U. S. DEPARTMENT OF THE INTERIOR
FISH AND WILDLIFE SERVICE

Species at Risk from Selenium Exposure in the San Francisco Estuary

Final Report to the U. S. Environmental Protection Agency

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Prepared by

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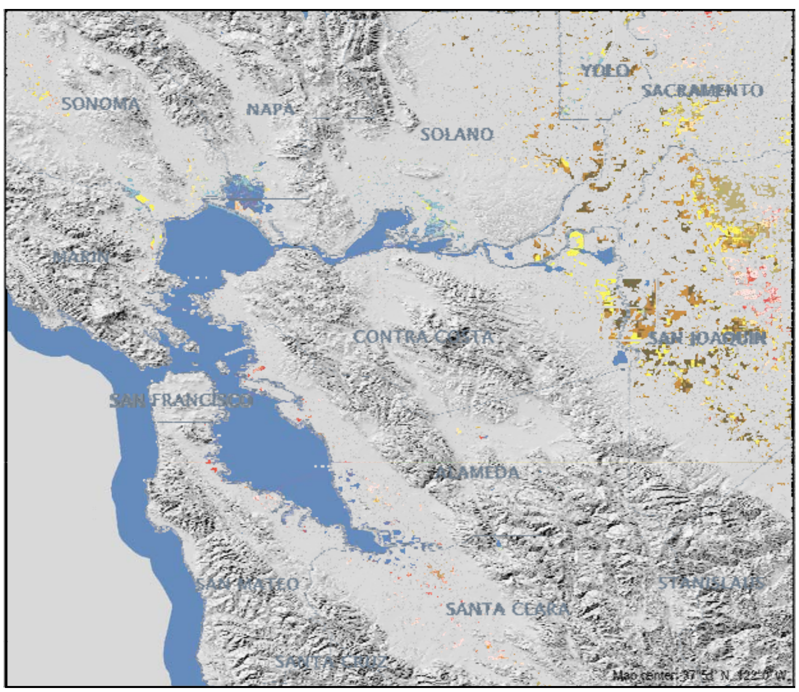

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Table of Contents

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INTRODUCTION

Discharges of selenium from Bay Area oil refineries and in the subsurface drainwater from Central Valley farms eventually reach the San Francisco Bay/Delta estuary, where uptake by food chains is particularly efficient (Presser and Luoma 2006). Selenium concentrated through food chains may have a variety of adverse effects including reduced growth and embryonic deformities in wildlife species at risk (Eisler 1985). As part of an effort to revise water quality criteria to be protective of federally listed species in California, the U. S. Environmental Protection Agency (USEPA) is engaged in an interagency project to determine selenium dietary exposure benchmarks that would be protective of wildlife, including aquatic-dependent wildlife, in the San Francisco Bay/Delta estuary. This document is the initial contribution of the U. S. Fish and Wildlife Service (USFWS) to this interagency project. The role of the USFWS in this initial task is to review and compile existing pertinent data (diet, body weight, food ingestion rate, natural history) on the species that are most likely to be at risk due to dietary selenium exposure in the San Francisco Bay/Delta.

SPECIES CONSIDERED

Species considered for evaluation of selenium exposure risk in the San Francisco Bay/Delta are listed in Table 1. To keep this list reasonably manageable but representative, it was subject to a preliminary screening process that eliminated plants, most invertebrates, and most vertebrates that are members of strictly or mainly terrestrial food chains. In compiling this list, species protected by federal legislation were given priority.

Table 1. List of species considered for evaluation of selenium exposure risk in the San Francisco Bay/Delta.

Common Name	Scientific Name	Federal Status	California State Status	Potential to be adversely affected by selenium in Bay/Delta*
Mammals				
salt marsh harvest mouse	*Reithrodontomys raviventris*	endangered	protected	As a terrestrial herbivorous mammal, unlikely to be among the most exposed and sensitive of wildlife species; therefore not likely to be a "species most at risk."
Birds				
American white pelican	*Pelecanus erythrorhynchos*	MBTA	concern	SF Bay is North end of West Coast distribution of non-breeders. Preys on some bottom-feeding fish as well as schooling fish, but not likely to be a "species most at risk.".
California brown pelican	*Pelecanus occidentalis californicus*	endangered	protected, endangered	SF Bay is North end of W Coast distribution. Feeds mainly on surface-schooling fish; therefore, not part of benthic-based food chain and not likely to be a "species most at risk."
white-faced ibis	*Plegadis chihi*	concern	concern	Breeds and winters in San Joaquin Valley. Inhabits mainly freshwater wetlands, but also estuarine wetlands. Eats aquatic and moist soil invertebrates. At some risk but not likely to be a "species most at risk."
double-crested cormorant	*Phalacrocorax auritus*	MBTA	concern	Winters in Central Valley and SF Bay/Delta. Feeds on bottom-dwelling fish and invertebrates as well as schooling fish. At some risk but not likely to be a "species most at risk."
American bittern	*Botaurus lentiginosus*	concern	none	Feeds mainly in freshwater marshes, eating mainly insects and small vertebrates; therefore not likely to be a "species most at risk."
western least bittern	*Ixobrychus exilis hesperis*	concern	concern	Breeds in SF Delta. Feeds in fresh and brackish water marshes, eating mainly small fish and insects; therefore not likely to be a "species most at risk.".
Aleutian Canada goose	*Branta canadensis leucopareia*	delisted, MBTA	none	Winters in California, feeding primarily in upland crops and fallow fields. Sensitive to selenium but unlikely to be exposed in estuary; therefore not likely to be a "species most at risk."
greater scaup	*Aythya marila*	MBTA	none	SF Bay is one of 2 major wintering areas on W coast of N America. Feeds on benthic mollusks that efficiently bioaccumulate selenium in the SF Bay/estuary, therefore likely to be a "species most at risk."
lesser scaup	*Aythya affinis*	MBTA	none	SF Bay is an important wintering area; feeds on clams; therefore likely to be a "species most at risk."
black scoter	*Melanitta nigra*	MBTA	none	Winters along California coast, diving mainly for mollusks; therefore likely to be a "species most at risk."
white-winged scoter	*Melanitta fusca*	MBTA	none	Winters along California coast and estuaries, diving mainly for mollusks; therefore likely to be a "species most at risk."

Common Name	Scientific Name	Federal Status	California State Status	Potential to be adversely affected by selenium in Bay/Delta*
surf scoter	*Melanitta perspicillata*	MBTA	none	Winters along California coast, diving mainly for mollusks; therefore likely to be a “species most at risk.”
osprey	*Pandion haliaetus*	MBTA	concern	High trophic level piscivore; not at risk overall and exposure well represented by bald eagle. Therefore not treated here as a “species most at risk.”
bald eagle	*Haliaeetus leucocephalus*	delisted, MBTA,BGEPA	protected, endangered	High trophic level piscivore; at risk overall and exposed to aquatic food chain in the SF Bay/Delta; therefore likely to be a “species most at risk.”
northern harrier	*Circus cyaneus*	MBTA	concern	High trophic level but less exposed to aquatic food chain than bald eagle; therefore not likely to be a “species most at risk.”
white-tailed kite	*Elanus leucurus*	concern	protected	Feeds mainly on terrestrial mammals; minimal exposure to aquatic selenium; therefore not likely to be a “species most at risk.”
American peregrine falcon	*Falco peregrinus anatum*	delisted, MBTA	protected, concern	Delisted but monitored for population status and contaminants. Exposed to selenium in aquatic food chain as predator on piscivorous birds, but exposure generally diluted by terrestrial component of diet; therefore not likely to be a “species most at risk.”
prairie falcon	*Falco mexicanus*	MBTA	concern	Winters along California coast; high trophic level but in mainly terrestrial food chain; therefore not likely to be a “species most at risk.”
California black rail	*Laterallus jamaicensis coturniculus*	MBTA	protected, concern	Inhabits tidal marsh in SF Bay estuary. Feeds on invertebrates, including snails, but also seeds; therefore not likely to be a “species most at risk.”
California clapper rail	*Rallus longirostris obsoletus*	endangered	protected, endangered	Subspecies endangered and endemic to SF estuary; feeds on benthic invertebrates, including filter-feeders that bioaccumulate selenium; therefore likely to be a “species most at risk.”
marbled murrelet	*Brachyramphus marmoratus*	threatened	endangered	Forages in bays along Pacific coast in summer, but not recorded in SF Bay/Delta. Dives for pelagic food: schooling fish and euphausiids (krill). Therefore not likely to be a “species most at risk.”
California least tern	*Sterna antillarum browni*	endangered	protected, endangered	Breeds primarily in Central San Francisco Bay but can nest throughout estuary. Feeds throughout estuary, mainly on surface fish, not part of the benthic mollusk-based food chain; therefore not likely to be a “species most at risk.”
black tern	*Chlidonias niger*	concern	concern	Breeds in C Valley including SF Delta. Feeds on marine and freshwater surface fish and insects; therefore not likely to be a “species most at risk.”
Caspian tern	*Sterna caspia*	MBTA	none	Preys heavily on juvenile salmonids, but not endangered overall; therefore not likely to be a “species most at risk.”
western snowy plover	*Charadrius alexandrinus*	threatened	concern	Terrestrial component of diet likely provides dietary dilution of aquatic system selenium exposures; have been shown to be very tolerant of selenium exposure; therefore not likely to be a “species most at risk.”

Common Name	Scientific Name	Federal Status	California State Status	Potential to be adversely affected by selenium in Bay/Delta*
mountain plover	*Charadrius montanus*	concern	concern	Winters in agricultural fields of Sacramento/San Joaquin Valley. Diet mainly terrestrial; therefore not likely to be a "species most at risk."
tricolored blackbird	*Agelaius tricolor*	concern	concern	Nests colonially, mainly in freshwater marshes. Feeds on terrestrial as well as freshwater insects; therefore not likely to be a "species most at risk."
Reptiles				
giant garter snake	*Thamnophis gigas*	threatened	threatened	Aquatic predator, but not known to inhabit the estuary; therefore not likely to be a "species most at risk" in the estuary.
Fish				
Chinook salmon	*Oncorhynchus tshawytscha*	endangered/ threatened	endangered/ threatened	Sensitive to selenium; most sensitive life stages occur in rivers and estuary; therefore likely to be a "species most at risk."
steelhead	*Oncorhynchus mykiss*	threatened	none (in Central Valley)	Sensitive to selenium; most sensitive life stages occur in rivers and estuary; therefore likely to be a "species most at risk."
delta smelt	*Hypomesus transpacificus*	threatened	threatened	Endemic to the Bay/Delta estuary. Feeds on zooplankton, not a pathway of greatest exposure, but threatened overall, so included as a "species most at risk."
longfin smelt	*Spirinchus thaleichthys*	concern	endangered	SF Bay/estuary is S end of distribution. Prefers more saline water than delta smelt. Overall less threatened and probably less exposed than delta smelt so adequately represented by that species. Therefore not treated here as a "species most at risk."
green sturgeon	*Acipenser medirostris*	threatened	concern; fishing prohibited	Threatened overall, and vulnerable to selenium as a clam-eating bottom feeder in the SF estuary; therefore likely to be a "species most at risk." Emergency regulations issued by CDFG March 2006--Zero (0) bag limit for green sturgeon year-round in all areas.
white sturgeon	*Acipenser transmontanus*	none	limited fishing	Population in the SF estuary not federally listed, but vulnerable to selenium as a clam-eating bottom feeder. Therefore, treated here as a "species most at risk." The daily bag and possession limit established by CDFG is one fish that must be between 46 inches and 72 inches total length. The yearly limit is three.
river lamprey	*Lampetra ayresi*	none	watch list	Anadromous; feeds on young salmon. Recorded from lower Sacramento and San Joaquin Rivers. Not federally listed; therefore not considered to be a "species most at risk."
Sacramento perch	*Archoplites interruptus*	concern	concern	Fry feed primarily on bottom-dwelling crustaceans, insect larvae, snails, and fish. One captured in the Delta in 1992, not likely to represent an established population there. Therefore not considered to be a "species most at risk" in the Delta.
Sacramento splittail	*Pogonichthys macrolepidotus*	concern	threatened	Vulnerable to selenium as clam-eating bottom feeder in the SF estuary; therefore likely to be a "species most at risk."

Common Name	Scientific Name	Federal Status	California State Status	Potential to be adversely affected by selenium in Bay/Delta*
striped bass	*Morone saxatilis*	none	none	Introduced sport fish in California. Population in Delta declined sharply in early 2000s, but species overall not threatened. Therefore not considered to be a “species most at risk.”
threadfin shad	*Dorosoma pretenense*	none	none	Introduced in California as food for game fish. Population in Delta declined sharply in early 2000s, but species overall not threatened. Therefore not considered to be a “species most at risk.”
tidewater goby	*Eucyclogobius newberryi*	endangered	endangered	Bottom-dwelling carnivore. Prefers semi-closed estuaries. Potentially exposed, but not found recently (since 1984) in the Bay area; therefore not considered to be a “species most at risk” in the SF Bay/Delta.
California halibut	*Paralichthys californicus*	none	none	Bottom dweller inhabiting the SF Bay, but overall not threatened; therefore not likely to be a “species most at risk.”
leopard shark	*Triakis semifasciata*	none	none	Bottom dweller inhabiting the SF Bay, but overall not threatened; therefore not likely to be a “species most at risk.”
starry flounder	*Platichthys stellatus*	none	none	Bottom dweller inhabiting the SF Bay. Population in bay declined sharply since 1980, but overall not threatened; therefore not likely to be a “species most at risk.”
Invertebrates				
Dungeness crab	*Cancer magister*	none	none	Estuary is nursery for this ocean-breeding bottom feeder, but overall not threatened; therefore not likely to be a “species most at risk.”

Federal Status: Endangered: listed as endangered under the Federal Endangered Species Act; Threatened: listed as threatened under the Federal Endangered Species Act; Proposed threatened: proposed as threatened under the Federal Endangered Species Act; Concern: designated a species of concern; Delisted: removed from the list of endangered and threatened species under the Federal ESA; MBTA: protected under Migratory Bird Treaty Act; BGEPA protected under the Bald and Golden Eagle Protection Act.

California State Status: Endangered: listed as endangered under the California Endangered Species Act; Threatened: listed as threatened under the California Endangered Species Act; Concern: designated by the California Department of Fish and Game as a species of concern**;** Protected: Fully protected under the Fish and Game Code of California predating the California Endangered Species Act

* Assessment based upon population status, dependence upon benthic food web, and sensitivity to selenium. Aquatic dependent species feeding directly in the benthic food web of the San Francisco Estuary were considered to be at greater risk to selenium exposure than those species feeding in a pelagic/planktonic food web. This assumption is based upon the work of Stewart et al. (2004).

Table 2. Species most at risk from selenium exposure in the San Francisco Estuary: summary data.

Common Name	Scientific Name	Probable critical life stage for selenium effects[1]	Food ingestion rate at critical life stage (g wet weight per day)[2]	Food ingestion rate at critical life stage (g dry weight per kg body weight per day)[3]	Body weight at critical life stage (g)[4]	Diet	Mainly clam-based food chain?[5]	Percent of diet that is clam-based (worst case)
bald eagle	*Haliaeetus leucocephalus*	Adult female (egg laying)	644	249	5275 (female)	fish, birds, mammals	no	22.8[6]
California clapper rail	*Rallus longirostris obsoletus*	Adult female (egg laying)	172	46.8	346	mussels, spiders, clams, crabs, snails, marsh cordgrass seeds	yes	64.1
greater scaup	*Aythya marila*	Adult male and female (migration)	313	85.8	1054 (male)	clams, snails, other mollusks, crustaceans, algae	yes	80.7
lesser scaup	*Aythya affinis*	Adult male and female (migration)	246	67.5	734 (male)	clams, other mollusks, aquatic insects, crustaceans, plants	yes	96
white-winged scoter	*Melanitta fusca*	Adult male and female (migration)	465	127.3	1917 (male)	clams, other mollusks, crustaceans, aquatic insects	yes	75[7]
surf scoter	*Melanitta perspicillata*	Adult male and female (migration)	314	86.0	1059 (male)	mussels, other mollusks, plants, crustaceans	yes	86[8]
black scoter	*Melanitta nigra*	Adult male and female (migration)	325	89.1	1117 (male)	mussels, clams, snails, barnacles	yes	80[9]

Chinook salmon	*Oncorhynchus tshawytscha*	Migrating/rearing juvenile		23.3	0.5-18	insects, crustacea, juvenile fish	no	0[10]
steelhead	*Oncorhynchus mykiss*	Migrating/rearing juvenile		19.9	31-105	insects, annelids, *Daphnia*	no	0[10]
green sturgeon	*Acipenser medirostris*	Juvenile or adult female		20	1300 (average caught)	benthic crustacea, mollusks and fish	probably substantially	See white sturgeon
white sturgeon	*Acipenser transmontanus*	Juvenile or adult female		15-20	6280 (mode)	benthic mollusks and crustacea	substantially	41.1[11]
delta smelt	*Hypomesus transpacificus*	Juvenile or adult female		114	0.32 (average Jun-Aug)	copepods, cladocerans, amphipods, insect larvae	no	0
Sacramento splittail	*Pogonichthys macrolepidotus*	Juvenile or adult female		33.7	121 (mode)	benthic detritus, clams, other mollusks, mysids	substantially	34

1 For most species it is premature and speculative to designate a critical life stage at this time. Such designation prejudges the outcome of a thorough search of the toxicology literature.
2 Food ingestion rates based on wet weight can be calculated from available parameters (Nagy 2001) for birds, mammals, reptiles and amphibians, but not, in general for fish.
3 For birds, the food ingestion rate as dry weight is calculated from the regression parameters for dry matter intake per day from Table 3 in Nagy (2001), using categories of birds used to calculate food ingestion rate in terms of wet weight as described in the text below.
4 See note 1 above. For anadromous species, a range of body weights is given corresponding to the period spent rearing in the estuary.
5 We interpret "clam-based" broadly to mean filter-feeding benthic mollusk-based.
6 For the worst case, we assume that all birds consumed are those waterfowl (scaups and scoters) that primarily feed on benthic mollusks (clams, etc.).
7 Percent of mollusks in gizzards of 819 adults and 4 juveniles collected in coastal Maine and Washington (Cottam C. *U.S. Dep. Agric. Tech. Bull*. 643) .
8 Wet weight percents of summer and winter gizzard contents, British Columbia salt water (Vermeer K. 1981 *Wildfowl* 32:107-116; Vermeer and Bourne 1984 as summarized in Appendix 1 of Savard *et al*. 1998).
9 Percent mussels, winter, coastal New England (reviewed in Bordage and Savard 1995).
10 Although the diets of salmon, steelhead and delta smelt are not known to be clam-based, these species may still be at risk from selenium because of greater sensitivity to selenium. The sensitivity of salmon and steelhead is documented below. The sensitivity of delta smelt to selenium is unknown; population numbers are alarmingly low, so this species is particularly vulnerable to any adverse effect.
11 Percentage clams by volume, fall, Suisun Bay and Carquinez Strait (Table 10 below).

