmon canneries in operation on Sacramento River
1884	Federal injunction banned use of hydraulic mining unless sediment was controlled
1892	Congress establishes California Debris Commission to remove mining debris from rivers and navigable waters
1893	Modesto Irrigation District and Turlock Irrigation District constructed La Grange Dam on Tuolumne River
1895	Debris dams constructed along Sacramento River tributaries, including the American and Yuba rivers
1900	Bear River Dam completed
1902	Union Dam completed on North Fork Stanislaus River
1912	Goodwin Dam completed on Stanislaus River
1913	California Legislature passed Raker Act allowing San Francisco to divert water from Tuolumne River
1913	Almanor Dam completed on North Fork Feather River
1914	Sacramento River Flood Control Project levees constructed to minimize flooding due to increased elevation of riverbed caused by mining debris

TABLE II-3. CONTINUED

Year	Event
1915	Oakdale Irrigation District and South San Joaquin Irrigation District began diversions from Stanislaus River
1916	Main Strawberry Dam completed on South Fork Stanislaus River
1919	Merced Irrigation District constructed Exchequer Dam and Power Plant on Merced River
1923	Modesto Irrigation District and Turlock Irrigation District constructed Don Pedro Reservoir on Tuolumne River
1924	Melones Dam constructed on Stanislaus River
1924	(Old) Bullards Bar Dam completed on Yuba River
1925	Calaveras Dam completed on Calaveras River
1927	Lake Briton Dam and Pit River No. 3 and No. 4 dams completed on Pit River
1929	Pardee Dam and Mokelumne Aqueduct completed on Mokelumne River
1930	Lyons Dam completed on South Fork Stanislaus River
1930s	Fertilizers and vector poisons introduced on farmlands
1931	The federal government and California State Water Resources Commission (Hoover-Young Commission) recommended that the federal government construct the Central Valley Project and that the state operate the facilities
1931	Salt Springs Dam completed on North Fork Mokelumne River
1933	State of California authorized bonds for $170 million for the Central Valley Project (Shasta Dam and Power Plant, Friant Dam and Power Plant, Contra Costa Canal, Madera Canal, Friant Kern Canal, other dams and pumps on the San Joaquin River, transmission lines from Shasta to Antioch, and a pump between the Sacramento and San Joaquin rivers
1934	San Francisco constructed Hetch Hetchy Aqueduct from Tuolumne River
1940	Congress reauthorized the Central Valley Project, restated the purposes of the project, and allowed for construction of local distribution systems as part of Central Valley Project construction projects
1940	Water diversions began at Contra Costa Canal from the Delta (first CVP diversion)
1944	Shasta Dam completed and water diversions began on Sacramento River
1947	Friant Dam completed on San Joaquin River
1949	Friant Kern Canal completed
1950	Keswick Dam completed on Sacramento River
1950	Anderson-Cottonwood diversion began
1951	Delta Cross Channel, Delta-Mendota Canal, and Tracy Pumping Plant completed
1954	Pine Flat Dam completed on Kings River
1954	Isabella Dam completed on Kern River
1955	Nimbus Dam and Power Plant completed on American River
1956	Folsom Dam completed on American River

TABLE II-3. CONTINUED

Year	Event
1957	Beardsley and Donnel dams completed on Middle Fork Stanislaus River
	Tulloch Dam completed on Stanislaus River
	Tracy fish collection facility completed
1959	Putah South Canal diversions began
1961	California Department of Water Resources establishes Interagency Delta Committee to evaluate solutions to Delta problems
1961	Success Dam completed on Tule River
1962	Terminus Dam completed on Kaweah River
1962	South Bay Aqueduct completed
1963	Whiskeytown Dam completed
1963	New Hogan Dam completed on Calaveras River
1963	Camanche Dam completed on Mokelumne River
1963	Lewiston Dam, Carr Power Plant, and Clear Creek Tunnel completed
1964	Trinity Dam completed on Trinity River
1964	Red Bluff Diversion Dam completed on Sacramento River
1967	San Luis Canal and Dam completed
1967	New Exchequer Dam completed on Merced River
1967	California State Water Resources Control Board adopted Water Quality Control Plan for Sacramento-San Joaquin Delta pursuant to Federal Water Pollution Control Act of 1965
1967	State Water Project Delta Pumps and California Aqueduct completed
1967	Oroville Dam completed on Feather River
1969	New Bullards Bar Dam completed on Yuba River
1970	New Don Pedro Dam completed on Tuolumne River
1971	California State Water Resources Control Board adopted Water Rights Decision 1379 establishing Delta water quality standards
1971	Tehama-Colusa Canal and Pumping Plant completed
1971	Tehama-Colusa Canal Fish Facility completed
1976-1977	Drought
1978	California State Water Resources Control Board adopted Water Rights Decision 1485
1978	New Melones Dam completed on Stanislaus River
1986	Coordinated Operations Agreement adopted
1987	North Bay Aqueduct completed

estuary. A federal injunction in 1884 banned hydraulic mining unless sediments were prevented from washing into streams.

In the late 1800s, dikes and levees were built to develop and protect low-lying agricultural lands. Conversion of wetlands to agricultural land was encouraged by federal and state laws during this period. In the southern Central Valley, Tulare and Buena Vista lakes supported a small commercial fishery before they were drained for farmland (Moyle, 1976). As early as 1920, nearly all the Delta's 350,000 acres of freshwater marsh had been diked and converted to farmland. However, serious flooding continued and levees were built and reinforced along the river channels. Removal of riparian vegetation increased bank erosion and sedimentation. Bank protection projects stabilized the riverbanks and reduced sedimentation, but resulted in poorly vegetated, steeply sloped riverbanks covered with rock. Land reclamation and levee construction severely degraded riparian habitat by increasing depth and flow velocity and reducing cover and habitat diversity (California Department of Water Resources [DWR], 1984). Riparian vegetation, marshes, and temporarily flooded land are important rearing habitats for many fish species, and these activities significantly reduced available habitat to numerous species (particularly native species).

Water quality in the Sacramento-San Joaquin River system was degraded by several development activities. Agricultural drainage increased salinity and concentrations of pesticides and other toxic substances in the rivers and Delta. Early cities, towns, and homesteads dumped untreated sewage into rivers, sloughs, and bays, causing eutrophication, oxygen depletion, and disease. Industries discharged toxic substances into the rivers and bays. Removal of riparian vegetation increased sedimentation and water temperature in small streams used by chinook salmon and trout for spawning and rearing.

Although the largest dams in the Central Valley were constructed after 1939, early dams and diversions built by miners and farmers obstructed hundreds of miles of habitat without allowing for fish passage. Debris dams were constructed in the late 1800s and early 1900s on the American and Yuba rivers to block downstream transport of mining sediments; however, these dams also largely blocked spring-run chinook salmon and steelhead migrations (Sumner and Smith, 1940). The Anderson-Cottonwood Irrigation District (ACID) diversion dam, constructed in 1917 on the upper Sacramento River near Redding, created an important impediment to migrating chinook salmon. By the 1920s, at least 80 percent of the Central Valley chinook salmon and steelhead spawning areas had been cut off by dams and other human-made barriers (Clark, 1929).

Although the dams built in this period blocked spawning migrations of chinook salmon and steelhead on most Central Valley streams and rivers, many of the smaller dams were eventually fitted with fish ladders, some of which were moderately successful. Beginning with the Baird Hatchery in 1872, hatcheries were constructed to augment chinook salmon production; however, the hatcheries replaced only a limited portion of the production that was lost from dams and other impacts.

In addition to blocking spawning migrations of chinook salmon and steelhead, dams may have disrupted spawning migrations of sturgeon, striped bass, and American shad; however, these species were probably only marginally affected because they generally do not ascend the rivers as

far as the dam sites. Because resident fish populations above dams were reproductively isolated, outbreeding and natural recolonization in the event of extinction were no longer possible. Diversions, especially for irrigation, entrained and killed many young chinook salmon, steelhead, and other fish species annually. Diversions also dewatered sections of streams to the point that insufficient water prevented migrating adult salmon and steelhead from passing through, eggs from incubating, or juvenile salmon and other fish from rearing successfully.

Species Introductions. Exotic species were first introduced in the Sacramento-San Joaquin River system shortly after the gold rush. Most of the species were introduced to improve fishing or provide forage for game species. Several hundred striped bass were introduced from the Atlantic coast in 1879 and 1882, and the population quickly multiplied to millions of adults. American shad were introduced from New York between 1871 and 1881 and were well established by 1879. Largemouth and smallmouth bass, catfishes, and sunfishes were also introduced.

In some instances, exotic species were introduced because the native fishes were no longer productive in habitats altered by human activities. The introduced species adversely affected native species through predation, competition for habitat and food, and hybridization. Introduced species became more abundant than native species in some parts of the Sacramento-San Joaquin River system.

Overfishing. The first organized commercial fishery in the Sacramento-San Joaquin River system was developed between 1848 and 1850. Chinook salmon were taken in gill nets and seines in the rivers, the Delta, and the Bay; and sardines, herring, and flatfishes were captured with seines in the Bay (Clark, 1929; Skinner, 1962). Commercial fishing expanded rapidly after the gold rush. From 1873 to 1910, more than 20 canneries processed 5 million pounds of chinook salmon annually from the Sacramento and San Joaquin River system. The river fishery for salmon peaked in 1910 at more than 10 million pounds and then began a somewhat continuous decline.

An ocean salmon fishery was developed in the 1890s and early 1900s and largely replaced the collapsed river fishery. Commercial fishing may have been a major contributor to the depletion of chinook salmon stocks following the gold rush (Clark, 1929). Commercial and sport fisheries for striped bass also developed in the late 1800s and early 1900s, but a decline in the population resulted in an end to the commercial fishery in the 1930s.

1940-1992

Habitat Modifications. Most of the factors adversely affecting fish habitats before 1940 continue today. The principal exception is uncontrolled hydraulic mining, which was banned in 1884. The CVP and SWP, which include the largest dams, diversions, export pumps, and canals in the Central Valley, produced profound new changes to fish habitats. CVP and SWP facilities and operations are described in the technical appendix for surface water supplies and facilities operations and Attachment B, Fish Habitat Water Quality Technical Information.

The many large dams of the CVP and SWP permanently eliminated most of the chinook salmon and steelhead spawning and rearing habitat in the Central Valley. Although relatively small

dams had disrupted spawning migrations of chinook salmon and steelhead on most Central Valley streams and rivers before the major dams were constructed, several of the small dams were fitted with fish ladders, some of which were moderately successful. The large dams built after 1939 blocked passage permanently and completely.

Shasta and Keswick dams, built on the Sacramento River in 1944 and 1950, respectively, blocked hundreds of miles of spawning habitat on the upper Sacramento, Pit, and McCloud river drainages. The RBDD, built in the mid-1960s on the Sacramento River 60 miles downstream of Keswick Dam, has been a major impediment to upstream migration of adult salmon. Friant Dam, built on the San Joaquin River in 1947, blocked thousands of migrating chinook salmon and steelhead. CVP and SWP dams blocked chinook salmon and steelhead passage to the principal spawning reaches of the American and Feather rivers, respectively. Large dams constructed by local water districts, power companies, and the U.S. Army Corps of Engineers (COE) blocked passage to spawning reaches of other major rivers in the Central Valley.

The dams permanently altered the chinook salmon stock composition in the Central Valley and virtually eliminated wild stocks of spring-run chinook salmon, once the dominant race of chinook salmon in California. Steelhead runs also were reduced to a fraction of historical levels. Available spawning habitat was restricted to main river channels and small tributaries at lower elevations, habitat best suited for fall- and late-fall-run chinook salmon. The dams also blocked recruitment of spawning gravels from upstream sources to the portions of rivers still available for salmon and steelhead spawning. Gravel mining continued to eliminate remaining spawning habitat on rivers below the dams.

The large dams altered the temperature regime in the rivers downstream, which affected chinook salmon, steelhead trout, and other game and nongame species. For a period after the large dams were constructed, reservoirs were kept relatively full and the cold water released from the hypolimnion provided cooler summer temperatures in the downstream reaches. Fall-run chinook salmon populations responded to the colder flows and began to spawn earlier than historical salmon runs. More recently, however, the reservoirs have been drawn down farther because of increased water demands, resulting in warmer water releases and high egg mortalities. Winter-run chinook salmon, which spawn in spring and summer, have been especially harmed by the warmer water temperatures.

CVP and SWP facilities that greatly modified fish habitat in the Delta include the export pumps in the South Delta and the DCC. These facilities have greatly altered flow patterns in the Delta and cause entrainment and mortality of young fish of all resident and anadromous species. Additional diversions by agricultural, municipal, and industrial interests in the Delta and the Sacramento and San Joaquin rivers entrain many millions of young fish annually. During spring and early summer, many emigrating juvenile chinook salmon and steelhead die at improperly screened and unscreened irrigation diversions and pumping facilities.

Operations of dams and diversions in the Sacramento-San Joaquin River system and the Delta have altered natural flow regimes by changing the frequency, magnitude, timing, and direction of flow. These changes potentially affect all fish species in the rivers, Delta, and Bay. Abundances of many Delta and Bay species are strongly related to the magnitude of flows into and out of the Delta (Jassby, 1993). The timing and magnitude of high flows in fall, winter, and spring

influence migrations and spawning of many fishes in the rivers and estuary. Flows also affect food supply and water temperature and therefore influence growth. Reverse flows in the Delta may distort normal transport and migration patterns of many fish species, resulting in high mortalities from predation, entrainment, and other causes. Flow regulation can create complex adverse impacts that reduce fishery potential in a cumulative fashion.

Modifications of fish habitat resulting from levee construction and channel dredging have continued into the present period. Fish species diversity is generally much lower in channelized sections of streams than in undisturbed sections (Moyle, 1976). By the mid-1960s, the Sacramento River Flood Control Project consisted of more than 440 miles of river, canal, and stream channels; 1,000 miles of levees; and 95 miles of bypasses. Additions to the project are ongoing and proposed for the future.

Contamination of fish habitats by agricultural drainage, urban runoff, and industrial and municipal discharges are continuing problems. High salinity and pesticide runoff from agricultural lands have been especially severe in the San Joaquin River. Pesticides in drainage from rice fields have been implicated in the mortality of striped bass and may have played a role in their decline (DWR and Reclamation, 1993). Dredging to maintain ship channels in the Bay and Delta increases turbidity and resuspends contaminants present in the sediments.

Species Introductions. The introduction of exotic species has continued into the present period. Introduced species are now more abundant than native species in many areas of the Central Valley. The major difference between recent introductions and those of the earlier period is that most of the recent introductions have been accidental. Many species were introduced into the estuary by ships discharging ballast water that contained exotic species.

Introduced zoobenthos and zooplankton in the Delta may be major factors affecting the Delta ecosystem and fish species that use the Delta. *Potamocorbula*, a clam species, was accidentally introduced to the Bay about 1986 and now occurs throughout the Bay and into Suisun Bay. The *Potamocorbula* population consumes enormous quantities of phytoplankton and may adversely affect food chains of many fish species in the estuary. A native copepod species, *Eurytemora*, has declined significantly, and an introduced and less desirable species, *Sinocalanus*, has increased. The predator avoidance mechanisms displayed by *Sinocalanus* make it harder for fish to prey upon them, than upon displaced native zooplankton species. Species introductions have markedly changed the food web in the Sacramento-San Joaquin River system with unknown effects on fish species.

Large-scale hatchery production of chinook salmon and steelhead to mitigate impacts from large dams have adversely affected the remaining wild stocks, primarily by affecting their genetic composition and facilitating high harvest rates. Massive releases of hatchery fish at distant release sites without imprinting have resulted in high straying rates and spawning of hatchery-produced salmon in non-natal rivers. These hatchery fish may alter gene pools and adversely affect genetic fitness of wild populations.

Overfishing. Fishing has contributed to the declines of salmon, steelhead, striped bass, and American shad in the present period. Commercial salmon fishing effort and effectiveness have increased substantially since the earlier period. Ocean harvest of Central Valley chinook salmon

was at an all-time high in 1988, both in terms of harvest rate and number of salmon caught. High harvests result in lower returns and smaller sizes of spawning salmon. Recent harvests of chinook salmon, however, have been well below average (see the Recent Conditions for Chinook Salmon section of this chapter).

Commercial fishing for striped bass was banned in 1935; however, annual striped bass landings by the sport fishery may have been much larger than the commercial harvest (see the Recent Conditions for Striped Bass section of this chapter). The striped bass population declined in the 1930s and was severely depleted by 1970.

Although it is clear that the catches have significant adverse effects on the number of returning adults each year, to what extent fishing mortality affects future year-classes has not been adequately addressed. The extent to which fishing contributed to the decline of the striped bass and chinook salmon populations is unknown.

HISTORICAL IMPACTS OF CVP FACILITIES

CVP facilities fall into three general categories: upstream storage facilities, upstream diversion facilities, and Delta diversion (and export) facilities. This section briefly discusses historical impacts of major CVP facilities in each river system. More detailed information is provided in discussions of recent conditions for various fish species.

Sacramento River

In dry and critically dry water years, water temperature conditions in the upper Sacramento River are frequently unsuitable for chinook salmon. Reclamation has operated CVP reservoirs to provide water of suitable temperature, balancing reservoir releases into the Sacramento River from CVP facilities in the Trinity and Sacramento river systems and releasing water through the low-level outlets at Shasta Dam. Reclamation has constructed a temperature control device at Shasta Dam and began operation in early 1997. This device will enable the release of water of suitable temperature from specific water levels through the power-generating facilities.

Shasta Dam has blocked gravel recruitment for downstream salmon spawning and incubation. The problem of gravel availability in the Sacramento River is exacerbated downstream by other non-CVP-related activities, such as dam construction in Sacramento River tributaries, bank protection measures in the mainstem of the Sacramento River, and gravel mining enterprises. In 1988, Reclamation and DFG cooperated in an action to place salmon spawning gravel in approximately 3 miles of the Sacramento River between Keswick Dam and ACID diversion dam. Reclamation and DFG participated in another gravel placement activity in 1989, when gravel was stockpiled on the right bank of the Sacramento River immediately downstream from Keswick Dam.

Construction of the RBDD was completed in 1964. The only identified mitigation requirement was to replace 3,000 fall-run chinook salmon expected to have been lost because of inundation of salmon spawning habitat in the Sacramento River by Lake Red Bluff immediately upstream of the dam. The Tehama-Colusa Canal (TCC) fish facilities were constructed in part to mitigate the presence and operation of the RBDD. The fish facilities (i.e., dual- and single-purpose canals)

failed to meet production goals and ultimately ceased operation when the RBDD was operated with its gates raised to improve salmon migration past the dam.

Additional impacts related to the RBDD have been identified during the period the dam has been in operation. These impacts include impeded passage of upstream-migrating adult salmon past the dam, excessive losses of downstream-migrating juvenile fish due to predators in the vicinity of the dam, and passage of small fish through the fish screens into the TCC fish facilities.

To improve passage of adult winter-run chinook salmon upstream past the RBDD, Reclamation raises the gates at the dam from September through May. Reclamation also replaced the louver fish screen facility at the TCC fish facilities with a more efficient rotating drum screen facility.

Other non-CVP-related operations adversely affect survival of anadromous fish in the Sacramento River. These effects include inadequate fish passage facilities for adult salmon at ACID diversion dam, losses of juvenile fish at the GCID diversion facility, and entrainment of young fish in the unscreened diversions along the Sacramento River.

Coleman National Fish Hatchery was constructed to partially mitigate the reduction in the anadromous fish populations attributable to the loss of fish habitat upstream of Keswick Dam. In addition, a minimum-flow release schedule was developed to provide in-river flows for the Sacramento River anadromous fish populations downstream of Keswick Dam.

The Clear Creek fishery has been affected by several CVP- and non-CVP-related activities. Saeltzer Dam effectively blocks the upstream migration of anadromous fish and also prevents the movement of gravel downstream past the dam site. Gravel mining downstream of Saeltzer Dam has removed much of the spawning gravel potentially available for the fish. Whiskeytown Dam, constructed in 1963 approximately 10 miles upstream from Saeltzer Dam, has altered the flow regime in Clear Creek by stabilizing annual flow conditions. Fall flows have been reduced and summer flows have been increased. Although Whiskeytown Dam traps much of the sediment that normally will be transported downstream, the reduced high-flow events have prevented the movement of sediments out of the stream system, thereby resulting in sediment compaction in the gravels downstream of Saeltzer Dam.

American River

The American River fishery has been affected by CVP- and non-CVP-related activities. Non-CVP-related activities in the basin dating back to the 19th century include extensive hydraulic mining operations and construction of water development projects, including water storage reservoirs and diversion facilities. These activities removed salmon spawning gravel from the river and blocked anadromous fish migration. CVP-related effects are associated with Folsom and Nimbus dams.

The Nimbus Fish Hatchery was constructed to mitigate for the construction of Nimbus Dam and subsequent reduction in the anadromous fish populations attributable to the loss of American River fish habitat upstream of the dam. In addition, a minimum-flow release schedule was developed to provide in-river flows for the American River anadromous fish populations downstream of Nimbus Dam. During normal water years, the production of chinook salmon at

Nimbus Fish Hatchery typically exceeded the mitigation goal originally identified for Folsom and Nimbus dams.

Delta Waterways

Impacts on fish and wildlife in the Delta waterways attributable to CVP operations are difficult to identify because of the complex interaction of federal, state, and local water development activities, all of which affect the same resources. In addition, water development activities upstream of the Delta can affect conditions in the Delta. Operation of the Tracy Pumping Plant hinders survival of San Joaquin and Sacramento River juvenile salmon. Most of the young salmon salvaged at the Tracy Fish Collection Facility originated from the San Joaquin River. Many other fish species are also affected.

Stanislaus River

Water resources development in the Stanislaus River preceded construction of CVP facilities in the basin. Goodwin Dam, a major water diversion facility owned and operated by private water users, blocks anadromous fish migration past the dam. This dam was in place and operational before the CVP facilities were constructed at New Melones Dam, which is sited upstream of Goodwin Dam.

San Joaquin River

The San Joaquin River fishery has been affected by CVP- and non-CVP-related water development activities. Non-CVP-related water development that began in the 19th century eliminated the fall-run chinook salmon that had spawned in the mainstem of the river. This run was eliminated before Friant Dam was constructed. Friant Dam operation, however, has been associated with the elimination of spring-run chinook salmon in the San Joaquin River upstream of the Merced River.

RECENT CONDITIONS FOR CHINOOK SALMON

SACRAMENTO RIVER

Stock Abundance and Distribution

Historical Population Trends (1850-1966). Early trends in Central Valley chinook salmon (*Oncorhynchus tshawytscha*) populations were monitored indirectly by commercial catch records dating back to 1874, when complete records of commercial gill net landings were first available. These records are of limited use in determining population trends for specific streams or runs, but they indicate major trends in Central Valley chinook salmon abundance.

Early accounts indicate that the commercial salmon fishery in California began around 1850. The gold rush and the ensuing human population growth in California led to its rapid expansion. Hydraulic gold mining, logging, agricultural, and grazing activities also increased rapidly, leading to the first major human impacts on stream habitat and fish populations in the

Sacramento River Basin (Buer et al., 1984). Later, construction of agricultural, power generation, and debris dams accelerated declines in chinook salmon populations by preventing access to historical spawning and rearing habitat or substantially reducing the amount of available habitat (Clark, 1929).

Between 1874 and 1910, total gill net landings fluctuated between 2 million and 11 million pounds and averaged approximately 6 million pounds. A distinct downward trend after 1910 led to a period of extremely poor catches between 1926 and 1943, in which annual yields ranged from 0.4 to 2.5 million pounds per year and averaged approximately 1 million pounds per year (Skinner, 1962) (Figure II-1). This decline coincided with a decline in the number of adult salmon returning to hatchery facilities in the Sacramento River between 1915 and 1924 (Clark, 1929). The California ocean troll fishery, the dominant commercial salmon fishery by 1916, also experienced reduced catches from approximately 6 million pounds per year before 1920 to an average of approximately 4 million pounds per year during the 1920s and 1930s, despite increasing effort. Clark (1929) concluded that the Sacramento-San Joaquin River salmon fishery was in a "state of serious depletion" by 1929, citing overfishing, loss of spawning areas from dam construction, loss of young salmon in overflow basins, and losses to predatory fishes as principal causes. Following a brief increase to approximately 6.5 million pounds in 1946, annual gill net landings returned to an average of approximately 1 million pounds per year through the 1950s (Skinner, 1962).

Baird Fish Hatchery was constructed on the McCloud River in 1872, marking the beginning of artificial propagation of chinook salmon in the Central Valley. (Skinner, 1962.)

Four races of chinook salmon, recognized by the season of their upstream migration, occur in the Sacramento basin: fall-, late fall-, winter-, and spring-run.

Fall-Run Chinook Salmon. Counts of adult salmon were obtained as they passed through the ACID dam fish ladder as early as 1937, but comprehensive estimates of fall-run chinook salmon abundance in the Sacramento River and its major tributaries were not made until 1953 (Hallock, n.d.). Annual estimates of spawning escapement (i.e., the total number of adult salmon, age 2 years and older, that escape the ocean fishery and return to spawn) in the mainstem Sacramento River reveal a gradual but steady decline during the 1950s and 1960s; annual run size declined from an average of 179,000 adults during 1953-1966 to an average of 77,000 adults during 1967-1991 (Figure II-2).

Fall-run chinook salmon in the major Sacramento River tributaries exhibited variable abundance patterns during the 1950s and 1960s. Between 1953 and 1966, annual spawning escapement in the Feather River fluctuated widely and averaged approximately 41,000 adults (Figure II-3). Annual run size in the Yuba River, a major tributary of the Feather River, increased markedly from an average level of 5,000 adults in the 1950s to a peak of 37,000 adults in 1963. Average annual spawning escapement in the Yuba River between 1953 and 1966 was approximately 14,000 adults (Figure II-3). Annual run size in the American River averaged approximately 26,000 adults before construction of Folsom Dam and Nimbus Fish Hatchery in 1955 (Fry, 1961). By 1966, average run size, including river and hatchery spawners, had increased to approximately 39,000 adults. Average annual spawning escapement in the American River

C-081360

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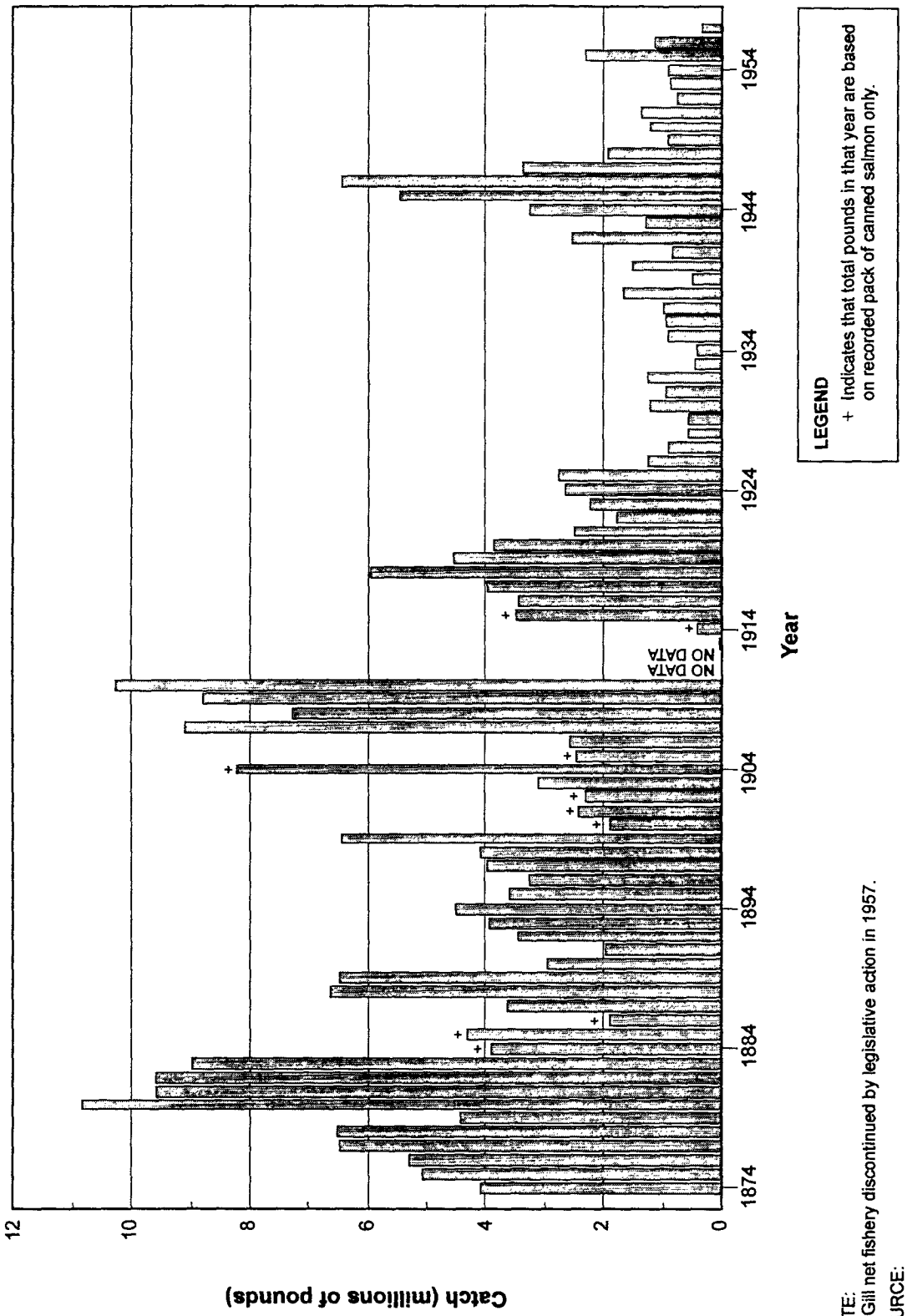
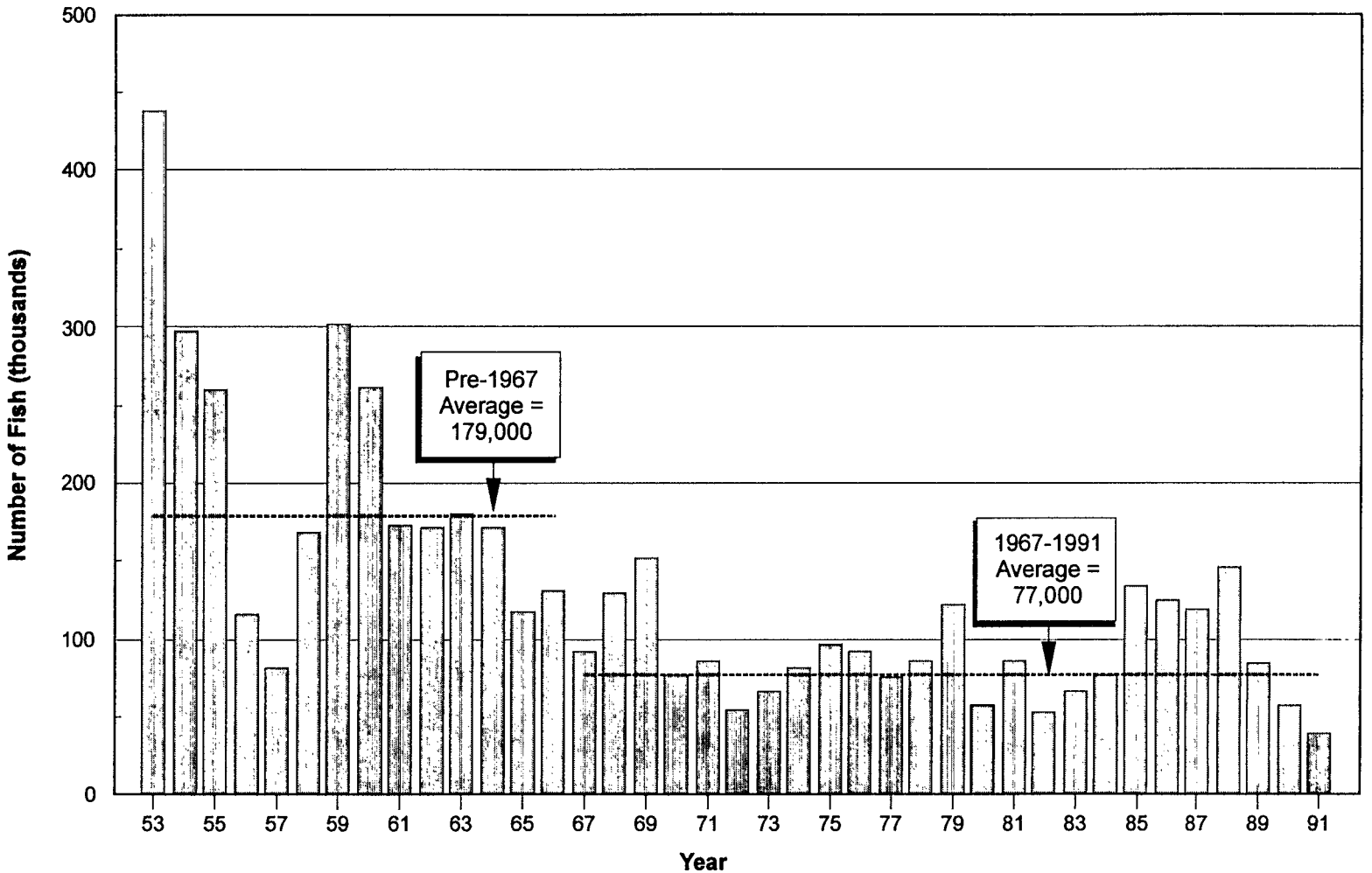


NOTE:
Gill net fishery discontinued by legislative action in 1957.

SOURCE:
Skinner, 1962.

FIGURE II-1

SACRAMENTO-SAN JOAQUIN RIVER COMMERCIAL GILL NET SALMON LANDINGS (1874-1957)



NOTE:
Population estimates prior to 1967 may not be comparable to post-1967 estimates because counting methods changed after completion of RBDD.

SOURCES:
Hallock, n.d.; Mills and Fisher, 1993.

FIGURE II-2

ESTIMATES OF FALL-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN THE MAINSTEM SACRAMENTO RIVER (1953-1991)

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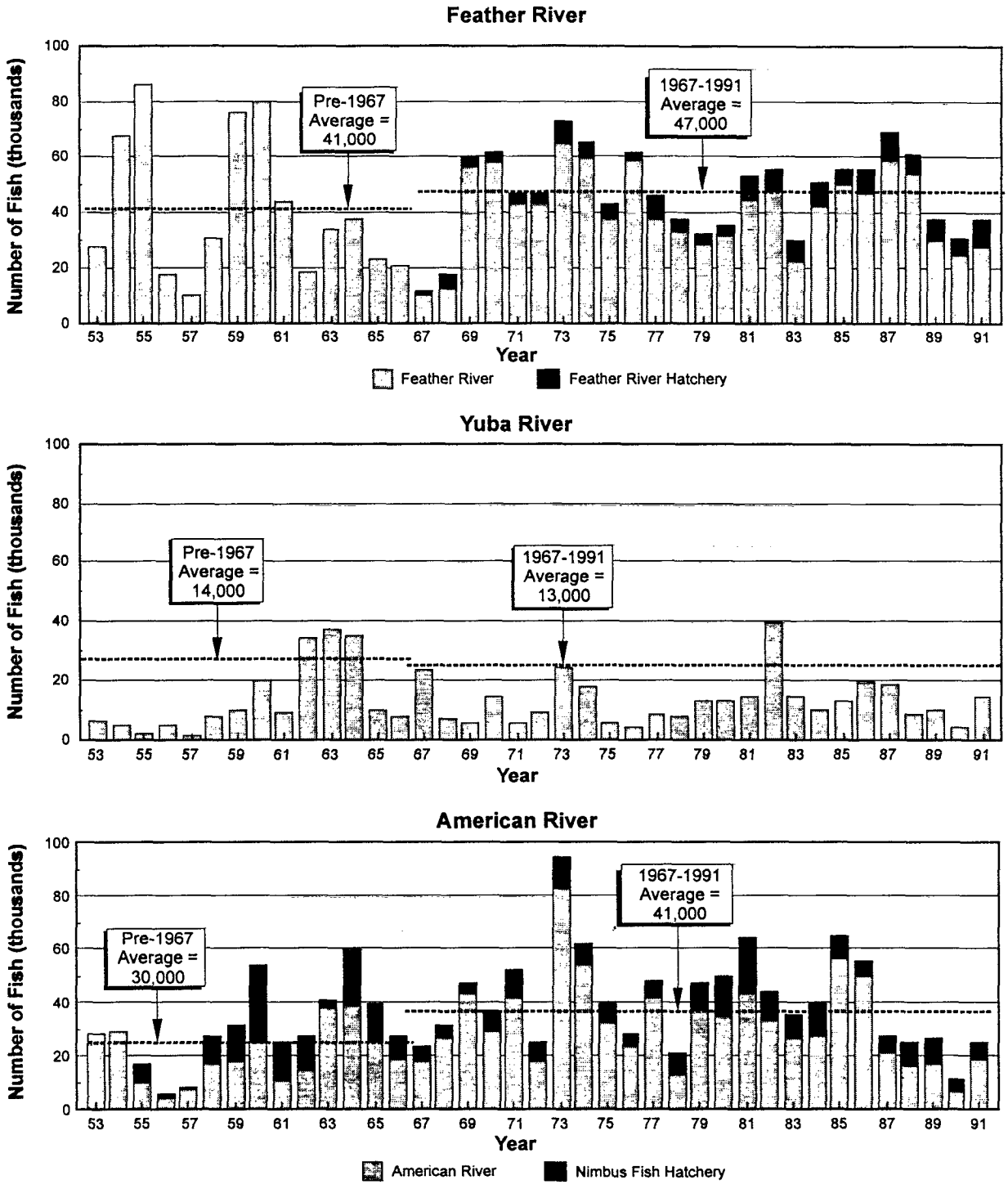
C-081361



SOURCES:
Hallock, n.d.; Mills and Fisher, 1993.

FIGURE II-3

ESTIMATES OF FALL-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN THE FEATHER, YUBA, AND AMERICAN RIVERS (1953-1991)

between 1953 and 1966 was approximately 30,000 adults; on average, 19,000 adults spawned in the river and 11,000 were spawned in the hatchery (Figure II-3).

Fall-run chinook salmon runs in minor Sacramento River tributaries, including Clear, Cow, Cottonwood, Antelope, Mill, and Deer creeks, were not monitored regularly, although declines in abundance have been evident since 1953 (Figure II-4). Annual spawning escapement in Battle Creek between 1953 and 1966 exhibited a general decline similar to the pattern observed in the mainstem Sacramento River. Total run size averaged 17,000 adults, with an average of 9,000 adults spawning in Battle Creek and 8,000 spawning in Coleman National Fish Hatchery (Figure II-5).

Late Fall-Run Chinook Salmon. Because of high flows and turbid conditions that generally prevail during the late fall-run chinook salmon spawning period, annual abundance estimates were possible only after construction of the RBDD and its associated fish-counting facilities in 1967.

Winter-Run Chinook Salmon. Before construction of Shasta and Keswick dams in 1945 and 1950, respectively, winter-run chinook salmon were reported to spawn in the upper reaches of the Little Sacramento, McCloud, and lower Pit rivers (Moyle et al., 1989). Specific data relative to historical run sizes prior to 1967 are sparse and mostly anecdotal. Slater (1963) is frequently cited indicating that winter-run populations were small and limited to the McCloud River before construction of Shasta Dam. Recent DFG research in the California State Archives indicates that the winter-run chinook salmon population could have numbered more than 200,000 (Fox and Rectenwald, pers. comms.). Cold hypolimnetic releases from Shasta Reservoir enabled the run to spawn successfully in the Sacramento River below Keswick Dam. Under these favorable habitat conditions, the run was maintained at more than 80,000 adults by the mid-1960s (Reclamation, 1986a).

Spring-Run Chinook Salmon. Historically, spring-run chinook salmon was the most abundant salmon race in the Central Valley. The principal holding and spawning areas were in the middle reaches of the San Joaquin, Feather, upper Sacramento, McCloud, and Pit rivers upstream of the present location of major dams. Smaller runs occurred in tributaries large enough and cold enough to support adults during the summer holding period.

Gold mining, agricultural diversions, logging, and overharvest caused the first major declines in spring-run chinook populations. By 1930, agricultural and sediment control dams on tributary streams resulted in severe declines and extirpation of tributary stocks by preventing spring-run adults from reaching critical summer holding and spawning habitat. Further extirpations followed construction of major storage reservoirs on the Sacramento River and major tributaries in the 1940s and 1950s. By 1966, only remnant populations of spring-run chinook salmon were present below these dams.

Considerable overlap in the spawning period with fall-run on the mainstem Sacramento River and major tributaries has probably resulted in significant introgression (i.e., loss of genetic purity) of spring-run stocks (Slater, 1963). Genetically pure stocks occur only in three minor Sacramento River tributaries, Butte, Mill, and Deer creeks (Mills and Ward 1996).

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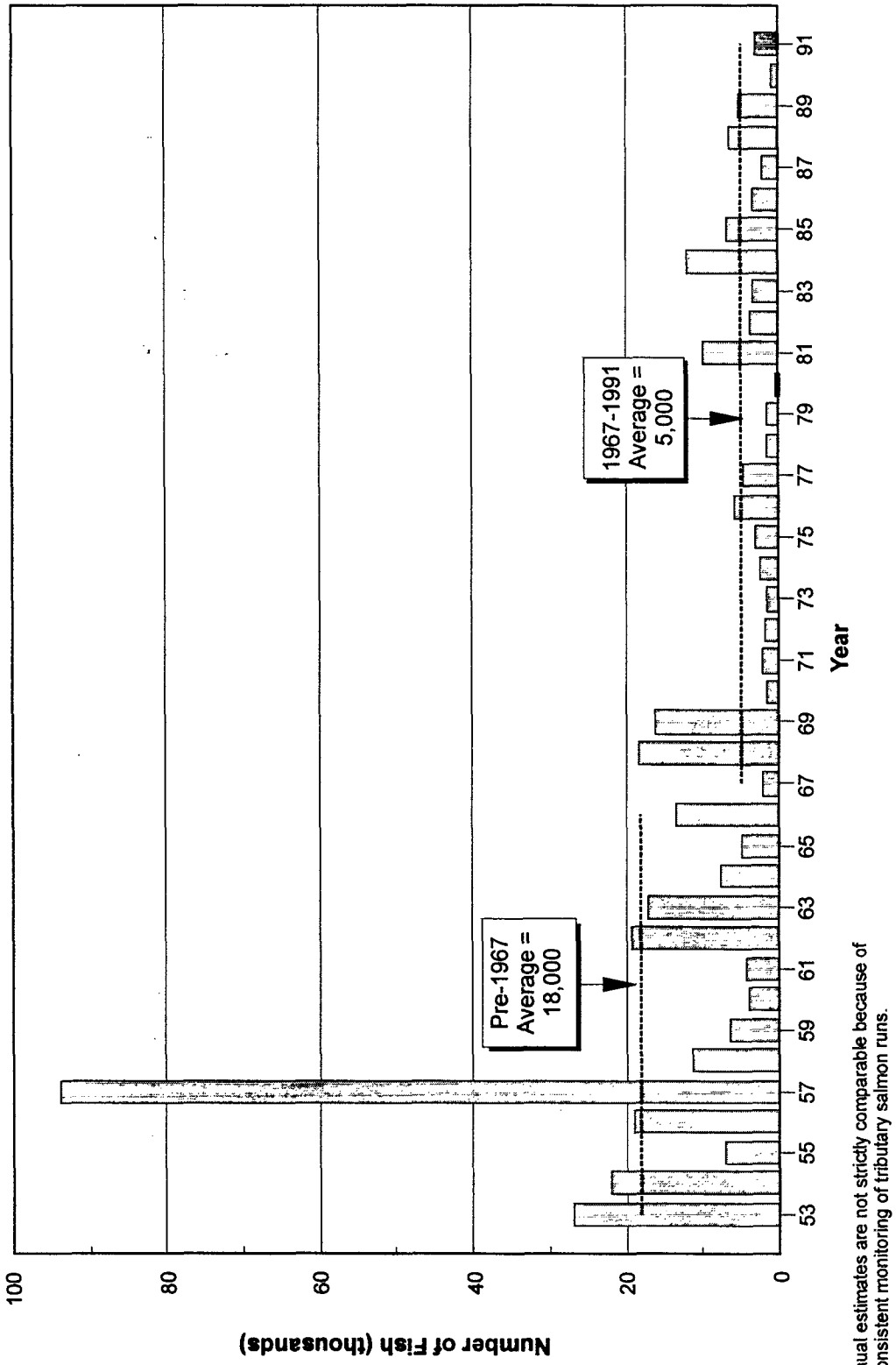


NOTE:
Annual estimates are not strictly comparable because of inconsistent monitoring of tributary salmon runs.

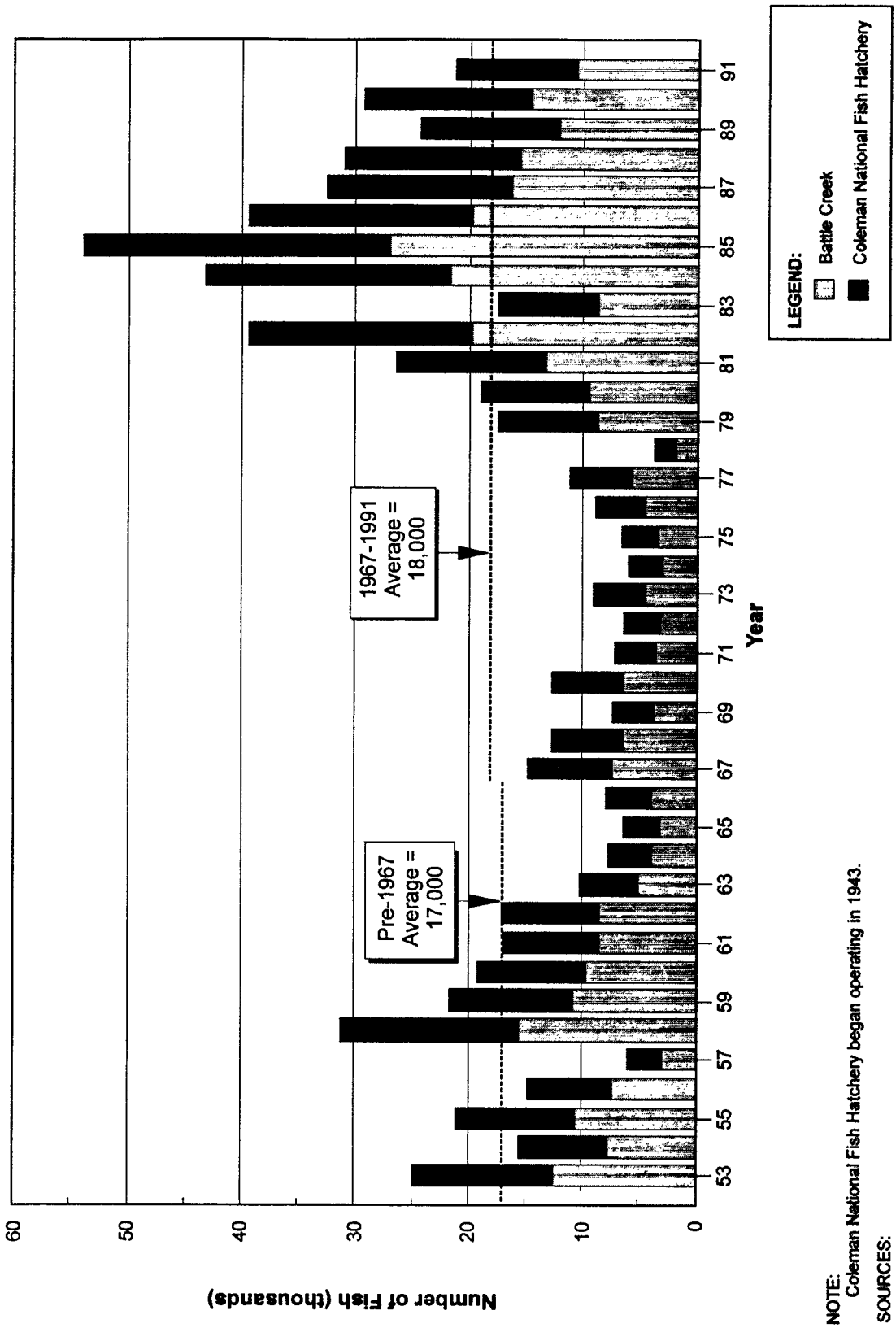
SOURCES:
Hallock, n.d.; Mills and Fisher, 1993.

FIGURE II-4

ESTIMATES OF FALL-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN MINOR SACRAMENTO RIVER TRIBUTARIES (1953-1991)

C-081365

C—0 8 1 3 6 5



NOTE:
Coleman National Fish Hatchery began operating in 1943.

SOURCES:
Hallock, n.d.; Mills and Fisher, 1993.

FIGURE II-5

ESTIMATES OF FALL-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN BATTLE CREEK AND COLEMAN NATIONAL FISH HATCHERY (1953-1991)

Current Population Trends (1967-1991)

Fall-Run Chinook Salmon

Sacramento River Mainstem. The overall decline in mainstem Sacramento River fall-run chinook salmon during the 1950s and 1960s was followed by low but relatively stable population levels during the 1970s and 1980s. (Note: Population estimates before 1967 could not be comparable to post-1967 estimates because counting methods changed after completion of the RBDD.) A decline during the recent drought, however, led to a record low spawning escapement of approximately 29,000 adults in 1991. Average annual spawning escapement of fall-run chinook salmon in the Sacramento River between 1967 and 1991 was approximately 77,000 fish (Figure II-2).

Feather and Yuba Rivers. Annual fall-run chinook salmon spawning escapement in the Feather River increased and became less variable following completion of Oroville Dam and Feather River Salmon and Steelhead Hatchery in 1968; average run size increased sharply in 1969 and remained relatively high through 1991. Annual spawning escapement of fall-run chinook salmon in the Feather River between 1967 and 1991 averaged approximately 47,000 adults; on the average, 41,000 adults spawned in the river and 6,000 were spawned in the hatchery (Figure II-3). Annual spawning escapement in the Yuba River during 1967-1991 averaged approximately 13,000 adults with no apparent trend (Figure II-3).

American River. Fall-run chinook salmon spawning escapement in the American River between 1967 and 1991 averaged 41,000 adults; on the average, 32,000 adults spawned in the river and 9,000 were spawned in Nimbus Fish Hatchery (Figure II-3).

Minor Sacramento River Tributaries. Estimates of fall-run chinook salmon spawning escapement in minor Sacramento River tributaries (excluding Battle Creek) are incomplete for the 1967 to 1991 period. No trends in run size are apparent for Clear, Cow, Cottonwood, Paynes, Antelope, Mill, Deer, and Butte creeks, although record low escapements occurred in most of these creeks in recent years (Figure II-4). Annual spawning escapement in Battle Creek between 1967 and 1991 averaged approximately 18,000 adults; on the average, approximately 8,000 adults spawned in Battle Creek and 10,000 were spawned in Coleman National Fish Hatchery (Figure II-5). Increases in production capacity and improved water quality, temperature, and disease control techniques at Coleman National Fish Hatchery resulted in record run sizes in recent years (Reclamation, 1985).

Late Fall-Run Chinook Salmon. Counts of chinook salmon passing the RBDD since 1967 provide the most reasonable indication of overall trends in late fall-, winter, and spring-run chinook salmon abundance in the upper Sacramento River. The number of late fall-run chinook salmon passing the RBDD declined from an average of 35,000 adults in the late 1960s to an average of 7,000 adults in recent years (Figure II-6). Hatchery returns to Coleman National Fish Hatchery during this period have fluctuated between 200 and 3,000 fish, with record low returns in 1990 and 1991 (Figure II-7).

Small runs of late fall-run chinook salmon may be present on the Yuba and Stanislaus rivers. However, for the purpose of the Draft PEIS, the late fall-run were not addressed in either river.

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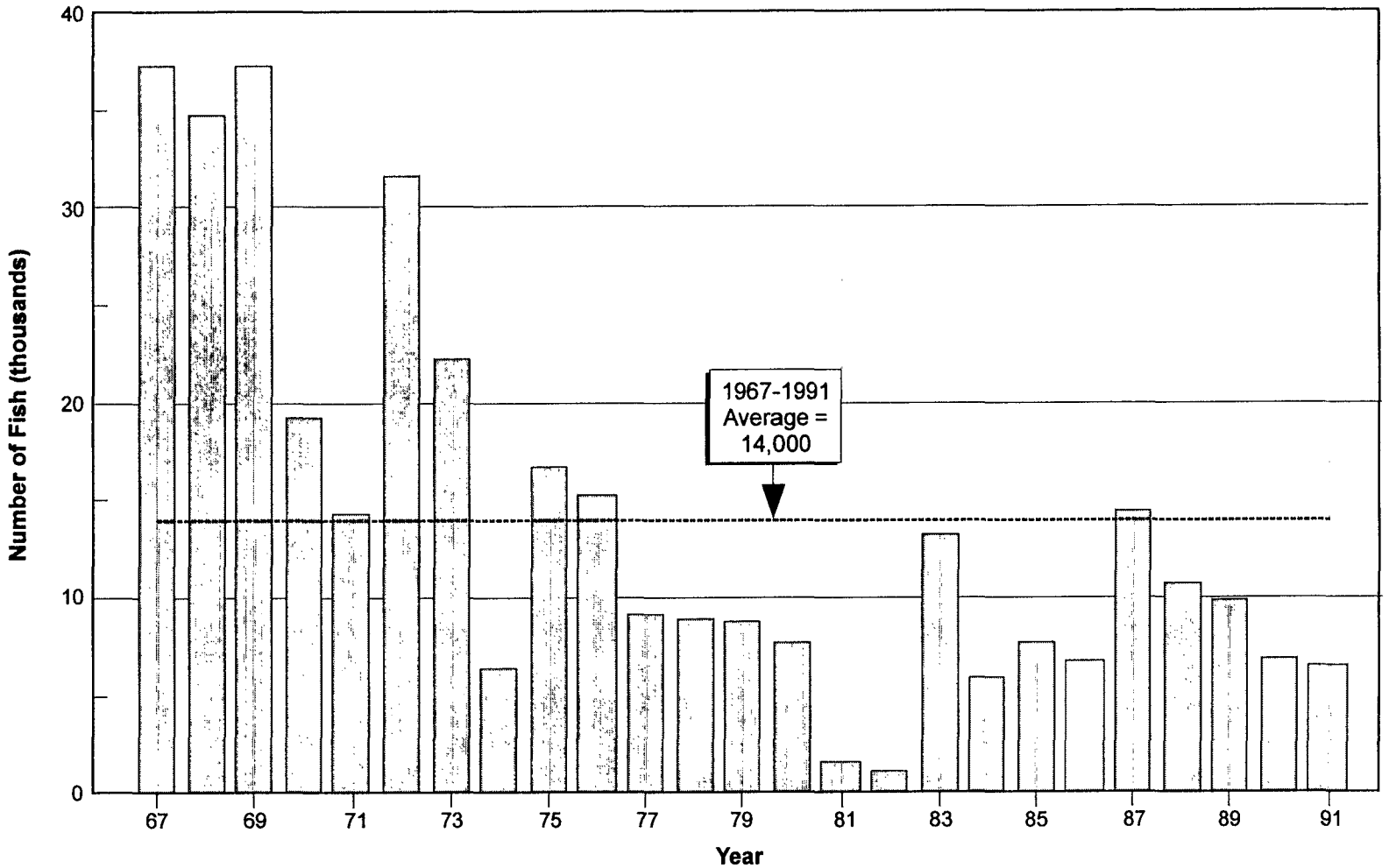
C—081367



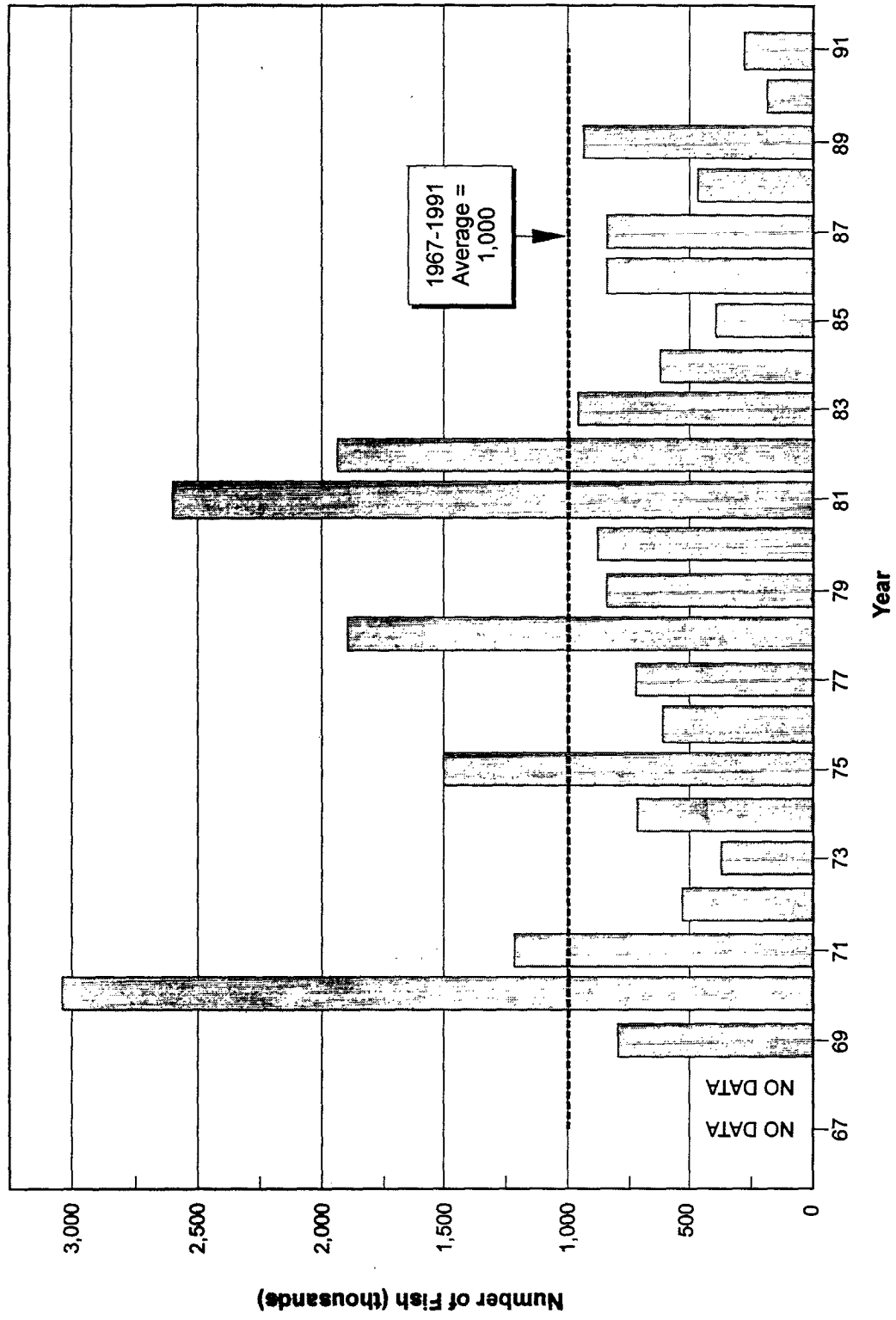
SOURCE:
Mills and Fisher, 1993.

FIGURE II-6

ESTIMATES OF LATE FALL-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN THE MAINSTEM SACRAMENTO RIVER (1967-1991)

C-081368

C—0 8 1 3 6 8



SOURCE:
Mills and Fisher, 1993.

FIGURE II-7

ANNUAL RETURNS OF LATE FALL-RUN CHINOOK SALMON TO COLEMAN NATIONAL FISH HATCHERY (1967-1991)

Winter-Run Chinook Salmon. Winter-run chinook salmon suffered a precipitous decline from an average of approximately 80,000 adults in the late 1960s to estimated run sizes of 547, 441, and 191 in 1989, 1990, and 1991, respectively (Figure II-8). Estimated run sizes in 1992 and 1993 were 1,180 and 341, respectively. Factors contributing to this decline include water temperature impacts associated with operation of Shasta Reservoir and, to a lesser extent, of Keswick Reservoir and Whiskeytown Lake, adult and juvenile passage problems at the RBDD, modification and loss of spawning and rearing habitats, predation, pollution, and entrainment in water diversions on the Sacramento River and in the Delta. The recent drought in California (1987-1992) exacerbated these impacts. (NMFS, 1992.)

The return of an estimated 550 adults in 1989 prompted listing of the winter-run chinook salmon as an endangered species by the State of California and as a threatened species by the federal government. Another record low spawning escapement of 191 fish in 1991 prompted review and subsequent reclassification of the winter-run chinook salmon to endangered status under the federal ESA (NMFS, 1992).

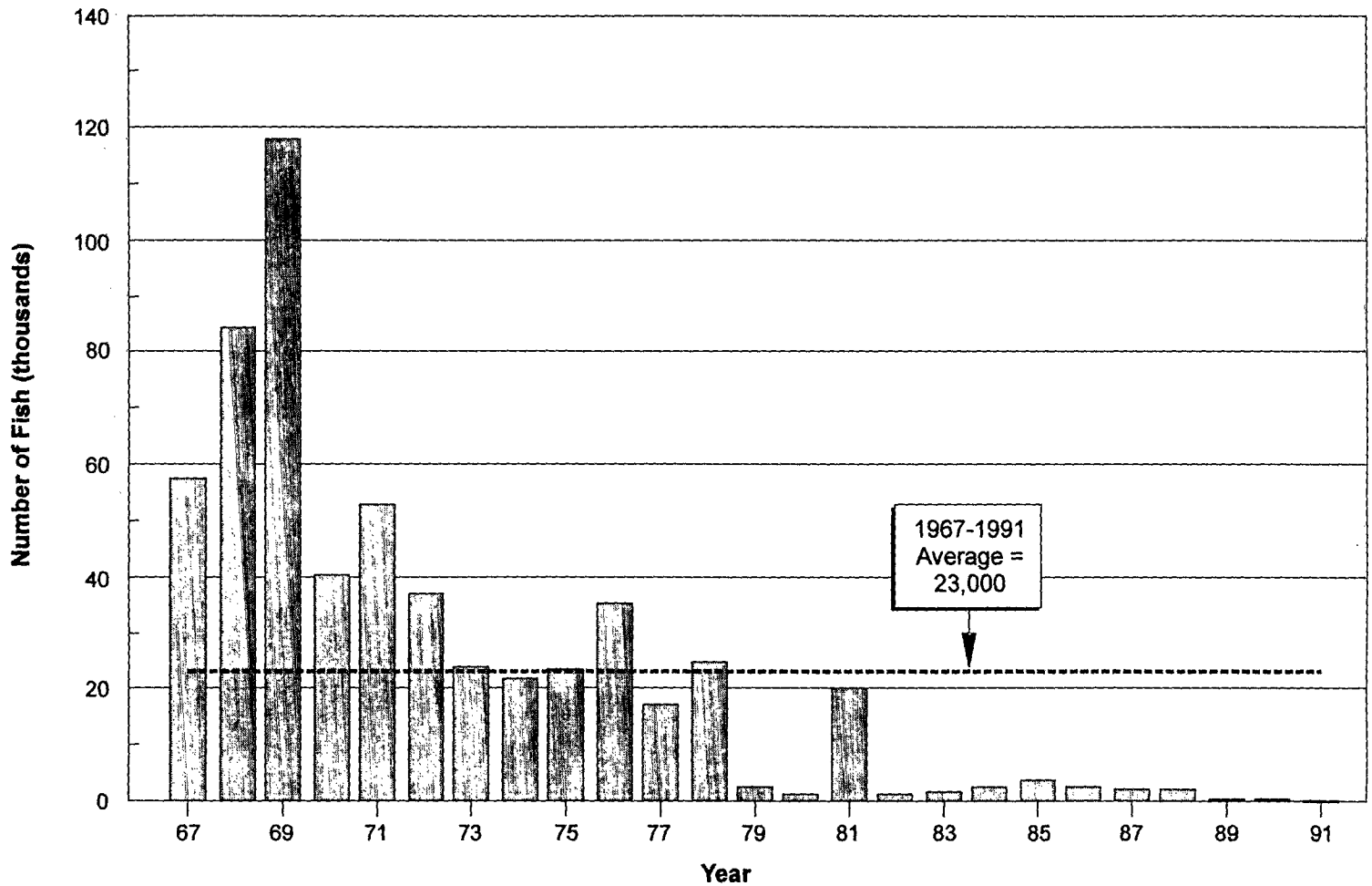
Spring-Run Chinook Salmon. The number of adults passing the RBDD has fluctuated between highs of more than 25,000 fish to a record low of 773 fish in 1991 (Figure II-9). An average of approximately 11,000 fish migrated past the dam between 1967 and 1991.

Numbers of spring-run chinook salmon entering Feather River Fish Hatchery increased from an average of approximately 300 adults between 1967 and 1981 to an average of approximately 2,000 adults between 1982 and 1991 (Figure II-10). Increased returns are associated with the recent practice of trucking and releasing large numbers of hatchery smolts in the lower Sacramento River and Delta. Annual hatchery returns are based on the assumption that all salmon entering the hatchery before October 1 are spring-run fish. Fish entering after that date are considered fall-run fish. Small numbers of spring-run chinook salmon migrate into the Yuba River, but these fish appear to be primarily strays originating from Feather River Fish Hatchery.

The 1967-1991 average spawning escapement of spring-run chinook salmon in Deer and Mill creeks was 1,300 and 900, respectively (Figure II-11). Since 1967, run sizes have declined by 85 percent in Mill Creek and 80 percent in Deer Creek. A small run averaging approximately 400 fish spawns in Butte Creek (Figure II-11). This run has been supported by natural reproduction and plants of chinook salmon smolts from Feather River Fish Hatchery. Big Chico Creek previously supported a run size of 500 spring-run chinook salmon, but now only a remnant population remains.

Life History

Because of overlapping run timing, spawning periods, and early life history phases, the upper Sacramento River supports all freshwater life stages of chinook salmon during all months of the year. General chinook salmon life history traits are described below, followed by a review of traits that distinguish each of the four races. Figure II-12 illustrates the general chinook salmon life cycle. Figure II-13 shows the location of major spawning and rearing areas for each chinook salmon race. Figures II-14, II-15, and II-16 summarize the timing and abundance of chinook



SOURCE:
Mills and Fisher, 1993.

FIGURE II-8

ESTIMATES OF WINTER-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN THE MAINSTEM SACRAMENTO RIVER (1967-1991)

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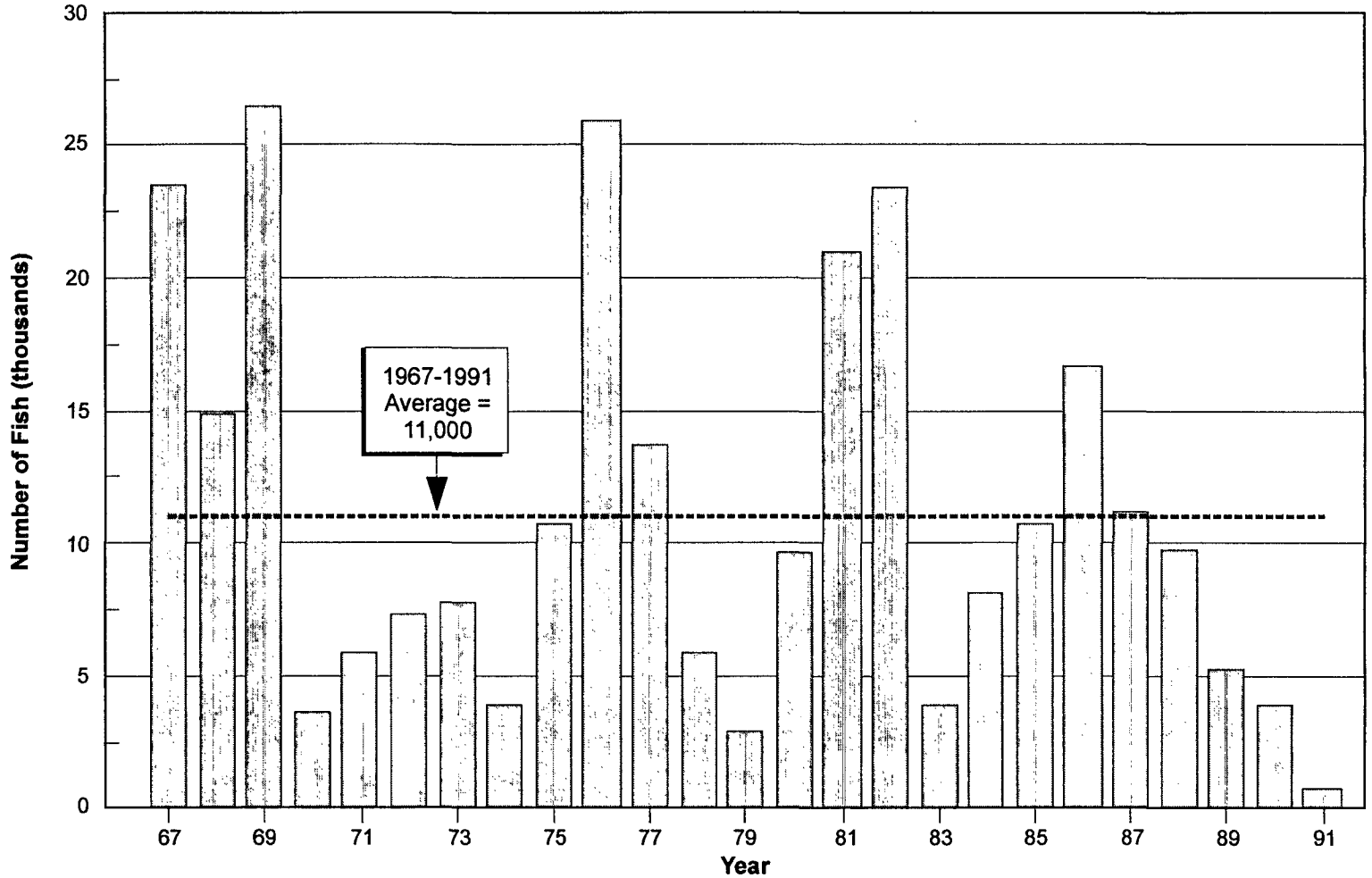
C-081370



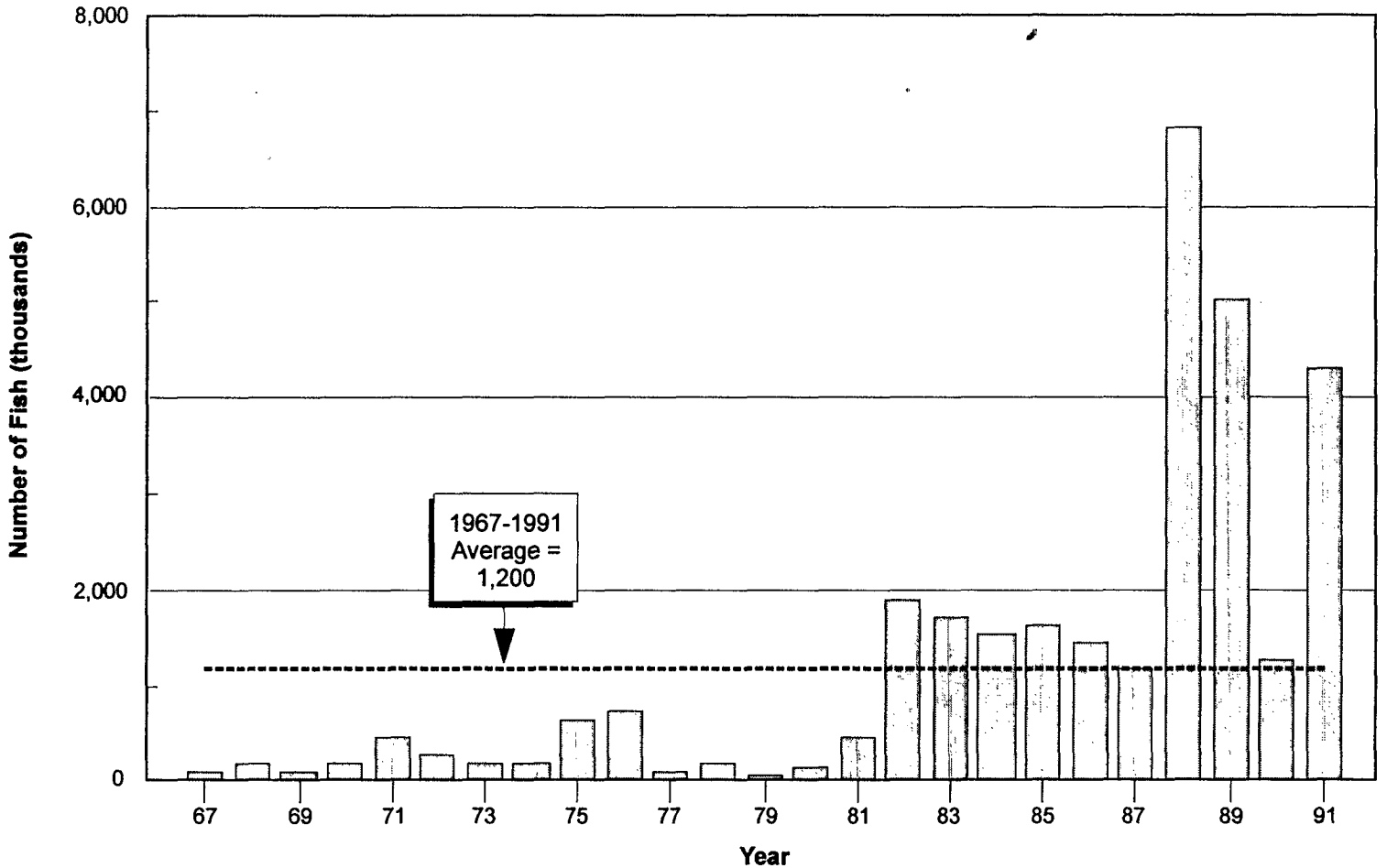
SOURCE:
Mills and Fisher, 1993.

FIGURE II-9

ESTIMATES OF SPRING-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN THE MAINSTEM SACRAMENTO RIVER (1967-1991)

C-081371
C—081371

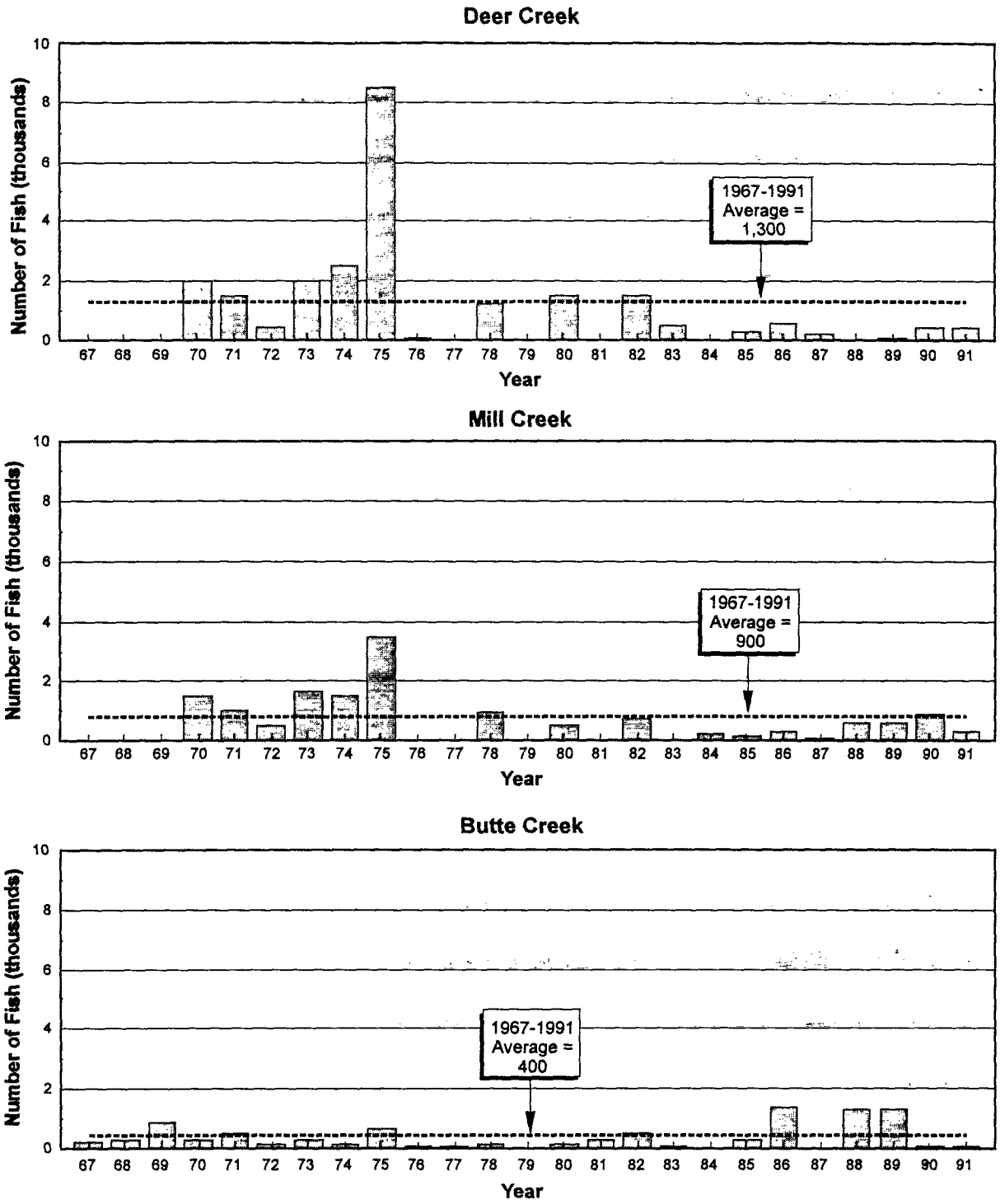
C-081372
C—081372



SOURCE:
Mills and Fisher, 1993.

FIGURE II-10

ANNUAL RETURNS OF SPRING-RUN CHINOOK SALMON TO FEATHER RIVER FISH HATCHERY (1967-1991)



SOURCE:
Mills and Fisher, 1993.

FIGURE II-11

ESTIMATES OF SPRING-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN DEER, MILL, AND BUTTE CREEKS (1967-1991)

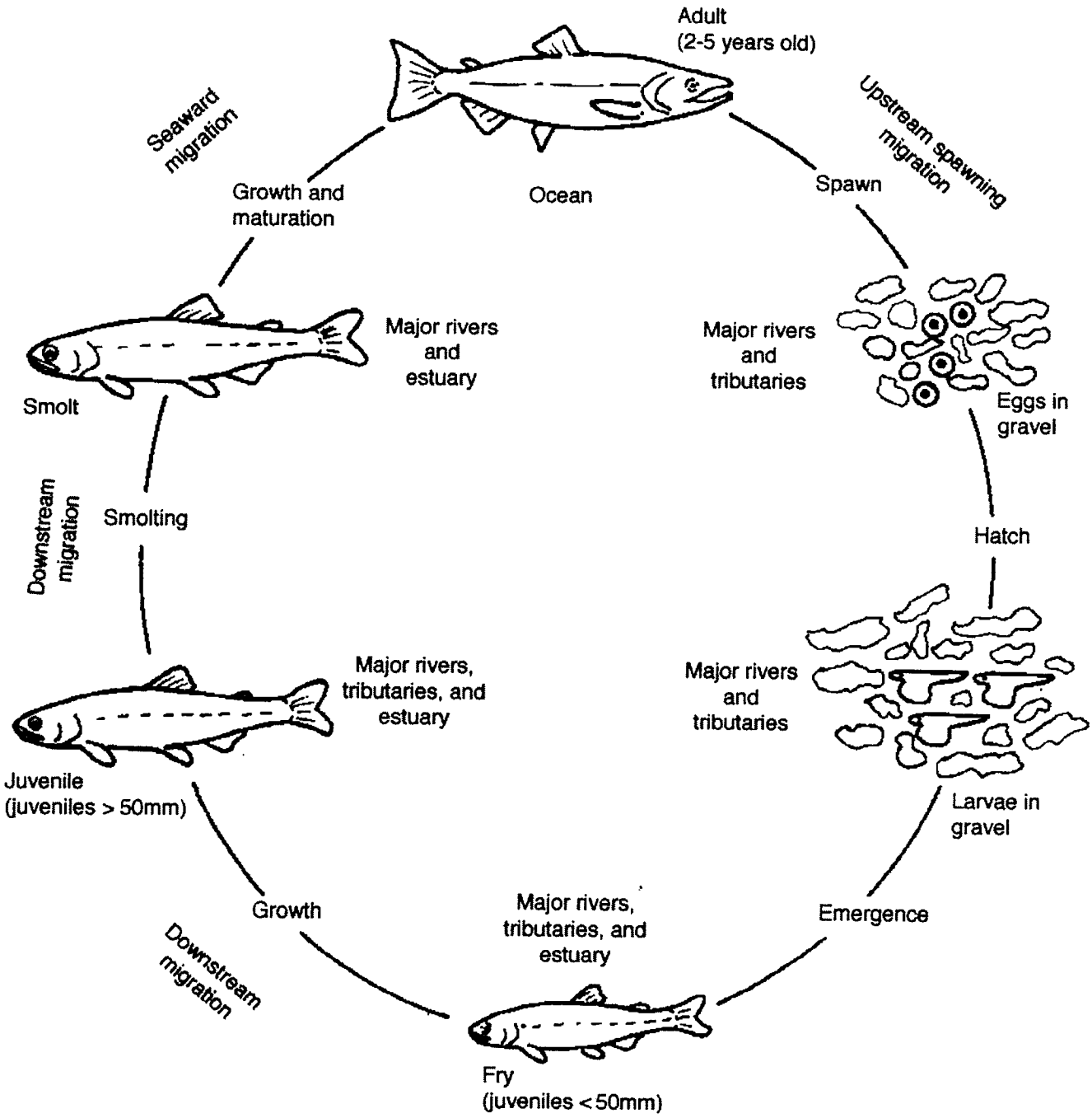


FIGURE II-12

LIFE HISTORY OF CHINOOK SALMON

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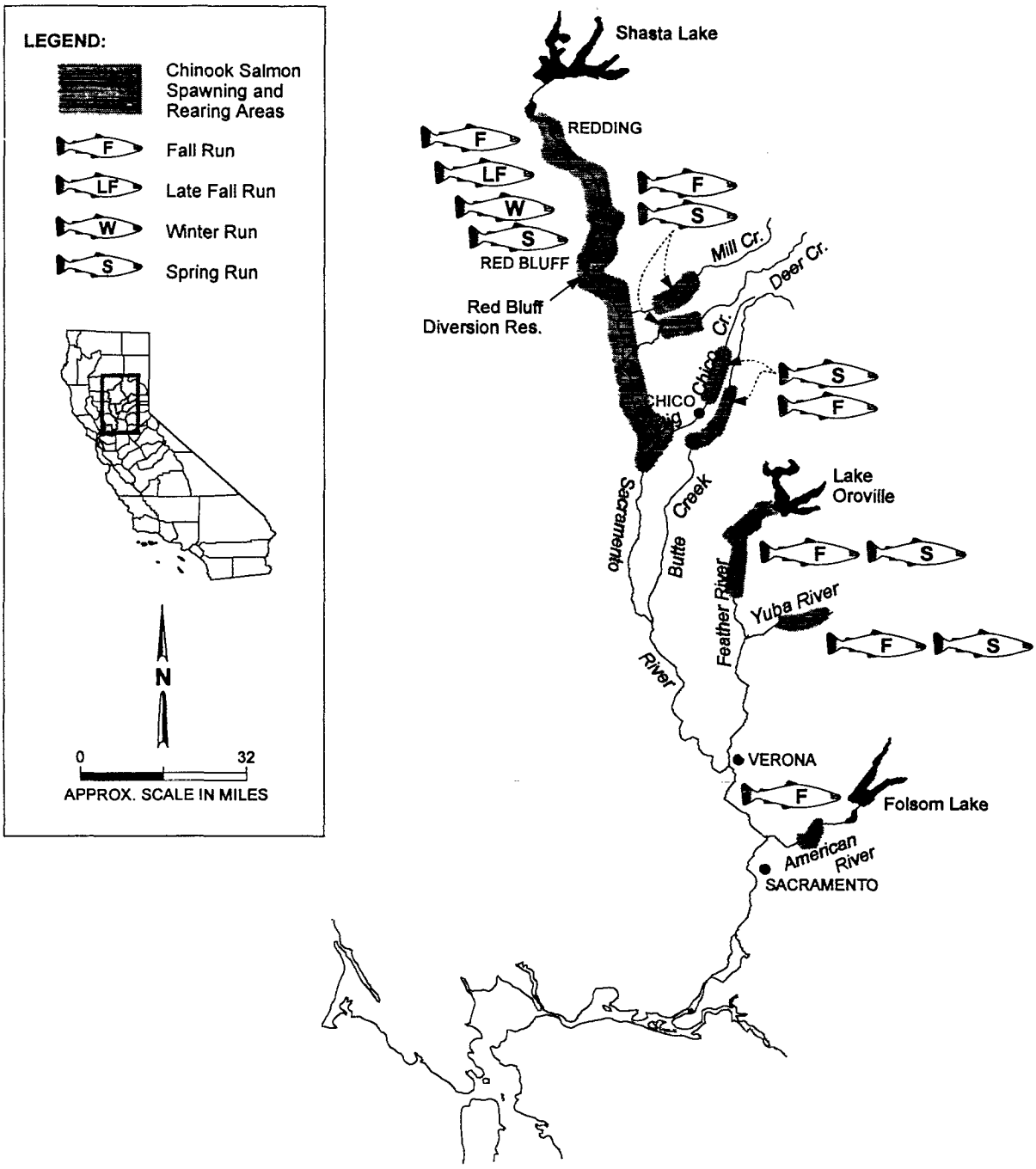
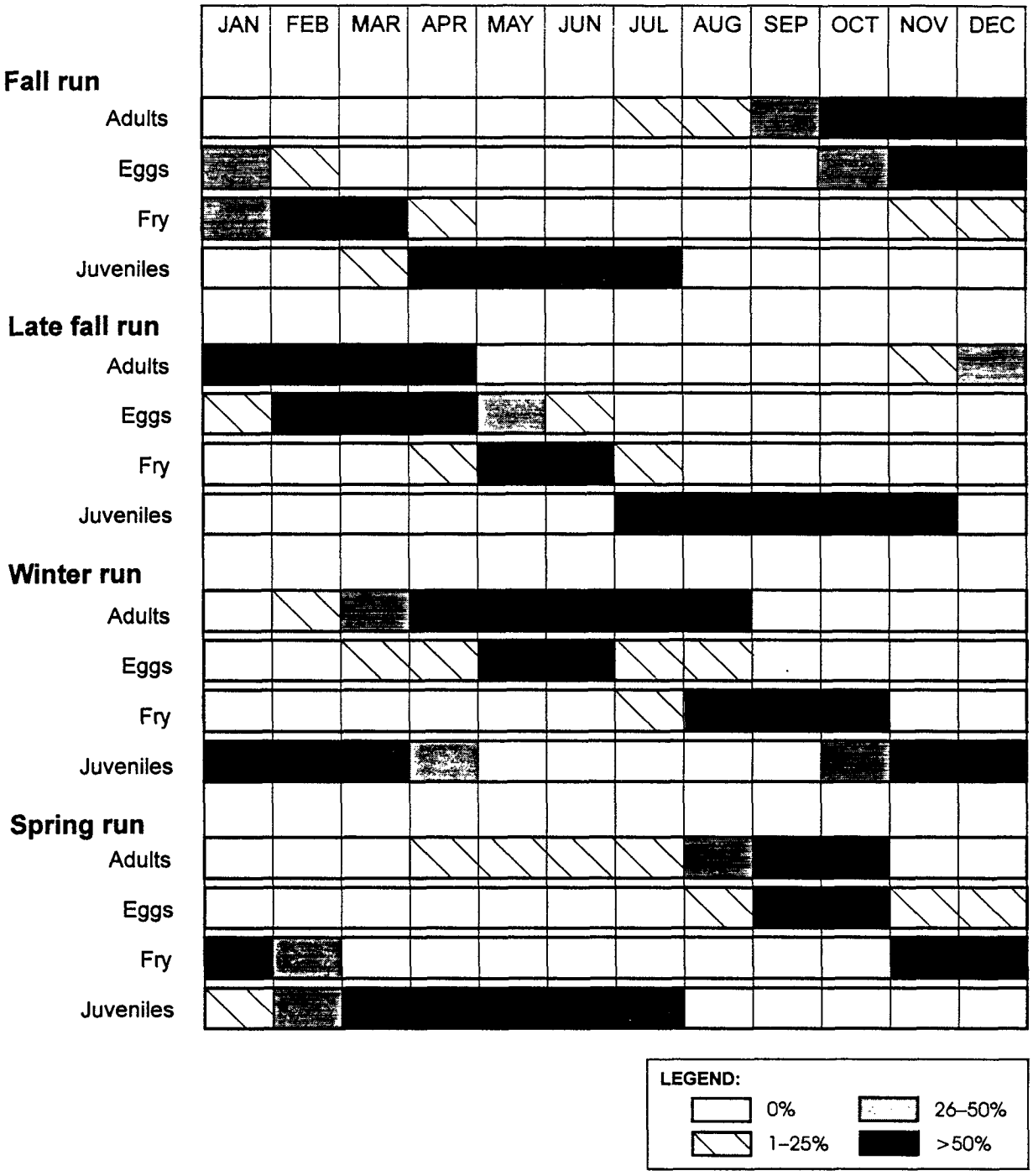
C-081374



SOURCE:
Jones & Stokes Associates, 1992a.

FIGURE II-13

MAJOR CHINOOK SALMON SPAWNING AND REARING AREAS IN THE SACRAMENTO RIVER BASIN



NOTES:
Adults are in cumulative percent.
Other life stages are the percent of year's brood.

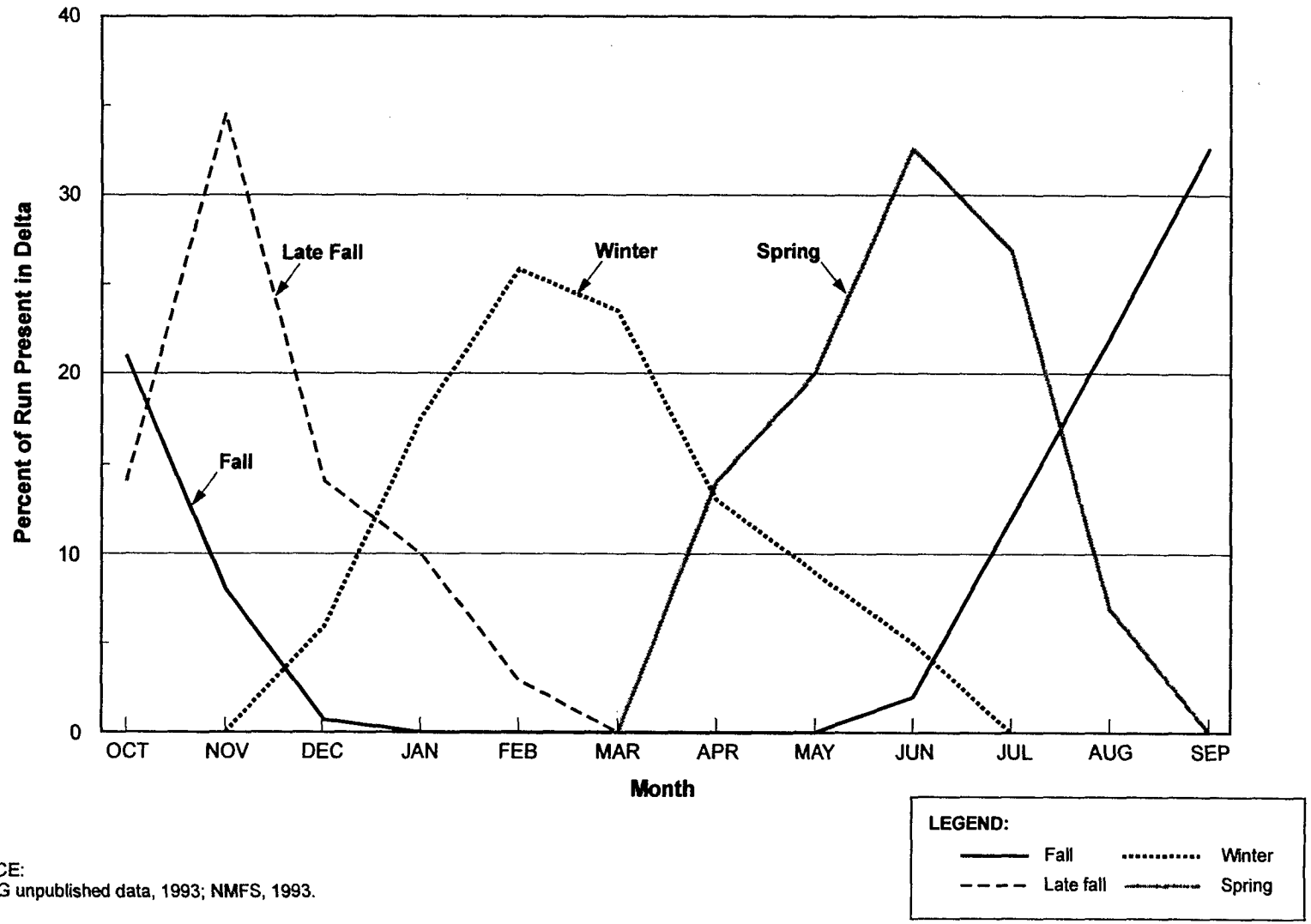
FIGURE II-14

OCCURRENCE OF CHINOOK SALMON BY LIFE STAGE IN THE SACRAMENTO RIVER BASIN

C—081376

C-081376

C-081377

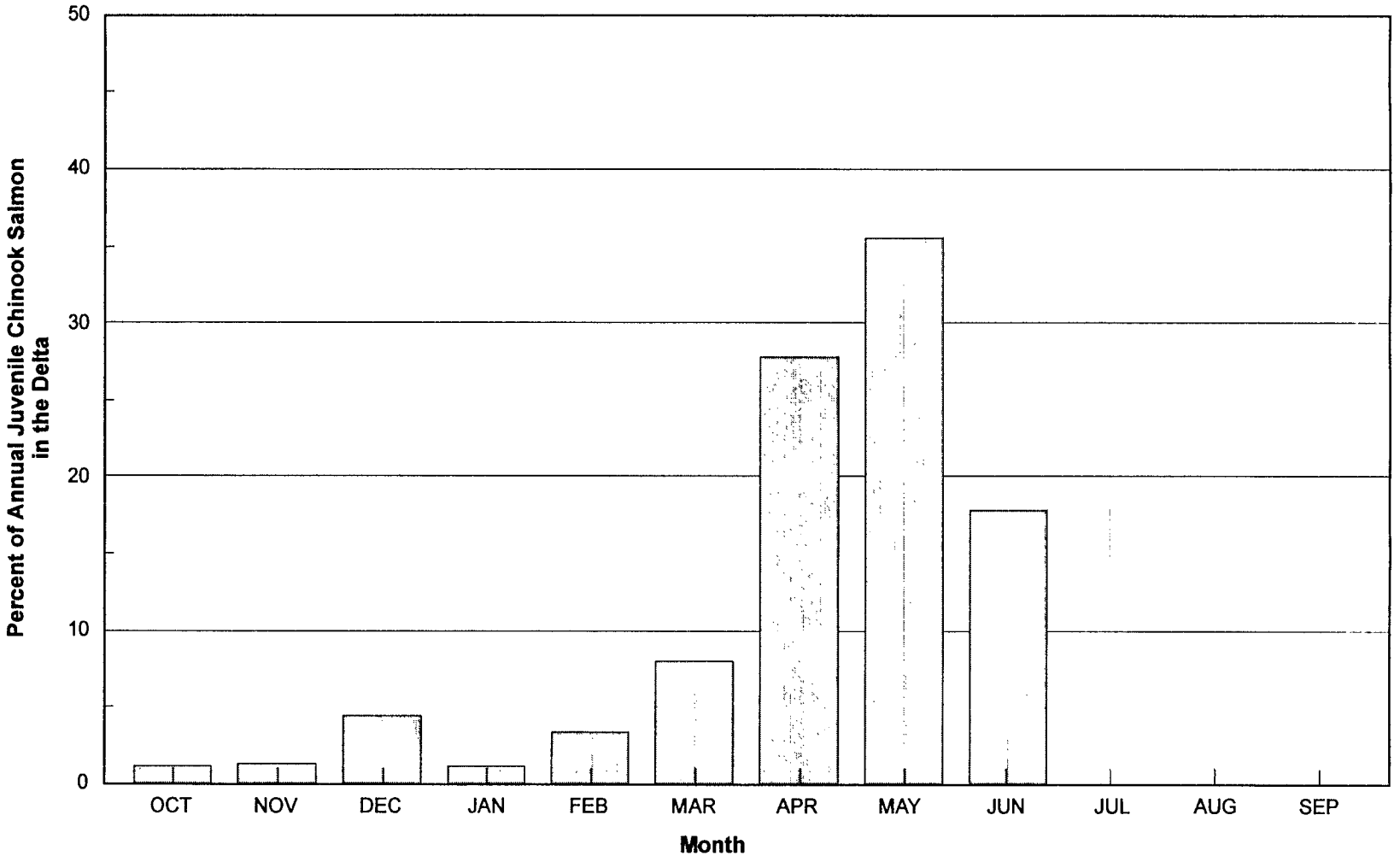


SOURCE:
DFG unpublished data, 1993; NMFS, 1993.

FIGURE II-15

MONTHLY ABUNDANCE OF ADULT CHINOOK SALMON, BY EACH SPAWNING RUN IN THE SACRAMENTO-SAN JOAQUIN DELTA

C—081377



SOURCE:
State Water Project and Federal Water Project fish salvage data, 1981-1988.

FIGURE II-16

MONTHLY ABUNDANCE OF JUVENILE CHINOOK SALMON IN THE SACRAMENTO-SAN JOAQUIN DELTA

C-081378
C-081378

salmon races by life stage in the Sacramento River Basin, the timing of adult upstream migration through the Delta, and the general timing and abundance of juvenile chinook salmon in the Delta, respectively.

Chinook salmon require cold, freshwater streams with suitable gravel for reproduction. Female chinook salmon deposit their eggs in redds, or nests, which they excavate in the gravel bottom in areas of relatively swift water. The eggs are fertilized by one or more males. Fecundity varies among different populations and with body size. All adult chinook salmon die after spawning.

Spawning typically occurs at the lower end of a pool or head of a riffle. Females generally prefer gravel ranging from 1 to 6 inches in diameter, depths exceeding 0.5 foot, and water velocities ranging from 1.5 to 2.5 feet per second (Vogel and Marine, 1991).

Incubation time is inversely related to water temperature. Eggs generally hatch in approximately six to nine weeks, and newly emerged fry remain in the gravel for another two to four weeks until the yolk is absorbed. Maximum survival of incubating eggs and larvae occurs at water temperatures between 41 and 56 degrees Fahrenheit.

After emerging, chinook salmon fry begin to feed and grow in the stream environment. Chinook salmon fry tend to seek shallow, nearshore habitat with low water velocities and move to progressively deeper, faster water as they grow. In streams, chinook salmon fry feed mainly on drifting terrestrial and aquatic insects, but zooplankton become more important in the lower river reaches and estuaries. Juveniles typically rear in freshwater for two to three months before migrating to sea.

Chinook salmon in the Sacramento River appear to exhibit stream-type (spring run) and ocean-type (fall-run) behavior (Healy, 1991). An ocean-type life history pattern is characterized as having juveniles that migrate seaward as smolts in their first year of life. Juvenile chinook salmon typically spend two to three months in freshwater before emigrating as smolts; this is the dominant pattern. Stream-type behavior is indicative of chinook salmon that remain in freshwater for at least one year prior to emigrating as smolts. During the smolting process, juvenile chinook salmon undergo physiological, morphological, and behavioral changes that stimulate emigration and prepare them for ocean life.

Chinook salmon spend two to four years maturing in the ocean before returning to their natal streams to spawn. Most chinook salmon mature at two (primarily males) and three years of age, while a smaller proportion matures at four.

Fall-Run Chinook Salmon. Adult fall-run chinook salmon migrate through the Sacramento-San Joaquin River Delta and into Central Valley rivers from July through December and spawn from October through December. Peak spawning usually takes place in October and November. Egg incubation begins in October and can extend into March, but in some years could occur as late as mid-May.

Chinook salmon fry (juveniles less than 2 inches long) generally emerge from December through March, with peak emergence by the end of January. Generally, fry emigrate from December

through March and smolt from April through June, and a small proportion of the population emigrates as yearlings from October through December.

Two principal movements of juvenile fall-run chinook salmon into the Sacramento-San Joaquin River Delta estuary have been identified. Fry begin entering the estuary in January, with peak abundance occurring in February and March. In general, fry abundance in the Delta increases following high winter flows. A later emigration of smolts takes place from April through June. Fry continue to rear in the upper estuary and emigrate as smolts during the normal smolt emigration period. Smolts arriving in the estuary from upstream rearing areas migrate quickly through the Delta and Suisun and San Pablo bays.

Late Fall-Run Chinook Salmon. Late fall-run salmon migrate into the Sacramento River from October through April and spawn from January through April. Peak spawning activity in February and March is followed by egg incubation from January through June and fry emergence from April through June. Rearing and emigration of fry and smolts occur from April through December.

Winter-Run Chinook Salmon. Adult winter-run chinook salmon migrate through the Delta and up the Sacramento River from December through June. They remain in the river up to several months before spawning from April through July, with peak spawning activity occurring in May and June. Incubation occurs from April through October; however most fry emerge by the end of September. Rearing and emigration begin in July and typically continue through March, but in some years could occur as late as mid-May. Smolts migrate through the Delta mainly from January through March, but in some years could remain as late as mid-May. Figure II-17 summarizes the distribution and relative monthly abundance of winter-run chinook salmon by life stage and location.

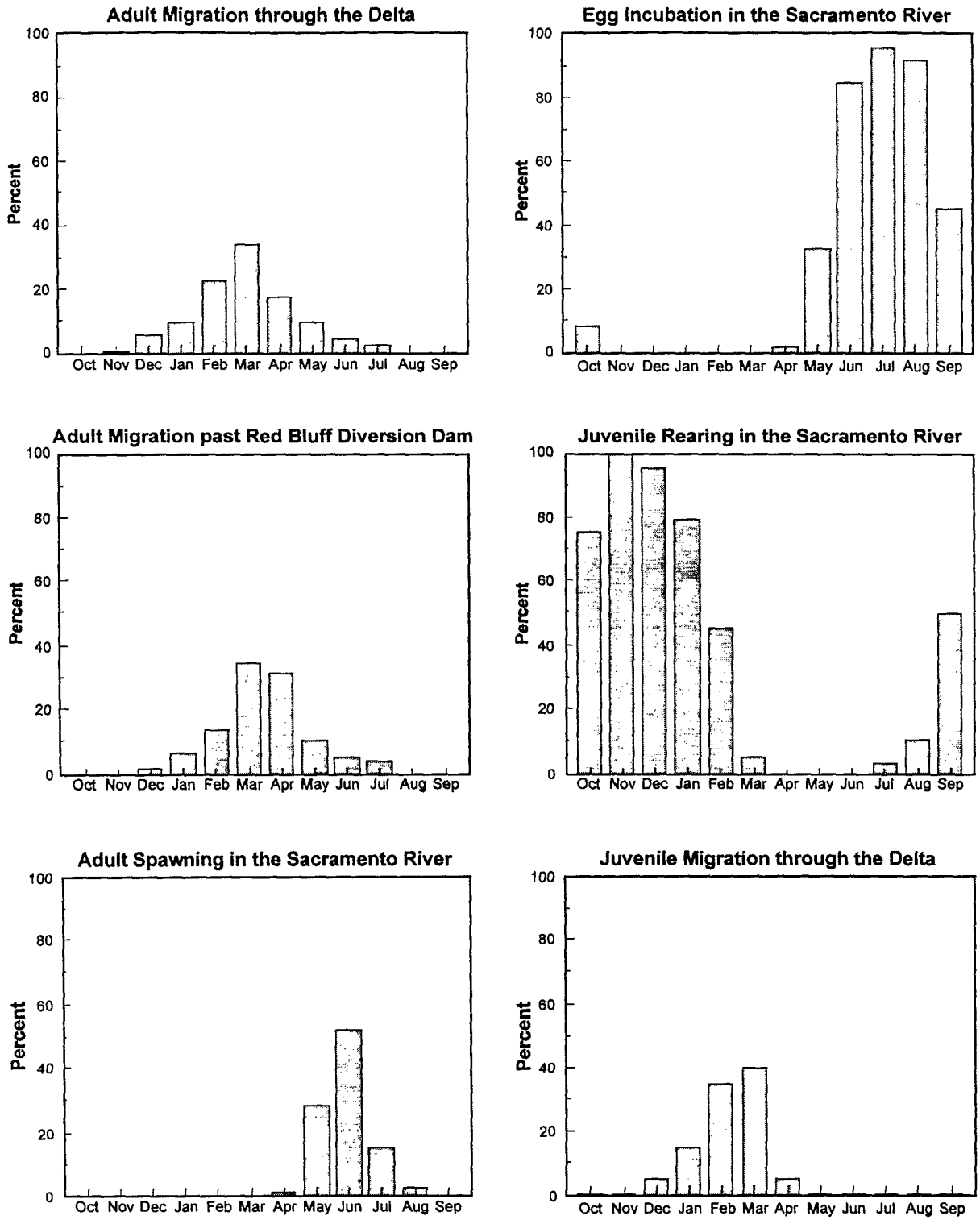
Spring-Run Chinook Salmon. Spring-run chinook salmon adults enter the Sacramento River from March through September and remain there for several months before spawning from August through October; peak spawning activity occurs in September. Incubation takes place from August through December, and most fry emerge from November through January. Rearing and emigration begin in November and continue through April.

Factors Affecting Abundance

Upstream Migration

Flow. Reservoir operations have altered the natural flow regime of Central Valley streams by changing the frequency, magnitude, and timing of flow. These changes potentially affect all chinook salmon life stages. See Attachment B, for a description of hydrologic changes associated with reservoir operation.

Seasonal increases in streamflow provide an important migration cue for adult chinook salmon. Higher flows and associated lower water temperatures in the fall stimulate upstream migration of fall-run chinook salmon. Conversely, low flows and higher water temperatures can inhibit or delay migration to spawning areas. Extremely low or high flows can block or delay migration to spawning areas by preventing passage over shallow riffles or creating excessive water velocities.



SOURCE:
Jones & Stokes Associates, 1992b.

FIGURE II-17

DISTRIBUTION AND RELATIVE MONTHLY ABUNDANCE OF WINTER-RUN CHINOOK SALMON BY LIFE STAGE AND LOCATION

High export pumping and diversion of Sacramento River water into the central and south Delta could increase the number of adult salmon reaching the Sacramento River via the Mokelumne River and DCC or Georgiana Slough. During upstream migration, adult salmon primarily use their sense of smell to find their home stream. Therefore, salmon destined for the Sacramento River that are drawn into the central Delta could be delayed by the longer migration distance and greater number of channels that must be negotiated in this portion of the Delta. Large volumes of Sacramento River water and reverse flows in the lower San Joaquin River can also inhibit or delay migration of San Joaquin River spawners (Hallock et al., 1970). The frequency, magnitude, and timing of reverse flows between 1967 and 1991 are discussed in Attachment B.

Hatchery juveniles that have been transported and released at sites other than the hatchery or stream of origin could fail to imprint properly, which causes high straying rates upon their return as adults. These adults tend to migrate where streamflows are greatest. Cramer et al. (1990) estimated mean straying rates of 7 percent to 86 percent for Feather River and Coleman National Fish Hatchery adults, depending on release location.

Water Quality. Low dissolved oxygen levels in the lower San Joaquin River and Delta are known to delay or block migration of fall-run chinook salmon into the San Joaquin River. See the corresponding San Joaquin River section in this chapter for further discussion.

Water Temperature. Water temperature affects the timing of chinook salmon spawning migrations, although the migratory response to water temperature could differ among chinook salmon races. Upstream migrations of fall-run chinook salmon generally coincide with decreasing water temperatures in the fall. Water temperatures during upstream migration usually range from 51 and 67 degrees Fahrenheit (Bell, 1973). Hallock et al. (1970) found that chinook salmon initiated migration into the lower San Joaquin River as water temperatures declined from 72 to 66 degrees Fahrenheit.

Migration Barriers

Red Bluff Diversion Dam. The RBDD on the upper Sacramento River (Figure II-18) is a major impediment to upstream migration of adult salmon (Hallock et al., 1982; Vogel et al., 1988). After completion of the RBDD in 1966, the proportion of fall-run chinook salmon spawning above the dam declined from an estimated average of 94 percent between 1964 and 1968 to an average of 63 percent between 1977 and 1981 (Reclamation, 1985). The extent of delay and blockage was found to increase with increasing river discharge as a result of decreases in the proportion of total discharge passing through or adjacent to the fish ladders (Figure II-19). Blockage of fall-, late fall-, winter-, and spring-run chinook salmon ranged from 8 percent to 44 percent and can be related to the extent of delay (Figure II-20). Vogel et al. (1988) concluded that adult salmon passage problems at the RBDD were caused primarily by insufficient attraction flows in the fish ladders, operation and maintenance problems, and improper configuration of the fish ladder entrances.

Potential effects of blocked or delayed migration of adult chinook salmon include prespawning mortality, reduced egg viability, and shifts in spawning distribution. Obstructions can cause excessive delay and energy expenditure, which can result in prespawning mortality of adults and reduced fecundity. Fall-run and late fall-run chinook salmon are probably most susceptible to

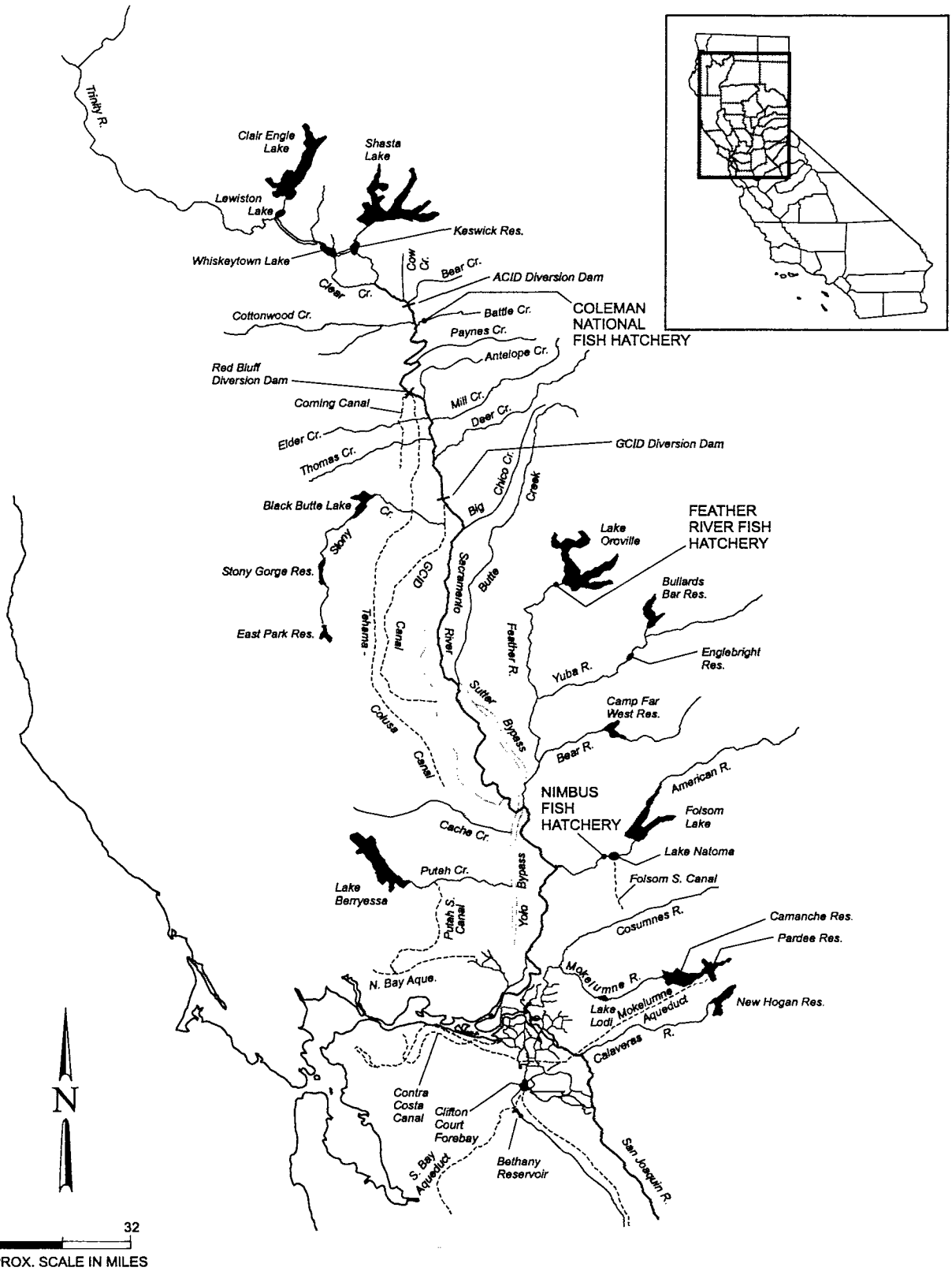
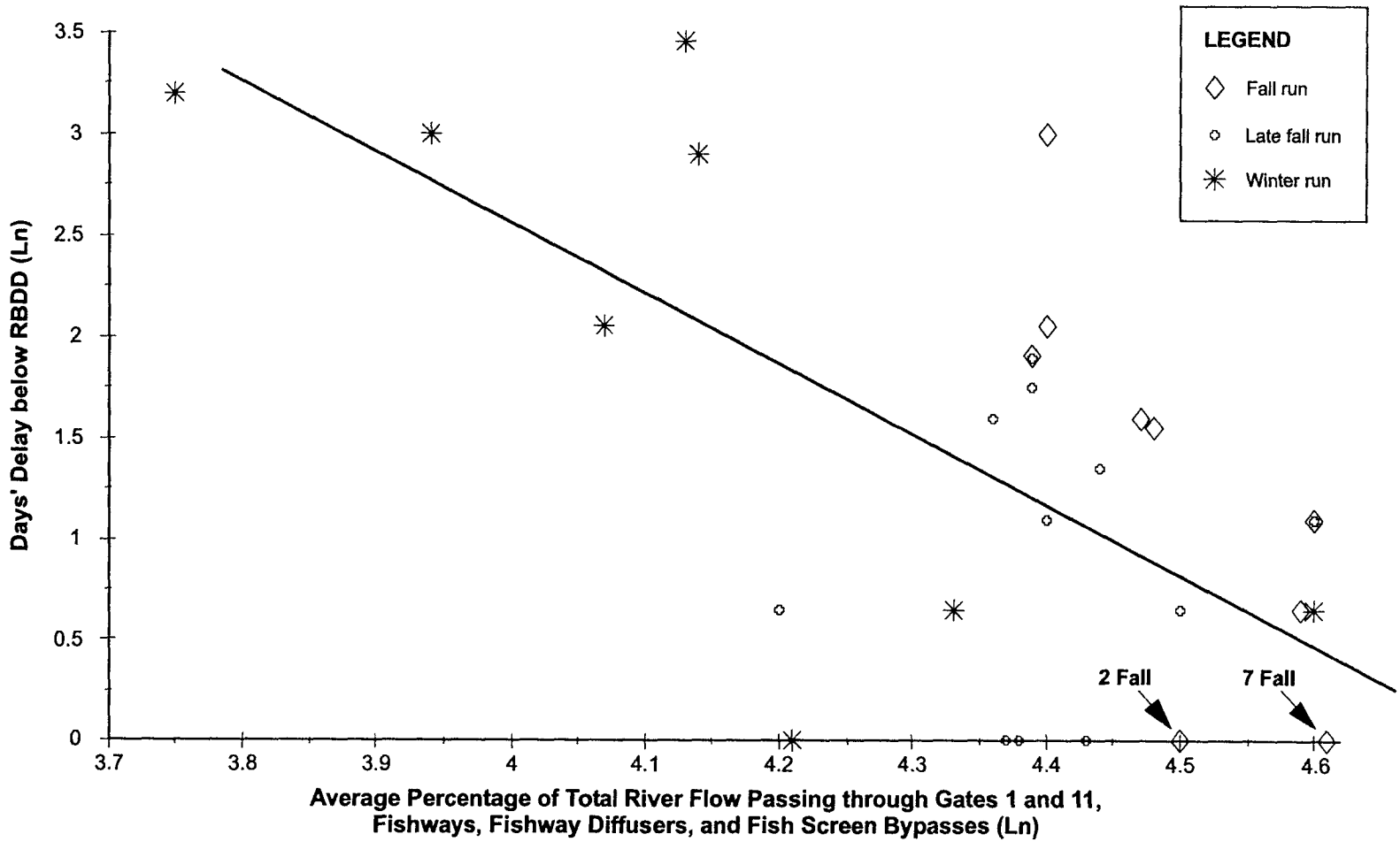


FIGURE II-18

SACRAMENTO RIVER BASIN



NOTE:
All data transformed to natural logarithms.

SOURCE:
Hallock et al., 1982.

FIGURE II-19

RELATIONSHIP BETWEEN DELAY OF RADIO-TAGGED SALMON THAT PASSED RED BLUFF DIVERSION DAM AND MEAN PROPORTION OF TOTAL RIVER FLOW PASSING THROUGH OR NEAR THE FISHWAYS DURING THE PERIOD WHEN THE TAGGED SALMON WERE MIGRATING

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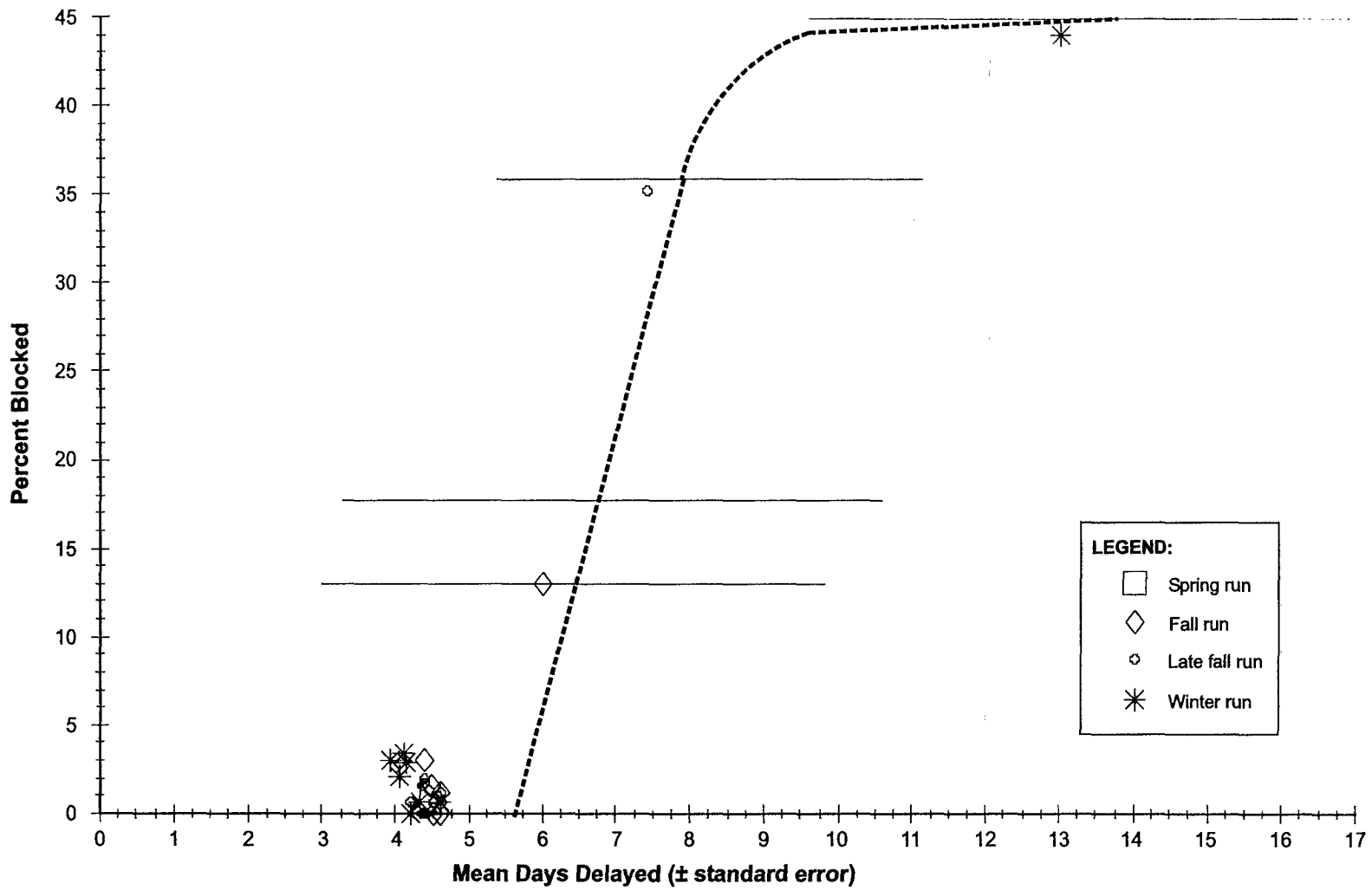

SOURCE:
Vogel and Marine, 1992.

