Appendix V Other Stressors on Delta Smelt

This appendix is a compilation of supplemental material that summarizes five main potential areas of non-Project effects to delta smelt, specifically effects of: predation by non-native recreational fisheries; contaminants; water diversions; reduced habitat; and aquatic invasive species. These elements are generally discussed in the body of the Biological Assessment as factors that both directly and indirectly affect delta smelt, and that it is the summation of many factors that is theorized to contribute to the overall Pelagic Organism Decline.

The material in this appendix illustrates the contribution of predation as one of many factors that may be suppressing the delta smelt population and impeding species recovery. In addition, the documentation provides information about contaminants found in water samples taken from the Sacramento-San Joaquin Delta that can cause chronic toxicity to standard test species. The higher level of toxicity coupled with increased invasive exotic species is a significant concern. Finally this appendix provides additional detail regarding the potential effects of diversions and reduced habitat.

Effects of Contaminants on Delta Smelt

The acute and chronic effects of contaminants on aquatic organisms and the presence of contaminants in the Sacramento and San Joaquin rivers and Sacramento-San Joaquin Delta are well documented. While the data on the direct and indirect effects of contaminants found within the Delta on aquatic organisms are limited, some suggest that the available information on contaminants are a significant problem to the biota of the Delta (USFWS 1996).

Numerous sampling programs have detected contaminants in Delta water and sediment samples, occasionally at toxicologically relevant concentrations and often with multiple other contaminants that in combination could be toxicologically relevant. Contaminants enter the Sacramento and San Joaquin rivers and Delta by urban and agricultural runoff, atmospheric deposition, municipal water treatment effluent, recreational and commercial boating activities, and navy operations, as well as the legacy from historical mining operations. Researchers have reported elevated levels of several contaminants, such as metals and pesticides, hydrocarbons, chlorinated organic compounds (DDT and metabolites DDD, DDE; PCBs; chlordane), and organometals (tributyl tin) in the Delta (Guo et al. 2007, Smalling et al. 2007, Amweg et al. 2006, CVRWQCB 2007, Bennett et al. 2001, Kuivila and Moon 2004, Bennett 1996, SWRCB 1990, USFWS 1996, USGS 1993). A number of these pollutants have been detected in areas inhabited by delta smelt at concentrations that are known to be lethal or sublethal to delta smelt and the organisms upon which they feed. Contaminant concentrations vary seasonally and many are highest following rain events (Connor et al. 1993).

Significant evidence exists that water and sediment samples from within the Sacramento-San Joaquin Delta cause acute and chronic toxicity to standard test species (CVRWQCB 2007; Amweg et al. 2006; Werner 2005, Werner et al. 2006, 2007a, 2007b; Weston et al. 2004). For example, monitoring from May 2004 to October 2006 for the Irrigated Lands Program from the Coalition Group Monitoring, University of California, and Surface Water Ambient Monitoring

Program (SWAMP) showed significant toxicity to at least one test species at 59% of the sites tested on at least one occasion (CVRWQCB 2007). Another study tested sediment samples from the Sacramento area and found that 22 of the 33 samples caused significant toxicity to *Hyalella azteca* and seven of the eight creeks tested had toxic samples on at least one occasion (Amweg et al. 2006). While there is very little known about the sensitivity of delta smelt to many of the contaminants detected, where they have been tested, delta smelt appear to be more sensitive than the standard test species. For example, Werner found that three-month old delta smelt are 10-12 times more sensitive to copper than three-month old striped bass (Werner 2005), and may be more sensitive to unionized ammonia than other fish species (Werner et al. 2006).

Sublethal adverse effects, such as impaired growth and reproduction, behavioral changes, and even population collapse are well documented in numerous aquatic organisms in response to chronic exposure to low concentrations of one or more toxicants. Exposure to multiple compounds can have additive or synergistic adverse effects, which may be compounded by other environmental stressors. There is extensive documentation of these sublethal and synergistic effects on other aquatic organisms (Ward et al. 2008, Oros et al. 2005, Clifford et al. 2005). For example, juvenile Chinook salmon exposed to 0.1 μg/l esfenvalerate (a pyrethroid insecticide) for 96 hours and to infectious hematopoietic necrosis virus had higher mortality and died sooner than those exposed to either agent separately (Clifford et al. 2005). Similarly, goldfish exposed to sublethal concentrations of copper and to *Mycobacterium marinum* exhibited greater immune suppression (measured by macrophage function) than fish exposed to the copper or bacteria alone (Jacobson et al. 1999).

Behavioral changes in fishes, such as decreased ability to detect prey and avoid predation, have been associated with exposure to very low concentrations of contaminants (Linbo et al. 2006, Lurling et al. 2007, Sandahl et al. 2004, Sandahl et al. 2007, Raloff et al. 2007, and Ward et al. 2008). In an aquatic system in which available sources of food are declining and nonnative predators are increasing, such as is the case for delta smelt, these behavioral changes could be critical to survival. Conditions of decreased food supply (*Eurytemora affinis, Pseudodiaptomus forbesi*) and increased nonnative species (*Micropterus salmoides, Dorosoma petenense, Menidia berrylina, Corbula amurensis, Microcystis aeruginosa*), which prey on delta smelt or compete with delta smelt for food and/or habitat, exist in areas known to be inhabited by delta smelt (Bennett 2005, Bennett and Moyle 1996). In particular, feeding success, which for delta smelt is facilitated by turbidity (Lindberg et al. 2006), is impaired by colonies of *Microcystis* (Lehman et al. 2005a,b) and upstream water channelization and management (for examples, dams) (BDCP Options Evaluation Report 2007). Decreased feeding success and reduced growth often lead to reduced survival and reproductive success. Notably, mean length of adult delta smelt captured by the fall midwater trawl survey has declined significantly from 63.0 mm in 1975-1991 to 53.9 mm in 1992-1997 (http://www.dfg.ca.gov/habcon/cgi-bin/read_one.asp?specy=fish&idNum=62). Embryotoxicity caused by direct exposure to contaminants or maternal transfer of contaminants to eggs, combined with the loss of suitable spawning substrate and increased predation of eggs by nonnative species, can negatively affect the production of delta smelt and other Delta fish larvae.

Following is a detailed description of the significant contaminants of concern in the Delta, empirical data regarding their presence, and information regarding their lethal and sublethal effects on native fishes including delta smelt.

Metals

A diverse array of metals – aluminum, arsenic, cadmium, copper, chromium, lead, mercury, nickel, silver, vanadium, and zinc – have been detected in the Sacramento and San Joaquin rivers and Delta and downstream, from Stockton (SWRCB 1990). Exposure to metals, even at low concentrations often measured in the environment, can exert toxic effects, such as changes in feeding, growth, and swimming behavior, on aquatic organisms, especially on sensitive early life stages. Below are brief discussions of the toxicological effects of copper, mercury, and aluminum.

The use of copper in the Delta has been extensive. There are multiple sources of copper in the Delta including fertilizers, aquatic weed control agents, industrial point and nonpoint discharge, acid mine drainage, wastewater treatment plant effluent, urban runoff (motor vehicles, residential use), and marinas. Concentrations of copper in delta smelt in the Sacramento River have been measured at 6.5 mg/kg (wet weight), which is over 32 times higher than normal background concentrations (Bennett et al. 2001). Copper in northern California streams has been measured at 3.4 to 64.5 µg/L following storm events (cited in Sandahl et al. 2007). The Sacramento River Watershed Program has detected copper as high as 21.5 µg/L in the Colusa Basin Drain, and 18.9 µg/L in the Sacramento River near Hamilton City (Bay Delta and Tributaries Project 2008). The Department of Water Resources' Environmental Monitoring Program (EMP) detected up to 478 µg/L total copper and 149 µg/L dissolved copper, average 27 µg/L and 16 µg/L, respectively (n=42 and 25) in Suisun Bay between 1975 and 1993 (Bay Delta and Tributaries Project 2008). Between 1975 and 2002, total copper concentrations averaged 13 µg/L in the San Joaquin river and 10 vg/L in the Sacramento River (n=114 and 202, respectively) (Bay Delta and Tributaries Project 2008). Concentrations in Mosher Slough and Duck Creek have been reported as high as 500 µg/L and 670 µg/L, respectively (City of Stockton Annual Report 2004-2005).

Copper is highly toxic to all elements of the food web that supports smelt – microbes, algae, invertebrates, and the fish themselves (especially early life stages). Exposure may be waterborne or dietary, or both. Importantly, concentrations as low as 2 µg/L can inhibit the ability of juvenile coho salmon to locate prey and avoid predation even after only three hours of exposure (Sandahl et al. 2007). Copper at 24.7 µg/L caused mortality to 50% of three-month old delta smelt in 7-day lab exposures (Werner 2005).

Mercury contamination is one of the primary water quality issues in the Delta region and is largely due to the extraction of millions of kilograms from the California Coastal Range and its use in Sierra Nevada gold mining during the mid 1800s to early 20th century. Large amounts of mercury are still annually transported to the Bay-Delta, contaminating sediment, water, and biota. Adverse health effects of mercury (discussed in further detail below) are dependent on numerous factors, most importantly the conversion by bacteria from the relatively nontoxic, elemental form to the highly toxic, bioavailable form (methylmercury) by a process known as methylation. In a recent study, Heim et al. (2007) examined temporal and spatial variation in concentrations of total mercury and monomethylmercury (MMHg) in sediments of various ecosystem types in the Bay-Delta. Total mercury concentrations in sediment in the central Delta were generally .100-.200 µg/g and increased westward through Suisun Bay to .250-.350 µg/g, whereas MMHg concentrations in the central Delta were between .001 and .003 µg/g. Although total mercury sediment concentrations varied as a function of location and not season, MMHg concentrations varied seasonally, increasing from .001 µg/g during winter months to .006 µg/g during spring and summer. Total mercury appears to be a key factor controlling MMHg sediment concentrations within marsh habitat; however, other factors such as temperature may play an important role, which would explain higher concentrations in warmer months. Ecosystem type also partly determines MMHg concentrations in sediment. Heim et al. reported higher MMHg concentrations in marsh habitat compared with open water habitat. Slotton et al. investigated mercury methylation potential at over 60 varied sites in the Bay-Delta and concluded flooded wetland sediments exhibited 200% and 3000% greater mercury methylation potential compared with sediments of adjacent channels and flats.

Between 1993 and 2001, the CALFED Mercury Transport, Cycling and Fate Study measured total mercury in water samples collected from the Sacramento and San Joaquin watersheds. Unfiltered total mercury ranged between 1.58-2210 ng/L, average 69.9 ng/L (n=342), dissolved mercury ranged between 0.1-5.39 ng/L, average 1.06 ng/L (n=151) (Bay Delta and Tributaries Project 2008). Between 2000 and 2002, the Sacramento River Watershed Project and the CALFED Mercury Transport, Cycling and Fate Study measured methyl mercury concentrations in water samples collected throughout the Sacramento and San Joaquin River watersheds. Filtered methyl mercury concentrations ranged between 0.007-1.183 ng/L, average 0.007 ng/L (n=248). Unfiltered methyl mercury concentrations ranged between 0.14-1.213 ng/L, average 0.13 ng/L (n=280) (Bay Delta and Tributaries Project 2008). The highest concentrations were measured in Arcade Creek. In Mosher Slough, exceedances in mercury concentrations in water were measured as high as 1900 ng/L, while average mercury exceedances in the Stockton area were 20 times the permitted level of 50 ng/L (City of Stockton Annual Report 2004-2005).

