V. AQUATIC RESOURCES

Populations of many aquatic resources that exist in the Bay-Delta Estuary have undergone significant declines over the past several decades. These population reductions have led to concerns about the condition of the estuarine system, as well as petitions and listings for endangered status for some species. The simultaneous declines of many estuarine species suggest that they are responding to common stresses (Jassby et al. 1994).

The following discussion is divided into three sections. First, the general causes contributing to the decline of aquatic resources are reviewed. Second, the population trends and, if relevant, causes of declines of specific aquatic resources are presented. Third, the degree to which the general causes contributing to the decline are controllable is discussed.

A. GENERAL CAUSES OF DECLINES

Numerous factors are thought to be responsible for the decline of fish and invertebrate species that live in and migrate through the Bay-Delta Estuary. The conditions in the Estuary may be only partially responsible for the decline of those species that utilize the Estuary for only a part of their life cycle. The general causes of decline for most of the species utilizing the Estuary fall within the following categories: (1) natural hydrologic variability; (2) water development; (3) introduction of non-native aquatic organisms; (4) food limitations; (5) land reclamation and waterway modification (diking, dredging, and filling); (6) pollution; (7) harvesting; and (8) oceanic conditions. These factors can cause direct, indirect, and cumulative effects on the various species in the Estuary (DFG 1994b, SFEP 1992a). The most significant factors are the human-induced factors, and of these, water development, land use practices, and harvesting of aquatic species are the most significant factors causing declines in aquatic species.

1. Natural Variability of Precipitation and Hydrology

The flow of fresh water to the Bay-Delta Estuary is determined mainly by the amount and timing of precipitation in the Central Valley watershed. Under natural conditions in an average year, flows increase in late fall as rains swell streams. Flows continue to increase throughout the winter and peak in the spring when warm temperatures melt the Sierra snowpack. After the spring snow melt, flows decline to low levels until the fall (SFEP 1992b).

Just as total precipitation varies each year, the volume of water annually flowing into the Delta has been highly variable. During the past 70 years, in years of high precipitation, the volume of inflow to the Delta may exceed 50 MAF; in years of low precipitation, Delta inflow may be less than 8 MAF. For planning and regulatory purposes, the SWRCB has developed water year classification systems that provide a relative estimate of the amount of water originating in the Sacramento and San Joaquin hydrologic basins from seasonal runoff and reservoir storage. Each system has five kinds of water years: wet, above normal, below normal, dry, and critical. Table V-1 shows the water year types for the Sacramento and San Joaquin river systems for the period 1922-1992 (SFEP 1992b). The past 20 years have included the wettest year (1983), as well as the driest and longest droughts (1976-1977 and 1987-1992), on record (NHI 1992a). In addition to year-to-year variations in flow, extreme fluctuations occur on a seasonal basis. For example, May of 1990, a critical year, was the wettest May on record (CUWA 1994).

Many of the Estuary's native aquatic species are adapted to an ecosystem characterized by this high seasonal variation in freshwater flows. One of the most important aspects of the natural flow pattern was the large volume of water that entered the Estuary in the winter and spring. These flows repelled sea salts from the Delta, ensuring appropriate water quality for freshwater wetlands. They also washed nutrients into the Estuary, stimulating growth of organisms at the base of the food web, and enabled fish to migrate, spawn, and rear successfully (SFEP 1992b).

Variation in the amount of flow to the Bay-Delta is the most commonly cited control on the abundance, distribution, and reproductive success of many species fish in the Estuary. Drought and low flow conditions can have wide-ranging impacts on aquatic resources, depending on the species and life stage requirements. For many species, drought conditions can reduce available physical habitat, elevate water temperatures, reduce the food supply, increase susceptibility to predation, and degrade spawning and rearing habitats. Poor habitat conditions in one year will likely result in reduced egg and young survivals for that year, resulting in a poor year class in the adult population. If conditions continue for multiple years, the ability of the species to recover may be reduced (CUWA 1994).

There is little doubt that the combination of floods and severe droughts have contributed to, and accelerated, the declines in populations of aquatic resources in the Bay-Delta Estuary, particularly in recent years. However, the effects of variable precipitation patterns, particularly sustained drought and low flow conditions, on aquatic species must be considered in the context of ongoing operations of water projects and other diversions (NHI 1992a, SFEP 1992b). As discussed below, the natural pattern of freshwater flow into the Bay-Delta Estuary has been changed significantly by water development.

2. Water Development

There has been extensive water development throughout the Central Valley. There are numerous direct and indirect effects on downstream water quantity and water quality, from operations of reservoirs upstream to the export pumping in the Delta.

Until the mid-1800's, the waters of the Central Valley and the Bay-Delta Estuary, and its aquatic resources, were essentially undisturbed by water development. After the discovery of gold in 1848, the diversion of water from northern Sierra streams for hydraulic mining began to modify the Estuary's freshwater flows. By 1860, more than half of the State's population of 380,000 lived around the Estuary or in its watershed. The increasing demand for food

Table V-1
Water Year Types for the Sacramento and San Joaquin River Basins [1]

1922-1992

Water Year	Sacramento River	San Joaquin River
1922	AN	W
1923	BN	AN
1924	C	C
1925	D	BN
1926	D	D
1927	W	AN
1928	AN	BN
1929	C	C
1930	D	C
1931	C	C
1932	D	AN
1933	C	D
1934	C	C
1935	BN	AN
1936	BN	AN
1937	BN	W
1938	W	W
1939	D	D
1940	AN	AN
1941	W	W
1942	W	W
1943	W	W
1944	D	BN
1945	BN	AN
1946	BN	AN
1947	D	D
1948	BN	BN
1949	D	BN
1950	BN	BN
1951	AN	AN
1952	W	W
1953	W	BN
1954	AN	BN
1955	D	D
1956	W	W
1957	AN	BN
1958	W	W
1959	BN	D
1960	D	C
1961	D	C
1962	BN	BN
1963	W	AN
1964	D	D
1965	W	W
1966	BN	BN
1967	W	W
1968	BN	D
1969	W	W
1970	W	AN
1971	W	BN
1972	BN	D
1973	AN	AN
1974	W	W
1975	W	W
1976	C	C
1977	C	C
1978	AN	W
1979	BN	AN
1980	AN	W
1981	D	D
1982	W	W
1983	W	W
1984	W	AN
1985	D	D
1986	W	W
1987	D	C
1988	C	C
1989	D	C
1990	C	C
1991	C	C
1992	C	C

[1] *Sacramento River sorted by 40-30-30 water year classification; San Joaquin River sorted by 60-20-20 water year classification*

prompted the conversion of native habitats to farmland. As agriculture became established in the Central Valley, significant volumes of water from streams were diverted to irrigate crops (SFEP 1992b).

Although hydraulic mining was ceased in 1884, the expansion of irrigated agriculture in the Central Valley continued well into this century. By 1929, more than 1.2 million acres of valley lands, excluding the Delta, were irrigated with water diverted directly from the Sacramento and San Joaquin river systems upstream of the Delta. The early-1900's was also a period of urban and industrial growth. To support the economic growth of the region, private and public water development projects were constructed in the Estuary's watershed to provide electrical generation, flood control, and water for municipal, industrial, and agricultural uses. The Mokelumne Aqueduct began delivering water from the Mokelumne River drainage to the East Bay in 1929, and the Hetch Hetchy Aqueduct began transfers of water from the Tuolumne River to San Francisco in 1935. The federal CVP, with dams on the Sacramento and San Joaquin rivers, began providing water in 1945 with the operation of the Contra Costa Canal. The CVP began transporting Delta water in the Delta-Mendota Canal in 1951. The main features of the Sacramento River Flood Control Project were completed in the mid-1940's. The SWP, which was authorized in 1959, began delivering water, via the California Aqueduct, to north of the Tehachapis in 1968; by 1972, SWP facilities were supplying water to southern coastal areas of California.

The extensive water development in the Central Valley and Delta has adversely affected fish and wildlife habitat in the Estuary and upstream areas (SFEP 1992b). An overview of impacts resulting from water development are discussed below under the following headings: upstream impacts; inflows to the Delta; Delta outflow; Delta diversions and entrainment; reverse flows; and the Delta Cross Channel and Georgiana Slough.

a. Upstream Impacts. Large multi-purpose reservoirs have been constructed on all of the Central Valley's major streams except the Cosumnes River. More than 100 reservoirs each have a storage capacity of at least 50 TAF, and the ten largest each store more than 1 MAF of water. Together, valley reservoirs can store about 27 MAF, which is about 60 percent of the State's average annual runoff (SFEP 1992b).

The construction of dams for water storage on nearly all of the Estuary tributary streams in the Central Valley has eliminated habitats for numerous aquatic species. Dams have also blocked access to thousands of miles of cool water spawning and rearing habitats for migratory species, such as salmon and steelhead trout, which rely on the upper tributaries to complete their life cycle. Upstream water development has reduced the stream spawning habitat available to salmon and steelhead from 6,000 miles to 300 miles, a 95 percent reduction from historic levels. Approximately 50 percent of the available spawning grounds in the Sacramento River were eliminated by the construction of Shasta Dam alone, and Friant Dam eliminated all salmon spawning on the main stem of the San Joaquin River above Friant (DFG 1993).

Reservoirs not only block access to cooler water upstream, but also act as heat storage facilities in the summer months (DFG 1994b). Impoundments increase stream water temperatures by releasing water that was heated in the reservoir and by reducing instream flows below the dam. Water temperature is also affected by overhanging vegetation which shades and cools the water. Shaded riverine aquatic habitat has been significantly altered through bank protection and flood control projects.

Agricultural return flows, such as those from the Colusa Basin Drain into the Sacramento River, are also major contributors of warm water to the rivers (DFG 1993). Flows in the Colusa Basin Drain occasionally exceed 2,000 cfs with water temperatures in excess of 80°F; whereas typical summer Sacramento River flows are 15,000 cfs at temperatures of 68°F. Consequently, water temperatures in the Sacramento River can exceed 70°F below Knights Landing during May and June. In all three major tributaries of the San Joaquin River system, the Merced, Stanislaus, and the Tuolumne rivers, warm water temperatures have exceeded critical temperatures for salmonid spawning, incubation, and rearing, especially in dry years (DFG 1993).

Dams also impede the replenishment of gravel necessary for salmon and steelhead spawning by preventing the movement of new gravel from upstream areas. Furthermore, gravel replacement from stream banks is limited by erosion control and bank stabilization activities. These activities have also reduced the amount of riparian habitat in the upstream areas, reducing usable fish habitat and contributing to the warming of the rivers (DFG 1994b).

In addition to the direct impacts associated with loss of habitat, the operations of upstream water projects have altered the natural flow conditions in the lower rivers. Upstream water development causes variations in stream flows which differ from the natural seasonal variation in freshwater flows to which the Estuary's native aquatic species are adapted. Central Valley reservoirs are operated primarily for flood control in the winter and for capturing the spring snowmelt runoff to be released in the summer for agriculture. Although the timing of flow releases varies from reservoir to reservoir, the overall effect of storage operations is to reduce the volume of water flowing downstream throughout the late fall, winter, and spring, and to increase flows during the summer and early fall. Under natural conditions in an average year, flows increase with rainfall in late fall, continue to increase throughout the winter, peak in the spring with the Sierra snow melt, and then decline to low levels until the fall (CUWA 1994, SFEP 1992b).

Changes in the amount and timing of flows as a result of water development have impacted aquatic resources in upstream areas by influencing the amount and quality of habitat available. Rapid reductions in flow can expose incubating eggs or strand young fish which use the edge of the stream channel. Adequate flows, particularly in the San Joaquin River system, are often not provided to maintain adequate temperatures during the salmon smolt outmigration period (DFG 1993). Delays in the transport of migratory species can increase mortality rates through increased predation and losses to diversions (CUWA 1994).

Upstream water diversions impact aquatic resources in various ways. Diversion structures, such as those on Butte, Clear, Mill, and Deer creeks in the Sacramento River basin, can cause barriers to upstream spawning habitat and delay migration (DFG 1993). Thousands of unscreened and inadequately screened diversions in upstream areas entrain aquatic organisms and increase mortality. Upstream diversions remove large volumes of fresh water from Central Valley streams that are tributary to the Delta and cause reductions in stream flow. The amount of water diverted upstream of the Delta has increased markedly since the turn of the century when slightly more than 1 MAF was removed. Today, upstream diversions reduce Delta inflow by an estimated 9.4 MAF, or about one-third of the historic average annual Delta inflow. CVP diversions account for about 4.5 MAF of the upstream depletion; the Hetch Hetchy and Mokelumne aqueducts combined remove about 470 TAF, and thousands of other agricultural and urban diversions account for the remainder (SFEP 1992b).

b. Inflows to the Delta. The total annual volume of freshwater inflow to the Estuary is highly variable. During the past 70 years, annual inflow has ranged from more than 70 MAF to 5.9 MAF, with an average of about 21 MAF. This variability is the result of precipitation patterns and upstream water development, primarily storage reservoirs and diversions (SFEP 1992b). Inflows to the Delta principally come from three Central Valley sources: the Sacramento River basin, the San Joaquin River basin, and the Central Sierra basin. These river basins contributed approximately 80, 15, and 5 percent of the average annual Delta inflow from the Central Valley (SFEP 1992b).

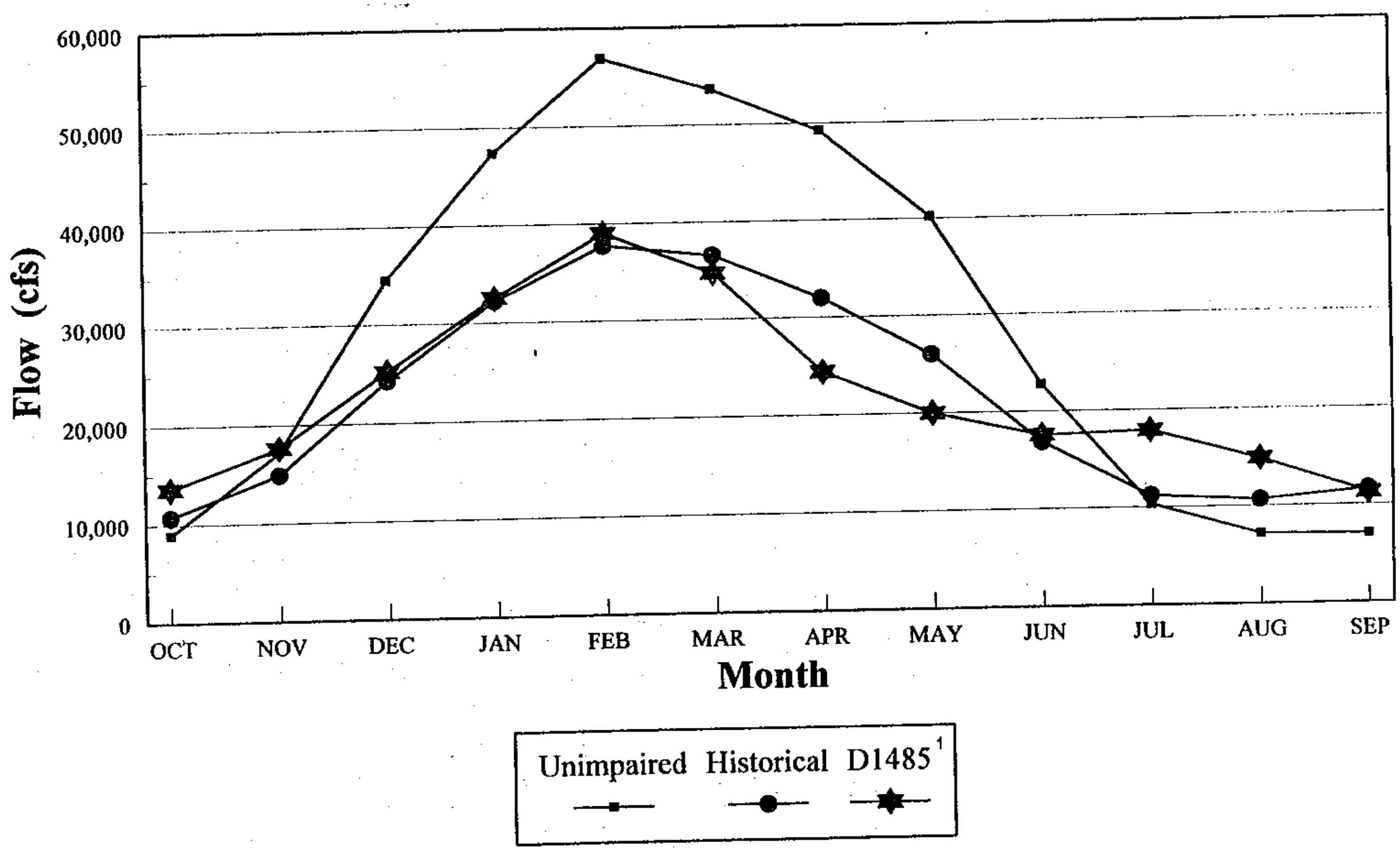
The Sacramento River basin inflow to the Delta comes from four major river systems: the Sacramento, Feather, Yuba, and American. The unimpaired flows from these river systems, often referred to as the Sacramento River Basin Four Rivers Index, represent approximately 47, 25, 13, and 15 percent, respectively, of the total flow from the Sacramento River Basin. Flows to the Delta from the Sacramento River basin are measured at Freeport. Figure V-1 shows average monthly unimpaired, historical, and D-1485 flows for the Sacramento River at Freeport over the 1930-1992 hydrological period. Unimpaired flows are those that would exist in the absence of upstream impoundments and diversions in the presence of existing channel configurations. Historical flows are those that actually occurred and were measured over the historical hydrological period; historical flows reflect upstream impoundments and diversions in the presence of existing channel configurations. D-1485 flows, which were derived from a DWRSIM operation study at the 1995 level of development, are those flows that would have occurred had the flow and operation requirements of D-1485 been implemented over the 63-year hydrological period.

Unimpaired flows in the Sacramento River at Freeport (Figure V-1) were high from January through May and low in July to September until rains began in October or November. Historical and D-1485 flows are below unimpaired flows in wetter months due to upstream diversions and storage, and are higher than unimpaired flows in the drier months due to flow requirements for meeting water quality objectives and export demands.

Figure V - 1

Sacramento River Average Monthly Flow at Freeport

1930-1992 Average Monthly Hydrology



1 - Derived from DWRSIM operation study at 1995 level of development

The San Joaquin River basin inflow to the Delta comes from the San Joaquin, Merced, Tuolumne, and Stanislaus river systems. Peak stream flows above the reservoirs on these streams, which depend on snow melt, typically occur later in spring than in the Sacramento River basin because the San Joaquin River basin mountain ranges are generally higher than those in the Sacramento basin. Flows to the Delta from the San Joaquin River basin are measured at Vernalis. Figure V-2 shows average monthly unimpaired, historical, and D-1485 flows for the San Joaquin River at Vernalis over the 1930-1992 hydrological period. The primary reasons for the differences between annual average unimpaired flows and historical and D-1485 flows are storage in the upstream reservoirs and consumptive water use by San Joaquin Valley agriculture during the irrigation season, which is generally April-September. About 77,000 acres in the San Joaquin River basin have subsurface agricultural drainage systems which discharge to the San Joaquin River, primarily via Mud and Salt sloughs. During the irrigation season, and occasionally following the flushing of the agricultural drainage water from duck clubs in January and February, agricultural drainage makes up a significant portion of San Joaquin River inflow. Low flows of poor quality in the lower San Joaquin River interfere with the upstream migrations of salmon (due to lack of attraction flows, high water temperatures, and low dissolved oxygen) and spawning of striped bass (due to high salinities). The operation of reservoirs on the four major rivers in the San Joaquin River basin has raised flows in September and October above unimpaired flow levels (DFG 1993).

The Central Sierra basin includes the Delta and watersheds of the Mokelumne, Cosumnes, and Calaveras rivers. Inflow to the Delta from this basin comes from the Mokelumne and Cosumnes river systems, sometimes called the "eastside streams" (SWRCB 1988).

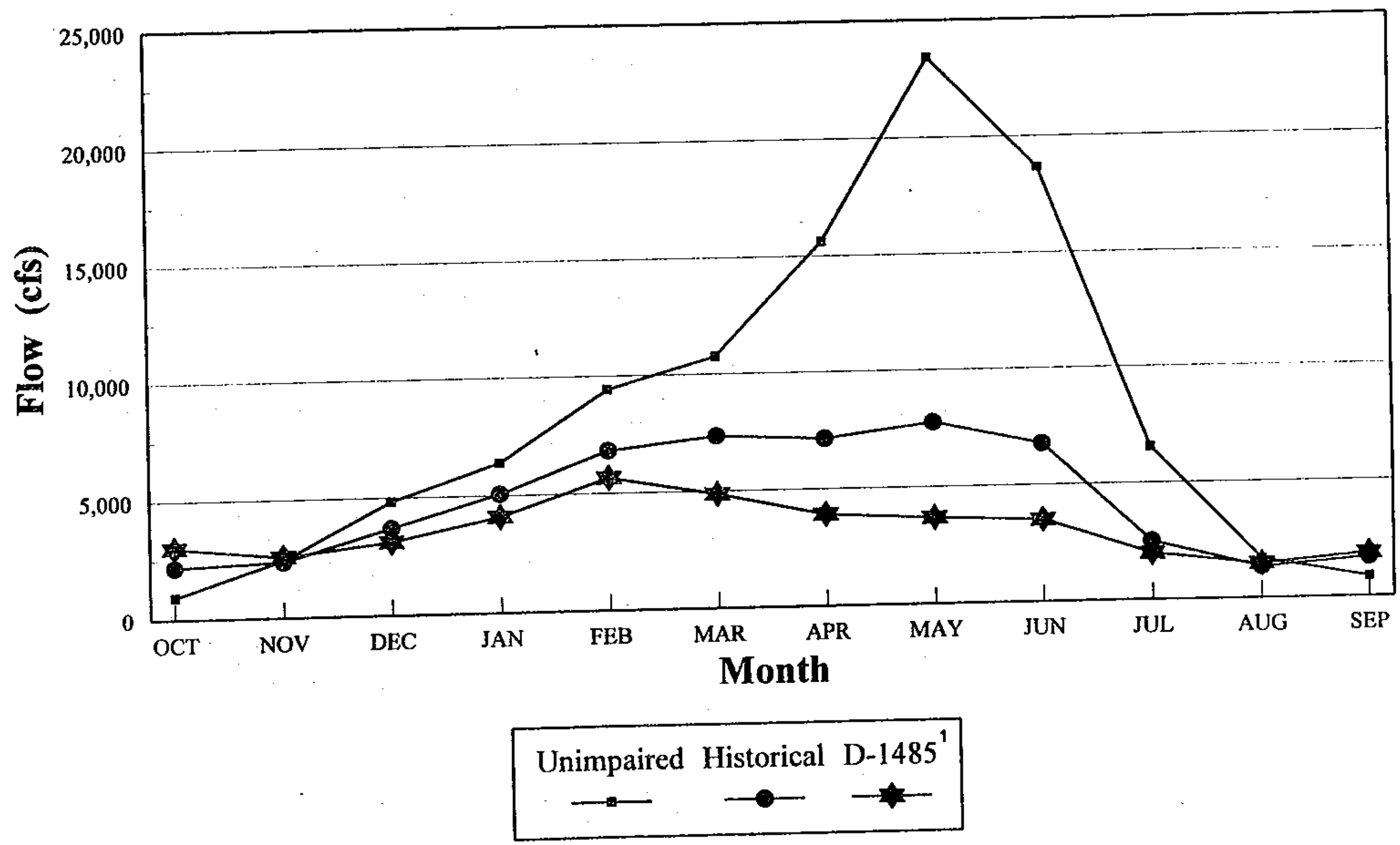
It is evident that water project operations, particularly since 1940, have altered the unimpaired flow conditions by changing the timing of flows and preventing significant volumes of fresh water from reaching the Estuary (SFEP 1992b). The overall effect of water development, in many years, is that inflow to the Delta is generally higher in the summer and early fall and considerably lower during the remainder of the year, particularly in the spring. The effect is less pronounced in wetter years. This disruption of unimpaired inflows to the Delta also contributes to the causes of the declines in aquatic species that are affected by Delta outflow.

c. Delta Cross Channel and Georgiana Slough. The Delta Cross Channel was constructed by the USBR in 1951 to improve water conveyance through the Delta. This gated channel diverts water from the Sacramento River into the eastern Delta channels, including the north and south forks of the Mokelumne River. During periods of high flow in the Sacramento River (above 25,000 cfs at Freeport), the gates are closed to limit flooding in the interior Delta channels. Georgiana Slough, a natural ungated channel located about 1 mile downstream of the Delta Cross Channel, conveys Sacramento River water to the San Joaquin River (DWR 1992a).

Figure V - 2

San Joaquin River Flow at Vernalis

1930-1992 Average Monthly Hydrology



1 - Derived from DWRSIM operation study at 1995 level of development

The Delta Cross Channel and Georgiana Slough can divert fish from the Sacramento River into the central Delta. Up to 70 percent of the Sacramento River flow can be diverted through these two channels. Studies show that fish which migrated through the central Delta experienced a higher mortality rate than those that stayed in the main river channel. Survival of fish released downstream of the gates has been about twice that of fish released above the gates (DWR 1992b, USFWS 1992).

The Delta Cross Channel is not screened to prevent fish from entering the central Delta. An interagency salmon management study concluded that screening the Delta Cross Channel was not a technically feasible alternative (DWR 1992a). Therefore, closure of the Delta Cross Channel gates is the only method available to prevent fish from being diverted into the channel. Investigation into the feasibility of a temporary rock barrier at Georgiana Slough was suspended. Studies are presently underway to determine the feasibility and effectiveness of an acoustic fish barrier to prevent diversion into Georgiana Slough.

d. Delta Outflow. Delta outflow is the calculated amount of fresh water that flows past Chipps Island into Suisun Bay. During this century, the annual depletion in the Estuary's freshwater supply due to upstream and Delta diversions has grown from about 1.5 MAF to nearly 16 MAF. Of the 16 MAF diverted, about 7 MAF are diverted from the Delta for local use and export. These Delta diversions consist of numerous agricultural diversions for Delta farmlands and exports by the CVP and SWP (SFEP 1992b).

About 1,800 unscreened agricultural diversions remove water directly from Delta channels for irrigation and leaching. The volume of water diverted each year for in-Delta farming is significant but has not changed much over the years. Taking into account agricultural return flows, Delta farms deplete Delta outflow by an average of about 960 TAF each year. During the summer, when irrigation of Delta farmlands is at a peak, these agricultural diversions may exceed 4,000 cfs; this is about the same rate at which the CVP removes water from the Delta in the summer (SFEP 1992b).

The two largest diverters of Delta water are the CVP and the SWP. Annual diversions at the CVP's Contra Costa Canal averaged about 35 TAF during the first decade of operations and about 130 TAF in 1987-1989. Major diversions from the Delta began in 1951 with the pumping of water by the CVP's Tracy Pumping Plant to the Delta-Mendota Canal (DMC). The volume of water pumped into the DMC each year has increased from an average of about 700 TAF in the 1950's to more than 2.8 MAF in 1989. In 1989, the total CVP diversion from the Delta through both canals was over 3.0 MAF. Since the SWP's Banks Pumping Plant began operation in 1968, annual SWP Delta diversions have increased steadily, reaching a peak of more than 3 MAF in 1989. In 1990, annual exports of water from the Delta by the CVP and SWP totaled nearly 6 MAF (SFEP 1992b).

Despite the long-term trend of increased annual diversions, there is disagreement concerning whether annual Delta outflow has decreased or increased. It may be that, despite increases in the volume of water diverted, average annual Delta outflows have remained fairly high due to an increasing trend in precipitation and changes in hydrological conditions (e.g., increased runoff from land use changes, water imported from outside the watershed, redistribution of ground water) that have occurred in the watershed since the 1920's. Nevertheless, it is primarily the seasonal pattern of Delta outflow, rather than the average annual volume of Delta outflow, that influences the populations of aquatic organisms which are dependent on the Estuary (SFEP 1992b).

Seasonal flows strongly affect physical variables and biological processes in the Bay-Delta Estuary, such as water temperature, salinity, pollutant concentrations, and the migration and transport of many life stages of organisms (SFEP 1992b). Changes in Delta outflow may affect estuarine and anadromous species by altering the time it takes them to move upstream or downstream. A reduction in transport time may adversely affect Delta species that spawn upstream and depend on currents to carry their eggs and larvae to downstream nursery areas (DWR 1992a). Flows during the months of April, May, and June are especially important for the reproductive success and survival of many species found in the Estuary (SFEP 1992b).

Seasonal trends in Delta outflow are illustrated in Figure V-3. Figure V-3 shows average monthly Delta outflow under unimpaired, historical, and D-1485 conditions (described above for Figures V-1 and V-2) over the 1930-1992 hydrological period. A comparison of the unimpaired Delta outflow to the historical and D-1485 levels of Delta outflow reveals that water development has drastically altered seasonal Delta outflow. Water storage and diversions generally reduce Delta outflow in every month except September and October. The reduction in outflow is especially pronounced in April, May, and June, when flows are critical for aquatic resources in the Bay-Delta Estuary. Therefore, it is widely held that the reduction of spring outflow is one of the most significant adverse impacts of water development on aquatic resources in the Estuary.

Entrapment Zone. In addition to water quality and migration/transport factors, Delta outflow, in part, influences the location of low salinity habitat in the Estuary. Understanding of the low salinity habitat and hydrodynamics of the Estuary, and their influence on the estuarine biota, is continually evolving.

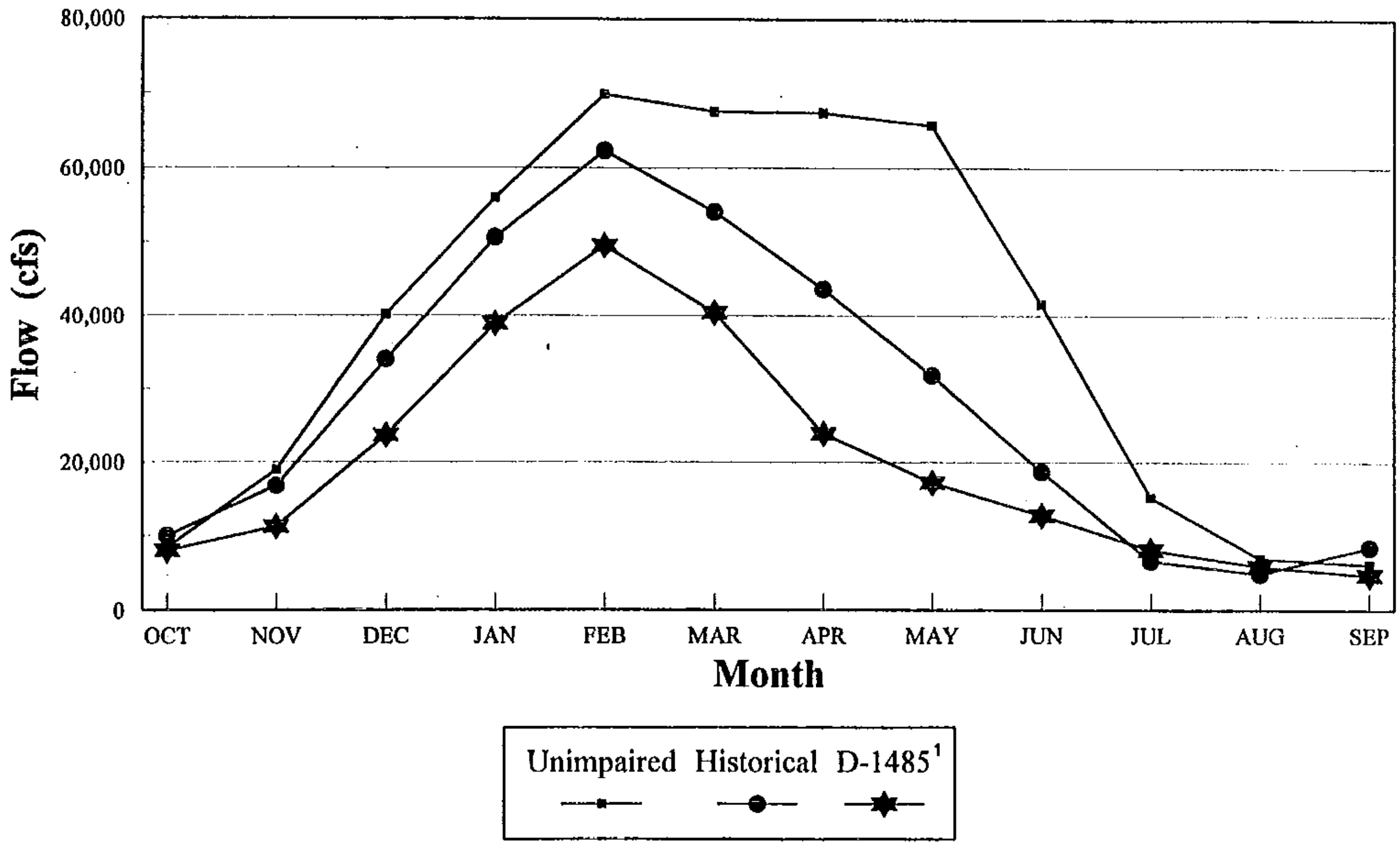
Conventional thought has been that the entrapment zone is a transient region of the Estuary where freshwater and saltwater flows interact, creating relatively low salinity habitat. The entrapment zone was believed to form principally as a result of two-layered flow, resulting in elevated concentrations of particulate matter. As fresh water flowed downstream over the more dense, landward-flowing salt water, some of the water in each layer moved vertically due to frictional forces between the layers. The combination of vertical mixing between the fresh- and salt-water layers, and the horizontal flows within these layers would trap particles with certain settling velocities.

In 1994, the USGS, with others, investigated the relationships between low salinity habitat, hydrodynamics, suspended sediment, and biology, and found evidence that disrupted the

Figure V - 3

Delta Average Monthly Outflow

1930-1992 Average Monthly Hydrology



1 - Derived from DWRSIM operation study at 1995 level of development

theory of the entrapment zone. They found that two-layered flow, called gravitational circulation, occurred in the fall, but did not occur in the spring of that year. Also, they found that gravitational circulation does not necessarily occur just downstream of the location of low salinities, but can occur much farther downstream. Therefore, gravitational circulation is not necessarily associated with the low salinity habitat. Additional investigation is continuing in 1995 to further discover the relationships between the hydrodynamics, salinity, and the distribution and abundance of the biota of the Estuary (Jon Burau, USGS, pers. comm., March 1995). (Because this discovery is very new, many recent publications cited in the following sections refer to the entrapment zone, without making a distinction between low salinity habitat and gravitational circulation.)

Nonetheless, freshwater outflows and antecedent conditions determine the location of the low salinity habitat in the Estuary. Other factors, including exports and upstream reservoir operations, may alter the location of the low salinity habitat. The location and size of low salinity habitat are also affected by the magnitude of tidal flow, bottom topography, and wind (DWR 1992a).

The entrapment zone provides habitat for species that reside in or near it, and may also serve as a food supply region for consumer species such as zooplankton and fish. It has been found to contain elevated concentrations of juvenile striped bass and some species of phytoplankton and zooplankton (SFEP 1992b).

Phytoplankton production is increased with increased outflow, in general. The phytoplankton growth rate is influenced by the location of the zone of gravitational circulation. When gravitational circulation is farther downstream, the phytoplankton have a longer residence time in the shoals and, therefore, a higher growth rate. Within the zone of gravitational circulation, phytoplankton production is decreased because of increased turbidity. Phytoplankton biomass is highest when gravitational circulation is adjacent to the shoal areas, in San Pablo and Suisun bays, due to the exchange of phytoplankton cells from the shoals (where productivity is highest) to the zone of gravitational circulation (driven by winds and tidal exchange), which then traps the cells and accounts for the higher biomass (but not higher productivity). In Suisun Bay, Delta outflows in the 5,000 to 8,000 cfs range historically have been associated with maximum phytoplankton production (SFEP 1992b, DWR 1995c).

An operational definition based on 2 ppt salinity measured on the bottom (commonly known as X2) has been used to define the approximate location of the upstream edge of the entrapment zone in the Estuary. Relationships between measures of abundance for certain aquatic species and entrapment zone position indicate that, when X2 is upstream, annual abundance indices are lower (DWR 1992a). For certain other species, this is not the case (CUWA 1994). Figure V-4 shows the entrapment zone position from 1972-1989 relative to the Golden Gate Bridge, and Figure V-5 shows the relationship between entrapment zone position and Delta outflow (Kimmerer 1992).