FIGURE II-20

RELATIONSHIP BETWEEN DELAY AND BLOCKAGE OF CHINOOK SALMON AT RED BLUFF DIVERSION DAM

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C—081385

this source of mortality because they spawn immediately after migration. Winter-run chinook salmon that do not reach spawning areas above the dam generally have poor spawning success because water temperatures in the Sacramento River below the RBDD frequently exceed tolerance levels for eggs and fry during the summer incubation period (Hallock and Fisher, 1985).

The RBDD gates are raised during the nonirrigation season (November 1 until April 30) to facilitate upstream passage of adult winter-run chinook salmon. Reclamation is investigating alternatives that will permit the RBDD gates to be raised permanently or for longer periods (such as found in Alternative 6 [September 15 to May 14]) to provide unimpeded passage of adult and juvenile chinook salmon.

Anderson-Cottonwood Irrigation District Diversion Dam. The ACID diversion dam, a seasonal flashboard dam on the Sacramento River near Redding (Figure II-18), has caused fish passage problems since its construction in 1917. A fish ladder, completed in 1927 and still in place today, does not effectively attract and convey upstream migrating chinook salmon past the dam (Reclamation, 1983a). A new fishway was recently installed on the opposite side of the dam, but its passage effectiveness has not yet been evaluated. The ACID dam is usually installed in early April and removed in late October or early November, resulting in potential delay and blockage of winter-, spring-, and fall-run chinook salmon to upstream spawning areas.

Passage at Natural Riffles. Minimum depths are necessary for successful upstream migration of adult salmon. For chinook salmon, Thompson (1972) recommended that a minimum depth of 0.8 foot extend over at least a 10 percent continuous portion of the stream's cross-sectional profile. In addition, the minimum depth should extend over at least 25 percent of the stream's cross-sectional profile overall. Application of these criteria to shallow riffles in the Yuba River indicated that streamflow should exceed 100 cfs to ensure adequate passage depths (Beak Consultants, 1989). Passage at natural riffles has been identified as a potential problem, as in many of the smaller tributaries such as Mill and Deer creeks (Service, 1995).

Suisun Marsh Salinity Control Structure. The Suisun Marsh Salinity Control Structure, designed to improve water quality in Montezuma Slough and Suisun Marsh during periods of low to moderate Delta outflow, could delay upstream migration of adult chinook salmon and other anadromous species when it is operating (Herrgesell, 1993) (Figure II-21). Studies are underway to examine whether any impacts occur.

Sport Fishing and Poaching. From 1977 to 1981, the average sport catch of fall-run chinook salmon in the Sacramento River was 1.8 percent of the total estimated run (Allen and Hassler, 1986).

DFG initiated a four-year program in 1990 to estimate annual angler effort and catch of salmon and steelhead trout in seven river reaches covering 420 miles of the Sacramento River Basin, including the reach from the Carquinez Bridge to Sacramento. Table II-4 shows estimated annual catch of chinook salmon (excluding fish released) for each survey reach and period.

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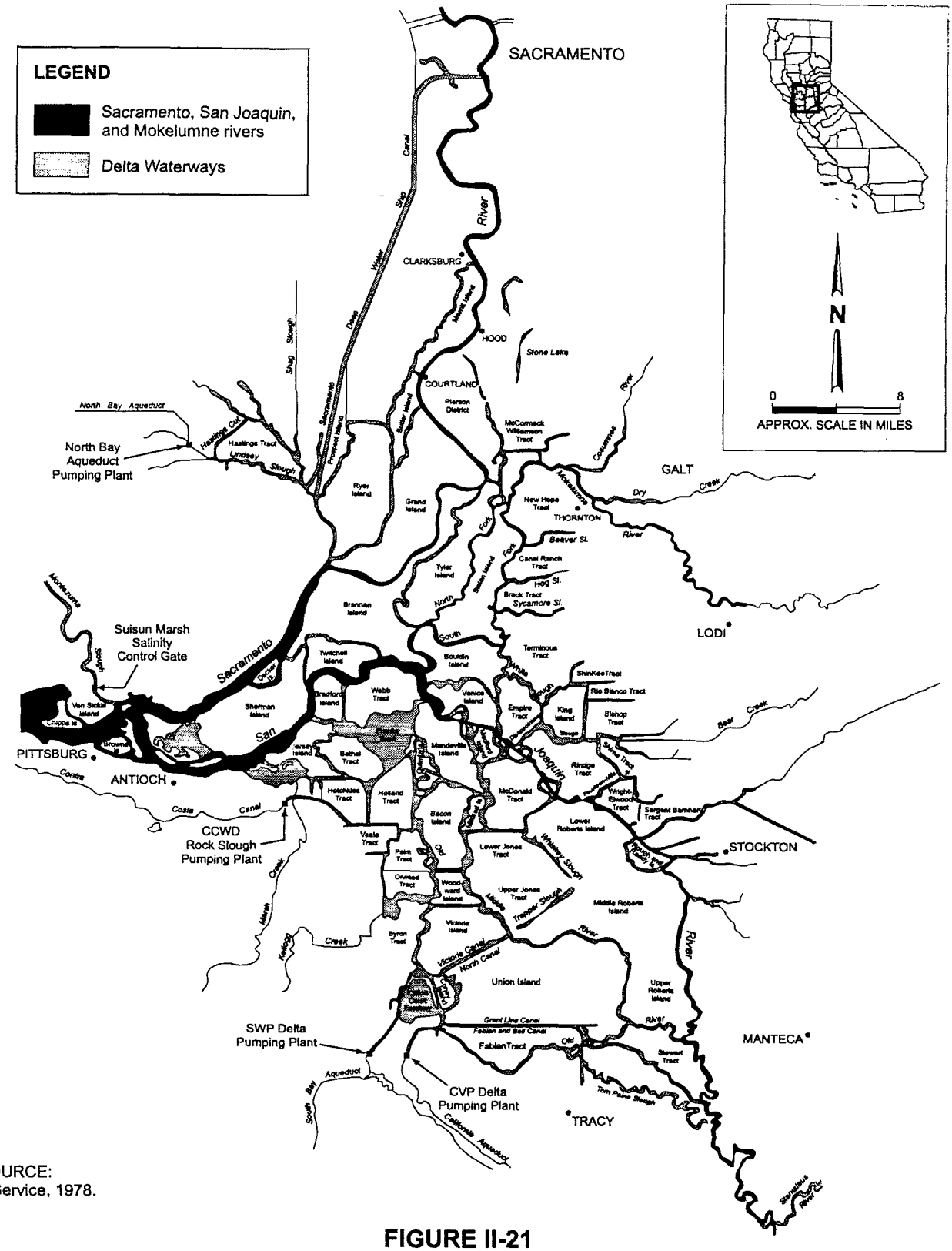

SOURCE:
Service, 1978.

FIGURE II-21

SACRAMENTO-SAN JOAQUIN DELTA

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TABLE II-4
ESTIMATED CHINOOK SALMON SPORT LANDINGS FOR SIX SACRAMENTO RIVER BASIN REACHES, 1990-1993

Period	Carquinez Bridge to Sacramento	Sacramento to Colusa	Colusa to Red Bluff	Red Bluff to Redding	Feather River	American River
January 1, 1991-December 31, 1991	1,588	1,402	4,302	6,167	9,150	12,448
January 1, 1991-December 31, 1992	2,079	1,323	2,911	3,278	4,694	5,788
January 1, 1992-December 31, 1993	3,191	2,093	4,105	2,804	8,476	17,054

NOTE:
Numbers of fish landed exclude fish released.

SOURCE:
DFG, 1995.

Poaching, particularly during low flows, is another source of mortality for upstream migrating chinook salmon.

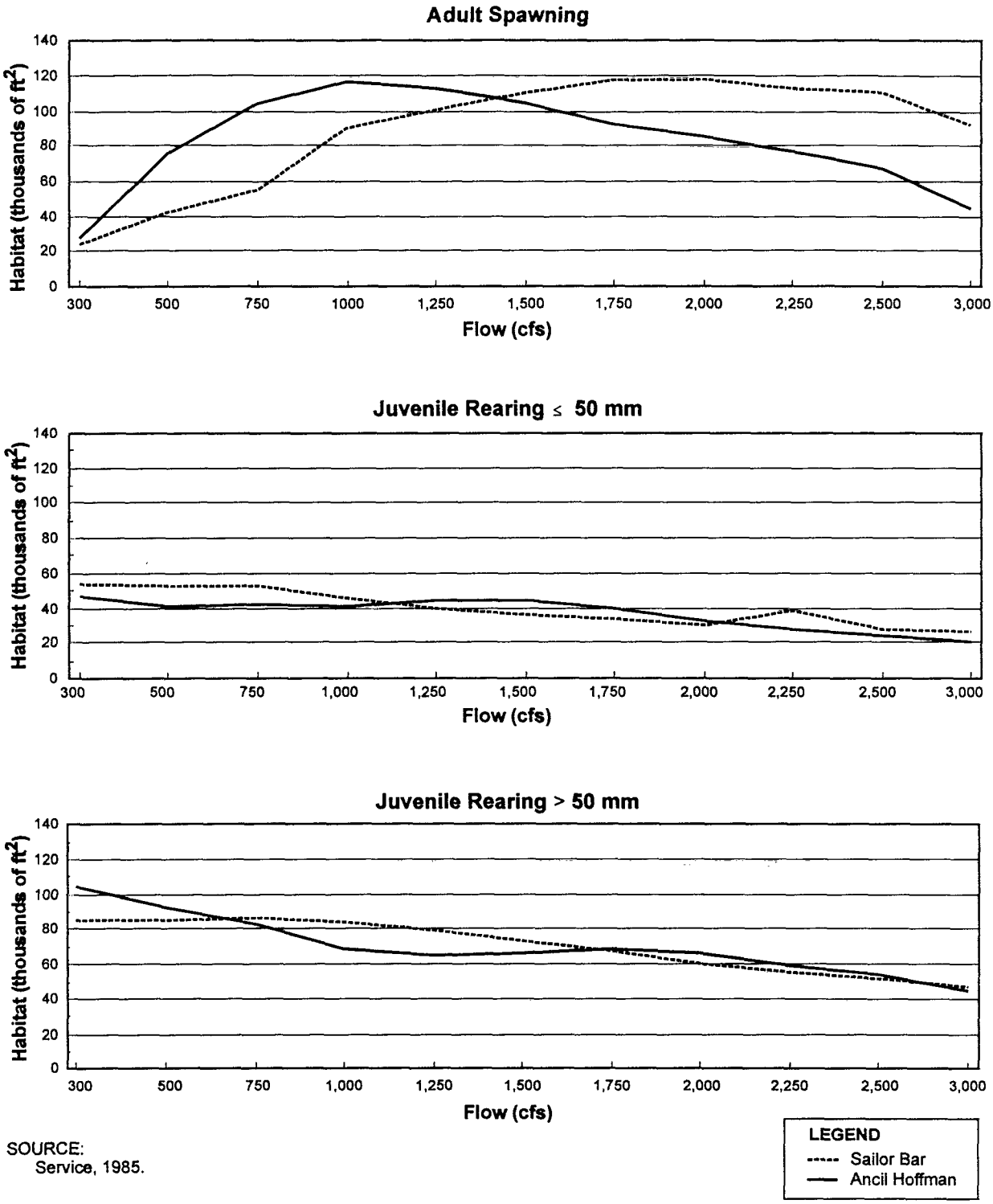
Spawning

Flow. Streamflow influences the quantity, quality, and distribution of chinook salmon spawning habitat. Streamflow directly affects the amount of available spawning habitat by determining the stream area with appropriate combinations of water depths, velocities, and streambed characteristics (e.g., substrate composition). Indirect effects of flow on spawning habitat include effects on water quality; water temperature, which can influence the longitudinal extent and seasonal availability of suitable spawning habitat; flow reductions during the incubation period that cause inadequate intragravel flow and dewatering; and flushing flows that remove harmful quantities of sediment and plant growths from the spawning gravels.

Relationships between streamflow and chinook salmon spawning habitat availability have been developed for several streams in the Sacramento River Basin through application of the instream flow incremental methodology (IFIM) (Bovee, 1982) and related techniques. The results have formed the basis for assessing instream flow requirements or evaluating alternative operations and reservoir release schedules. Habitat-discharge relationships are currently available for the American (Service, 1985) and Yuba (Beak Consultants, 1989) rivers (Figures II-22 and II-23) and are being developed for the Feather and upper Sacramento rivers.

Water Temperature. Mature females exposed to water temperatures above 60 degrees Fahrenheit for prolonged periods experience poor survival and produce less viable eggs than females subjected to lower water temperatures. Extremely cold water (less than 38 degrees Fahrenheit) also results in poor adult survival and egg viability (Hinze, 1959).

Water temperatures limit the geographic range where chinook salmon can successfully spawn. In the upper Sacramento River, high water temperatures observed during the summer and fall limit the range of successful spawning for winter-, spring-, and fall-run salmon during the July through October period (Vogel and Rectenwald, 1987). The downstream limit of suitable water



SOURCE:
Service, 1985.

FIGURE II-22

RELATIONSHIP BETWEEN FLOW AND HABITAT AVAILABILITY FOR CHINOOK SALMON SPAWNING AND JUVENILE REARING IN SELECTED REACHES OF THE LOWER AMERICAN RIVER

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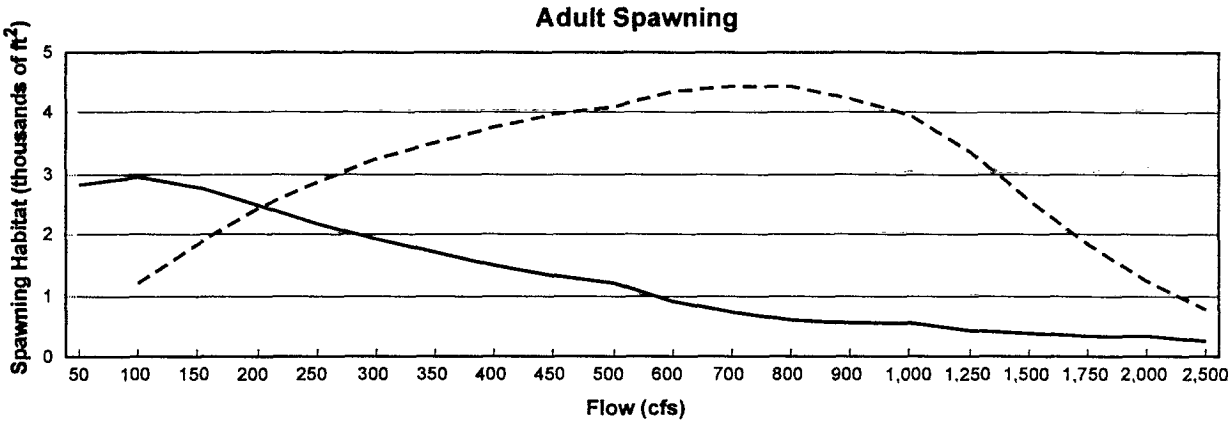

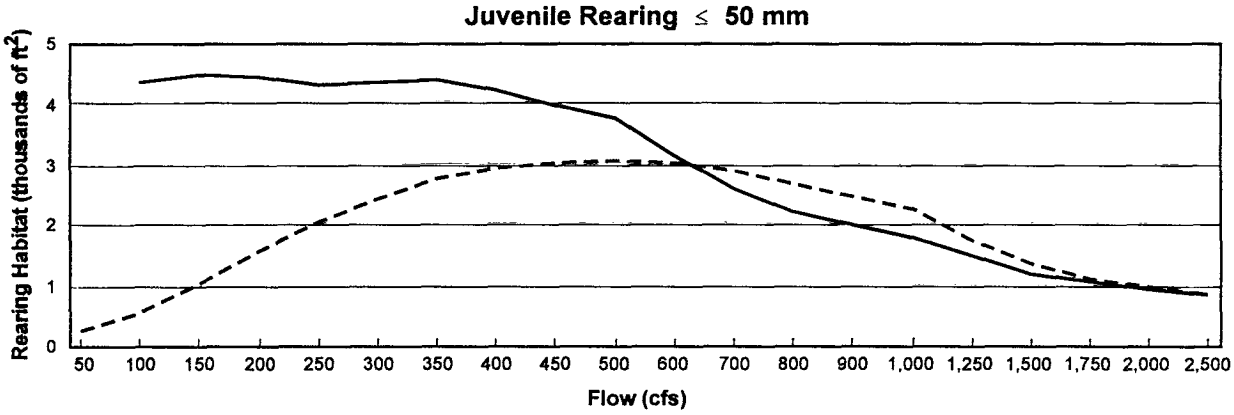

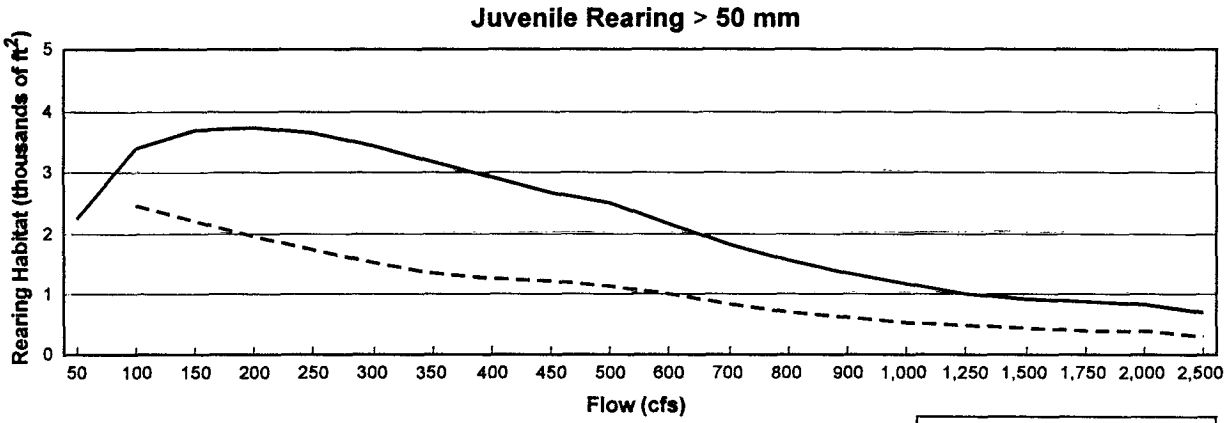

LEGEND
- - - Garcia Gravel Pit
——— Daguerre Point Dam

SOURCE:
Beak Consultants, 1989.

FIGURE II-23

RELATIONSHIP BETWEEN FLOW AND HABITAT AVAILABILITY FOR CHINOOK SALMON SPAWNING AND JUVENILE REARING IN SELECTED REACHES OF THE YUBA RIVER

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temperatures for fall-run chinook salmon in most years is near Hamilton City, whereas suitable temperatures for winter- and spring-run salmon are typically limited to the reach above the RBDD (Figure II-18). Fall-run chinook salmon in the American River are adversely affected by water temperature during dry years and low water elevations at Folsom Reservoir. High prespawning water temperatures are also a problem in the Yuba, Mokelumne, Tuolumne, Merced, and San Joaquin rivers. High temperatures in the San Joaquin River could delay the upstream return of spawners to the Merced, Tuolumne, and Stanislaus rivers for up to several weeks.

Water temperature effects on reproductive success are discussed further under the Egg Incubation and Emergence section of this chapter.

Spawning Gravel. Construction of Shasta and Keswick dams blocked the recruitment of spawning gravels from upstream sources to the upper Sacramento River. Lack of gravel recruitment and increases in the average size of streambed materials have degraded spawning habitat below Keswick Dam to at least Clear Creek. Below Clear Creek, tributary streams increase in importance as a source of spawning gravels to the Sacramento River. Intensive gravel mining in most of these tributaries has reduced gravel recruitment to the mainstem Sacramento River by more than 50 percent. Below Red Bluff, gravel recruitment occurs principally from the natural erosion of historical deposits along the banks of the Sacramento River. Bank protection and levee projects in the middle and lower Sacramento River have substantially reduced gravel recruitment into these reaches. (Buer et al., 1984.) Impoundments on the tributaries have also decreased the gravel recruitment from the tributaries.

Existing gravel supplies are adequate to support current population levels of chinook salmon in the upper Sacramento River. With future population increases, however, spawning gravel could become limited and gravel restoration will be necessary. Recent restoration efforts by DFG and DWR (1992) have included placement of spawning gravel to restore degraded spawning riffles in the upper Sacramento River above Clear Creek, and gravel restoration projects to replace or improve riffles in the Merced and Tuolumne rivers. Minor projects have been completed on the Mokelumne, American, and Feather rivers.

Egg Incubation and Emergence

Water Temperature. Appropriate water temperatures for egg incubation and emergence are a critical concern for Sacramento River chinook salmon. Historically, early fall water temperatures were warmer in the lower reaches of the upper Sacramento, Feather, Yuba, and American rivers, particularly during dry water years. Spring-run chinook salmon was the dominant race and spawned at higher elevations where temperatures were not a major limiting factor. Fall-run chinook salmon spawned at lower elevations in the fall to avoid lethal water temperatures. Nevertheless, water temperatures likely affected early spawning fall-run chinook salmon, increasing mortality of eggs during all but wet years. In general, immediately after dam construction, reservoirs were kept relatively high and provided colder water in the lower reaches of these rivers. Fall-run chinook salmon populations responded to the colder flows earlier in the year, mixed genetically with hatchery salmon, and began to spawn much earlier than historical salmon runs. Coincidentally with these earlier runs, most Sacramento River Basin reservoirs have, over time, reached lower elevations because of greater demands for spring and summer

releases for agricultural and municipal demands. These lower elevations, particularly during dry water years, now frequently result in warm water being released from the reservoirs, which causes high mortalities to incubating fall-run chinook salmon eggs.

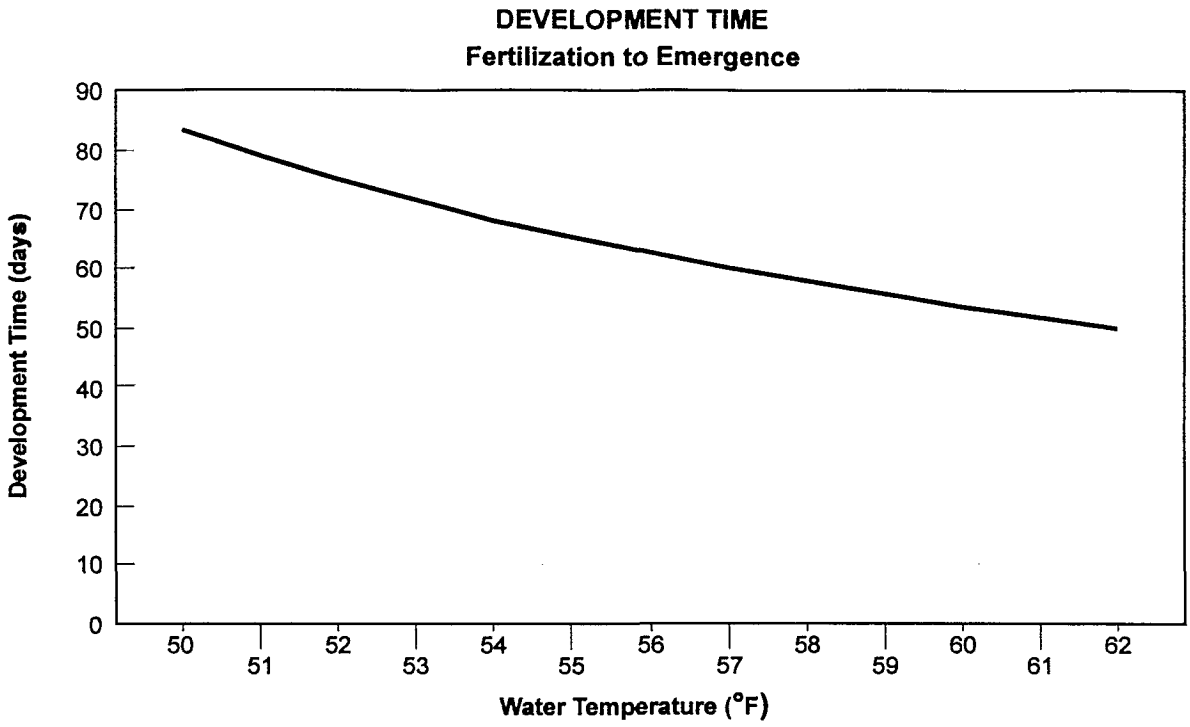
Incubation time declines with increasing water temperatures (Figure II-24). Maximum survival of incubating eggs and yolk-sac larvae occurs at water temperatures between 41 and 56 degrees Fahrenheit. At constant water temperatures, survival through emergence decreases at water temperatures exceeding 56 degrees Fahrenheit, with no survival occurring at 62 degrees Fahrenheit or higher (Figure II-24). The effects of hourly or daily fluctuations in water temperature above 56 degrees Fahrenheit on eggs and yolk-sac larvae are largely unknown.

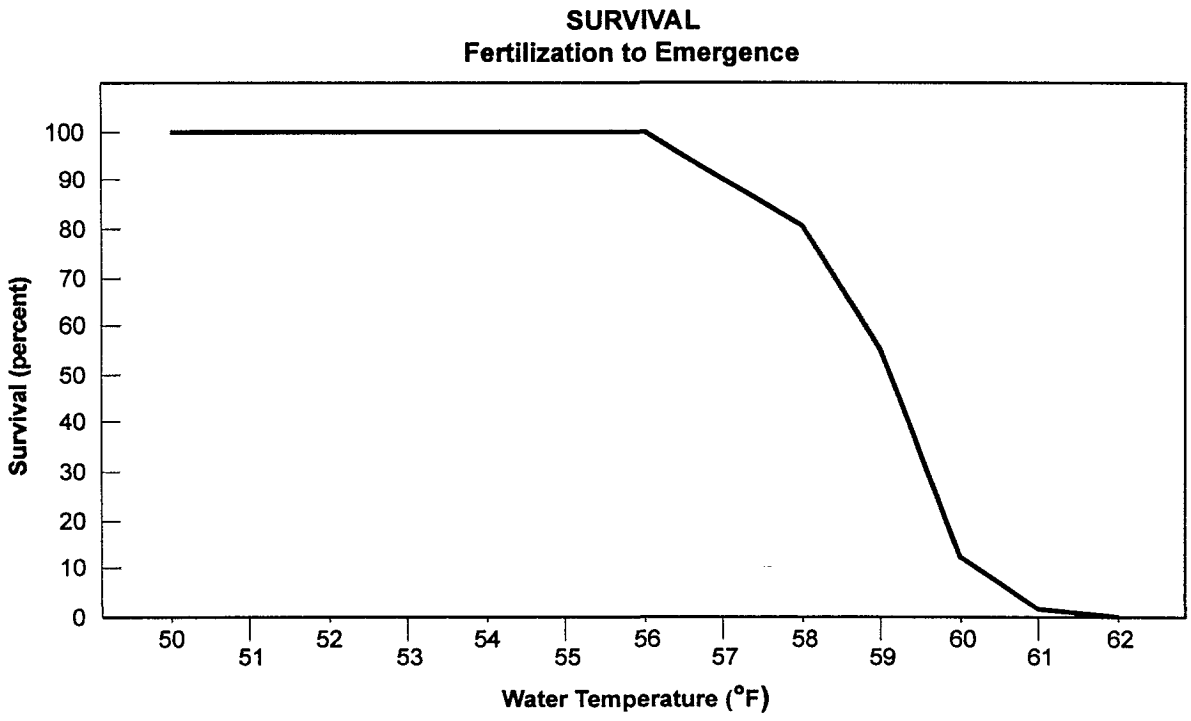
Increasing water demands and prevailing drought conditions in recent years have limited the ability to maintain suitable water temperatures in the principal winter-run chinook salmon spawning area in the upper Sacramento River. Changes in the frequency of deleterious water temperatures between 1967 and 1991 during the summer incubation period are described in Attachment B. To reduce the severity and extent of deleterious water temperatures during the recent drought period, Reclamation initiated alternative reservoir operations, including increases in the relative amount of cold water from the Trinity River system and low-level bypass releases at Shasta Dam. Operation of the temperature control structure at Shasta Dam will improve Reclamation's ability to control water temperatures and significantly benefit winter-run, spring-run, and early spawning fall-run chinook salmon without forgoing power generation. Temperature control devices have also been installed at Lewiston and Whiskeytown lakes.

Water Quality. Hatching success is adversely affected by reductions in dissolved oxygen and increases in metabolic waste products resulting from inadequate water flow through the redd. Inadequate intragravel flow could be caused by streamflow reductions following spawning or by increases in the quantity of fine sediments in the gravel. Incubating eggs and larvae require dissolved oxygen at saturation levels. Optimum levels equal or exceed 8 milligrams per liter (mg/l) at temperatures between 44 and 50 degrees Fahrenheit and equal or exceed 12 mg/l at temperatures above 50 degrees Fahrenheit (Raleigh et al., 1986).

Water quality impacts on aquatic resources vary by location and season in response to variable streamflows and pollutant levels in point and nonpoint agricultural, municipal, and industrial discharges. Although largely unquantified, water quality impacts on fish populations in the Sacramento River and its tributaries include effects related to heavy metal pollution, high suspended sediment levels, and elevated nutrient, herbicide, and pesticide levels from agricultural drainage.

Simpson Paper Company, which operates a pulp and paper mill near Anderson, has achieved an approximate 98 percent reduction in the discharge rate of dioxins and related compounds in recent years. As a result, dioxin concentrations in fish tissues from the Sacramento River have been reduced 80 to 90 percent, and the current health advisory on consumption of fish taken from the Sacramento River between Redding and Red Bluff was lifted. (Sacramento River Information Center, 1993.)





SOURCE:
Reclamation, 1992a.

FIGURE II-24

CHINOOK SALMON EGG AND LARVAL DEVELOPMENT TIME AND SURVIVAL VERSUS WATER TEMPERATURE

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Flow Fluctuations. Lowering river levels during the chinook salmon incubation period can cause mortality of eggs and alevins by dewatering redds, reducing flow rates through the redd, or increasing water temperatures. For example, fall-run chinook salmon redds are subject to potential dewatering as a result of streamflow reductions during the reservoir storage phase, which could begin during the winter incubation period. Redd dewatering impacts have been assessed using stage-discharge relationships for known spawning areas and chinook salmon spawning depth criteria.

Juvenile Rearing

Flow. Streamflow is a dominant variable affecting chinook salmon rearing habitat. Streamflow directly determines the amount of physical habitat with appropriate combinations of depth, velocity, substrate, and cover for chinook salmon rearing. Streamflow also influences the extent of suitable water temperatures, water quality conditions, and habitat for production of aquatic invertebrates, a major food source for juvenile salmonids in freshwater.

The habitat preferences of juvenile chinook salmon change with increasing body size; newly emerged chinook salmon fry typically occur along the shallow edges of streams but seek faster, deeper water as they grow (Lister and Genoe, 1970; Everest and Chapman, 1972). Generally, chinook salmon fry prefer depths of 0.5 to 3 feet and water velocities of 0.1 to 1 foot per second (Raleigh et al., 1986).

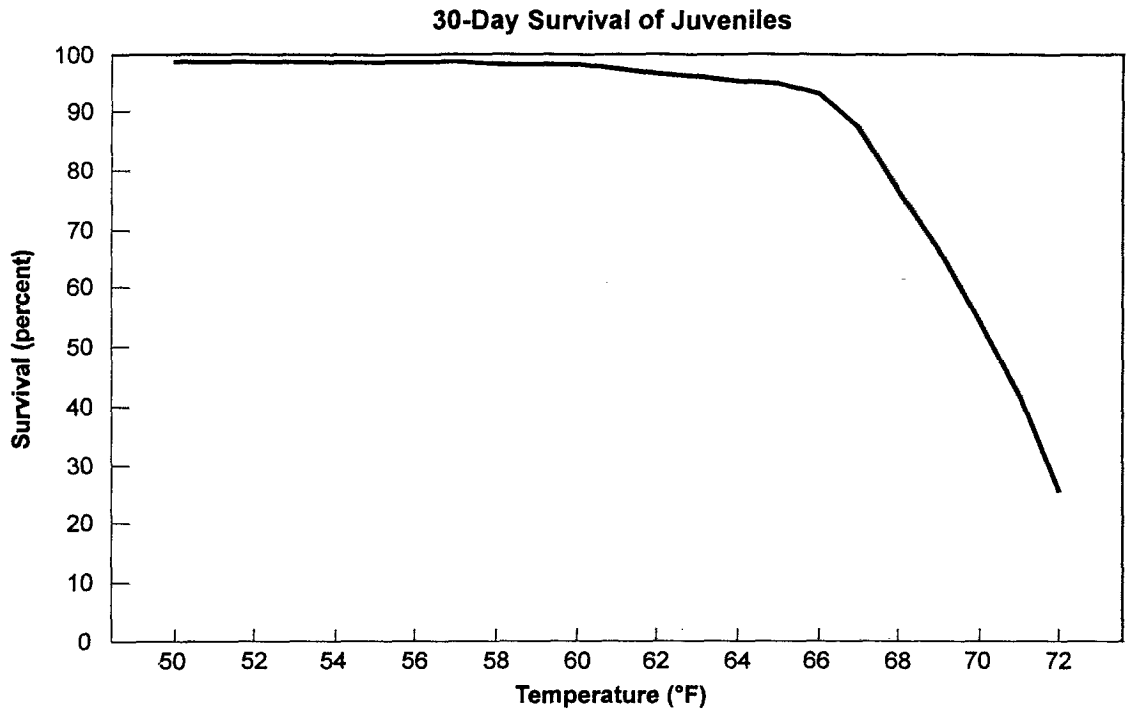
Relationships between streamflow and juvenile rearing habitat have been developed for the American and Yuba rivers through application of Physical Habitat Simulation (PHABSIM) (Bovee, 1982) (Figures II-22 and II-23).

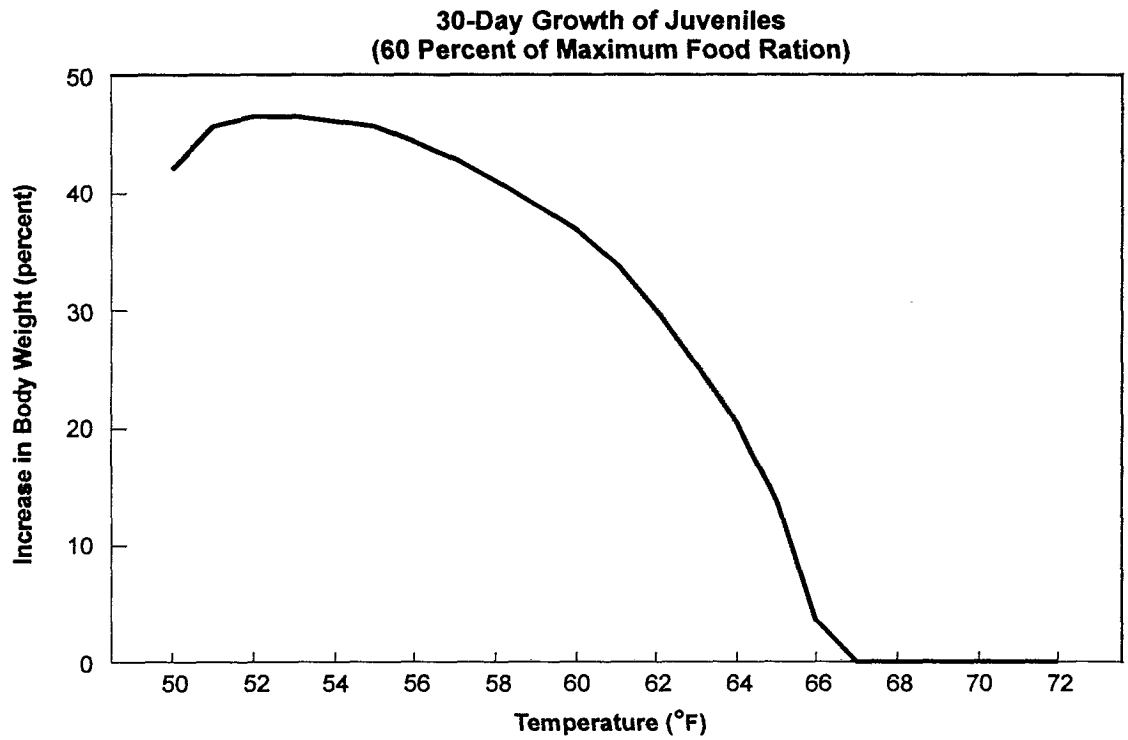
Water Temperature. In general, juvenile chinook salmon tolerate water temperatures from 32 to 75 degrees Fahrenheit, but the optimal range for survival and growth (provided an adequate food supply exists) ranges from 53 to 64 degrees Fahrenheit (Raleigh et al., 1986).

In the natural environment, water temperature affects juvenile chinook salmon growth and survival through complex physiological responses that can be modified by acclimation and behavior. In general, responses to water temperature vary depending on fish size; the duration and frequency of exposure to a given water temperature; physical habitat conditions; food availability; and the presence of competitors, predators, or disease.

Elevated water temperatures affect juvenile survival directly through acute (i.e., lethal) effects and indirectly through chronic (i.e., sublethal) effects. Water temperature becomes lethal at 75 degrees Fahrenheit. Chronic temperature effects occur at lower temperatures and include physiological stress, reduced growth rates, and increased vulnerability to disease and predation. Under laboratory conditions, American River juvenile chinook salmon experienced increasing levels of chronic thermal stress as water temperatures increased from 60 to 75 degrees Fahrenheit (Rich, 1987).

Figure II-25 presents survival and growth rates of juvenile chinook salmon fed maximum rations and exposed to different water temperatures under laboratory conditions. Because maximum





SOURCE:
Jones & Stokes Associates, 1992b.

FIGURE II-25

JUVENILE CHINOOK SALMON GROWTH AND SURVIVAL VERSUS WATER TEMPERATURE

feeding levels are probably seldom realized in the natural environment, the growth curve was modified based on a 60 percent ration level.

Flow Fluctuations. Rapid flow fluctuations can cause stranding of juvenile chinook salmon and subsequent mortality of juveniles unable to return to the river. Causes of mortality include elevated water temperatures, low dissolved oxygen levels, and predation. Fish losses due to stranding have not been well monitored or documented in Central Valley streams. Stranding of juvenile winter-run chinook salmon has occurred in the upper Sacramento River following rapid flow reductions associated with operation of the ACID dam. Since 1970, limitations on flow reduction rates at Keswick Dam have minimized stranding losses (Reclamation, 1983a). Daily flow fluctuations below Keswick Reservoir for both actual and unimpaired conditions are illustrated for the period 1967-1991 in Attachment B.

Flood control structures on the Sacramento River (Moulton, Colusa, Tisdale, and Fremont weirs) divert Sacramento River water from the main river into the Butte Creek Basin and the Sutter and Yolo bypasses during major flood events. As a result, juvenile chinook salmon and other anadromous species migrating down the Sacramento River can be diverted into the bypasses, where they are subject to potential migration delays or entrapment as flood flows recede. Although juvenile fall-, spring-, and winter-run chinook salmon are likely to be found in the bypasses during major winter floods, survival rates associated with these migration routes are unknown. Adult salmon entering the bypasses during their upstream migration could be delayed or blocked by control structures in the bypass channels. However, efforts have been made to alleviate passage problems by installing or upgrading fish ladders at known obstructions to allow the fish to escape before dewatering.

Riparian Habitat. Riparian vegetation performs a variety of critical functions in stream ecosystems by maintaining bank stability, providing overhead and instream cover for aquatic organisms, moderating water temperatures, contributing nutrients and energy, and providing habitat diversity. The presence of riparian vegetation along streambanks greatly enhances the quality of nearshore aquatic habitat for juvenile chinook salmon. Overhanging and submerged branches and root systems provide favorable hydraulic characteristics for resting and feeding; food inputs (primarily terrestrial insects); and shelter from strong light, from swift currents, and from predators. In addition, naturally eroding streambanks are a valuable source of large woody material (e.g., fallen trees) to the stream, which provides important instream cover and contributes to channel and habitat diversity.

Riparian vegetation has been significantly reduced along much of the Sacramento River and its major tributaries as a result of agricultural conversion, urbanization, timber and fuel harvesting, channelization, levee construction, streambank protection, streamflow regulation, bank erosion, and other land use activities. Existing riparian woodlands along the Sacramento River constitute less than 5 percent of historical acreage, and river edge vegetation makes up less than 50 percent of its historical extent (California Resources Agency, 1989). Approximately 5 to 15 percent of the historical acreage remains on tributary streams (Mills and Fisher, 1993).

Riparian loss has been greatest in the middle and lower reaches of the Sacramento River and Delta as a result of levee construction and bank protection projects. The most significant fisheries impacts are caused by bank protection projects, which typically require removal of

nearshore riparian vegetation, grading of the bank slope, and placement of rock revetment over the graded slope. Shaded riverine aquatic (SRA) habitat is of greatest concern because of the unique fishery values associated with this habitat type and substantial losses that have already occurred. Loss of the river's meander belt through replacement of naturally eroding banks with rock revetment has been shown to reduce densities of juvenile chinook salmon locally; chinook salmon densities in undisturbed areas are typically 4 to 12 times higher than in riprapped sites (Michny and Hampton, 1984; Michny and Deibel, 1986).

Levees and other flood control structures have drastically reduced the occurrence and extent of temporarily flooded terrestrial habitat that seasonally provided thousands of acres of potential rearing habitat for juvenile chinook salmon.

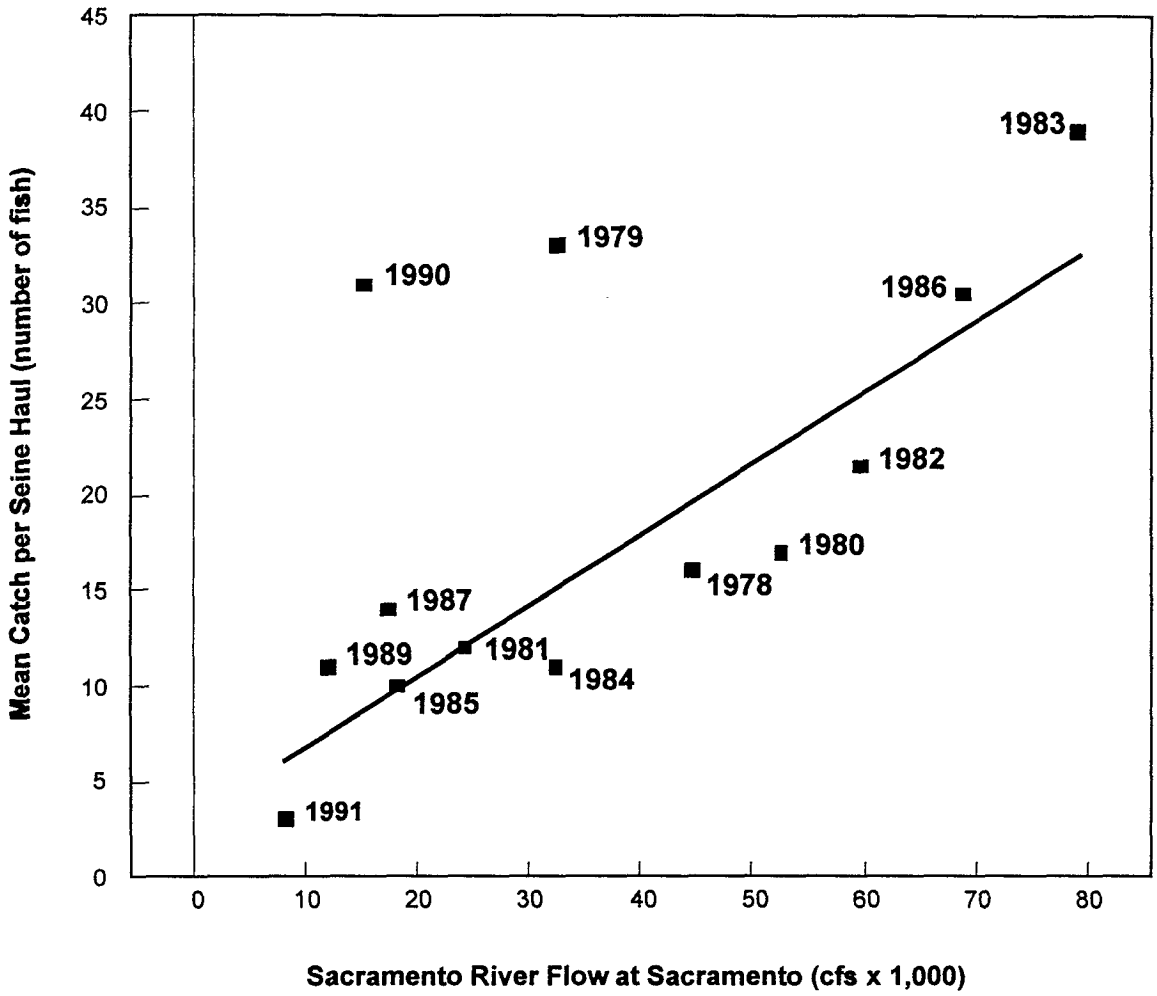
Since 1971, the COE incorporated several features into the Chico Landing to Red Bluff Bank Protection Project to mitigate project impacts on fish and wildlife resources. Primary mitigation measures consisted of using rock fill to save riparian vegetation that will otherwise be removed, replanting affected areas with riparian vegetation, and constructing artificial rearing benches or fish slopes.

Food Production. Little information is available to assess food availability for juvenile chinook salmon in relation to environmental variation. Comparative studies of invertebrate production in revetted versus natural bank areas have not been conducted. Drift densities of invertebrate prey species were not substantially different between revetted and natural banks (Schaffter et al., 1983).

Juvenile Emigration

Flow. Flow influences distribution, abundance, and survival of emigrating juvenile salmonids. Generally, continuous higher flows improve survival and migration success of juvenile salmonids by reducing exposure to diversions (i.e., reducing the proportion of flow diverted), and maintaining favorable water quality conditions. In recent years, increased flow releases from Keswick Reservoir (up to 14,000 cfs) and reduced diversions in May have been designed to assist the downstream migration of hatchery juveniles released in the upper Sacramento River (Reclamation, 1986a). Correlations between Sacramento River flows during the chinook salmon smolt emigration period and the number of adults returning to Sacramento River tributaries (Dettman et al., 1987) indicate that flow, or factors related to flow, significantly affect chinook salmon survival and abundance.

The timing and distribution of chinook salmon emigration in the Sacramento River system are affected by runoff conditions. In general, high flows during the early rearing period result in downstream displacement or active migration of large numbers of fry. Under low-flow conditions, most of the fry remain in upstream rearing areas and emigrate during the normal smolt emigration period. Fall-run chinook salmon fry abundance in the lower Sacramento River and northern Delta during the winter months generally increases as Delta inflow increases (reflected in the relationship between river flow and juvenile salmon captured in seine hauls, (Figure II-26). Peak numbers of fry in the lower Sacramento and Delta are associated with high winter flows or flow pulses in the Sacramento River (Service, 1993).



SOURCE:
Service, 1993.

FIGURE II-26

RELATIONSHIP BETWEEN RELATIVE ABUNDANCE OF CHINOOK SALMON FRY IN THE SACRAMENTO-SAN JOAQUIN DELTA AND SACRAMENTO RIVER FLOW IN FEBRUARY (1978-1991)

Factors influencing smolt emigration timing appear to be closely related to growth rate, fish size, and water temperature, although increased flow and turbidity could act to stimulate downstream migration (Wedemeyer et al., 1980). Downstream movement of juvenile chinook salmon also could be triggered by declining flow and rising water temperatures during the late spring months. Emigration rates generally peak at night or during periods of high turbidity (Vogel et al., 1988).

Mark-recapture studies of fall-run chinook salmon smolts demonstrated that smolt survival through the Delta was positively correlated with Sacramento River flows and negatively correlated with water temperatures and the fraction of Sacramento River flow diverted into the DCC and Georgiana Slough during the April-June emigration period (Service, 1987). Further studies designed to estimate the independent effects of these variables indicated that water temperature and diversions were key causal factors affecting smolt survival (Service, 1988). A regression model was developed to estimate smolt mortality in the Delta as a function of Sacramento River water temperatures at Freeport, the fraction of Sacramento River flow diverted at Walnut Grove, and total SWP and CVP exports in the south Delta (Kjelson et al., 1989). Figure II-27 illustrates model predictions for various combinations of water temperature, export pumping rates, and diversion fractions.

Water Temperatures. Water temperature influences chinook salmon emigration timing. In the Yuba River, an extended period of cold water into summer delays smolt emigration. Later emigrating smolts could experience higher water temperatures and increased mortality upon reaching the lower Sacramento River and Delta (Jones & Stokes Associates, 1992c).

The delta chinook salmon smolt mortality model includes three predictive relationships describing changes in smolt mortality as a linear function of water temperature for three major Delta reaches (Figure II-28). Based on multiple regression analysis, water temperature was found to be the best predictor of smolt mortality among the major environmental variables thought to influence smolt survival in each of the three reaches (Kjelson et al., 1989). Fall-run chinook salmon smolt survival appears to decline at temperatures above 60 degrees Fahrenheit, indicating that sublethal effects could be occurring at relatively low water temperatures in the Delta.

The frequency of suboptimum water temperatures for juvenile chinook salmon in the lower Sacramento River appears to have generally increased since the mid-1970s (Reuter and Mitchell, 1987); however, more than half those years were dry or critical. See Attachment B.

Diversions. Water diversions reduce survival of emigrating juvenile salmonids through direct losses at unscreened or inadequately screened diversions and indirect losses associated with reduced streamflows. Fish screening and salvage efforts at major agricultural diversions have met with variable success, and many smaller unscreened or inadequately screened diversions continue to operate. Fish losses at diversions can result from physical injury, impingement, entrainment, or predation. Delayed passage, increased stress, and increased vulnerability to predation also contribute to mortality at diversions.

Diversion impacts on anadromous fish populations depend on diversion timing and magnitude, river discharge, species (i.e., race), life stage, and other factors. Fall-run chinook salmon juveniles are particularly vulnerable to diversion-related mortality because the fall-run smolt

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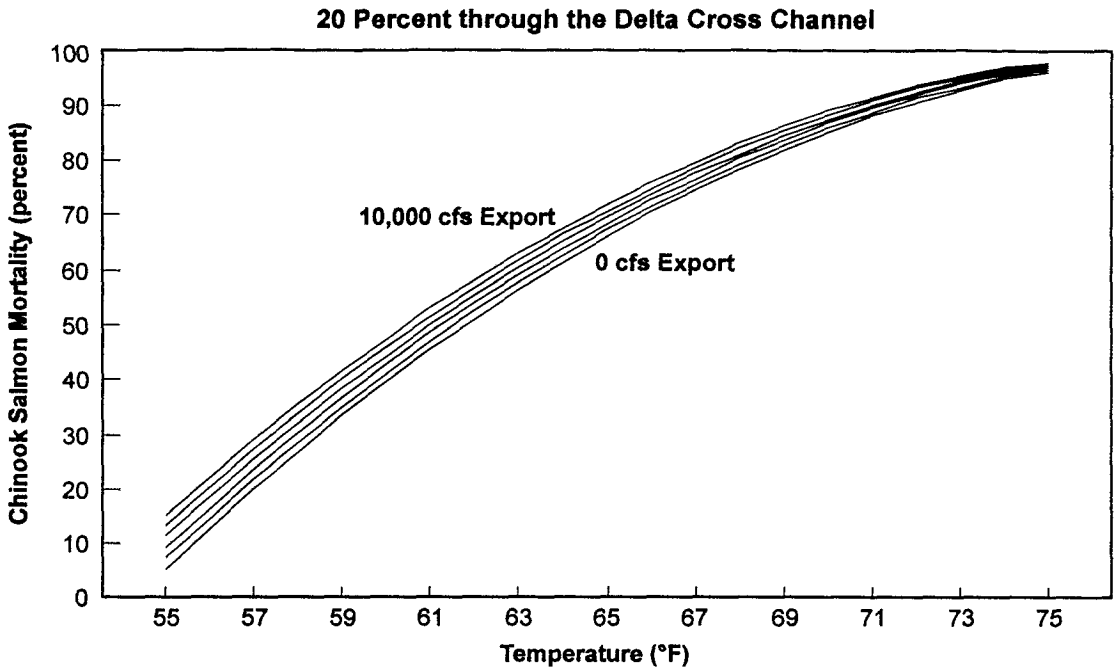
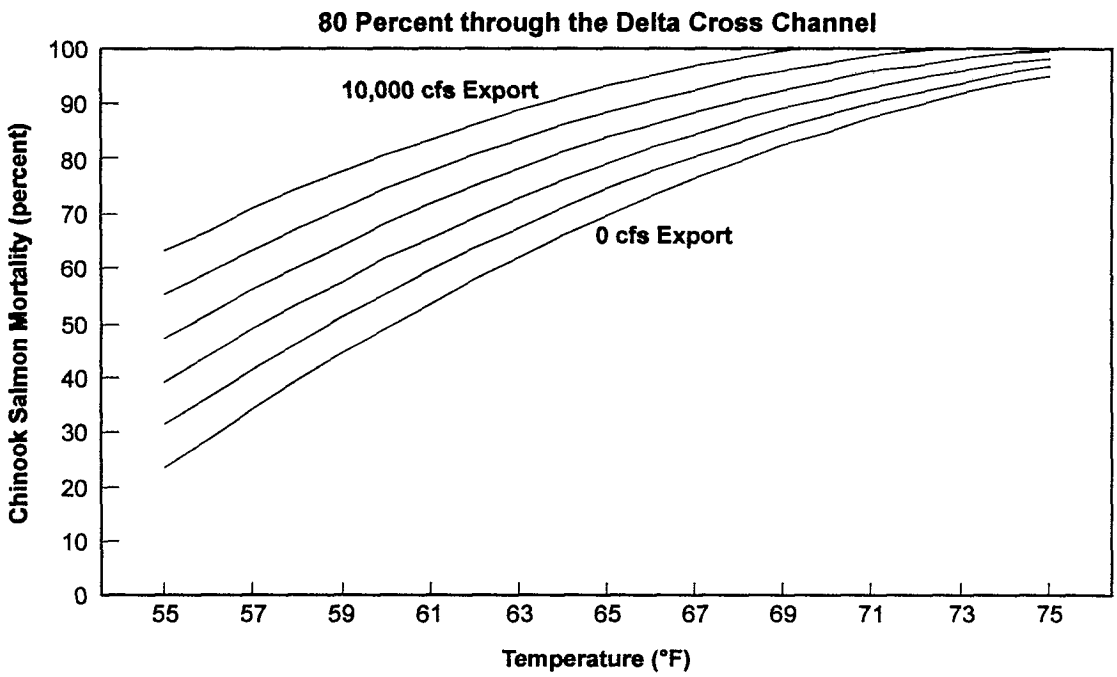
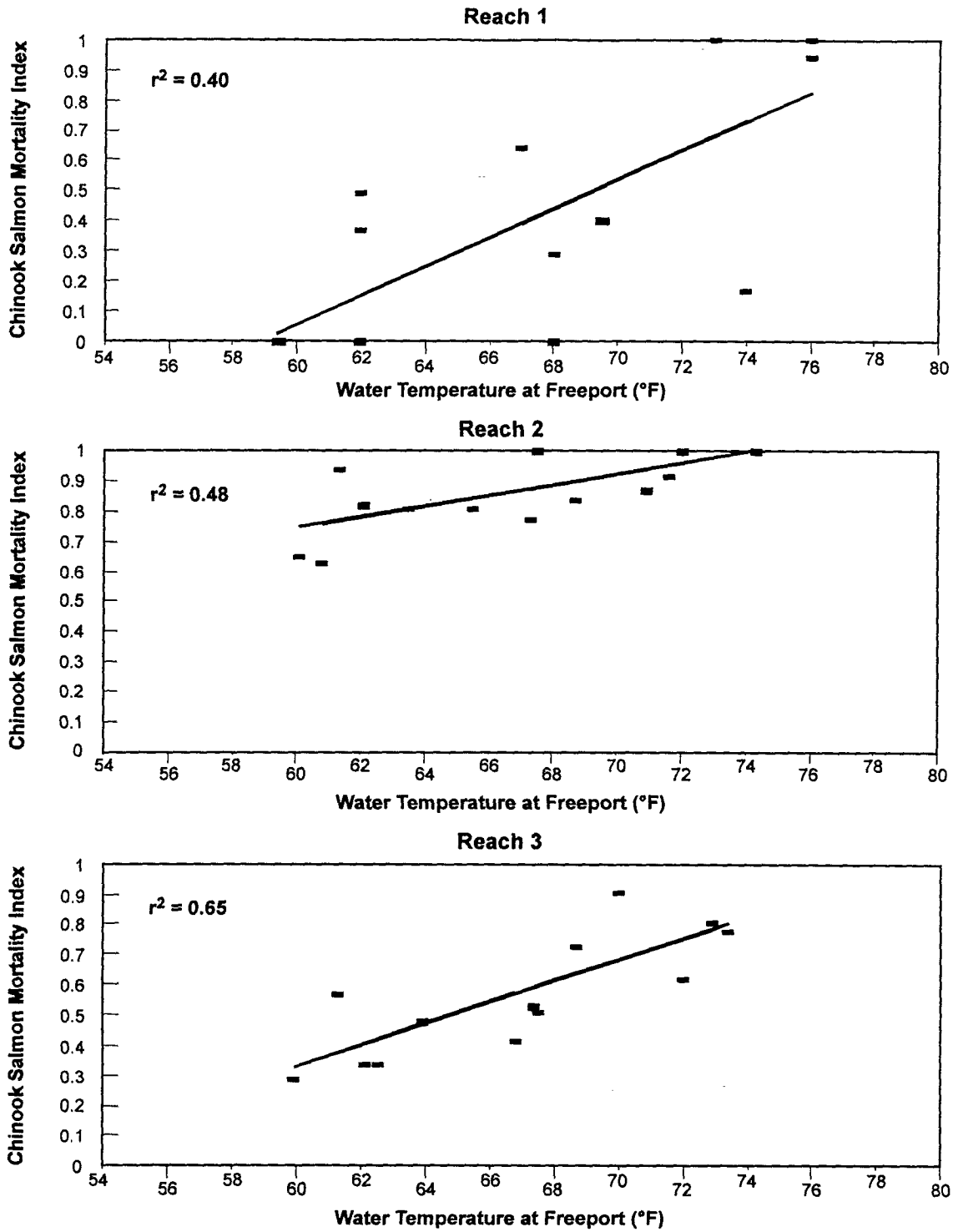




SOURCE:
Kjelson et al., 1989.

FIGURE II-27

RELATIONSHIP BETWEEN PREDICTED SACRAMENTO RIVER CHINOOK SALMON SMOLT MORTALITY THROUGH THE DELTA AND SACRAMENTO RIVER WATER TEMPERATURE AND DELTA EXPORT PUMPING RATES



SOURCE:
Kjelson et al., 1989.

FIGURE II-28

RELATIONSHIP BETWEEN ESTIMATED CHINOOK SALMON SMOLT MORTALITY AND AVERAGE DAILY WATER TEMPERATURE AT FREEPORT ON RELEASE DAY, REACHES 1-3

emigration period (April-June) generally coincides with the onset of the irrigation season (April-October). Chinook salmon losses are decreased during the summer irrigation season because juvenile salmon do not actively migrate during summer. Winter-run chinook salmon are subject to diversion losses during the beginning of the irrigation season (April and May) and the latter part of the irrigation season (September-October), after which diversions are negligible. Figure II-29 compares seasonal timing and magnitude of diversions with seasonal smolt migration patterns of chinook salmon in the Sacramento River and the Delta.

Annual variation in runoff conditions also affects the magnitude of diversion losses. High river flows during the winter or early spring could displace large numbers of fall-run juveniles downstream of most of the unscreened diversions on the Sacramento River before diversion activity begins. Continued high spring flows delay the onset of diversions and maintain favorable survival conditions, including a high ratio of river discharge to volume diverted. Fish losses are generally increased under low-flow conditions because of little downstream displacement, earlier diversion activity, and less favorable survival conditions.

The location and capacities of Sacramento River diversions are shown in Attachment B. Sacramento River diversions total 3.2 million acre-feet per year. Ten diversions account for most of the water diverted from the Sacramento River, and only three of these (at the RBDD, and ACID and GCID diversion dams) have fish screens or bypass systems. More than 300 unscreened diversions account for 1.2 million acre-feet of water diverted annually in the Sacramento River. Annual losses of juvenile salmon in these diversions could reach 10 million fish (California Resources Agency, 1989).

Reclamation initiated a Pilot Fish Screen Demonstration Program in 1993 to help diverters screen existing unscreened diversions along the Sacramento River. The main objective of the program is to participate with diverters in demonstrating approved fish screen technologies and experimenting with other technologies to evaluate their effectiveness in guiding fish safely past water diversions.

Anderson-Cottonwood Irrigation District Canal. The ACID diversion canal is screened but requires frequent maintenance and inspection. In general, potential impacts on downstream migration are considered minor because of the small proportion of juvenile salmon produced in the Sacramento River above the ACID diversion canal (Reclamation, 1986a). Impacts on juvenile winter-run chinook salmon could be significant because of the limited abundance of the population (NMFS, 1992).

Tehama-Colusa Canal. Vogel et al. (1988) concluded that losses of chinook salmon migrating downstream past the TCC and the RBDD during the winter and spring chinook salmon emigration period result from entrainment through the TCC headworks, physical injury as juveniles pass through the TCC headworks fish bypass system, and predation as juvenile salmon pass under the RBDD gates or through the fish bypass system (Vogel et al., 1988). Maximum estimated losses attributable to entrainment and physical injury were 0.6 percent and 4.1 percent, respectively. Predation presumably accounted for the remainder of estimated losses, ranging from 16 percent to 55 percent.

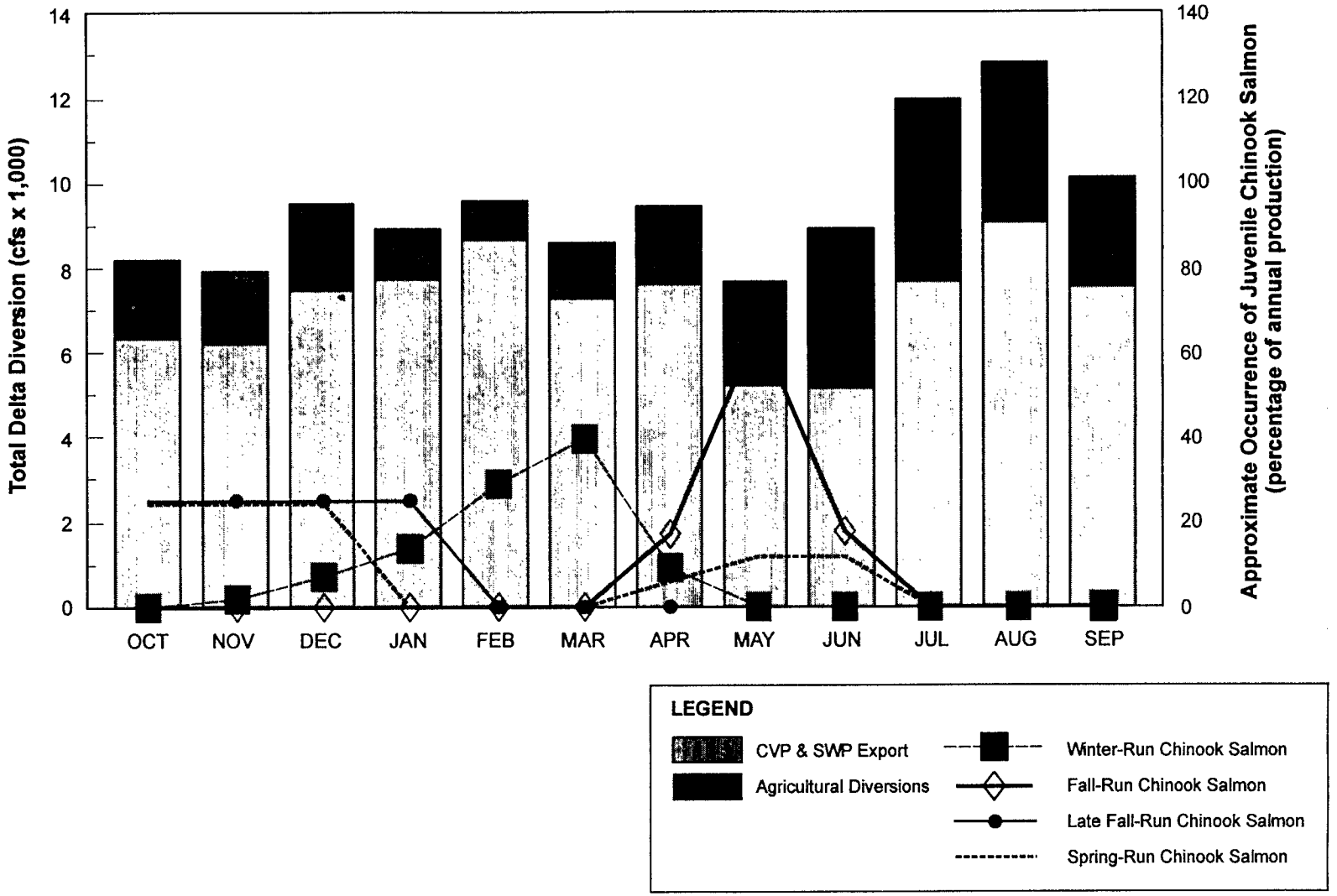


FIGURE II-29

COMPARISON OF MONTHLY JUVENILE CHINOOK SALMON MIGRATION WITH MONTHLY DIVERSION PATTERNS (1981-1990) FOR THE LOWER SACRAMENTO RIVER AND THE DELTA

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The RBDD gates are currently raised during the nonirrigation season to facilitate upstream passage of adult winter-run chinook salmon. Downstream migrating juvenile salmon (primarily late fall- and winter-run) also benefit from this measure because of unimpeded flow conditions past the dam, although predation rates during this period are thought to be low. The TCC headworks louver fish screens and bypass system were replaced with state-of-the-art rotary drum screens and an improved fish bypass system in 1990.

Glenn-Colusa Irrigation District Diversion. Past evaluations of screen efficiency and fish mortality at the GCID diversion near Hamilton City have identified major problems in design and operation of the facility that have caused significant losses of downstream migrating salmonids. These problems included an inadequate bypass system, excessive approach velocities, and inadequate bypass flows. After construction of the present fish screens in 1972, natural degradation of the Sacramento River channel lowered the water elevation at the fish screen by 4 feet, causing excessive water velocities (up to 0.78 foot per second) at the screen face (relative to DFG's current criterion of 0.33 foot per second) at pumped flows over 1,500 cfs. (GCID et al., 1989.)

Recent mark-recapture studies using fall-run chinook salmon juveniles showed that the survival rates (i.e., fish bypass efficiencies) were negatively correlated to pumping flows (Figure II-30), indicating that fish losses were caused by impingement, entrainment, or predation at the screen. The data also indicated that chinook salmon fry (less than 50 millimeters long) were more vulnerable to loss than larger juveniles or smolts; in general, fish bypass efficiency increased as fish size increased (Cramer et al., 1990).

An injunction obtained by the NMFS against GCID for the illegal take of winter-run chinook salmon requires GCID to operate the diversion within specific criteria designed to avoid or minimize losses of winter-run chinook salmon. An environmental impact report/environmental impact statement is being prepared to identify a permanent solution to diversion impacts on all anadromous fish species. Potential solutions being evaluated include improving the existing screens and bypass system, constructing new screens, relocating the intake, restoring the gradient of the Sacramento River at the head of the GCID diversion channel, or some combination thereof (58 Federal Register 194, October 8, 1993).

Delta Diversions. Annual losses of chinook salmon at the SWP and CVP Delta export facilities have typically ranged from 400,000 to 800,000 salmon in recent years (Figure II-31), assuming 75 percent mortality in Clifton Court Forebay. Salvage records from the SWP pumping plant indicate salmon fry and smolts are entrained year-round, but levels generally peak in late winter and spring when fall-run chinook salmon pass through the Delta (Figure II-16). Although fall-run chinook salmon produced in the Sacramento River constitute approximately 80 percent of the total number of chinook salmon passing through the estuary, only a small percentage of chinook salmon juveniles released in the Sacramento River typically reach the CVP and SWP export pumps (Service, 1987).

Juvenile chinook salmon salvaged at the SWP export facility between December 1992 and April 1993 were classified by race based on size criteria developed by DFG.

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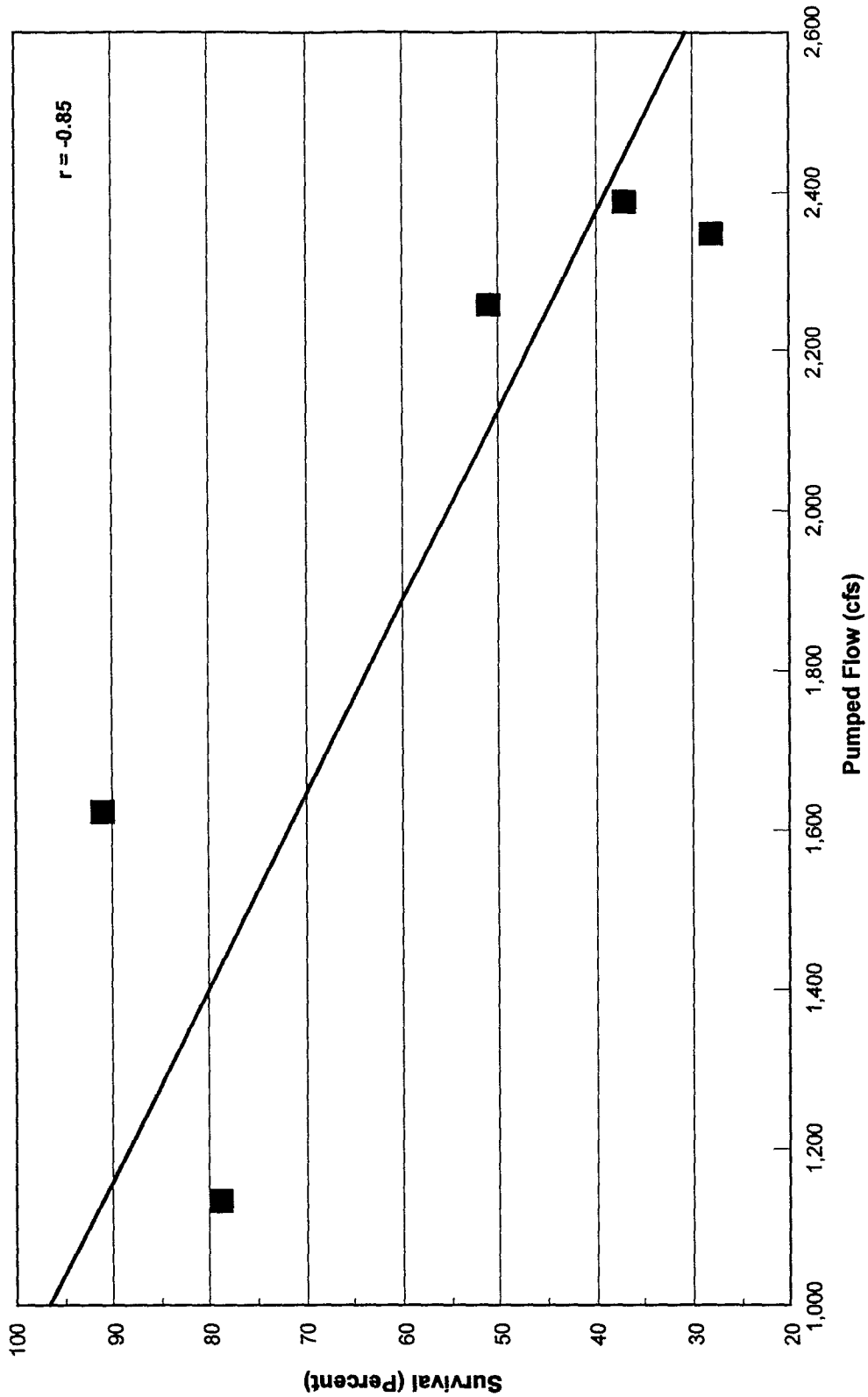
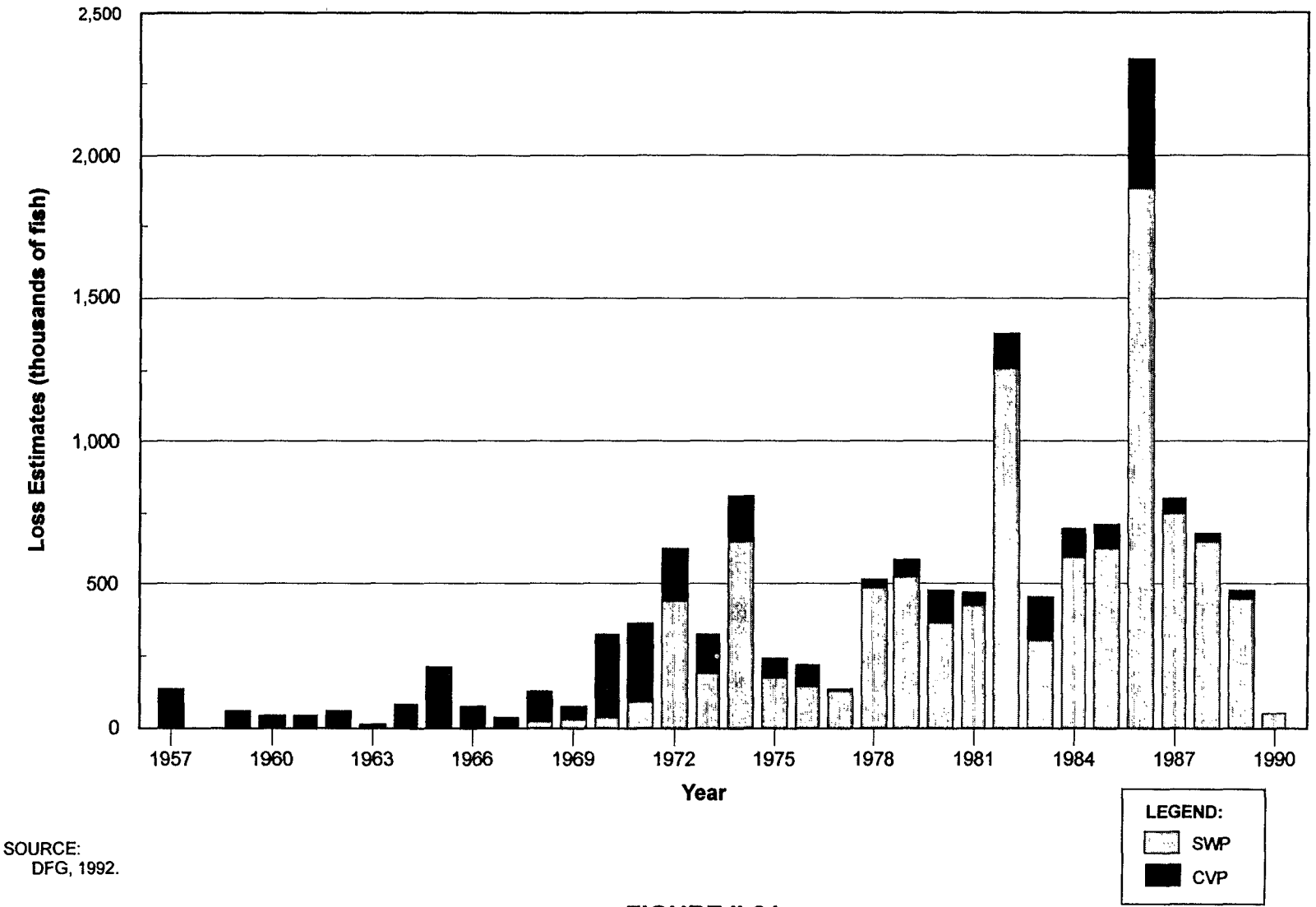


SOURCE:
Cramer et al., 1990.

FIGURE II-30

RELATIONSHIP BETWEEN ESTIMATED CHINOOK SALMON SURVIVAL RATE THROUGH GLENN-COLUSA IRRIGATION DISTRICT DIVERSION CHANNEL AND PUMPED FLOW



SOURCE:
DFG, 1992.

FIGURE II-31

CHINOOK SALMON LOSS ESTIMATES FOR THE SWP AND CVP FACILITIES (1957-1990)

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Unknown numbers of salmon are also entrained in other Delta diversions, including more than 1,800 unscreened agricultural diversions; the Contra Costa Canal; the City of Vallejo diversion; and western Delta industry diversions (DWR, 1993). The location and capacities of Delta diversions are shown in Attachment B.

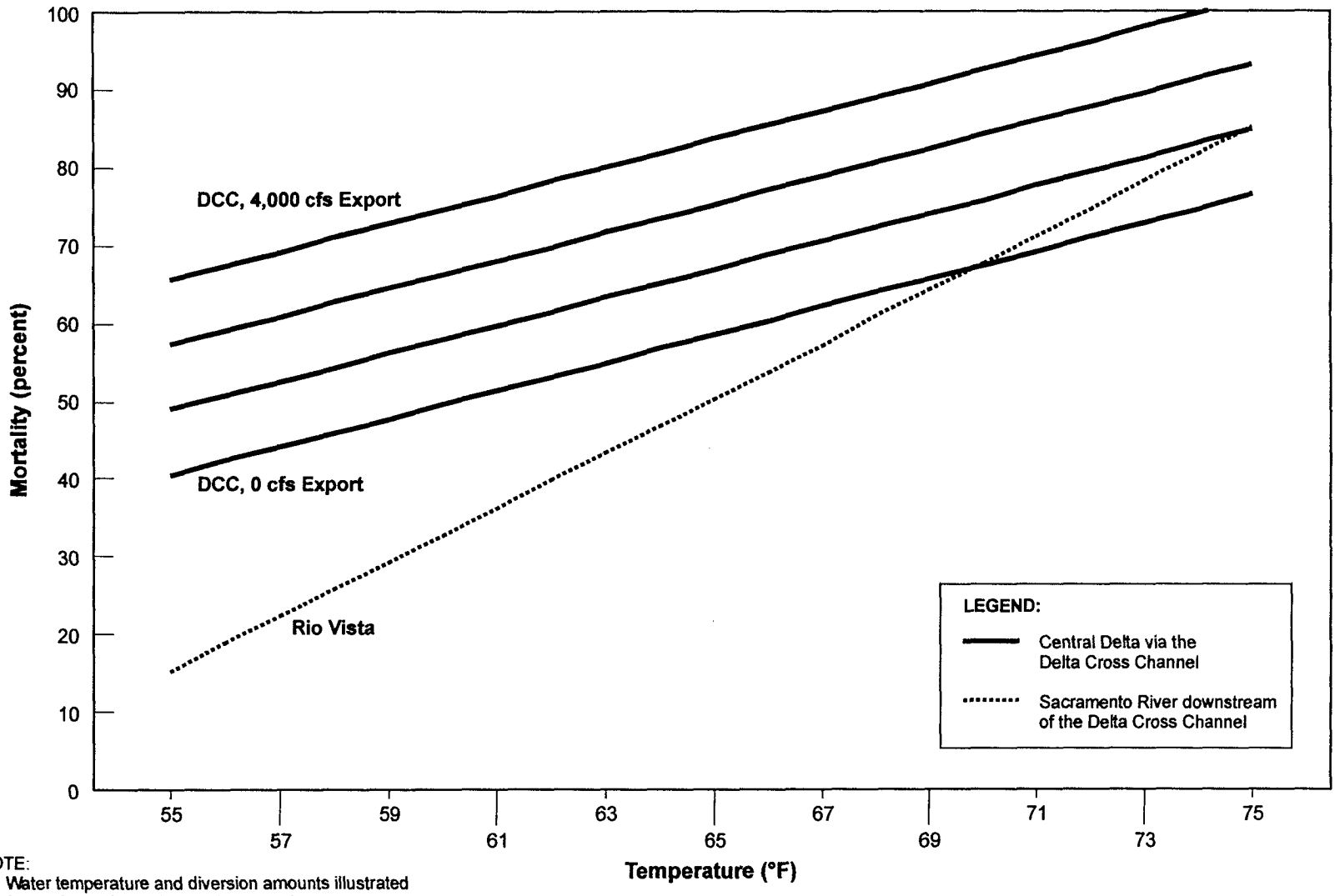
Delta Flows and Exports. The SWP (Banks) and CVP (Tracy) export facilities in the south Delta adversely affect anadromous fish survival in the Delta through direct entrainment losses and indirect effects related to changes in the magnitude and direction of flow in the Delta channels. Increases in upstream storage and diversions over the last 20 years have significantly reduced inflow to the Delta. Reduced inflow in combination with increased diversions from the Delta (illustrated in Attachment B) has caused increasing, adverse impacts on anadromous and resident species by reducing net flow through the Delta and Delta outflow. Unscreened Delta diversions have contributed to fish losses.

Fall-run salmon smolts diverted from the Sacramento River into the central Delta via the DCC or Georgiana Slough have higher mortality rates than smolts that remain in the Sacramento River (Figure II-32); at a given water temperature, the survival of hatchery fall-run chinook salmon smolts that enter the DCC averages approximately 50 percent less than for smolts released in the Sacramento River below the DCC diversion when Delta exports total approximately 3,000 cfs. Poor survival of smolts diverted into the central Delta is attributed to increased migration time, high water temperatures, predation, entrainment in unscreened agricultural diversions, and exposure to reverse flows in the central and south Delta channels. The proportion of diverted Sacramento River flow and total Delta exports are important regression variables in the Service's Delta chinook salmon smolt mortality model (Kjelson et al., 1989). Recent mark-recapture experiments provide some evidence that a positive net flow at Jersey Point increases the survival of salmon migrating down both the Sacramento and San Joaquin rivers, including those migrants that are diverted from the Sacramento River into the central Delta and move to the San Joaquin via the Mokelumne River (Service, 1993) (Figure II-33). In Figure II-33, temperature-corrected survival is defined as the effects of QWEST on survival with the effects of temperature removed. Other factors also could influence survival, such as inflow, tides, and exports.

Delta flow and operational criteria established by the NMFS for protection of winter-run chinook salmon for the period February 15, 1993, through February 15, 1994, included closing the DCC gates during the main emigration period through the Delta, and operating the CVP and SWP Delta export facilities to maintain specific minimum running average QWEST (i.e., computed net flow at Jersey Point) values during the Delta rearing and emigration periods (NMFS, 1993).

Predation. Predation on emigrating salmonids is probably of minor significance in unobstructed portions of the Sacramento River system, but predator efficiency increases at human-made structures and impoundments where fish are concentrated, stressed, or delayed in their downstream migration (Reclamation, 1983b).

Vogel et al. (1988) concluded that predation is the primary cause of downstream migrant salmon mortality at the RBDD, accounting for losses ranging from 16 percent to 55 percent. Disorientation of downstream migrants as they pass under the dam gates or through the TCC headworks fish bypass system increases their vulnerability to predators. Predation by squawfish



NOTE:
Water temperature and diversion amounts illustrated for the Delta Cross Channel route only.

SOURCE:
Kjelson et al., 1989.

FIGURE II-32

RELATIONSHIP BETWEEN PREDICTED SACRAMENTO RIVER CHINOOK SALMON SMOLT MORTALITY FOR TWO DELTA MIGRATION PATHWAYS AND SACRAMENTO RIVER WATER TEMPERATURE

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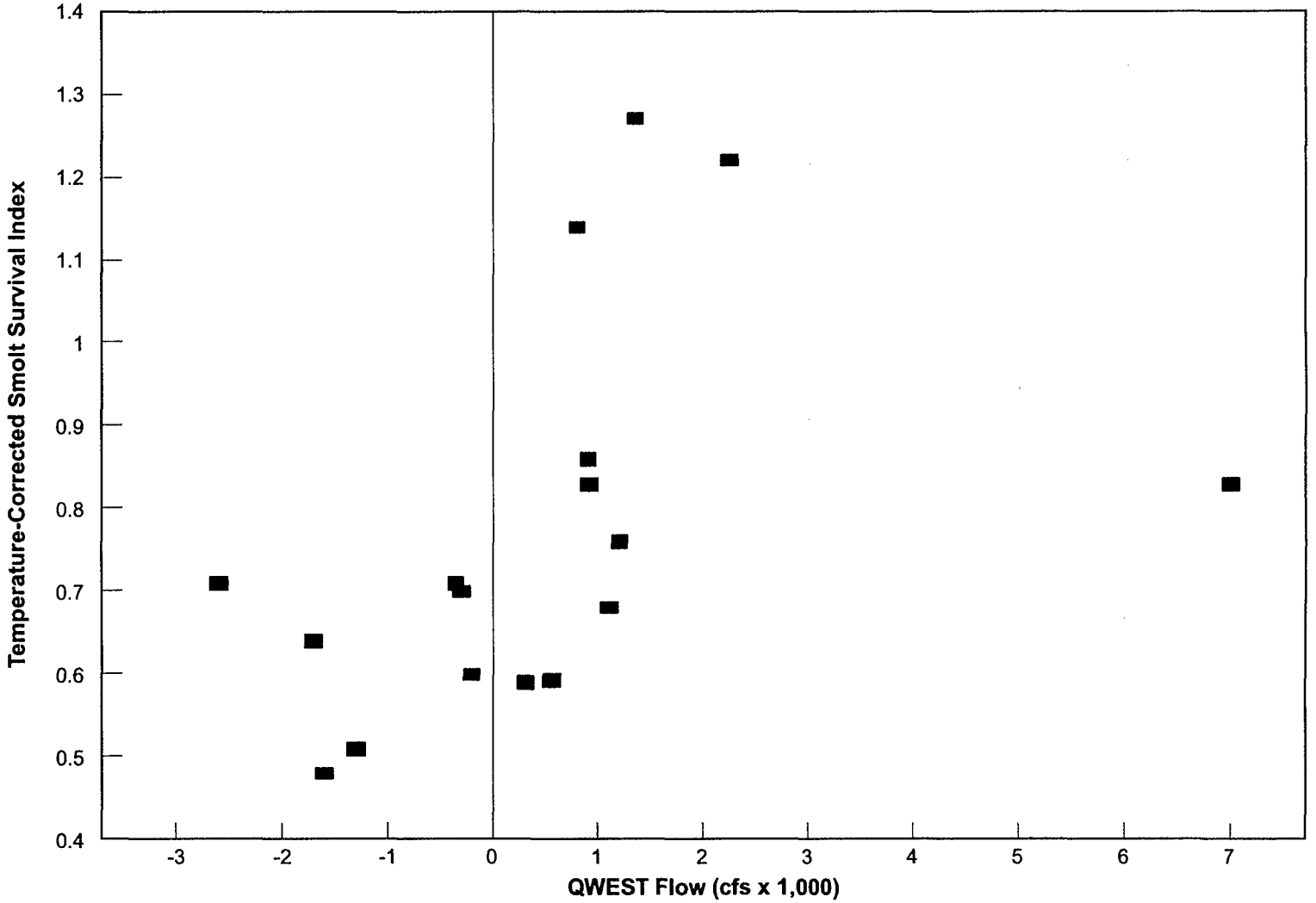


SOURCE:
Service, 1993.

FIGURE II-33

RELATIONSHIP BETWEEN TEMPERATURE-CORRECTED SURVIVAL FOR FISH RELEASED AT RYDE AND QWEST FLOW (1984-1992)

C-081409
C—081409

is particularly evident in spring when adult squawfish congregate at the RBDD during the fall-run chinook salmon emigration period.

Predation by striped bass is considered the primary cause of high prescreening mortality of juvenile chinook salmon at the SWP export facility in the south Delta. Although data are limited, estimated losses of juvenile chinook salmon entrained into Clifton Court Forebay range from 63 to 86 percent (DFG, 1987a). Predation losses at the CVP export facility are assumed to be lower because of the absence of extensive predator habitat. The significance of predation at other diversion facilities and in the Delta has not been adequately evaluated.

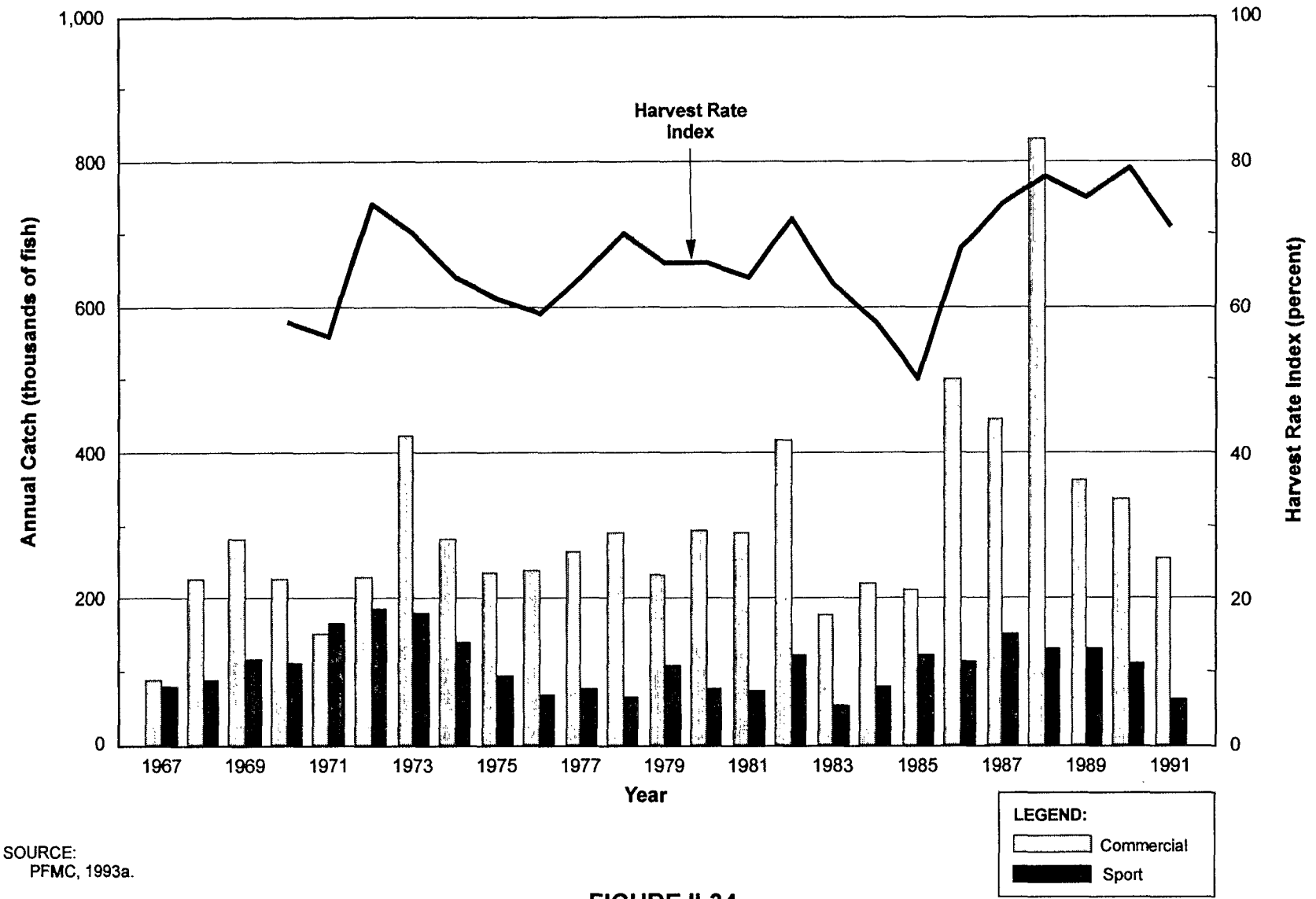
Ocean Residence

Commercial and Sport Fishing. Total commercial and sport annual landings of chinook salmon from 1967 to 1991 ranged from 358,000 pounds in 1983 to 1,489,000 pounds in 1988 and averaged 707,000 pounds (PFMC, 1993a). Since 1988, total landings have steadily decreased to levels near the historical minimum for the period of record (Figure II-34). Catch-per-unit effort, roughly approximated by the number of chinook salmon landed per number of days fishing, was computed for commercial landings (1978-1990) and for sport landings (1962-1990) (Figure II-35). Catch-per-unit effort for the chinook salmon sport fishery remained relatively constant during this period, while the commercial fishery trended upward over the last 13 years. From 1986 to 1989, catch-per-unit effort for commercial landings more than doubled, reflecting a large increase in ocean salmon abundance during these years.

Hatchery Fish. Accurate estimates of the Central Valley hatchery contribution to ocean chinook salmon landing have not been developed because of the lack of a consistent hatchery marking program in California. The Service (1988) estimated that 21 percent of the smolts passing Chipps Island in 1988 were of hatchery origin. Cramer et al. (1990) estimated that hatchery fish composed approximately one-third of the spawning escapement to the American and Feather rivers. This fraction is significantly lower than previous estimates developed by Dettman and Kelley (1987).

Because of increased survival from eggs to smolts under hatchery conditions, fewer adults are needed to maintain a hatchery run. Consequently, a harvest rate based on hatchery fish will tend to eliminate wild fish in a mixed fishery of wild and hatchery stocks (Hilborn, 1992). Current harvest rates of Central Valley chinook salmon are high enough to adversely affect the natural production in some rivers.

Intensive harvest of natural chinook salmon stocks for many years has resulted in a shift in age composition toward smaller, earlier maturing individuals. Historically, adult spawning populations in California appear to have been dominated by four- and five-year-olds with smaller proportions of two-, three-, and six-year-olds. Today, spawning runs typically consist largely of two- and three-year-olds with smaller numbers of four-year-olds and very few five-year-olds (Dettman et al., 1987). Changes in age composition have been accompanied by a decrease in average size of fish landed in the California troll fishery since 1950 (Reisenbichler, 1986). Major reasons for declining age and size of chinook salmon stocks include the selective harvest of larger or faster growing individuals; higher fishing mortality of later maturing fish that are



SOURCE:
PFMC, 1993a.

FIGURE II-34

ANNUAL HARVEST RATE INDEX AND LANDINGS FOR CALIFORNIA COMMERCIAL AND SPORT OCEAN CHINOOK SALMON FISHERIES (1967-1991)

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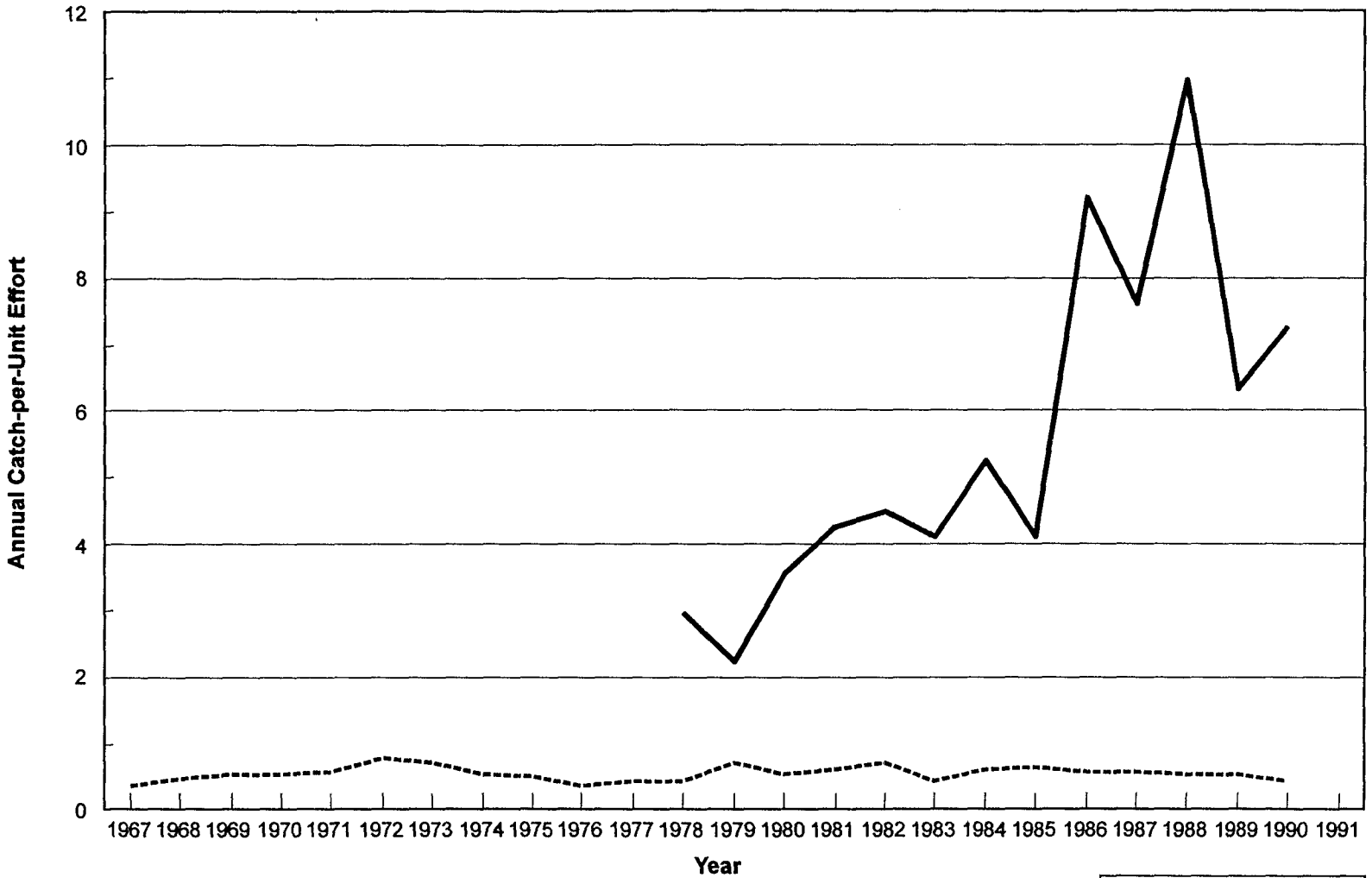
C-081411



NOTE:
No effort data are available for commercial fishery from 1967-1977.

SOURCE:
PFMC, 1993a.

LEGEND:
Commercial
Sport

FIGURE II-35

ANNUAL CATCH-PER-UNIT EFFORT FOR CALIFORNIA COMMERCIAL AND SPORT OCEAN SALMON FISHERIES (1967-1991)

C-081412
C-081412

exposed to ocean harvest for more years than earlier maturing fish; and the resulting long-term genetic selection for smaller, younger fish (Ricker, 1980).

Ocean Habitat Conditions. Overall salmon production depends on both freshwater conditions (factors affecting adult migration, spawning, incubation, rearing, and emigration) and ocean conditions (factors affecting ocean salmon growth, survival, and migration back to freshwater). Much more is known about the freshwater life history, biology, and environmental requirements of salmon. The ocean ecology of salmon has been generally neglected, and studies of the factors affecting chinook salmon populations in the ocean have only recently been initiated (Pearcy, 1992).

Ocean survival of salmon depends on a complex interaction of oceanographic, meteorologic, and biologic factors. Increased marine survival of salmon is commonly associated with upwelling events that bring cold nutrient-rich water from deep ocean layers to the surface along the eastern Pacific coast during spring and summer (Lichatowich, 1993). El Niño events, which transport warm, low-salinity water from subtropical regions, can suppress or reduce the intensity of upwelling, leading to poor marine survival and reduced abundance of adult salmon. In addition, periodic, southward transport of subarctic waters enhances productivity off California. Increased marine exploitation of important forage species (e.g., California sardine, hake, and anchovy) has also likely affected ocean salmon production. Overall, forage fish biomass in the California current declined from approximately 25 million tons in 1905 to 4.5 million tons by 1950 and has remained well below historical levels. Before the collapse of the California sardine market, the sardine could have been an exceptionally rich energy source for salmon and a buffer against predation during the species' first summer at sea (Lichatowich, 1993).

According to Lichatowich (1993), if the California current has undergone a "change in state" that influences salmon production, then the state of the freshwater links in the chain could become more important. Healthy freshwater habitats could become more critical when oceanic productivities are lower and marine mortality higher. The degradation of freshwater habitat combined with cyclic changes in ocean productivity and high harvest rates could have had the effect of "burning the candle at both ends". Cycles of ocean productivity can mask, at the very least, the effects of improvements in freshwater habitat or hatchery production or cause biologists to falsely attribute increased marine survival to restoration effects in freshwater. However, important additive or multiplicative consequences of freshwater habitat degradation could occur in the troughs of ocean productivity cycles.

Hatchery Production

Five hatcheries produce chinook salmon in the Central Valley (Table II-5). The three largest fish hatcheries (Coleman National, Feather River, and Nimbus) are located in the Sacramento River Basin; the smaller ones are on the Mokelumne and Merced rivers in the San Joaquin River Basin. DFG operates six other salmon hatcheries in northern California outside the Central Valley, including Trinity River Fish Hatchery. Most of these hatcheries were constructed between 1940 and 1970 as mitigation for specific dams or water projects. Only Nimbus and Coleman National fish hatcheries had significant production prior to 1967. The salmon hatcheries are funded by hatchery-specific mitigation agreements with state, federal, and public agencies and monies collected from commercial salmon anglers.

TABLE II-5
CENTRAL VALLEY CHINOOK SALMON HATCHERIES

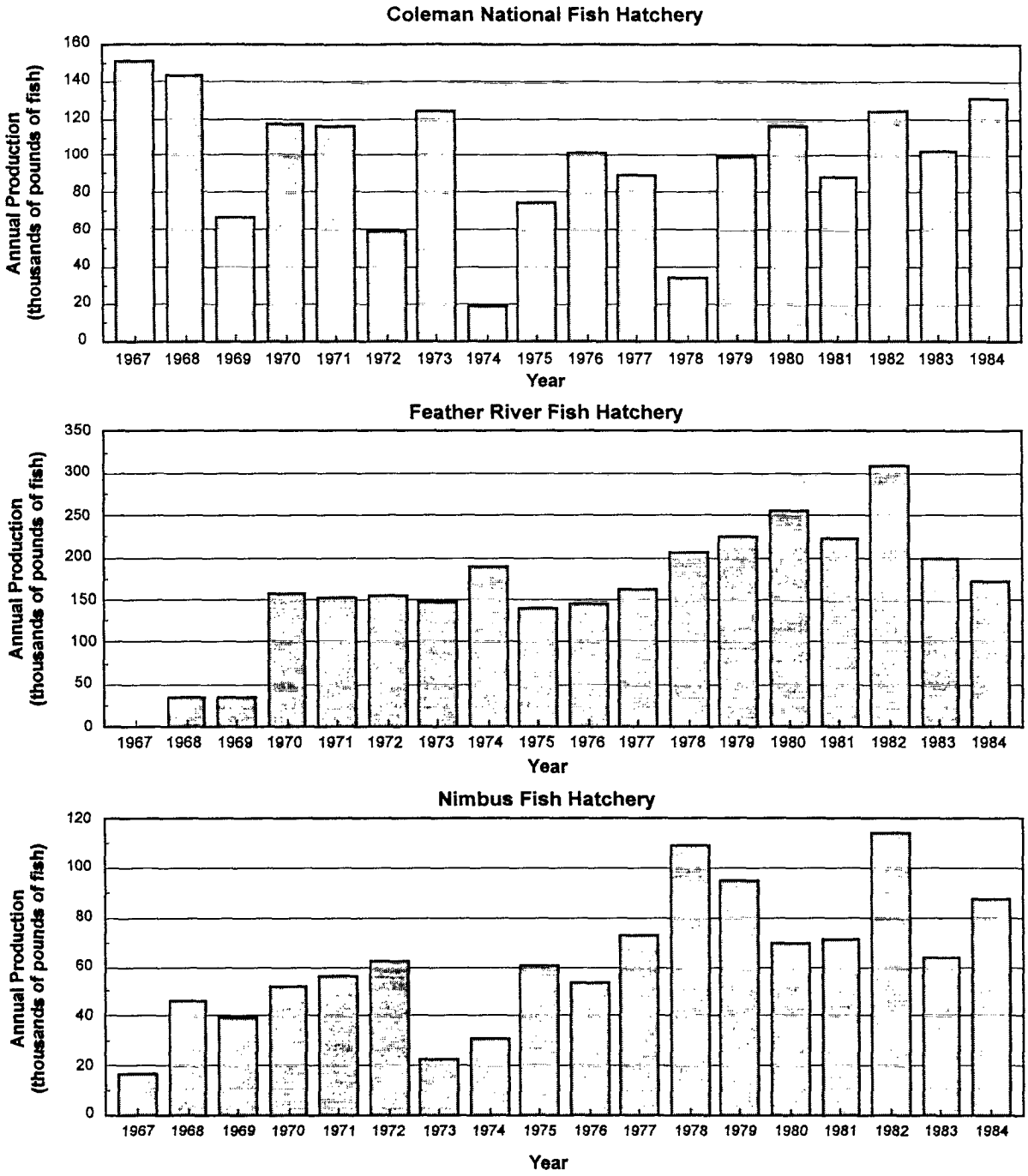
Hatchery	Location	First Year of Operation	Operator	Primary Funding Source	1984-85 Salmon Production (lb/year)
Coleman National Fish Hatchery	Battle Creek near Cottonwood	1942	Service	Service	130,958
Feather River Fish Hatchery	Feather River at Oroville	1967	DFG	DWR	203,388
Merced River Fish Hatchery	Merced River near Snelling	1970	DFG	DFG	49,188
Mokelumne River Fish Hatchery	Mokelumne River below Camanche Dam	1964	DFG	EBMUD	26,650
Nimbus Fish Hatchery	American River below Nimbus Dam	1955	DFG	Reclamation	146,176

DFG hatchery production data were obtained from annual DFG reports (1960-n.d.). The release numbers reported by Cramer et al. (1990) for Coleman National Fish Hatchery were converted to weights using the average weight of each release type (e.g., fingerling). The study used annual production of chinook salmon from 1967 to 1984. Annual production from the Feather, Nimbus, Mokelumne, and Merced river hatcheries exhibited a general increase, while Coleman showed no clear trend (Figures II-36 and II-37). Total Central Valley salmon production nearly doubled during this period (Figure II-38).

Changes in Hatchery Practices. Traditionally, Central Valley hatcheries have released fish directly into the river. To reduce downstream mortality, some of the hatcheries have trucked fish to locations nearer the ocean. At Nimbus Fish Hatchery, fish were predominantly released during the early 1970s, at Rio Vista during the late 1970s, and in the estuary during the 1980s. At Feather River Fish Hatchery, most chinook salmon were released in the estuary after 1983. Survival was observed to be significantly greater for fish released farther downstream. For 1980 paired releases from Feather River Fish Hatchery, survival rates for fish released at Port Chicago were four times higher than those released at the hatchery and two times higher than those released at Discovery Park (Cramer et al., 1990). The increase in survival depended on the time of release, river temperature, oceanic conditions, and size of fish. Fish released off-station have a higher tendency to stray on return than fish released on-station, where station refers to a hatchery facility.

Increased survival of hatchery fish has also been achieved by releasing juveniles at larger sizes. Survival significantly increases for fingerlings (1 to 5 grams) and smolts (5 to 10 grams) as compared with fry (less than 1 gram) (Cramer et al., 1990).

Since 1967, increased production and survival of hatchery chinook salmon has increased contributions of hatchery fish to adult spawning escapements. Annual contributions of hatchery fish to runs in the American and Feather rivers in recent years range from 33 percent to 80 percent (Dettman and Kelley, 1987; Cramer et al., 1990).



SOURCES:
Cramer et al., 1990; DFG, 1968-n.d.

FIGURE II-36

ANNUAL PRODUCTION OF CHINOOK SALMON AT COLEMAN NATIONAL FISH HATCHERY, FEATHER RIVER FISH HATCHERY, AND NIMBUS FISH HATCHERY (1967-1984)

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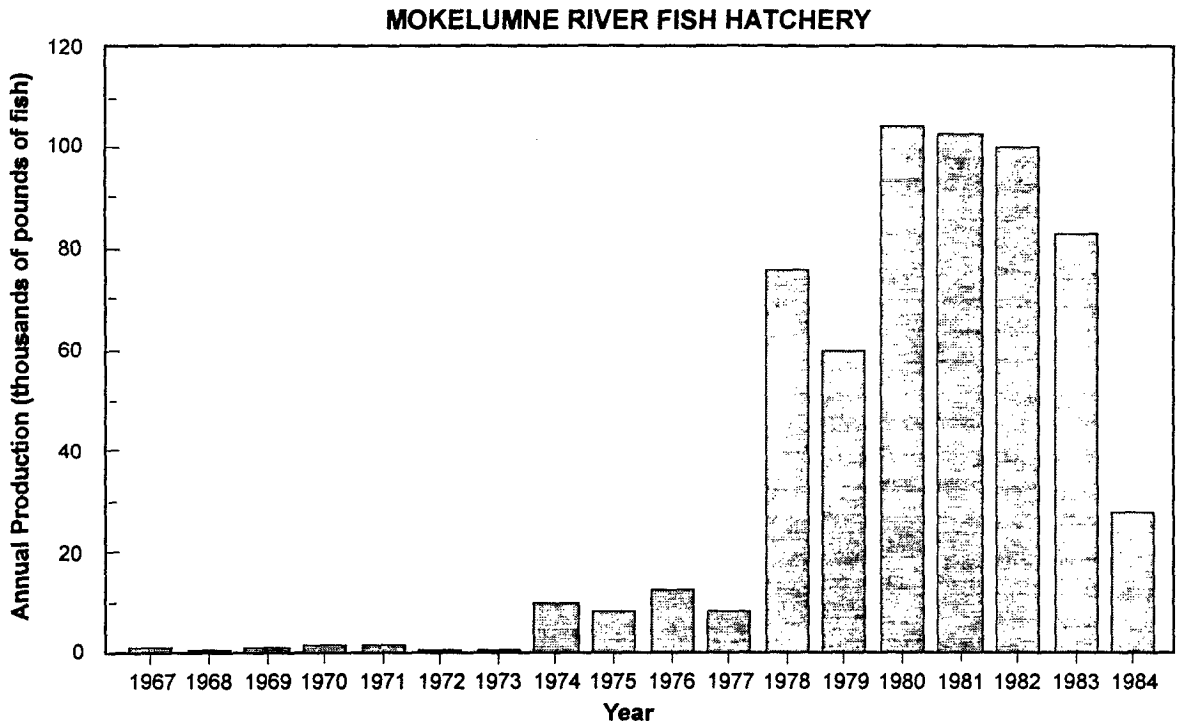
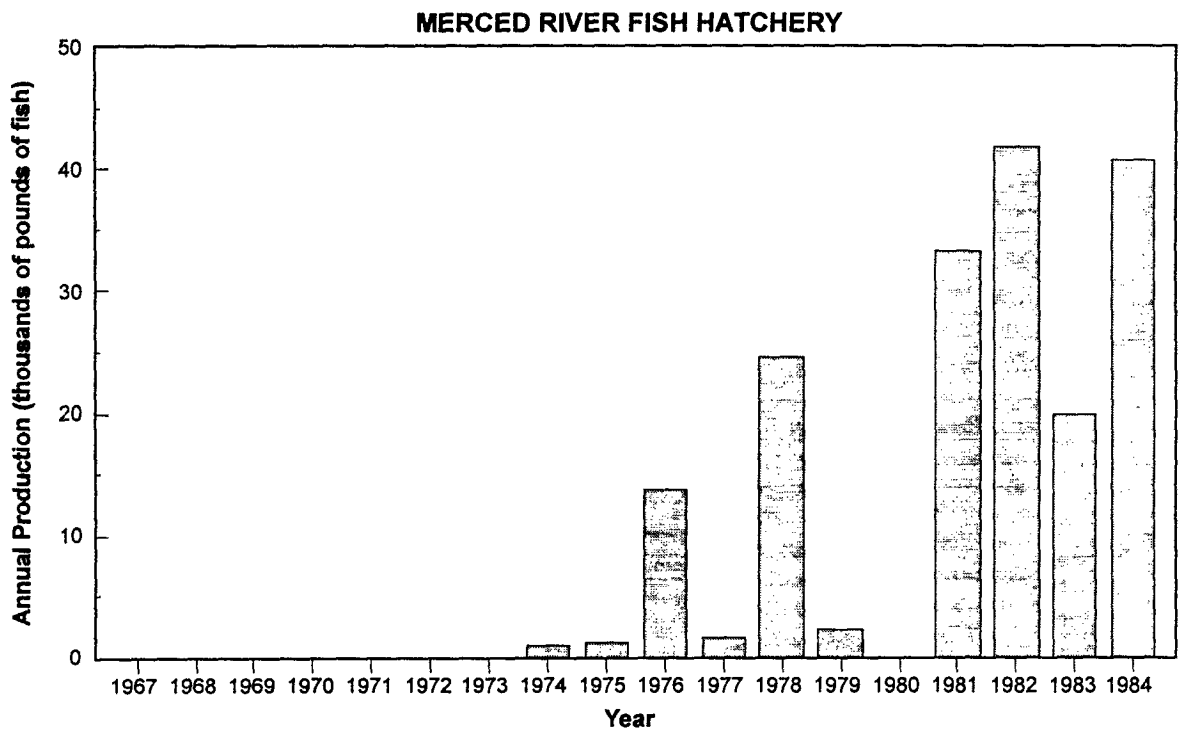
C-081415





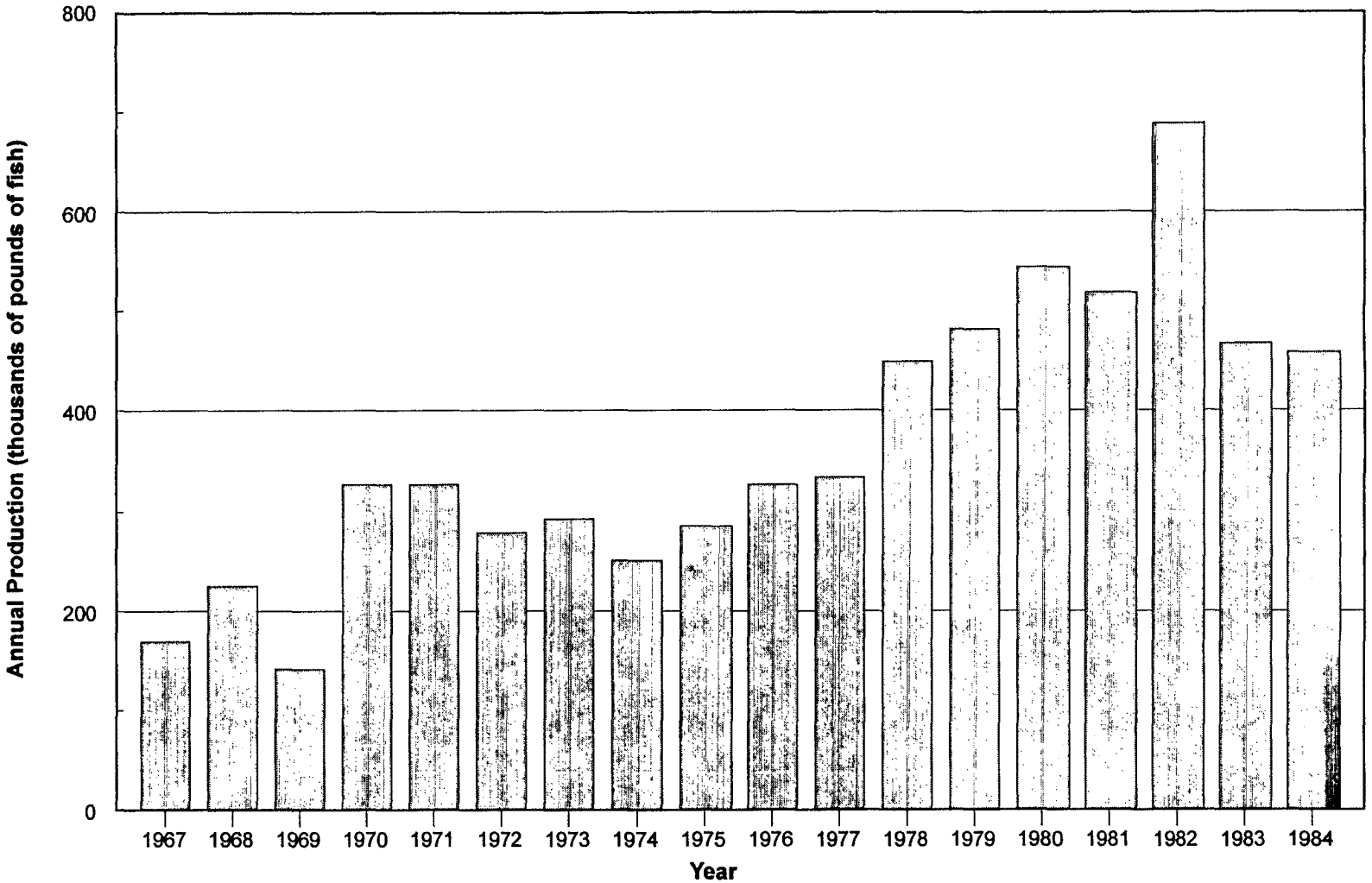
SOURCE:
DFG, 1969-n.d.

FIGURE II-37

ANNUAL PRODUCTION OF CHINOOK SALMON AT MOKELUMNE AND MERCED RIVER HATCHERIES (1967-1984)

C—081416

C-081416



SOURCE:
Cramer et al., 1990; DFG, 1969-n.d.

FIGURE II-38

TOTAL ANNUAL HATCHERY PRODUCTION OF CHINOOK SALMON IN THE CENTRAL VALLEY (1967-1984)

C-081417
C-081417

Impact of Hatchery Fish on Wild Populations. Concern is growing that the release of large numbers of hatchery fish can threaten wild fish populations. Potential impacts include direct competition for food and other resources between wild and hatchery fish, predation of hatchery fish on wild fish, genetic dilution of wild fish stocks by hatchery fish allowed to spawn in rivers, and increasing fishing pressure on wild stocks due to hatchery production. Because of increased survival from eggs to smolts under hatchery conditions, fewer adults are needed to maintain a hatchery run. In a mixed fishery of hatchery and wild fish, a harvest rate based on the hatchery fish will tend to eliminate the wild fish (Hilborn, 1992). Upper Battle Creek has been blocked by the Coleman National Fish Hatchery fish barrier dam.

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

Ongoing projects to monitor chinook salmon populations in the Sacramento River Basin include the following:

- annual counts of fall-, winter-, and spring-run salmon at the RBDD;
- annual counts of fall- and spring-run salmon at hatcheries;
- snorkel surveys to count spring-run chinook salmon in summer holding pools of minor tributaries;
- mark-recapture surveys of adult salmon carcasses to estimate fall-run chinook salmon spawning escapement in the American, Feather, and Yuba rivers and selected tributaries of the Sacramento River, including Clear, Battle, and Cow creeks;
- aerial surveys of salmon redds in the upper Sacramento River;
- angler surveys in the upper Sacramento, Feather, and American rivers; and
- seining surveys and screwtrap operations for juvenile chinook salmon throughout the Sacramento River and its tributaries.

Outside the Sacramento River Basin, ocean salmon commercial and sport fisheries are monitored annually to estimate total pounds and numbers landed at major California ports.

Fisheries restoration, enhancement, and habitat improvement efforts in the Central Valley are largely focused on chinook salmon and steelhead trout with incidental benefits accruing for other anadromous species. DFG administers a major fisheries enhancement program that uses special bond funds and annual budget appropriations to restore or create spawning and nursery habitat and produce fish in hatcheries. The Upper Sacramento River Fisheries and Riparian Habitat Management Plan, developed jointly by DFG and DWR, outlines proposed fisheries and riparian restoration actions for the upper Sacramento River and minor tributaries (California Resources Agency, 1989). Elements of this plan are being implemented as resources allow.

The Central Valley Salmon and Steelhead Restoration and Enhancement Plan outlines DFG's restoration and enhancement goals for salmon and steelhead trout resources in the Sacramento

and San Joaquin basins and provides direction for DFG restoration, enhancement, and monitoring activities (Reynolds et al., 1990). The plan was prepared in response to the state's Salmon, Steelhead Trout, and Anadromous Fisheries Program Act of 1988; plan goals are provided below:

- restore all depleted salmon and steelhead habitat to a condition capable of sustaining population goals;
- at least double natural salmon production by the year 2000;
- develop an annual steelhead trout run in the Sacramento River Basin of 100,000 fish;
- ensure proper mitigation and compensation for losses of, or damage to, salmon and steelhead trout populations and habitat associated with existing projects;
- ensure that future projects either avoid adverse impacts or provide compensation where impacts cannot be avoided; and
- enhance the quality of fishing opportunities for inland sport and ocean sport and commercial fisheries and maintain populations at levels capable of sustaining year-round angling opportunities.

Specific fisheries protection, restoration, and enhancement actions currently underway or proposed in the Sacramento River Basin include the following actions:

- replenishing and improving spawning gravels in the upper Sacramento River and several minor tributaries;
- operating the Shasta Dam temperature control device to improve Reclamation's ability to meet water temperature objectives in the upper Sacramento River;
- raising the RBDD gates during winter-run chinook salmon migration periods;
- operating the Red Bluff Pilot Pumping Plant Project to evaluate the feasibility of using Archimedes screw pumps or helical pumps to improve fish passage at RBDD;
- conducting instream flow, water temperature, and other habitat studies, to determine habitat conditions and restoration measures needed to restore salmon populations;
- trucking hatchery salmon smolts to the estuary to enhance survival during emigration;
- releasing flow pulses to facilitate upstream migration of adult salmon and downstream migration of hatchery and naturally produced juveniles;
- limiting water diversions during crucial migration times; and
- acquiring and preserving riparian systems and associated communities.

Other restoration actions are being considered for the basin (Reynolds et al., 1990). Fisheries restoration in the Sacramento River Basin will benefit from efforts in the Delta to maintain the State Water Resources Control Board's draft Water Quality Control Plan (1995) and to meet Delta flow and operational criteria designed to protect the winter-run chinook salmon and delta smelt, which are federally and state listed as endangered species.