Mercury toxicity in fish is well documented and includes decreased appetite, ability to catch food, visual activity, and growth; lethargy; loss of equilibrium; gill hyperplasia and reduced respiration; neurotoxicity; nephrotoxicity; and teratogenic and reproductive effects (Reimschuessel 2001, Rodgers and Beamish 1982, Matida et al. 1971, Weis and Weis 1991, others). Reproductive effects from mercury exposure are a particularly sensitive endpoint, and can begin with the maternal transfer of mercury to embryos via the yolk (Weis and Weis 1991). Experimental studies have shown that embryo survival can be substantially reduced by very low concentrations of mercury from waterborne exposure or maternal transfer (Birge et al. 1979). For example, rainbow trout embryos experienced 100% mortality after 8-day exposures to 100 ng/L of inorganic mercury, resulting in about .07 to .10 µg/g (wet weight) in fertilized eggs, which is less than 1% of the tissue residues in the adult fish. Effects on feeding behavior and growth are also a sensitive endpoint of mercury exposure. Field studies comparing northern pike from a mercury-contaminated lake in northwestern Ontario to a relatively, uncontaminated reference lake reported fish from the contaminated lake were emaciated, had low hepatic fat stores, and showed clinical signs of starvation (low levels of total protein, glucose, cholesterol, and alkaline phosphatase) (Lockhart et al. 1972). Accumulation of mercury tends to be higher in smaller fish with higher metabolic rates, such as smelt (Reinert et al. 1974). Concentrations of mercury in delta smelt in the Sacramento River have been measured at 600g/kg (dry weight) (Bennett et al. 2001).

Aluminum concentrations between 1996 and 1998 in Barker and Lindsey Slough, near delta smelt winter spawning habitat, averaged 74 µg/L dissolved aluminum and 2400 µg/L total aluminum (n=135 and 64, respectively) (Department of Water Resources Municipal Water

Quality Investigations Program). Peak concentrations were measured at 729 µg/L dissolved aluminum and 11,000 µg/L total aluminum (Bay Delta and Tributaries Project 2008). Aluminum exceedances in Calaveras River and Duck Creek were measured as high as 120,000 µg/L, or 600 times the permitted level of 200 µg/L (City of Stockton Annual Report 2004-2005). Excluding these very high values, average exceedances in the Stockton area are slightly more than six times the permitted level (City of Stockton Annual Report 2004-2005).

Experimentally, Lewis (1987) determined that the highest concentration of aluminum at which no adverse effects can be detected (NOEC) was less than 57 µg/L at pH 6.6. Early life stages are more sensitive to aluminum exposure which causes decreased hatch success, changes in feeding behavior (age 1-30 days), and decreased growth and survival (age 60+ days).

Pesticides

In the Central Valley of California, approximately 20 and 42 million pounds of pesticides were reported in the California Department of Pesticide Regulation's Pesticide Use Reporting database in 2006 for the Sacramento and San Joaquin River Watersheds, respectively. This poundage does not include residential consumer use. Pesticide detections in streams and other waterways are generally highest following pulse inputs from high flows after rain events, and frequently exceed California Department of Fish and Game's water quality criteria to protect aquatic life (Menconi and Cox 1994, Menconi and Paul 1994, Menconi and Gray 1992). Monitoring data from May 2004 to October 2006 for the Irrigated Lands Program from the Coalition Group Monitoring, University of California and Surface Water Ambient Monitoring Program (SWAMP) exceeded the Central Valley Regional Board's triggers for pesticides at 57% of the sites tested on at least one occasion(Table 1).

Table 1. Pesticide Exceedence Detection

Zone	Pesticides Detections		
	Number of sites exceeding trigger level for ≥ 1 pesticide on one occasion	Total number of sites tested	Percent of sites exceeding trigger level at least once
Zone 1	23	57	40.4%
Zone 2	28	46	60.9%
Zone 3	40	55	72.7%
Total Zone 1-3	91	159	57.2%

Table compiled from data within CVRWQCB, 2007.

During this same sampling program, significant toxicity was observed in at least one standard test species at 59% of the sites tested on at least one occasion (Table 2).

Table 2. Observed Significant Toxicity

Species tested	Number of sites with ≥ 1 toxic sample	Number of sites tested	Percent of sites with at least one toxin
Pimephales promelas	26	186	14.0%
Ceriodaphnia dubia	69	185	37.3%
Selenastrum capricornutum	60	157	38.2%
Hyalella azteca	54	139	38.8%
All species combined	119	201	59.2%

Table compiled from data within CVRWQCB, 2007.

Pesticides are of particular concern to delta smelt because their spawning season (from February to June) corresponds with the rainy season in the Central Valley, and with peak pesticide application to orchards, alfalfa, and rice.

Pesticides in water samples from the Delta were analyzed during a three-year USGS study (from 1998 through 2000); of 28 dissolved pesticides subject to testing, 23 were detected. All water samples showed evidence of multiple pesticides, ranging from two to 14 in each sample (with a median of five). Metolachlor was the most frequently detected pesticide (present in 95% of all water samples), followed by rice pesticides molinate (74%) and thiobencarb (61%). These three pesticides overlapped temporally (≥ 2-3 weeks) and spatially with the period of peak densities of larval and juvenile smelt. Metolachlor is slightly to moderately toxic to freshwater invertebrates (LC50 ranges from 1.1 to 25 mg/L) and freshwater fish (LC50 3.2-15 mg/L) (USEPA 2007). Sublethal effects reported in experimental studies of rainbow trout include lethargy, loss of equilibrium, and erratic swimming behavior (USEPA 2007). The only chronic study on freshwater fish exposed to metolachlor was a registrant-submitted study on fathead minnow (USEPA 2007). The NOAEC for the most sensitive endpoint (dry weight of larval fish) was 0.030 mg/L and the LOAEC was 0.056 mg/L. Molinate toxicity varies in its effects on different fish species, but frequently causes behavioral changes, impaired growth, and in some cases anemia. In addition, molinate is highly toxic to invertebrates, which are critical food sources for fishes of conservation concern. Thiobencarb has been associated with fish kills in agricultural drains near the Sacramento and San Joaquin rivers and the Delta. Because rice field pesticides, such as molinate and thiobencarb, are applied in the Sacramento Valley during the delta smelt spawning season, potential toxicity to smelt embryos and larvae is of particular concern. Experimentally, thiobencarb has been found to be embryotoxic in medaka (*Oryzias latipes*) (Villalobos et al. 2000).

In a more recent study, Guo et al. (2007) monitored the Sacramento River and its tributaries for 26 pesticides following a storm event in January 2005 and detected five pesticides and one pesticide metabolite. Diazinon, diuron, and simazine were found in every stream sampled. Diazinon concentrations in the Feather River and Colusa Basin Drain exceeded the water quality criterion of 0.17 µg/L (Guo et al. 2007). Similarly, Smalling et al. (2007) detected 13 currently used pesticides in both sediment samples and water samples from the Yolo Bypass and its tributaries in 2004 and 2005. Simazine and hexazinone were detected in all areas, including 2.5 µg/L hexazinone in water from Willow Slough.

Diazinon is a major dormant spray organophosphate insecticide used in orchards during winter months. Application of diazinon coincides with periods of high-energy demand when adult delta smelt migrate upstream and spawn in freshwater. Diazinon accumulates in and is toxic to aquatic life, particularly invertebrates, causing changes in feeding behavior, growth, predation avoidance, reproduction, biochemistry and enzyme function (USEPA ECOTOX database). In a 1993 USGS study (Kuivila 1993), all concentrations of diazinon measured in the Sacramento and San Joaquin rivers and San Francisco Bay were above the maximum surface water concentration guideline (0.009 µg/L) recommended by the National Academy of Sciences (1973) and the acute and chronic freshwater quality criteria for diazinon (0.170 µg/L) recommended by EPA for the protection of aquatic life [Federal Register: February 23, 2006. Volume 71, Number 36)]. Pulses of diazinon have been recorded at concentrations as high as 0.393 and 1.071 µg/L in the Sacramento and San Joaquin rivers, respectively.

A marked trend of replacement of carbamate and organophosphate insecticides with pyrethroids, a newer class of synthetic chemical insecticides that act in a similar manner to natural pyrethrins derived from chrysanthemum flowers, has occurred in the past decade in the Sacramento-San Joaquin watershed. Pyrethroid use in the Delta system nearly quadrupled from 1990 to 2006 from approximately 27,000 kg/yr to over 101,000 kg/yr in 2006 (California Department of Pesticide Regulation). In addition, there has been a shift in recent years from permethrin, one type of synthetic pyrethroid, to more toxic forms, such as bifenthrin and cypermethrin, which are 21 and 29 times more toxic, respectively, to aquatic life compared with permethrin (Amweg et al. 2005). Amweg et al. calculated the permethrin equivalents of current use pesticides by adjusting for the relative toxicity of the compounds and found a 58% increase in application of permethrin equivalents from 2001 to 2002.

Pyrethroids can enter the aquatic environment from agricultural urban runoff, direct application to water bodies, and drift during aerial spray procedures. More than 20 pyrethroids are currently registered for use in California and are applied to a wide variety of crops including alfalfa, corn, rice, and tomatoes, as well as to most orchard crops. Agricultural use of pyrethroids is greatest during the months of May to October, with peak use occurring in July, corresponding with the time when delta smelt are in the height of their development.

Residential (nonagricultural) use of pyrethroids for pest control has also steadily risen, in part because of the phasing out of diazinon and chlorpyrifos. Pyrethroids are the most common active ingredient in commercial and residential ant and insect repellents. According to the Department of Pesticide Regulation, urban professional application accounts for the largest nonagricultural use of pyrethroids (based on total quantity and permethrin equivalents) and most of these applications are for structural pest control. The most common outdoor use is for ant control around buildings, which is more likely to involve application to impervious surfaces and higher runoff, compared with application to pervious surfaces.

The toxicity of pyrethroids is relatively low in birds and mammals, but is extremely high in fish and other aquatic species with 96-h LC50 values in the ng/L range (Bradbury and Coats 1989, Haya 1989). Chronic exposure studies that measure pyrethroid effects in fish for a complete life cycle indicate that newly hatched larvae and early juveniles are the most sensitive of all life stages. Exposures of fish to sublethal concentrations of pyrethroids have resulted in decreased growth and impaired swimming performance (Haya 1989), increased susceptibility to viral infection (Clifford et al. 2005), and impacts to olfactory response (Sandahl et al. 2004). Acute exposure, even as brief as one hour, to low levels of pyrethroid esfenvalerate during the early larval stage of midge can have measurable population level effects on larval survival and development rates (Forbes and Cold 2005). Acute toxicity tests that evaluated the effects of cypermethrin (concentrations .0001 to 8 µg/L) on the embryos and larvae of the common carp reported a statistically significant dose-response increase in dead embryos and decrease in hatching success, even at the lowest concentration tested (Aydin et al. 2005).