SPECIES MOST AT RISK

Species were selected from among the species considered (Table 1) representing those species that are both (1) generally endangered, threatened or of concern, and (2) most at risk specifically for selenium toxic effects in the San Francisco Bay/Delta estuary. Information on the diet, body weight, and ingestion rate of each of the species most at risk is summarized in Table 2 and outlined in greater detail below.

Bald Eagle (*Haliaeetus leucocephalus*)

Status: The bald eagle was federally listed as endangered on February 14, 1978 (43 FR 6233) in all of the coterminous United States except Minnesota, Wisconsin, Michigan, Oregon, and Washington, where it was classified as threatened. On August 15, 1995 (60 FR 36010), the bald eagle was down-listed to threatened throughout its range. On July 6, 1999, the USFWS published a proposed rule to remove the bald eagle from the federal list of threatened and endangered species (64 FR 36454). On July 9, 2007 the species was removed (delisting effective August 8, 2007) from the Federal List of Endangered and Threatened Wildlife (72 FR 37346). The protections provided to the bald eagle under the Bald and Golden Eagle Protection Act (BGEPA) and the Migratory Bird Treaty Act (MBTA) will continue to remain in place after the species is delisted. To help provide more clarity on the management of bald eagles after delisting the USFWS published a regulatory definition of ''disturb'', the final National Bald Eagle Management Guidelines and a proposed rule for a new permit that would authorize limited take under the BGEPA.

Size: The bald eagle was a representative species used for the derivation of wildlife criteria in the Great Lakes Initiative (GLI) (USEPA 1995). For that effort, the bald eagle body weight used in criteria calculations (4.6 kg) was based on the mean of average male and female eagle body weights; although it was noted that female eagles are approximately 20 percent heavier than males. Because a bald eagle reference dose for selenium should be based on the most sensitive endpoint, which most likely is adverse reproductive effects manifested by laying females, it is more appropriate to use average female body weights in the calculation of wildlife values.

In the GLI, the USEPA presented an average body weight of 5.2 kg for female bald eagles. This value was based on the weights of 37 birds, taken from Snyder and Wiley (1976). Dunning (1993) presented an average female body weight of 5.35 kg, also based on the weights of 37 birds, taken from Palmer (1988). The average of these two (equally weighted) averages is **5.275 kg**.

Diet: The bald eagle diet has been extensively studied throughout North America. Although generally known as a piscivorous species, bald eagles are opportunistic predators and carrion scavengers (Buehler 2000). Various birds, mammals, reptiles, amphibians, and crustaceans may serve as additional bald eagle prey (Buehler 2000). In Northern California bald eagle diets commonly consist of (but are not limited to) Sacramento sucker (*Catostomus occidentalis*), hardhead (*Mylopharodon conocephalus*), Sacramento pikeminnow (*Ptychocheilus grandis*), brown bullhead (*Ameiurus nebulosus*), common carp (*Cyprinus carpio*), tui chub (*Gila bicolor*), rainbow trout (*Onchorhyncus mykiss*), largemouth bass (*Micropterus salmoides*), Sacramento perch (*Archoplites interruptus*), American coot (*Fulica americana*), mallard (*Anas platyrhynchos*), western grebe (*Aechmophorus occidentalis*), gulls (*Larus* spp.), pied-billed grebe (*Podilymbus podiceps*), common merganser (*Mergus merganser*), and other diving ducks (Hunt *et al*. 1992; Jackman *et al*. 1999).

Food ingestion rate: California supports both wintering and resident bald eagles, with a broad array of suitable foraging habitats. Because of this variety, eagle diets in California span a wide range of possible food types and trophic level combinations. Attempting to quantify a specific dietary composition for bald eagles is more difficult than for other species with a narrower range of prey types, and is further confounded by the fact that food preferences may vary both geographically and temporally. It is not possible in the scope of this analysis to determine all the potential bald eagle diets in California and evaluate them with regard to the selenium criterion, however, a review of several approaches is provided and an average FIR for bald eagles in northern California is considered.

Since bald eagles are well described as "carnivorous birds", it is possible to calculate the bald eagle food ingestion rate (FIR) using the generic allometric equation for carnivorous birds from Nagy (2001). This equation is based on body mass and for the reasons stated above we use the average body mass of females: 5275 g.

FIR (wet weight) = $3.048 \times (\text{body weight in g})^{0.665}$
FIR = $3.048 \times 5275^{0.665}$
FIR = 911 g/day wet weight

Alternatively, when information is available on the dietary composition and free-living metabolic rate (FMR) of a species, USEPA (1993) recommends that this information be used to calculate an estimate of the food ingestion rate. Several alternative bald eagle diets are considered below using this method.

The dietary composition of bald eagles was investigated as part of the GLI (USEPA 1995). Using information on bald eagles nesting on islands and along the shore of Lake Superior in Wisconsin (*from* Kozie and Anderson 1991), and adjustment factors to estimate the relative number of birds and fish delivered to a nest based on the prey remains found under the nest, the USEPA determined that 92 percent of the dietary biomass comprised fish and 8 percent comprised birds or mammals. The adjustment factor was developed to account for the inherent error in estimating a dietary composition based solely on the analysis of prey remains. Using this dietary composition of 92 percent fish and 8 percent birds and mammals, along with information about the energetic needs of adult eagles and their ability to assimilate the caloric content of these food types, the GLI presented estimates of the amount of each food type ingested daily: 464 g fish and 40 g birds/mammals for a total of 504 g/day (USEPA 1995). This analysis depends upon diets of bald eagles in the Great Lakes area; however, there is sufficient dietary data available on bald eagles in California to calculate a FIR representative of local diets.

The Service's evaluation of the threatened and endangered wildlife protectiveness of USEPA's human health criterion for methylmercury in California (USFWS 2003) includes a detailed discussion of the trophic composition of the diet of Northern California bald eagles. This assessment utilized detailed bald eagle dietary data from Jackman *et al*. (1999) who studied 56 nests across Northern California. Although the FIR calculated in the Service document (600 g/day) was based upon a diet representative of Northern California bald eagles (83 % fish, 17% birds), the assessment was based upon risks and assumptions associated with various trophic level diets relevant to mercury biomagnification. This diet represented the highest combined percentage of trophic level 4 fish and aquatic-dependent birds from within the Jackman *et al*. (1999) dataset and may not be appropriate for assessing selenium exposure in the Bay area.

Based on the dietary analysis presented by Jackman *et al*. (1999) a generic composition for Northern California bald eagle diet can be estimated as 71.2 percent fish, 22.8 percent birds, and 6 percent mammals. These figures represent an average dietary composition for all bald eagles in the study area and may be the most appropriate estimate to use for this analysis in lieu of Bay area specific bald eagle dietary information.

When information is available on the dietary composition and free-living metabolic rate (FMR) of a species, USEPA (1993) recommends that this information be used to calculate an estimate of the food ingestion rate (FIR) of the species using the equation:

$$FIR = FMR / ME_{avg}$$

Where ME_{avg} is the weighted average metabolizable energy of the items in the diet.

FMR can be calculated using the allometric relationship between weight and FMR developed by Nagy (1987) for non-passerine birds:

$$\begin{aligned} FMR\ (kcal/day) &= 1.146\ Wt^{0.749}\ (g) \\ &= 1.146 \times 5275^{0.749} \\ &= 703.3\ kcal/day \end{aligned}$$

The generic bald eagle diet proportions (71.2 % fish, 22.8 % birds, and 6 % mammals) from Jackman *et al*. (1999) are then used to calculate average metabolizable energy following the method outlined in USEPA (1993):

Dietary item	Proportion of diet P	Gross Energy (kcal/g wet wt) GE*	Assimilation efficiency AE†	Metabolizable energy (kcal/g wet wt) ME=GE x AE	P x ME (kcal/g wet wt)
Fish	0.712	1.2	0.79	0.948	0.675
Birds	0.228	1.9	0.78	1.482	0.338
Mammals	0.06	1.7	0.78	1.326	0.080

* from Table 4-1 in USEPA (1993)
† from Table 4-3 in USEPA (1993)

The sum of the right-most column in the table above provides the weighted average metabolizable energy:

$$ME_{avg} = \Sigma (P \times ME)$$
$$= 0.675 + 0.338 + 0.080$$
$$= 1.09 \text{ kcal/g wet weight (following procedure on p. 4-17 EPA 1993)}$$

The food ingestion rate is then calculated from the above estimates of free-living metabolic rate and average metabolizable energy in the diet:

$$FIR = FMR/ ME_{avg}$$
$$= 703.3/1.09$$
$$= \mathbf{644\ g/day} \text{ wet weight}$$

General life history: Breeds in coastal and aquatic habitat with forested shorelines or cliffs in North America, including the Pacific Northwest and the northern Sierra Nevada Mountains in California (Buehler 2000). More recently breeding pairs occur along reservoirs in the Coast Ranges of central and southern California including the Channel Islands (CDFG 2005). Wintering areas include coastal estuaries and river systems of northern California (Buehler 2000).

Risk of selenium exposure: Lillebo *et al*. (1988) derived levels of selenium to protect various species of waterbirds. Based on an analysis of bioaccumulation dynamics and an estimated critical dietary threshold for toxicity of 3 µg/g, they concluded that piscivorous birds would be at substantially greater risk of toxic exposure than mallards (*Anas platyrhynchos*). The calculated water criterion to protect piscivorous birds was 1.4 µg/L as opposed to 6.5 µg/L for mallards. It should also be noted that the 6.5 µg/L calculated criterion for mallards exceeds the actual threshold point for ducks in the wild which is somewhere below 4 µg/L (Skorupa 1998). Thus, the 1.4 µg/L calculated criterion for piscivorous birds may be biased high compared to the wild as well.

Applying an energetics modeling approach, modified from the Wisconsin Department of Natural Resources, Peterson and Nebeker (1992) calculated a chronic criterion specifically for bald eagles. Peterson and Nebeker's estimate of a protective criterion is 1.9 µg/L. Peterson and Nebeker calculated a mallard criterion (2.1 µg/L) that was much closer to their bald eagle criterion than Lillebo *et al.*'s (1988) results would suggest. Peterson and Nebeker's mallard criterion is consistent with real-world data (cf. Skorupa 1998) and therefore their bald eagle criterion may also be reliable.

As potential prey for the bald eagles wintering or breeding in the Bay area, diving ducks such as surf scoter and scaup would provide a significant source of selenium (see discussion on these species in this document). Surf scoter and scaup are abundant in San Pablo and Suisun Bays in the winter (Goals Project 2000). A pair of bald eagles has begun nesting at San Pablo Reservoir (ESA 2006, The Quail 2006) about 5-6 miles from North and Central San Francisco Bay.

California Clapper Rail (*Rallus longirostris obsoletus*)

Status: The California clapper rail was federally listed as endangered on October 13, 1970 (35 FR 16047-16048).

Size: In the only literature found for this particular subspecies that provided body weights, nineteen female California clapper rails from South San Francisco Bay were examined as part of a Master's Degree thesis (Albertson 1995). Weights ranged from 300 to 400 g, with a mean weight of **346 g.**

Diet: The most comprehensive assessment of the California clapper rail diet is presented by Moffitt (1941). Stomach contents from 18 birds were examined and the food items identified and measured as a volumetric percentage. On average, animal matter accounted for approximately 85 percent of the diet, with the remainder composed of seed and hull fragments of marsh cordgrass. Over half (56.5%) of the overall diet comprised plaited horse mussels (*Modiolus demissus*). Spiders of the family Lycosidae (wolf spiders) accounted for 15 percent of the diet, while the remaining important dietary items were little macoma clams (*Macoma balthica*) (7.6%), yellow shore crabs (*Hemigrapsis oregonensis*) (3.2%), and worn-out nassa snails (*Ilyanassa obsoletus*) (2.0%). Worms, insects, and carrion combined accounted for a total of 1.1 percent of the remaining diet found by Moffitt (1941) in the 18 clapper rail stomachs. The importance of crabs in the clapper rail diet was confirmed by Varoujean (1972), who observed rails eating striped shore crabs (*Pachygrapsus crassipes*).

Although Moffitt (1941) reported that plant matter accounted for approximately 15 percent on average of the clapper rail diets, the author stated that this percentage probably represented the maximum of a vegetable diet. This conclusion was based on the fact that the birds were collected in early February, a time when animal food items would typically be at lowest abundance. However, it is important to note that this reported average for plant food (~15%) was calculated from a wide range of percentages in the 18 birds examined (0% - 58% plant food). As with other omnivorous species, the amount of any particular food item consumed at any given time may vary substantially depending on a number of factors. While clapper rails most likely do not eat a set amount of plant matter daily, it is clear from Moffitt (1941) that vegetation generally constitutes a substantial dietary item over time.

Based on Moffitt's (1941) assumption that his mid-winter gut analyses represented a maximum for vegetation in the clapper rail diet, and the knowledge that clapper rails nest during a time when animal foods would be in greater abundance (mid-March - July) (USFWS 1984), the overall rail diet for is assumed to be 10 percent vegetation and 90 percent animal matter.

Food ingestion rate (FIR): Clapper rails may consume a wide variety of foods. Values for the gross energy content for some of these foods (*e.g.*, shell-less bivalves, shelled crabs) and the efficiency at which rails assimilate them can be found in the *Wildlife Exposure Factors Handbook* (USEPA 1993). However, because rails do not consume set amounts of these food types, FIR must be estimated using one of the generic allometric equations from Nagy (2001). Out of the 17 avian categories for predicting FIRs presented by Nagy (2001), Charadriiformes is the taxonomic order most closely related to rails (Gill 1995). In addition, the rail's feeding ecology most closely resembles that of birds in the Charadriiformes category (*i.e.*, shore birds, gulls, auks). Therefore, the FIR for California clapper rails was calculated using the following equation:

FIR (wet weight) = $1.914 \times (\text{body weight in g})^{0.769}$
FIR = $1.914 \times 346^{0.769}$
FIR = **172 g/day wet weight**

General life history: The California clapper rail inhabits salt marshes surrounding the San Francisco Bay, California. Principal habitats are low portions of coastal wetlands dominated by cordgrass and pickleweed. Nesting habitat in San Francisco Bay is characterized by tidal sloughs, abundant invertebrate populations, pickleweed, gum plant, and wrack in upper zone. Individuals do not migrate far from the breeding grounds (Eddleman and Conway 1998).

Risk of selenium exposure: California clapper rails feed largely on benthic invertebrates, including filter-feeding mussels and clams, a well-documented pathway for bioaccumulation of selenium (Pease *et al*. 1992, Stewart *et al*. 2004). Lonzarich *et al*. (1992) reported that eggs of California clapper rails collected from the north bay in 1987 contained up to 7.4 µg/g selenium. Water data from this time and location are not available. The *in ovo* threshold for selenium exposure that causes toxic effects on embryos of California clapper rails is unknown. For another benthic-foraging marsh bird, the black-necked stilt, the *in ovo* threshold for embryotoxicity is 6 µg/g selenium (Skorupa 1998). The most widely-used hormetic model (Brain and Cousens 1989) applied to Heinz *et al*. (1989) data from laboratory experiments with mallard reproduction indicates that in mallards, a selenium concentration of 7.4 µg/g (dry weight) in the eggs would be associated with a 32 percent reduction in hatchability of the eggs (Figure 1).