SAN JOAQUIN RIVER

Stock Abundance and Distribution

Historical Population Trends (1850-1966). Early impacts on chinook salmon in the San Joaquin River Basin were caused by gold mining activities, agricultural and power diversions, and overfishing (Clark, 1929). Prior to construction of water development projects in the San Joaquin River Basin, spring-run chinook salmon were the most abundant chinook race. Spring-run chinook salmon on the Stanislaus, Tuolumne, and Merced rivers were probably eliminated by 1930 as a result of dam construction. Large spring-runs migrating past the Merced River confluence were completely eliminated after 1947 following construction of Friant Dam, which blocked access to historical holding and spawning habitat and severely reduced flows in the San Joaquin River downstream from the dam (DFG, 1987b).

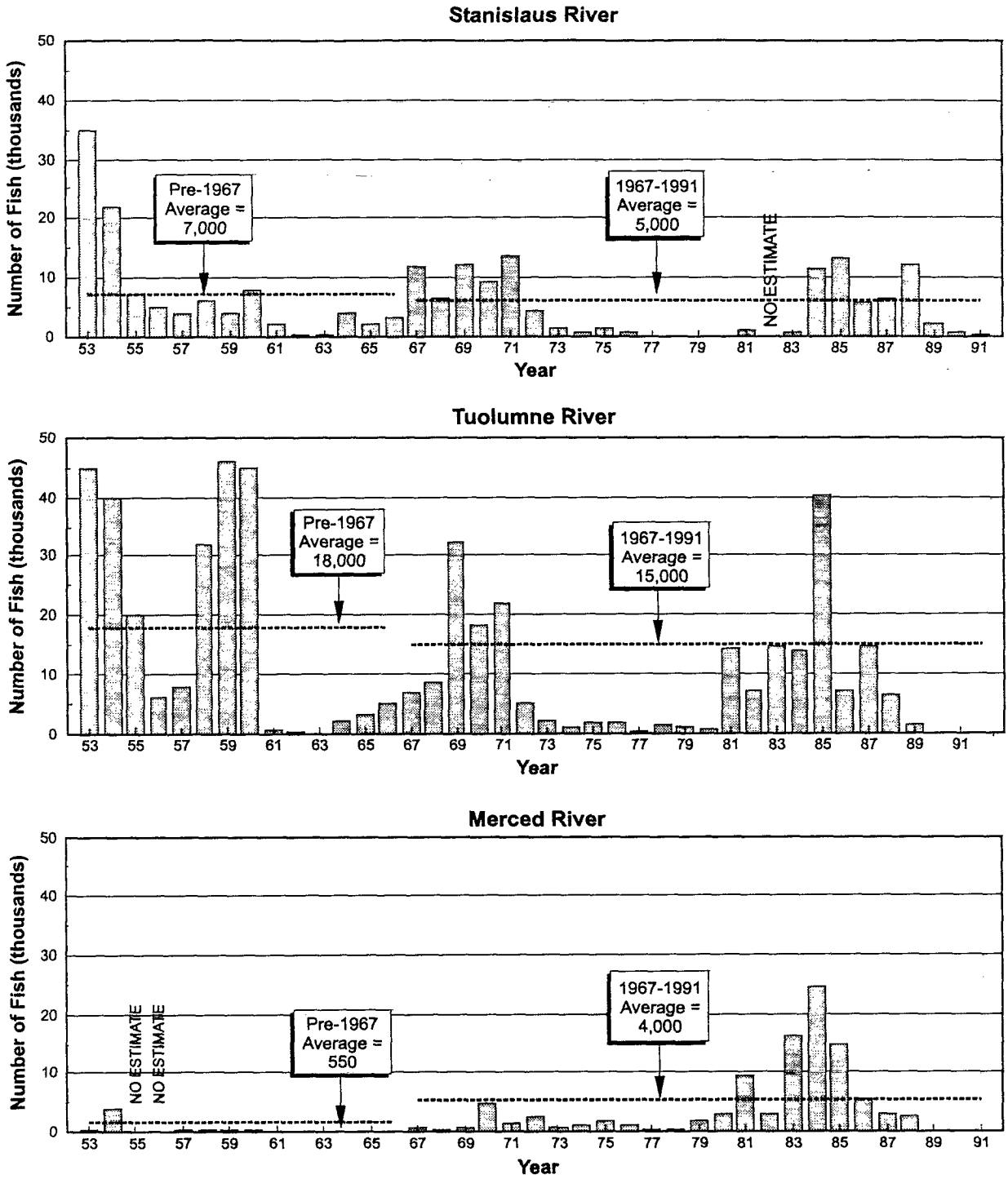
Annual spawning escapement of fall-run chinook salmon in the San Joaquin River Basin has been estimated since 1940, but early estimates are often incomplete and based on subjective methods (Reclamation, 1986b). Fall-run chinook salmon have been seriously reduced since the 1940s but have persisted as small but fluctuating populations below major dams on the Merced, Tuolumne, and Stanislaus rivers (Figure II-39). Low returns of fall-run chinook salmon to all three tributaries in 1961 were attributed to a fall migration barrier caused by low San Joaquin River flows, flow reversals, and low dissolved oxygen levels in the lower San Joaquin River and south Delta channels. Nearly complete run failures in 1962 and 1963 appeared to be related to low spring flows in 1959, 1960, and 1961 rather than to fall migration conditions (Hallock et al., 1970).

Current Population Trends (1967-1991). Recent spawning escapement levels of fall-run chinook salmon in the Merced, Tuolumne, and Stanislaus rivers show considerable annual variability (Figure II-39). Peak abundance generally follows high spring-runoff years. Conversely, spawning escapements are generally small following below normal or dry runoff years. Very low spawning escapements since 1990 are related to recent drought conditions (1987-1992).

The Merced River run has been partially sustained by production of yearling fall-run chinook salmon at the Merced River Fish Hatchery since 1970. The hatchery contribution to San Joaquin River chinook salmon stocks is less than 5 percent (DFG, 1987b).

Life History

See the Sacramento River Basin/Life History section of this chapter.



SOURCE:
Reclamation, 1986b; Mills and Fisher, 1993.

FIGURE II-39

ESTIMATES OF FALL-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN THE STANISLAUS, TUOLUMNE, AND MERCED RIVERS

Factors Affecting Abundance

The following discussion highlights those factors that have been specifically identified as having important effects on chinook salmon abundance in the San Joaquin River Basin. Other factors generally affecting chinook salmon abundance are not addressed here but are discussed in general in the Sacramento River section of this chapter.

Upstream Migration

Flow. For many years, attraction flows from the Merced River have proved inadequate during October, resulting in straying of adult salmon into agricultural drainage ditches, primarily Mud and Salt sloughs (Figure II-40). Barriers (electrical and physical) were installed across the San Joaquin River upstream of the Merced River confluence in 1992 to prevent salmon migration into these sloughs and help guide them into the Merced.

Water Temperature. Hallock et al. (1970) found that chinook salmon migrated into the lower San Joaquin River as water temperatures declined from 72 to 66 degrees Fahrenheit.

Water Quality. Low dissolved oxygen levels (less than 5 parts per million) and high water temperatures (greater than 66 degrees Fahrenheit) in the San Joaquin River near Stockton delayed or blocked the migration of adult chinook salmon during the 1960s (Hallock et al., 1970). Since 1964, fall migration problems have been reduced by improved wastewater treatment and installation of a physical barrier at the head of Old River in dry years to direct most of the San Joaquin flows down the main channel past Stockton. Despite these efforts, low dissolved oxygen levels recurred during recent drought conditions. Proposed remedial measures include increasing tributary outflow, evaluating and monitoring dredging activity in the Delta, and further evaluating the fall barrier at Old River (San Joaquin River Management Council, 1992).

Spawning

Flow. Relationships between chinook salmon spawning habitat availability and flow for the Merced, Tuolumne, and Stanislaus rivers are discussed in the Environmental Consequences chapter of this TA. Monthly flows in those three streams for the period 1967 to 1991 are presented in Attachment B.

Water Temperature. Water temperatures below major reservoirs in the San Joaquin River tributaries frequently do not permit successful spawning of fall-run chinook salmon until November. Monthly water temperatures in the Merced, Tuolumne, and Stanislaus rivers for the 1967-1991 period are presented in Attachment B.

Spawning Gravel. Although spawning habitat does not appear to be limiting recovery of fall-run chinook salmon stocks in the San Joaquin River Basin, spawning gravel restoration could be needed in the future to offset gravel depletions below dams and to provide sufficient spawning habitat to accommodate future adult populations. Several gravel restoration projects are currently being undertaken.

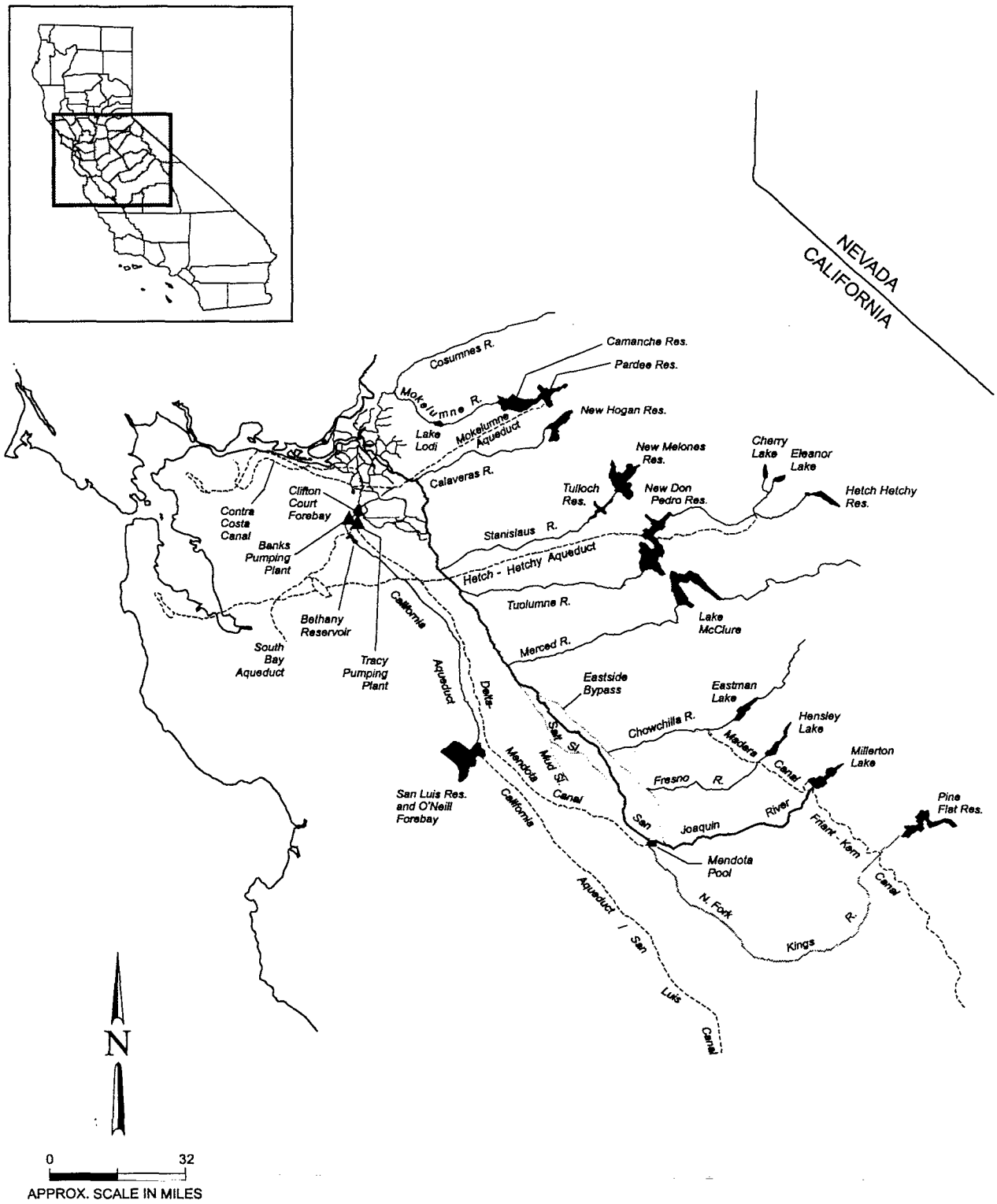


FIGURE II-40

SAN JOAQUIN RIVER BASIN

Egg Incubation and Emergence. The fishery management agencies have proposed an interim temperature objective of 42 to 56 degrees Fahrenheit throughout the designated chinook salmon spawning reaches in the Tuolumne, Merced, and Stanislaus rivers during the fall-run spawning and incubation periods. Special water operations using this objective were implemented on the Stanislaus River in 1991 and 1992 (San Joaquin River Management Council, 1992). Monthly water temperatures in the Merced, Tuolumne, and Stanislaus rivers for the 1967-1991 period are included in Attachment B.

Juvenile Rearing

Flow. Streamflow has been identified as the primary factor affecting abundance of chinook salmon stocks in the San Joaquin River Basin. Streamflow reductions after April and May in the Merced and Tuolumne rivers result in poor survival conditions for chinook salmon juveniles that remain in these tributaries beyond these months. High mortality generally results from reduced living space, high water temperatures, and increased predation. Current interim instream flow requirements in the Stanislaus River provide adequate flow conditions through the chinook salmon rearing period.

Relationships between chinook salmon rearing habitat availability and flow for the Merced, Tuolumne, and Stanislaus rivers are discussed in the Environmental Consequences chapter.

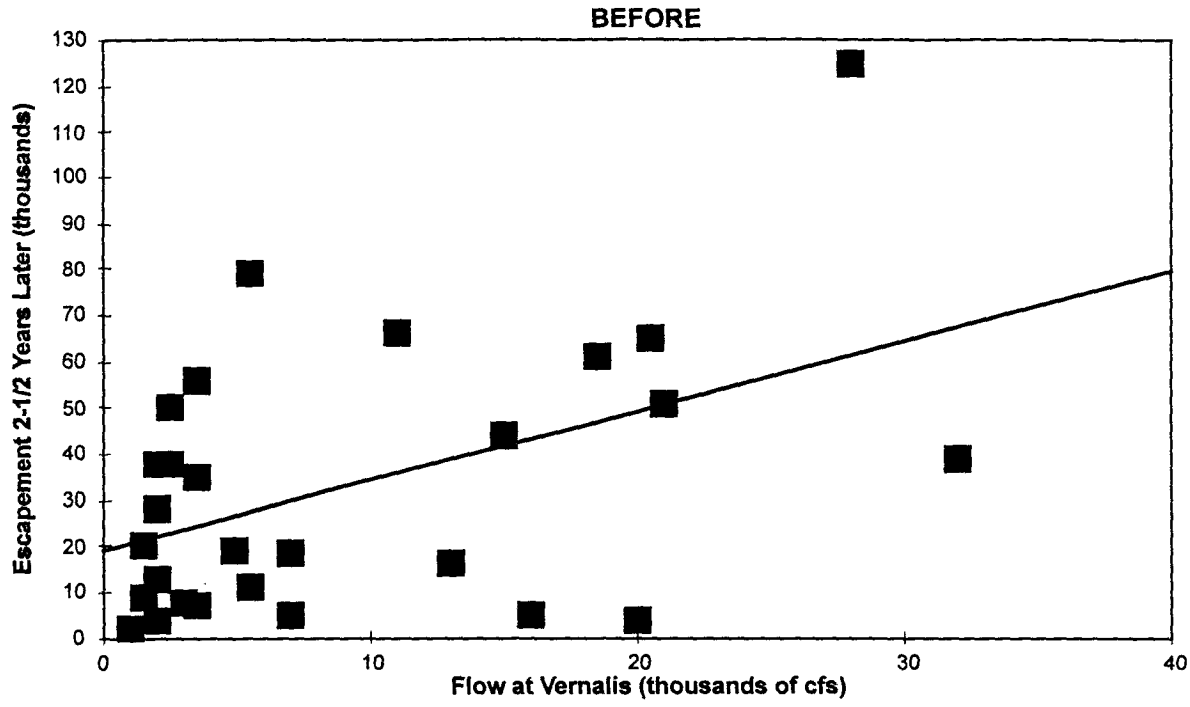
Water Temperature. Generally, water temperatures below major dams on the San Joaquin River tributaries become unsuitable for chinook salmon rearing in May or June, causing high mortality of juvenile chinook salmon that have not yet emigrated. In the Stanislaus River, however, releases of cold hypolimnetic water from New Melones Reservoir have improved water temperatures during the late spring rearing period relative to pre-impoundment conditions (Reclamation, 1986b).

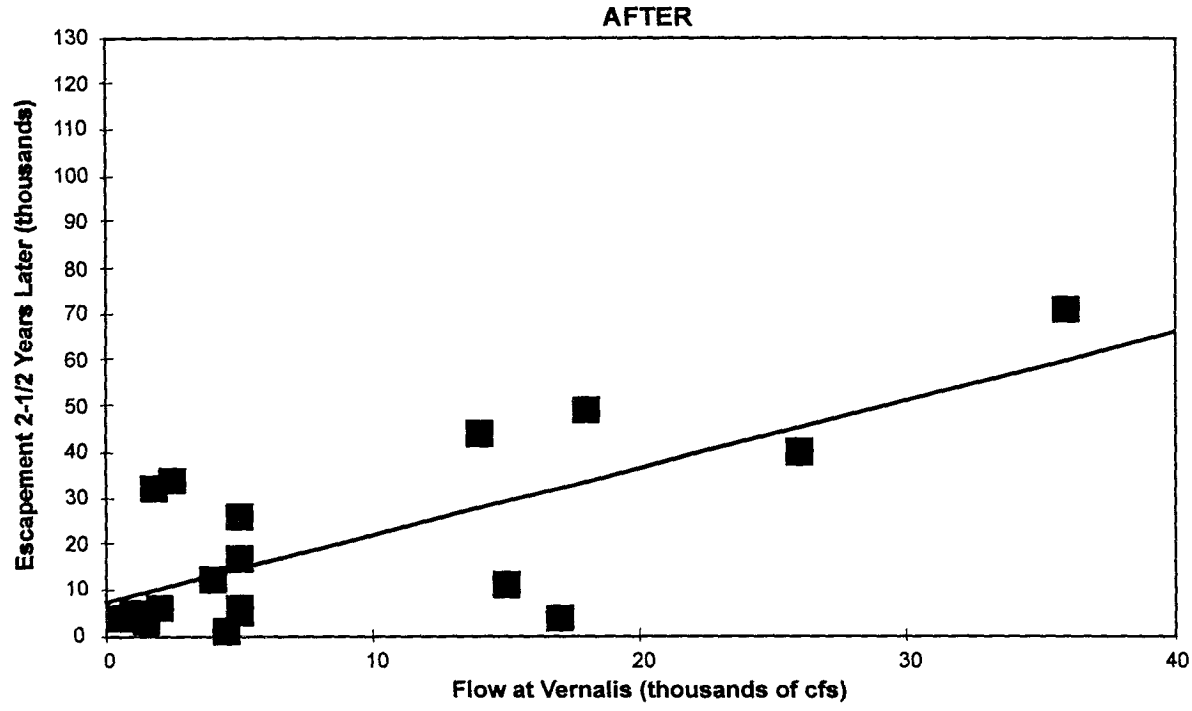
Monthly water temperatures in the Merced, Tuolumne, and Stanislaus rivers for the 1967-1991 period appear in Attachment B.

Water Quality. Selenium in agricultural drainage water poses a potential risk to juvenile chinook salmon in the San Joaquin River. Selenium is directly toxic to fish at elevated levels in the water column and through bioaccumulation in body tissues. Growth and survival of juvenile chinook salmon are adversely affected by exposure to dissolved and dietary selenium, but harmful levels have not been detected in the major rearing areas of the San Joaquin River and its tributaries (DFG, 1987b).

Juvenile Emigration

Flow. Spring flows in the San Joaquin River and major tributaries during the chinook salmon emigration period appear to have a major influence on the number of adults returning to the San Joaquin River Basin. Positive correlations exist between spring flows in the San Joaquin River and total chinook salmon spawning escapement 2.5 years later (Figure II-41). Greater inflow has been required to maintain chinook salmon escapement after the operation of the SWP (Figure II-41). Similar relationships for San Joaquin River tributary stocks indicate that the flow required to maintain a given spawning escapement level increased following operation of the





SOURCE:
DFG, 1987b.

FIGURE II-41

RELATIONSHIP BETWEEN CHINOOK SALMON ESCAPEMENT IN THE SAN JOAQUIN DRAINAGE AND VERNALIS FLOWS BEFORE AND AFTER THE EXISTING SWP IN THE SOUTH DELTA AND MAJOR STORAGE INCREASES IN THE SAN JOAQUIN DRAINAGE

CVP and SWP. Over time, increases in the significance of other mortality factors, such as increased Delta exports, have diminished the positive effects of incremental increases in spring flows. (DFG, 1987b.)

Smolts migrating down the San Joaquin River and through the southern Delta frequently encounter low flows, high temperatures, and high diversion rates. Currently proposed spring outflow recommendations for the Merced, Tuolumne, and Stanislaus rivers are designed to improve survival of juvenile salmon migrating down the tributaries, the mainstem San Joaquin River, and through the Delta. Recent evaluations have focused on the effectiveness of releasing short-duration, high-amplitude flows (i.e., pulsed flows) from tributary streams in conjunction with reduced Delta exports.

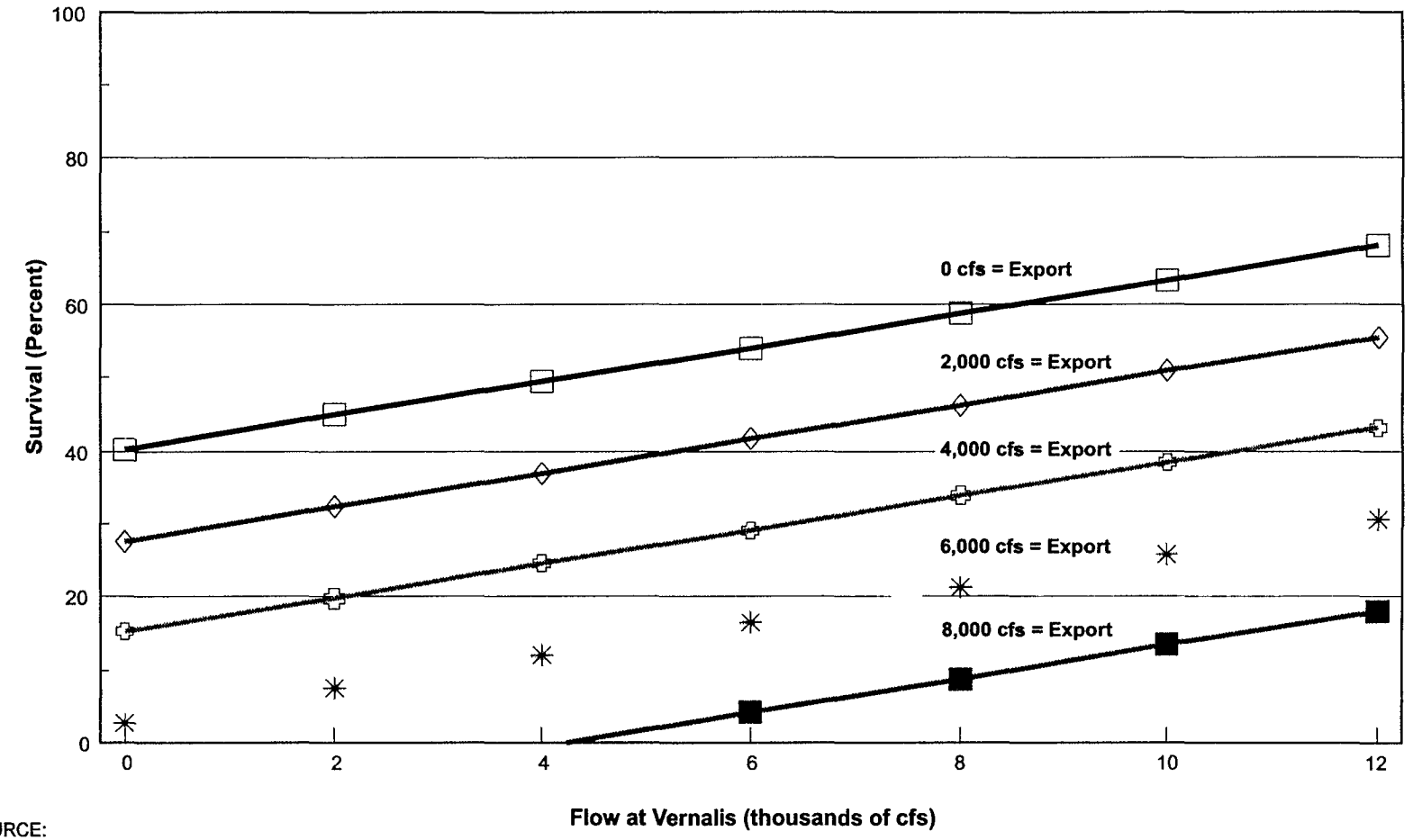
Water Temperature. Declining streamflow during the spring emigration period of fall-run chinook salmon coincides with rising air temperatures and increased agricultural return flows to the San Joaquin River, often resulting in deleterious water temperatures along much of the emigration route in the lower San Joaquin River. In May, water temperatures in the San Joaquin River near Vernalis often reach high chronic stress levels (greater than 67.6 degrees Fahrenheit) at flows of 5,000 cfs or less. Under these conditions, up to half the production of San Joaquin River chinook salmon can be subjected to harmful water temperatures. (DFG, 1987b.)

Delta Flows and Exports. Data indicate that pumping by the CVP and SWP export facilities in the south Delta has a major impact on survival of emigrating juvenile chinook salmon (Figure II-42). High juvenile mortality in the lower San Joaquin River and Delta is associated with low spring outflows and corresponding increases in the proportion of San Joaquin River flow diverted by CVP and SWP export facilities. At low San Joaquin River flow, high diversion rates increase the proportion of San Joaquin River flow drawn toward the pumps via Old River (shown in Attachment B). Juvenile salmon, diverted with the flow, experience reduced survival associated with increased migration time, high water temperatures, predation, entrainment in unscreened agricultural diversions, and Delta export pumping. Mark-recapture studies since 1985 demonstrated that chinook salmon smolts released in the San Joaquin River downstream of the head of Old River survived better than those released into upper Old River (Service, 1987 and 1990). Maximum survival benefits are expected as a result of reduced exports, increased San Joaquin flows, and a barrier at the head of Old River during the spring emigration period (Service, 1993) (Figure II-43).

Most chinook salmon reaching the CVP and SWP export facilities in the south Delta are from the San Joaquin River Basin (Reclamation, 1986b). Monthly salvage estimates at the CVP and SWP export facilities indicate the primary periods when juvenile chinook salmon are vulnerable to direct entrainment losses and mortality associated with salvage operations (Figure II-31).

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

Fall-run chinook salmon abundance in the Stanislaus, Tuolumne, and Merced rivers is estimated annually based on mark-recapture surveys of adult carcasses in the major spawning areas. Direct counts are made at the Merced River Fish Hatchery.



SOURCE:
Herrgesell, 1993.

FIGURE II-42

RELATIONSHIP BETWEEN PREDICTED SAN JOAQUIN RIVER CHINOOK SALMON SMOLT SURVIVAL AND FLOW AT VERNALIS AT FIVE DIFFERENT RATES OF COMBINED CVP/SWP EXPORTS

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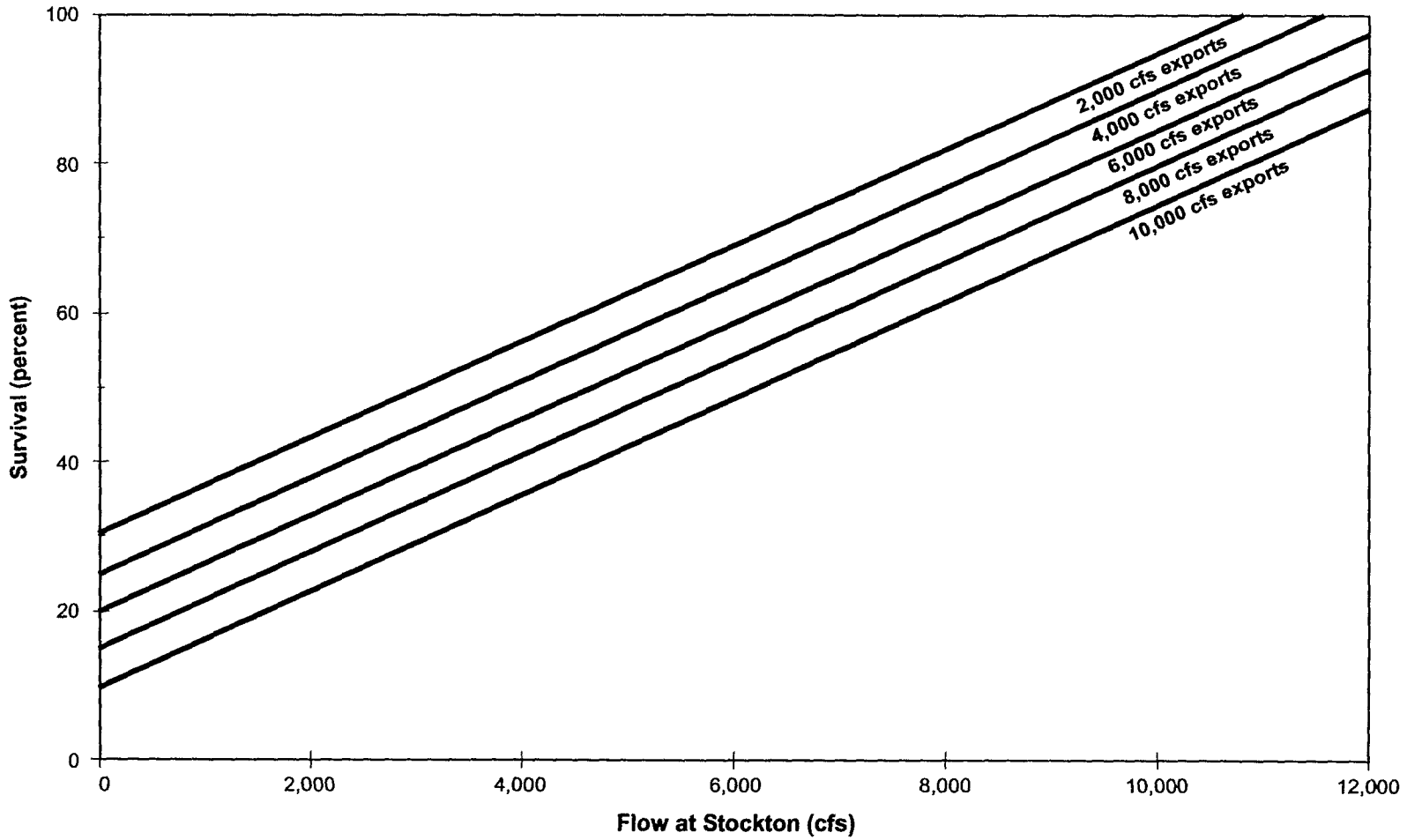


SOURCE:
Herrgesell, 1993.

FIGURE II-43

RELATIONSHIP BETWEEN PREDICTED SAN JOAQUIN RIVER CHINOOK SALMON SMOLT SURVIVAL THROUGH THE DELTA AND FLOW AT STOCKTON AT FIVE DIFFERENT RATES OF COMBINED CVP/SWP EXPORTS WITH A BARRIER AT THE HEAD OF OLD RIVER

C-081428
C—081428

Fisheries restoration and enhancement opportunities in the San Joaquin River Basin have been identified in the Central Valley Salmon and Steelhead Restoration and Enhancement Plan (see corresponding discussions in the Sacramento River section of this chapter), but implementation is not moving as rapidly relative to efforts in the Sacramento River Basin. Proposed actions include those provided below:

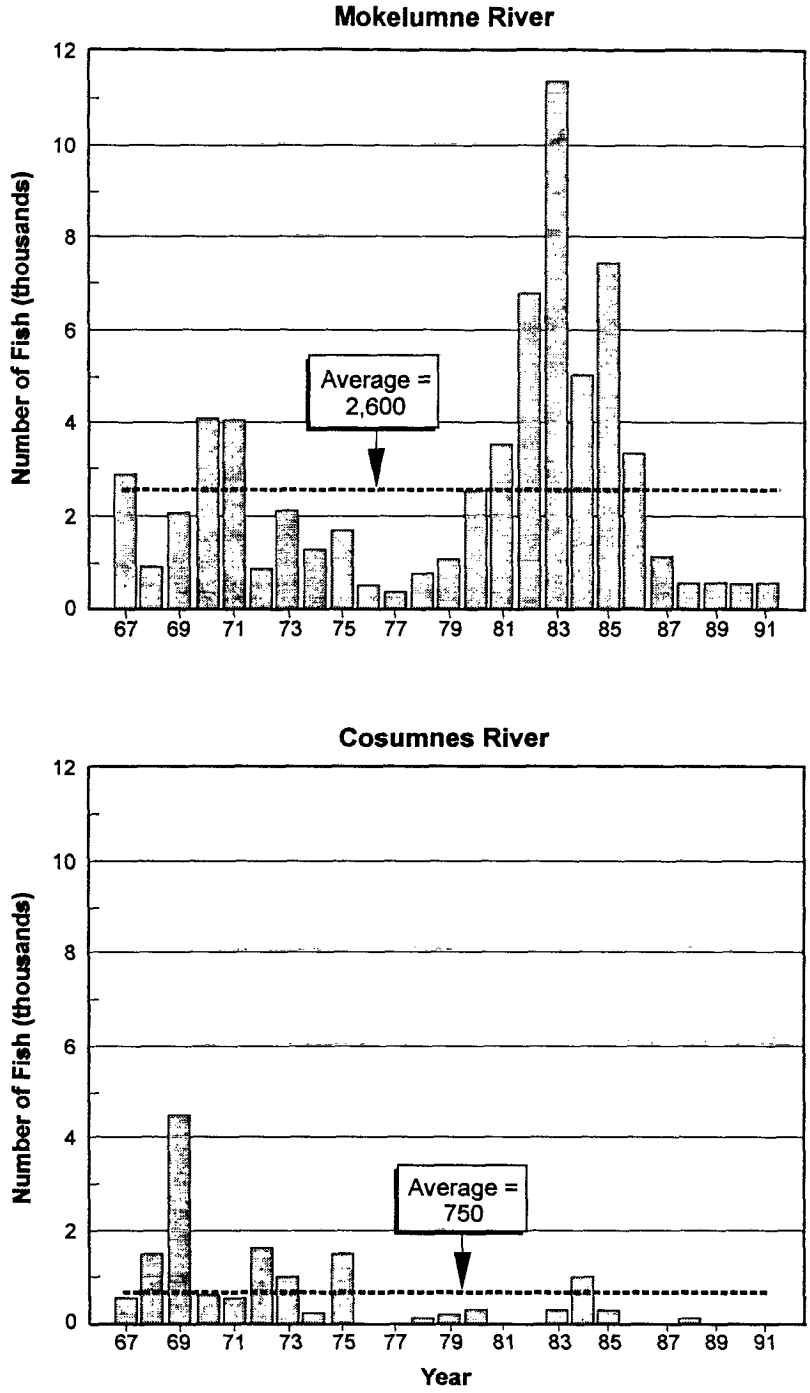
- improving flow and water temperatures in the Stanislaus, Tuolumne, and Merced rivers during the fall spawning migration and spring rearing and emigration periods;
- coordinating operation of major reservoirs and Delta export facilities to improve flow and water quality conditions in the San Joaquin River and Delta during the fall spawning migration and spring emigration periods;
- installing a physical barrier at the head of Old River in spring and fall to reduce diversion of smolts to the Delta export facilities and straying of adult salmon away from principal migration routes;
- installing and operating a fish barrier on the San Joaquin River at the Merced River confluence to prevent straying of adults into irrigation canals in western Merced County;
- replenishing and improving spawning gravels below tributary dams; and
- constructing new hatchery facilities on the Tuolumne River with an egg-taking station and imprinting pond on the Stanislaus River.

DELTA TRIBUTARIES

Stock Abundance and Distribution

Historical Population Trends (1850-1966). The earliest records indicate the presence of fall-run chinook salmon in the Mokelumne, Cosumnes, and Calaveras rivers (Clark, 1929). Spring-run chinook salmon were probably present in the Mokelumne River before the construction of Pardee Dam in 1929. Dams, poaching, and sedimentation caused by gold mining eliminated the spring-run in the Mokelumne River (Reynolds et al., 1990).

Declines in fall-run chinook salmon stocks probably paralleled declines in major San Joaquin tributaries. Since the early 1900s, chinook salmon in the lower Mokelumne River were adversely affected by poor water quality associated with winery and mine wastes, fish losses at unscreened diversions, and migration barriers due to dams (DFG, 1991). Runs up to 12,000 fish were recorded in the early 1940s. Since 1953, fall-run chinook salmon run size has varied considerably with peak salmon abundance, generally corresponding to similar peaks in the Stanislaus, Tuolumne, and Merced rivers. Annual spawning escapement in the Mokelumne River fluctuated between 100 fish in 1961 and 7,000 fish in 1958, with an average of 2,600 between 1967 and 1991 (Figure II-44). The Mokelumne River Fish Hatchery was constructed in 1964 as mitigation for the loss of spawning habitat between Camanche and Pardee reservoirs.



SOURCES:
Mills and Fisher, 1993.

FIGURE II-44

ESTIMATES OF FALL-RUN CHINOOK SALMON ANNUAL SPAWNING ESCAPEMENT IN THE MOKELUMNE AND COSUMNES RIVERS (1967-1991)

Between 1953 and 1966, annual fall-run chinook salmon spawning escapement in the Cosumnes River ranged from zero in 1961 to approximately 5,000 fish in 1954 and averaged approximately 2,500 fish (Figure II-44); however, the average escapement between 1967 and 1991 was 750 (Figure II-44).

A small population of fall-run chinook salmon could have existed in the Calaveras River before the construction of New Hogan Dam in 1963 (White, pers. comm.). Historically, chinook salmon production in the Calaveras River was limited by low, intermittent flows during summer and fall.

Current Population Trends (1967-1991). Since 1967, annual fall-run chinook salmon spawning escapement in the Mokelumne River has fluctuated between 250 and 11,000 fish and averaged about 2,600 fish (Figure II-44). Increased abundance during the 1980s has been attributed to increased smolt survival resulting from several high spring-runoff years and increased production of juvenile salmon at the Mokelumne River Fish Hatchery. Annual hatchery returns averaged approximately 500 chinook salmon adults between 1967 and 1991. Annual run size declined steadily following a peak in 1982 and has remained low during the recent drought period (1987-1992).

Annual fall-run chinook salmon spawning escapement in the Cosumnes River since 1967 has ranged from zero to 4,400 fish and averaged approximately 750 fish (Figure II-44). Since 1987, three years of no streamflow during the spawning season have precluded perpetuation of a natural run (Reynolds et al., 1990).

Operation of New Hogan Reservoir since 1963 resulted in sustained flows in the lower Calaveras River during summer and fall. Several hundred winter-run chinook salmon and smaller runs of fall-run chinook salmon and steelhead trout were thought to have entered the Calaveras River before the recent drought period. Since 1987, low flows and high water temperatures appear to have eliminated these runs (White, pers. comm.).

Life History

Refer to the Sacramento River Basin/Life History section of this chapter.

Factors Affecting Abundance

Nearly all information on factors affecting abundance in Delta tributaries pertains to the Mokelumne River. For a discussion of general impact mechanisms potentially affecting chinook salmon abundance, refer to the corresponding Sacramento River section of this chapter.

Upstream Migration

Flow. Inadequate attraction and migration flows (generally less than 50 cfs) below Woodbridge Dam during October and November have resulted in poor adult returns to the Mokelumne River and Mokelumne River Fish Hatchery. The failure of returning adults to detect Mokelumne River outflow could be exacerbated by diversion of proportionately large volumes of

Sacramento River water into the lower Mokelumne River via the DCC and by reverse flows in the lower San Joaquin River and south Delta channels.

Monthly flows in the Mokelumne, Cosumnes, and Calaveras rivers for the 1967-1991 period are presented in Attachment B.

Water Temperature. Upstream migration of adult chinook salmon in the Mokelumne River can be delayed by high water temperatures below Woodbridge Dam, which can persist until early November, even during a normal water year (DFG, 1991).

Monthly water temperatures in the Mokelumne, Cosumnes, and Calaveras rivers for the 1967-1991 period are presented in Attachment B.

Migration Barriers. The major barrier to upstream migration of chinook salmon adults on the Mokelumne River is Woodbridge Dam. This flashboard dam, constructed on the lower Mokelumne River in 1910, contained no fish ladder until 1925. Fish passage depended on river flows and the length of the irrigation season. Upstream migration of adult chinook salmon was generally possible only after the flashboards were removed at the end of the irrigation season (October). The fish ladder proved to be ineffective and was reconstructed in 1955. Recent analyses of passage conditions indicate that migration of adult chinook salmon past the dam is potentially impaired by spills that attract fish away from the fish ladder (DFG, 1991).

Using Thompson's (1972) criteria, DFG identified a shallow portion of the Mokelumne River near Thornton as a migration barrier to adult chinook salmon at flows less than 60 cfs (DFG, 1991).

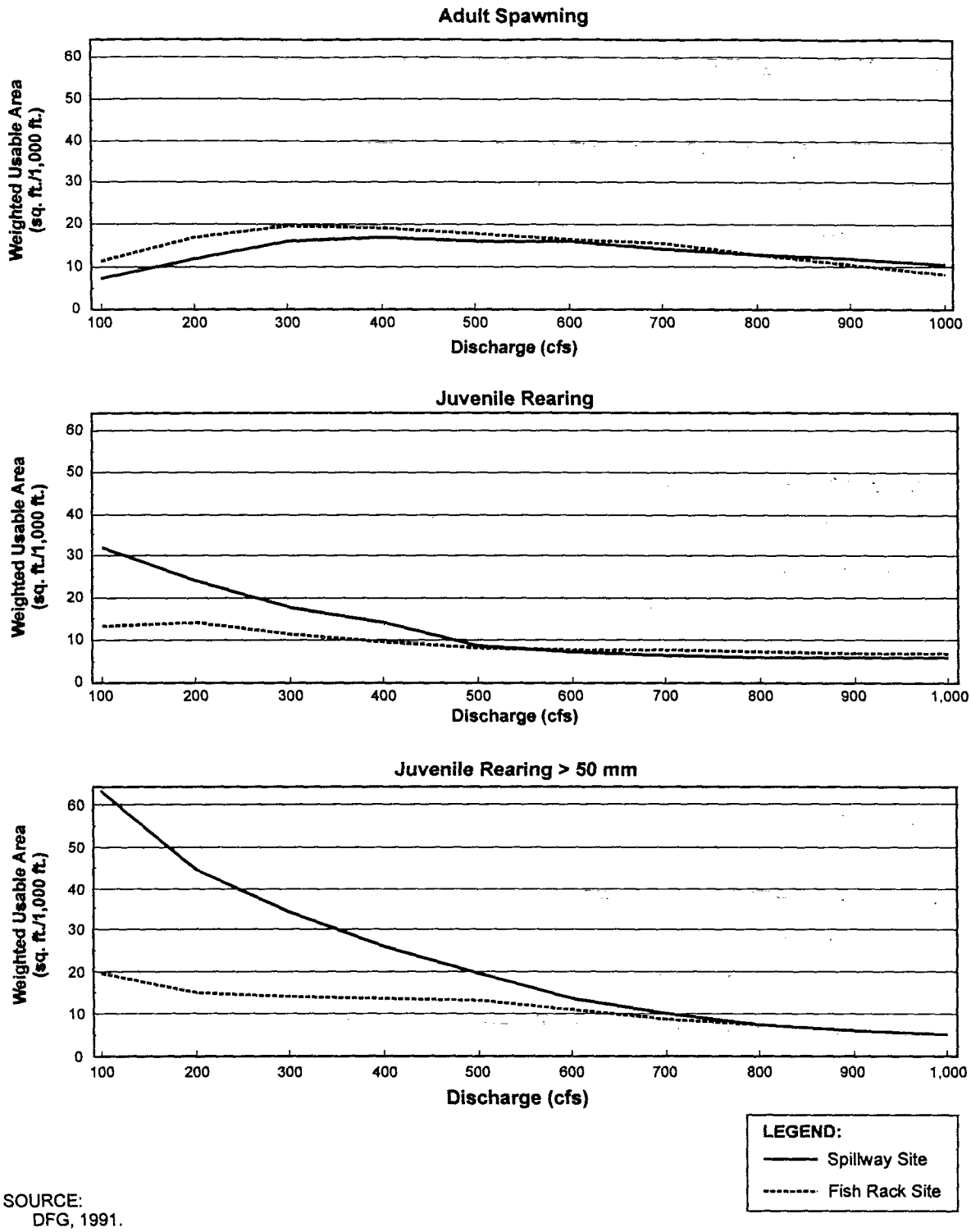
Water Quality. Poor water quality conditions below Camanche Reservoir could adversely affect chinook salmon by inhibiting upstream migration of adult chinook to spawning areas.

Water quality problems in the Mokelumne River have been associated with drought conditions and Pardee and Camanche reservoir operations. Recent fish kills at Mokelumne River Fish Hatchery were attributed to Camanche Reservoir discharges containing low dissolved oxygen levels. These conditions were associated with low inflows from Pardee Reservoir, record low reservoir levels, and hypolimnetic mixing which could have mobilized sediments during the late summer and fall turnover of the reservoir (DFG, 1991). DFG (1991) recommended water quality standards to protect aquatic resources in the receiving waters below Camanche Dam.

Spawning

Flow. Figure II-45 presents relationships between estimated chinook salmon spawning habitat availability and flow for the Mokelumne River.

Monthly flows in the Mokelumne, Cosumnes, and Calaveras rivers for the 1967-1991 period are presented in Attachment B.



SOURCE:
DFG, 1991.

FIGURE II-45

RELATIONSHIP BETWEEN FLOW AND HABITAT AVAILABILITY FOR CHINOOK SALMON SPAWNING AND JUVENILE REARING IN SELECTED REACHES OF THE MOKELUMNE RIVER

Water Temperature. Water temperatures for chinook salmon spawning in the Mokelumne River below Camanche Dam are generally not suitable until early November during a normal water year (DFG, 1991).

Monthly water temperatures in the Mokelumne, Cosumnes, and Calaveras rivers for the 1967-1991 period are presented in Attachment B.

Water Quality. Water quality standards recommended by DFG are designed to protect aquatic resources, including adult chinook salmon spawners.

Spawning Gravel. Camanche Dam prevents the natural recruitment of gravel from upstream sources to spawning areas below the dam. Net losses of spawning gravels and a general increase in the size of streambed materials have reduced the amount of suitable spawning area. In addition, armoring or compaction of spawning substrate has reduced spawning gravel quality.

Egg Incubation and Emergence

Water Temperature. Water temperatures for chinook salmon incubation and emergence in the Mokelumne River below Camanche Dam are generally not suitable until early November during a normal water year (DFG, 1991).

Monthly water temperatures in the Mokelumne, Cosumnes, and Calaveras rivers for the 1967-1991 period are presented in Attachment B.

Water Quality. Water quality standards recommended by DFG are designed to protect aquatic resources, including incubating eggs and fry.

Juvenile Rearing

Flow. Figure II-45 shows the relationships between chinook salmon rearing habitat availability and flow for the Mokelumne River.

Water Temperature. Water temperatures in the Mokelumne River below Camanche Dam remain within suitable levels for juvenile rearing and emigration through June during a normal water year. Water temperatures exceed suitable levels by April to early May at the Cosumnes River confluence and March to early June at Woodbridge Dam during all water year types examined. Under existing project operations, water temperatures at Woodbridge Dam are strongly influenced by air temperatures (DFG, 1991.).

Water Quality. Water quality standards recommended by DFG are designed to protect aquatic resources, including juvenile chinook salmon.

Flow Fluctuations. Potential stranding of juvenile salmonids as a result of flow fluctuations was evaluated in several reaches downstream of Camanche Dam based on predicted changes in wet surface area over a range of flows. The stranding potential increased at flows below 400 cfs. Rapid flow reductions also increased the stranding potential. (DFG, 1991.)

Juvenile Emigration

Flow. Dry-year flows in the lower Mokelumne River below Woodbridge Dam during the spring chinook salmon emigration period are not sufficient to effectively convey juvenile chinook salmon migrants downstream and through the Delta. Juvenile chinook salmon in the Mokelumne River are allowed to migrate naturally to the ocean in wet years but are trapped at Woodbridge Dam and trucked to Rio Vista in drier years. In general, peak adult returns to the Mokelumne River indicate favorable rearing and emigration conditions during preceding wet years. Nearly all chinook salmon produced at Mokelumne River Fish Hatchery are trucked as yearlings to release locations in the western Delta.

Diversions. Major diversions affecting juvenile chinook salmon emigrants from the Mokelumne River are the Woodbridge Canal diversion and the south Delta SWP and CVP export facilities. The Woodbridge Canal diversion was screened in 1968 and currently operates from April to October, depending on irrigation demands. The Woodbridge Canal fish screen does not meet current DFG fish screen velocity and design criteria but has not been shown to result in significant losses of downstream migrants. See the Sacramento River section of this chapter for a discussion of the effects of Delta export facilities on juvenile salmon.

Delta Flows and Exports. Smolts migrating naturally from the Mokelumne River are exposed to Delta flow patterns in the central and south Delta. Mark-recapture studies indicate that juvenile chinook salmon released in the lower Mokelumne River have higher mortality rates than those released in the Sacramento River below the DCC under dry-year conditions (Service, 1987). Reverse flows caused by CVP and SWP export pumping in the south Delta contribute to poor survival of juvenile chinook salmon that enter the central Delta from the Mokelumne River or from the Sacramento River via the DCC or Georgiana Slough. Other mortality factors associated with this migration route are high water temperatures, predation, unscreened agricultural diversions, and direct entrainment losses at the south Delta pumps. These factors will also affect downstream migrant chinook salmon from the Cosumnes and Calaveras rivers.

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

The fall-run chinook salmon population in the Mokelumne River is estimated annually based on direct counts at the Woodbridge Diversion Dam, on mark-recapture surveys of adult carcasses, and on counts at Mokelumne River Fish Hatchery.

Ongoing protective measures in the Mokelumne River include trapping of downstream migrating juveniles at Woodbridge Dam and trucking of these fish to release sites below major Delta water diversions. Hatchery juveniles are trucked directly from the hatchery. Flow and habitat restoration and enhancement measures proposed for the San Joaquin River Basin tributaries and Delta generally apply to eastside Delta tributaries.

RECENT CONDITIONS FOR STEELHEAD TROUT

SACRAMENTO RIVER

Stock Abundance and Distribution

Historical Population Trends (1850-1966). Historically, steelhead trout (*Oncorhynchus mykiss*) spawned and reared in the most upstream portions of the upper Sacramento River and most, if not all, of its perennial tributaries. Because they have greater swimming and leaping abilities than chinook salmon, steelhead trout could migrate farther into headwater streams where water temperatures were generally cooler. According to Hanson et al. (1940), an estimated 187 miles of accessible rivers and streams were blocked to chinook salmon by Keswick and Shasta dams alone; even more miles will have been blocked for steelhead trout. Dams and diversions for water supply, flood control, and sediment control were located on each of the major tributaries and blocked steelhead migrations to preferred spawning and rearing habitats.

Throughout the Central Valley, a 95 percent reduction (from 6,000 to 300 miles) of river available to anadromous fish (Reynolds et al., 1993) affects steelhead trout the most because of its migratory prowess. Although in some cases dams created favorable temperature conditions downstream, the physical habitat in the lower portions of these streams is not conducive to steelhead spawning and rearing as are stream reaches higher in the watersheds.

Few specific data are available regarding historical steelhead trout abundance. A commercial fishery for steelhead trout has never existed, and quantitative estimates of population abundance were not developed until the 1950s (and later than that in most streams).

Sacramento River Mainstem. Annual estimates of total (natural spawning and hatchery returns) Sacramento River steelhead runs upstream of the Fremont Weir (above the American and Feather rivers) ranged from 14,340 to 28,400 between 1953 and 1959, and averaged 20,500 (Skinner, 1962). The average estimated natural spawning portion of these runs was 88.6 percent.

Feather and Yuba Rivers. No definitive population estimates exist for steelhead in the Feather or Yuba River. It is likely that both river systems supported large steelhead runs in the 1800s. Hydraulic mining and diversion and storage dams on both rivers significantly reduced steelhead populations. For example, from 1910 to 1949, Daguerre Point Dam completely or almost completely blocked upstream migration (Dunn et al., 1992).

Pre-Oroville Dam Feather River steelhead populations were estimated to average approximately 1,000 fish above the dam site (Reynolds et al., 1990). Wooster and Wickwire (1970) estimated that approximately 200 steelhead trout spawned annually in the Yuba River prior to 1970.

American River. No definitive historical population estimates exist for steelhead in the American River. The steelhead run is estimated to have exceeded 100,000 fish annually before completion of Folsom and Nimbus dams in 1955, but before 1970, steelhead runs were estimated to average approximately 5,000 fish (Reynolds et al., 1993).

Minor Sacramento River Tributaries. Historically, steelhead runs were sustained in all tributaries with adequate flow and habitat qualities although no firm estimates of steelhead abundance exist. Counts conducted before 1967 enumerated populations in excess of 1,000 in both Mill and Deer creeks (Mills and Fisher, 1993). Average estimates for the 1950s and 1960s were approximately 300 steelhead in Antelope Creek and 150 steelhead in Big Chico Creek. These general estimates, however, were developed after barriers were erected and water diversions and habitat degradation had occurred on most sections of these streams; steelhead runs were likely much larger in these streams prior to the 1900s.

Current Population Trends (1967-1991). The average annual total steelhead run in the Sacramento River system was estimated by DFG in 1990 to be approximately 35,000 fish, primarily composed of hatchery-produced fish from Coleman National, Feather River, and Nimbus fish hatcheries. More than 90 percent of the annual steelhead run in the Central Valley is the result of hatchery-raised fish stocked as smolts or fingerlings (Reynolds et al., 1990).

Sacramento River Mainstem. Following completion of RBDD in 1964, steelhead runs could be counted at that location, but counts underestimate the total spawning run in the drainage since an unknown number remain below RBDD and spawn in the lower river and tributaries (Figure II-46). With that limitation, and subtracting the number of steelhead returning to Coleman National Fish Hatchery, an estimated average of 3,465 steelhead spawned naturally in the Sacramento River system above RBDD in the 1967-1991 period. Maximum and minimum estimated runs were 13,248 fish in 1968 and 0 fish in 1989, respectively. A distinct decline has occurred, with the estimated average run size decreasing from 15,055 fish in the first five years of the 25-year period to only 1,714 fish in the last five years.

Average steelhead returns to Coleman National Fish Hatchery during the same 25 years averaged 1,910 fish, 3,498 fish in the first five years and 979 fish in the last five years, a decline of nearly 75 percent (Figure II-47). Coleman National Fish Hatchery produces approximately 65 to 70 percent of the steelhead run to the upper Sacramento River (Reclamation, 1985; Reynolds et al., 1990).

Feather and Yuba Rivers. Steelhead currently spawn in the Feather River up to the fish barrier dam below Lake Oroville and in the Yuba River up to Englebright Dam. Steelhead in the Feather River are maintained from Feather River Fish Hatchery (Meyer, 1992). Overall, hatchery returns averaged 858 fish in the 1967-1991 period, with an increasing trend from an average of 790 in the first five years of the period to 1,386 fish in the last five years (Figure II-47). Annual angler catches of steelhead in the Feather River have been estimated as high as 7,875 fish in the past 10 years (Reynolds et al., 1993).

Limited information indicates that steelhead populations have increased on the Yuba River since the 1970 construction of New Bullards Bar Dam and Reservoir, which provided cooler summer rearing temperatures. DFG planted hatchery-raised steelhead smolts and fingerlings in most years from 1971 through 1983, and DFG estimated the 1975 run at 2,000 fish (Rogers, pers. comm.).

American River. Steelhead migrate up the American River to Nimbus Dam and the Nimbus Fish Hatchery, 23 miles upstream from the confluence with the Sacramento River. The

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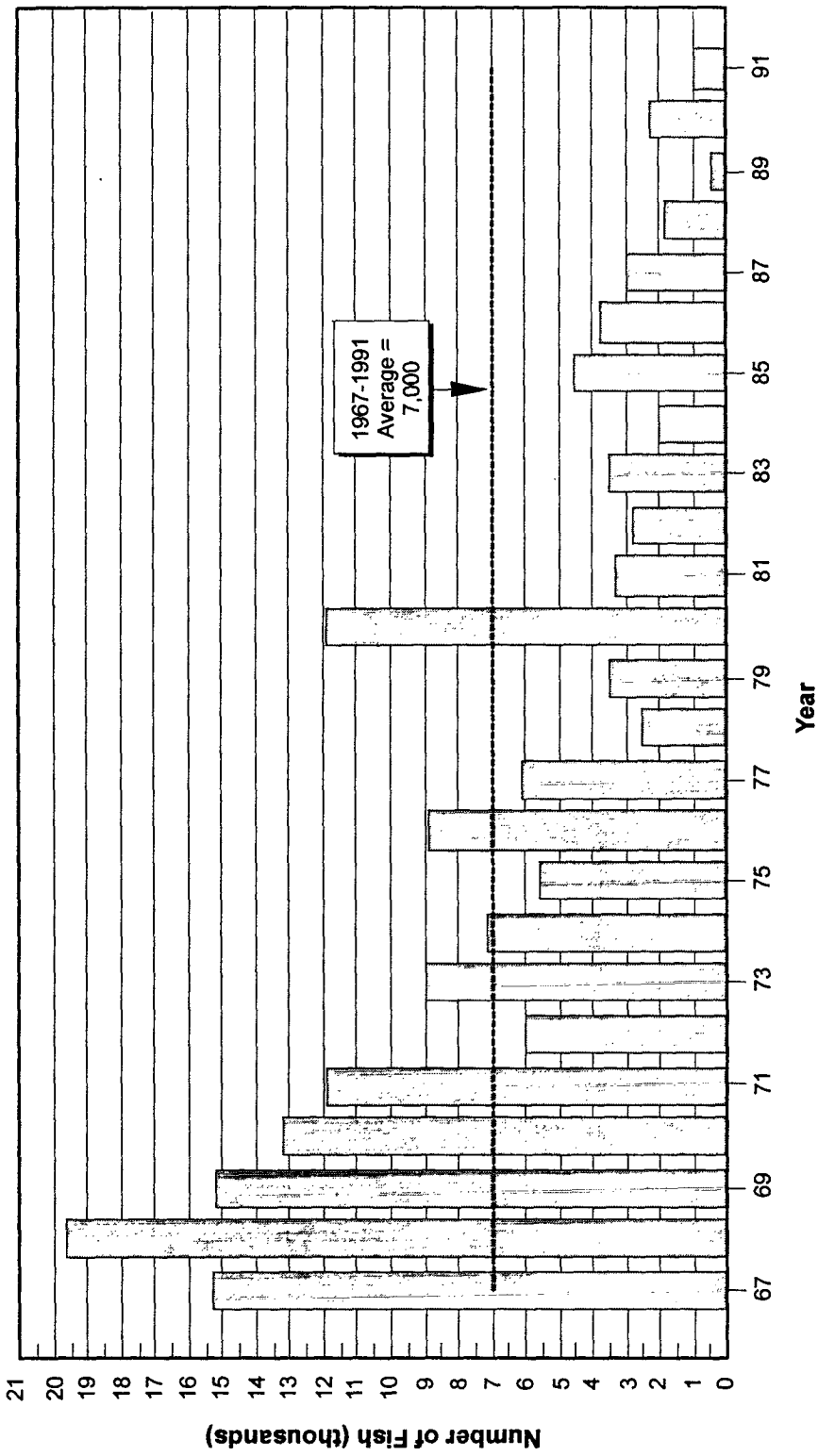
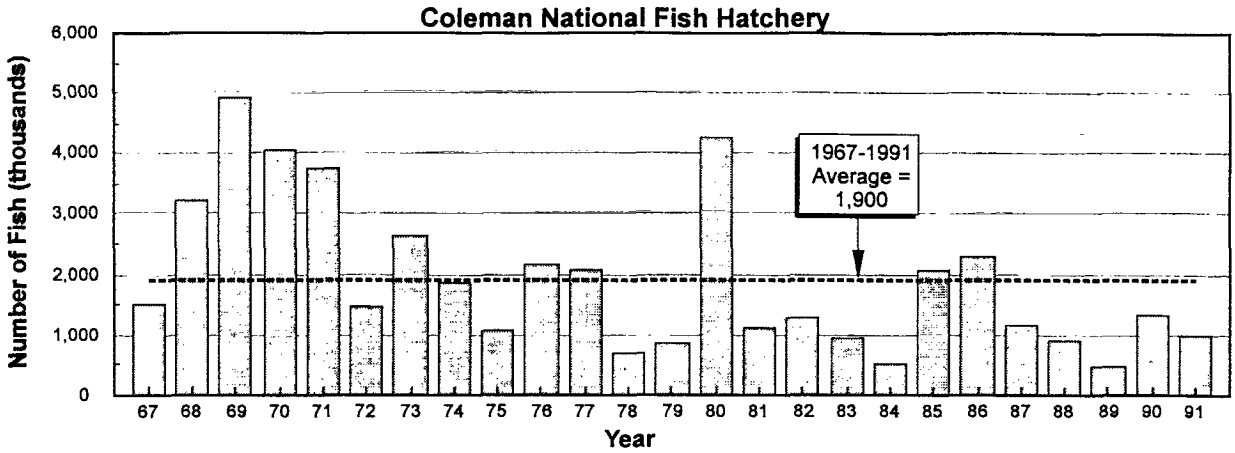
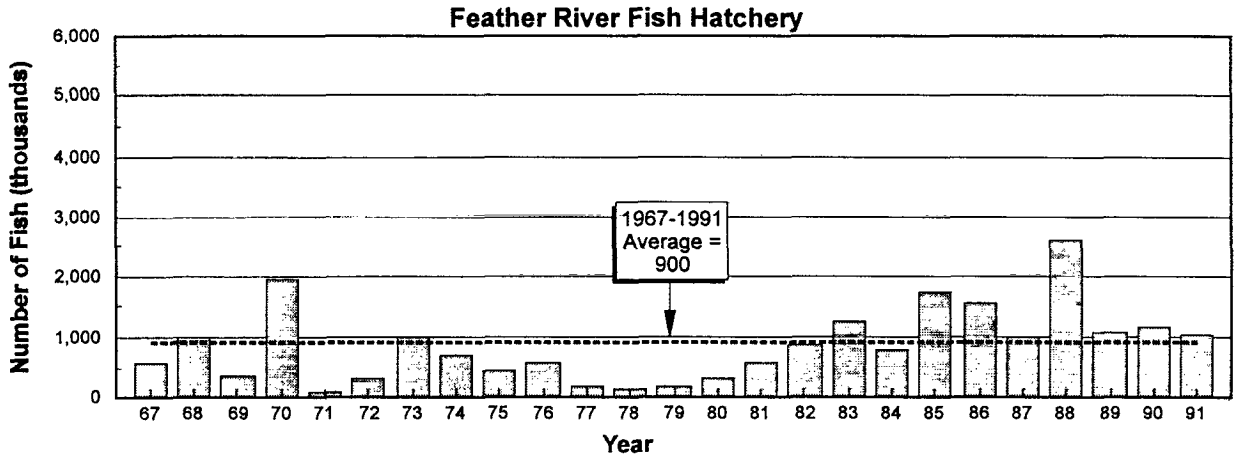


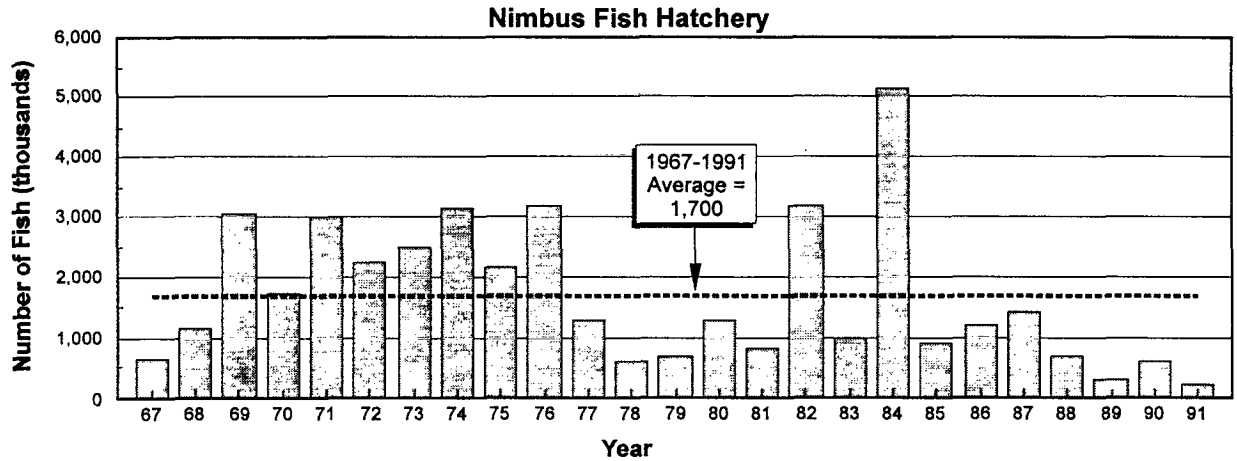
SOURCE:
Mills and Fisher, 1993.

FIGURE II-46

ESTIMATES OF ADULT STEELHEAD TROUT ANNUAL ABUNDANCE IN THE UPPER SACRAMENTO RIVER (1967-1991)







SOURCE:
Mills and Fisher, 1993.

FIGURE II-47

ANNUAL RETURNS OF ADULT STEELHEAD TROUT TO SACRAMENTO RIVER HATCHERIES (1967-1991)

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number of adults returning to the Nimbus Fish Hatchery averaged 1,694 fish in the 1967-1991 period, with no particular trend until the decline during the last four years (Figure II-47). Nearly all steelhead in the American River are hatchery produced, and many of the steelhead produced at Coleman National Fish Hatchery and Feather River Fish Hatchery stray and return to the American River.

Minor Sacramento River Tributaries. Because counts of steelhead generally come only from hatcheries or are incidental to counts of chinook salmon, no firm estimates of steelhead run sizes exist for minor Sacramento River tributaries. Steelhead trout runs are believed to have declined since the 1950s and 1960s in most of these streams. Runs in the larger tributaries, Big Chico, Mill, Deer, and Antelope creeks, are probably approximately 50 to 200 fish annually. Even smaller but unknown numbers of steelhead also use Clear, Cow, Cottonwood, Battle (in addition to those going to Coleman National Fish Hatchery), Paynes, and Butte creeks and Bear River (Reynolds et al., 1993).

Life History

Steelhead are generally classified into two races, depending on whether they begin their upstream migration in winter or summer. Historically, only winter steelhead trout are native to the Sacramento River Basin. However, summer steelhead have been introduced into the basin, as have strains of winter steelhead from the Eel, the Mad, the Rogue (Oregon) and the Washougal (Washington) river basins. Because of these introduced individuals, the genetic composition of the native steelhead trout could have been modified. Adult steelhead can be found in freshwater during every month of the year due to the influence of introduced genetic strains, modified and unnatural flow and/or temperature regimes throughout the basin. The general life history pattern of a typical steelhead is described in this section and presented in Figure II-48.

Upstream Migration. As an anadromous species, steelhead trout migrate to sea as juveniles and typically return to inland waterways as two- to four-year-old adults to spawn. Upstream migration begins when the fish enter freshwater, which occurs generally from July through February, depending on prevailing flow and temperature conditions. Relatively early attraction of steelhead trout into the tributaries can be triggered by occasional reservoir releases of cold water and natural high-water conditions in major Sacramento River tributaries. The upstream migration run can consist of both sexually mature adults and immature individuals who have spent only a few months at sea.

The smaller steelhead, sometimes called fall steelhead, enter the river in July, peak in November, spawn primarily in late December and January, and complete spawning by mid-February. The larger winter steelhead migrate upstream during mid-December through February, spawn in late January through early March, and complete spawning by April 1.

Steelhead stocks in the Sacramento River appear to respond to environmental conditions to a greater degree than do pure native stocks.

While adult steelhead are in freshwater, they rarely eat and consequently grow very little (Pauley et al., 1986).

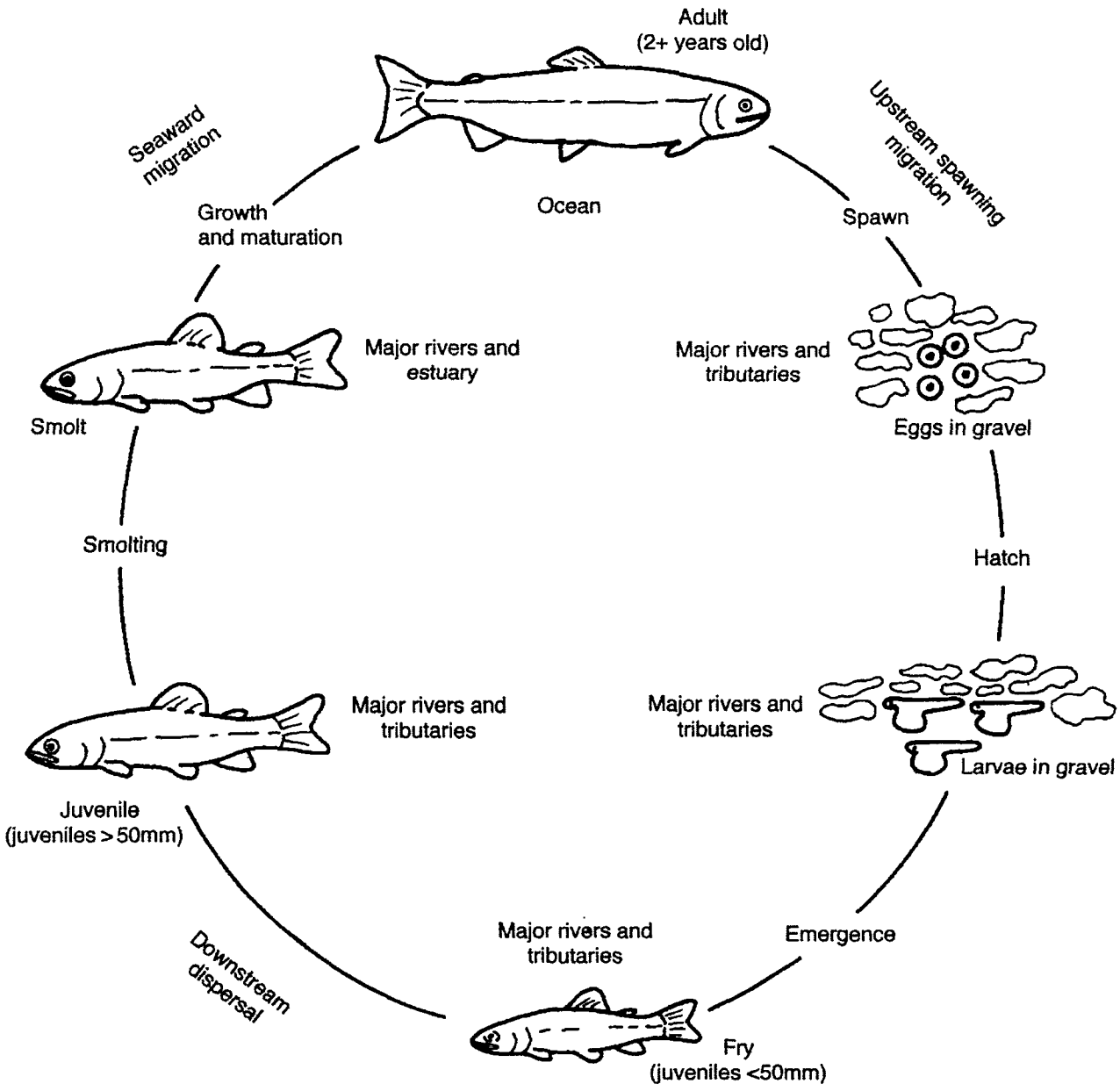

FIGURE II-48

LIFE HISTORY OF STEELHEAD TROUT

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Spawning. Natural spawning of steelhead in the Sacramento River system has been greatly reduced from historical levels. This reduction is due, in part, to installation of dams and other physical barriers that prevent steelhead from migrating to historical spawning grounds; to reduced spawning flows; and to other forms of habitat degradation in the stream reaches to which they currently have access. As a result, steelhead are highly dependent on hatchery operations to maintain their populations.

Spawning in the Sacramento River Basin takes place primarily from December through April, with most spawning occurring from January through March. Unlike chinook and other Pacific salmon, most steelhead do not die after spawning, and a small portion survive to become repeat spawners. During spawning, the female digs a redd (i.e., nest) and deposits her eggs, which are then fertilized by the male. The number of eggs a female produces is largely a function of her size. Female steelhead in the American River contain an average of 3,500 eggs, with a range of 1,500 to 4,500 eggs (Mills and Fisher, 1993). Female steelhead in the Sacramento River are smaller and contain an average of approximately 1,500 eggs (Bell, 1990). Females could deposit from a few hundred to more than 1,000 eggs per redd and require six or seven redds to complete spawning (Skinner, 1962). Females have a higher survival rate than males during and after spawning, and a few females could spawn as many as four times. Spawning males usually spawn with more than one female, remain in the stream up to two weeks longer than females after spawning, and expend more physical energy (Barnhart, 1986). Individual adult steelhead that survive spawning return to the sea between April and June (Mills and Fisher, 1993).

Egg Incubation and Emergence. Egg incubation time in the gravel is determined by water temperature, varying from approximately 19 days at an average water temperature of 60 degrees Fahrenheit to approximately 80 days at an average temperature of 40 degrees Fahrenheit. Under favorable conditions, up to 90 percent of the eggs hatch (Skinner, 1962). Steelhead eggs seem to be less tolerant of fine sediment than chinook salmon eggs, probably because oxygen requirements for developing embryos are higher for steelhead trout (Reynolds et al., 1990). Positive correlations have been demonstrated between steelhead egg and embryo survival and both the percolation rate of water through gravels and the oxygen content of the water (Reynolds et al., 1990). Steelhead fry usually emerge from the gravel two to eight weeks after hatching (Barnhart, 1986; Reynolds et al., 1993), which usually occurs in April and May on the American River (McEwan and Nelson, 1991).

Juvenile Rearing. Following emergence from the gravel, steelhead fry usually live in small schools in shallow water along streambanks. Mortality is high in the first few months after emergence. As the steelhead grow, the schools break up and the fish establish individual feeding territories. Though most live in riffles in their first year of life, some of the larger steelhead live in deeper, faster runs or pools. Their appearance and life-history traits are similar to those of nonanadromous resident rainbow trout.

Juvenile steelhead feed on a variety of aquatic and terrestrial insects and other small invertebrates, and newly emerged fry are sometimes preyed on by older steelhead.

Juvenile Emigration. With most stocks of steelhead, juveniles emigrate downstream to the ocean in November through May (Schaffter, 1980); however, most Sacramento River steelhead migrate in spring and early summer (Reynolds et al., 1993). Sacramento River steelhead

generally migrate as one-year-olds at a length of 6 to 8 inches (Barnhart, 1986; Reynolds et al., 1993). Emigration rates are influenced by water temperatures and current velocities. Although some steelhead have been collected in most months at the state and federal pumping plants in the Delta, the peak numbers salvaged at these facilities have been primarily in March and April in most years.

Ocean Residence. Much of the life of steelhead in the ocean is not fully understood. Steelhead can live one to four years in the ocean, with low survival in the 1-2 year class. They grow rapidly, reaching an average length of 23 inches after two years in the ocean. Immature grilse (two-year-old sexually mature steelhead) grow approximately 1 inch each month they are in the ocean.

Steelhead migration patterns at sea are not well known. Steelhead apparently tend to migrate north and south along the Continental Shelf and at least some spend part of their ocean life in the Alaskan gyre (Barnhart, 1986; Pauley et al., 1986).

Factors Affecting Abundance

Upstream Migration

Flow, Water Temperature, and Barriers to Fish Passage. The timing of upstream steelhead migration coincides with the timing of upstream migration of fall-, late fall-, and winter-run chinook salmon. In general, steelhead trout are attracted to high, cold flows, and such conditions provide optimal migration opportunities. Consequently, flow, water temperature, and passage-related factors affecting upstream migration of adult steelhead in the Sacramento River Basin are similar to those affecting chinook salmon (see the Recent Conditions for Chinook Salmon section of this chapter for more detailed discussion).

Passage at natural riffles is not as difficult for steelhead trout as it is for chinook salmon because steelhead are smaller and better swimmers and can better negotiate natural riffles and partial barriers. Nonetheless, minimum migration flows during major migration months are necessary to ensure that steelhead reach the preferred upstream spawning habitats.

Major dams have blocked access to most steelhead habitat in Central Valley rivers and streams. Smaller dams contribute to migration delay. Hallock (1989) estimated that passage problems at RBDD alone had reduced annual adult steelhead runs in the upper Sacramento River system by approximately 6,000 fish. Subsequent recorded declines in steelhead counts at RBDD could indicate continuing adult migration problems at RBDD. Steelhead trout production, however, is likely more limited by inadequate flows and temperature constraints influenced by holding capacity of reservoirs.

Sport Fishing. Sport fishing and illegal poaching affect migrating adult steelhead in the Sacramento River system in much the same way they affect chinook salmon (see the Recent Conditions for Chinook Salmon section of this chapter for more detailed discussion). However, poaching of steelhead is incidental compared to poaching of chinook salmon because steelhead are smaller, more difficult to catch, and generally less accessible to poachers. Unlike salmon,

steelhead generally do not die after spawning and are threatened by sport fishing on their return to the ocean.

The estimated annual sport catch of steelhead in the upper Sacramento River system above Big Chico Creek ranged as high as 11,000 fish in the 1950s and as high as 7,000 fish in the late 1960s; the present actual total population counts at RBDD averaging 1,714 fish between 1987 and 1991 extrapolate to estimated catches of less than 1,100 fish (Reynolds et al., 1990). In the lower Sacramento River system from Big Chico Creek downstream, approximately 8,000 steelhead are harvested in the Feather River during approximately 30,000 angler days per year, and an additional 1,000 to 2,000 Feather River fish are harvested downstream in the Sacramento River system and in the American River. An estimated 20,000 angler days each year result in an estimated catch of 5,000 to 8,000 steelhead on the American River. Hundreds of American River fish, along with steelhead from other sources, are also estimated to be caught incidentally downstream in the Delta and Carquinez Strait sport fisheries for other species. No estimates of steelhead harvests in the Yuba River and other Sacramento River system tributaries are available for the present period (Reynolds et al., 1990).

Spawning

Flow. Successful steelhead spawning flow requires appropriate water depths and current velocities over suitable spawning gravels. The water also must be of sufficient temperature and quality. Barnhart (1986) reported steelhead spawning in water depths of 5 to 28 inches, and Bovee (1978) reported an average water depth of 14 inches. Barnhart (1986) also reported steelhead spawning in water velocities of 0.5 to 3.6 feet per second, and Bovee (1978) reported a preferred velocity of 2.0 feet per second. Reynolds et al. (1993) reported a spawning velocity preference of 1.5 feet per second.

Several instream flow studies conducted in the Sacramento River Basin have developed spawning habitat-discharge relationships for steelhead trout incidental to studies for chinook salmon. Flows providing optimal spawning habitat could or could not increase steelhead abundance, depending on the limiting factors in each drainage. Arguably, spawning habitat could not be a limiting factor for steelhead production in most of the Sacramento River Basin; however, much of the historic spawning habitat is now inaccessible.

Water Temperature. From various experiments and literature sources, Leidy and Li (1987) reported the following temperature ranges for steelhead spawning:

Optimum	46.0-52.0 degrees Fahrenheit
Chronic Low Stress	52.1-57.5 degrees Fahrenheit
Chronic Medium Stress	57.6-61.0 degrees Fahrenheit
Chronic High Stress	Greater than 61.0 degrees Fahrenheit

Because steelhead spawning in the Sacramento River and its tributaries occurs from December through April (primarily January through March), water temperature is not considered a limiting factor for steelhead spawning in most of the Sacramento River Basin.

Spawning Gravels. Spawning redd sites selected by steelhead generally have gravel particle sizes that are 0.25 to 3.0 inches in diameter (Reynolds et al., 1993). The average redd size for Sacramento River Basin steelhead appears to be smaller than the average redd size in California streams reported as 56 square feet (Reynolds et al., 1993). Spawning success (egg hatching and fry emergence) is highly dependent on flow, temperature, and dissolved oxygen surrounding the developing embryos. Barnhart (1986) reported gravels with high permeability and few fines (less than 5 percent sand and silt by weight) in highly productive steelhead spawning streams. Most of the natural production of steelhead occurs in tributaries to the upper Sacramento River because mainstem spawning is limited by the shortage of smaller gravel, principally in the wide, braided areas of the river (Reynolds et al. 1990).

Although steelhead generally select somewhat smaller spawning gravels than do chinook salmon, the factors affecting spawning gravels for steelhead production in the Sacramento River system are similar to those affecting spawning gravels for fall-run chinook salmon production (see the Recent Conditions for Chinook Salmon section of this chapter), particularly in the larger stream systems and downstream of the larger dams. In some of the minor tributaries where passage is available during the spawning season, some steelhead ascend higher in the watershed than salmon, where they find suitable pockets of gravel to spawn.

Egg Incubation and Emergence. Flow fluctuation, water temperature, and water quality-related factors affecting successful egg incubation and emergence for steelhead are basically the same as for steelhead spawning and are discussed in detail in the Recent Conditions for Chinook Salmon section of this chapter. Physical and chemical conditions must remain fairly constant within the indicated ranges throughout the approximate two-month period that the eggs and pre-emergent fry are in the gravel. Flow fluctuation factors, in particular, can significantly reduce egg incubation and fry emergence success. Eggs are most susceptible to mortality during the early stages of development, and sudden changes in water temperature, oxygen rates, or percolation rates around the eggs can increase mortalities.

Juvenile Rearing. Juvenile steelhead generally rear in freshwater for up to 2 years before emigrating, usually in spring. Habitat and other related factors affecting juvenile steelhead in the Sacramento River system are similar to those described for juvenile chinook salmon (see the Recent Conditions for Chinook Salmon section of this chapter). Chinook salmon generally emigrate within a few months after emergence, however, and steelhead rear to a larger size than salmon. Consequently, juvenile steelhead depend more on large and abundant food resources than do salmon and also use deeper and faster runs and pools as they grow to larger sizes before emigration.

Flow. Another major difference between salmon and steelhead juvenile rearing is that steelhead juveniles must have suitable summer habitats; juvenile chinook salmon generally are not found in tributary streams during summer. Juvenile steelhead summer rearing habitat in the form of suitable flows and water temperatures is generally characterized as the major factor limiting steelhead trout abundance. Upstream barriers, typically large dams, also limit steelhead rearing to certain physical habitats (typically large, mainstem tributary rivers) that are not optimal or suitable for steelhead rearing.

Rearing flows must be adequate to provide the physical habitat needed by steelhead fry and juveniles, as well as that needed to produce the aquatic insects and other invertebrates on which they feed. Bovee (1978) shows steelhead fry using water approximately 2 to 15 inches deep but preferring depths of approximately 8 inches. Suitable water velocities are generally 0.3 to 1.0 foot per second, with optimal velocities approximately 0.6 foot per second. Bovee (1978) shows steelhead juveniles using deeper and faster water with water depths approximately 7 to 24 inches (optimal depths approximately 14 inches), and velocities approximately 0.3 to 1.5 feet per second (optimal velocities approximately 0.9 foot per second). The existence of pools can be especially important in streams that are naturally or artificially subjected to low-flow conditions in summer and fall.

Because steelhead trout rear year-round, suitable flows must be provided year-round, although in most streams the critical limiting factors occur during summer. Unlike juvenile salmon, steelhead are also susceptible to flow fluctuations and other flow characteristics year-round and are therefore exposed to in-river mortality factors for a longer period.

Water Temperature. Based on various experiments and literature sources, Leidy and Li (1987) generated the following temperature ranges for steelhead fry and juvenile rearing in the American River:

Optimum	55.0-60.0 degrees Fahrenheit
Chronic Low Stress	60.1-68.0 degrees Fahrenheit
Chronic Medium Stress	68.1-72.5 degrees Fahrenheit
Chronic High Stress	Greater than 72.5 degrees Fahrenheit

The actual effects of chronic low-, medium-, or high-stress temperatures on abundance, however, are influenced by a number of factors, such as exposure duration, acclimation factors, food availability, water quality, and groundwater dynamics. Numerous other water temperature criteria are available for steelhead trout fry and juvenile that are not presented here or are the basis of the criteria developed by Leidy and Li (1987).

Water temperature is obviously related to flow and is most likely the factor currently limiting natural steelhead rearing on many streams. Although cold-water releases occur below some dams, the amount and quality of habitat available for steelhead rearing below these dams are a fraction of what they were prior to human disturbances. In addition, cold-water releases are not available below many migration barriers or are only possible when reservoirs are full. Appropriate water temperature regimes in river reaches below many dams are not consistently maintained as they were naturally in the well-shaded upper watersheds before human disturbances. For example, lower American River water temperatures are commonly 60 to 77 degrees Fahrenheit from July through October and are not conducive to juvenile steelhead survival. Steelhead generally do not survive the extended warm waters in many years and move prematurely out of the American River to seek cooler water (McEwan and Nelson, 1991). These temperatures explain why natural production now contributes less than 5 percent of the adult steelhead population in the American River. Low summer flows and high temperatures have also been identified as creating unfavorable conditions in the Bear River (Reynolds et. al., 1993) and Clear, Cottonwood, Mill, Deer, and Butte creeks (California Resources Agency, 1989).

Sport Fishing. Juvenile steelhead are indistinguishable from resident rainbow trout in appearance, feeding, and other activities, and many are caught by sport anglers fishing for resident trout. On a statewide basis, DFG estimated that the fishing pressure on juvenile steelhead in 1965 exceeded that for adult steelhead (Barnhart, 1986).

Juvenile Emigration

Flow, Water Temperature, Diversions, and Predation. The effects of these factors on juvenile steelhead emigration in the Sacramento River system are similar to their effects on fall-run chinook salmon (see the Recent Conditions for Chinook Salmon section of this chapter) because of similarities in the timing and environmental needs of these two species during downstream migration. The principal difference between the two species is that steelhead juveniles are larger than most salmon emigrants. Except for their greater swimming ability, which can help them avoid or escape some sources of mortality better than salmon, steelhead are subject to the same sources of mortality and mechanisms as are salmon. For the most part, steelhead trout emigrate during spring.

Leidy and Li (1987) reported the following temperature ranges for steelhead emigration and smoltification:

Optimum	44.4-52.3 degrees Fahrenheit
Chronic Low Stress	52.4-59.3 degrees Fahrenheit
Chronic Medium Stress	59.4-63.2 degrees Fahrenheit
Chronic High Stress	Greater than 63.2 degrees Fahrenheit

Again, these are general ranges; the actual effects of these temperature ranges on steelhead trout emigration survival relate to numerous other factors. Additional sources of information that cite temperature criteria or preferences for steelhead trout are available.

Raleigh et al. (1984) reported that photoperiod (i.e., the number of daylight hours best suited to the growth and maturation of an organism) appeared to be the dominant triggering mechanism for transformation to smolts, with temperature affecting the rate of transformation, and that levels of smoltification of juvenile steelhead kept in water warmer than 55.4 degrees Fahrenheit from March through June were reduced. However, reduced flows and coincident warming spring water temperatures (a natural phenomena prior to dams) and high flows or freshets could also trigger juvenile emigration. Extended cold-water releases below dams could actually retard emigration until late spring, when increasing water temperatures and diversions in the mainstem Sacramento River and Delta constitute a larger mortality factor for steelhead smolts.

Delta Flows and Exports. Delta flows and exports could affect the abundance of downstream migrating steelhead much the same way as they affect fall-run chinook salmon (see the Recent Conditions for Chinook Salmon section of this chapter for more information). A portion of the water flowing down the Sacramento River is diverted into Georgiana Slough, the DCC, and Threemile Slough and into the lower San Joaquin River. A portion of the juvenile steelhead migrating down the Sacramento River enter these channels, and many are subsequently drawn toward the SWP Banks Pumping Plant and the CVP Tracy Pumping Plant. Although both pumping plants have louver-type fish screens that could be 90 percent effective for downstream

migrating steelhead, assuming that efficiencies for salmon (DFG, 1992a) are similar to efficiencies for steelhead, prescreening losses of 75 percent have been estimated at SWP pumping facilities, mostly due to predation in Clifton Court Forebay; while losses at Tracy are 15 percent.

The average annual number of steelhead salvaged at the SWP intake between 1968 and 1980 was 2,453 (DFG, 1981). Table II-6 lists the number of juvenile steelhead salvaged at these two pumping plants during the months of February through May, including migration peak salvage during March and April. Table II-7 lists losses of yearling equivalent steelhead at the SWP intake estimated by a formula negotiated between DFG and DWR. Salvaged steelhead are trucked to either the north or south side of Sherman Island or near Antioch. Some of these fish are lost to predation and stress associated with handling and trucking. Reverse flows (negative QWEST) in Delta channels caused by pumping operations could have adverse effects on steelhead similar to effects seen for salmon (Service, 1993). Unscreened diversions at the Contra Costa Water District (CCWD) intake at Rock Slough and at more than 1,500 unscreened agricultural water diversions in the Delta also cause unknown losses of emigrating steelhead. No steelhead have been caught in routine entrainment and impingement sampling at the screened intakes of Pacific Gas and Electric Company (PG&E) powerplants at Antioch and Pittsburg in the western Delta (Running, pers. comm.).

Hatchery Releases. More than 90 percent of the adult steelhead (greater than 15 inches in length) in the Central Valley are produced in hatcheries (Reynolds et al., 1990). Therefore, the number and survival to adulthood of hatchery-released steelhead presently has far more bearing on steelhead run sizes than does natural production. The sizes, timing, and points of release of hatchery-reared juvenile steelhead, as well as the same factors affecting naturally produced fish in the same physical environments, affect their survival rates. A major difference is that survival of eggs, fry, and juveniles before release is much higher for hatchery-produced fish. Because high survival rates of hatchery releases are desired, hatchery fish are released during periods and at sites most conducive to survival, whereas natural fish cannot be controlled in such a manner. Consequently, the survival of juvenile hatchery fish in particular could be higher than that of naturally produced juveniles, at least upon entering the ocean.

In operation since 1943, Coleman National Fish Hatchery has a capacity to raise approximately 1 million yearling steelhead, which grow to approximately seven fish per pound before being released to the upper Sacramento River near the mouth of Battle Creek, or in Battle Creek, in December and January (California Resources Agency, 1989; Reynolds et al., 1990). Feather River Fish Hatchery, in operation since 1967, and Nimbus Fish Hatchery, in operation since 1955, each have a capacity to raise approximately 400,000 yearling steelhead to a size of three to four fish/pound, after which the Feather River Fish Hatchery fish are planted in the Feather River below Yuba City, most by the end of March, and the Nimbus Fish Hatchery fish are trucked and released in the Carquinez Strait (Reynolds et al., 1990).

Ocean Residence

Little is known about the life of steelhead and the factors affecting their abundance during the one or two years that most spend in the ocean. Mortalities during this period are almost exclusively from natural conditions in the ocean environment. There is no commercial or sport

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TABLE II-6

NUMBER OF STEELHEAD TROUT SALVAGED AT SWP AND CVP DELTA PUMPING PLANTS FEBRUARY-MAY (1979-1991)

Year	February		March		April		May	
	SWP Intake	CVP Intake	SWP Intake	CVP Intake	SWP Intake	CVP Intake	SWP Intake	CVP Intake
1979	25	372	454	444	1,407	1,080	969	0
1980	835	0	74	90	118	243	210	126
1981	1,509	1,258	3,088	1,008	4,902	168	0	267
1982	1,432	0	1,110	0	10,965	0	2,441	297
1983	89	0	0	0	0	0	256	0
1984	0	0	41	146	357	187	18	70
1985	325	83	1,221	134	1,165	127	647	101
1986	139	524	54	127	1,328	505	446	238
1987	69	112	3,387	718	976	776	446	275
1988	2,403	0	823	491	2,116	1,039	426	1,646
1989	499	252	4,767	5,051	2,105	3,139	404	1,212
1990	1,317	1,085	3,115	2,139	1,039	786	19	0
1991	23	109	5,799	4,412	2,692	1,263	91	98

SOURCE:
DFG, 1981.

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TABLE II-7

ESTIMATED ANNUAL LOSSES OF STEELHEAD TROUT AT THE SWP DELTA INTAKE (1982-1991)

Year	Young-of-Year	Yearling
1982	0	73,748
1983	0	2,945
1984	0	1,713
1985	0	15,621
1986	0	15,663
1987	747	21,266
1988	0	25,080
1989	253	32,571
1990	0	19,187
1991	0	38,430

NOTE:
Estimates use the formula established under the 1986 agreement between DWR and DFG to offset direct fish losses in relation to the Harvey O. Banks Delta pumping plant.

SOURCE:
DWR, 1993.

fishery for steelhead in the ocean, and for unknown reasons they are rarely taken by commercial or sport salmon trollers (Skinner, 1962).

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

Monitoring of steelhead populations in the Sacramento River system is limited to the direct counts made at the RBDD, Coleman National Fish Hatchery, Feather River Fish Hatchery, and Nimbus Fish Hatchery. Steelhead outmigration is being assessed on the Feather River by DWR and on the American and upper Sacramento rivers by DFG. Angler surveys have also given sporadic steelhead harvest estimates for portions of the upper Sacramento, Feather, and lower American rivers.

Enhancement and habitat improvement actions benefiting steelhead are generally tied to overall anadromous fishery plans targeting chinook salmon races rather than steelhead (see the Recent Conditions for Chinook Salmon section of this chapter for more information). Various habitat studies, instream flow studies, evaluations of impacts of facilities on fish, hatchery program evaluations, fish screening and passage programs, and habitat restoration plans have been undertaken through the years. Because of the differences in life history, these improvement actions often fail for steelhead.

The upper Sacramento River Fisheries and Riparian Habitat Management Plan was prepared in response to 1986 state legislation (California Resources Agency, 1989). In 1989, the state legislature declared that it was state policy to implement actions recommended in the plan. The

plan reviews problem areas in the basin and recommends two riparian habitat restoration plan elements and 20 fisheries restoration plan elements.

Restoring Central Valley Streams: A Plan for Action (Reynolds et al., 1990) was the first step in developing a series of basin plans for anadromous fish waters in California. It was prepared in response to California's Salmon, Steelhead Trout, and Anadromous Fisheries Program Act of 1988, which makes it state policy to attempt to double the natural production of salmon and steelhead by the end of the century, to encourage a broad spectrum of public and private involvement in the restoration and enhancement programs, and to reduce the destruction of salmon and steelhead habitat. The 1990 plan inventories and identifies restoration needs for salmon and steelhead in the Sacramento-San Joaquin River system and states that, among other goals, DFG will develop an annual steelhead run of 100,000 fish in the Sacramento River system - 50,000 in the upper Sacramento River and its tributaries, and 50,000 in the lower Sacramento River tributaries. In response to the 1988 act and other actions, DFG prepared a Steelhead Restoration Plan for the American River (McEwan and Nelson, 1991).

DFG subsequently prepared a draft Central Valley Anadromous Fisheries and Associated Riparian and Wetland Areas Protection and Restoration Action Plan (Reynolds et al., 1993). This plan reviews anadromous fish resources of the Central Valley, discusses statutes and funding sources for restoration activities, and presents individual stream action plans for streams in the Central Valley Basin, including the Sacramento River and all of its major and most of its minor tributaries.

DFG developed a statewide steelhead management plan that identifies impacts on the state's steelhead resources and focuses mostly on habitat restoration and stock recovery. The plan was completed in February 1996 (McEwan and Jackson, 1996).

Other protection and restoration activities underway to benefit steelhead in the drainage include Reclamation's efforts to improve cool water releases from Shasta Dam for winter-run chinook salmon (in 1993, low-level cool water was released for 87 days) and ongoing programs to reduce predation, as well as the take of threatened or endangered species at the SWP and CVP pumping plants in the Delta, and other fish impact problems.

SAN JOAQUIN RIVER AND DELTA TRIBUTARY SYSTEMS

Stock Abundance and Distribution

Historical Population Trends (1850-1966). Historically, steelhead trout had sustained annual runs up the San Joaquin, Mokelumne, Stanislaus, Tuolumne, and Merced rivers. Steelhead runs will also have occurred in any other smaller tributaries with accessible headwaters, cool water temperatures, and appropriately sized gravels. Water development facilities and operations and other forms of habitat loss and degradation substantially reduced steelhead trout resources to remnant levels.

Current Population Trends (1967-1991). Few viable, naturally produced steelhead populations exist in the San Joaquin River system. Consistent hatchery-maintained steelhead

runs now occur only in the Mokelumne River, with sporadic runs occurring in the Merced River (Reynolds et al., 1993). Steelhead may have been present on the Calaveras River during this period, and a small but viable population of steelhead still exists in the Stanislaus River.

Steelhead migrate 64 miles up the Mokelumne River to the Mokelumne River Fish Hatchery (in operation since 1965) at Camanche Dam (completed in 1963). Between 1967 and 1991, hatchery returns were from 0 to 134 fish, with an average of only 40 fish. Efforts to create a naturally producing steelhead run have been unsuccessful to date (Reynolds et al., 1993), and there is no known recent natural spawning of steelhead in the Mokelumne River (Richardson, pers. comm.). Steelhead fry and juveniles have been known to rear only in the upper river reaches below Camanche Dam where temperatures are coolest (DFG, 1991).

The present program for the Mokelumne River calls for approximately 30,000 yearlings or older steelhead planted on a weekly basis in the river during the recreation season of April through September. This has provided a fishery for 12- to 20-inch trout that is popular with anglers, although few of these fish return to the Mokelumne River as adults (Reynolds et al., 1990).

Life History

Life history aspects of the few steelhead in these systems are likely similar to those described for the Sacramento River system. Upstream spawning migration runs in the Mokelumne River extend from September through January (DFG, 1991).

Factors Affecting Abundance

Factors affecting steelhead trout abundance in the San Joaquin River Basin are similar to those described in detail for San Joaquin River fall-run chinook salmon (see the Recent Conditions for Chinook Salmon section of this chapter). The primary factors limiting abundance and distribution are dams, water diversions, poor water quality, and riparian impacts. Low summer flows and concurrent high water temperatures preclude the necessary year-round rearing habitat for steelhead trout below the impassable dams (Friant, Crocker Huffman, LaGrange, Goodwin, and Camanche) on the mainstem San Joaquin River and its major tributaries.

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

Monitoring of steelhead populations in the San Joaquin River Basin is limited to direct counts made at the Mokelumne River Fish Hatchery and at Woodbridge Dam, both on the Mokelumne River. DFG has a goal of 2,000 adult steelhead spawners to return annually to Mokelumne River Fish Hatchery. The hatchery, which has the capacity to raise 100,000 yearling steelhead, has a goal to raise 40,000 yearling steelhead annually for release into the Mokelumne River. Because the target numbers of adult spawners do not currently reach Mokelumne River Fish Hatchery, the hatchery is supplied primarily from surplus Feather River Fish Hatchery and Nimbus Fish Hatchery eggs (Reynolds et al., 1993). Plants of Mokelumne River Fish Hatchery-raised steelhead typically return as adults to the American River (Reynolds et al., 1990).

Various habitat and instream flow studies, evaluations of facility impacts on fish, hatchery program evaluations, fish screening and passage programs, and habitat restoration plans have been undertaken through the years. Enhancement and habitat improvement actions benefitting steelhead are generally tied to overall anadromous fishery plans targeted at San Joaquin River fall-run chinook salmon. Because of the differences in life history, these improvement actions often fail for steelhead.

The San Joaquin River Management Program (SJRMP) Advisory Council and Action Team were created by Assembly Bill 3603 in 1990 to stem further deterioration in the San Joaquin River system and develop consensus-based solutions to water use problems within the system (San Joaquin River Management Program Advisory Council, 1992). The Fisheries Subcommittee of the SJRMP Advisory Council has prepared an action plan for chinook salmon in the drainage that will also aid in restoring steelhead populations if implemented. The Recent Conditions for Chinook Salmon section of this chapter provides greater detail about ongoing activities through the SJRMP.

The Central Valley Salmon and Steelhead Restoration and Enhancement Plan inventories and identifies restoration needs for steelhead in the Sacramento-San Joaquin River Basin and describes DFG's goal of attaining an annual steelhead run in the San Joaquin River system of 20,000 fish, equally divided between natural and hatchery production (Reynolds et al., 1990). The Recent Conditions for Chinook Salmon section of this chapter provides greater detail on the Central Valley Salmon and Steelhead Restoration and Enhancement Plan.

DFG's Lower Mokelumne River Fisheries Management Plan (DFG, 1991) identifies problems and recommends flows and other improvements for anadromous fish in that river. The draft Central Valley Anadromous Fisheries and Associated Riparian and Wetland Areas Protection and Restoration Action Plan (Reynolds et al., 1993), described in the Recent Conditions for Chinook Salmon section of this chapter, presents individual stream action plans for the San Joaquin River and its tributaries. DFG's statewide steelhead management plan focuses primarily on habitat restoration and stock recovery, including stocks in the San Joaquin drainage (McEwan and Jackson, 1996).

RECENT CONDITIONS FOR WHITE STURGEON AND GREEN STURGEON

WHITE STURGEON

Sacramento-San Joaquin River System

Stock Abundance. Population estimates and trends in abundance of white sturgeon (*Acipenser transmontanus*) in the Sacramento-San Joaquin River system have been conducted intermittently since 1954 (Pycha, 1956; Miller, 1972a; Kohlhorst, 1979, 1980; Kohlhorst et al., 1991; Brennan and Cailliet, 1991). The estimates relied on the recovery of tagged adult sturgeon and catch statistics from the sport fishery (fish longer than 40 inches).

Historical Population Trends (1850-1966). Prior to the late 1870s, sturgeon in the Sacramento-San Joaquin River system were harvested for their notochord primarily by Chinese

residents; the remainder of the carcass was discarded. Some Native American harvest and consumption have also been noted (PSMFC, 1992).

The earliest records of sturgeon abundance are 1875 catch data from the commercial fishery for sturgeon (Skinner, 1962). However, these records (annual reports of the U.S. Commissioner of Fisheries) often contradicted each other from year to year. Sturgeon catches between 1875 and 1892 averaged just under 500,000 pounds, with the largest catch reported as 1,658,000 pounds in 1885. Between 1892 and 1901, annual catch averaged between 100,000 and 200,000 pounds (Skinner, 1962). This intensive commercial fishery for sturgeon meat and roe used set lines and gill nets.

The commercial fishery produced the collapse of the fishery in the late 1800s. A substantial decline in abundance was noted in the 1890s, resulting in the total closure of all sturgeon fisheries in 1901 (Kohlhorst, 1980). The fishery was reopened twice in 1910 (and closed in 1912) and 1916 (and closed in 1917). No commercial or sport harvests of sturgeon were allowed in the Sacramento-San Joaquin River system between 1917 and 1954. However, some incidental take of sturgeon was associated with gill net fisheries for salmon and shad until gear restrictions were imposed in 1957 (Skinner, 1962).

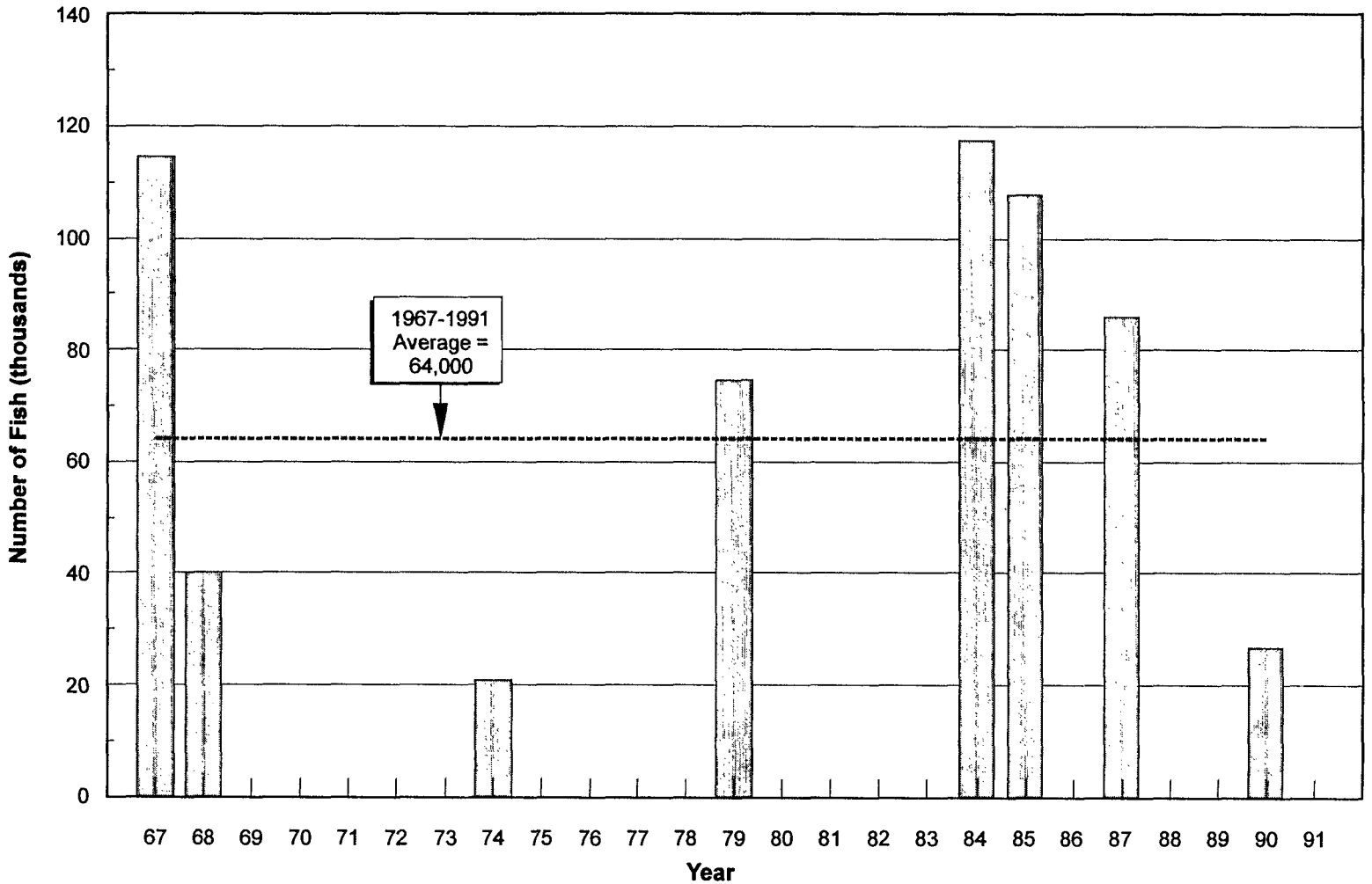
In 1954, a sport fishery was opened for sturgeon in the Sacramento-San Joaquin River system. Between 1954-1956 and 1963-1990, the sport fishery allowed a year-round fishery for adult fish greater than 40 inches and a bag limit of one sturgeon per angler per day. Between 1956 and 1962, the minimum size limit was increased to 50 inches. Based on aging studies, 40-inch sturgeon entered the fishery at eight to nine years of age (Shirley, 1987).

The first population estimate available for white sturgeon in the Sacramento-San Joaquin River system was reported for 1954. Pycha (1956) estimated an adult sturgeon population of 11,540 in San Pablo Bay; however, this estimate is considered low because of the methods employed in the study (Miller, 1972a). Population estimates made by Miller (1972a) for 1967 were much higher (114,667) than the 1954 estimates. Even allowing for the low 1954 estimate, Miller (1972a) was convinced that the population increased between 1954 and 1967.

Current Population Trends (1967-1991). Population trend information is based on legal-size adult sturgeon (longer than 40 inches). Population estimates for the period between 1967 and 1991 are presented in Miller (1972a), Kohlhorst (1979, 1980), and Kohlhorst et al. (1991) and summarized in PSMFC (1992) and Mills and Fisher (1993). Population estimates for 1967-1991 are presented in Figure II-49 and Table II-8.

Each year, a portion of the adult population moves up river to spawn. Based on studies conducted by DFG, the majority of the spawning run from the Delta apparently congregates between Knights Landing and just upstream of the town of Colusa. The primary spawning area is above Colusa; however, some white sturgeon spawn at least as far upstream as the RBDD (Kohlhorst et al., 1991).

Tag recoveries and catches in the sport fishery indicate that some adult sturgeon also migrate into the San Joaquin River. Adult sturgeon are caught in the sport fishery between Mossdale and the mouth of the Merced River in late winter and early spring, suggesting this could be a spawning



NOTE:
Average based on estimated and interpolated (not shown) annual values.

SOURCE:
Mills and Fisher, 1993.

FIGURE II-49

ESTIMATES OF ADULT WHITE STURGEON ANNUAL ABUNDANCE IN THE CENTRAL VALLEY, 1967-1991

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TABLE II-8

ANNUAL ESTIMATES OF ADULT WHITE STURGEON AND GREEN STURGEON IN THE CENTRAL VALLEY (1967-1991)

Year	White Sturgeon Population	Green Sturgeon Population	Years Population Estimated	Ratio White:Green
1967	114,700	1,850	X	62.0:1
1968	40,000	1,040	X	38.6:1
1969	36,783	900		
1970	33,567	760		
1971	30,350	620		
1972	27,133	480		
1973	23,917	340		
1974	20,700	200	X	101.9:1
1975	31,460	444		
1976	42,220	688		
1977	52,980	932		
1978	63,740	1,176		
1979	74,500	1,420	X	52.6:1
1980	83,120	1,378		
1981	91,740	1,336		
1982	100,360	1,294		
1983	108,980	1,252		
1984	117,600	1,210	X	106.3:1
1985	107,700	760	X	127.3:1
1986	96,850	635		
1987	86,000	510	X	163.7:1
1988	66,267	520		
1989	46,553	530		
1990	26,800	540	X	49.7:1
1991	--	--		
Average	63,501	867		

SOURCE:
Mills and Fisher, 1993.

run (Kohlhorst, 1976). Based on the ratio of tags recovered, Kohlhorst et al. (1991) estimated that approximately 10 percent of the Sacramento-San Joaquin River system stock migrate up the San Joaquin River. No studies have been conducted to definitively determine whether and where sturgeon spawn in the San Joaquin River.

One tagged adult sturgeon was recovered in the Feather River in the late 1960s (Miller, 1972a), indicating that sturgeon could also migrate into that system. However, no one has documented spawning or the presence of larval or juvenile sturgeon in the Feather River. Based on existing data, use of the lower portions of the Feather River system or other large Sacramento River tributaries by white sturgeon is minimal and not important for overall population maintenance or enhancement.

Some coastal migrations have been noted for adult sturgeon. Sturgeon tagged in the Sacramento-San Joaquin River system have been captured in Oregon estuaries (the Yaquina and Umpqua rivers and Tillamook Bay) and in Washington (the Columbia, Chehalis, and Willapa rivers) (Chadwick, 1959; Kohlhorst et al., 1991). Recent increases in tag recoveries of Sacramento-San Joaquin river sturgeon in distant coastal systems could be related to drought conditions, which have persisted over the last several years (Kohlhorst, pers. comm., cited in PSMFC, 1992).

Eggs, Fry, and Juveniles. Juvenile white sturgeon nursery areas extend down river from the spawning area to areas within the Delta. The primary factor influencing distribution of juvenile white sturgeon within the Sacramento River system is river flow, with distribution extending farther down river during wet years (Stevens and Miller, 1970; Kohlhorst, 1976). Eggs and larvae have been primarily collected near Colusa, Knights Landing, and the mouth of the Feather River; however, larvae have been found as far upstream as Hamilton City (Stevens and Miller, 1970; Kohlhorst, 1976). Larvae and young-of-the-year have been found in the Delta between Collinsville and Rio Vista and as far down river as Suisun Bay (Radtke, 1966; Stevens and Miller, 1970).

Life History

The white sturgeon, the largest freshwater or anadromous fish species in North America, can reach record sizes over 1,300 pounds. Historically, white sturgeon populations ranged from Alaska to central California (Scott and Crossman, 1973); however, the major spawning populations are now limited to the Fraser River (British Columbia, Canada), the Columbia River (Washington), and the Sacramento-San Joaquin River system.

Habitat utilization varies among populations. Portions of populations are considered anadromous, using fresh, brackish, and marine waters during different phases of their life history. In the Sacramento-San Joaquin River system, some of the mature adult sturgeon move upstream to freshwater environments to spawn during late winter and early spring. After hatching, juvenile sturgeon rear in fresh or slightly brackish waters for some period of time, dispersing downstream with the river currents. Subadults commonly rear in river sloughs, estuaries, or bays during summer and can move into deeper freshwater areas upstream, into the marine environment, or remain in the estuary in fall and winter. The proportion of sturgeon that uses marine environments is unknown, but tagging studies have demonstrated that some adults migrate along

the coast. Tag recoveries indicate that sturgeon migrating from the Sacramento-San Joaquin River system generally travel north (Kohlhorst et al., 1991). White sturgeon life history is depicted in Figure II-50.

White sturgeon are long-lived and mature some time after 10 years. Their longevity allows them to reach large sizes, reportedly weighing as much as 1,300 pounds at more than 100 years of age. The California sport fishing record is a 468-pound fish that was probably 40 to 50 years old when caught in the mid-1980s. Most females spawn for the first time at approximately age 15 and could spawn as infrequently as every five years thereafter (Kohlhorst et al., 1991).

Compared to salmon and steelhead, little is known regarding white sturgeon life history. For this reason, specific information available for the Sacramento-San Joaquin River system is augmented with other available information about other populations in the Pacific Northwest.

Upstream Migration. Upstream spawning migrations of white sturgeon in the Sacramento-San Joaquin River system occur between February and May (Miller, 1972a; Kohlhorst, 1976; Kohlhorst et al., 1991). Only a portion of the total adult sturgeon population migrates upstream from the Delta each year. Sturgeon that do move upstream are believed to be mature and ready to spawn. Kohlhorst et al. (1991) found that the mean size of tagged sturgeon recaptured up river was significantly greater (57 inches) than the mean size of all fish tagged (49 inches). Haynes et al. (1978) found a significant difference in the size of sturgeon that migrated upstream (72 inches) in the Columbia River system, in comparison to sturgeon that demonstrated only local (51 inches) or downstream movements (37 inches). No white sturgeon tagged in the Columbia River have been recaptured in California. Female and male white sturgeon tend to mature at 55 to 59 inches and 47 to 51 inches, respectively (Chapman, 1989).

Based on the recoveries of tagged adult sturgeon between 1974 and 1988 and collection of sturgeon eggs, larvae, and juveniles, most white sturgeon migrating up the Sacramento River

congregate and spawn between Knights Landing and a point just above Colusa; however, juvenile sturgeon have been found as far as the RBDD by the Service.

The environmental cues that initiate upstream migration are not well understood. Mature fish could be stimulated to migrate upstream by cues triggering the final stages of gonadal development, such factors as flow, velocity, photoperiod (i.e., the number of daylight hours best suited to the growth and maturation of an organism), or temperature (PSMFC, 1992).

Spawning. Sturgeon spawn in the Sacramento River between mid-February and late May, with a peak in spawning (93 percent) occurring between March and April (Kohlhorst, 1976). Not all adults migrate upstream to spawn each year. Sexual cycles in sturgeon are complex because these fish mature at a late age and adults do not spawn every year. It is likely that only mature sturgeon migrate up river to spawn and that most immature fish or fish in resting stages remain in the estuary.

Chapman (1989) studied sexual maturation in 836 white sturgeon collected from the Delta over several years. The sex ratio of immature males to immature females was approximately 1:1. The ratio of mature males to mature females was 2:1. The size range of adult sturgeon was bimodal,

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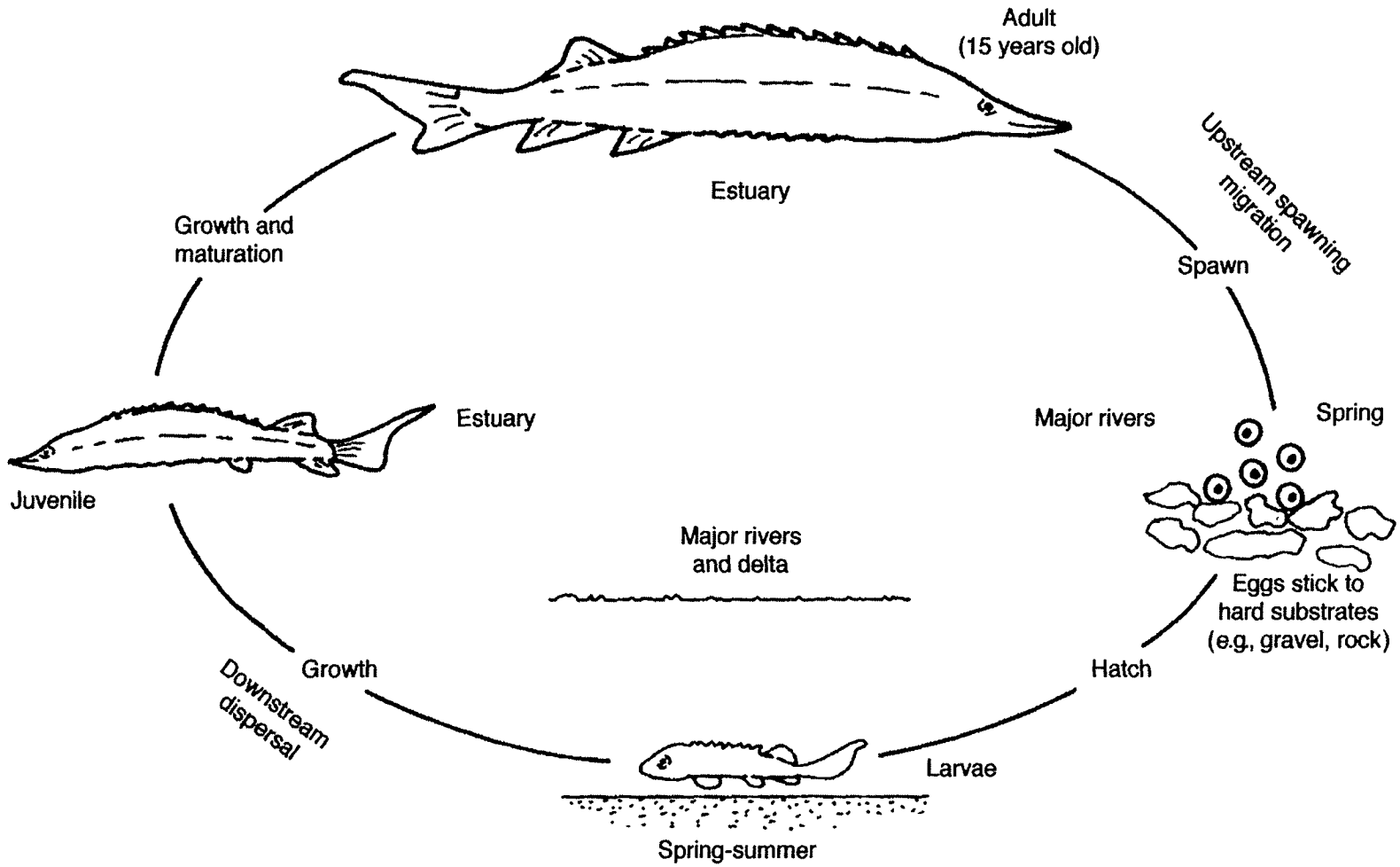

FIGURE II-50

LIFE HISTORY OF WHITE STURGEON

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with the average length of males (52 inches) being less than that of females (57 inches). Fish less than 35 inches long showed no gonadal development. No fish less than 39 inches in length with mature gonads were found. Of the fish studied, 44 percent were immature or in a resting phase of gonadal development, 31 percent were showing active egg and sperm development, and 28 percent contained mature gonads. The youngest mature fish were a 12-year-old male and a 14-year-old female. A higher percentage of the males (37 percent) were riper than females (15 percent). Chapman (1989) hypothesized that the timing of first sexual maturation depends on metabolic body size and body nutrient composition, with poor nutrition resulting in delayed sexual maturation.

Fecundity and periodicity of spawning of female sturgeon appear to depend on female age or size (PSMFC, 1992). Mature female sturgeon could carry 0.1 million to 7 million eggs, 7 to 30 percent of the female's weight. Spawning periodicity apparently varies. Estimates of spawning frequency range from 2 to 11 years (Stockley, 1981; Cochnauer, 1983; Doroshov, 1985). Fish age and size could also influence spawning frequency, with young, mature females spawning more frequently than older, larger females. Spawning habitat requirements for white sturgeon in the Sacramento-San Joaquin River system have not been definitively identified. Other sturgeon stocks are known to spawn in areas with gravelly or rocky bottoms, where currents are moderate to fast (Dees, 1961; Nikolskii, 1961), and with depths exceeding 9 feet (Galbreath, 1979; Doroshov, 1985).

Few observations of wild sturgeon spawning have been reported. Apparently sturgeon broadcast spawn in swift water. It is not known if eggs are fertilized in the water column or after they contact the bottom. The current initially disperses the adhesive eggs, which sink and adhere to gravel and rock on the bottom. The adhesive properties of the eggs are adaptive to spawning and retention of eggs within swift current environments. Sediments can reduce this adhesiveness of eggs (Conte et al., 1988).