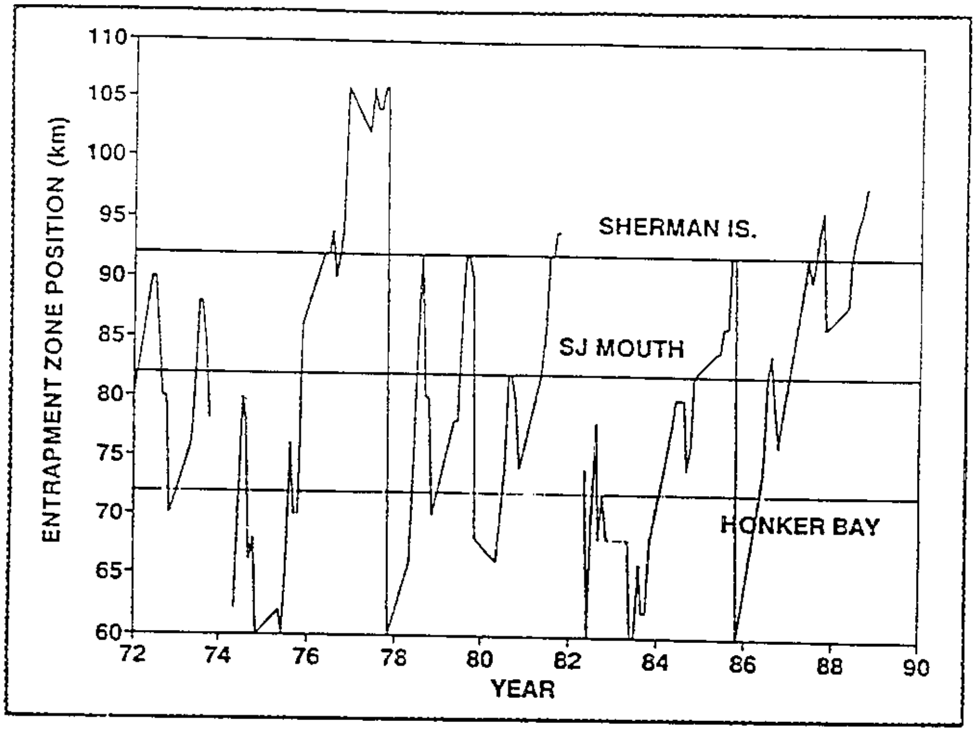


Figure V-4. **Entrapment zone position (kilometers from the Golden Gate) versus time. (Source: Kimmerer 1992)**

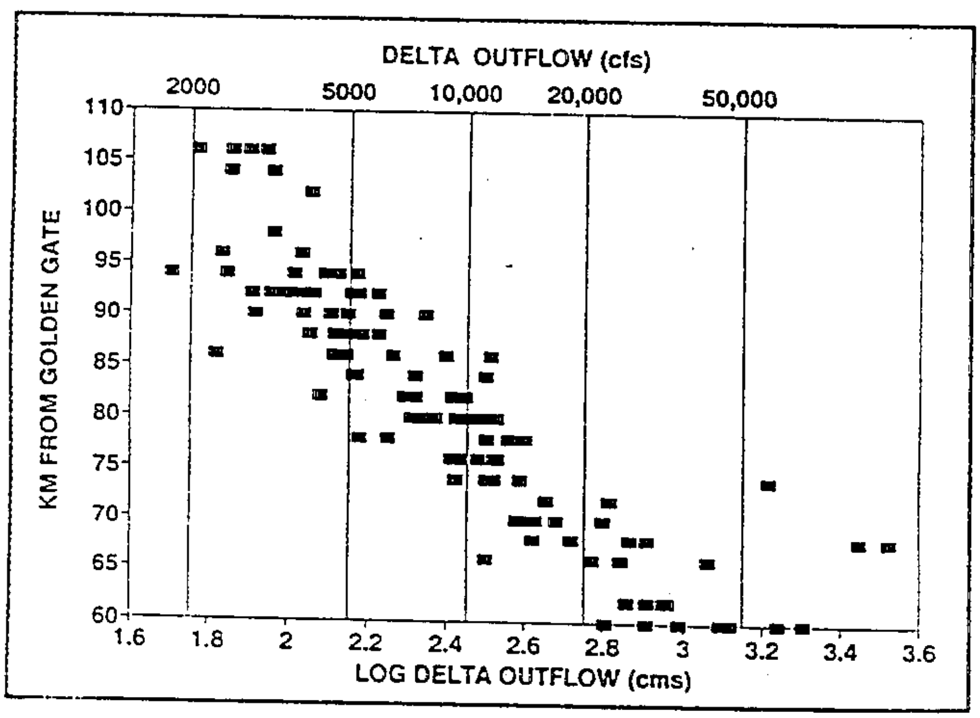


Figure V-5. **Entrapment zone position (kilometers from the Golden Gate) versus log Delta outflow (shown in cubic meters per second and cubic feet per second). (Source: Kimmerer 1992)**

Whether it is actually salinity or outflow that influences the abundance of certain species, and whether it is more effective to regulate one or the other, have been issues of much discussion. The DFG's submittal to the Bay-Delta Oversight Council (BDOC) states that the evidence indicates that the biological phenomena of primary interest are driven by flow rather than salinity (BDOC 1994).

e. Delta Diversions and Entrainment. Each year, as Delta water is diverted to SWP and CVP aqueducts and to Delta farmlands, millions of fish eggs, larvae, and juveniles are diverted, or entrained, as well. Delta diversions also remove nutrients, phytoplankton, zooplankton, and higher organisms from the Estuary; however, the impacts of such entrainment are not well understood (SFEP 1992b).

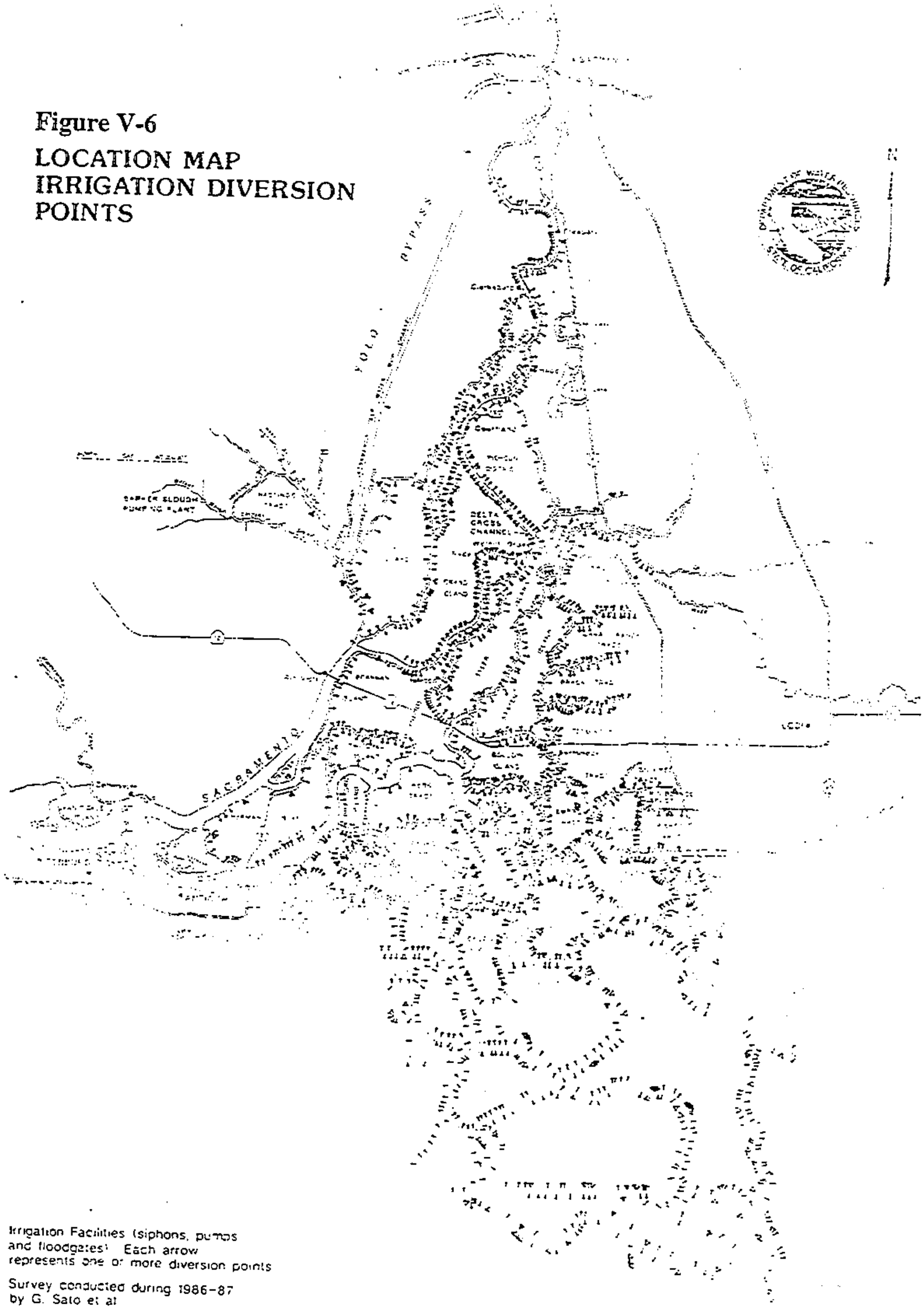
The State and federal pumps are screened to minimize the passage of juvenile and adult fish; however, neither the SWP nor the CVP is able to prevent removal of the millions of fish eggs and larvae that are pulled from Delta channels. Of the approximately 1,800 siphons and pumps that divert water to Delta farms (Figure V-6), at least one, and maybe six, is screened to prevent the removal of fish from the channel. The one that is known to be screened is a 16-inch siphon on Bacon Island. The effects of this diversion and the efficiency of the screen are being studied by the DWR under the IEP's Agricultural Diversion Study (CUWA 1994, SFEP 1992b, DWR 1995c).

The export operations of the CVP and SWP draw water and fish out of the central and southern Delta. The term "entrainment" is used to describe the situation of fish having entered the projects' facilities. At the CVP, fish are entrained when they approach the log boom and trashrack; at the SWP, entrainment begins when fish enter Clifton Court Forebay. The term "loss" is used to identify those fish which do not survive the entrainment and salvage process. The salvage process is the successful recovery of fish entrained at the CVP and SWP fish collecting facilities (the Tracy and Skinner fish facilities, respectively). These facilities use louver fish screens to separate the fish from the water being exported. The fish that are separated from the diverted water are diverted into holding tanks. The fish are then trucked to the western Delta, beyond the immediate influence of the pumps, and released. The SWP screens are relatively efficient for larger fish; however, they are not efficient for small fish less than about 38 mm (DFG 1987).

Clifton Court Forebay at the SWP export facility causes increased losses before the fish get to the fish screen (DFG 1987), primarily due to predation (DWR and USBR 1993). It has been estimated that 75 percent of entrained fish will be lost crossing the forebay (DWR 1992a). Pre-screening losses of entrained salmon are estimated at 75 percent for the SWP and 15 percent for the CVP. Estimates of predation or efficiency of louver screens for other fish are not available (DWR and USBR 1993).

Other factors that contribute to mortality associated with SWP and CVP exports include: size of fish, water velocities at the screens, and handling and trucking losses associated with the salvage operation. Since it is impossible to count all the salvaged fish, estimates are

Figure V-6
LOCATION MAP
IRRIGATION DIVERSION POINTS



– Irrigation Facilities (siphons, pumps and floodgates). Each arrow represents one or more diversion points

Survey conducted during 1986-87 by G. Sato et al

Source: California Department of Water Resources made by subsampling periodically during the day and extrapolating the results to the entire day, which results in large but uncalculated errors. The DFG assumed control of the counting and salvage operations in 1992 and salvage data prior to 1980 are generally not used (DWR 1992a, 1992b).

Salvage records from the SWP and CVP indicate that fish are entrained year-round with peaks for various species occurring during the period that a particular life stage is vulnerable to the export pumps (DFG 1987). Pumping losses at the SWP and CVP facilities are a significant cause of mortality for many species of fish. During 1976 through 1986, pumping operations killed an annual average of 6.5 million juvenile striped bass greater than 20 mm in length. This includes a 15 percent loss rate to predators in front of the fish screens, losses to entrainment, and losses due to handling and trucking (SFEP 1992b). Estimated chinook salmon salvaged, which does not include those lost to predation and handling mortality, between 1981 and 1992 averaged 54,007 at the SWP and 79,197 at the CVP (DWR 1992a). Virtually all the species found in the Delta are salvaged during some portion of the year at the export pumps. Table V-2 shows estimated numbers of all species of fish salvaged at the CVP's Tracy fish facility and the SWP's Skinner fish facility in 1986, 1989, and 1992.

In addition to losses at the SWP and CVP pumps, agricultural diversions may well account for significant fish losses in the Delta. The peak agricultural diversion season in the Delta is April through August, coinciding with months when large number of young chinook salmon, striped bass, American shad, Delta smelt and other fish are present. The estimated total average diversion rate from Delta channels during the growing period ranges from 2,500 to 5,000 cfs (DWR 1992a). The annual removal of water from these diversions is estimated at about 2.3 MAF (NHI 1992a). It is estimated that several hundred million striped bass (less than 16 mm long), as well as tens of thousands of juvenile chinook salmon, are lost to agricultural diversions. Agricultural diversions impact Delta fish, but the magnitude of the impact is unknown (DWR 1992b). However, it is also possible that aquatic organisms have increased exposure to these diversions due to changes in flow patterns in the Delta caused by CVP and SWP pumping (NHI 1992a).

The Pacific Gas and Electric (PG&E) Company power generating facilities in the Estuary, at Pittsburg and Antioch, entrain fish less than about 38 mm in size and impinge larger fish with the intake of cooling water. Entrainment for some fish, particularly striped bass, may not be fatal. As mitigation for these losses, PG&E releases striped bass in the Estuary.

It is not certain how the operation of these facilities has affected the fish populations of the upper Estuary over the past 20 years. The available information suggests that larval and juvenile smelt of the family Osmeridae were historically one of the most abundant fish taxa in the area. PG&E, during the period of peak striped bass entrainment (May to mid-July), operates the power generation units based on fish monitoring data. This program has reduced entrainment losses of larval and juvenile striped bass by more than 75 percent. Incidental benefits to other species may be occurring as well (NHI 1992a, DWR and USBR 1994).

TABLE V-2

ESTIMATED NUMBERS OF FISH SALVAGED AT TRACY AND SKINNER FISH FACILITIES FOR THE YEARS 1986, 1989, AND 1992[1]

Fish	1986	Fish	1989	Fish	1992
Striped Bass	18,544,652	Striped Bass	10,549,877	Striped Bass	4,411,064
Sacramento Splittail *	2,391,588	Sacramento Splittail *	60,584	Sacramento Splittail *	12,082
Threadfin Shad	1,763,815	Threadfin Shad	315,867	Threadfin Shad	1,291,772
Chinook Salmon *	1,187,272	Chinook Salmon *	149,196	Chinook Salmon *	63,878
American Shad	1,139,342	American Shad	644,696	American Shad	710,154
White Catfish	997,009	White Catfish	320,621	White Catfish	228,350
Yellowfin Goby	777,627	Yellowfin Goby	283,921	Yellowfin Goby	77,355
Channel Catfish	384,309	Channel Catfish	18,475	Channel Catfish	36,636
Inland Silverside	64,689	Inland Silverside	47,363	Inland Silverside	115,595
Prickly Sculpin *	37,160	Prickly Sculpin *	54,655	Prickly Sculpin *	14,903
Bluegill	30,508	Bluegill	11,286	Bluegill	22,437
Lampreys (all spp.) *	17,023	Lampreys (all spp.) *	1,418	Lampreys (all spp.) *	1,592
Sacramento Blackfish *	11,171	Sacramento Blackfish *	18	Sacramento Blackfish *	154
Black Crappie	9,877	Black Crappie	5,487	Black Crappie	5,394
Bigscale Logperch	8,380	Bigscale Logperch	9,929	Bigscale Logperch	5,488
Mosquitofish	7,711	Mosquitofish	480	Mosquitofish	1,047
Delta Smelt *	6,380	Delta Smelt *	20,074	Delta Smelt *	6,178
Tule Perch *	5,507	Tule Perch *	5,756	Tule Perch *	3,159
Miscellaneous	4,836	Miscellaneous	4,387	Miscellaneous	25,557
Steelhead Rainbow Trout	4,746	Steelhead Rainbow Trout	17,475	Steelhead Rainbow Trout	18,745
Warmouth	3,998	Warmouth	494	Warmouth	266
Riffle Sculpin *	3,648	Riffle Sculpin *	0	Riffle Sculpin *	1,767
Goldfish	2,978	Goldfish	0	Goldfish	0
Carp	2,496	Carp	431	Carp	238
Hardhead *	2,422	Hardhead *	0	Hardhead *	4
Longfin Smelt *	2,296	Longfin Smelt *	67,545	Longfin Smelt *	3,590
Golden Shiner	2,050	Golden Shiner	1,148	Golden Shiner	4,861
Green Sunfish	1,788	Green Sunfish	0	Green Sunfish	108
Largemouth Bass	991	Largemouth Bass	1,045	Largemouth Bass	19,704
Staghorn Sculpin *	929	Staghorn Sculpin *	1,455	Staghorn Sculpin *	295
Redear Sunfish	828	Redear Sunfish -	122	Redear Sunfish	276
Starry Flounder *	758	Starry Flounder *	3	Starry Flounder *	108
Yellow Bullhead	755	Yellow Bullhead	0	Yellow Bullhead	71
White Sturgeon *	666	White Sturgeon *	17	White Sturgeon *	62
Black Bullhead	502	Black Bullhead	258	Black Bullhead	155
Pumkinseed	249	Pumkinseed	0	Pumkinseed	86
Smallmouth Bass	209	Smallmouth Bass	0	Smallmouth Bass	498
White Crappie	191	White Crappie	0	White Crappie	928
Sacramento Perch *	187	Sacramento Perch *	0	Sacramento Perch *	0
Sacramento Sucker *	121	Sacramento Sucker *	0	Sacramento Sucker *	0
Green Sturgeon	49	Green Sturgeon	0	Green Sturgeon	164
Hitch *	48	Hitch *	0	Hitch *	0
Brown Bullhead	34	Brown Bullhead	364	Brown Bullhead	546
Blue Catfish	28	Blue Catfish	7,199	Blue Catfish	72
White Bass	0	White Bass	0	White Bass	18
Chameleon Goby	0	Chameleon Goby	13,020	Chameleon Goby	22,307
Total	**27,421,823**	**Total**	**12,614,666**	**Total**	**7,107,664**
Percent natives	**13.4%**	**Percent natives**	**3.0%**	**Percent natives**	**1.8%**
Percent introduced	**86.6%**	**Percent introduced**	**97.0%**	**Percent introduced**	**98.2%**

* Native Species

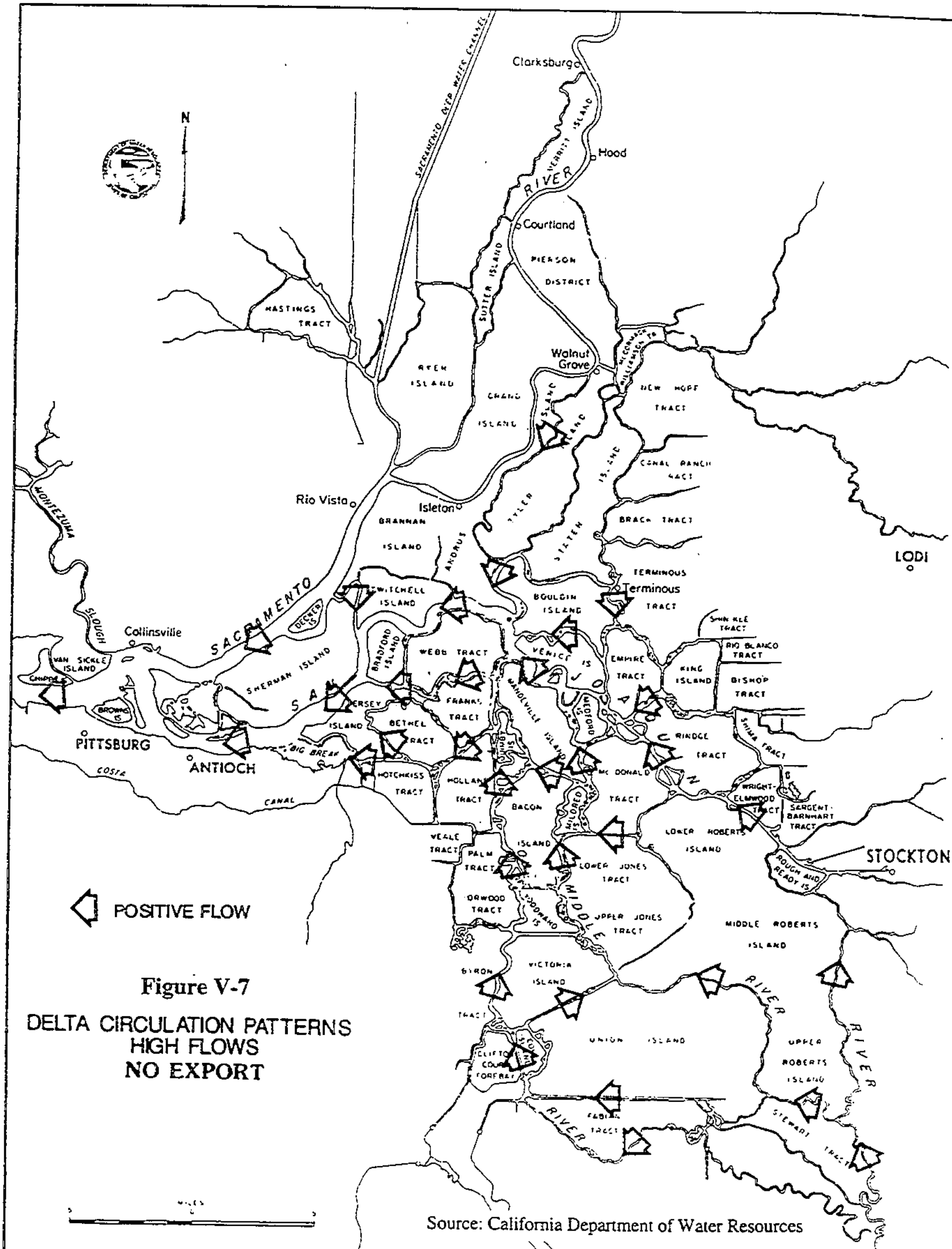
1 Fish and Game Bulletin Board, "Fish Facilities Salvage Project California Department of Fish & Game", Phone No. 1-209-948-7347.

f. <u>Reverse Flows</u>. Tidal flows dominate water movement in this Estuary. In the western Delta the average tidal flow is 180,000 cfs and ranges from -300,000 to +300,000 cfs twice daily. The concept of reverse flows deals with net flow during the day in the same way the Delta outflow is a calculated net daily flow. The importance of reverse flow is controversial and is presented here for completeness.

Water supplies for CVP and SWP exports are obtained from Delta inflow. Typically, when export rates are high and inflow is low, Sacramento River water is pulled in an upstream direction around Sherman Island, at the confluence with the San Joaquin River. As water travels around Sherman Island, it mixes with saltier ocean water entering as tidal inflow, and is drawn upstream into the San Joaquin River and other channels that feed the CVP and SWP pumping plants. This situation, which causes a net upstream flow of water in the lower San Joaquin River toward the export pumps, is known as reverse flow. During periods of high Delta inflow and high export, there is some reverse flow, but enough water is available from the San Joaquin River, the Central Sierra Basin (eastside streams), and the Sacramento River via the Delta Cross Channel to meet export demands. Figures V-7 and V-8 show the net direction of normal (high flows, no exports) and reverse (low flows, high exports) flows, respectively.

The hydraulic capacities of the Delta Cross Channel and Georgiana Slough provide a physical limitation to the quantity of Sacramento River water that can be moved toward the SWP and CVP pumping plants in the southern Delta. These physical constraints cause reverse flows when pumping plus internal Delta demand exceeds the sum of cross-Delta flows and San Joaquin River inflows (DWR 1992a).

Reverse flows reportedly disorient anadromous fish as they migrate either upstream or downstream following the salinity gradient. The USFWS reported a weak relationship between salmon smolt survival and QWEST (USFWS 1992). QWEST is an index of San Joaquin River flow which serves as an indicator of reverse flows conditions; QWEST is calculated by subtracting Delta exports and 65 percent (representing the Delta channel depletion that occurs in the central and southern Delta areas) of net Delta consumptive use from central Delta inflow. CUWA (1994) reviewed the literature describing the effects of reverse flows on fish. According to this review, reverse flows may influence the number of fish lost via entrainment into the CVP and the SWP pumping plants. Reverse flows may carry young fish into the central or southern Delta, where habitat may not be as good or where they may be more susceptible to entrainment at local agricultural, municipal, and industrial diversions, and to SWP and CVP exports (DWR 1992a). Table V-3 shows the months during the period from 1978 to 1989 in which the average calculated flow, QWEST, was negative. As the drought continued, the numbers of months with reverse flows increased.



Source: California Department of Water Resources

Figure V-7

DELTA CIRCULATION PATTERNS
HIGH FLOWS
NO EXPORT

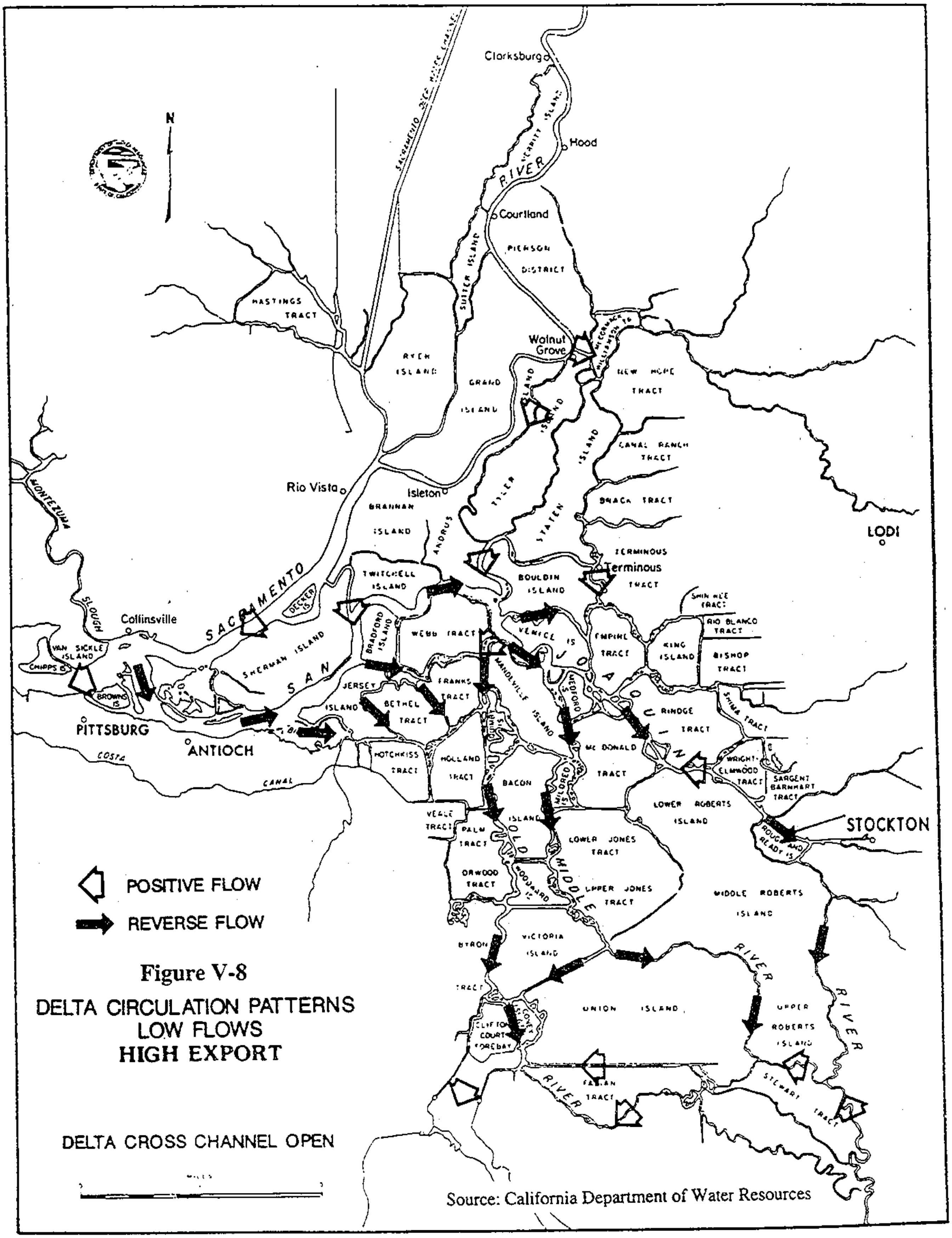


Figure V-8

DELTA CIRCULATION PATTERNS
LOW FLOWS
HIGH EXPORT

Source: California Department of Water Resources

Table V-3. Months during water years 1978-1989 in which the average calculated net flow on the San Joaquin River past Jersey Point (QWEST) was negative.

Water Year	Months with Negative Flow	Water Year	Months with Negative Flow
1978	July-August	1984	July-August
1979	July-September	1985	July-December
1980	August, November	1986	July-September
1981	April, July-September	1987	January, June-December
1982	None	1988	All but April, November, December
1983	None	1989	All but March

Source: DWR (1992b)

QWEST has been used as a regulatory parameter to limit movement of winter-run chinook salmon and Delta smelt toward the CVP and the SWP export pumps. The use of QWEST is partly driven by the perception that the transport of small fish is largely dictated by QWEST. This issue is being examined because there is some evidence that QWEST is not a good indicator of entrainment losses in the interior Delta. The DWR Particle Tracking Model indicates that the export pumps have a "zone of influence" and a large percentage of modeled particles (assumed to represent young fish) within it are likely to be entrained into the CVP and the SWP facilities regardless of QWEST. Further model studies are being designed to characterize the zone of influence (DWR and USBR 1993).

3. Introduced Species

The Bay-Delta Estuary is home to more than 150 introduced aquatic species of plants and animals. About 28 of these species are non-native fish and over 100 species are non-native invertebrates (BDOC 1994). A list of the more notable introduced species in the Estuary is presented in Table V-4.

Between 1850 (when documentation of introductions of organisms to the Estuary began) and 1950, some introductions were deliberate attempts to diversify the native fish fauna of the Estuary. Intentional introductions by government agencies occurred when species such as striped bass, American shad, carp, eastern oyster, and Japanese oyster were introduced to expand the opportunities for angling, commercial fishing, or aquaculture. Species such as threadfin shad were introduced to increase the forage base for predators, and mosquitofish were introduced in an effort to control pest populations (BDOC 1994, DWR 1992a).

Table V-4. Introduced species (and dates of introduction) in the Bay-Delta Estuary.

Year	Common Name	Scientific Name
1850	Isopod	*Sphaeroma quoyanum*
1869	Eastern Oyster	*Crassostrea virginica*
1871	American Shad	*Alosa sapidissima*
1872	Carp	*Cyprinus carpio*
1873	Gribbles	*Limnoria* spp.
1874	Black Bullhead	*Ictalurus melas*
1874	Brown Bullhead	*Ictalurus nebulosus*
1874	Largemouth Bass	*Micropterus salmoides*
1874	Soft Shell Clam	*Mya arenaria*
1874	White Catfish	*Ictalurus catus*
1874	Yellow Bullhead	*Ictalurus natalis*
1879	Striped Bass	*Morone saxatillis*
1891	Golden Shiner	*Notemigonus crysoleucas*
1891	Green Sunfish	*Lepomis cyanellus*
1900	Goldfish	*Carassius auratus*
1908	Black Crappie	*Pomoxis nigromaculatus*
1908	Bluegill	*Lepomis macrochirus*
1913	Shipworm	*Teredo navalis*
1921	Warmouth	*Lepomis gulosus*
1922	Mosquitofish	*Gambusia affinis*
1930	Japanese Oyster	*Crassostrea gigas*
1940	Channel Catfish	*Ictalurus punctatus*
1946	Asian Clam	*Corbicula fluminea*
1946	Japanese Littleneck	*Tapes japonica*
1949	Redear Sunfish	*Lepomis microlophus*
1950	Fathead Minnow	*Pimephales promelas*
1951	White Crappie	*Pomoxis annularis*
1953	Bigscale Logperch	*Percina macrolepida*
1953	Threadfin Shad	*Dorosoma petenense*
1963	Yellowfin Goby	*Acanthogobius flavimanus*
1966	Copepod	*Oithona davisae*
1968	Inland Silverside	*Menidia beryllina*
1968	Snail	*Littorina littorea*
1978	Copepod	*Sinocalanus doerrii*
1979	Blue Catfish	*Ictalurus furcatis*
1979	Copepod	*Limnoithona sinensis*
1982	Clam	*Theora fragilis*
1983	Amphipod	*Gammarus daiberi*
1986	Asian Clam	*Potomocorbula amurensis*
1986	Crustacean	*Hemileucon hinumennsis*
1986	Copepod	*Pseudodiaptomus marinus*
1987	Copepod	*Pseudodiaptomus forbesi*
1988	Snail	*Malanoides tuberculata*
1989	Polychaete	*Potamilla* sp.
1991	European Green Crab	*Carcinus maenas*
1991	Polychaete	*Spionid* sp.

Source: CUWA (1994)

The inland silverside, which was transported to the Estuary in runoff from Clear Lake where it was introduced in an attempt to control gnats (DFG 1994b), is the only known unauthorized deliberate introduction of a fish in California (BDOC 1994).

Although intentional introductions to the Bay-Delta Estuary have decreased since 1950, accidental introductions probably have not (DFG 1994b). Accidental introductions in the Estuary have occurred incidental to other activities. Many early introductions of invertebrate species occurred incidental to the intentional transplanting of oysters in the 1870's and early-1900's. Most recent introductions of aquatic species, such as yellowfin goby, chameleon goby, and many invertebrates, have occurred when ballast water from ships was released into the Estuary (BDOC 1994).

As a result of intentional and unintentional introductions, aquatic resources in the Bay-Delta Estuary have changed dramatically. Introduced species which become established due to favorable conditions can affect native species through a wide variety of mechanisms, including: competition for food and space; predation; habitat alteration; disturbance; hybridization; and acting as pathways for and sources of diseases (BDOC 1994).

The successful establishment of non-native organisms has greatly altered the relative abundance and composition of species in the Estuary. For fish, a shift from native to introduced species has been more pronounced in the freshwater portions of the Estuary (DFG 1994b). The SWP's fish salvage facilities, which are probably the best sampler of Delta biota (DWR 1994), produce data which illustrate the relative abundances of native and introduced fish.

In 1980, 17 of the 30 species salvaged at the SWP fish screens were introduced, with 13 of them having been introduced prior to 1950. Data for 1986, 1989, and 1992 indicate that 29 of the 45 identified species salvaged were introduced species (see Table V-2, above). In 1986, 1989, and 1992, introduced species comprised 86.6, 97.0, and 98.2 percent, respectively, of the total number of identified organisms salvaged. This indicates that introduced fish species are becoming increasingly more numerous relative to native fish in the Estuary.

Changes in the composition of the Estuary's invertebrates have been more dramatic than those for fish. Several new species of zooplankton have significantly changed the species composition in the brackish and freshwater portions of the Estuary. For example, two introduced copepods, *Pseudodiaptomus forbesi* and *Sinocalanus doerrii*, have largely replaced the once dominant native copepod, *Eurytemora affinis*, which had been the principal food for young fish. The establishment of the highly efficient, filter-feeding Asian clam, *Potamocorbula amurensis*, in San Pablo and Suisun bays has also been identified as a factor in the decline of *Eurytemora* and the shift in the composition of benthic organisms in these portions of the Estuary (CUWA 1994, DFG 1994b, NHI 1992a). Another species of Asian clam, *Corbicula fluminea*, has become the dominant mollusk in the Delta since its introduction in 1946. Today, all but two of the common benthic mollusks in the Estuary are introduced species (CUWA 1994).

The introduction of aquatic plants also impacts the estuarine ecosystem. For example, the water hyacinth, *Eichhornia crassipes,* creates dense mats of vegetation that clog screens, block light, causing rooted submergent plants to die and shading phytoplankton, and provide cover for fish predators. Aquatic weeds can also increase siltation and affect water temperature and dissolved oxygen levels. Invasive introduced terrestrial plants can displace native plants and affect the habitat structure of the wetland habitat of the Estuary. For example, the eastern cordgrass, *Spartina alterniflora*, was introduced through a salt marsh restoration project in the Bay Area to mitigate for loss of wetlands. It has since spread and established itself in the higher and lower areas in the tidal zones. It is prolific, out-competing the native cordgrass and turning mudflat areas into cordgrass islands. Although it can provide additional habitat for such species as the endangered clapper rail, it diminishes mudflat communities which provide important food source for shorebirds (BDOC 1994).

The introductions of non-native species in the Bay-Delta Estuary have caused major changes in the fish fauna in the Estuary, particularly in fresh waters; however, the introductions have not coincided with the principal declines in certain fish populations, such as the striped bass and Delta smelt. Although there is no strong empirical case for recent introductions being a principal cause of the declines in some species (DFG 1994), it is likely that the establishment of non-native species in the Estuary has been a contributing factor (NHI 1992a). It is uncertain what effects the introductions may have had on some of the species and whether the introductions may make the recovery of previously abundant native species and striped bass more difficult (BDOC 1994, DFG 1994b). While few opportunities exist to effectively reduce or eliminate introduced species in the Estuary, management activities should focus on preventing additional incidental introductions and on managing the existing composition of species (BDOC 1994).

4. Food Limitations

Food supply is another factor that can affect the abundances of aquatic organisms at all trophic levels. Food may be limited in various ways, including decreased availability of nutrients, and decreased abundance and availability of food items.

Some scientists believe that a decrease in nutrients, which support the base of food webs (primarily phytoplankton), has contributed to declines in the aquatic resources of the Bay-Delta Estuary. Building dams, leveeing river channels, and diking and filling tidal wetlands have reduced the loadings of land-derived detritus, a primary nutrient source, to the Estuary (DWR 1994). Corresponding increases in water clarity may have resulted in the aperiodic blooms of the diatom, *Melosira granulata*, which is difficult for zooplankton to graze upon (NHI 1992a). In addition, reduced loadings of urban organic waste through increased treatment over the past 40 years may have also removed an important nutrient source for the base of the Estuary's food web (DWR 1994). Decreased sewage may have had a significant adverse effect on the estuarine biota, particularly in the upper Estuary. In fact, any nutrient contribution to the food web may have been cancelled by the effects of toxic pollutants associated with the sewage (NHI 1992a), both of which have now been greatly reduced.

Declines in the populations of phytoplankton, zooplankton, and fish have occurred at about the same time; however, food limitation has not yet been identified as a cause. Although zooplankton are a primary food for several species of fish, the studies that have been done to document food limitation have not been able to document such a phenomenon.

Most studies on the effects of food supply have been on striped bass. The copepod *Eurytemora affinis*, which is an important initial food for striped bass, declined following the introduction of non-native invertebrates. Although studies on food supply and striped bass production have shown that some degree of food limitation exists (probably through slowing growth and, thus, increasing mortality rates), no direct evidence of starvation of bass has been found. Young striped bass changed their diet when a newly-introduced amphipod, *Gammarus diaberi*, became a major food item for young striped bass and may have minimized the impact of reduction in *Eurytemora* (BDOC 1994). In feeding experiments, striped bass larvae, when they first start to feed, are much more adept at capturing the native *Eurytemora* and *Cyclops* than they are at capturing an introduced species of copepod, *Sinocalanus* (which have more effective escape responses). Histological analysis of striped bass larvae collected from the wild has failed to show any signs of starvation (SFEP 1992a). Although the composition of prey species has changed, no general relationships have been found between food supply and bass mortality. The changes in prey items, therefore, do not appear to be a major factor contributing to the decline in striped bass; however, it might inhibit the recovery of other fish species (BDOC 1994).

5. Land Reclamation and Waterway Modification

Land reclamation and waterway modification have caused major ecological changes both in the Estuary and throughout the Central Valley. They have destroyed most of the tidal marshes in the Estuary and the seasonally-flooded wetlands upstream of the Estuary. The vast majority of land reclamation occurred before 1920, so there is essentially no information available to estimate its consequences (DFG 1994b). Only about 3 percent of the historical acreage of wetlands (estimated at 545,000 acres) remains today, with most being reclaimed for agriculture (CUWA 1994).

An impact of the loss of wetland habitat is the reduced population sizes of fish, especially those that utilize shallow, back-water habitats, sloughs, and intertidal zones during all or part of their life cycle. Species that utilize flooded vegetation for spawning habitat have either gone extinct or have declined in abundance (CUWA 1994; DFG 1994b). The losses of habitat that have occurred throughout the Delta have probably reduced the resiliency of certain populations to respond to natural and man-induced perturbations, setting the stage for the declines in certain species. Marsh and other wetland habitat losses must be considered as one of the major factors that have served to shape and control existing populations (CUWA 1994).

The earliest, and probably most profound, cause of change in aquatic habitat in the Bay-Delta Estuary was the introduction of European methods of agriculture into the Central Valley. Diking the rivers and clearing riparian vegetation began to change the lower parts of the valley from seasonal freshwater marsh to dry cropland. Diking of islands in the Delta began in 1852. Dikes, which were constructed of dredged materials from the river or from the interior of the island, consisted of fine river sediments, easily degraded peaty soils, or a combination of both. Such diking led to weak dikes, depressed island interiors, and deeper, more U-shaped channels in the river. Water flows more quickly in dredged channels and the vertical walls are easily eroded (SFEP 1992a).