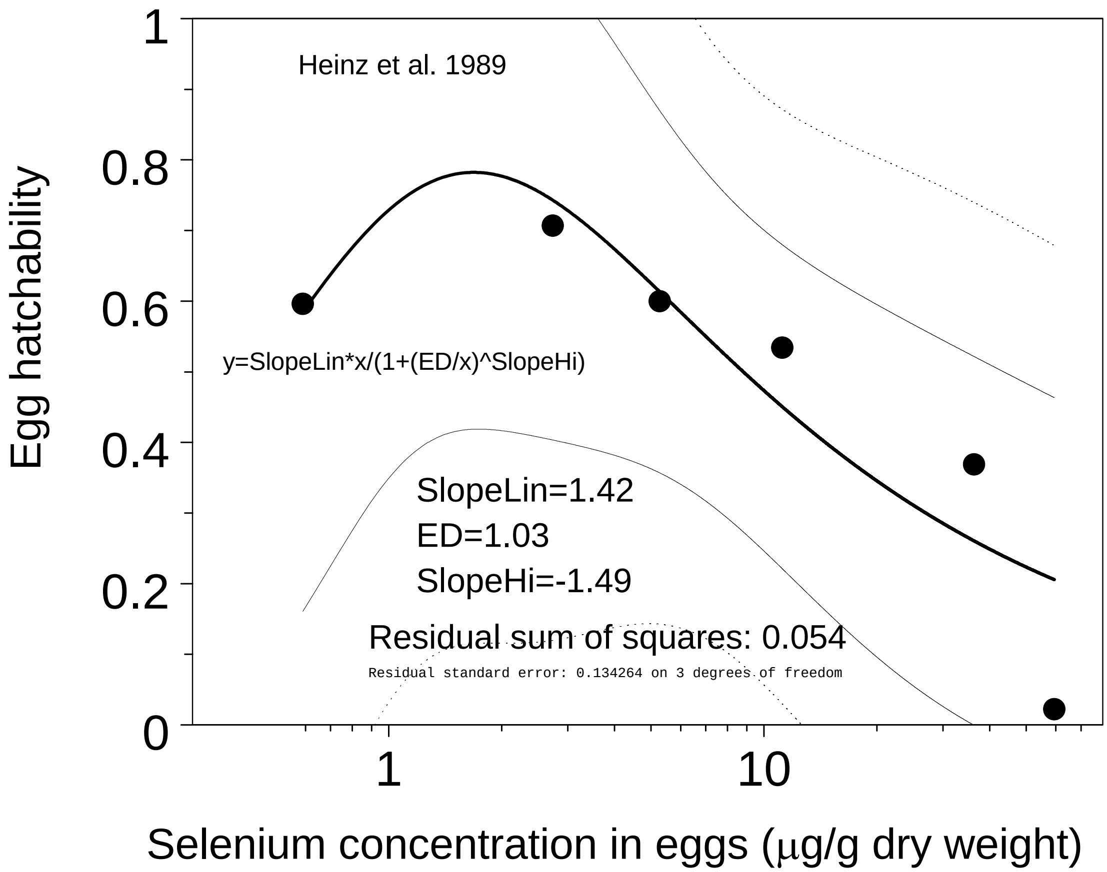

Figure 1. The percent hatching success of mallard eggs as a function of selenium concentration in the eggs, with the Brain-Cousens (1989) hormetic model fitted by least squares regression. Confidence intervals of 95% and 99% are shown.

In mallard ducks, interactive effects of selenium and mercury can be super-toxic with regard to embryotoxic effects (Heinz and Hoffman 1998). Lonzarich *et al*. (1992) also reported potentially embryotoxic concentrations of mercury in eggs of California clapper rails. Abnormally high numbers of nonviable eggs, 13.7-22.9 percent, potentially a result of mercury contamination, have also been reported for the California clapper rail (Schwarzbach 1994, Schwarzbach *et al.* 2006). Based, in part, on the data for California clapper rails, staff technical reports prepared for the San Francisco Bay Regional Water Quality Control Board recommend decreasing selenium loading to the estuary by 50 percent or more (Taylor *et al*. 1992; 1993). The California clapper rail is particularly vulnerable to any locally elevated effluent concentrations of selenium as the rail generally occupies small home ranges of only a few acres.

Greater Scaup (*Aythya marila*)

Status: The greater scaup is a relatively little-known duck with a circumpolar distribution (Kessel *et al*. 2002). The San Francisco Bay is one of the two main wintering grounds of this species along the west coast of North America. Regulated hunting of these birds is permitted under provisions of the Migratory Bird Treaty Act.

Size: The mean body mass of 345 adult male greater scaup on Long Island Sound, Connecticut, measured between October 1991 and January 1994, was **1,054 g** (range: 850-1,350; standard deviation: 105); the mean body mass of 104 adult females at the same location was 959 g (range: 688-1,210; standard deviation: 108) (Kessel *et al*. 2002).

Diet: Greater scaup dive in waters from one to ten meters deep to feed mainly on benthic invertebrates, especially bivalves in winter (Bellrose 1980). The stomach contents of 119 greater scaup in Long Island Sound in the winter of 1952-1953 consisted of 93.5% (by volume) animal material, including 56.4% bivalves, 6.1% snails, 24.3% unidentified mollusks, 4.4% crustaceans (Cronan 1957). The remaining diet was plant material (6.6%) consisting mainly of algae (3.6%). More recent studies suggest that diets vary substantially among different locations, and the dietary proportion of animal material has declined in Long Island Sound (review in Kessel *et al*. 2002). During the winters of 1998-1999 and 1999-2000, the introduced Asian clam, *Potamocorbula amurensis*, was the most important food of greater scaup in San Pablo Bay (Poulton *et al.* 2002). The Asian clam is an extraordinarily efficient bioaccumulator of selenium (Stewart *et al*. 2004).

Food ingestion rate: The most likely limiting effect of selenium on wintering greater scaup in the San Francisco Bay area is reduction in body mass, or reduction of the rate of increase in body mass, as the birds build up reserves in preparation for their migration to breeding grounds in Alaska and northern Canada. Because wintering males bioaccumulate selenium to higher concentrations than females (see below), their migration is more likely to be adversely affected. Therefore, for this study, we use the body mass and estimate the food ingestion rate (FIR) of males. Due to the variability of the diets of wintering greater scaup, we use a generic equation to estimate FIR based on body mass (Nagy 2001). No such equation is available specifically for clam-eating birds, but because greater scaup are largely carnivorous and mainly marine or estuarine, we use equations for carnivorous and marine birds (Nagy 2001):

For carnivorous birds:
$\text{FIR (wet weight)} = 3.048 \times (\text{body weight in g})^{0.665}$
$\text{FIR} = 3.048 \times 1054^{0.665}$
FIR = 312 g/day wet weight

For marine birds:
$\text{FIR (wet weight)} = 3.221 \times (\text{body weight in g})^{0.658}$
$\text{FIR} = 3.221 \times 1054^{0.658}$
FIR = 314 g/day wet weight

These estimates are in close agreement; we average them to yield an estimated food ingestion rate of **313 g/day** wet weight

General life history: Greater scaup breed mainly on the tundra of Alaska and northern Quebec, with some breeding sites scattered across Canada. They winter mainly along the northeast coast of North America. Along the west coast of North America they winter as far south as northern Baja California, but mainly around Vancouver Island and in the San Francisco Bay (Kessel *et al*. 2002).

Risk of selenium exposure: In San Pablo Bay, greater scaup feed primarily on bivalves, which are efficient selenium bioaccumulators (Stewart *et al*. 2004). Consequently, greater scaup can bioaccumulate selenium to high concentrations. Bioaccumulation evidently increases through the winter, and is greater in males than in females. Two adult males collected along the coast of California in early winter had liver selenium concentrations of 7.6 and 23.7 μg/g dry weight (geometric mean: 13.4 μg/g). In late winter five males had a geometric mean liver concentration of 47.9 μg/g and two females had liver concentrations of 8.9 and 31 μg/g (geometric mean: 16.61 μg/g) (Takekawa *et al*. 2002). Eighteen greater scaup collected from San Francisco Bay in March and April 1982 had liver selenium concentrations ranging from 6.7 to 31 μg/g dry weight (mean: 19.3 μg/g) (Ohlendorf *et al*. 1986). Males collected in San Francisco Bay in early winter of 1986-87 had a geometric mean liver selenium concentration of 20.7 μg/g; in late winter geometric means at two sites in San Francisco Bay were 32.7 and 27.4 μg/g (Hothem *et al*. 1998). Adult males collected from a reference site and a polluted site in San Francisco Bay in March, 1989 had liver selenium concentrations ranging from 7 to 23 μg/g (geometric mean: 13 μg/g) at the reference site, and ranging from 21 to 140 μg/g (geometric mean: 67 μg/g) at the polluted site (Hoffman *et al*. 1998). Reproductive impairment is associated with a liver concentration of 10 μg/g (dry weight), and concentrations above 33 μg/g (dry weight) are considered harmful to the health of mallards, *Anas platyrhynchos* (Heinz *et al*. 1989).

Lesser Scaup (*Aythya affinis*)

Status: The lesser scaup is the most abundant North American diving duck, but declining populations in recent decades (Austin *et al*. 1998) have raised concerns. Regulated hunting of these birds is permitted under provisions of the Migratory Bird Treaty Act.

Size: On average, male lesser scaup are heavier than females. The mean mass of six adult and hatch-year males migrating through the Klamath Basin in northern California in spring was **734 g** (standard deviation: 24); the mean mass of five females was 663 g (standard deviation: 109) (Gammonley and Heitmeyer 1990).

Diet: Wintering lesser scaup feed primarily by diving for clams, other mollusks, aquatic insects and crustaceans, but in some places, up to 99 percent of the diet consists of plant material (review in Austin *et al*. 1998). During the winters of 1998-1999 and 1999-2000, 96 percent of the dry mass of the esophagus contents of 13 lesser scaup in San Pablo Bay consisted of the introduced Asian clam, *Potamocorbula amurensis*, (Richman and Lovvorn 2004) which is an extraordinarily efficient bioaccumulator of selenium (Stewart *et al*. 2004).

Food ingestion rate: The most likely limiting effect of selenium on wintering lesser scaup in the San Francisco Bay area is reduction in body mass, or reduction of the rate of increase in body mass, as the birds build up reserves in preparation for their migration to breeding grounds in Alaska and northern Canada. It is likely that wintering males bioaccumulate selenium to higher concentrations than females because this been clearly shown for the closely related Greater Scaup (see above) and the more meager data for lesser scaup are in agreement (Takekawa *et al*. 2002). Therefore, the migration of males is more likely to be adversely affected by selenium. So, for this study, we use the body mass and estimate the food ingestion rate (FIR) of males. Due to the variability of the diets of wintering lesser scaup, we use a generic equation to estimate FIR based on body mass (Nagy 2001). No such equation is available specifically for clam-eating birds, but because wintering lesser scaup are largely carnivorous and mainly marine or estuarine, we use equations for carnivorous and marine birds (Nagy 2001):

For carnivorous birds:
FIR (wet weight) = $3.048 \times (\text{body weight in g})^{0.665}$
FIR = $3.048 \times 734^{0.665}$
FIR = 245.3 g/day wet weight

For marine birds:
FIR (wet weight) = $3.221 \times (\text{body weight in g})^{0.658}$
FIR = $3.221 \times 734^{0.658}$
FIR = 247.5 g/day wet weight

These estimates are in close agreement; we average them to yield an estimated food ingestion rate of **246 g/day wet weight**.

General life history: Lesser scaup breed mainly in Alaska and Canada. In the U.S. Pacific Northwest they breed as far south as the Klamath basin in northern California. They winter mainly along the Gulf coast and the Pacific coast of North and Central America on freshwater bays and wetlands as well as estuarine and marine habitats (Austin *et al*. 1998).

Risk of selenium exposure: In San Francisco Bay ducks, contaminants, including selenium, are associated with reduced body size. This suggests that ducks wintering in the more contaminated areas in the bay might be in poor condition when they reach their breeding grounds in the boreal forest, which may reduce nesting success (John Takekawa, USGS, quoted in Martin 2002). In addition, migration success may be reduced by lowered body condition associated with high body burdens of selenium.

In December 1986 one adult male Lesser Scaup in Lake Earl, northern California, had a selenium liver concentration of 11 µg/g dry weight; four juvenile males had a liver concentrations ranging from 2.8 to 12.2 µg/g (geometric mean: 6.3 µg/g); the liver concentration of one juvenile female was 3.5 µg/g (Takekawa *et al*. 2002). In March 1987 six adult males in Morro Bay, central California, had liver selenium concentrations ranging from 7.1 to 19 µg/g (geometric mean: 11.13 µg/g); two juvenile males had liver concentration ranging from 8.9 to 16 µg/g (geometric mean: 11.92 µg/g) (Takekawa *et al*. 2002). In these relatively uncontaminated sites, Takekawa *et al*. (2002) found a positive correlation between selenium and carcass mass, as would be expected for a biphasic (beneficial at low concentrations; toxic at high concentrations) contaminant such as selenium at the lower range of selenium concentrations.

White-winged Scoter (*Melanitta fusca*)

Status: The white-winged scoter is the best known of the world's three species of scoter. It is the sea duck most commonly taken by hunters, constituting 30 percent of all sea ducks hunted from 1961 to 1986 (USFWS 1988). Despite increasing hunting pressure (Brown and Fredrickson 1997), success in hunting white-winged scoters has declined since 1976 (USFWS 1988), an indication of declining populations.

Size: Twenty-one male white-winged scoters along the coast of British Columbia throughout the year had a mean mass of 1,722 g (standard deviation: 51); nineteen females had a mean mass of 1,437 g (standard deviation: 46) (Vermeer and Bourne 1984). White-winged scoters exhibit seasonal variation in weight, reaching maximum weights before breeding (Brown and Fredrickson 1997). Therefore, weight measurements taken in the non-breeding period are most relevant to white-winged scoters wintering in the San Francisco Bay area. The averages of weights of non-breeding white-winged scoters collected from November to April in Kachemak Bay, Alaska, were **1,917** g (28 males) and 1,732 g (10 females).

Diet: In wintering areas, white-winged scoters usually dive in water depths of less than five meters to feed on benthic mollusks (especially clams), crustaceans, and insects (review in Brown and Fredrickson 1997).

Food ingestion rate: The most likely limiting effect of selenium on wintering white-winged scoter in the San Francisco Bay area is reduction in body mass, or reduction of the rate of increase in body mass, as the birds build up reserves in preparation for their migration to breeding grounds in Alaska and northern Canada. It is likely that wintering males bioaccumulate selenium to higher concentrations that females, as this has been documented in related and ecologically similar diving ducks (see above). Therefore, the migration of males is more likely to be adversely affected by selenium, so, for this study, we use the body mass and estimate the food ingestion rate (FIR) of males. Due to the variability of the diets of wintering white-winged scoter, we use a generic equation to estimate FIR based on body mass (Nagy 2001). No such equation is available specifically for clam-eating birds, but because wintering white-winged scoters are largely carnivorous and mainly marine or estuarine, we use equations for carnivorous and marine birds (Nagy 2001):

For carnivorous birds:
FIR (wet weight) = $3.048 \times (\text{body weight in g})^{0.665}$
FIR = $3.048 \times 1917^{0.665}$
FIR = 464.5 g/day wet weight

For marine birds:
FIR (wet weight) = $3.221 \times (\text{body weight in g})^{0.658}$
FIR = $3.221 \times 1917^{0.658}$
FIR = 465.5 g/day wet weight

These estimates are in close agreement; we average them to yield an estimated food ingestion rate of **465 g/day wet weight**.

General life history: North American white-winged scoters breed in Alaska and western Canada and winter along the east, west, and gulf coasts of the United States (Brown and Fredrickson 1997).

Risk of selenium exposure: The risk to the migration of white-winged scoters due to selenium exposure in their wintering areas is similar to the risks for scaups (see above). In addition, some white-winged scoter females arrive in their northern breeding areas with sufficiently high body burdens of selenium to be a concern for survival and reproductive success. Reproductive impairment is associated with a liver concentration of 10 µg/g (dry weight), and concentrations above 33 µg/g (dry weight) are considered harmful to the health of mallards, *Anas platyrhynchos* (Heinz *et al*. 1989). The liver of a female white-winged scoter found dead at Cape Yakataga, Alaska, in August 1992, had a selenium concentration of 53 µg/g dry weight (Henny *et al*. 1995). A pooled sample of livers of 7 female white-winged scoters collected in the Old Crow/Porcupine River area in the northern Yukon Territory 1988-1994 had a selenium concentration of 20 µg/g wet weight. This is equivalent to 66.7 µg/g dry weight (average 70% moisture in the livers analyzed in this study) (Braune and Malone 2006).

Surf Scoter (*Melanitta perspicillata*)

Status: Until recently, the surf scoter has been the least known of North American ducks (Belrose 1980), although it is common in winter along the coast of California (Johnsgard 1975; Cogswell 1977).

Size: As with other scoters and scaup (see above) males are heavier than females. In May 1996, 21 males collected in the St. Lawrence estuary had a mean mass of **1059.3** g (standard deviation: 133.7); 16 females had a mean mass of 985.3 g (standard deviation: 118.6) (Savard *et al*. 1998).

Diet: Surf scoters feed primarily on mussels and other mollusks during all seasons of the year; plants and crustaceans are additional minor components of their diets (review in Savard *et al*. 1998).

Food ingestion rate: The most likely limiting effect of selenium on wintering surf scoter in the San Francisco Bay area is reduction in body mass, or reduction of the rate of increase in body mass, as the birds build up reserves in preparation for their migration to breeding grounds in Alaska and northern Canada. It is likely that wintering males bioaccumulate selenium to higher concentrations that females, as this has been documented in related and ecologically similar diving ducks (see above). Therefore, the migration of males is more likely to be adversely affected by selenium. So, for this study, we use the body mass and estimate the food ingestion rate (FIR) of males. Due to the variability of the diets of wintering surf scoter, we use a generic equation to estimate FIR based on body mass (Nagy 2001). No such equation is available specifically for clam-eating birds, but because wintering surf scoters are largely carnivorous and mainly marine or estuarine, we use equations for carnivorous and marine birds (Nagy 2001):

For carnivorous birds:
FIR (wet weight) = $3.048 \times (\text{body weight in g})^{0.665}$
FIR = $3.048 \times 1059^{0.665}$
FIR = 313.0 g/day wet weight

For marine birds:
FIR (wet weight) = $3.221 \times (\text{body weight in g})^{0.658}$
FIR = $3.221 \times 1059^{0.658}$
FIR = 315.0 g/day wet weight

These estimates are in close agreement; we average them to yield an estimated food ingestion rate of **314 g/day wet weight**.

General life history: North American surf scoters breed in northern Canada and Alaska and winter along the east, west, and gulf coasts of North America (Savard *et al*. 1998)(Figure 2). The San Francisco Bay estuary supports the southernmost large population of wintering surf scoters; this species constitutes more than 99 percent of the scoters in the estuary (Takekawa *et al*. 2002).

Risk of selenium exposure: The risk to the migration of surf scoters due to selenium exposure in their wintering areas is similar to the risks for scaups (see above). Livers from 22 surf scoters collected in March and April 1982 in south San Francisco Bay had a mean selenium concentration of 34.4 µg/g dry weight (Ohlendorf *et al*. 1986). The average selenium concentration in the livers of 79 adult male surf scoters collected in north and south San Francisco Bay in January and March 1985 was about 60 µg/g dry weight (Ohlendorf *et al*. 1991). Livers from ten adult male surf scoter collected in Suisun Bay in March 1989 had a mean concentration of 119 µg/g dry weight (Hoffman *et al*. 1998).