Egg Incubation and Emergence. The incubation and emergence of white sturgeon have been studied under laboratory conditions to determine protocols for hatchery rearing. Hatching time depends primarily on water temperature. Egg incubation can last 4 to 14 days post-fertilization; yolk depletion can occur 15 to 30 days post-fertilization (Wang et al., 1985; Conte et al., 1988). Optimum temperatures for incubation and hatching range from 52 to 63 degrees Fahrenheit; higher temperatures result in greater mortality and premature hatching (Wang et al., 1985, 1987).

Under culture conditions, white sturgeon eggs hatch at the same time (Brewer, 1987; Conte et al., 1988). Mass hatching of sturgeon eggs generally occurs during darkness (Brewer, 1987). Both synchronous hatching and hatching during darkness could be adaptive mechanisms to minimize predation on larvae.

Larvae and Juvenile Rearing. Laboratory studies indicate that larval sturgeon demonstrate three behavioral phases after emergence: swim-up and dispersal, hiding, and feeding (Brannon et al., 1986; Brewer, 1987; Duke et al., 1990; Miller et al., 1991). After hatching, yolk sac larvae swim up into the water column. The currents act as a dispersal mechanism, transporting larvae downstream of the spawning area. Larvae swim toward or to the surface, then passively sink to the bottom (Brewer, 1987). Either immediately or shortly after touching bottom, the larvae

repeat the swimming activity. The duration of this phase is variable and lasts from one to five days (Brewer, 1987). During periods of higher flow, larvae are dispersed farther downstream, which is thought to result in higher recruitment (Kohlhorst et al., 1991). However, Brewer (1987) indicates that larvae initiate the hiding behavior phase more rapidly at higher flow velocities (0.3 foot per second).

When larvae enter the hiding phase, they are still nourished from the yolk sac. To hide, larvae place their heads within substrates (either rock or vegetation) and maintain a constant tail beat to retain their position. Substrate preference of hiding larvae is related to the degree of darkness the substrate provides, a negative phototaxic (i.e., movement away from light) response. This hiding behavior is thought to provide protection from predation as the larvae develop. (Brewer, 1987.)

Larvae develop the mouth and olfactory morphology needed for feeding before the yolk sac is completely absorbed. Although larvae do not appear to forage actively in the hiding phase, they are capable of consuming exogenous food during the later stages of the hiding phase if it is present at the hiding site. (Brewer, 1987.)

Exogenous feeding occurs approximately 12 days after hatching at temperatures of 63 degrees Fahrenheit (Buddington and Doroshov, 1984). During this phase, the larvae move out of hiding to forage actively for food. Young sturgeon appear to be opportunistic, nonvisual feeders (Miller, 1987). The senses of smell and touch appear to be more important than vision for locating prey. Based on Miller's observations, sturgeon that have progressed to the feeding stage swim above the substrate with their barbels (i.e., fleshy whiskers) close to the substrate. At times, perhaps when encounters with prey are infrequent, the larvae swim into the water column and settle at another location to continue searching for food. Feeding larvae that detect the chemical odor of a prey in the water column will descend to the bottom and actively seek that prey (Brannon et al., 1984; Brewer, 1987). Larvae are also territorial during this phase (Brannon et al., 1984).

Under culture conditions, larvae develop a search "image" at the onset of exogenous feeding (Conte et al., 1988). The larvae probably imprint on chemical composition of foods. There have been no studies of the feeding habitats of sturgeon larvae in the wild. The bottom and near-bottom are probably where the dominant food items in the diet of sturgeon larvae are found, but they also feed on pelagic fish and zooplankton (Brannon et al., 1987).

The diet of sturgeon changes as the fish become larger. Young-of-the-year sturgeon (less than 8 inches long) feed on a number of prey, including small crustaceans and insect larvae, and can potentially consume small fish fry. *Corophium* spp. and *Neomysids* (Schreiber, 1962) are the most common prey of sturgeon captured in the Sacramento-San Joaquin River system. As the fish grow, the diet becomes more diverse and includes several benthic invertebrates and seasonally abundant food items, such as fish eggs or fry.

Daily food ration needs for wild fish are unknown. Under culture conditions promoting maximum growth, young sturgeon are fed 20 to 30 percent of their body weight per day until they reach 3 grams and 15 percent of their body weight until they reach 15 grams (Doroshov et al., 1983). Sturgeon that weigh more than 28 grams are fed 1 to 1.5 percent of their body weight per day.

Estuarine Residence. There is no defined age or size at which juvenile sturgeon from anadromous populations enter the estuarine environment (Binkowski and Doroshov, 1985). During high-flow years, larvae and young-of-the-year sturgeon have been found in Suisun Bay (Stevens and Miller, 1970).

Adult and subadult sturgeon inhabit estuarine areas year-round. Adult sturgeon are found in Suisun, San Pablo, and San Francisco bays and in the Delta (Miller, 1972b; Shirley, 1987; Kohlhorst et al., 1991). Distribution in the Delta is thought to depend primarily on river flow and consequent salinity regimes. The center of the population is up river during low river flow years and down river during high river flow years.

Shirley (1987) studied adult sturgeon in the estuary and found differences in age structure of fish from different regions of the estuary. Relatively young fish were captured from Suisun and Grizzly bays and near Candlestick Park in San Francisco Bay; older fish were caught in Carquinez Strait, in San Pablo Bay, and near Tiburon. Sturgeon captured near Tiburon, closer to the mouth of San Francisco Bay, had a significantly older age structure. Only four sturgeon older than 20 years were caught at locations other than Tiburon. At Tiburon, 34 fish were more than 20 years old, with the oldest fish estimated to be 27 years old. Age structures of all groups peaked in the age distribution between 11 and 15 years.

Sturgeon continue to be opportunistic feeders as adults. As they grow larger, their diet diversifies. McKechnie and Fenner (1971) found that adult sturgeon caught in San Pablo and Suisun bays feed primarily on benthic invertebrates, including clams, barnacles, crab, and shrimp. Seasonally, herring eggs and small fish, such as striped bass, flounder, goby, and herring, are important prey items. Although numerous in the estuary, worms, such as polycheates and nematodes, were seldom consumed.

Factors Affecting Abundance

Upstream Migration. As discussed earlier, only a fraction of the adult sturgeon population migrates upstream to spawn each year. The timing and extent of upstream migration are probably triggered by both biotic (i.e., sexual maturation) and abiotic (i.e., temperature, flow, and photoperiod) factors.

Flow. It is unknown whether river flow affects the timing or success of upstream migration. River flow velocity is thought to trigger spawning in female sturgeon, but its relation to the onset of migrational behavior is unknown.

Water Temperature. Little is known of the effects of water temperature on upstream migration of white sturgeon in the Sacramento-San Joaquin River system. Water temperature and photoperiod could promote the final stages of egg maturation and start upstream migration. Chapman (1989) found that temperature did affect sperm production and hypothesized that it also likely affected egg production. Although it has not been shown in the literature for the Sacramento-San Joaquin River system, a threshold temperature could initiate upstream migration and spawning in some populations. Haynes et al. (1978) found that sturgeon migrations in the

Columbia River occur only at temperatures above 55 degrees Fahrenheit. However, sturgeon in the Sacramento River migrate at temperatures as low as 46 degrees Fahrenheit (Kohlhorst, 1976).

Passage. Little information is available concerning the abilities of white sturgeon to negotiate upstream passage barriers. A recent literature search failed to locate information regarding cruising, sustaining, and darting speeds for white sturgeon (Jones & Stokes Associates, 1992). However, sturgeon do spawn in relatively swift water; water column velocities as high as 10 feet per second have been measured in areas where sampling has determined the presence of sturgeon eggs (Parsley, pers. comm.).

Sturgeon are bottom-oriented fish with limited jumping abilities and have little success passing barriers along the channel bottom. Warren and Beckman (1991) reported that modified fish ladders in the Columbia River that provided orifices through the weirs at the ladder floor increased passage of white sturgeon over several Columbia River dams.

Sport Fishing. Annual exploitation rates (e.g., sport harvest rate) of white sturgeon in the Sacramento-San Joaquin River system have increased dramatically since the early 1980s due to increased popularity of the fishery, the discovery of appropriate bait, and the use of more sophisticated means of locating and landing sturgeon. In the 1980s, the exploitation rate increased by 40 percent (Kohlhorst et al., 1991). Patterns in the size of landed sturgeon indicated that recruitment of fish to harvestable size declined during this period (PSMFC, 1992). However, since new size limits were imposed in 1990, the harvest rate has been dramatically reduced.

Spawning. The following sections describe physical factors that could affect spawning success after sturgeon have arrived at the spawning grounds. Little information relating environmental conditions to the initiation or success of spawning in sturgeon is available.

Flow. Few data relate flow with sturgeon spawning habitat or success. White sturgeon eggs have been found in water column velocities as high as 10 feet per second (Parsley, pers. comm.). Velocity criteria for white sturgeon spawning habitats will soon be published in a habitat suitability index model being developed by the Service (Parsley, pers. comm.). Some preliminary data suggest that flow velocity could be a factor that triggers spawning in female sturgeon (Schaffter, 1990). River flow acts to disperse eggs and prevent clumping of the adhesive eggs. Significant relationships between year-class strength and freshwater outflow in spring are discussed in the Juvenile Rearing and Emigration section of this chapter.

Water Temperature. Chapman (1989) found that temperature affects the rate and timing of sperm production and hypothesized that it also likely affected egg production. Sturgeon in the Sacramento-San Joaquin River system spawn within temperature ranges of 46 to 64 degrees Fahrenheit, with most fish spawning when water temperatures are 58 degrees Fahrenheit (Kohlhorst, 1976); however, Kohlhorst did not note a temperature effect on the intensity of spawning or a temperature threshold for spawning.

Water Quality. The influence of water pollution on sturgeon is not well documented. Sturgeon tissue has been found to contain polychlorinated biphenyls (PCBs), organochlorides, mercury, selenium, and dioxins (PSMFC, 1992). Egg tissues can also contain toxins, which

could reduce reproductive potential (Doroshov, 1990). Turbidity can affect the adhesiveness of eggs, which could displace eggs to less than optimum habitats during incubation.

Spawning Gravels. Because of the adhesive nature of sturgeon eggs, areas of silt-free gravel appear to be required for successful sturgeon spawning. The nature of spawning site selection and the availability of clean gravel spawning areas with sufficient flow are unknown.

Egg Incubation and Hatching. No published data relate environmental conditions to egg incubation and hatching in the wild. The following data are from laboratory studies.

Flow. River flow is important to maintain oxygen levels and remove waste products at the egg surface. After sturgeon larvae hatch, the currents act to disperse the larvae downstream from the spawning grounds.

Water Temperature. Several authors have reported the effects of temperature on incubation and early development of sturgeon (Wang, 1984; Wang et al., 1985; Doroshov, 1985; Conte et al., 1988). Wang (1984) found a strong inverse correlation between temperature (52 to 68 degrees Fahrenheit) and incubation period (-0.9567), and temperatures and yolk depletion (-0.9943) in the temperature range of normal development (Figure II-51). Egg incubation can last four to 14 days post-fertilization, while yolk depletion can occur 15 to 30 days post-fertilization. Optimum temperatures for white sturgeon incubation and larval development range between 52 and 63 degrees Fahrenheit (Wang et al., 1987). Higher mortality and premature hatching occurs at 64 to 68 degrees Fahrenheit. Temperatures of 73 to 79 degrees Fahrenheit are lethal to sturgeon embryos (Wang, 1984). A lower temperature limit has not been defined; however, Wang et al. (1987) suggest that it might range between 43 and 46 degrees Fahrenheit.

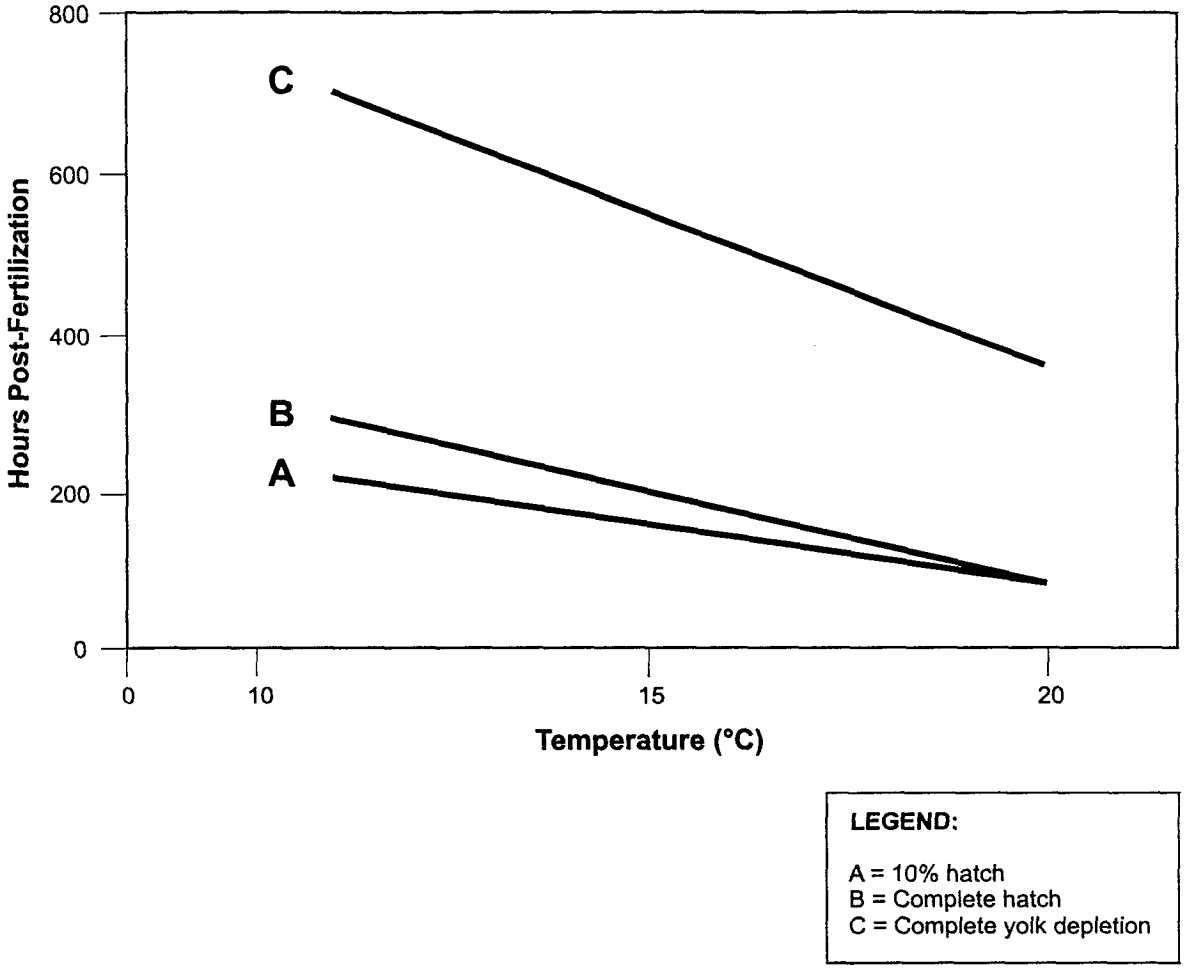
Based on Wang's (1984) correlations, incubation and yolk depletion at peak spawning season temperatures (58 degrees Fahrenheit) (Kohlhorst, 1976) will be approximately nine days and 24 days post-fertilization, respectively.

Water Quality. Effects of most water quality parameters on incubation and emergence are not well documented.

Juvenile Rearing and Emigration

Flow. Kohlhorst et al. (1991) found a significant positive correlation between a year-class strength index and Sacramento River outflow in spring and early summer (April to July). During years with high spring and early summer flow conditions (1982 and 1983), white sturgeon production (as evidenced by abundance of young-of-the-year) was greater than years with lower outflows (between 1975 and 1985) (Figure II-52). SWP data from 1968 to 1987 also indicate that the number of young sturgeon salvaged per acre-foot of water exported was associated with April-May Delta outflows (DWR, 1990) (Figure II-53). The strength of these relationships, however, is primarily driven by two very wet springs in 1982 and 1983.

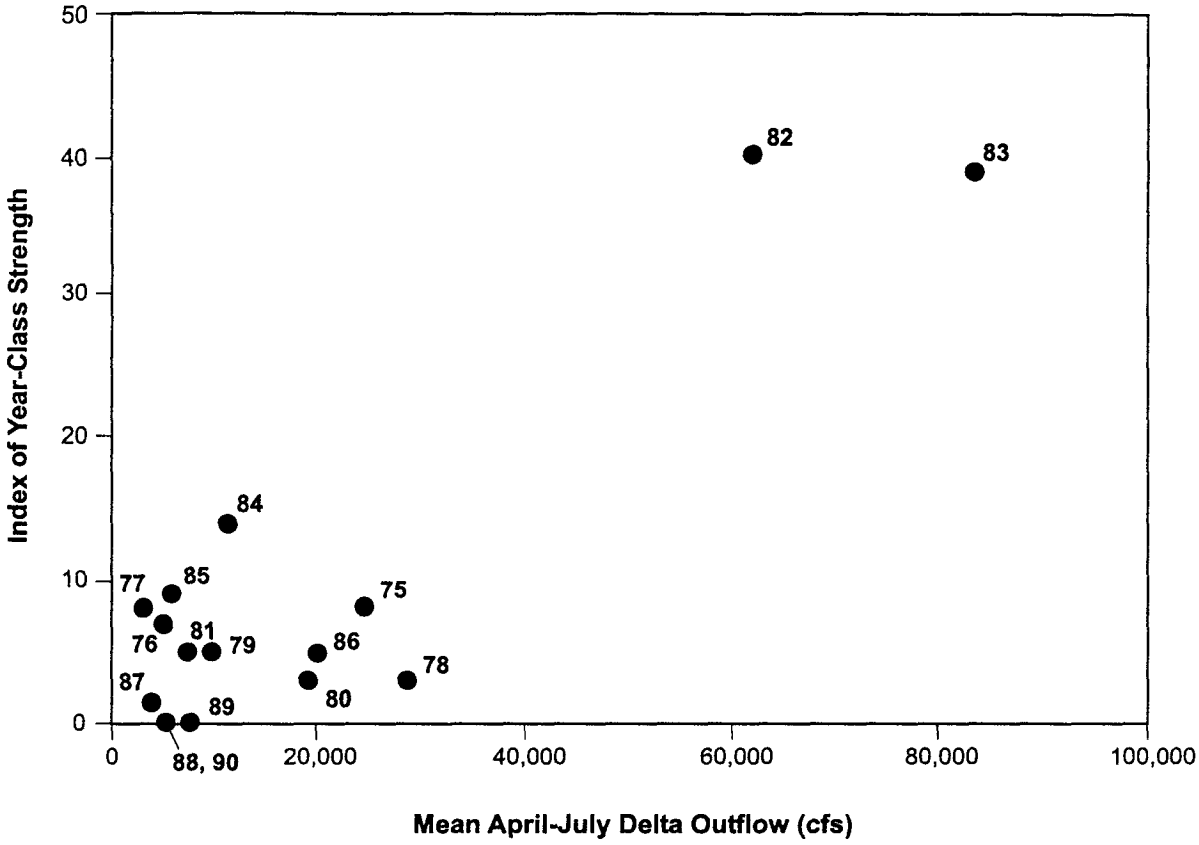
The mechanisms responsible for this increased recruitment are not well defined. Likely contributing factors include the dispersion of larvae by the currents to more productive or less



SOURCE:
Wang, 1984.

FIGURE II-51

RELATIONSHIP BETWEEN WATER TEMPERATURE AND EARLY LIFE CYCLE DEVELOPMENT OF WHITE STURGEON

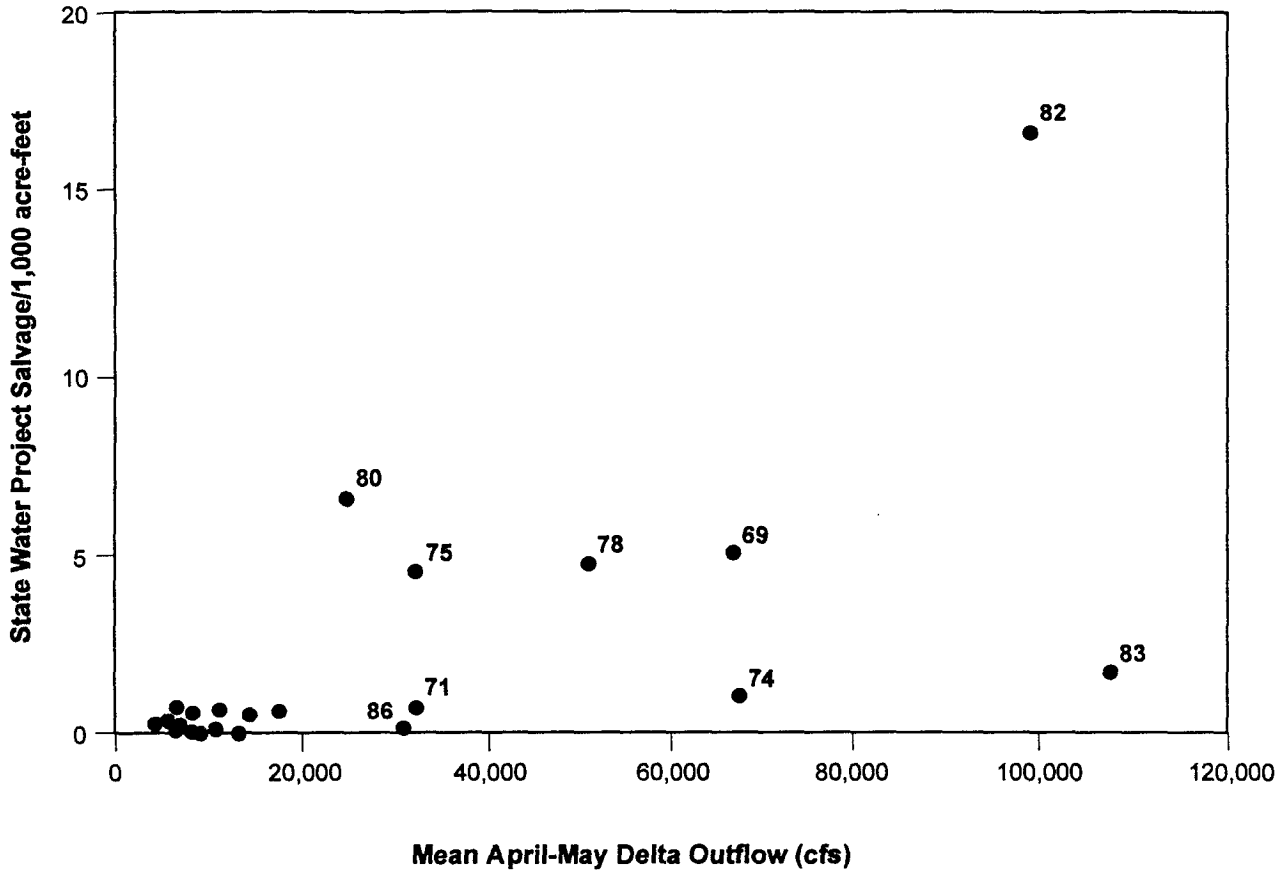


NOTE:
Year-class index determined from trawl catches.

SOURCE:
Kohlhorst et al., 1991.

FIGURE II-52

RELATIONSHIP BETWEEN WHITE STURGEON YEAR-CLASS INDEX AND MEAN DELTA OUTFLOW FOR APRIL THROUGH JULY (1975-1990)



NOTE:
Year-class index determined from salvage at State Water Project Skinner Fish Facilities.

SOURCE:
DWR, 1990.

FIGURE II-53

RELATIONSHIP BETWEEN WHITE STURGEON YEAR-CLASS INDEX AND MEAN DELTA OUTFLOW FOR APRIL THROUGH MAY (1968-1987)

utilized habitats, resulting in reduced competition; reduced vulnerability to diversions; and increased nutrient loading to the nursery environment from increased flows.

Water Temperature. Water temperature can affect the growth rate and health of juvenile sturgeon. Under laboratory conditions, maximum growth occurs at rearing temperatures of 68 degrees Fahrenheit, but rearing at lower temperatures (61 to 65 degrees Fahrenheit) reduces the incidence of disease (Cech et al., 1984; Conte et al., 1988).

Water Quality. Water quality effects on juvenile sturgeon are not well documented.

Salinity. Juvenile sturgeon are known to be sensitive to salinity (McEnroe and Cech, 1985; Brannon et al., 1985; Brewer, 1987). Young Sacramento River white sturgeon had low survival in 10 parts per thousand salinity (McEnroe and Cech, 1985). Salinity tolerance did not appear to change with age or size in larval and juvenile Columbia River white sturgeon (1 to 83 days post-hatching) (Brannon et al., 1985). Larvae and juveniles could not tolerate (less than 50 percent mortality) direct salinity increases to 11 parts per thousand, and no fish survived transfer to aquaria with 16 parts per thousand. Those fish that survived 11 parts per thousand salinity were sluggish in response. Acclimation of larger fish improved tolerance to 15 parts per thousand. Brannon et al. (1985) also demonstrated that sturgeon larvae and fry can respond to salinity gradients by avoiding higher salinity areas in aquaria.

Food Production. No data are published on the effects of reduced food production on the abundance of wild juvenile sturgeon. Because sturgeon primarily feed on benthic organisms, reduced populations of these organisms will likely have the most detrimental effect on sturgeon growth and survival.

Diversions. Larval and juvenile sturgeon are weak swimmers; they are transported downstream primarily by the currents. Consequently, larval and juvenile sturgeon are highly susceptible to entrainment and impingement on fish screens associated with water diversion projects in the Sacramento River and Delta. The magnitude of such losses and the effects on population abundance are unknown. Fish screen designs at diversions are important to pass juvenile sturgeon successfully at diversions and prevent impingement of sturgeon on the screens. Based on the work of Reading (1982), Ward (pers. comm.) suggested that maximum approach velocities of approximately 0.06 foot per second will be needed to protect juvenile sturgeon at diversions.

Predation. No data have been published on the effects of predators on juvenile sturgeon in the Sacramento River. Mass hatching in the dark, hiding behavior during yolk absorption, and avoidance of light are all adaptations to minimize predation. As sturgeon grow, they become less likely to be killed by predators. Adult sturgeon are not known to have any natural predators in freshwater. Benthic feeding fish are most likely to consume sturgeon eggs and larvae. It is conceivable that dramatic increases in these predators could adversely affect sturgeon recruitment.

Adult Estuarine Residence. Annual exploitation rates (e.g., sport harvest rate) have increased dramatically due to increased popularity of the fishery, the discovery of appropriate bait, and the use of more sophisticated means of locating and landing sturgeon. In the 1980s, the

exploitation rate increased by 40 percent (Kohlhorst et al., 1991). Most sturgeon are landed by private and charter boats rather than from shore.

Patterns in the size of landed sturgeon indicated that recruitment of fish to harvestable size is declining (PSMFC, 1992). Stricter angling regulations have been employed to provide additional protection to adult sturgeon (discussed in the following paragraphs).

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

As a means of increasing recruitment, stricter size limitations recently have been imposed on sport anglers. In 1990, the minimum size was increased to 42 inches, and for the first time, a maximum size limitation was imposed (72 inches). The minimum size limit was increased in 2-inch increments from 42 inches in 1990 to 46 inches in 1993. Some seasonal angling restrictions were also imposed in central San Francisco Bay to protect concentrations of sturgeon feeding on herring roe. These new sport fishing regulations were designed to meet the following management goals for the white sturgeon in the Sacramento-San Joaquin River system:

- reduce sturgeon harvest to 50 percent of that observed during the 1980s by March 1993;
- protect large fecund females from sport harvest;
- maximize sport angling opportunities consistent with the management plan; and
- maintain equal access to the resource for all sport anglers.

Ongoing monitoring of white sturgeon populations is being conducted by DFG (PSMFC, 1992). Recent projects include:

- tag recapture programs to estimate abundance, mortality rates, and movement patterns;
- trapping of juvenile sturgeon to determine abundance and year-class strength on a monthly basis; and
- identification of spawning habitats, spawning migrations, and specific spawning sites in the Sacramento-San Joaquin River system.

Enhancement efforts include private or academic aquaculture programs (PSMFC, 1992). Regional DFG biologists are attempting to reestablish white sturgeon in Shasta Lake.

Several white sturgeon aquaculture programs are in progress at the University of California, Davis, including the study of:

- nutrition
- reproductive endocrinology
- domestic broodstock development and spawning
- hatchery technology
- population genetics

- pathology and virology
- molecular biology
- environmental physiology
- age, size, and population structure (PSMFC, 1992)

At least two commercial aquaculture ventures are currently in operation.

GREEN STURGEON

Little is known about green sturgeon (*Acipenser medirostris*) stock abundance and distribution, life history, or factors affecting abundance. No monitoring, enhancement, and habitat improvement actions are ongoing. The brief available information is summarized primarily from Moyle (1976) and Kohlhorst et al. (1991).

Green sturgeon are found in the lower reaches of large rivers from the Sacramento-San Joaquin River Basin to the Eel, Mad, Klamath, and Smith rivers. Green sturgeon seem to be the most common sturgeon in the Klamath and Trinity rivers at present (Moyle, 1976). Green sturgeon are a minor component of the sturgeon populations in the Central Valley; ratios of adult green sturgeon to white sturgeon during tagging studies in the Delta have ranged from 1:39 to 1:164 (Mills and Fisher, 1993) (Table II-8).

Green sturgeon spend less time in estuaries and freshwater than do white sturgeon. Green sturgeon also make extensive ocean migrations; consequently, most recoveries of individuals tagged in San Pablo Bay have come from the ocean and from rivers and estuaries in Oregon and Washington. Juvenile fish have been collected in the Sacramento River, near Hamilton City, and in the Delta and San Francisco Bay. Adults and juveniles have been observed near RBDD in late winter and early spring. Juveniles inhabit the estuary until they are approximately four to six years old, when they migrate to the ocean. (Kohlhorst et al., 1991.)

The diet of adult green sturgeon appears to be similar to that of white sturgeon: bottom invertebrates and small fish (Ganssle, 1966). Juveniles in the Delta feed on opossum shrimp and amphipods, such as Corophium (Radtke, 1966). Little information is available about green sturgeon age and growth; they seldom exceed 4 feet in length from the Delta (Skinner, 1962; Moyle, 1976).

RECENT CONDITIONS FOR STRIPED BASS

STOCK ABUNDANCE AND DISTRIBUTION

Historical Population Trends (1850-1966)

Striped bass (*Morone saxatilis*) are native to the east coast of the United States. Juvenile striped bass were taken from rivers in New Jersey and introduced to California waters; approximately 130 juvenile fish were released in Carquinez Strait in 1879, and another 300 were released in Suisun Bay in 1882 (California Bureau of Marine Fisheries, 1949; Skinner, 1962). Successful

reproduction was observed prior to 1882, and the population quickly multiplied to several million adult bass.

A few of the fish planted in 1879 were reportedly caught in 1880, and striped bass weighing more than 16 pounds were caught in 1883 and 1884 (California Bureau of Marine Fisheries, 1949; Skinner, 1962). By 1888, striped bass supported a significant fishery in San Francisco Bay, and several thousand fish were available in local fish markets. A minimum size limit of 8 pounds, the first fishing regulation for striped bass in California, was enacted in 1890. State regulations set a minimum size limit of 3 pounds in 1897.

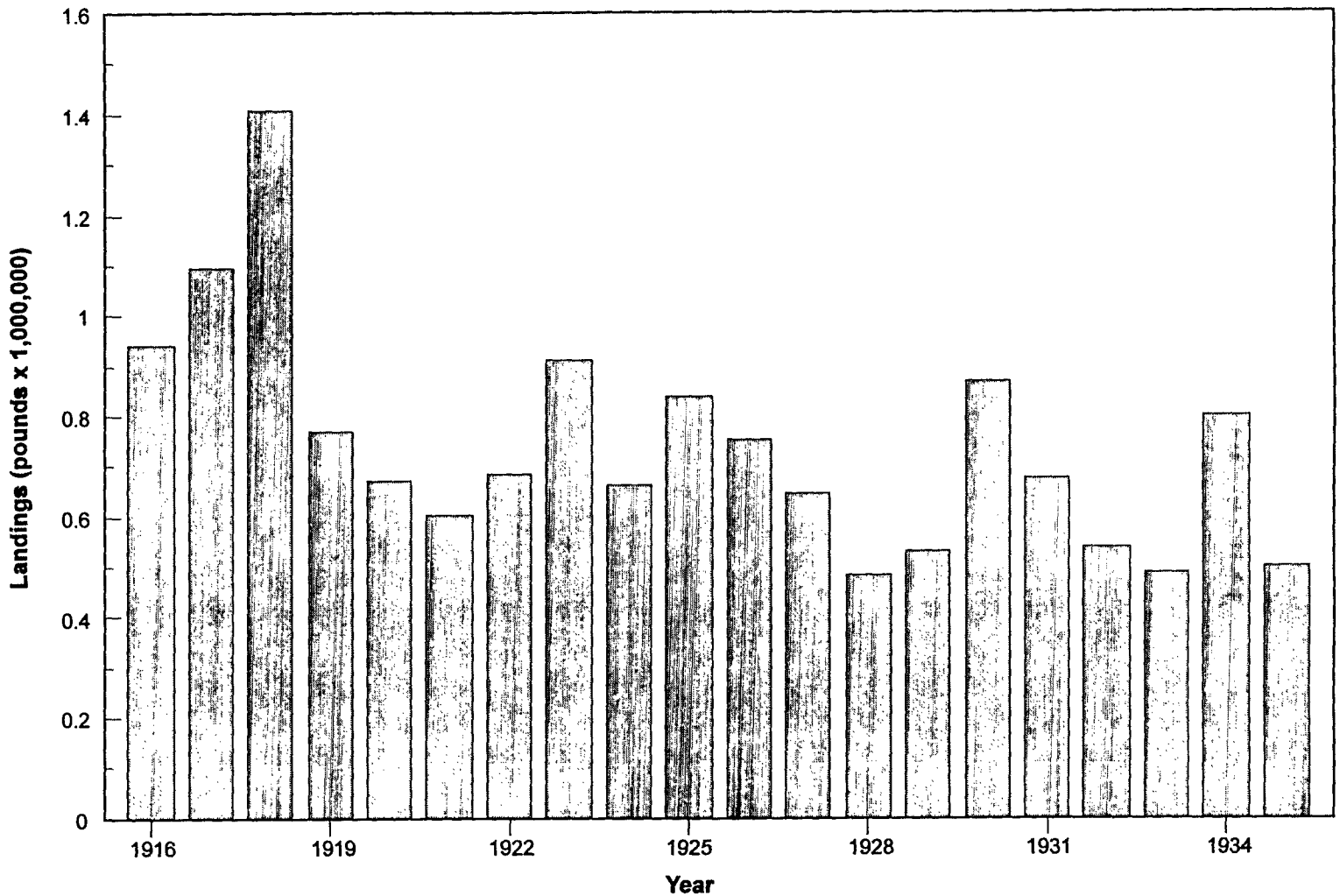
The 1899 commercial catch was reported as 1,234,000 pounds (Skinner, 1962), and commercial landings between 1916 and 1935 ranged from 0.5 million to 1.5 million pounds (Figure II-54). Sport fishing for striped bass became increasingly popular after 1895, leading to more restrictive commercial fishing regulations. Commercial fishing for striped bass with nets was prohibited in 1931, and all commercial striped bass fishing was prohibited after 1935.

Beginning in 1936, the striped bass fishery was reserved exclusively for sport anglers. Annual striped bass landings by the sport fishery were reported to be much larger than commercial striped bass landings had ever been (California Bureau of Marine Fisheries, 1949). By 1955, more than 200,000 anglers participated in the fishery, catching more than 1 million striped bass annually with an aggregate weight of approximately 4 million pounds (Skinner, 1962).

Analysis of sport catch records and other data showed a decline in the fishery after 1944 and a severely depleted adult striped bass population by 1970 (Skinner, 1962; DFG, 1989). Data from the sport fishery and mark-recapture studies indicate that the population declined from approximately 3 million bass in the early 1960s to a population level of approximately 1.7 million by the late 1960s.

Charter boat records provide the only continuous records on the striped bass fishery from 1938 to 1982, although catch was influenced by size limits, creel limits, and fishing methods. Catch-per-angler-day was greatest during the early years of the charter boat fishery and decreased over time (Figure II-55). The reduction in the catch-per-angler-day could indicate decreasing striped bass population abundance; however, changes in fishing regulations and sport fishing efforts affect catch-per-angler-day statistics.

Factors contributing to increased mortality prior to 1967 include fishing, entrainment in diversions, exposure to toxic materials, and habitat loss. Sport fishing annually removed 14 to 37 percent of the striped bass population longer than 16 inches. Incidental catch in net fisheries targeting other species could have caused annual mortality approaching 50,000 adult striped bass before net fisheries were prohibited in 1957. Entrainment in the Contra Costa Steam Plant (PG&E) and the CVP's Tracy Pumping Plant diversions could have reduced the juvenile striped bass population by more than 20 percent each year. Salvage operations at both facilities greatly reduced the number of fish destroyed, but losses continued to occur after 1957.



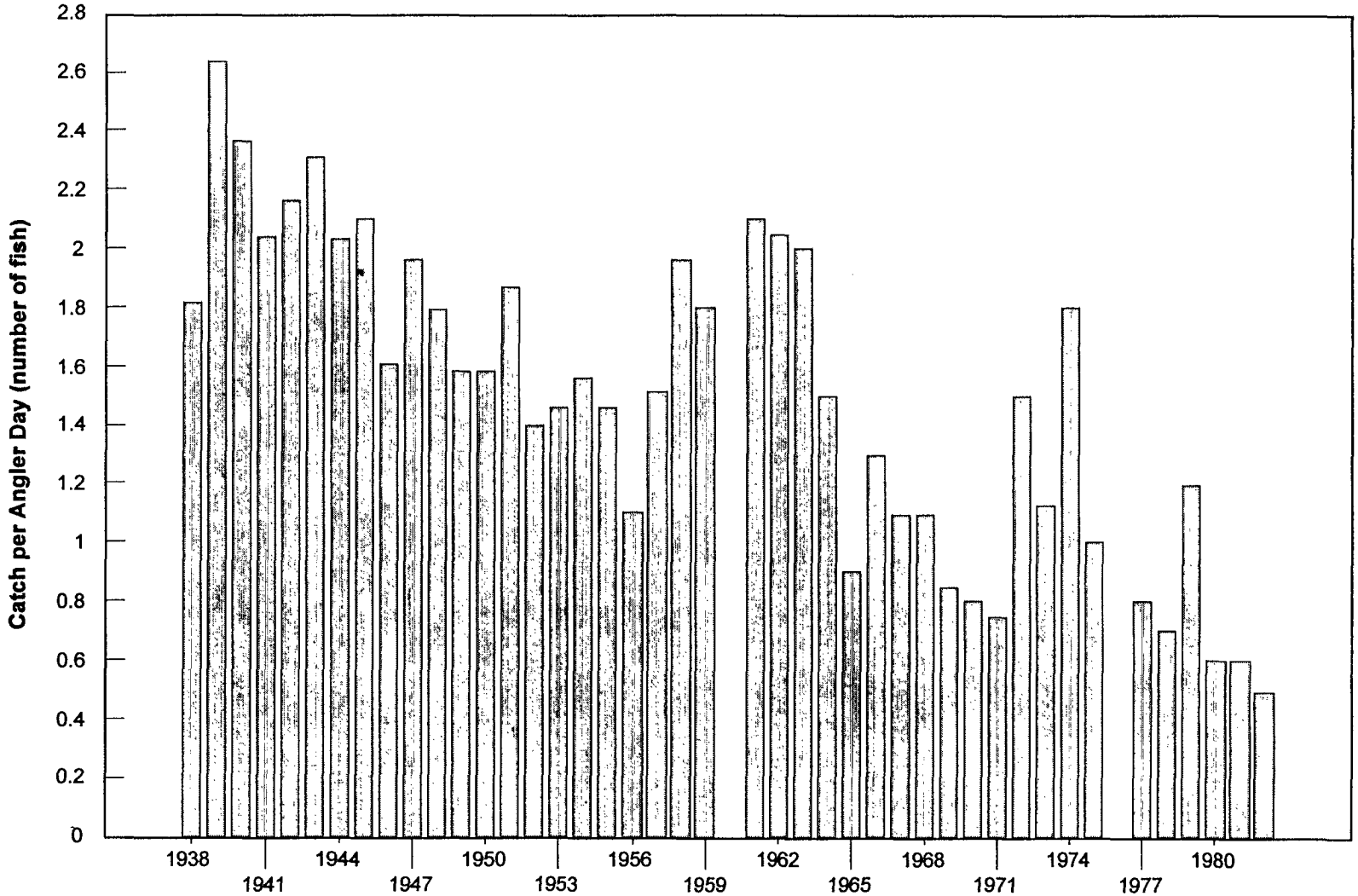
SOURCE:
California Bureau of Marine Fisheries, 1949.

FIGURE II-54

COMMERCIAL LANDINGS OF STRIPED BASS (1916-1935)

C-081472
C—081472

C-081473



SOURCES:
Skinner, 1962; State Water Contractors, 1987.

FIGURE II-55

CATCH OF STRIPED BASS PER ANGLER DAY FOR CHARTER BOATS (1938-1982)

C—081473

Anecdotal information indicates pollution in the Napa River and San Francisco Bay by tannery, chemical company, and garage discharges could have resulted in substantial mortality of striped bass as early as 1924.

Between 1860 and 1959, nearly half of the estimated 570 square miles of marsh and tidal habitat were filled and leveed off (DFG, 1989). Sloughs that formerly afforded good fishing and habitat were no longer accessible to striped bass. Diking and filling not only restricted striped bass habitat but also reduced tidal mixing (i.e., potential for reduced dilution of toxic materials) and overall estuary productivity.

Current Population Trends (1967-1991)

Although the striped bass population had declined from historic levels by 1967, the period over which the decline occurred is unclear (Turner, 1987). A more precipitous decline was documented after 1967 and continues to the present (Figure II-56). The average adult population size in the late 1960s and early 1970s of approximately 1.7 million striped bass declined to an average of less than 1 million in the 1980s (DFG, 1989). The average adult striped bass population size for the 1967 to 1991 period was approximately 1.25 million fish. A record low adult striped bass population of 624,000 fish was estimated in 1992, including approximately 90,000 bass that were raised in hatcheries and stocked in the Delta and Bay (DFG, 1992a).

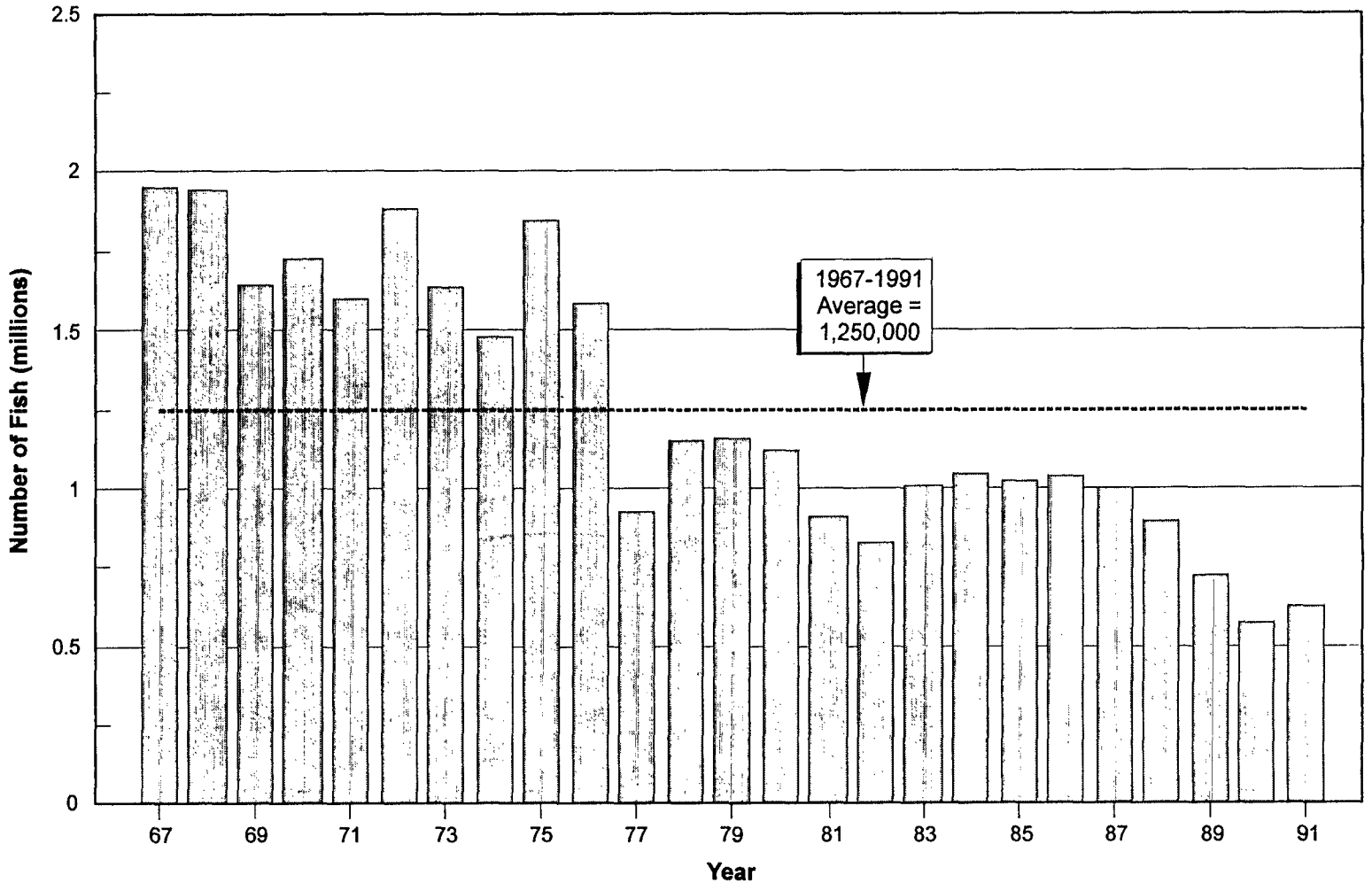
From 1981 to 1990, more than 10 million juvenile striped bass were raised in hatcheries and released in the Delta and Bay to supplement the wild population (Delisle, pers. comm.). The hatchery contribution to the total adult striped bass population increased from less than 1 percent in 1984 to more than 12 percent in 1991. The greater percentage contribution to the wild population is attributable to increased annual stocking of hatchery fish and to the declining population of wild fish.

The annual sport catch in the late 1980s was less than 150,000 fish, compared to more than 300,000 fish landed by anglers in the early 1970s. After 1967, harvest rates have ranged from 10 to 24 percent of the adult striped bass population. (DFG, 1992a.)

The adult population decline primarily reflects a decline in the number of new fish reaching legal size. The youngest and most numerous component of the adult striped bass population (i.e., three-year-old fish) had declined to record lows by 1990 (DFG, 1992a).

The summer tow net survey was initiated in 1959 to provide an index of young-of-the-year abundance (i.e., the 38-millimeter index). The 38-millimeter index declined coincidentally with the decline in adult abundance after the mid-1960s (Figure II-57). The peak 38-millimeter index was 117 in 1965; the lowest was 4.3 in 1991 (DFG, 1992a). The index averaged 38 for the 1967 to 1991 period. The fall mid-water trawl surveys initiated in 1967 also provided an index of young-of-the-year abundance during September-December (Figure II-57).

Reduced populations of larvae larger than 8 millimeters have contributed to the decline in the 38-millimeter index (DFG, 1987). Low abundance of 38-millimeter juveniles was preceded by low abundance of 8-millimeter juveniles or larger larvae (Figure II-58). Although low larval



SOURCE:
Mills and Fisher, 1993.

FIGURE II-56

ESTIMATES OF ADULT STRIPED BASS ANNUAL ABUNDANCE IN THE CENTRAL VALLEY (1967-1991)

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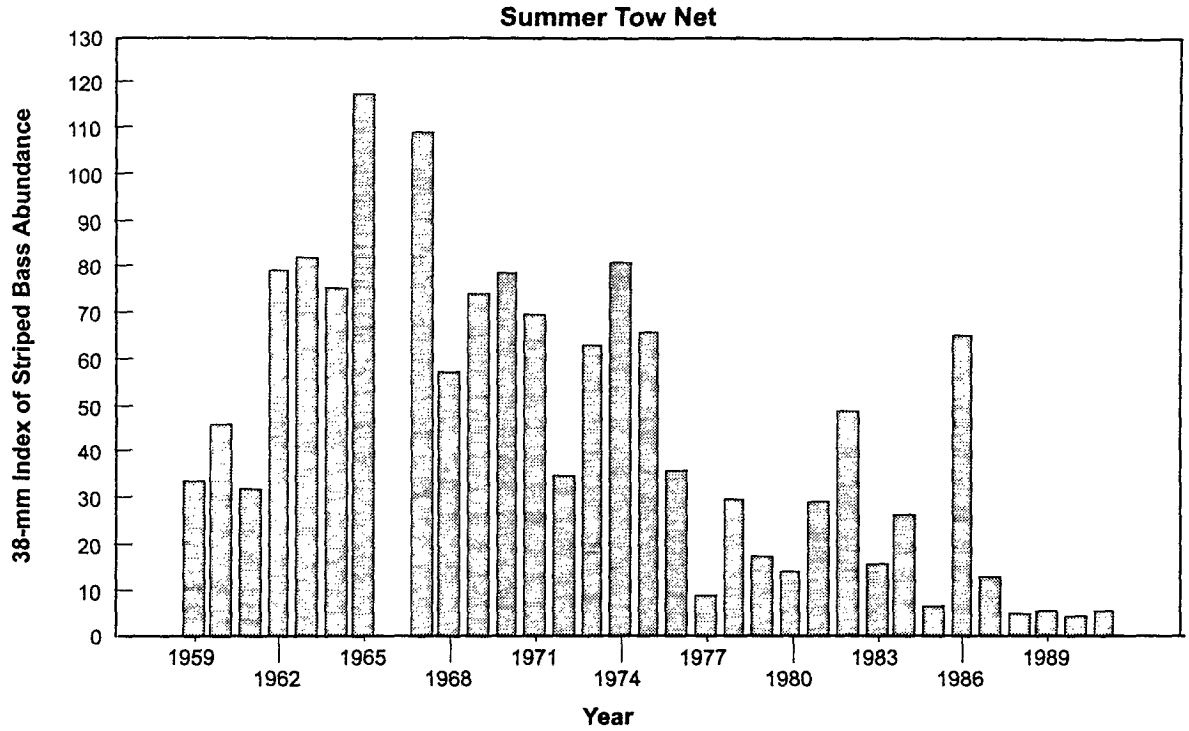
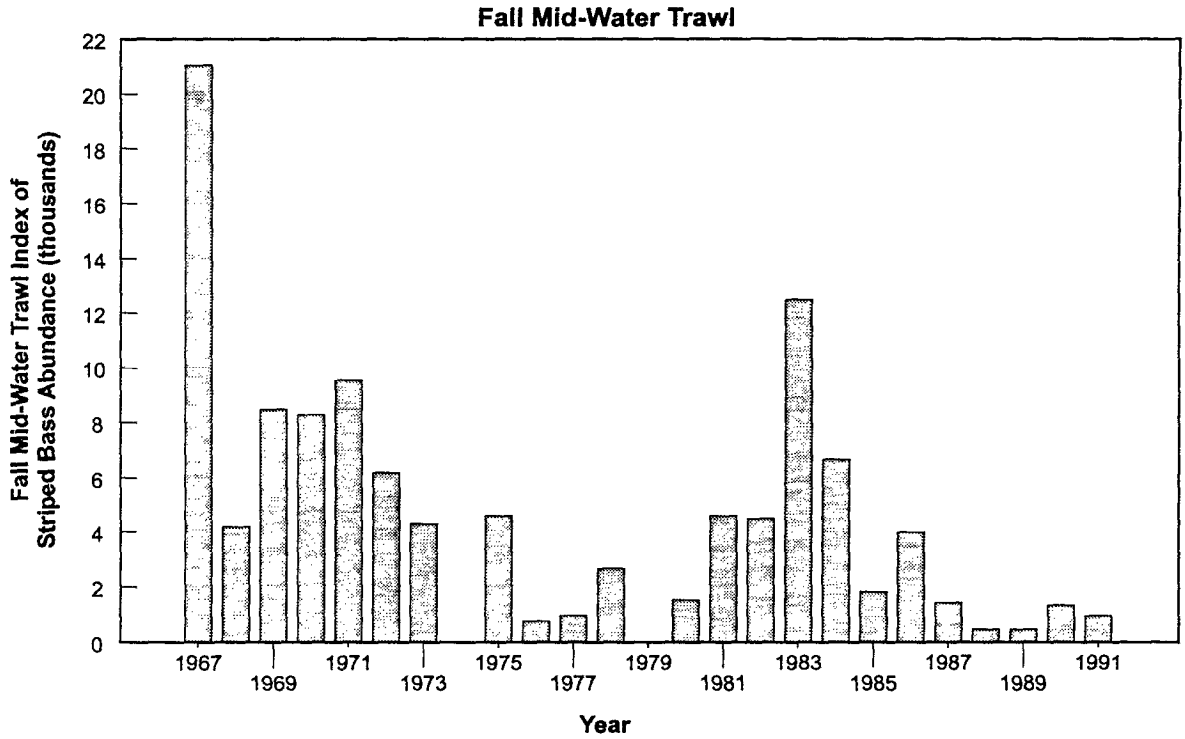
C—081475



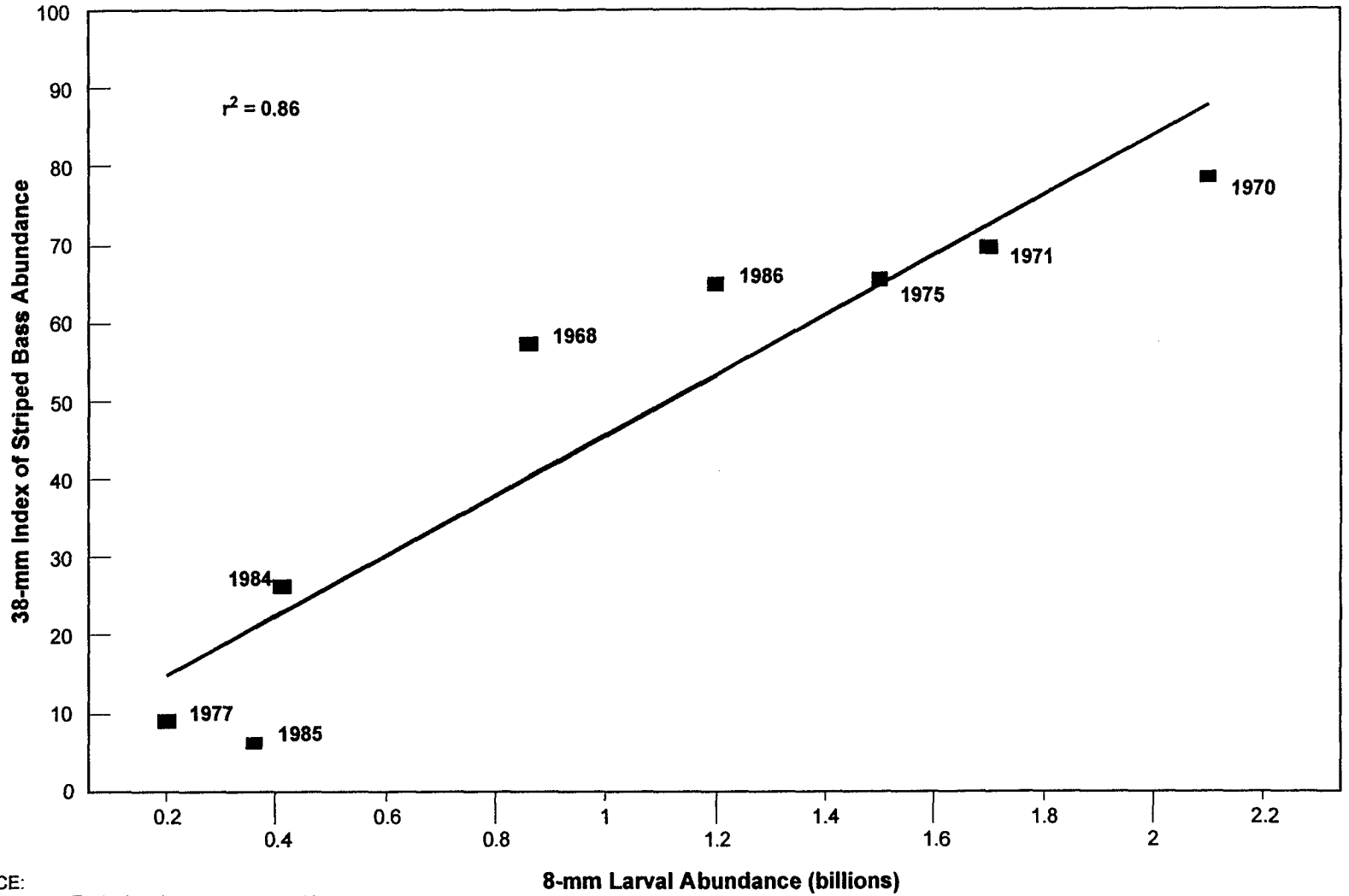


SOURCE:
Herrgesell, 1993.

FIGURE II-57

YOUNG-OF-YEAR STRIPED BASS ANNUAL ABUNDANCE INDICES IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY (1959-1991) CALCULATED FROM TWO SURVEY PERIODS

C-081477



SOURCE:
Interagency Ecological Studies Program, 1987.

FIGURE II-58

RELATIONSHIP BETWEEN 8-MM LARVAL STRIPED BASS ABUNDANCE INDEX AND 38-MM JUVENILE STRIPED BASS ABUNDANCE INDEX

C—081477

abundance could indicate that year-class abundance will remain low, high 38-millimeter indices likely reflect increased survival during and after the larval period.

LIFE HISTORY

Striped bass inhabit fresh and ocean waters (Figure II-59). They require tidal or riverine habitat for spawning, with turbulence and currents sufficient to keep the eggs suspended off the bottom (Moyle, 1976). Estuarine habitat with high invertebrate densities is needed to support larval and early juvenile bass. Adult bass survive and grow best in water bodies supporting a large prey base (i.e., large populations of forage fishes). The Sacramento and San Joaquin rivers, the Delta, Suisun Bay, San Francisco Bay, and the Pacific Ocean provided conditions that sustained the striped bass population for more than 100 years.

Adults

Striped bass are considered adults at three years old (approximately 38 centimeters long) and could live for more than 30 years (Moyle, 1976). Most adult striped bass in the Sacramento-San Joaquin Delta estuary are between three and eight years old. Female striped bass grow faster than males, and most six-year-old females are the same size as seven-year-old males (Figure II-60) (Collins, 1981). Growth mainly occurs between May and November.

Food preference is primarily a function of prey availability, which is dependent on habitat, changes in species composition, season, and other factors. In general, adult striped bass feed on fish, including smaller striped bass. In the Delta, adult bass prey primarily on threadfin shad, American shad, and young striped bass (Stevens, 1966). Anchovies, chinook salmon, delta smelt, and mysids are seasonally eaten in the lower Delta and Suisun Bay (Thomas, 1967). In San Pablo and San Francisco bays, anchovies, bay shrimp (*Crangon* sp.), and shiner perch are the primary prey items. When striped bass inhabit rivers, juvenile chinook salmon and carp are key prey species.

Migrating Adults. Adult bass are found throughout the year in rivers (the Sacramento, San Joaquin, and Mokelumne rivers and their major tributaries), the Delta, San Francisco Bay, and the Pacific Ocean, but they show definitive migration patterns. In fall, adult striped bass migrate upstream to Suisun Bay and the Delta, where they overwinter (Chadwick, 1967; Mitchell, 1987). During spring, bass disperse throughout the Delta and into the tributary rivers to spawn. They migrate back to the Delta, Suisun Bay, and San Francisco Bay during summer. After the mid-1960s, however, most striped bass have inhabited Suisun Bay and the Delta during summer and fall; migration to San Francisco Bay and the Pacific Ocean has declined.

Spawning Adults. Male striped bass could be sexually mature at the end of their first year, but most reach sexual maturity at two to three years (Moyle, 1976). Sexual maturity occurs at a later age in females, usually after four to six years.

Striped bass always spawn in freshwater (DFG, 1987). They spawn in the Sacramento River between Sacramento and Colusa (including the Feather River below Marysville [Wang, 1986]) and in the San Joaquin River part of the Delta between Antioch and Venice Island (Figure II-61). Spawning has also been recorded in the lower San Joaquin River above the Delta (Turner, 1976).

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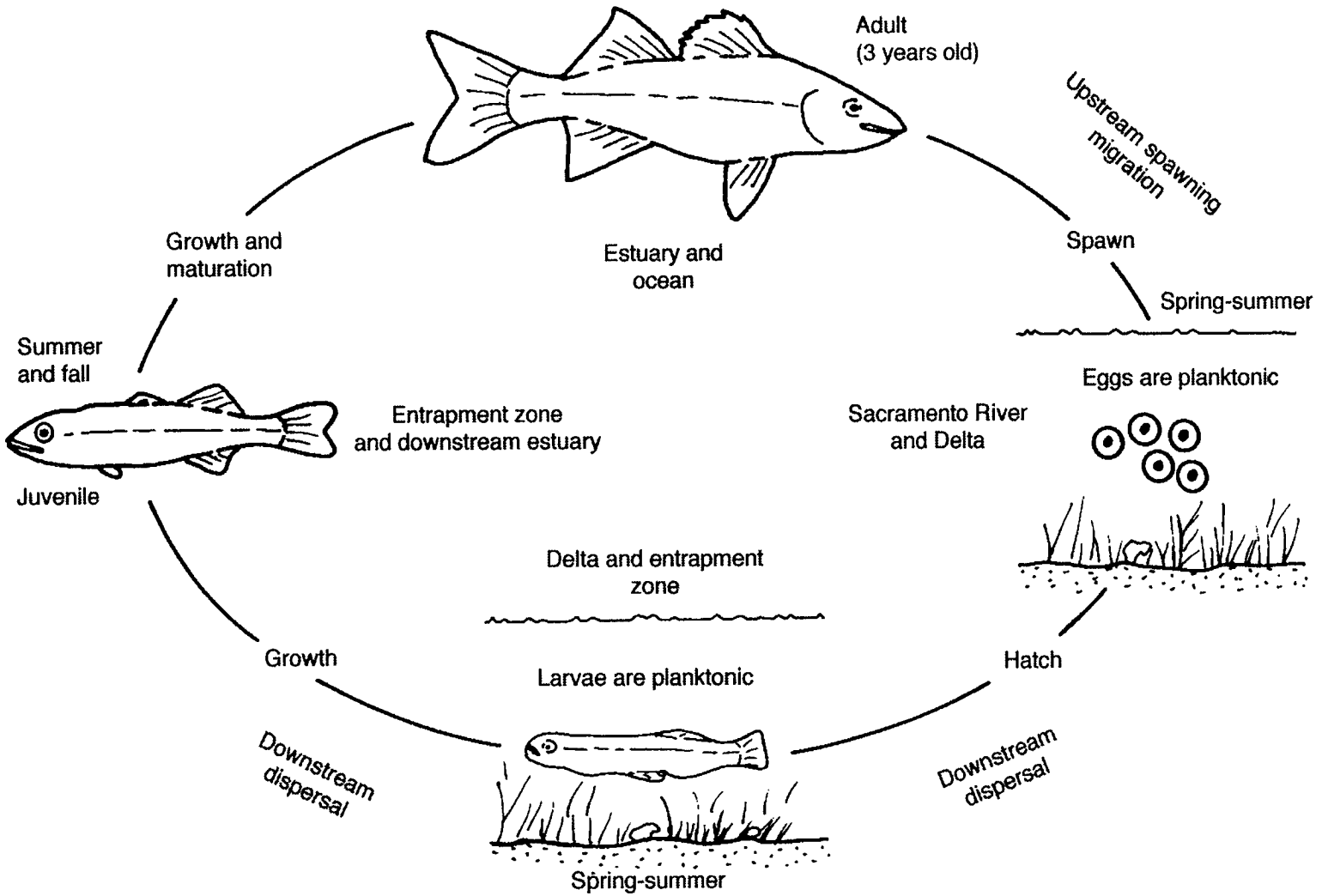
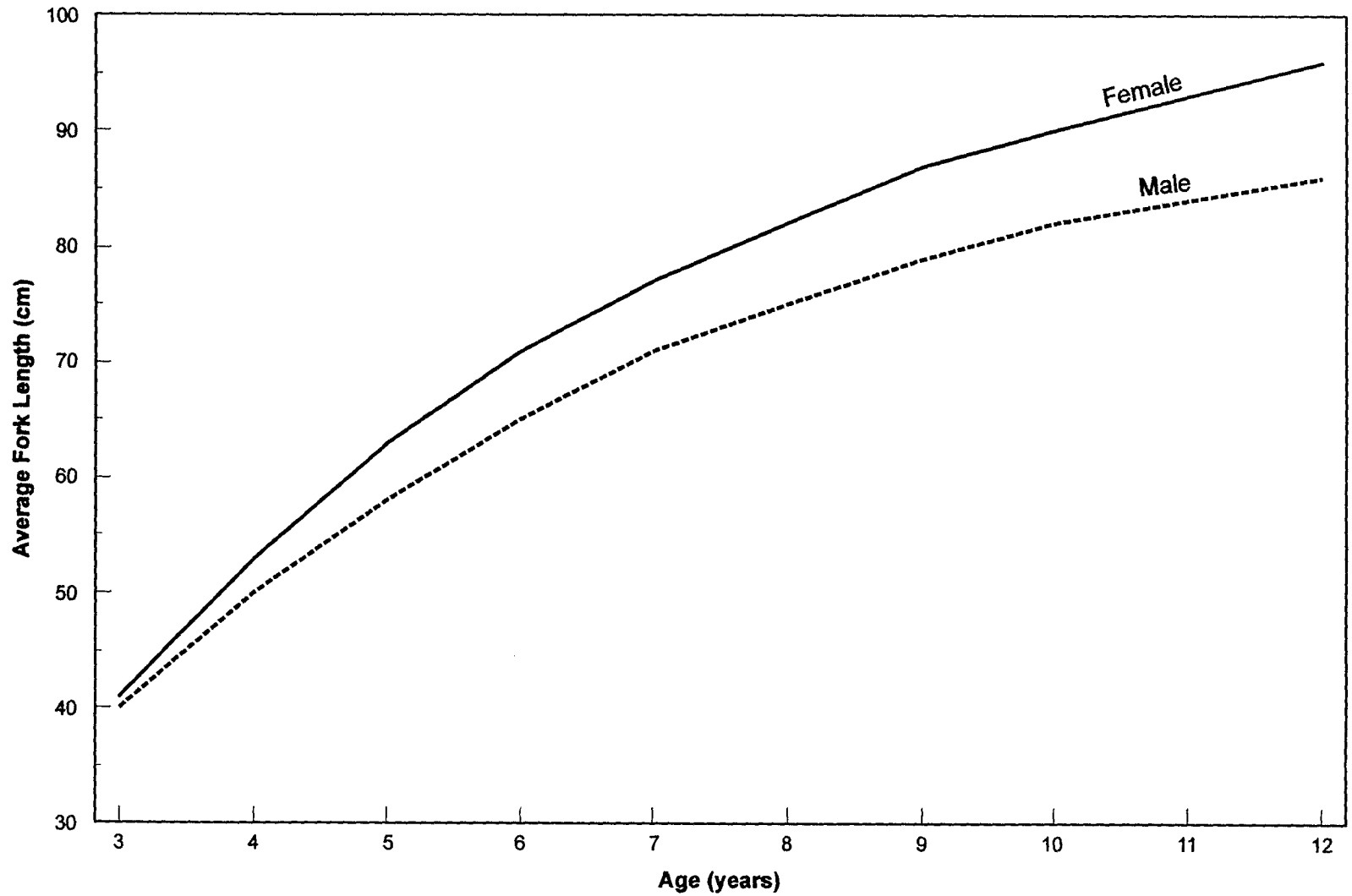


FIGURE II-59

LIFE HISTORY OF STRIPED BASS

C-081481

C-081481



SOURCE:
Collins, 1981.

FIGURE II-60

GROWTH OF ADULT STRIPED BASS IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY

C-081482
C—081482

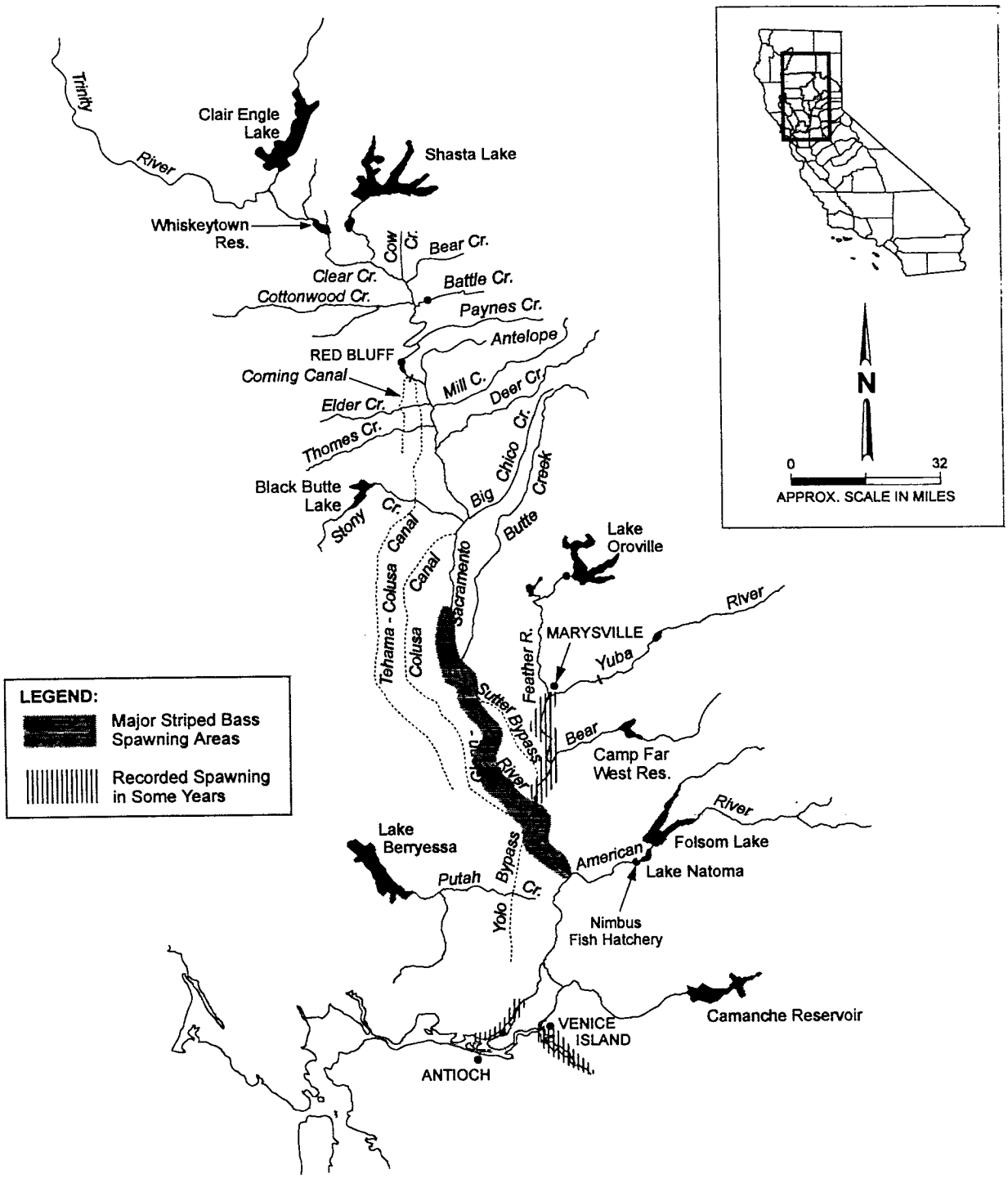


FIGURE II-61

STRIPED BASS SPAWNING AREAS IN THE SACRAMENTO-SAN JOAQUIN RIVER SYSTEM

Usually, approximately 60 percent of the spawning population uses the Sacramento River and 40 percent spawn in the Delta. The proportion spawning in each area varies annually, but 50 to 66 percent of the annual egg production is from the Sacramento River spawn. Spawning in the Sacramento River occurs farther upstream during years of high flow (Turner, 1976).

Spawning could begin after the water temperature exceeds approximately 58 degrees Fahrenheit and during, or immediately following, an average temperature rise of 2 to 4 degrees Fahrenheit (Turner, 1976). Striped bass generally spawn when temperatures are increasing and is most intense at water temperatures from 63 to 68 degrees Fahrenheit (Turner, 1976; Mitchell, 1987). Most eggs are spawned during peaks that could last one or several days (Interagency Ecological Studies Program, 1991 and 1993). During the spawning season, two to four peaks encompass most of the annual egg production. Spawning peaks in the Sacramento River and the Delta have occurred over a temperature range of 58 to 71 degrees Fahrenheit. The average water temperature during a peak spawning event was 64 degrees Fahrenheit.

Spawning begins first in the Delta, usually mid- to late April, and continues sporadically over three to five weeks (Mitchell, 1987; DFG, 1987). Spawning averages 15 days later in the Sacramento River than in the Delta and usually begins in early or late May and ends in early June (Turner, 1976). Cooler water temperatures delay spawning in the Sacramento River relative to the Delta. High flow dampens increases in temperature, and the delay period is greater during high-flow years.

Striped bass are mass spawners, broadcasting eggs and sperm into the water column (Moyle, 1976; Wang, 1986). Groups consisting of 5 to 30 striped bass, predominantly males, move into the main current of the river to spawn near the surface. Spawning can occur any time of day but generally takes place in the late afternoon and evening. Females are prolific, producing from 11,000 to more than 2 million eggs each. The number of eggs produced is a function of size. A four-year-old female produces more than 200,000 eggs; an eight-year-old female produces more than 1 million eggs; and a 12-year-old female produces more than 1.8 million eggs (DFG, 1987).

Planktonic Eggs and Larvae

Eggs are slightly denser than freshwater and, in the absence of current, sink slowly to the bottom (Moyle, 1976). In the Sacramento River near Verona, where flows are turbulent in the relatively narrow and shallow river, egg densities were variable but tended to be greatest at the surface (Fujimura, 1991). Farther downstream near Walnut Grove, eggs are generally most concentrated at mid-depth and near the bottom. The river near Walnut Grove is wider, deeper, and has more uniform laminar flow, and currents slow when flood tides back up against the downstream river flow (Fujimura, 1991). Eggs transported downstream from the spawning areas sink slowly and are generally concentrated within a few meters of the bottom (Turner, 1976; Wang, 1986).

Eggs hatch in approximately two days at 64 to 66 degrees Fahrenheit (Moyle, 1976; Wang, 1986). Larvae measuring approximately 3 to 5 millimeters long at hatching are sustained by their yolk sac for seven to nine days, after which they exceed 6 to 7 millimeters in length and begin feeding on small zooplankton. As larvae increase in size, their swimming ability and control over position in the water column increase (Fujimura, 1991). Until the transition to external feeding, however, larvae are weak swimmers and are passively dispersed by currents.

In the Sacramento River, eggs and larvae are transported downstream of Rio Vista within a few days and arrive in the Delta before larvae begin feeding (Low and Miller, 1986). The destination of egg and larval striped bass appears to be a function of flow conditions (Turner, 1987). Under high Sacramento River flow and high Delta outflow, eggs and larvae from both the Sacramento River and Delta spawnings are concentrated downstream in Suisun Bay. Under low-flow conditions, eggs and larvae are generally concentrated in the Delta.

The movement of eggs and larvae downstream in the Sacramento River is clearly a function of flow, with higher flows moving eggs and larvae more rapidly downstream. Once eggs and larvae are in the Delta, movement downstream could become more dependent on larval and juvenile behavior, tides, DCC operation, wind, exports, and the location of the entrapment zone (i.e., the zone where salinity is between 2,000 and 10,000 microsiemens electrical conductivity [EC]).

Larval striped bass accumulate in or upstream of the entrapment zone (i.e., near or upstream of salinity greater than 2,000 microsiemens EC) (Fujimura, 1991; Kimmerer, 1992). Larvae are concentrated in the entrapment zone and slightly upstream, consistent with larval behavior to avoid the surface and to concentrate at mid-depth and near the bottom (Fujimura, 1991). Striped bass do not appear to undergo diel (i.e., night and day) vertical movements to maintain position with their prey. Position in the water column could be a function of factors other than feeding.

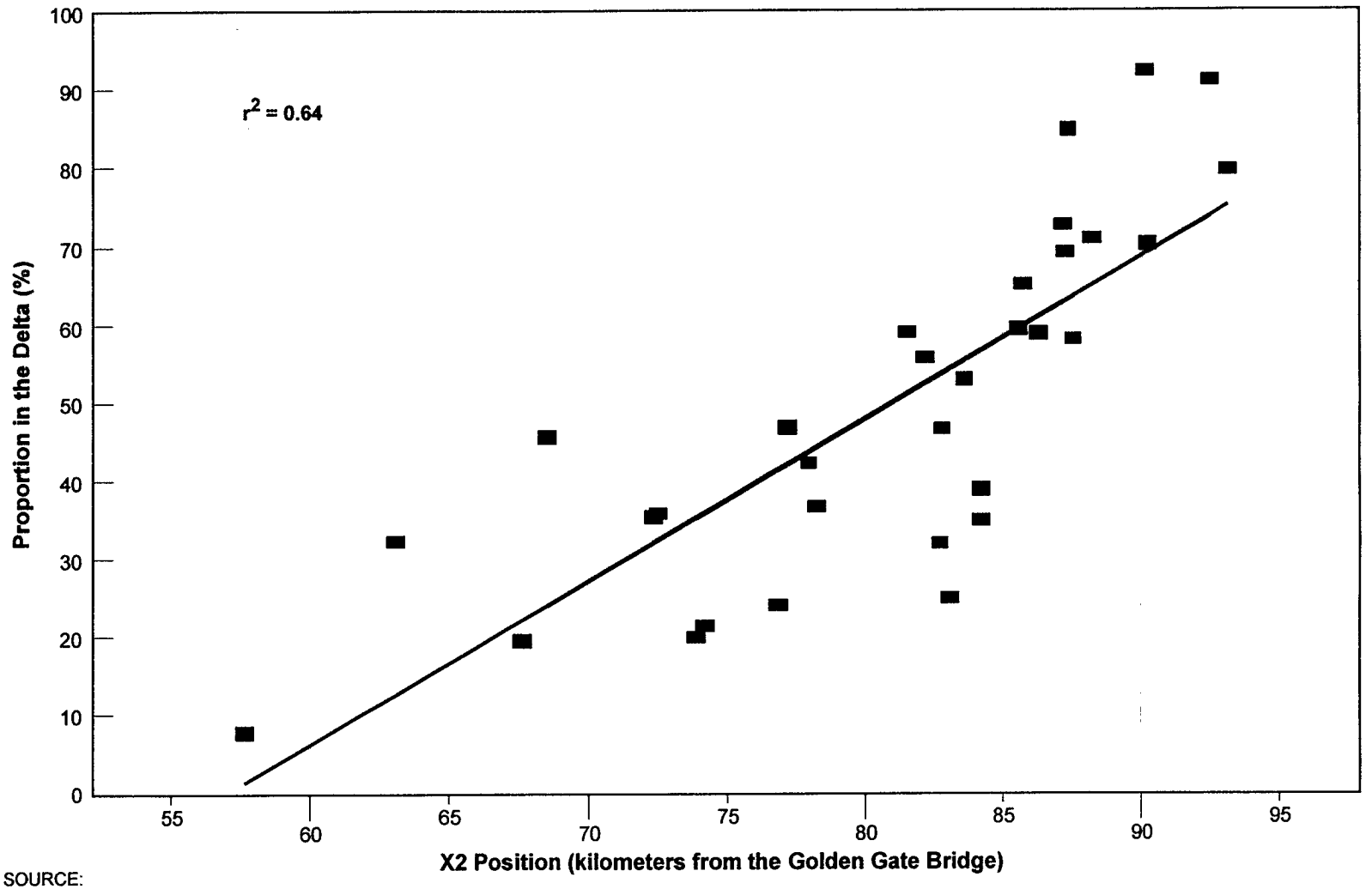
Striped bass larvae eat several species of copepods (including *Eurytemora* spp., *Sinocalanus* spp., and *Cyclopidae*), several species of cladocerans (including *Bosmina longirostrus* and *Daphnia* spp.), and the mysid *Neomysis* spp. The copepod *Eurytemora* spp. is the preferred food of larval striped bass in the Sacramento-San Joaquin Delta estuary. In the San Joaquin River portion of the Delta, the cladoceran *Bosmina longirostrus* is sometimes heavily selected as prey by striped bass larvae.

Larval striped bass generally select prey larger than 1 millimeter within each species and each species group. *Neomysis* is generally too large for larvae to consume but becomes progressively more important in the diet as larvae increase in size.

Rearing Juveniles

Larval stages last four to five weeks, and at approximately 18 millimeters long, the young bass have developed all the features characteristic of juveniles (Wang, 1986; DFG, 1987). Within another four to five weeks (usually in July), depending on water temperature and food availability, juvenile bass will have grown to 38 millimeters. By September, the length of the juveniles in the current year-class ranges from 50 to 120 centimeters (Sasaki, 1966b). By August of the following year, the juveniles will be from 120 to 230 millimeters long. By the end of their third year, the average length is 380 millimeters and the young bass are considered adults.

Like larvae, early juveniles at least 38 millimeters long accumulate in or upstream of the entrapment zone (i.e., near or upstream of salinity greater than 2,000 microsiemens EC) (Fujimura, 1991; Kimmerer, 1992) (Figure II-62). During high-flow years, the entrapment zone and the majority of young-of-the-year striped bass are located in Suisun Bay into fall (Turner and Chadwick, 1972). During low-flow years, the entrapment zone and the majority of young-of-the-year striped bass are found in the Delta. Young-of-the-year bass move out of the



SOURCE:
Herrgesell, 1993.

FIGURE II-62

PROPORTION OF THE YOUNG-OF-YEAR STRIPED BASS POPULATION (38-MM INDEX) IN THE DELTA RELATIVE TO THE LOCATION OF X2 (2 PPT SALINITY OR ABOUT 3,000 μS EC) IN JULY (1959-1991)

C-081486
C—081486

Delta and into Suisun and San Pablo bays during late fall and winter (Sasaki, 1966a, 1966b; Turner and Chadwick, 1972). Movement downstream is more apparent in low-flow years and obscured during high-flow years. After winter of the first year, movements of juvenile striped bass appear to be similar to those of adult bass.

Juvenile striped bass, like larvae, select progressively larger prey as they grow (Thomas, 1967). The primary prey of juvenile bass during their first year is *Neomysis* sp. and amphipods in the genus *Corophium* (Stevens, 1966). Later, the diet of juvenile bass shifts more to fish and becomes similar to the diet of adult striped bass.

FACTORS AFFECTING ABUNDANCE

The decline of the striped bass population in the Sacramento-San Joaquin Delta estuary has generated substantial evaluation of causal factors. The decline in population abundance is a result of increased mortality and reduced reproduction.

Factors that could have contributed to increased mortality after 1967 include the same factors that affected mortality before 1967 (i.e., fishing, entrainment in diversions, exposure to toxic materials, and habitat loss). Additional contributing factors include reduced Delta inflow and outflow, altered Delta flow patterns, dredging and spoil disposal, diseases and parasites, and introduction of exotic species.

Reduced reproduction because of fewer fertile eggs being produced by the population each year can be correlated with the abundance, size, and health of female striped bass. Mortality rates determine the abundance of female bass. Factors affecting size and health of female striped bass include accumulation of toxic materials by the female bass, diseases and parasites, and reduced food availability.

Egg production depends on the abundance and fecundity of adult female striped bass. From the early 1970s to the present, the number of eggs produced declined as the population of adult females declined (DFG, 1987). Average egg production between 1981 and 1986 was 17 percent of the 1969 to 1973 average egg production.

DFG (Kohlhorst et al., 1992) has suggested that a significant stock-recruit relationship exists for striped bass (i.e., the number of bass produced in any given year depends, to some extent, on egg production) (Figure II-63). If the stock-recruit relationship is valid, the existing adult striped bass population could be unable to produce sufficient numbers of eggs to sustain existing mortality rates at all life stages. Reduced recruitment to the adult population and increased mortality of the adult striped bass population will result in continued decline. However, reduced adult mortality in combination with improved habitat conditions could enhance the ability of the population to recover to historic levels.

The focus of the following section is on anthropogenic (i.e., human-caused) factors that could continue to affect abundance, especially factors that are affected by CVP facilities and operations or that could be affected by implementing the CVPIA. In addition, information is provided on environmental conditions that could exacerbate the effects of CVP operations and facilities on conditions that could suppress the benefits of actions implemented under the CVPIA.

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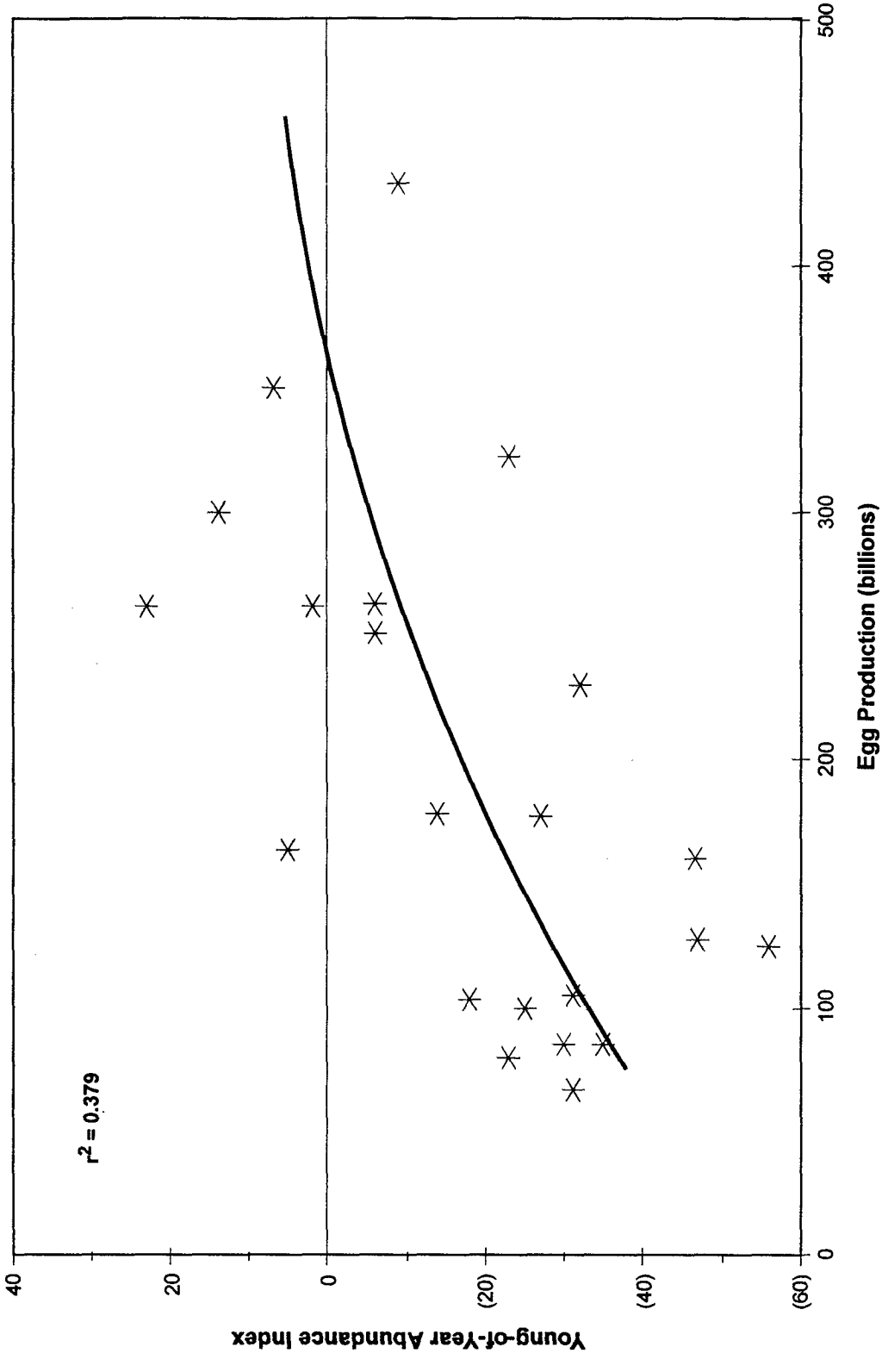


SOURCE: DFG, 1992a.

FIGURE II-63

RELATIONSHIP BETWEEN YOUNG-OF-YEAR STRIPED BASS ABUNDANCE INDEX AND EGG PRODUCTION IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY

Adults

Adult population abundance is correlated with the 38-millimeter index (Figure II-64), suggesting that reduced recruitment to the adult population has been the major cause of declining adult abundance (Kohlhorst et al., 1992). Lower recruitment is estimated to account for 75 percent of the adult decline, while lower adult survival rates account for the remaining 25 percent.

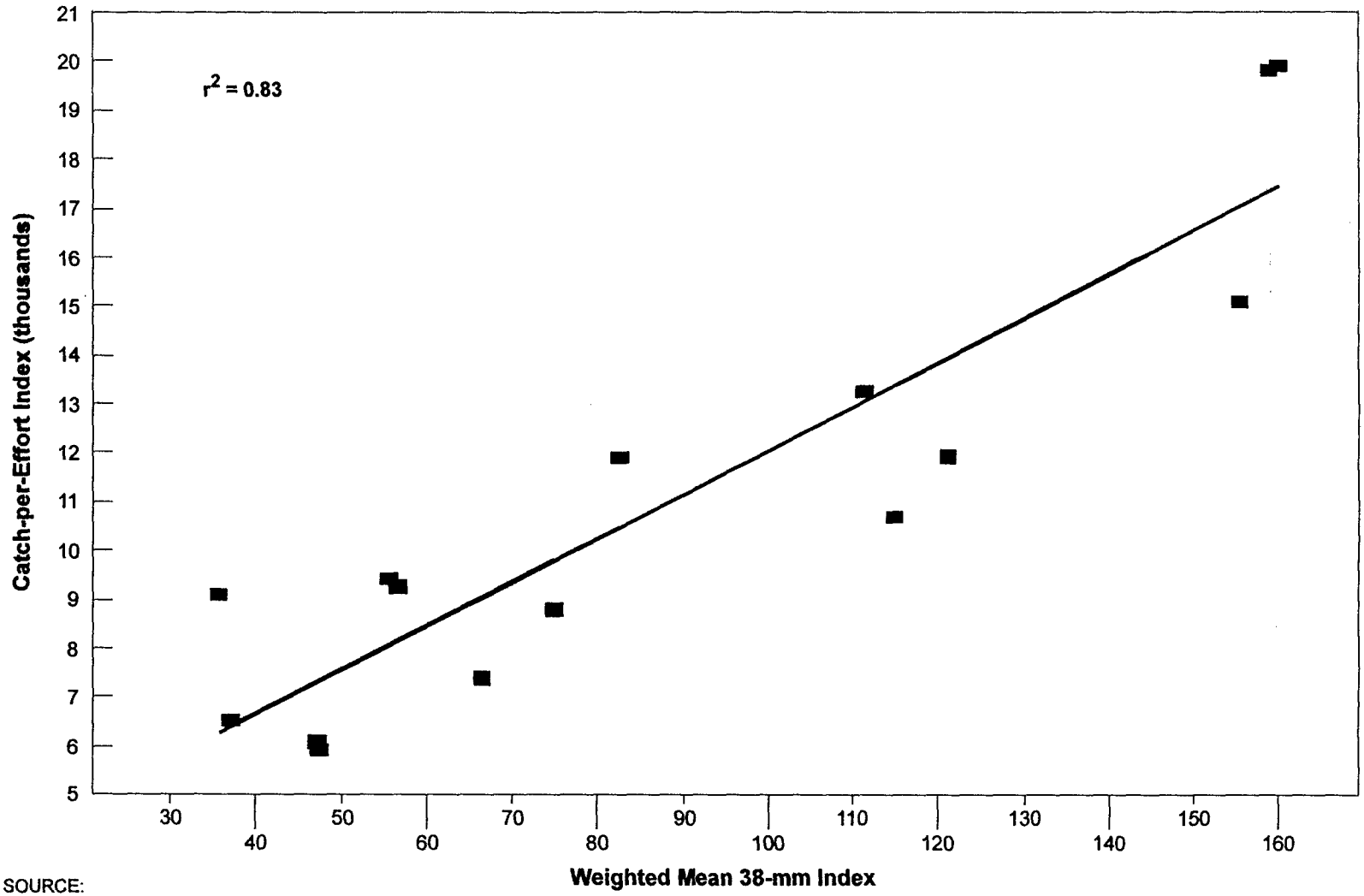
Annual adult striped bass mortality rates increased from approximately 40 percent in the early 1970s to 53 percent in recent years. The relationship between abundance and recruitment is complex, affecting some year classes more than others (DFG, 1987). This increase could be attributed to habitat loss, increased levels of toxic materials, sport and illegal fishing, and other factors.

Habitat. As noted previously, nearly half of the available marsh and tidal habitat has been filled and leveed off (DFG, 1989). In the Delta, less than 3 percent of the habitat remains in a state similar to that of 150 years ago (Herbold et al., 1992). Diking and filling have restricted striped bass habitat and reduced tidal mixing and overall estuary productivity. However, most diking and filling in the estuary preceded the recent precipitous 20-year decline in the population. Since 1970, only relatively small habitat areas have been lost to levee riprapping and additional filling. Although habitat loss does not account for the population decline, restoration of diked and filled wetlands, with subsequent reconnection to the estuary, could provide additional habitat for adult striped bass and increase overall productivity of the estuary.

Toxic Materials. Survival of adult striped bass could be affected by toxic materials entering the Sacramento-San Joaquin Delta estuary from agricultural runoff, discharge of industrial and municipal waste, and runoff from non-point sources (i.e., stormwater runoff). Adult striped bass tissues contain concentrations of toxics exceeding levels recommended for human consumption; however, data prior to the striped bass decline after 1970 are unavailable for comparison (Herbold et al., 1992). Relative to striped bass on the Atlantic coast and in other estuaries, striped bass from the Sacramento-San Joaquin Delta estuary appear to be in poor health and often have open lesions (reactions to parasite infection) (Brown, 1987).

Every year during May and June, hundreds to thousands of adult striped bass die and wash up along the shoreline of the estuary (Brown, 1992). The highest density of dead adults is found in Carquinez Strait. Livers from dead striped bass were found to be contaminated with higher concentrations of toxic materials than the livers of healthy fish taken from the Delta. A causative factor for the die-off has not been identified, but the relatively high concentration of toxic materials could contribute to factors resulting in the mortality.

The number of viable eggs is directly affected by contaminant levels in prespawning females, causing resorption of eggs or production of abnormal embryos (Brown, 1987; DFG, 1987). Analysis has not shown strong relationships between reproductive condition, parasite burdens, and pollutant concentrations. Striped bass in the Sacramento-San Joaquin Delta estuary, however, are less fecund than female bass from other estuaries. Reduced fecundity appears to be related to the effects of toxic materials, but the extent of reduced fecundity is unknown.



SOURCE:
Kohlhorst et al., 1992.

FIGURE II-64

RELATIONSHIP BETWEEN ADULT POPULATION ABUNDANCE (STRIPED BASS TAGGING STUDY CATCH-PER-EFFORT INDEX) AND WEIGHTED MEAN YOUNG-OF-YEAR INDEX 3-7 YEARS EARLIER

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Fishing. The recent (1988 to 1992) annual catch of striped bass is approximately 85,000 fish (i.e., approximately 9 to 14 percent of the adult population) (DFG, 1992a). Incidental take of striped bass in legal commercial fisheries increases the annual harvest rate by a negligible amount. Considering that fish populations can sustain high levels of fishing mortality and that striped bass populations on the Atlantic coast have sustained harvest rates greater than 40 percent, the existing harvest rate will likely have minimal effects on a healthy striped bass population. The precipitous decline in adult striped bass abundance over the past 20 years, however, indicates that the population is unhealthy (Figure II-56).

The declining status of the adult population has resulted in more stringent angling regulations, including an 18-inch minimum length and two-fish-daily bag limits (DFG, 1992a). Prior to 1982, the minimum legal length was 16 inches and the daily bag limit was three fish. More stringent sport fishing regulations and stricter enforcement could reduce adult mortality and increase egg production.

Planktonic Eggs and Larvae

As discussed in preceding paragraphs, lower recruitment is estimated to account for 75 percent of the adult population decline that has occurred since the late 1960s. Recruitment to the adult population depends on survival of eggs, larvae, and juvenile bass.

Studies have shown a significant relationship between the annual abundance of larval striped bass (8 millimeters long) and juvenile striped bass (38-millimeter index), and between juvenile striped bass and recruitment to the population three years later, indicating that year-class strength of the population is set early in the life cycle (Turner, 1987; Kohlhorst et al., 1992) (Figures II-59 and II-63). The number of 8-millimeter larvae is a function of the number of viable eggs spawned, spawning timing and location, flow conditions, direct diversion effects, and development rates (a function of water temperature).

Flow

Sacramento River. The survival (survival index) between the egg and the 6-millimeter larvae stage in the Sacramento River is low when Sacramento River flow is low (Figure II-65) (DFG, 1992a). Survival is always low when flow is less than 13,000 cfs. Mechanisms that could explain reduced survival at lower Sacramento River flows include the following:

- Eggs and larvae settle to the river bottom and die when they encounter near zero velocity in tidally affected reaches.
- Larval survival is reduced because arrival in higher quality downstream nursery areas is delayed.
- Larvae are subjected to increased exposure to toxic substances carried by the river.
- A higher proportion of larvae are drawn through the DCC, Georgiana Slough, and Threemile Slough into the central Delta where vulnerability to entrainment in diversions is greater.

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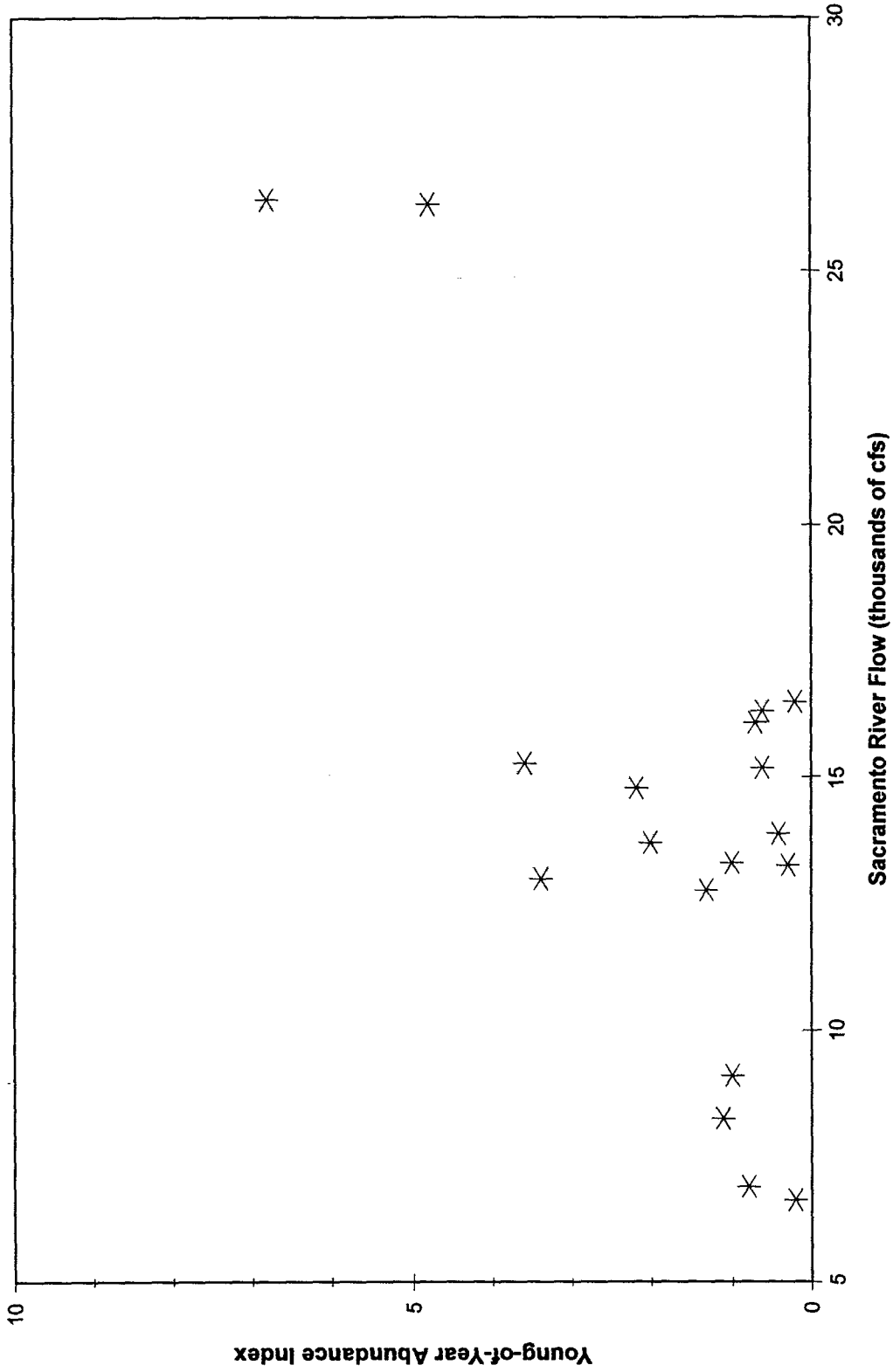


SOURCE:
DFG, 1992a.

FIGURE II-65

RELATIONSHIP BETWEEN SURVIVAL OF STRIPED BASS (EGGS TO 6MM) AND SACRAMENTO RIVER FLOW

Feeding efficiency, and hence growth and survival, could be greater in downstream reaches because the density of striped bass prey in the Sacramento River is higher in the reaches below Rio Vista (DFG, 1992a). Assuming that the proportion of eggs and larvae drawn into the DCC, Georgiana Slough, and Threemile Slough depends on the proportion of Sacramento River flow diverted, more eggs and larvae will be drawn into the central Delta at low flows than at high flows (see Attachment B for historical information on flows and proportion of flow diverted off the Sacramento River in the Delta).

In addition to flow effects on survival, diversions from the Sacramento River could entrain eggs and larvae and reduce river flow. In proportion to Sacramento River flow, diversions from the Sacramento River in the spawning reach (between Sacramento and Colusa) are small (see Attachment B). The effect of Sacramento River diversions on striped bass, although they contribute to the cumulative effect of total diversions and upstream storage, will also be expected to be relatively small.

Delta Outflow. Delta outflow is highly variable across years, seasonally, and weekly at times (see Attachment B). In general, month-to-month outflows in any given year are highly autocorrelated, whereas year-to-year outflows are not. This generally means that high outflows occur across several months in wet years (Herbold et al., 1992). In any given year, outflow has ranged from less than 10 million acre-feet to more than 50 million acre-feet.

Although dependent on the natural hydrology of the Sacramento-San Joaquin River system, the timing and volume of Delta outflow have been substantially modified by changes in system characteristics; channelization and flood control projects; and by operations of water project facilities, reservoirs, and diversions (Herbold et al., 1992) (see Attachment B). Channelization and flood control projects (not including reservoir storage) enable water to move more quickly to the Delta. Reservoir storage reduces peak flows and changes the timing of water movement down the rivers. Consumptive diversions remove water from the system.

In general, water projects have increased summer and fall outflow and reduced winter and spring outflow (Herbold et al., 1992). Total annual Delta outflow can be reduced by 50 to 60 percent of the outflow expected in the absence of storage and diversions, with less proportional change in wet years and greater in dry years.

Delta outflow and diversions are considered by DFG (1992a) to be the primary factors contributing to the continuing 20-year decline of striped bass in the Sacramento-San Joaquin Delta estuary. The decline in juvenile striped bass abundance correlates significantly with numerous flow-related variables, including water temperature, Delta inflow, Delta outflow, salinity, and diversions (Turner and Chadwick, 1972). Because the variables are highly interdependent, the mechanisms causing the decline are unclear.

Delta outflow affects the distribution of striped bass larvae. The location of X2 (i.e., 2 parts per thousand salinity, or approximately 3,000 microsiemens EC, measured one meter above the channel bottom) in the estuary is indicative of the level of Delta outflow; as outflow increases, X2 moves farther downstream (San Francisco Estuary Project, 1993). When X2 is in Suisun Bay, larvae density is greatest in Suisun Bay; when X2 is in the Delta, larvae density is greatest in the Delta. Figure II-62 shows a similar relationship for 38-millimeter striped bass juveniles.

The mechanism of distribution (i.e., whether outflow transports the larvae downstream or larvae actively maintain their position relative to the entrapment zone) is not known, but the location of larvae relative to X2 is consistent with larval avoidance of the surface.

Striped bass survival from egg to 38 millimeters long and from 9 millimeters to 38 millimeters long is higher at higher outflows (i.e., when X2 is farther downstream) (DFG, 1992a; San Francisco Estuary Project, 1993) (Figure II-66). High outflow could benefit larval striped bass by:

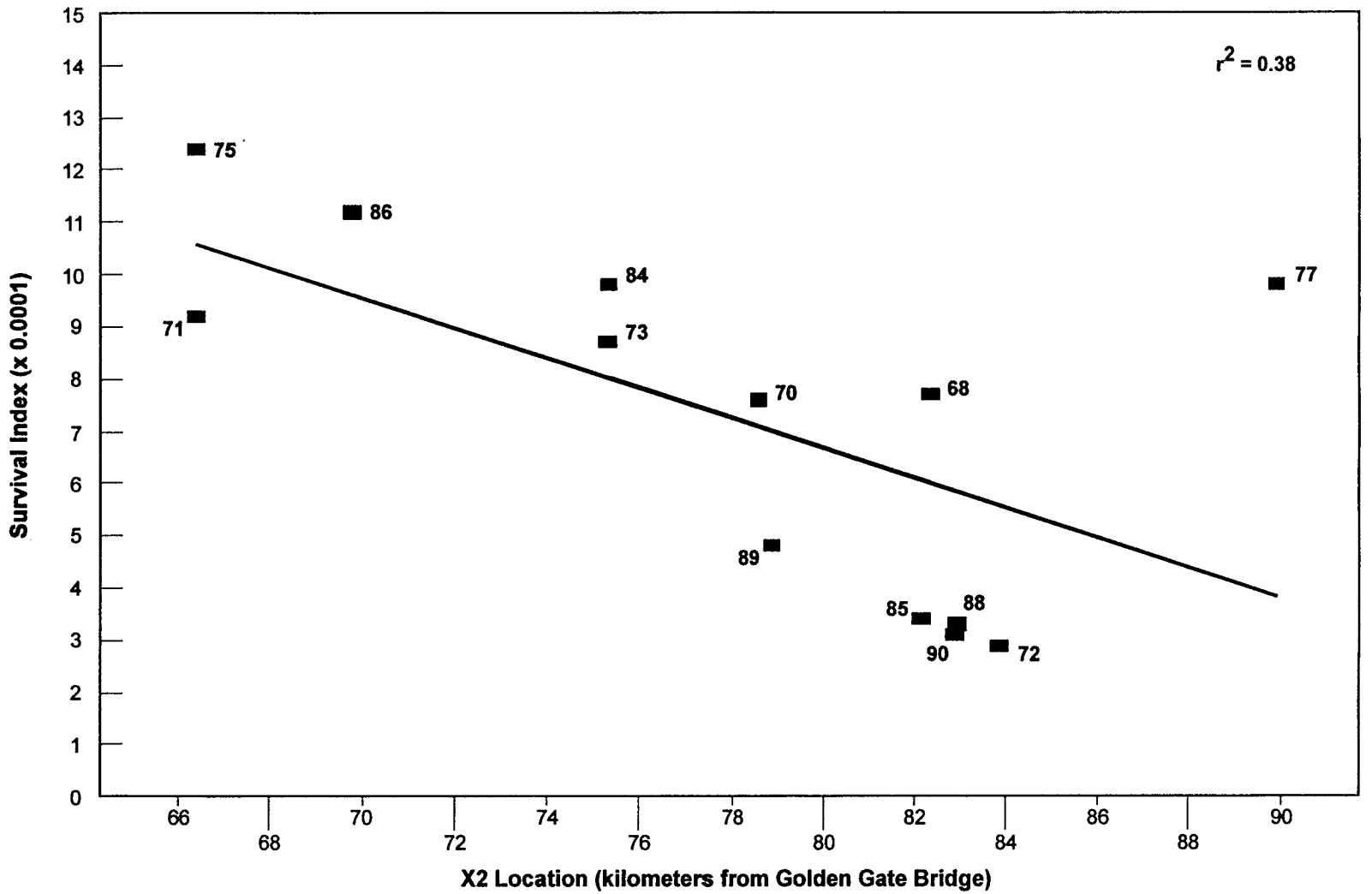
- increasing the nursery area and reducing intraspecific competition;
- increasing shallow habitat area and food abundance;
- diluting toxic materials;
- increasing turbidity and reducing predation; and
- reducing vulnerability to entrainment in Delta diversions (Herbold et al., 1992).

Delta Diversions. Consumptive diversions from the Delta include the CVP and SWP Delta pumping facilities, more than 1,800 agricultural diversions, CCWD's Rock Slough diversion, the North Bay Aqueduct, and numerous other municipal and industrial diversions (shown in Attachment B). Up to 4,600 cfs and 10,300 cfs can be diverted from the CVP and SWP Delta pumping facilities, respectively. CCWD has a maximum diversion capacity of approximately 300 cfs; the North Bay Aqueduct has a maximum capacity of approximately 140 cfs. Maximum agricultural diversions during the peak summer irrigation season could exceed 4,000 cfs (DWR, 1993a). Total diversions from the Delta can exceed 80 percent of the total Delta inflow (Turner and Chadwick, 1972; DWR, 1993b).

Diversions entrain striped bass (discussed under "Entrainment") and affect Delta outflow and flows in the Delta channels. Considering the historical magnitude (see Attachment B) and location of diversions relative to striped bass distribution and life history patterns, Delta diversions could have been a major factor contributing to reduced young (or larval) striped bass survival. Delta diversions, primarily by the CVP and SWP, are considered by DFG to be responsible for the depleted state of the striped bass population (DFG, 1992a).

Over the 1959-1990 period, the abundance of striped bass (38-millimeter index) was negatively correlated with the Delta diversions and positively correlated with Delta outflow (DFG, 1987, 1992a). If data for the entire 1959-1990 period are used to develop the regression equation, the total predicted abundance is generally less than the total measured abundance for 1959 to 1976 and greater than measured abundance for 1977 to 1990 (Figure II-67). When separate equations are used for those time periods for Suisun Bay (using Delta outflow only) and for the Delta (diversion and outflow), the predictions are greatly improved (Figure II-67).

DFG has hypothesized that the difference between the 1959-1976 and 1977-1990 relationships is attributable to the decline of the adult population to a level that caused egg production to become limiting (i.e., the stock-recruit relationship is partially controlling abundance) (Figure II-64) (DFG, 1992a). Changes in estuarine productivity, toxic materials entering the estuary, and other factors could also explain the change in the relationship between abundance and the combined effects of diversion and outflow during the 1970s.



SOURCE:
DFG, 1992a.

FIGURE II-66

RELATIONSHIP BETWEEN THE SURVIVAL INDEX (FOR 9-MM TO 38-MM STRIPED BASS) AND THE LOCATION OF X2

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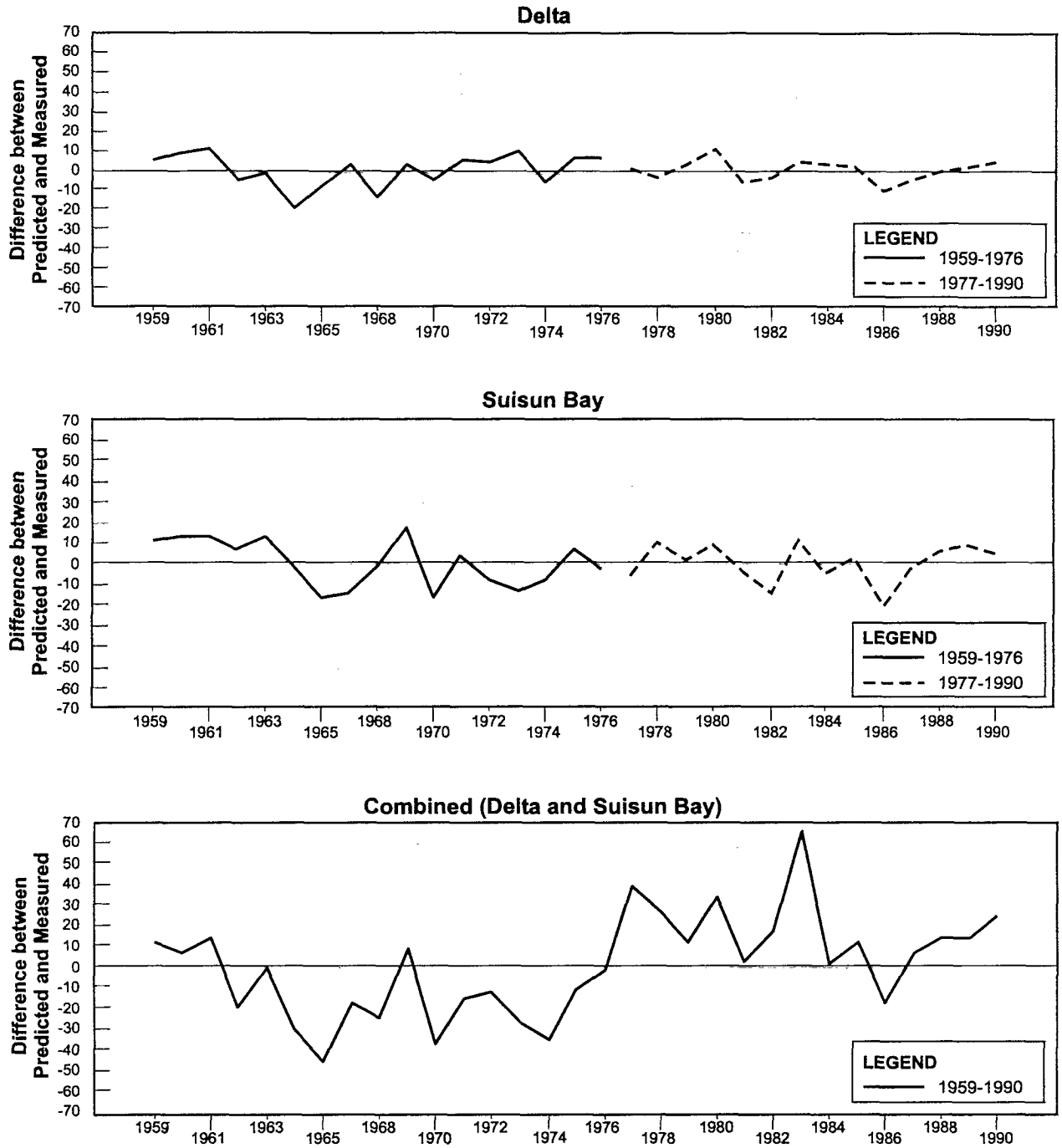

NOTE:
Different equations were used for 1959-1976 and 1977-1990 to calculate the Delta and Suisun Bay indices.

SOURCES:
DFG, 1992a; Herrgesell, 1993.

FIGURE II-67

DIFFERENCE BETWEEN THE PREDICTED AND MEASURED ANNUAL ABUNDANCE INDICES FOR STRIPED BASS IN THE DELTA, SUISUN BAY, AND IN BOTH AREAS COMBINED (1959-1990)

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C-081496

After 1970, striped bass survival in Delta habitats appears to have declined (DFG, 1992a). The difference in the relationships between the proportion of striped bass in the Delta and the location of X2 for the 1959-1969 and 1970-1991 periods (r^2 equals 0.85 and 0.62, respectively) indicates that use of the Delta as a nursery could have declined or that survival could have been lower for the 1970-1991 period (Figure II-68). The lower position of the line representing the 1970-1991 correlation between the proportion of striped bass in the Delta and the location of X2 indicates that fewer bass were in the Delta during similar outflow conditions (i.e., X2 locations).

If survival rates in the Delta in fact declined after 1970, SWP Delta pumping facilities could be responsible. The SWP began exporting water after 1968 and began significant pumping by 1970 (see Attachment B). Other factors could have contributed to the decline (e.g., toxic materials entering the estuary), but insufficient data could exist for evaluation.

Montezuma Slough. After installation and operation of the Suisun Marsh Salinity Control Structure in 1989 to improve water quality, the flow in Montezuma Slough greatly increased, averaging more than 2,000 cfs toward Suisun Marsh during operation of the structure (see Attachment B). The timing of operations extends through the striped bass egg and larval period. The effect on striped bass is currently unknown, but operations could reduce survival through increased predation at the control structure (Herrgesell, 1993) and exposure to conditions within Montezuma Slough that could be less conducive to survival than conditions in Suisun Bay.

Also, the effect of the diversion by the Suisun Marsh Salinity Control Structure on the location of X2 is unknown. If that diversion causes X2 to be located farther upstream, survival of striped bass could be reduced (Figure II-66).

Entrainment. Entrainment losses were at least partly responsible for the decline in striped bass after 1970. Entrainment losses appear to be greater in low-flow years, as evidenced by greater losses at the CVP Delta pumping facilities and by the close relationship between striped bass abundance and the percentage of inflow diverted (DFG, 1987).

CVP and SWP Delta Pumping Facilities. As discussed previously, the CVP and SWP Delta pumping facilities are the largest diversions from the Delta. Millions of striped bass eggs and larvae are lost to annual entrainment in export by the CVP and SWP Delta pumping facilities (Figure II-69). Using estimated egg and larval survival rates (Figure II-70), the adult equivalent loss amounts to thousands of yearling striped bass each year (Figure II-71).

Agricultural Diversions. Losses of striped bass to agricultural diversions are believed to be considerable (Odenweller, 1981) and have been estimated to be in the millions, possibly equivalent to entrainment loss to SWP and CVP diversions (Stevens et al., 1985; Brown, 1992). Actual loss estimates are unavailable (Brown, n.d.). Losses to agricultural diversions depend on the timing, size, and location (geographically and position in the channel) of individual diversions relative to the seasonal distribution and abundance of striped bass. Losses of egg and larval striped bass could be most effectively minimized by curtailing diversions in May and June. A major Interagency Ecological Program (IEP) (previously known as the Interagency Ecological Studies Program) study could provide future estimates of agricultural diversion effects.