In sediment studies, Amweg et al. (2006) observed significant toxicity to *Hyalella azteca* in 22 of the 33 sediment samples collected from the Sacramento area. Pyrethroid concentrations were high enough to explain the toxicity in 21 of the 22 toxic samples. Similarly, Weston detected pyrethroids in 75% of the 70 sediment samples collected from the Central Valley in 2002 and 2003. Forty-two percent of the sites caused significant mortality to *H. azteca* or *Chironomus tentans* on at least one occasion (Weston et al. 2004).

The growing concern for pyrethroid toxicity to aquatic organisms is evidenced by the recent (August 31, 2006) action by California's Department of Pesticide Regulation, which initiated a reevaluation of pesticide products containing pyrethroids. The reevaluation action was based on recent studies revealing the widespread presence of synthetic pyrethroid residues in the sediments in California waterways at levels toxic to aquatic crustaceans.

Ammonia

Ammonia can enter freshwater from several sources, including agriculture (fertilizer, livestock waste), atmospheric deposition (combustion processes), urban runoff (cleaning products, septic systems), and point sources (sewage effluent, metallurgic operations, strip mining, and others). In water, ammonia exists in two forms: NH_3 (unionized) and NH_4^+ (ionized). The toxicity of ammonia is primarily attributed to the unionized form and increases as pH and temperature increase. Compared with invertebrates, fish are more sensitive to NH_3, but toxicity varies among species of fish. Lethal concentrations of NH_3 range from 0.2 mg/L in rainbow trout (Thurston et al. 1981) to 3.8 mg/L in channel catfish (Roseboom and Richey 1977). Acute exposure to high concentrations of NH_3 can cause hyperexcitability and convulsions. Effects of chronic exposure to lower concentrations of NH_3 include decreased egg production, decreased egg viability, decreased growth, delayed spawning, gill hyperplasia (causing increased ventilation), and increased susceptibility to disease.

In addition to direct toxicity, there is evidence that ammonia levels in Delta water may inhibit phytoplankton production. Field studies and enclosure experiments conducted from 1999 to 2003, indicate that total ammonia levels greater than 4 µmol/L (0.056 mg N/L) inhibit nitrate uptake and bloom formation in the Suisun, San Pablo, and Central San Francisco Bays (Dugdale et al. 2007). Monitoring by the Environmental Monitoring Program of the Department of Water

Resources measured total ammonia levels in Suisun Bay and nearby Grizzly Bay ranging between 0.01 and 0.75 mg N/L, average 0.09 mg N/L (n=168) between 1975 and 2006. Dissolved ammonia ranged between 0.01 and 0.31 mg N/L, average 0.07 mg N/L (n=1323) (Bay Delta and Tributary Project 2008).

Total ammonia concentrations average 0.23 mg N/L on the Sacramento River near Hood and 0.10 mg N/L on the San Joaquin River at Vernalis (Heidel et al 2006). However, the Bay Delta and Tributaries Project database reports concentrations approaching 1 mg N/L on the Sacramento River at river mile 44 and greater than 1 mg N/L in Sacramento Slough. Effluent discharges from the Sacramento Regional County Sanitation District into Sacramento River ranged between 8.0 mg N/L and 29.0 mg N/L between 1999 and 2002 (Sacramento County Department of Environmental Review and Assessment 2003). While Sacramento's treated wastewater is discharged through a 10-foot diameter diffuser pipe, and river concentrations are significantly less than effluent concentrations, these discharges account for a significant percent of the ammonia measured downstream in the Sacramento River. Similarly, effluent discharges from the Stockton Wastewater Treatment Plant into the San Joaquin River often exceed 25 mg N/L total ammonia. Stockton brought a new treatment process on line in 2006 that is expected to reduce ammonia levels significantly.

Pharmaceuticals

The presence of human and veterinary pharmaceuticals in U.S. watersheds is also a concern for the health of zooplankton and fish. The number of U.S. human prescriptions rose to a record 3.7 billion in the past five years, with another 3.3 billion in nonprescription drug purchases (IMS Health and The Nielsen Co.). Prescription drugs enter the watershed by treated wastewater containing drug residues, septic tanks, and waste from livestock treated with veterinary pharmaceuticals, such as anabolic steroids, hormones for increased milk production, and antibiotics. Currently, there are no federally established permit levels or testing requirements.

The issue of pharmaceuticals in the environment recently gained national attention when the Associated Press (http://ap.google.com/article/ALeqM5hGsoyElv4ZL879LW 6z2aZS0Pix7AD8VA14500) released its recent findings from a five-month investigation, reporting that pharmaceuticals were detected in 28 of the 35 watersheds surveyed and in drinking water supplies of 24 major metropolitan areas. Some of the most commonly detected compounds in western states include estradiol (hormone), meprobamate (anxiolytic), phenytoin (anticonvulsant), carbamazepine (used to prevent seizures and to treat bipoloar disorder), sulfamethoxazole (antibiotic), caffeine, and acetaminophen.

Similar results were reported in a 1999-2000 USGS study that sampled 139 streams in 30 states. In this study, one or more organic wastewater contaminants were detected in 80% of the streams. Half the streams contained seven or more chemicals (Barnes et al. 2002). Six of the sites are in the Central Valley. Samples from the Sacramento River at Freeport had cholesterol at 383 ng/L, acetaminophen at 25 ng/L, and mestranol at 11 ng/L. Acetaminophen was estimated at 4 ng/L in Orestimba Creek and San Joaquin River at Vernalis. The Turlock Irrigation District Lateral 5 that drains to the San Joaquin River contained 2 ng/L 17 β-estradiol, 10 ng/L estriol, 113 ng/L 19-norethisterone, 1,110 ng/L cholesterol, 624 ng/L coprostanol (measure for the presence of human fecal matter), 390 ng/L acetaminophen, 680 ng/L caffeine, 17 ng/L diltiazem (potent vasodilator of peripheral and coronary vessels), 210 ng/L 1,7-Dimethylxanthine (caffeine metabolite), and 19 ng/L codeine (Barnes et al 2002). University of California researchers reported 17 ng/L estrone (estrogenic hormone) in a drainage canal near high-density dairy farms in the Central Valley following a storm in 2004 (Kolodziej et al. 2004). Kolodziej et al. also detected testosterone at 1.9 ng/L in an irrigation canal in August 2003.

Many of the pharmaceuticals detected in the environment are endocrine disrupting compounds—synthetic chemicals that mimic or block hormones and disrupt normal physiological functions. Evidence of their effects on wildlife has been documented as early as the 1960s. For example, following the introduction of DDT in the 1940s, it was discovered twenty years later that DDT, and its metabolite DDE, bioaccumulated in fish that were then consumed by bald eagles and osprey, preventing adequate calcium deposition during eggshell formation. The resulting thin eggshells easily cracked during incubation causing precipitous declines in the birds' populations. Also during the 1960s, farmed mink fed PCB-contaminated fish from Lake Michigan tributaries suffered high mortality rates and reproductive failure. In 1980, an accident at Tower Chemical Co. released the pesticide difocol into Lake Apopka, one of Florida's largest lakes, killing 90% of the alligators. After substantial cleanup efforts, remediation was considered complete by the late 1980s. However, persistent effects of the residual contaminants gained international attention when it was discovered that juvenile male alligators had significantly decreased testosterone concentrations, abnormal testes, and phalli one half to one third of normal size preventing successful reproduction. Juvenile female alligators had twice the normal estradiol concentrations and abnormal ovaries. Of the eggs that were fertilized, 80-95% failed to hatch, and the mortality rate for the hatched eggs was at least 10 times the expected rate.

Similar endocrine disruption effects and intersex characteristics have also been observed in fish. Huang et al. (2001) analyzed secondary wastewater effluent and detected concentrations of estrogenic hormones at levels that cause vitellogenesis (egg yolk production and storage) in fish. Effluent from secondary wastewater treatment contained 2.75-4.05 ng/l 17β-estradiol and 1.54-2.42 ng/l 17α-ethinyl estradiol. Vitellogenesis in fish has been observed at concentrations greater than 1 ng/l. Sacramento County currently discharges approximately 154 million gallons per day (average dry weather flow) of secondary treated wastewater to the Sacramento River. Williamson and May (2002) analyzed over 400 adult Chinook salmon collected from 13 locations in the Sacramento and San Joaquin River watersheds and found that up to 38% of the phenotypic female Chinook salmon tested positive for the Y-chromosome marker. Wastewater effluent can also have population level impacts as was observed by Kidd et al. (2007). Kidd conducted a 7-year whole lake experiment that showed that chronic exposure of fathead minnows to 5-6 ng/l 17α-ethinyl estradiol (a synthetic estrogen) led to feminization of males, impacts on gonadal development, altered oogenesis in females, and near extinction of the species from the lake after only two year's exposure.

Other endocrine disrupting compounds found in the Central Valley include bis(2-ethylhexyl)phthalate (DEHP), a phthalate used in the plastics industry. Exposure to DEHP causes reduces growth in brook trout at concentrations around 55 μg/L, and reproductive toxicity at lower concentrations. Exceedances of the permitted level for DEHP (1.8 μg/L) reported for Stockton averaged 5.65 μg/L (range 1.9 to 58 μg/L) (City of Stockton Annual Report 2004-

2005) and 8.5 µg/L (range 3.7 to 16 µg/L) in receiving waters (11/9/06 letter from Courtney Vasquez, Stormwater Program Manager to Greg Vaughn, CVRWQCB).

Effects of In-Delta Water Diversions on Delta Smelt

The narrow habitat range and short life cycle of delta smelt make them particularly vulnerable to human alterations of their habitats in the Sacramento-San Joaquin estuary. Diversion of freshwater is one alteration known to contribute to mortality in delta smelt and other fishes (Bennett 2005; Kimmerer 2004).

Several studies have discussed the potential effect of export pumping on delta smelt populations (Bennett 2005; Kimmerer 2004). Regulatory and recent court actions address the effects of entrainment caused by export pumping. Less is known regarding the effects of diversions for in-delta use,

Effects of in-Delta Agricultural Diversions

Entrainment by agricultural diversions is not frequently invoked as a factor in the decline of delta smelt. However, the large number of diversions, located throughout the delta, can contribute to delta smelt entrainment. Several sources have estimated the number of in-delta agricultural diversions. A California Department of Fish and Game survey counted 2,294 diversions. Of these diversions, only 216, or 9%, are screened. Only 0.6% of the diversions are screened for delta smelt. A paper by Herren and Kawasaki (2001) reported 2,209 water diversions within the delta during 1991-1997, only 17 of which were screened. Ninety percent of the diversion intakes measured between 12 and 24 inches (Herren and Kawasaki, 2001). Another calculation estimated 1,800 surface agricultural diversions within the Sacramento-San Joaquin Delta (California Water Plan 2005). The diversions are distributed throughout the delta, and those with screens are mainly in the delta perimeter (Figure 1).