Selenium is depurated fairly rapidly, with a half life of about 19 days in mallards (Heinz *et al*. 1990). Therefore, depuration during migration would be expected to mitigate the risk to reproduction caused by elevated loads of selenium carried from wintering areas to nesting areas. However, surf scoters bioaccumulate selenium to such high levels in the San Francisco Bay area, and their migration is so rapid (Savard *et al*. 1998) that some surf scoter females arrive in their northern breeding areas with sufficiently high body burdens of selenium to be a concern for hatching success and embryo toxicity. A pooled sample of livers of two female surf scoters collected at Old Crow in the northern Yukon Territory 1988-1994 had a selenium concentration of 13 µg/g wet weight. This is equivalent to 43.3 µg/g dry weight (average 70% moisture in the livers analyzed in this study). A pooled sample of four male surf scoter livers collected at Fort Good Hope in northern Northwest Territories, Canada, had a selenium concentration of 17 µg/g wet weight, equivalent to 56.7 µg/g dry weight (Braune and Malone 2006). Reproductive impairment is associated with a liver concentration of 10 µg/g (dry weight), and concentrations above 33 µg/g (dry weight) are considered harmful to the health of mallards, *Anas platyrhynchos* (Heinz *et al*. 1989).

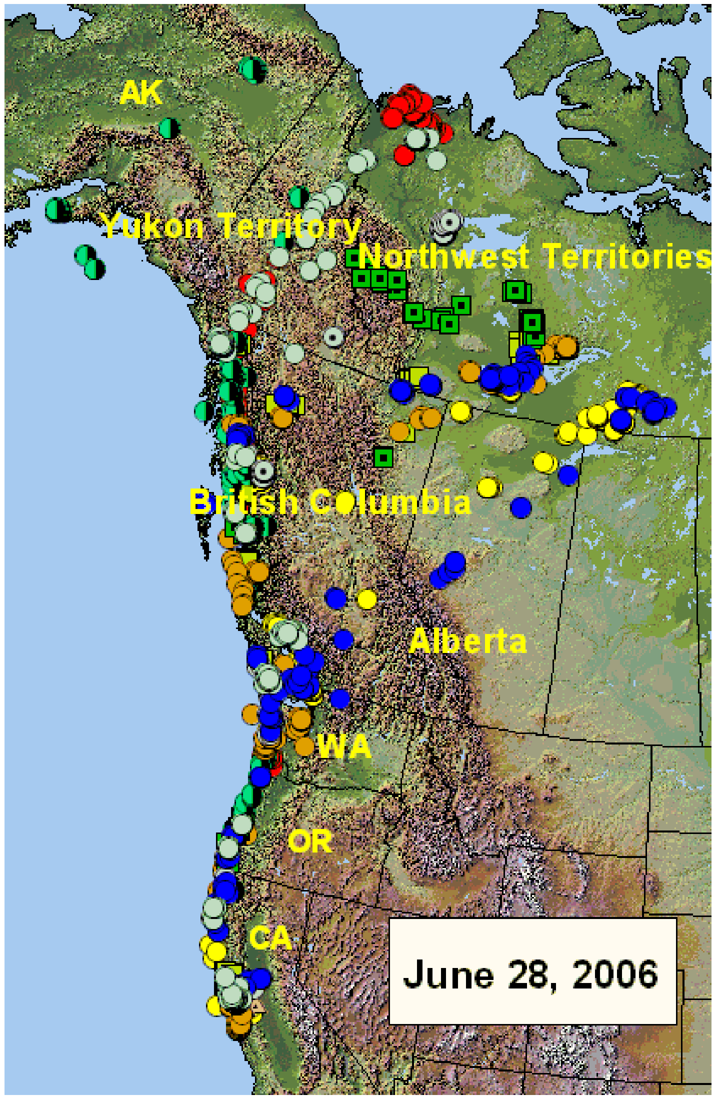

Figure 2. Tracking of individual surf scoters from their wintering grounds in the San Francisco Bay area and coastal North America to their breeding grounds in the north, by the US Geological Survey (http://www.werc.usgs.gov/scoter/2005/maps.html)

Black Scoter (*Melanitta nigra*)

Status: The Black Scoter is one of the least-known North American ducks (Bordage and Savard 1995). Declining populations of sea-ducks (USFWS 1993; Goudie *et al*. 1994) and of scoters in particular (Kehoe *et al*. 1994) have been a matter of concern. A lower ratio of immature to adult birds has been observed in black scoters relative to other scoters (Padding *et al*. 1992; 1993), suggesting that black scoters have been experiencing lower reproductive success.

Size: As with other scoters and scaup (see above) males are heavier than females. In 1990 in Quebec, 34 males had a mean mass of **1117.0 g** (standard deviation: 101.6); 21 females had a mean mass of 987.4 g (standard deviation: 110.1) (Consortium Gauthier & Guillemette—GREBE 1993).

Diet: Wintering and migrant black scoters feed primarily on mussels and clams; snails and barnacles are additional minor components of their diets (review in Bordage and Savard 1995).

Food ingestion rate: The most likely limiting effect of selenium on wintering black scoter in the San Francisco Bay area is reduction in body mass, or reduction of the rate of increase in body mass, as the birds build up reserves in preparation for their migration to breeding grounds in Alaska and northern Canada. It is likely that wintering males bioaccumulate selenium to higher concentrations that females, as this has been documented in related and ecologically similar diving ducks (see above). Therefore, the migration of males is more likely to be adversely affected by selenium, so, for this study, we use the body mass and estimate the food ingestion rate (FIR) of males. Due to the variability of the diets of wintering black scoter, we use a generic equation to estimate FIR based on body mass (Nagy 2001). No such equation is available specifically for bivalve-eating birds, but because wintering black scoters are largely carnivorous and mainly marine or estuarine, we use equations for carnivorous and marine birds (Nagy 2001):

For carnivorous birds:
FIR (wet weight) = $3.048 \times (\text{body weight in g})^{0.665}$
FIR = $3.048 \times 1117^{0.665}$
FIR = 324.3 g/day wet weight

For marine birds:
FIR (wet weight) = $3.221 \times (\text{body weight in g})^{0.658}$
FIR = $3.221 \times 1117^{0.658}$
FIR = 326.3 g/day wet weight

These estimates are in close agreement; we average them to yield an estimated food ingestion rate of **325 g/day wet weight**.

General life history: North American black scoters breed mainly in northern Quebec and Alaska and winter along the east, west, and gulf coasts of North America (Bordage and Savard 1995). Less than one percent of the scoters wintering in the San Francisco Bay estuary are black scoters (Takekawa *et al*. 2002).

Risk of selenium exposure: The risk to the migration of black scoters due to selenium exposure in their wintering areas is similar to the risks for scaups (see above). In addition, some black scoter females may arrive in their northern breeding areas with sufficiently high body burdens of selenium to be a concern for hatching success and embryo toxicity. Reproductive impairment is associated with a liver concentration of 10 µg/g (dry weight), and concentrations above 33 µg/g (dry weight) are considered harmful to the health of mallards, *Anas platyrhynchos* (Heinz *et al*. 1989). The livers of two weakened black scoter live-captured with a net at Cape Yakataga, Alaska, in August 1991, had selenium concentrations of 24 and 32 µg/g dry weight (Henny *et al*. 1995). An adult black scoter from Quebec had the highest selenium concentration in pectoral muscle (10.1 µg/g wet weight) among all 3,957 waterfowl and game birds collected and analyzed across Canada between 1987 and 1995 (Braune *et al*. 1999).

Chinook Salmon (*Oncorhynchus tshawytscha*)

Status: The National Marine Fisheries Service (NMFS) has identified 17 Evolutionarily Significant Units (ESUs) of Chinook salmon from Washington, Oregon, Idaho, and California (Myers *et al*. 1998; 63 FR 11482). Three of these use the San Francisco Estuary: the Sacramento River winter-run ESU, the Central Valley spring-run ESU, and the Central Valley fall/late fall-run ESU. The Sacramento River winter-run ESU was listed as endangered on January 4, 1994 (59 FR 440). On September 16, 1999, NMFS listed the Central Valley spring-run ESU as threatened (64 FR 50394). In the same rulemaking, NMFS also determined that the Central Valley fall/late fall ESU is not warranted for listing at this time.

Size: Chinook salmon fry remain in the estuarine nursery areas as they grow from about **0.5 g to about 18 g** (Figure 3). They migrate seaward from the estuary when they reach about 70 mm fork length (Healy 1991).

Diet: Young Chinook salmon in the Sacramento-San Joaquin River delta feed primarily on insects (especially chironomid larvae and dipterans) and crustacea (especially Neomysids (opossum "shrimp"), amphipods (mud "shrimp"), copepods, and cladocerans (water fleas)) (Sasaki 1966, Kjelson *et al*. 1982). Chinook salmon fry appear to be opportunistic feeders; hence their diet varies with locale, season, and available prey (Figure 4, Figure 5, Figure 6, Figure 7, and Figure 8). As they grow larger and move to deeper water within the estuary, they evidently shift dietary preference to larval and young fishes (Healey 1991).

Food ingestion rate: The food ingestion rates of wild Chinook salmon are entirely unknown. However, ingestion rates in the wild may be estimated from ingestion rates in captivity or from growth rates in the wild combined with food conversion efficiency (FCE = wet weight gain of fish / dry weight of food) in captivity. Fall-run Chinook salmon from the Cowlitz River had conversion rates of 1.32, 1.5, 1.3 and 1.36 in 1991, 1992, 1993 and 1994 respectively (WDFG 2001). The growth rates of marked fry in the wild (Vancouver Island estuaries) range from about 3% to 5% of body weight per day (Healey 1991, Figure 3). Based on the average FCE (1.37) from the Cowlitz, a medium growth rate (4% per day), and weights from Vancouver Island, Chinook salmon entering the estuary at about 1 g body weight consume about 0.055 g (dry weight) of food per day, and when they leave the estuary at about 18 g body weight they consume about 1.0 g (dry weight) of food per day.

MacFarlane and Norton (2002) estimated the growth rate of Chinook salmon in the San Francisco estuary to be about 0.02 g per day for young averaging about 7 g in weight (about 0.3% body weight per day). However, this estimate is not based on recaptures of tagged individuals but by comparing averages of all fish captured at different locations along the migration route. This method is known to underestimate growth rate because it does not account for mortality. However, if the fork length growth rate (0.47 mm/day for fish of about 60 mm fork length) of tagged Delta juveniles (Figure 9) is combined with seemingly good data from MacFarlane and Norton (2002) relating weight to fork length (best fit equation of Figure 10(A) differentiated and evaluated at 60 mm fork length yields 0.10 g/mm growth at 2.74 g), the calculated weight growth rate is 0.47 mm/day X 0.10 g/mm = 0.047 g/day. This growth rate

(1.7% of body weight per day for the 60 mm 2.74 g individual) is well below the growth rate of tagged individuals in Vancouver Island estuaries, but much higher and probably much more realistic than the MacFarlane and Norton (2002) estimate above. This growth rate combined with the Cowlitz food conversion efficiency of 1.37 yields a food (dry weight) ingestion rate of **2.33% of live body weight per day**.

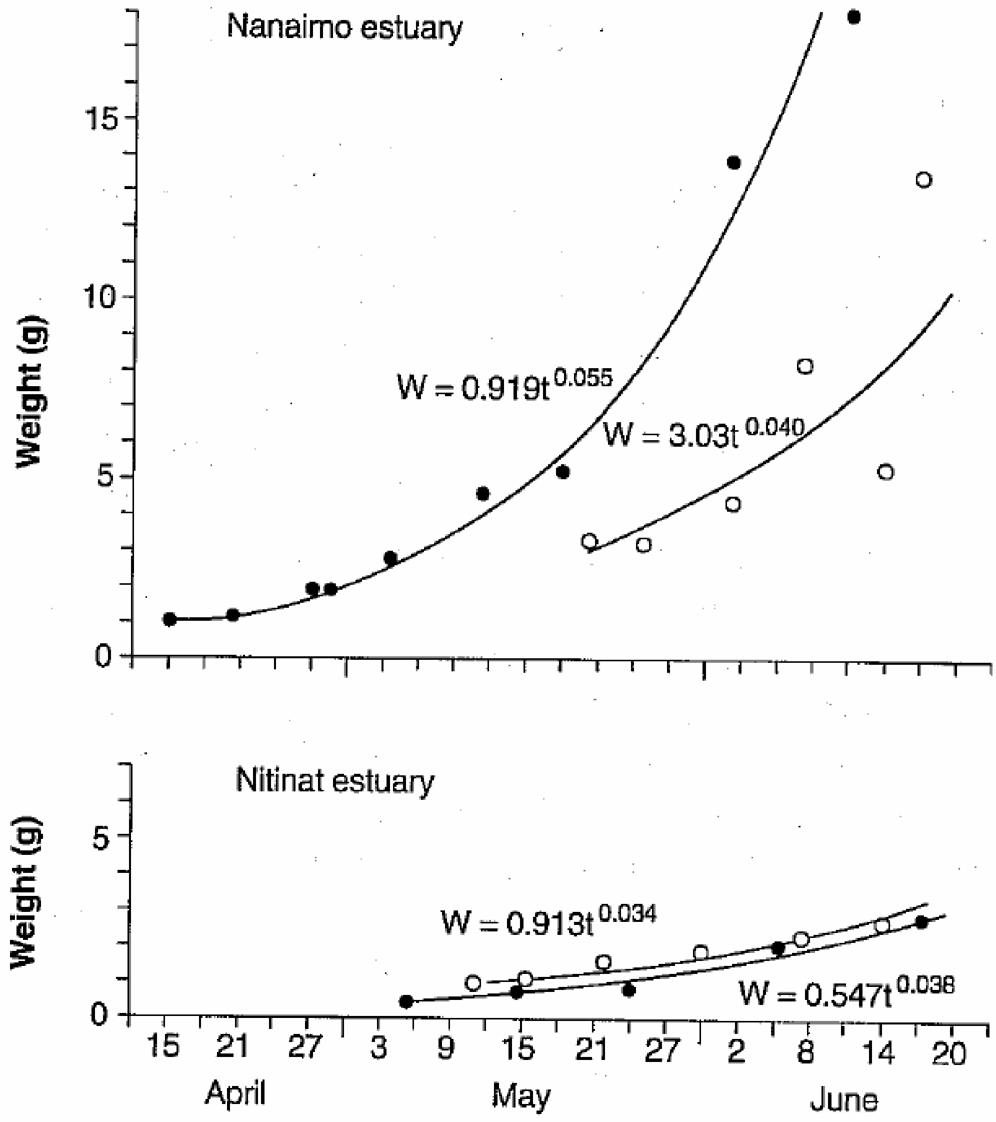

Figure 3. Growth of Chinook salmon fry in the Namaimo and Nitinat River estuaries, Vancouver Island, British Columbia. Closed and open circles designate different tagged groups of fry (Healey 1991).

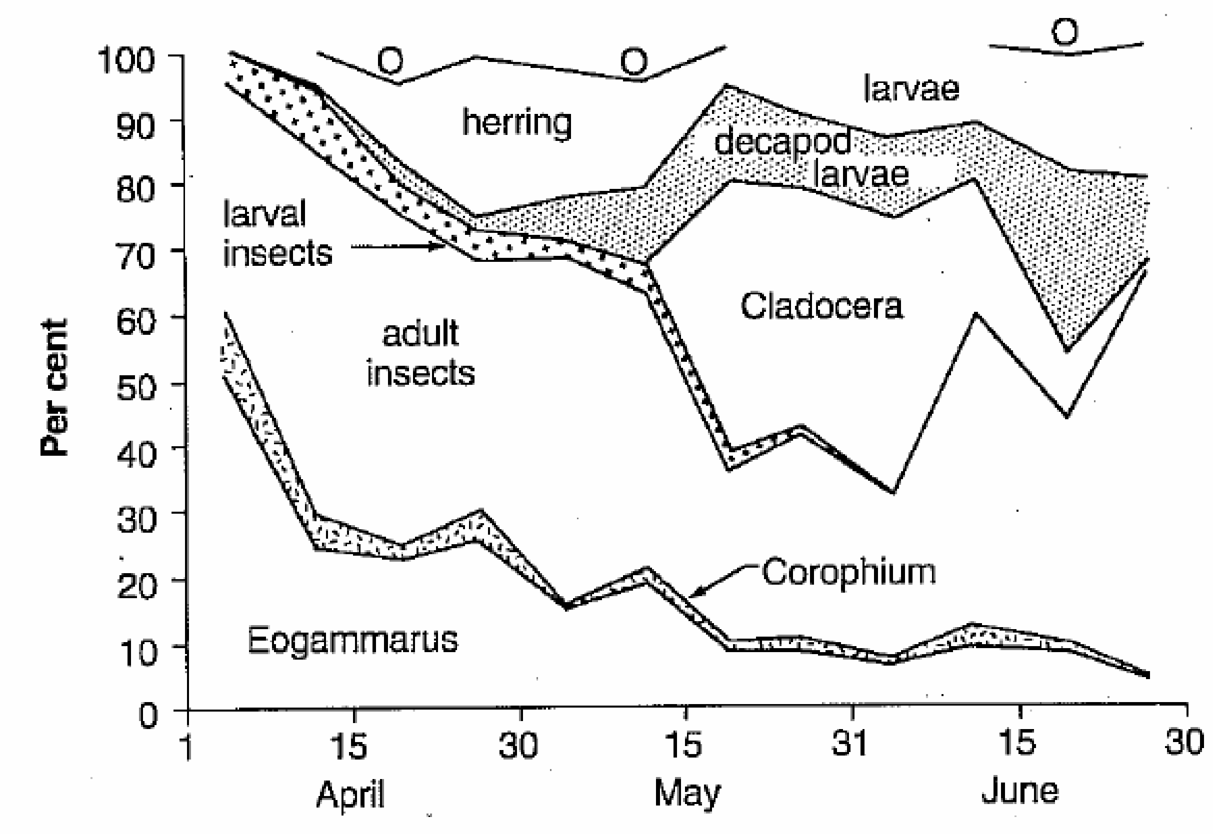

Figure 4. Seasonal variation in the diet of juvenile Chinook salmon in Nitinat Lake, Vancouver Island, British Columbia. At the top of the figure, O denotes other diet items (Healey 1982).