A secondary effect of diking was to change river habitats and primary productivity. Restriction of water to channels increased water velocity and led to decreased residence times of water in the Estuary, allowing less time for phytoplankton to grow. The transformation of vast areas of freshwater marsh into cropland effectively eliminated the contribution of marsh productivity to downstream food web organisms. Channelization removed the shallow margins of most river channels and prevented the growth of benthic algae (SFEP 1992a).

Almost concurrent with the first diking of Delta islands was the advent of hydraulic gold mining in the Sierras. The main impact of hydraulic mining on downstream sites was the introduction and transport of large quantities of silt. Before hydraulic mining was banned in 1884, an estimated 1.5 billion cubic yards of extra sediment was brought into the Estuary. Although the effects of mining on the aquatic resources of the Estuary are undocumented, the siltation and dewatering of spawning streams undoubtedly devastated salmonid populations (SFEP 1992a). Today, more than 6 million cubic yards of sediments enter the Estuary each year, mostly from the Sacramento and San Joaquin rivers. As many as 286 million cubic yards of existing sediments in the shallows of San Francisco Bay are resuspended by currents and wind-driven waves (SFEP 1992b).

Dredging of bottom sediments in the naturally-shallow Estuary frequently occurs to ensure water depths necessary for navigation and docking, to maintain flood control channel capacities, and for breakwater and bridge construction. The dredging and disposal of estuarine sediments temporarily increase turbidity, influence benthic communities at and near disposal sites, and may affect the behavior and physiology of fish and other organisms. These activities also may redistribute toxic pollutants and increase their availability to aquatic organisms (SFEP 1992b).

Flood control measures, such as alterations to channel configurations, removal of riparian vegetation, placement of rock revetment ("rip-rap") to reduce erosion, and construction of concrete channels, also adversely affect fish and wildlife habitat in the Estuary's tributaries. In the Delta, levee maintenance standards affect habitat conditions by limiting the extent of vegetation allowed on the levees (SFEP 1992b). The construction and maintenance of reclamation and flood control levees have also reduced detrital loading and the amount of shoal and wetland areas (DWR 1994).

Perhaps the most important and far-reaching aspect of waterway modification is the rise in sea level. Around the Bay-Delta Estuary, the relative increase in sea level will be even greater on low-lying lands where sediment-deposited soils are expected to subside from soil compaction and consolidation. For example, by the year 2037, the relative mean water level in Central Bay at Sausalito is projected to increase 0.3 to 0.48 feet above mean sea level; in South Bay at Alviso Slough, where greater land subsidence is expected, the relative mean water level is projected to rise 0.8 to 5.76 feet above mean sea level. Impacts to the Estuary that are associated with these projected increases include: saltwater intrusion in tidal marshes, freshwater tributaries, and ground water; submergence of tidal marshes in North and South bays; increased periodic flooding of previously protected low-lying areas around the bay and in the Delta; and increased shoreline and beach erosion. These conditions will adversely impact the Estuary's water quality, wetland habitat, and Estuary-dependent human activities (SFEP 1992b).

6. Pollution

The quality of water needed to support populations of freshwater, estuarine, and marine species in the Bay-Delta Estuary is dependent on more than a certain concentration of salinity at various locations. The release of pollutants which adversely affect the physical, chemical, and biological properties of water in the Estuary also impacts aquatic species.

In its natural state, the Bay-Delta Estuary exhibited few, if any, adverse effects of pollutants since the sediments and naturally-occurring chemicals that entered the Estuary from upstream were assimilated. As urban, industrial, and agricultural activities expanded throughout the watershed, pollutant loads and associated impacts to aquatic resources increased. By the end of the 1800's, untreated industrial and sewage wastes adversely affected water quality in many portions of San Francisco Bay. It is believed that pollution contributed to the decline seen in the Estuary's salmon, sturgeon, and striped bass commercial fisheries by the early 1900's (SFEP 1992b).

After World War II, the Bay-Delta Estuary was receiving large and mostly uncontrolled amounts of inadequately untreated sewage, industrial effluent, urban runoff, and agricultural wastes. The most obvious impacts were caused by the discharge of large quantities of nutrients, which resulted in increased biological oxygen demand (BOD) and suspended solids, and decreased dissolved oxygen levels. Efforts to control the effects of sewage in the Estuary were initiated in the early 1950's, when some publicly-owned wastewater treatment plants began primary treatment of municipal wastewater. Construction of facilities to enable secondary treatment, which removes a greater percentage of pollutants than primary treatment, began in the mid-1960's (SFEP 1992b).

With the implementation of the State Porter-Cologne Water Quality Control Act of 1969 and the federal Clean Water Act of 1972, rapid improvements in the quality of municipal and industrial effluent, and of the San Francisco Bay water, occurred in the 1970's (SFEP 1992b). The result of these improvements has been the steady decline of BOD loadings and suspended solids in the bay. It has been suggested that decreasing trends in abundance of the major zooplankton species correspond with the reductions of BOD loadings, which supply nutrients, in the bay (CUWA 1994).

With a decrease in nutrient loading over time, there has been an increase in chemical pollutants. Toxic chemical pollutants, which now pose the greatest pollution threat to the Estuary, include trace elements (e.g., mercury, selenium), organochlorines and other pesticides (e.g., DDT, dioxins), and petroleum hydrocarbons (e.g., benzene, chrysene). Today, 5,000 to 40,000 tons of toxic pollutants enter the Estuary each year. The bulk of these chemicals are carried in runoff from urban areas and farms. Effluent from municipal and industrial outfalls, riverine inputs, dredging and dredge material disposal, atmospheric deposition, accidental spills, marine vessel discharge, and leakage from waste disposal sites contribute the remainder. Although programs are in place to regulate the discharge of pollutants, large quantities of toxic chemicals continue to enter the Estuary (CUWA 1994, SFEP 1992b).

Pollutants are distributed within the Bay-Delta Estuary by a combination of physical, chemical, and biological processes. The loadings and concentrations of pollutants are dependent not only on the direct discharge of pollutants, but also the patterns of chemical use, land development, freshwater flows, and tidal action. Many persistent pollutants (i.e., those which do not degrade or degrade very slowly) become bound to particulate matter that settles near discharge points and accumulates in areas of sediment deposition, together with pollutants from past industrial activities. Although evidence indicates that loading rates of toxic pollutants have declined in the last 20 years, human activities (e.g., dredging) have increased rates of mobilization of toxicants previously discharged into the Estuary. Thus, although some pollutants have been banned, such as DDT and polychlorinated biphenyls (PCB's), or significantly reduced, they continue to pose potential hazards to biota. Some pollutants can become concentrated in organisms directly from the water column and by ingestion of contaminated food. These processes can result in high levels of pollutants in tissues, through bioaccumulation, even when concentrations in the water are low. The effects of selenium in causing deformities in waterfowl are well-known in this regard (SFEP 1991).

Pollutants have a wide range of effects on estuarine organisms, ranging from very subtle physiological changes to death. While it is possible to measure concentrations of pollutants in water, sediments, and animal tissue, it is often difficult to determine the overall effect of a given pollutant on individual organisms. Even more difficult to determine are the cause-and-effect relationships between pollutants and populations of a single species or the effects on the aquatic community as a whole. However, bioassays of the Estuary's water, sediments, and biota indicate that existing pollutant concentrations cause toxic effects (SFEP 1992b).

The results of bioassays and other studies on the effects of pollutants in the Estuary suggest that pollutants may be having significant effects (SFEP 1991). Examples of these affects include: high concentrations of PCB's in starry flounder have been linked to poor reproductive success and certain creeks, rivers, and some sediments are significantly toxic in bioassays; species diversity and abundance of benthic invertebrates have decreased in certain highly polluted areas; and high concentrations of silver and copper are found in shellfish in the South Bay (SFEP 1991). Researchers have also implicated pollutants as the cause of death, due to indications of liver disease, in studies of moribund adult striped bass found in Carquinez Strait. A variety of contaminants, including those from industrial, agricultural, and urban sources, were found in the livers from which the researchers concluded that the die-off may have occurred as a result of multiple stressors. Other toxicological investigations have found that the incidence of liver malformations in larval striped bass from the Sacramento River was much higher than that in larvae from other locations (DWR 1992a).

There is growing concern about nonurban runoff in the Estuary's watershed, particularly the agricultural component (SFEP 1992b). Agricultural drainage, which contains pesticides, trace elements, and solvents, may contribute over 30 percent of the total flow of the Sacramento River in May and June, and most of the flow of the San Joaquin River in the summer (SFEP 1991). The use of herbicides has raised widespread concerns over the possible toxicological effects to aquatic biota in the Delta, especially striped bass (CUWA 1994). Rice herbicides in the Sacramento River and western Delta were found to be toxic to larval striped bass. Associated chemicals are toxic to the bass' principal food organisms, resulting in a lower ration and poorer survival for larval fish. It is hypothesized that between 1973 and 1986, pesticides may have been a factor in determining the annual recruitment of 38 mm striped bass (DWR 1992a). However, since 1986, rice herbicide loads have been decreased by 99 percent in a cooperative effort of the Central Valley RWQCB and local rice growers. This decrease in rice herbicide loads has not resulted in increases in survival of young striped bass.

Most recently, the dormant spray pesticide, diazinon, which is applied to orchards in the winter, has been identified in the Sacramento and San Joaquin rivers and the upper Estuary at levels which cause lethality in organisms. The elevated concentrations, which are highest in the San Joaquin River, immediately follow rainfall events when runoff from agricultural and urban areas occurs (DWR 1994, SFEP 1992b). Studies to further determine the impacts of this chemical are ongoing.

In addition to being a source of pesticides, agricultural drainage can increase the salinities of receiving waters to levels which adversely affect some aquatic species. This situation occurs in the lower San Joaquin River where striped bass spawning habitat is impacted as a result of a combination of saline drainage and reduced freshwater flows (which can lower salinity through dilution) due to upstream water development.

Another type of pollution is one that is created by the discharge or release of relatively warm water. Thermal pollution can be caused by the discharge of cooling water from power plants or the release of warm water from reservoirs. Warm water can be an additional stress factor for species such as salmon, which depend on cool water temperatures for successful reproduction and survival. Conversely, warm water outfalls may provide temporary refuges for certain warm water species, yet such species are adversely impacted when water temperatures near such outfalls fluctuate (SFEP 1992a).

Given the major pollutant abatement actions that have occurred during the last 20 years, it is unlikely that pollution is the principal cause of the widespread declines in fishery resources during that same time period (DFG 1994b). Nevertheless, the Estuary's biota continues to be exposed to toxic levels of pollutants and the available evidence indicates that many organisms are being adversely affected (SFEP 1992b). It is, therefore, reasonable to conclude that toxic pollutants have been, and continue to be, among the factors which contribute to the decline of some species.

7. Harvesting

Many of the mollusks, crustaceans, and fish of the Bay-Delta Estuary have been heavily harvested by humans. There is little doubt that overexploitation of species such as chinook salmon, white sturgeon, softshell clam, and crangonid shrimp has contributed to their declines in the early part of this century. In fact, the sturgeon and shrimp populations showed dramatic recoveries once commercial fisheries for these organisms were eliminated or reduced (SFEP 1992a). Although most declining species are not harvested (NHI 1992a), they may be impacted by harvest techniques (e.g., seining, gill netting) targeted at exploited species (CUWA 1994).

The legal harvest of various fish undoubtedly decreases the number of spawning adults and the average age of adults. It is unclear whether legal harvest is sufficient to inhibit a population's ability to maintain itself or if it is responsible for observed changes in abundance. The possibility of overharvesting is greatest for striped bass, white sturgeon, and chinook salmon. The DFG is confident, however, that fishing regulations for striped bass and sturgeon are preventing overharvest of these species. Management of the salmon fishery is more complicated because of the sport and commercial fishery in the ocean, the presence of several regulatory bodies, and the support of populations by hatchery production. Although ocean harvests of salmon substantially reduce spawning escapement, it is believed that the fishery is not the principal limiting factor for salmon abundance. However, it is possible that the increase in fishing effort supported by hatchery production has resulted in overharvesting of wild salmon stocks (DFG 1994b).

Illegal harvest, which is more difficult to estimate than legal harvest, potentially is of greatest concern for striped bass and chinook salmon. While the DFG believes that illegal harvest of salmon does not have a significant effect on the resource as a whole, it is very likely that illegal harvest does adversely impact striped bass populations. It is estimated that about 500,000 sublegal bass are harvested each year (DFG 1994b). This is equivalent to at least 125,000 legal-sized adults lost each year. In comparison, SWP operation is estimated to result in an average loss of an equivalent of 86,000 legal-sized bass each year, which is mitigated (DWR 1992a). The DFG concluded that, although it is very likely that illegal take reduces the production of adult bass, it seems unlikely that the harvest of sublegal bass is the dominant factor causing the decline in adult bass abundance since the collection of annual harvesting data began in 1969 (DFG 1994b).

Where harvest rates have been measured for fish populations inhabiting the Bay-Delta Estuary, no evidence was found indicating that the rates were either excessive or primarily responsible for recent declines in fish stocks (DFG 1994b). It appears that overharvest has played a minor role in the long-term declines of the Estuary's aquatic resources (SFEP 1992a) and has affected fish populations mainly after they have already suffered a severe decline (NHI 1992a).

8. Oceanic Conditions

Generally, the California coast is under the influence of the Davidson Current, which brings subtropical waters northward to Point Conception, and the California Current, which brings subarctic waters southward to Point Conception. These very different currents produce profound differences in the biological communities associated with them. Near San Francisco Bay, the oceanic conditions respond markedly to the shifting strengths of the Davidson and California currents, particularly resulting in fluctuations in the coastal zooplankton populations (SFEP 1992a).

Year-to-year changes in oceanic conditions are results of large-scale meteorological activities. In most years, the conditions vary through three seasonal stages: the upwelling period, the oceanic period, and the Davidson Current period. The most significant ecological impact is associated with the strength of the upwelling period from March through August. The strength of upwelling, which is strongest near San Francisco Bay during June and July, is closely tied to the abundance and species composition of the near-shore zooplankton community. The oceanic period marks a shift in climatic conditions in September and October, when there is a lull in winds and water flows. In November, southerly winds and the north-flowing Davidson Current produce a downdraft of surface waters along the coast. The vertical movement of water causes surface temperatures to decline during upwelling and deeper water temperatures to rise during late fall and winter (SFEP 1992a).

A failure of this seasonal progression can be associated with *El Niño* events in which warmer tropical waters at the surface produce density differences between surface and bottom waters. Consequently, there is little upwelling, and productivity at all trophic levels is reduced. *El Niño* conditions have occurred during the drought of 1976-1977 and during 1983, a wet year. The high outflows generally lead to short water residence times, low productivity, and the low salinity habitat downstream of its normal position. Thus, in 1983, low oceanic productivity lowered the marine contribution of productivity to the Estuary at the same time that riverine production was small (SFEP 1992a).

Annual variations in oceanic conditions, particularly upwelling, are thought to control recruitment success in a number of marine species. However, there does not appear to be any periodicity to the strength of upwelling while there is obvious periodicity in the populations of certain marine and anadromous species (SFEP 1992a). Therefore, it may be concluded that oceanic conditions are a contributing, rather than a major, cause in the decline of the Estuary's aquatic resources.

9. Conclusion

All of the factors described above have contributed to the declines in aquatic resources in the Bay-Delta Estuary; however, quantification of the declines has only been accomplished for a few factors such as outflow and diversions.

B. POPULATION TRENDS AND CAUSES OF DECLINES

There has been a general decline in aquatic resources in the Bay-Delta Estuary which spans all trophic levels. Although the conditions of estuarine fish populations have received the most attention, trends in the abundance of organisms from other levels of the food web are also important and indicate broad ecological changes that have occurred in the Estuary. The following discussion of the population trends in aquatic resources begins with phytoplankton, followed by zooplankton, benthos, and shrimp, and then ends with freshwater, estuarine, marine, and anadromous fish. The species addressed in this chapter do not include all the species in decline in the Estuary, such as most species of surfperch, jacksmelt, and topsmelt; nor do they include all of the species in the Estuary which show increasing population trends, such as some marine species (e.g., white croaker, California halibut, chameleon goby) (DFG 1994b).

The primary sources of information on the organisms addressed in this chapter are the results of the DWR's phytoplankton monitoring, the DFG's zooplankton monitoring, the DFG's fall mid-water trawl fish surveys from the Delta to San Pablo Bay, the DFG's summer tow-net survey from the Delta to San Pablo Bay, the Delta Outflow/San Francisco Bay Study (Bay Study) of mid-water and otter trawls from South San Francisco Bay to the western Delta, the DWR/University of California Suisun Marsh fish survey, and salvage data from the CVP and SWP facilities in the southern Delta, as presented primarily by BDOC (1993, 1994), DFG (1994a, 1994b), DWR (1992a), and the San Francisco Estuary Project (SFEP 1992a). Some of the fish surveys were designed to monitor specific species, such as striped bass and salmon, yet information on other species was obtained incidentally. Other surveys were designed to monitor fish populations in specific areas. Therefore, the sampling programs have relative strengths and weaknesses with respect to various species, depending on such factors as gear selectivity, the geographic and channel area sampled, and the season and time of day sampled. Some of the data obtained from these monitoring programs were provided by the DFG and the DWR, and are included in this chapter in graphical form to illustrate general population trends.

Population Trend Graphs. Much of the variability seen in the abundance of a given species can be explained by the variability associated with salinity among sampling stations and seasonal changes over the sampling period. This is particularly true for phytoplankton and zooplankton. By removing or accounting for the effects of salinity and season as known factors which influence the abundances of estuarine species, long-term population trends (which would otherwise be obscured by a population's response to variations in salinity and season) become apparent. The calculation of anomalies is a way to transform data so that the influence of relatively short-term factors, such a salinity and season, is dampened.

Therefore, long-term population trends represented by anomaly values reveal the variance that is due to factors which are not removed by the calculation. Thus, while population trend data for most of the aquatic organisms addressed in this chapter are graphically presented in terms of catch or abundance indices, the graphs for phytoplankton and three groups of zooplankton are presented as anomalies. A discussion of the derivation and interpretation of anomaly values follows.

An anomaly is generally defined as the deviation of a particular data point from the mean of all data within some range. Data on chlorophyll *a* (which serves as a measure of phytoplankton biomass) are expressed in terms of concentration (e.g., $\mu g/l$); data on the three types of zooplankton are expressed in terms of abundance. Thus, anomalies for these types of data are expressed as either concentration anomalies or abundance anomalies. In both cases, the data for the period of record (1972-1993) are converted to $\log_{10}$ and grouped by month and salinity classes to account for (i.e., eliminate) variability due to season and salinity. Sampling of phytoplankton and zooplankton occurs in March-November (and occasionally December-February) at 35 core (consistently sampled) stations throughout the upper Estuary (Suisun Bay through the Delta). The salinities measured among the various stations over the period of record were grouped into 20 salinity classes with approximately equal numbers of stations per class.

For each combination of month and salinity class (Mar./class 1, Mar./class 2, ... , Nov./class 20), averages were calculated using data for the entire period of record (called long-term means). Then, the data for each year of sampling were grouped by month/salinity class, and the corresponding long-term mean was subtracted from each individual observation (i.e., a data point which represents a concentration or abundance measurement) in the database. For example, the "May/class 15" long-term mean was subtracted from the "May/class 15" observation for 1976. The difference between these two values is an anomaly (i.e., anomaly value = observed value - long-term mean). Thus, an anomaly value was calculated for each observation in the period of record. Finally, the anomalies within each year, regardless of month or salinity class, are averaged (called annual anomalies).

The anomaly value, zero (0), indicates where the annual mean equalled the long-term mean. Anomalies greater than zero (positive values) indicate that the annual mean was greater than the long-term mean; anomalies less than zero (negative values) indicate that the annual mean was less than the long-term mean. Therefore, bars above the zero line are positive anomalies, indicating that the annual mean population for that year is greater than the long-term mean "population"; bars below the line are negative anomalies, indicating that the annual mean population for that year is less than the long-term mean "population".

While anomaly values have the same unit or count value of the data from which they are derived (e.g., concentration in $\log_{10}$ $\mu g/l$, or actual or estimated $\log_{10}$ abundance), they are best used as relative values that show trends, rather than quantified values, over the period of record. Therefore, anomaly values, which are very small due to compression of the data through conversion to $\log_{10}$ values, serve best as a type of index rather than actual or estimated concentration or abundance. In addition, the relatively low values of anomalies compared to the absolute values of the original data do not indicate low variability; instead, highly variable data are compressed and averaged to reveal long-term trends unrelated to factors which are known to cause variability (in this case, salinity and season). With the influences of salinity and season removed through the calculation of anomalies, population trends in these graphs are more apparent. Furthermore, increasing or decreasing trends in the populations of these organisms, as illustrated by the anomaly graphs, are primarily due to factors other than salinity and season.

1. Phytoplankton

Phytoplankton are very small, usually microscopic, algae which are suspended in water and drift with the currents. The major phytoplankton groups in estuaries are diatoms, dinoflagellates, and cryptomonads. As primary producers, which mostly convert the energy of sunlight into food through photosynthesis, phytoplankton comprise an important part of the food web base in the Bay-Delta Estuary. As a component of particulate organic carbon (POC), phytoplankton serve as food for zooplankton and other animals.

Total organic carbon, which is comprised of POC and dissolved organic carbon fractions, is used as a measure of food at the base of the estuarine food web. Sources of organic carbon include: phytoplankton, benthic microalgae, macroalgae, and photosynthetic bacteria produced in the Estuary; river-borne organic loads; tidal marsh export; point sources; runoff; atmospheric deposition; spills; ground water; and animal migration. Much of the POC appears to be phytoplankton and phytoplankton-derived detritus produced in and upstream of the Estuary.

Phytoplankton productivity and abundance are influenced by several factors, including light, temperature, nutrients, and grazing by aquatic animals. These factors can be influenced by hydrologic conditions in the Estuary which in turn affect various conditions, such as the location of the low salinity habitat. Phytoplankton abundance is estimated by direct counts or by measuring the chlorophyll produced (DFG 1994b). As part of the D-1485 water quality monitoring program, the DWR routinely samples the phytoplankton composition and biomass in San Pablo and Suisun bays, and in the Delta. Estimates of phytoplankton biomass are derived from measurements of the concentrations of chlorophyll *a*, a green pigment found in all plants. Measured chlorophyll concentrations are used primarily to document abrupt changes in phytoplankton concentrations, called "blooms" (DWR 1992a).

a. __Population Trends__. Between 1976 and 1991, phytoplankton blooms occurred in all regions of the upper Estuary (the western, central, northern, and southern Delta, and Suisun and San Pablo bays). These blooms, which typically occur during the spring and fall, are most often dominated by one of four diatom genera: *Skeletonema*, *Thalassiosira*, *Cyclotella*, and *Melosira*. Blooms have been most intense in the southern Delta, where chlorophyll *a* concentrations have exceeded 300 μg/l, and least intense in the San Pablo Bay ship channel, where chlorophyll *a* concentrations have not exceeded 26 μg/l (DWR 1992a).

Both the frequency and intensity of phytoplankton blooms have decreased in many regions of the upper Estuary. Throughout the upper Estuary, substantially fewer blooms occurred between 1987 and 1991 than in any other 5-year period examined. Beginning in the mid- to late-1980's, a decreasing trend in bloom intensity has occurred in all monitored regions of the upper Estuary, except the southern Delta. During the drought years of 1977 and 1987-1991, as well as during the extremely wet year of 1983, phytoplankton biomass was substantially depressed, often below the background level of 10 μg/l. In the southern Delta, however, peak levels of phytoplankton biomass increased during periods of drought compared to other years (DWR 1992a). These levels may have developed in response to increases in water residence time, which can occur during periods of reduced inflow, combined with the eutrophic conditions that generally exist in this region (Hymanson et al. 1994).

The southern Delta, which is dominated by warm nutrient-rich waters of the San Joaquin River, supports high concentrations of phytoplankton. Because of higher salinities due to recirculated agricultural water, the southern Delta phytoplankton communities are similar to those of the western Delta. The northern Delta, which receives most of its water from the Sacramento River and the Yolo Bypass, supports the lowest phytoplankton concentrations in the area (SFEP 1992a).

Chlorophyll *a* levels in the central Delta increased in 1982-1986, and decreased in 1978-1981 and 1987-1990. In the western Delta, chlorophyll *a* levels increased in 1978-1982, and decreased in 1983 and after 1986 (Hymanson et al. 1994). Prior to 1976, phytoplankton blooms in the western and central Delta were dominated by *Skeletonema* spp., *Melosira* spp., *Thalassiosira* spp., or *Cyclotella* spp. Since the May 1976 bloom, almost all large blooms in the western and central Delta have been due to *Melosira granulata* (SFEP 1992a), a phytoplankton species that is not a preferred food source of zooplankton (DFG 1994b).

In Suisun Bay, chlorophyll levels generally have declined since the mid-1970's. During the 1976-1977 drought, extremely low phytoplankton levels were observed in San Pablo and Suisun bays while the highest levels were observed entering the Delta with Sacramento and San Joaquin river inflows. Since 1978, however, such high in-flowing levels of phytoplankton have not been observed (DFG 1994b). Long-term data for chlorophyll *a* at shoal stations in Grizzly and Honker bays suggest that phytoplankton productivity in Suisun Bay was low in 1977 and has been depressed since about 1983 (SFEP 1992a). From 1980 through 1990, *Thalassiosira* spp. dominated the phytoplankton populations in Suisun Bay (Hymanson et al. 1994).

Long-term chlorophyll *a* data are insufficient to adequately characterize the interannual variability in phytoplankton production in Central and San Pablo bays (SFEP 1992a). Based on the sources of organic carbon for 1980, phytoplankton production constituted about 60 percent of the total organic carbon in the South Bay (below the Bay Bridge). In the North Bay (i.e., San Pablo Bay to Chipps Island), where phytoplankton production provided only about 20 percent of the total organic carbon, the sources were dominated by the loading of organic carbon from the Sacramento and San Joaquin rivers. During 1975-1989, phytoplankton-derived particles in Suisun Bay that were attributed to river loading ranged from 20 to 90 percent, suggesting that the dominant source changes from year to year (IEP 1994b).

Unlike phytoplankton or benthic microalgae, some of this river-borne organic matter (both dissolved and particulate forms) may be metabolically inert and not capable of being incorporated into the food web. BOD measurements in the Sacramento River over many years correspond, on average, to only about 10 percent of the total organic carbon concentration, suggesting that most of the organic matter is not readily useable (IEP 1994b). Although BOD values, which are obtained for point source waste discharges, correspond to the metabolizable fraction of the organic carbon load, it is necessary to convert them to organic carbon to compare with the contributions from other sources (SFEP 1992a).

Mean chlorophyll *a* concentrations in San Pablo Bay, Suisun Bay, and the Delta for 1975-1991 are shown in Figure V-9. Because trends are less evident in data which do not account for variations in salinity and season, anomaly values (explained above) were calculated for some of the chlorophyll *a* data. Figure V-10 presents chlorophyll *a* $\log_{10}$ concentration anomalies for Suisun Bay and the Delta from 1972 through 1993. This graph illustrates the overall decline in phytoplankton biomass throughout the upper Estuary. From 1972 through 1982, chlorophyll *a* levels were relatively high, although lower levels were observed during the 1977-1978 drought. Then, overall chlorophyll *a* levels declined in 1983 (a wet year), rebounded slightly in 1984, and steadily decreased between 1985 and 1993.

b. Causes of Decline. With the exception of the southern Delta, phytoplankton production in the upper Estuary has decreased during extremely dry and wet years, and has shown a steady decline overall. The effects of Delta outflow on phytoplankton production have been linked to the location of the entrapment zone, the area in the Estuary where fresh water and saline water flow converge, resulting in the concentration of particulate matter, including phytoplankton (Arthur and Ball 1980). The concept of the entrapment zone currently is separated into two components, low salinity habitat and gravitational circulation (see section V.A.2.d).

A model of the theoretical mechanisms by which Delta outflow influences phytoplankton productivity is described by Kimmerer (1992). Generally, phytoplankton production is increased with increased outflow. The phytoplankton growth rate is higher overall in the shoals. Phytoplankton production is also influenced by the location of the entrapment zone. When the entrapment zone is farther downstream, the phytoplankton have longer residence times in the shoals and, therefore, a higher growth rate. Within the entrapment zone, phytoplankton production is decreased because of increased turbidity.

Phytoplankton biomass is highest when the entrapment zone is adjacent to the shoal areas, in San Pablo and Suisun bays. This is due to the exchange of phytoplankton cells from the shoals (where productivity is highest) to the entrapment zone (area of gravitational circulation), driven by winds and tidal exchange. The phytoplankton is trapped in this area, which accounts for the higher biomass. In Suisun Bay, Delta outflows in the 5,000 to

Figure V-9
**Chlorophyll *a* Concentrations
San Pablo Bay, Suisun Bay, and Delta
(1975-1991)**

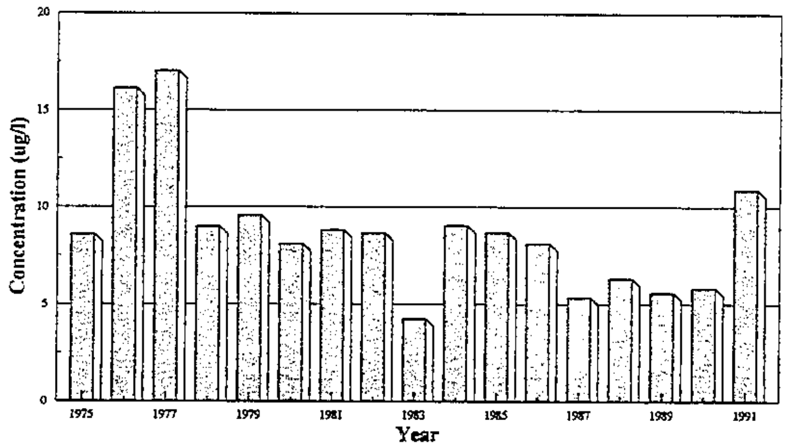
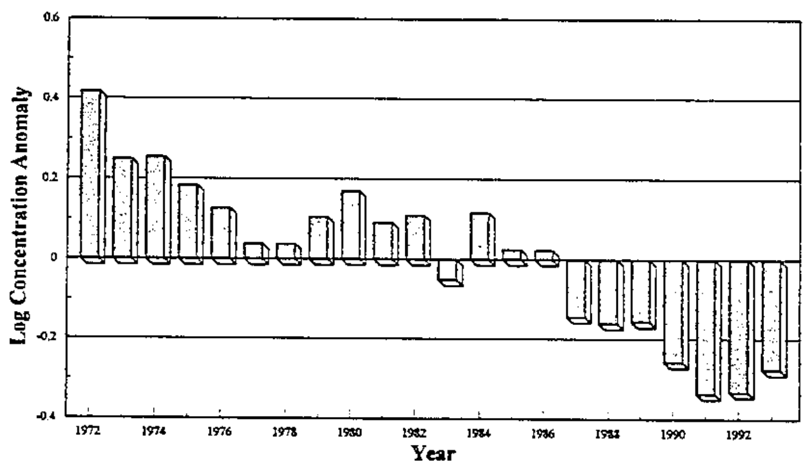


Figure V-10
**Chlorophyll *a* Concentration Anomalies
Suisun Bay and Delta
(1972-1993)**



8,000 cfs range historically have been associated with maximum phytoplankton production. When Delta outflow is less than 5,000 cfs, the entrapment zone moves upstream into the deeper Delta waters which reduces phytoplankton production in the shoals downstream (SFEP 1992b).

Based on the organic carbon budget work of Jassby (SFEP 1992a), a positive relationship between POC to Suisun Bay (including phytoplankton production and riverine loading of algal-derived particulate matter) and Delta outflow for the period 1975-1989 was demonstrated. This relationship is illustrated in Figure VI-3 of Chapter VI.

The drought-associated increases in phytoplankton biomass in the southern Delta suggest that SWP exports have not adversely impacted phytoplankton activity in this part of the Estuary during droughts. Additionally, short-term studies have found no enhancement of phytoplankton biomass during periods of curtailed exports. The central Delta is the region where phytoplankton levels could most likely be impacted by SWP operations. Increases in channel water velocities and changes in flow patterns (e.g., cross-Delta flows and reverse flows) result in reduced residence times (DWR 1992a).

Changes in sewage treatment practices and loadings could also affect the abundances of phytoplankton by reducing the amount of nutrients entering the Estuary (DWR 1992a). However, nutrients apparently do not limit the growth of phytoplankton at least until biomass reaches extremely high levels during summer blooms (Kimmerer 1992).

Finally, low phytoplankton biomass during extended drought periods could be due to increased benthic grazing that results from the gradual landward penetration of marine benthic grazers (Kimmerer 1992). However, since its discovery in 1986, the introduced Asian clam (*Potamocorbula amurensis*), a highly efficient suspension feeder that has become established at high concentrations in San Pablo Bay, Suisun Marsh, and Suisun Bay, may have also caused sustained reductions in phytoplankton biomass in some regions of the Estuary, such as Grizzly Bay (Figure V-10) (DWR 1992a). *P. amurensis* is discussed further under the section on benthos, below.

2. Zooplankton

Zooplankton are small, sometimes microscopic, aquatic invertebrate animals that drift with water currents, although they have some swimming ability. Zooplankton usually occupy intermediate trophic levels in the estuarine food web, where they may feed on phytoplankton, bacteria, protozoans, and organic detritus (e.g., POC), and are fed upon by organisms such as mollusks, shrimp, and various life stages of estuarine fish. Important zooplankters in the Bay-Delta Estuary include the rotifera, cladocera, and the copepoda, as well as the opossum shrimp.

Rotifers are microscopic, multicellular invertebrates that are most common in fresh waters, although a few purely marine species are known. Omnivorous feeding on both living and dead particulate organic matter is typical, but some species prey on protozoa, other rotifers, and other zooplankters. Dominant rotifer genera in the Bay-Delta Estuary include *Synchaeta* and *Keratella*. *Synchaeta* is most common where salinities greater than 5-10 ppt occur (e.g., in South Bay and in the western Delta in the fall). *Keratella*, which is found in fresher water, occurs in the eastern Delta and in the western Delta in the spring (SFEP 1992a).

Cladocerans, or water fleas, are often the most abundant crustaceans in fresh water. They seldom occur in waters where salinity is greater than 1 ppt and are, therefore, more abundant in the Delta than in Suisun Bay. Cladocerans are efficient feeders on a wide variety of materials from throughout the water column, including phytoplankton, bacteria, and colloidal suspensions. Among the most common cladoceran genera in the Estuary are *Bosmina*, *Daphnia*, and *Diaphanosoma*. *Bosmina* is the most widely distributed in the Estuary and is the dominant cladoceran in the Delta. *Daphnia* is also found in the Delta and Suisun Bay, but in less abundance than *Bosmina*. *Diaphanosoma* has the most restricted distribution of these three native cladocerans. The densities of all three species are highly correlated with water temperature and, except for *Diaphanosoma*, with chlorophyll *a* concentrations (SFEP 1992a).

Copepods are small crustaceans that are a major food item of plankton-feeding shrimp and fish in the Bay-Delta Estuary (NHI 1992a, SFEP 1992a). Copepods, which feed on detritus and phytoplankton, occur in a much larger range of salinities than cladocerans. In the Estuary, the abundant native copepods are sharply separated primarily by salinity and season. The dominant native copepod genera in the Estuary include *Acartia* and *Eurytemora*. Prior to the introduction of *Pseudodiaptomus*, *Cyclops* was also abundant in the Estuary. In addition to *Pseudodiaptomus*, several other copepods species were unintentionally introduced into the Estuary in the late-1970's and 1980's, including *Sinocalanus*, *Limnoithona*, and *Oithona*. *Acartia* and *Oithona* are most abundant in the more saline regions of the Estuary (e.g., South and Suisun bays); *Cyclops*, *Sinocalanus*, and *Limnoithona* are primarily freshwater copepods and can be found in the upper Estuary. *Eurytemora affinis*, an estuarine species, can be found in Suisun Bay and is the dominant native copepod in the Sacramento and San Joaquin rivers (SFEP 1992a).

The opossum shrimp (*Neomysis mercedis*) is a native mysid shrimp that is an important food source for many estuarine fish, especially young striped bass. *N. mercedis* is found in greatest abundance in Suisun Bay and the western Delta, although it occurs as far upstream as Sacramento and the lower reaches of the Mokelumne River. The diet of *N. mercedis* consists of phytoplankton, rotifers, and copepods, particularly *E. affinis* (SFEP 1992a).

a. **Population Trends.** Zooplankton populations in the Estuary are regularly sampled only in the Delta and Suisun Bay; therefore, trends in zooplankton abundance in South, Central, and San Pablo bays are not known. Abundances of 12 of the 20 zooplankton taxa routinely monitored in the Estuary have declined significantly between 1972 and 1988. Seven taxa showed no trend in abundance, and one introduced copepod, *Oithona davisae*, increased in abundance. In general, declines in zooplankton abundance occurred throughout the upper Estuary, but were more prevalent in the Sacramento and San Joaquin rivers than in Suisun Bay (DWR 1992a).

For the Delta and Suisun Bay from 1972 to 1993: Figures V-11 and V-12 present mean abundance and $\log_{10}$ abundance anomalies, respectively, for rotifers; Figures V-13 and V-14 present mean abundance and $\log_{10}$ abundance anomalies, respectively, for cladocerans; Figure V-15 presents the abundances of native and introduced copepods; and Figures V-16 and V-17 present mean abundance and $\log_{10}$ abundance anomalies, respectively, for opossum shrimp (*Neomysis*). Like the anomalies presented for phytoplankton, above, the anomaly values for zooplankton show population trends which generally ignore the effects of salinity and seasonal variability.

Rotifers. Overall, rotifer abundance in the Delta and Suisun Bay has steadily declined between 1972 and 1993 (Figures V-11 and V-12). Since the early 1970's, rotifer populations have declined sharply throughout the Delta (DFG 1994b), particularly in the San Joaquin River where they were formerly most abundant. Between 1972 and 1979, the rotifer populations in the Delta declined to less than 10 percent of their initial measured densities. In Suisun Bay, where rotifers were never very abundant, the decline was less severe. Since 1979, there has been no consistent difference in the abundance of rotifers in the Delta and Suisun Bay. Rotifer abundance in the Delta appears to be strongly associated with chlorophyll *a* concentrations (SFEP 1992a).

Cladocerans. The average abundance of cladocerans since the early 1970's has shown a long-term decline similar to that of the rotifers, but at a more gradual rate (Figures V-13 and V-14) (DFG 1994b). The decline in cladocera, which varies within different parts of the Estuary, is apparent in most genera except *Bosmina*. Examination of the patterns of abundance of cladocerans over time for Suisun Bay, and for Delta areas dominated by Sacramento River water and San Joaquin River water, shows the importance of Delta outflow on cladoceran abundance and distribution. Very high outflows of 1983 produced peak abundances of most cladoceran genera in Suisun Bay; moderately high outflows of 1986 produced peaks in abundance for all genera within the Delta, but had little effect on Suisun Bay populations (SFEP 1992a).