The shoreline location and timing of most agricultural diversions may also contribute to effects on delta smelt. Delta smelt spawning is thought to occur in shallow and shoreline waters from February through June. Specific geographic spawning locations vary depending on hydrological conditions and temperature (Bennett 2005). Agricultural diversions are mostly active from late March through September when water is needed for spring and summer crops (Brown 1982). This period overlaps with the spring spawning cycle of delta smelt and the subsequent appearance of yolk-sac larvae and larval delta smelt. These early life stages possess limited motility and are located in the shallow and shoreline waters in which they hatched.

During this time, delta agricultural diversions may collectively divert water at an estimated mean rate of over 4,000 cfs (California Water Plan 2005). Approximately 1.3 million acre-feet annually is diverted to support delta agriculture (California Water Plan 2005).

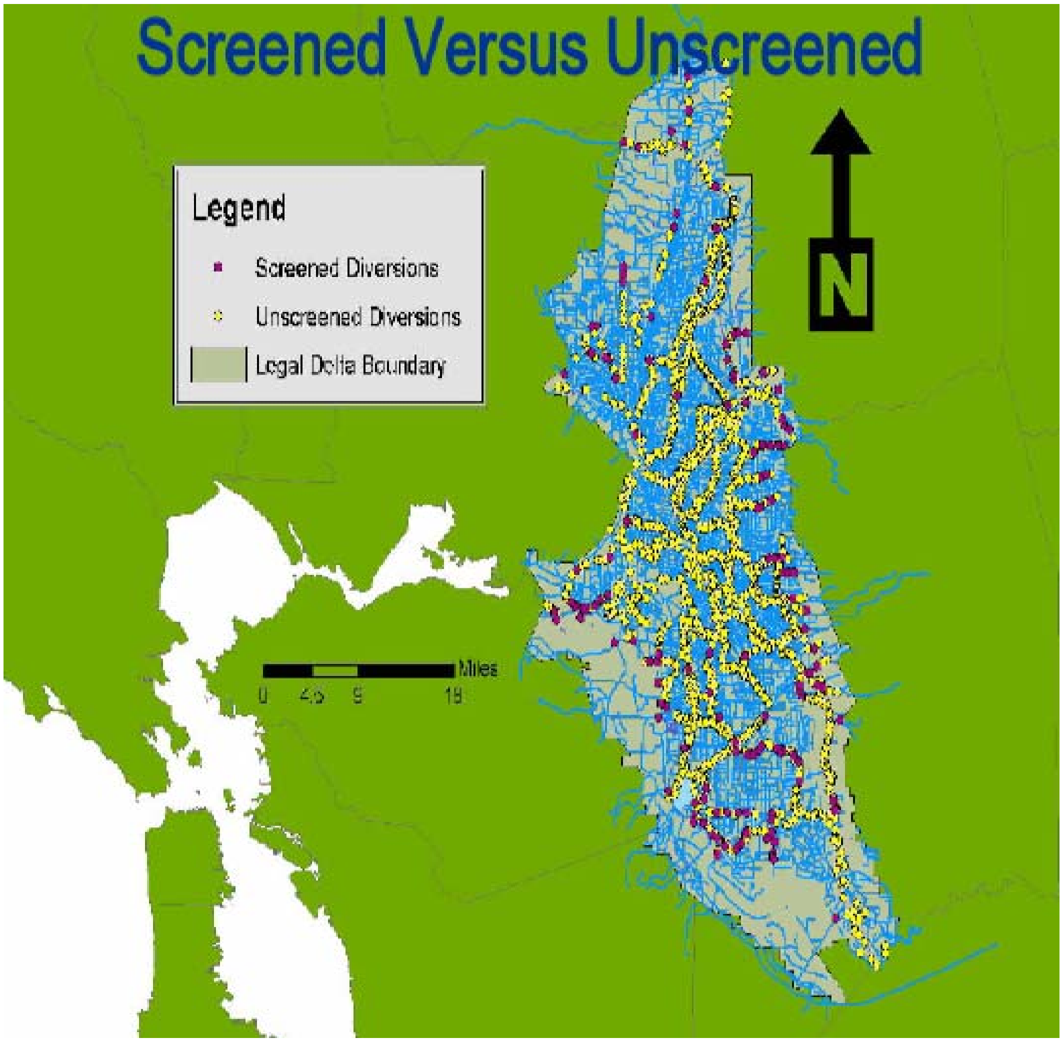


Figure 1. Screened and unscreened diversions in the Sacramento-San Joaquin Bay Delta

Entrainment by in-delta diversions was evaluated in two studies: Cook and Buffaloe (1998) and Nobriga et al. (2004). Cook and Buffaloe (1998) evaluated five sites in the west and south delta using a 2.4 mm egg and larval net staked in the diversion outlet ditch just downstream of the area of turbulence; a fyke net was used to capture later life stages. They found that a large diversity of fish species can be entrained by small agricultural diversions, especially young-of-year fish present from May through August. Catch per unit effort (CPUE) for delta smelt suggested that relatively lower densities of these were entrained compared to other fishes; however, at one site a CPUE of 5.0 was calculated for early-life stage delta smelt, indicating five delta smelt entrained for each acre-foot of water sampled. Extrapolated over the delta smelt habitat area in the western and southern delta, this could represent a significant impact on the population.

Nobriga et al. (2004) evaluated entrainment in a screened and an unscreened diversion in the vicinity of Decker Island, along Horseshoe Bend in July 2000 and July 2001 (Figure 2). The screened diversion was designed to exclude delta smelt and other fishes longer than 25 mm total length. No delta smelt were found in the screened diversion, whereas diversion losses were detected in the unscreened diversion. Specifically, 43 delta smelt were entrained during 69 hours of unscreened sampling of 170,839 m^3 of water. Even greater entrainment of other ecologically similar, open-water species (threadfin shad, inland silverside, striped bass) was observed. The authors refer to data from the 20-mm Delta Smelt Survey (a mid-channel trawling survey) that suggests delta smelt were, in fact, relatively abundant in the vicinity of the experimental diversion. These data, along with their own sampling, led Nobriga et al. (2004) to conclude that delta smelt are distributed farther from shore than the other species. However, Nobriga's study was limited to the month of July and was therefore related to post-larval delta smelt and not earlier larval stages, which display limited motility and great shallow water affinity. Since spawning and larval development is likely to occur in shallow shoreline locations, entrainment of these life stages by agricultural diversions may be more significant. In addition larval and post-larval individuals, which measure less than 25 mm in total length, would not have been excluded by the screens used for this study.

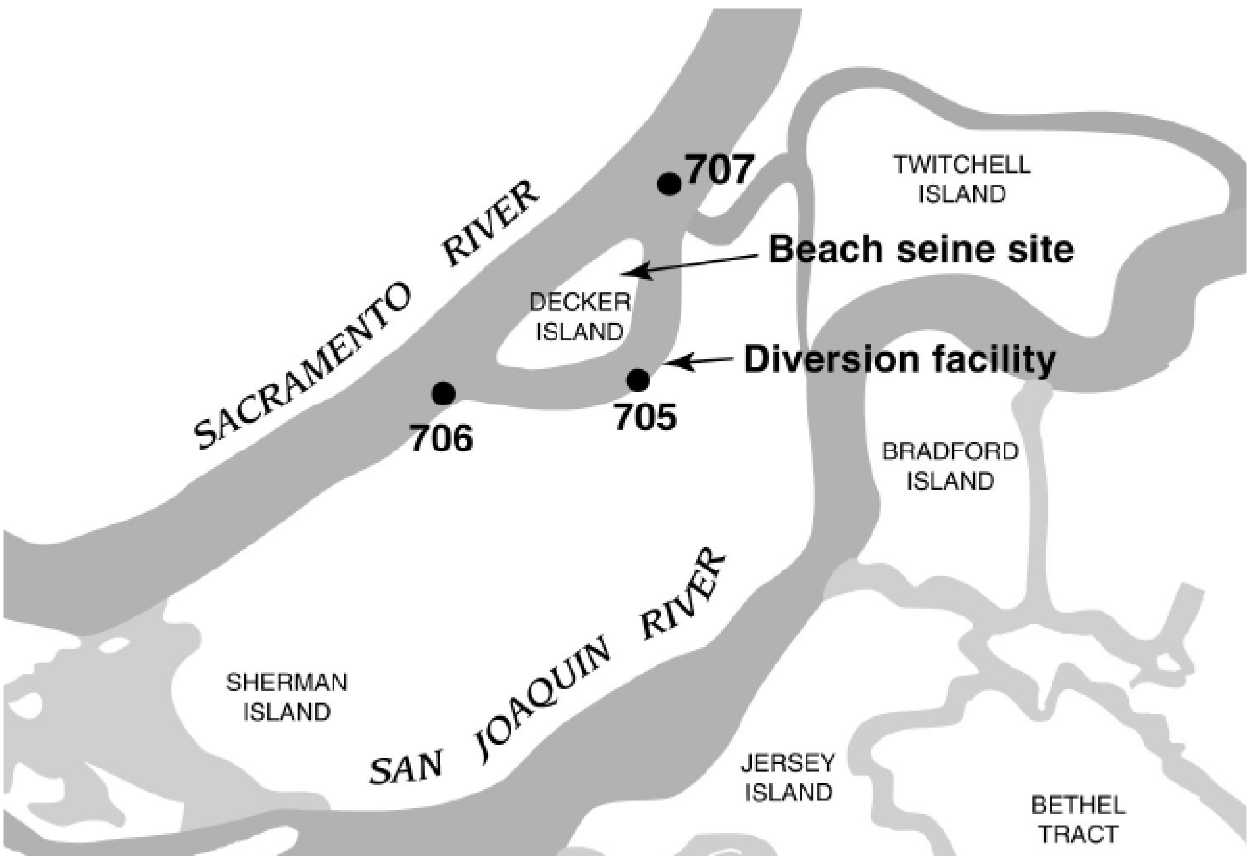


Figure 2. Water diversion sampled by Nobriga et al. 2004 that document entrainment in agricultural diversions.

The screened diversion in the Nobriga et al. (2004) study successfully excluded fish larger than 25 mm in length and reduced entrainment of threadfin shad by up to 450% compared to the unscreened diversion (Table 3). Although few delta smelt were sampled in this study, the results for ecologically similar species suggest that screens may reduce juvenile and adult delta smelt entrainment at some agricultural diversions.

Table 3. Numbers of threadfin shad, inland silverside, striped bass, and delta smelt collected, and their length ranges from screened and unscreened diversion samples in Horseshoe Bend, 12-14 July 2000 and 9-11 July 2001. (From Nobriga et al., 2004)

Year	Species	Screened	FL (mm)	Unscreened	FL (mm)
2000	Threadfin shad	1	19	59	13-59
	Inland silverside	0	--	0	--
	Striped bass	2	11-18	300	13-33
	Delta smelt	0	--	12	19-30
2001	Threadfin shad	17	10-22	7,824	9-42
	Inland silverside	0	--	160	15-37
	Striped bass	3	12-16	115	9-35
	Delta smelt	0	--	31	16-45

The California Department of Fish and Game’s (DFG) 1993 action plan for protection of salmonids emphasizes the importance of screening or installing fish protective devices on diversions. Likewise, the Ecosystem Restoration Program Plan (CALFED 1999) lists unscreened diversions as an important stressor on populations of salmonids and other fishes, and the plan indicated that elimination of unscreened diversions should be a high priority action.