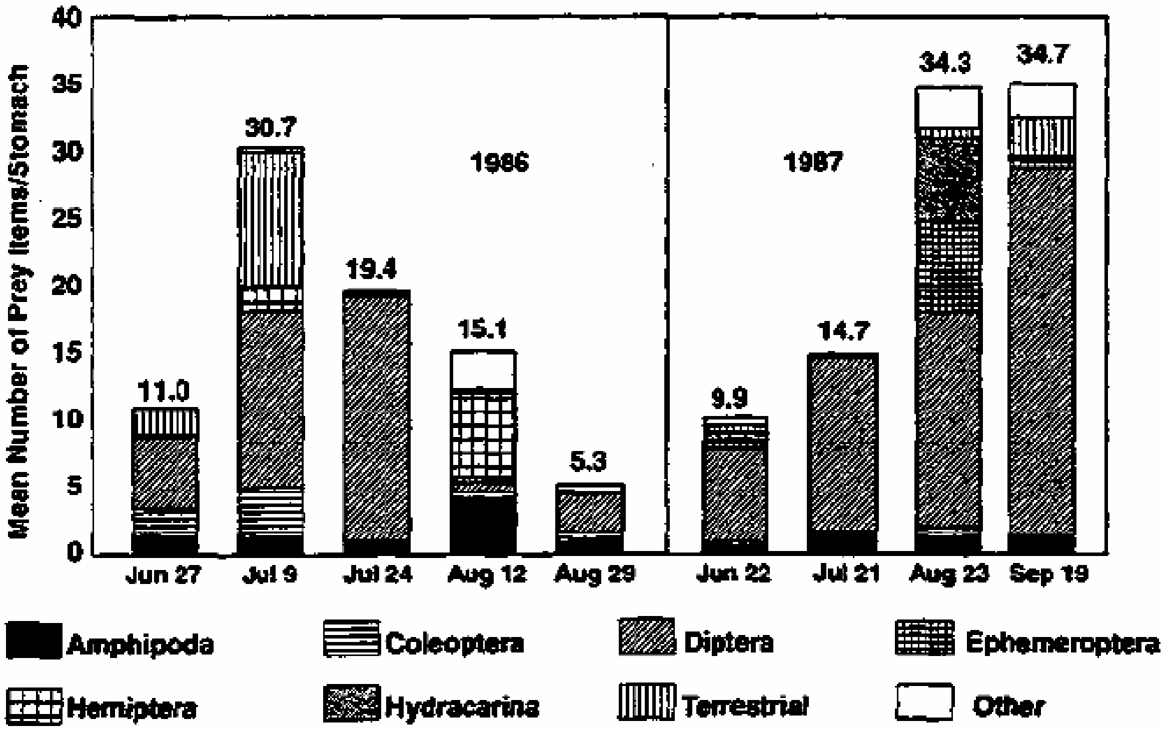

Figure 5. Mean numbers of prey organisms per juvenile Chinook salmon stomach in the Mattole River estuary and lagoon, California, 1986 and 1987 (Fig. 6 in Busby and Barnhart 1995. Salmon fork lengths ranged from 64 to 103 mm.

Food Item	Percent Frequency of Occurrence					
	Sacramento River at Isleton	Sacramento River at Sherman Isl.	Lower San Joaquin River	Flooded Islands	Other Areas	Average and Total
Microplankton	5.0	--	10.9	22.2	0.7	5.0
Mysid shrimp (*Neomysis awatschensis*)	5.0	31.4	31.2	61.1	0.7	14.0
Amphipods (*Corophium*)	16.7	8.6	34.4	61.1	11.0	18.9
Terrestrial Arachnids	1.7	2.9	4.7	--	--	3.1
Tendipedids	46.7	2.9	6.2	11.1	9.7	16.1
Other insects	70.0	60.0	67.2	33.3	89.0	73.9
Fishes	--	8.6	1.6	--	2.1	1.9
Seeds	--	--	--	--	0.7	0.3
Stomachs examined	75	68	88	22	216	469
Stomachs containing food	60	35	64	18	145	322

Figure 6. Stomach contents of young Chinook salmon (322 stomachs containing food out of 469 stomachs examined) caught in the Sacramento-San Joaquin Delta from September 1963 to August 1964 (Table 3 in Sasaki 1966).

Location and prey species	%*N*	%*V*	%*FO*	*IRI*
km 68, Chipps Island (*n*=21)				
Malacostraca				
Decapoda				
Caridean shrimp	2.6	5.7	4.8	39.3
Crab megalopae	7.7	10.0	19.0	336.3
Mysidacea				
Unidentified	7.7	4.6	14.3	175.9
Amphipoda				
Gammaridea	2.6	5.7	4.8	39.8
Corophium spp.	30.8	26.4	33.3	1905
Eusirdae unidentified	2.6	5.7	4.8	39.8
Isopoda				
Gnorimosphaero luta	2.6	0.6	4.8	15.4
Insecta				
Hymenoptera				
Unidentified	2.6	5.7	4.8	39.8
Homoptera				
Aphid	2.6	0.3	4.8	13.9
Diptera				
Flies unidentified	2.6	1.1	4.8	17.8
Culicidae	2.6	2.6	4.8	25.0
Unidentified	7.7	6.3	19.0	266.0
Unidentified	10.3	9.7	14.3	286.0
Algae				
Unidentified	5.2	11.4	9.5	157.7
Unidentified	7.7	4.0	4.8	56.2
km 46, Carquinez Strait (*n*=8)				
Malacostraca				
Mysidacea				
Acanthomysis spp.	3.9	17.0	12.5	257.5
Unidentified	2.0	6.0	12.5	100.
Amphipoda				
Gammaridea	3.9	3.0	12.5	86.3
Cumacea	5.9	5.0	12.5	136.3
Copepoda				
Calanoida				
Eucalanus californicus	5.9	15.0	12.5	261.3
Insecta				
Coleoptera				
Unidentified	2.0	14.0	12.5	200.0
Hemiptera				
Hesperocorixa spp.	72.5	1.0	12.5	918.8
Unidentified	2.0	20.0	12.5	275.0
Pisces				
Unidentified	2.9	19.0	12.5	273.8
km 26, San Pablo Bay (*n*=20)				
Crustacean				
Unidentified	-	2.9	5.0	-
km 26, San Pablo Bay (*n*=20) *continued*				
Malacostraca				
Decapoda				
Crab megalopae	0.5	0.3	35.0	28.0
Mysidacea				
Unidentified	0.5	0.5	5.0	5.0
Cumacean				
Unidentified	61.7	17.9	50.0	3980
Amphipoda				
Gammaridea				
Ampelisca abdita	1.6	1.2	10.0	28.0
Corophium spp.	3.6	23.2	5.0	134.0
Corophium spinicorne	0.5	4.2	5.0	23.5
Maera spp.	0.5	1.5	5.0	10.0
Unidentified	1.0	1.3	3.6	8.3
Cirripedia				
Thoracic				
Barnacle cirri	1.6	1.3	10.0	29.0
Insecta				
Coleoptera				
Unidentified	1.0	1.1	5.0	10.5
Hemiptera				
Unidentified	0.5	0.5	5.0	5.0
Homoptera				
Flatidae	0.5	0.3	5.0	4.0
Diptera				
Unidentified	1.6	1.0	15.0	39.0
Lepidoptera	1.3	1.0	5.0	11.5
Orthoptera	0.5	0.3	5.0	4.0
Unidentified	10.4	12.7	25.0	577.5
Polychaeta				
Phyllodocida				
Nereidae	1.0	4.7	10.0	57.0
Unidentified	1.6	8.2	15.0	147.0
Pisces				
Unidentified larvae	0.5	2.4	5.0	14.5
Unidentified	0.5	1.6	5.0	10.5
Algae				
Unidentified	8.8	8.7	30.0	10.5
Unidentified	0.5	0.3	5.0	4.0
km 3, Central Bay (*n*=10)				
Crustacean				
Unidentified	-	1.2	10.0	-
Malacostraca				
Decapoda				
Crab megalopae	1.2	0.1	10.0	13.0

continued

Figure 7. Stomach contents of juvenile Chinook salmon in the San Francisco estuary and the Gulf of the Farallones (Table 2 in MacFarlane and Norton 2002). %N is the numerical percentage; %V is the percent relative volume; %FO is the frequency of occurrence percentage; and IRI is the index of relative importance, (%N + %V)%FO.

km 3, Central Bay (*n*=10) *continued*				
Cumacean				
Unidentified	45.1	11.1	20.0	1124
Amphipoda				
Gammaridea				
Ampelisca abdita	20.7	8.6	60.0	1758
Copepoda				
Siphonostomatoida	1.2	1.2	10.0	24.0
Cirripedia				
Thoracica				
Barnacle cirri	1.2	1.2	10.0	24.0
Insecta				
Unidentified	8.5	1.2	20.0	194.0
Polychaeta				
Phyllodocida				
Nereidae	1.2	2.5	10.0	37.0
Unidentified	1.2	3.7	10.0	49.0
Pisces				
Unidentified larvae	14.6	55.4	60.0	4200
Unidentified	1.2	10.5	10.0	117.0
Algae				
Unidentified	3.7	3.1	10.0	68.0
Gulf of the Farallones (*n*=23)				
Gastropoda	0.3	0.2	4.3	2.2
Malacostraca				
Decapoda				
Cancer magister (juv)	1.5	6.9	26.0	218.4
Gulf of the Farallones (*n*=23) *continued*				
Crab megalopae	2.1	7.3	21.7	204.0
Crab zoea	6.8	6.6	26.0	348.4
Caridean shrimp	0.9	1.2	13.0	27.3
Euphausiacea				
Thysanoessa gregaria	6.2	2.6	13.0	114.4
Unidentified	24.9	18.5	47.8	2075
Amphipoda				
Caprella californica	0.3	-	4.3	-
Insecta				
Coleoptera				
Unidentified	0.3	0.3	8.7	5.2
Hymenoptera				
Unidentified	2.1	1.2	8.7	2.3
Homoptera				
Aphid	2.1	0.9	17.4	52.2
Diptera				
Culicidae	0.3	0.1	4.3	1.7
Unidentified	5.3	3.7	21.7	195.3
Arachnida				
Araneae				
Unidentified	0.3	0.2	4.3	2.2
Polychaeta				
Phyllodocida				
Nereidae	0.3	0.5	4.3	3.4
Pisces				
Unidentified larvae	46.3	49.9	70.0	6545

Figure 8. Stomach contents of juvenile Chinook salmon in the San Francisco estuary and the Gulf of the Farallones (Table 2 in MacFarlane and Norton 2002). %N is the numerical percentage; %V is the percent relative volume; %FO is the frequency of occurrence percentage; and IRI is the index of relative importance, (%N + %V)%FO.

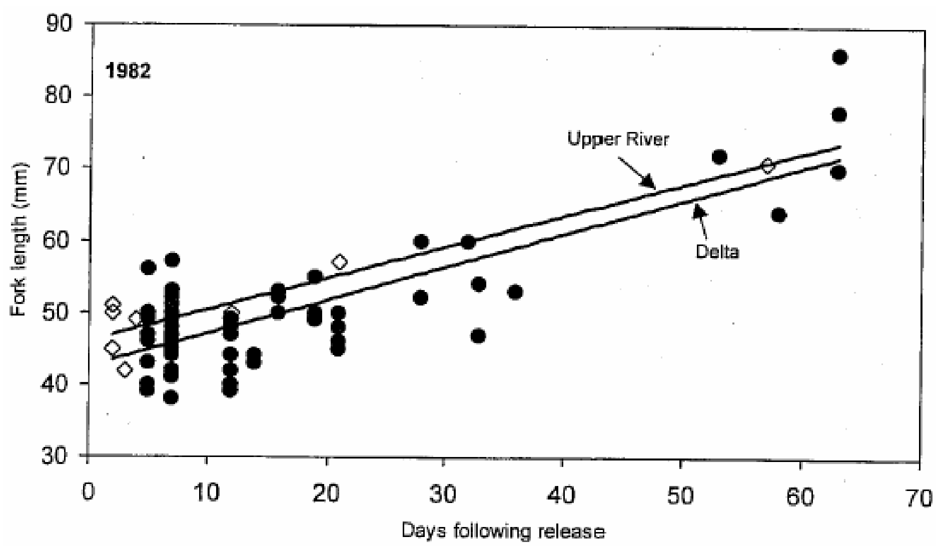

Figure 9. Growth of tagged Chinook salmon fry from the Coleman Fish Hatchery released and recaptured in the Delta (circles) and upper Sacramento River (diamonds) between February 7 and April 28, 1982 (Brandes and McLain 2001).

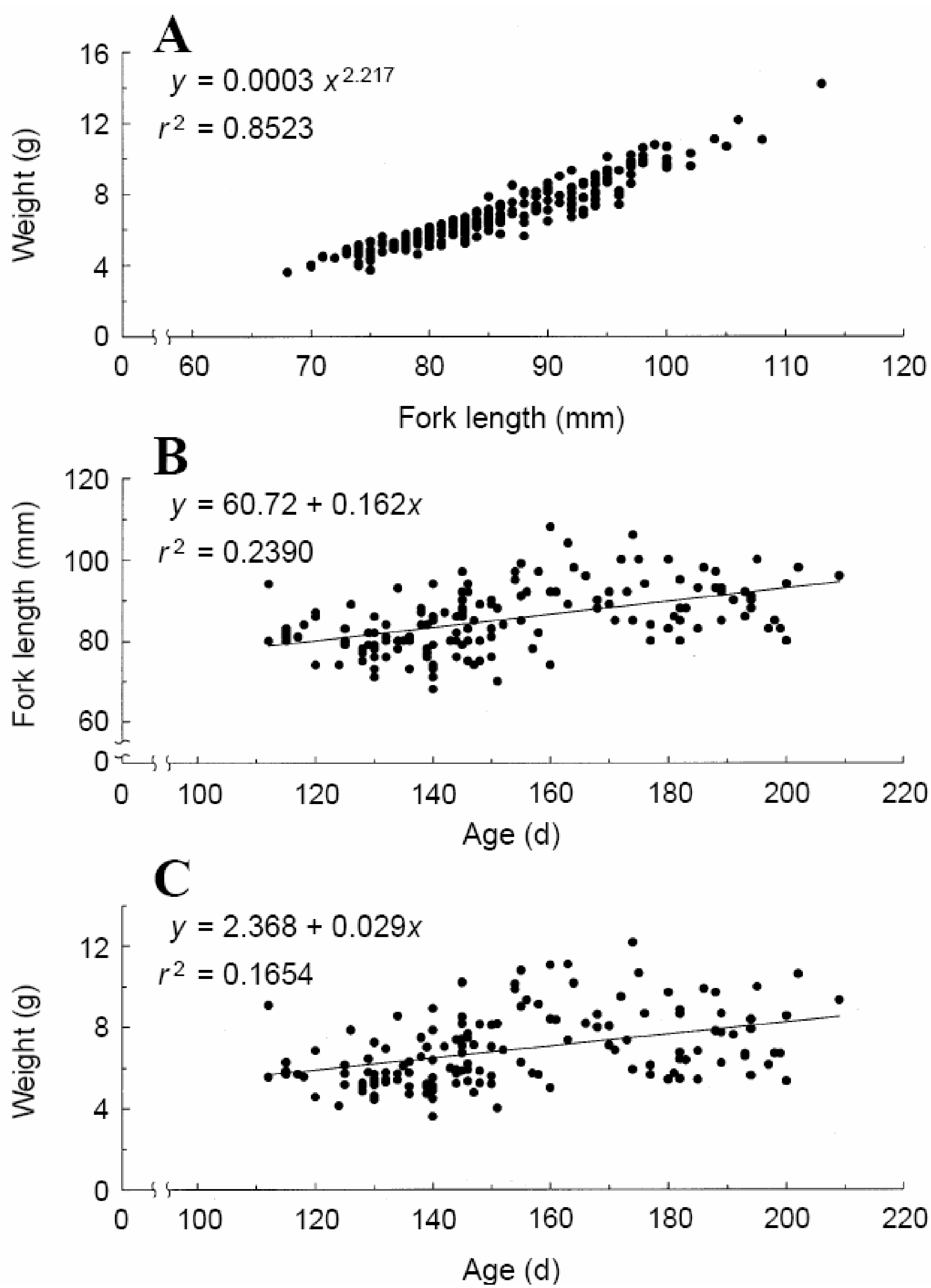

Figure 10. Fork lengths, weights, and ages of all juvenile Chinook salmon collected from the San Francisco Estuary and Gulf of the Farallones in 1997 (Figure 2 in MacFarlane and Norton 2002).

General Life History

Summary: Chinook salmon are anadromous and semelparous. That is, as adults they migrate from a marine environment into the fresh water streams and rivers of their birth (anadromous) where they spawn only once and die (semelparous). Juvenile Chinook may spend from 3 months to 2 years in freshwater after emergence before migrating to estuarine areas as smolts, and then into the ocean to feed and mature. The timing and duration of the migratory movements of Chinook salmon are important in assessing their exposure to selenium and estimating consequent risks. Natal streams and estuary rearing habitat vary seasonally in selenium concentration and the salmon evidently vary in sensitivity to selenium across stages in their life histories.

Freshwater migration: Once their downstream migration begins, Chinook salmon fry may stop migrating and take up residence in the stream for a period of two weeks to a year or more (Healey 1991).