Copepods. Overall copepod abundance has remained fairly stable in recent years. However, native copepods, particularly *E. affinis*, have suffered large declines in abundance while non-native species (e.g., *Sinocalanus doerrii* and *Pseudodiaptomus forbesi*) have increased in abundance since their introductions in the late-1970's and 1980's (Figure V-15). The net result is that copepods have been at least as abundant since the late-1970's as they were previously (DFG 1994b, SFEP 1992a). In the Sacramento and San Joaquin rivers, introduced copepods are now more abundant than native copepod species, whereas in Suisun Bay, native copepods are more abundant. In the Delta, the once abundant *Cyclops* has been replaced by *Pseudodiaptomus* as the dominant copepod. However, due to increases in the populations of the introduced freshwater copepods, the average densities of copepods in the rivers are still high in most years. Within Suisun Bay, which usually supports copepod densities about twice those found in the Delta, only *E. affinis* shows a consistent pattern of decline over time. The abundance of *E. affinis* declined following the invasion of the western Delta and Suisun Bay by *S. doerrii* in 1978 and *P. forbesi* in 1987. Although

Figure V-11
Rotifera Annual Mean Abundance
Suisun Bay and Delta
(1972-1993)

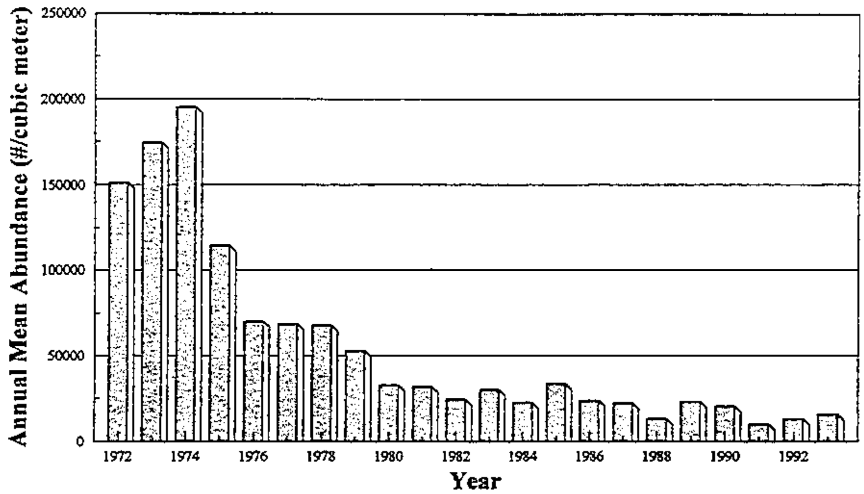


Figure V-12
Rotifera Abundance Anomalies
Suisun Bay and Delta
(1972-1993)

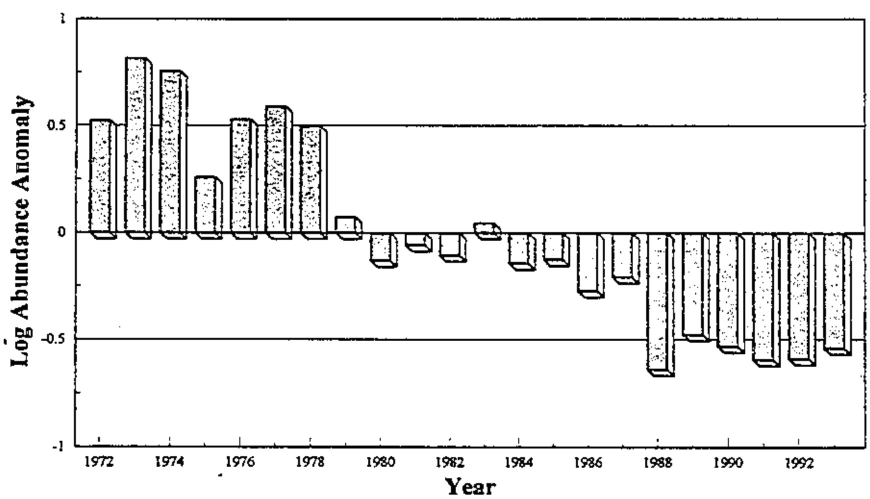


Figure V-13

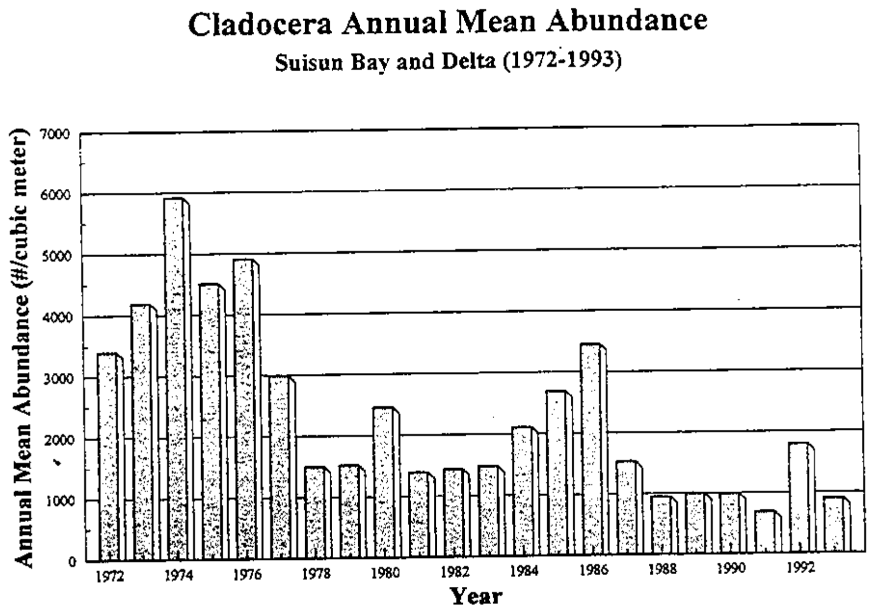


Figure V-14

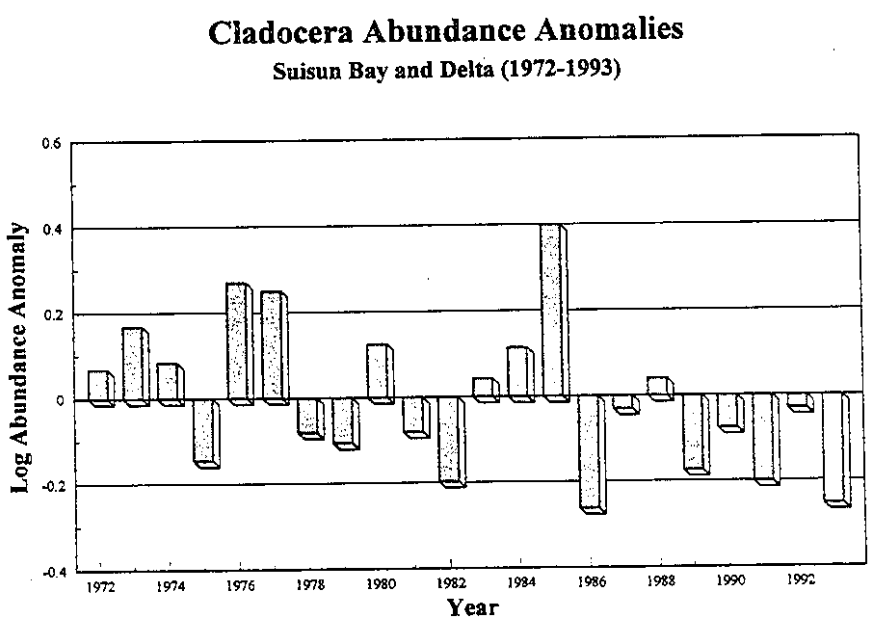


Figure V-15

Native and Introduced Copepod Abundance

Suisun Bay and Delta (1972-1993)

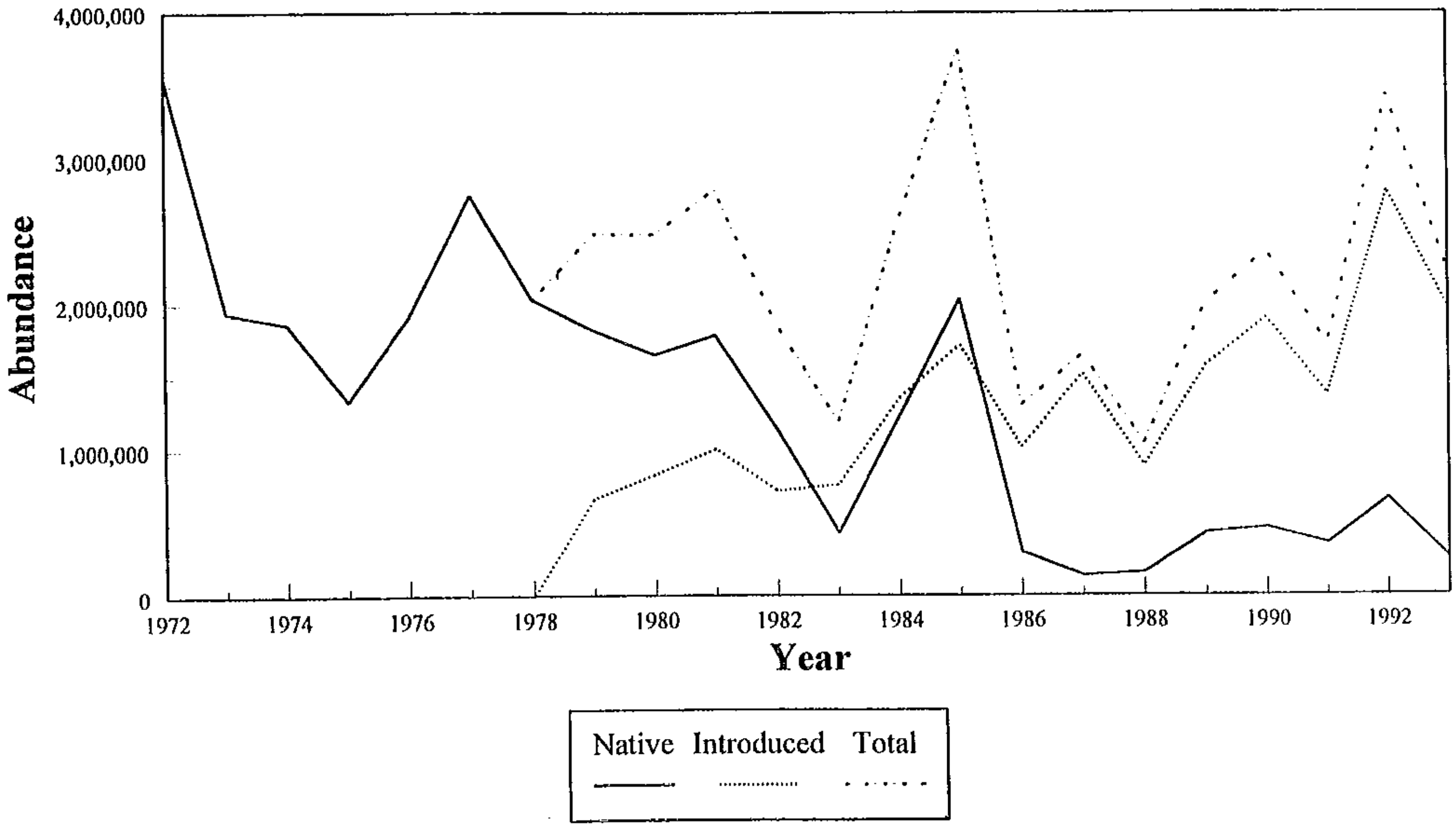


Figure V-16

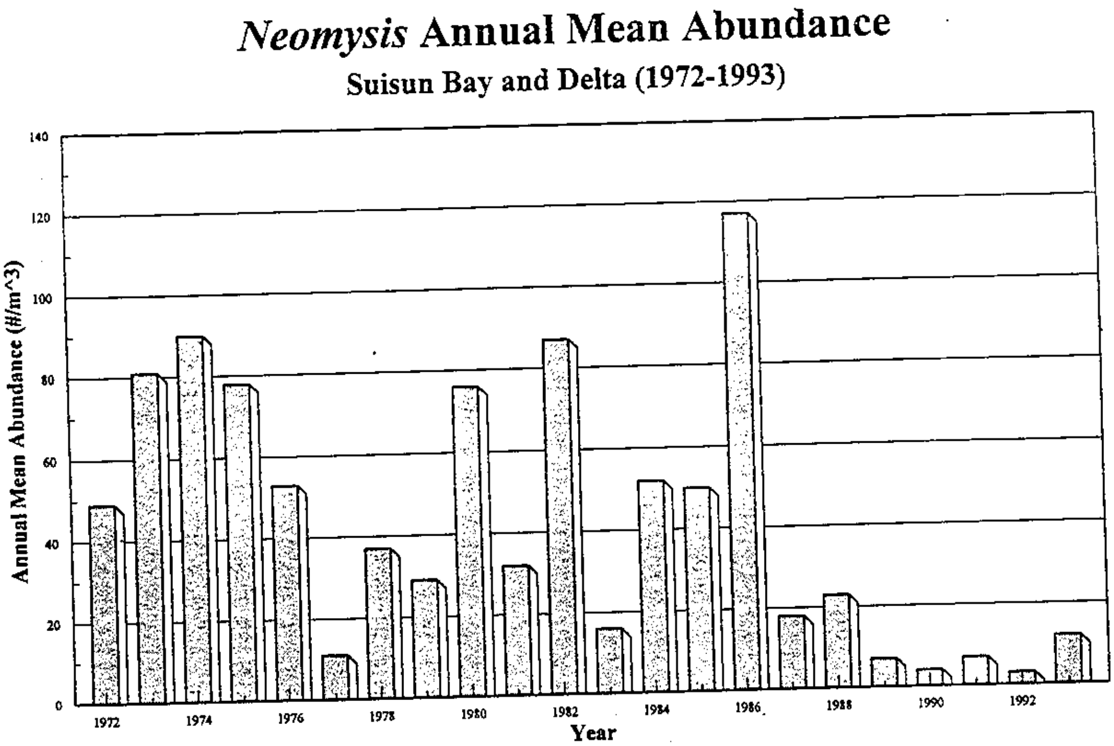
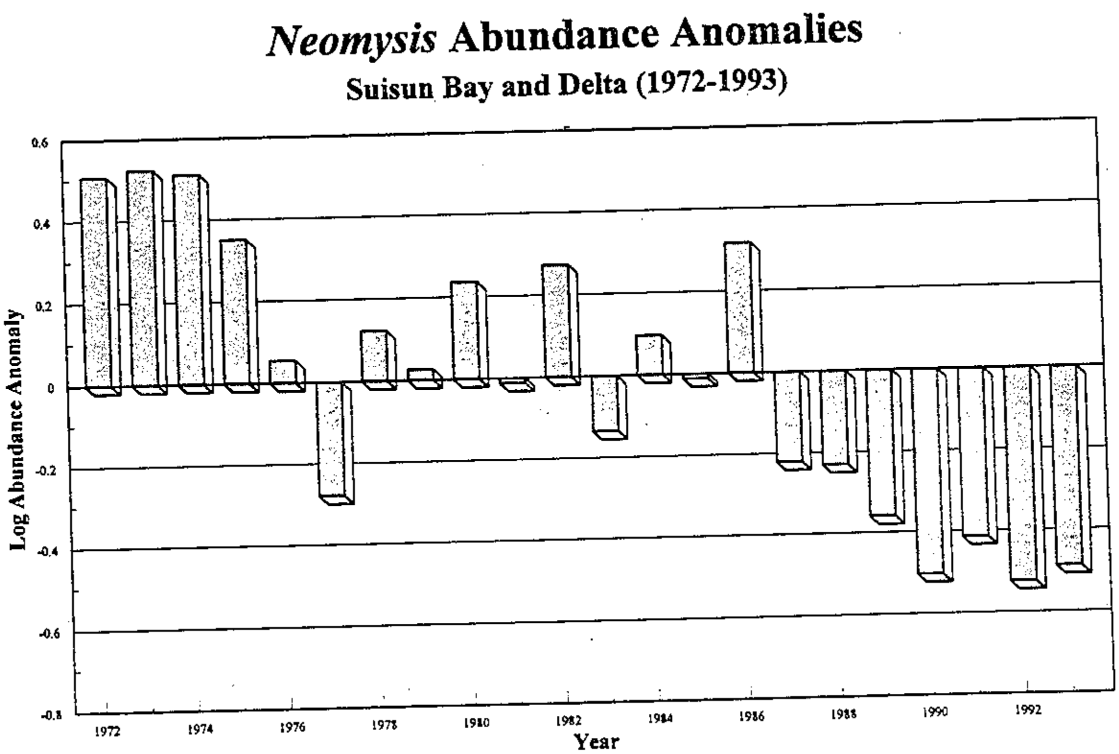


Figure V-17



introduced copepod species generally are not a large part of the populations in Suisun Bay, they typically increase in abundance in the bay in response to periods of high outflow (SFEP 1992a).

While most species of copepods have undergone severe, long-term declines in abundance, the marine species, *Acartia*, shows no evidence of a trend through time. This species is least abundant in the Delta and Suisun Bay during years of high outflow and is usually most abundant when salinity in Suisun Bay is greatest (SFEP 1992a).

Opossum Shrimp. During most of the 1980's, the opossum shrimp (*N. mercedis*) population varied considerably, but at a lower level of abundance than existed in the early 1970's (Figures V-16 and V-17). *N. mercedis* abundance fell dramatically after 1986 and remained at very low levels from 1990 to 1993 (DFG 1994b). Populations of *N. mercedis* have declined substantially in Suisun Bay, yet they have occasionally rebounded to high levels (BDOC 1993).

b. Causes of Declines. Reasons for the system-wide declines of several zooplankton taxa in the Bay-Delta Estuary are not known. Although the declines occurred at about the same time as declines in phytoplankton and various fish species, no cause-and-effect relationships have been established (DWR 1992a). However, several factors have been identified which are believed to have some influence on the decline of zooplankton in the Estuary.

Decrease in food supply has been associated with the decline in abundance of rotifers and the copepod, *E. affinis*. The decline of rotifers in the Delta appears to be strongly associated with declining concentrations of chlorophyll *a*, which formerly characterized the areas of greatest rotifer abundance (SFEP 1992a). However, chlorophyll and many zooplankton species have similar spatial distributions, and correlations between the two groups can arise through movement of the low salinity habitat in the Estuary. Also, while it is commonly assumed that chlorophyll is a good measure of food availability for zooplankton, *E. affinis* can subsist on detrital matter and requires larger particles than those that make up total chlorophyll. In addition, small zooplankton could provide food for many of the larger zooplankton species (Kimmerer 1992). Consistently low *E. affinis* abundance in recent years has been named as a factor that has probably contributed to the decline of *N. mercedis* (SFEP 1992a).

Introduced species have also been named as a potential cause for the decline in zooplankton abundance. For example, the introduction of *Sinocalanus* has been identified as a possible cause of the decline in abundance of *E. affinis* (Kimmerer 1992), although the introduced copepod does not have the same habitat requirements as the native copepods (NHI 1992a). However, based on the known feeding habits of a related species of *Sinocalanus*, *S. doerrii* may prey on native copepods (SFEP 1992a). In addition, predation by the introduced Asian clam, *Potamocorbula amurensis*, has been suggested as a factor in the decline of rotifer (SFEP 1992a) and *E. affinis* populations. *E. affinis* abundance in Suisun Bay decreased substantially when the clam became abundant there in 1988 (DWR 1992a). Kimmerer et al.

(1994) hypothesized that direct predation by *P. amurensis* is the cause of the reduced survival of *E. affinis* nauplii which has, therefore, depressed the abundance of the adults.

The decline in the abundance of *N. mercedis* and other zooplankton species (e.g., *E. affinis*) that are found in the low salinity habitat in relatively high abundances has been correlated with Delta outflow (see Figure VI-4 in Chapter VI). It is presumed that low outflow reduces *N. mercedis* abundance by: (1) restricting the low salinity habitat to deeper, more upstream channels which are less likely to promote high densities of *N. mercedis*; and (2) producing weaker landward currents along the bottom so that the ability of *N. mercedis* transported downstream to return to the low salinity habitat is reduced. It has also been presumed that larger numbers of *N. mercedis* may be exported through the CVP and SWP pumps as a result of the increased proportion of inflow diverted during drought years when the low salinity habitat is upstream in the Estuary. The location of the low salinity habitat within the lower river channels during dry years increases the vulnerability of *N. mercedis* to such displacement (SFEP 1992a). However, analyses by Kimmerer (1992) suggest that exports by the water projects are not a major source of losses for *N. mercedis* and *E. affinis* populations, primarily due to the small percentage of low salinity habitat volume (and low salinity habitat organisms) diverted. Depending on the timing, location, and quantity of withdrawals, in-Delta water diversions, whose net consumption is on the same order of export flows, may result in a higher rate of loss to resident zooplankton populations than export pumping.

Pollutants may be another factor in the decline of zooplankton in the upper Estuary. For example, rice herbicides have been shown to be toxic to *N. mercedis* (DWR 1992a). However, rice herbicides are largely confined to the Sacramento River, not the entire Estuary. No Estuary-wide decline in planktonic crustaceans has been associated with the timing of herbicide occurrence in the river (NHI 1992a).

3. Benthos

Benthic organisms (benthos) are animals that live in or on the bottom of an aquatic habitat. Most benthic organisms feed by straining phytoplankton and non-living organic matter from the water column. The benthos in the Bay-Delta Estuary include mollusks, such as oysters and clams, and benthic crustaceans, such as crabs, crayfish, and shrimp. With few exceptions, all of the common benthic species in the Estuary have been intentionally or accidently introduced (BDOC 1993).

The factors which most affect the abundance, composition, and health of the benthic community include local runoff, pollution, and Delta outflow. The importance of pollution in controlling benthic communities has been assumed to be very high. Lower outflows are also associated with lower phytoplankton biomass and, therefore, lower productivity during periods of low flow in parts of the Bay complex. High outflows lead to lower salinities, which particularly control the species abundance and composition in shallow areas where animals are exposed to less saline surface waters (SFEP 1992a).

In the northern reach of the Estuary, the abundance and distribution of benthic species are greatly affected by salinity variation. Historically, during high outflow years, some brackish water species decline; during low flow years, species associated with more saline water occur more frequently. However, in 1987, following several years of very low flow and high salinity, Suisun Bay was not colonized by more marine benthic species as expected. Rather, the newly-introduced Asian clam, *Potamocorbula amurensis*, (discussed below) remarkably increased in abundance (DFG 1994b).

a. Mollusks. With the exception of one or two species (i.e., the bay mussel, *Mytilus edulis*, and, possibly the clam, *Macoma balthica*), the common benthic mollusks of the Bay-Delta Estuary are introduced. Within the Delta, the dominant mollusk is the introduced Asiatic clam, *Corbicula fluminea* (SFEP 1992a). Introduced into California in the late 1940's, *C. fluminea* quickly became a dominant member of the benthos in the Estuary. *C. fluminea* is a suspension-feeding, freshwater clam that filters phytoplankton and organic detritus from the water column. Recent studies suggest that *C. fluminea* is able to filter a significant portion of the phytoplankton from the water column. Immature clams are readily dispersed in the Estuary by flowing water. Increased outflows result in *C. fluminea* being found throughout the upper Estuary, but salinity levels in Suisun Bay prevent the establishment of permanent populations there. Established populations appear to exist in the central Delta and, to a lesser extent, in the western Delta (Hymanson et al. 1994).

The most recently introduced mollusk in the Estuary is the Asian clam, *Potamocorbula amurensis*. Native to the estuaries along the east coast of Asia, this clam is thought to have been introduced into Suisun Bay as larvae through the discharge of ship ballast water (Hymanson et al. 1994).

Potamocorbula amurensis. Like *C. fluminea*, *P. amurensis* is a suspension-feeding clam. It is capable of consuming phytoplankton, bacterioplankton, particulate organic matter, and immature zooplankton (Hymanson et al. 1994). This small clam, which grows to a maximum size of 1 inch, has high feeding and-reproductive rates. At densities as great as 25,000 individuals per square meter, the *P. amurensis* population is able to filter substantial volumes of water as it feeds (SFEP 1992a). It has been calculated that densities of *P. amurensis* in the Estuary are so high that the entire water column of San Pablo and Suisun bays can be filtered within a 24-hour period (CUWA 1994).

Population Trends. Since its discovery near Carquinez Strait in 1986, *P. amurensis* has become the most abundant benthic organism in several regions of the upper Estuary (CUWA 1994, DWR 1992a). By 1990, *P. amurensis* was well established in a variety of habitats throughout San Pablo and Suisun bays, and Suisun Marsh (Hymanson et al. 1994). Before the introduction of *P. amurensis*, shifts to more saline conditions in the Estuary, as during the low flow years of 1976 and 1977, resulted in the increase in abundance of the introduced softshell clam, *Mya arenaria* (CUWA 1994), which was first noted in the Estuary in 1874 (SFEP 1992a). It is thought that *P. amurensis* prevented the recolonization of Suisun Bay by *Corbicula* following the return to lower salinities there after the drought conditions in 1984 and 1985 (CUWA 1994). During the drought period, 1987-1992, this species spread throughout the more saline portions of the Estuary and into Suisun Bay (BDOC 1994). The persistently low salinity in the central Delta probably prevents the establishment of *P. amurensis* in this region (Hymanson et al. 1994).

Potamocorbula amurensis has altered the benthic community in Grizzly Bay and the Sacramento River near the confluence, where it has been dominant since 1988 (Hymanson et al. 1994). In Suisun Bay, the previous benthic community largely disappeared as *P. amurensis* multiplied. During this time, normal summertime phytoplankton blooms have failed to occur and chlorophyll *a* densities have remained at some of the lowest recorded values (Figure V-10) (SFEP 1992a). This species' extremely high filter-feeding rate has resulted in dramatic reductions in phytoplankton density and shifts in POC loadings. Such reductions are likely having a direct influence on the population dynamics of zooplankton and planktivorous fish (CUWA 1994).

Causes of Increase. The establishment and spread of *P. amurensis* indicate that this introduced species has found the conditions of the Estuary to be conducive to its propagation and growth, and that it apparently has a wide niche partition. As a filter feeder, it is able to remove and process phytoplankton from all waters that it inhabits. There has been a dramatic reduction in phytoplankton and chlorophyll *a* densities since its introduction. This has ecological significance for a number of planktivorous fish species in the Estuary which rely on both phytoplankton and zooplankton as a major food source (CUWA 1994).

While the establishment of *P. amurensis* may have increased the competition with other benthic organisms for space and food, it does provide a new and abundant food source for bottom-feeding crabs, fish, and birds (Hymanson et al. 1994). However, this clam can bioaccumulate high concentrations of selenium, which could result in higher tissue concentrations in organisms that feed on it (DWR 1992a).

b. **Benthic Crustaceans**. Unlike the mollusks, the benthic crustaceans are comprised of many native species, particularly young Dungeness crabs and other smaller crabs, as well as caridean shrimp. However, in the upper Bay complex, the epibenthos (unattached benthic organisms) consist entirely of introduced species, particularly the crayfish. The benthic epifauna, except for the Dungeness crab, is probably the least studied community of animals in the Estuary (SFEP 1992a). The DFG has also monitored the abundance of true shrimp (Caridea) in recent years. Therefore, the Dungeness crab and the caridean shrimp are discussed below as representative species of the benthic crustaceans in the Bay-Delta Estuary.

Dungeness Crab. The most familiar member of the benthic community in the Estuary is the Dungeness crab (*Cancer magister*). This native species reproduces at sea, enters San Francisco Bay as juveniles during May or June, and leaves the bay by August or September of the following year (SFEP 1992a). Bay-reared Dungeness crabs grow about twice as fast as, and contribute to the commercial and sport ocean fishery 1 to 2 years sooner than, ocean-reared crabs. Dungeness crab fishing is not allowed in San Francisco Bay (DFG 1987). The bay population contributes as much as 83 percent of the crabs in the Central California fishery (SFEP 1992a).

Population Trends. The Dungeness crab is generally most abundant from Richardson's Bay upstream through Suisun Bay, with the most consistently high number of juveniles in San Pablo Bay. No crabs are found where bottom salinities are less than 10.2 ppt, and the onset of high outflows from winter storms results in a mass movement of crabs to more downstream locations (SFEP 1992a).

For the first 60 years of this century, Dungeness crabs were an increasingly important fishery for San Francisco. Historical trends in Dungeness crab landings (Figure V-18) indicate that the catches rose until the late-1950's. The May and June abundance indices of juvenile crabs in the bay have varied widely since monitoring efforts of the Bay Study began in 1980 (Figure V-19). Low abundances occurred in 1983 and 1986, two years with the highest outflows ever recorded; then, they attained higher abundances in the following years, 1984 and 1987 (SFEP 1992a). The crab expanded its distribution in the bay during the low outflow years of 1981, 1984, and 1985 (DFG 1987). Overall, the species exhibits a declining trend in population size, with low abundances occurring in the late 1980's and early 1990's (DFG 1994b).

Causes of Decline. Oceanic conditions in 1959 caused the population and catch of Dungeness crabs to drop dramatically. Although oceanic conditions are probably the strongest control on the size of Dungeness crab populations (SFEP 1992a), Delta outflow has been correlated with juvenile crab abundance. There is a negative relationship between outflow and juvenile crab abundance in the bays. The estuarine flows during high outflow years may carry larval crabs too far offshore, and possibly too far north, to allow them to return to the vicinity of the bay (DFG 1987); however, the actual mechanism for transporting larval crabs to the coast is unclear. The number of crabs entering the bay is primarily a function of larval crab abundance in the ocean and, perhaps, the strength of landward-flowing bottom currents. High outflows, which appear to reduce the transport of crabs into the bay, are frequently associated with *El Niño* events and other oceanic conditions that are suspected of reducing larval crab abundance (SFEP 1992a).

Another factor which has been considered in the reductions of Dungeness crab abundance is cannibalism. Because juvenile crabs generally remain in the bays for about 15 months, two year classes (i.e., the newly-arrived juveniles that entered the Estuary in May and June, and the older juveniles that entered the Estuary in May and June of the previous year) occur together during the summer. Therefore, an abundant year class of larger juveniles could reduce the subsequent year class size of smaller juveniles through cannibalism (SFEP 1992a).

Caridean Shrimp. Five species of caridean shrimp (*Crangon franciscorum*, *C. nigricauda*, *C. nigromaculata*, *Heptacarpus stimpsoni*, and *Palaemon macrodactylus*), which seldom exceed 70 mm in total length, dominate the smaller benthic fauna in the Bay-Delta Estuary (SFEP 1992a). *Crangon* spp. are commonly called "bay shrimp" and *Palaemon* is known as "pile shrimp"; collectively, they are often referred to as "grass shrimp". The three species of *Crangon*, as well as the less abundant *H. stimpsoni*, are native shrimp, whereas *P. macrodactylus* was introduced to the Bay-Delta Estuary in the 1950's (DFG 1994b). The crangonid shrimp are common food items for many estuarine fish (SFEP 1992a). The

Figure V-18

Dungeness Crab Landings

San Francisco Area, Bodega Bay to Princeton (1941-1992)

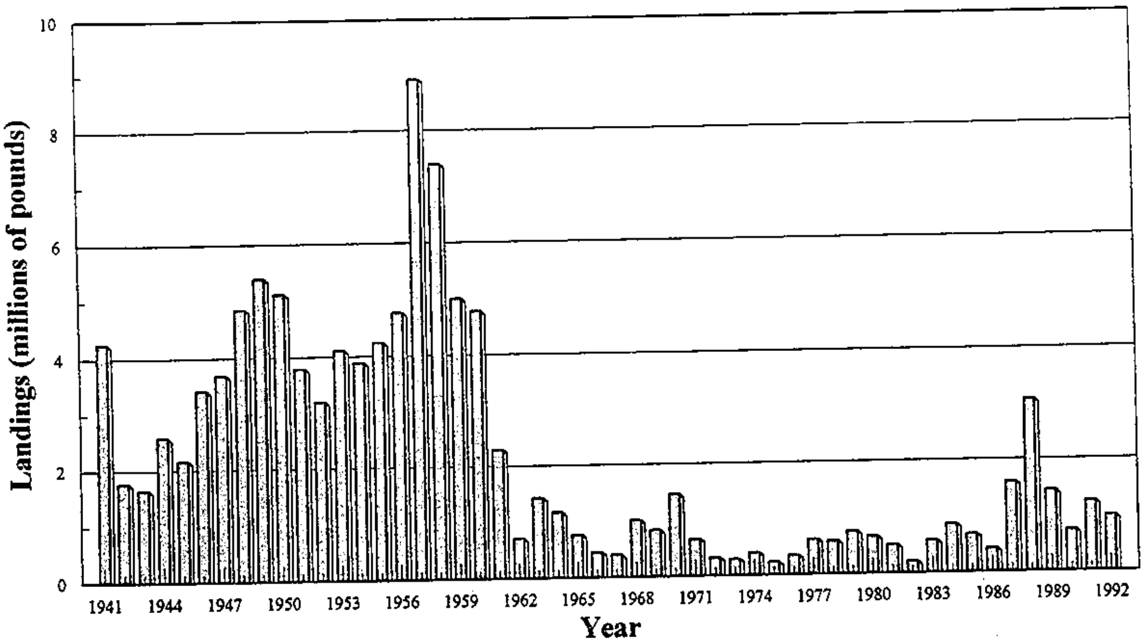
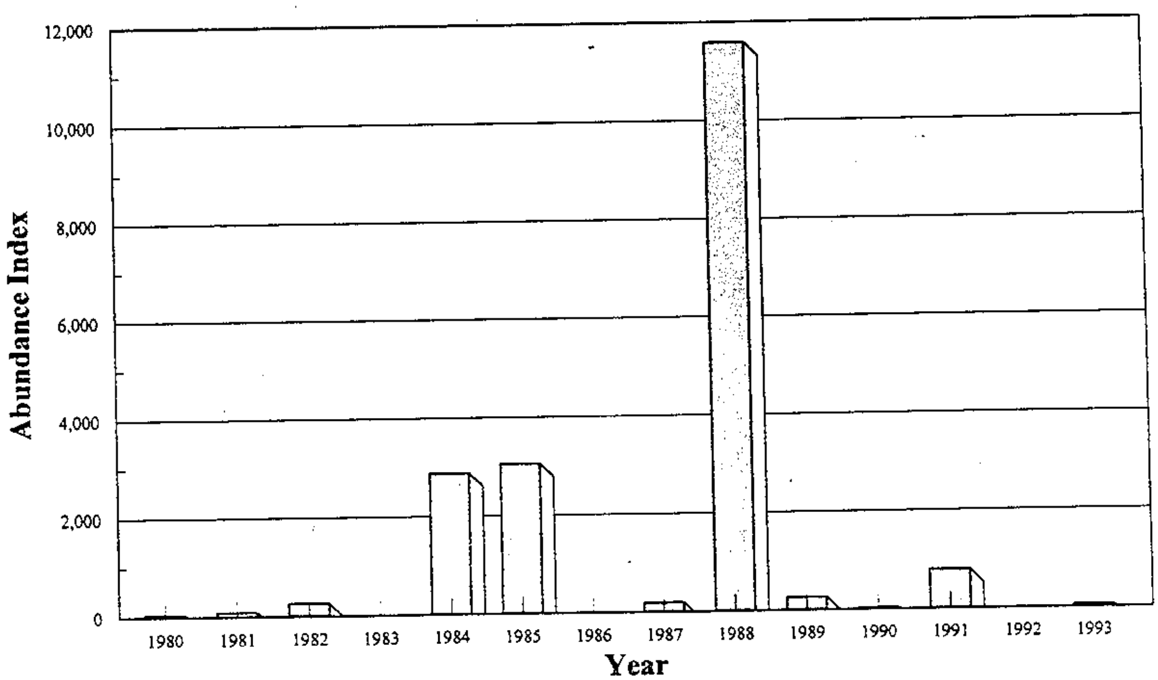


Figure V-19

Dungeness Crab (Age 0+) Abundance Indices

May-July Otter Trawl Survey (1980-1993)



California bay shrimp, *C. franciscorum,* moves between marine and brackish water during its life cycle. The larvae hatch in relatively high salinity water. The post-larvae and juveniles migrate upstream to lower salinity nursery area where they grow for 4-6 months. Mature shrimp, which live between 1 and 2 years, migrate downstream to higher salinity water to complete the life cycle (DFG 1992c).

Each of the shrimp species uses the Estuary as a nursery area to varying degrees. *P. macrodactylus* and *C. franciscorum* are Estuary-dependent. *P. macrodactylus* is most common in Suisun Bay, the western Delta, and areas adjacent to freshwater sources, such as the mouths of creeks in South and San Pablo bays. *C. franciscorum* is found in brackish, relatively warm water, *C. nigricauda* is found in higher salinity and cooler water, and *C. nigromaculata* is primarily a coastal, shallow water species that is most commonly found in the nearshore ocean area adjacent to San Francisco Bay. *H. stimpsoni* is also considered a coastal species, although it is locally abundant in the bay (DFG 1994b).

Population Trends. *Crangon* spp. and *Palaemon* support a commercial fishery in the bays. Early in the century, when there was a large market for dried shrimp, over 3 million pounds per year were landed (Figure V-20). Since 1980, this fishery has landed between 100,000 and 200,000 pounds of shrimp annually. To protect juvenile striped bass, shrimp fishing has been prohibited upstream of Carquinez Strait since 1985 (DFG 1994b).