Three sections of the DFG Code require fish screens on diversions (Odenweller 1994). These sections were promulgated primarily to protect salmonids. NOAA Fisheries and USFWS often require screening to protect fish species listed as threatened or endangered under the ESA, the Federal Power Act, and the Fish and Wildlife Coordination Act. A major justification for screening under the ESA is that any removal of individuals of federally listed species by a diversion constitutes “take” that is prohibited by the ESA, even if there is no demonstrable effect on the species at the population level.

Effects of In-Delta Power Plant Diversions

Two power plants operate diversions for once-through cooling systems in the Delta. The power plants, owned by Mirant Delta LLC, are located near Pittsburg and Contra Costa, near post-larval and juvenile delta smelt habitat. The power plants have the capacity to divert 1,4600 cubic feet per second for cooling purposes. Actual diversions are often significantly lower.

In a 1979 study commissioned by DFG and Pacific Gas and Electric, consultants estimated total average annual entrainment of smelt (longfin and delta) at 86 million (California Department of Water Resources 2005a). Given the current reduced abundance of both smelt species and changes in power plant operations, entrainment numbers today are likely much lower (Lund, et al. 2007).

Delta Smelt Habitat

Delta smelt (*Hypomesus transpacificus)* is a physically small fish species, with most adults in the range of 50 to 80 mm in total length, that is endemic to the Sacramento-San Joaquin Delta (hereinafter "Delta"). Delta smelt is predominantly an annual species, but a fraction, less than 10 percent, of the individuals that survive to maturity do live a second year.

Delta smelt have a stage-structured life history and their different life-cycle stages have very distinct habitat requirements. As with virtually all fishes, nonsenescence-related risk of death decreases with increased size; that is, as an individual increases in size, risk of predation is decreased and ability to withstand food shortages or suboptimal environmental conditions is increased. There are, however, occasions of high mortality in the life cycle that are not linked to the fish's specific size. Periods of morphological or behavioral transitions are typically highly vulnerable periods during an individual fish's life cycle. Accordingly, habitat specific conservation actions need to address the entire delta smelt life cycle. Each stage has its own combination of physical and biological requirements that must be met in order to ensure population survival through that stage.

Delta smelt are found in the Delta primarily below Isleton on the Sacramento River, below Mossdale on the San Joaquin River, and in Suisun Bay. They move into freshwater when spawning (ranging from February through June) and can occur in: (1) the Sacramento River as high as Sacramento, (2) the Mokelumne River system, (3) the Cache Slough region, (4) the Delta, (5) Montezuma Slough, (6) Suisun Bay, (7) Suisun Marsh, (8) Carquinez Strait, (9) Napa River, and (10) San Pablo Bay. Since 1982, surveys indicate that the center of delta smelt abundance has been the northwestern Delta in the channel of the Sacramento River. In any month, two or more life stages of delta smelt may be present in Suisun Bay (DWR and Reclamation 1994, Moyle 1976, Wang 1991).

Three abiotic factors – salinity, temperature, and water clarity – are indicators of delta smelt habitat suitability (Nobriga et al. 2008). Delta smelt are dependent on areas of freshwater or very low salinity for reproduction and early growth. Adult and juvenile smelt are restricted to waters low in salinity (0.5 to 6 practical salinity units (psu)), also referred to as the low salinity zone (LSZ). Delta smelt cannot tolerate salinities above about 19 psu or temperatures above about 25^{o}C, and more than 90 percent of sampled or observed individuals have been found in circumstances with salinities below 9 psu and temperatures below 22°C (Bennett 2005). Furthermore, increased water clarity may constrict delta smelt habitat quality by diminishing feeding success and increasing predation pressures (Baskerville-Bridges et al. 2004, Nobriga et al. 2008). Wright and Schoellhamer (2004) report that sediment transport to the estuary from the Sacramento River has declined by 50% since 1957. Other contributors to increased water clarity in the Delta include the proliferation of aquatic vegetation (i.e., freshwater marcophytes) (Nobriga et al. 2008).

Another indicator of habitat suitability, which may play an important role in age-class strength for the delta smelt along with the abiotic habitat factors referenced above, is the relative density of potential prey. Delta smelt are planktivores, generally feeding on copepods and other small zooplankton; early exogenous feeding includes rotifers and other very small-bodied microorganisms. Much of the Delta has experienced a decrease in densities of delta smelt prey organisms, including the calanoid copepods *Eurytemora affinis* and *Pseudodiaptomus forbesi* (Bennett 2005, Bennett and Moyle 1996). Historically delta smelt fed primarily on *E. affinis* in the spring and summer (Nobriga 2002). Prey density of *E. affinis* is associated with higher recruitment rates in striped bass *Morone saxitilis* (Tsai 1991). In the northern and western Delta and Suisun Bay, chlorophyll-a concentrations, used as a proxy for phytoplankton biomass, are characterized by declines throughout the 1970s-80s and a step change in 1987-1988 (Kimmerer 2004) (Figure 3). Phytoplankton production decreased 43 percent from 1975 to 1995 in Suisun Bay (Jassby and Cloern 2000).

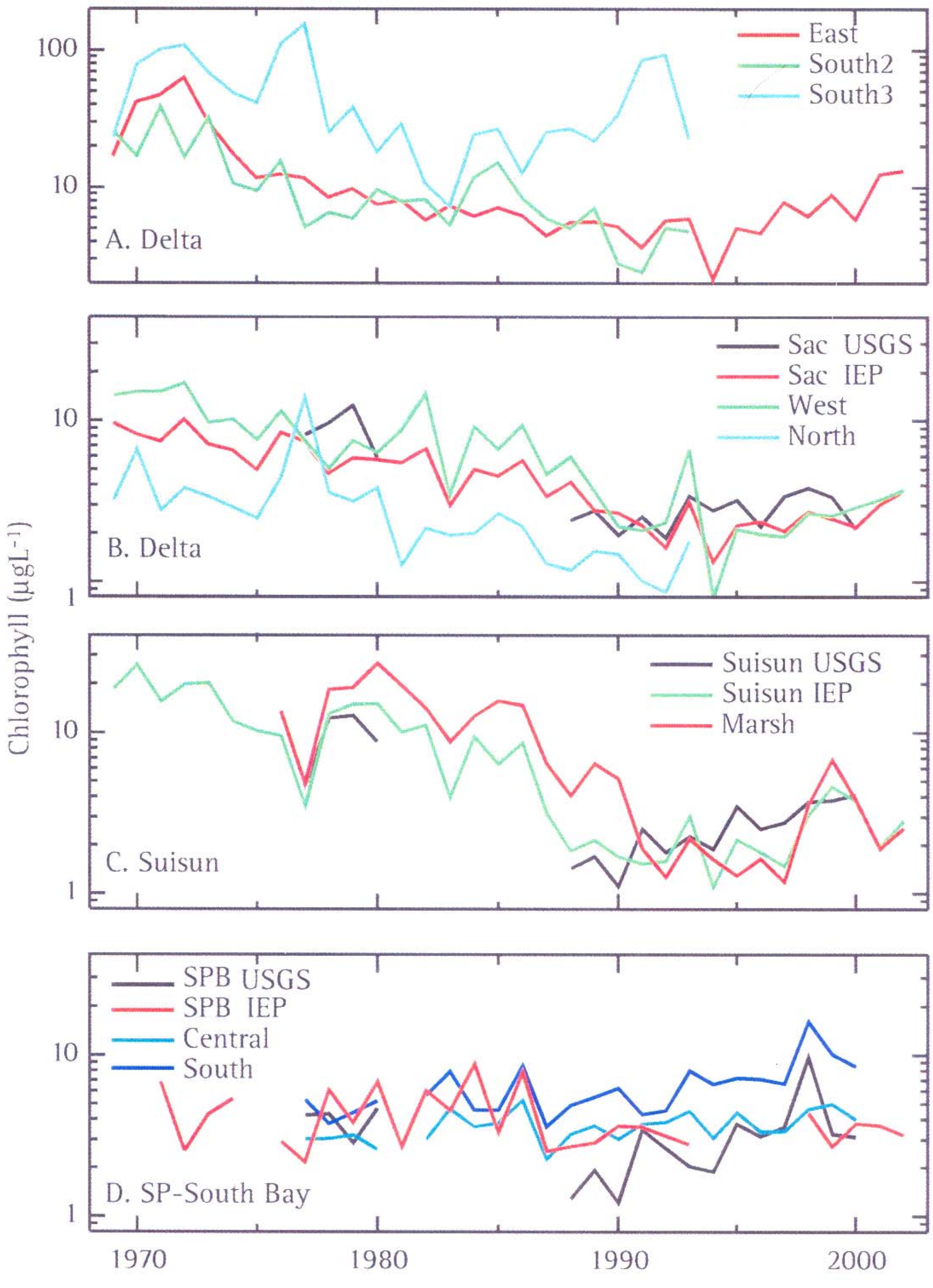


Figure 3. Time course of annual mean of monthly mean chlorophyll concentration for March-October from IEP and USGS monitoring programs, and for different regions of the estuary. The top two panels refer to regions of the Delta identified by Jassby et al. (2002): A. Two regions of the southern Delta and the eastern Delta; B. The lower Sacramento River (below Rio Vista), the western Delta, and the northern Delta (at Hood on the Sacramento River); C. Suisun Bay and channels of Suisun Marsh; D. San Pablo, Central, and South bays.

Although most of the date were from stations sampled consistently, the monitoring programs have had numerous additions and deletions of stations throughout the time period.

After more than a century of land use changes in the Delta, there is now a general lack of shallow-water habitats for delta smelt and their associated food web organisms. In fact, over the past 150 years, 95 percent of the estuary's historic wetlands have been removed by marsh reclamation for agricultural, municipal, and industrial uses (Sommer et al. 2007). Furthermore, decreases in and diminution of suitable spawning and rearing habitat have limited the carrying capacity of the Delta for delta smelt. With a limited habitat range, a 1-year life cycle, low fecundity, and planktonic larvae, delta smelt are unusually sensitive to changes in estuarine conditions. (Moyle et al. 1992).

Habitat that produces higher densities of delta smelt prey organisms could improve the likelihood of first feeding and stock recruitment. In the northern Delta, floodplain habitats have been shown to produce food web organisms at levels up to an order of magnitude higher than channel habitats (Sommer et al. 2004).