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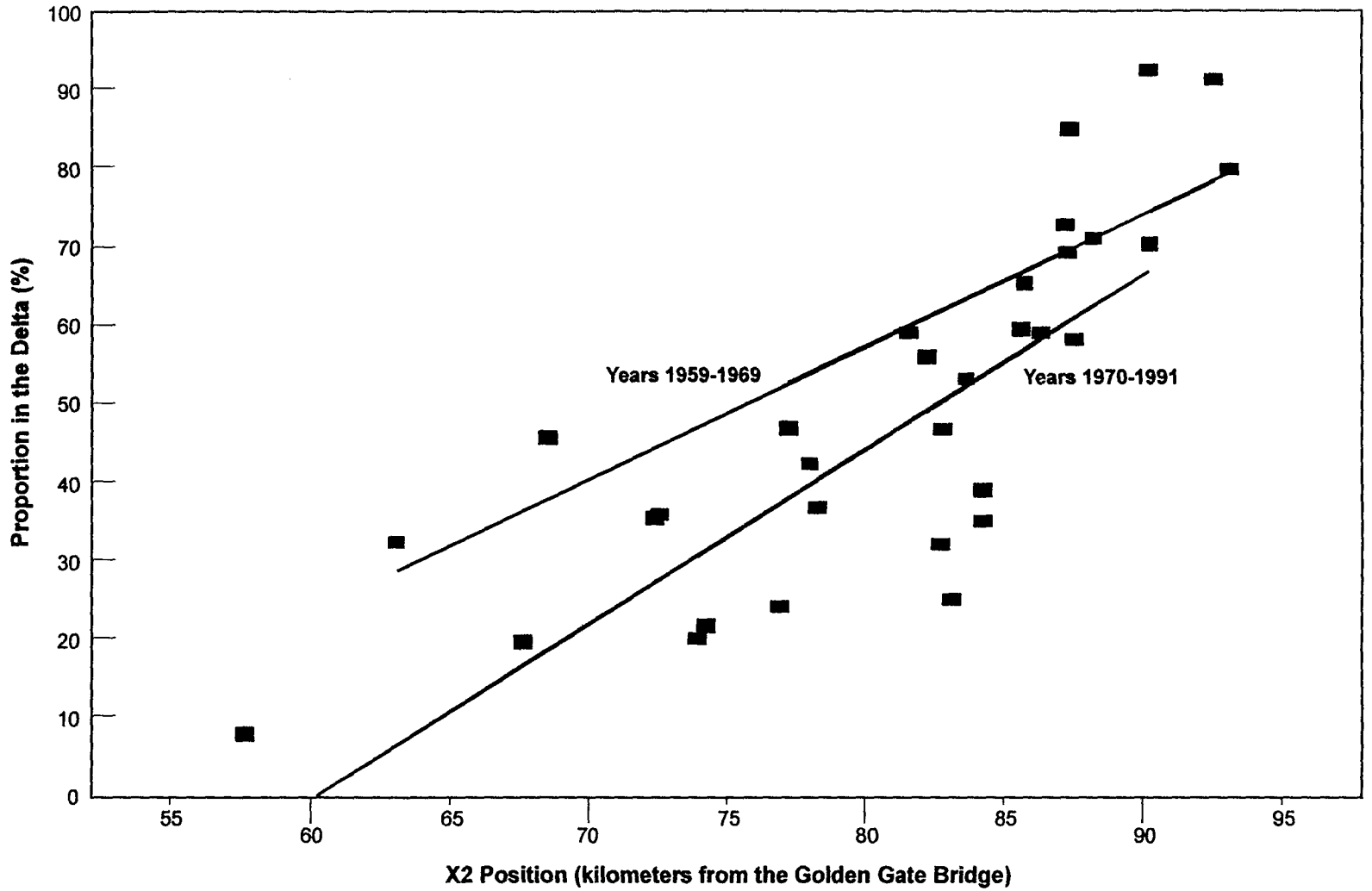
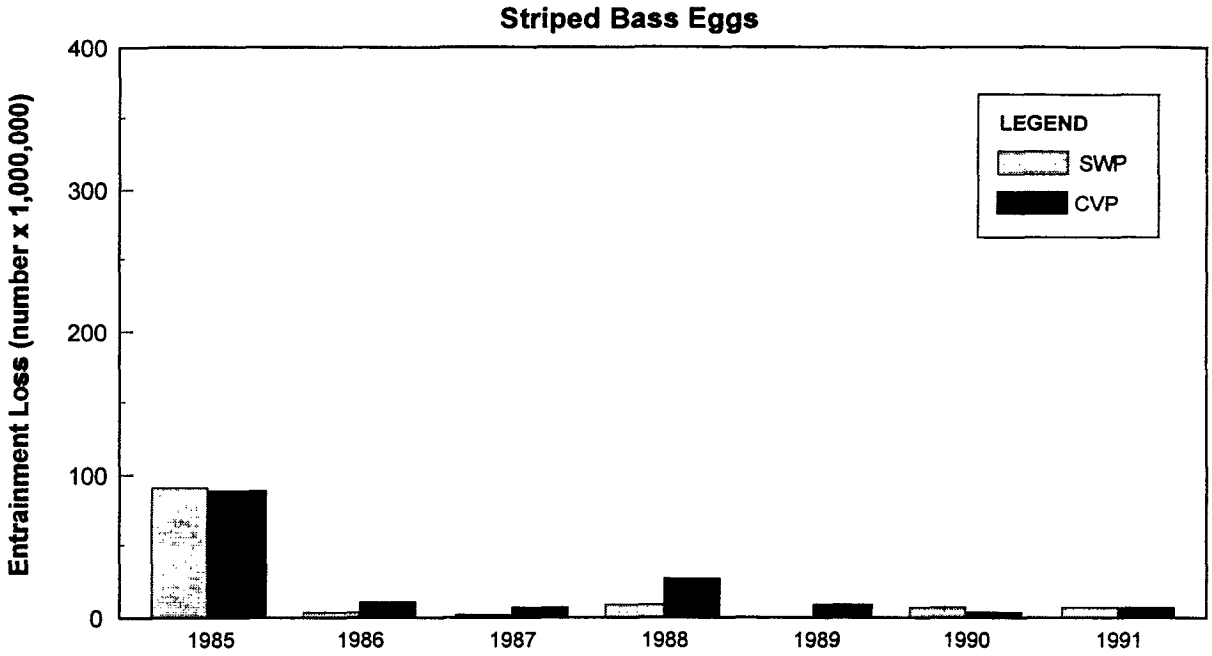
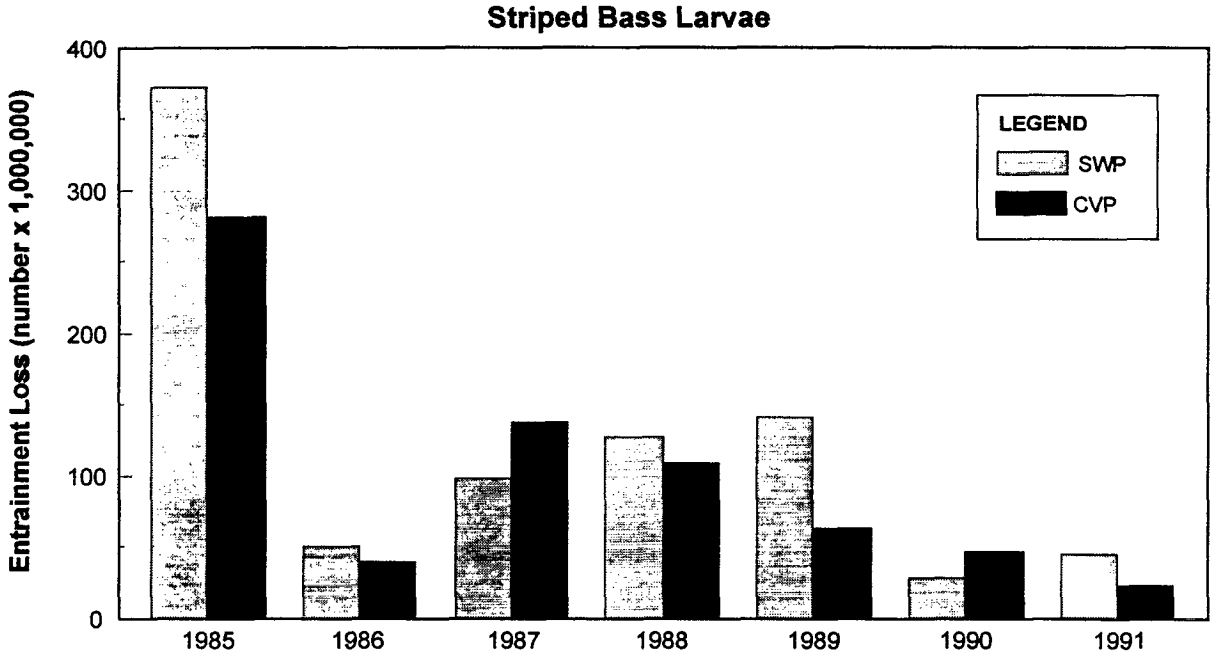


SOURCE:
Herrgesell, 1993.

FIGURE II-68

COMPARISON OF THE 1959-1969 AND 1970-1991 RELATIONSHIPS BETWEEN THE PROPORTION OF STRIPED BASS IN THE DELTA AND THE LOCATION OF X2





SOURCE:
Herrgesell, 1993.

FIGURE II-69

ENTRAINMENT LOSS OF STRIPED BASS EGGS AND LARVAE IN DIVERSIONS BY THE SWP AND CVP DELTA PUMPING FACILITIES (1985-1991)

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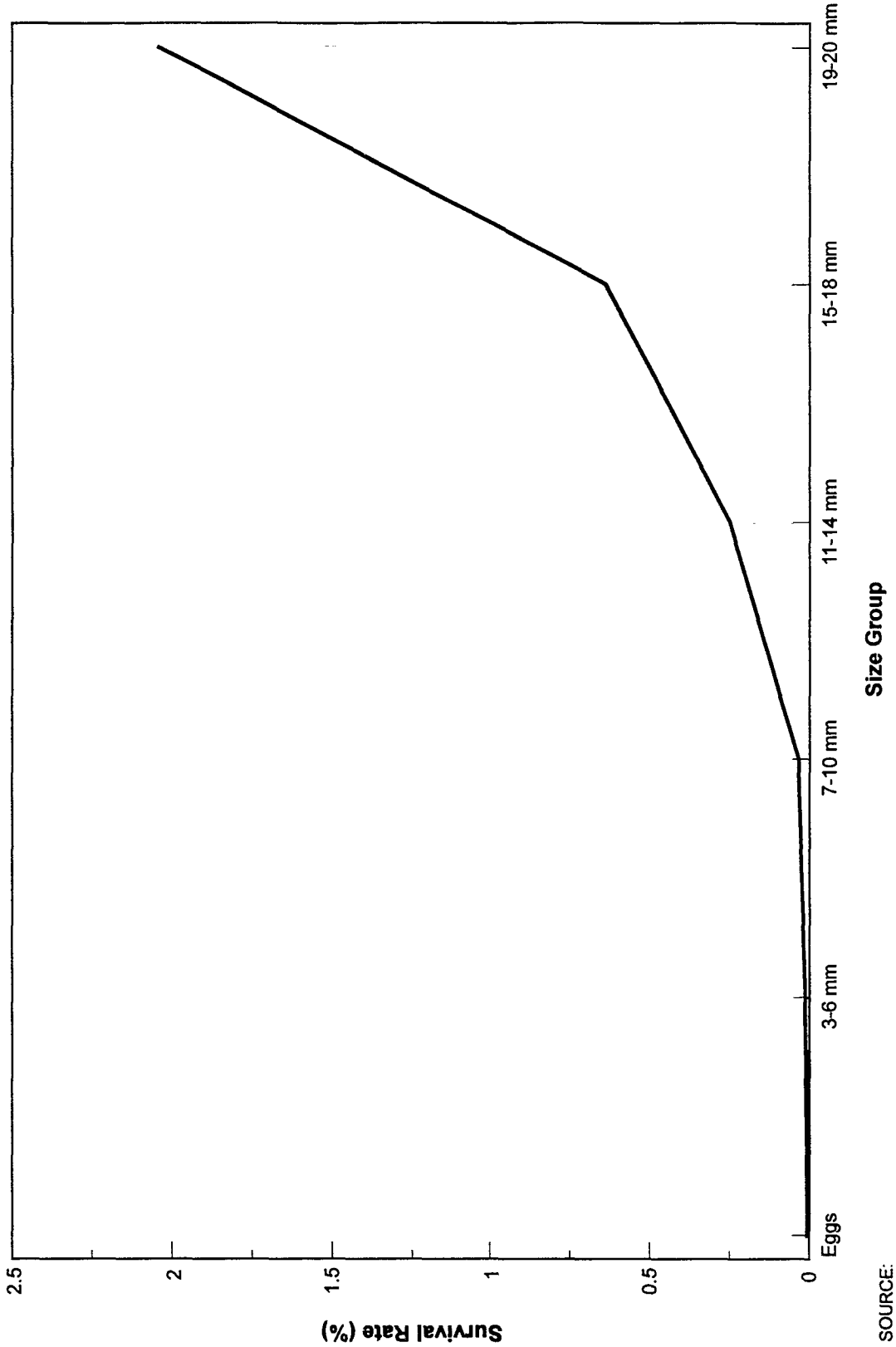
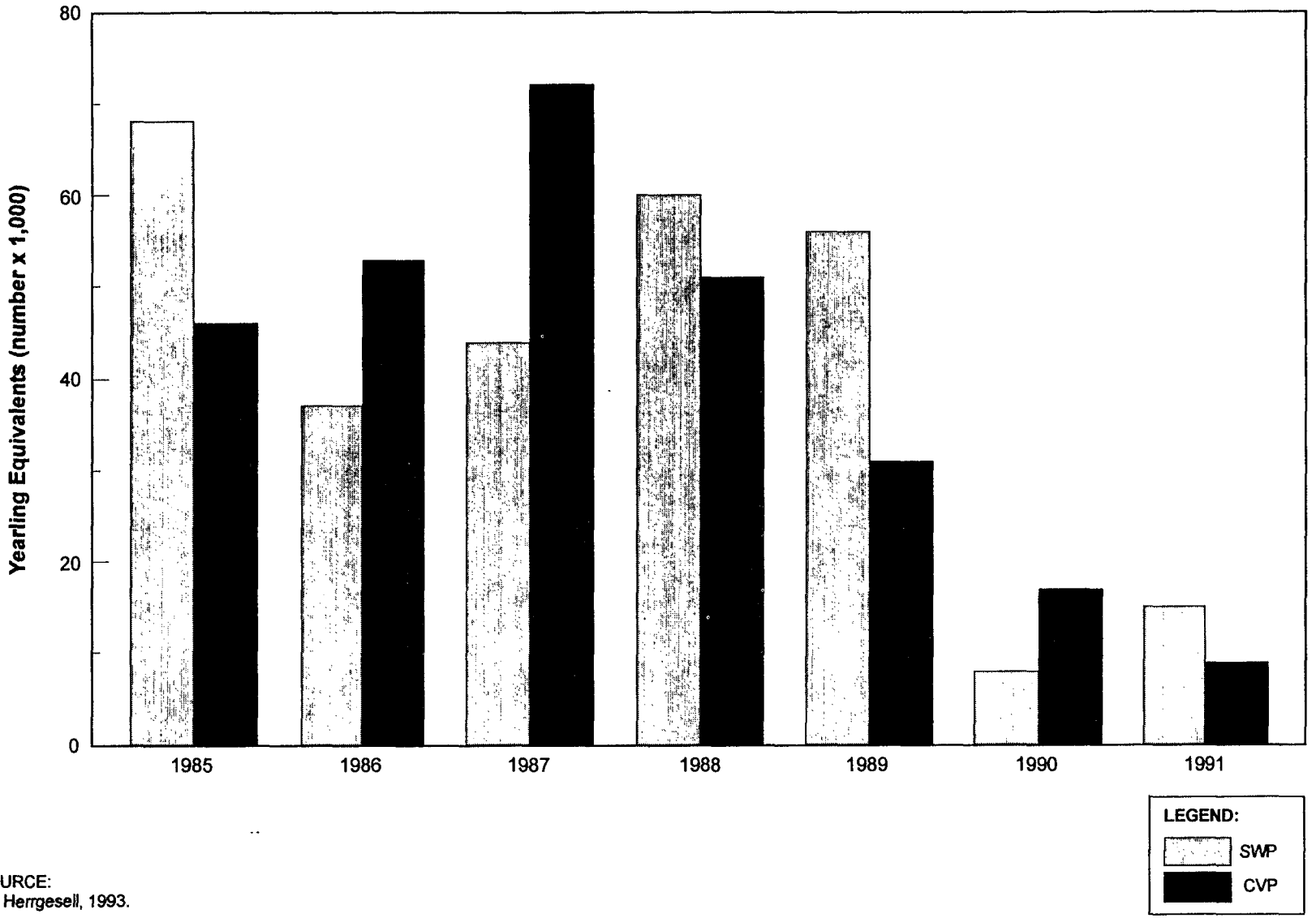


SOURCE:
Herrgesell, 1993.

FIGURE II-70

SURVIVAL RATES USED TO CONVERT EGG AND LARVAL ENTRAINMENT LOSSES TO YEARLING EQUIVALENTS



SOURCE:
Herrgesell, 1993.

FIGURE II-71
YEARLING EQUIVALENTS FOR STRIPED BASS EGGS AND LARVAE LOST IN DIVERSIONS BY THE SWP AND CVP DELTA PUMPING FACILITIES (1985-1991)

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Power Generation Facility Diversions. Two of the largest nonconsumptive diversions in the Sacramento-San Joaquin Delta are PG&E's Contra Costa and Pittsburg powerplants. Considering the location of the facilities' intakes in the striped bass rearing area (near Antioch and Pittsburg) and the size of the diversions (nearly 1,500 cfs at each powerplant, depending on power generation needs), substantial numbers of egg and larval striped bass could be entrained and lost in the diversions (PG&E, 1985). From 1984 to 1989, 10,000 to 61,000 striped bass yearling equivalents were killed annually at the two powerplants (PG&E, 1990).

Losses of striped bass, however, have been reduced from previous operations. Annual variability in water temperature (a factor controlling bass mortality) and variability in the availability of alternative power supplies have prevented the powerplants from additional reductions in striped bass losses. PG&E has participated in the juvenile striped bass stocking program to mitigate losses.

Other Diversions. Other diversions also entrain and kill striped bass eggs and juveniles. The largest diversions not previously discussed are the North Bay Aqueduct diversion and CCWD's Rock Slough diversion. Losses of eggs and juveniles to diversions other than those described in previous sections are unquantified.

Although eggs and larvae occur around CCWD's Rock Slough diversion, annual entrainment losses of eggs and larvae to the diversion are unknown. The diversion is not located near the main striped bass spawning area. High entrainment losses of striped bass eggs and larvae do occur at the SWP and CVP Delta pumping facilities. Old River transports water and striped bass eggs and larvae to the SWP and CVP Delta pumping facilities. Diversion during the striped bass egg and larval period draws water and the associated eggs and larvae off Old River and to the Rock Slough diversion. Entrained eggs and larvae could also come from nearby channels (e.g., Grant Line Canal).

Salinity

Egg Distribution. Approximately 40 percent of the striped bass population spawns in the Delta, generally in the lower San Joaquin River, from Venice Island downstream to Antioch. Historically, striped bass spawned in the lower San Joaquin River from Vernalis to Chipps Island. Striped bass spawn within areas that have a preferred salinity range, where these conditions are typically found in the lower San Joaquin River. Salinity in the western Delta affects the spawning distribution in the Delta (DFG, 1987). The lowest salinity occurs immediately downstream of the confluence of the San Joaquin and Mokelumne rivers, where freshwater from the Mokelumne and Sacramento rivers enters the San Joaquin River. To the east, salty agricultural drainage discharges contaminate water to the San Joaquin River and alter the salinity composition such that adult striped bass avoid these areas during spawning.

Although spawning in the Delta has occurred when salinity (from seawater intrusion) exceeded 1,500 microsiemens EC, the effect on egg and larva survival is unknown (DFG, 1987). Laboratory studies indicate that salinities less than 1,500 microsiemens EC do not adversely affect egg survival.

The downstream extent of spawning is usually near Antioch, but in years when salinity intruded into the Delta, spawning occurred several miles farther upstream (DFG, 1987). The shift in spawning has not always avoided higher than normal salinity, and spawning has been recorded in salinities exceeding 1,500 microsiemens EC. Striped bass generally return to the same spawning area each year, but regular occurrence of high salinities could gradually reduce the use of the lower San Joaquin River in the Delta as a spawning area because of the preference for freshwater for spawning.

The farther upstream X2 is located, the farther upstream spawning generally occurs (Figure II-72). Eggs spawned upstream in the Delta (in the lower San Joaquin River) are more vulnerable to entrainment in water exports from the south Delta (DFG, 1992a).

Larval Distribution and Survival. Larval distribution and survival in relation to salinity (X2 as a function of Delta outflow) were discussed previously under "Delta Outflow" and "Delta Diversions."

Habitat. The effects of habitat loss on eggs and juvenile striped bass are currently unknown. Effects on overall estuary productivity, however, could have substantially reduced larval survival, but this does not account for the population decline after 1970.

Toxic Materials. Larval striped bass survival could have been reduced by the toxic effects of insecticides, herbicides, trace elements, and other toxic materials that have entered the estuary from agricultural runoff and municipal and industrial discharge. Toxic materials can affect larval bass directly and indirectly, causing mortality within a short period (days) or adversely affecting growth and development, thereby limiting chances for survival (Brown, 1987).

Recent studies indicate that striped bass larvae from the Sacramento River have a higher incidence of liver malformation than larvae from other areas of the estuary. Contamination of the Sacramento River increased substantially in the mid-1970s when application of rice pesticides increased (Herbold et al., 1992). Measured toxic concentrations were sufficient to kill fish in sloughs draining rice fields, and estimated toxic concentrations for the Sacramento River between 1970 and 1988 could have deleteriously affected striped bass larvae (Bailey, 1992). Discharge of contaminated rice field water coincides with striped bass spawning and could account for part of the decline in striped bass abundance. Pesticide application has correlated with young striped bass abundance, but direct relationships are inconclusive.

Although the decline in striped bass abundance that has occurred during the last 20 years is not attributable to toxic materials alone, toxics could have contributed substantially to reduced survival of striped bass. The issue of toxic materials needs to be addressed in much greater detail to determine the effect on striped bass abundance.

Prey Availability. Decline in the copepod *Eurytemora,* the preferred prey of larval striped bass, began in 1988 after the period that striped bass declined in abundance (DFG, 1992a; Obrebski et al., 1992). The composition and abundance of larval striped bass prey have changed dramatically since 1979; some species increased in abundance while others declined (see "Recent Conditions for the Estuarine Invertebrate Community"). Although the introduced *Sinocalanus* has replaced declining populations of *Eurytemora* (Herbold et al., 1992), striped bass larvae (less

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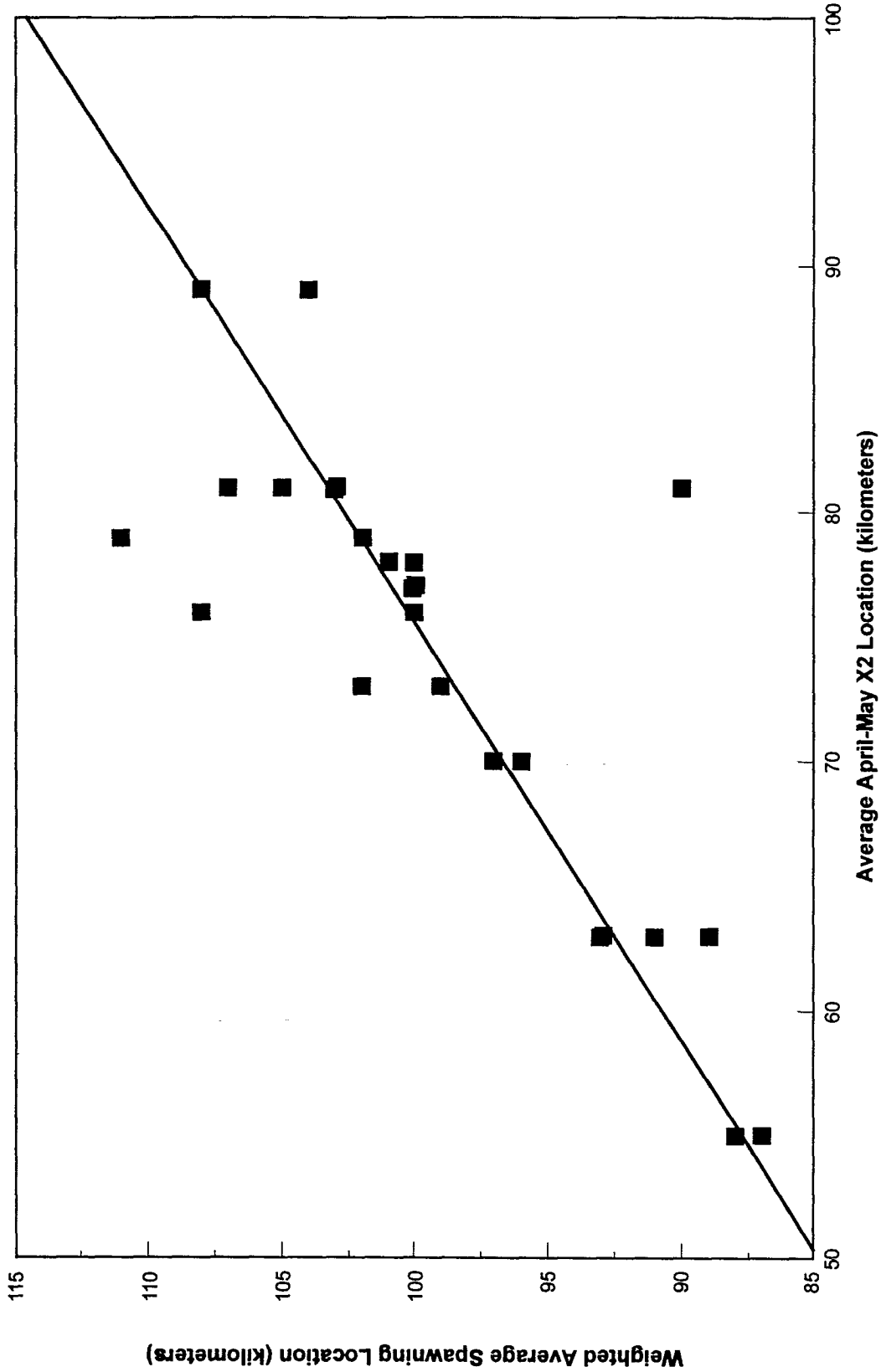


FIGURE II-72

RELATIONSHIP BETWEEN THE LOCATION OF STRIPED BASS SPAWNING IN THE LOWER SAN JOAQUIN RIVER PORTION OF THE DELTA AND THE LOCATION OF X2 (KILOMETERS FROM THE GOLDEN GATE BRIDGE) IN THE ESTUARY

than 10 millimeters) do not effectively feed on the recently abundant *Sinocalanus* (Interagency Ecological Studies Program, 1994). *Pseudodiaptomus forbesi*, an introduced copepod, is found in freshwater at concentrations greater than those of *Eurytemora* and is a food source for larval striped bass (Interagency Ecological Studies Program, 1994).

Laboratory experiments show that striped bass mortality is negatively correlated with prey density (Herbold et al., 1992). Field studies indicated that prey density in the estuary was low relative to densities needed to support high survival in the laboratory. Larvae collected from the estuary do not show signs of starvation, but low densities of prey could result in slower striped bass larval growth rates and increased mortality from predation. Experiments indicate that larval mortality in the estuary was estimated to be higher than larval mortality for similar prey densities in the laboratory.

Reduced abundance of striped bass attributable to reduced prey abundance should be reflected in reduced larval survival rates for any given level of outflow and diversion (DFG, 1992a). Larval survival over the historical period (1969-1990), however, appeared to be unchanged, except for the effects of diversion and outflow.

Competition and Predation. The effects of competition and predation are difficult to evaluate in wild populations. Parallel trends (i.e., abundance declines of one species during the same period that abundance of a competing or predator species increases) will suggest competition or predation effects. A consistent increase in the abundance of species that compete with or prey on striped bass is not apparent from analysis of available data (DFG, 1987).

Introduction of exotic organisms has substantially altered the biological structure of the estuary. Exotic organisms affect striped bass through competition, predation, and change in trophic dynamics (i.e., the availability of prey). Although numerous introduced fish and invertebrate species have become abundant (Brown, 1992), the effect on striped bass survival is unknown (see "Recent Conditions for the Estuarine Invertebrate Community").

Temperature. High water temperature has not caused substantial direct mortality of eggs and larvae and has not played a major role in the recent decline of young striped bass in the Sacramento-San Joaquin River system (Mitchell, 1987).

Rearing Juveniles

Many of the factors affecting abundance of eggs and larvae equally apply to the early juvenile stages and have been discussed in previous sections. Except for entrainment loss to diversions, the focus of this section is primarily on juvenile striped bass greater than 38 millimeters long.

Although year-class strength of the population is set early in the life cycle of striped bass, perhaps before the juvenile life stage, survival of juveniles ultimately determines the number of bass recruited to the adult population. Losses of juvenile striped bass are important in determining adult abundance (Kohlhorst et al., 1992).

Flow. Other than the relationship of Delta inflow to exports and Delta outflow to location of X2, flow likely has minimal direct effects on juvenile striped bass. Wendt (1987) showed that

flow in the lower San Joaquin River (along with export volume and striped bass abundance and size) was significantly correlated with entrainment losses at the CVP and SWP Delta pumping facilities. Lower San Joaquin River flow, however, is determined by Delta inflow and export, as is the location of X2 in the estuary (San Francisco Estuary Project, 1993). For juvenile striped bass, their location in the estuary could be more important than flow in determining the effect of other factors (i.e., entrainment). The location of juvenile striped bass relative to X2 was discussed previously under "Salinity."

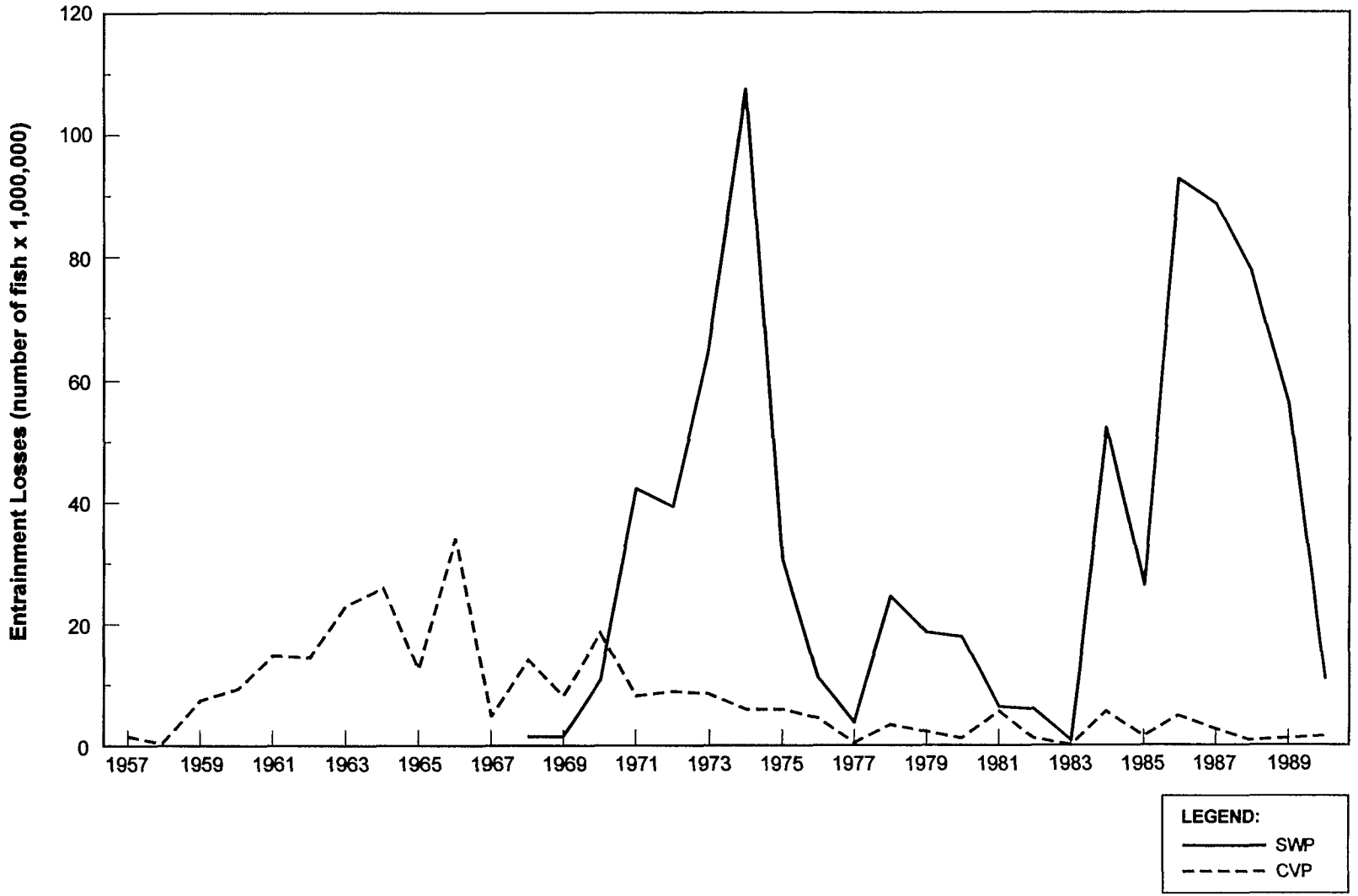
Entrainment. High adult abundance results from year classes that experience minimal late summer through winter losses to export pumping (Kohlhorst et al., 1992). The magnitude of juvenile striped bass losses is potentially affected by the abundance and distribution of juvenile bass and the magnitude of exports (Wendt, 1987; Kohlhorst et al., 1992).

CVP and SWP Delta Pumping Facilities. Millions of juvenile striped bass (greater than 20 millimeters long) are entrained in diversions at the CVP and SWP Delta pumping facilities each year (Figure II-73). Most entrained bass are lost, although 5 to 30 percent of all juvenile bass entrained at the SWP were salvaged and returned to the Delta alive (DFG, 1992b). The proportion salvaged depended on screen efficiency (a function of screen design and pumping volume), fish size, predation rates, and handling and trucking mortality. Salvage rates for the CVP facilities are estimated to exceed SWP salvage rates, primarily because of SWP losses in Clifton Court Forebay.

Entrainment loss of large bass has a more adverse effect on the population than the loss of the same number of small bass. Conversion to yearling equivalents for the SWP pumps shows the relative annual loss for all sizes combined, including eggs and larvae (Figure II-74). The bulk of entrainment loss is composed of early juvenile life stages (prior to 38 millimeters) and occurs during May-August (Figure II-75). Substantial losses of young-of-the-year bass have also occurred between November and January and could be a function of young bass distribution (i.e., relative to the location of X2).

Agricultural Diversions. Juvenile striped bass could have the swimming ability to avoid entrainment in small intakes, but losses have been documented. The magnitude of entrainment losses to agricultural diversions is currently unknown. Entrainment of juvenile bass in agricultural diversions is a function of diversion location (including location in the channel relative to distance from shore and depth); diversion volume and design; and distribution, size, and behavior of young striped bass. Most agricultural diversion takes place in the interior Delta where there are generally fewer bass; therefore, the effect could be less than for other diversions (Cannon, 1982). A major IEP study could provide future estimates of agricultural diversion effects.

Power Generation Facility Diversions. PG&E's Contra Costa and Pittsburg powerplants have fish screens, but their efficiency and the loss of juvenile bass could not be determined with available data. As discussed previously for eggs and larvae, losses to the powerplant diversions are likely substantial because of the location of the intakes in proximity to striped bass rearing areas (Cannon, 1982).



SOURCE:
DFG, 1992b.

FIGURE II-73

ENTRAINMENT LOSS OF JUVENILE STRIPED BASS IN DIVERSIONS BY THE SWP AND CVP DELTA PUMPING FACILITIES (1957-1990)

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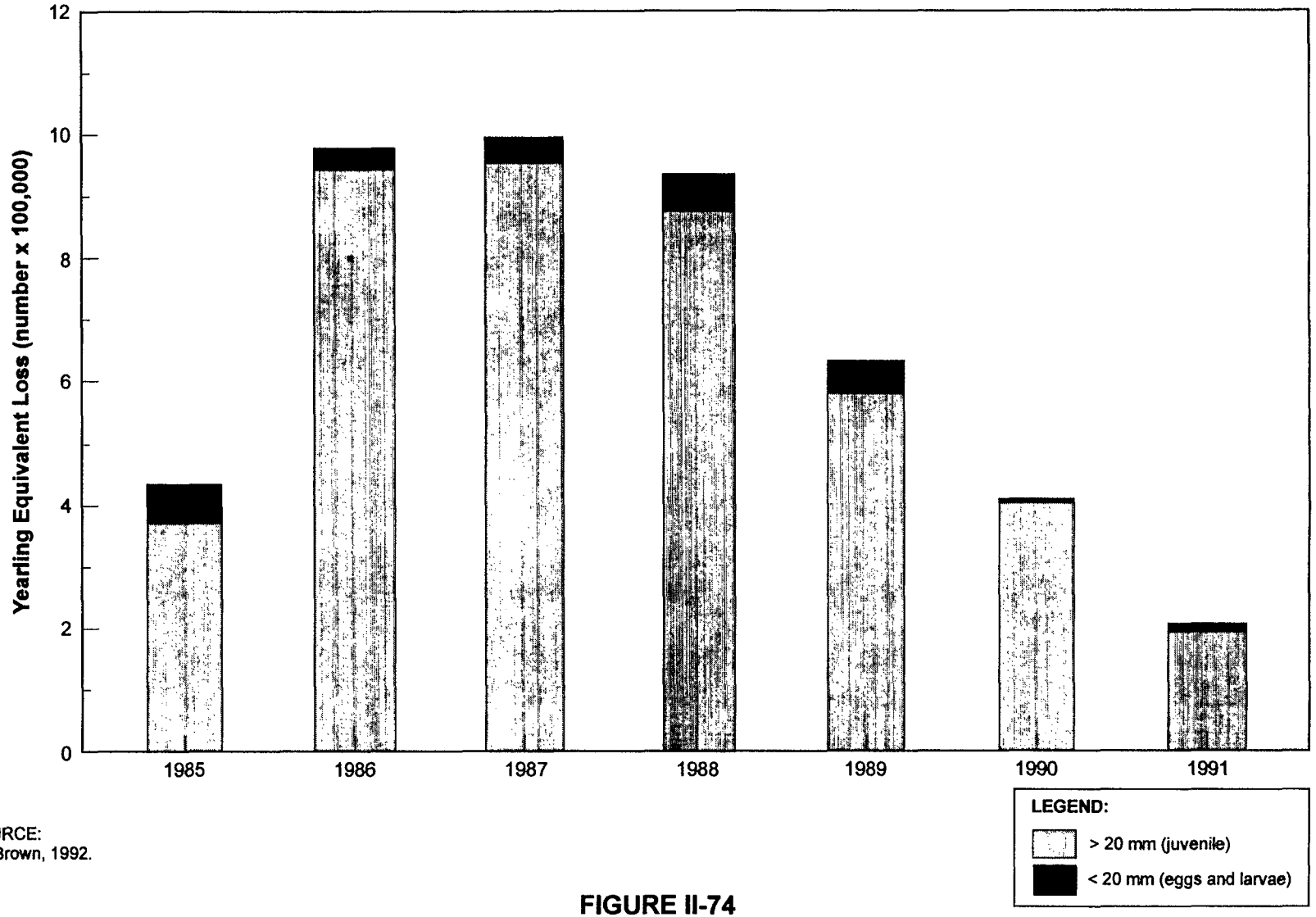

SOURCE:
Brown, 1992.

FIGURE II-74

YEARLING EQUIVALENT ENTRAINMENT LOSS OF STRIPED BASS AT THE SWP FACILITY (1985-1991)

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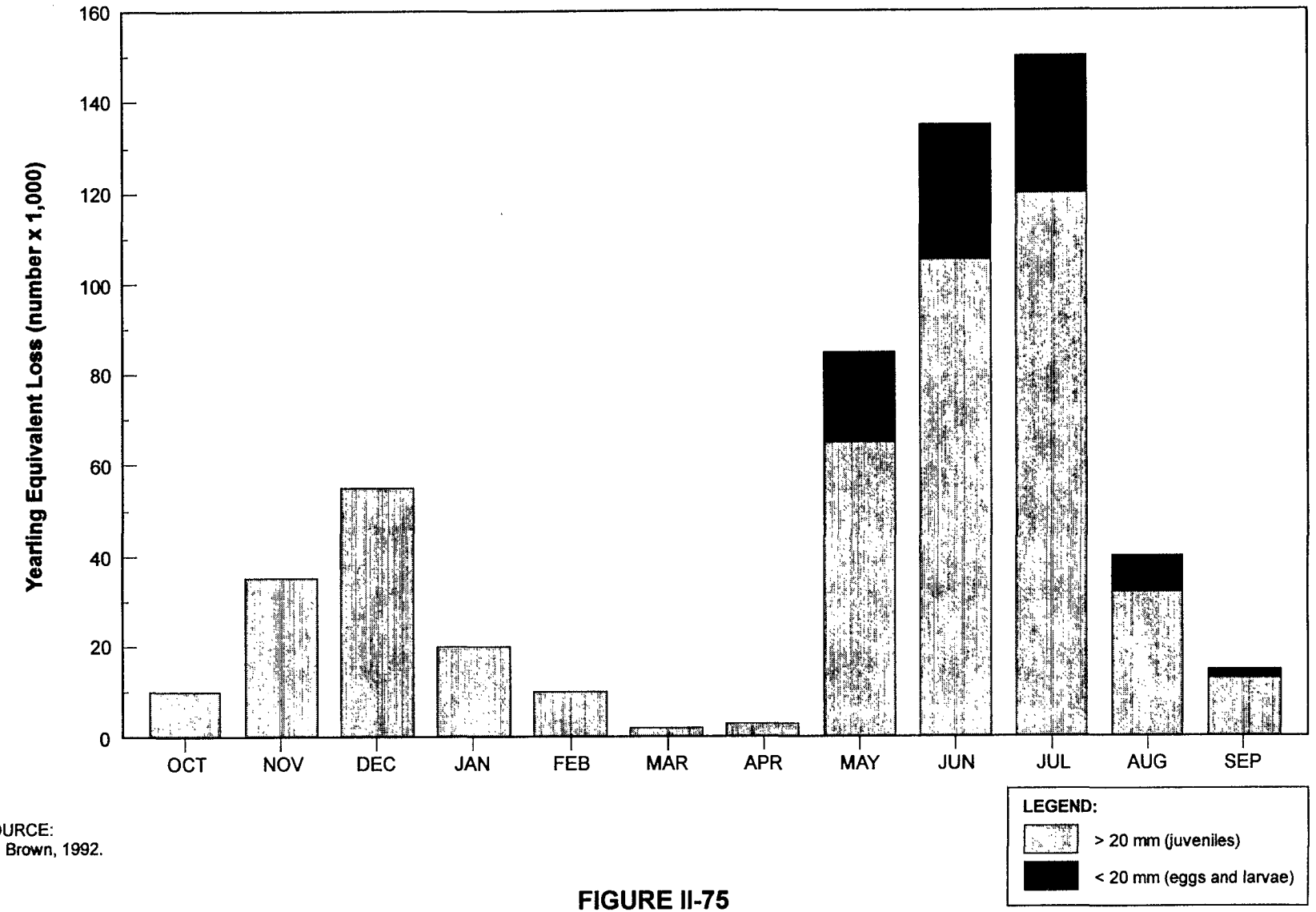

SOURCE:
Brown, 1992.

FIGURE II-75

AVERAGE ANNUAL ENTRAINMENT PATTERN FOR STRIPED BASS AT THE SWP FACILITY (1986-1991)

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Other Diversions. The fish screen at the North Bay Aqueduct diversion prevents entrainment of juvenile striped bass into the diversion. Indirect losses (i.e., predation and other factors associated with the screen) have not been determined. Relative to other diversions, the effect on juvenile bass is probably minimal because of the location relative to the main striped bass rearing areas.

Annual entrainment losses to CCWD's Rock Slough diversion could have historically exceeded 1 million juvenile striped bass (Odenweller, 1992). Sampling of striped bass entrainment, however, has not been consistent, and actual entrainment losses are unknown. The diversion is not located near the main striped bass rearing areas, but striped bass juveniles are abundant in some years in Old and Middle rivers, which transport water to the SWP and CVP Delta pumping facilities (as supported by high entrainment losses of juveniles at those facilities). The Rock Slough diversion draws water off the Old River channel.

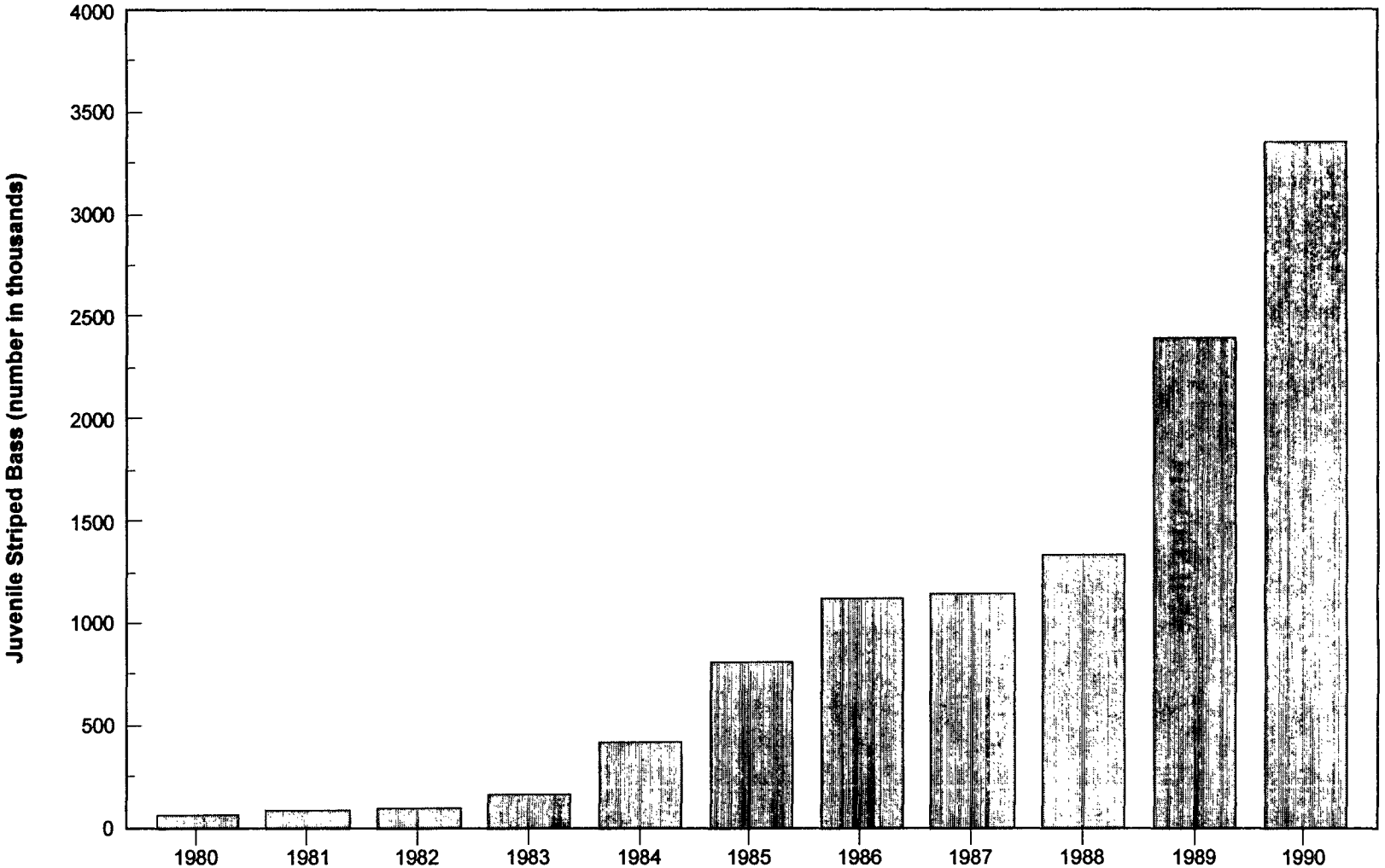
Habitat. Habitat factors for juveniles are the same as those discussed for adult and larval striped bass.

Toxic Materials. The effects of toxic materials on juveniles are the same as those discussed for larval striped bass.

Hatchery Releases. More than 3 million juvenile striped bass were released into the estuary in 1990 (Figure II-76). If habitat and food availability are limiting juvenile survival, release of hatchery juveniles could have a detrimental effect on the wild juvenile population. Available data do not indicate any detrimental effects of hatchery releases on wild striped bass survival. The release of hatchery-produced juvenile striped bass was discontinued by DFG after 1991 as part of the effort to avoid the risk of adverse effects on winter-run chinook salmon (Ford, pers. comm.). Low numbers (32,000) of juvenile striped bass were released to the Sacramento-San Joaquin Delta estuary in 1992 as part of the pen-rearing project.

Fishing. Illegal fishing could kill thousands of juvenile striped bass, possibly equivalent to the death of at least 125,000 legal-sized bass each year (Brown, 1987). This level of illegal fishing could equal or exceed the annual legal sport catch of approximately 85,000 adult striped bass (DFG, 1992a). As discussed previously, healthy fish populations can sustain high levels of fishing mortality, but the precipitous decline in adult striped bass abundance during the past 20 years indicates that the population is unhealthy (Figure II-56).

Prey Availability. As discussed for larvae, additional studies are needed to resolve questions on prey availability and the effect on striped bass survival. Juvenile striped bass (especially during their first year of life) feed primarily on the mysid *Neomysis* (see earlier "Life History" discussion). *Neomysis* declined in abundance during the 1970s, but declines were significant only during fall (Obrebski et al., 1992). Reduced abundance of prey could slow the growth of striped bass and increase mortality from predation.



SOURCE:
Delisle, pers. comm.

FIGURE II-76

RELEASES OF HATCHERY-REARED JUVENILE STRIPED BASS TO THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY (1980-1990)

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ONGOING MONITORING, ENHANCEMENT, AND HABITAT IMPROVEMENT ACTIONS

Striped Bass Management Program

Because of the popularity of the sport fishery, DFG has focused considerable attention on monitoring striped bass and developing a management plan. Ongoing monitoring, enhancement, and habitat improvement actions for striped bass in the Sacramento-San Joaquin Delta estuary are included in the Striped Bass Management Program (DFG, 1991).

The purpose of the Striped Bass Management Program is to describe ongoing and proposed actions designed to restore and improve the striped bass population. The guidelines require DFG to review the Striped Bass Management Program annually, receive public review and comment every two years, and revise the program every two years.

The specific striped bass resource goals are to stabilize, restore, and improve the striped bass fishery of the Sacramento-San Joaquin Delta estuary. Specific objectives are as follows:

- restore a self-sustaining Bay-Delta striped bass population to levels of more than 3 million adult fish by 2000;
- provide Bay-Delta striped bass that, when eaten, will not endanger human health through contamination from chemicals or trace metals; and
- provide striped bass angling, aesthetic, and educational use opportunities.

Major aspects of the Striped Bass Management Program are listed in Table II-9.

Striped Bass-Related Recommendations by the U.S. Environmental Protection Agency

The U.S. Environmental Protection Agency (USEPA) has proposed water quality standards for surface waters of the Sacramento River, San Joaquin River, San Francisco Bay, and Delta that will directly improve the habitat of striped bass (USEPA, 1994). The standards include:

- salinity criteria to protect the estuarine habitat and other designated fish and wildlife uses;
- salinity criteria to protect striped bass spawning habitat in the lower San Joaquin River; and
- salmon smolt survival index criteria to protect fish migration and cold freshwater habitat uses in the estuary (i.e., additional spring Delta inflow and reduced diversions).

Similar standards are established in the State Water Resources Control Board's draft Water Quality Control Plan (1995).

TABLE II-9

SUMMARY OF THE STRIPED BASS MANAGEMENT PROGRAM

		Status	Agency
I.	Develop Public Participation in Plan Preparation and Implementation		
	A. Submit draft of the plan to public, private, and government entities	U	S
	B. Develop recommendations for tasks to be conducted by public, private, and government entities	U	S
	C. Prepare information to increase public awareness	U	S
II.	Resolve Problems Detrimental to Striped Bass		
	A. Minimize entrainment losses of bass eggs, larvae, and young in Delta water diversions, including diversions by:		
	1. SWP Delta pumping facilities: two-agency fish protective agreement	P, U	S
	2. CVP Delta pumping facilities: agreement between Reclamation and DFG to reduce and offset direct fish losses	P, U	S, F
	3. Contra Costa Water District	P, U	S, F
	4. PG&E: operating permit for PG&E from the Central Valley Regional Water Quality Control Board (Contra Costa powerplant) and the San Francisco Bay Regional Water Quality Control Board (Pittsburg powerplant)	P, U	S, P
	5. Agriculture	P, U	S, F, P
	B. Eliminate reverse flows in the Delta east of Antioch when bass eggs and larvae are present (same participants as in IIA, above)	P	S, F, P
	C. Increase Delta outflow in spring and early summer (same participants as in IIA, above)	P, U	S, F, P
	D. Increase residence time in secondary Delta channels (i.e., not including the Sacramento and San Joaquin rivers) (same participants as in IIA above)	P	S, F, P
	E. Reduce quantities of toxic materials contained in municipal, industrial, and agricultural discharges		
	1. DFG Aquatic Toxicology Laboratory	P, U	S
	2. DFG Regions 2 and 3 are to continue monitoring and evaluating waste discharges	P, U	S
	F. Reduce bass losses during fish screen salvage, handling, and fish release operations at SWP and CVP facilities		
	1. DFG assumes operations of the fish protection facilities	P, U	S
	2. Upgrade fish holding facilities	P, U	S, F
	G. Install fish screens on larger Delta agricultural diversions		
	1. Roaring River diversion in Suisun Marsh	P	S
	2. Other	P	S, F, P
	H. Improve existing fish screens	P	S, F

TABLE II-9. CONTINUED

	Status	Agency
I. Consolidate and relocate Delta agricultural diversions to areas of lower bass abundance	P	S
J. Reduce predation at major water intake structures		
1. SWP Delta pumping and fish facilities and Clifton Court Forebay	P	S
2. CVP Delta pumping and fish facilities	U	S, F
K. Curtail channel dredging and prohibit dredge spoil disposal in Delta channels		
1. DFG review of COE dredging permits	U	S, F
2. DFG review of COE dredging spoils disposal permits	U	S, F
L. Eliminate future Bay-fill projects	U	S
M. Reduce illegal take and poaching		
1. Resolve illegal commercialization	P	S
2. Increase law enforcement activities	U	S, F
N. Reduce bass diseases and parasitic infestations	N	
O. Reduce the annual summer bass die-off near Carquinez Strait	P	S
P. Minimize kill of small bass by the commercial bay shrimp fishery	P	S
Q. Halt introductions of exotic aquatic organisms from maritime shipping		
1. Federal regulations and legislation to restrict discharge of ship ballast	P	S, F
2. High seas exchange of ballast	P, U	S, F
III. Resolve Problems of Human Use of Striped Bass		
A. Continue hatchery-reared striped bass stocking program	U	S, P
B. Improve pond production at state hatchery	N	
C. Maintain sport fishing and commercial regulations to protect the resource and allow angling opportunities	P, U	S
D. Reduce methyl mercury contamination of adult bass	U	S, F
E. Reduce diseases and parasitic infestations	N	S
F. Reduce tainting of bass flesh	N	
IV. Conduct Fishery and Environmental Studies		
A. Develop techniques to better detect large masses of bass eggs and larvae as they drift downstream		
1. Reclamation	U	F
2. DFG egg and larval survey	U	S
B. Continue survey of annual production of bass eggs, larvae, juveniles, and adults	U	S, F, P
C. Improve annual larval bass growth and mortality estimates	U	S

TABLE II-9. CONTINUED

	Status	Agency
D. Survey waste discharges to locate sources of toxic materials in the estuary		
1. Rice herbicides and insecticides	U	S, F, P
2. Colusa Basin Drain studies	U	S, F, P
3. Toxics and trace metals studies	U	S, F, P
E. Continue testing impacts of toxic materials on young bass and their food organisms	U	S
F. Develop a striped bass population model to evaluate factors affecting the bass population abundance	P, U	S, F, P
G. Analyze bass food production in spring and determine if food is limited	U	S, F
H. Extend toxicology testing	U	S, F
I. Improve DFG ability to estimate striped bass egg and larval entrainment losses	U	S, F
J. Compare prey suitability of introduced and native copepods	N	S
K. Determine the effect of toxic materials on egg viability	N	P, S
L. Continue monitoring abundance of fish, invertebrates, and aquatic plants as indicators of adverse conditions for striped bass	U	S
M. Evaluate merits of adding Atlantic coast bass stocks for improved growth and Sacramento-San Joaquin stock condition	N	
N. Develop improved model of striped bass mortality	N	S
O. Evaluate bass predation	U	S, F
P. Determine results of stocking hatchery-reared striped bass		
1. Stocking of tagged bass	U	S
2. Creel census	U	S
Q. Evaluate new stocking locations for tagged bass	N	
R. Evaluate potential of bass "grow-out" facilities	U	S

SOURCE:
DFG, 1991.

LEGEND:
Status
U = construction/operation underway
P = planning underway
N = no activity
Agency
S = state
F = federal
P = private

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Endangered Species Act Consultations

Prior to continuation of the striped bass stocking program, DFG was asked by NMFS to initiate Section 10 consultation under the federal ESA, specifically in regard to the potential effect on the endangered winter-run chinook salmon (Ford, pers. comm.). DFG was also asked by the Service to initiate Section 10 consultation on the threatened delta smelt. The results of the consultation will determine the immediate future of the striped bass stocking program.

RECENT CONDITIONS FOR AMERICAN SHAD

STOCK ABUNDANCE AND DISTRIBUTION

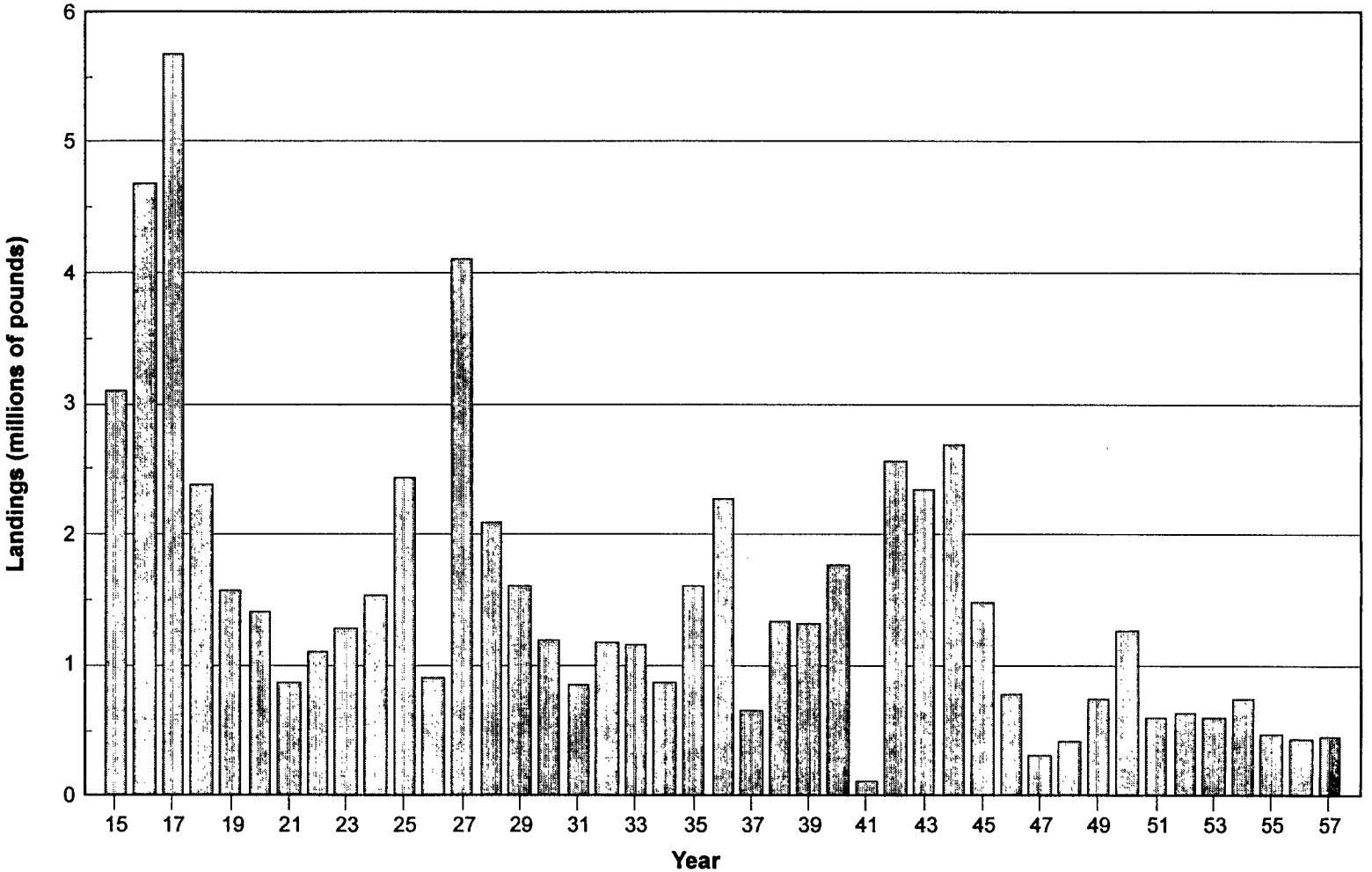
Historical Population Trends (1850-1966)

American shad (*Alosa sapidissima*) are native to the east coast of the United States. Juvenile shad were transported from New York to California in 1871, when approximately 10,000 juveniles were released in the Sacramento River near Tehama (Painter et al., 1980). An additional 824,000 juvenile shad were introduced into California from 1873 to 1881 (Skinner, 1962). The shad quickly multiplied and were found as far north as the Columbia River in Washington by 1880 (Fry, 1973). A commercial fishery for shad developed by 1879; by 1886, the State Board of Fish Commissioners estimated that 1 million mature fish were taken (Skinner, 1962).

Commercial catch data from 1899 to 1914 are limited but indicate that commercial landings ranged from 620,891 to 1,169,000 pounds (Skinner, 1962). Commercial landings from 1915 to 1945 ranged from approximately 0.1 to 5.5 million pounds; however, landings below 1 million pounds were rare (Figure II-77). After 1945, commercial shad landings exceeded 1 million pounds only once (Figure II-77). The commercial gill net fishery in the Sacramento-San Joaquin Delta estuary was eliminated through legislation in 1957 (Skinner, 1962).

It is not known when sport fishing for shad first occurred, although some angling was reported in the 1930s and 1940s (Painter et al., 1980). After 1950, sport fishing for shad became extremely popular. One popular method of taking shad, called "bumping," was conducted from boats by using hand-held nets. Anecdotal information indicates that 2,500 anglers operating from a single recreational fishing business caught 30,000 shad using this method in 1954 (Skinner, 1962). No reliable sport catch records are available to determine the relative proportion of the fishery caught by sport anglers; however, by the mid-1960s, an estimated 100,000 angler days per year were spent sport fishing for shad (Painter et al., 1980).

Analyzing commercial and sport catch data to determine shad abundance is difficult because commercial landings were influenced more by market, economic, and angling factors than by shad abundance (California Bureau of Marine Fisheries, 1949).



SOURCE:
Skinner, 1962.

FIGURE II-77

COMMERCIAL LANDINGS OF AMERICAN SHAD (1915-1957)

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C—081517

Current Population Trends (1967-1991)

Currently, American shad are found on the Pacific coast from Todos Santos Bay in Baja California northward to Alaska. In California, anadromous shad populations are found seasonally in the Sacramento and San Joaquin rivers and Delta; the Feather, Yuba, and American rivers; the Mokelumne and Stanislaus rivers; and the Klamath, Russian, and Eel rivers (Figure II-78). In the Sacramento River drainage, shad migrate up the Sacramento River as far upstream as the Red Bluff Diversion Dam, the Feather River as far upstream as Oroville, the Yuba River as far upstream as Daguerre Point Dam, and the American River as far upstream as Nimbus Dam. Shad are occasionally seen upstream of the RBDD and Daguerre Dam. Smaller shad runs occur in the Mokelumne River, Stanislaus River, sloughs of the south Delta, and the San Joaquin River (Stevens, 1972; Moyle, 1976). A landlocked population also exists in Millerton Lake (Fresno and Madera counties).

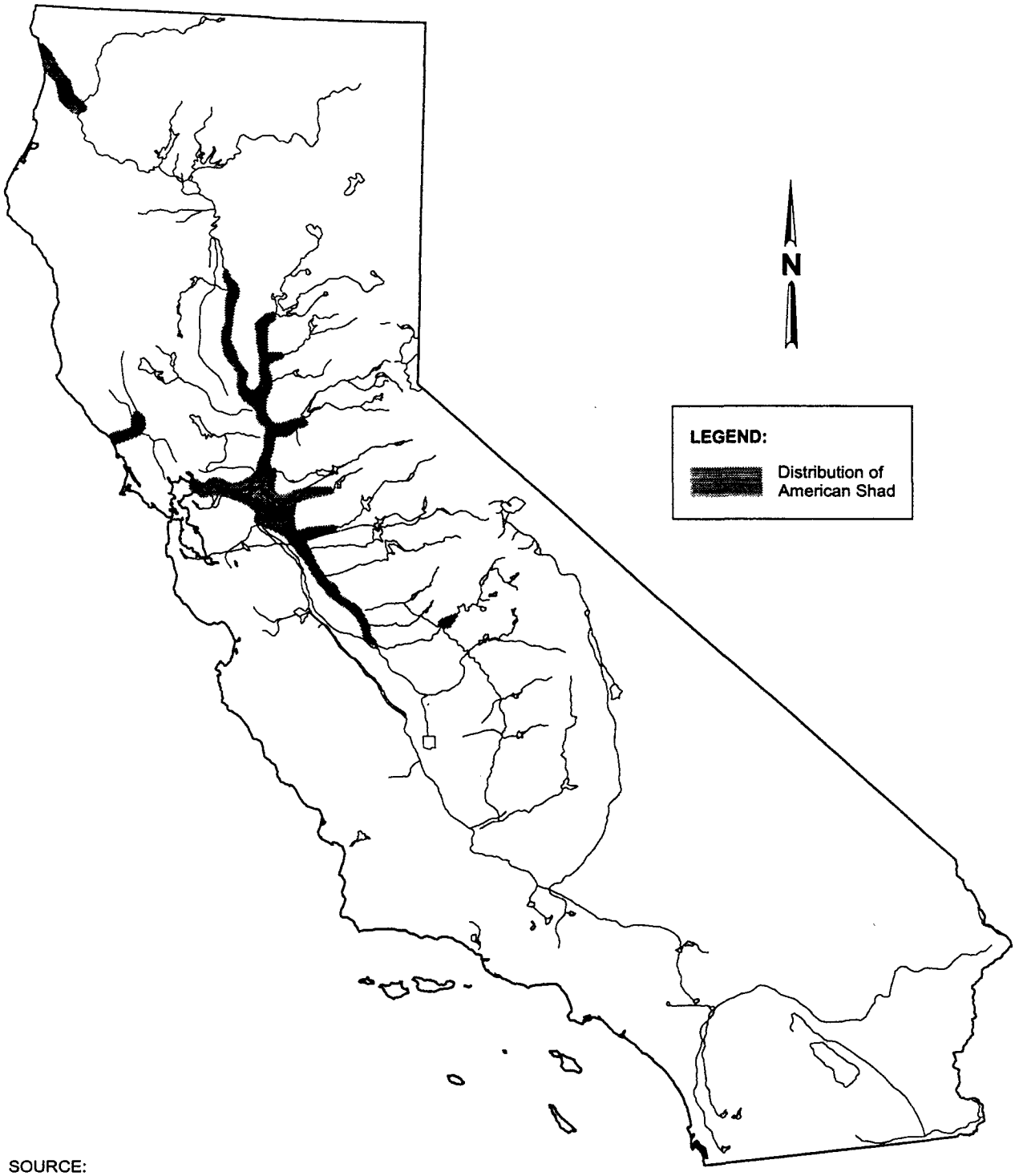
Sport fishing for shad continues to be popular in the Sacramento, American, Feather, and Yuba rivers, with a smaller, less consistent fishery in the San Joaquin River and its tributaries (Painter et al., 1980). Evidence suggests that both shad catch and angler effort have declined; however, it is unclear whether this reflects of a change in shad abundance or angler interest. Between 1976 and 1978, the mean annual sport catch ranged from 86,200 to 152,000 adult shad, and angling effort ranged from 35,000 to 55,000 angler-days (Meinz, 1981). During this period, 60 percent of the annual catch was taken from the Sacramento River (Meinz, 1981).

DFG conducted population estimates in 1976 and 1977 using mark-recapture techniques to estimate the size of the spawning run of adult shad. Fish were captured using gill nets and marked with tags that ensured a reward to anglers as an incentive to return the tags. These studies were the only specific attempt to estimate adult shad abundance. DFG estimated the shad population at 3.04 million adults and 2.79 million adults in 1976 and 1977, respectively. DFG further estimated that these population numbers are approximately one-third to one-half the number present during 1917, based on commercial catch data. (DFG, 1987.)

Fall mid-water trawl (MWT) surveys provide a measurement of young-of-year abundance in the Delta between September and December (Figure II-79). These annual surveys have been conducted since 1967 and provide the longest, most accurate measurement of shad abundance. The peak abundance index was 5,386 in 1982; the lowest was 334 in 1972. The index averaged 2,070 during the 1967-1991 period, and the median index was 1,596 in 1985. However, according to Herbold et al. (1992), the fall MWT begins as most shad have begun to migrate out of the Delta. Therefore, the fall MWT could be underrepresenting the actual shad abundance.

LIFE HISTORY

With only a few exceptions, American shad are anadromous, spending most of their lives in the ocean and returning as adults to spawn in freshwater rivers. Spawning occurs in riverine habitats with moderate currents sufficient to keep the eggs suspended above the bottom. The young can migrate downstream to the ocean soon after hatching and linger in the Delta for several weeks to several months or rear for several months in the major rivers before moving downstream.



SOURCE:
Moyle, 1993.

FIGURE II-78

DISTRIBUTION OF AMERICAN SHAD IN CALIFORNIA

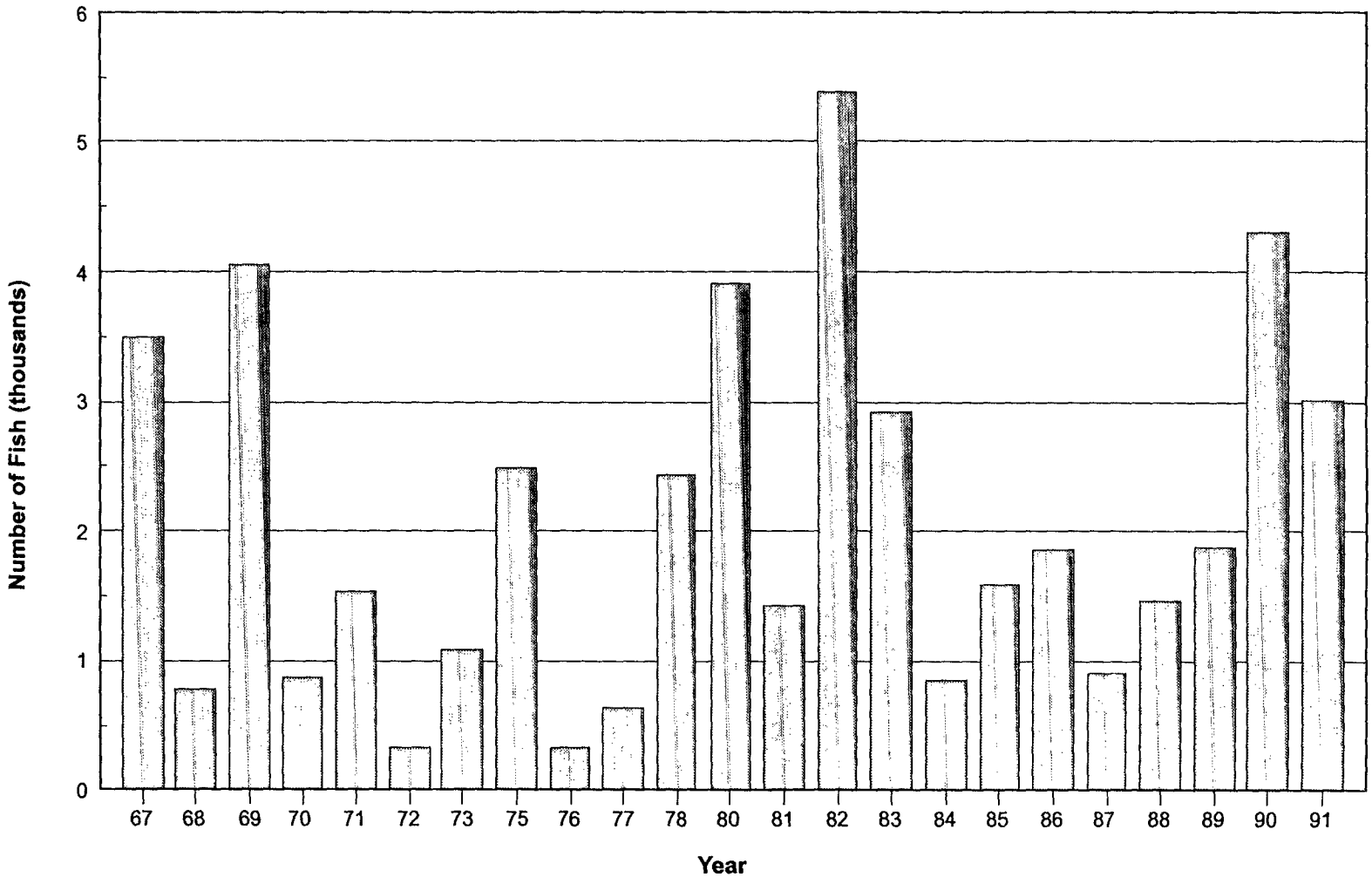


FIGURE II-79

ANNUAL ABUNDANCE INDEX OF YOUNG-OF-YEAR AMERICAN SHAD IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY (1967-1991)

C-081520
C—081520

Information presented on American shad life history is based primarily on Moyle (1976), Painter et al. (1980), Stier and Crance (1985), Wang (1986), and Jones & Stokes Associates (1990). American shad life history is depicted in Figure II-80.

Adults

American shad become sexually mature while in the ocean at an average age of three to five years; the oldest shad on record lived to be 11 years old (Painter et al., 1980). At maturity, male shad typically average 3 pounds, and female shad average almost 4 pounds; shad as large as 6 to 8 pounds are rare (Skinner, 1962). Although shad are strongly anadromous, they are capable of surviving and reproducing while landlocked in freshwater reservoirs (Moyle, 1976). In California, all American shad except the Millerton Lake shad populations have an anadromous life cycle.

Little is known about the shad ecology and behavior in the ocean. As stated earlier, shad are found in the Pacific Ocean from Baja California to Alaska; however, they are seldom found south of Monterey, California (Fry, 1973). Their wide distribution along the Pacific coast suggests that shad in the Pacific Ocean could exhibit migrational patterns similar to those of Atlantic Ocean shad (Moyle, 1976; Painter et al., 1980).

Unlike shad on the Atlantic coast, adult shad in the Delta feed while in freshwater, probably because of the abundance of large zooplankters. However, not all adult shad feed while in the Delta, and most feeding ceases once they enter the main rivers (Moyle, 1976). In the Delta, adult shad feed primarily on opossum shrimp (*Neomysis mercedis*) and also on copepods, cladocerans, and amphipods (*Corophium* sp.) (Moyle, 1976). The presence of these zooplankters in shad stomachs appears to be directly related to zooplankton concentrations in the Delta (Stevens, 1966). On occasion, adult shad have been known to prey on clams and fish larvae.

Migrating Adults. Adult American shad begin their spawning migration as early as February; however, most adults do not initiate migration into the Delta until March or early April (Skinner, 1962). Typically, most migrating adults need three months (March-May) to pass through the Sacramento-San Joaquin Delta estuary (Painter et al., 1980). The timing of shad migration appears to be regulated by water temperatures in the ocean and natal rivers. Typically, adult shad do not enter freshwater until water temperatures approach 52 degrees Fahrenheit.

Peak migration into spawning habitats occurs when water temperatures are much higher (59 to 68 degrees Fahrenheit), usually in late May or early June (Moyle, 1976). During studies in the western Delta (1976-1977), DFG tagged the majority of migrating shad when water temperatures were between 57 and 66 degrees Fahrenheit (Painter et al., 1980). Despite the importance of temperature, studies on both the Feather River (Painter et al., 1977, 1980) and the Yuba River (Jones & Stokes Associates, 1990) suggest that increased flows, not water temperatures, were the primary factors responsible for attracting shad into these streams. Migration appears to decline after water temperatures exceed 68 degrees Fahrenheit, usually in early July (Moyle, 1976). Peak migration in the Sacramento River upstream of the Feather River occurs in May, and angling surveys indicate that peak migration in the Feather and Yuba rivers occurs during June (Stevens, 1972; Jones & Stokes Associates, 1990).

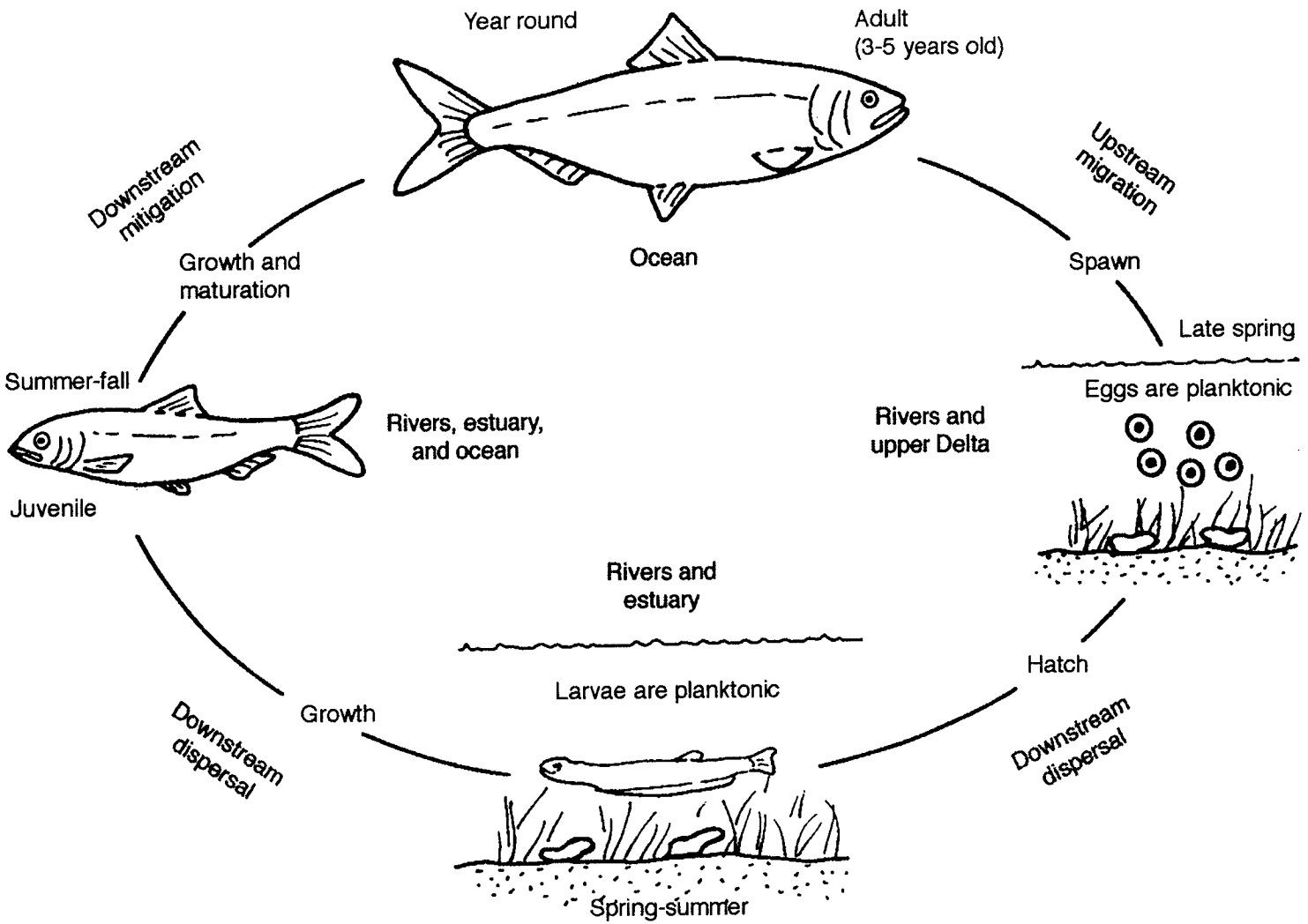


FIGURE II-80

LIFE HISTORY OF THE AMERICAN SHAD

C-081522
C—081522

Unripe, male shad constitute the majority of the early run, and smaller, unripe females are known to precede the larger, later migrating ripe females (Moyle, 1976; Painter et al., 1980). The ratio of males to females on the Yuba River was found to be 1:1 during the first half of the season and over 3:1 during the last half of the season (Jones & Stokes Associates, 1990). Most of the migrating shad are composed of three-year-old males and four-year-old females and range in size from 12 to 30 inches (Wixom, 1981). Approximately 70 percent of the shad run in central California is composed of fish spawning for the first time (i.e., virgin spawners) (Painter et al., 1980).

American shad ascend most of the major rivers in the Sacramento-San Joaquin River drainage; the greatest proportion of the population is found in the Sacramento River drainage (Skinner, 1962). The upstream limit of shad migration is dictated by impassable barriers such as dams and water diversion structures. Adult shad do not appear to use fish ladders to any appreciable extent, although it is believed that passage over these barriers is possible given proper hydraulic conditions (Skinner, 1962).

Spawning Adults. American shad reach sexual maturity at three to five years; most males reach maturity at three to four years, and most females become sexually mature at four to five years (Painter et al., 1980). However, some shad have been found to spawn as young as two years and as old as seven years.

American shad spawn exclusively in freshwater, although spawning could be possible in brackish water (Wang, 1986). American shad spawn in the main channels of the Sacramento River from Red Bluff downstream to Hood; the American, Feather, and Yuba rivers; the lower reaches of the San Joaquin River; and the Mokelumne and Stanislaus rivers (Wang, 1986). It is not known if shad return to their natal rivers to spawn.

Although fish can spawn at any time of day, spawning usually occurs at night as a mass affair, often among small schools. Spawning is initiated when a male swims alongside a female and the two adults swim rapidly side by side. The males fertilize the eggs as the female releases them into the water column. Each fish spawns repeatedly; some survive the spawn and return the following year after emigrating to the ocean. Post-spawning adults emigrate through the Delta and Suisun Bay as late as August and September. Spawning mortality appears to be greater at higher water temperatures, especially above 68 degrees Fahrenheit (Moyle, 1976).

Water temperature appears to be the most important factor determining the timing of shad spawning. Water temperatures could be as low as 50 degrees Fahrenheit, but the general range appears to be 55 to 75 degrees Fahrenheit. The optimum range is likely 62 to 68 degrees Fahrenheit (Skinner, 1962). In the Feather River, shad spawning does not occur until water temperatures reach 60 degrees Fahrenheit and peaks at 70 degrees Fahrenheit (Painter et al., 1977). In the Yuba River, shad spawning did not occur until mean daily temperature reached 61 degrees Fahrenheit (Jones & Stokes Associates, 1990). Most shad spawning occurs in May and June.

Shad typically spawn over sand to gravel substrates in depths of 3 to 30 feet (Painter et al., 1980). Jones & Stokes Associates (1990) concurred with these depth findings but found spawners concentrated within a specific range of mean water velocities of 1.5 to 2.4 feet per second on the

Yuba River. Because shad spawning is pelagic (in open water) and not limited to a fixed site, as is salmon and steelhead spawning, it can occur repeatedly at the same locations without any apparent adverse effect on egg survival.

Planktonic Eggs and Larvae

American shad eggs are slightly heavier than water and are suspended in the water column by the slightest current. Although shad eggs can be found throughout the water column, the greatest concentration appears to be near the river bottom. The eggs drift with the current and hatch in three to six days at water temperatures of 74 and 57 degrees Fahrenheit, respectively (Stevens, 1972). Although hatching occurs sooner at higher water temperatures, egg survival is reduced.

Lengths of larval shad range from 5.7 millimeters to 10 millimeters at hatching and the shad grow rapidly, tripling their length in the first month. The newly hatched larvae are pelagic, are most abundant at the water surface, and feed on zooplankton within four to five days of hatching (Painter et al., 1980; Wang, 1986). Larval shad initially prey predominantly on cladocerans but increasingly feed on ostracods, insects, insect larvae, and copepods as they grow. Shad larvae usually consume food items that are most readily available (Painter et al., 1980). Growth is rapid, and larval stages last approximately four to six weeks. Newly hatched larvae are found downstream of spawning areas and can be rapidly transported downstream by river currents because of their small size.

Rearing Juveniles

Larval stages last approximately 30 to 40 days, and the young shad have developed adult features and are classified as juveniles at lengths of 24 to 28 millimeters (Painter et al., 1980). Depending on water temperature and food availability, young-of-year shad in the Delta are an average length of 30 millimeters in July, 81 millimeters in September, and 114 millimeters in November (Stevens, 1972). By the time they enter saltwater, shad range from 80 to 180 millimeters long.

Season-long rearing of juvenile shad occurs in the Mokelumne River in the vicinity of the DCC to the San Joaquin River, the lower Sacramento River below Knights Landing, the Feather River below Yuba City, and the Delta. No rearing occurs in the American and Yuba rivers. (Painter et al., 1980.)

While in the Delta, juvenile shad are opportunistic feeders and prey on *Neomysis* spp., copepods, amphipods, chironomid midge larvae, and surface insects (Moyle, 1976). Some juvenile shad apparently remain in the Delta for up to a year or more before emigrating to the ocean. However, it appears that all juvenile shad eventually emigrate to the ocean because immature shad greater than 200 millimeters long are rarely caught in the Delta (Moyle, 1976). Seaward migration of juvenile shad in the Delta begins in late June and continues through November, peaking between September and November (Stevens, 1972; Painter et al., 1980). Little is known about the ecology and behavior of juvenile shad once they enter the ocean.

FACTORS AFFECTING ABUNDANCE

Since the early 1900s, the shad population is believed to have declined gradually. Evidence suggests that this decline is attributable primarily to human-caused factors, such as water development, that likely continue to affect abundance. The rapid increase in American shad abundance and distribution shortly after their introduction indicates that habitat and environmental conditions historically were ideal for shad. Although the rivers and Delta were largely leveed and many of the wetlands were diked and filled soon after the introduction of shad, the Delta environment and river flow patterns were relatively unmodified compared to current conditions.

Undoubtedly, many factors have combined to decrease California's American shad populations, and historical conditions for successful shad spawning, growth and development, and emigration have been impaired. Although knowledge of American shad ecology and specific factors limiting shad abundance in California have been limited primarily to DFG's Feather River studies in the mid-1970s, additional information being developed in the context of other studies will assist in understanding factors affecting shad abundance.

Adult Migration and Spawning

Flow Relationships. Although shad on the East Coast are known to exhibit a tendency to spawn in their natal streams, river flow appears to be largely responsible for the distribution of virgin (first-time) spawners in the Sacramento River system (Painter et al., 1980). Within the system, the relative magnitude of tributary flow to the mainstem rivers apparently determines the relative percentage of virgin spawners using those tributary rivers (Painter et al., 1980).

Based on 1975-1978 data, flow relationships have been developed indicating that virgin shad are attracted to the upper Sacramento, Yuba, and American rivers when flows in these rivers relative to the Feather and Sacramento rivers, respectively, are relatively large during May and June (Table II-10) (Painter et al., 1980). Such a relationship does not exist in the Feather River, however, where it is believed the longer rearing time allows juveniles to become imprinted for homing (DFG, 1987). The lack of such a relationship has recently been verified using 1990-1993 shad data from the Sacramento, Feather, and Yuba rivers (Sommer, pers. comm.). Strong relationships also exist in the 1975-1978 data between the percentage of virgin shad attracted into the upper Sacramento, Yuba, and American rivers and total May-June flows in these rivers, without consideration of the flow percentages between any two rivers (Jones & Stokes Associates' file data).

Despite these strong relationships, the effect of the relative distribution of virgin spawners on young-of-year shad abundance and overall shad populations is unknown. Specifically, it is unclear whether survival from shad spawning in the major tributaries is greater than from spawning in the Sacramento River. It is unknown whether young-of-year abundance is a function of the distribution of flows (and therefore spawners) or increased flows in general. For instance, fall MWT survey data suggest that young-of-year abundance is greater during years with high freshwater Delta inflow. However, during years of high Delta inflow, relatively more young-of-year shad could be washed downstream into the Delta compared to years with lower Delta inflows, causing the abundance index to be higher than it actually is Adult passage into

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TABLE II-10

RELATIONSHIP BETWEEN RIVER FLOW AND VIRGIN SHAD ABUNDANCE IN THE UPPER SACRAMENTO, FEATHER, YUBA, AND AMERICAN RIVERS (1975-1978)

Year/ Coefficient	Upper Sacramento		Feather		Yuba		American	
	%Q (1)	%V	%Q (2)	%V	%Q (3)	%V	%Q (4)	%V
1975	65.8	72.7	34.2	62.7	33.8	70.45	19.0	96.8
1976	79.5	90.8	21.5	29.0	10.3	32.61	10.5	71.7
1977	76.8	85.4	23.2	82.2		N/A	5.4	58.8
1978	60.1	63.9	39.9	80.1	38.9	80.06	18.2	91.9
Coefficient of Correlation	0.9971		0.5020		0.9997		0.9978	

NOTES:

Percent virgins (%V) each year from Wixom (1981). Percent Virgins are defined as the percentage of first time spawning shad in the spawning population.

Percent flow (%Q) based on mean May-June flows, U.S. Geological Survey data. The percent flow is the comparison of flow for a given river, relative to the combined Sacramento and Feather rivers.

Predictive equations (y = percentage flow, x = percentage virgins) are as follows:

Upper Sacramento River	$y = 1.3284\ x - 15.5171$
Feather River	$y = 1.3991\ x + 21.9482$
Yuba River	$y = 1.6440\ x + 15.5572$
American River	$y = 2.7208\ x + 43.6819$

(1) Upper Sacramento River flow as a percentage of total flow below confluence with the Feather River.
(2) Feather River flow as a percentage of Sacramento River flow below confluence with the Feather River.
(3) Yuba and Feather River flow as a percentage of Sacramento River flow below confluence with the Feather River.
(4) American River flow as a percentage of Sacramento River flow below confluence at Sacramento.

SOURCE:
Painter et al., 1980.

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tributary streams is also important in determining the distribution of spawning adults. Relatively low flows during spring could reduce or restrict adult access to spawning areas in tributary rivers at riffle habitats. Critical riffle habitats exist when decreasing flows cause water depths to be too low to pass migrating adult shad. Reduced or restricted access to spawning areas could cause adult shad to spawn where habitat or environmental conditions are less favorable, thereby reducing reproductive success.

Temperature. The timing of spawning migrations correlates with water temperatures.

Upstream migration of adult shad generally occurs as water temperatures increase during spring. However, adult shad could discontinue their upstream migration if water temperatures exceed 68 degrees Fahrenheit (Stier and Crance, 1985). Furthermore, water temperatures exceeding 68 degrees Fahrenheit are known to increase mortality among post-spawning adults (Moyle, 1976). The initiation of spawning also correlates with water temperatures; spawning is generally delayed until water temperatures exceed 60 degrees Fahrenheit.

Water temperatures in tributary rivers can be affected by storage levels in upstream reservoirs and changes in the timing and magnitude of reservoir releases. Large volumes of cold-water releases (i.e., deep-water releases from reservoirs during periods of high water surface elevations) can create relatively cooler water temperatures in tributary rivers. Similarly, low instream flows and warm-water releases (i.e., surface releases during periods of low reservoir water surface elevations) can cause relatively warmer water temperatures downstream of reservoirs.

Habitat. Dams could have restricted access to upstream spawning and rearing habitats and modified or reduced freshwater flows that provide the necessary conditions for optimal shad migration, spawning, egg incubation, and rearing. Diking and dredging have eliminated an estimated 96 percent of the wetland habitats in the lowland areas (50 Code of Federal Regulations, Part 17). Diking and filling of wetlands in the Delta have restricted shad habitat and, in combination with reductions in freshwater flows, have reduced tidal mixing and overall estuary productivity. Although many of these modifications occurred before the initial introduction of shad in California, more recent anthropogenic factors could exacerbate the effects of wetland filling and diking, thereby contributing to the decline in shad abundance.

Toxic Materials. Adult American shad could be affected by toxic materials entering the Sacramento-San Joaquin River system from agricultural runoff, discharge of industrial and municipal waste, and runoff from non-point sources (e.g., urban stormwater runoff). In the Delta, pollutants of particular concern are trace elements (e.g., selenium, copper, cadmium, and chromium) and agricultural chemicals and their derivatives, which are used extensively in the Central Valley.

Although no specific information is available on how toxic materials are affecting shad populations in the rivers or Delta, the effects of toxics on adult shad could be similar to known effects on other Delta fish species. For instance, toxics exceeding levels considered safe for human consumption have been found in tissue samples of adult striped bass and appear to reduce fecundity in female striped bass (see "Recent Conditions for Striped Bass"). Although toxic materials likely have an adverse effect on adult shad, no evidence exists to suggest that these materials are causing a decline in shad abundance. Because American shad spend most of their

lives in the ocean rather than in the estuary (as striped bass do), they have been less exposed to toxic materials. Toxic materials could affect adults either directly or indirectly, thereby reducing reproductive success and survival.

One of the complicating factors in understanding the effects of toxics on ecological processes in the estuary is the complex distribution of "hot spots" (i.e., areas with high concentrations of toxics), both spatially and temporally (Herbold et al., 1992). These hot spots could cause adults to avoid biologically important habitat or alter movements.

Salinity. American shad have a wide range of salinity tolerances; however, studies suggest that adults require two to three days to adapt to freshwater (Stier and Crance, 1985). In addition, shad spawn only in freshwater; however, there does not appear to be a specific distance upstream of brackish water required for spawning to occur (Painter et al., 1980).

Upstream water storage projects, diversions, and Delta export pumping have reduced and modified Delta outflows. Reduced outflow increases salinities in Suisun Marsh, Suisun Bay, and the lower Delta. Consequently, changing salinities in the estuary could influence migrating adult shad and Delta spawning. No evidence exists to suggest that changing salinities in the estuary are decreasing shad abundance.

Sport Fishing. American shad have not been harvested commercially in the Delta since 1957. Currently, shad are harvested only by sport anglers. Angler surveys in 1977 and 1978 determined that sport anglers harvested 79,000 and 140,000 shad, respectively (DFG, 1987). Although the present sport harvest limit is 25 shad per day, most sport anglers typically release all or most of their catch (DFG, 1987). Although it is unknown if caught-and-released fish have significantly higher prespawning mortality, shad are delicate fish and the slightest physical injury usually results in death (Skinner, 1962). More recently, it appears that more shad caught in the Feather River are being kept, and many anglers catch and keep their limits on consecutive days during the peak of the spawning runs. If the number of spawned eggs significantly affects overall adult abundance, further increases in the number of fish caught and kept could affect population levels.

Planktonic Eggs and Larvae

Flow. Young-of-year shad abundance appears to be positively correlated with flow during the primary spawning months (April-June) (Painter, 1979). Analysis of the 1967-1991 MWT abundance indices indicates that young-of-year shad are most abundant in years when April-June Delta outflows are greater (Figure II-81). Seining surveys conducted during the 1975-1978 period collected a greater number of juvenile shad in 1975 and 1978 than during the 1976-1977 drought years (Painter et al., 1980).

The precise environmental mechanism responsible for increasing young-of-year shad abundance during years with increased April-June flows is unknown. However, possible explanations of reduced abundance at lower Delta outflows include the following:

- Eggs and larvae are more likely to settle to the river bottom and die because water velocities that are necessary to suspend eggs off the bottom are reduced.

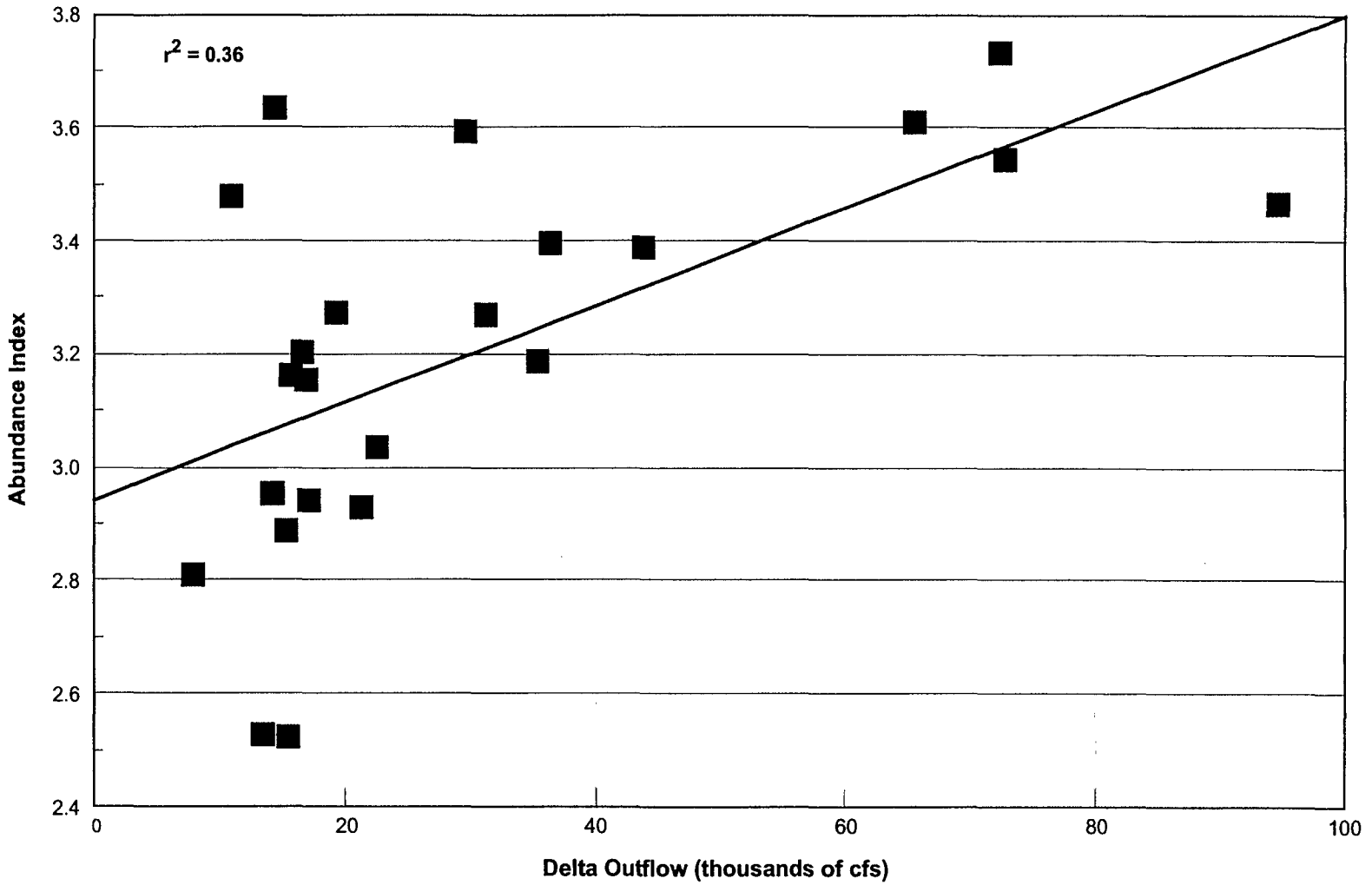

FIGURE II-81

RELATIONSHIP BETWEEN YOUNG-OF-YEAR AMERICAN SHAD ANNUAL ABUNDANCE INDEX AND MEAN ANNUAL APRIL-JUNE DELTA OUTFLOW (1967-1991)

C-081529
C-081529

- Egg and larval survival is reduced because of warmer water temperatures associated with reduced river flows.
- Eggs and larvae are more susceptible to exposure to toxic substances in the rivers and Delta.
- A lower proportion of larvae are carried to the Delta where feeding efficiency and survival rates could be increased.
- A higher proportion of larvae are drawn into the central and south Delta where vulnerability to entrainment in diversions is greater.

Entrainment. Entrainment losses depend on the timing, size, and location of individual diversions relative to the seasonal distribution and abundance of American shad. Losses of larval shad could be most effectively minimized by reducing diversions in July and August.

CVP and SWP Delta Pumping Facilities. CVP and SWP Delta pumping facilities are the largest diversions in the Delta, and young shad are vulnerable to diversion by these and other facilities. Thousands of American shad are salvaged annually by CVP and SWP fish protection facilities (Figure II-82), and thousands more are lost to the diversions. American shad are the third most common fish salvaged at the SWP screens (DFG, 1987).

Young shad spawned in the south Delta and Mokelumne River channels are drawn into the pumps as larvae and small juveniles; Sacramento River system juveniles are drawn through the

DCC and across the Delta during their downstream migration (DFG, 1987). Salvage data from the CVP and SWP pumping facilities indicate that larval shad (less than 28 millimeters long) are entrained from May through September (Figure II-83). Many of the entrained larvae are lost in the diversions. Entrainment losses, including predation, handling, and trucking mortality, have not been quantified.

Delta Agricultural Diversions. Losses of larval shad to agricultural diversions are probably considerable because these diversions account for approximately one-third of the volume of water diverted from the Delta. Losses to agricultural diversions depend on the timing, size, and location (geographically and position within the channel) of individual diversions relative to the seasonal distribution and abundance of larvae. Entrainment losses to agricultural diversions have not been quantified.

Power Generation Facility Diversions. PG&E's Contra Costa and Pittsburg powerplants have fish screens, but little information on entrainment rates, salvage efficiency, and associated losses of larval shad is available (PG&E, 1981a, 1981b). Shad larvae are known to occur in the Delta and Suisun Bay and are probably susceptible to entrainment as they pass near the intakes to these powerplants.

Other Diversions. The magnitude of larval entrainment losses at the North Bay Aqueduct and CCWD's Rock Slough diversions is unknown. Losses at upstream diversions in rivers where shad spawn, rear, and emigrate undoubtedly occur but are not quantified. As the

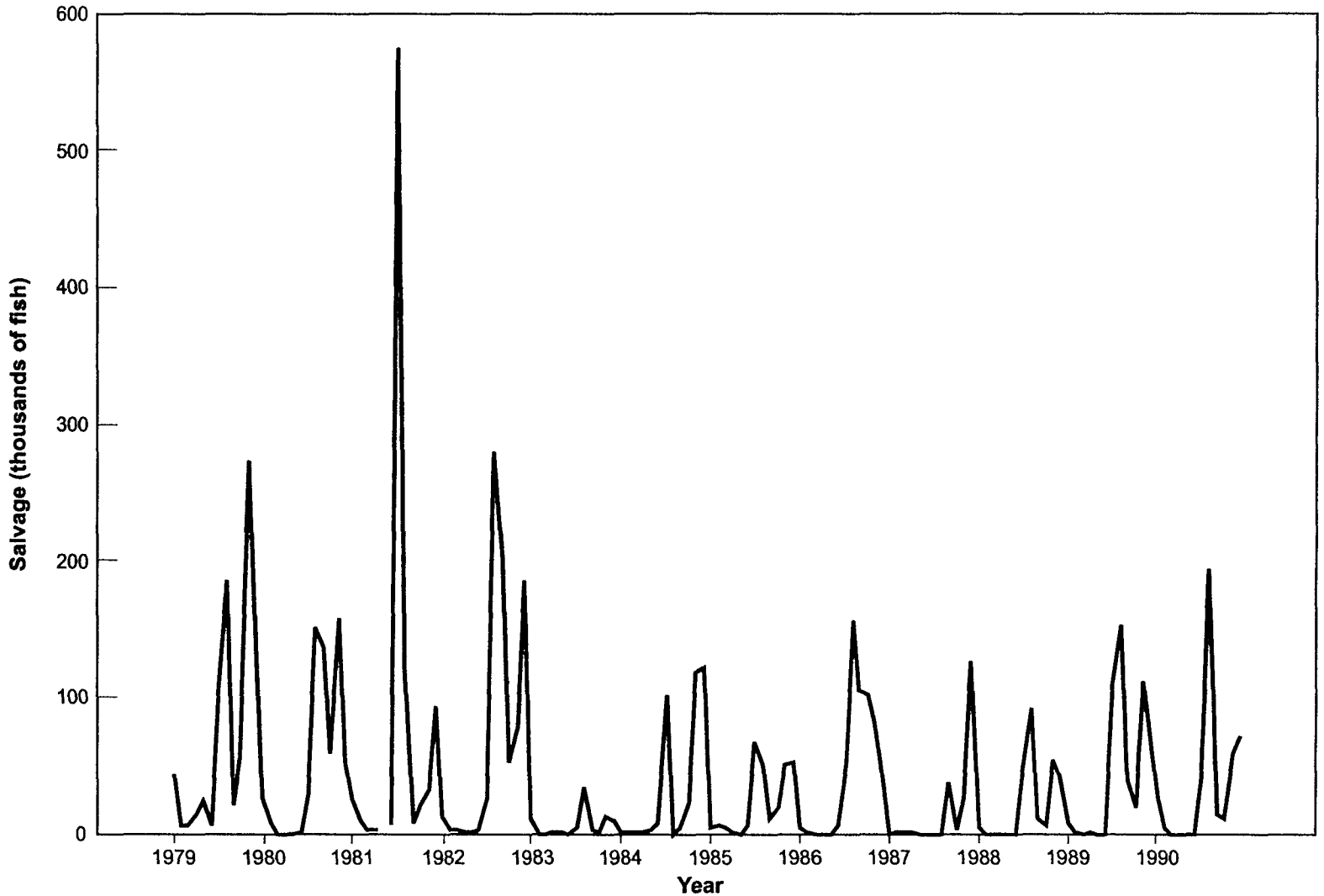

FIGURE II-82

SALVAGE OF AMERICAN SHAD AT THE SWP AND CVP FISH PROTECTION FACILITIES (1979-1990)

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C-081531

C-081532
C—081532

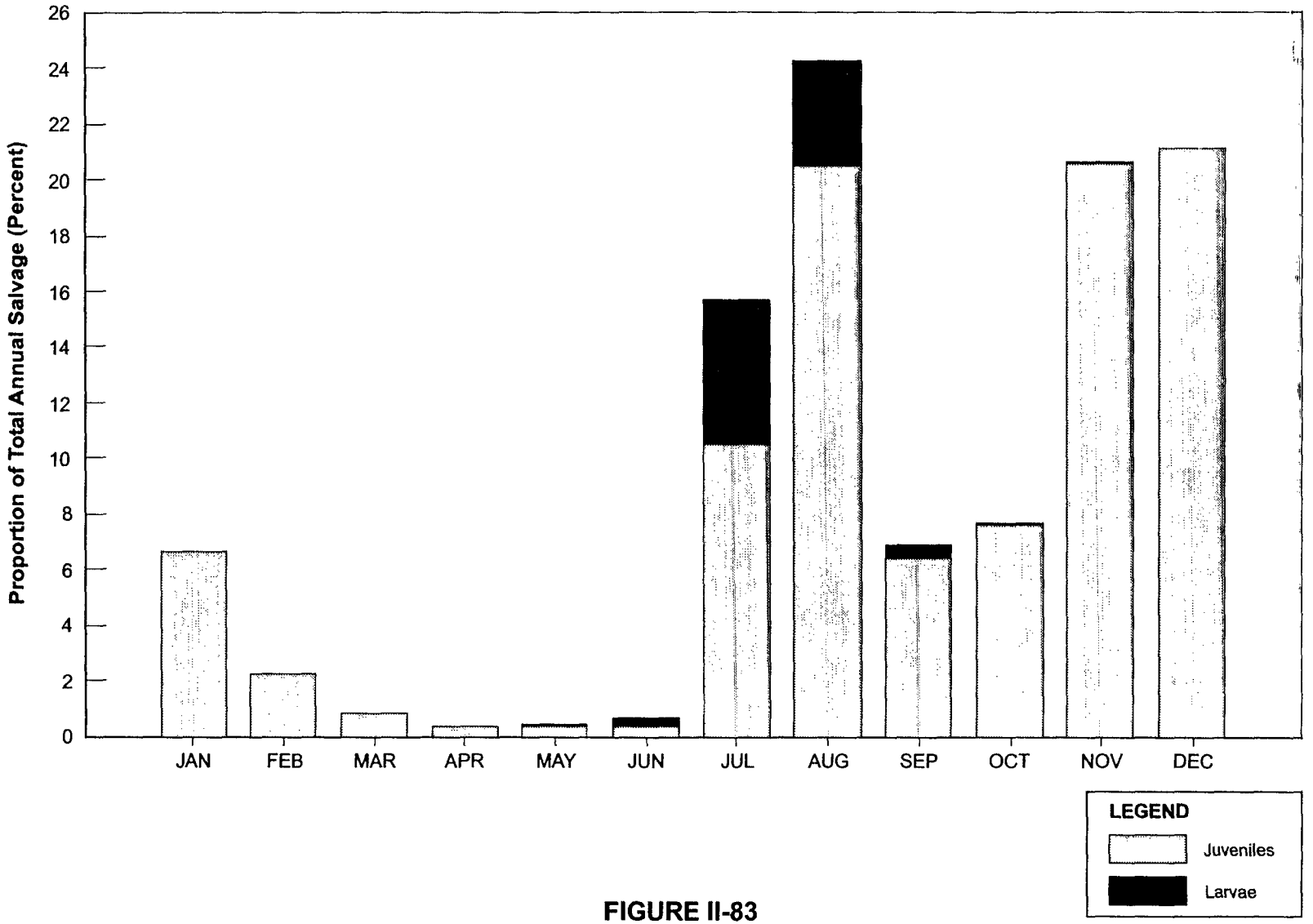


FIGURE II-83

AVERAGE MONTHLY PROPORTION OF ANNUAL AMERICAN SHAD SALVAGE AT THE SWP AND CVP FISH PROTECTION FACILITIES (1979-1990)

proportion of river flow diverted is increased, American shad egg, larvae, and juvenile survival generally will be expected to decrease if these life stages resided in the area of the river where the diversions were occurring.

Habitat. Habitat modifications have had the greatest effect on shallow-water habitats particularly important to developing larvae. Important shallow-water habitats provide optimal water temperatures necessary for growth and proper development and excellent conditions for food production. As noted previously, levee construction, river channelization, dredging, and the diking and filling of flood basins have drastically reduced the number of shallow-water habitats available to young shad both in the major river systems and in the Delta.

Temperature. Operation of large upstream reservoirs has altered historical water temperature regimes in tributary rivers. The survival of shad eggs and larvae is closely related to water temperatures. Exceedingly low water temperatures (less than 52 degrees Fahrenheit) can reduce hatching success of shad eggs (Stier and Crance, 1985). Similarly, exceedingly high water temperatures (greater than 80 degrees Fahrenheit) can be unsuitable for hatching of eggs and eventual development of larvae (Stier and Crance, 1985). Less than optimal water temperatures could cause poor development, reduced growth rates, and increased mortality of developing larvae.

Salinity. As stated earlier, upstream water storage projects, diversions, and Delta export pumping have reduced Delta outflow and periodically increased salinity in Suisun Marsh and Bay. Because larval shad appear to be highly tolerant of salinity and salinity changes (Stier and Crance, 1985), increased salinity in the estuary does not appear to affect young shad directly.

However, increased salinity in the estuary could influence other environmental and biological factors such as prey availability, thereby affecting shad abundance indirectly.

Toxic Materials. Although shad spawn when flows are typically high and resulting pollutant concentrations are probably relatively low (because of the diluting effects of high freshwater flows), localized populations of young shad and eggs could be disproportionately affected by pollutants if developing eggs and larvae encounter discharges containing high pollutant concentrations. Developing eggs and larvae in the vicinity of these discharges could be subject to poor development, reduced growth rates, and increased mortality, but specific data are unavailable to ascertain the importance of toxic materials in determining shad abundance.

Prey Availability. Prey availability for larval shad appears to be adversely affected by human-induced factors. Removal of riparian and streamside vegetation in the Sacramento River system upstream of the Delta potentially reduces the recruitment of terrestrial insects. Young shad in these upstream areas often rely on terrestrial insects as a food source, which has been decreasing as more leveed river sections have bank protection. (DFG, 1987.)

Water development has affected zooplankton abundance in the Delta. This could occur because the use of Delta channels to convey Sacramento River water to the south Delta has reduced water residence times in the Delta and increased the volume of zooplankton-deficient Sacramento River water that is transported to the central and south Delta (DFG, 1987).

The introduced Asiatic clam (*Potamocorbula* sp.) could affect young shad abundance because the clam has become extremely abundant in Suisun Bay where it could compete for food resources with opossum shrimp, a prey item of American shad. Introduced species of copepods and cladocerans could have similar effects on young shad abundance (see "Recent Conditions for the Estuarine Invertebrate Community").

Competition and Predation. The effects of increased competition and predation resulting from species introductions are difficult to evaluate in wild populations. Competition and predation are likely factors that affect shad abundance. Competition-predation effects will be distinguishable if a concomitant decline in abundance of shad matched an increase in the abundance of an introduced species. No firm evidence exists of a decline in shad populations.

Rearing Juveniles

Flow. As discussed earlier, young-of-year shad abundance is positively correlated with April-June Delta outflow (Figure II-81). However, the precise environmental mechanism that determines shad abundance is unknown. Several mechanisms that could contribute to reduced abundance of egg and larval stages likely apply to juvenile shad and were discussed in the previous section.

Entrainment

CVP and SWP Delta Pumping Facilities. Thousands of juvenile shad (2.8 to 30 centimeters long) are entrained in diversions at the CVP and SWP Delta pumping facilities each year and constitute the majority of entrained shad. Although the bulk of juveniles are entrained from July through December, salvage records indicate that the juvenile shad are entrained year round (Figure II-83).

The relative proportion of entrained juveniles that are salvaged and returned to the Delta alive has not been quantified. Evaluations of screening efficiency comparable to studies for striped bass and salmon have not been conducted for American shad; however, it is believed that larger fish in fall are screened more efficiently than those in late spring and early summer (DFG, 1987).

Entrainment losses result from predation near the screening facilities and stress associated with handling and trucking. Salvaged American shad suffer mortality rates in excess of 50 percent during summer, with slightly lower mortality rates during the cooler fall (DFG, 1987).

Delta Agricultural Diversions. Entrainment of juvenile shad to agricultural diversions is a function of fish size, location of the diversions (geographically and position within the channel), and the volume and design of the diversions. Although juvenile shad could be capable of avoiding smaller intakes, entrainment is likely. The volume of entrainment losses of juveniles to these diversions is currently unknown and depends on juvenile abundance and distribution in addition to the factors mentioned in the preceding sections.

Power Generation Facility Diversions. PG&E's Contra Costa and Pittsburg powerplants have fish screens, but little information on entrainment rates, salvage efficiency, and

associated losses of juvenile shad is available (PG&E, 1981a, 1981b). Juvenile entrainment could be substantial because of the proximity of the intakes to juvenile rearing areas.

Other Diversions. The efficiency of the fish screens and the entrainment losses of juvenile shad at the North Bay Aqueduct and losses to CCWD's Rock Slough diversions are unknown. Diversions in known juvenile rearing areas in the rivers will have an adverse effect similar to that described for eggs and larvae but substantially diminished because of the swimming capabilities of the larger juvenile fish.

Habitat, Toxic Materials, Prey Availability, and Other Factors. These factors generally affect juvenile shad as previously described for adult and larval shad.

Ocean Rearing

Little is known about the oceanic ecology and behavior of juvenile and adult American shad. In general, overall shad production depends on both freshwater conditions (factors affecting adult migration, spawning, egg incubation, rearing, and emigration) and oceanic conditions (factors affecting ocean shad growth, survival, and migration back to freshwater). More is known about the freshwater life history, biology, and environmental requirements of shad. The oceanic ecology of shad in the Pacific Ocean has been generally neglected. Oceanic conditions should not be entirely dismissed as a factor affecting abundance, however, because DFG viewed the 1982-1983 El Niño conditions in the ocean as having detrimental impacts on shad populations (Messersmith, pers. comm.), and oceanic conditions are being found to have greater effects on salmon populations than once thought (see "Recent Conditions for Chinook Salmon").

ONGOING MONITORING, ENHANCEMENT, AND HABITAT IMPROVEMENT ACTIONS

Because of the popularity of the sport fishery, DFG originally planned to focus on monitoring American shad and developing a detailed management plan for this species in the late 1970s, with the principal goal of maintaining and enhancing the adult shad population present at that time. Funding for further research on American shad and development of the detailed management plan was substantially reduced, ending the program and resulting in a management plan being developed (Painter et al., 1980) based on the available data only. Since that time, little progress has been made on basic shad research and management; however, many of the programs described for other anadromous species in this chapter will provide benefits to American shad. For completeness, the original goals and recommendations for managing American shad are described below (Painter et al., 1980).

Specific objectives of the management plan were to:

- identify factors affecting the survival of juvenile shad during their rearing and outmigration periods;
- determine the role and relative importance of the lower Sacramento-San Joaquin Delta and Bay in the growth of juvenile shad and the maintenance of adult shad populations;

- develop and implement methods to reduce entrainment losses at water diversions; and
- plan and implement studies to monitor shad population abundance and sport harvest rates periodically.

Recommendations proposed in the management plan focused on maintaining suitable habitat conditions (i.e., water temperature and instream flows). Specific recommendations included the following:

- maintain the highest practicable level of activities and studies to preserve and maintain shad habitat and implement program objectives;
- maintain a normal distribution of adult shad in tributary rivers by maintaining instream flows during May and June so that the Feather River flow is at least 34 percent of the Sacramento River flow, the Yuba River flow is at least 33 percent of the Feather River flow, and the American River flow is at least 10 percent of the Sacramento River flow at Sacramento; and
- maintain water temperatures between 60 and 70 degrees Fahrenheit in the upper Sacramento, Feather, Yuba, and American rivers during May and June.

RECENT CONDITIONS FOR DELTA AND LONGFIN SMELT

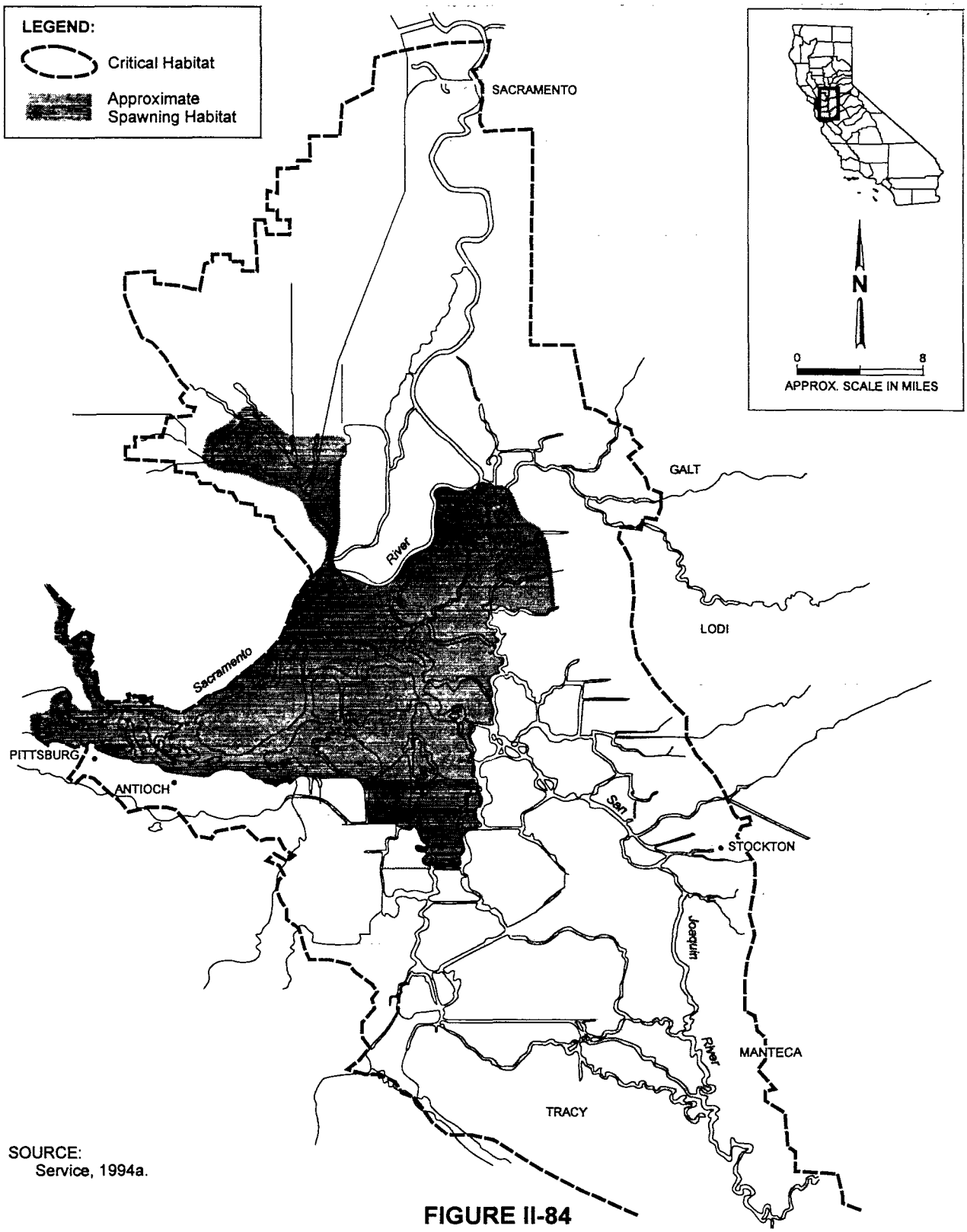
DELTA SMELT

Stock Abundance and Distribution

Delta smelt (*Hypomesus transpacificus*) are found only in the Sacramento-San Joaquin Delta estuary. Before 1961, it was thought to be the same species as the pond smelt (*Hypomesus olidus*), which is widely distributed. Declines in delta smelt abundance led to listing the species as threatened in 1993 under the federal ESA (Federal Register 58:12854-12862, March 5, 1993).

Delta smelt are euryhaline and inhabit a narrower range of salinities than longfin smelt (see subsequent section on longfin smelt) and rarely occur in water with more than 14 parts per thousand salinity (Moyle et al., 1992). Except for spawning adults and recently hatched larvae and juveniles, delta smelt primarily inhabit the region of the estuary with salinities between approximately 0.45 and 4.4 parts per thousand (DWR and Reclamation, 1993). The location of this region varies from year to year depending on the volume of freshwater outflows, but it is generally in the western Delta and Suisun Bay. Delta smelt have been collected in the Sacramento River as far upstream as Verona, near the mouth of the Feather River, and in the San Joaquin River as far upstream as Mossdale (Wang, 1991). Their normal downstream limit appears to be the western Suisun Bay, but they are sometimes washed into San Pablo Bay during high outflows. The critical habitat for delta smelt as proposed by the Service includes all of Suisun Bay and the Delta (Figure II-84).

The Interagency Ecological Program for the Sacramento-San Joaquin Estuary (IEP) includes several surveys that sample delta smelt in addition to other fish species. The summer tow net



SOURCE:
Service, 1994a.

FIGURE II-84

SPAWNING HABITAT AND CRITICAL HABITAT AS DESIGNATED UNDER THE FEDERAL ENDANGERED SPECIES ACT FOR DELTA SMELT

survey (TNS) samples juvenile smelt while the fall mid-water trawl (MWT) samples adult smelt. These surveys encompass most of the habitat of delta smelt and have relatively long and continuous sampling records.