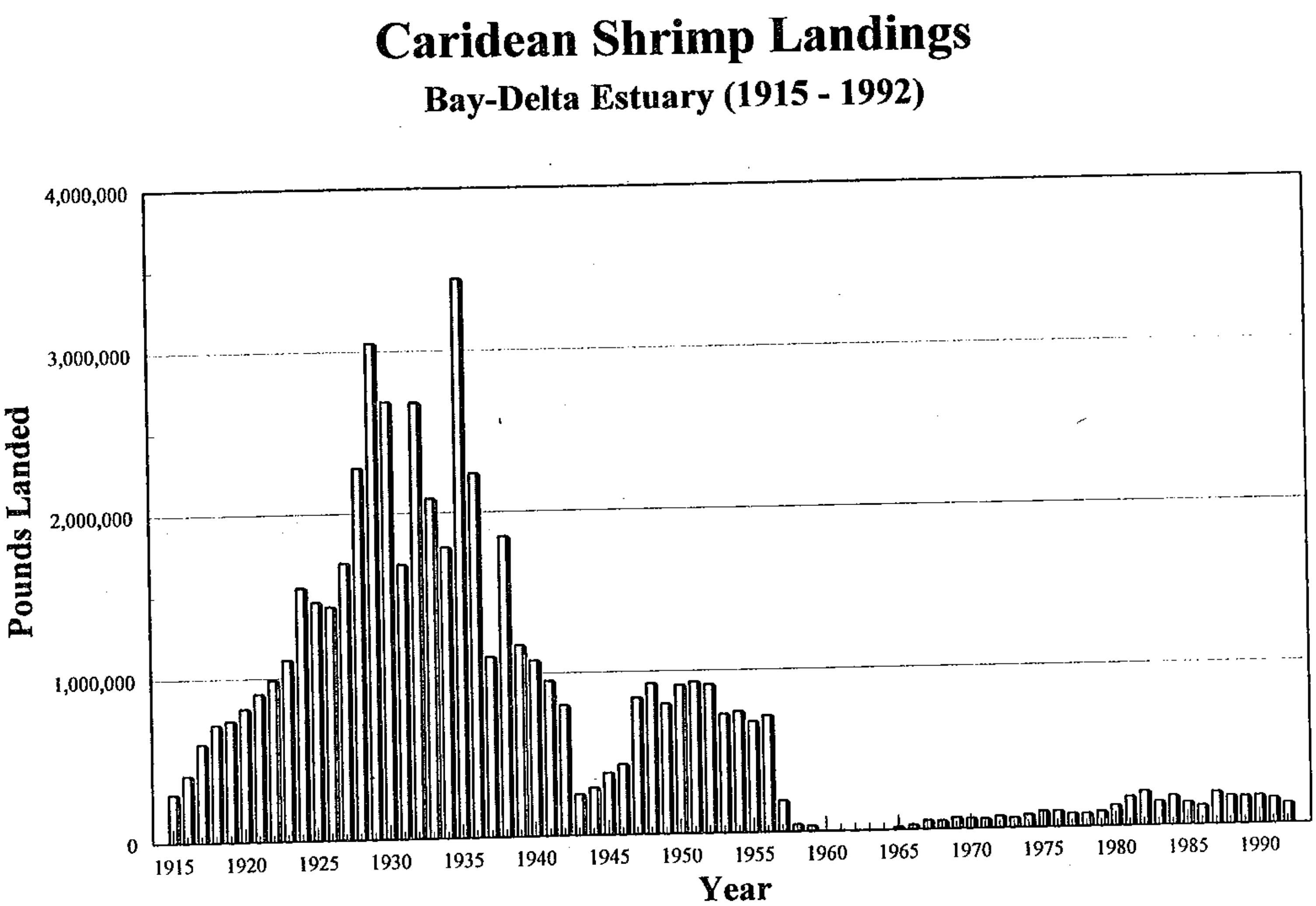
Aside from the commercial catch data, dependable abundance indices for shrimp are only available since 1980 (Figure V-21). Since that time, there has been a change in species composition in the catches. In the early-1980's, *C. franciscorum* dominated the catches; but in the late-1980's and early-1990's, *C. nigricauda* was dominant, and *C. nigromaculata* and *H. stimpsoni* increased in abundance. This change was caused in part by the relatively stable, high salinities associated with the drought, resulting in increased habitat for species that prefer higher salinities, but decreased habitat for *C. franciscorum*, which prefers lower salinities. Abundance data for *P. macrodactylus* are inconclusive (survey methods probably are inadequate for this species) (DFG 1994b).

Reflecting this change in species composition, the contribution of shrimp catches in San Pablo and Suisun bays to the total abundance index declined, while the contribution of Central Bay catches increased. In 1992, the Suisun Bay index decreased to a study period low with only a 3 percent contribution to the total index (DFG 1994b).

Biomass indices, which serve as a relative measure of the weight of shrimp available as a food source, have declined since 1986 (Figure V-21). The divergence between the abundance and biomass indices during the recent drought is due to an increase in abundance of juveniles and species that do not grow as large as *C. franciscorum* (DFG 1994). Figure V-22 illustrates the decline in immature *C. franciscorum* abundance indices since the early-1980's.

Causes of Decline. Unlike the other caridean shrimp, *C. franciscorum* decreased in abundance in recent years. *C. franciscorum*, which can be found at a wide range of salinities

Figure V-20



Caridean Shrimp Landings
Bay-Delta Estuary (1915 - 1992)
4,000,000
3,000,000
2,000,000
1,000,000
0
Pounds Landed
1915
1920
1925
1930
1935
1940
1945
1950
1955
1960
1965
1970
1975
1980
1985
1990
Year

Figure V-21

Caridean Shrimp Abundance and Biomass Indices

Bay-Delta Estuary (1980-1993)

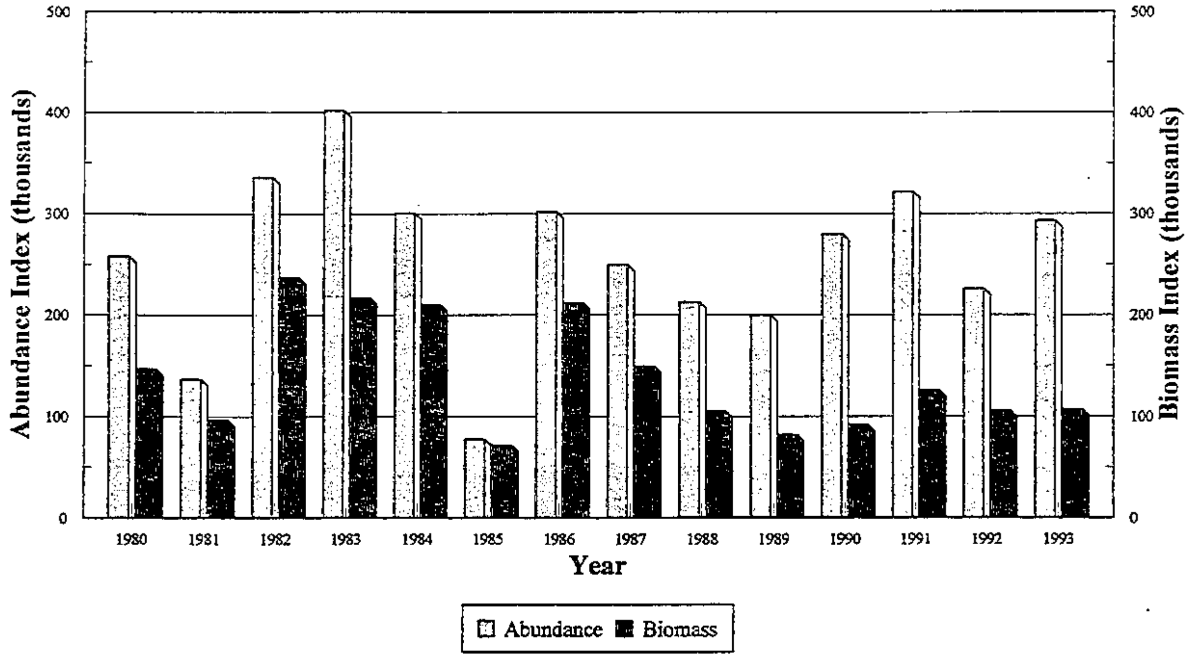
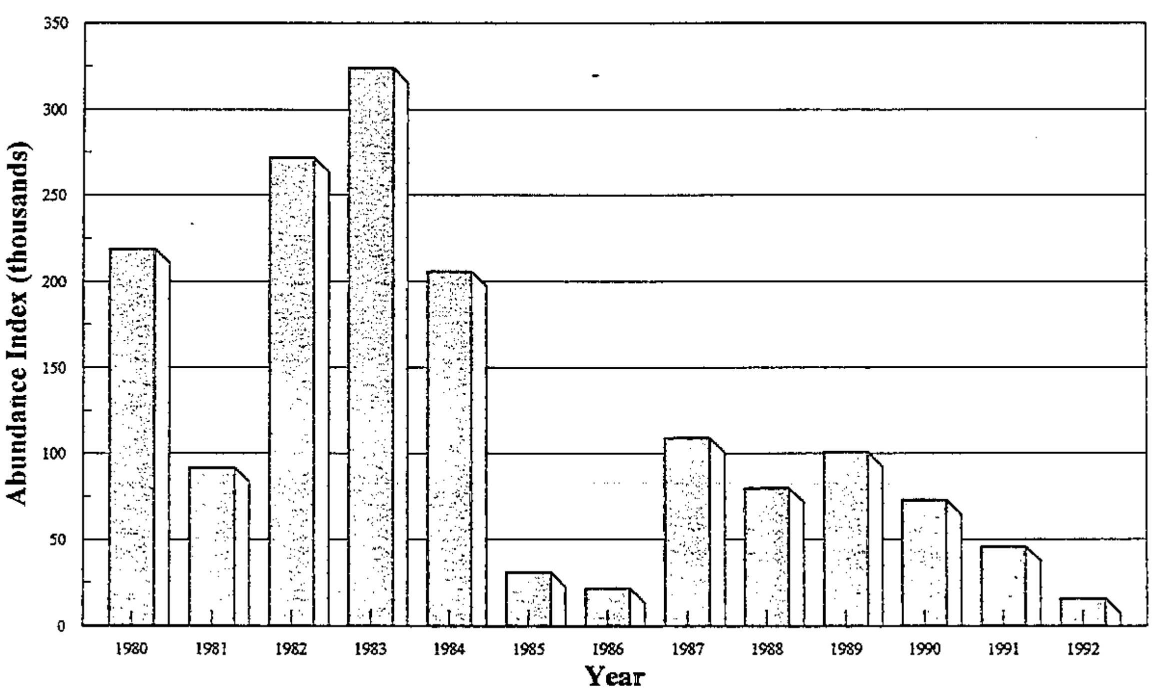


Figure V-22

Immature *Cragon franciscorum* Abundance Index

Bay-Delta Estuary (1980-1992)



and temperatures, exhibits a straightforward response to outflow alone, whereas other species of shrimp appear to respond more to salinity (SFEP 1992a). The response of *C. franciscorum* to outflow has been attributed to two flow-related mechanisms. First, higher river inflows result in larger landward-flowing currents, transporting the small post-larval shrimp into the bay and dispersing them upstream. Second, higher river inflows reduce bay salinity and increase the amount of suitable nursery habitat for juvenile shrimp (Jassby et al. 1994; SFEP 1992a).

The period March to May has been identified as the most critical period for freshwater outflow in the establishment of a strong year class of immature *C. franciscorum* in the bay. There is also a strong positive relationship between the annual abundance of mature *C. franciscorum* and freshwater outflow the previous spring (March-May) when they were recruited to the bay. Therefore, an increase in outflow in March to May should result in an increase in the abundance of *C. franciscorum*. Significant relationships between abundance and outflow were not found for the other species of shrimp. The other species of *Crangon* and *Heptacarpus* are much less estuarine-dependent than *C. franciscorum*, which is affected by freshwater outflows its entire life cycle, and their abundance is affected more by ocean conditions (DFG 1992c).

The decreased food abundance (e.g., *N. mercedis*) in Suisun Bay in recent years may also have played a role in reducing the abundance of *C. franciscorum* since it is the only crangonid found in abundance that far upstream (SFEP 1992a). Also, as with the zooplankters, *Eurytemora* and *Neomysis*, the decline of *C. franciscorum* has also been associated with the introduction of the zooplankters, *Sinocalanus doerrii* and *Pseudodiaptomus forbesi*, and the Asian clam, *Potamocorbula amurensis*, as well as pumping by the SWP and the CVP (NHI 1992a).

4. Freshwater Fish

The Bay-Delta Estuary has both native and introduced freshwater fish species. Most native fish are large minnows, such as the Sacramento splittail, Sacramento squawfish, hitch, Sacramento blackfish, and hardhead. The Sacramento splittail is discussed here as a representative native freshwater species in the Estuary. Among the many introduced species in the Estuary are centrarchids (sunfish such as bluegill and smallmouth bass), catfish, carp, threadfin shad, and inland silverside. Because there is more information on the population trends of white catfish than for any other resident freshwater species, it will be discussed here as a representative of introduced freshwater species in the Delta.

a. **Sacramento Splittail**. The Sacramento splittail (*Pogonichthys macrolepidotus*) is a large minnow endemic to the Bay-Delta Estuary. Historically, it was found through low elevation lakes and rivers of the Central Valley from Redding to Fresno. Data from recent surveys indicate that the splittail is still found in many of the major Central Valley tributaries, specifically, the American, Tuolumne, San Joaquin, and Mokelumne rivers. Splittail are also found in Suisun Bay, Suisun Marsh, Napa Marsh, and the Delta. Although the Sacramento splittail is considered a freshwater species, the adults and sub-adults have an unusually high tolerance for saline water, up to 10-18 ppt (Meng 1993), for a member of the minnow family (DFG 1994b). Therefore, the Sacramento splittail is often considered an estuarine species. The salt tolerance of splittail larvae is unknown (DFG 1992b).

The Sacramento splittail, which has a high reproductive capacity, can live 5-7 years and generally begin spawning at 1-2 years of age (Hanson 1994a). Spawning, which seems to be triggered by increasing water temperature and day length, occurs over beds of submerged vegetation in slow-moving stretches of water, such as flooded terrestrial areas and dead-end sloughs. Adults spawn from March through May. Hatched larvae remain in shallow, weedy areas until they move to deeper offshore habitat later in the summer. Young splittail may occur in shallow and open waters of the Delta and San Pablo Bay, but they are particularly abundant in the northern and western Delta (DFG 1992b, DWR 1992a).

Splittail are benthic foragers that feed extensively on opossum shrimp (*Neomysis mercedis*) and opportunistically on earthworms, clams, insect larvae, and other invertebrates. They are preyed upon by striped bass and other predatory fish in the Estuary (NHI 1992b).

Population Trends. Abundance indices of the Sacramento splittail, based on fall mid-water trawl catches, have varied over the years (Figure V-23). The indices, based on sampling juvenile splittail, were relatively high in the late 1960's (e.g., 66.3 in 1967) and then declined severely until 1977. After 1977, splittail abundances increased to a record high of 153.2 in 1983, after which the index declined to 3.6 in 1992. Likewise, the Bay Study indices for splittail were highly variable. Maximum abundances were attained in 1982, 1983, and 1986, all wet years; but abundance indices declined through the late 1980's, slightly increased during the early 1990's (DFG 1994b), and declined again in 1992 (Cech and Young 1994).

Because of the apparent reduced abundance and distribution of the Sacramento splittail, it is considered a species of special concern by the DFG, and the USFWS is contemplating its listing as a threatened species under the federal ESA (DFG 1994b).

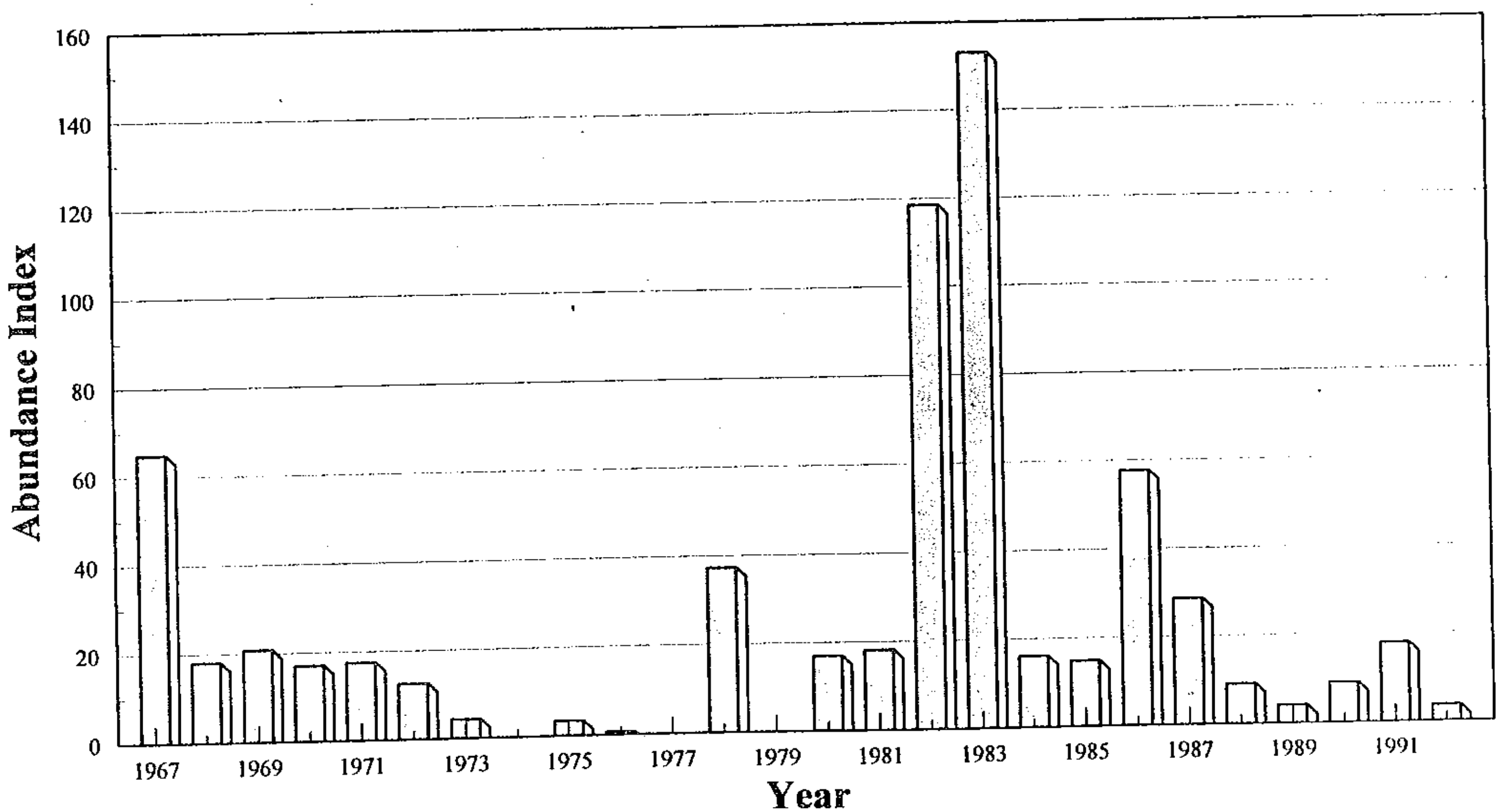
The DWR analyzed the various catch records available and made the following findings. The number of young splittail in the Estuary may have declined over the 6-year drought; however, some recent data suggest that recent levels have improved. In upstream areas, beach seine results indicate that recent young-of-the-year (YOY) abundance levels are similar to, or perhaps greater than, pre-decline levels. Adult abundance trends also indicate that the number of spawners has not declined, except in the region of Suisun Marsh and Chipps Island. The ability of the population to recover does not appear to have been compromised by conditions in the past decade. In 1993, adult indices were fairly strong in most surveys and YOY production appeared to have been substantial, based on catches upstream in the Sacramento River (DWR 1995a).

Causes of Decline. The Sacramento splittail has declined in abundance because of loss or alteration of lowland habitats following dam construction, water diversion, and agricultural development (Cech and Young 1994). The Sacramento splittail has lost much of

Figure V-23

Sacramento Splittail Abundance Indices

Fall Mid-Water Trawl Survey (1967-1992)



Note: Not sampled in 1974 and 1979.

its original foraging and spawning habitats through losses of marshlands due to land reclamation activities (CUWA 1994, DFG 1992b).

Within the Estuary, it appears that the decline in splittail abundance is a result of habitat constriction associated with the reduction of Delta outflow and changes in hydrodynamics due to Delta exports. Shallow-water habitat is important for rearing of young, and freshwater outflow may be important for the dispersion of young to appropriate nursery areas in Suisun Bay (Meng 1993). Although little data exist regarding its environmental requirements or tolerances, it is likely that high salinity restricts the downstream range of the splittail (Cech and Young 1994).

Sacramento splittail populations fluctuate on an annual basis depending on spawning success and year class strength (NHI 1992b). Successful reproduction is strongly associated with high outflows preceding, during, and following spawning, as demonstrated by high correlations between abundance of splittail in the fall mid-water trawl survey and various monthly combinations of Delta outflow from the previous winter through early summer (DFG 1992b). The DFG's statistical relationship between the juvenile splittail abundance indices and March-May (the spawning period) Delta outflow for the years 1967-1993 indicates that increased outflow in the spring corresponds with increased splittail abundance indices (see Figure VI-8 in Chapter VI).

Abundance is also correlated with the duration of floodplain inundation, which may provide a large amount of additional spawning, rearing, and foraging habitat in wet years. Except for 1993 and the current water year, little flooding has occurred in the range of splittail since 1986, perhaps leading to a series of weaker year-classes in the Estuary. The DWR states that: "Although hydrology appears to be important to the production of young splittail, the USFWS beach seine data and recent egg and larval analyses show that spawning can be successful in many areas of the Sacramento and San Joaquin rivers and the northern and central Delta in both wet and dry years" (DWR 1995a).

The strong correlation of the abundance of young Sacramento splittail with freshwater outflows (NHI 1992b; DFG 1992b) during the late winter and spring accounts, in part, for the observed decline in juvenile production during the recent drought period (Hanson 1994a). The corresponding relationship for adult splittail is very weak, indicating that the relationship between splittail and outflow is particularly important for reproduction (Meng 1993). Because a strong stock-recruitment relationship has not been established, the relationship between the observed decline in juvenile splittail abundance indices and the abundance, age structure, reproductive capacity, and population dynamics for the adult splittail population is unknown (Hanson 1994a).

The major factor cited in reducing splittail abundance is loss of spawning and nursery habitat due to reclamation activities (DFG 1992b), bank protection, and channelization. In addition, introduced species (i.e., planktonic copepods and the Asian clam, *Potamocorbula*) may have reduced the splittail's favored prey, *Neomysis mercedis*, and, therefore, are also possible factors in the decline of Sacramento splittail populations in the Estuary (NHI 1992b).

b. White Catfish. The white catfish (*Ictalurus catus*) was introduced into the Bay-Delta Estuary in 1874 and rapidly increased in abundance. In recent years, the white catfish has supported an important sport fishery (BDOC 1993). In the Estuary, they are most abundant in areas of slow currents and dead-end sloughs. White catfish, which can live in salinities as high as 11 to 12 ppt, are the only catfish common in Suisun Bay (Moyle 1976). As bottom-feeders, they are known to eat the eggs of other fish species (BDOC 1994).

Population Trends. Based on a 1978-1980 tagging study, the adult (≥ 7 inches) white catfish population was estimated at 3-8 million fish. Although population estimates of adult white catfish have not been made since that study, there is evidence that the abundance of white catfish has declined severely since the mid-1970's. For example, incidental catches of young (≤ 4.5 inches) white catfish in the summer tow-net survey (designed for sampling YOY striped bass) ranged from one to four fish per tow from 1969 to 1975; since 1975, the catch has not exceeded one fish per tow and, in several years, has been less than 0.06 fish per tow (Figure V-24) (DFG 1994b).

Likewise, the fall mid-water trawl survey indicates a general decline in white catfish abundance since the early 1970's before the population rebounded in 1992 (Figure V-25). Furthermore, CVP and SWP fish salvage data show that salvaged white catfish have declined dramatically since the late 1960's. Compared to about 8 million catfish salvaged in 1967, in 1990, 203,000 and 33,000 catfish were caught at the CVP and SWP export facilities, respectively (DFG 1994b).

Causes of Decline. Available evidence indicates that catfish reproduction has been concentrated in the southern and eastern Delta, and that this source of recruitment to the overall population has greatly diminished since the early 1970's (BDOC 1993). It is believed that southern Delta water exports have caused the decline in white catfish abundance for the following reasons: (1) the water project intakes draw water from the key reproductive areas for white catfish; (2) the water projects entrain large numbers of catfish; and (3) screening efficiencies for white catfish are low compared to other fish species. Negative correlations between white catfish abundance and the water exports support the hypothesis that losses of catfish to water exports in the southern Delta have depleted the catfish population (DFG 1992f).

5. Estuarine Fish

A completely estuarine species of fish in the Bay-Delta Estuary is the Delta smelt. All other Bay-Delta fish species maintain at least part of their population outside of the Estuary. Because the longfin smelt, which is similar to the Delta smelt, occurs in the Estuary and rarely outside the Golden Gate, it will be considered here following Delta smelt.

a. Delta Smelt. The Delta smelt (*Hypomesus transpacificus*) is a small, short-lived native fish which is found only in the Bay-Delta Estuary. This schooling species inhabits open surface and shoal waters of main river channels and Suisun Bay (DWR 1992a, SFEP 1992a). It was assumed that Delta smelt prefer shallow water; however, a study conducted to

Figure V-24
White Catfish Mean Catch Per Tow
Summer Townet Survey (1969-1992)

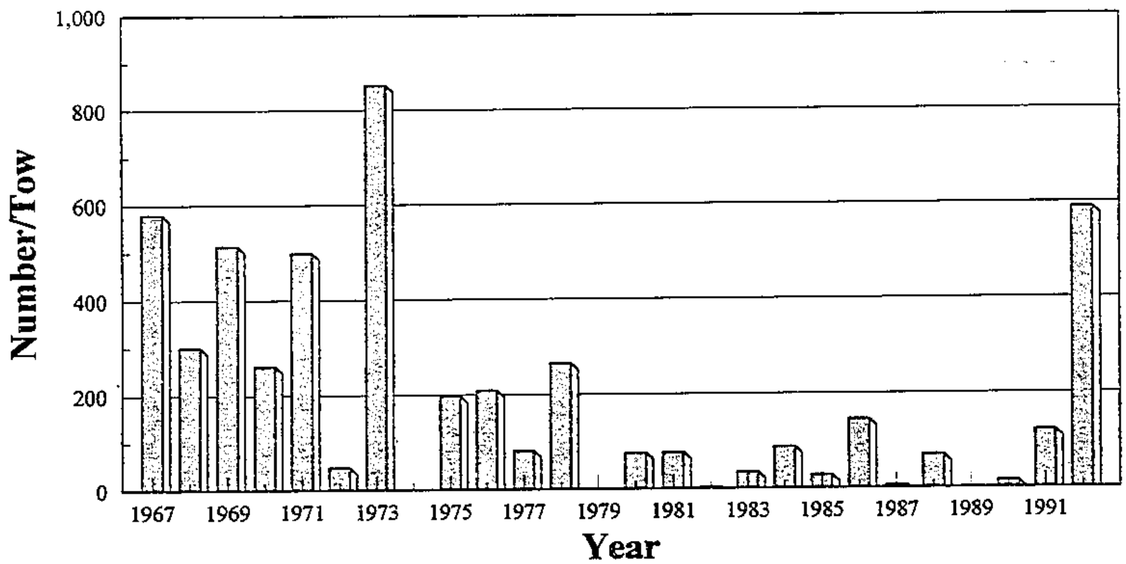
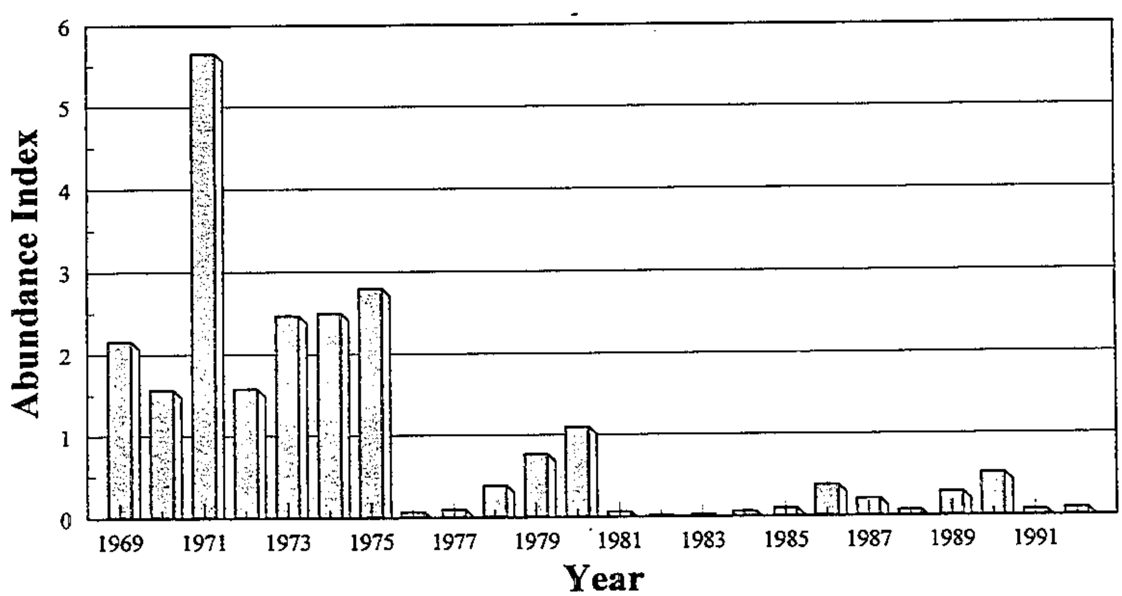


Figure V-25
White Catfish Abundance Indices
Fall Mid-Water Trawl Survey (1967-1992)



determine whether or not this is true was not conclusive. In June 1994, the IEP conducted deep and shallow water sampling in the San Joaquin River, Sacramento River, and Suisun Bay. Delta smelt densities were not significantly different between shallow (less than 10 feet deep) and deep (10-45 feet deep) water areas within the San Joaquin River and Suisun Bay; however, densities were significantly different between shallow and deep water habitats in the lower Sacramento River (Hanson 1994b, DWR 1995b).

Delta smelt have been found as far upstream as Sacramento on the Sacramento River and Mossdale on the San Joaquin River. Their normal downstream limit appears to be western Suisun Bay although, during periods of high outflow, they can be washed into San Pablo and San Francisco bays, but they do not establish permanent populations there (SFEP 1992a). They often inhabit the upper portion of the water column and at salinities ranging from 2-10 ppt (DFG 1992d). Overall, Delta smelt concentrate near or immediately upstream of the low salinity habitat. Their concentration in the low salinity habitat may simply reflect that it is the only remaining area with dense enough populations of their primary prey, copepods (SFEP 1992a).

The Delta smelt has low fecundity and is primarily an annual species, although a few individuals may survive a second year (SFEP 1992a). The location and season of Delta smelt spawning vary from year to year. Spawning, which occurs in shallow fresh or slightly brackish water in or above the low salinity habitat (DFG 1992d, USFWS 1994), has been known to occur at various sites within the Delta, including the lower Sacramento and San Joaquin rivers and Georgiana Slough, and in sloughs of the Suisun Marsh (USFWS 1994). It appears that few Delta smelt spawn in the southern Delta. Based on egg and larval trawls over the last few years, it appears that, at least in low-flow years, a significant portion of Delta smelt spawning now takes place in the northern and western Delta (DWR 1992a).

Spawning may occur from late winter (December) to early summer (July). In 1989 and 1990, peak spawning occurred in late-April and early-May (USFWS 1994). The adhesive eggs descend through the water column and likely attach to submerged substrates such as tree roots, vegetation, and gravel (DFG 1992d). After hatching, the planktonic larvae are transported downstream to the low salinity habitat where they feed on zooplankton (USFWS 1994).

After hatching, many Delta smelt may be transported downstream to the low salinity habitat while many also remain upstream to rear in the channels of the lower Sacramento and San Joaquin rivers. The mid-water trawl results, for the period of 1967-1981, show an average of 37 percent of the Delta smelt were caught in Suisun Bay and 63 percent were caught in the Delta. During the period of 1969-1981, more Delta smelt were caught in the Delta than in Suisun Bay. The summer tow-net index indicated an average of 45 percent of the smelt reared in Suisun Bay, while 55 percent reared in the upstream areas (DWR 1995b).

Population Trends. Seven surveys, although not specifically designed to gather data on Delta smelt populations in the Estuary, have charted the abundance of Delta smelt. The summer tow-net survey, which began in 1959 and was primarily designed to measure striped bass abundance, is considered one of the best measures of Delta smelt abundance because it covers much of the species' habitat and represents the longest historical record. Although the abundance indices vary considerably (Figure V-26), they have generally remained low between 1983 and 1993, although the 1993 index is the highest since 1982 (DWR and USBR 1993). The recent increase may be due to an artifact of the sampling program and recent smelt distribution patterns (NHI 1992a). The reduced population levels during the 1980's appear to have been consistent throughout the Delta and Suisun Bay, but declines may have occurred as early as the mid-1970's in the eastern and southern portions of the Delta (DWR and USBR 1993).

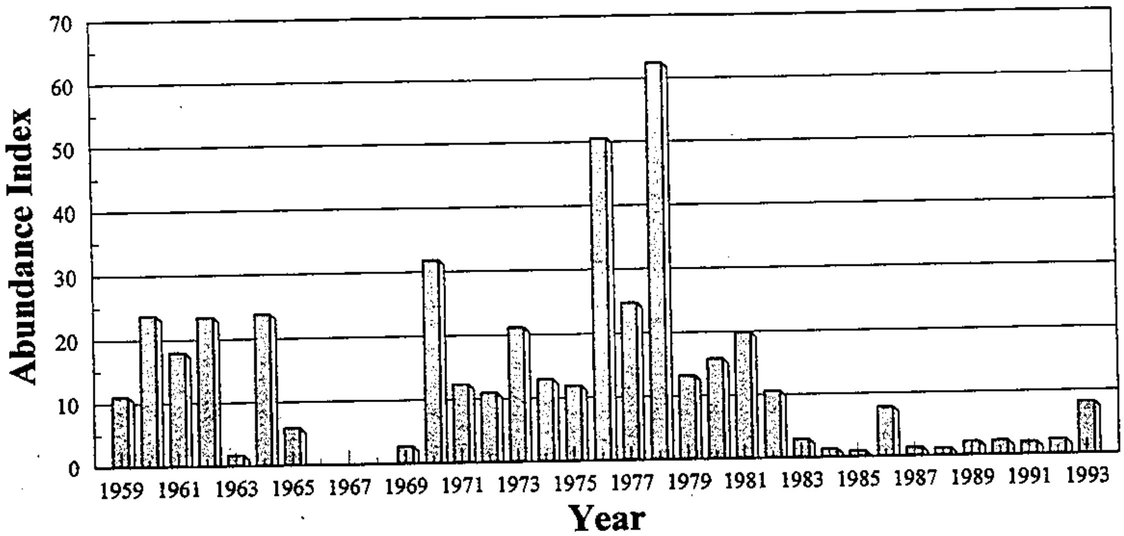
Information from the other six independent data sets have demonstrated a dramatic decline in the Delta smelt population, with particularly low levels since 1983 (DFG 1994b). The fall mid-water trawl survey, which measures relative abundance of adult smelt, yields mean monthly catches of Delta smelt that vary from month to month and from year to year. From 1967 through 1975, fall catches were generally greater than 10 smelt per trawl per month (in 6 of 8 years); from 1976 through 1989, catches were generally less than 10 smelt per trawl per month (in 13 of 14 years). Since 1986, catches have averaged considerably less than one smelt per trawl per month. The frequency of occurrence of Delta smelt in the trawls has also declined. Prior to 1983, Delta smelt were found in 30 percent or more of the fall trawl catches. In 1983-1985, they occurred in less than 30 percent of the catches, and since 1986, they have been caught in less than 10 percent of the trawls (SFEP 1992a). Figure V-27 presents the fall mid-water trawl survey data as abundance indices for adult Delta smelt. Unlike the summer tow-net survey indices, the mean catches of Delta smelt have not declined in the mid-water trawl survey. The smelt population is more dispersed in the summer than in the fall. The summer populations have decreased in average densities while the fall populations have decreased numbers of schools (DFG 1992d). Data from the Bay Study and the Suisun Marsh study show sharp declines in Delta smelt at about the same time. The exact timing of the decline is different in most of the sampling programs but falls between 1982 and 1985 (SFEP 1992a).

As a result of the sharp decline in abundance since the early 1980's, the Delta smelt was listed as a federal threatened species by the USFWS in March 1993 and as a State threatened species by the DFG in December 1993.

Causes of Decline. There are a number of theories that attempt to explain the decline in Delta smelt. Some of the theories have been disputed and still other theories are in the early stages of development. The following section presents the various theories and cites the sources of the information.

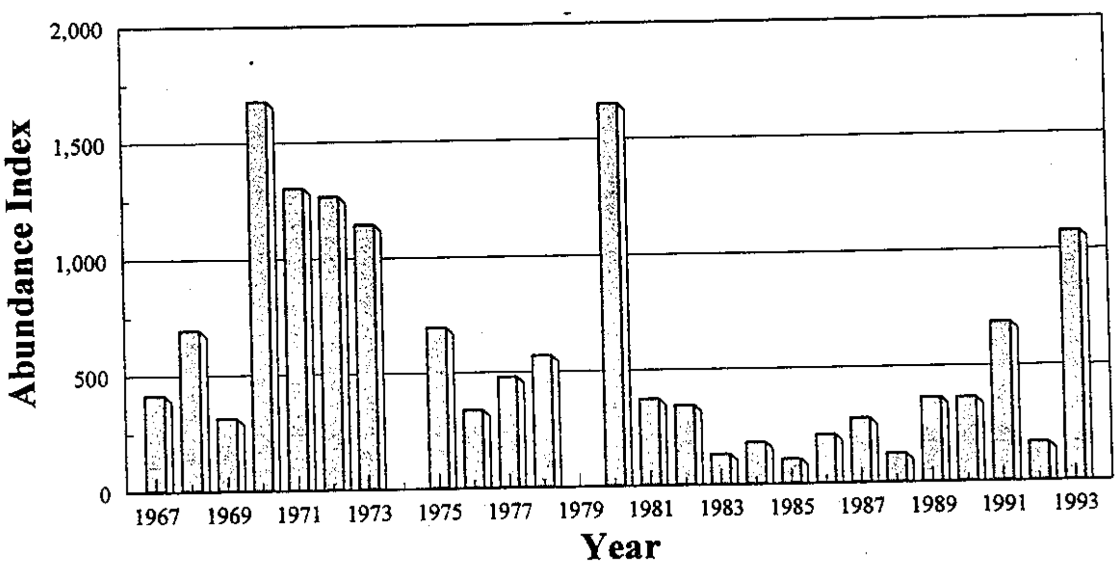
Declines in Delta smelt have been attributed primarily to restricted habitat and increased losses through entrainment by Delta diversions (DWR 1992a, SFEP 1992a, USFWS 1994). Reduced available habitat and increased entrainment occur when the low salinity habitat moves out of the productive shallows of Suisun Bay and into the channels of the lower Sacramento and San Joaquin rivers as a result of low Delta outflow. The theory is that the movement of the low salinity habitat upstream to the river channels decreases the amount of

Figure V-26
Delta Smelt Abundance Indices
Summer Townet Survey (1959-1993)



Note: Not sampled in 1966, 1967, and 1968.

Figure V-27
Delta Smelt Adult Abundance Indices
Fall Mid-Water Trawl Survey (1967-1993)



Note: Not sampled in 1974 and 1979.

area that can be occupied by smelt, and probably results in decreased phytoplankton and zooplankton as well (SFEP 1992a). When the Delta smelt are upstream in the Delta, they are more vulnerable to entrainment by the pumps of the SWP and the CVP, as well as local agricultural diversions (DWR 1992a, NHI 1992a, SFEP 1992a). Diversions in the northern and central Delta, where smelt are most abundant, are likely the greatest source of entrainment (USFWS 1994).

Increasing diversions of fresh water from the Estuary have shifted the location of the low salinity habitat and have altered the flow patterns of the Delta during most months of the year. Prior to 1984, largely before the sharp decline in Delta smelt abundance, the location of the low salinity habitat was generally in Suisun Bay during October through March, except during months with very high outflows or during years of extreme drought; during April through September, the low salinity habitat was mainly in the river channels. Since 1984, the low salinity habitat has been located mainly in the channels of the rivers during all months of the year (SFEP 1992a).