Use of estuarine habitat: On their migration downstream, many Chinook salmon fry take up residence in the river estuary where they rear to smolt size (about 70 mm fork length) before resuming their migration to the ocean. The proportion of fry that rear in the estuary is not known. On Vancouver Island, BC, about 30% of the estimated downstream migrants could be accounted for in the estuary; the fate of the remaining 70% is unknown, but they probably suffered mortality due to unknown agents (Healey 1991). The maximum residence time of Chinook salmon fry in the Sacramento-San Joaquin River delta was estimated to be 64 days in 1980 and 52 days in 1981 (Kjelson *et al.* 1981)

Life history types: Chinook salmon exhibit two generalized freshwater life history types (Healey 1991). "Stream-type" Chinook salmon, enter freshwater months before spawning and reside in freshwater for a year or more following emergence, whereas "ocean-type" Chinook salmon spawn soon after entering freshwater and migrate to the ocean as fry or parr within their first year. Spring-run Chinook salmon exhibit a stream-type life history. Adults enter freshwater in the spring, hold over summer, spawn in fall, and the juveniles typically spend a year or more in freshwater before emigrating. Winter-run Chinook salmon are somewhat anomalous in that they have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter freshwater in winter or early spring, and delay spawning until spring or early summer (stream-type). However, juvenile winter-run Chinook salmon migrate to sea after only four to seven months of river life (ocean-type). Adequate instream flows and cool water temperatures are more critical for the survival of Chinook salmon exhibiting a stream-type life history due to over summering by adults and/or juveniles.

Runs: Salmon runs (separate ESUs) are designated on the basis of adult migration timing; however, distinct runs also differ in the degree of maturation at the time of river entry, thermal regime and flow characteristics of their spawning site, and the actual time of spawning (Myers *et al.* 1998). Both spring-run and winter-run Chinook salmon tend to enter freshwater as immature fish, migrate far upriver, and delay spawning for weeks or months. For comparison, fall-run Chinook salmon enter freshwater at an advanced stage of maturity, move rapidly to their spawning areas on the mainstem or lower tributaries of the rivers, and spawn within a few days or weeks of freshwater entry (Healey 1991).

Run-specific downstream migration: Winter-run Chinook salmon fry begin to emerge from the gravel in late June to early July and continue through October (Fisher 1994). Spring-run Chinook salmon fry emerge from the gravel from November to March and spend about 3 to 15 months in freshwater habitats prior to emigrating to the ocean (Kjelson *et al.* 1981). Post-emergent fry disperse to the margins of their natal stream, seeking out shallow waters with slower currents, finer sediments, and bank cover such as overhanging and submerged vegetation, root wads, and fallen woody debris, and begin feeding on small insects and crustaceans.

When juvenile Chinook salmon reach a length of 50 to 57 mm, they move into deeper water with higher current velocities, but still seek shelter and velocity refugia to minimize energy expenditures. In the mainstems of larger rivers, juveniles tend to migrate along the margins and avoid the elevated water velocities found in the thalweg of the channel. When the channel of the river is greater than 9 to 10 feet in depth, juvenile salmon tend to inhabit the surface waters (Healey 1982). Emigration of juvenile winter-run Chinook salmon past RBDD may begin as early as mid-July, typically peaks in September, and can continue through March in dry years (Vogel and Marine 1991; NMFS 1997). From 1995 to 1999, all winter-run Chinook salmon outmigrating as fry passed RBDD by October, and all outmigrating pre-smolts and smolts passed RBDD by March (Martin *et al.* 2001). The emigration timing of Central Valley spring-run Chinook salmon is highly variable (CDFG 1998). Some fish may begin emigrating soon after emergence from the gravel, whereas others over summer and emigrate as yearlings with the onset of intense fall storms (CDFG 1998). The emigration period for spring-run Chinook salmon extends from November to early May, with up to 69 percent of the young-of-the-year fish outmigrating through the lower Sacramento River and Delta during this period (CDFG 1998).

As Chinook salmon fry and fingerlings mature, they prefer to rear further downstream where ambient salinity is up to 1.5 to 2.5 parts per thousand (Healy 1980, 1982; Levings *et al.* 1986). Juvenile winter-run Chinook salmon occur in the Delta from October through early May based on data collected from trawls, beach seines, and salvage records at the Central Valley Project (CVP) and State Water Project (SWP) pumping facilities (CDFG 1998). The peak of listed juvenile salmon arrivals in the Delta generally occurs from January to April, but may extend into June. Upon arrival in the Delta, winter-run Chinook salmon spend the first two months rearing in the more upstream, freshwater portions of the Delta (Kjelson *et al.* 1981, 1982). Data from the CVP and SWP salvage records indicate that most spring-run Chinook salmon smolts are present in the Delta from mid-March through mid-May depending on flow conditions (CDFG 2000).

Winter-run Chinook salmon fry remain in the estuary (Delta/Bay) until they reach a fork length of about 118 mm (*i.e.*, 5 to 10 months of age) and then begin emigrating to the ocean perhaps as early as November and continuing through May (Fisher 1994; Myers *et al.* 1998). Little is known about estuarine residence time of spring-run Chinook salmon. Juvenile Chinook salmon were found to spend about 40 days migrating through the Delta to the mouth of San Francisco Bay and grew little in length or weight until they reached the Gulf of the Farallones (MacFarlane and Norton 2002). Based on the mainly ocean-type life history observed (*i.e.*, fall-run Chinook salmon) MacFarlane and Norton (2002) concluded that unlike other salmonid populations in the

Pacific Northwest, Central Valley Chinook salmon show little estuarine dependence and may benefit from expedited ocean entry. Spring-run yearlings are larger in size than fall-run yearlings and are ready to smolt upon entering the Delta; therefore, they are believed to spend little time rearing in the Delta.

Risk of selenium exposure: California Central Valley Chinook salmon evidently are among the most sensitive of fish and wildlife to selenium. They are especially vulnerable during juvenile life stages when they migrate and rear in selenium-contaminated Central Valley rivers and the San Francisco Bay/Delta estuary.

In a laboratory experiment, measurements were made of the selenium bioaccumulation, weight and survival of juvenile (initially swim-up larvae) San Joaquin River fall run Chinook salmon that were exposed for 90 days in fresh water to two parallel graded series of dietary selenium treatments (Hamilton *et al*. 1990). In one series, the food was spiked with seleno-DL-methionine (SeMet); in the other series, the source of selenium was mosquitofish collected from the San Luis Drain (SLD), which carried seleniferous agricultural drainwater from a subsurface tile drainage system in the Westlands Water District in the San Joaquin Valley of California. Although the SLD mosquitofish diets may have included other contaminants, such as pesticides, the results of this experiment indicate that, once selenium is incorporated into fish tissue, there is no difference in the tissue concentration-response relationship due to the different sources of selenium (SLD or SeMet). Therefore, all data from both diet series were combined in the analysis presented here.

The effects of selenium on animals (including fish) are well known to be biphasic (beneficial at low doses; toxic at high doses; see for example Hilton *et al*. 1980), and in the Hamilton *et al*. (1990) experiment, the 90-day survival data appear to confirm a biphasic dose-response relationship with respect to the survival endpoint (Figure 11). Therefore, we fitted a biphasic model (Brain and Cousens 1989) to the data by least squares regression. This regression provides a weight-of-evidence estimate of the maximum survival rate (0.7, or 70 percent) of young salmon under these experimental conditions at the estimated optimal selenium concentration in the fish (about 1 µg/g whole body dry weight). It also provides an estimate of the survival rate at any given selenium concentration above the optimum. Any such survival rate estimate can be compared to the maximum survival rate to yield an estimate of the mortality (inverse of survival) specifically attributable to selenium. For example, at a fish tissue concentration of 7.9 µg/g (whole body dry weight) the regression curve predicts a survival of 0.29 (29 percent). As a proportion of the maximum survival this is 0.29/0.7 = 0.41, or 41 percent. Therefore our best weight-of-evidence estimate of the mortality due to selenium toxicity at a tissue concentration of 7.9 µg/g is the inverse of 0.41, which is 0.59, or 59 percent. Similarly, the model predicts that fish with a selenium concentration of 2.5 µg/g (whole body dry weight) after 90 days of exposure would experience 20 percent mortality due to selenium (Figure 11).

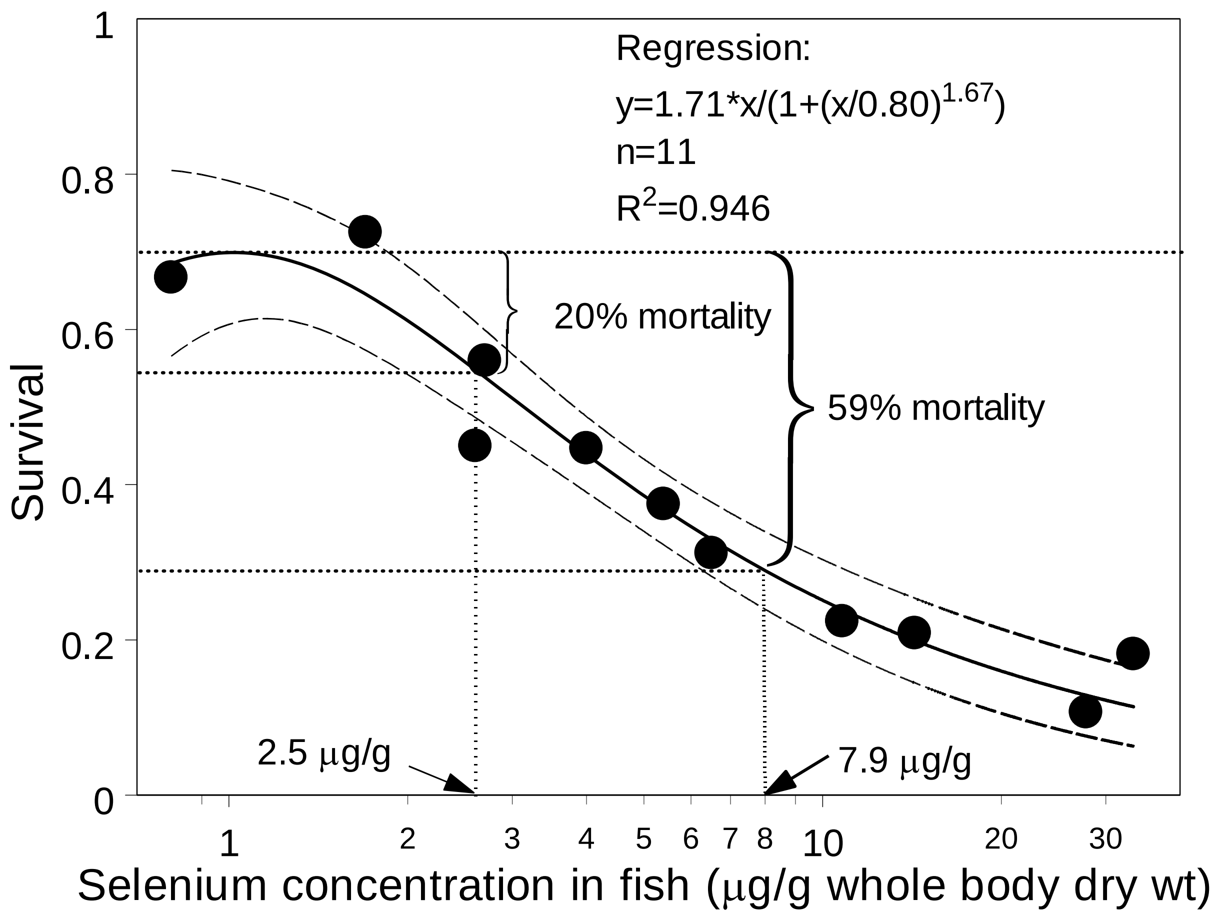

Figure 11. Survival as a function of selenium concentration in tissue of juvenile Chinook salmon after 90 days of exposure to dietary selenium. A biphasic model (Brain and Cousens 1989) was fitted to the data by least squares regression (see text). Dashed lines indicate 95% confidence bands around the regressions in this and following figures.

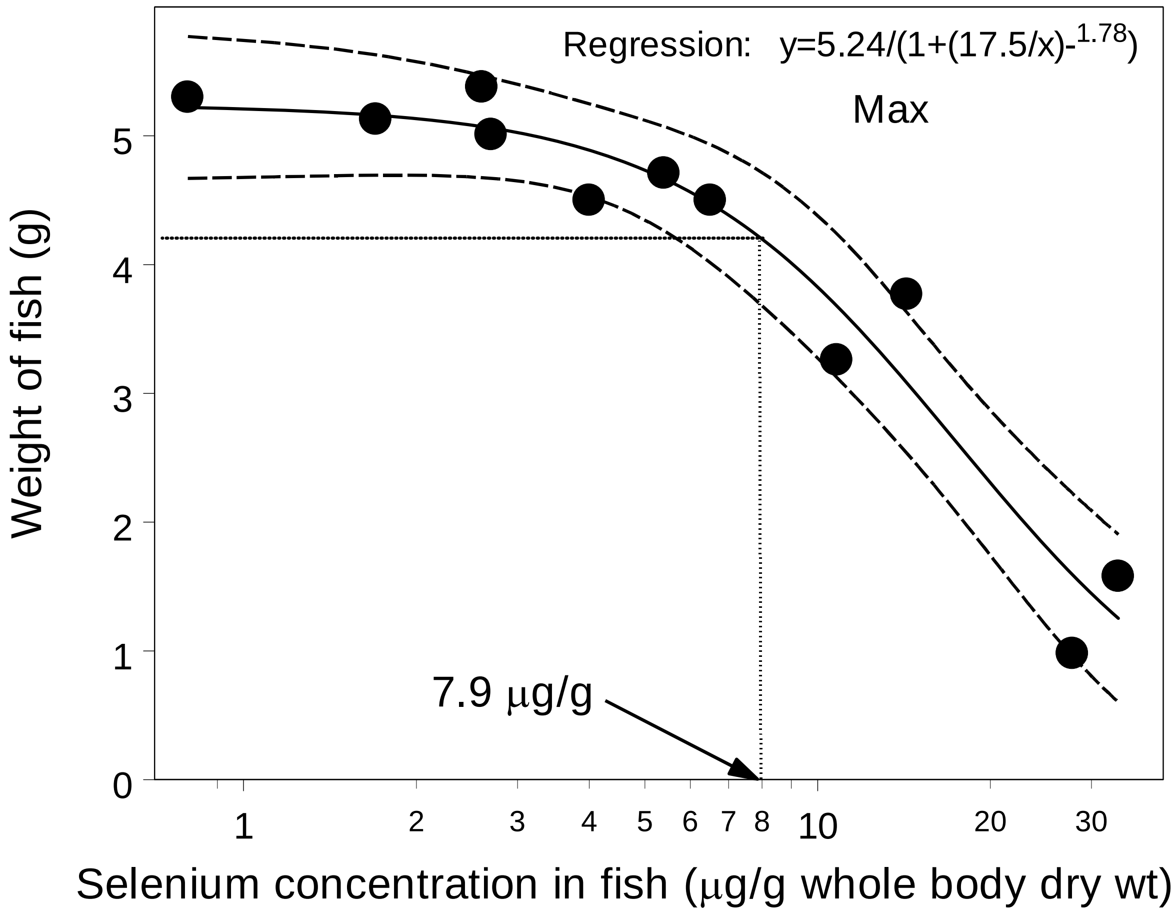

Figure 12. Juvenile fall run Chinook salmon weight 90 days after swim up, in fresh water with dietary exposure to selenium.

In the Hamilton *et al*. (1990) experiment, the weight measurements after 90 days of exposure do not exhibit clear evidence of a biphasic dose-response relationship (**Figure 12**). This suggests that none of the dietary treatments in this experiment was low enough in selenium to be substantially deficient with respect to weight gain. This is consistent with the results of an experiment in which rainbow trout juveniles where exposed to diets spiked with sodium selenate for 20 weeks (Hilton *et al*. 1980). In that experiment, rainbow trout exhibited measurable impairment of growth due to selenium deficiency at tissue selenium concentrations of less than 0.65 µg/g (carcass dry weight). In the Hamilton *et al*. (1990) experiment, the average selenium concentration in fish after 90 days in the lowest selenium treatment was 0.8 µg/g, above the 0.65 µg/g rainbow trout threshold for the effect of selenium deficiency on growth.

The considerations outlined above suggest that a biphasic model is not necessary for adequately describing the Hamilton *et al*. (1990) weight data. Therefore, the weight data were modeled with a log-logistic function, which is commonly used to model monotonic dose-response data. The log-logistic model provides an estimate of the most likely maximum weight (5.24 g), which can be used as the basis of comparison for the predicted weight corresponding to any given elevated selenium concentration. For example, the model projects that a juvenile Chinook salmon tissue concentration of 7.9 µg/g (whole body dry weight after 90 days of exposure to dietary selenium) would most likely be associated with a weight of 4.21 g. This is about a 20 percent reduction from the maximum weight of 5.24 g (**Figure 12**).

We conclude that, 90 days after swimup, Chinook salmon juveniles that bioaccumulate selenium to a concentration of 7.9 µg/g will most likely suffer 59 percent mortality due to selenium (Figure 11), and the survivors will most likely be reduced in weight by 20 percent due to selenium (**Figure 12**).

Hamilton *et al*. (1990) also reared San Joaquin River fall run Chinook salmon fingerlings in reconstituted brackish water with dietary exposure to selenium for 120 days, simulating the rearing of young salmon in the San Francisco estuary. The results of this portion of the experiment indicate that salmon fingerlings with a concentration of 7.9 µg/g selenium after rearing in brackish water for120 days are likely to experience a 2.3 percent reduction in growth due to selenium (Figure 13).

After rearing the young salmon in brackish water, Hamilton *et al*. (1990) simulated the passage of the salmon from the estuary into the ocean by challenging them with 10 days of emersion in reconstituted seawater. The results of this final phase of the experiment suggest that upon entering the ocean, young salmon with a tissue concentration of 7.9 µg/g are likely to experience 15 percent mortality within 10 days, due to selenium (Figure 14).