Delta smelt abundance indices are computed from results of the summer TNS and the fall MWT survey. The catch of delta smelt in each sampling area, consisting of the total catch from one station for the summer TNS and the mean catch from all stations in the sampling area for the fall MWT survey, is multiplied by the estimated total water volume of the sampling area, and the products are summed over all areas (Stevens et al., 1990). An abundance index is calculated for each survey visit, and an annual index of abundance is computed as the mean for the summer TNS and as the sum for the fall MWT survey.

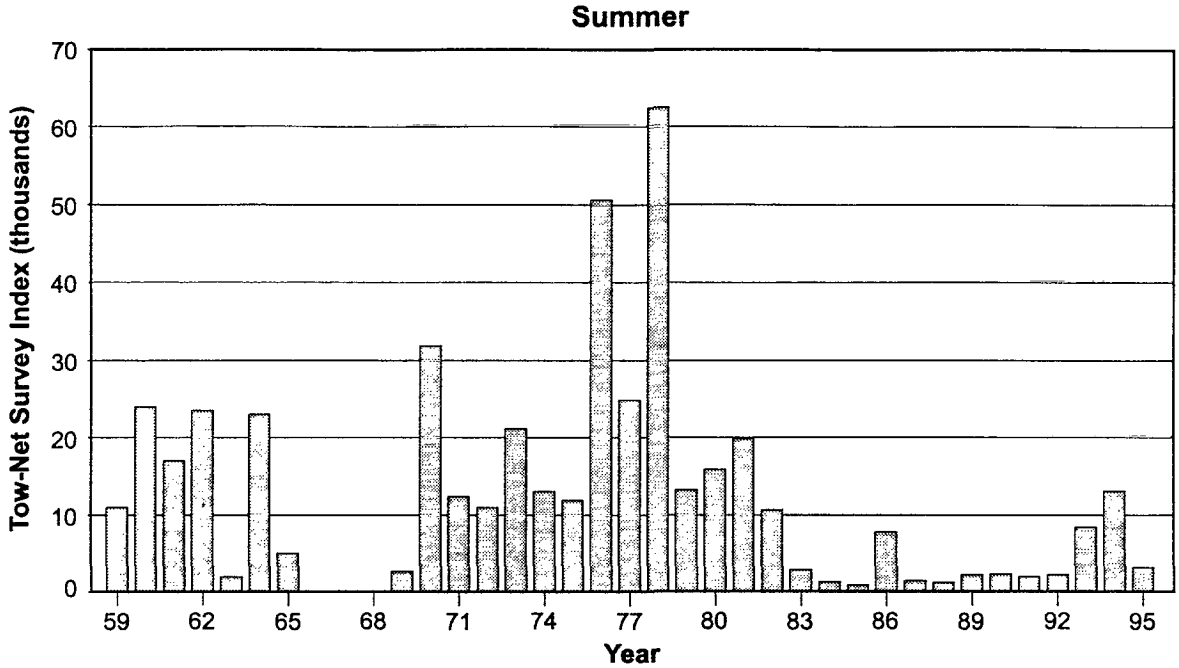
Historical Population Trends (1850-1966). Delta smelt are characterized as historically abundant in the estuary (Moyle, 1976), but specific data on abundance are unavailable for years prior to 1959. DFG sampled delta smelt in the summer TNS during 1959-1965 (the summer TNS was not conducted in 1966, and delta smelt catch data were not recorded during 1967-1968). The summer TNS annual abundance indices for five of seven years during 1959-1965 were above the median for the 32-year period of record (median annual abundance index for 1959-1993 equals 11) (Figure II-85). The index for 1963, however, was 1.7, which is the fifth lowest index on record.

Current Population Trends (1967-1991). Delta smelt abundance, as measured by either the summer TNS annual abundance index or the fall MWT survey annual abundance index, has fluctuated widely since 1967 (Figure II-85). Both indices were generally high through the 1970s and early 1980s, dropping sharply in 1983. All summer TNS indices for 1969-1981, except the 1969 index, were above the median value for 1969-1993 (median equals 10.6), whereas all indices for 1983-1993 were below the median (1982 had the median index). All fall MWT survey indices for 1967-1981 exceeded the median value for 1967-1993 (median equals 375), except the indices for 1969, 1976, and 1981 (1981 had the median index), and all indices for 1982-1993 were below the median, except those for 1991 and 1993. Both the summer TNS and fall MWT survey indices provide strong evidence of a substantial decline in delta smelt abundance since 1982.

Life History

Delta smelt are small (usually less than 3.5 inches long), plankton-feeding fish that typically live for only one year. They have no direct commercial value but could have been one of the most abundant fish species in the Sacramento-San Joaquin Delta and could potentially have been an important prey for commercially valuable predator species, such as striped bass. Food habit studies in the 1960s, when striped bass and delta smelt were both abundant, indicated that striped bass preyed little on delta smelt (Stevens et al., 1990). Delta smelt life history is depicted in Figure II-86.

Adults. Delta smelt begin to mature in fall, seven to nine months after hatching. Prespawning adults are found near the entrapment zone in Suisun Bay or in the Delta as early as September. During the months preceding spawning, the smelt grow little in length because they allocate most growth energy to gonadal development. Adults typically reach 2 to 3 inches in length.



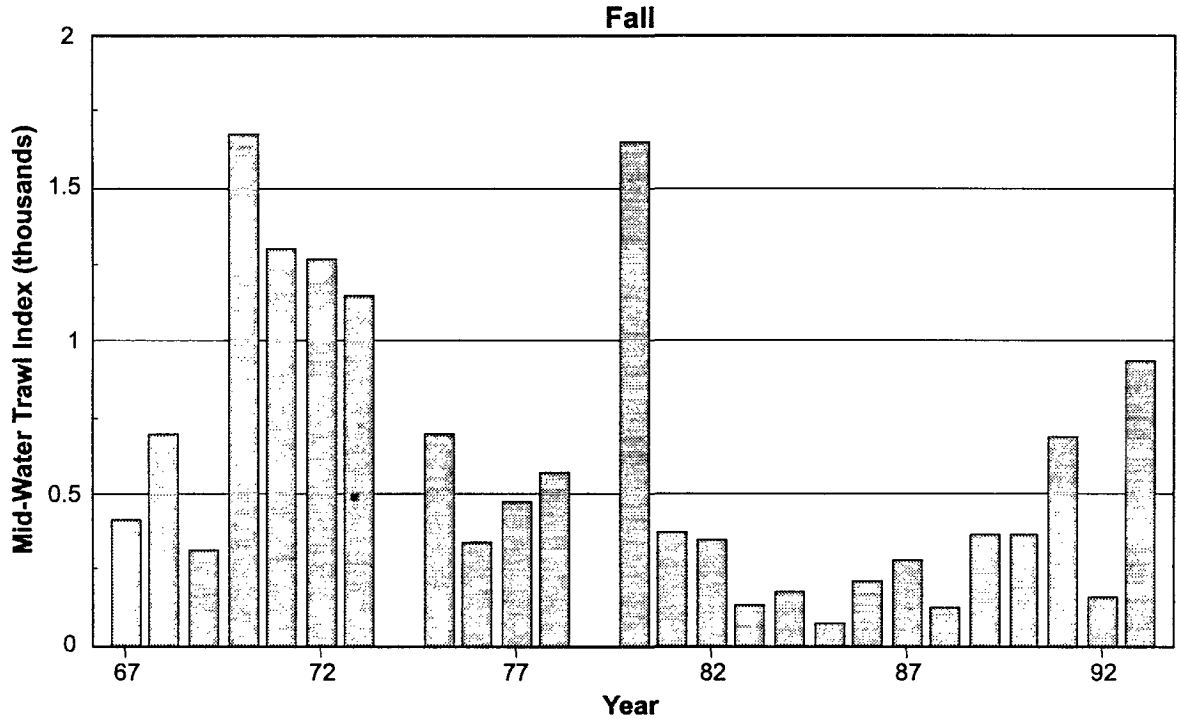


FIGURE II-85

DELTA SMELT ABUNDANCE INDICES IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY CALCULATED FROM TWO SURVEY PERIODS (1959-1995 AND 1967-1992)

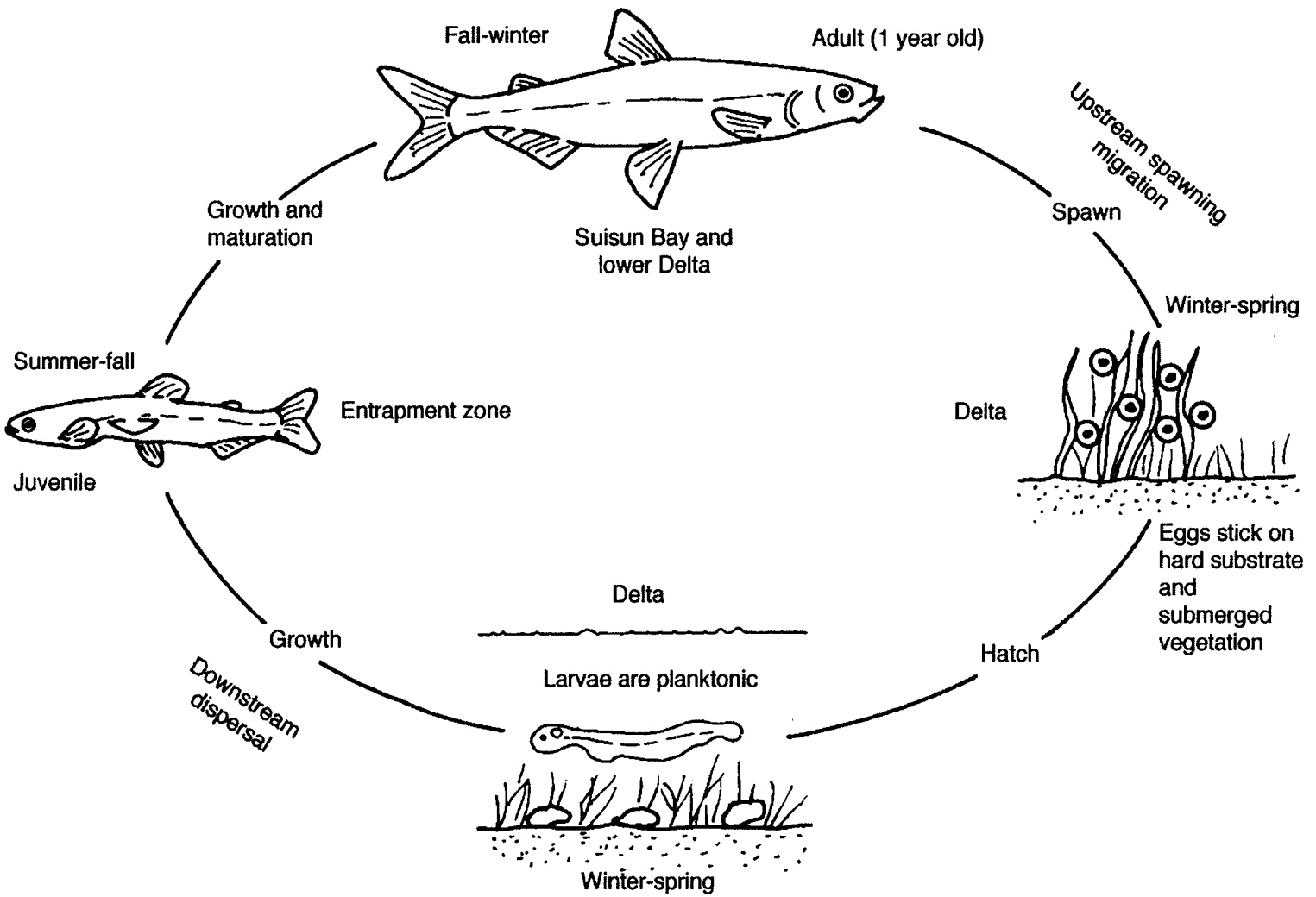


FIGURE II-86

LIFE HISTORY OF DELTA SMELT

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Smelt that survive to spawn a second year could grow to be as large as 5 inches long, but most die during or shortly after their first spawning season. Prespawning adults and juvenile delta smelt are generally most abundant above the upstream end of the entrapment zone, at salinities of approximately 0.45 to 4.40 parts per thousand (DWR and Reclamation, 1993). In years with high Delta outflows, the entrapment zone is generally downstream of the confluence of the Sacramento and San Joaquin rivers; in years with low outflows, the entrapment zone is in the Delta. Consequently, the delta smelt population is concentrated in the estuary west of the confluence in high outflow years and in the Delta in low outflow years (Stevens et al., 1990; Moyle et al., 1992).

Delta smelt adults and older juveniles live principally in shallow water or near the surface in deeper water, where they feed on zooplankton, particularly copepods (*Eurytemora affinis*, *Pseudodiaptomus forbesi*, and others) (Moyle et al., 1992). Mysids (*Neomysis mercedis*), cladocerans, and amphipods could be important food items, depending on prey availability and the size of the smelt. When shallow-water habitat is available, such as in Suisun Bay, trawl catches of delta smelt in shallow water (less than 13 feet deep) are generally higher than catches in deep water (Moyle et al., 1992). Although this difference suggests that the smelt prefer shallow-water habitat, the difference could also reflect differences in the capture efficiency of the trawl in deep and shallow water.

Spawning. Initiation of delta smelt upstream migrations and spawning could be triggered by freshwater outflows (Baxter, pers. comm.). Delta smelt usually spawn during February through May, although ripe females have been collected as early as December. A female deposits approximately 1,200 to 2,600 eggs at one time (Moyle et al., 1992). Fecundity is low compared to that of most fish species.

Spawning delta smelt and spawned eggs have not been observed in the field. Based on the distribution of gravid females and young larvae, spawning is believed to take place primarily in shallow edgewaters and river areas under tidal influence with moderate to fast velocities (Wang, 1991). Dead-end sloughs have been indicated as additional spawning habitat (Wang, 1991), but spawning in such areas has not been documented and is inconsistent with current knowledge about larval distribution. Laboratory observations indicate that delta smelt are broadcast spawners, spawning in currents at night (DWR and Reclamation, 1993). The eggs are adhesive and are probably deposited on rocks or aquatic plants.

Delta smelt disperse in the Delta during spawning migrations. The spawning distribution is probably related to the salinity gradient. In most years, smelt spawn primarily in the upper end of Suisun Bay, in Montezuma Slough, and in the lower and central Delta (Figure II-84). In the Delta, they spawn mostly in the Sacramento River channel and adjacent sloughs. Delta smelt generally spawn in freshwater but could also spawn in slightly brackish water in or above the entrapment zone (Wang, 1991).

During the 1987-1992 drought, when saline water intruded into the Delta, delta smelt spawned mostly in the central Delta, particularly in the Sacramento River near Rio Vista and the San Joaquin River near Twitchell Island (Wang, 1991; Jones & Stokes Associates, 1993). Adults and larvae have been collected in Barker Slough, Lindsey Slough, Cache Slough, Georgiana Slough, Prospect Slough, Beaver Slough, Hog Slough, Sycamore Slough, the Mokelumne River,

Fisherman's Cut, and False River (Wang and Brown, 1993). They have also been found as far upstream as Mossdale on the San Joaquin River, at the CVP and SWP pumps in the South Delta, and above the City of Sacramento on the Sacramento River. In normal and high outflow years, some spawning occurs in the channels and sloughs of Suisun and Honker bays (Wang, 1991; DWR and Reclamation, 1993). The areas of the estuary used for spawning by delta smelt are similar to the spawning areas of longfin smelt (see Figure II-92 under "Longfin Smelt").

Egg Incubation and Larval Rearing. After they are released during spawning, delta smelt eggs sink toward the bottom and adhere to any available hard substrate. The eggs hatch in 9 to 14 days at 63 degrees Fahrenheit and the larvae begin feeding four to five days after hatching (DWR and Reclamation, 1993). Larvae generally peak in abundance during March, April, or May (DFG, 1992a), although earlier peaks could occur in years with early freshwater outflow events (Baxter, pers. comm.).

Unlike longfin smelt larvae, which develop a gas bladder almost immediately after hatching, delta smelt larvae do not develop a gas bladder until they are one month to one and one-half months old (Wang, 1991; Mager and Baxter, pers. comms.). However, laboratory observations indicate that delta smelt larvae are buoyant after hatching because of the buoyancy of the oil globule in their yolk sac (Baxter, pers. comm.). They also actively swim toward the surface during this stage (DWR and Reclamation, 1993). The oil globule is absorbed in approximately 8 to 10 days (DWR and Reclamation, 1993), and the larvae lose their buoyancy and become more evenly distributed in the water column. In the Delta, the larvae are presumably transported downstream during the brief period of buoyancy. Downstream transport is probably reduced after the larvae lose their buoyancy. When the larvae are approximately one to one month and one-half months old, their gas bladders inflate, allowing them to control their buoyancy and drift downstream toward the entrapment zone. Unfortunately, little is known about how the larvae are distributed in the water column at different stages of their development.

The difference in the buoyancy of delta smelt and longfin smelt larvae could significantly influence their downstream transport and their ultimate distribution in the estuary. Reduced buoyancy probably improves the chances that delta smelt larvae are retained in or upstream of the entrapment zone by localized circulation (Kimmerer, 1992). The more buoyant longfin smelt larvae could remain near the surface, which will improve their chances of being carried over the entrapment zone to brackish water farther downstream. As discussed later, the entrapment zone, or the area just upstream of it, is the principal habitat of delta smelt; longfin smelt are principally found in more brackish water habitats.

Delta smelt larvae and young juveniles collect in and near the entrapment zone, where mixing saltwater and freshwater currents apparently keep the larvae circulating with the abundant zooplankton that also occur in this zone (Herbold et al., 1992; Kimmerer, 1992; Jassby, 1993). Little information exists on food habits of delta smelt larvae; however, fish larvae of most species feed on phytoplankton and small zooplankton, such as rotifers and copepod nauplii (Hunter, 1981). In the laboratory, delta smelt larvae began feeding four to five days after hatching and fared best on a diet of rotifers until they were 10 to 14 days old (DWR and Reclamation, 1993).

Juvenile Rearing. Metamorphosis of delta smelt from larval to juvenile form occurs when the smelt reach a length of approximately 0.7 inch (Wang, 1991). The juveniles grow rapidly and reach adult length (approximately 2 inches) within six to nine months.

The distribution and food habits of delta smelt juveniles are similar to those of the prespawning adults, although small juveniles probably feed less than do adults on relatively large prey such as mysids and amphipods.

Factors Affecting Abundance

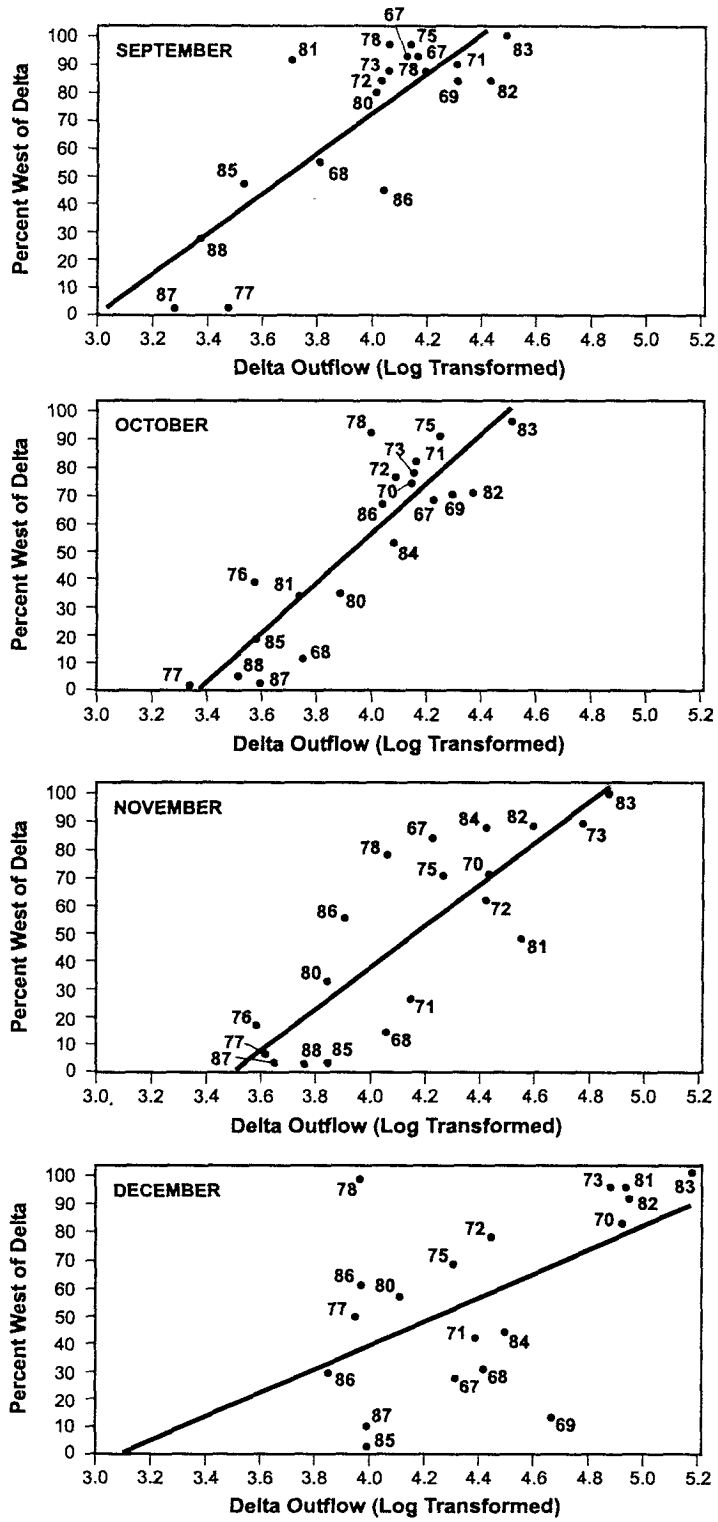
Adults and Spawning Migration

Delta Outflow. The distribution of delta smelt prespawning adults and juveniles is related to Delta outflow. Linear regression analyses by DFG indicated that the percentage of delta smelt in the fall MWT survey collected west of the confluence of the Sacramento and San Joaquin rivers is positively related to Delta outflow (DFG, 1992a) (Figure II-87). The regressions were conducted using arcsine-transformed percentages and log-transformed outflow and were carried out separately for each month of the survey (September through December) using the outflow estimate for that month. The coefficient of determination, r^2, was high September through November (r^2 equals 0.64-0.76), but lower for December (r^2 equals 0.34) (r^2 estimates the proportion of the variation in the percentage of delta smelt west of the confluence that can be explained by variations in Delta outflow). As noted earlier, the relationship between the distribution of delta smelt and outflow largely reflects the fact that delta smelt prespawning adults and juveniles are concentrated near the upper end of the entrapment zone, and the position of the entrapment zone is largely determined by Delta outflow. The lower r^2 for December probably results from the fact that in most years adults leave the entrapment zone to begin their upstream spawning migration during December.

Delta outflow probably affects where delta smelt adults spawn in the estuary. The smelt spawn primarily in freshwater, and the downstream distribution of freshwater is determined by the amount of flow in the Sacramento and San Joaquin rivers (see Attachment B). The freshwater in high outflow years in the Upper Suisun Bay could encourage spawning in Suisun Bay. In low outflow years, the adult smelt must migrate into the Delta to reach freshwater (Wang and Brown, 1993).

When outflow is low and exports at the CVP and SWP pumps are high, the net flow in the lower San Joaquin and Sacramento rivers could be toward the pumps rather than downstream. These reverse flows, which contain relatively freshwater drawn from the Sacramento River, could encourage greater than normal upstream migration of the adults in the South Delta. In the South Delta, adults and their larvae are vulnerable to entrainment and other sources of mortality. Despite the potential liabilities of reverse flows, however, a variety of analyses have failed to discover any relationship between the number of days of reverse flows and delta smelt abundance (DWR and Reclamation, 1993).

Delta outflow could influence the timing of delta smelt spawning. Recent evidence suggests that delta smelt migrate to spawning areas and begin spawning earlier in years with early high outflows than in years with low outflows (Baxter, pers. comm.).



SOURCE:
DFG, 1992a.

FIGURE II-87

RELATIONSHIP BETWEEN THE PERCENTAGE OF THE DELTA SMELT POPULATION OCCURRING WEST OF THE DELTA AND LOG-TRANSFORMED DELTA OUTFLOW DURING THE SURVEY MONTH (1967-1987)

Entrainment. Thousands of diversions exist in the Delta, but adult delta smelt are probably entrained primarily at four major diversions: the CVP and SWP pumping facilities in the South Delta (maximum diversion capacities of 4,600 cfs and 10,300 cfs, respectively) and PG&E Contra Costa and Pittsburg generating facilities (approximately 1,500 cfs) (DWR and Reclamation, 1993; PG&E, 1985). Total diversions from the Delta can exceed 80 percent of the total Delta inflow and thus greatly reduce Delta outflow (Turner and Chadwick, 1972).

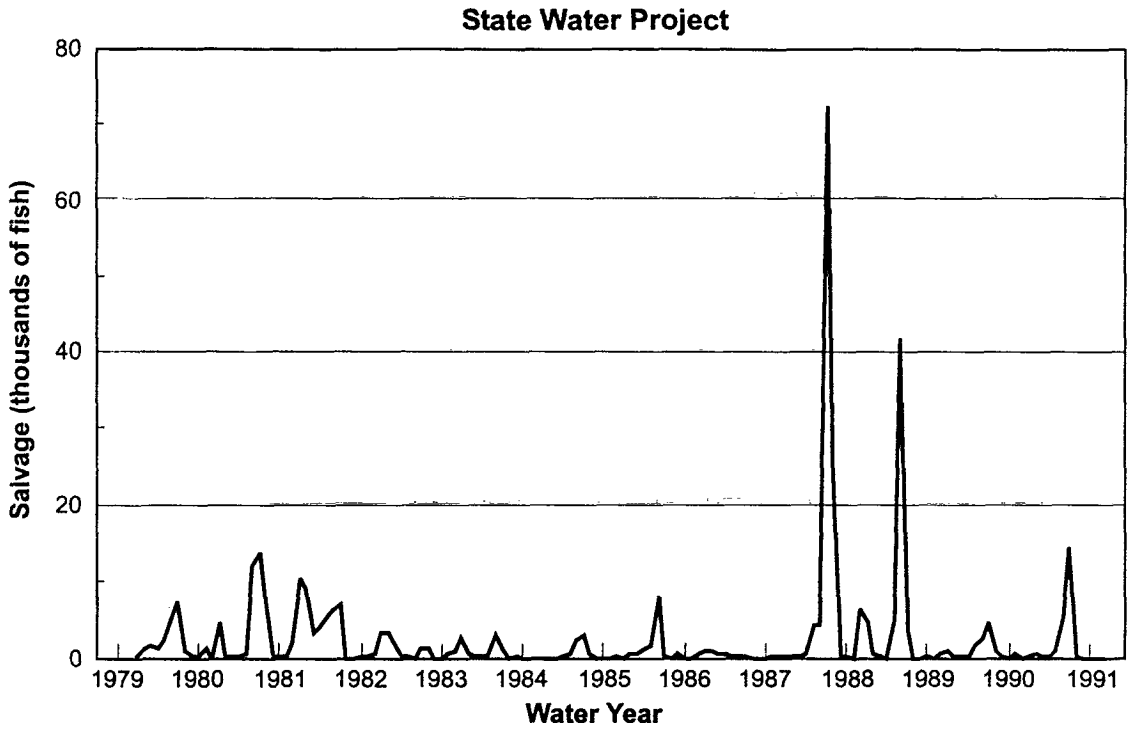
Entrainment of delta smelt by the CVP and SWP pumps primarily affects spawning adults, larvae, and young juveniles. Prespawning adults and older juveniles inhabit the western Delta and Suisun Bay, which are probably beyond the influence of the CVP and SWP pumps. Although delta smelt entrained at the CVP and SWP facilities are "salvaged" and returned by truck to downstream areas in the Delta, survival of the salvaged smelt is believed to be low (DWR and Reclamation, 1993). Salvage is often less than total entrainment because fish smaller than approximately 1 inch long pass through the fish screens at the salvage facilities and cannot be salvaged. Total entrainment, similarly, is probably much less than export loss because prescreening losses (e.g., predation) are probably high at the SWP.

Annual salvage of delta smelt at both the CVP and SWP facilities has varied greatly (Figure II-88). Despite the reduced delta smelt abundance during 1984-1991, salvage remained high, particularly at the SWP pumps (Figure II-88). The high entrainment during this period was the result of low Delta outflows and high exports (see Attachment B). A significant negative correlation exists between Delta outflow and salvage of delta smelt (DWR and Reclamation, 1993). Low outflow and high exports could increase entrainment because a greater proportion of Delta inflow is exported.

Entrainment of adult delta smelt has a potentially greater adverse effect on the population than entrainment of larvae and young juveniles because the reproductive value of adults (Ricklefs, 1973) is much greater than that of younger fish unless they have already spawned. The number of adult delta smelt salvaged at the CVP and SWP pumps was estimated by analyzing data obtained from DFG on lengths of delta smelt salvaged during 1981-1990 (Barrow, pers. comm.). Salvaged fish greater than 50 millimeters in length were assumed to be adults (DWR and Reclamation, 1993). On average, nearly all fish salvaged during December through March were adults that migrated and spawned during these months (Figure II-89). May through November, juveniles from this spawn dominated the salvage. The main size of the juveniles increased after May because of their growth and because numbers of new recruits decreased.

Adult delta smelt are vulnerable to entrainment at diversions other than the CVP and SWP pumps, but entrainment at other diversions has not been studied in detail. Adults appear to spawn in and near Barker and Lindsey sloughs, which are near the intake for the North Bay Aqueduct, at least in dry years (BioSystems Analysis, 1993; DWR and Reclamation, 1993). Adults have rarely been found in CCWD's Rock Slough diversion, but existing information is inadequate to rule out spawning activity in the area. Agricultural diversions are numerous in the Delta and, in total, could divert in excess of 4,000 cfs during the peak summer irrigation season (April through August). Entrainment loss to agricultural diversions is currently unknown.

PG&E operates powerplants located near Antioch and Pittsburg that divert cooling water at rates of up to 1,500 cfs and 1,600 cfs, respectively. Antioch and Pittsburg are located near the



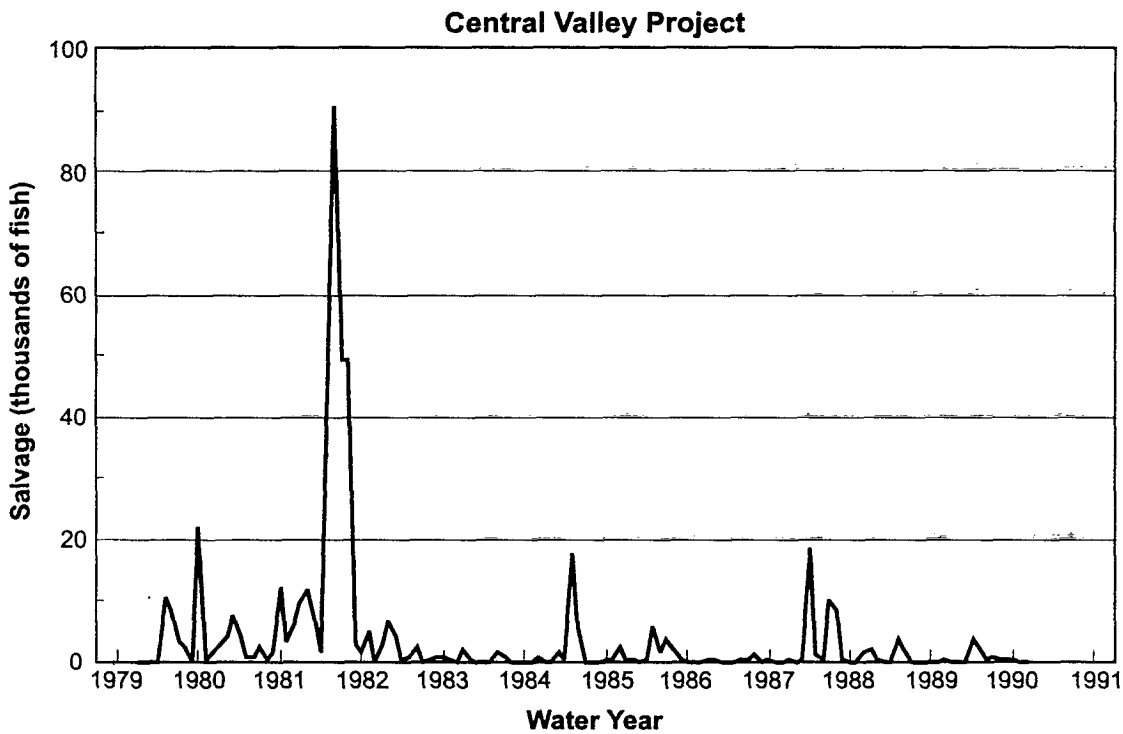


FIGURE II-88

SALVAGE OF DELTA SMELT AT THE SWP AND CVP FACILITIES (1979-1991)

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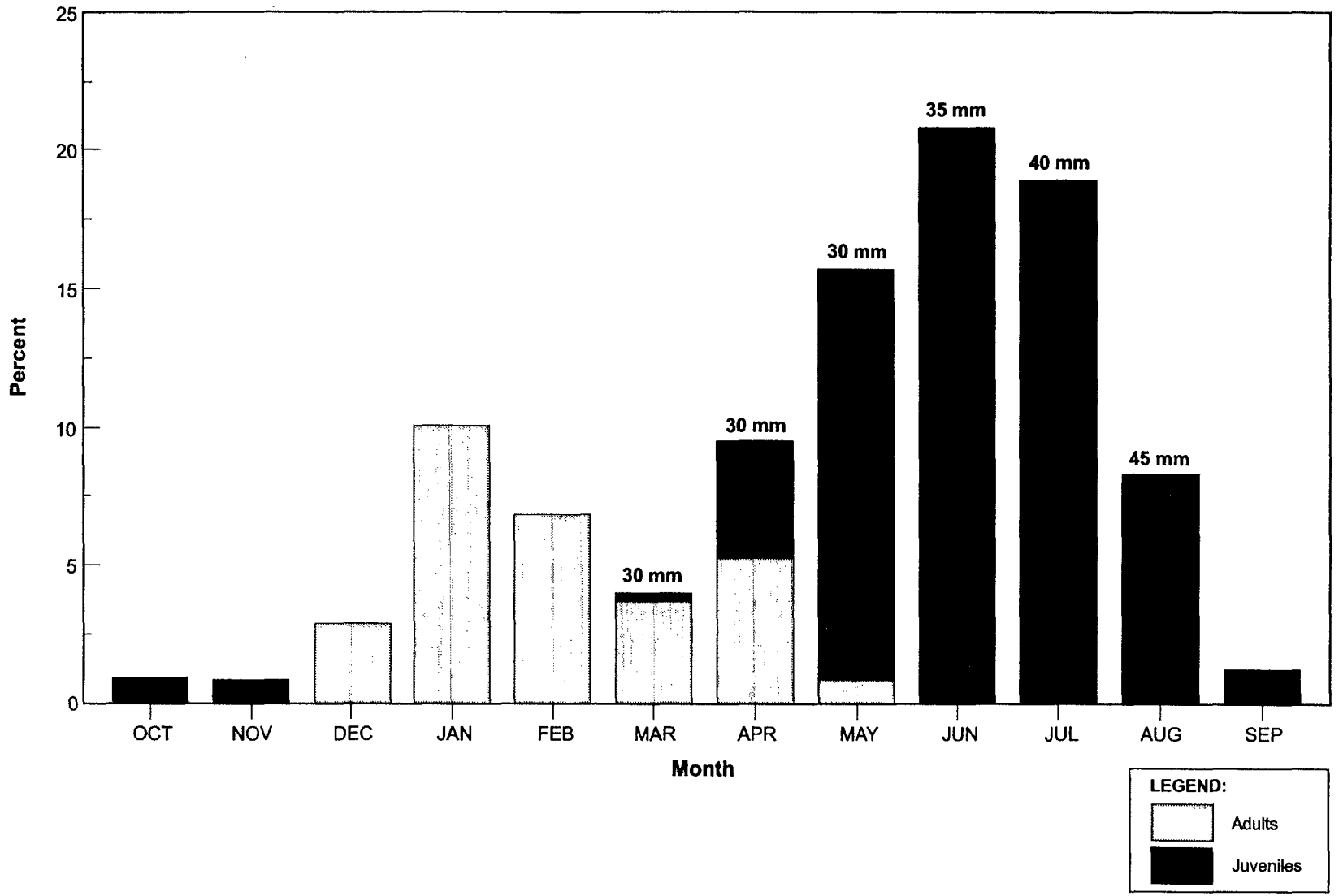

FIGURE II-89

PERCENT ADULT AND JUVENILE DELTA SMELT OF TOTAL FISH SALVAGED ANNUALLY AT SWP AND CVP FACILITIES

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confluence of the Sacramento and San Joaquin rivers, an important rearing and spawning area for delta smelt. The diverted cooling water is returned to the estuary, but nothing is known about survival of the smelt that are entrained in the water. The powerplant intakes are screened, which protects most adult and older juvenile delta smelt from entrainment, but these life stages could be impinged on the screens. In 1978-1979, estimates of delta smelt impingement were 10,000 smelt at the Pittsburg plant and 6,400 smelt at the Contra Costa (Antioch) plant (Stevens et al., 1990). Effects of thermal discharges from the plants on delta smelt are unknown (DWR and Reclamation, 1993).

Entrainment of delta smelt by the CVP and SWP pumps and other diversions probably contributed to the recent decline in delta smelt abundance, but analyses by Stevens et al. (1990) found no significant relationship between CVP and SWP exports and the delta smelt abundance indices. Reduced Delta outflow (which is influenced by exports) probably affects delta smelt abundance more than entrainment does.

Other water project facilities that could affect delta smelt include the Suisun Marsh Salinity Control Gates, the DCC, and the south Delta temporary barriers. Effects of these facilities on delta smelt are unknown; however, altered hydraulics from these projects could affect delta smelt.

Food Availability. Adult delta smelt prey chiefly on copepods, but they seasonally prey on cladocerans and mysids as well. *Eurytemora affinis*, one of the principal prey species of delta smelt, has substantially declined in recent years (Obrebski et al., 1992), and this decline could have contributed to the recent decline in delta smelt abundance (Stevens et al., 1990). However, an introduced copepod, *Pseudodiaptomus forbesi*, became abundant in the estuary in 1988, and diet studies indicate that it is now the main food item of delta smelt. Consequently, overall food availability for delta smelt could not have declined (Herbold et al., 1992).

Water Quality

Salinity. Delta smelt are found at salinities of 0 to 14 parts per thousand (Moyle et al., 1992), but they are most abundant at salinities between approximately 0.45 and 4.40 parts per thousand (DWR and Reclamation, 1993). The latter salinity range typically encompasses the upper part of the entrapment zone, so it is not clear whether delta smelt actually prefer this salinity range or whether this association results from the smelt gathering in the entrapment zone for other reasons (e.g., prey concentrations). Delta smelt abundance between 1984 and 1991 was negatively correlated with salinity in Suisun Bay (DWR and Reclamation, 1993). Whether this is a direct or indirect effect is unclear because delta smelt abundance is significantly related to the number of days that the entrapment zone is in Suisun Bay. No evidence that salinity has any direct effect on adult delta smelt exists, but as noted above, freshwater flow patterns could influence spawning migration.

Water Temperature. Delta smelt are found in waters ranging from 43 to 73 degrees Fahrenheit (Moyle et al., 1992). Water temperature during delta smelt spawning is reported to be 45 to 59 degrees Fahrenheit (Wang, 1986), although water temperature during the period of peak larval abundance is typically 59 to 73 degrees Fahrenheit (DFG, 1992a). No long-term trend in water temperature in the estuary during the past 20 years could account for the

recent decline in delta smelt abundance (DWR and Reclamation, 1993). Regression analysis found no relationship between delta smelt abundance and water temperature (Stevens et al., 1990).

Water Transparency. Increased water transparency could affect survival of delta smelt by increasing their visibility to predators and by the resulting decrease in nutrient availability. Secchi disk measurements during the past 20 years suggest a general increase in water transparency in some regions of the upper estuary (Kimmerer, 1992; DWR and Reclamation, 1993). Correlation analyses indicate that delta smelt abundance is negatively correlated with water transparency, particularly during winter and spring (DWR and Reclamation, 1993). However, trends in water transparency and smelt abundance could result from trends in other factors (e.g., reduced Delta outflow), so the significant correlations do not necessarily indicate that increased water transparency contributed to reduced delta smelt abundance.

Pollution. Agricultural chemicals (including pesticides and herbicides), heavy metals, petroleum-based products, and other waste materials toxic to aquatic organisms enter the estuary through nonpoint runoff, agricultural drainage, and municipal and industrial discharges. Pesticides have been found in the Sacramento River in recent years at concentrations potentially harmful to fish larvae (Herbold et al., 1992). Recent bioassays by the Central Valley Regional Water Quality Control Board indicate that water in the Sacramento River is periodically toxic to larvae of the fathead minnow, a standard USEPA test organism (Stevens et al., 1990). Recent evidence shows that pollutants in the Sacramento River adversely affect striped bass larvae (see "Recent Conditions for Striped Bass") and that periodic kills of adult striped bass, particularly in the Carquinez Strait, could be caused by contaminants (DWR and Reclamation, 1993). However, the short life span of delta smelt and their relatively low position in the food chain probably reduce the accumulation of toxic materials in their tissues and make them less susceptible to injury than species that live longer.

Introduced Species. Many exotic species have invaded the estuary in recent years and could compete with or prey on delta smelt. The recently introduced Asiatic clam (*Potamocorbula amurensis*) is currently abundant in Suisun and San Pablo bays, where it feeds on zooplankton and phytoplankton and thus could compete directly with delta smelt for prey organisms and indirectly with the smelt's zooplankton prey. Several introduced fish species, including the inland silverside, the yellowfin goby, and the chameleon goby, could also compete with or prey on delta smelt. No single invasion of exotic species parallels the decline in delta smelt closely enough to suggest that competition or predation by exotic species was a primary cause of the delta smelt's recent decline. However, the effects of multiple-species invasions, which have occurred in the estuary, are difficult to assess.

The successful invasion of the estuary by many introduced species is attributable to a variety of factors, including reduced competition and predation because of reduced abundances of many native species. One factor that could have contributed to the success of the invasions is reduced seasonal variability of Delta outflows. The abundance of many native species of the estuary, including delta smelt, is strongly influenced by seasonal patterns of Delta outflow. Development of large-scale water projects in the Central Valley and Delta have led to a 50 to 60 percent reduction in Delta outflow, and the operation of diversions and upstream dams has smoothed the

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annual pattern of outflow so that while winter flows are generally lower than in the past, summer and fall flows are much higher (Herbold et al., 1992). Variability of the physical environment greatly affects the adaptations of a species (Ricklefs, 1973). Native species of the estuary are adapted to exploit highly variable flow conditions, so a reduction in the natural variability of flows could provide a competitive advantage to less specialized exotic species.

Size of Spawning Stock. The delta smelt has low fecundity compared to many fish species, so the abundance of delta smelt in one year is potentially limited by the number of spawning adults present in the previous year. The relationship between the abundance of progeny (recruits) and the abundance of their parents (spawning stock) is known as the stock-recruitment relationship. Several stock-recruitment models have been developed to analyze this relationship (Ricker, 1975). A significant stock-recruitment relationship indicates that the abundance of recruits is determined in part by the abundance of the parent stock (i.e., the size of the population in a previous year). Conversely, if the stock-recruitment relationship is not significant, the abundance of recruits is not greatly influenced by the population's previous abundance and, therefore, a given set of environmental conditions will result in a set number of recruits regardless of the number of parent fish. It will clearly be useful to know the stock-recruitment relationship of a threatened species in developing a recovery plan.

Recent analyses of delta smelt abundances have led to the conclusion that a significant stock-recruitment relationship exists for delta smelt (DWR and Reclamation, 1993). However, it is probably not valid to use a stock-recruitment model to analyze the delta smelt population. Stock-recruitment models were derived from studies of commercially exploited fish populations. These populations differ from the delta smelt population because the various life stages of the commercially exploited populations are affected by different mortality factors (e.g., fishing versus natural mortality), whereas all life stages of delta smelt could be affected by the same important mortality factors. For instance, survival of all life stages of delta smelt could be affected by Delta outflow. If so, and if years tend to run in series of dry or wet years, then abundances of the stock and recruit year-classes will be positively correlated regardless of whether the abundance of adults has any effect on abundance of young.

The resurgence of the delta smelt population in 1993 to its highest level in 13 years, after a 12-year period of low abundance, suggests that a stock-recruitment relationship plays a small role, if any, in determining delta smelt abundance (Herbold, 1994). With or without a significant stock-recruitment relationship, the risk of extinction during a year is related to the population abundance in that year.

Larvae and Downstream Transport

Delta Outflow. The distribution and survival of delta smelt larvae are probably strongly affected by Delta outflow. As noted earlier, delta smelt larvae are buoyant for a brief period after hatching, which could improve their chances of being transported from their upstream spawning habitats to nursery areas near the entrapment zone.

Low Delta outflow and high export at the CVP and SWP pumps in the south Delta produce reverse or low net flow conditions that could keep delta smelt larvae in the Delta where they are vulnerable to entrainment and could be more vulnerable (than larvae downstream of the Delta) to

predation or other sources of mortality. DWR recently produced evidence, based on its particle tracking model, that neutrally buoyant particles in the Delta interior (east of Antioch on the San Joaquin side of the Delta) are drawn to the CVP and SWP pumps even under high positive net flow conditions (Service, 1995). The similarity of delta smelt larvae to neutrally buoyant particles is unknown, but the risk of entrainment in south Delta diversions is greater when larvae are located in the San Joaquin River side of the Delta.

When Delta outflows are low, the entrapment zone forms in the western Delta (see Attachment B), and the larvae collect in this region. It is not known whether overall survival of larvae and young juveniles near the entrapment zone is higher than survival elsewhere, even if the entrapment zone is in the Delta.

Very high Delta outflow could transport delta smelt larvae beyond nursery areas near the entrapment zone into potentially less suitable areas. In 1983, for instance, very high flows probably flushed most of the delta smelt larvae into San Pablo Bay or beyond (Stevens et al., 1990). The 1983 smelt year class apparently survived poorly because the 1983 summer TNS and fall MWT survey abundance indices were very low (Figure II-85).

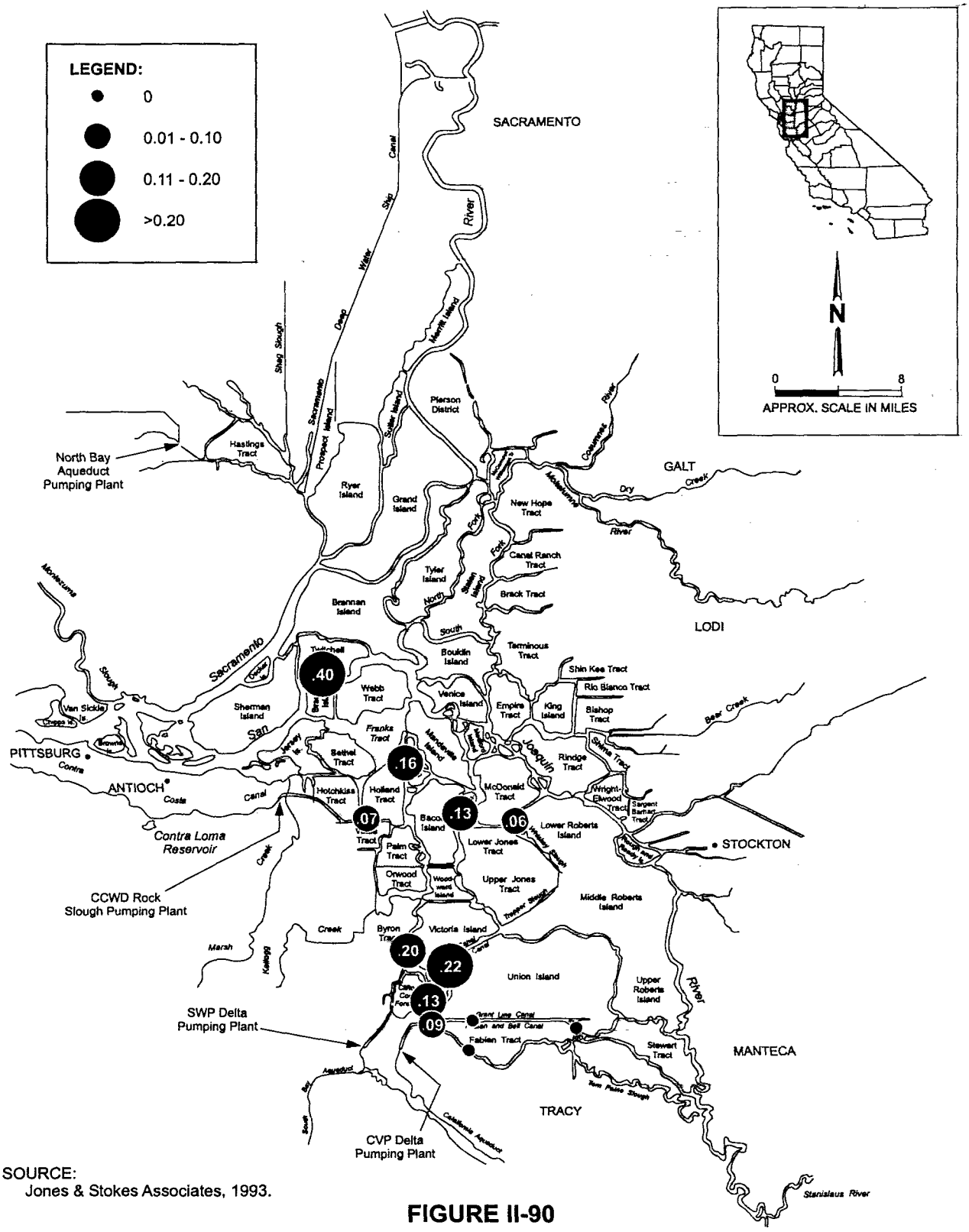
Entrainment. Since about 1985, delta smelt larvae have been sampled in the south Delta, near the CVP and SWP pumps, as part of the egg and larval survey, but they were not distinguished from longfin smelt larvae until 1989. Catch rates of delta smelt larvae near the pumps were fairly high in 1992 (Figure II-90). During 1990-1992, the larvae near the pumps peaked in abundance in April or May (Jones & Stokes Associates, 1993).

Given the high salvage rates of juveniles at the CVP and SWP pumps in some years (Figures II-88 and II-89), it is possible that a great many delta smelt larvae are also entrained. As noted earlier, fish larvae are too small to be removed by the fish screens and are not salvaged.

Entrainment of smelt larvae at PG&E's powerplants in 1978 and 1979 was estimated at approximately 50 million at the Pittsburg plant and 16 million at the Contra Costa plant. The estimates include delta smelt and longfin smelt because larvae of the two species were not differentiated. An unknown percentage of the entrained larvae could survive because the powerplants return diverted cooling water to the estuary, although other factors could reduce survival (e.g., temperature and abrasions).

Delta smelt larvae could be highly vulnerable to entrainment in agricultural diversions during April through June, when the irrigation season begins and larvae are relatively abundant in the Delta. However, few delta smelt larvae were recovered in agricultural diversions during 1992 and 1993 in DWR's agricultural diversions study (DWR and Reclamation, 1993). The low recovery rate could result from low abundance of delta smelt larvae or from larval behavior that makes them less vulnerable to entrainment. Delta smelt larvae could remain close to the bottom after they lose their buoyancy (Mager, pers. comm.), which will probably make them less vulnerable. Future sampling by the IEP will provide improved information on entrainment in agricultural diversions.

Few studies have been conducted on the entrainment of delta smelt larvae at the North Bay Aqueduct and CCWD diversions. An estimated 432 to 4,320 delta smelt larvae per day could be



SOURCE:
Jones & Stokes Associates, 1993.

FIGURE II-90

CATCH RATES OF DELTA SMELT LARVAE DURING APRIL AND MAY NEAR THE CVP AND SWP PUMPS (1990-1992)

entrained in the North Bay Aqueduct (DWR and Reclamation, 1993). Relatively few delta smelt larvae were collected at the Rock Slough station near CCWD's intake during the egg and larval survey in 1992 and 1993 (DWR and Reclamation, 1993), so entrainment of larvae by this diversion could be low.

Food Availability. Food availability could be an important factor affecting survival of delta smelt larvae. Abundance of rotifers and phytoplankton has declined in recent years (Obrebski et al., 1992); these prey are small and could be important in the diet of delta smelt larvae (DWR and Reclamation, 1993) and other fish larvae (Hunter, 1981). Year-class strength of many fish populations, particularly species with planktonic larvae, is believed to be strongly influenced by concentrations of small-sized prey during the larval life stage (Lasker, 1981).

Water Quality

Salinity. Delta smelt larvae are generally found in freshwater or slightly brackish water. They are probably less tolerant of high salinities than are older smelt.

Water Temperature and Transparency. Effects of water temperature and transparency on larvae and downstream transport are similar to effects described previously for adults.

Pollution. Effects of contaminants on larvae are similar to effects described previously for adults. Larvae, however, could be more sensitive to toxic substances.

Introduced Species. The effects of introduced species on larvae, which are largely unknown, are similar to effects described previously for adults. Different species and impact mechanisms are likely involved but are unknown.

Juveniles and Rearing Habitat

Delta Outflow. The area just upstream of the upper end of the entrapment zone is the principal rearing habitat of juvenile delta smelt. The position of the entrapment zone in the estuary could have a major influence on the quality of the rearing habitat and is determined by Delta outflow. However, there is no simple relationship between Delta outflow and abundance of delta smelt (Stevens and Miller, 1983). Much of the following discussion applies to adult delta smelt as well as juveniles, but it is likely that the position of the entrapment zone primarily affects adult abundance through effects on survival of larvae and juveniles.

The means of the fall MWT survey and summer TNS delta smelt annual abundance indices for dry years were not significantly different from the means for wet years (DWR and Reclamation, 1993). Delta smelt abundance was severely depressed during very high outflow years, such as 1967, 1969, 1982, 1983, and 1986, and during very low outflow years, such as 1987-1990 and 1992. Moyle et al. (1992) suggested that delta smelt benefited when Delta outflows were moderate, but they found no statistically significant effect of moderate flows on the fall MWT survey abundance index.

Recent evidence suggests that delta smelt abundance is related to the position of X2. X2 is a generally reliable index of the geographic position of the upstream end of the entrapment zone and is largely determined by Delta outflow (see Attachment B). It is measured as the distance in kilometers from the Golden Gate Bridge. Herbold (1994) demonstrated that the fall MWT survey abundance index is positively correlated with the number of days during the spawning season (February through June) that X2 is located in Suisun Bay (i.e., Chipps Island to Benicia [see Attachment B]). The highest correlation coefficient (r equals 0.56) was obtained when the analysis was limited to the reach of Suisun Bay between kilometer 65 and kilometer 70 (see Attachment B). No significant correlation was obtained when the summer TNS abundance index was used rather than the fall MWT survey index (Figure II-91) (Herbold, pers. comm.).

A further test of the hypothesis that delta smelt abundance is related to X2 was conducted by comparing abundance indices for years when X2 during February through June is mostly in Suisun Bay with abundance indices for years when X2 is mostly upstream or downstream of Suisun Bay (see Attachment B). Years were divided into three groups according to frequency of location of February-June X2: years with February-June X2 most frequently located in Suisun Bay, years with February-June X2 most frequently located upstream of Suisun Bay, and years with February-June X2 most frequently located downstream of Suisun Bay. The means of the fall MWT survey index and of the summer TNS were compared for the three groups using analysis of variance (ANOVA) (SAS Institute, 1990b). The abundance indices were log-transformed to correct for hetero-scedasticity (Green, 1979).

The fall MWT survey index was significantly higher (p is less than 0.05) for years when X2 was most frequently in Suisun Bay during February through June than for years when X2 was most frequently upstream or downstream of Suisun Bay (Table II-11). The summer TNS index was significantly higher for years when X2 was most frequently in Suisun Bay during February through June than for years when X2 was most frequently upstream of Suisun Bay, but was not significantly different than for years when X2 was most frequently downstream of Suisun Bay (Table II-11). These results corroborate Herbold's results for the fall MWT survey index. However, the results conflict with his results for the summer TNS index because he found no significant correlation between the number of days that X2 is in Suisun Bay and the summer TNS index. One reason that the analyses using the summer TNS index gave contradictory results is that two years with anomalous summer TNS index values, 1976 and 1977 (Figure II-91), had a greater influence on the results of the correlation analyses than on the results of the ANOVA. It is not clear which analysis procedure is more appropriate for testing the hypothesis that delta smelt abundance is related to the number of days that X2 is in Suisun Bay (it depends mainly on whether the effect of the number of days with X2 in Suisun Bay is considered continuous or discontinuous; either could be reasonably argued). In any case, the results suggest that usually both the summer TNS index and the fall MWT survey index are positively related to the number of days that X2 is in Suisun Bay.

Herbold's 1994 results and the results in Table II-11 indicate that the delta smelt population increases in years when X2 is located in Suisun Bay for a number of days during February through June, but the analyses do not elucidate the mechanism responsible for the increase in abundance. Possible explanations for the relationship include the following: rearing conditions for larvae and juveniles are optimal when the entrapment zone is in Suisun Bay (Moyle et al., 1992), transport conditions for larvae are optimal when flows are such as to maintain X2 in Suisun Bay, and production of prey species of delta smelt larvae and juveniles is maximized

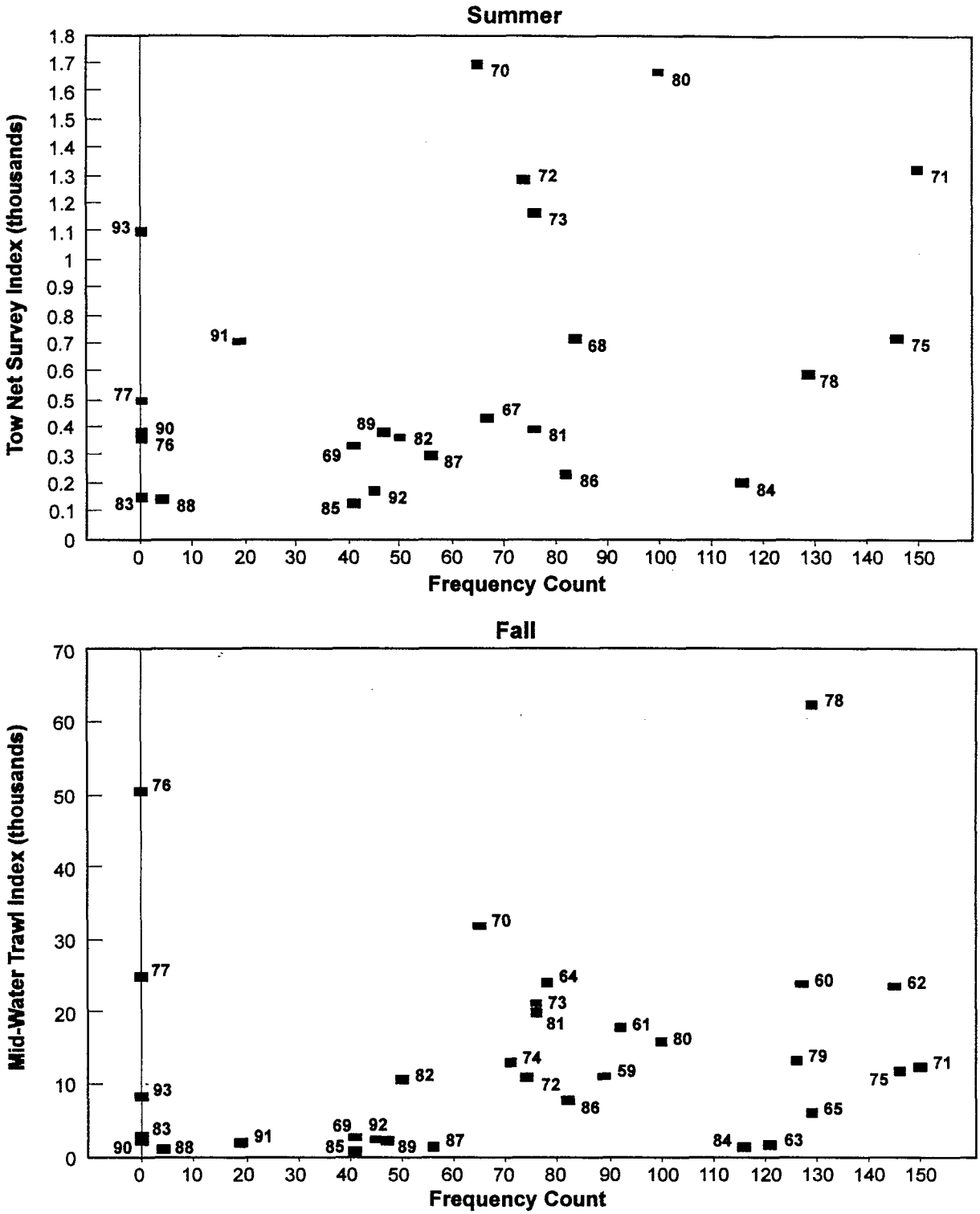


FIGURE II-91

PLOTS OF DELTA SMELT ABUNDANCE INDICES VERSUS DAYS X2 IN SUISUN BAY FOR FEBRUARY THROUGH JUNE ONLY (1959-1993)

TABLE II-11

ANALYSIS OF VARIANCE RESULTS COMPARING DELTA SMELT ABUNDANCE INDICES FOR YEARS WITH X2 MOST FREQUENTLY IN SUISUN BAY, UPSTREAM OF SUISUN BAY, OR DOWNSTREAM OF SUISUN BAY

Source of Variation	df	Mean Square	F Value	*p* Value
Log-transformed fall MWT survey index				
Area with X2	2	4.135	3.79	0.039
Suisun Bay vs. downstream	1	2.452	4.49	0.046
Suisun Bay vs. upstream	1	3.134	5.74	0.026
Error	22	0.546		
Log-transformed summer TNS index				
Area with X2	2	4.957	4.17	0.026
Suisun Bay vs. downstream	1	2.004	1.69	0.204
Suisun Bay vs. upstream	1	9.488	7.98	0.008
Error	29	1.188		

NOTE:
Significant effect ($p < 0.05$).

when outflows are such as to maintain X2 in Suisun Bay. All these explanations require that early juveniles increase in abundance in years when X2 is most frequently in Suisun Bay, but, as discussed previously, results of the summer TNS analyses are ambiguous about whether such a relationship exists. An increase in delta smelt abundance when X2 is in Suisun Bay during February through June will be particularly difficult to explain if it applied to the fall MWT survey index but not the summer TNS index.

Entrainment. Typically, in years with relatively high Delta outflows, delta smelt larvae and young juveniles are transported sufficiently far downstream to avoid being entrained in the major diversions of the south Delta. Once in the entrapment zone, the juveniles are generally not vulnerable to entrainment at the CVP and SWP pumps. During the recent drought, however, outflows have been low and exports have been high, so many juveniles have remained in the Delta and have been entrained at the CVP and SWP pumps. Most juveniles are entrained from April through August (Figure II-89). Entrainment of juveniles at other diversions has not been studied, but entrainment at PG&E powerplants and agricultural diversions in the western Delta could be high because of the facilities' proximity to the entrapment zone.

Food Availability, Water Quality, and Introduced Species. Food availability, water quality, and introduced species that affect juvenile delta smelt are the same as those that affect other life stages. These factors were discussed in previous sections.

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

For many years, delta smelt were sampled incidentally in programs designed to study other species, particularly striped bass. The fall MWT survey and summer TNS are the two most important of these programs. Recently, however, the IEP initiated an ambitious field and laboratory study program for delta smelt: the Delta Smelt Study Plan.

The Delta Smelt Study Plan was developed by DFG in consultation with other agencies and organizations. Its purpose is to monitor and investigate factors potentially affecting delta smelt population levels to ensure the species' long-term survival and to address the management and recovery objectives set forth by the California Fish and Game Commission in 1990 (Sweetnam and Stevens, 1991). It has also provided the Service with information relevant to the federal listing of delta smelt as a threatened species. The plan encompasses ongoing sampling programs, such as the fall MWT survey and summer TNS, and new studies designed to improve knowledge of delta smelt spawning, feeding, growth, survival, habitat preferences, genetic structure, and population dynamics and other aspects of delta smelt ecology. The studies are expected to produce information about delta smelt adults, juveniles, and larvae that will facilitate efforts to protect delta smelt and improve habitat conditions.

The MWT survey has been expanded to include all months and several new survey stations. The additional sampling will provide a more complete understanding of the timing and distribution of delta smelt life history events and more reliable information about trends in abundance.

Two new or expanded studies address questions concerning distribution of adult delta smelt and spawning locations. The beach seine and artificial substrate surveys provide information on location and timing of spawning. The beach seine survey was recently expanded to include additional potential delta smelt spawning locations.

Many of the new studies of the Delta Smelt Study Plan will address questions about delta smelt larvae and early juveniles. As noted earlier, survival of the early life stages is believed to be critical in determining year-class abundance. The egg and larvae survey has been expanded in recent years to include stations in areas potentially important to young delta smelt. This survey will help identify factors controlling survival and abundance of the smelt and their food supply. Zooplankton samples and analyses of smelt diet will be used to assess feeding conditions in different areas of the Delta and under different hydrologic conditions. Food supply could be an important determinant of larval survival.

A study of otoliths (i.e., ear bones) from delta smelt larvae and juveniles initiated in 1992 will provide estimates of daily growth rates. Growth information will be used to calculate the time of smelt hatching and, combined with environmental information such as outflows, diversions, food supply, salinity, water temperature, and other factors, it is expected to reveal how environmental conditions affect delta smelt growth and survival.

A new study will use histologic (tissue analysis) and morphometric (body size measurement) methods to compare the condition of larvae in the field to that of larvae exposed to starvation, toxicity, and parasites in the laboratory. This information could help identify factors affecting growth and survival in different areas of the Delta.

LONGFIN SMELT

Stock Abundance and Distribution

Longfin smelt (*Spirinchus thaleichthys*) are widely distributed in estuaries on the Pacific Coast. They have been collected from numerous river estuaries in California from San Francisco Bay to Oregon, but the largest population inhabits the Sacramento-San Joaquin Delta estuary. Until 1963, the population in the estuary was considered a distinct species known as Sacramento smelt. All longfin smelt populations are now classified as one species. Apparent declines in longfin smelt abundance in the estuary led to a petition to list the longfin smelt under the federal ESA. The Service determined that the petitioned action is not warranted at this time (Federal Register 59:4, January 6, 1994).

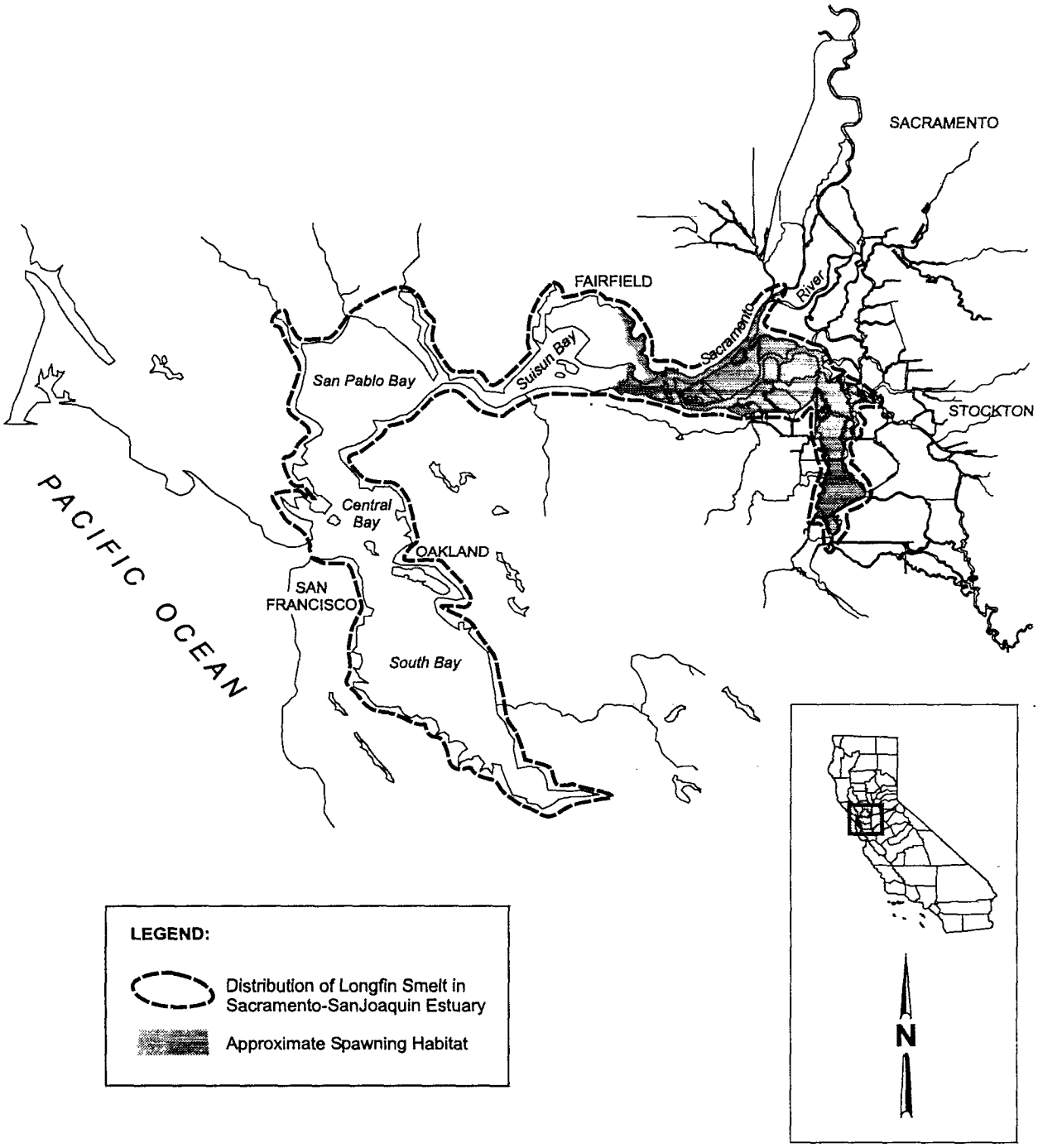
Longfin smelt are euryhaline (i.e., adapted to a wide range of salinities) and anadromous. Spawning adults occur seasonally as far upstream in the Delta as Rio Vista, Medford Island, and the CVP and SWP pumps (Figure II-92). Except when spawning, longfin smelt are most abundant in Suisun and San Pablo bays, where salinities generally range between 2 and 20 parts per thousand (Natural Heritage Institute, 1992). Prespawning adults and yearling juveniles are generally most abundant in San Pablo Bay and downstream areas, whereas the young-of-year occur primarily in San Pablo and Suisun bays. Adults are found seasonally as far downstream as South Bay and are occasionally collected in the open ocean. Before Shasta Dam was constructed in 1944, saline water intruded in dry months as far upstream in the Delta as the City of Sacramento; therefore, it is likely that longfin smelt periodically ranged much farther upstream during that time than they currently do (Herbold et al., 1992).

Several IEP surveys sample longfin smelt in addition to other fish species. However, none of these surveys encompass the entire geographic range of longfin smelt in the estuary. DFG's monthly bay survey initiated in 1980 samples 42 locations in the lower estuary from the South Bay upstream to the confluence of the Sacramento and San Joaquin rivers. This survey uses MWT surveys and otter trawls, which catch primarily adult and juvenile smelt.

DFG's summer TNS survey, initiated in 1959 and conducted in June and July, samples 30 stations in the Delta and Suisun Bay and one station in eastern San Pablo Bay. This area encompasses the spawning habitat of longfin smelt and much of the nursery habitat. The TNS survey collects only young smelt because older juveniles and adults can escape the tow net.

DFG's fall MWT survey, initiated in 1967 and conducted during September through December, samples approximately 90 stations extending from Stockton and Walnut Grove in the Delta to the middle of San Pablo Bay. This survey primarily samples young-of-year longfin smelt because the young-of-year constitute the bulk of the catches in fall of most years (DFG, 1987).

Results of the MWT survey are generally used to demonstrate longfin smelt abundance trends because the survey has a long and nearly continuous sampling record, encompasses more of the geographic range of longfin smelt in the estuary than the other surveys, and samples older fish than the summer TNS. DFG computes longfin smelt abundance for each month and for 17 subregions of the estuary. The longfin smelt annual abundance index is computed by weighing



SOURCE:
Service, 1994a.

FIGURE II-92

DISTRIBUTION OF LONGFIN SMELT AND THE APPROXIMATE LOCATION OF SPAWNING HABITAT

the mean catch per trawl for each subregion by the estimated total water volume of the subregion and summing the weighted means over all subregions and months.

Historical Population Trends (1850-1966). Longfin smelt are characterized as being abundant historically in the estuary (Moyle, 1976), but specific data on abundance or abundance indices are unavailable.

Current Population Trends (1967-1991). Before about 1984, longfin smelt was one of the most abundant fish species in the Sacramento-San Joaquin Delta estuary, but abundance fluctuated widely between drought years (e.g., 1976-1977) when abundance was low and wet years (e.g., 1982-1983) when the population quickly recovered (Figure II-93). Longfin abundance was very low from 1987 to 1992, with 1992 having the lowest index on record. Abundance increased somewhat in 1993. Longfin smelt young-of-year abundance estimates for 1980-1990 derived from the Bay survey closely parallel the fall MWT survey (DFG, 1992b).

Life History

Longfin smelt are small (usually less than 5 inches long), plankton-feeding fish that typically live for two years (Moyle, 1976). They are commercially harvested with other smelt species in and near the Bay. Between 1985 and 1988, the total annual catch of all smelt species ranged between 20,000 and 40,000 pounds, but it is not known what portion of the catch consisted of longfin smelt (DFG, 1992b). Longfin smelt have little direct commercial value but are potentially important prey for commercially important predator species, such as salmon and steelhead. Longfin smelt life history is depicted in Figure II-94.

Adults. Longfin smelt begin to mature late in their second summer (August or September) of life (Figure II-94). As they mature, the smelt begin migrating upstream from the San Francisco and San Pablo bays toward Suisun Bay and the Delta. Most longfin smelt spawn and die at two years of age. Occasionally, young-of-year could spawn, and a few fish (mostly females) live a third year and grow to approximately 5 inches long (DFG, 1992b; Natural Heritage Institute, 1992). Larger longfin smelt spawn later in the season than smaller smelt (Wang, 1986).

Adult longfin smelt appear to inhabit shoal and channel areas equally (DFG, 1987). The primary prey of adult longfin smelt is the opossum shrimp (*Neomysis mercedis*) (Natural Heritage Institute, 1992). A variety of fish, birds, and mammals prey on smelt (DFG, 1992b).

Spawning. Most longfin smelt spawn from February through April (Figure II-94), although some spawning could occur any time from November through June (Natural Heritage Institute, 1992). Given the length of the spawning season, it is possible that individual smelt spawn more than once per year. Spawning occurs at night, with a female depositing approximately 5,000 to 24,000 eggs at one time (Natural Heritage Institute, 1992). The eggs are adhesive and are probably deposited on rocks or aquatic plants.

Longfin smelt spawn in freshwater primarily in the upper end of Suisun Bay and in the lower and middle Delta. In the Delta, they spawn mostly in the Sacramento River channel and adjacent sloughs (Figure II-92) (Wang, 1991). During the 1987-1992 drought, when saline water intruded into the Delta, larval longfin smelt were found near the CVP and SWP pumps, well upstream of

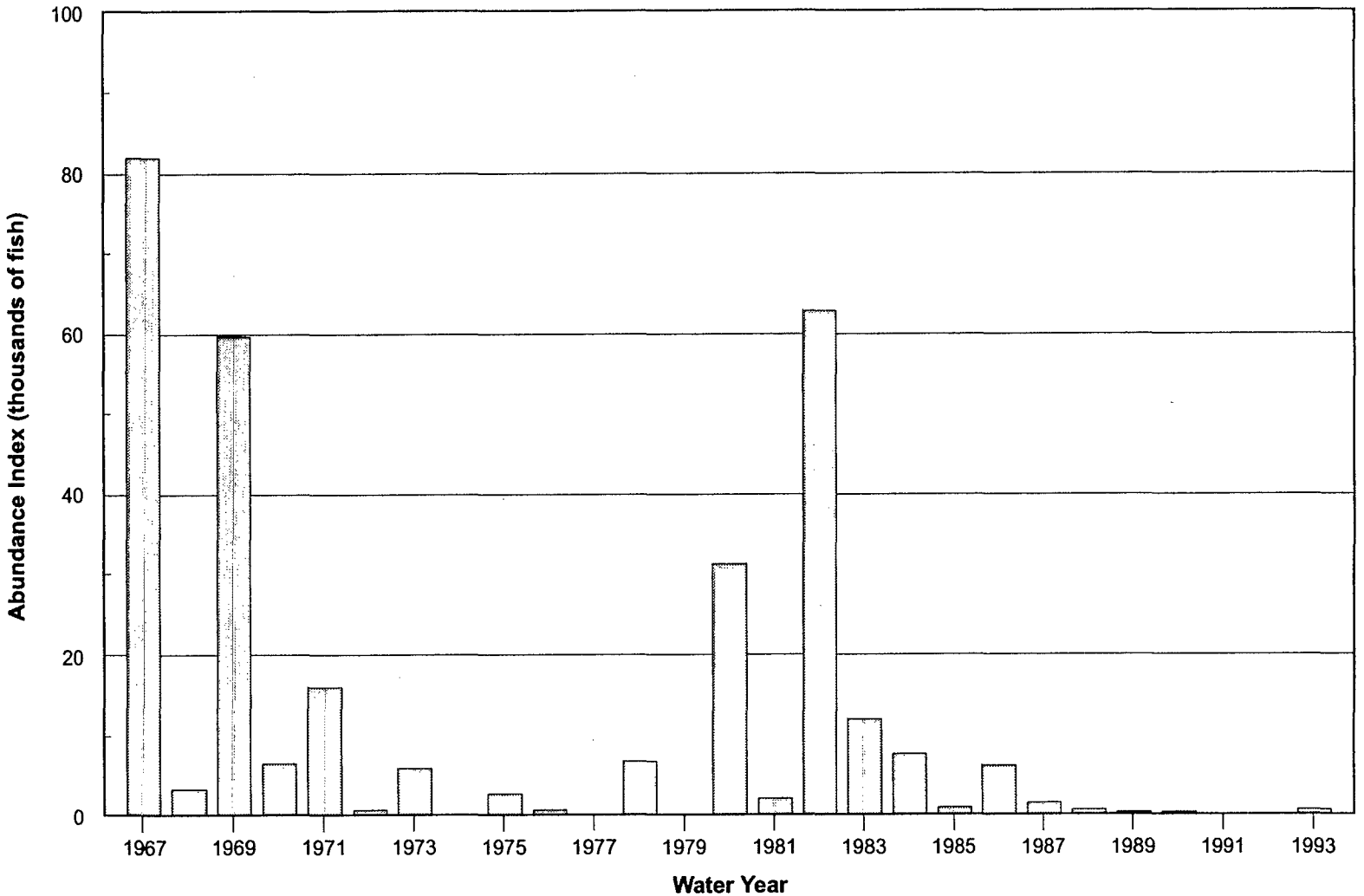


FIGURE II-93

ANNUAL ABUNDANCE INDEX OF LONGFIN SMELT IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY (1967-1993)

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C-081561

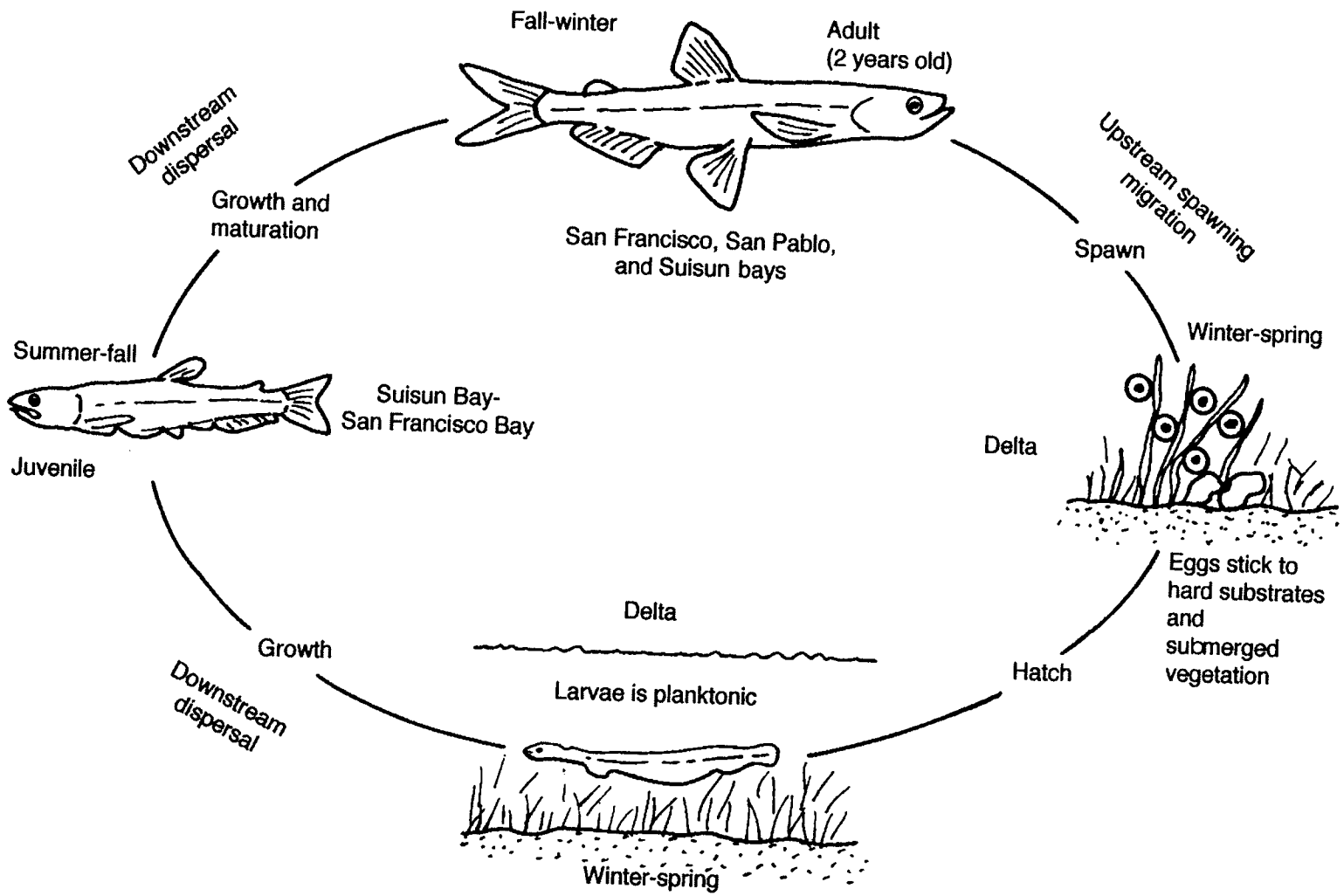

FIGURE II-94

LIFE HISTORY OF LONGFIN SMELT

C-081562

C-081562

the usual spawning habitat (Wang, 1991). The areas of the estuary used for spawning by longfin smelt are similar to the spawning areas of delta smelt (Figure II-84).

Spawning. Most longfin smelt spawn from February through April (Figure II-94), although some spawning could occur any time from November through June (Natural Heritage Institute, 1992). Given the length of the spawning season, it is possible that individual smelt spawn more than once per year. Spawning occurs at night, with a female depositing approximately 5,000 to 24,000 eggs at one time (Natural Heritage Institute, 1992). The eggs are adhesive and are probably deposited on rocks or aquatic plants.

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Egg Incubation and Larval Rearing. In Lake Washington (the State of Washington), longfin smelt eggs hatch in 37 to 47 days at 45 degrees Fahrenheit (DFG, 1992b). The peak of larval abundance in the Sacramento-San Joaquin Delta estuary occurs from February through April (DFG, 1992b).

Shortly after hatching, longfin smelt larvae develop a gas bladder that allows them to remain near the water surface (Wang, 1991). The larvae do not migrate vertically and remain near the surface during both the flood and ebb tides (DFG, 1992b). Larvae in near-surface waters are swept downstream into nursery areas in the western Delta and Suisun and San Pablo bays (Figure II-94) (Baxter, pers. comm.; DFG, 1987). The ultimate distribution of longfin smelt larvae in the estuary is determined by Delta outflow during the period of larval development (February through May), with larvae being dispersed farther downstream in high outflow years than in low outflow years (DFG, 1992b).

As noted earlier, delta smelt larvae do not develop a gas bladder until approximately one to two months after hatching (Mager and Baxter, pers. comms.). The difference in the time of gas bladder development of longfin smelt and delta smelt could be of major ecological importance. Delta smelt larvae are buoyant for four to five days after hatching because of the buoyancy of the oil globule in the yolk sac; after this period, they sink to the bottom. Longfin smelt larvae, however, remain buoyant after yolk sac absorption. Early gas bladder development by longfin smelt causes the larvae to remain near the surface much longer than delta smelt larvae and could explain why the longfin smelt larvae are dispersed farther downstream in the estuary than delta smelt larvae (Baxter, pers. comm.).

Little information on food habits of longfin smelt larvae is available, but fish larvae of most species, including delta smelt, feed on phytoplankton and small zooplanktons, such as rotifers and copepod nauplii (Hunter, 1981; DWR and Reclamation, 1993).

Juvenile Rearing. Metamorphosis of longfin smelt from larval to juvenile form probably begins 30 to 60 days after hatching, depending on water temperature (Natural Heritage Institute,

1992). The distribution of young-of-year juveniles is largely determined by the final distribution of the larvae (DFG, 1992b). During their first summer and fall, the juveniles occur primarily in the western Delta and in Suisun and San Pablo bays (DFG, 1992b). In the following winter and spring, when outflows are highest, the juveniles disperse farther downstream. Therefore, yearlings are generally abundant in San Francisco, San Pablo, and Suisun bays (DFG, 1987). The yearlings occasionally venture into the ocean.

Juvenile longfin smelt appear to inhabit shoal and channel areas almost equally (DFG, 1987) but could sometimes favor intermediate and bottom depths over the surface layer (DFG, 1992b). In Lake Washington, longfin smelt migrate vertically between the upper water column at night and deep water during the day (Natural Heritage Institute, 1992).

Juvenile longfin smelt feed on copepods and cladocerans, but mysids (*Neomysis mercedis*) are their most important prey, particularly for large juveniles and adults. Juveniles have also been found feeding on fish larvae (Baxter, pers. comm.). Most smelt growth in length occurs during their first summer, when they typically reach 2.5 inches in length. During the second summer, lengths reach 4 inches and weight gain could be considerable because of gonadal development (Natural Heritage Institute, 1992).

Factors Affecting Abundance

Adults and Spawning Migration

Delta Outflow. The distribution of adult longfin smelt is related to winter and spring Delta outflow. In high outflow years, adult longfin smelt are most abundant in San Francisco, San Pablo, and Suisun bays, but in low outflow years they are concentrated in the Delta and Suisun Bay (Natural Heritage Institute, 1992). The distribution of adults could result primarily from their dispersal as larvae by outflow events during their first winter and spring and as juveniles during their second winter and spring.

Yearling longfin smelt begin to mature in late summer or early fall. It is unlikely that Delta outflow affects the timing of maturation because outflow at this time of year is relatively low and constant. However, it is possible that outflow influences the timing of the spawning migration, which could begin as early as November (Natural Heritage Institute, 1992). As noted previously, evidence exists that delta smelt begin spawning earlier in early season high outflow years than in low outflow years. Although this same effect has not been studied, it could also occur for longfin smelt.

Delta outflow probably affects where in the Delta or Suisun Bay longfin smelt adults migrate to spawn. In high outflow years, upper Suisun Bay has freshwater, which could encourage spawning in Suisun Bay, while in low outflow years, the smelt must migrate into the Delta to reach conditions suitable for spawning (see Attachment B). When outflow is low and exports at the CVP and SWP pumps are high, the net flow in the lower San Joaquin and Sacramento rivers could be toward the pumps rather than downstream. These reverse flows, which contain freshwater drawn from the Sacramento River, could encourage greater than normal upstream migration of the adults in the south Delta. In the south Delta, adults and their larvae are vulnerable to entrainment and other sources of mortality.

Salinity. Longfin smelt adults are generally found at salinities ranging from seawater to freshwater. When stations of the Bay survey were grouped according to salinity class, the mean catch per trawl of adult longfin smelt during 1980-1985 was highest at a salinity of approximately 7 parts per thousand (DFG, 1987); however, mean catches were high at salinities ranging as high as 27 parts per thousand. No evidence exists that salinity has any direct effect on adult longfin smelt, but, as noted above, freshwater flow patterns could greatly influence spawning migrations.

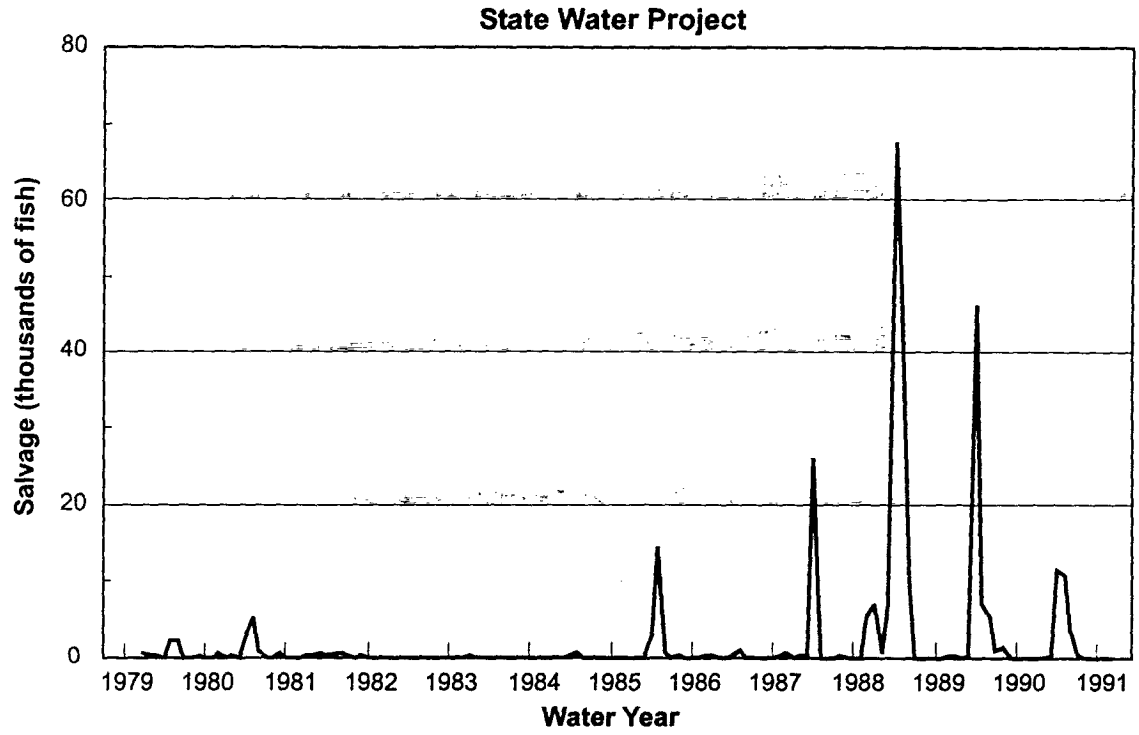
Entrainment. Entrainment of longfin smelt by diversion and export facilities, such as the CVP and SWP pumps, affects only spawning adults, larvae, and early juveniles. Older juveniles and prespawning adults inhabit downstream portions of the estuary (Suisun and San Pablo bays), which have smaller diversions. Although longfin smelt entrained at the CVP and SWP facilities are salvaged and returned by truck to downstream areas in the Delta, survival of the salvaged smelt is believed to be low (Natural Heritage Institute, 1992).

Annual salvage at both the CVP and SWP facilities has varied greatly (Figure II-95). Salvage represents entrainment but is often much less than total entrainment because fish smaller than approximately 1 inch long pass through the fish screens at the salvage facilities and cannot be salvaged. With the exception of 1986 (a wet year), the annual salvage of longfin smelt at the CVP and SWP pumps was much higher during 1984-1990 than during 1979-1983 (Figure II-95) despite substantially higher abundance during 1980, 1982, and 1983 (Figure II-93). The increased entrainment resulted from lower Delta outflows and greater exports (see Attachment B). Low outflow and high exports could increase entrainment because of the greater proportion of Delta inflow that is exported.

Figure II-93 shows that the longfin smelt population declined substantially after 1984. This decline could be partially attributable to increased entrainment by the CVP and SWP pumps and other diversions, but reduced Delta outflow (which is influenced by exports) is probably more important.

Entrainment of adult longfin smelt has a potentially greater adverse effect on the population than entrainment of larvae and young juveniles because, unless the adults have already spawned, their reproductive value (Ricklefs, 1973) is much greater than that of younger fish. The number of adult longfin smelt salvaged at the CVP and SWP pumps was estimated by analyzing data obtained from DFG on lengths of longfin smelt salvaged during 1981-1990 (Barrow, pers. comm.). Salvaged fish greater than 50 millimeters in length were assumed to be adults (DWR and Reclamation, 1993). On average, nearly all fish salvaged during November through February were adults, which migrate and spawn during these months (Figure II-96). Later, juveniles from this spawn dominated the salvage. The mean size of the juveniles increased after May because of growth and because numbers of new recruits decreased.

Food Availability. Abundance of opossum shrimp (*Neomysis mercedis*) and other zooplankton prey of longfin smelt has declined in recent years (Obrebski et al., 1992). Significant habitat changes and the introduction of exotic species likely have a major effect on estuarine food webs and longfin smelt prey abundance and distribution. It is not known, however, what specific effects declines in prey abundance have had on longfin smelt adult populations.



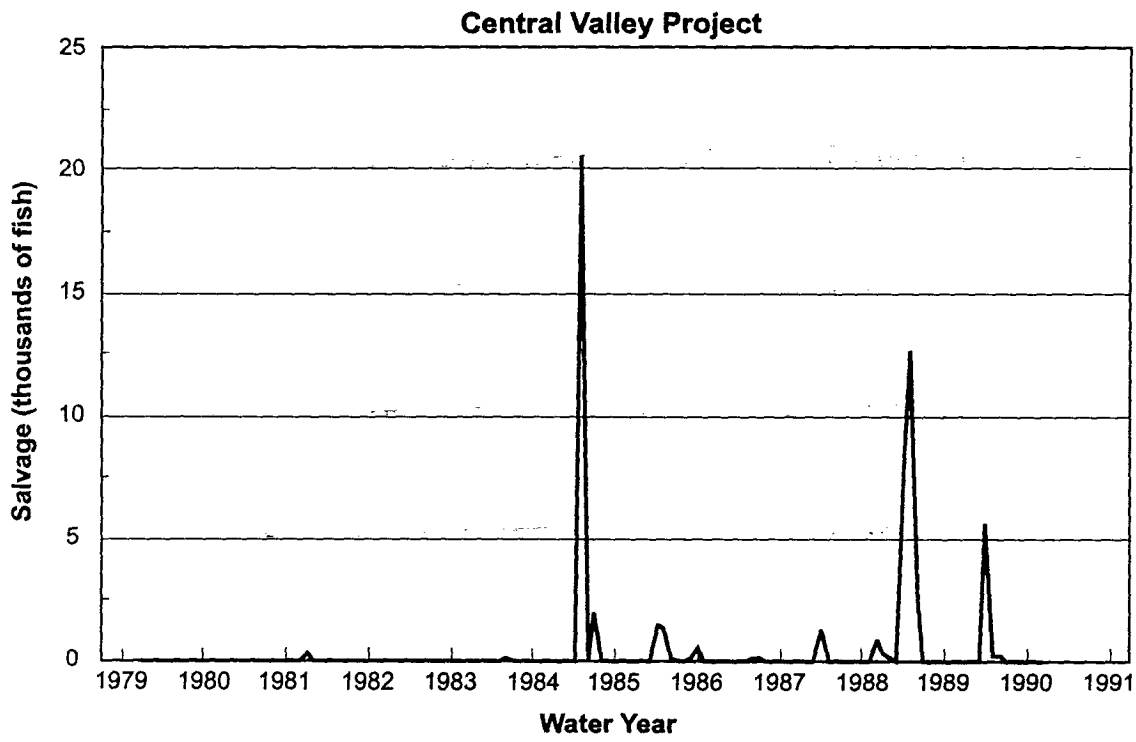


FIGURE II-95

SALVAGE OF LONGFIN SMELT AT THE SWP AND CVP FACILITIES (1979-1991)

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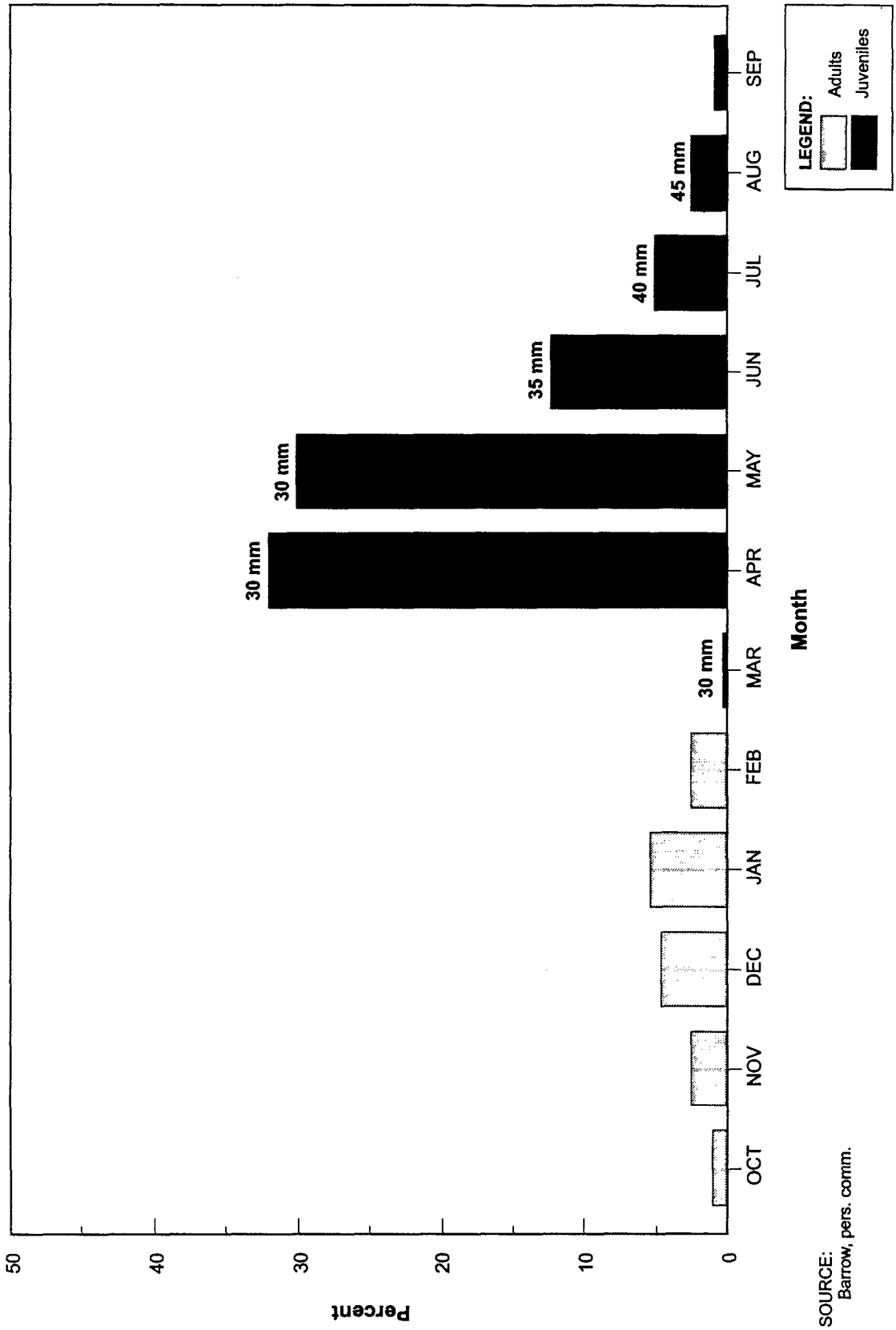

SOURCE:
Barrow, pers. comm.

FIGURE II-96

PERCENT ADULT AND JUVENILE LONGFIN SMELT OF TOTAL FISH SALVAGED ANNUALLY AT SWP AND CVP FACILITIES

Water Quality. Agricultural chemicals (including pesticides and herbicides), heavy metals, petroleum-based products, and other waste materials toxic to aquatic organisms enter the estuary through non-point runoff, agricultural drainage, and municipal and industrial discharges. Although the effects of toxic substances have not been tested on longfin smelt, recent bioassays by the Central Valley Regional Water Quality Control Board indicate that water in the Sacramento River is periodically toxic to larvae of the fathead minnow, a standard USEPA test organism (Stevens et al., 1990). However, the short life span of longfin smelt and their relatively low position in the food chain probably reduce the accumulation of toxic materials in their tissues and make them less susceptible to injury than many other fish (Natural Heritage Institute, 1992).

Introduced Species. Many exotic species have invaded the estuary in recent years and could compete with or prey on longfin smelt. The recently introduced Asiatic clam (*Potamocorbula amurensis*) is now abundant in Suisun and San Pablo bays, where it feeds on zooplankton and phytoplankton, and thus could compete directly with longfin smelt for prey organisms and indirectly with the smelt's zooplankton prey. Several introduced fish species could also compete with or prey on longfin smelt. No single invasion of exotic species parallels the decline in longfin smelt closely enough to suggest that competition or predation by the species was a primary cause of the longfin smelt's recent decline; however, the effects of multiple-species invasions, which have occurred in the estuary, are difficult to assess.

Larvae and Downstream Transport

Delta Outflow. The distribution and survival of longfin smelt larvae are strongly related to Delta outflow. As noted previously, the early development of an air bladder equips longfin smelt larvae for efficient transport from their spawning habitat in the Delta to nursery areas in the bays.

Results of sampling surveys indicate that higher outflows lead to greater downstream dispersion and higher survival of longfin smelt larvae. In low outflow years (1981, 1985, 1987, and 1988), longfin smelt larvae were found primarily in the western Delta and Suisun Bay, while in high outflow years (1980, 1982-1984, and 1986) (Figure II-97), the larvae were equally or more abundant in San Pablo, Central, and South bays. An index of survival of larvae and young juveniles, computed as the ratio of the fall MWT survey abundance index to the abundance of longfin smelt larvae in the previous spring, was strongly correlated (r equals 0.95) with December through August Delta outflow (DFG, 1987). This relationship suggests that Delta outflow or factors associated with outflow affect survival of larvae and early juveniles.

Low Delta outflow and high export from the south Delta produce reverse or low net flow conditions that keep longfin smelt larvae in the Delta for longer periods. Larvae in the Delta are vulnerable to predation, entrainment, and other sources of mortality.

Salinity. Although all life stages of longfin smelt are euryhaline, larvae are generally not found in full seawater and could be less tolerant of high salinities than older smelt.

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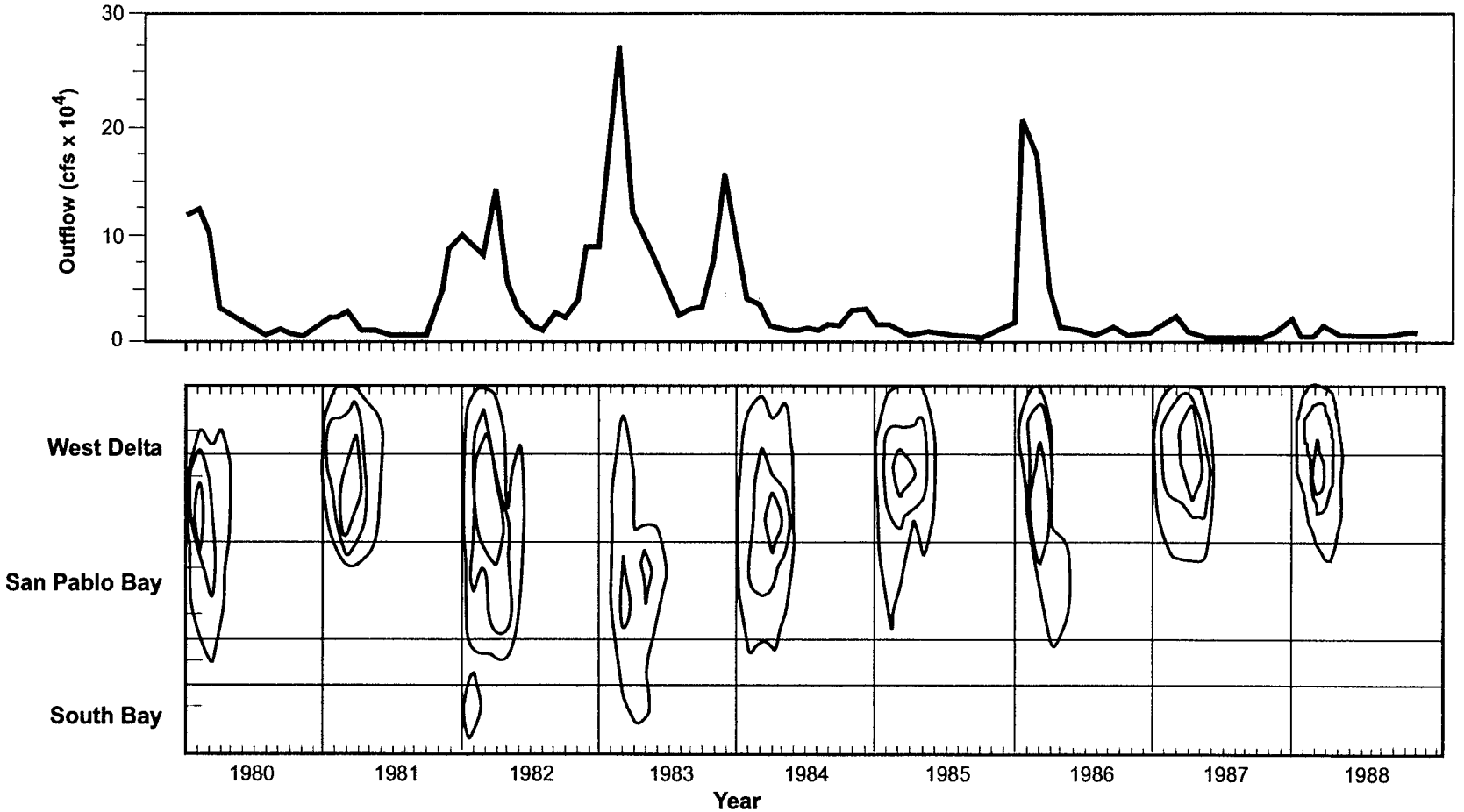

NOTE:
Average monthly outflow at Chipps Island (cfs x 10,000) from 1980 through 1988 with topographic representation of the geographical (vertical axis) and temporal (horizontal axis) longfin smelt larval distribution in the San Francisco Bay. Contour lines represent average larval densities of 50, 150, and 500 larvae per 1,000 m^3 of water filtered by the plankton net.

SOURCE:
DFG, 1992b.

FIGURE II-97

LONGFIN SMELT LARVAL DENSITIES AND DISTRIBUTION IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY (1980-1988)

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Entrainment. Longfin smelt larvae have been sampled in the south Delta near the CVP and SWP pumps as part of the egg and larval survey since about 1985, although longfin smelt larvae were not distinguished from delta smelt larvae until 1989. Total catch from samples collected in 1988, 1990, and 1992 had fewer than 10 longfin smelt larvae (Spaar, 1990, 1993; Wang, 1991). Nevertheless, given the high salvage rates of young-of-year juveniles at the pumps in some years (Figures II-95 and II-96), it can be assumed that many thousand longfin smelt larvae were entrained during those years. As noted earlier, fish larvae are too small to be removed by the fish screens and therefore are not salvaged.

Entrainment of longfin smelt larvae at PG&E powerplants, agricultural diversions, and other diversions has not been studied.

Food Availability. Food availability could be important for survival of longfin smelt larvae because abundance of rotifers and phytoplankton has declined in recent years (Obrebski et al., 1992). These are small-sized prey required for feeding by delta smelt larvae (DWR and Reclamation, 1993) and other fish larvae (Hunter, 1981). Year-class strength of many fish populations, particularly species with planktonic larvae, is believed to be strongly influenced by feeding conditions during the larval life stage (Lasker, 1981).

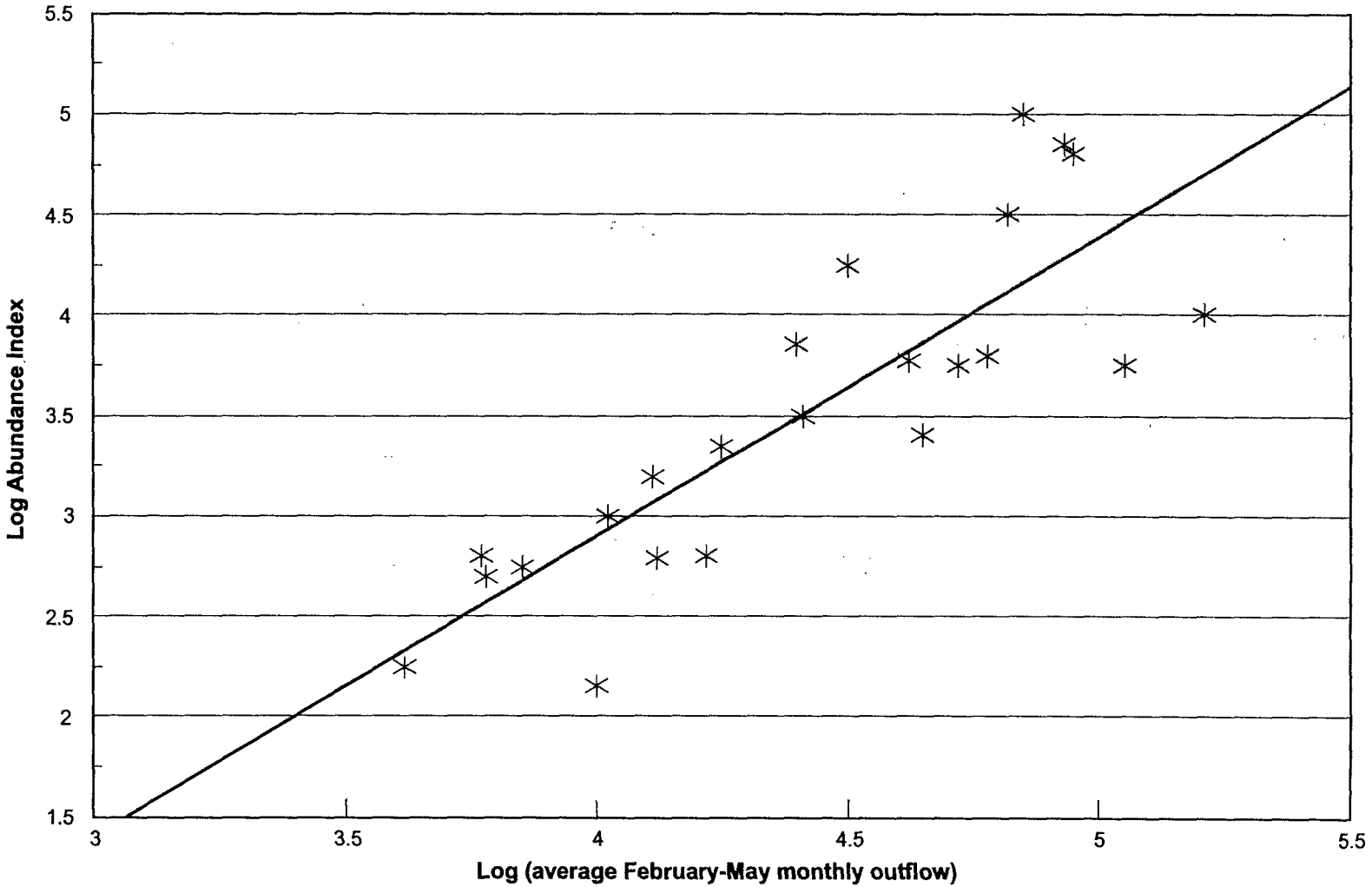
Water Quality. Water quality effects on larvae are similar to effects described previously for adults. Larvae could be even more susceptible, however, to toxic substances.

Introduced Species. The effects of introduced species on larvae are similar to effects described previously for adults. Although different species and impact mechanisms are likely involved, longfin smelt larvae are affected to an unknown degree by the numerous exotic invertebrate and fish species currently thriving in the estuary's ecosystem.

Juveniles and Rearing Habitat

Delta Outflow. The distribution of young-of-year juveniles could be largely determined by how the larvae were dispersed by Delta outflows in winter and spring (DFG, 1992b). During their second winter and spring, juveniles could be further dispersed by outflows, so yearling juveniles and adults generally have a more downstream distribution in the estuary than do young-of-year (Baxter, pers. comm.). Distribution of juveniles could not result from dispersion but could be the result of juvenile response to the salinity gradient that is a function of outflow.

The abundance of young-of-year longfin smelt is strongly related to Delta outflow (Stevens and Miller, 1983; DFG, 1987, 1992b). Linear regression analysis indicated that the fall MWT survey abundance index, which primarily reflects young-of-year juvenile abundance, is positively related to Delta outflow. For the analysis, the fall MWT survey index was log-transformed and regressed on log-transformed February through May Delta outflow at Chipps Island. February through May was considered the main period of larval development. Because the abundance index was unavailable for 1974 and 1979, these years were excluded from the analysis. The regression analysis indicated that 74 percent of the variation in the index is explained by changes in Delta outflow (Figure II-98).



SOURCE:
DFG, 1992b.

FIGURE II-98

RELATIONSHIP BETWEEN ANNUAL MEAN FEBRUARY-MAY MONTHLY OUTFLOW AT CHIPPS ISLAND AND LONGFIN SMELT ABUNDANCE INDEX (1967-1991)

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The significant relationship between the fall MWT survey abundance index and Delta outflow probably results primarily from effects of outflow on survival of larvae and early juveniles. As noted earlier, survival between spring larvae and fall young-of-year is correlated with Delta outflow. Assuming that the number of larvae produced is considerably more constant among years than survival rate, the survival of larvae and young juveniles largely determines variations in young-of-year abundance, and by extension, Delta outflow determines young-of-year abundance. The number of longfin smelt adults is a relatively constant fraction of young-of-year abundance (DFG, 1987), so year-class strength could be largely determined by survival of the early life stages.

Possible explanations for a positive effect of high Delta outflow on survival of larvae and young juveniles include increased area of suitable brackish water rearing habitat (DFG, 1992b), reduced salinity driving saltwater fish species from the bays and minimizing competition with and predation on longfin smelt (Baxter, pers. comm.), reduced predation on young longfin smelt because the smelt are more dispersed and the water is more turbid (Stevens and Miller, 1983), increased phytoplankton and zooplankton production (DFG, 1993), and reduced entrainment losses. Any or all of these mechanisms could be responsible for the observed relationship between Delta outflow and longfin smelt abundance.

Low outflows and high exports increase the probability that juvenile longfin smelt will remain in the Delta for longer periods. Larvae in the Delta could be more vulnerable to predation, entrainment, and other sources of mortality.

Salinity. Although longfin smelt tolerate a wide range of salinities, indirect effects of salinity could influence longfin smelt abundance. DFG found that approximately 90 percent of early juvenile longfin smelt (i.e., less than 2 inches long) were collected in water with bottom salinity of less than 18 parts per thousand (DFG, 1992b). Critical nursery habitat was therefore defined as regions of the estuary with bottom salinities of less than 18 parts per thousand during March through June.

Regression analysis indicated that the fall MWT survey abundance index is positively related to the total volume of the critical nursery habitat. Both the abundance index and the habitat volume were log-transformed. This relationship suggests that longfin smelt abundance is influenced by the volume of brackish water habitat available. However, the volume of brackish water habitat and Delta outflow are also highly correlated (DFG, 1992b), so it is difficult to determine which of these factors (or other factors that could be correlated with Delta outflow) directly affect the longfin smelt population.

As discussed in the section on delta smelt, the position of X2 is a fairly accurate predictor of delta smelt abundance. Delta smelt abundance tends to be highest when X2 has an intermediate value (i.e., X2 is in upper Suisun Bay). X2 is also a good predictor of longfin smelt abundance, but longfin smelt abundance is highest when X2 has minimal values (i.e., X2 is in lower Suisun Bay) (Jassby, 1993). However, X2 is largely determined by Delta outflow, so the relationship between it and longfin smelt abundance could simply reflect effects of outflow or other correlates of outflow on longfin smelt abundance.

Entrainment. In most years, longfin smelt larvae and young juveniles are quickly transported out of the Delta and therefore are unlikely to be entrained in diversions. During the 1987-1992 drought, however, outflows have been low and exports have been high, so many juveniles have remained in the Delta and have been entrained at the CVP and SWP pumps (Figure II-95). Entrainment at other diversions has not been studied but probably also increased during the drought.

Food Availability, Water Quality, and Introduced Species. Food availability, water quality, and introduced species that affect juvenile longfin smelt are the same as those that affect other life stages. These factors were discussed in previous sections.

Ongoing Monitoring, Enhancement, and Habitat Improvement Actions

Ongoing monitoring, enhancement, and habitat improvement actions for longfin smelt are largely the same as those described previously for delta smelt. Longfin smelt have a wider distribution in the estuary than delta smelt, however, and additional sampling survey programs are necessary to properly monitor longfin smelt populations. In particular, the bay survey sampling programs are essential for monitoring longfin smelt distribution and abundance because all the bays are important habitat for longfin smelt. The MWT survey and otter trawl bay sampling programs, which were described previously, monitor adult and juvenile longfin smelt abundances in the bays, and a sampling survey using an egg and larval net monitors abundance of longfin smelt larvae.

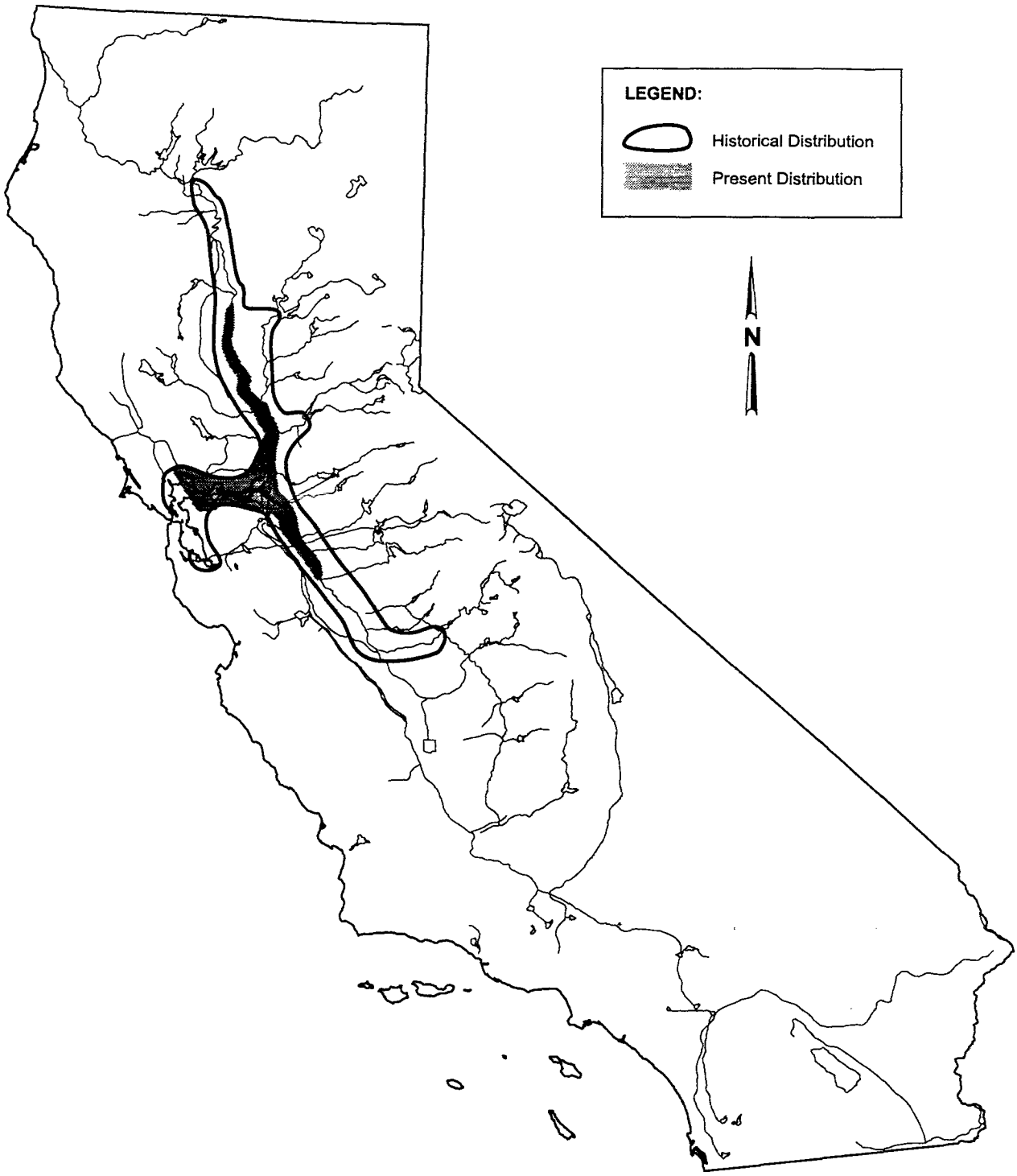
RECENT CONDITIONS FOR SACRAMENTO SPLITTAIL

STOCK ABUNDANCE AND DISTRIBUTION

Historical Population Trends (1850-1966)

Sacramento splittail (*Pogonichthys macrolepidotus*) are large cyprinids (minnow family) endemic to the lakes and rivers of the Central Valley of California (Moyle et al., 1989). Historically, they were collected from the Sacramento River as far north as Redding (Shasta County), from the San Joaquin River as far south as Fresno (Fresno and Madera counties), and from the Delta (Figure II-99). Early fish surveys indicate that splittail were also present in the Feather River near Oroville, the American River near Folsom, and the Merced River near Livingston (Natural Heritage Institute, 1992). In the early 1900s, splittail were also found in southern San Francisco Bay and lower Coyote Creek in Santa Clara County; however, recent surveys indicate that splittail could no longer be present in these locations (Moyle et al., 1989). DFG sampling surveys from September 1963 to August 1964 determined that splittail were the most evenly distributed cyprinid in the Delta (Turner, 1966).