The decline in Delta smelt also coincides with increases in the proportion of water diverted in recent years. Since 1984, the proportion of the water diverted at the export pumps from October through March has been higher, and has stayed higher for longer periods of time than during any previous period, including the severe 1976-1977 drought. Because high levels of diversions draw Sacramento River water across the Delta and into the channels of the San Joaquin River downstream of the pumps, the lower San Joaquin River has a net flow upstream during these periods. The number of days of net reverse flow of the San Joaquin River has consequently increased in recent years, especially during the Delta smelt spawning period. During the months when Delta smelt are spawning, the changed flow patterns resulting from Delta diversions presumably draw larvae into the Delta channels, where they can be exported through the pumps along with locally-produced larvae (Moyle et al. 1992, SFEP 1992a).

The DWR disagrees with the hypothesis that the decline in Delta smelt is coincident with increases in the proportion of water diverted in recent years (DWR 1995b). The DWR argues that although Moyle and Herbold (1989) indicated that low Delta smelt abundance indices (fall mid-water trawl data) were associated with the number of days of negative values of QWEST, there was found no statistical association between Delta smelt abundance and the number of days of reverse flows. Nevertheless, it was observed that years of high smelt abundance usually had positive flow in the lower San Joaquin River and years of low smelt abundance usually had a higher number of days of reverse flows. Moyle and Herbold (1989) concluded that the frequency of reverse flow in the lower San Joaquin River was probably limiting smelt recruitment, but that it was not a simple direct relationship. Furthermore, results of statistical analyses between reverse flows and smelt abundance are confounded by both the inability to measure reverse flows and autocorrelations with other environmental variables.

Moyle et al. (1992) found that, until 1984, water years with 100 days of reverse flow were sporadic and rarely occurred during the Delta smelt spawning season (February-May). From

1985 to 1989, reverse flows have characterized the lower San Joaquin River for more than 150 days of the year and, in every year except 1986, reverse flows have occurred for 15-85 days of the spawning season. This pattern continued in 1990 through 1992. The DWR could not find a statistical relationship between the number of days of reverse flow and the Delta smelt mid-water trawl index (1967-1992) or the tow-net index (1959-1993) (DWR 1995b).

The relationship between Delta outflows and smelt abundance is not a simple one (Moyle et al. 1992). In fact, high outflows, such as those that occurred in February 1986, may have flushed Delta smelt out of the Estuary (SFEP 1992a). Unlike striped bass, longfin smelt, and other species with planktonic larvae, the Delta smelt does not show a strong correlation in abundance with outflows (DWR 1992a, NHI 1992, SFEP 1992a). The substantial annual variation in abundance of smelt probably masks any long-term trends linked to outflows (NHI 1992a). It is believed that February-June Delta outflows are needed to transport larval and juvenile Delta smelt away from the influence of the export pumps and into low salinity, productive rearing habitat in Suisun Bay and Suisun Marsh (USFWS 1994).

Hanson (1994c) conducted an analysis to specifically test the hypothesis that the abundance of adult Delta smelt in the fall is dependent upon geographic distribution of rearing juvenile Delta smelt earlier in the year. The importance of the geographic distribution of Delta smelt during the rearing period has been linked to: (1) the transport and distribution of early lifestages into areas downstream of the Sacramento-San Joaquin confluence; and (2) the importance of habitat within productive shallow-water areas in Suisun Bay. Under these theories, the expected result is increased abundance of sub-adult and adult Delta smelt the following fall. Hanson found no significant relationship between the percentage of juvenile Delta smelt collected downstream of the Sacramento-San Joaquin River confluence and the corresponding fall mid-water trawl abundance index. This finding does not support the theory that distributing larval and juvenile Delta smelt into Suisun Bay will result in a large fall mid-water trawl index.

Other contributing factors in the precipitous decline in the Delta smelt population may be: the presence of toxic compounds in the water (DFG 1992d, SFEP 1992a); displacement of native copepods by introduced species (DFG 1992d, SFEP 1992a); invasion of the Estuary by the Asian clam, *Potamocorbula amurensis* (SFEP 1992a); predation (USFWS 1994); very high outflows; and low spawning stock (DFG 1992d).

Pesticides in the Sacramento River at concentrations potentially harmful to larval fish and zooplankton have been recorded in recent years by the Central Valley RWQCB. Though their effects on the Delta smelt are unknown, these pesticides have occurred at high levels in fresh water prior to the most recent decline of the smelt. However, the concentration of smelt in the low salinity habitat may have allowed them to avoid the effects of pesticides through the dilution of the contaminated fresh water by inflowing seawater (SFEP 1992a).

The 1988 decline of *Eurytemora affinis*, a copepod which has been the primary food supply of Delta smelt, has been identified as a possible factor in the decline of smelt in the Estuary

(DFG 1992d). However, it may be that the decline in *E. affinis* abundance, due to the introduction of other copepod species, is not an important factor because the smelt has shifted its diet and now consumes *Pseudodiaptomus forbesi*, which was introduced into the Estuary in 1986. The clam, *Potamocorbula amurensis*, may have an indirect effect on smelt populations by reducing its food supply (SFEP 1992a, Kimmerer et al. 1994).

Predation by striped bass and other predatory fish which occur at the pumping plants and other diversions which entrain fish has also been named as a possible factor in the decline of Delta smelt (USFWS 1994). However, it is questionable if this is an important factor when both striped bass and Delta smelt were abundant in the 1960's, and the smelt was not a significant prey of the bass (DFG 1992d). It is also possible that predation on Delta smelt larvae by inland silversides, whose introduction and population explosion occurred concurrently with the early declines in Delta smelt abundance, is a contributor to the declines in smelt populations; however, research on the inland silverside in the Estuary is lacking (CUWA 1994).

Spawning stock does not appear to have a major influence on Delta smelt year class success. However, the low fecundity of this species, combined with planktonic larvae which likely have high rates of mortality, requires a large spawning stock if the population is to perpetuate itself. This may not have been an important factor in the decline of Delta smelt, but it may be important for its recovery (DFG 1992d).

b. Longfin Smelt. The longfin smelt (*Spirinchus thaleichthys*) is a small, planktivorous fish that is found in several Pacific coast estuaries from San Francisco Bay to Prince William Sound, Alaska. Until 1963, the population in San Francisco Bay was thought to be a distinct species. Within California, longfin smelt have been reported from Humboldt Bay and the mouth of the Eel River. However, data are infrequently collected from Humboldt Bay, and there are no recent records from the Eel River (SFEP 1992a). In California, the largest longfin smelt reproductive population inhabits the Bay-Delta Estuary (DFG 1992c).

Longfin smelt can tolerate salinities ranging from fresh water to sea water. Spawning occurs in fresh to brackish water over sandy-gravel substrates, rocks, or aquatic vegetation (Meng 1993). In the Bay-Delta Estuary, the longfin smelt life cycle begins with spawning in the lower Sacramento and San Joaquin rivers, the Delta, and freshwater portions of Suisun Bay (SFEP 1992a). Spawning may take place as early as November and extend into June, with the peak spawning period occurring from February to April (Meng 1993). The eggs are adhesive and, after hatching, the larvae are carried downstream by freshwater outflow to nursery areas in the lower Delta and Suisun and San Pablo bays (SFEP 1992a). The principal nursery habitat for larvae are the productive waters of Suisun and San Pablo bays. Adult longfin smelt are found mainly in Suisun, San Pablo, and San Francisco bays, although their distribution is shifted upstream in years of low outflow (Meng 1992).

With the exceptions that both longfin smelt and Delta smelt spawn adhesive eggs in river channels of the eastern Estuary and have larvae that are carried to nursery areas by freshwater outflow, the two species differ substantially. Consistently, a measurable portion of the longfin smelt population survives into a second year (SFEP 1992a). During the second year of life, they inhabit San Francisco Bay and, occasionally, the Gulf of the Farallones; thus, longfin smelt are often considered anadromous. Longfin smelt are also more broadly distributed throughout the Estuary, and are found at higher salinities, than Delta smelt. Because longfin smelt seldom occur in fresh water except to spawn, but are widely dispersed in brackish waters of the Bay, it seems likely that their range formerly extended as far up into the Delta as salt water intruded. The easternmost catch of longfin smelt in the fall mid-water trawl was at Medford Island in the central Delta. They have been caught at all stations of the Bay Study. A pronounced difference between the two species in their region of overlap in Suisun Bay is by depth; longfin smelt are caught more abundantly at deep stations (>10 m), whereas Delta smelt are more abundant at shallow stations (<3 m) (SFEP 1992a).

Population Trends. The longest index of longfin smelt abundance in the Estuary comes from the fall mid-water trawl survey which began in 1967. The index represents at least two year classes; however, YOY are usually predominant. Since 1967, the longfin smelt abundance index has fluctuated widely from year to year (Figure V-28). The abundance index was high in 1980, low in 1981, and high again in 1982. Since 1982, when the index was 63,000, the indices have declined precipitously. In 1992, the longfin smelt abundance index was about 14 (DFG 1994b). As recently as 1983, the longfin smelt was one of the most abundant species in San Francisco Bay (NHI 1992b). Yet since 1984, the fall mid-water trawl data indicate a 90 percent decline in the longfin smelt population (Meng 1993).

Data from the Bay Study mid-water and otter trawl sampling effort (Figure V-29), which began in 1980, substantiate the decline detected by the fall mid-water trawl program. These data show that YOY longfin smelt were generally much more abundant during the early- and mid-1980's than from 1987 to 1993 (DFG 1994b).

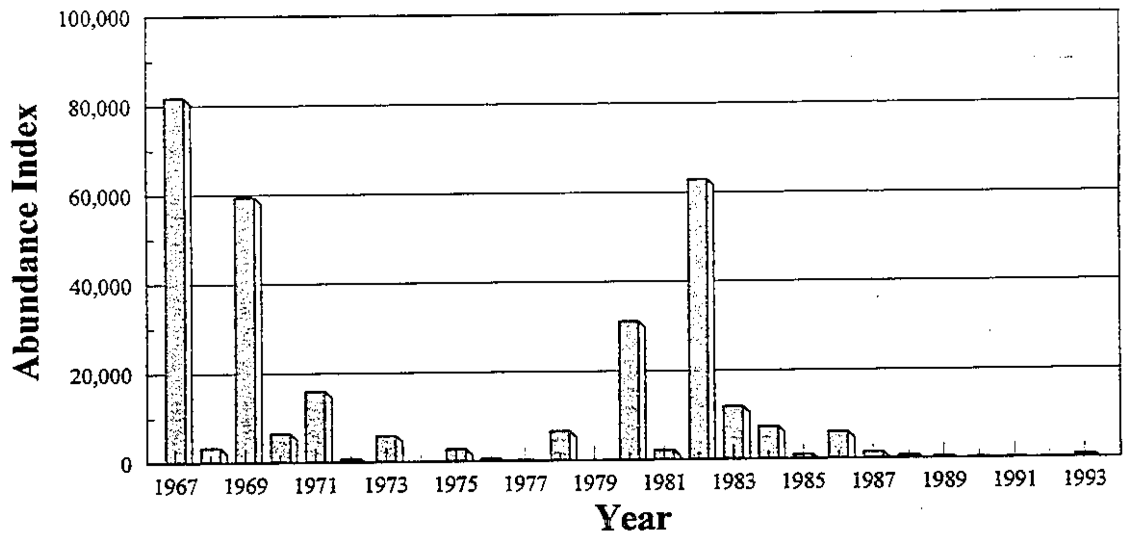
In both the South and Central bays, a brief dominance by longfin smelt occurred in the mid-water catch in 1983. In San Pablo and Suisun bays, their abundance in 1983 was lower than their abundance in 1982; thus, supporting the idea of washout from upstream. Longfin smelt failed to recover in 1986, nominally a wet year, because record flows in February presumably flushed a high percentage of mature adults out of the Estuary. Unlike Delta smelt, which declined in frequency of occurrence but not in abundance at the stations at which they are still caught, longfin smelt have retained most of their earlier distribution but their catch at each station has declined. Longfin smelt have nearly disappeared from San Pablo Bay (SFEP 1992a).

Although longfin smelt populations were known to be affected by freshwater inflow to the Estuary, there has been little concern for their persistence in the Estuary as they have been regarded as abundant and widely distributed, with additional populations in other California estuaries. A recent compilation of fish species of special concern for California (Moyle et al. 1989), for example, does not list longfin smelt (SFEP 1992a). However, the recent dramatic decline in longfin smelt abundance has prompted a petition to the USFWS to list

Figure V-28

Longfin Smelt Abundance Indices

Fall Mid-Water Trawl Survey (1967-1993)

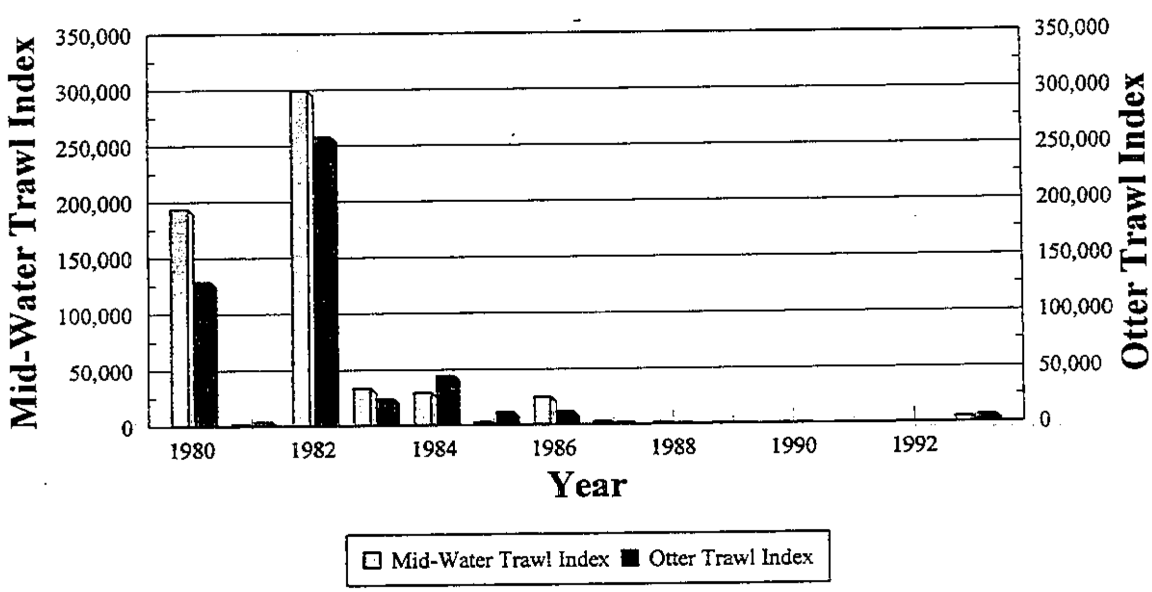


Note: Not sampled in 1974 and 1979.

Figure V-29

Longfin Smelt Young-of-Year Abundance Indices

Otter and Mid-Water Trawl Surveys (1980-1993)



this fish as a threatened species in California (NHI 1992b). The USFWS has determined that listing of the longfin smelt is not warranted at this time because, although the southernmost populations are declining, the species may be surviving and reproducing in numerous other bays and estuaries along the Pacific Coast north of San Francisco Bay. Furthermore, based on current knowledge, the Bay-Delta Estuary population does not seem to be biologically significant to the species as a whole (Federal Register, Vol. 59, No. 4, January 6, 1994).

Causes of Decline. The factor most strongly associated with the recent decline in the abundance of longfin smelt has been the increase in water diverted by the SWP and the CVP during the winter and spring months when the smelt are spawning (NHI 1992a). A major effect of the SWP on longfin smelt appears to be due to entrainment at Clifton Court Forebay (DWR 1992a). The pumping changes the hydrology of the Delta and increases the exposure of larval, juvenile, and adult longfin smelt to predation and entrainment (NHI 1992b). Salvage data indicate that longfin smelt have been more vulnerable to pumping operations since 1984. This increase in vulnerability may be due to the concentration of longfin smelt populations in the upper Estuary, within the zone of influence of the pumps, as a result of reduced Delta outflow. Also, decreases in outflow fail to disperse the larvae downstream to Suisun Bay nursery areas, away from the effects of Delta pumping (Meng 1993).

The abundance index of longfin smelt is closely correlated with total Delta outflow (DFG 1992b, DWR 1992a, Meng 1993). The decline in 1981, a dry year, (for which Delta smelt remained at relatively high numbers) reflects the dependence of longfin smelt on high outflows (SFEP 1992a).

Correlation analyses for almost all combinations of months between December and August indicate significant positive relationships between average monthly flow into the Delta and longfin smelt abundance from the fall mid-water trawl surveys. It was determined that the most critical outflow period for longfin smelt is December through May. Most larvae begin feeding and complete fin development (which facilitates feeding efficiency and predator avoidance), and mortality is likely to be highest, during the February-May period. Estuarine conditions in December and January, prior to downstream movement of young, are also important to survival. A model of longfin smelt abundance for the December-May period shows a positive relationship between Delta outflow and smelt abundance (see Figure VI-6 in Chapter VI) (Randy Baxter, DFG, pers. comm., October 1994; DFG 1992b).

Reduced outflow during the winter may decrease the amount of spawning area in the lower Delta, and changes in spring outflow could alter the transport time for young smelt to reach downstream bays or affect the availability of rearing habitat. It is unclear, however, whether total outflow or short-term peak flows are biologically important during this period (DWR 1992a). Reduced outflow may also affect longfin smelt abundance through increased predation which occurs when water clarity increases and the young are concentrated in small volumes, and, as mentioned above, through increased losses of fish at the CVP and SWP export facilities, as well as in agriculture diversions. Higher outflows likely benefit the longfin smelt by providing increased larval dispersal and volume of nursery habitat, and possibly increase nutrients that form the base of the food chain (DFG 1992b, BDOC 1993).

Like the Sacramento splittail, the strong outflow-abundance relationship for longfin smelt appears to be breaking down, suggesting that factors besides flow are affecting abundance. It is possible that longfin smelt stocks are so depressed that there are not enough spawners to produce a good year class (Meng 1993).

Other factors which affect longfin smelt populations include entrainment into irrigation diversions and power plant cooling systems, predation from introduced species (e.g., striped bass), competition for zooplankton from introduced planktivorous fish and invertebrates, and droughts and floods. However, most of these factors have been operating prior to the recent decline in longfin smelt abundance (NHI 1992b).

6. Marine Fish

Marine fish species can be divided into those that are seasonally present in the Bay-Delta Estuary and those with at least part of their populations in the Estuary year-round. The seasonal species comprise many of the most abundant fish in the bay. Northern anchovy and Pacific herring are the first and second most abundant, respectively, of the seasonal marine fish in the bay. Other species which are found seasonally in the bay include the starry flounder, English sole, and white croaker. Resident marine species, which often fluctuate in abundance in the bay from year to year, include the native shiner perch, bay goby, and staghorn sculpin, and the introduced yellowfin goby and chameleon goby (SFEP 1992a). The Pacific herring and the starry flounder are addressed here as representative marine species.

a. Pacific Herring. The Pacific herring (*Clupea harengus*) is a native, plankton-feeding marine fish that spawns in estuaries (Moyle 1976). Adults enter San Francisco Bay in the fall and generally spawn from November through March. Most of the spawning occurs in intertidal and shallow habitats of the Tiburon Peninsula and Angel Island, although some spawning occurs on aquatic vegetation near Berkeley and Richmond (SFEP 1992a). Pacific herring use San Francisco Bay as a nursery area for approximately 6 to 8 months before migrating to the ocean (DFG 1994b). Smaller young tend to be widely distributed in shallower habitats in South, Central, and San Pablo bays. As they grow, young Pacific herring are found in deeper waters closer to the Golden Gate and leave the bay between April and August (SFEP 1992a). Pacific herring return to the bay as 2- and 3-year olds (DFG 1994b), where they support a large fishery (BDOC 1993).

Population Trends. YOY Pacific herring abundance is estimated from the Bay Study which began in 1980. YOY herring were abundant in the bay in 1980, declined through the 1983 *El Niño* year, increased to high abundance in 1986, then decreased again through the early 1990's (Figure V-30). YOY abundance was particularly low in 1990 (DFG 1994b).

Information regarding the abundance of adult Pacific herring in the bay comes from the estimated spawning biomass (Figure V-31). The spawning population of Pacific herring has been relatively stable, with the exception of a very low spawning biomass associated with the

Figure V-30

Pacific Herring Young-of-Year Abundance Indices

April-September Mid-Water Trawl (1980-1993)

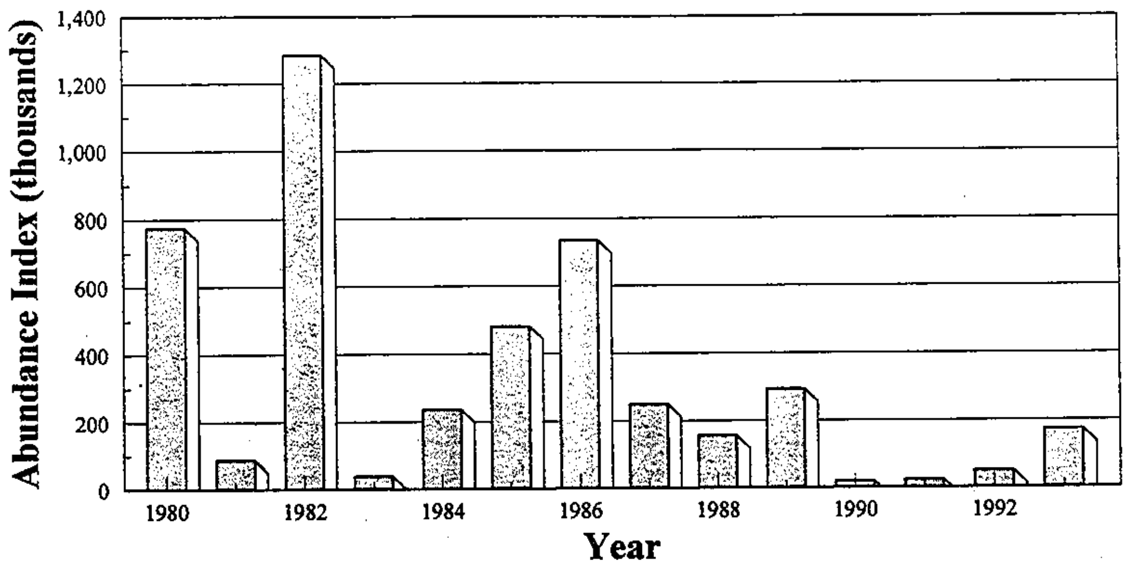
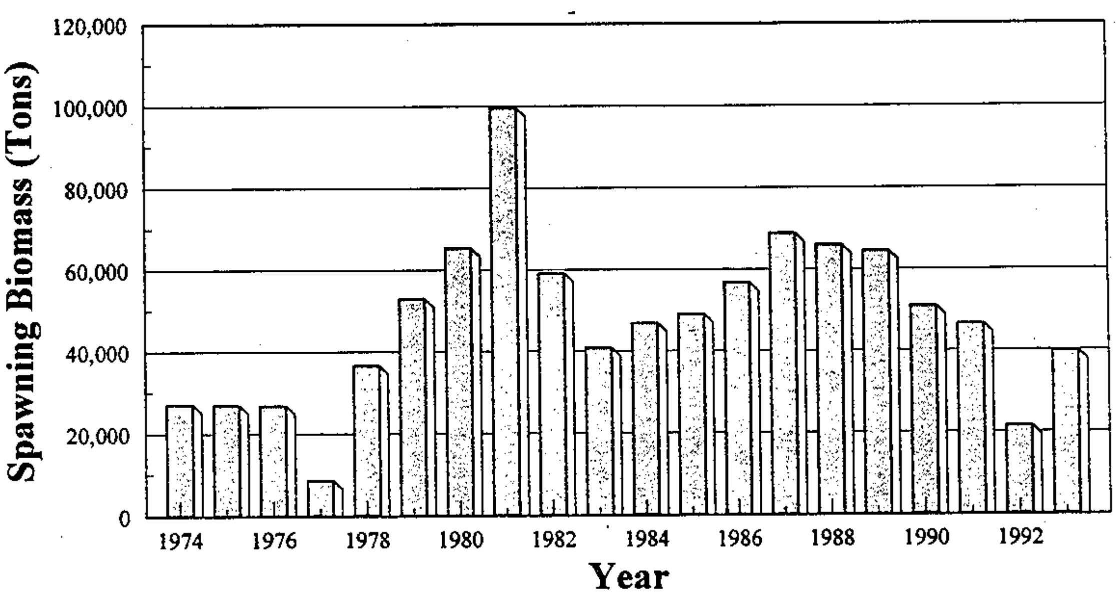


Figure V-31

Pacific Herring Estimated Spawning Biomass

San Francisco Bay (1974-1993)



1977-1978 *El Niño* condition and unusually low abundance in 1992-1993 which reflected poor recruitment from the 1990 and 1991 year classes (DFG 1994b), both critical years.

Causes of Decline. The decline in catch of YOY Pacific herring during 1983 was apparently due, at least in part, to a reduced oceanic herring population in response to reduced productivity during *El Niño* conditions (SFEP 1992a). Overall, the Pacific herring population, which supports a large fishery in San Francisco Bay, has remained relatively stable (BDOC 1993). However, the recent decline in herring biomass in the bay has prompted investigations of possible causes (e.g., increases in salinity due to drought conditions and increases in temperature due to *El Niño* conditions) (IEP 1994c).

b. Starry Flounder. The starry flounder (*Platichthys stellatus*) is a flatfish that feeds on benthic organisms (Moyle 1976). This native fish can be found in the Bay-Delta Estuary throughout the year (SFEP 1992a). Adults inhabit shallow, coastal marine waters, whereas the juveniles appear to be estuarine-dependent and seek out fresh to brackish waters of bays and estuaries as a nursery ground (DFG 1994b). Starry flounder are most abundant and most diverse in sizes in San Pablo Bay, although many young are found in Suisun Bay (SFEP 1992a).

The starry flounder spawn in near-shore areas between November and February. The pelagic eggs and young larvae are found mostly in the upper water column. About two months after hatching, the larvae settle to the bottom (DFG 1992c). Bottom density and tidal currents transport the young into San Francisco Bay (BDOC 1993, Jassby et al. 1994, SFEP 1992a), where they rear for one or more years. As they grow, juveniles move to water of higher salinity within the Estuary. During the late fall and winter, mature starry flounder probably migrate to coast waters to spawn (DFG 1992c).

Population Trends. Because the starry flounder supports a moderately important sport fishery in California (BDOC 1993), the longest historical record of abundance in San Francisco Bay come from charter boat logs. Most of the Estuary's starry flounder catch has occurred in San Pablo and Suisun bays (DFG 1994b). A sharp decline in starry flounder catches, most notably in San Pablo Bay, has occurred since 1983 (SFEP 1992a). Figure V-32 presents the total catch and catch per angler hour data for San Pablo Bay only. In general, catch and catch per hour increased between 1964 and 1971, and decreased to 1964 levels by 1976. In 1976, the total starry flounder catch and catch per hour declined rapidly and, except for a brief period in the mid-1980's, has not recovered to anywhere near previous levels (DFG 1992c).

The Bay Study otter trawl data demonstrate a dramatic decline in YOY and one-year-old starry flounder abundance since sampling began in 1980 (Figure V-33) (DFG 1994b). Such continued low abundances indicate that recruitment to and/or survival of starry flounder in the bay has been very poor for the past five years.

Causes of Decline. Like Delta smelt, longfin smelt, and striped bass, the resident population of starry flounder depends on hydrologic and other environmental conditions in

Figure V-32

Starry Flounder Catch and Catch/Angler-Hour

San Pablo Bay, January to May (1964-1990)

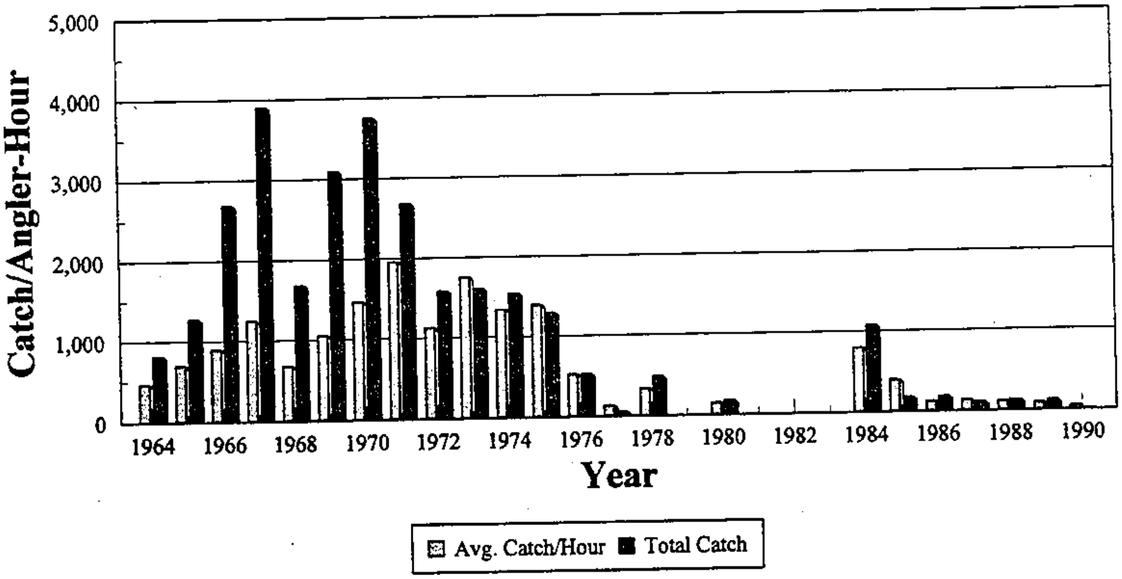
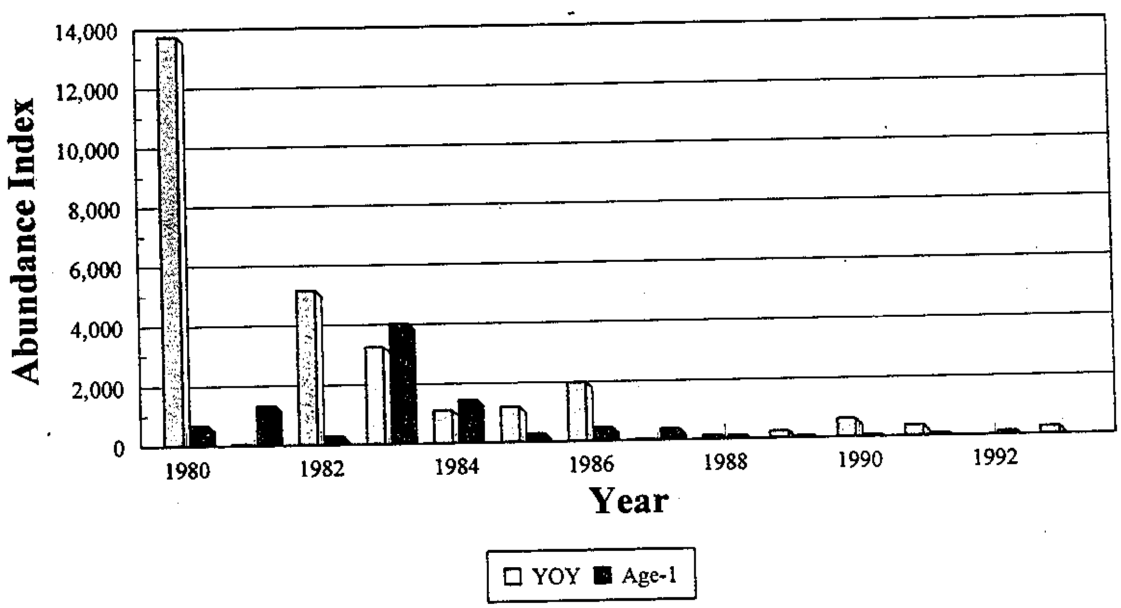


Figure V-33

Starry Flounder Young-of-Year and Age-1

Abundance Indices from Otter Trawl Catches (1980-1991)



San Pablo and Suisun bays (SFEP 1992a). It is probable that reduced Delta outflow during the winter and spring is the principal cause of the long-term decline of starry flounder (BDOC 1993). It is expected that, because bottom currents transport young flounders into the bay, higher net Delta outflows (which strengthens bottom currents) should result in higher abundance of one-year-old fish the following year (Jassby et al. 1994).

The most critical period for starry flounder has been determined to be March through June, when most of the larvae and juvenile immigration occurs. Also, the amount and location of shallow, brackish water nursery habitat for recently settled and small juveniles are most important during this period. The $\log_{10}$ average March-June outflow at Chipps Island and the $\log_{10}$ average 1-year-old starry flounder abundance index the following year have a significant positive relationship (see Figure VI-7 in Chapter VI). Good recruitment of larvae to nearshore areas is possible during both high and low outflow years, but poor recruitment only occurs when outflow is low (DFG 1992c). This indicates that starry flounder abundance in the bay probably also depends on ocean conditions, as well as other lesser known factors.

Although young starry flounder can be found upstream of Suisun Bay, especially in years of low flow, their overall distribution is such that diversion plays a minor role, if any, in their variability (Jassby et al. 1994). The decline in starry flounder abundance in Suisun Bay principally reflects a reduced production of young. Although the even sharper decline in the abundance of San Pablo Bay flounders is not explained, the concentrations of organic contaminants (e.g., PCBs) in adult starry flounder from San Pablo Bay have been shown to be sufficient to reduce reproductive success. Also, the decline of the San Pablo Bay starry flounder population coincides with increased presence of English sole, another bottom-foraging flatfish that spawns outside the Golden Gate and immigrates into the bay with bottom currents. Although the starry flounder is present in the Estuary year-round and the English sole is found seasonally, biotic interactions between the two species may be occurring (SFEP 1992a).

7. Anadromous Fish

Anadromous fish are those which migrate from the ocean to fresh water to spawn. Anadromous fish that spawn in the Sacramento and San Joaquin river basins use the Bay-Delta Estuary as a route of passage to the spawning grounds and, in some cases, as a nursery area. Native anadromous species that may be found in the Estuary include chinook salmon, steelhead trout, white sturgeon, and green sturgeon. The anadromous striped bass and American shad are introduced species in the Estuary (SFEP 1992a).

a. **Chinook Salmon**. The chinook salmon (*Onchorhynchus tshawytscha*), also called king salmon, is the largest and has the broadest geographic range of the five Pacific salmon species. In spite of its wide distribution, the chinook salmon is the least abundant of Pacific salmon species, yet it is an important recreational and commercial species throughout most of its range. In San Francisco Bay, the chinook salmon population is open only to sport fishing. Populations of this native, anadromous species, which is distinguished by its highly variable life history and multiple stocks, are maintained to a large extent by hatchery production (DWR 1993, SFEP 1992a).

Chinook salmon migrate to the ocean early in their life, mature in the ocean, and return inland as adults to spawn in freshwater streams (SFEP 1992a). Acceptable water temperatures for the upstream migration of adults range from 57°F to 67°F. Spawning generally occurs in swift, relatively shallow riffles or along the edges of fast runs where there is an abundance of loose gravel. The females dig spawning redds in the gravel into which their eggs are deposited and buried after fertilization by the male. The adults die a few days after spawning (DFG 1993).

Spawning requires well-oxygenated cool water that percolates through the gravel and supplies oxygen to developing embryos. The preferred temperature for chinook salmon spawning is generally 52°F, with lower and upper threshold temperatures of 42°F and 56°F. Temperatures above this range result in reduced viability of eggs or heavy mortality of developing juveniles. Total egg mortality normally occurs at 62°F. The eggs usually hatch in 40-60 days, depending on water temperature within the appropriate temperature range. The young sac-fry remain in the gravel for an additional 4-6 weeks until the yolk sac is absorbed. Thus, at 50°F, the total time from spawning to emergence is approximately 79 days. After emergence, chinook salmon fry feed in low velocity slack water and back eddies. They move to higher velocity areas as they grow larger and, eventually, migrate to the ocean as smolts. The length of rearing and migration timing varies among the various chinook salmon runs. Young salmon remain in the ocean until their third or fourth year, at which time they return to their home stream to spawn. Two- and 5-year-old fish also participate in the spawning run in small numbers (DFG 1993).

The Central Valley supports the largest population of chinook salmon in the State (SFEP 1992a). The Bay-Delta Estuary serves as a migratory corridor for upmigrating adults and outmigrating smolts, and serves as rearing habitat for salmon fry. Four distinct races of chinook salmon, distinguished by their timing of upstream migration and spawning season, enter the Estuary. Named for the season during which the adults enter fresh water, the four runs of chinook salmon are: fall-run, late-fall-run, winter-run, and spring-run.

All four races of chinook salmon spawn in the upper Sacramento River. Fall-run chinook salmon usually spawn within a few weeks of their arrival in the fall. Late-fall-run chinook salmon spawn in the winter. Spring-run chinook salmon spend the summer in deep, cool pools and spawn in early fall. Winter-run fish enter the river in the winter and spawn early the following summer. The San Joaquin River system supports fall-run, and possibly a small population of late-fall-run, chinook salmon. The fall runs of the Sacramento and San Joaquin river systems may be genetically distinct and may constitute separate races (DFG 1993). All of the runs are supplemented to some degree by hatchery production; however, the fall- and late-fall-run chinook salmon populations are principally augmented by hatchery production (DFG 1993, SFEP 1992a).

Adult fall-run chinook salmon migrate into the river systems from July through December, and spawn from early October through late December. Peak spawning occurs in October and November, although timing of the runs varies from stream to stream. Egg incubation occurs from October through March, and juvenile rearing and smolt emigration occur from January through June. Although the majority of young salmon migrate to the ocean during the first few months following emergence, a small number may remain in fresh water and migrate as yearlings (DFG 1993).

Adult late-fall-run chinook salmon migrate into the Sacramento and San Joaquin rivers from mid-October through mid-April, overlapping the mid-October through December fall-run salmon spawning migration. Late-fall-run salmon spawn from January through mid-April. Incubation occurs from January through June, and rearing and emigrations of fry and smolts occur from April through mid-October. Significant emigration of naturally-produced juveniles occurs through November, into December, and possibly January. Emigration of hatchery-produced juveniles occurs well into February (DFG 1993).

Adult winter-run salmon enter the Estuary from about November through May, and pass the Red Bluff Diversion Dam on the upper Sacramento River from December through early August. Historically, winter-run chinook salmon spawned from April through August in the upper reaches of Sacramento River tributaries, including the McCloud, Pit, and Little Sacramento rivers, and Battle Creek. Now, winter-run salmon spawn in the main stem of the Sacramento River below Keswick Dam from mid-April through August, when water storage project releases provide cool water temperatures. Egg and larval incubation occurs from mid-April through mid-October. Emigration of fry and smolts extends from July through March at Red Bluff Diversion Dam, and from September through June in the Delta (DFG 1993), but peak emigration extends from late-January through April (DFG 1994a).