Stage-specific Delta Smelt Habitat Requirements and Limitations

Eggs and Yolk-sac Larvae

Spawning occurs seasonally in areas of freshwater or in areas with very low salinity, including the tidal portions of rivers, creeks, sloughs, and backwaters. Water temperature is a key stimulus for spawning, with spawning occurring in the spring when the water temperature is within the range of 15 degrees to 20 degrees Celsius. The precise location of spawning is not well documented, but it is very likely that spawning occurs in shallow water close to the shoreline, making those landscape conditions essential to delta smelt survival and recovery. Spawning in tidally-influenced shallow water is typically linked to the phases of the moon (Moyle and Cech 1996). It is very likely that delta smelt eggs are somewhat tolerant of changing water depths; however, delta smelt eggs probably need to remain moist, if not completely submerged. As the sites where delta smelt spawn are tidally influenced, coordinating spawning with the monthly tidal highs and lows helps eggs deposited in very shallow water avoid desiccation and risk of predation. The adhesive eggs apparently are preferentially scattered over non-vegetated substrata and hatch after 9-14 days, depending on water temperature and possibly the amount of sunlight reaching the eggs (Bennett 2005). Eggs may be laid over an extended period from February through June (see Figure 4; also Brown and Kimmerer 2001) and as physical and biological conditions change through the season, it is likely that cohorts of individuals derived from eggs laid at different times of the season experience dramatically different rates of survival.

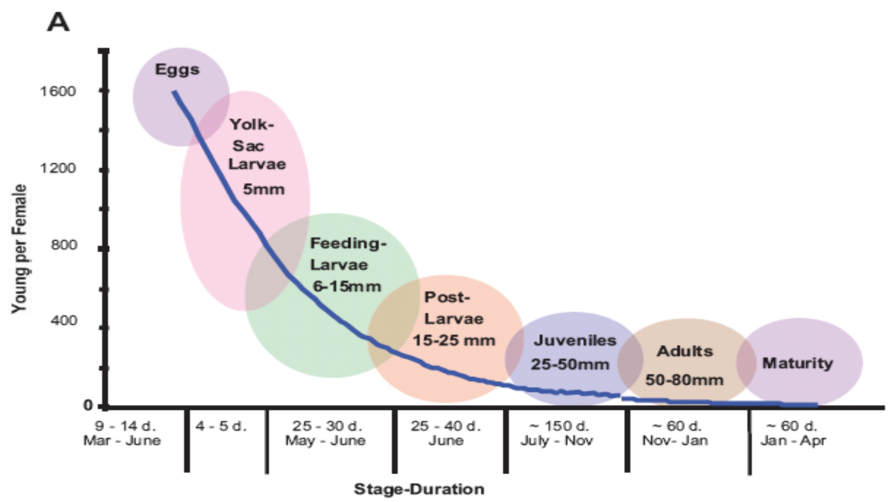


Figure 4. The delta smelt life cycle (Bennet 2005)

The geographic location of delta smelt spawning is variable and dependent on the hydrologic conditions of the particular year. In comparatively dry years, spawning mostly occurs in the north Delta region, where inflows create a zone of low salinity and fresher waters. In comparatively wet years, the low salinity conditions required for spawning are found spread throughout much of the Delta and extending west to at least the Napa River. Yolk-sac larvae generally do not depend on extrinsic food sources and have limited dispersal capabilities; therefore, for 4 to 5 days these larvae are essentially dependent on the physical conditions of their spawning site for survival; if conditions decline in suitability, larvae are not likely able to move to more hospitable habitat and may perish. Increases in salinity during this time or large changes in temperature can result in high rates of mortality.

Landscape areas suitable for delta smelt spawning have diminished greatly over the past century, with well-documented human actions reducing such habitat in large portions of Suisun Bay and shallow water areas where the Napa River and Cache Creek meet the Delta.

Free-feeding Larvae

Delta smelt use endogenous energy supplies (the yolk sac) until they reach ~5 mm total length, about 4 or 5 days after hatching. The transition from that endogenous food supply to dependence on exogenous food is an exceedingly vulnerable period for many species of fishes. During this critical period, an inability to find, capture, and process sufficient amounts of suitable prey rapidly results in starvation. For many species, in many years, age-class strength is determined by survival through this transition period. Motility is typically fairly limited during his phase and it is doubtful that larvae venture far from the spawning sites, either to find suitable prey or physical conditions. Primary prey items during this first phase of exogenous feeding are subadult copepods (Bennett 2005). The limited motility of delta smelt free-feeding larvae means feeding success depends on practically bumping into prey items. This strongly suggests that feeding success is dependent on prey densities (Nobriga 2002).

Because developing larvae are restricted in their abilities to seek out the physical conditions upon which they depend for survival, conditions during spawning must provide stable and suitable conditions for the duration of this life cycle phase – if conditions change too drastically, either too rapidly or too great a magnitude, the larvae will not survive.

Post-larval Stages

After morphogenesis is complete at ~15 to 20 mm smelt total length, post-larval individuals have increased functional swimming and foraging capabilities. With increased body size, delta smelt become more able to withstand brief periods of limited food and inhospitable water conditions and can move to more suitable habitat areas. Delta smelt in this stage can swim and track environmental features; they are known to exhibit diel migrations, traveling from surface waters during the day to deeper waters at night. As they mature, the post-larval smelt move to areas subject to greater marine influences, although they remain in the shifting area of the LSZ (Bennett *et al*. 2002); post-larval smelt are typically fully resident in the LSZ by July in most years (Bennett et al. 2002).

Individuals in the post-larval stage require several distinct habitat situations; the mostly freshwater shallows where the eggs were laid, the more saline and open LSZ, and areas of intermediate conditions. Post-larvae are much better at swimming than the earlier life cycle stages, but they cannot be considered strong swimmers and entrainment at water diversion facilities may cause mortality.

Juvenile Stage

As post-larvae grow and further mature physically, they become classified as juveniles at ~25 mm in total length. Although much more widely dispersed than post-larvae, juvenile delta smelt remain in the LSZ zone through the dry season, which extends from mid-summer through fall. During this period, the juveniles need to double in size in order to transition into adults. Growth rates may be food limited during this stage; food limitation can extend the duration that an individual spends in this stage. Bennett (2005) found that approximately 60% of juveniles examined exhibited characteristics of growth impairments during the juvenile stage, which can be attributed to poor feeding success.

Adult Stage

When total body length reaches ~50 mm and fall shifts to winter, juveniles transition into adults, reproductive organs mature, and fish move towards freshwater spawning locations. Prior to spawning, as with juveniles, adult delta smelt stay in the LSZ and feed on formerly abundant zooplankton. With winter influxes of freshwater, adult delta smelt are queued to move into shoreline and shallow circumstances where spawning will occur. Individuals can spawn repeatedly through the winter into early spring. Most adults die after one year, but a small proportion of the population survives to reproduce a second season. This group of individuals surviving into a second year has potentially significant implications in regard to delta smelt population levels. Egg production in fishes is inevitably linked with body size. Two year-old female delta smelt are on average larger than one year-old females; these year-two females produce greater numbers of eggs than year-one females – and more importantly produce more eggs earlier in the spawning season (Bennett 2005).

Habitat Loss

Historic losses of shallow freshwater and tidal marsh habitat areas have been extensive; substantial efforts to rehabilitate them are mandatory for successful recovery of delta smelt. It is generally appreciated that anthropogenic changes in the Delta have reduced suitable habitat for estuarine species (Nichols et al. 1986; Bennett and Moyle 1996). Over the past 150 years, 95 percent of the estuary’s historic wetlands have been removed by marsh reclamation for agricultural, municipal, and industrial uses (Sommer et al. 2007). Invasions by the alien clams *Corbula amurensis* and *Corbicula fluminea* have put stress on available habitats because of their prolific population growth and capacity for filtering the water column of delta smelt food resources and food web organisms (Bennett 2005, Carlton et al. 1990). Until recently, these clams were most abundant in the Suisun Bay and south Delta regions, but now occur throughout the western Delta.

Historical sampling for delta smelt indicates the species was several fold more abundant in northern Suisun Bay and Suisun Marsh channels than in areas of southern Suisun Bay and Delta, with the highest catches consistently occurring near Sherman Island and Decker Island in the lower Sacramento River (Bennett et al. 2002, Bennett 2005).

Jassby and Cloern (2000) identified floodplain restoration and creation of more inundated habitat as the most effective approaches to improve food availability for aquatic organisms. The Delta Smelt Action Plan (2005) states that increased production of zooplankton for delta smelt in all life stages is critical to their survival.

Aquatic Invasive Species in the Delta: Effects on Delta Smelt

Invasive exotic species pose a significant threat to the continued existence of the delta smelt. The Sacramento-San Joaquin Delta is considered one of the most “invaded” estuaries in the world (Cohen and Carlton 1998). At the same time, a substantial portion of the area’s native plant and animal diversity has declined, and a number of native species are threatened with extinction. The addition of non-native species and the loss of native species have been so extensive that restoration of the Bay-Delta ecosystem to a pre-settlement condition is not possible.

Non-native species threaten delta smelt in numerous ways, both directly and indirectly. Some believe predation of delta smelt by non-native species, most importantly striped bass and largemouth bass, poses a significant direct threat to that species. Those two predatory species annually consume a notable fraction of the dwindling delta smelt population.

It should be noted that predation of delta smelt by non-native species is undoubtedly not limited to the two aforementioned species. Inland silversides (*Menidia beryllina*) have been implicated in preying on newly free-swimming delta smelt (IEP minutes October 1997). Predation on such a vulnerable life history stage can readily impact recruitment of the prey species. Other non-native species of fishes, including a number of species of sunfish and catfish, seasonally overlap with delta smelt in areas of low salinity, in areas where delta smelt reproduce. While few species are as completely piscivorous as striped bass and largemouth bass, nearly all invasive exotic fishes will devour eggs and non-motile fry. Since these non-native species are exceedingly well naturalized in the Delta, elimination is virtually possible, and control will be difficult.

Predation on delta smelt by non-native invertebrate species is also likely. Predatory copepods have been noted to feed on larval fishes in other systems (Garcia and Alejandre 1995). Given there has been a nearly complete turnover in the copepods in the Bay-Delta system, it is possible that some of the exotic species pose a threat to small delta smelt.

Since delta smelt is essentially a broadcast spawner, no nest is constructed or defended, there is likely little direct competition for nest sites between delta smelt and other species. Likewise, since juvenile and adult delta smelt are found relatively high in the water column, feeding on relatively dispersed and undefendable prey – it is unlikely that significant direct interference competition between delta smelt and other fishes occurs.