Historical information on population trends of splittail is limited to commercial harvest records and anecdotal information. Commercial harvest records indicate that approximately 449,000 pounds of splittail were harvested from 1916 to 1947 (California Bureau of Marine Fisheries, 1949). The actual commercial catch of Sacramento splittail during this period is probably lower because these catch records include the now extinct Clear Lake splittail. Before the construction



SOURCE:
Moyle et al., 1989.

FIGURE II-99

HISTORICAL AND PRESENT DISTRIBUTION OF SACRAMENTO SPLITTAIL IN CALIFORNIA

of Folsom and Oroville dams, anglers in Sacramento reported catching 50 or more splittail in a single day (50 Code of Federal Regulations, Part 17).

Current Population Trends (1967-1991)

Splittail are currently confined largely to the Delta, Suisun Bay, Suisun Marsh, and Napa Marsh and will use the Sutter, Sacramento, and Yolo bypasses when flooded (Moyle et al., 1989; Natural Heritage Institute, 1992; Jones & Stokes Associates, 1993). They are most abundant in Suisun and Grizzly bays (50 Code of Federal Regulations, Part 17). Since 1985, splittail have been rare in San Pablo Bay, indicating that their range could be declining further. In the Delta, splittail are most abundant in the western and northern portions (Moyle et al., 1989); however, in recent years their distribution appears to have shifted to the lower Sacramento River and South Delta, possibly because of reductions in Delta outflow (50 Code of Federal Regulations, Part 17).

Splittail are caught occasionally in the lower Sacramento, Feather, and American rivers (50 Code of Federal Regulations, Part 17). Recently, seining surveys of the lower reaches of the Sutter Bypass near Karnak (Sutter County) recorded the occurrence of both juvenile and adult splittail (Jones & Stokes Associates, 1993). In the San Joaquin River Drainage, splittail appear to be less common than in the Sacramento River drainage. Limited numbers of splittail were collected in the late 1980s from the San Joaquin River near the confluences of the Tuolumne and Merced rivers and in Mendota Pool (which receives water pumped from the Delta via the Delta-Mendota Canal) (Jones & Stokes Associates, 1987; Natural Heritage Institute, 1992). On occasion, splittail are caught in San Luis Reservoir (Merced County), which receives Delta water through the Delta-Mendota Canal and the California Aqueduct (Natural Heritage Institute, 1992).

DFG estimates current young-of-year splittail abundance in the Delta to be only 35 to 60 percent of 1940 levels (DFG Bay-Delta Hearings, 1992); however, these estimates appear to have been derived by extrapolating data from a regression equation that describes the relationship between splittail abundance indices and Delta outflow during March through May. The abundance index is based on DFG's 1967-1990 fall MWT surveys of San Pablo Bay and the Delta. Fall MWT surveys provide the longest, most accurate index of splittail abundance. Results of the fall MWT surveys indicate that splittail abundance has been highly variable from year to year, with highs and lows in the index values occurring during wet and dry periods, respectively (Figure II-100). After 1983, the splittail population steadily declined. Reduced Delta outflow, water diversions, entrainment in diversions, dams, introduced aquatic species, loss of wetlands and shallow-water habitats, and the recent drought appear to have contributed to the decline in splittail distribution and abundance (Service, 1993). The decline in abundance has prompted DFG to designate splittail as a species of special concern and the Service to propose the species for federal listing as threatened. (Note: Splittail abundance rebounded in 1995, indicating the resilience of the species in response to high-flow conditions.)

LIFE HISTORY

Existing information on the life history of Sacramento splittail is limited and based largely on Moyle (1976); Daniels and Moyle (1983); Wang (1986); Moyle et al. (1989); and 50 Code of Federal Regulations, Part 17. The life history of the Sacramento splittail is presented in Figure II-101.

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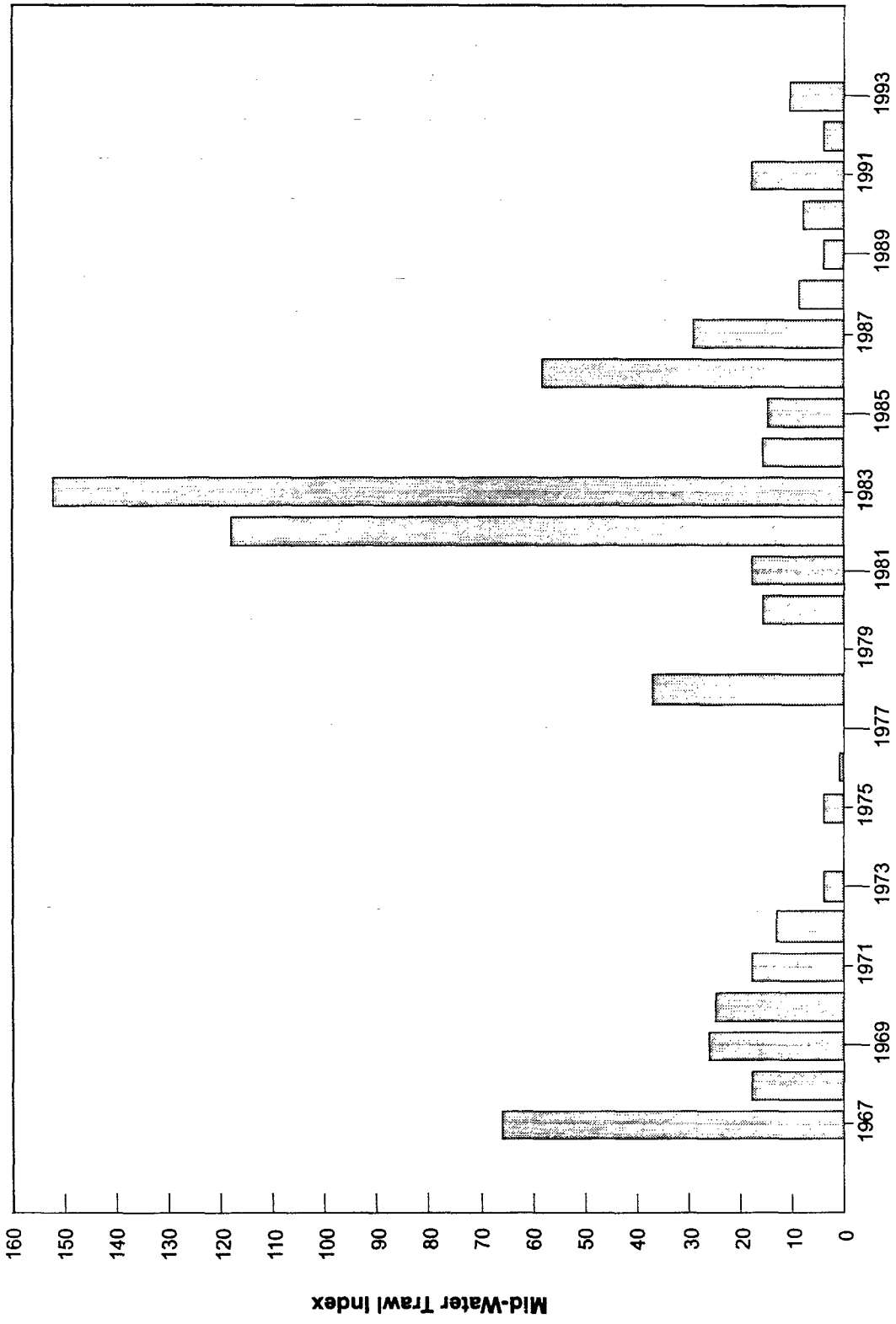

FIGURE II-100

ANNUAL ABUNDANCE INDEX OF SPLITTAIL IN THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY (1967-1993)

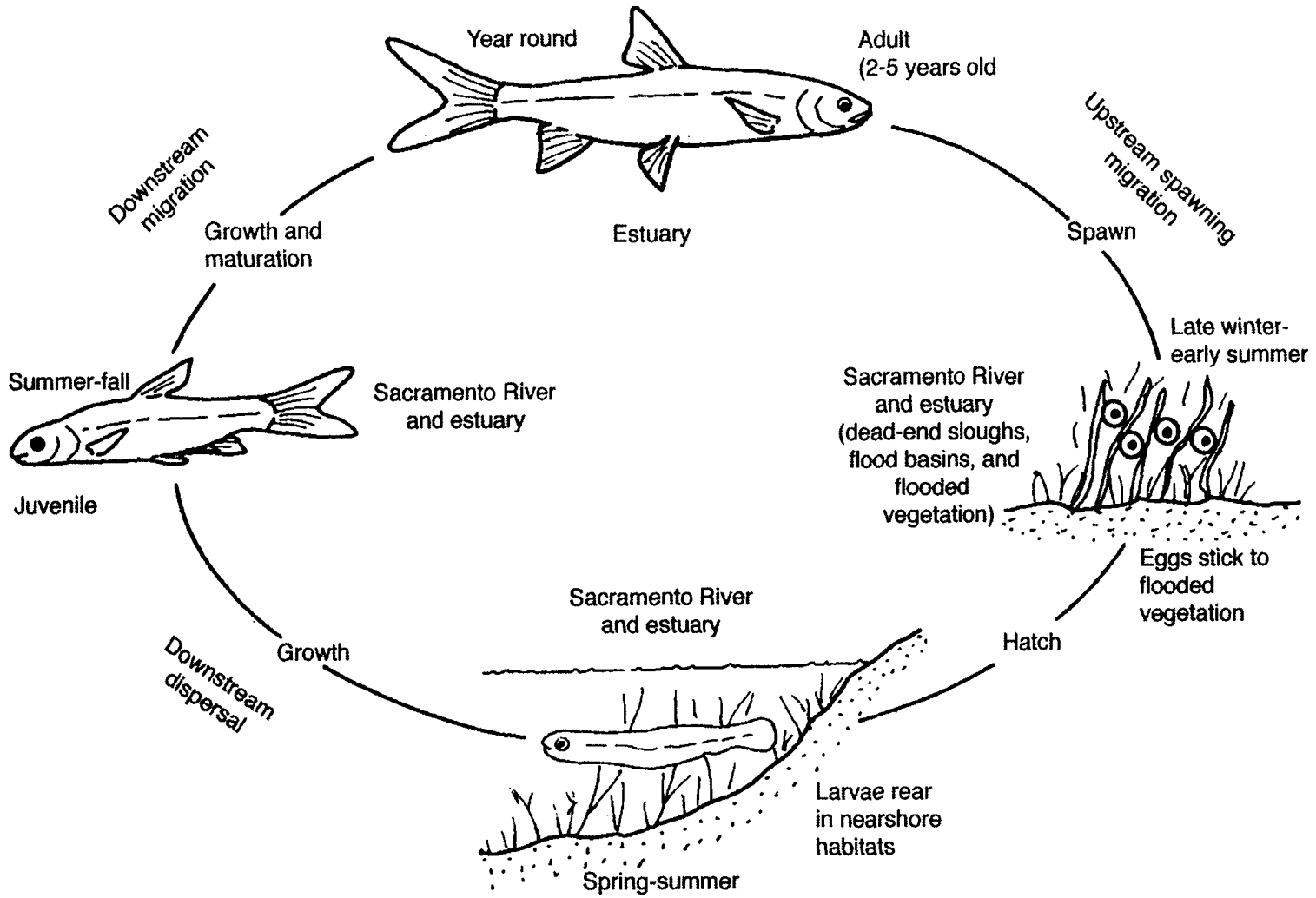

FIGURE II-101

LIFE HISTORY OF SACRAMENTO SPLITTAIL

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Adults

Sacramento splittail are freshwater fish capable of tolerating moderate levels of salinity (10 to 18 parts per thousand) (50 Code of Federal Regulations, Part 17). They grow up to 16 inches long and live approximately five years. Adult splittail are primarily benthic (i.e., bottom) foragers with a limited diet, including detritus (i.e., decayed organic material) earthworms, clams, insect larvae, and other invertebrates. Opossum shrimp (*Neomysis mercedis*) appear to be the dominant prey item, although detritus constitutes a large proportion of their stomach contents (Daniels and Moyle, 1983). Male and female splittail become sexually mature by their second winter at minimum lengths of approximately 10 centimeters. Female splittail are capable of producing more than 100,000 eggs per year (Daniels and Moyle, 1983; Moyle et al., 1989).

Migrating Adults. Little information exists on the occurrence, timing, and pattern of migration of adult splittail. Incidental catches of large splittail in large fyke traps set by DFG to catch migrating striped bass in the lower Sacramento River during spring suggest that splittail could migrate from lower river reaches to upstream spawning habitats. Seasonal sport catches indicate adult migration during winter and spring.

Spawning Adults. The onset of spawning appears to be associated with increasing water temperatures and increasing day length (Daniels and Moyle, 1983). Spawning occurs in late April and May in the Suisun Marsh and between early March and May in the upper Delta (Moyle et al., 1989). Spawning in the tidal freshwater and oligohaline (0.3 to 5.0 parts per thousand) habitats of the Sacramento-San Joaquin River Delta estuary has been observed as early as January and continuing through July (Wang, 1986). Spawning also occurs in the lower reaches of the Sacramento and San Joaquin rivers. Adult splittail collected from the Sutter Bypass in mid-February and early March 1993 were presumed to be spawning because of the later appearance of juvenile splittail (Jones & Stokes Associates, 1993), and successful spawning was observed in the lower Tuolumne River during wet years in the 1980s (Natural Heritage Institute, 1992).

Splittail apparently spawn in the evening over flooded streambank or aquatic vegetation when water temperatures are between 48 and 68 degrees Fahrenheit (Moyle, 1976; Wang, 1986). Spawning success appears to be correlated with freshwater outflow and the availability of shallow-water habitats with submerged vegetation (Daniels and Moyle, 1983).

Planktonic Larvae

Larval splittail are commonly found in the shallow, weedy areas where spawning occurs and eventually move into deeper, open water habitats in the summer (e.g., near the Pittsburgh powerplant at New York Slough and in Montezuma Slough in Suisun Marsh). The larvae are occasionally found in San Pablo Bay and have been collected near the Berkeley Marina in San Francisco Bay, indicating that they are somewhat tolerant of salinity. (Wang, 1986.)

Rearing Juveniles

Juvenile splittail are commonly found in Delta sloughs in late winter and spring and are particularly abundant in the vicinity of Montezuma Slough. As summer progresses, juvenile splittail gradually move from shallow, inshore habitats to the deeper, open water habitats of Suisun and San Pablo bays (Wang, 1986). In upstream areas, they are found in shallow, flooded areas where warmer water temperature and low water velocity persist and can be expected to occur in any inundated flood basin, such as the Sutter and Yolo bypasses. Juvenile splittail have been collected from the lower reaches of the Sutter Bypass during spring (April) (Jones & Stokes Associates, 1993) and from the Sacramento River near Colusa and Princeton in Colusa County (Baxter, pers. comm.). Juvenile splittail feed primarily on algae and invertebrates and are often preyed upon by Sacramento squawfish and striped bass (Wang, 1986).

FACTORS AFFECTING ABUNDANCE

Adults

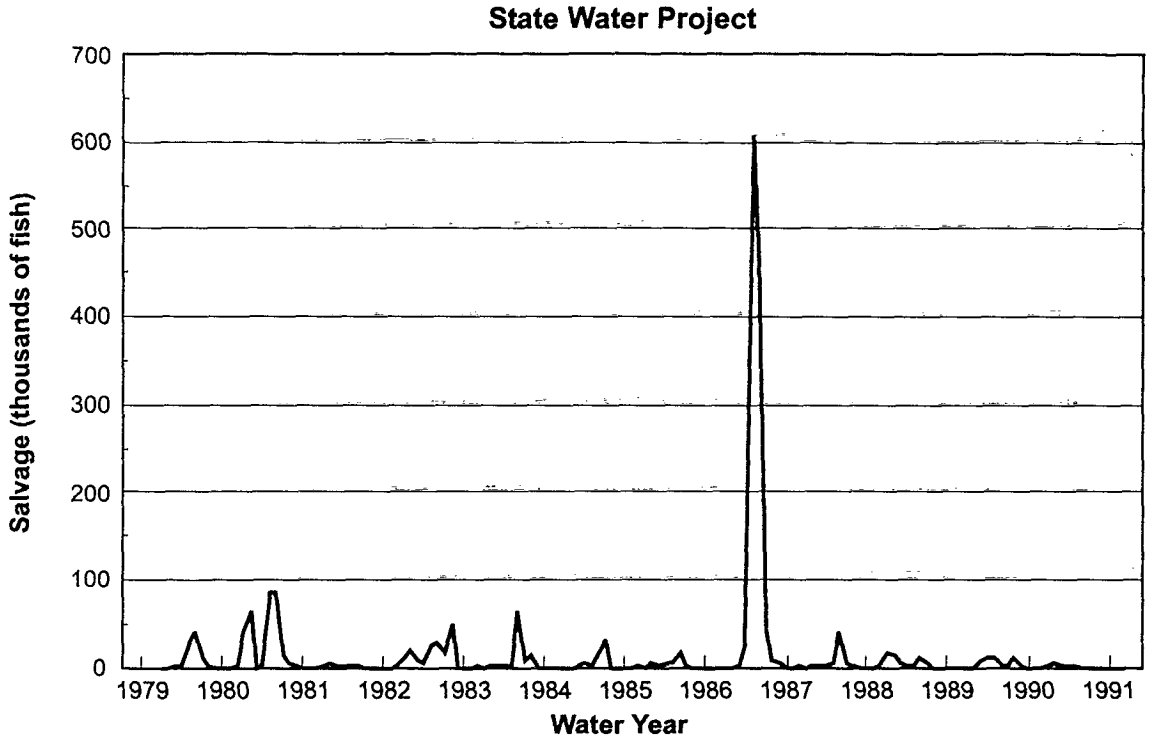
Entrainment. Delta diversions total approximately 9 million acre-feet per year. Combined federal (CVP) and state (SWP) projects export approximately 6 million acre-feet per year when sufficient water is available, and in-Delta agricultural diversions could total approximately 3 million acre-feet per year (50 Code of Federal Regulations, Part 17). After 1983, diversions during October through March have been higher than in earlier years (50 Code of Federal Regulations, Part 17). Diversions during late winter and spring have coincided with splittail migration and spawning. Entrainment of splittail also occurs in the approximately 1,800 in-Delta diversions, although no data are available to estimate their contribution to total entrainment losses.

CVP and SWP Delta Pumping Facilities. As stated earlier, the CVP and SWP Delta pumping facilities are the largest diversions from the Delta. Thousands of splittail larvae, juveniles, and adults are salvaged annually by CVP and SWP fish protection facilities (Figure II-102), and thousands more are lost to the diversions.

An unknown proportion of adult splittail entrained in the CVP and SWP diversions are salvaged and returned to the Delta alive. The proportion salvaged depends on screen efficiency (a function of screen design and pumping volume), fish size (salvage increases with increasing size), predation rates, and handling and trucking mortality. These factors vary between CVP and SWP facilities. Mean annual salvage (1979-1989) of splittail at the CVP and SWP facilities is approximately 213,000 and 190,500, respectively.

Salvage data from the CVP and SWP pumping facilities indicate that adult splittail are entrained at the pumping facilities year-round (Barrow, pers. comm.). However, most of the adults (greater than 18 centimeters) are entrained during January through April, which coincides with the migration and spawning season.

Agricultural Diversions. Losses of splittail to agricultural diversions could be considerable given that these diversions account for approximately one-third of the volume of



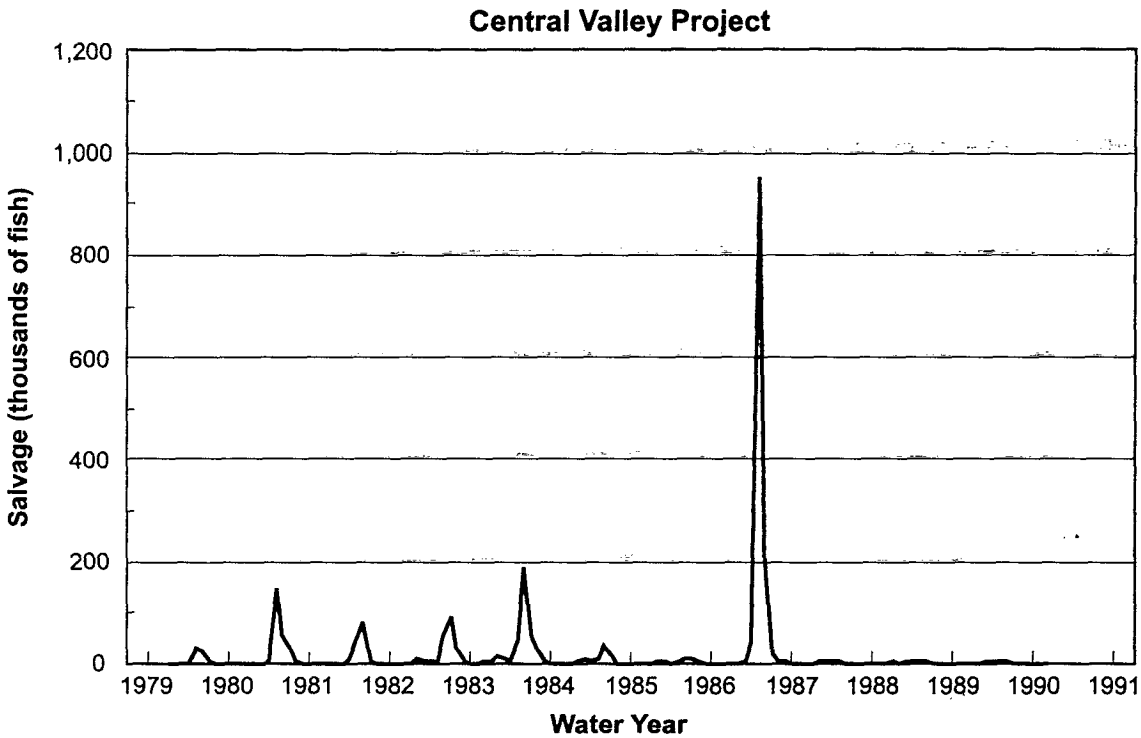


FIGURE II-102

SALVAGE OF SPLITTAIL AT THE SWP AND CVP FACILITIES (1979-1991)

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water diverted from the Delta. Losses to agricultural diversions depend on the timing, size, and location (geographically and position in the channel) of individual diversions relative to the seasonal distribution and abundance of splittail. The ability of splittail to avoid agricultural diversions and losses to those diversions is currently unknown.

Power Generation Facility Diversions. Two of the largest nonconsumptive diversions in the Delta are PG&E's Contra Costa and Pittsburg powerplants. Considering the location of the facilities' intakes (near Antioch and Pittsburg) and the size of the diversions (nearly 1,500 cfs at each facility depending on power generation demands), substantial numbers of larval, juvenile, and adult splittail are likely entrained and lost in the diversions. However, splittail entrainment losses at these facilities have not been quantified, and existing fish screens could reduce losses of adult splittail.

Other Diversions. Other diversions also entrain splittail larvae, juveniles, and adults. The largest diversions not previously discussed above are the North Bay Aqueduct diversion and CCWD's Rock Slough diversion. Diversion during the splittail spawning and early rearing period will likely entrain the largest numbers of splittail. The relative contribution of entrainment losses at these diversions to the overall total entrainment of splittail has not been quantified.

Habitat. Habitat modification is probably the largest single factor contributing to the decline of splittail (DFG, 1992). Land reclamation, flood control facilities, and agricultural development have eliminated and drastically altered much of the splittail habitat in the lowland areas. Dams have restricted access to upstream spawning and rearing habitats. Diking and dredging have eliminated an estimated 96 percent of the wetland habitats that splittail apparently require (50 Code of Federal Regulations, Part 17). Most of the diking and filling of wetlands preceded the recent decline in splittail abundance. During the past 20 years, only relatively small habitat areas have been lost to levee riprapping and wetland filling.

Delta Outflow. Delta diversions, including CVP and SWP diversions and upstream diversions and storage, have affected splittail habitat by reducing freshwater flow and altering the availability of flooded habitat. Increased salinity in San Pablo and Suisun bays in response to reduced outflow could have forced splittail into the lower Sacramento River and South Delta where habitat conditions for splittail are less favorable (50 Code of Federal Regulations, Part 17).

Toxic Materials. Survival of adult splittail could be affected by toxic materials entering the Sacramento-San Joaquin River system from agricultural runoff, discharge of industrial and municipal waste, and runoff from non-point sources (e.g., urban stormwater runoff). In the Delta, pollutants of particular concern are trace elements (e.g., selenium, copper, cadmium, and chromium) and agricultural chemicals and their derivatives, which are used extensively in the Central Valley.

No specific information exists on how toxic materials are affecting splittail populations in the Delta; however, the effects of toxics on adult splittail could be similar to known effects on other Delta fish species. For instance, toxics exceeding levels considered safe for human consumption have been found in tissue samples of adult striped bass and appear to reduce fecundity in female

striped bass (see "Recent Conditions for Striped Bass"). Toxic materials likely have an adverse effect on adult splittail, but there is no evidence to suggest that they are the cause of the recent decline in splittail abundance. Toxic materials could affect adults either directly or indirectly, thereby reducing reproductive success, growth, and survival.

One of the complicating factors in understanding the effects of toxics on ecological processes in the estuary is the complex distribution of "hot spots" (areas of high concentrations of toxics), both spatially and temporally (Herbold et al., 1992). These hot spots could cause adults to avoid biologically important habitat or alter movements.

Fishing. Splittail have not been commercially harvested in the Delta since the 1950s. Currently, splittail are harvested only as food and bait by sport anglers, probably on a limited basis. No evidence exists to suggest that the sport harvest has contributed to the recent decline in abundance (50 Code of Federal Regulations, Part 17).

Introduced Species. The effects of increased competition and predation resulting from species introductions are difficult to evaluate in wild populations. Competition-predation effects will be distinguishable if there were a concomitant increase in the abundance of an introduced species with the decline in abundance of splittail.

Striped bass are known to prey on splittail; however, they are not likely to be the primary cause of the recent decline in splittail abundance because splittail and striped bass have coexisted since striped bass were introduced. Furthermore, striped bass abundance has been declining along with that of splittail (Natural Heritage Institute, 1992).

The introduced Asiatic clam, *Potamocorbula* sp., could affect splittail abundance because the clam has become extremely abundant in Suisun Bay where it could compete with opossum shrimp, a favorite prey item of splittail (Natural Heritage Institute, 1992). The Asiatic clam did not cause the initial decline in splittail abundance, however, because it was not introduced until after 1986 (Natural Heritage Institute, 1992).

Climatic Conditions. Extreme climatic conditions, such as prolonged droughts and devastating floods, also contribute to the fluctuations in splittail abundance. Relatively low Delta outflow and river flows during the recent drought could have reduced available habitat. Although splittail historically persisted despite extreme climatic conditions, recent human-induced factors make it more difficult for splittail populations to rebound from the effects of drought during more "normal" hydrologic conditions.

Migrating Adults

Flow. The precise environmental cue that triggers splittail to migrate to their spawning areas is unknown. However, it is likely that increasing streamflow in winter is one environmental cue that is responsible for initiating spawning migrations. Increasing flows are important for providing flooded habitats that splittail apparently require and could improve passage conditions into those habitats. Before the impoundment and channelization of the Central Valley's rivers, many habitats within the Central Valley flooded annually. Splittail timed their migration and

spawning with the occurrence of these floodflows to take advantage of conditions that permitted access to these flooded basins, marshes, and sloughs.

Aside from habitat modifications, upstream water storage facilities and water diversions have affected migrating splittail by reducing the incidence and duration of floodflows that inundate lowland areas. Consequently, passage into inundated habitats is compromised, resulting in reduced spawning success because splittail are unable to exploit former spawning areas. Furthermore, reduced flood duration increases the risk that splittail become stranded in temporarily inundated habitats, such as those in the Sutter and Yolo bypasses.

Entrainment, Toxic Materials, Fishing, Introduced Species, and Climatic Conditions. All of these factors affect migrating adults as described previously.

Spawning Adults

Flow. Splittail abundance has been shown to be strongly associated with high Delta outflows during primary spawning months (March through May) (Figure II-103) (DFG, 1992). High Delta outflows during late winter and spring correlate with increased total surface area of shallow-water habitats containing submerged vegetation, both within and upstream of the Delta. Adult splittail spawn in areas of submerged vegetation where they lay their adhesive eggs. During years of severely reduced Delta outflow, such as the recent drought, spawning success could have been greatly reduced, contributing to reduced abundance.

Delta diversions, including CVP and SWP pumping facilities, coupled with upstream storage reservoirs, could have adversely affected spawning adults by reducing freshwater flow and the availability of temporarily flooded habitat. Consequently, spawning adults are forced to use less favorable spawning habitat, thereby decreasing reproductive success.

Habitat. Habitat modifications discussed previously have had the greatest effect on spawning habitat. Levee construction, bank stabilization practices (i.e., riprapping), river channelization, dredging, and the diking and filling of historical flood basins have drastically reduced shallow-water habitats available to spawning adults. Splittail typically spawn in dead-end sloughs and slow reaches of large rivers over submerged vegetation. An estimated 96 percent of historical wetland habitats either are unavailable to splittail or have been eliminated. Although spawning habitat loss does not appear to be the primary mechanism for the recent decline in abundance, it is likely a contributing factor.

Salinity. Upstream water storage projects, diversions, and Delta export pumping have reduced Delta outflow. Reduced outflow increases salinities in Suisun Marsh, Suisun Bay, and the lower Delta. Although adult splittail could spawn successfully in habitats containing low (0.3 to 5.0 parts per thousand) salinities (Wang, 1986), excessive salinity likely inhibits use of spawning habitat downstream of the Delta.

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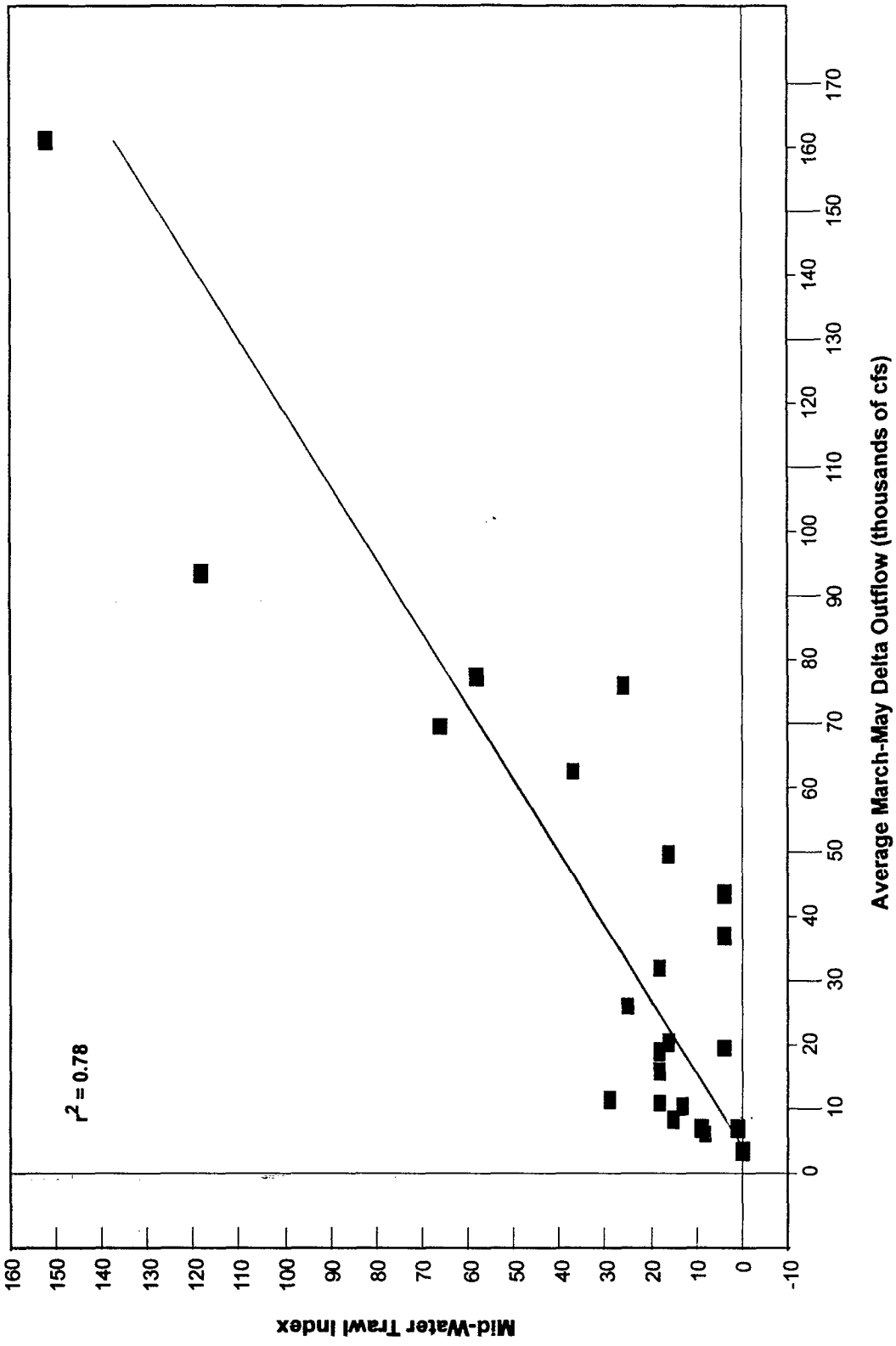

FIGURE II-103

RELATIONSHIP BETWEEN ABUNDANCE INDEX FOR ADULT SPLITTAIL AND AVERAGE MARCH-MAY DELTA OUTFLOW

Planktonic Eggs and Larvae

Flow. As stated earlier, splittail abundance is positively correlated with high flows. Because larval splittail are commonly found in the shallow, weedy areas where spawning occurs, flow duration could be an important factor determining egg and larval survival.

Upstream water storage facilities and water diversions have reduced the magnitude and duration of floodflows, thereby reducing the time that wetland habitats are inundated. Reduced duration of flooding during spawning and early rearing could degrade conditions necessary for optimal egg and larvae development, or worse, could desiccate these habitats before larvae are able to move to where more permanent rearing conditions are found.

Entrainment

CVP and SWP Delta Pumping Facilities. Thousands of larval splittail (less than 20 millimeters) are entrained in CVP and SWP exports annually. Most larvae are entrained from April through July (Barrow, pers. comm.). Most of the entrained larvae are lost in the diversions. Additional losses to entrainment, including predation, handling, and trucking mortality, have not been quantified.

Agricultural Diversions. Entrainment of splittail larvae in agricultural diversions is a function of the location of the diversions (geographically and position within the channel) and the volume and design of the diversions. The magnitude of larval entrainment losses to these diversions is unknown and depends on larval abundance and distribution in addition to those factors mentioned previously.

Power Generation Facility Diversions. PG&E's Contra Costa and Pittsburg powerplants have fish screens, but entrainment rates, salvage efficiency, and associated losses of larval splittail are not available. Splittail larvae are known to occur near the intakes to the Pittsburg powerplant (Wang, 1986).

Other Diversions. The magnitude of larval entrainment losses at the North Bay Aqueduct and CCWD's Rock Slough diversions is unknown.

Habitat. Habitat modifications have had the greatest effect on shallow-water habitats particularly important to developing eggs and larvae. Important shallow-water habitats contain abundant vegetation that provides attachment sites for developing eggs, protection from predators, optimal water temperatures necessary for growth and proper development, and excellent conditions for food production. As noted previously, levee construction, river channelization, dredging, and the diking and filling of historical flood basins have drastically reduced the number of shallow-water habitats available to splittail.

Toxic Materials. Although splittail spawn when flows are relatively high and pollutant concentrations are probably relatively low (because of the diluting effects of high freshwater flows), localized populations of young splittail and eggs could be disproportionately affected by pollutants if spawning occurs near discharges containing high pollutant concentrations.

Developing eggs and larvae in the vicinity of these discharges could develop poorly, with reduced growth rates and increased mortality. Furthermore, toxic hot spots could force larvae out of the shallow, weedy habitats and into less favorable habitats where they are subjected to increased predation and competition for food, thereby reducing survival.

Salinity. Upstream water storage projects, diversions, and Delta export pumping have reduced Delta outflow and periodically increased salinity in Suisun Marsh and Bay. Larval splittail appear to be somewhat tolerant of low salinities (Wang, 1986); however, the effects of salinity on developing eggs are unknown. Increased salinity probably affects egg and larval survival, but the available information does not indicate that increased salinity caused the recent decline in abundance.

Introduced Species and Climatic Conditions. The effects of introduced species and climatic conditions on planktonic eggs and larvae are similar to the effects discussed for adult splittail.

Rearing Juveniles

Flow. Flow factors affecting rearing juvenile splittail are similar to the effects discussed for adult splittail.

Entrainment

CVP and SWP Delta Pumping Facilities. Splittail entrainment in CVP and SWP Delta diversions varies annually (Figure II-102) and juveniles account for the majority of splittail that are entrained. Salvage records indicate that juvenile splittail are entrained year-round, although the bulk of them are entrained from April through August (Barrow, pers. comm.). The relative proportion of entrained juveniles salvaged and returned to the Delta alive has not been quantified. Entrainment-related losses result from predation in the vicinity of the screening facilities and from stress associated with handling and trucking. Salvage rates vary between the two pumping facilities due to differences in salvage operations and other factors (e.g., prescreening predation).

Agricultural Diversions. Entrainment of juvenile splittail in agricultural diversions is a function of fish size, the location of the diversions (geographically and position within the channel), and the volume and design of the diversions. Juvenile splittail could be capable of avoiding smaller intakes, but entrainment likely occurs in many agricultural diversions. The magnitude of juvenile entrainment losses to diversions is currently unknown and also depends on juvenile abundance and distribution, in addition to those factors mentioned previously.

Power Generation Facility Diversions. PG&E's Contra Costa and Pittsburg powerplants have fish salvage facilities, but entrainment rates, salvage efficiency, and associated losses of juvenile splittail are not available. Juvenile entrainment could be substantial because of the proximity of the intakes to splittail-rearing areas.

Other Diversions. The efficiency of the salvage facilities and the entrainment losses of juvenile splittail at the North Bay Aqueduct and losses to CCWD's Rock Slough diversions are unknown.

Habitat, Toxic Materials, Fishing, Introduced Species, and Climatic Conditions

These factors generally affect juvenile Sacramento splittail as previously described for adult and larval splittail.

ONGOING MONITORING, ENHANCEMENT, AND HABITAT IMPROVEMENT ACTIONS

The Service has recently proposed listing the status of the Sacramento splittail as threatened pursuant to the federal ESA and also has recently issued a biological opinion for delta smelt that included technical assistance recommendations for splittail (Service, 1993). The Service has recommended that transport and habitat flows be maintained in the Sacramento River downstream of the confluence of the Feather River and in the Napa River from March 1 to July 31. The recommendation is intended to provide juvenile and larval splittail with adequate transport and habitat flows during downstream migration to suitable rearing areas in Suisun Bay.

The Service recognizes that diversions on the Sacramento River below the confluence of the Feather River could entrain juvenile and larval splittail. The Service has recommended that these diversions be screened or operated to minimize entrainment.

The Suisun Marsh Salinity Control Structure, when in operation, could prevent splittail movement into and out of Montezuma Slough. The Service has recommended that the DWR, in consultation with Reclamation, develop and implement a program of investigations designed to evaluate operational effects of the control structure on splittail. These investigations will address the diversion rate of adult splittail into Montezuma Slough and predation at the control structure.

RECENT CONDITIONS FOR RESERVOIR FISHERIES COMMUNITIES

GENERAL RESERVOIR CHARACTERISTICS

Reservoirs have become one of the major fish habitats in the Sacramento-San Joaquin River Basin since the advent of the CVP and SWP. The nature of each reservoir and its fish fauna is determined by its elevation, size, location, and water quality. In general, reservoirs are much less productive per surface acre than lakes because their generally deep, steep-sloped basins and fluctuating water levels greatly limit habitat diversity. CVP and other Central Valley reservoirs lie at mid-elevations in the foothills and have characteristics of both warm-water and cold-water impoundments (Moyle, 1976). Tables II-12 and II-13 provide general information about and physical features of selected reservoirs in the Central Valley.

Central Valley reservoirs are generally either warm-water reservoirs, suitable for black bass, sunfish, and catfish, or "two-story" reservoirs that contain a zone of deep, well-oxygenated water

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TABLE II-12

GENERAL INFORMATION FOR SELECTED CENTRAL VALLEY RESERVOIRS

Reservoir	County	Impounded Stream	Owner	Reservoir Purpose	Drainage Area (square miles)	Year Storage Began
Black Butte Lake	Tehama	Stony Creek	COE	FC/IR	738	1963
Camanche Reservoir	Amador/San Joaquin/ Calaveras	Mokelumne River	EBMUD	WS	613	1963
East Park Reservoir	Colusa	Little Stony Creek	Reclamation	IR	98	1910
Eastman Lake	Madera	Chowchilla River	COE	FC/IR/RC	236	1975
Englebright Lake	Nevada/Yuba	Yuba River	COE	DC	1,108	1941
Folsom Lake	Sacramento/Placer/ El Dorado	American River	Reclamation	FC/IR/HP/WS	1,861	1955
Hensley Lake	Madera	Fresno River	COE	FC/IR/RC	237	1975
Jenkinson Lake	El Dorado	Sly Park Creek	Reclamation	IR/WS	16	1954
Keswick Reservoir	Shasta	Sacramento River	Reclamation	HP	6,468	1948
Lake Berryessa	Napa	Putah Creek	Reclamation	FC/IR/WS	566	1956
Isabella Lake	Kern	Kern River	COE	FC/IR	7,074	1954
Lake Kaweah	Tulare	Kaweah River	COE	FC/IR	560	1962
Lake McClure	Mariposa	Merced River	MCD	IR/WS	1,020	1967
Lake Natoma	Sacramento	American River	Reclamation	HP/RR	1,887	1955
Lake Oroville	Butte	Feather River	DWR	FC/HP/IR/RC/WS	3,607	1967
Lake Red Bluff	Tehama	Sacramento River	Reclamation	FC/IR	8,900	1964
Shasta Lake	Shasta	Sacramento River	Reclamation	FC/HP/IR/NV/WS/WQ	6,400	1944
Millerton Lake	Fresno	San Joaquin River	Reclamation	FC/IR	1,638	1944
New Bullards Bar Reservoir	Yuba	Yuba River	YCWA	IR/WS	481	1970
New Don Pedro Reservoir	Tuolumne	Tuolumne River	TID/Modesto ID	IR/WS	1,542	1971
New Hogan Lake	Calaveras	Calaveras River	COE	FC/IR	362	1963

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TABLE II-12. CONTINUED

Reservoir	County	Impounded Stream	Owner	Reservoir Purpose	Drainage Area (square miles)	Year Storage Began
New Melones Reservoir	Calaveras/Tuolumne	Stanislaus River	COE		900	1978
O'Neill Forebay	Merced	San Luis Creek	Reclamation/DWR	IR	0	1966
Pine Flat Reservoir	Fresno	Kings River	COE	FC/IR	1,545	1951
San Luis Reservoir	Merced	San Luis Creek	Reclamation/DWR	HP/IR	83	1967
Stony Gorge Reservoir	Glenn	Stony Creek	Reclamation	IR	301	1928
Lake Success	Tulare	Tule River	COE	IR/FC	393	1961
Thermalito Afterbay	Butte	Feather River tributary	DWR	HP/IR	6	1967
Thermalito Forebay	Butte	Cottonwood Creek	DWR	HP/IR	4	1967
Whiskeytown Lake	Shasta	Clear Creek	Reclamation	HP	200	1963

SOURCE:
Leidy and Myers, 1984.

LEGEND:

Owner code letters:
COE: U.S. Army Corps of Engineers
Reclamation: U.S. Bureau of Reclamation
DWR: California Department of Water Resources
YCWA: Yuba County Water Agency
EBMUD: East Bay Municipal Utility District
TID: Turlock Irrigation District
Modesto ID: Modesto Irrigation District
MCID: Merced Irrigation District

Reservoir purpose code letters:
DC: debris control
FC: flood control
HP: hydropower
IR: irrigation
NV: navigation
RC: recreation
RR: reregulation
WS: water supply

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TABLE II-13

PHYSICAL DESCRIPTION OF SELECTED CENTRAL VALLEY RESERVOIRS

Reservoir	Surface Elevation (feet msl)	Surface Area (acres)	Reservoir Volume (acre-feet)	Total Annual Discharge (acre-feet)	Surface Ratio	Mean Depth (feet)	Maximum Depth (feet)	Thermocline Depth (feet)	Water-Level Fluctuation (feet)	Shoreline Length (miles)	Shoreline Develop-ment	Total Dissolved Solids (mg/l)	Total Phosphorus (mg/l as P)
Black Butte Lake	448	2,449	52,600	433,000	0.12	21	73	0	35	23	3.3	234[1]	0.09[1]
Camanche Reservoir	236[2]	7,700	431,500	NA	NA	NA	171	NA	NA	NA	NA	NA	NA
East Park Reservoir	1,177	986	19,100	54,600	0.40	19	65	30	33	21	4.8	430	0.10
Englebright Lake	520	765	64,500	704,200	0.09	84	230	NA	21	20	4.3	68	0.01
Folsom Lake	435	10,000	713,000	2,970,800	0.24	71	238	30	66	72	5.2	46	0.02
Hensley Lake	483	767	22,800	70,700	0.32	30	78	36	35	15	3.9	105	0.06
H.V. Eastman Lake [3]	506	975	39,700	44,600	0.89	41	96	22	37	12	2.7	202	0.08
Jenkinson Lake	3,448	518	27,500	4,500	6.11	53	137	NA	31	7	2.0	32	NA
Keswick Lake	585	625	22,300	5,177,400	0.00	36	105	0	3	19	5.4	103	0.03
Lake Berryessa	424	17,731	1,299,000	342,100	3.80	73	232	17	16	136	7.3	181	0.03
Lake Isabella	2,560	5,800	163,000	465,700	0.35	28	113	0[4]	23	31	2.9	67[1]	0.06[1]
Lake Kaweah	616	855	31,100	373,200	0.08	36	106	40[1]	101	13	3.0	68[1]	0.02[1]
Lake McClure	879[2]	7,127	1,026,000	NA	NA	NA	479	NA	NA	NA	NA	NA	NA
Lake Natoma [5]	124	500	8,300	201,500	0.04	17	54	0	3	19	6.1	41	0.02
Lake Oroville	861	13,866	2,954,500	411,700	7.17	213	708	37	75	140	8.5	52[6]	0.02[6]
Lake Red Bluff	252	499	3,674	9,505,000	0.00	7	51	0	2	19	5.9	100	0.04
Lake Success	615	917	24,300	107,900	0.22	27	57	18[1]	43	11	2.5	191[1]	0.03[1]
Lewiston Lake	1,910[2]	750	14,600	NA	NA	NA	80	NA	NA	NA	NA	NA	NA
Millerton Lake	524	3,676	294,100	1,600,000	0.18	80	224	NA	59	45	5.3	36	NA
New Bullards Bar Reservoir	1,965[2]	4,810	969,600	NA	NA	NA	635	NA	NA	NA	NA	NA	NA
New Don Pedro Reservoir	855[2]	12,960	2,030,000	NA	NA	93	568	NA	NA	NA	NA	NA	NA
New Hogan Lake	664	2,800	146,100	1,400,000	1.06	53	130	28	30	35	4.7	114[1]	0.06[1]
New Melones Reservoir	1,135[2]	12,500	2,400,000	NA	NA	NA	625	NA	NA	NA	NA	NA	NA
O'Neill Forebay	221	2,159	46,300	251,400	0.18	21	46	0	5	14	1.9	225	0.08
Pine Flat Reservoir	877	4,526	613,100	2,083,800	0.29	135	314	50[1]	96	57	6.1	30[1]	0.04[1]
San Luis Reservoir	508	11,700	1,595,700	763,000	2.09	136	238	25	64	47	3.1	370	0.06

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TABLE II-13. CONTINUED

Reservoir	Surface Elevation (feet msl)	Surface Area (acres)	Reservoir Volume (acre-feet)	Total Annual Discharge (acre-feet)	Surface Ratio	Mean Depth (feet)	Maximum Depth (feet)	Thermocline Depth (feet)	Water-Level Fluctuation (feet)	Shoreline Length (miles)	Shoreline Develop-ment	Total Dissolved Solids (mg/l)	Total Phosphorus (mg/l as P)
Shasta Lake	1,065	29,500	4,500,000	6,396,947(7)	0.89	152	490	30	55	365	14.8	102	NA
Stony Gorge Reservoir	818	825	25,300	18,700	1.35	31	88	14	42	10	2.8	281	0.01
Thermalito Afterbay	142(2)	4,550	57,500	NA	NA	NA	38	NA	NA	NA	NA	NA	NA
Thermalito Forebay	231(2)	600	11,400	NA	NA	NA	75	NA	NA	NA	NA	NA	NA
Whiskeytown Lake	1,204	3,081	221,600	396,900	0.56	72	239	20	15	37	4.8	52	NA

NOTES:
Mean annual parameter values are presented for the period 1970 to 1979 unless otherwise noted.
(1) 1974 to 1979 data.
(2) Design data.
(3) All values for the period 1976 to 1979.
(4) 1974 to 1975 data.
(5) All values for the period 1968 to 1977.
(6) 1974 to 1978 data.
(7) 1968 to 1969 data.

SOURCE:
Leidy and Myers, 1984.

LEGEND:
NA = no information currently available.

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in summer suitable for trout, topped by warm-water surface waters suitable for black bass, sunfish, and catfish.

Warm-water reservoirs usually have low fertility and yield relatively small crops of game fish. Because of extensive drawdowns, the inshore zones inhabited by warm-water species are usually relatively unproductive. The deep, open water portion of a large reservoir also does not provide satisfactory habitat for most game fish.

Big, low-elevation reservoirs, such as Shasta Lake, Lake Oroville, Pine Flat Reservoir, and Lake Berryessa, are two-story reservoirs, essentially warm-water lakes resting on top of trout lakes. These reservoirs provide greater fishing diversity, although extensive drawdowns limit species dependent on shallow-water habitat, such as black bass and sunfish.

GENERAL ABUNDANCE AND DISTRIBUTION

Reservoir construction in California has greatly increased game fish production; however, angling in these reservoirs is generally not outstanding. Most reservoirs are relatively artificial ecosystems that rarely meet all the needs of the species present. Consequently, large, self-sustaining game fish populations are uncommon (Leidy and Myers, 1984).

In most reservoirs, fish populations decline with the aging of the reservoirs. For a variety of reasons, new reservoirs often develop outstanding populations of fish that gradually decline as the reservoir matures. Loss of cover in the form of inundated vegetation is a major contributing factor. Many of the Central Valley reservoirs are more than 25 years old, and these factors are likely reducing reservoir fish communities from population levels sustained shortly after these same reservoirs were filled.

The mid-elevation reservoirs support a mixture of native fishes that lived in the streams prior to dam construction and introduction of exotic species. In many cases, the native species have become uncommon after an initial 5 to 10 years of abundance. Over time, a variety of exotic species tend to dominate the fish fauna in these reservoirs. The exact species composition in each reservoir is related to the history of introductions, but some species are almost universal in their occurrence: bluegill, largemouth bass, carp, golden shiner, black crappie, brown bullhead, mosquitofish, and rainbow trout (hatchery strains). A few native species, such as prickly sculpin, Sacramento sucker, hitch, and tui chub, are permanently established in a number of Central Valley reservoirs (Moyle, 1976). Table II-14 identifies game species present in selected reservoirs.

It is possible to divide the typical reservoir into four ecological zones, each with a more or less distinct fish assemblage: the littoral zone, the epilimnetic zone, the hypolimnetic zone, and the deep-water benthic zone (Moyle, 1976). These zones are present primarily during summer, and could, even then, be disrupted by extreme drawdowns. Some species move freely between the zones, probably because of the habitat instability. Zonation can also disappear in winter when low temperatures reduce fish activity. The following habitat descriptions are excerpted from Moyle (1976).

TABLE II-14

GAME FISH FOUND IN SELECTED CENTRAL VALLEY RESERVOIRS

Reservoir	RT	BT	LB	SB	C	SF	CF/ BH	STB/ AS	KS	CH/ CHS
Black Butte Lake			X	X	X	X	X			
Camanche Reservoir	X	X	X		X	X	X			
East Park Reservoir			X		X	X	X			
Englebright Lake	X	X	X		X	X	X			
Folsom Lake	X		X	X	X	X	X			
Hensley Lake			X		X	X	X			
Eastman Lake			X		X	X	X			
Jenkinson Lake	X		X		X	X	X			
Keswick Reservoir	X	X	X		X	X	X			
Lake Berryessa	X	X	X	X	X	X	X		X	X
Isabella Lake	X	X	X		X	X	X			
Lake Kaweah		X	X		X	X	X			
Lake McClure		X	X		X	X	X			
Lake Natoma	X	X	X	X	X	X	X			
Lake Oroville	X	X	X		X	X	X			X
Lake Red Bluff	X		X	X	X	X	X			
Lake Success			X		X	X	X			
Millerton Lake	X	X	X	X	X	X	X	X	X	
New Bullards Bar Reservoir	X	X	X		X	X	X			
New Don Pedro Reservoir	X	XX	X	X	X	X	X			
New Hogan Lake	X		X		X	X	X			
New Melones Reservoir	X	X	X	X	X	X	X			
O'Neill Forebay			X	X	X	X	X	X		
Pine Flat Reservoir	X	X	X	X	X	X	X			
San Luis Reservoir			X	X	X	X	X	X		
Shasta Lake	X	X	X	X	X	X	X		X	X
Stony Gorge Reservoir			X		X	X	X			
Thermalito Afterbay			X	X	X	X	X			
Thermalito Forebay			X	X	X	X	X			
Whiskeytown Lake	X	X	X	X	X	X	X		X	

LEGEND:

AS: American shad
BH: bullhead
BT: brown trout
C: crappie
CF: catfish
CH: chinook salmon
CHS: coho salmon
KS: kokanee salmon
LB: largemouth bass
RT: rainbow trout
SB: smallmouth bass
SF: sunfish
STB: striped bass

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The littoral zone occurs along the edges, down to the depth of light penetration, or to the upper limits of the thermocline, whichever occurs first. It is the zone most severely affected by water-level fluctuations, which are common in most Central Valley reservoirs. Large areas could be flooded or dewatered in a short period of time. Despite the fluctuations, large numbers of fish are found in this zone. Bluegill, largemouth bass, and golden shiners live close to the shore near the surface. Brown bullhead, white catfish, and carp stay close to the bottom. Black crappie cluster around submerged boulders and logs during the day, moving into open water to feed on plankton and fish in the evening. Reproduction is a problem for most of these species; a sudden drop in water surface elevation could expose nests of eggs, or a sudden rise could submerge nests to unfavorable depths.

The epilimnetic zone occupies the well-lit, well-oxygenated surface waters away from the shore and above the thermocline. Its fish fauna is the most variable from reservoir to reservoir. Zooplankton is the primary food source, and three main types of fishes are present: plankton-feeding larvae of littoral zone fishes, such as bluegill and other centrarchids; plankton-feeding adult fishes, such as threadfin shad; and fish that prey on plankton feeders, such as striped bass.

Threadfin shad is the most common permanent plankton-feeding species of the epilimnetic zone. Other plankton grazers, such as hitch, tui chub, inland silverside, and American shad, could be found in reservoirs lacking threadfin shad. Striped bass assume the role of chief epilimnetic predator in a number of reservoirs, although their inability to spawn in most reservoirs means that they must be planted on a regular basis. Fish from other zones prey on epilemnetic fishes.

The hypolimnetic zone occupies the cold (less than 68 degrees Fahrenheit) water below the thermocline in deep reservoirs that stratify during summer. The main inhabitants are rainbow trout, which often enter the epilimnetic zone at night to feed on whatever forage fish are most abundant. Kokanee salmon are also abundant in reservoirs where they have been planted, but they stay in the hypolimnetic zone during summer to feed on zooplankton.

The deep-water benthic zone is on the bottom, below the thermocline and usually below the limits of light penetration. It is the one zone in which native fish, especially prickly sculpin and Sacramento sucker, are abundant. White and channel catfish also live in this zone but move into the littoral zone at night to feed.

LIFE HISTORIES

Life histories, distribution, and feeding preferences of game species in Central Valley reservoirs have been summarized primarily from Moyle (1976).

Rainbow Trout

Rainbow trout are the most abundant and widespread salmonid in Central Valley reservoirs. They are successful because they are adaptable to a wide variety of aquatic habitats and are flexible in their life history patterns.

The cold, deep waters of reservoirs provide suitable habitat for rainbow trout although they do not spawn in reservoirs; suitable spawning tributaries to the reservoirs must be present for self-sustaining populations. Most reservoirs have hatchery plants of rainbow trout regularly to sustain populations and sport fishing harvest rates. Catchable-sized trout are typically stocked. Rainbow trout usually spawn in spring, with specific timing varying primarily with reservoir elevation and water temperatures. Trout then typically enter reservoirs from the upstream spawning streams as fry or juveniles.

In reservoirs, rainbow trout seldom live for more than six years or grow more than 18 inches long. Although benthic invertebrates and zooplankton seem to be preferred prey items, terrestrial insects will be consumed when other food is scarce. Rainbow trout in reservoirs, particularly trout more than 12 inches long, also have a greater proclivity for consuming fish. The preferred forage fish in most reservoirs are threadfin shad and pond smelt, although sculpins and suckers will also be eaten. Trout growth has been outstanding in Shasta Lake and Pine Flat Reservoir, where threadfin shad have been introduced (Borgeson, 1966). Optimum temperatures for growth and for completion of most stages of their life histories seem to be 55 to 70 degrees Fahrenheit.

Brown Trout

Brown trout were established in California in the late 1800s and are now present in most large reservoirs. They are generally not stocked in reservoirs and are much less abundant than rainbow trout. Most brown trout populations in reservoirs require streams with gravel riffles to spawn; however, some spawning could take place in reservoirs on gravel bars close to shore.

Brown trout are sexually mature in their second or third year. Spawning normally takes place in November and December when water temperatures fall to 43 to 50 degrees Fahrenheit. The eggs typically hatch in seven to eight weeks, depending on temperature. The alevins emerge from the gravel and begin feeding three to six weeks later. The fry tend to live in quiet waters close to shore, among large rocks, or under overhanging plants.

In reservoirs, small juvenile brown trout feed on aquatic insect larvae. As they grow, they shift to a fish diet. Optimal temperatures for good growth and survival are 54 to 66 degrees Fahrenheit (Raleigh et al., 1986).

Largemouth Bass

Largemouth bass were first introduced into California in 1874 and have since spread to most suitable waters. They are abundant in reservoirs and are normally found in warm, quiet waters with low turbidity and beds of aquatic plants. Largemouth bass provide an important sport fishery component of the Central Valley reservoirs and are one of the most sought after warm-water game fish in California.

Largemouth bass spawn for the first time during their second or third spring. Spawning activity usually begins in April, when water temperature reaches 61 degrees Fahrenheit, but could continue through June. Males build nests in sand, gravel, or debris-littered bottoms at a depth of 3 to 6 feet. Rising reservoir levels could submerge active nests up to 15 feet during spring. The

eggs adhere to the substrate and hatch in two to five days. The sac fry usually spend five to eight days in or around the nest.

For the first month or two, fry feed mainly on rotifers and small crustaceans. By the time they are 2 to 3 inches long, they feed primarily on aquatic insects and fish fry. After reaching a length of 4 inches, largemouth bass feed primarily on fish and large aquatic invertebrates. Optimal temperatures for growth are 68 to 86 degrees Fahrenheit, and 15-inch large bass in California reservoirs are four to five years old.

Food availability for largemouth bass could be affected by competition and by the amount of cover available to prey. Competition effects are likely to be most severe for young-of-the-year bass because they feed on zooplankton and other small invertebrates favored by many other fishes. In reservoirs, threadfin shad could depress the growth and survival of young bass, presumably by reducing invertebrate populations.

The overall quality of bass fishing in California reservoirs has declined since the reservoirs were constructed due to three main factors: overfishing, reservoir aging, and competition from threadfin shad and other plankton-feeding fishes (Von Geldern, 1974). Largemouth bass are extremely vulnerable to angling, and at least half the population of legal-size fish are caught annually in many reservoirs. Over time, the catch rate declines and the fish caught are smaller on the average. Reservoir aging reduces cover and forage fish, which reduces largemouth bass populations. Competition between young bass and other plankton-feeding fish, primarily threadfin shad, also reduces largemouth bass populations.

Smallmouth Bass

Smallmouth bass were first introduced into California in 1874 and have since spread to most suitable waters. They have become established in large, two-story reservoirs and are normally found in cool waters, often near the upstream end of the impoundments. Compared to largemouth bass, smallmouth bass are of minor importance as sport fish. Their populations are scattered and mostly small, but the populations in the upper reaches of Pine Flat Reservoir and Millerton and Folsom lakes can provide excellent opportunities to catch large, fast-growing fish.

Smallmouth bass spawn for the first time during their third or fourth year. Spawning activity usually begins in April, when water temperature reaches 55 to 61 degrees Fahrenheit. Males build nests in rocky bottoms at a depth of 3 feet in the reservoir or in the lower portions of tributary streams of the larger rivers. The male guards the nest until the eggs hatch in 3 to 10 days. The sac fry usually spend 3 to 4 days in the nest. The male herds and guards the fry for an additional one to three weeks; then they disperse into shallow water.

For the first month or two, fry feed mainly on rotifers and small crustaceans. By the time they are 2 to 3 inches long, they feed primarily on aquatic insects and fish fry. Once smallmouth bass exceed 4 inches, they feed primarily on fish and large invertebrates. Growth in Central Valley reservoirs is excellent, so four-year-old fish are typically approximately 15 inches long and larger fish are not uncommon (Emig, 1966). Optimum temperatures for growth and survival are approximately 68 to 81 degrees Fahrenheit.

Crappie

Both black crappie and white crappie are abundant in Central Valley reservoirs since their successful introduction into the state in 1917 and 1908, respectively. The white crappie is more tolerant of turbidity, high temperatures, and lack of cover than the black crappie and could be displacing black crappie in some areas. Crappie are an abundant and popular game fish in many reservoirs. They provide particularly good fishing in spring and early summer. (Moyle, 1976.)

Crappie tend to be found in highly localized schools that spend their days around large, submerged objects and their evenings and early mornings foraging in open water. Crappie are sexually mature in their second to third year. Spawning begins in March or April as temperatures exceed 57 to 63 degrees Fahrenheit and could continue through July. Males build nests in water less than 3 feet deep in or near beds of aquatic plants. The newly hatched fry are guarded by the male for a few days.

Crappie are opportunistic, mid-water feeders; zooplankton and small insect larvae are primary food sources for small fish. Adult crappie, fish longer than 4 inches, tend to eat mainly fish. Optimum temperatures are in the range of 63 to 81 degrees Fahrenheit (Edwards et al., 1982a, 1982b).

Sunfish

Sunfish are represented by species in the genus *Lepomis*, such as bluegill, redear sunfish, green sunfish, and warmouth. Sunfish were introduced into California in the late 1800s and early 1900s and are now found in every Central Valley reservoir. They maintain popular sport fisheries, and fishing usually has little effect on their populations because of their high reproductive rates.

Bluegill is the most widespread and abundant sunfish found in the reservoirs. They have the ability to survive and reproduce under a wide variety of conditions. Bluegill become too abundant in some reservoirs, and severe competition limits individual growth rates, resulting in a large population of stunted fish. Such large populations can limit populations of other game fishes both by eating eggs and young and by eating food that other young game fish need to survive.

Redear sunfish are not as fecund as bluegill and rarely have stunted populations in California. They are harder to catch than bluegill, live in deeper water, and are generally underexploited in California. Green sunfish are usually abundant only in areas of reservoirs with shallow, weedy areas that exclude most other species. Warmouth are relatively uncommon and are not a dominant sunfish in reservoirs.

In general, sunfish spawn in the spring when water temperatures reach 64 to 70 degrees Fahrenheit. Spawning fish could be one year old but are usually two or three years old. Males build nests in sand, gravel, or debris-littered bottoms and guard the nest until the eggs hatch in two to three days. The fry then disperse into aquatic plant beds in shallow water. Bluegill spend most of their lives in a rather restricted area, even in the large reservoirs.

Sunfish are generally highly opportunistic feeders, taking whatever animal food is abundant in their shallow-water habitat. They typically feed on aquatic invertebrates, planktonic crustaceans, terrestrial insects, snails, small fish, fish eggs, and crayfish. Sunfish will also eat aquatic plants and algae when other food is scarce. Goodson (1965) found that bluegill in Pine Flat Reservoir fed largely on fish eggs, midge larvae, and cladocerans in spring, switching to flying insects in summer, and going back to midge larvae and cladocerans in winter. Growth is most rapid in water between 59 and 77 degrees Fahrenheit.

Catfish and Bullhead

Several catfish and bullhead species are found in California reservoirs, having been introduced into California in the late 1800s. Like sunfish, catfish and bullhead have the ability to survive and reproduce under a wide variety of conditions, and they are an abundant game fish in all reservoirs.

The most important catfish and bullhead species are brown bullhead, channel catfish, and white catfish. Brown bullhead are abundant in all foothill reservoirs, where they are usually associated with the deep end of the littoral zone, aquatic plants, and muddy or sandy bottoms. Channel catfish are common in nearly all, if not all, Central Valley reservoirs.

Catfish and bullhead typically begin spawning during their third year. Spawning generally occurs in late spring from April through July when water temperatures increase over 68 degrees Fahrenheit. A sudden rise in water temperature often triggers spawning. A shallow nest depression is constructed where eggs are laid (with the exception of channel catfish, which spawn in cavelike sites). Eggs hatch within a few days at the warm-water temperatures and are typically guarded by one or both of the parents.

Catfish and bullhead are omnivorous and opportunistic bottom feeders. Younger, smaller fish feed primarily on chironomid larvae and small crustaceans. As they grow, fish and larger prey become more important parts of their diets. Catfish and bullhead prefer warm-water temperatures in the range of 68 to 86 degrees Fahrenheit.

Striped Bass and American Shad

Striped bass and American shad populations in reservoirs are limited to Millerton Lake, O'Neill Forebay, and San Luis Reservoir. The Millerton Lake populations are maintained by spawning in the upper reaches of the San Joaquin River. The populations in San Luis Reservoir are maintained by eggs, fry, and small fish being pumped in with water from the Delta. Striped bass populations maintained by planting are now found in San Antonio Reservoir and Lake Mendocino. These fisheries are quite small but provide a valuable trophy fishery in these reservoirs.

Age at maturity for female striped bass is four to six years and two to four years with American shad. Spawning occurs in May and June when temperatures are between 59 and 68 degrees Fahrenheit. Striped bass and American shad move into the San Joaquin River from Millerton Lake and spawn in open water. The eggs are slightly heavier than freshwater, so they sink slowly

as they are transported downstream. They hatch in approximately two days, and the larvae are helpless for seven or eight days. The larvae then begin feeding on small zooplankton.

Adult striped bass are open water predators and opportunistic feeders at the top of the food web, feeding on threadfin shad, smaller striped bass, and any other fish they can catch. By the end of their fourth year, Millerton Lake striped bass are typically 20 inches long and grow approximately 2 inches each year thereafter. American shad typically feed on large zooplankton and other invertebrates. The preferred temperature range for both species is 61 to 68 degrees Fahrenheit.

Kokanee Salmon

Kokanee salmon are the non-anadromous form of sockeye salmon. Kokanee have been successfully introduced into several cold-water reservoirs in California beginning in the 1940s, and they are now present in systems including Lake Berryessa, Clair Engle Lake, Millerton Lake, Shasta Lake, and Pardee Reservoir.

Kokanee mature in two to four years. Most of the recent kokanee transplants into California have been from populations with two-year life cycles because these populations also seem to have superior growth rates. Kokanee salmon are usually 6 to 8 inches long at maturity. They generally spawn between early August and early February, the time being determined largely by the genetic background of the fish and stream and reservoir temperatures. Spawning requires temperatures of 43 to 55 degrees Fahrenheit. Most spawning takes place in tributary streams, but some lake spawning has been reported. Kokanee build redds, and fry emerge in April through June and move downstream immediately. (Moyle, 1976.)

Kokanee are zooplankton grazers, although they will occasionally eat larval fish. The food sources of kokanee salmon change little as the fish grow. Feeding ceases in the winter and just prior to spawning. Kokanee will inhabit surface waters of the reservoirs as long as temperatures remain in the preferred range (approximately 43 to 59 degrees Fahrenheit) or colder. As surface waters get warmer, the fish gradually move deeper. In Shasta Lake, kokanee stay in the hypolimnion during summer. Occasionally, heavy kokanee mortality will occur if the hypolimnion becomes depleted of oxygen.

Angling for kokanee has become popular only in recent years. Most populations are probably underexploited because catching them requires specialized techniques. This low fishing mortality, combined with the small plankton populations in some California lakes and reservoirs, such as Clair Engle Lake, has led to stunted kokanee populations. Kokanee populations declined significantly in Shasta Lake coincident with the threadfin shad boom (Borgeson, 1966).

Chinook Salmon and Coho Salmon

Chinook and coho salmon have been introduced to Lake Oroville, Shasta Lake, and Lake Berryessa. Coho salmon are still present in Lake Berryessa although such populations will not reproduce naturally. In Lake Berryessa, planted coho feed almost exclusively on threadfin shad.

FACTORS AFFECTING ABUNDANCE

Fishery management problems limiting optimal sport fishery development in Central Valley reservoirs were appraised as part of the Central Valley Fish and Wildlife Management Study (Leidy and Meyers, 1984). Water-level fluctuation was the most frequently cited factor adversely affecting fishery production. The second principal environmental problem limiting sport fish production in reservoirs was the limited cover habitat available to fish as shelter. The effects of fishing on reservoir fish communities are not well understood, although overfishing of naturally reproducing populations of game fishes seldom seems to be a problem (Moyle, 1976). The following discussions primarily summarize Leidy and Myers (1984). Fishery management problems at individual reservoirs are summarized in Table II-15.

Water-Level Fluctuations

Extreme water-level fluctuation in reservoirs is perhaps the most significant environmental factor influencing reservoir fish population productivity and is a direct result of reservoir management priorities. The direct and indirect effects of fluctuating water levels are responsible to a large degree for other fishery management problems, such as limited cover habitat, limited littoral habitat, and shoreline erosion. Reservoirs in the Central Valley operate to store water during winter and spring and release water in summer and fall. This pattern of storage and releases results in variable, seasonal availability of water in reservoirs. Surface water fluctuations in some Central Valley reservoirs could exceed 100 feet annually.

Reservoir drawdown rates and their effects on spawning success for largemouth bass, smallmouth bass, and redear sunfish were developed based on existing literature and professional judgment for Nacimiento and San Antonio reservoirs (Jones & Stokes Associates, 1990). These two reservoirs vary dramatically in their topography, and the general criteria developed for these reservoirs are probably generally applicable to Central Valley reservoirs. For largemouth bass, reservoir drawdown rates of less than 4, 4 to 6, 7 to 10, and greater than 10 feet per month were rated as excellent, good, fair, and poor, respectively, for spawning success. For smallmouth bass, reservoir drawdown rates of less than 7, 7 to 12, 13 to 18, and greater than 18 feet per month were rated as excellent, good, fair, and poor, respectively, for spawning success. For redear sunfish, reservoir drawdown rates of less than 6, 6 to 10, 11 to 15, and greater than 15 feet per month were rated as excellent, good, fair, and poor, respectively, for spawning success. Daily reservoir drawdown rates based on percent of successful nests have also been developed for largemouth and smallmouth bass in Folsom Lake and New Don Pedro Reservoir (Lee, pers. comm.).

Fluctuations can affect reservoir productivity directly in several ways. Water-level changes affect physical, chemical, and biological parameters, which in turn directly or indirectly affect fish populations as follows:

- Changes in Surface Area. Most primary production occurs near the surface, and a greater surface area will yield higher total primary production.

TABLE II-15

COMPARISON OF FISHERY PROBLEMS AT SELECTED CENTRAL VALLEY RESERVOIRS

Reservoir	No Fishery Mgmt Plan	Limited Fisheries Data	Water-Level Fluctuation	Excessive Harvest	Under-harvest	Limited Cover Habitat	Limited Spawning Habitat	Limited Littoral Habitat	Low Water Fertility	Water Quality Problems	Limited Fishery	Forage Fish-Related Problems	Undesir-able Species	Shore-line Erosion	Multiple Use Conflict	Angler Identified Per Access	Number of Problem Reservoirs
Black Butte Lake	X	X	X		X	X					X	X	X	X			8
Camanche Reservoir																	
East Park Reservoir	X	X	X			X							X	X	X		7
Englebright Lake	X	X				X			X			X				X	6
Folsom Lake	X	X	X	X	X	X	X		X								8
Hensley Lake	X	X								X	X						4
H.V. Eastman Lake	X	X	X	X													4
Jenkinson Lake	X	X				X				X				X	X		6
Keswick Reservoir	X	X								X							3
Lake Berryessa			X			X									X		3
Lake Isabella		X	X			X	X						X	X			6
Lake Kaweah	X	X	X			X							X	X	X		7
Lake McClure																	
Lake Natoma	X	X	X					X	X							X	6
Lake Oroville	X	X	X			X		X			X	X					7
Lake Red Bluff	X	X															2
Lake Success	X	X	X	X		X				X	X						7
Millerton Lake	X		X	X		X											4
New Bullards Bar Reservoir																	
New Don Pedro Reservoir																	
New Hogan Lake	X	X	X		X	X				X							6
New Melones Reservoir																	
O'Neill Forebay	X	X	X		X											X	5
Pine Flat Reservoir	X	X	X		X	X			X					X	X		8
San Luis Reservoir	X	X	X			X					X						5
Shasta Lake	X		X	X		X				X	X	X	X				8
Stony Gorge Reservoir	X	X	X			X							X	X	X		7
Thermalito Afterbay																	
Thermalito Forebay																	
Whiskeytown Lake	X	X							X			X	X				5
Number of reservoirs with identified problems	21	20	17	5	5	16	2	2	5	6	6	4	7	7	6	3	

SOURCE:
Leidy and Myers, 1984

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- Changes in Mean Depth. Depth could influence the degree of stratification, and as a result, temperature, oxygen, and total dissolved solids profiles. Additionally, the extent of littoral areas could be altered.

- Changes in Reservoir Volume. In some reservoirs, greater volume results in greater species diversity and greater total biomass and abundance of plankton and fish.

- Changes in Flushing Rate. A high flushing rate could continually draw plankton and fish out of the reservoir, resulting in lower total productivity, as found in Lake Natoma and Lake Red Bluff.

- Changes in Outlet Depth. Epilimnial outlets tend to draw off the most productive layer of water in the reservoir. Hypolimnial outlets could break down temperature stratification and affect mineral and dissolved gas distributions.

- Magnitude of Water-Level Fluctuation. The degree of depth change directly affects the quantity of littoral area exposed. Also, the greater the magnitude of fluctuation, the greater the physical and chemical changes to which organisms, macrophytes, and fishes must adjust.

- Annual Changes in Fluctuation Pattern. Long-range or annual changes in water-level fluctuation could be reflected in changes in reservoir flora and fauna composition. Random fluctuations in water level over time could inhibit stabilization of the fish community of the reservoir.

- Changes in Temperature Stratification. Fluctuating water levels could affect temperature stratification, which in turn could influence the extent of water mixing, oxygen content of epilimnion and hypolimnion, and distribution of incoming nutrients. The temperature regime will affect biological production in the reservoir and the distribution of plankton and fish.

- Changes in Dissolved Oxygen Content. Low reservoir volumes due to water-level fluctuations coupled with high water temperatures and organic decay of benthic materials could produce deoxygenated conditions lethal to fishes. Low hypolimnial oxygen levels due to limited mixing could restrict fish and plankton distribution to the epilimnion.

Minimum Pool

Black Butte Lake, East Park Reservoir, and Stony Gorge Reservoir have no guaranteed minimum pool for the maintenance of fishes during low water periods. In cases of drought or extreme water-level fluctuations, these reservoirs could be completely drained, eliminating the fish populations.

Excessive Harvest

Excessive harvesting of certain species or sizes of fish by anglers results in adverse effects on the fish population. In the long run, excessive angler harvest restricts the availability of fish to the angler. Five reservoirs have excessive fish harvest problems (Table II-15). All of these

reservoirs reported overharvesting of largemouth and smallmouth bass. Excessive harvest of the largest bass results in a smaller average size of bass. The larger fish are mature adults that constitute much of the reproductive potential of the population. Removing these fish could reduce bass reproduction significantly. Overharvesting results in a shift in the population to smaller, though mature, fish as the fishery becomes dominated by the progeny of small largemouth bass.

Underharvest

Five reservoirs have reported that anglers were underharvesting catfish populations (Table II-15). Catfish were the only group of sport fish that were not harvested by anglers to the degree that they could be. Two reasons have been identified for the limited angler interest in fishing of catfish: lack of angler access to the catfish resource and lack of knowledge by anglers about how to fish for catfish. Opportunities for bluegill capture also cannot be fully used by anglers in certain reservoirs.

Limited Cover Habitat

The lack of adequate quantity or quality of cover habitat in reservoirs is a significant factor limiting production of warm-water fishery resources. Sixteen reservoirs have serious cover habitat problems (Table II-15). The lack of cover restricts development of black bass, sunfish, and crappie populations. Cover provides shelter for these fish for spawning and rearing. Cover habitat also improves food availability.

Cover could take many forms, but as a rule, reservoir cover implies structural relief, which fish can use for shelter and feeding. During the construction of many reservoirs, trees and brush were cleared from the reservoir basin. Clearing of vegetation resulted in the removal of potential cover habitat; therefore, in many of these reservoirs, rocks and boulders were the only cover remaining. This limited type of cover, coupled with extreme water-level fluctuation, has severely limited the productivity of reservoir fish populations dependent on cover.

The lack of established rooted aquatic vegetation is another problem common to reservoirs. A variety of factors, including fluctuating water levels, shoreline erosion, and cattle grazing, prevent vegetation from becoming established. Studies have shown a positive correlation between flooded vegetation during the spring spawning period and highly successful reproduction. Sheltered areas during spring and summer drawdown are essential for providing escape cover for juvenile fish.

Limited Spawning Habitat

Water-level fluctuations during the spawning period at Isabella and Folsom lakes greatly reduce the productivity of largemouth bass (Table II-15). Spawning habitat is degraded from sedimentation caused by shoreline erosion and exposure and desiccation because of fluctuating water levels.

Limited Littoral Habitat

Extensive and well-developed littoral habitats are critical factors in determining primary and secondary productivity in reservoirs. Factors that influence the development of littoral habitat include water depth, vertical extent of light penetration, wave action, nutrient supply, substrate texture, and water-level fluctuation.

Lake Natoma and Lake Oroville have limited fish productivity because of limited littoral habitat (Table II-15). In Lake Natoma, the development of littoral habitat is limited by low water fertility, high flushing rates, limited shallow-water habitat, and large daily water-level fluctuations. Littoral habitat at Lake Oroville is limited by low water fertility, limited shallow-water habitat, large seasonal water-level fluctuations, and wave action.

Low Water Fertility

Water fertility is a function of the geochemical nature of the watershed and land uses in the watershed. The quantity and quality of dissolved solids often determine the variety and abundance of plants and animals in a given aquatic environment. Total dissolved solids (TDS) concentrations less than 100 milligrams per liter generally indicate low fertility. Five reservoirs identified in Table II-15 have limited fisheries production because of low water fertility.

Water Quality Problems

Water quality problems affect fish production at six reservoirs (Table II-15). Low dissolved oxygen concentrations affect fish populations at Hensley Lake, New Hogan Lake, and Lake Success during summer. Hydrogen sulfide production is directly related to low dissolved oxygen. Decomposition of vegetation on the reservoir bottom consumes oxygen and releases hydrogen sulfide, which is highly toxic to fish.

High water temperatures during summer at New Hogan Lake and Lake Success limit salmonid fisheries. Low temperatures at Keswick Reservoir limit the total productivity of the aquatic ecosystem, which is reflected in limited fish production.

Shasta Lake and Keswick Reservoir are affected by heavy metal pollution. High concentrations of copper, zinc, and cadmium have been linked to fish kills in these reservoirs. These metals also precipitate out of solution and accumulate in bottom sediments, possibly affecting benthic organisms.

Limited Fishery Problems

Problems associated with a limited fishery are related to the availability of desirable species of appropriate size to the angler. Six reservoirs, identified in Table II-15, have limited fishery problems:

- Availability of largemouth in Black Butte, Shasta, and Hensley lakes is limited.

- Striped bass populations are declining in San Luis Reservoir, which is probably a reflection of the general decline of stripers in the Delta.

- Chinook and coho salmon planted in Lake Oroville remain in the deep-water benthic zone during summer and are consequently inaccessible to the average angler.

- The bass fishery at Lake Success has been affected by overharvesting.

Forage Fish-Related Problems

Forage fish provide a food source for larger fish-eating game fish. Four reservoirs, identified in Table II-15, have problems associated with limited forage fish:

- Englebright and Whiskeytown lakes have limited forage fish resources.

- Underutilization of forage fish and forage fish competition with game fish occur at Lake Oroville.

- Widely fluctuating forage fish populations affect the availability of game fish food in Shasta Lake.

Non-Native Species

Productivity of game fishes could be limited or reduced by non-native species through competition for food, cover, or spawning habitat as well as predation, disease, and habitat modification. The presence of non-native species has been identified in seven reservoirs (Table II-15):

- Whiskeytown Lake, Black Butte Lake, Shasta Lake, East Park Reservoir, and Stony Gorge Reservoir have an overabundance of nongame fishes that compete with or prey on game fishes.

- The large Asiatic clam (*Corbicula* sp.) population in Isabella Lake reduces plankton populations used by juvenile game and forage species.

- White bass, a non-native game species, has been established in Lake Kaweah by an unauthorized introduction.

Shoreline Erosion

Shoreline erosion has been identified as a fishery management problem at seven reservoirs (Table II-15). Shoreline erosion reduces fishery production by silting spawning areas and limiting the establishment of vegetation in the littoral zone. The reservoirs affected and the primary causes of shoreline erosion are listed as follows:

- Black Butte Lake, Jenkinson Lake, Stony Gorge Reservoir, and East Park Reservoir have shoreline erosion associated with wave action resulting from wind and recreational boating.
- Pine Flat Reservoir and Lake Kaweah have shoreline erosion associated with excessive cattle grazing and trampling.
- Isabella Lake and East Park Reservoir are affected by seasonal water-level fluctuations.

Multiple Use Conflicts

The multiple uses of recreational boating and livestock grazing conflict with sport fishery management at six reservoirs (Table II-15). Boating activities resulted in the following problems:

- angler disturbance in East Park Reservoir and Lake Berryessa;
- decreased bass production because of wave action at Lake Berryessa; and
- aggravated shoreline erosion affecting fish survival and aesthetics at Jenkinson Lake and Stony Gorge Reservoir.

Livestock grazing resulted in increased soil erosion and turbidity and trampling and removal of vegetative cover at Lake Berryessa, Lake Kaweah, and Pine Flat Reservoir.

Angler Access

Two distinct angler access problems are present: limited access to the reservoir and lack of access to fish concentrations in the reservoir. Englebright Lake and Lake Natoma have severe access problems. Most of Englebright Lake is accessible by boat only. Lake Natoma has very steep banks on the north shoreline and is closed to the public near Folsom State Prison. Lake Oroville, Black Butte Lake, and O'Neill Forebay are affected by lack of access to concentrations of fish, which is associated with the limited fishery and underharvesting problems described previously.

ONGOING MONITORING, ENHANCEMENT, AND HABITAT IMPROVEMENT ACTIONS

Central Valley reservoirs have greatly increased the warm-water game fish production in California; however, large, self-sustaining game fish populations are relatively uncommon. Periodic stocking is required to maintain some game fish populations because reproductive success in reservoirs is limited by water-level fluctuations and steep-sloped basins that greatly limit spawning habitats.

Leidy and Myers (1984) pointed out that 21 of the 23 reservoirs investigated in 1984 lacked a specific written fishery management plan. Few additional management plans have been prepared since 1984. Without plans that identified specific management goals and objectives, evaluating

the fishery management needs of each reservoir is difficult. A second problem is the limited amount of fishery data usable for management purposes. Much of the available data are fragmented and of limited aid to the fishery biologist attempting to manage reservoirs on the basis of scientific principles (Leidy and Myers, 1984).

Enhancement and habitat improvement actions have been implemented in a few reservoirs, but not to such a degree that noticeable improvements in fish populations have been realized.

RECENT CONDITIONS FOR BAY FISHERIES COMMUNITIES

GENERAL ABUNDANCE AND DISTRIBUTION

Historical Population Trends (1850-1966)

The San Francisco Bay and Sacramento-San Joaquin Delta were virtually undisturbed until the mid-1800s. From the California gold rush to the present, the abundance and distribution of Bay fisheries resources have been modified by human activities, resulting in habitats and conditions becoming increasingly unsuitable for native species. The entire Bay-Delta estuary, including phytoplankton, zooplankton, benthic, and fish communities, has significantly changed since 1850 (DFG, 1993). The changes include the introduction of more than 100 non-native invertebrate species and 50 non-native fish species into the estuary in the last 150 years (DFG, 1993). Overfishing of some Bay species has eliminated or reduced their abundance and distribution.

Many species, including bay shrimp, have fluctuated in abundance partly because of commercial harvest. The commercial catch of this species diminished from the harvest of more than 2,270 tons annually in 1882-1892 to approximately 200 tons annually by 1916 (Herbold et al., 1992). Bay shrimp abundance rebounded and commercial landings increased to approximately 1,000 tons per year by 1935. After the collapse of the export fishery in the 1930s, bay shrimp have been harvested primarily for the bait industry, and populations have not been heavily exploited.

Northern anchovy abundance dramatically increased in the Bay in response to the overfishing of Pacific sardine in the 1930s and 1940s (Herbold et al., 1992). Northern anchovy was never heavily exploited, and populations have not been greatly affected by commercial harvest. Pacific herring populations were heavily exploited in the early 1900s, but its market was eliminated by the Federal Reduction Act of 1919. Since that time, and until recently, the herring fishery has never been large because of lack of consumer acceptance. Populations of both the northern anchovy and Pacific herring in the Bay have fluctuated widely but have not generally declined (Herbold et al., 1992).

Some Dungeness crabs were harvested from inside the Bay prior to the 1900s (Herbold et al., 1992). From early in the century through 1960, Dungeness crab landings rose from approximately 1 to 2 million pounds per year in 1925 to 4 to 8 million pounds per year in 1945-1959. Dramatic changes in oceanic conditions in 1959 resulted in drastic declines in crab catches (Herbold et al., 1992). Since that time, the landings have generally been less than 1 million pounds per year, with the commercial take in 1984-1985 being 384,000 pounds (DFG, 1987).

Current Population Trends (1967-1991)

At least 200 species of fish, shrimp, and crabs are known to inhabit the Bay and Delta (DFG, 1992). More than 120 fish species have been identified in DFG's San Francisco Bay sampling program (Herrgesell, 1993). For the following discussion, "Bay" refers to Suisun Bay, San Pablo Bay, Central Bay, and South Bay to allow the most accurate description of current trends in the abundance and distribution of Bay fishery resources.

Each embayment supports a distinct association of fish and invertebrate species. Extensive studies summarized by the San Francisco Estuary Project have led to fish associations being characterized within each embayment (Herbold et al., 1992). The Suisun, San Pablo, Central, and South bays are dominated by 16 fish species (Table II-16), including northern anchovy and Pacific herring, which are the most abundant species within all embayments.

The IEP has monitored for trends in annual distributions and abundances of fish, crabs, and shrimp. Annual abundance indices have been calculated for many of the monitored species. The following discussion of population abundance and species distributions trends focuses on several of these species, including northern anchovy and Pacific herring (planktivorous fish), white croaker (predatory fish), English sole and starry flounder (bottom fish), and Dungeness crab and caridean shrimps (invertebrates). These species represent significant and commercially important fishery resources of the Bay estuary.

Recent trends in population abundance and distribution of many Bay species have been reviewed by Herbold et al. (1992), DFG (1992, 1993), and Herrgesell (1993). It must be recognized that drought conditions in California since the early 1970s, particularly those of 1976-1977 and 1986-1992, have intensified the downward trend in abundance and have greatly influenced distributions of many of the aquatic species in the estuary (DFG, 1993). The influence of these droughts has been felt at all community levels, including phytoplankton productivity and zooplankton and benthic abundances.

The reductions of these food organisms during drought conditions have greatly influenced the general decline of many higher trophic-level species within the estuary.

Suisun Bay. Phytoplankton productivity has been low in Suisun Bay since 1983, based on chlorophyll *a* concentrations, a measure of aquatic primary productivity. Since 1987, benthic grazing of phytoplankton by the introduced Asiatic clam (*Potamocorbula amurensis*) has resulted in low phytoplankton concentrations in Suisun Bay (Herrgesell, 1993) (Figure II-104).

As a result of high densities of this clam, zooplankton densities within Suisun Bay since 1987 have also been extremely low (Herbold et al., 1992).

Overall, abundance declines of caridean shrimp and fish were greatest in Suisun Bay compared to other embayments during the 1988-1990 years of the drought (DFG, 1992). Furthermore, except for 1985, bay shrimp abundance has continued to decline significantly from peaks in 1982-1983 (Figure II-105) (Herrgesell, 1993). In 1988, *Crangon nigricauda*, a shrimp tolerant of high salinity, replaced the bay shrimp (*C. franciscorum*) for the first time as the most abundant caridean shrimp species in Suisun Bay.

TABLE II-16
COMMONLY OCCURRING FISH SPECIES IN SAN FRANCISCO BAY AS DETERMINED FROM TRAWLS

Species Name		Embayment			
Common	Scientific	South	Central	San Pablo	Suisun
American shad	*Alosa sapodissma*				M
Bay goby	*Lepidogoblus lepidus*	O	O		
Speckled sand dab	*Citharichthys stigmaeus*	O			
Delta smelt	*Hypomesus transpacificus*				M
English sole	*Parophys vetulus*	O	O		
Jacksmelt	*Atherinopsis californiensis*	M	M	M	
Longfin smelt	*Spirinchus thaleichthys*		M, O	M, O	M, O
Plainfin midshipman	*Porichthys notatus*	M			
Northern anchovy	*Engraulis mordax*	M, O	M, O	M, O	M, O
Pacific herring	*Clupea harengeus*	M	M	M	M
Striped bass	*Morone saxatilis*			M, O	M, O
Starry flounder	*Platichthys stellatus*			O	O
Shiner perch	*Cymatogaster aggregata*	M, O	M, O	M, O	
Sacramento splittail	*Pogonichthys macrolepidotus*				M
Staghorn sculpin	*Leptocottus armatus*	O	O	O	O
Topsmelt	*Atherinops affinis*	M			

SOURCE:
Herbold et al., 1992.
LEGEND:
M = mid-water trawl.
O = other trawl.

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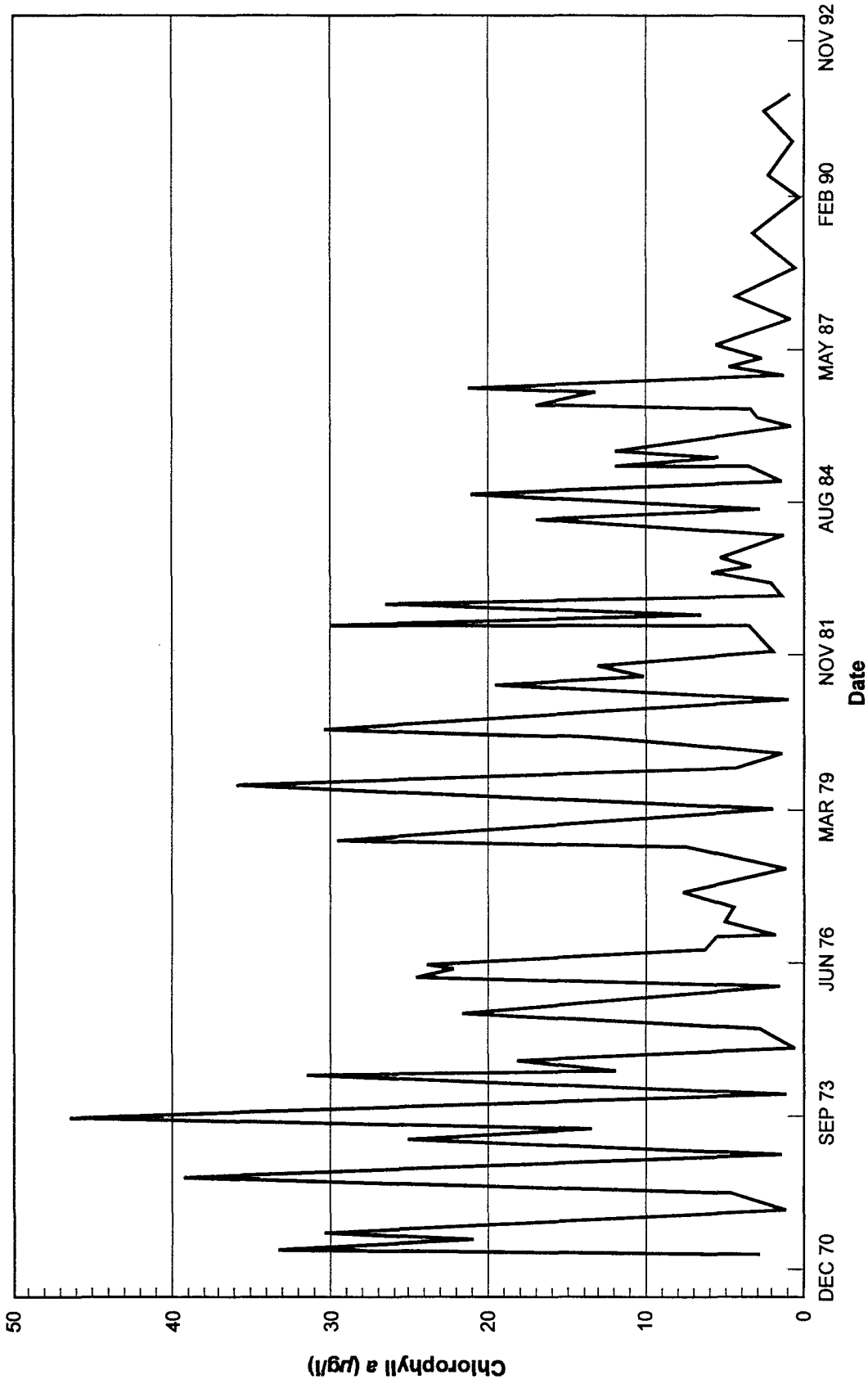
C-081610

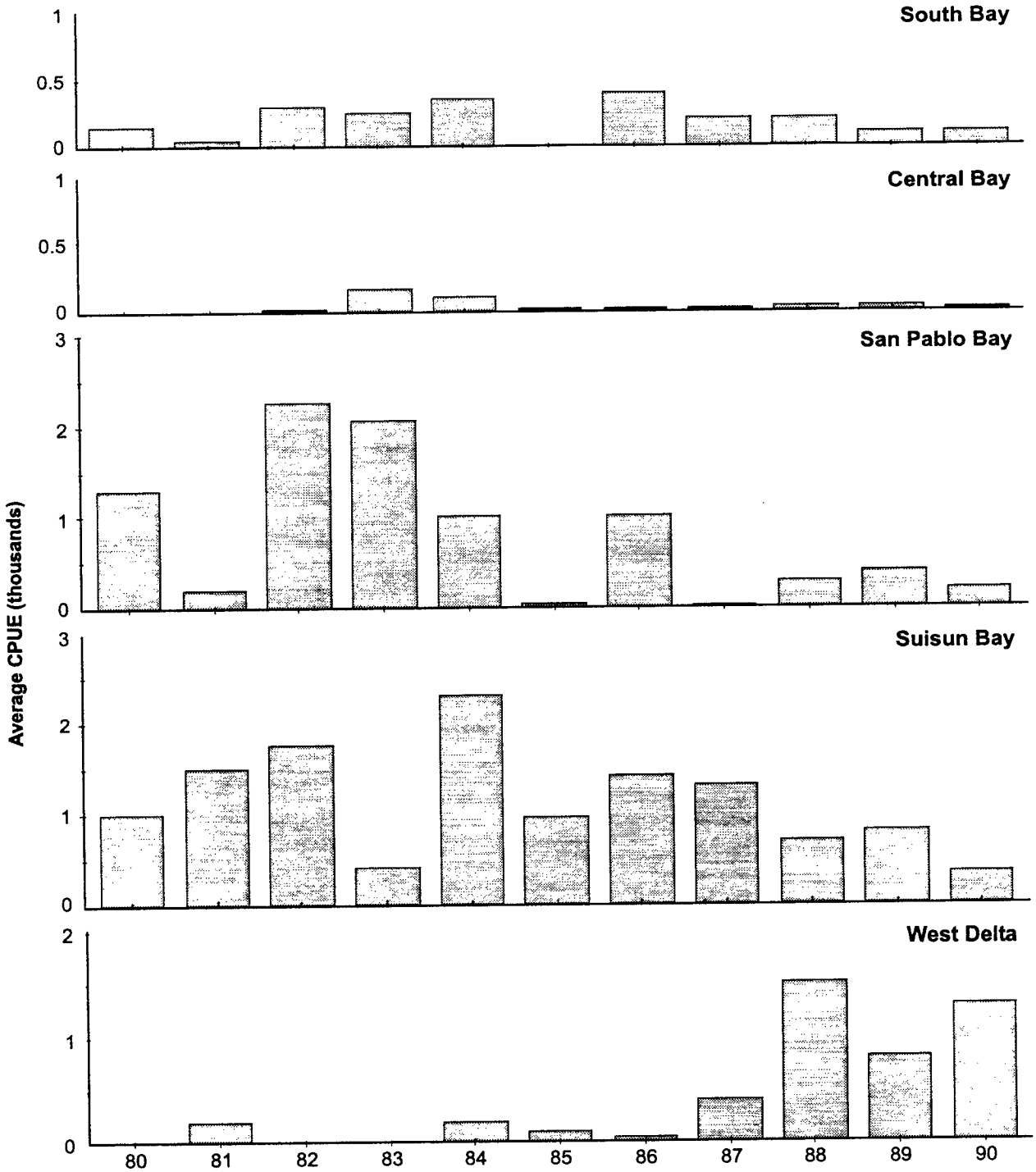
C—081610



SOURCE:
Herrgesell, 1993.

FIGURE II-104

MEAN MONTHLY CHLOROPHYLL *a* CONCENTRATIONS IN SUISUN BAY (1971-1991)



NOTE:
Average CPUE is for all sizes from May to August.

SOURCE:
Herrgesell, 1993.

FIGURE II-105

CATCH PER UNIT EFFORT OF CALIFORNIA BAY SHRIMP (*Crangon franciscorum*) IN FIVE DIFFERENT REGIONS OF THE SACRAMENTO-SAN JOAQUIN DELTA ESTUARY

Populations of many fish species that use Suisun Bay as a nursery area have declined during the past 20 years, including delta smelt, longfin smelt, striped bass, and yellowfin goby. Since 1984, northern anchovy has constituted 70 percent of the catch in Suisun Bay, a significant increase over previous years (Herbold et al., 1992). White sturgeon and the important fish prey organism, opposum shrimp, have remained abundant in areas of Suisun Bay (Herbold et al., 1992). Suisun Marsh, a portion of Suisun Bay, has recently supported high densities of Sacramento splittail and tule perch, both native California fish species. During recent sampling, English sole young-of-the-year were found farther upstream in Suisun Bay than in previous years (Herrgesell, 1993).

San Pablo Bay. Bay shrimp (*C. franciscorum*) abundance in San Pablo Bay has generally declined since 1982-1984 (Figure II-105). In addition, population abundance indices have increased since 1986 for those caridean shrimp species with preferences for higher salinity conditions (*C. nigricauda, C. nigromaculata,* and *Heptacarpus cristatus*) (Herrgesell, 1993).

Phytoplankton productivity in San Pablo Bay has been variable during 1970-1991 (Figure II-106). Phytoplankton productivity during the early 1980s was much greater than during the early 1970s and in 1987-1989 (Herrgesell, 1993). Productivity in San Pablo Bay during 1991 was somewhat greater than the years from 1987 through 1989 but significantly less than during the early 1980s.

Patterns of juvenile Dungeness crab abundance within the Bay during 1980-1991 were related to oceanic conditions during those years. Years with frequent storms (1980, 1982, 1983, and 1986) and warm ocean currents (1982, 1983, and 1987) resulted in poorer-than-normal year classes of Dungeness crabs. Except for some fluctuations in distribution within the upper San Pablo Bay and lower Suisun Bay (the nursery area for juvenile Dungeness crabs), no obvious trends have been observed in Dungeness crab distribution in the Bay (Herrgesell, 1993).

The abundance of many fish species characterizing San Pablo Bay (Table II-16) has declined in recent years, including striped bass and longfin smelt (Herrgesell, 1993). DFG (1992) has determined that the abundance of Bay shrimp and some species of demersal and pelagic fish declined in San Pablo Bay during several years in the period 1979-1988. Recent sampling indicates greater numbers of adult white croakers within San Pablo Bay than in past years (Herbold et al., 1992).

Starry flounder abundance has greatly diminished within San Pablo Bay in recent years. Since 1976, total starry flounder catch has declined rapidly except for 1984-1986 (DFG, 1992) (Figure II-107). English sole young-of-the-year distribution was generally restricted to the Central Bay during several years (1980, 1982-1984, and 1986) but spread farther upstream into San Pablo Bay during 1981, 1985, 1987, and 1988 (Herrgesell, 1993).

Northern anchovy and Pacific herring populations have fluctuated widely in San Pablo Bay, but there have been no obvious trends in abundance from recent sampling (Herbold et al., 1992).

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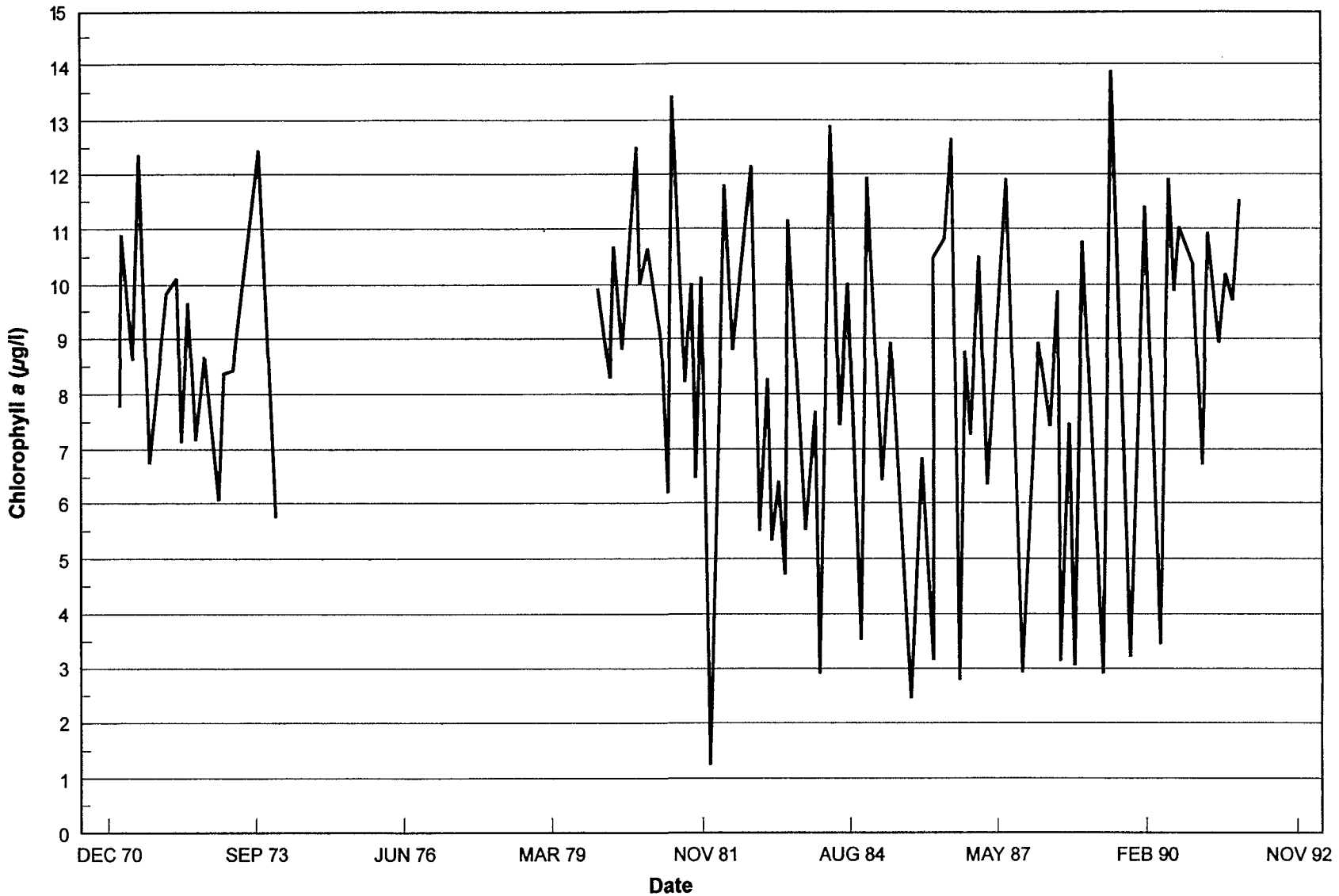

SOURCE:
Herrgesell, 1993.

FIGURE II-106

MEAN MONTHLY CHLOROPHYLL *a* CONCENTRATIONS IN SAN PABLO BAY (1971-1991)

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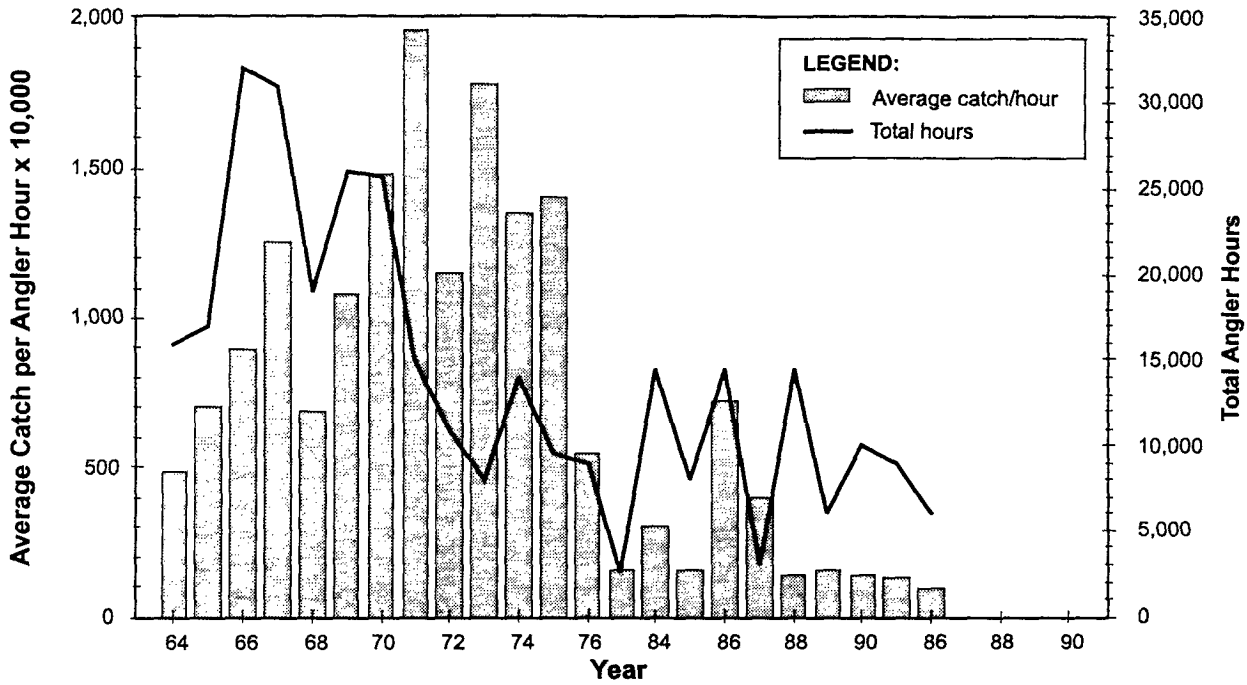
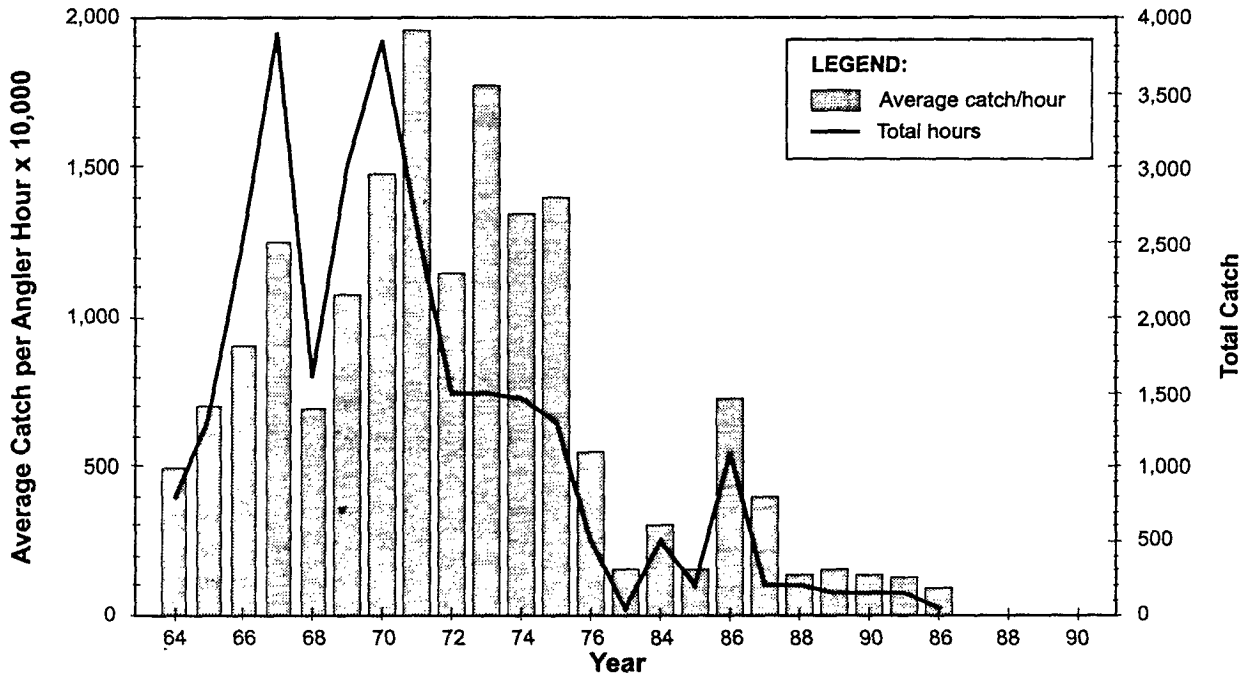
C-081613





NOTE:
Starry flounder catch and fishing effort in San Pablo Bay based on commercial passenger fishing vessel log data from 1964-1990. A January through May period was used to calculate annual data. Data for 1979 and 1981-1983 were not available during analyses and were not plotted.

SOURCE:
DFG, 1992.

FIGURE II-107

RELATIONSHIP OF STARRY FLOUNDER AVERAGE CATCH PER HOUR WITH TOTAL ANGLER HOURS AND WITH TOTAL CATCH (1964-1986)

Central Bay. Abundance of several of the more salinity-tolerant caridean shrimps has increased in the Central Bay since 1987 (DFG, 1992). Recent studies have shown stability in annual fish species compositions and a very high species diversity during all seasons for the Central Bay (Herbold et al., 1992). Dispersal of South Bay species has increased during recent drought conditions, with a decline in catch of formerly abundant South Bay species in the Central Bay. However, fish population abundances for the Central Bay during 1980-1991 have shown no trends through time (Herbold et al., 1992).

South Bay. Several South Bay fish species have increased in abundance since 1987, including white croaker and plainfin midshipman (Herbold et al., 1992). Populations of two other fishes that characterize the South Bay, jacksmelt and shiner perch, did not increase in 1980-1991. Population trends for topsmelt have been variable during 1980-1991. Pacific herring populations fluctuated somewhat during the early 1980s but have stabilized at intermediate levels (Herbold et al., 1992).

Total fish catches in the South Bay have fluctuated over the monitoring period, but total catch has generally increased. Species composition has remained relatively constant during 1980-1991. Except for an increase in white croaker abundance, there has been little change in the composition or abundance of the fish fauna characteristic of the South Bay (Herbold et al., 1992).

LIFE HISTORIES

Northern Anchovy

Of the three distinct subpopulations of this species, the most central population inhabits waters of northern California (Herbold et al., 1992). A subspecies of northern anchovy (*Engraulis mordax nanus*) is thought to be restricted to San Francisco Bay (Emmett et al., 1991). The migration pattern for northern anchovy is primarily one of inshore-offshore movement. Northern anchovies mature at less than one year (3 to 4 inches in length) to four years, depending on population location and size (Emmett et al., 1991). Northern anchovy use San Francisco Bay extensively as a nursery and for limited spawning (DFG, 1987). This species spawns in groups in nearshore coastal waters from January through April (Herbold et al., 1992). Spawning has been documented within San Francisco Bay (Herbold et al., 1992), however, and takes place from mid-June through mid-August in nearshore waters (Herbold et al., 1992). Although northern anchovy peak spawning occurs in the spring and summer, DFG has collected eggs and larvae in all months (White, pers. comm.).

Anchovy eggs are widely distributed in the Bay in clear, thermally stratified waters with low zooplankton abundance (Herbold et al., 1992). The planktonic embryos and larvae are then distributed by currents into the Bay in different patterns. Greatest northern anchovy larval abundance was found in Bay waters that contained high zooplankton abundance and lower water clarity (Herbold et al., 1992). Eggs are tolerant of salinities of 32 to 35 parts per thousand (Emmett et al., 1991). Young anchovies are first caught in the Bay prior to, or simultaneously with, the catch of older fish and are believed to be transported into the Bay from the ocean (Herbold et al., 1992). Northern anchovy are seasonally present in San Francisco Bay, with entry in April and emigration from the Bay taking place in fall (Figure II-108). In the less saline waters

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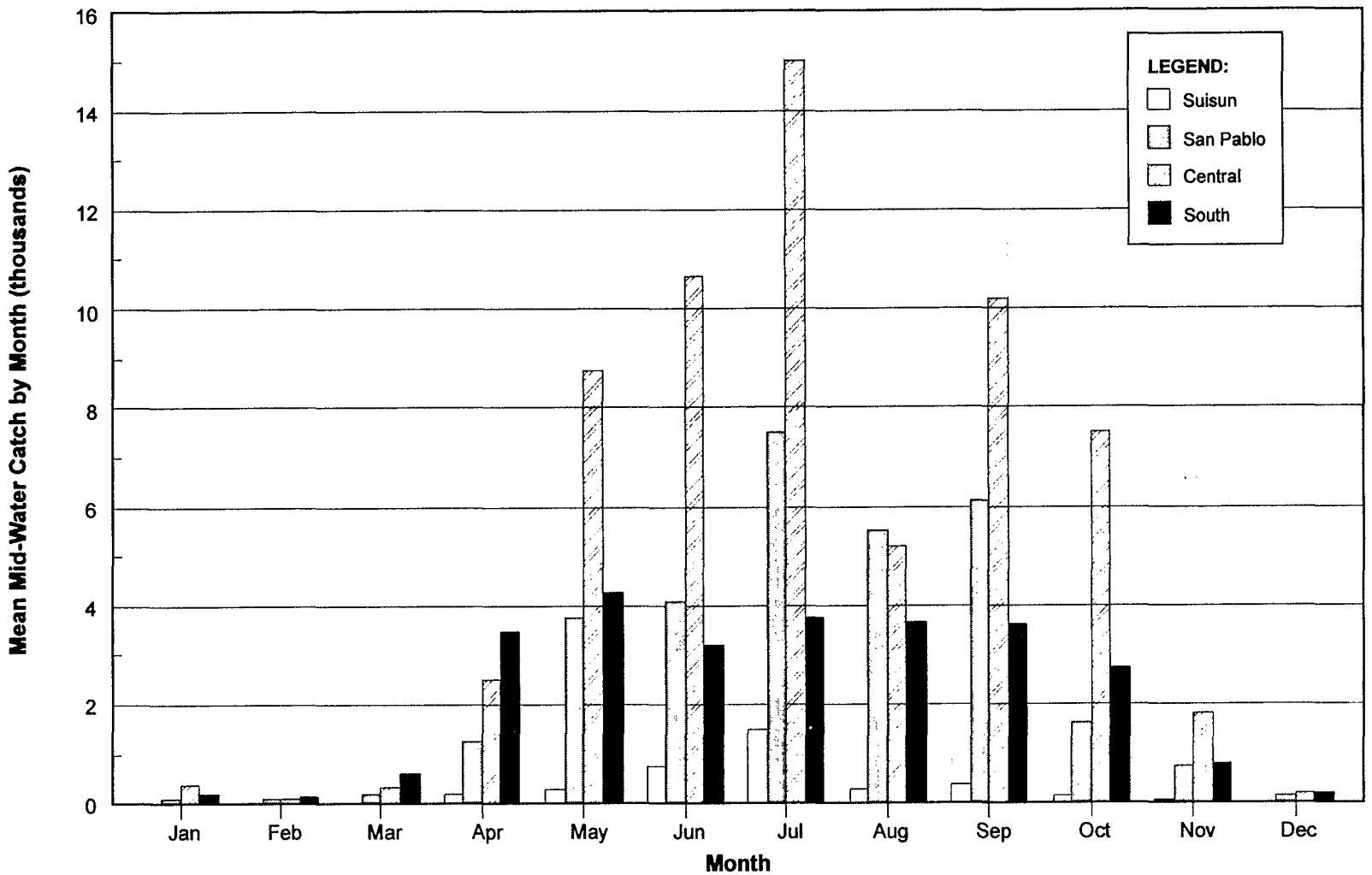

SOURCE:
Herbold et al., 1992.

FIGURE II-108

MEAN MONTHLY CATCH OF NORTHERN ANCHOVY IN MID-WATER NETS IN THE SUISUN, SAN PABLO, CENTRAL, AND SOUTH BAYS OF THE SAN FRANCISCO BAY (1980-1988)

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of Suisun and San Pablo bays, northern anchovy abundance peaks later, and they emigrate sooner than those in the Central and South bays (Herbold et al., 1992).

Pacific Herring

Pacific herring do not make extensive coastal migrations but migrate onshore to spawn and offshore to feed (Emmett et al., 1991). San Francisco and Tomales bays attract the largest spawning aggregations in California (Herbold et al., 1992). Adult Pacific herring move onshore in fall and spawn within San Francisco Bay from November through March (Herbold et al., 1992). Spawning takes place in restricted intertidal and shallow-water habitats near Tiburon Peninsula, Angel Island, Berkeley, and Richmond (Herbold et al., 1992). Spawning takes place on eelgrass, algae, tubeworms, oysters, rocks, and other substrates, with the eggs adhering to these substrates until hatching is complete (Emmett et al., 1991).

Pacific herring eggs can tolerate salinities of 3 to 33 parts per thousand. Larvae are tolerant of salinities of 2 to 28 parts per thousand. Increased turbidity could increase survival. Larval Pacific herring are selective planktivores and consume diatoms, invertebrate and fish embryos, crustacean and mollusk larvae, and zooplankton. Juvenile herring are also selective planktivores, consuming zooplankton and fish and other larvae (Emmett et al., 1991). Juvenile Pacific herring are widely distributed in shallower habitats of South, Central, and San Pablo bays. As the juveniles grow and mature, they migrate into deeper water habitats in the Central Bay and emigrate from the Bay between April and August (Herbold et al., 1992).

White Croaker

White croaker are common and abundant in San Francisco Bay, generally not abundant north of San Francisco Bay, and rare north of California (Emmett et al., 1991). This species enters the Bay through three life stages. Adult white croaker spawn in April in the Gulf of the Farallones or in the Central Bay by broadcasting eggs, which are carried to the upstream reaches of the Bay. Young-of-the-year white croaker spawned in the ocean enter the Bay in May, distributing themselves throughout the Bay. Juveniles from the previous year class also immigrate into the deep-water channel areas of the South Bay from the ocean during spring (Herbold et al., 1992). Generally, the young-of-year and juvenile fish emigrate from the Bay in fall and return the following spring. Deeper South Bay habitats also contain a year-round population of older white croaker.