Adult spring-run chinook salmon enter the Sacramento River from late-March through September. Many early arriving adults hold in cool water habitats through the summer, then spawn in the fall. Spawning occurs from mid-August through early-October, with the peak in September, overlapping with the fall-run in the main stem Sacramento River in early-October. Incubation occurs from mid-August through mid-March. Rearing and emigration of fry and smolts begin in late-November and continue through April. A significant migration of yearlings from the upper tributaries also occurs in September through December. It is likely that some individual spring-run salmon have interbred with fall-run salmon in the main stem Sacramento River and the Feather River. A genetically uncontaminated strain of spring-run chinook salmon may still exist in Deer and Mill creeks, where they are geographically separated from the fall run. Spring-run salmon are also present in Antelope, Battle, Cottonwood, Big Chico, and Butte creeks (DFG 1993).

Chinook salmon originally spawned throughout the tributaries or upper reaches of the Sacramento and San Joaquin river basins. However, dams have reduced the amount of historic river and spawning habitat available to chinook salmon by 95 percent (from about 6,000 miles to less than 300 miles). As a consequence, in both the Sacramento and San Joaquin river basins, some runs of chinook salmon have been almost totally eliminated.

About half of the potential spawning habitat in the Sacramento River basin was blocked by construction of Shasta Dam in 1942, which prevented access of enormous runs of salmon to the upper Sacramento, Pit, and McCloud rivers. Unfortunately, only sparse or incomplete population estimates are available for years prior to 1953 (DFG 1993). The construction of Red Bluff Diversion Dam in 1966 later reduced access to spawning areas below Shasta Dam. Completion of Folsom and Nimbus dams in 1955 blocked access to the historical spawning and rearing habitat on the American River. By 1965, Oroville Dam and other facilities prevented most salmon, including the wild spring-run, from reaching historic spawning grounds on the Feather River. A population of spring-run chinook salmon in the San Joaquin River was lost when Friant Dam, completed in 1949, dried up sections of the river. Friant Dam blocked access and totally eliminated salmon from the main stem and upper tributaries. Dams on the Merced, Tuolumne, and Stanislaus rivers, the major downstream tributaries to the San Joaquin River, have reduced access to chinook salmon habitat. In addition, numerous other projects have been constructed that directly or indirectly affected salmon habitat (DFG 1993, SFEP 1992a).

Four hatcheries (Mokelumne River, Nimbus, Feather River, and Coleman) were constructed in the Sacramento River basin to mitigate for habitat loss as a result of water project construction (DFG 1993). Since the early 1970's, juvenile chinook salmon produced at these hatcheries have augmented natural salmon populations (BDOC 1993). A small hatchery on the Merced River is the only mitigation for upstream salmon habitat losses in the San Joaquin River basin (DFG 1993, SFEP 1992a).

Population Trends. Historical chinook salmon abundance in the Bay-Delta Estuary prompted massive fishing efforts and the opening of the world's first salmon cannery in 1864 (SFEP 1992a). Based on commercial harvest data, it is estimated that, prior to 1915, peak chinook salmon runs in the Sacramento River system may have been as large as 800,000 to 1 million fish, with an average run size of about 600,000 fish (DFG 1993).

Chinook salmon production in the San Joaquin-River system historically approached 300,000 adults and probably averaged about 150,000 fish. Large runs of salmon in the San Joaquin River during the 1940's were predominantly spring-run fish until this run was extirpated after the construction of Friant Dam. The San Joaquin River system now supports an important population of fall-run chinook salmon which is only a remnant of its former size (DFG 1993).

Since 1953, annual estimates of spawning chinook salmon in the major river systems of the Estuary have been made. These are estimates of spawning "escapement" since they describe the numbers of chinook salmon, from both natural and hatchery production, that have escaped the ocean fisheries and returned inland to spawn (DFG 1994b). Although chinook salmon escapement in the San Joaquin River system has been monitored since 1939, these data are sparse or incomplete prior to 1953. Since 1967, following completion of the Red Bluff Diversion Dam in 1966, accurate estimates of all salmon runs to the upper Sacramento River have been possible (DFG 1993).

Since the regular counts of chinook salmon abundance began in 1953, the spawning runs from all river systems have fluctuated greatly. Total runs decreased from over 600,000 in 1953 to 120,000 in 1957, then up to almost 500,000 by 1960 (DFG 1994b). In the last 20 years, the total runs have averaged about 250,000 to 300,000 fish (BDOC 1993). From 1967 to 1991, total escapement averaged 247,100 natural spawners and 28,500 hatchery spawners (DFG 1994b).

Most estimates of chinook salmon abundance indicate that most runs have been severely reduced compared to the 1967-1991 average (DFG 1994b). Wild stocks of chinook salmon have suffered very large declines in the Central Valley (SFEP 1992a). The stream systems that are supported by effective hatchery programs, such as the Feather and American rivers, have maintained adequate populations. Fall-run salmon are presently the most abundant of the four races. Approximately 80 percent of the Central Valley chinook salmon spawners are fall-run fish. About 90 percent of the Central Valley chinook salmon are produced in the Sacramento River basin. The chinook salmon runs of greatest concern are the winter-run, spring-run, and San Joaquin River basin fall-run (DFG 1994b).

Fall-Run. The fall-run comprised an average of 83 percent of all chinook salmon spawning stocks in the Central Valley from 1986-1990. The fall-run is the largest run of chinook salmon in the Sacramento River with an average spawning population of 108,000 fish since 1980. This exceeds the combined total of the other three runs and is the mainstay for the ocean commercial and recreational troll fishery (USBR 1994). An estimated 107,300 adult fall-run chinook salmon returned to the Sacramento River basin in 1992, and an estimated 147,500 returned in 1993. These recent estimates are 53 percent and 73 percent, respectively, of the average escapement of 201,100 from 1967-1991. In comparison, the 1985 and 1986 spawning escapements for the Sacramento River basin, including the Feather, Yuba, and American rivers, were 295,200 and 274,000 adults, respectively (DFG 1994b). Figure V-34 shows the annual estimated run sizes for fall-run chinook salmon, only for the upper Sacramento River above Red Bluff Diversion Dam.

The fall-run of chinook salmon in the San Joaquin River system spawn in the tributary streams; no spawning occurs on the main stem of the San Joaquin River. The fall-run populations in the Merced, Tuolumne, and Stanislaus river tributaries are now at dangerously low levels (Figure V-35). Since annual population surveys began in 1953, fall-run chinook salmon escapement in the San Joaquin River basin has fluctuated widely. In 1985, the escapement was estimated to be 76,100 (BDOC 1993). The 1991 estimate of 620 fall-run chinook salmon was the lowest escapement recently; an escapement of 320 in 1963 was the lowest ever observed in the San Joaquin River basin (USFWS 1992). The 1992 and 1993 escapements were estimated to be about 2,000 and 3,200 fish, respectively. These recent returns are much lower than the average of 20,700 adults for the 1967-1991 period (DFG 1994b).

Figure V-35 also shows estimated chinook salmon run sizes for the San Joaquin River for several years prior to the construction of Friant Dam in 1949. These fish represent the spring-run salmon that were entirely eliminated when the dam dried up parts of the river.

Figure V-34

Sacramento River Basin

Annual Estimated Chinook Salmon Run Size Above Red Bluff Diversion Dam (1967-1993)

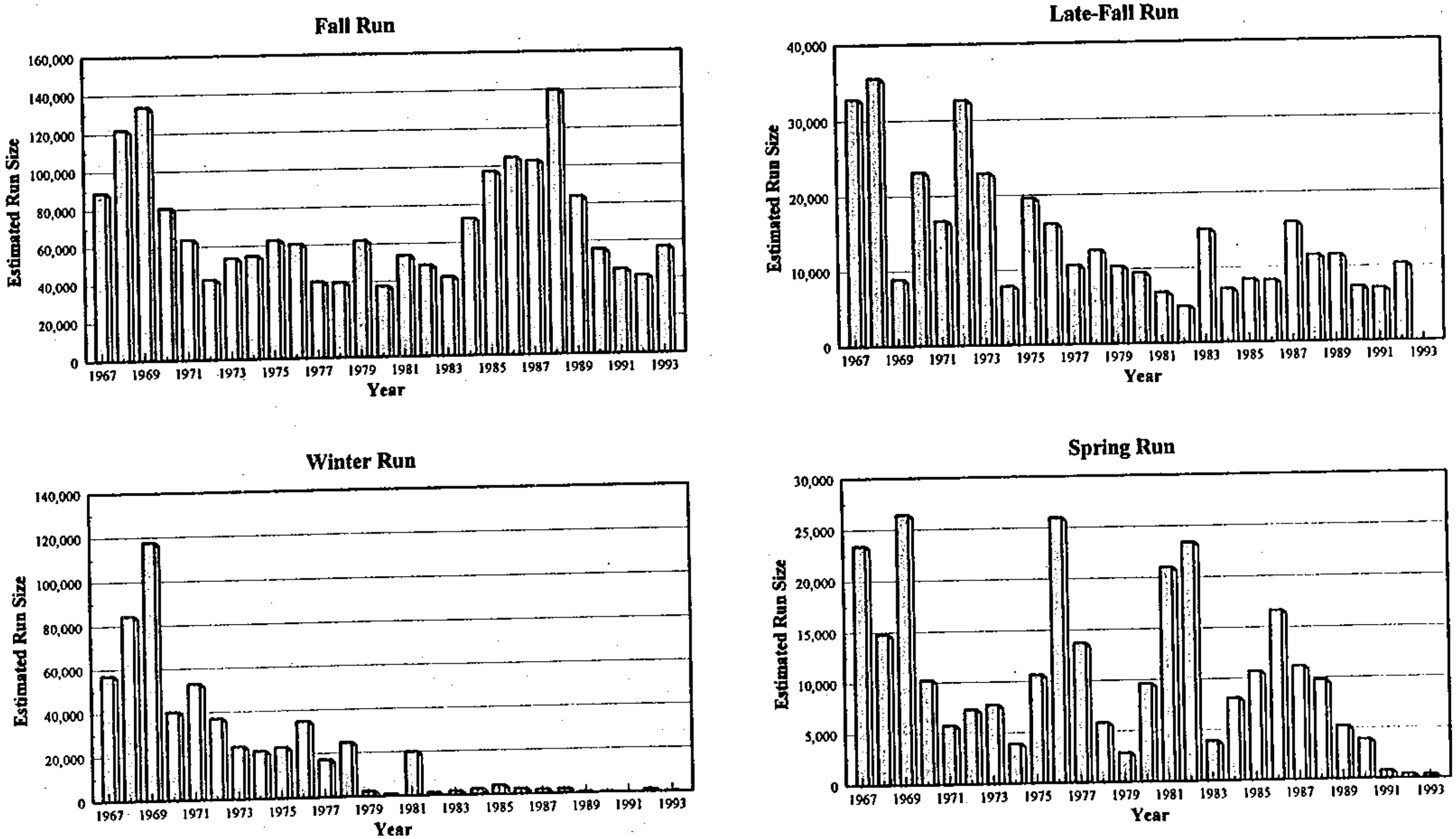
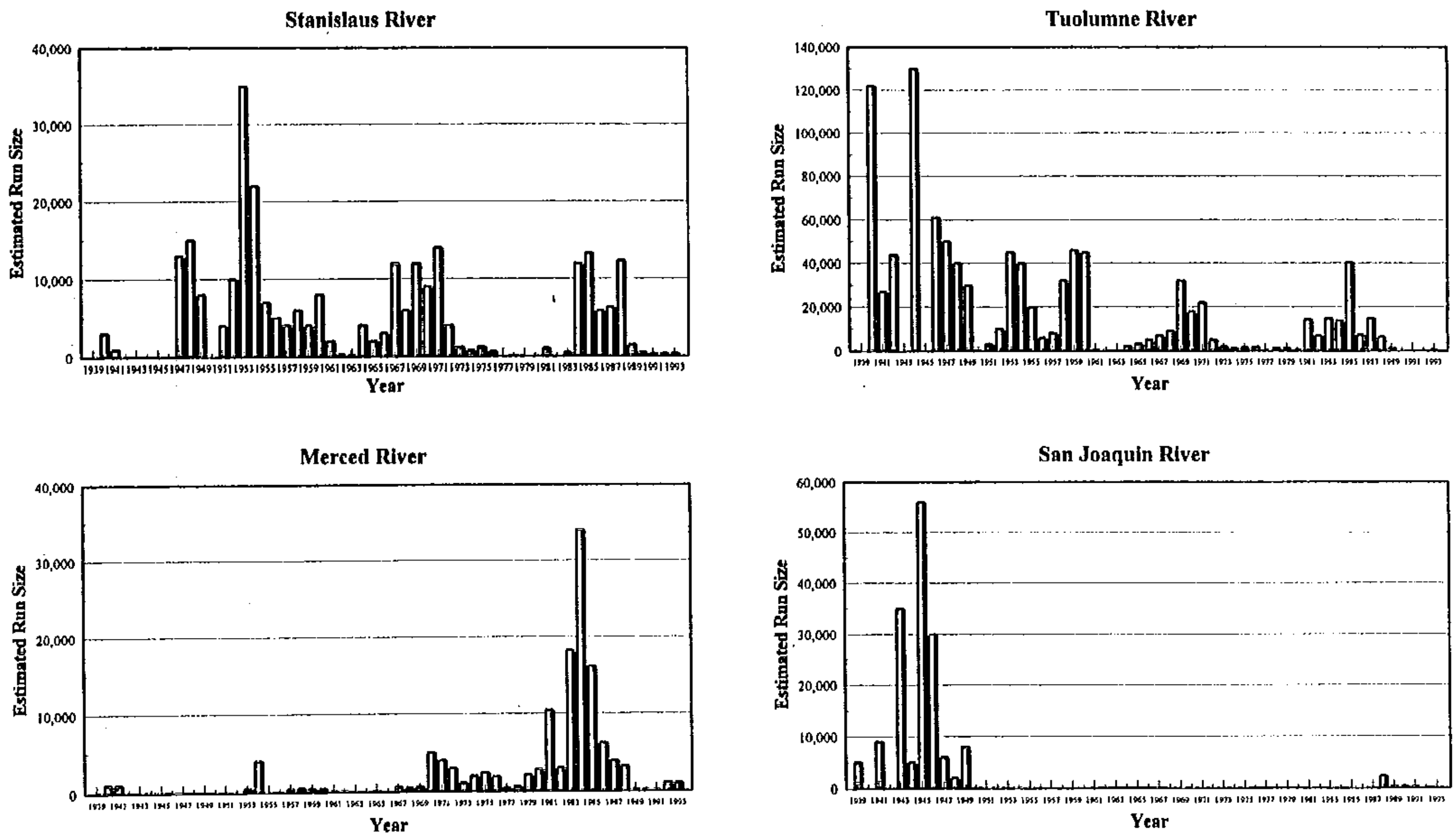


Figure V-35

San Joaquin River Basin

Annual Estimated Chinook Salmon Run Size (1939-1993)



The small populations that appear from 1988 to 1991 represent fall-run chinook salmon that strayed into Mud and Salt sloughs, tributary to the San Joaquin River, during the drought when flows in the lower tributaries were low. In the fall of 1992, a temporary fish barrier was installed across the San Joaquin River just upstream of the confluence with the Merced River and below the confluence with the sloughs to prevent salmon from straying into westside canals (DFG 1993).

Late-Fall-Run. Recent escapements of late-fall-run chinook salmon in the Sacramento River basin are below the average 14,100 late-fall escapements from 1967-1991. In 1991, an estimated 8,600 late-fall spawners returned to the upper Sacramento River and, in 1992, the estimate was 10,400 fish (Figure V-34). Operation of the Red Bluff Diversion Dam, in response to the NMFS biological opinion on the endangered winter-run chinook salmon, precludes estimating late-fall run chinook salmon escapement after 1992, until a new estimation method is developed (DFG 1994b).

Although a small population of late-fall-run chinook salmon spawns in the San Joaquin River basin, there have not been formal inventories of this stock (DFG 1994b).

Winter-Run. The Sacramento River has the only remaining winter-run chinook salmon population in the world. When completion of Shasta and Keswick dams in the early 1940's blocked access to the upper Sacramento tributary streams, the population began declining but recovered dramatically during the 1940's and 1950's, apparently by taking advantage of cool water released from the reservoirs in the summer (DFG 1993). Since estimates of winter-run salmon escapement began in 1967, the numbers of adults have steadily declined from about 118,000 fish in 1969, to an estimated 1,200 fish in 1992 and 300 fish in 1993 (Figure V-34). The average escapement from 1967 to 1991 was 23,100 winter-run chinook (DFG 1994b). The winter-run salmon returning in 1994 are the progeny of the 1991 run which was the lowest on record (191) (DFG 1994a).

The NMFS believes the sizes of the winter-run are dangerously low because it has been estimated that a run size of 400 to 1,000 fish is necessary to maintain genetic diversity in the winter-run salmon population (DFG 1994a). The State listed the winter-run chinook salmon as endangered in 1989; the NMFS listed the winter-run chinook salmon as threatened in November 1990 and endangered in 1994 (NMFS 1993). Although conservation measures have been implemented since 1987, specifically to improve habitat conditions for the winter-run, the population has continued to decline precipitously (USBR 1994).

Spring-Run. Spring-run chinook salmon were, perhaps, historically the most abundant run in the Central Valley (DFG 1993). Run sizes have varied greatly since the early 1970's (Figure V-34), averaging around 13,000 fish annually from 1967-1991. In 1992, fewer than 1,200 spring-run salmon used the Sacramento River basin. The escapement in 1993 was estimated to be 1,400 fish (DFG 1994b). Present wild spring-run salmon populations are less than 0.5 percent of the historic runs. Because of their continuing decline, the spring-run chinook salmon may be considered as a candidate for listing as an endangered species (NHI 1992a).

Causes of Declines. The loss of 95 percent of historical habitat for chinook salmon due to dams and habitat degradation has been a significant cause in the decline of salmon populations in the Central Valley. Salmon habitat loss and degradation began with hydraulic mining in the mid-1800's. By 1929, declines in the abundant spring- and fall-run chinook salmon populations in the upper Sacramento River were noted. These declines were thought to have resulted from overharvest, blockage by irrigation dams, and habitat degradation through activities such as reclamation, flood control, and logging. This period of severe loss and degradation of salmon habitat culminated with the completion of the major water project developments in the 1970's (DFG 1993).

Much of the area in which fall-run chinook salmon historically spawned was downstream from the major dam sites; therefore, this race was not as severely affected by early water project developments as were spring- and winter-run salmon, which historically spawned at higher elevations. The construction of dams that barred migration of adult spawners to upstream areas also created higher water temperatures and altered stream flows. This situation resulted in the elimination of spring-run chinook salmon in the San Joaquin River system and most other Central Valley tributaries (DFG 1993). The runs currently of greatest concern are winter-run, spring-run, and the San Joaquin River fall-run, due to low escapements and future low projections, based on population trends (DFG 1994b).

There are a number of factors in the upstream areas that affect the number of naturally-produced chinook salmon each year. These include spawning habitat access, availability and condition of habitat, water quality conditions including temperature and pollution, flow fluctuations, water diversion entrainment, and high predation rates. Survival through the Delta is critical, especially for the naturally-produced salmon and those hatchery fish released in the upstream areas. Factors which influence survival in the Delta include temperature and entrainment. The relative importance of such factors to chinook salmon survival and production varies between the Sacramento and San Joaquin river basins, and among the various salmon runs.

San Joaquin River Basin. The San Joaquin River system supports an important population of fall-run chinook salmon, which is now only a remnant of its former size. Spawning populations and production vary widely from year to year, depending upon the timing and magnitude of flows available for upstream migration, spawning, rearing, and emigration. San Joaquin River basin salmon populations also can be severely affected by pumping operations in the Delta, which may capture all of the San Joaquin River outflow (DFG 1993). Cumulative effects of prolonged drought, poor water quality, habitat deterioration, water diversion, and ocean harvest have caused greatly reduced population levels of fall-run populations in the Merced, Tuolumne, and Stanislaus river tributaries. However, low population levels occurred historically and the population rebounded in the 1980's, in association with high flows (DWR 1993).

Streamflow releases below Friant Dam are insufficient to support salmon passage, spawning, or rearing in the San Joaquin River. The dam also damaged runs to the San Joaquin River tributaries by significantly reducing total basin outflow. The reduction in fall attraction flows and spring outflows on the main stem San Joaquin River significantly reduced adult returns, and reduced production and survival of salmon throughout the system. Since Friant Dam went into operation, low spring outflows from the basin, in most years, have been a major factor contributing to low salmon production (DFG 1993).

San Joaquin River basin emigrating smolt losses can be attributed to high water temperatures, low flows, high predation losses, unscreened water diversions, and SWP and CVP diversions. Elevated water temperatures during the spring emigration period (April-June) probably reduce smolt survival in the main stem of the river and tributaries. Typical flow and water quality conditions in the Delta are detrimental to the survival of San Joaquin salmon smolts due to low inflow from the San Joaquin River and high exports by Delta water diversions. Survival of smolts migrating down the main stem San Joaquin River is higher than the survival of smolts migrating down upper Old River toward the export pumps (DFG 1993).

Chinook salmon fry and smolt losses occur at the CVP and SWP export pumps year-round, but peak levels generally occur in late winter and spring, when the most abundant salmon race, the fall-run, passes through the Delta (DWR 1992a). The proportion of outmigrants from the San Joaquin River system that show up at the CVP and SWP intakes is greater (20-70 percent) than the proportion of Sacramento River system outmigrants that show up at the intakes (2 percent) (BDOC 1993). Peak chinook salmon losses due to SWP pumping from 1980 to 1987 occurred in April-June. The majority of SWP salmon losses have been attributed to predation by striped bass in Clifton Court Forebay. Other factors associated with the water projects, such as screen efficiencies and salvage operations, also influence salmon survival (DWR 1992a).

The upstream migration of adult salmon into the San Joaquin River basin is probably delayed due to the lack of attraction flow, elevated water temperatures, and low dissolved oxygen levels, which commonly occur in the San Joaquin River in the fall. Increases in agricultural return flows in recent years, such as in Mud and Salt sloughs, have attracted significant numbers of adults salmon into sloughs and irrigation canals, where there is no suitable spawning habitat available. In the fall of 1991, an estimated 35 percent of the San Joaquin River basin salmon strayed into westside canals. Installation of a temporary fish barrier in the fall, which began in 1992, has prevented salmon straying into the westside irrigation canals and sloughs (DFG 1993). Beginning in 1995, from October through December, a fish barrier will be installed annually for 15 years on the San Joaquin River near its confluence with the Merced River to prevent the salmon from migrating upstream and into the irrigation canals and sloughs (Steve Ford, DWR, pers. comm., April 1995).

Water temperature and dissolved oxygen can vary considerably according to stream flow, water depth, and water quality. Adults migrating up the San Joaquin River in September through December must deal with warm water temperatures, which can range in the mid-70's°F in September and October, and extremely low dissolved oxygen levels. Low dissolved oxygen levels are a result of reduced flow, warm water temperatures, dredging activities in the Stockton Ship Channel and turning basin, and effluent discharges. A temporary barrier is installed each fall by the DWR at the head of Old River to improve water quality and help adult salmon migration in the lower reaches of the San Joaquin River (DFG 1993, DWR 1993).

Minimum flows in the San Joaquin River at Vernalis during the spring outmigration would improve salmon smolt survival into and through the Delta (BDOC 1993, DFG 1993). When spring outflow in the San Joaquin River at Vernalis is high, the total adult salmon escapement in the San Joaquin River basin 2.5 years later is increased (DFG 1993). Increased flows in the fall would also benefit upmigrating adults by providing attraction flow, lower water temperatures, and higher dissolved oxygen levels. Figure VI-11 in Chapter VI, shows the relationship between water temperature at Jersey Point, San Joaquin River flow, and exports.

Sacramento River Basin. Conditions in the Estuary, impacting Sacramento River basin chinook salmon, affect primarily the emigrating smolts rather than the immigrating adults. Current understanding of smolt survival in the Sacramento River and through the Delta is based primarily on studies using hatchery-reared fall-run chinook salmon (IEP 1994a). Based on the habitat requirements of fall-run chinook salmon and the USFWS salmon smolt survival model, water temperatures and diversions, rather than flow, are the principal factors affecting salmon smolt survival in the Delta (BDOC 1993). Factors found to affect smolt survival during the fall-run outmigration include water temperature, SWP and CVP export rates, percent of flow diverted into the central Delta via the Delta Cross Channel gates and Georgiana Slough (which have the combined capacity to divert about 70 percent of the flow in the main stem Sacramento River), and size of the fish (DWR 1992a, IEP 1994a). These factors, possibly excluding water temperature, likely affect the survival of the other three runs of chinook salmon as well.

During their passage through the Delta, fall-run smolts are particularly liable to suffer increased mortality if they enter the central Delta (IEP 1994a). Salmon smolts may follow Sacramento River water that is diverted into the lower San Joaquin River via the Delta Cross Channel, Georgiana Slough, and Three Mile Slough. Experiments have shown that young hatchery-reared salmon released in the Sacramento River below Walnut Grove have a survival rate twice that of smolts released upstream of Walnut Grove and diverted through the Delta Cross Channel or Georgiana Slough. Since 2 percent or less of Sacramento River salmon show up at the SWP and CVP fish screens in the southern Delta, most of the mortality is assumed to occur in central Delta channels (BDOC 1993).

Passage through the central Delta is detrimental to smolts because of warmer temperatures, increased predation rates, longer migration routes, areas of reverse flow in river channels, and entrainment by agricultural and export pumps. At the CVP and SWP export facilities in the Delta, causes of mortality include predation and entrainment. Smolts released into the Sacramento River downstream of both the Delta Cross Channel and Georgiana Slough can be entrained at the pumps even when the Delta Cross Channel gates are open and QWEST is positive. Closing the Delta Cross Channel gates can result in increased negative (reverse) flows in the central and western Delta, particularly when export rates are high (IEP 1994a).

Three years of sampling chinook salmon at the Golden Gate indicated that salmon smolts migrate through the lower Estuary faster than net flow would transport them. In those three years, smolt survival rate in that area was not related to the magnitude of Delta outflow (BDOC 1993). However, Sacramento River system fall-run smolt survival through the Delta was found to be significantly correlated with Delta outflow, although the increased survival was probably due to cooler water temperatures. Smolt survival apparently is not related to reverse flows, which tend to occur more frequently in summer and fall, after the period of peak outmigration (DWR 1992a). However, smolt survival is significantly affected by operation of the Delta Cross Channel gates. Figures VI-9 and VI-10 in Chapter VI illustrate that, for a given water temperature and smolt survival index, when the Delta Cross Channel gates are closed, exports can be higher than when the gates are open.

The release of most hatchery fish in the lower Estuary, rather than in the river, has substantially increased smolt survival (DFG 1994b). However, when Feather River hatchery and Nimbus hatchery smolts are released many miles downstream in or near the Delta, straying of these fish, when they return as adults to spawn, is substantial, resulting in fewer fish returning to the hatchery. At Coleman National Fish Hatchery, where smolts are released near the hatchery, straying is much less (DFG 1993). Increased survival, from the releasing of fish in the Delta, has enabled a relatively intense ocean fishery to continue, even with reduced natural salmon populations. However, the success of the hatchery program increases the risk of over-harvesting natural stocks (DFG 1994b). Although ocean harvests clearly reduce spawning escapement substantially, it is not the principal factor limiting salmon production. Evidence of this is that reduced San Joaquin River stocks can rebound after a wet spring, which would not be possible if overharvesting were a significant factor in salmon abundance (BDOC 1993).

Some upstream migrating adult salmon use the lower San Joaquin River, Mokelumne River, Delta Cross Channel, and Georgiana Slough on their way to the Sacramento River system spawning grounds. It is believed that this is not detrimental to the salmon if the channels are not blocked (BDOC 1993).

Numerous factors affect chinook salmon survival in the upstream areas of the Sacramento River. These include: fish passage delay and fish losses associated with Red Bluff Diversion Dam; losses associated with inadequate fish screens at Anderson-Cottonwood Irrigation District's and Glenn-Colusa Irrigation District's diversions; hundreds of unscreened diversions; bank protection and flood control projects which reduce useable instream habitat; excessive flow fluctuations and elevated water temperatures below Keswick Dam; industrial, municipal, agricultural, and mining discharges of chemical waste; and poor quality, warm agricultural drainage water from the Colusa Basin Drain. Salmon runs in the lower American River have declined significantly due to the combined effects of project-induced low flows, severe flow fluctuations which expose and dry redds and strand juveniles, and high water temperatures. Inadequate flows and elevated water temperatures are also problems for chinook salmon in the Feather River (DFG 1993).

Many of the factors that are known to affect juvenile fall-run chinook salmon survival in the spring also impact the other runs of salmon, at slightly different times of the year. Although upstream effects are responsible for the significant decline in the spring-run chinook salmon, conditions in the Estuary may contribute to their continuing decline. A key factor in the recovery of spring-run salmon is adequate Delta outflows during the smolt outmigration period to reduce their vulnerability to entrainment and Delta predators (NHI 1992a). Winter-run salmon are believed to be less vulnerable than fall-run fish to predation and temperature factors due to their greater size and the relatively cool water temperatures during their outmigration. Despite this, the survival of winter-run smolts that are diverted into the central Delta, is similarly low. Therefore, closure of the Delta Cross Channel gates, as well as measures to prevent smolts from entering Georgiana Slough, must be considered to prevent further decline of winter-run chinook salmon (IEP 1994a).

b. Steelhead Trout. The native steelhead trout (*Oncorhynchus mykiss*) is an anadromous strain of rainbow trout that is generally distributed along the Pacific Coast. Within California's Central Valley, a viable population of naturally-produced steelhead is found only in the Sacramento River (above Red Bluff Diversion Dam) and its tributaries, primarily Mill and Deer creeks (DFG 1993; Dennis McEwan, DFG, pers. comm., September, 1994). Steelhead trout comprise an important recreational fishery within the Sacramento River system (DWR 1993). No significant steelhead populations now occur in the San Joaquin River system (DFG 1993).

Steelhead trout have a life history similar to chinook salmon, although the timing and duration of different stages varies. In the Sacramento River, upstream migration occurs from early August through November, with the peak in mid-September. Spawning in the Sacramento River and its tributaries usually occurs from January through March. Unlike chinook salmon, many steelhead do not die after spawning, but return to the ocean. Individuals that survive return to the ocean between April and June, where they remain for 1 or 2 years. Egg incubation takes place from January through April. Unlike chinook salmon which typically outmigrate soon after emerging from the gravel, steelhead in the Sacramento River generally emigrate as 1-year olds, at a larger size than salmon. Average monthly SWP fish salvage data, for the years 1980-1991, indicate most steelhead are salvaged in the late winter and early spring, with the peak occurring in March and April (Steve Ford, DWR, pers. comm., April 1995). In addition, all freshwater life stages of steelhead, except rearing, require lower temperatures than chinook salmon. The preferred temperatures for steelhead trout in the Sacramento River are between 50°F and 58°F (DFG 1993).

Population Trends. With natural spawning greatly reduced in the Sacramento and San Joaquin river systems, steelhead trout populations are primarily maintained by hatcheries (DFG 1993). Approximately 15 percent of the annual steelhead runs in the Sacramento River are the result of stocked fish released as smolts or fingerlings. Steelhead escapement in the lower American River is supported entirely by hatchery production (DWR 1993).

Both natural and hatchery-maintained steelhead stocks in the Central Valley are declining (DFG 1993). Figure V-36 illustrates the combined estimates of runs of wild and hatchery steelhead above Red Bluff Diversion Dam from 1967-1994. Figure V-37 shows the number of hatchery returns for the Coleman, Feather River, and Nimbus hatcheries during the same period of record.

Causes of Decline. Because spawning usually occurs from January through March, the temperature-sensitive egg and sac-fry life stages of steelhead are not present in the main stem Sacramento River and tributaries during the warmest period of the year (USBR 1994). Summer rearing temperatures, however, can and do preclude their survival in some areas. Natural production is limited because of the lack of sufficient cold water habitat during spring and summer months (DWR 1993).

Declines in natural and hatchery-maintained steelhead stocks in the Central Valley are due mostly to water development, inadequate instream flows, rapid flow fluctuations, high summer water temperatures in streams immediately below reservoirs, diversion dams which block access, and entrainment of juveniles into unscreened or poorly-screened diversions. The operations of the SWP and the CVP, particularly the Delta pumping plants, have had a detrimental effect on steelhead smolts emigrating through the Delta to the ocean. Reverse flows, entrainment of fish into the pumping facilities, and increased predation at water facilities are major problems (DFG 1993). Although these are the same factors that affect chinook salmon, it is possible that steelhead smolts are less susceptible to reverse flows, entrainment, and predation since they are larger than salmon smolts during their migration through the Delta (DWR 1992a).

c. **Sturgeon.** Two species of sturgeon are found in the Bay-Delta Estuary: the green sturgeon (*Acipenser medirostris*) and the white sturgeon (*Acipenser transmontanus*). These native fish are long-lived and late-maturing, making them extremely vulnerable to overfishing. Historical accounts indicate that a commercial fishery greatly reduced the estuarine white sturgeon population in the late-1800's. All sturgeon fishing was prohibited in 1917. The sturgeon sport fishery was reopened in 1954 (BDOC 1993).

The Bay-Delta Estuary contains the southernmost of the three known spawning populations of the green sturgeon (NHI 1992a). The green sturgeon is much less common in the Estuary than the white sturgeon and comprises a minor component of the sturgeon sport fishery. They make extensive ocean migrations and enter estuaries on the Pacific coast to spawn. Green sturgeon are known to spawn in the Sacramento River. Juveniles inhabit the Estuary until they are about 4-6 years old, at which time they migrate to the ocean. Little is known about the life history of green sturgeon (DFG 1992e).

The white sturgeon is the more common sturgeon in the Estuary and supports an important sport fishery. It apparently makes less extensive ocean migrations than the green sturgeon, and spends most of its life in river and estuarine environments. In the Bay-Delta Estuary, spawning, which appears to be triggered by increasing freshwater flows, occurs in both the Sacramento and San Joaquin rivers (BDOC 1993). Tag returns suggest that spawners in the

Figure V-36
Steelhead Trout Annual Estimated Run Size
Sacramento River Above Red Bluff Diversion Dam (1967-1993)

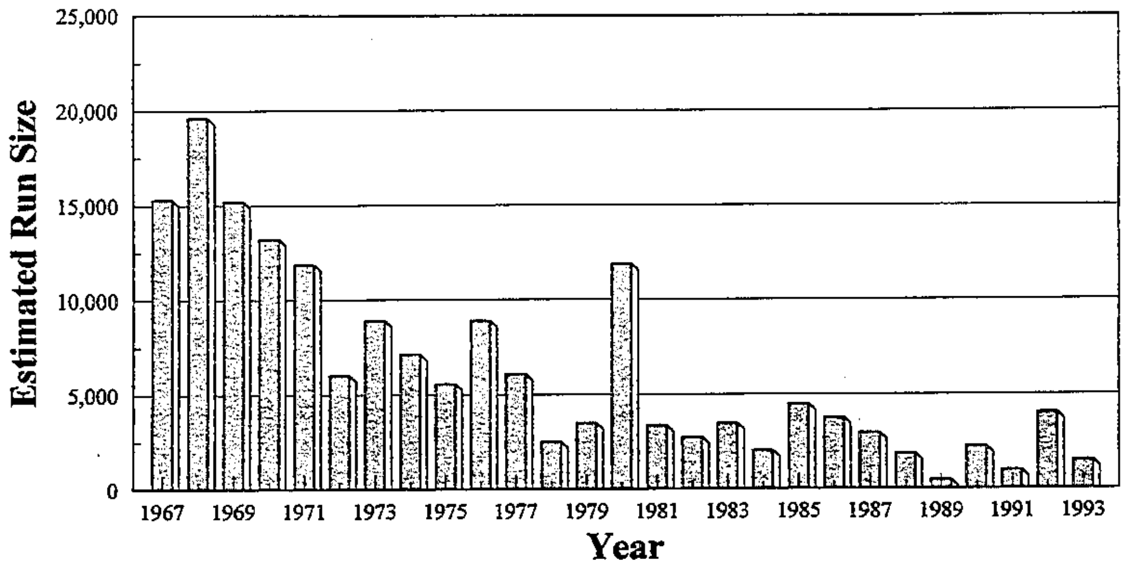
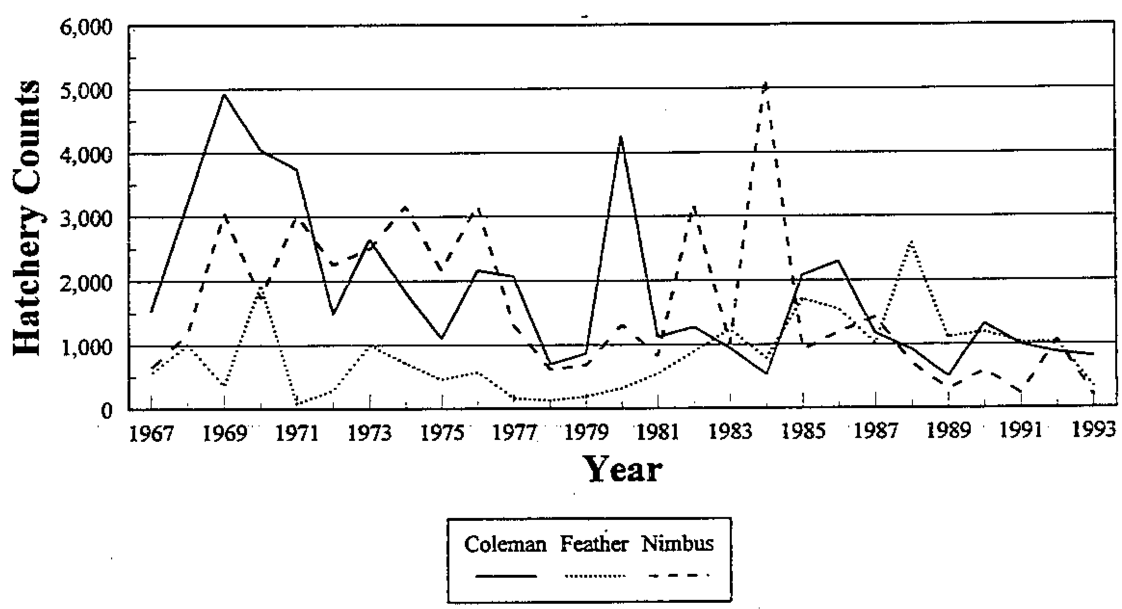


Figure V-37
Steelhead Trout Hatchery Returns
(1967-1993)



Sacramento River are about ten times more abundant than spawners in the San Joaquin River. In the Sacramento River, the spawning season extends from late February through May, with most spawning occurring in March and April at water temperatures of 46-64°F. The eggs sink to the bottom and adhere to solid substrate until they hatch in 5-10 days, depending on water temperatures. Larval movement and dispersal are also dependent on river flow; therefore, the location of the sturgeon nursery area appears to move farther downstream as flows increase (DFG 1992e).

Young white sturgeon grow rapidly, reaching 12 inches at age 1 and 18 inches at age 2. They attain 46 inches, currently the minimum legal size for the sport fishery, at age 11. White sturgeon are long-lived and can reach a large size; reportedly over 100 years old and as large as 1,300 pounds. Most females spawn for the first time at about age 15 and may spawn as infrequently as every 5 years thereafter. Food habits vary with size. Up to 1 year old, white sturgeon feed primarily on benthic invertebrates and *Neomysis*. As they grow, their diet becomes more diverse and includes clams, shrimp, crabs, polychaetes, fish, and fish eggs (DFG 1992e).