Indirect impacts on delta smelt by exotic species are very problematic, very significant, and manifest in two main categories: food availability and spawning habitat alteration. A seminal workshop on delta smelt, its status, trends, and conservation challenges, identified food availability as potentially having “major implications” for the species’ recovery. It has subsequently become increasing clear that declines in the abundance of the primary food sources used by delta smelt have co-occurred with recent declines in the smelt abundance. Those declines in food availability may be attributable to the expansion of a wide variety of non-native fishes that that have been implicated as likely primary competitors for those resources and also due to changes in the composition and abundance of plankton

The one invasive species that may have had the most dramatic impact on the smelt over the past decade is the overbite clam (*Corbula [Potamocorbula] amurensis*). This species of Asian origin rapidly invaded the region, often reaching stultifying densities – many thousands per square meter. Presumably the clam arrived in the 1980s in ship ballast water, and subsequently spread from the brackish waters of the lower Delta into less saline up-stream circumstances. In Suisun Bay and elsewhere in the Bay-Delta system it may be severely depressing the phytoplankton species that provide the foundation of the food web supporting delta smelt. Overbite clams can filter huge amounts of river water, taking from it algae, zooplankton, and other minute organisms upon which juvenile smelt and other pelagic fishes feed. So prodigious is the feeding capacity of overbite clams that they are able to daily filter up to a dozen times the water column present above them – meaning that in areas where the seabed is virtually all covered with these invasive clams, all the water in the area passes through a clam every two hours. Given this unprecedented rate of filtration, it is not surprising that the entire food web has been altered. The decline of all plankton-feeding pelagic fishes in the Delta is tied to a dramatic shift in the food web; wherein most energy and carbon in the system once flowed through plankton and fishes, they now flow through the overbite clam. Even if not taken to the logical extreme, the filter-feeding overbite clam feeds on a number of the same plankton species that serve as key forage for delta smelt and other at-risk pelagic fishes. The overbite clam is a threat to the longterm persistence of delta smelt.

Additionally even in areas where the overbite clam has not become dominant, IEP monitoring data show declines in several native zooplankton species that serve as important food sources for both longfin smelt and delta smelt; two calanoid copepods, *Eurytemora affinis* and

Pseudodiapomus forbesi, have suffered particularly steep declines concomitant with increases of a non-native copepod, *Limnosthona tetraspina*, which apparently does not serve as food for larger juvenile delta smelt, and is a predator on native copepods that smelt prefer. It appears that non-native copepod species have replaced, either via competitive exclusion or predation, the majority of the native copepod community. Unfortunately this replacement has not been a suitable replacement in terms of delta smelt – the new non-native species are for a number of reasons not utilized by delta smelt as a food source in the same ways as are native copepods. The newest reported research on food selection by delta smelt shows that fish size (age) and food size are tightly linked. Smaller smelt feed on small copepods, and larger smelt on larger copepods. Both small native copepods and the small non-native copepod, *Limnosthona*, which may be replacing the native species in the Delta ecosystem, are eaten by the smallest juvenile smelt. However, at larger sizes, smelt appear to require the larger native copepod that the small non-native species may be replacing. While the smaller juvenile stages of delta smelt may be relatively unaffected by invasive copepods, larger smelt may be greatly impacted.

Another exotic organism that has recently invaded the Sacramento and San Joaquin system, including the Delta, which is impacting the food web, is the cyanobacterium, *Micocystis aeruginosa.* The invasion of this species into the Bay-Delta has immense implications for potential impacts on smelt, as well as serious potential to affect human health. Algal blooms of this species have been shown to negatively impact fish populations in estuaries subject to eutrophication. First observed in the Delta system in 1999, this freshwater blue-green algae species is abundant during the months from July to November. Its biomass reaches greatest levels in shallow freshwater areas in the central Delta that are critical to pelagic fish production; the toxicity of *Microcystis* has been recorded as highest in portions of the lower Sacramento River. The algae produces toxins that have been found to enter the Delta food web, and are expected to bioaccumulate throughout the food web. To date histological studies do not show *Microcystis* toxins in pelagic fishes in the Delta, or that harmful algal blooms of this or other species are associated with recent pelagic fish declines; but a recent POD Panel Review described *Microcystis* as “an additional symptomatic response to environmental stressors associated with human uses within the watershed,” warranting focused studies to identify potential impacts on delta smelt and its food resources.

While not completely understood, competition for food with the exotic inland silverside has been suggested by some as a possible contributing factor to the observed decline in delta smelt. This exotic species now occurs in large numbers in shallow water habitats with Delta smelt. The rapid expansion in range and abundance of inland silversides in the Delta since its introduction in the middle 1970s coincides with declines in the abundance of delta smelt. Inland silversides increased dramatically in population size in the decade preceding the Delta’s pelagic fishery crash; and, similar to circumstances elsewhere after the introduction of silversides, native fish populations suffer declines. Bennett (2005) describes inland silversides as a near perfect candidate for “inter-guild” competition, reflecting great similarities in morphology and ecology of the two fish species.

Non-native pathogens have resulted in the complete or regional extinction of numerous species. If delta smelt has been spared the direct impacts from non-native pathogens, they still are very likely being indirectly impacted, and some species in the food web of the Bay-Delta have been eliminated or reduced by due to non-native pathogens.

Other indirect impacts of exotic species on delta smelt associated with invasive exotic species include alteration of physical conditions of habitats required by smelt. Invasive species of plants, in particular, the Brazilian waterweed (*Egeria densa*) may be altering large tracts of the shallow waters of the Delta. These plant species typically expand and contract in abundance according to season, with highest densities occurring during the warmer parts of the year. These plants have the effect of actually filling many areas of shallow waters and by altering the turbidity and temperature of the remaining open water. Both of these impacts, choking of shallow waters and changing the conditions of the remaining water, are potentially very problematic for smelt reproduction. These non-native plant species may also alter the habitat to favor other non-native species, including those that prey on or compete with delta smelt.

Non-Natvie, Recreational Fishery Predation Effects on Delta Smelt

Introduced recreational fisheries have contributed to the decline of native fishes (Brown and Moyle 2005, Calamusso et al. 2005, Mueller et al. 2005). Though predation is not considered one of the principal drivers of delta smelt population decline (U.S. Dept. of the Interior 1996), it is one of many factors that may be suppressing the population and impeding its recovery (IEP 2008; U.S. Dept. of the Interior 1996). Striped bass, a predator of the delta smelt, are also considered a major impediment to shallow-water habitat restoration in the Delta (Brown 2003) and studies on the East Coast have recently demonstrated that adult striped bass populations released from angler harvest may cause substantial decline among prey species (Hartman 2003; Uphoff 2003).

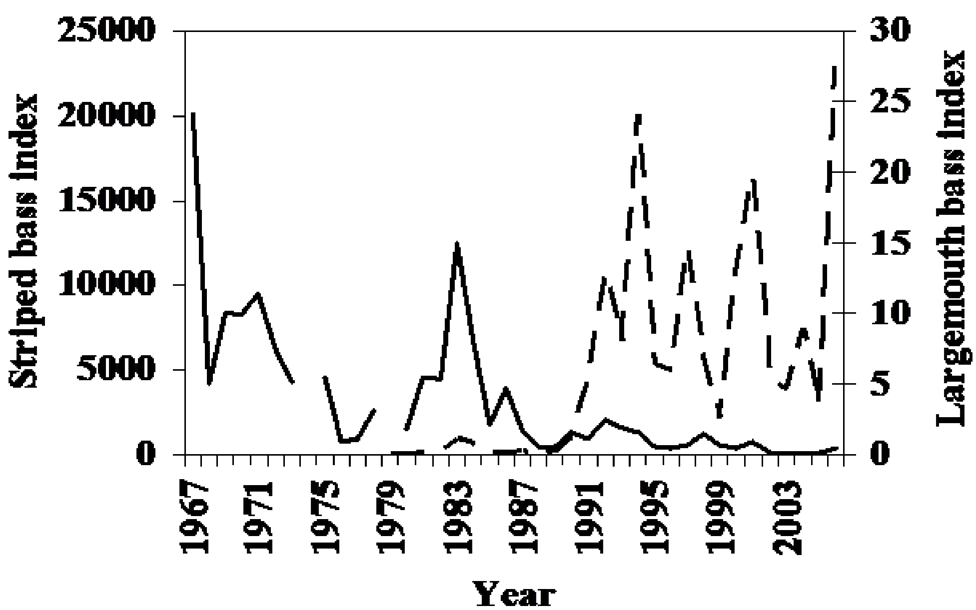
Striped bass, *Morone saxatilis*, and another predator of the delta smelt, the largemouth bass, *Micropterus salmoides*, were both intentionally introduced and have successfully naturalized in the Delta and its watershed (Moyle 2002). Although striped bass have successfully coexisted with delta smelt for most of their history, the Delta ecosystem has undergone radical changes in more recent years (see IEP 2008), which has altered this relationship.

Over the past several decades, striped bass and largemouth bass have exhibited divergent trends in abundance in the Delta (Figure 1). Age-0 striped bass abundance has declined. However, sampling from pelagic habitats shows that young piscivorous striped bass have apparently declined more slowly and adult striped bass abundance has increased somewhat since the 1990s, so elevated striped bass predation pressure on smaller pelagic fishes in recent years is a likely outcome (IEP 2008). At the same time that Age-0 striped bass abundance has been declining, largemouth bass abundance has increased. In fact, within the last decade an exceptionally popular largemouth bass sport fishery has developed, and the Delta has become a destination for national bass fishing tournaments.

Striped Bass

Striped bass were introduced in the late 1800s, and are now the most broadly distributed and abundant large, piscivorous fish in the Delta (Nobriga & Feyrer 2007, and Figure 5). Adult striped bass generally move into freshwater in the fall, spend winter in the Delta, and migrate up Central Valley tributaries in the spring. They spawn in freshwater before moving back to salt water bays (Moyle 2002). However, recent strontium tracer studies indicate that a large number of adult striped bass may never migrate to salt water bays (Ostrach 2008). Instead, they may congregate year-round in areas of higher prey densities, such as water diversion facilities, and salvage release points.

Striped bass are typically found in turbid, open-water habitats, which also support special-status native fishes such as delta smelt, Chinook salmon (*Oncorhyncus tschawytscha*), and splittail (*Pogonichthys macrolepidotus*). Foraging striped bass tend to form and move in groups, which apparently aids them in locating and capturing prey. Striped bass consume a wide variety of invertebrates and fishes (Schaefer 1970, Rulifson and McKenna 1987), including both salmonids (Lindley et al. 2007) and delta smelt (DFG 1999). Various studies in the Delta suggest that striped bass are adept at finding and feeding upon concentrations of outmigrating juvenile salmon (DFG 1999, EBMUD unpublished). Elevated predation rates on outmigrating salmonids appear to be particularly problematic at artificial structures (Warner and Kynard 1986, Liston et.al.1994, DFG 1999).



Striped bass abundance indices are based on the DFG Fall Midwater Trawl, 1967-2006 (http://www.delta.dfg.ca.gov/data/mwt/). Largemouth bass abundance indices are summed daily salvage acre feet^{-1} of water diverted by the SWP and CVP in the Sacramento-San Joaquin Delta, 1979-2006.

Figure 5. Relative abundance trends for age-0 striped bass (solid line) and largemouth bass (dashed line) in the SSJD. The indexes shown for these two species are not comparable since they are determined using different methods. However this figure shows the relative increase and decline for each species.

Historically, striped bass recruitment was best when freshwater inflows to the estuary were high (Stevens et al. 1985). The freshwater flow effect is still apparent on the survival of striped bass larvae (Kimmerer 2002), but it is no longer discernable by the end of the first year of life (Sommer et al. 2007). Based on extensive analyses of multi-decade datasets, Kimmerer et al. (2000) concluded the relative abundance of cohorts recruiting to the fishery is set somewhere between the first summer of life and recruitment in year three.