The pelagic eggs of this species drift into the shallow waters of the Bay on incoming tides, and the larvae settle near the bottom after hatching. The larvae, juveniles, and adults are omnivorous and feed primarily at night near the bottom. White croaker larvae consume rotifers, copepods, and invertebrate eggs. Small juvenile white croaker primarily consume zooplankton organisms, such as cladocerans, mysids, copepods, and fish larvae. Larger juveniles and adults consume clams, crabs, worms, shrimp, and a variety of fish, including northern anchovy (Emmett et al., 1991).

Adult white croaker reach sexual maturity at one to four years, with approximately 50 percent reaching maturity at one year (Emmett et al., 1991). Species abundance in the Bay increases in response to greater immigration and recruitment from coastal populations during high Delta outflow years or in years in which adults spawn within the Bay. However, emigration from the Bay merges these juveniles into the larger population in the ocean; therefore, catches of subsequent years of juveniles or adults in the Bay are not necessarily higher (Herbold et al., 1992).

English Sole

English sole in California are most abundant from Point Conception northward, with juveniles most abundant in estuaries north of Elkton Slough (Emmett et al., 1991). English sole are bottom-foraging flatfish that spawn outside San Francisco Bay during November through May (Herbold et al., 1992). The ages at maturity are approximately two and three years old for males and females, respectively (Emmett et al., 1991). English sole eggs and larvae are pelagic and remain near the surface of the water column until the larvae are approximately 6 to 10 weeks old. At that time, they descend in the water column and are transported into the Bay by landward-flowing bottom currents (Herbold et al., 1992). The Bay acts as a nursery for English sole in that the young-of-the-year rear on the soft mud and sand substrates in the Central Bay and often move into higher salinity areas of San Pablo and South bays (Herbold et al., 1992).

English sole larvae metamorphose into juveniles in spring and early summer and rear until fall and early winter, when they move into deeper waters and emigrate from the Bay (Herbold et al., 1992; Emmett et al., 1991). Larvae of English sole are planktivorous and consume copepods and other small plankton. Small juveniles feed primarily on harpacticoid copepods and benthic crustaceans until they reach approximately 2 inches in length, at which time they begin to feed primarily on polychaete worms (Emmett et al., 1991). Adults feed primarily on benthic organisms, including worms, clams, amphipods, and crustaceans.

Starry Flounder

Starry flounder are common in estuarine areas from Morro Bay northward. Like English sole, starry flounder generally spawn outside San Francisco Bay. Unlike English sole, the starry flounder in all its life stages, including adults, could be found within the Bay (Herbold et al., 1992). Some starry flounder also are thought to spawn inside the northern reaches of the Bay (Emmett et al., 1991; Herbold et al., 1992). Spawning generally takes place from November through February (DFG, 1992), depending on location. Starry flounder eggs and larvae are pelagic, and larvae move into the Bay on currents and disperse into the upper reaches of the Suisun and San Pablo bays during May through October (Herbold et al., 1992). The younger and smaller larvae tend to be distributed farther upstream than the slightly older and larger larvae. As they grow and mature, starry flounder juveniles move into the more saline waters of San Pablo Bay.

Larval starry flounder consume phytoplankton and zooplankton. Juveniles smaller than 4 inches in length eat copepods and other small crustaceans. Larger juveniles and adults consume crustaceans, including *Crangon* spp., Dungeness crabs, worms, clams, and occasionally fish, including northern anchovy (Emmett et al., 1991). Starry flounder are most abundant and diverse

in size in San Pablo Bay, but Suisun Bay is a very important nursery area for young-of-the-year (Herbold et al., 1992).

Dungeness Crab

Dungeness (i.e., market) crabs can be found in coastal waters and most bays and estuaries from Morro Bay to Puget Sound (Emmett et al., 1991). San Francisco Bay provides a significant nursery for the Dungeness crab population of central California. Reproduction of this species takes place entirely in the ocean waters of the Gulf of the Farallones, where peak spawning occurs during October and November (Herbold et al., 1992). The mature (i.e., two- to three-year-old) female Dungeness crabs retain the developing eggs on their pleopods until January when most eggs hatch and the larvae enter the water column. The timing of egg hatch depends on water temperature. Larval development continues in coastal waters until approximately March or April, when transformation to crab-like megalopa takes place. The megalopa and early stage juveniles are transported into the mouth of San Francisco Bay during May and June, where they settle to the bottom (Herbold et al., 1992).

Juvenile Dungeness crabs are distributed in Central and South San Francisco bays and in San Pablo and Suisun bays in areas with salinities greater than approximately 10 parts per thousand (Herbold et al., 1992). Juvenile crabs prefer soft substrates containing eelgrass and oyster shell. Studies have indicated that the Bay as a nursery area permits a more rapid rate of growth and maturity than do offshore areas (Herbold et al., 1992). Juvenile crabs consume fish, clams, benthic crustaceans, and shrimp (Emmett et al., 1991). In general, crabs eat bivalve mollusks their first year, *Crangon* shrimp their second year, and fish their third year (Emmett et al., 1991). The onset of high winter outflows results in the mass movement of juvenile crabs into downstream locations with more favorable salinity conditions, primarily within San Pablo Bay. Juvenile Dungeness crabs emigrate from the Bay during August and September after approximately one year in the estuary (Herbold et al., 1992).

Caridean Shrimp

The caridean shrimp of San Francisco Bay include four commonly occurring estuarine species, which are known locally as grass shrimp. These four species are California bay shrimp (*C. franciscorum*), the largest species in the Bay; blacktail bay shrimp (*C. nigricauda*); black-spotted bay shrimp (*C. nigromaculata*); and an introduced prawn (*Palaemon macrodactylus*) (Herbold et al., 1992). Each species has its own salinity preferences and tolerances. Bay shrimp are most abundant in lower salinities, and young of this species are found in nearly total freshwater during late spring through early summer. During maturation, bay shrimp move to progressively more saline water. Blacktail shrimp prefer salinities of 25 parts per thousand or greater and, as with bay shrimp, juvenile abundance occurs in spring through early summer. Black-spotted shrimp is primarily a coastal shallow-water species preferring marine-like salinities above 30 parts per thousand, with juvenile abundance within the Bay occurring from May through November. *Palaemon* has a wide tolerance to salinity but is generally found only in the freshwaters of the upper estuary, particularly Suisun Bay and Marsh and at the mouths of creeks in San Pablo and South bays (DFG, 1992; Herbold et al., 1992).

An additional species (*Heptacarpus cristatus*) is also known from the Bay, but information on its life history is not as well known (Herbold et al., 1992). It is assumed that juvenile *Heptacarpus* prefer lower salinities than do the adults (DFG, 1992).

Adult egg-bearing females of all species migrate into higher salinity waters to release their eggs. Newly hatched larvae are planktonic, and post-larvae and juveniles migrate into lower salinity nursery areas of the estuary where they grow and mature for four to six months (DFG, 1992). Adults of all species are sexually mature at one year. *C. franciscorum* is a good indicator of changes in estuarine temperature and salinity regimes because its distribution, recruitment, growth, and survival respond to riverine discharge conditions (Emmett et al., 1991).

FACTORS LIMITING ABUNDANCE

Many factors that affect the distribution and abundance of Bay resources have been greatly affected by human activities, including, but not limited to, Delta outflow, Bay hydrodynamics, salinity gradients, nutrients, food availability, interspecific competition, and predation. The following sections briefly discuss the most salient points regarding these factors and how they affect the abundance and distribution patterns of the Bay's key resources. Many of these limiting factors are interrelated and driven by the effects of Delta outflow.

Delta Outflow

Delta outflow and its effect on San Francisco Bay biota have only recently been investigated. Almost all factors that could act to limit Bay fishery resources are influenced by the magnitude and timing of the freshwater flows from the Delta. Delta outflow is the major source of the freshwater inflow to the San Francisco Bay estuary. In an estuarine system, the total freshwater flow can be an important factor regulating biological, physical, and chemical processes (Herrgesell, 1993). Delta outflow has been significantly reduced by literally thousands of upstream river and Delta diversions and exports, including CVP and SWP diversions. CVP interbasin transfers from the Trinity River Basin Region have increased Delta inflow but not necessarily Delta outflow. Storage impoundments have also caused major changes to the timing, duration, and frequency of different types of Delta outflow events.

The effects of Delta outflow on Bay aquatic resources have been studied since 1980 by the IEP. One of the objectives of these studies is to determine how the abundance and distribution of fish, shrimp, and crabs are affected by the amount and timing of freshwater flows to the Bay (Herrgesell, 1993). To evaluate the effects of Delta outflow, annual abundance indices are calculated for several important aquatic species based on field sampling data from throughout the estuary. Annual abundance indices for 1980-1991 have been calculated for caridean shrimp, Dungeness crab, northern anchovy, Pacific herring, white croaker, English sole, starry flounder, plainfin midshipman, speckled sand dab, California halibut, Bay goby, jacksmelt, shiner perch, staghorn sculpin, and longfin smelt (Herrgesell, 1993).

The abundances of several recreational and commercially important species for 1980-1991 are shown in Table II-17 (caridean shrimp) and Figures II-109 (Dungeness crab), II-110 (white croaker), II-111 (English sole), II-112 (northern anchovy), II-113 (Pacific herring), and II-114 (starry flounder). Based on data analyzed by DFG, the relationships between Delta outflow and abundance and distribution of several of these species are shown in Table II-18.

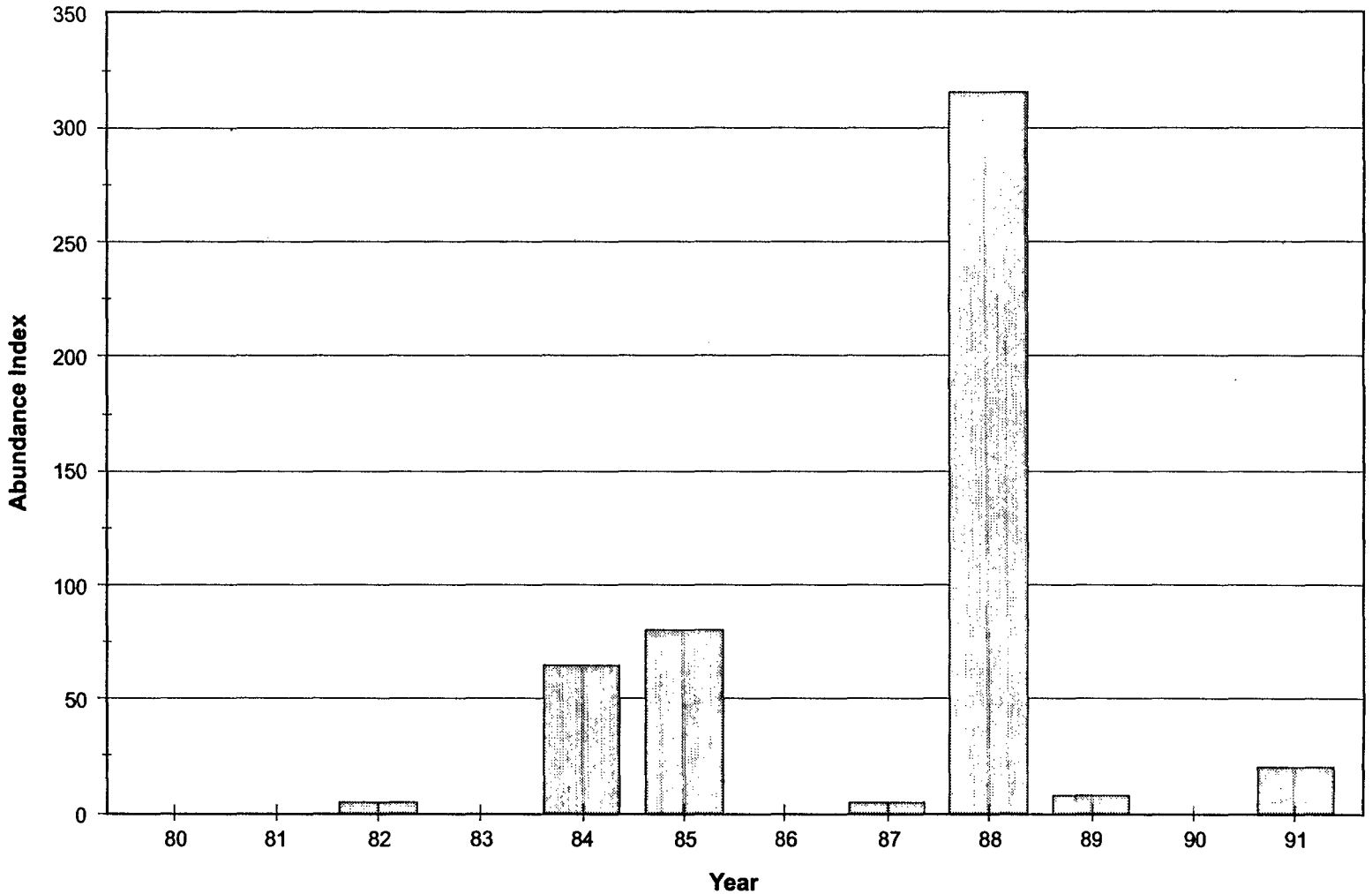
TABLE II-17
ANNUAL ABUNDANCE INDICES FOR THE SIX MOST COMMON SPECIES OF CARIDEAN SHRIMP COLLECTED IN SAN FRANCISCO BAY, 1980-1991

Year	*C. franciscorum*	*C. nigricauda*	*C. nigromaculata*	*Palaemon*	*Heptacarpus*	*Lissocrangon*	Total
1980	5,343	1,255	49	121	94	0	6,862
1981	2,587	651	15	127	36	0	3,415
1982	8,498	495	31	83	75	0	9,182
1983	9,304	1,152	363	33	80	0	10,932
1984	6,820	355	152	171	67	80	7,645
1985	1,329	505	68	91	92	22	2,108
1986	6,061	1,344	188	106	92	140	7,930
1987	3,345	2,716	300	58	328	56	6,804
1988	2,300	2,856	288	39	299	7	5,789
1989 (1)	2,859	3,350	347	137	671	3	7,367
1990 (2)	1,820	5,118	980	103	561	19	8,602
1991 (2)	1,294	5,349	1,332	150	1,448	8	9,580
All	4,438	1,937	311	99	283	30	7,099

NOTES:
(1) January to August.
(2) February to October.

Delta outflow to the Bay affects every life stage of caridean shrimp. Outflow affects the distribution of adult shrimp and transports young larval life stages downstream into the nearshore coastal areas. Outflow also affects upstream migration of late-stage larvae and post-larvae into nursery areas of the Bay and the size and location of the nursery areas, the abundance and types of predators and food organisms, and the downstream migration of the adult shrimp (Herrgesell, 1993).

A positive relationship of abundance of *Crangon franciscorum* and spring Delta outflow has been established (Herrgesell, 1993). However, the studies conducted by the IEP have not demonstrated a significant relationship, either positive or negative, for the abundance of the other caridean shrimps and Delta outflow (Herrgesell, 1993). DFG (1992) reported that the total abundance of all species of caridean shrimps was slightly greater for high outflow years (1980, 1982, 1983, and 1986) and was 1.5 times greater during the drought years of 1988-1991 than total abundances for the other low outflow years. DFG (1992) attributed this to greater abundance of the more salinity-tolerant *Crangon* and *Heptacarpus* species, particularly in the South and Central bays.

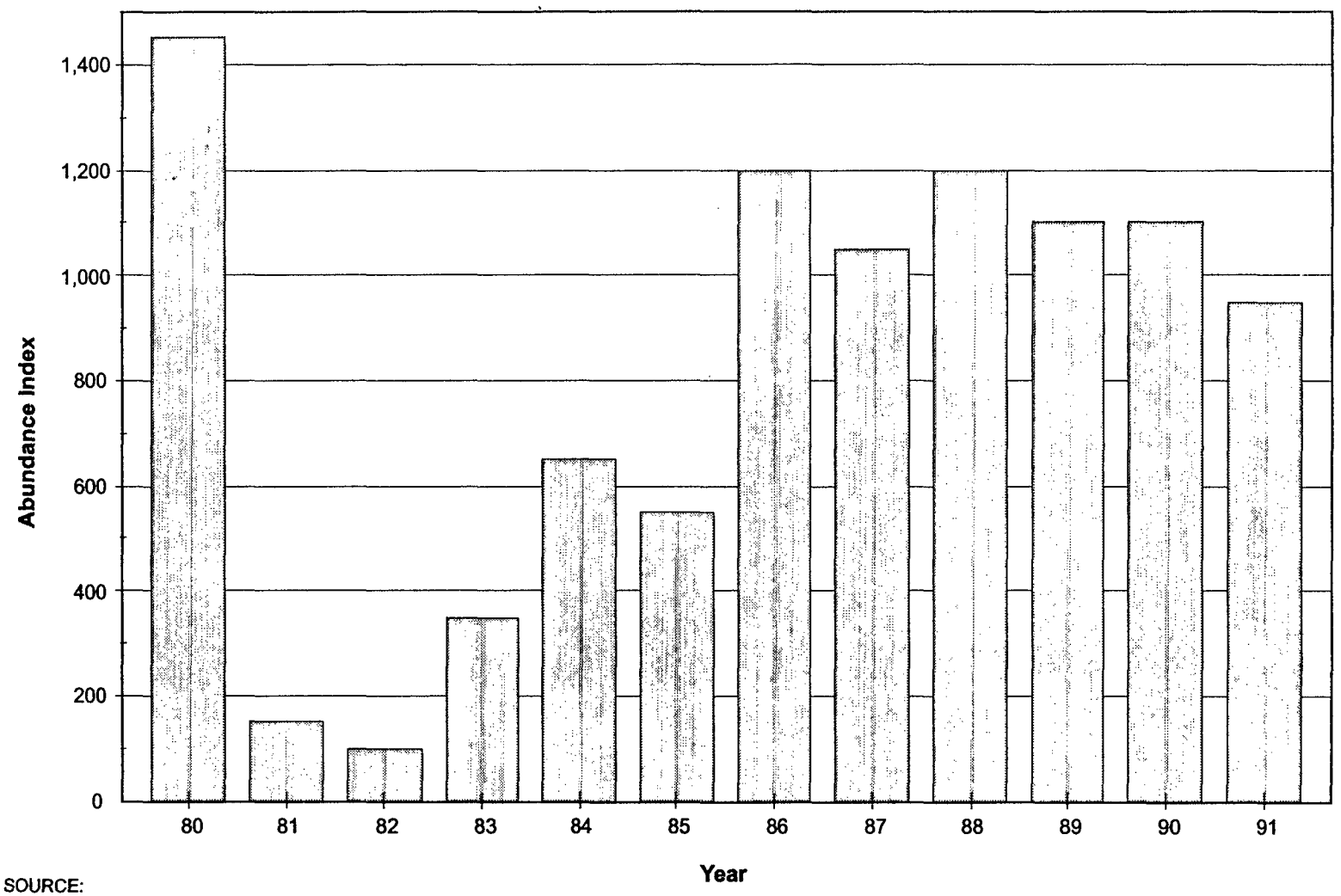


SOURCE:
Herrgesell, 1993.

FIGURE II-109

ANNUAL ABUNDANCE INDEX FOR DUNGENESS CRAB (MAY-JULY, OTTER TRAWL 1980-1991)

C-081622
C-081622



SOURCE:
Herrgesell, 1993.

FIGURE II-110

ANNUAL ABUNDANCE INDEX FOR WHITE CROAKER (ALL SIZES, FEBRUARY-OCTOBER, OTTER TRAWL 1980-1991)

C-081623
C-081623

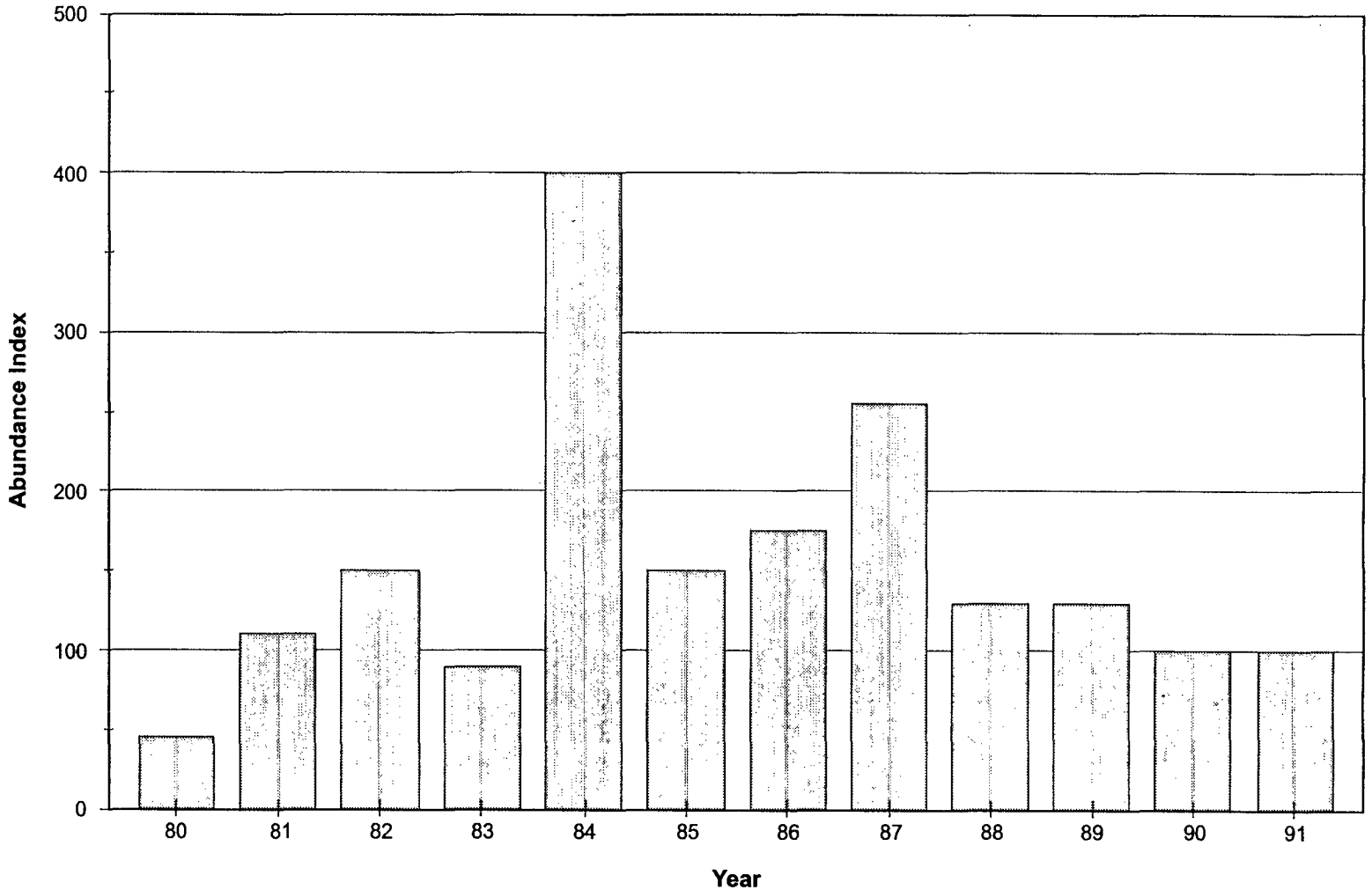
C-081624

C—0 8 1 6 2 4

SOURCE:
Herrgesell, 1993.

FIGURE II-111

ANNUAL ABUNDANCE INDEX FOR ENGLISH SOLE (FEBRUARY-OCTOBER, OTTER TRAWL 1980-1991)

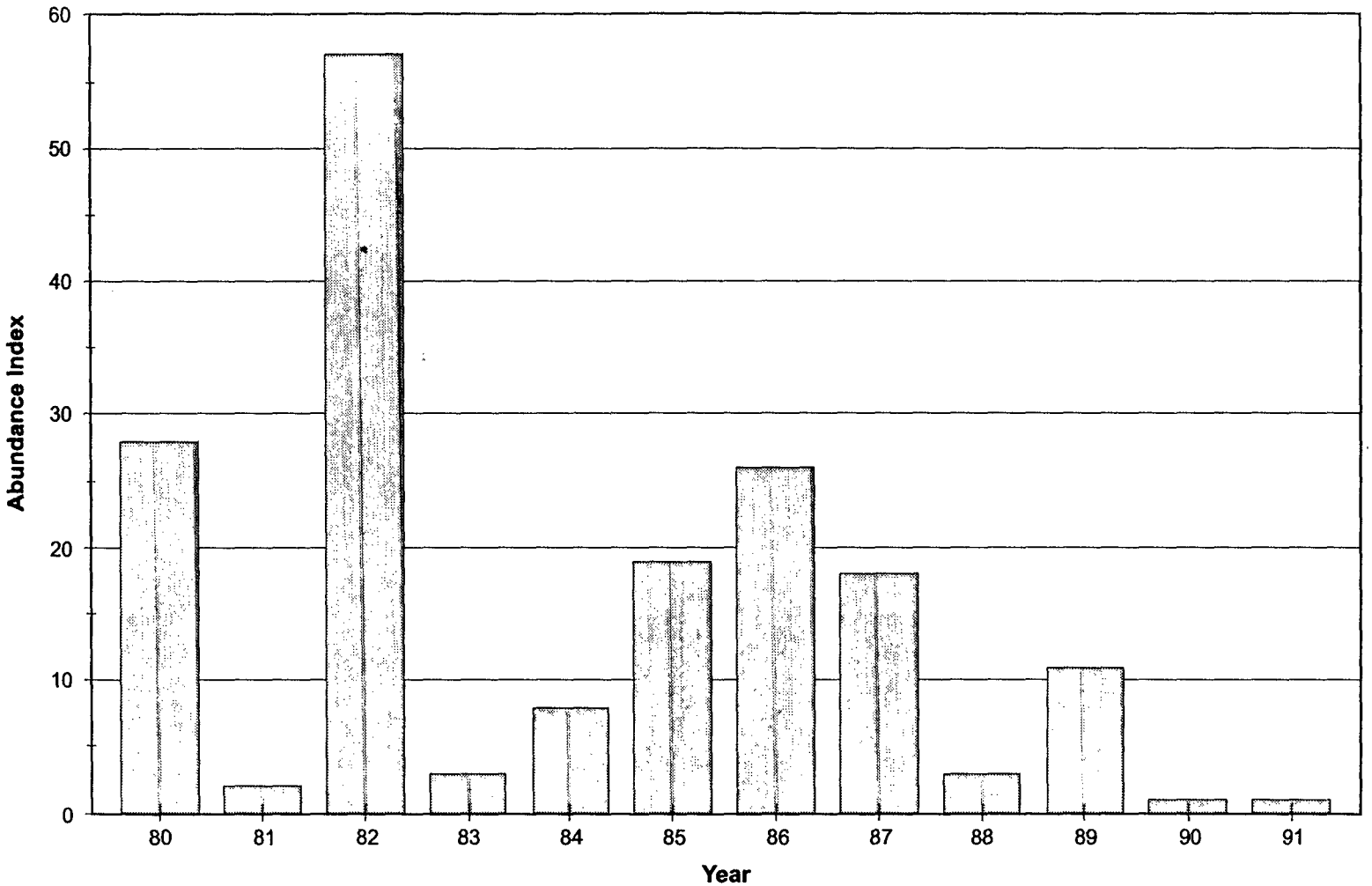


SOURCE:
Herrgesell, 1993.

FIGURE II-112

ANNUAL ABUNDANCE INDEX FOR NORTHERN ANCHOVY (ALL SIZES, APRIL-OCTOBER, MID-WATER TRAWL 1980-1991)

C-081625
C—081625



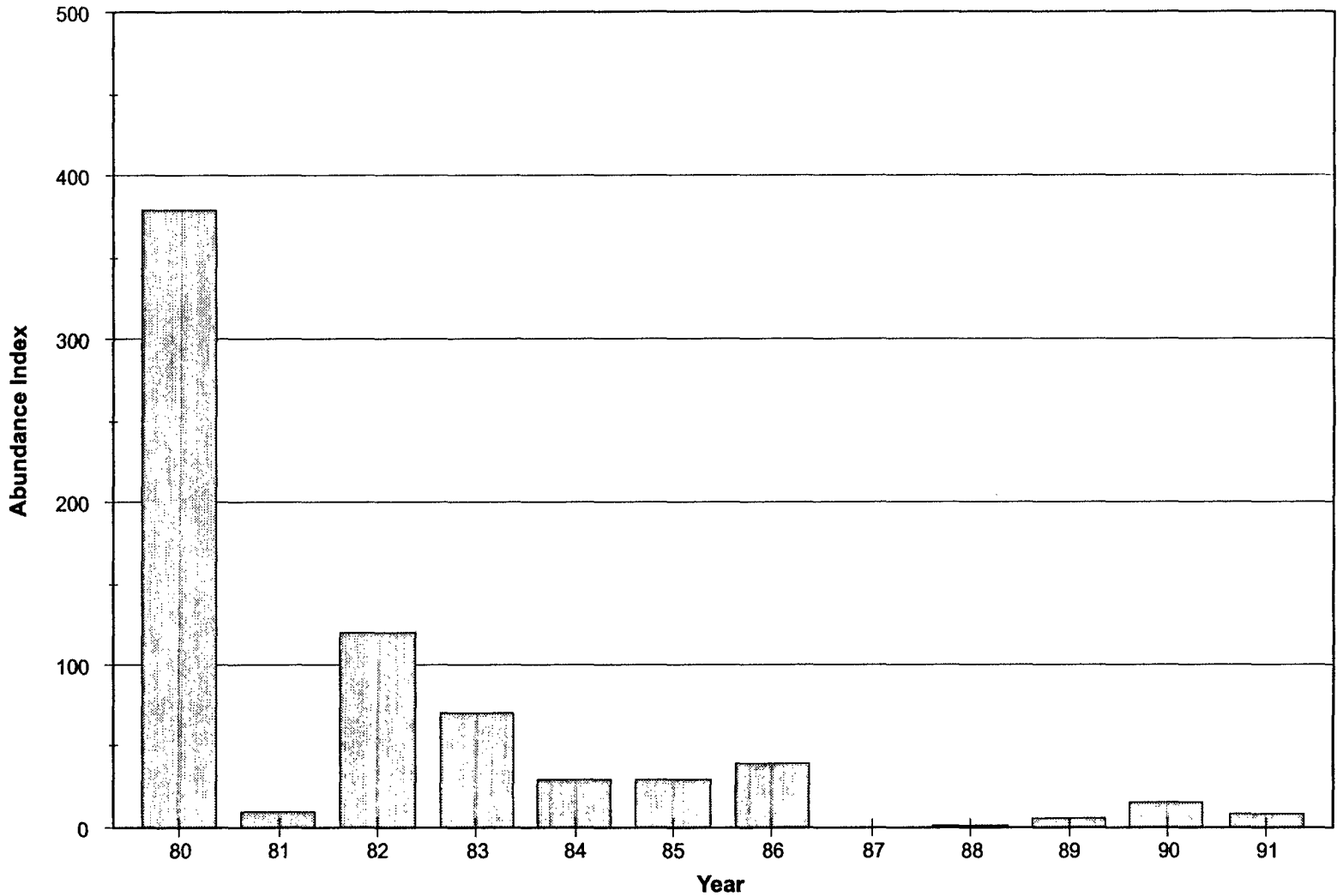
SOURCE:
Herrgesell, 1993.

FIGURE II-113

ANNUAL ABUNDANCE INDEX FOR PACIFIC HERRING (MAY-AUGUST, MID-WATER TRAWL 1980-1991)

C-081626
C-081626

C-081627



SOURCE:
Herrgesell, 1993.

FIGURE II-114

ANNUAL ABUNDANCE INDEX FOR STARRY FLOUNDER (MAY-OCTOBER, OTTER TRAWL 1980-1991)

C—081627

C-081628

TABLE II-18

RELATIONSHIPS BETWEEN DELTA OUTFLOW AND ABUNDANCE AND DISTRIBUTION OF VARIOUS LIFE STAGES OF SEVERAL RECREATIONAL AND COMMERCIALLY IMPORTANT BAY SPECIES

		Abundance Changes with Increasing Delta Outflow				Bay Distribution with Increasing Outflows			
Species	Life Stage	Winter	Spring	Summer	Fall	Expand	Decrease	Shift	None
Crangon franciscorum	Larval	-	-				X		
	Juvenile		+				X		
	Adult		+				X		
Crangon nigricauda	Larval	-	-				X		
	Juvenile		+				X		
	Adult		+				X		
Palaemon macrodactylus	Larval			-				X	
	Juvenile			-				X	
	Adult			-				X	
Crangon nigromaculata	Juvenile		+				X		
	Adult		+				X		
Dungeness crab	Juvenile		-				X		
Pacific herring	Larval								X
	Juvenile						X		
	Adult						X		
Northern anchovy	Larval								X
	Juvenile						X		
	Adult						X		
Starry flounder	Juvenile		+	+		X			
	Adult								
English sole	Larval	+				X			
	Juvenile		-				X		

SOURCE:
DFG, 1991.

LEGEND:
- = decrease in abundance
+ = increase in abundance

C-081628

The IEP studies on the relationship of the abundance of juvenile Dungeness crabs in the Bay to Delta outflow have not determined any positive or negative relationships. Similar studies of abundances of English sole, staghorn sculpin, and speckled sand dab also have failed to demonstrate either positive or negative relationships with Delta outflow (Herrgesell, 1993). Bay goby abundance consistently increased during 1980-1991, including the 1987-1991 drought years (Herrgesell, 1993). Fish species that have shown a trend of increased abundance during drought years with low Delta outflow were white croaker, plainfin midshipman, speckled sand dab, and California halibut (Herrgesell, 1993).

The abundances of other fish species monitored since 1980, including English sole and northern anchovy, were relatively stable during the drought years, and relationships between Delta outflow and abundance were not demonstrated. The relationship of low Delta outflows and low abundances of other species, including Pacific herring, shiner perch, and jacksmelt, species using higher salinity areas, is unexplainable (Herrgesell, 1993).

DFG (1992) abundance and Delta outflow models have demonstrated a strong positive relationship of starry flounder abundance with Delta outflow from March through June. However, studies indicated that for 70 species monitored within the Bay between 1980 and 1988, a majority (55.5 percent) showed no differences in abundance between wet and dry years (DFG, 1992). A majority of the species, which had no apparent difference in abundance between wet or dry years, were marine species. For estuarine, anadromous, or freshwater fish species monitored, more than 66 percent were significantly more abundant in wet years than in dry years (DFG, 1992). As a generalization, species that use the lower salinity waters of the Bay as a nursery area had low abundance during years of low freshwater outflow (Herrgesell, 1993).

Bay Hydrodynamics

The patterns of currents, tides, water residence times, and freshwater inputs act to create a complex hydrodynamic system within the Bay. These factors affect the abundances and influence the distribution of some Bay species. Circulation patterns within the Bay distribute and transport both adult and young life stages of many species. For those species that are spawned offshore and rear in the Bay, gravitational circulation could play an important role in immigration and recruitment into the Bay fishery (DFG, 1992). While tides distribute and limit the abundance of Bay species on a shorter temporal basis (i.e., daily or seasonally) than gravitational currents, Delta outflows and effects on these gravitational currents can be directly responsible for yearly differences in species distributions and abundances. If tidal transport were the only mechanism for larval fish and invertebrate distributions within the Bay, these distributions will be similar from year to year, like tidal patterns (DFG, 1992). The abundance indices and species distribution studies indicate that there are significant trends in the yearly distributions and abundances for several Bay species. These trends could be a result of currents and tidal factors, which act to limit distribution, and thus affect abundance.

The velocity, duration, and geographical extent of the landward-flowing bottom current, responsible for transporting eggs and larvae of offshore spawning species like English sole, starry flounder, and Dungeness crab greatly affect the success of year classes of those species. The extent to which these life stages are transported into the upper reaches of Suisun and San Pablo

bays could determine their survival, growth rates, and their eventual contribution to subsequent year classes.

Tidal flow, along with freshwater outflows from the Delta, affects the location of the entrapment zone in the upper estuary. The position of this entrapment zone is critical to productivity and food availability for many organisms in upper Suisun Bay and the western Delta (DFG, 1987; Herbold et al., 1992).

Salinity

Salinity can either directly or indirectly influence species distribution and could affect species abundance in the Bay. When salinity changes, larger fish and mobile invertebrates often respond by moving to another portion of the estuary where salinity is more favorable (DFG, 1992). Thus, salinity preference plays a role in determining the distribution of most organisms in the Bay. Species distributions are determined by behavioral responses to salinity gradients to meet their physiological and ecological requirements. Species will relocate, if they are capable, to salinities that are physiologically compatible with their needs. Species within the Bay estuary are either freshwater, estuarine, or marine. Each of these types of fish and invertebrates species has its own preference and tolerance range to salinity.

Long-term salinity changes in a particular habitat can directly affect species abundance by displacing that species into habitats that are more suitable physiologically but are less suitable ecologically for their survival, competition, optimal growth, and reproductive success. This displacement also removes the emigrating species as potential food for those more salinity-tolerant species not forced to relocate.

Salinity stratification can increase algal productivity in layers of water overlying benthic communities that cannot benefit from the resulting zooplankton abundances (DFG, 1992). The net result of this salinity stratification will be a change in food availability to epibenthic- and benthic-dependent organisms.

Generally, long-term salinity increases caused by reduced Delta outflow could redistribute marine species into areas formerly inhabited by estuarine species. While distributions of marine species could change with increased salinity, studies on the abundance and distribution of marine species in the Bay indicate that abundance of marine species did not increase with salinity increases (DFG, 1992). Conversely, long-term salinity decreases could redistribute estuarine species into areas formerly occupied by marine species. Studies conducted by DFG (1992) were only partially successful in confirming this hypothesis. The greatest effect of long-term salinity on species distribution and perhaps abundance will be in the upper reaches of Suisun Bay, where estuarine species could potentially displace freshwater species from their habitats.

Nutrient Levels

Contributions of organic material from Delta discharge depend on the discharge volume and on the riverine concentrations of organic materials (Herbold et al., 1992). Reduced freshwater inflow into the Bay could limit nutrient inputs into Suisun Bay and the upper reaches of the Bay. The immediate result of nutrient limitations will be an overall reduction in Bay fertility.

Flow-regulated nutrient levels can cause significant changes in the organism abundance in estuaries (DFG, 1992). Primary productivity will be reduced by limiting nutrients into the estuary. Furthermore, reduced primary productivity will limit consumer productivity (e.g., zooplankton, shrimp, fish, and benthic organisms). Thus, the entire food web will become affected by productivity changes caused by decreased nutrient input into the Bay.

Food Availability

Food availability for Bay fishery resources is related to many factors, including Delta outflow, nutrient inputs, primary and consumer productivity, and estuarine hydrodynamics. Many biologists suspect that food limitations could have played a role in declining fish populations in the estuary, particularly striped bass populations (Chadwick, 1993). Chadwick (1993) states that certain elements of the zooplankton community are less abundant, which could have contributed to the observed decline in certain fisheries. Studies of the abundance of certain zooplankton species during 1972-1988 indicated that most native zooplankton taxa had decreased (Herbold et al., 1992). Larval and young-of-the-year life stages of Bay fish and invertebrates can be highly dependent on zooplankton distribution and abundance. Studies have shown that native species composition and abundance have changed in the upper estuary from the introduction of new species (i.e., the Asiatic clam, *P. amurensis*) (Herrgesell, 1993). These studies reveal that 12 of 20 species of zooplankton have declined in abundance, 7 are unchanged in abundance, and 1 has increased in abundance.

Study results of organic carbon sources for the Bay's food web indicate that a decline of total fish production in the Bay, particular the northern reach, could have accompanied the decline in organic carbon sources since the 1980s, but empirical generalizations could not be made. Those studies did indicate, however, that relatively more of the organic carbon supply could be shunted through the benthic community during drought conditions, thereby favoring demersal fish species (Herbold et al., 1992).

Interspecific Competition

Competition between species can limit fish and invertebrate abundance. If spatial distributions of certain species are reduced and ranges overlap, competition for food, spawning habitat, or preferred cover habitats could increase. This increase in competition could reduce abundance and distribution of less competitive species. DFG (1992) found that outflows or pulse flows could increase interspecific competition due to restrictions in the areal distributions of more competitive species.

Predation

As discussed for interspecific competition, the effects of restricting the distributions of certain species could increase predation rates by highly predatory species and thereby decrease prey species abundance. The same factors of decreased Delta outflow, increased salinity, reduced nutrient input, decreased productivity, and decreased food availability could create a number of temporary or long-term shifts in food web dynamics and change natural predator-prey relationships.

ONGOING MONITORING, ENHANCEMENT, AND HABITAT IMPROVEMENT ACTIONS

The most significant ongoing monitoring activities relating to Bay fisheries are elements of the IEP. Ongoing research within the Delta Outflow/San Francisco Bay Study constitutes a significant species monitoring activity within San Francisco Bay. Funding for this program element is shared by DWR and Reclamation. The objectives of the program element are to determine how species abundance and distribution of fish, shrimp, and crabs are affected by changes in the amount and timing of Delta outflow to the Bay; identify the outflow-related mechanisms underlying any significant relationships between Delta outflows and abundance or distribution; and quantify impacts of water development on these mechanisms (Herrgesell, 1993).

Field sampling, data collection, and data analyses to accomplish these goals are ongoing. These data are used to continually update yearly species abundance indices and distribution patterns. The level of effort in data collection diminished from year-round in 1980-1989 to February through October since that time. The activities include calculating annual abundance indices for Dungeness crab, 6 species of caridean shrimp, and 13 species of pelagic and demersal fish, including estuarine and marine species. All of the species use the Bay as a nursery, although some could use the coastal waters for juvenile rearing. Data analysis of relationships between Delta outflow and abundance is continuing (Herrgesell, 1993).

Additional elements of the IEP that have direct links to Bay fisheries are several of the environmental factors tasks within the water quality/fisheries element. Within this element, ongoing studies relating to Bay fisheries include phytoplankton monitoring and productivity estimation; zooplankton species composition, abundance, and distribution monitoring and analyses; aquatic vegetation distribution and abundance studies; and food chain information analyses. While these studies and tasks are primarily focused on the Delta portion of the estuary, study results also likely will be applicable, in many cases, to understanding the physical, chemical, and biological factors affecting Bay species. The long-term monitoring and data analyses of these tasks provide vital information for management decisions affecting the Bay fishery.

RECENT CONDITIONS FOR THE ESTUARINE INVERTEBRATE COMMUNITY

GENERAL ABUNDANCE AND DISTRIBUTION

Historical Population Trends (1850-1966)

Native invertebrate communities within the San Francisco Bay, Suisun Bay, and Sacramento-San Joaquin Delta have been affected by introduced species since the 1850s (Chadwick, 1993). In the last century alone, more than 100 species of non-native aquatic invertebrate species have been introduced into the estuary (DFG, 1993). Many, if not all, components of the aquatic communities within the estuary have been disturbed by these introductions. Impacts of water development, pollution, and land reclamation and dredging have seriously affected the abundance and distribution patterns of aquatic invertebrates in the Delta and Suisun Bay. The trend over this period has been an increase in the diversity of the aquatic invertebrate community

within the estuary, with a concurrent and dramatic decline in abundance and changes in distribution patterns of many native invertebrate species.

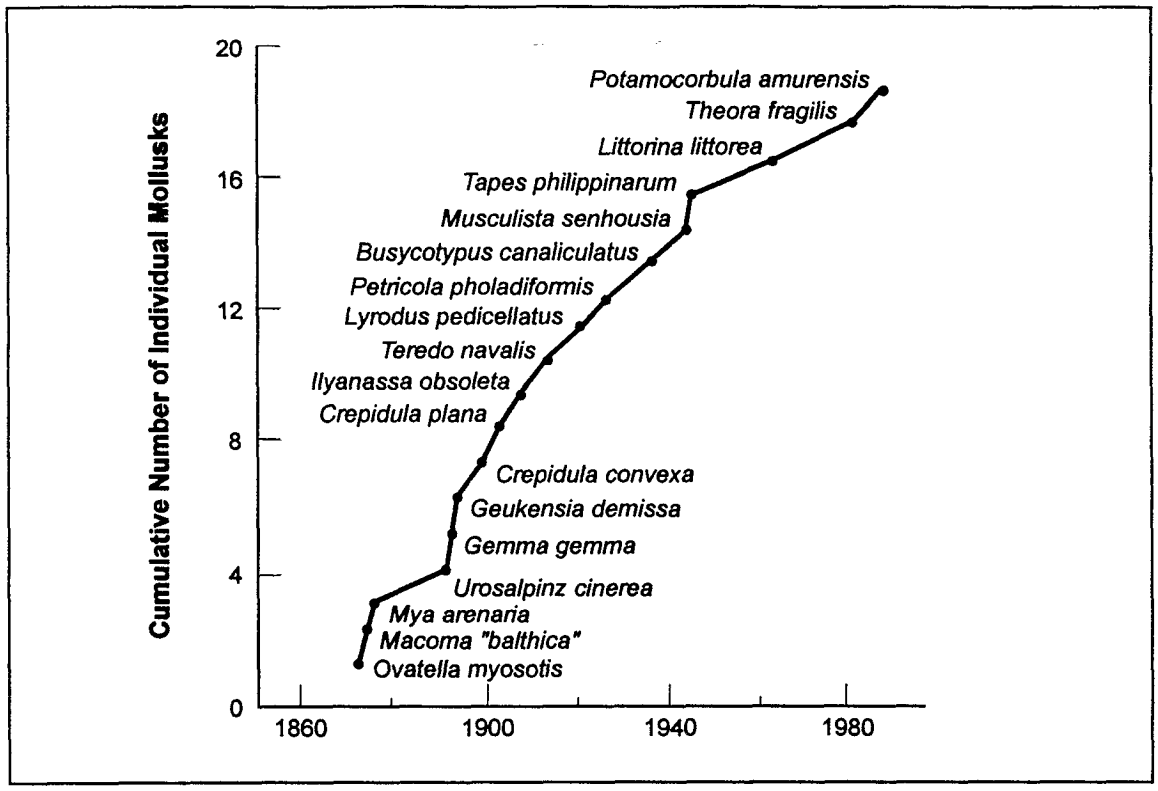
The rate of change in the estuary's aquatic invertebrate community structure has accelerated following the recent introduction of several species from Asia and elsewhere. An example of such introductions of non-native species into the estuary is shown in Figure II-115. Although many of these species are found in the higher salinity reaches of the San Francisco Bay only, Figure II-115 demonstrates the large number of exotic species introduced into the benthic community, just one faunal component of the estuary.

Current Population Trends (1967-1991)

Delta. Rotifer populations sharply declined throughout the Delta during the 1970s (DFG, 1987; Orsi et al., 1991). Densities of the most abundant rotifer species during 1972-1988 for the Sacramento River, San Joaquin River, and Suisun Bay are shown in Figure II-116. The decline of rotifers in the Delta has been associated with declining concentrations of chlorophyll *a*, a measure of phytoplankton productivity (DFG, 1987). The most abundant planktonic rotifer genera in the Sacramento and San Joaquin rivers prior to the 1970s was *Keratella* spp., with *Polyarthra* and *Trichocerca* spp. also common but less abundant (Herbold et al., 1992). The least abundant of the major species in the Delta, *Synchaeta*, did not decline precipitously as did these other rotifer genera. Since 1979, there have been no consistent differences in rotifer abundances between the Delta and Suisun Bay (Herbold et al., 1992).

Many populations of native copepod species have undergone severe declines in abundance since 1972 (DFG, 1987). The sharpest of these declines has been in the freshwater portion of the Delta, particularly in the Sacramento and San Joaquin rivers (Herbold et al., 1992). Figure II-117 shows mean densities of the four most abundant species of copepods in the Sacramento River, San Joaquin River, and Suisun Bay, respectively. The abundance of several copepod species has been at very low levels since the late 1970s. The abundance of *Cyclops vernalis,* which by the late 1970s had become very rare in the Sacramento River, has declined significantly. The abundance of other cyclopoid copepods in the San Joaquin River also diminished during this time. *Diaptomus* spp. abundance declined during the same time as did *Cyclops* but did not reach the same low levels of abundance until after 1981. These species are currently at very low levels of abundance.

Eurytemora populations in both rivers continued to decline throughout 1972-1988 (Figure II-117). *Eurytemora* abundances in the Sacramento and San Joaquin rivers have declined since 1978 to average densities of less than 500 per liter from more than 1,000 per liter in the early 1970s (Herbold et al., 1992). At the same time that the native *Eurytemora, Cyclops*, and *Diaptomus* species were declining in the Delta, the abundances of introduced copepod species were increasing dramatically. Because of the presence of introduced species in the Sacramento and San Joaquin rivers, the total abundance of copepods is still high in most years (Figure II-118).



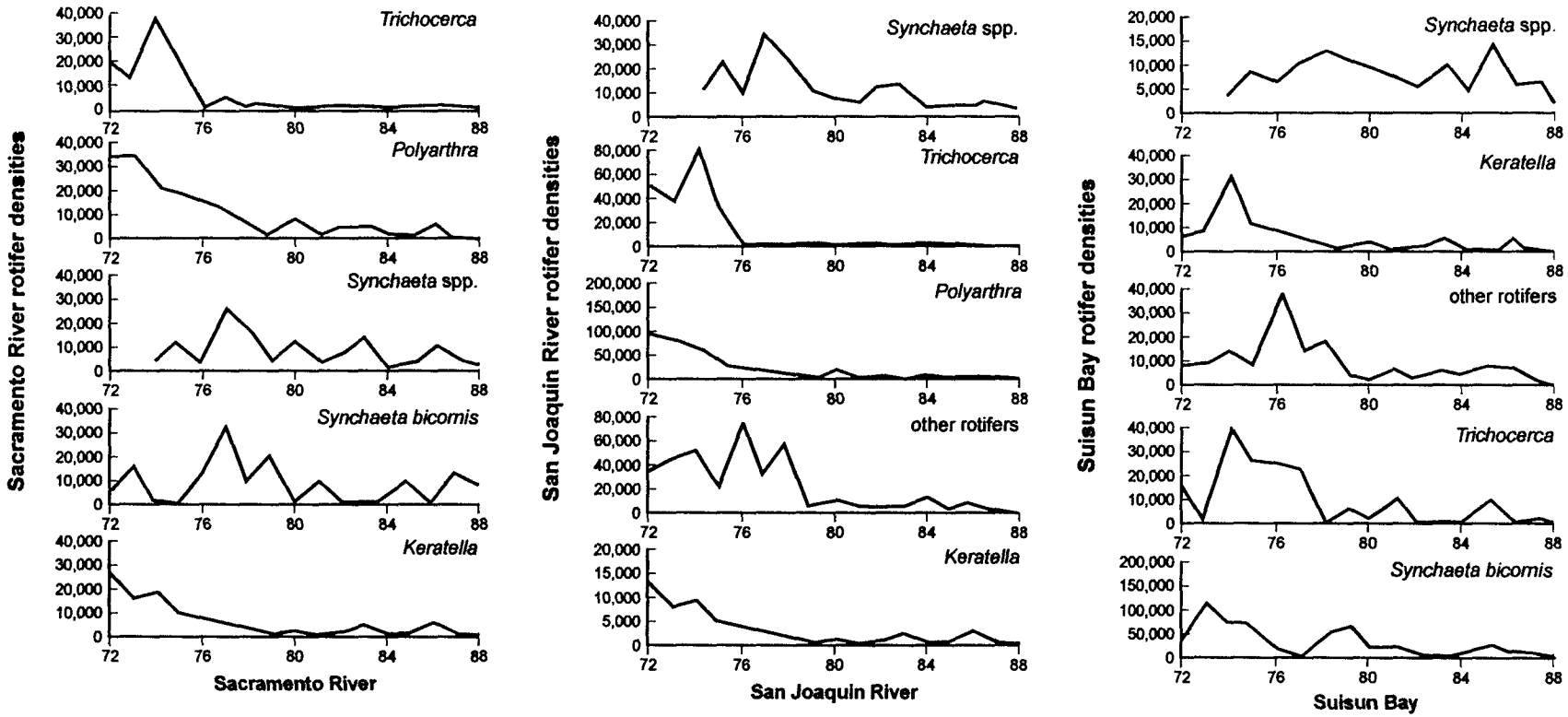
SOURCE:
Hymanson, 1991.

FIGURE II-115

INTRODUCED MOLLUSKS IN SAN FRANCISCO BAY

C—081634

C-081634



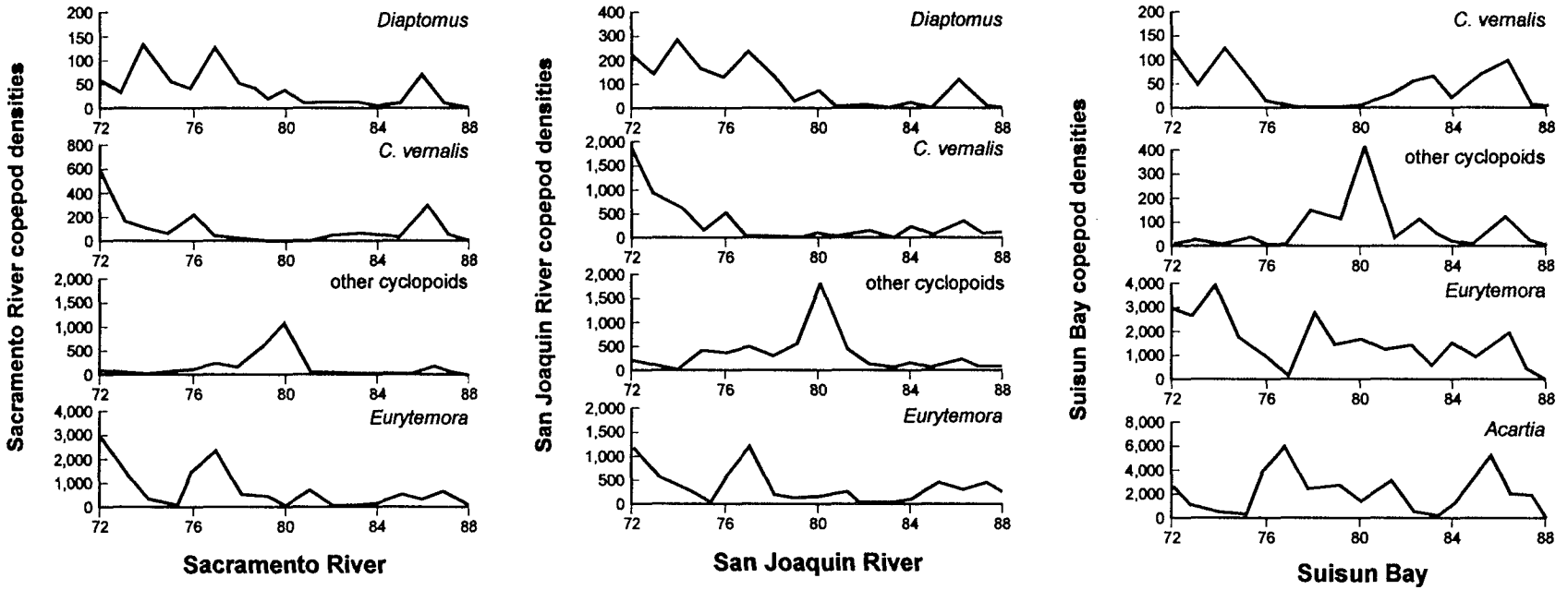
NOTE:
Data provided by DFG.

SOURCE:
Herbold et al., 1992.

FIGURE II-116

MEAN DENSITIES PER CUBIC METER OF THE ABUNDANT SPECIES OF ROTIFERS BY YEAR IN THE SACRAMENTO RIVER, SAN JOAQUIN RIVER, AND SUISUN BAY

C-081635
C—081635



NOTE:
Data provided by DFG.

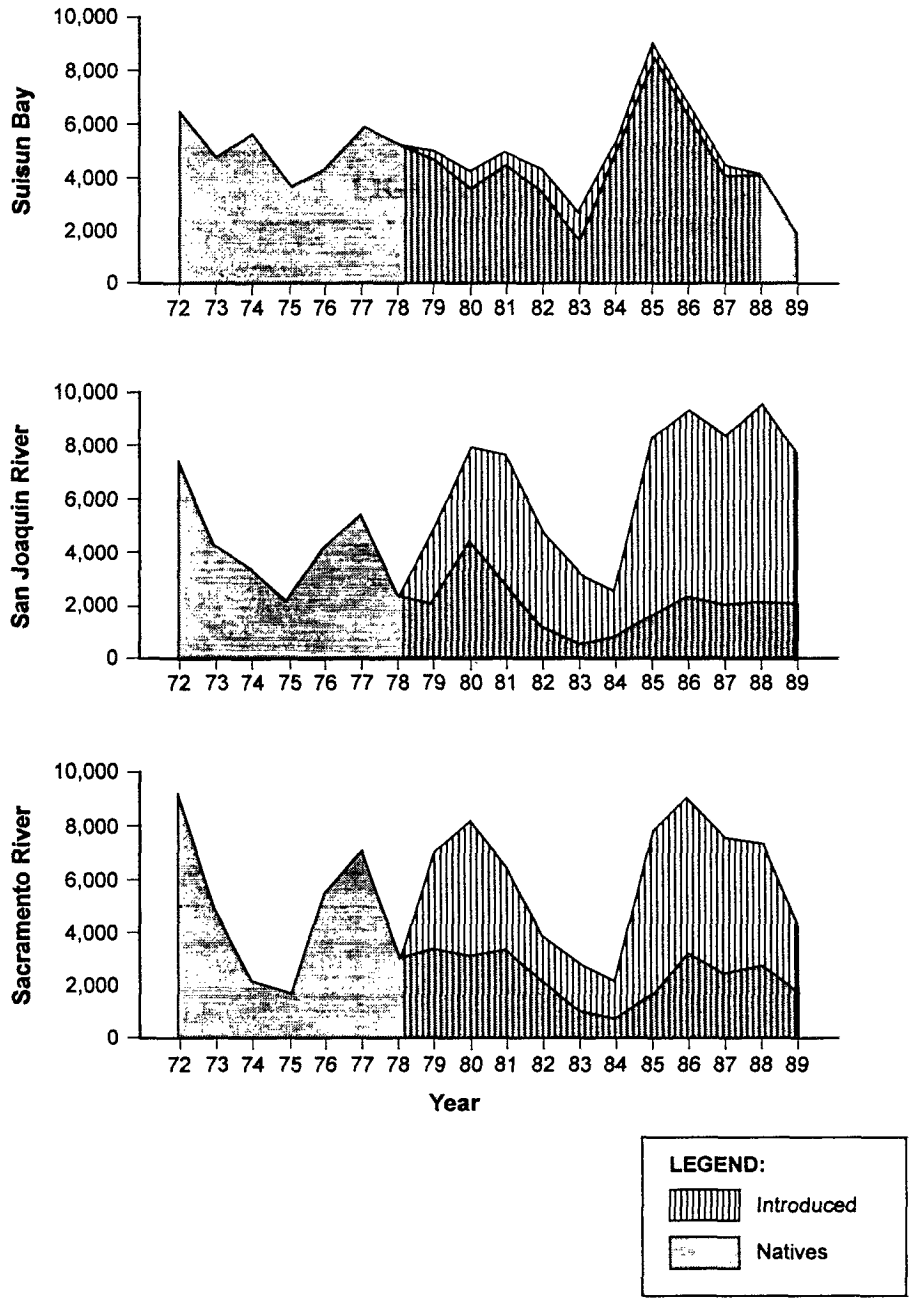
SOURCE:
Herbold et al., 1992.

FIGURE II-117

MEAN DENSITIES OF THE FOUR MOST ABUNDANT SPECIES OF COPEPODS IN THE SACRAMENTO RIVER, SAN JOAQUIN RIVER, AND SUISUN BAY (no. per cubic meter)

C-081636

C-081636



NOTE:
Data provided.by DFG.

SOURCE:
Herbold et al., 1992.

FIGURE II-118

COMPARISON OF DENSITIES (MEAN NUMBER PER CUBIC METER) OF NATIVE AND INTRODUCED COPEPODS IN THE SACRAMENTO RIVER, SAN JOAQUIN RIVER, AND SUISUN BAY

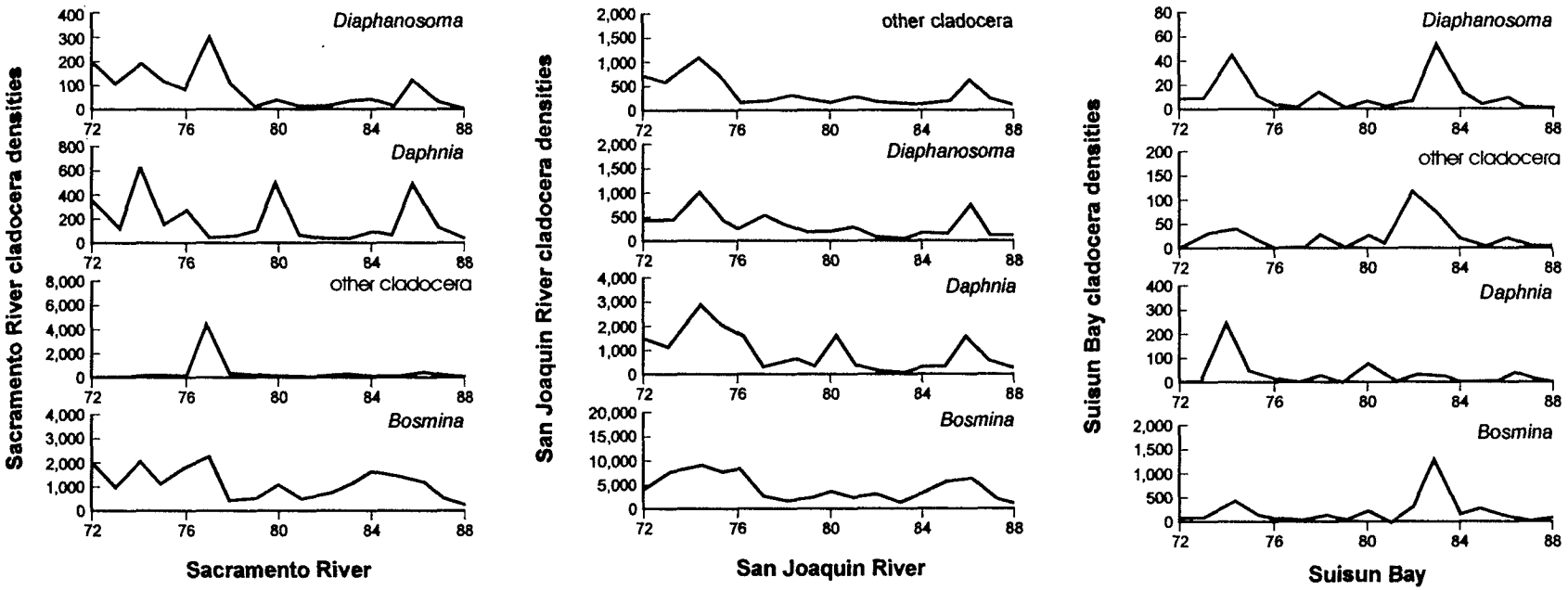
Populations of the cladoceran *Diaphanosoma* declined significantly in the Sacramento and lower San Joaquin river locations during 1972-1988; summer and fall *Daphnia* populations declined in the lower San Joaquin River and the western Delta during the same period (Obrebski et al., 1992). However, *Daphnia* and *Diaphanosoma* abundance did not decline within the entrapment zone (Obrebski et al., 1992). *Bosmina* population abundances did not show any long-term trends for any location within the Delta, but significant negative correlations between *Bosmina* abundance and Delta outflow were found for the period 1972-1990 (Orsi, 1992). Densities of the most abundant cladoceran species in the Sacramento and San Joaquin rivers during 1972-1988 are shown in Figure II-119.

Low Delta outflow, high salinity at Chipps Island, low chlorophyll *a* concentrations in Suisun Bay, and low densities of *Eurytemora* have been shown to be associated with low abundance of the mysid shrimp, *Neomysis mercedis* (Herbold, et al., 1992; Kimmerer, 1992). These factors were unfavorable for mysid populations during 1972-1988 (Herbold et al., 1992). *N. mercedis* populations declined during fall, 1972-1990 (Obrebski et al., 1992). During summer and fall, abundances of *N. mercedis* in the entrapment zone have generally declined (Obrebski et al., 1992). However, mysid populations have not shown consistent declines in abundance as many other zooplankton species have during the same period (Herbold et al., 1992). The timing of peak abundances of mysids has changed since 1974. The peak seasonal abundance of this species occurred earlier and has been followed by a rapid population decline through the summer in the years since 1974.

Little trend information is available on the abundance and distribution of gammarid amphipods in the Delta. Studies of the abundance and distribution of *Corophium* spp. in the Delta during 1975-1981 found that during the dry years of 1976 and 1977, peak population densities occurred in spring; in the wet year, 1978, peak population densities also occurred in fall (Markmann, 1986). Other research has shown that these amphipod species are limited by saltwater intrusion (Hazel and Kelley, 1966; Siegfried et al., 1978, as cited by Markmann, 1986). The recently introduced amphipod species *Gammarus daiberi* is now a food organism for larval striped bass (Chadwick, 1993). The common use of this species by striped bass indicates that the abundance of *G. daiberi* could be increasing.

Suisun Bay. Declines in rotifer abundance in Suisun Bay, where rotifers are generally less abundant than in the Delta, have been less severe (Herbold et al., 1992). Densities of the most abundant rotifer species in Suisun Bay are shown in Figure II-116. The most abundant rotifer in Suisun Bay, *Synchaeta*, did not decline as precipitously in the 1970s as did many other Suisun Bay rotifer species. However, populations of this species declined to low levels by the late 1980s. Record low abundance was recorded in 1988 and coincided with the establishment of the Asiatic clam (*Potamocorbula amurensis*) in Suisun Bay (Figure II-115) (Herbold et al., 1992).

The abundances of most native copepod species have also declined to record low levels in Suisun Bay since the introduction of the Asiatic clam (Figure II-117). Within Suisun Bay, *Eurytemora* has shown a consistent pattern of decline through time (Herbold et al., 1992). The decline of *Eurytemora* and other species in Suisun Bay, however, is not as severe as that in the Sacramento and San Joaquin river portions of the Delta. Until recently, *Acartia* abundance had increased during years of low Delta outflow (Figure II-117).



NOTE:
Data provided by DFG.

SOURCE:
Herbold et al., 1992.

FIGURE II-119

MEAN DENSITIES OF THE THREE MOST ABUNDANT SPECIES OF CLADOCERANS IN THE SACRAMENTO RIVER, SAN JOAQUIN RIVER, AND SUISUN BAY (no. per cubic meter)

C-081639
C-081639

Patterns of Delta outflow and cladoceran abundance and distributions show a positive relationship with abundances in Suisun Bay (Herbold et al., 1992). Sustained high Delta outflows during 1983 produced peak abundances of cladocerans in Suisun Bay. Moderate outflows during 1986 had little effect on cladoceran abundance. Densities of the most abundant cladoceran species in Suisun Bay during 1972-1988 are shown in Figure II-119.

During years of intermediate Delta outflow, abundance of mysids (*N. mercedis)* in the western Delta and Suisun Bay was greater than during either low or high outflows. However, recent drought conditions and low chlorophyll *a* concentrations have contributed to lower populations of mysids in Suisun Bay. The positioning of the entrapment zone in Suisun Bay has been shown to be an important factor affecting the abundance of *N. mercedis* (Herbold et al., 1992).

Following its introduction into the estuary in 1986, the Asiatic clam has increased tremendously in abundance and distribution. This species has been linked with the dramatic reduction of phytoplankton productivity and possibly zooplankton abundance in Suisun Bay (Herrgesell, 1993). The biomass of *P. amurensis* as compared to other mollusk species in Grizzly Bay, located in northern Suisun Bay, is shown in Figure II-120. Since 1986, the relative biomass of *P. amurensis* has risen tremendously. The abundance and patterns of distribution of *P. amurensis* could have significant ecological implications for both the benthic and pelagic communities of the western Delta and Suisun Bay (Hymanson, 1991).

LIFE HISTORIES

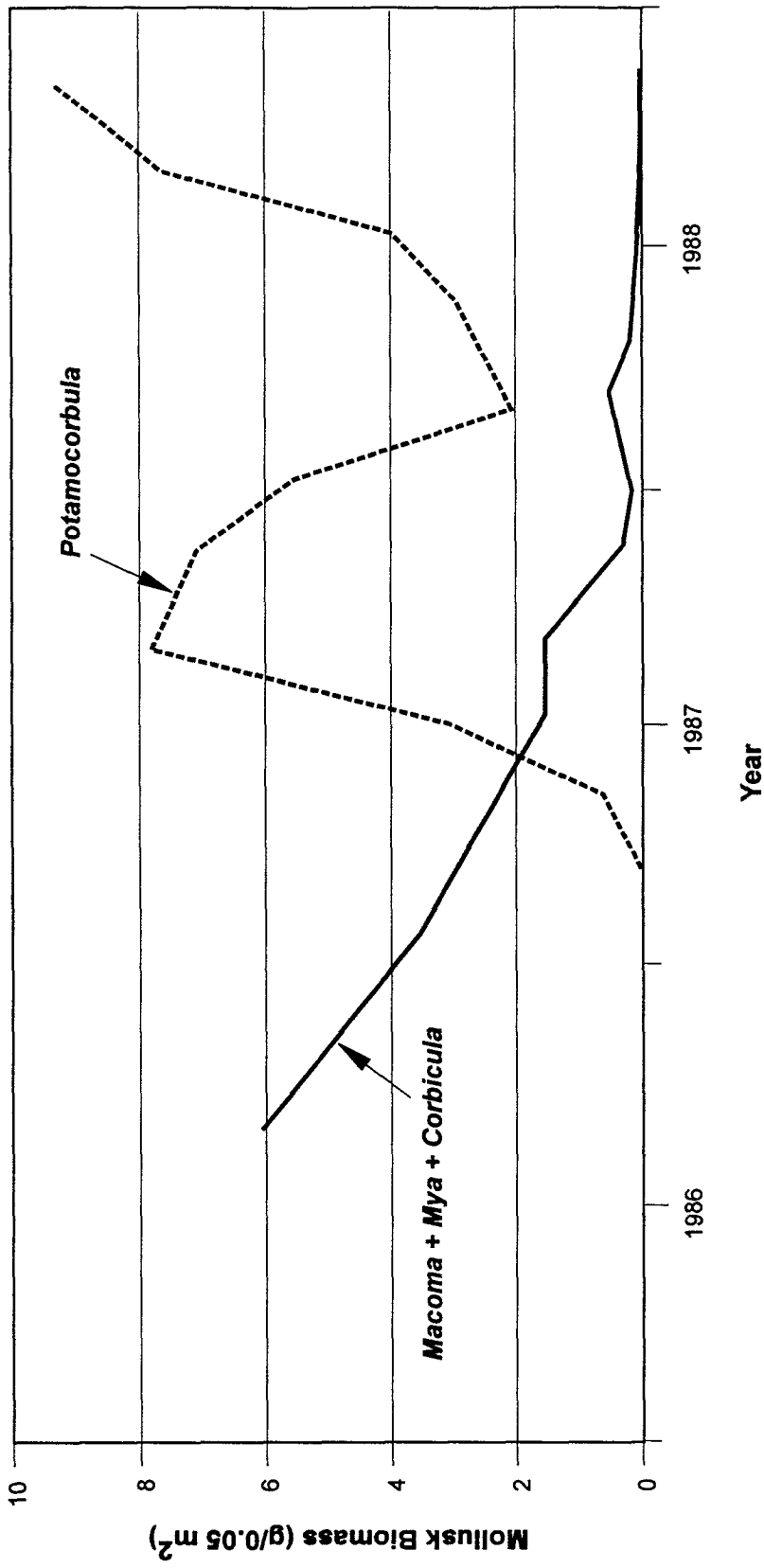
Rotifers

Rotifers (phylum *Rotiferia*) are a common and diverse group of freshwater and marine organisms called wheeled animalcules. Their name is derived from the presence of a ciliated crown, which when eating gives the appearance of a rotating wheel (Barnes, 1974). These organisms use cilia to direct food into their mouth and for locomotion (Herbold et al., 1992). There are both omnivorous rotifers, which consume both living and dead particulate organic matter, and predatory rotifers, which prey on protozoans and other zooplankton. Rotifers are microscopic and can be sessile (attached directly by its base) or planktonic. Planktonic species of rotifers constitute a significant portion of freshwater zooplankton communities of the Delta (Herbold et al., 1992). Reproduction is generally parthenogenic, with asexual reproduction accomplished by females only. Some species have the means for sexual reproduction involving short-lived males.

The dominant genera in the Bay-Delta estuary are *Synchaeta, Keratella,* and *Brachionus* (Herbold et al., 1992). Rotifers in the freshwater portions of the Delta undergo seasonal changes in abundance because of salinity fluctuations. The dominant freshwater rotifer genus *Keratella* is generally abundant in the western Delta only in spring during low-salinity conditions. In the eastern Delta beyond saltwater intrusion, the rotifer assemblage consists of at least eight planktonic genera that are herbivorous and omnivorous and one predatory rotifer genus (*Asplanchna*) (Herbold et al., 1992). Many additional rotifers are benthic and sessile and are abundant throughout the Delta.

C-081641

C—081641



SOURCE:
DFG, 1993.

FIGURE II-120
BIOMASS OF *POTAMOCORBULA* AND OTHER MOLLUSKS IN GRIZZLY BAY, 1986-1988

Copepods

Copepods are small crustaceans (many of which are planktonic); they feed on phytoplankton and form an important link in many marine and freshwater food webs (Barnes, 1974). Copepods are considered the most abundant life form on earth (Herrgesell, 1993). These zooplankton are the primary food for many small fish and other organisms in the estuary. Copepods reach their greatest diversity in marine environments, but many species are found in freshwater habitats as well. Two planktonic orders of copepods, Calanoida and Cyclopoida, are abundant in the Bay-Delta estuary. Cyclopoids are generally found in Suisun Bay to San Francisco Bay, and calanoids are found throughout the Delta and in the higher salinity waters below Chipps Island (Herbold et al., 1992; White, pers. comm.). Cyclopoid copepods move in a jerky motion within the water column by using their flattened abdominal appendages and the first pair of antennae. Calanoid copepods swim in a smooth manner propelled by movements of their mouthparts with an occasional jerky motion by use of their legs, antennae, and or mouth parts (Herbold et al., 1992).

All copepods in the estuary are sexual and cannot reproduce parthenogenetically like many other members of the zooplankton community. Development, incubation, and maturation are rapid for copepods, with one to two weeks required to reach sexual maturity for most species. After egg hatch, copepods undergo development through a series of nauplii and larval stages. Declining temperatures and photoperiods could cause eggs and larvae of some copepods species to encyst, fall to the bottom of the water column, and overwinter as a resting form (Herbold et al., 1992). In addition, cyclopoids could undergo a summer encysting stage during high water temperatures. Native estuarine copepod fauna are distributed primarily by salinity and season (Ambler et al., 1985, as cited by Herbold et al., 1992). Suisun Bay usually supports copepod densities approximately twice that of the Delta; average densities in Suisun Bay and the Delta range from 2,000 to 10,000 and 1,000 to 4,000 organisms per liter, respectively (Herbold et al., 1992).

Eurytemora. *Eurytemora affinis* is one of the two most dominant native calanoid copepods in the estuary (the other is *Acartia*). *Eurytemora* is known to concentrate in the entrapment zone at salinities of less than 10 parts per thousand (Cannon, 1982). This herbivorous copepod feeds on a variety of diatoms including *Skeletonema*, *Thalasiosira*, and *Melosira granulata*; blue-green algae; and flagellated protozoans (Obrebski et al., 1992). The greatest abundance of *Eurytemora* occurs in spring as phytoplankton and water temperatures increase in the Delta (Cannon, 1982). Abundance of this species is related to chlorophyll *a* and phytoplankton concentrations (Cannon, 1982). Adult *Eurytemora* are the first prey taken by larval striped bass (Obrebski et al., 1992), and also are known to be an important prey species for newly hatched Pacific herring (Cannon, 1982).

Sinocalanus. *Sinocalanus doerrii* is a calanoid copepod native to China and introduced into the Delta in 1978 from ballast water (Brown, 1992). *S. doerrii* has a broader salinity tolerance than *Eurytemora* and ranges farther into brackish water. This species is omnivorous, eating not only phytoplankton (diatoms and blue-green algae) and protozoans, but also competing with or preying on other zooplankton species. A closely related species, *S. tenellus,* feeds on rotifers and nauplii of crustaceans, so it is suspected that *S. doerrii* also could be predatory (Hada and Uye, 1991, as cited by Obrebski et al., 1992). In larval striped bass feeding studies, capture rates of *Sinocalanus* by young bass were found to be lower than those of either *Eurytemora* or *Cyclops,*

the native copepod species. These differences in capture rates are thought to be due to behavioral differences; *Sinocalanus* swims in unpredictable directions and/or remains motionless in the presence of young bass (Herrgesell, 1993).

Acartia. *Acartia californiensis* and *A. clausi* are common calanoid copepods abundant downstream of the entrapment zone in the estuary. These copepods are euryhaline marine species that reach their highest density in San Francisco Bay (Obrebski et al., 1992). Both species undergo complementary seasonal successional abundances in the San Francisco Bay (Herbold et al., 1992). *Acartia* is the most dominant copepod in Suisun Bay (Herbold et al., 1992) but is least abundant in Suisun Bay during years of high Delta outflow and low salinity conditions (DFG, 1987). Like *Eurytemora*, *Acartia* abundance depends on increasing phytoplankton concentrations and temperature in spring. These species are important prey to carnivorous fish and invertebrates (Cannon, 1982).

Pseudodiaptomus. Two species of *Pseudodiaptomus, P. marinus* and *P. forbesi*, were introduced to Suisun Bay and the Delta in ship ballast water from Asia in 1986 and 1987, respectively (Brown, 1992). *P. forbesi* is found in the entrapment zone during summer and fall (Herrgesell, 1993). Its range extends upstream in the Delta to Stockton in the San Joaquin River, where it is very abundant (Herrgesell, 1993). Peak abundance of *P. forbesi* appears to occur in June, late in the striped bass spawning season. The timing of its peak abundance could assist this copepod by reducing the opportunity for larval bass predation (Herrgesell, 1993).

Limnoithona and Oithona. *Limnoithona sinensis* and *Oithona davisae* are two additional introduced copepod species that have inhabited the Delta estuary since at least 1979 and 1963, respectively, when they were likely released with ballast water from Asia. Like several of the introduced Asian copepods, *Limnoithona* prefers freshwater but has a tolerance for brackish waters (Obrebski et al., 1993). *Limnoithona* is a carnivorous copepod. *Oithona* is a salinity-tolerant marine species with greatest abundance in the South Bay (Obrebski et al., 1993).

Diaptomus. *Diaptomus* spp. are native calanoid copepods found in the lower salinity portions of the western Delta above the entrapment zone and in the Sacramento and San Joaquin rivers. Although *Diaptomus* spp. were common in the Delta, their densities since 1972 have been less than 100 individuals per cubic meter in the Sacramento River and less than 300 individuals per cubic meter in the San Joaquin River (DFG, 1987).

Cyclops. Cyclopoid copepods of the genus *Cyclops*, including *Cyclops vernalis*, are native carnivorous freshwater species found in the most upstream portions of the Delta. *Cyclops* spp. were extremely abundant in the San Joaquin River near Stockton during dry water years 1972, 1976, and 1977 when abundances of these copepods were low elsewhere in the Delta (Orsi, 1992). These species are known to be important prey species for larval striped bass (Herrgesell, 1993).

Cladocerans

Cladocerans, commonly called water fleas, are small freshwater planktonic crustaceans. Most species are widely distributed within the Delta, including the Sacramento and San Joaquin rivers.

These crustaceans seldom occur in abundance in salinities above 1 part per thousand (Herbold et al., 1992). All cladoceran species in the Delta inhabit common areas with no separation by genera.

Cladocerans are seasonally abundant, and their reproductive habits change with the season. Cladocerans reproduce parthenogenetically during warmer months, and populations can increase rapidly (Herbold et al., 1992). As temperatures and photoperiod decline, male cladocerans and larger eggs called ephippia are produced. The male cladocerans fertilize the ephippia, which then sink to the bottom and overwinter. As conditions improve with warmer water and longer photoperiod in spring, the ephippia develop into fully functioning parthenogenetically reproducing females.

The most common and abundant species in the Delta are *Bosmina longirostris*, *Daphnia pulex*, *Diaphonosoma leuchtenbergianum*. *D. leuchtenbergianum* is the least abundant of the three cladoceran species in the Delta. A larger portion of the *D. leuchtenbergianum* population is found in areas of high salinity within the Delta than other cladoceran species (Herbold et al., 1992). *Bosmina* is the most abundant species in the freshwater portions of the Delta. The *Daphnia* spp., the second most abundant group of cladocerans, is restricted to a narrower range of salinity than *Bosmina* (Herbold et al., 1992). During years of high Delta outflow, these cladoceran species occupy portions of Suisun Bay but usually at densities much lower than those in the upper Delta estuary. Generally, abundance of cladocerans is associated with increased temperatures and chlorophyll *a* concentrations (Herbold et al., 1992).

Cladocerans swim by using their antennae and are efficient filter feeding organisms. They feed on phytoplankton, detritus, bacteria, and suspended colloidal materials. Cladoceran species are an important prey organism for many fish, particularly in the upper portions of the Delta.

Mysids

Mysids, commonly called opossum shrimp, are crustaceans of the family Mysidae. These organisms are referred to as opossum shrimp because the females carry their eggs and developing young in an external pouch at the base of their last pair of legs. Young post-larvae are released from this pouch at a relatively well-developed stage and, unlike many true shrimp species, these stages do not undergo further larval development. Numerous species of mysids are found worldwide in marine, estuarine, and freshwater habitats. These organisms are very important prey species for fish and other larger invertebrate species.

N. mercedis, Alienacanthomysis macropsis, and an unidentified mysid species are known to inhabit the San Francisco Bay and Delta (Herbold et al., 1992). Of these species, only *N. mercedis* is found in abundance and is primarily restricted to salinities less than 1 to 4 parts per thousand (Sitts, 1978, as cited by Herbold et al., 1992). *N. mercedis* has been found to be important in the diets of almost all Delta fishes (Herbold et al., 1992).

N. mercedis are found in great abundance in fall and winter in the Delta at upstream freshwater locations. From March through September, they are most abundant in areas within the Delta where salinity is up to 2 parts per thousand (Herbold et al., 1992). Upstream limits of mysid distribution have been shown to be determined by the intensity of light and the resulting light

attenuation within the water column. Water depths exceeding 10 feet are generally required to provide light attenuation to the levels preferred by *N. mercedis* (Heuback 1969, as cited by Herbold et al., 1992). At night, mysids migrate from the bottom and distribute themselves within the entire water column.

In addition to daily (diel) migrations, mysids migrate in response to tidal flow in the Delta (Herbold et al., 1992). Adult mysids remain near the bottom during ebb tides and rise to the surface during flood tides. This behavior tends to distribute the adult mysids farther into the freshwater areas of the estuary (Orsi and Knutson, 1979; Siegfried et al., 1979, as cited by Herbold et al., 1992). The result is that juvenile mysids, which tend to remain near the surface of the water column, are moved farther downstream into the entrapment zone. The entrapment zone also concentrates nutrients, phytoplankton, zooplankton, and other detritus and therefore becomes an ideal nursery for juvenile *N. mercedis* (Siegfried et al., 1979, as cited by Herbold et al., 1992). Flow velocities exceeding 0.4 foot per second provide a barrier to upstream distribution and migration of *N. mercedis* (Turner and Heubach, 1966; Orsi and Knutson, 1979 as cited by Herbold et al., 1992).

Mysid abundance fluctuates throughout the year with seasonal densities of less than 10 individuals per cubic meter in winter to almost 1,000 individuals per cubic meter in spring (Herbold et al., 1992). Reproduction of the larger, overwintering adults in spring is responsible for large abundances in spring and early summer. A second generation is produced in summer by the spring generation, but in much smaller numbers. The spring population is responsible for the production of the subsequent overwintering generation (Herbold et al., 1992).

The diet of *N. mercedis* depends on food type, food availability, season, and their location in the estuary. Older adults prefer copepods, particularly *Eurytemora.* Mysids smaller than 3 millimeters prefer phytoplankton and rotifers (Baldo et al., 1975, as cited by Herbold et al., 1992). Depending on season, various phytoplankton species constitute the dominant food for smaller mysids. Selection of zooplankton prey has been demonstrated for larger adult mysids.

Gammarid Amphipods

Amphipods, commonly known as scuds, are small bottom-dwelling crustaceans. They are detritivorous filter-feeders and live on and feed within the sediments of the western and central Delta (Markmann, 1986). Two species of tube-building encrusting amphipods, *Corophium stimpsoni* and *C. spinicorne,* are commonly found in the low-salinity portions of the estuary, including eastern portions of Suisun Bay and the Delta (Cannon, 1982). These species prefer low velocities, and their abundances are greatest at water velocities from 0.05 to 0.2 foot per second (Cannon, 1982). Highest population densities for both species occur in brackish water, but *C. spinicorne* probably tolerates higher salinity conditions within the Delta. *C. spinicorne* prefers shallow water with peat or cobble substrates; *C. stimpsoni* prefers deeper channels with sandy bottoms (Cannon, 1982).

These amphipods produce two generations per year with the first, the spring-summer population, blooming as the water temperature increases in March. This spring population could become limited by temperature, food, and available habitat (Cannon, 1982). High flows in Delta

channels and salinity intrusion into Suisun Bay probably limit their preferred habitat and *Corophium* abundance in the Delta (Cannon, 1982).

C. stimpsoni was found to be the most abundant benthic species during 1975-1981. Densities reached 20,000 to 50,000 per square meter in samples from some sites within the western Delta. Although found at high densities along with *C. stimpsoni* in the western Delta, *C. spinicorne* was collected at low densities in the central Delta. This difference in abundances in the central Delta was thought to be due to substrate preferences of each species (Markmann, 1986).

An additional species of gammarid amphipod, *Gammarus daiberi*, was introduced into the estuary in 1983 from eastern North America (Brown, 1992). This species is known to be a major food organism for young striped bass in the interior Delta (Chadwick, 1993).

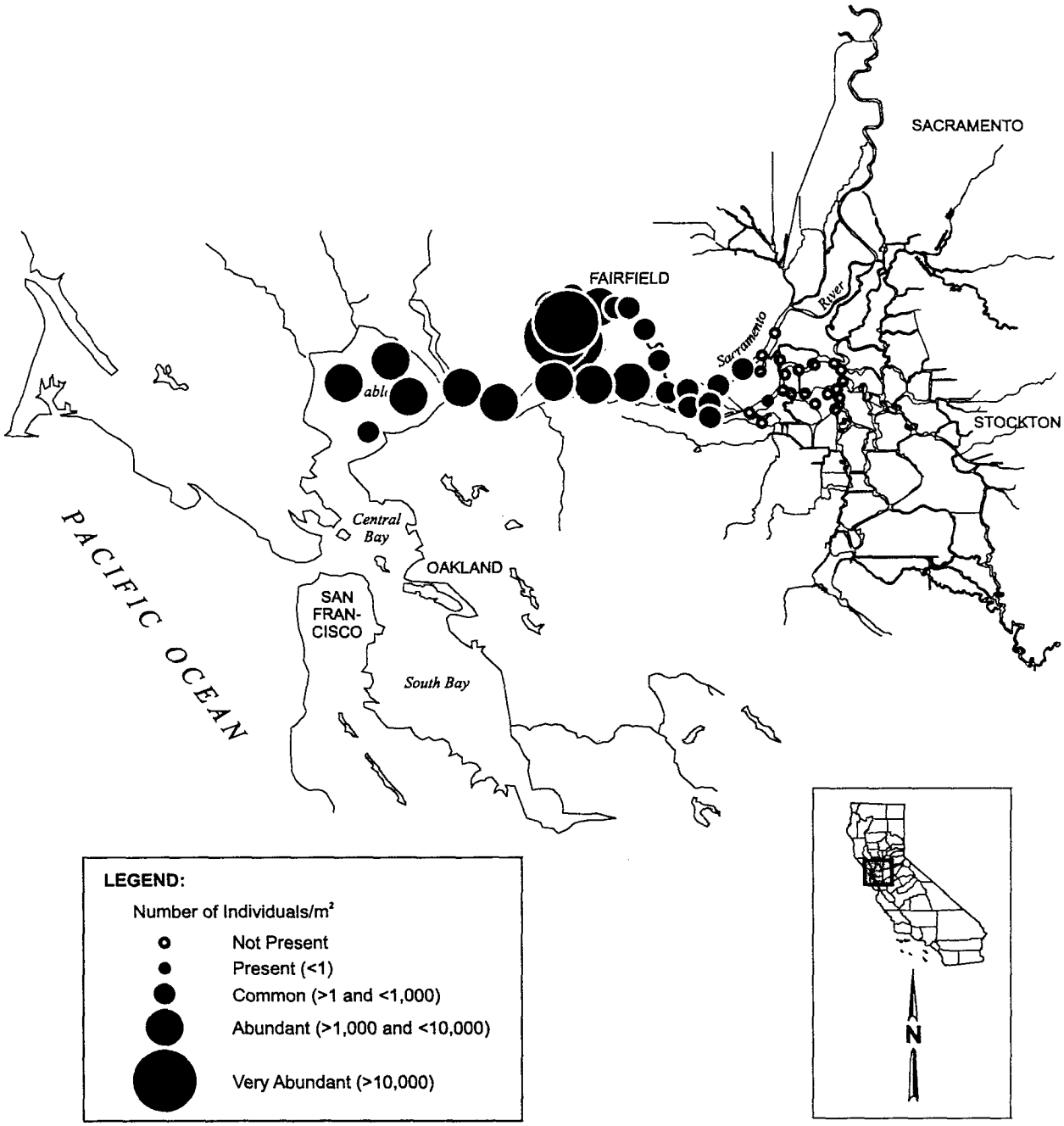
Bivalve Mollusks

Corbicula fluminea. The Asiatic clam *Corbicula fluminea* was introduced to the estuary in 1945 in ballast water from Southeast Asia (Brown, 1992). This clam is intolerant of saline water; it is seldom collected west of Port Chicago and was never found in large numbers in the western Delta during the 1975-1981 benthic sampling (Markmann, 1986). Within the interior Delta, however, *C. fluminea* is the dominant mollusk species (Herbold et al., 1992).

Corbicula spawn from April through October in the Delta at temperatures exceeding 61 degrees Fahrenheit (Eng, 1975, as cited by Markmann, 1986). Immature clams are held in a marsupial pouch within the adult for approximately one month before larval clams are released into the water (Eng, 1977, as cited by Markmann, 1986). Larval clams are transported by rapid flows into regions of lower velocity where they settle on channel banks. After release, larval clams settle from the water column within 48 hours and either attach to or burrow into the substrate. Data collected by DWR suggest that the persistent adult populations of *Corbicula* in the interior Delta could be the source of larval recruitment of this species into the western Delta (Markmann, 1986). Rising salinities in fall and velocities in winter could prevent establishment of large permanent colonies in the western Delta (Markmann, 1986).

Seasonal abundance of this species is a single peak in late spring or early summer in the western Delta and a bimodal peak in spring-summer and late fall in the interior Delta. DWR found densities of *Corbicula* to range from 2,000 to 12,000 per square meter in the interior Delta (Markmann, 1986). The highest densities were collected at sites with slow to moderate velocities, moderate to high levels of suspended fines and organic materials, and slightly brackish to freshwater (Markmann, 1986).

Potamocorbula amurensis. The Asiatic clam *Potamocorbula amurensis* was introduced to the estuary in 1986 in ballast water from Asia. Within four years of its introduction, this clam had become the most abundant benthic organism in many regions of the upper estuary and is among the most widely distributed (Brown, 1992). The highest concentration of this species is in Suisun Marsh, where mean densities of up to 19,200 clams per square meter have been measured (Hymanson, 1991). High densities of *P. amurensis* have been found in both San Pablo and Suisun bays, with much lower abundances in the western Delta where distribution was patchy (Hymanson, 1991). Figure II-121 shows the areas of sample abundance for *P. amurensis* for



NOTES:
Aug-Sept 1990 samples from the upper San Francisco Bay Estuary.

Data from at least four sites (n>4) were combined to calculate relative mean abundances at different locations within regions.

SOURCE: DFG, 1993.

FIGURE II-121

REGIONAL MEAN ABUNDANCE ESTIMATES OF *P. AMURENSIS* OVER THE SURVEY AREA

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August through September 1990. In all areas where these clams were abundant, substrate was predominantly fines consisting of silt and clay.

Studies of the food clearance rates and efficiencies of *P. amurensis* have shown that a clam population in 1 meter of water could effectively filter the overlying water almost 13 times per day (Hollibaugh and Werner, 1991). These studies also found that both phytoplankton and bacteria were good sources of food for *P. amurensis*, with 53 percent of the ingested phytoplankton and 45 percent of the ingested particle-bound bacteria biomass being incorporated into tissue (Hollibaugh and Werner, 1991).

Other Mollusk Species. Other mollusks, both introduced and native, are found in the estuary but are predominantly found downstream of Suisun and San Pablo bays. Species of *Macoma nasuta* (bed-nosed clam) and the introduced *Mya arenaria* (Atlantic soft-shelled clam) were commercially harvested from the San Francisco Bay prior to the 1930s. *Tapes japonica* (Japanese littleneck clam), introduced in the 1930s, now constitutes a clam fishery in the San Francisco Bay. Seasonal changes in the salinity of Suisun Bay historically inhibited clam populations. During the 1976-1977 drought conditions, the abundance of *Mya arenaria* increased dramatically in Suisun Bay. After a large storm event in February 1986, however, most of the native benthic community in Suisun Bay was removed and the Asiatic clam (*P. amurensis*) populations greatly increased in Suisun Bay (Herbold et al., 1992).

FACTORS LIMITING ABUNDANCE

Delta Inflow and Outflow

The most commonly cited factor controlling the abundance, distribution, and reproductive success for many fish species in the estuary is the quantity of the river flow through the estuary (Herbold et al., 1992). It is likely that the freshwater flow into and out of the Sacramento-San Joaquin Delta is also an important limiting factor to the pelagic and benthic invertebrate communities residing in the Delta. River water carries nutrients into the Delta; therefore, flow volume and residence time within the Delta can affect the productivity within the Delta and Suisun Bay (Herbold et al., 1992). This flow also transports phytoplankton produced both upstream of the Delta and, more importantly, within the Delta to Suisun Bay and beyond.

The volume and pattern of Delta inflow and outflow affect zooplankton in different ways. Greater Delta outflow disperses planktonic life stages of *Eurytemora* farther downstream into the more saline estuary (Orsi, 1992). These same freshwater flows also create a saline reverse-flowing bottom current that disperses older life stages of *Eurytemora* farther upstream. Therefore, the volume of freshwater outflow can affect dispersal and distribution and ultimately could affect the abundance of zooplankton populations. During extremely high Delta outflows, the water column can be so strongly stratified that it could prevent the early life stages of planktonic zooplankton from reaching the surface. During high outflows, these life stages could also be transported so far downstream that they are removed from the population within the Delta (Orsi, 1992). The effects of outflow-related dispersal could, therefore, change species and life stage distributions, species relative abundances, and species compositions for both planktonic and benthic communities.

Neomysis abundance is significantly related to Delta outflow during August through November. At very high outflows, *Neomysis* abundance was reduced; at intermediate outflows the abundance could be expected to be high; and at low outflows *Neomysis* populations will not concentrate and will be more dispersed throughout the Delta. The abundance of a cladoceran, *Bosmina*, was limited by high outflows. This species was transported downstream into more saline water where conditions were unfavorable for this freshwater species. Overall, moderately high Delta outflows from approximately 10,000 to 35,000 cfs were associated with the highest annual abundance of *Neomysis, Bosmina,* and possibly *Cyclops*. The abundances of other zooplankton species, *Daphnia* and *Diaptomus,* were not related to Delta outflow (Orsi, 1992).

The Sacramento and San Joaquin rivers are the major sources of freshwater inflow into the estuary. In any estuarine system, total freshwater outflow can be an important factor regulating the biological, physical, and chemical processes (Herrgesell, 1993). Fluctuating Delta inflows and outflows change the environmental conditions in the Delta and downstream in Suisun Bay. These changing conditions could limit invertebrate abundance and distribution by influencing nutrient loading, salinity, and the position of the entrapment zone. Those changes could also further affect food availability, interspecies competition, predation, and community structure.

Delta Hydrodynamics

The Delta is a complex hydrodynamic system. Changing patterns of freshwater inflow and outflow volumes, velocities, gravitational currents, prevailing winds, CVP and SWP pumping influences, and tidal flows all affect the quality and the suitability of the habitat for invertebrate species within the Delta and Suisun Bay. Upstream of Suisun Bay, freshwater volume and velocities limit the distribution of many, if not most, of the invertebrate species in the Delta. During much of the year, freshwater streamflows and tidal flows interact most intensively in Suisun Bay, and these factors influence the pattern of distribution and abundance of invertebrate species in Suisun Bay (Herrgesell, 1993).

A limiting factor thought to have a significant effect on invertebrate distribution and abundance within the Delta is the diversion of freshwater to the southern Delta by export pumping. The rate of freshwater pumped at the state and CVP facilities can result in a net reverse flow within the lower San Joaquin River. High diversion rates can thereby extend periods of anomalous flow patterns within the Delta. These reverse flow periods can intensify the effects of drought conditions on the fauna and limit the abundance and distribution of phytoplankton and invertebrate communities within the Delta and Suisun Bay. Reverse flows have recently occurred throughout the year and during natural spring abundance periods of certain zooplankton species. Under such conditions, zooplankton could be moved upstream in the Delta and directed toward and into the San Joaquin River channel and potentially entrained into the pumps (Herbold et al., 1992).

Salinity

Salinity gradients within the Delta and Suisun Bay are both horizontal and vertical. The penetration of saline water from the San Francisco Bay into Suisun Bay and the western Delta depends on tidal flow, freshwater outflow, and prevailing wind direction. In addition, a vertical salinity gradient can be established in the deep-water channels of the western Delta and Suisun

Bay in which more buoyant freshwater will flow outward over saltwater, which is counterflowing landward by gravitational current. These salinity gradients can affect the distribution of mobile species, which will attempt to find areas of physiologically suitable habitat.

Those species that are sessile (e.g., clams, attached rotifers) or planktonic (e.g., zooplankton, cladocerans) are unable to move to areas of more suitable salinity conditions. Therefore, the abundance of these species could be adversely affected by less-than-optimal salinity conditions. The changing salinity of Suisun Bay could be a major factor preventing the development of a large benthic fauna there (Nichols and Pamatmat, 1988, as cited by Herbold et al., 1992). The ultimate effect of this salinity fluctuation in Suisun Bay could favor organisms with a wide salinity tolerance, including the recently introduced Asiatic clam (*P. amurensis*) (Herbold et al., 1992).

Entrapment Zone

As discussed in the preceding section, water column stratification in the Delta and sometimes downstream in Suisun Bay occurs when buoyant surface freshwater flows outward and denser saltwater flows inward from tidal and gravitational countercurrent bottom flows (Herrgesell, 1993). The area of maximum concentration of suspended matter in this stratified layer is termed the entrapment zone. The entrapment zone moves horizontally on a daily and seasonal basis. Phytoplankton and zooplankton abundances are known to be higher in the entrapment zone, and the phytoplankton biomass is higher when the entrapment zone is in Suisun Bay (Arthur and Ball, 1978, 1979, 1980, as cited by Kimmerer, 1992). The position of the entrapment zone could act to regulate phytoplankton productivity and invertebrate abundance and distribution within the Delta and Suisun Bay.

Zooplankton species *Eurytemora* and *Neomysis* and rotifer species tend to be most abundant in or near the entrapment zone (Kimmerer, 1992). *Pseudodiaptomus forbesi,* the recently introduced copepod, also tends to be most abundant in or near the entrapment zone, whereas the introduced copepod *Sinocalanus doerrii* is most abundant in the area above the entrapment zone (Kimmerer, 1992). Other species can be abundant in the entrapment zone but also could be abundant downstream. These organisms include *Acartia* and other planktonic copepods and the larvae of many other invertebrate species.

Food Availability

Chlorophyll *a* concentration provides a measure of the phytoplankton productivity in the estuary (Figure II-122). Except for 1977, the mean monthly chlorophyll *a* concentrations at the lower Sacramento River sampling locations (northern Delta) remained relatively stable during 1970-1991. This suggests that phytoplankton productivity has been relatively stable. Except for 1991, chlorophyll *a* concentrations in the lower San Joaquin River (southern Delta) have remained low since the late 1970s, suggesting that phytoplankton productivity here has diminished somewhat since the late 1970s. The mean monthly chlorophyll *a* concentration measured in the central Delta fluctuates on a seasonal basis and is typically high in late summer and early fall and relatively stable during the remaining months. In contrast to the northern and central Delta, the mean monthly chlorophyll *a* concentration in Suisun Bay has declined during 1970-1991.

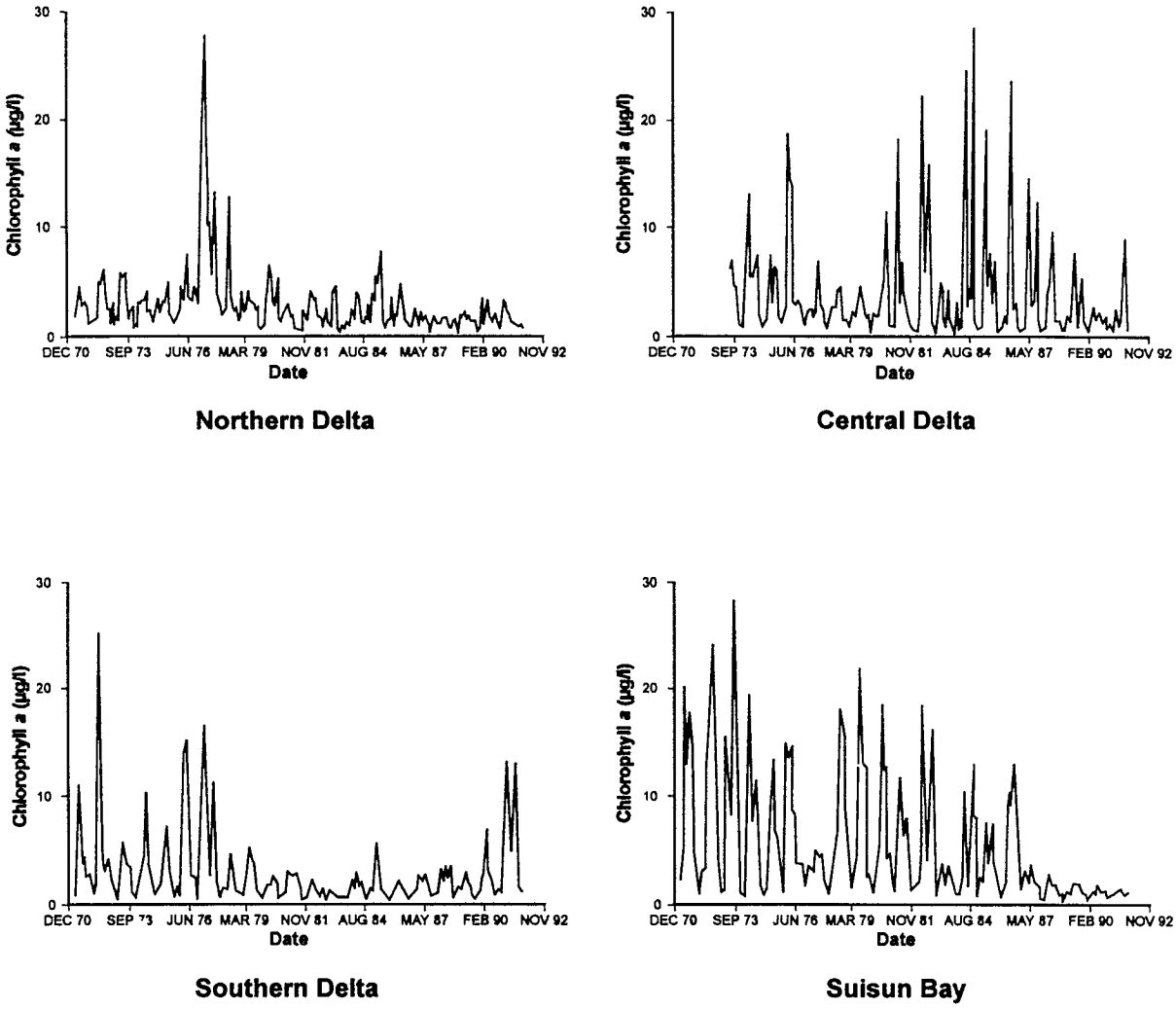

FIGURE II-122

MEAN MONTHLY CHLOROPHYLL *a* CONCENTRATIONS IN THE NORTHERN DELTA, CENTRAL DELTA, SOUTHERN DELTA, AND SUISUN BAY, 1971-1991

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These trends indicate that the phytoplankton productivity of water flowing into the Delta has remained relatively stable, phytoplankton productivity in the central Delta fluctuates on a seasonal basis, and phytoplankton concentrations in Delta outflow and Suisun Bay have greatly declined since 1970. The consequences of these trends are that phytoplankton production, a primary food source for zooplankton and mollusks, in the northern and central Delta has been relatively stable but could be seasonally limiting. Furthermore, intense grazing by the introduced clam *P. amurensis* could be responsible for reducing phytoplankton concentrations in Suisun Bay and limiting the availability of phytoplankton for zooplankton, thereby affecting zooplankton abundance in Suisun Bay.

It has been assumed that because the food web depends on phytoplankton production, factors that increase or enhance phytoplankton must likewise increase or enhance the rest of the food web. A simplified correlation has been established by DFG for phytoplankton chlorophyll *a* production and *Eurytemora* and *Neomysis* abundance in the Delta. Some evidence indicates that zooplankton could also use organic detritus in the entrapment zone (Heinle et al., 1977 as cited by Kimmerer, 1992). However, experiments determined that food was not limiting *Eurytemora* abundance. The question of food limitation as it affects invertebrate abundance and distribution is as yet unanswered (Kimmerer, 1992).

Interspecific Competition and Predation

The presence of the Asiatic clam *P. amurensis* is thought to severely reduce the phytoplankton standing crop in Suisun Bay (Herbold et al., 1992). The reduction in food availability for other phytoplankton-grazing species, including benthic and zooplankton species, could have reduced these species' abundances. Recent studies have also indicated that *P. amurensis* is probably capable of feeding on the nauplii stages of *Eurytemora* and possibly other copepods found in the estuary (Herrgesell, 1993).

Competition between species has accelerated because of the recent introductions of several copepod and clam species into the estuary. This has led to the decline of *Eurytemora affinis,* formerly the dominant copepod in the Delta. The introduced copepod *Sinocalanus doerrii* has replaced *E. affinis* as the dominant copepod, and two other introduced species, *Limnoithona* and *Pseudodiaptomus forbesi,* are becoming more abundant than other native species, such as *Cyclops* and *Diaptomus* (Herbold et al., 1992). It is believed that *Sinocalanus* can either compete with *Eurytemora* for its algal food or prey on its young life stages (Orsi, 1992).

Although the total abundance of copepods could not have significantly diminished over the past 20 years (Chadwick, 1993), the species composition has changed, with competition from introduced species likely a major contributing factor. This change in species composition could affect the availability of copepods as prey organisms for fish. *Sinocalanus,* which has replaced *Eurytemora* as the most abundant copepod in the Delta, generally occupies a position farther upstream in the Delta than does *Eurytemora* (Herbold et al., 1992). Introduced copepods have been found to be less easily preyed upon than native copepod species (Herrgesell, 1993). Therefore, although present densities and total abundances of copepods could be relatively the same as they were in the period prior to the 1970s, the available prey organisms for certain fishes and invertebrates could have diminished in the Delta.

ONGOING MONITORING, ENHANCEMENT, AND HABITAT IMPROVEMENT ACTIONS

The most significant ongoing monitoring activities relating to Delta invertebrate abundance and distribution are elements of the IEP. Ongoing research in the water quality element of the Bay/Delta Study constitutes a significant monitoring and analysis activity within the Delta and Suisun Bay. Activities relating to the monitoring and health of the invertebrate communities within the Delta and Suisun Bay include the phytoplankton monitoring program, which collects and analyzes samples at 19 sites throughout the Delta and Suisun Bay. Zooplankton monitoring and analyses continue to be conducted as part of the Bay/Delta Study. The Bay/Delta Study includes information obtained by the Food Chain Group whose goal is to promote an enhanced understanding of the food chains of important fishery resources of the Sacramento-San Joaquin Delta estuary.

To accomplish this goal, the following objectives have been put forth (Herrgesell, 1993):

- to detect significant and important changes that could have occurred at the various levels of the food chain through a systematic examination of the existing data;
- to investigate the underlining causes of the identified changes by developing and evaluating hypotheses relevant to the changes and to relationships among other elements of the food chain;
- to design and promote the implementation of applied research projects to test the above hypotheses and to acquire information on aspects of the food chain not available through routine monitoring programs;
- to provide a forum for critical scientific discussion and peer review of food chain investigation results obtained by agency and outside scientists and through the process associated with the above activities; and
- to provide up-to-date information on changes within the food chain group to interagency program management and others for use in establishing management objectives and recommendations.

RECENT CONDITIONS FOR SPECIAL-STATUS AQUATIC SPECIES

Special-status aquatic species are animals that are legally protected under federal and state ESAs or other regulations and species that the scientific community considers sufficiently rare to qualify for such listing. Special-status aquatic animals are species in the following categories:

- animals listed or proposed for listing as threatened or endangered under the federal ESA (50 Code of Federal Regulations 17.11 [listed animals] and various notices in the Federal Register [proposed species]);

- animals that are candidates for possible future listing as threatened or endangered under the federal ESA (54 Federal Register 554, January 6, 1989);
- animals listed or proposed for listing by the State of California as threatened or endangered under the California ESA (14 CCR 670.5); and
- animals considered to be species of special concern by the State of California.

Information on the status, habitats, and distribution of special-status aquatic animals in the project area was obtained primarily from DFG's Natural Diversity Data Base (1993), Moyle et al. (1989), Moyle (1976). Special-status aquatic animals were selected for inclusion in this analysis if they occur in counties within the study area. Given the large study area species occurring some counties within California were excluded. Those counties excluded include Siskiyou, Modoc, Lassen, Plumas, Sierra, Alpine, Mono, Inyo, and Imperial counties; the eastern portions of San Bernardino, Riverside, and San Diego counties; and the central and northern portions of Del Norte County were excluded.

Table II-19 summarizes information on the legal status, geographic distribution, and habitat requirements of special-status aquatic species. Many of these species will probably not be affected by the CVPIA, but excluding such species will require detailed information on local species distributions and local land use changes resulting from the CVPIA, which cannot be determined in a program-level EIS. Additional detailed information on special-status species is presented in the previous sections for winter-run and spring-run chinook salmon, summer steelhead trout, delta smelt, and Sacramento splittail.

Table II-19 omits fairy shrimp and aquatic beetles (coleopterans); these species are included in Table A-1 of Attachment A of the Vegetation and Wildlife Technical Appendix. In addition, Table II-19 excludes numerous caddisfly (Trichopteran) species because data were unavailable regarding their habitats and distribution, and it was not known whether these species occur within the study area.

CONCLUSIONS

Habitat modification, species introductions, and overfishing adversely affected fisheries resources in the CVPIA study area before and after construction and operation of CVP facilities. Major human-induced modifications of fish habitats began during the settlement of California that followed the 1849 gold rush. Historically, agricultural development, flood control, grazing, mining, logging, navigation, and urban development activities degraded water quality in almost all fish habitats. These same activities continue to degrade water quality in fish habitats. Debris control, flood control, and water supply projects (including CVP facilities) blocked most major rivers in the Central Valley. Water storage and diversions continue to alter natural river and Delta flow patterns, and fish are entrained at water diversions. Exotic species that were accidentally or intentionally introduced into the Sacramento-San Joaquin River system adversely affected native species through predation, competition for habitat and food, and hybridization. Total harvest of fishes has contributed to the declines of salmon, steelhead, striped bass, and American shad, although the extent of the contribution is unclear.

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TABLE II-19

LEGAL STATUS, HABITAT REQUIREMENTS, AND GEOGRAPHIC DISTRIBUTION OF SPECIAL-STATUS AQUATIC SPECIES

Common Name Scientific Name	Status Federal/State	Habitat Requirements	Geographic Distribution
Crayfish and Shrimp			
Shasta crayfish *Pacifastacus fortis*	E/E	Sandy bottoms among rubble in cool, clear waters associated with large springs	Fall River, Hat Creek, and sections of Pit River between these creeks in northeastern Shasta County
California freshwater shrimp *Syncaris pacifica*	E/E	Shallow, low-gradient coastal streams with good riparian cover and undercut banks	East Austin (tributary to Russian River), Salmon, Lagunitas, Sonoma, and Huachica creeks in Marin, Napa, and Sonoma counties
Fishes			
River lamprey *Lampetra ayresi*	--/SSC	Anadromous species spending much of its adult life in estuaries; spawns in small, clean tributary streams, and rears in silty backwaters of such streams	Widely distributed in northern California coastal areas south to San Francisco and probably most abundant in Sacramento-San Joaquin River system
Kernbrook lamprey *Lampetra hubbsi*	--/SSC	Silty backwaters of large rivers in foothill regions	Friant-Kern Canal and lower reaches of Merced, Kaweah, Kings, and San Joaquin rivers in San Joaquin Valley
Little Kern golden trout *Oncorhynchus aguabonita whitei* and *O. mykiss whitei*	T/--	Small, high-elevation and cold-water lakes and streams, generally without other salmonids	Found in more than 300 mountain lakes and streams in Sierra Nevada, including Little Kern River
Volcano Creek golden trout *Oncorhynchus mykiss aguabonita*	--/SSC	Small, high-elevation and cold-water lakes and streams, generally without other salmonids	Found in more than 300 mountain lakes and streams in Sierra Nevada, including South Fork Kern River
Kern River rainbow trout *Oncorhynchus mykiss gilberti*	--/SSC	Small- to medium-sized cold-water streams with suitable spawning gravels	Upper Kern River and tributaries in Kern County, including North Fork Kern River and Rattlesnake, Osa, Soda, and Hells Hole creeks
McCloud River redband trout *Oncorhynchus mykiss* ssp.	C/SSC	Clear, cold, spring-fed streams	McCloud River above Middle Falls and its tributaries, such as Sheephaven, Tate, Edson, Moosehead, Swamp, and Trout creeks in Shasta County

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TABLE II-19. CONTINUED

Common Name Scientific Name	Status Federal/State	Habitat Requirements	Geographic Distribution
Coastal cutthroat trout *Oncorhynchus clarki clarki*	--/SSC	Small, low-gradient, cool, and well-shaded coastal streams with small-sized gravel substrate and estuarine habitats	Small coastal streams from the Eel River northward
Steelhead trout *Oncorhynchus mykiss gairdneri*	P/SSC	Anadromous species favoring large cold-water tributaries with deep pools for oversummering habitat	Distributed throughout coastal and central California
Chinook salmon (spring run) *Oncorhynchus tshawytscha*	--/SSC	Anadromous species favoring large, cold-water tributaries with deep pools for oversummering habitat	Major isolated tributaries to the Trinity, Klamath, and Upper Sacramento rivers, including Mill, Deer, and Butte creeks and the Feather River
Chinook salmon (winter run) *Oncorhynchus tshawytscha*	E/E	Anadromous species favoring clean, coldwater over gravel beds during summer months and appropriate habitats for rearing and smolt emigration through the Delta	Sacramento River, with successfull spawning limited to the reach immediately downstream of Keswick Reservoir
Pink salmon *Oncorhynchus gorbuscha*	--/SSC	Anadromous species spawning in small to large coastal streams	Periodically occurs in San Lorenzo, Sacramento, Russian, Garcia, Mad, and Klamath rivers, as well as smaller north coastal streams
Coho salmon - Central Coast *Oncorhynchus kisutch*	T/SSC	Anadromous species preferring coastal streams with well-shaded pools, cool temperatures, and overhead cover such as woody debris	Klamath, Trinity, Mad, Noyo, and Eel rivers, with other populations in smaller coastal streams south to San Lorenzo River in Santa Cruz County
Bull trout *Salvelinus confluentus*	--/E	Cold-water rivers with deep pools	McCloud River in Shasta County
Delta smelt *Hypomesus transpacificus*	T/T	Euryhaline habitats with salinities no more than 10-12 ppt in both main Delta channels and open waters	Delta, Suisun Bay, and Suisun Marsh
Longfin smelt *Spirinchus thaleichthys*	--/SSC	Prefers euryhaline habitats with salinities mostly greater than 10 ppt although migrates from nearly pure seawater to completely freshwater	Delta; Suisun Marsh; and Suisun, San Pablo, and San Francisco bays with smaller populations in lower Eel River
Sacramento splittail *Pogonichthys macrolepidotus*	P/SSC	Primarily freshwater but tolerant of salinities of 10-12 ppt in slow-moving sections of rivers and sloughs with flooded vegetation for spawning	Delta; Suisun Bay; and Suisun Marsh, Napa Marsh, and other parts of the estuary

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TABLE II-19. CONTINUED

Common Name Scientific Name	Status Federal/State	Habitat Requirements	Geographic Distribution
Arroyo chub *Gila orcutti*	--/SSC	Slow-water stream sections with mud or sand substrates	Southern California streams as follows: Santa Ynez, Santa Maria, Santa Margarita, West Fork San Gabriel, and Mojave river systems; and Malibu
Clear Lake hitch *Lavinia exilicaudachi*	--/SSC	Limnetic zone of Clear Lake used by adults and near-shallow water by juveniles, with tributaries used for spawning	Clear Lake and its intermittent tributaries in Lake County
Navarro roach *Lavinia symmetricus navarroensis*	--/SSC	Shallow waters and pools in small coastal streams and their tributaries	Navarro River and its tributaries in Mendocino County
Gualala roach *Lavinia symmetricus parvipinnis*	--/SSC	Shallow waters and pools in small coastal streams and their tributaries	Gualala River and its tributaries in Solano County
San Joaquin roach *Lavinia symmetricus* ssp.	--/SSC	Shallow waters and pools in small intermittent streams	Intermittent tributaries to San Joaquin River
Tomales roach *Lavinia symmetricus* ssp.	--/SSC	Shallow waters and pools in small coastal streams and their tributaries	Tributaries to Tomales Bay in Marin County
Hardhead *Mylopharadon conocephalus*	--/SSC	Undisturbed streams with clear, deep pools with sand-gravel-boulder substrates and slow-water velocities and mid-elevation reservoirs	Widely distributed in low- to mid-elevation streams in Sacramento-San Joaquin drainage and Russian River drainage and mid-elevation reservoirs
Santa Ana speckled dace *Rhinichthya osculus* ssp.	--/SSC	Small, permanently flowing streams with runs and riffles, cover, gravel-cobble substrates, and water temperatures of 63 to 68 degrees Fahrenheit	Limited to headwaters of Santa Ana, San Gabriel, and Los Angeles River systems in Southern California
Santa Ana sucker *Catostomus santaanae*	P/SSC	Small- to medium-sized streams with clear water and generally coarse substrates with algae	Santa Ana River, Big Tujunga Wash in Los Angeles River drainage, three forks of San Gabriel River, and Santa Clara River
Unarmored threepine stickleback *Gasterosteus aculeatus williamsoni*	E/E	Quiet pools with abundant aquatic vegetation, backwater areas, and stream margins with low-velocity, cool, and clean water	Upper Santa Clara River in Los Angeles County and San Antonio Creek in Santa Barbara County

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TABLE II-19. CONTINUED

Common Name Scientific Name	Status Federal/State	Habitat Requirements	Geographic Distribution
Santa Ana threepine stickleback *Gasterosteus aculeatus santaanae*	--/SSC	Quiet pools with abundant aquatic vegetation, backwater areas, and stream margins having low-velocity, cool, and clean water	Shay Creek, tributary to Baldwin Reservoir in San Bernardino County
Sacramento perch *Archoplites interruptus*	--/SSC	Warm-water reservoirs, ponds, and a few streams where it has been introduced	Several small ponds and lakes widely distributed in California and Alameda Creek in Alameda County
Russian River tule perch *Hysterocarpus traski pomo*	--/SSC	Clean, flowing water with abundant cover such as beds of aquatic macrophytes, submerged tree branches, and overhanging plants	Russian River and its tributaries in Sonoma and Mendocino counties
Tidewater goby *Eucyclosobius newberryi*	E/SSC	Shallow lagoons and lower stream reaches in brackish to fresh water that is slow moving, with high dissolved oxygen, sand and mud substrates, and abundant emergent and submerged vegetation	Widely distributed in brackish water habitats along the California coast from Agua Medionda Lagoon in San Diego County to the Smith River in Del Norte County
Rough sculpin *Cottus asperrimus*	--/T	Deep, cool, clear, fast-flowing large streams with a variety of substrates, particularly fine-grained bottoms	Pit and Fall river systems in Shasta County
Bigeye marbled sculpin *Cottus klamathensis macrops*	--/SSC	Large, cool, clean, spring-fed streams with moderate velocities, abundant aquatic vegetation, and coarse substrates, including a few reservoirs	Middle reach of Pit River system in Shasta County below Britton Reservoir and in Britton Reservoir, Tunnel Reservoir, lower Hat Creek, Sucker Springs Creek, and Clark Creek

LEGEND:

Federal:
- E = listed as endangered under the federal Endangered Species Act.
- T = listed as threatened under the federal Endangered Species Act.
- C = candidate for federal listing; includes species for which the Service has on file enough substantial information on biological vulnerability and threat to support proposals to list them.
- P = proposd for listing as either a threatened or an endangered species.
- -- = no status designation.

State:
- E = listed as endangered under the California Endangered Species Act.
- T = listed as threatened under the California Endangered Species Act.
- P = proposed for listing as either a threatened or an endangered species.
- SSC = species of special concern.
- -- = no status designation.

NA = data not available.
ppt = parts per thousand.

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