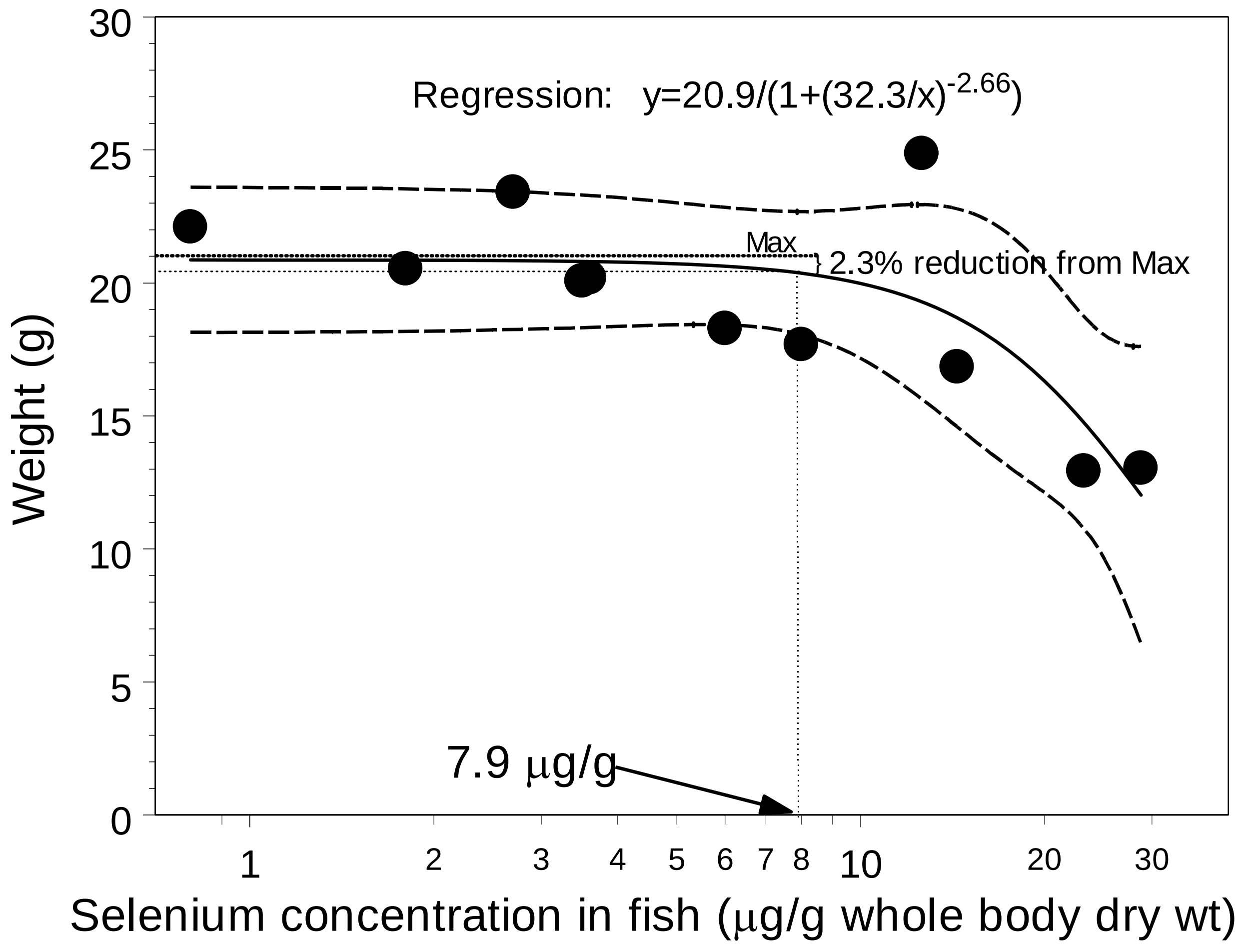

Figure 13. Juvenile fall run Chinook salmon weight after 120 days of rearing in brackish water with dietary exposure to selenium.

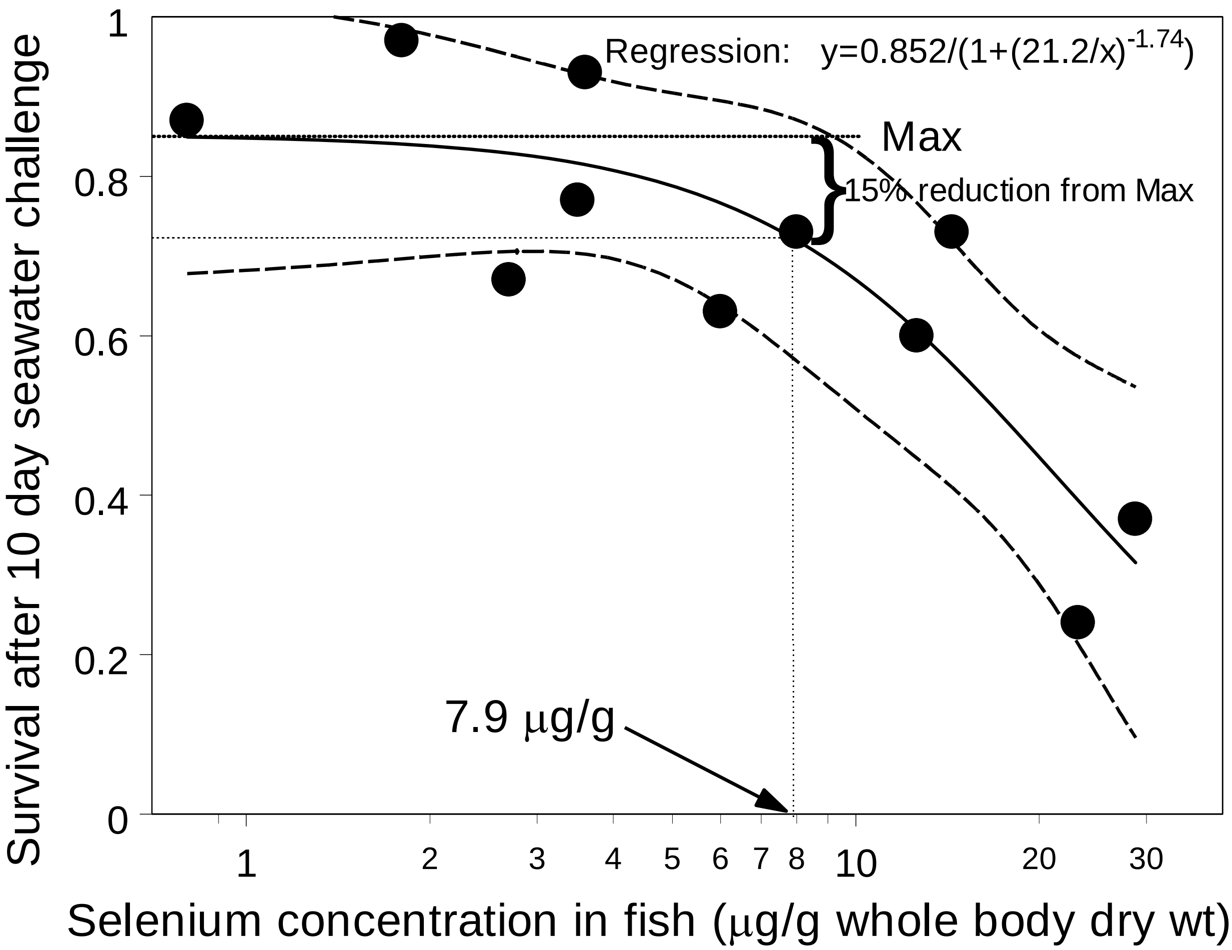

Figure 14. Survival of juvenile fall run Chinook salmon after 10 day seawater challenge following rearing for 120 days in brackish water with dietary exposure to selenium.

Steelhead Trout (*Oncorhynchus mykiss*)

Status: Steelhead trout are the anadromous form of the rainbow trout species. Central California Coast steelhead were listed as threatened under the ESA on August 18, 1997 (62 FR 43937). This Evolutionarily Significant Unit (ESU) includes steelhead in San Francisco and San Pablo Bays and their tributaries (Figure 15). Central Valley steelhead were listed as threatened under the ESA on March 19, 1998 (63 FR 13347). This ESU consists of steelhead populations in the Sacramento and San Joaquin River (inclusive of and downstream of the Merced River) basins in California's Central Valley (Figure 15).

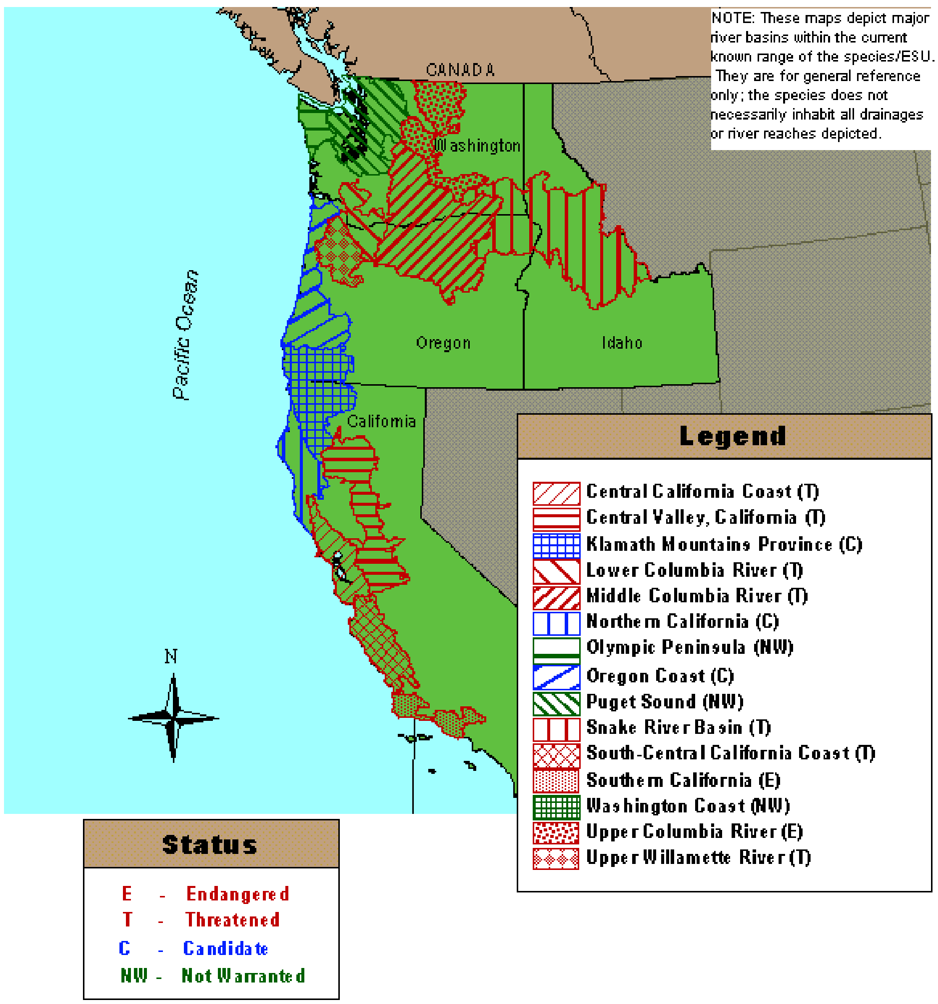

Figure 15. Status of west coast steelhead (http://swr.nmfs.noaa.gov/psd/stlesu.htm).

The breeding of wild steelhead in the Central Valley is mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks and a few wild steelhead are produced in the American and Feather Rivers (McEwan and Jackson 1996).

Until recently, steelhead were thought to be extirpated from the San Joaquin River system. Recent monitoring has detected small self sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be devoid of steelhead (McEwan 2001).

Historically, steelhead were abundant in San Francisco Bay area streams, such as the Napa River and its tributaries; however, populations have declined. The South-Central California Coast steelhead ESU critical habitat includes all river reaches and estuarine areas accessible to listed steelhead in coastal river basins from the Russian River to Aptos Creek, California (inclusive), and the drainages of San Francisco and San Pablo Bays. Also included are all waters of San Pablo Bay westward of the Carquinez Bridge and all waters of San Francisco Bay from San Pablo Bay to the Golden Gate Bridge (65 FR 7764).

Size: Barnhart (1986) reported that steelhead smolts in California range in size from 140 to 210 mm (fork length). This corresponds to a weight range of **31-105 g** according to the fork length-weight regression equation shown in Figure 16 for central California coastal steelhead.

Diet: Juvenile steelhead feed on a wide variety of aquatic and terrestrial insects (Figure 17 and Table 3), and emerging fry are sometimes preyed upon by older juveniles.

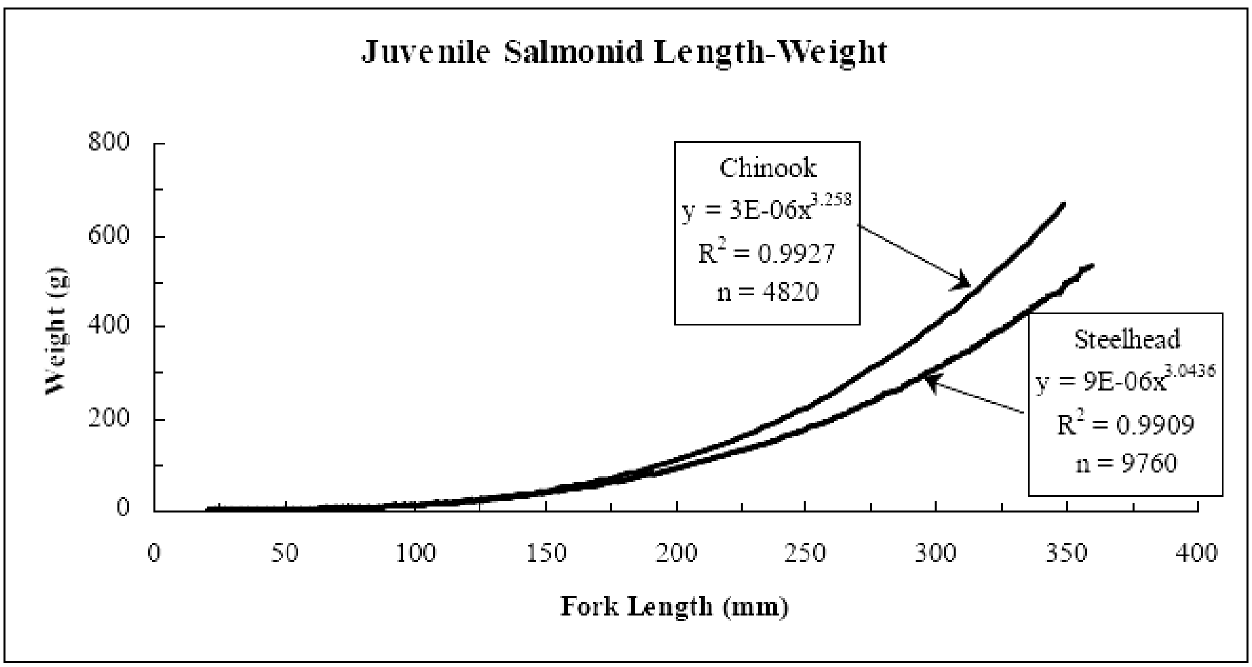

Figure 16. Fork length – weight relationships for Central Valley Chinook salmon and central California coastal steelhead. Chinook were collected in San Francisco Estuary (Fig. 6 in Klimley 2004). Data from MacFarlane (unpublished)

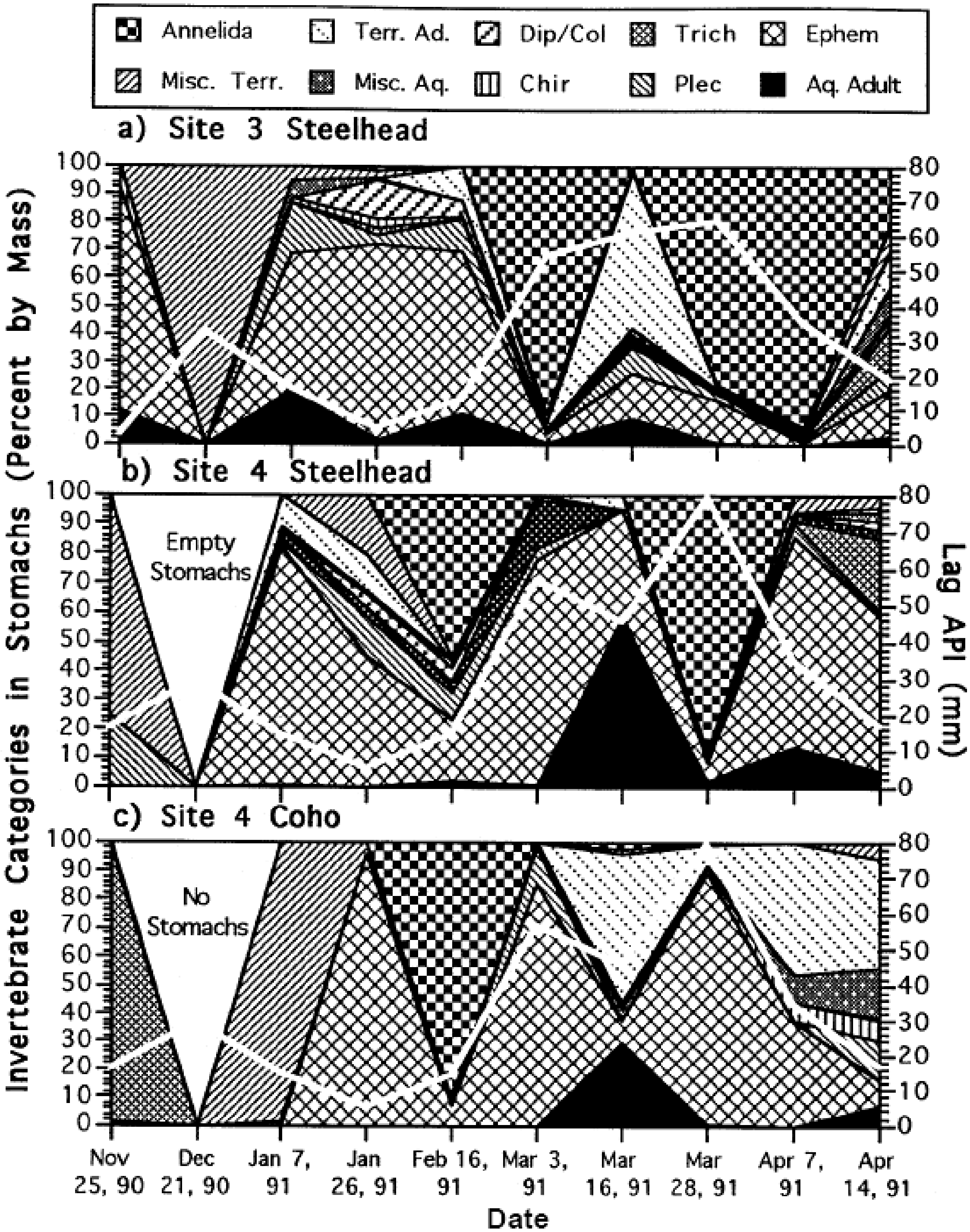

Figure 17. Diet composition, shown as % mass of invertebrate food categories, of juvenile steelhead and coho salmon from November 24, 1990 to April 14, 1991 in two sites along Pudding Creek, California (Fig. 13 in Pert 1987). Aq. Adults = Aquatic Adults, Ephem = Ephemeroptera, Plec = Plecoptera, Trich = Trichoptera, Chir = Chironomidae, Dip/Col = Diptera (other than Chironomidae) and Coleoptera, Misc. Aq. = Miscellaneous Aquatics, Terr. Ad. = Terrestrial Adults, Misc. Terr. = Miscellaneous Terrestrials, and Annelids.