Despite the recent decline in age-zero population abundance, striped bass may contribute to mortality for delta smelt. Analysis of age-zero striped bass data indicates that individuals are substantially larger in the fall compared with 40 years ago (Figure 6). The larger size of age-zero striped bass allows for earlier piscivory relative to other Delta predatory fishes (Nobriga 2008, in press). Among age-one striped bass, however, piscivory is more related to season than to size of striped bass (Nobriga & Feyrer 2007). Striped bass are typically most piscivorous during summer and fall regardless of size, consistent with the findings of Stevens (1966) who showed intra-annual waxing and waning of striped bass piscivory superimposed on the gradual increase in piscivory between young-of-year and adults. The apparent association of striped bass piscivory with the seasonal cycle of juvenile fish production has significant implications for understanding and managing predatory impact on native fish species including delta smelt.

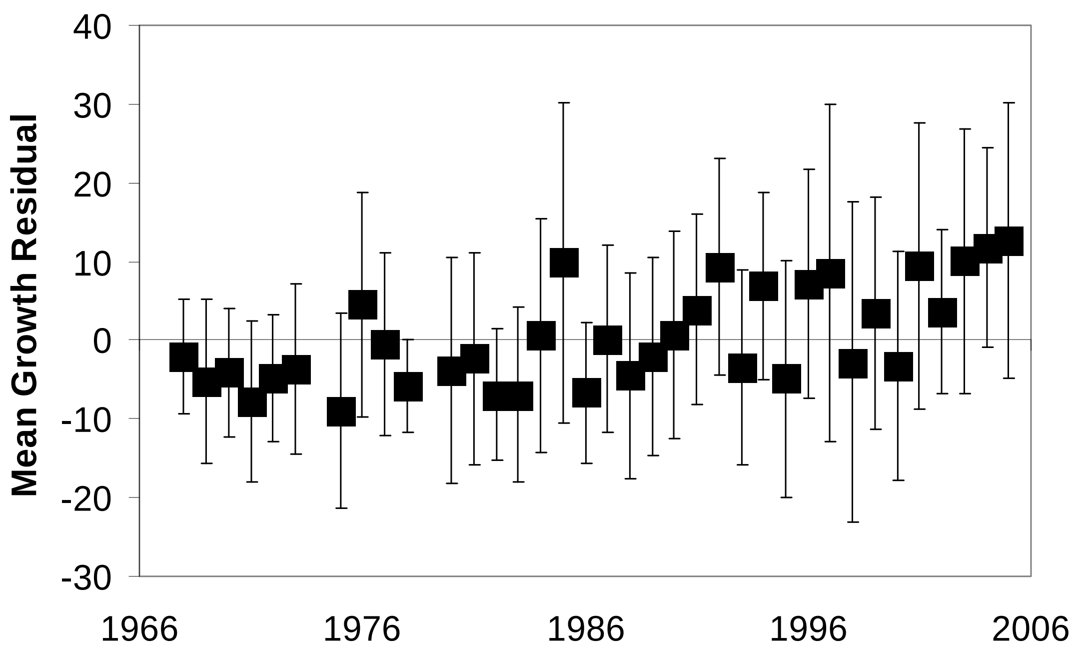


Figure 6. Plot of mean residual growth among age-0 striped bass captured by DFG sampling programs (specifics) in the fall of each. IEP unpublished.

Largemouth Bass

Largemouth bass were also introduced to the Delta in the late 1800s (Dill and Cordone 1997). They typically do not undergo long distance spawning migrations, so most largemouth bass inhabiting the Delta probably spawn there. Peak largemouth bass spawning occurs from May through July. Larval largemouth bass are guarded by an attending parent and thus do not typically disperse very far. Juvenile largemouth bass may be dispersed by tidal and river currents, but most remain closely associated with submerged vegetation along channel edges and in shallow portions of flooded agricultural tracts (Nobriga et al. 2005).

In contrast to the availability of extensive striped bass datasets, less is known about Delta largemouth bass population dynamics. Recruitment success was historically low (Moyle 2002), but abundance of adult largemouth bass in the Delta has increased in the last decade (Brown and Michniuk 2007) and now supports a significant sport fishery (Lee 2000). There is strong evidence that rapid increases in habitat have facilitated population growth (Nobriga et al. 2005, Brown and Michniuk 2007). Specifically, the rapid expansion of the aquatic weed *Egeria densa* has improved the carrying capacity for largemouth bass by providing structure and perhaps even by altering water clarity. Feyrer et al. (2007), for example, describes a long-term reduction in high turbidity habitat for striped bass via “filtration” of sediments by *E. densa*. While reduced turbidity may be harmful to some Delta fishes, it may improve foraging success among largemouth bass.

Fish biomass is higher in the Delta’s vegetation dominated shorelines than its open water shorelines (Nobriga et al. 2005). This suggests these habitats are productive relative to the open waters, and appear to have a fairly self-contained food web, decoupled from the open water habitats frequented by striped bass (Grimaldo 2004). It is likely that recent increases of largemouth bass are due to their integration into this vegetation-based food web, which could reduce their reliance on the estuary's depleted pelagic food web. Estimates of first-year largemouth bass growth in the Delta suggest growth is very slow (Schaffter 1998), probably as a result of intense competition among other co-occurring centrarchids (Nobriga 2008); thus, largemouth bass growth may be slowed by strong density-dependence, rather than from limited bottom-up productivity. In contrast, food limitation for striped bass is much more likely to be a ‘bottom-up’ phenomenon due to chronically low abundance of pelagic prey (Kimmerer et al. 2000, Feyrer et al. 2003, Sommer et al. 2007).

Largemouth bass appear to have the greatest impact on nearshore fishes, including native fishes (Nobriga & Feyrer 2007), which they consume well into summer months. Incidence of piscivory is predominantly a function of size (Figure 7). With largemouth bass becoming predominantly piscivorous at sizes smaller than those of the native predator, Sacramento pikeminnow (about 115 mm versus about 190 mm, respectively -- Nobriga & Feyrer 2007), largemouth bass could be considered as a significant potential predator of delta smelt.

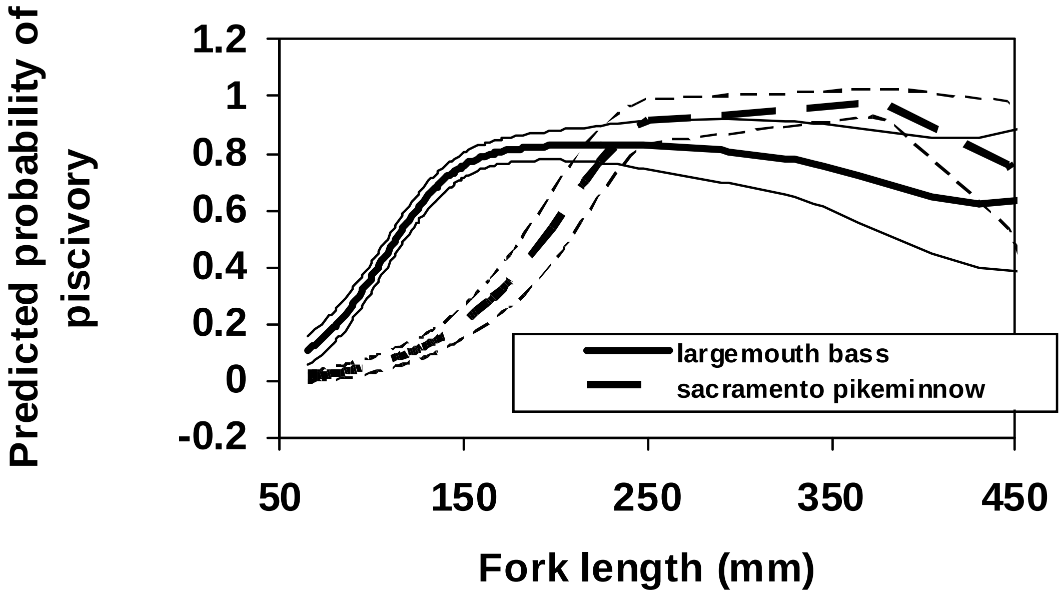


Figure 7. Relative incidence of piscivory among largemouth bass and Sacramento pikeminnow (Nobriga and Feyrer 2007).

Predator Effects

The relative effect of predation in the Delta is controlled through the interaction of factors influencing prey-encounter and capture probabilities. These probabilities are location and species specific. For example, some piscivores forage most efficiently in open water (unstructured) environments, whereas others forage most efficiently in structured environments (Greenberg et al. 1995, Buckel and Stoner 2000). In general, striped bass and Sacramento pikeminnow are examples of the former; largemouth bass is an example of the latter (Moyle 2002). In addition to location (habitat) and species specific effects, piscivorous fishes shift their diets as they grow and as prey availability changes (Buckel et al. 1999).

Accordingly the population-level impact of striped bass and largemouth bass predation on delta smelt is likely variable and difficult to document or characterize. For example, the impact of largemouth bass on pelagic native fishes and/or their pelagic early life stages may be mitigated by its limited use of open water and brackish habitats (Nobriga et al. 2005, Matern et al. 2002). However, the degree to which largemouth bass may opportunistically forage in open waters is unknown, but it almost certainly occurs when foraging conditions are favorable. Given largemouth bass abundance and wide distribution, even infrequent predation events could have dramatic demographic effects where largemouth bass consume locally concentrated spawning delta smelt. New acoustic tagging, sonar and videography techniques could be applied to better understand largemouth bass foraging behavior and potential for impact on listed fish species.

That said, striped bass likely remain the most significant predator of Chinook salmon (Lindley and Mohr 2003) and delta smelt (Stevens 1966), due to their wide distribution and tendency to aggregate around water diversion structures where other fishes may be entrained (Brown et al. 1996).

The Interagency Ecological Program, Pelagic Organisms Decline Progress Report (IEP 2008) stated that the striped bass and largemouth bass predation could have significant effects on delta smelt if:

> "…piscivorous fishes became more abundant relative to the POD fishes; pelagic fish distribution shifted to locations with higher predation risk (e.g. habitat changes); or the POD fishes became more vulnerable to predation as a consequence of their extremely low population size… or increases in water clarity. Predation-driven Allee effects can arise from diminished anti-predator behavior or increased predator swamping of individuals in smaller prey groups (Berec et al., 2006). They are most likely to occur with generalist predators in situations where predation is a major source of mortality, and predation refuges are limited (Gascoigne et al. 2004). In this situation individuals of depleted populations continue to be consumed even though they are at low density. More specialized predators often switch between abundant prey and consequently reduce consumption of rare prey species. As will be described below, the combination of a widely distributed pelagic piscivore (striped bass), an efficient littoral piscivore (largemouth bass), cumulative entrainment losses of multiple life stages, and decreased habitat suitability suggest the conditions listed by Gascoigne et al. (2004) could apply in the Delta."

Data indicate that striped bass and other predators may limit the potential for recovery of delta smelt. This potential has been acknowledged by the U.S. Fish and Wildlife Service (U.S. Dept. of the Interior 1996), the California Department of Fish and Game (DFG 1999), and most recently by the Interagency Ecological Program, Pelagic Organisms Decline evaluation team (IEP 2008).

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