Population Trends. Since the sturgeon fishery was reopened to sport fishing in 1954, white sturgeon life history and population dynamics have been studied intermittently. Mark-recapture abundance estimates for white sturgeon ≥ 40 inches (the former legal size for the sport fishery) are available from intermittent tagging efforts between 1954 and 1991 (Figure V-38). Estimated abundance was 114,700 fish in 1967, 20,700 fish in 1974, 117,700 fish in 1984, and 26,800 fish in 1990. These data show that large white sturgeon abundance, which has varied over the last 35 years, has declined dramatically since 1984 (DFG 1994b).

The Bay Study's monthly otter and mid-water trawl sampling from South Bay to the western Delta estimated that the production of white sturgeon year classes declined between 1980 and 1990 (Figure V-39). Estimated production from the 1982 and 1983 year classes was substantially greater than production between 1987 and 1990. Both 1982 and 1983 were years of very high spring and early summer freshwater outflow from the Estuary; 1987-1990 were drought years with very low outflow. These data indicate a strong correlation between year class index and outflow between April and July, with spring flows being more important. Salvage data from 1968 to 1987 also indicate that the production of young sturgeon is dependent on spring outflow, especially in April and May (DFG 1992e). Therefore, recruitment in white sturgeon appears to be greatest in years of very high outflow during the spawning and nursery period (April-May) (SFEP 1992a).

Fall abundance estimates of green sturgeon have ranged from about 200 fish in 1954 and 1974, to about 1,850 fish in 1967 (Figure V-38). Overall, green sturgeon abundance in the Estuary has steadily decreased since 1979 (DFG 1994b).

Causes of Decline. It appears that white sturgeon abundance has declined since 1984 due to both low recruitment between 1975 and 1982, and high harvest rates in the mid- to late-1980's (DFG 1994b). Evidence suggests that recruitment to the adult population is

Figure V-38

White and Green Sturgeon Abundance Indices

Fall Mark-Recapture Estimates (Intermittent 1954-1990)

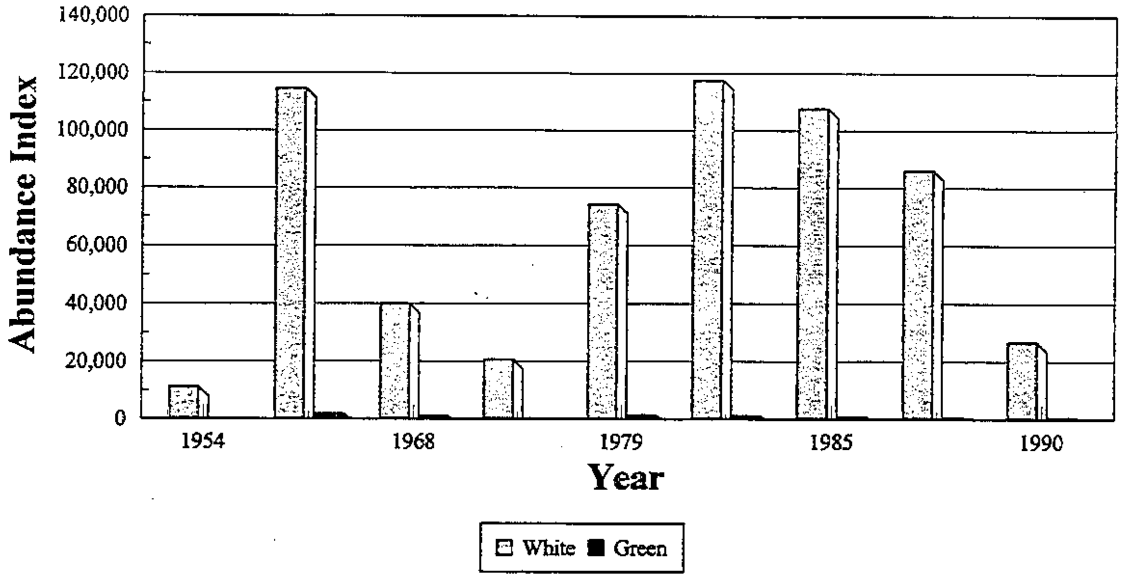
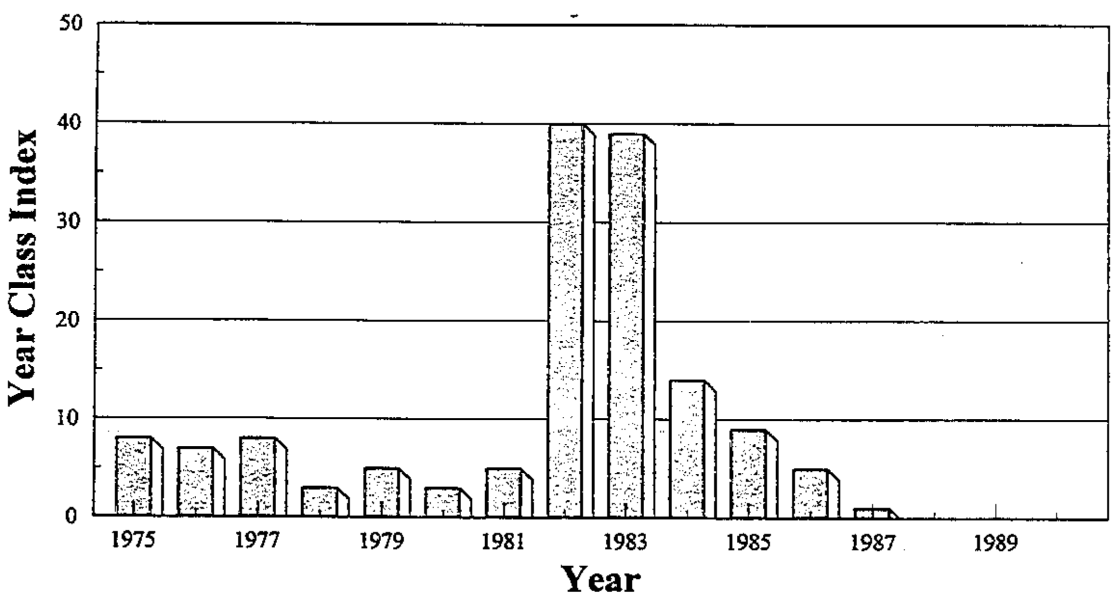


Figure V-39

White Sturgeon Year Class Indices

South Bay to Western Delta (1975-1990)



increased following years of high outflow (e.g., 1982-1983) and decreased following years of low outflow (e.g., 1987-1990). High flows may improve young sturgeon survival by transporting larvae to areas of greater food availability, dispersing larvae over a wide area, quickly moving larvae downstream of the influence of the pumps, and increasing nutrients to enhance productivity in the nursery area. Additionally, adults may experience a stronger attraction to upstream spawning areas and spawn in greater numbers in high flow years (DFG 1992e).

In addition to the effects of variable outflows on white sturgeon populations, exploitation rates in the late-1980's were about 40 percent higher than in the preceding two decades. This increase in harvesting, which occurred as a result of more sophisticated fishing techniques, may have reduced annual survival rates, abundance, and egg production (SFEP 1992a). Due to concerns about the status of the white sturgeon population, angling regulations were changed in 1990 to increase the minimum size limit from 40 inches to 46 inches and to impose a maximum size limit of 72 inches. These new restrictions have reduced the harvest rate to about one-third of the late-1980's level (DFG 1994b).

Although the green sturgeon population in the Estuary has shown a gradual decline, it is uncertain if conditions in the Estuary are affecting this species. Apparently, the green sturgeon is being overexploited throughout its range (NHI 1992a).

d. Striped Bass. The striped bass (*Morone saxatilis*) is native to streams and bays of the Atlantic Coast. It was first introduced into the Bay-Delta Estuary in 1879. Within 10 years, this highly fecund and voracious predator was supporting a commercial fishery in the Estuary (SFEP 1992a). In the Delta channels, adult striped bass primarily feed on fish. In the more saline portions of the Estuary, principal foods include anchovy, shiner perch, herring, and bay shrimp (BDOC 1994).

California striped bass spend most of their life in the Bay-Delta Estuary and along the Pacific Coast, within a few miles north and south of the Golden Gate (DWR 1992a). This anadromous fish resides in the ocean and brackish waters and enters the fresher waters of the Estuary to spawn (BDOC 1994). Approximately one-half to two-thirds of the striped bass spawn in the Sacramento River system, while the remainder spawn in the lower San Joaquin River. Important spawning areas include the main stem Sacramento River from Sacramento to Colusa, and in the San Joaquin River, between Antioch Bridge and the mouth of Middle River. Striped bass begin spawning in the Delta in spring, during April and May, when water temperatures reach about 60°F; most spawning occurs when water temperatures are between 61 and 69°F (BDOC 1993). Further up the Sacramento River, spawning occurs from about mid-May though mid-June. The difference in timing is due to temperatures rising more slowly in the Sacramento River than the lower San Joaquin River (DWR 1993).

Striped bass spawn in fresh water where there is moderate to swift currents. With slower currents, many eggs, which are slightly heavier than water, sink to the bottom and die (DFG 1993). The semi-buoyant striped bass eggs drift with river currents and are carried downstream. Larvae hatch two to three days after spawning. Initially, the larvae receive nourishment from the yolk sac, which is absorbed in five to ten days. As they move downstream toward the Delta, larvae begin feeding on small zooplankton. Upon reaching the western Delta, which is presently their primary rearing area, larvae are large enough to begin feeding or larger organisms such as the opossum shrimp (*Neomysis mercedis*). *Neomysis* remains the main food source until the striped bass reach their second year when they become large enough to feed on bay shrimp and small forage fish. They reach maturity at 3 to 4 years of age and may live to 20 to 30 years of age. In recent years, most of the adult striped bass in the Bay-Delta system are in the 4 to 7 year age classes. The older, more fecund fish, are no longer present in great numbers (DWR 1993).

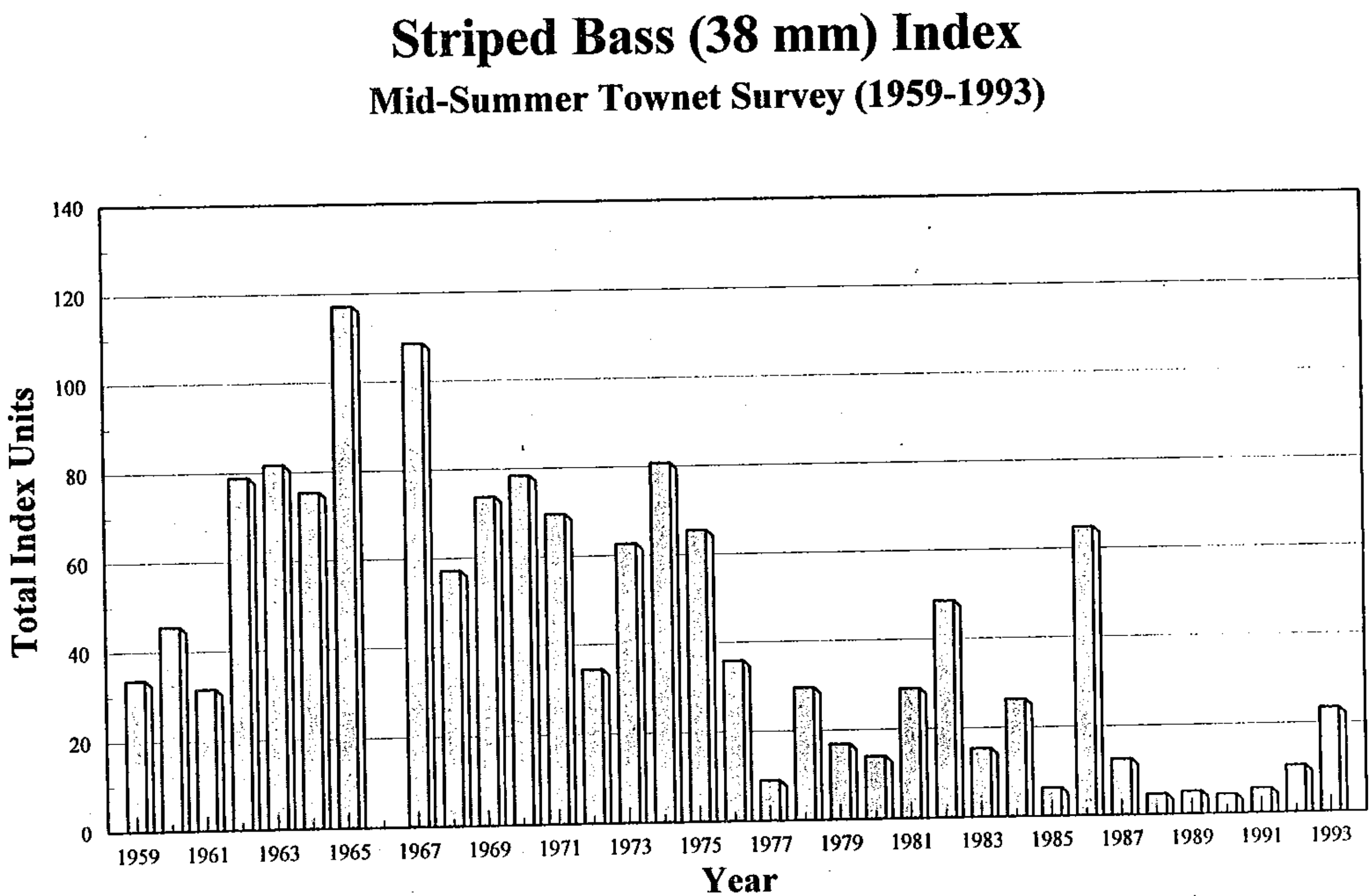
Beginning in 1982, the DFG stocked striped bass in the Estuary, largely as mitigation for various projects, in an effort to maintain the population. The stocking was stopped in 1992 due to concerns that the effort was adding predators which might eat the endangered winter-run chinook salmon (BDOC 1994).

Population Trends. The striped bass population in the Bay-Delta Estuary began with a planting of 132 fish in 1879. A subsequent planting of 300 fish was made in 1882. By 1888, a commercial fishery for striped bass was established (Moyle 1976), reflecting the enormous fecundity of this species. By 1889, the striped bass fishery was landing more than 454 tons each year until 1915. Either through overfishing, habitat degradation, or the usual decline in abundance following the successful introduction of a species, the population of striped bass appears to have begun declining in the early years of the 20th century. Finally, in 1935, the commercial fishing for striped bass was banned. Although the striped bass population decline persisted, the recreational fishery continued to attract a large number of anglers until the late 1970's (SFEP 1992a).

Monitoring of the striped bass population began with the DFG's mid-summer tow-net survey in 1959 (DFG 1994b, SFEP 1992a). This survey, which provides data for a striped bass index, based on the abundance of 38 mm young, peaked at 117.2 in 1965 (Figure V-40). The four lowest indices occurred from 1988 to 1991 when the average index was 4.9. From 1959-1976, the average abundance index was 66.6; since 1977, the average has been 19.4 (DFG 1994b). The declines have been more pronounced in the Delta than in Suisun Bay (SFEP 1992a).

Adult population estimates (Figure V-41) are made through extensive tagging of legal-sized striped bass during their spring migration to the Delta from the ocean and bays (BDOC 1993). Based on Petersen mark-recapture population estimates, the number of legal-sized adult striped bass was 624,000 fish in 1992. The 1992 abundance estimate for naturally-produced striped bass, excluding hatchery fish, was about 533,000 fish. This indicates a decline from approximately 1 million fish in the 1980's and 1.7 million fish in the late 1960's and early 1970's (DFG 1994b). For the years prior to 1976, estimates for the total population of adults in the Estuary were between 1,480,000 to 1,880,000; since 1977, the population ranged from 520,000 to 1,160,000 fish (SFEP 1992a). Population estimates of legal-sized 3-year-old fish (Figure V-42), which are the youngest and most numerous component of the adult population, have declined to record lows since 1988 (DFG 1994b).

Figure V-40



Notes: 1983 underestimated due to very high Delta outflows.
Not sampled in 1966.

Figure V-41
Striped Bass Legal-Size Adult (Age 3+)
Mark-Recapture Population Estimates (1969-1992)

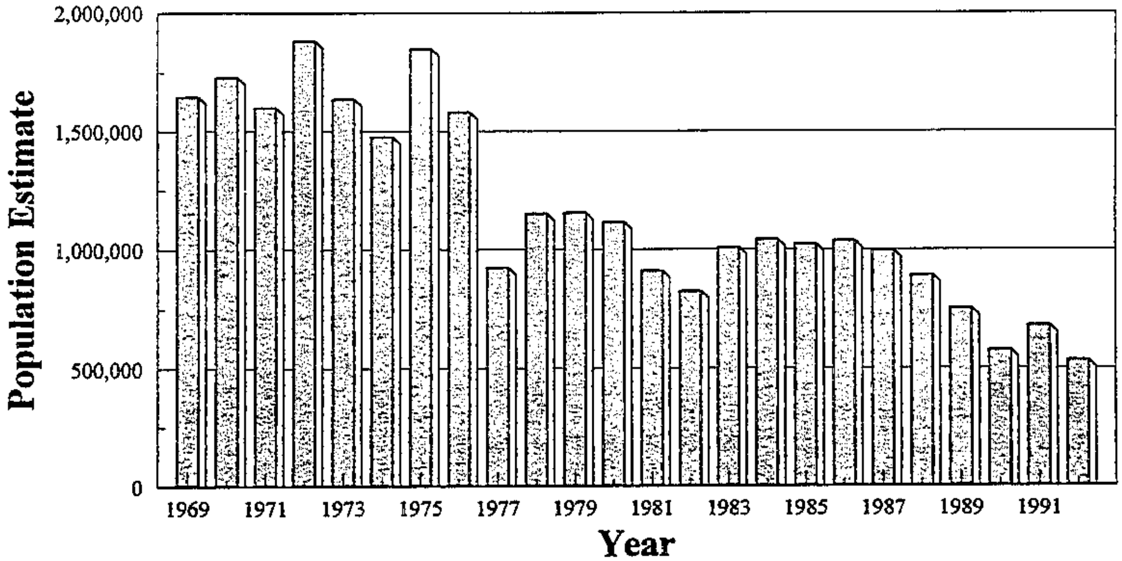
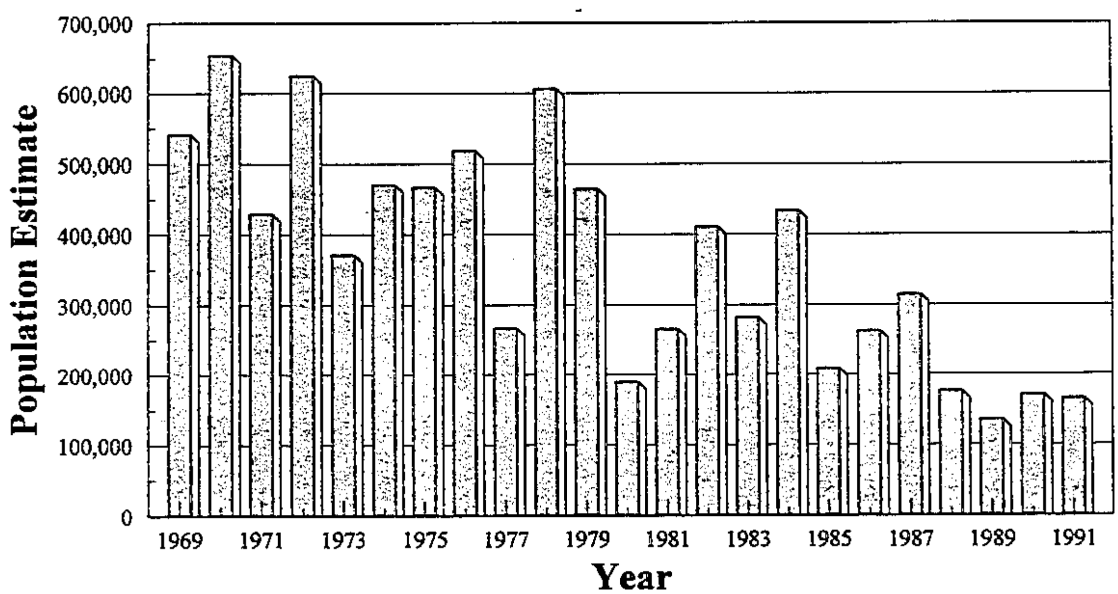


Figure V-42
Striped Bass Legal-Size Age 3
Mark-Recapture Population Estimates (1969-1991)



Causes of Decline. The adult striped bass population decline is a reflection of reduced recruitment. The decline in the adult striped bass population has resulted primarily from the irregular but steady decline of young striped bass (38 mm index) since the mid-1960's (DFG 1994b). It is believed that the decline in young bass predominately is due to a decreased survival rate during the first year of life. Increased mortality of striped bass eggs and larvae is attributed mainly to increased losses through entrainment by the CVP and SWP pumping operations and decreased outflows during the recent 6-year drought (DFG 1992a, DWR 1995c). Agricultural diversions in the Delta also impact fish (DWR 1992a).

Losses to export and entrainment are controlled by freshwater diversion, specifically by the proportion of water diverted for export and within-Delta use (Jassby et al. 1994). Higher outflows move a higher percentage of eggs and larvae out of reach of entrainment, and higher diversions lead to higher percentages of entrainment of eggs and embryos (SFEP 1992a). Higher outflows may also shift the low salinity habitat to a location downstream of the Delta, where larval striped bass appear to survive better (DWR 1992a).

The decline in recruitment due to entrainment by water project operations may have produced an adult population size that does not produce enough eggs to maintain the population. The fact that the percentage of eggs and larvae taken is independent of the numbers present, coupled with ever smaller numbers of eggs produced, makes the interaction of outflows and diversion rates the only adequate explanation for the decline of the striped bass population and its inability to rebound (SFEP 1992a).

Most entrainment of striped bass eggs and larvae at the SWP pumping plant occurs during May, June, and July. With some exceptions, such as during the 1976-1977 drought, the number of bass entrained appears to decrease rapidly from September to December. Losses occur due to passage of eggs and larvae through the fish screens, predation in Clifton Court Forebay, and handling and hauling of salvaged bass. Also, reverse flows in the San Joaquin River could impact striped bass by drawing young fish to the export pumps from spawning and nursery areas in the central and western Delta. There is a significant inverse relationship between flow in the lower San Joaquin River and the number of young bass salvaged at the SWP pumping plant in June and July (DWR 1992a).

Measurements, dating back to 1959, indicate that young striped bass survival increases in proportion to Delta outflow during April through July. There is also evidence that Delta outflow continues to influence bass survival through December. The DFG's statistical model for striped bass indicates that the survival of striped bass during their first year depends on the magnitudes of Delta outflow and State and federal exports in the southern Delta, and that these first year conditions determine subsequent abundance of adult bass (BDOC 1993). Figures VI-1 and VI-2 in Chapter VI show the relationship between mean exports and outflow during April-July and August-March, respectively, to maintain a striped bass population of 1 million, assuming various YOY indices. These figures represent a simplification of the DFG's striped bass model and illustrate how outflows and exports may be managed to maintain striped bass populations in the Estuary.

Besides reducing the likelihood of entrainment into diversions, higher outflows are thought to provide additional benefits for striped bass, including increasing: low salinity nursery habitat in Suisun Bay; primary productivity (food supply); turbidity (reduces predation on young); and providing dilution of pollutants. These factors relate to other possible causes for the continuing decline in striped bass abundance (e.g., food availability, competition, and toxics) (SFEP 1992a).

It is possible that a reduction in food supply has had an effect on striped bass abundance in the Estuary. Since the introduction of the Asian clam (*Potamocorbula amurensis*), zooplankton populations have failed to attain their normal densities. Also, the introduction of the copepod, *Sinocalanus doerrii*, which is less easily captured, has largely replaced *Eurytemora affinis*, a copepod which had comprised a major portion of the young striped bass diet (SFEP 1992a). Although laboratory experiments have demonstrated that the food density for larval striped bass in the Estuary is sometimes low enough to have an effect on both the growth and mortality rates of young bass (Jassby et al. 1994), no direct evidence of starvation has been found (BDOC 1994). Therefore, it is unlikely that decreased food supply and relatively higher abundance of less easily captured prey species has had a significant role in the striped bass decline, but these factors may make recovery of the population more difficult (SFEP 1992a).

There is a potential for competition for food between young striped bass and the introduced inland silverside. Both species have a preference for *Neomysis mercedis*. Although the inland silverside is an inshore feeder and the striped bass is a pelagic feeder, the food source and feeding sites of these two species overlap in the channels of the San Joaquin system and Suisun Bay and Marsh (CUWA 1994).

Agricultural drainage waters that enter the Sacramento and San Joaquin rivers have been acutely toxic to *Neomysis mercedis*, a major prey of young striped bass. There is also evidence that suggests that toxicity adversely affects some bass larvae. However, it is believed that toxicity is not responsible for the striped bass decline. The "background mortality" which results from toxicity, however, has not changed appreciably over the past 30 years (DFG 1992a). However, a study of the effects of rice pesticides on larval striped bass recruitment concluded that during the years investigated (1973-1986), the discharge of water, containing pesticides from rice culture, had adversely affected the striped bass population in the Estuary. The annual die-off of striped bass during May and June is apparently caused by liver deterioration associated with exposure to industrial, agricultural, and urban pollutants (DWR 1992a). Considering that toxic pollutants do impact striped bass to some degree, decreasing the effects of toxics through dilution is consistent with the concept that young striped bass survival improves with increasing outflow (DFG 1992a).

Illegal harvest of undersized striped bass may cause a serious loss to the population. It is estimated that the equivalent of at least 125,000 legal-sized adults are lost each year to poaching; whereas, an average annual loss of an equivalent of 86,000 legal-sized bass occurs due to the SWP pumping plant operations (DWR 1992a). However, the fact that illegal harvest of striped bass is not a new problem, and it is well documented that operation of the export facilities causes mortality to young bass, it is unlikely that the harvest of undersized striped bass has been the dominant factor causing the decline in adult bass abundance since 1969 (DFG 1994b).

e. American Shad. The American shad (*Alosa sapidissima*), a member of the herring family, was first introduced into the Sacramento River in 1871 and was supporting a commercial fishery by 1879 (DFG 1993). American shad are oceanic as adults except for a brief spawning run in fresh water. Most central California adults spawn in the Sacramento River or its tributaries (SFEP 1992a) from late April to early July. Shad enter the San Joaquin River and its tributaries in years when May and June outflow is high (DFG 1993).

American shad do not enter fresh water until water temperatures exceed 50°F. Peak runs and spawning occur at water temperatures between 59°F and 68°F (Moyle 1976). Spawning occurs where there is a good current in tidal fresh water or farther upstream. The fertilized eggs, which are slightly heavier than water, drift with the current, near the bottom. After hatching, from about May to late July, some young shad move downstream into brackish water, but large numbers remain in fresh water into November. By December, most young shad have left fresh water (DFG 1993). Many adults die after spawning, but some return to the ocean and spawn again in later years (SFEP 1992a). Large numbers of dead shad are particularly noticeable when spawning occurs above 68°F (Moyle 1976).

Population Trends. The DFG sampling programs do not encompass many of the times or locations where American shad occur; however, it is still possible to determine some patterns in the data. It appears that American shad recruitment increases in wetter years. Fall mid-water trawl survey data (Figure V-43) indicate that lower catches of American shad have generally occurred during drought periods (e.g., 1976-1977 and late 1980's). Runs of American shad in the Sacramento River have been estimated at 3.04 million fish in 1976 and 2.79 million fish in 1977, but populations in the early part of the century were likely two to three times as large. No recent estimates of spawning numbers seem to exist (SFEP 1992a).

Trawl data from the Bay Study show that catches of American shad fluctuated in the first five years of the study (1980-1984); however, 1981, a dry year, was not the lowest catch in this period. The four lowest catches occurred during the last four years (1985-1988) (SFEP 1992a) which, except for 1986, were dry years .

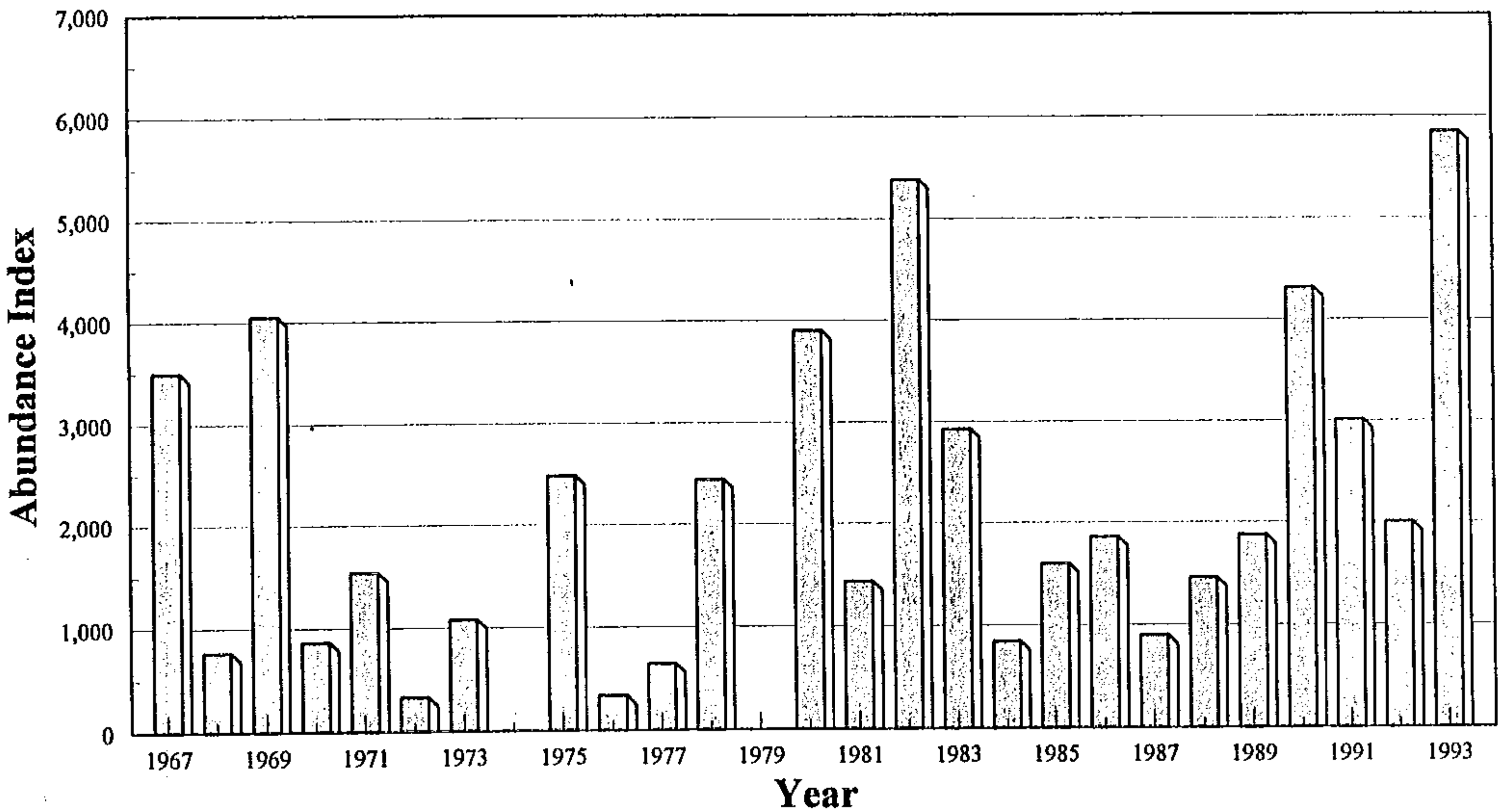
Peak salvage of young shad, at the Skinner Fish Facility, generally occurs during the main outmigration period, between July and December (DWR 1992a).

Causes of Decline. The American shad population data, though limited, appears to indicate that shad recruitment is lower during drier years (i.e., lower Delta outflow). A mechanism that may explain the linkage of shad abundance with outflow is the effect of outflow on water temperature. Drought conditions are often accompanied by increases of temperature, in the smaller volume of water, so that young shad are stressed, most likely in the Delta or upstream. Water temperatures over 68°F are known to cause increased

Figure V-43

American Shad Abundance Indices

Fall Mid-Water Trawl Survey (1967-1993)



Note: Not sampled in 1974 and 1979.

mortality in young shad. Increased entrainment during dry years probably also contributed to the decline (SFEP 1992a). It is likely that the factors that affect striped bass (e.g., Delta outflow and entrainment) also affect young American shad. It also appears that upstream conditions (e.g., flow and temperature in the rivers, where spawning occurs) play a critical role in recruitment to the population (DWR 1992a).

C. CONTROLLABLE CAUSES OF DECLINES

The factors that have been identified as causing or contributing to the declines of various aquatic species in the Bay-Delta Estuary can be grouped into two broad categories: controllable and uncontrollable. Uncontrollable factors include: climatic changes; variations in natural hydrology, due to seasonal and annual variability in precipitation; oceanic conditions; and permanent conditions, such as dams and other major constructed facilities. Although such factors undoubtedly influence the health of the Bay-Delta Estuary ecosystem, they are generally beyond the reasonable control of the people of the State. However, adverse effects of many uncontrollable factors may be offset, at least in part, by addressing controllable factors.

Controllable factors can be defined as those which can reasonably be influenced by human actions. Among the controllable factors that influence aquatic resources in the Estuary are: (1) freshwater flows; (2) entrainment; (3) water temperature; (4) pollution; (5) introduced species; and (6) harvesting. The extent to which controls can be exerted varies among these factors. Furthermore, some of these factors are outside the authority of the SWRCB. Yet, it is crucial to the success of a comprehensive approach for protection of the Bay-Delta Estuary that each factor be addressed to the extent possible.

1. Freshwater Flows

Freshwater flows to and through the Bay-Delta Estuary are primarily influenced by the amount of precipitation that occurs in the Estuary's watershed, and the existence and operations of water development project facilities. While extended drought periods are known to adversely impact aquatic resources in the Estuary, the amount and timing of precipitation is beyond human control. However, although the existence of major water project facilities are considered permanent, uncontrollable factors, their operations are controllable to a great extent.

Under its water right authority, the SWRCB can specify the amount, timing, and conditions of instream flows and water diversions in the Estuary's watershed to the extent that they are within the control of the water right holders in the basin. Thus, specific terms and conditions can be placed on water right permits and licenses toward meeting the conditions necessary for the reasonable protection of beneficial uses.

2. Entrainment

The diversion of water for offstream use or in-Delta pumping results in the entrainment and mortality of numerous aquatic organisms in the Estuary. Besides the direct mortality that occurs with physical entrainment, additional losses are incurred through predation at intakes and fish salvage facilities, and through the salvage process itself.

As part of water right permits and licenses, the SWRCB can include requirements on pumping rates and the installation of fish screens to reduce the numbers of organisms entrained by diversions. In conjunction with these requirements, losses due to entrainment can be minimized through the efforts of other entities, including: (1) designing, installing, and effectively operating fish screens or other protective devices at unscreened diversions associated with fish mortality; (2) improving screening efficiencies and salvage operations at the SWP and CVP facilities; (3) continuing the predator control program for Clifton Court Forebay; and (4) designing, installing, and effectively operating gates or other barriers at channel openings known to be associated with entrainment losses (e.g., continue evaluation of the effectiveness of an acoustic barrier on Georgiana Slough).

3. Water Temperature

Water temperatures in the rivers and the Delta have been primarily affected by changes in flow regimes and loss of streamside (riparian) vegetation. As a result, warm water temperatures that are detrimental to species that require cool water for successful spawning and migration, occasionally occur in some portions of the Bay-Delta Estuary watershed.

Under present conditions, water temperatures can only be minimally controlled. The most likely effective control is a combination of both maximizing reservoir control of cool water reserves to the benefit of downstream fisheries and increasing riparian vegetation to provide shading. These measures would improve water temperatures, primarily in the tributaries, and could provide a slight effect on Delta water temperatures. The SWRCB can encourage water project operators to evaluate and implement possible operational and structural modifications to their facilities to reduce water temperatures downstream of their projects. Such actions may include releasing water from the lower levels of the reservoir, maximizing cool water reserves, and installing temperature curtains. In addition, the SWRCB can encourage other State, federal, and local entities to undertake efforts to increase riparian vegetation along the riverine corridors.

4. Pollution

Through the efforts of the SWRCB and RWQCBs, and the USEPA, significant progress has been made in controlling the discharge of pollutants to the Bay-Delta Estuary and tributary streams. Most of the reduction in pollutant loading has occurred in the point source discharges of municipal wastewater treatment plants and industrial discharges. The most serious pollution problems in the Estuary today arise from nonpoint sources such as agricultural drainage, urban runoff, and mine drainage.

The control of pollution can be advanced through: (1) the adoption of water quality objectives/criteria for additional pollutants that adversely impact beneficial uses; (2) improvements in source control and pretreatment programs; (3) expediting the clean-up of toxic hot spots; and (4) improvements in management practices. As the waste discharge requirements of point source discharges are reissued, new water quality standards and pollution control measures are implemented. The management and control of agricultural drainage, urban runoff, and other nonpoint sources of pollution are being addressed by the SWRCB's Nonpoint Source Management Program, established in 1987. This program establishes a systematic management approach to the difficulties of nonpoint source pollution, by developing an inventory and ranking of nonpoint source problems, a statewide assessment, and management recommendations. Application of this general approach, which resulted in the significant reductions that have occurred in the concentrations of rice herbicides in the Sacramento River since 1991, is continuing with the cooperation of various entities.

5. Introduced Species

It is generally infeasible to effectively reduce or eliminate introduced species from the Bay-Delta Estuary; however, some degree of control is possible with certain non-native species, such as carp and water hyacinth. Because most introduced species cannot be completely eliminated from the Estuary, it is more desirable to focus efforts on preventing additional non-native species from being introduced. These efforts should include: prohibiting the intentional introduction of non-native species, including those intended for scientific and commercial purposes; developing, implementing, and enforcing stringent regulations to control discharges of ship ballast water within the Estuary and adjacent waters; controlling invasive terrestrial plants; restoring native plants; and investigating the feasibility of biological control for invasive non-native aquatic plants.

Although the SWRCB does not have direct control over the sources or management of introduced species, the SWRCB encourages and supports the efforts of other State and federal entities to that end.

6. Harvesting

There is no doubt that both legal and illegal harvesting of aquatic resources in the Bay-Delta Estuary reduces the abundances of populations; however, the significance of such impacts is difficult to determine. Although harvest regulations are not within the control of the SWRCB, the SWRCB would support a review and modification, if necessary, of harvest regulations for species of concern. Furthermore, the SWRCB would encourage strengthening programs to reduce the illegal harvest of aquatic species.

7. Conclusion

The management of controllable factors associated with the decline of aquatic resources is necessary. However, the relative effects of the controllable and uncontrollable factors have not been quantified. Therefore, management of controllable factors may not significantly improve the condition of the aquatic resources in the Estuary, due to the effects of the uncontrollable factors, but such efforts should be made with this uncertainty in mind.

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