Table 3. Number (No.), percent abundance (Abund.), percent weight (Wt.), and percent occurrence frequency (Occur.) of prey items from stomachs of three steelhead, lower Willamette River, 2002-2003 (Appendix Table 2 from Vile *et al*. 2004). L = larvae; P = pupae.

		Percent		
Prey taxa	No.	Abund.	Wt.	Occur.
Cladocera				
Daphnia spp.	2	10	<1	33
Coleoptera				
Carabidae	2	10	29	33
Diptera				
Chironomidae L	4	19	6	67
Chironomidae P	5	24	12	33
Terrestrial insect	7	33	53	33
Unknown	1	5	<1	33

Food ingestion rate: The food ingestion rates of wild steelhead trout are unknown. However, ingestion rates in the wild may be estimated from ingestion rates in captivity or from growth rates in the wild combined with food conversion efficiencies in captivity.

The growth rates of wild steelhead can be estimated from measurements of steelhead (Table 4) that were marked and recaptured in the Feather River in 2002 (CDWR 2003). Assuming a constant specific growth rate (growth rate as a percentage of body weight), growth is modeled by

$$w = e^{a+bt}$$

Where *w* is fish "weight" (mass) in grams, *t* is time in days, and *a* and *b* are fitted parameters representing respectively the intercept and slope of the straight line relationship when weight is log-transformed:

$$ln\, w = a + bt$$

Given two data points, (w_1, t_1) when a fish was marked, and (w_2, t_2) when the fish was recaptured, then

$$b = \frac{ln\, w_1 - ln\, w_2}{t_1 - t_2}$$

and specific growth rate is given by

$$SpecificGrowthRate = 1 - e^b$$

Table 4. Measurements of all steelhead marked and recaptured in the Feather River in 2002 (CDWR 2003). Specific locations are BP=Bedrock Park, HD=Hatchery Ditch, SR=Steep Riffle.

Tagging Location	Tag Date	Recap Date	Recap Site	Original FL (mm)	Recap FL	Original Wt (g)	Recap Weight	Length Diff	Weight Diff	Days to recap	Length growth/ day (mm)	Weight growth/ day (g)
BP	7/30	8/22	SR	73	103	4.18	16.68	30	12.5	36	0.83	0.35
BP	6/20	8/21	BP	83	113	7.02	17.36	30	10.34	36	0.83	0.29
BP	6/20	7/25	BP	58	82	2.61	6.81	24	4.2	36	0.67	0.12
HD	6/18	6/24	MR	54	64	2.1	2.75	10	0.65	35	0.29	0.02
HD	6/18	7/24	HD	44	68	1.5	4.34	24	2.84	35	0.69	0.08
HD	6/18	6/20	BP	58	58	1.8	1.78	0	-0.02	2	0.00	-0.01
HD	6/18	7/24	HD	47	57	1.3	2.06	10	0.76	6	1.67	0.13
HD	7/24	8/21	HD	68	83	3.93	7.68	15	3.75	28	0.54	0.13
HD	7/24	8/21	HD	59	67	2.32	3.5	8	1.18	28	0.29	0.04
HD	7/24	8/21	HD	55	65	2.05	3.59	10	1.54	28	0.36	0.06
HD	7/24	8/21	HD	52	57	1.55	2.74	5	1.19	28	0.18	0.04
HD	7/24	8/21	HD	55	59	1.7	2.76	4	1.06	28	0.14	0.04
HD	7/24	8/21	HD	69	75	4.12	5.83	6	1.71	28	0.21	0.06
HD	7/24	8/21	HD	53	68	1.82	3.89	15	2.07	28	0.54	0.07
HD	7/24	8/21	HD	53	68	2.15	3.88	15	1.73	28	0.54	0.06
HD	7/24	8/21	HD	63	83	3.06	6.86	20	3.8	28	0.71	0.14
HD	7/24	8/21	HD	69	76	4.51	5.61	7	1.1	28	0.25	0.04
HD	7/24	8/21	HD	70	83	3.79	7.78	13	3.99	28	0.46	0.14
HD	7/24	8/21	HD	63	82	2.83	7.16	19	4.33	28	0.68	0.15
HD	6/18	7/24	HD	55	86	2.4	7.46	31	5.06	62	0.50	0.08
SR	6/25	7/30	SR	94	105	11.9	17.03	11	5.13	23	0.48	0.22
SR	7/30	8/22	SR	109	121	18.08	22.56	12	4.48	23	0.52	0.19

Applying this procedure to the wild steelhead weight data in Table 4 yields the specific growth rates listed in Table 5, with an average specific growth rate of 2.29%.

Table 5. Specific growth rates calculated from the data in Table 4.

b	**exp(*b*)**	**Specific growth rate g/day/g body wt**
0.0384	1.0392	3.92%
0.0252	1.0255	2.55%
0.0266	1.027	2.70%
0.0077	1.0077	0.77%
0.0304	1.0308	3.08%
-0.0056	0.9944	-0.56%
0.0767	1.0797	7.97%
0.0239	1.0242	2.42%
0.0147	1.0148	1.48%
0.02	1.0202	2.02%
0.0203	1.0206	2.06%
0.0173	1.0175	1.75%
0.0124	1.0125	1.25%
0.0271	1.0275	2.75%
0.0211	1.0213	2.13%
0.0288	1.0293	2.93%
0.0078	1.0078	0.78%
0.0257	1.026	2.60%
0.0332	1.0337	3.37%
0.0183	1.0185	1.85%
0.0156	1.0157	1.57%
0.0096	1.0097	0.97%
	Average:	**2.29%**

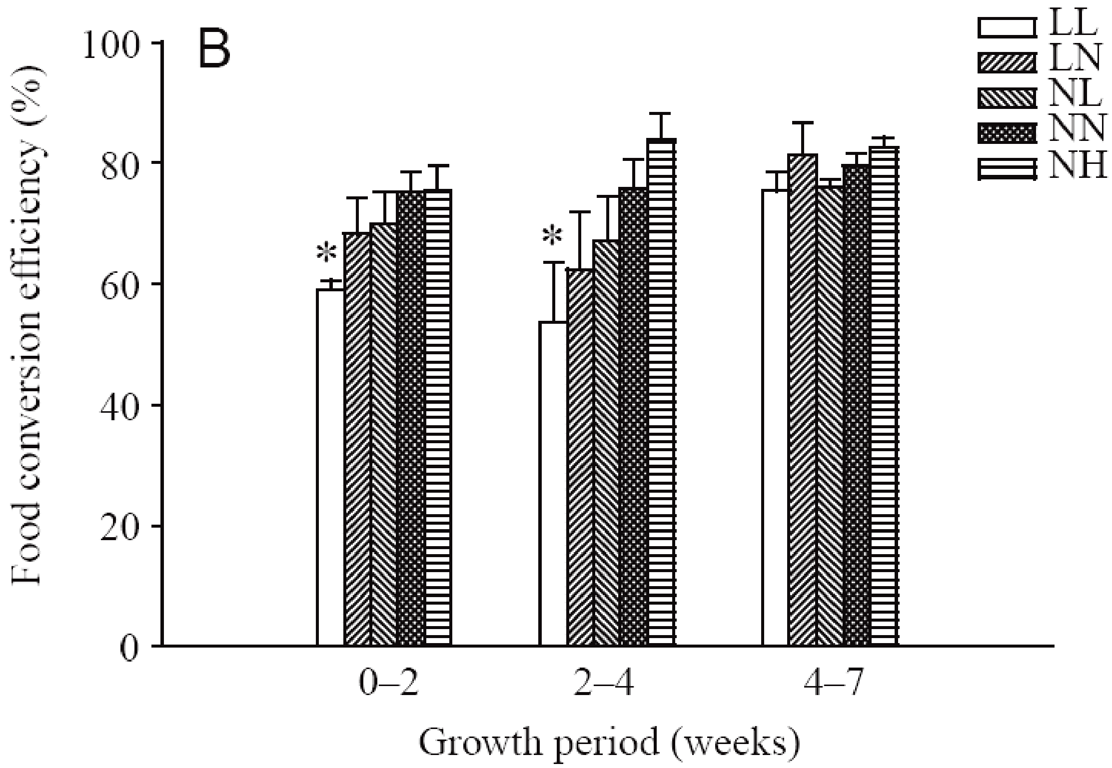

Figure 18. Effects of waterborne and dietary copper exposure on food conversion efficiency in juvenile rainbow trout obtained from Humber Spring Trout Hatchery, Mono Mills, Ontario, Canada (Fig. 1(B) in Kamunde *et al*. 2002). Values are means ± S.E.M. on a per tank basis, N=3 per data point. LL, low waterborne Cu and low dietary Cu; LN, low waterborne Cu and normal dietary Cu; NL, normal waterborne Cu and low dietary Cu; NN, normal waterborne Cu and normal dietary Cu; NH, normal waterborne Cu and high dietary Cu level. *Significant difference relative to group NN on normal water Cu and normal dietary Cu.

Food conversion efficiencies of about 0.8 were reported for captive juvenile rainbow trout raised with exposure to low concentrations of copper in ambient water and food (Kamunde *et al*. 2002, Figure 18). This agrees with the food conversion efficiency of 0.80 recorded by Bear (2005) for rainbow trout raised for 60 days in 75 liter aluminum test tanks at 20˚ C (Table 6), a water temperature commonly encountered in the shallow waters of the Delta.

Table 6. Feed consumption (±SE), and feed conversion efficiency (±SE) for westslope cutthroat (WCT) and rainbow trout (RBT). Rainbow trout were raised from eggs obtained from the Ennis National Fish Hatchery in Montana (Bear 2005).

Temperature	Feed consumption (% body weight)		Feed conversion efficiency (g growth / g consumed)	
°C	WCT	RBT	WCT	RBT
8	0.73(0.04) A	0.86(0.07) A	1.27(0.05) A	1.17(0.03) A
12	0.87(0.04) A	1.03(0.05) A	1.33(0.06) A	1.15(0.04) A
14	0.99(0.03) A	1.08(0.05) A	1.27(0.03) A	1.15(0.02) A
16	1.13(0.06) B	0.92(0.10) A	1.13(0.03) A	1.07(0.02) A
20	0.75(0.10) C	0.93(0.09) A	0.45(0.08) B	0.80(0.06) B +

If we use the food conversion efficiency (FCE) of 0.80 and the average specific growth rate (SGR) of 2.29% (Table 5) to calculate the food ingestion rate (FIR) for steelhead trout:

FIR = SGR/FCE
FIR = 2.29%/0.80
FIR = 2.86% (dry weight percent of live body weight per day)

This ingestion rate is substantially higher than the rate of consumption (0.93% of body weight per day at 20˚ C) of rainbow trout fed to satiation in the experiment reported by Bear (2005). It is also substantially higher than the rate at which Coos River Winter Steelhead smolts are reported to be fed (1% of body weight per day) at the Bandon Hatchery in Oregon (ODFW 2005), and the rate at which rainbow trout are fed (0.8% of body weight per day) at the Lake Roosevelt Net Pen Program in Washington (LRDA 2000).

If rearing steelhead actively seek out cooler waters that are closer to the temperature of optimal rainbow trout growth (about 13-14˚ C, Figure 19, see Natural History below), then the Bear (2005) experiment indicates that food conversion efficiency would be about 1.15 (Table 6). This is close to the FCE (1.0) reported to be typical for rainbow trout at the Lake Roosevelt Net Pen Program (LRDA 2000). Using an FCE of 1.15:

FIR = SGR/FCE
FIR = 2.29%/1.15
FIR = **1.99% (dry weight percent of live body weight per day)**

This is closer to the FIR actually recorded in the Bear (2005) experiment (1.08 at 14˚ C, Table 6) and closer to hatchery feeding rates, and it may be more likely to represent typical food ingestion rates of steelhead rearing in the range of water temperatures experienced in the Delta.

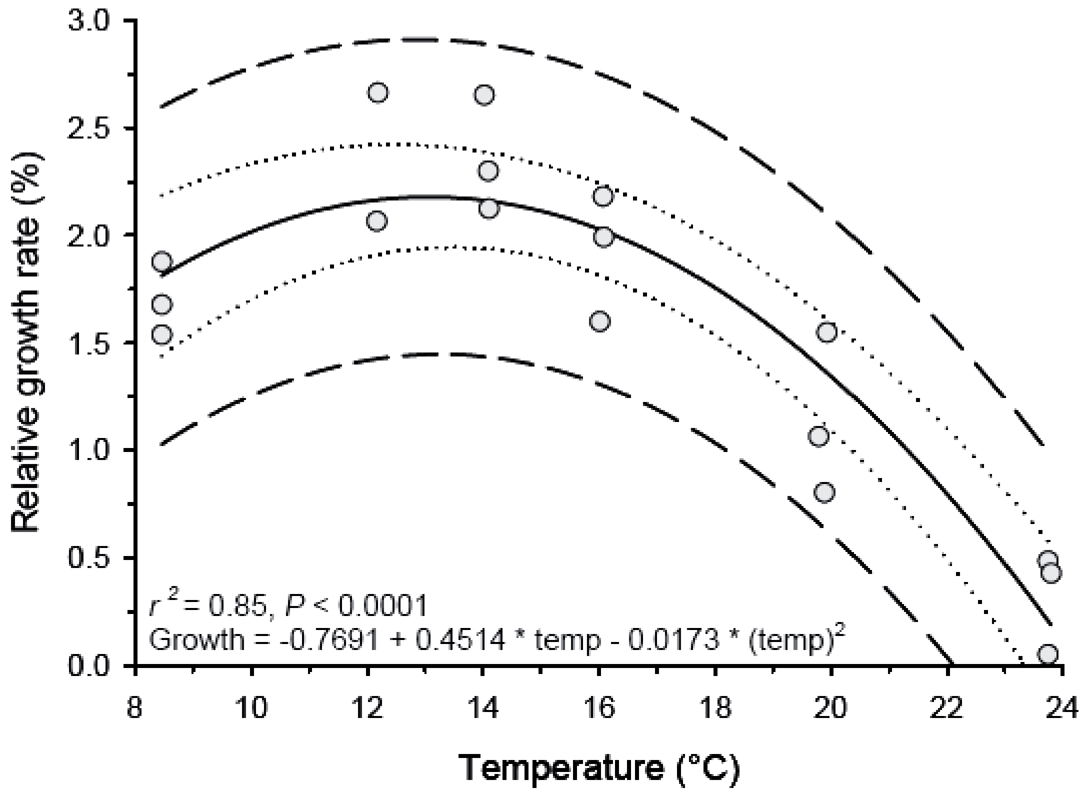

Figure 19. Growth of age-1 rainbow trout over 60 days in relation to temperature (Figure 8 bottom in Bear 2005). Each circle represents the relative growth rate (%) per tank with three tanks tested at each temperature. Dotted lines indicate the 95% confidence interval of the regression line and dashed lines indicate the 95% confidence interval of the data.

General Life History: Steelhead can be divided into two life history types, stream-maturing and ocean-maturing, based on their state of sexual maturity at the time of river entry and the duration of their spawning migration. Stream-maturing steelhead enter freshwater in a sexually immature condition and require several months to mature and spawn, whereas ocean-maturing steelhead enter freshwater with well-developed gonads and spawn shortly after river entry. These two life history types are more commonly referred to by their season of freshwater entry (*i.e.* summer [stream-maturing] and winter [ocean-maturing] steelhead). Only winter steelhead currently are found in the rivers and streams of Central Valley and San Francisco Bay area (McEwan and Jackson 1996).

Winter steelhead generally leave the ocean from August through April, and spawn between December and May (Busby *et al.* 1996). Timing of upstream migration is correlated with higher flow events, such as freshets or sand bar breaches, and associated lower water temperatures. In general, the preferred water temperature for adult steelhead migration is 46 °F to 52 °F (McEwan and Jackson 1996; Myrick 1998; and Myrick and Cech 2000).

Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby *et al.* 1996). However, it is rare for steelhead to spawn more than twice before dying; most that do so are females (Nickleson *et al.* 1992; Busby *et al.* 1996). Iteroparity is more common among southern steelhead populations than northern populations (Busby *et al.* 1996).

Although one-time spawners are the great majority, Shapovalov and Taft (1954) reported that repeat spawners are relatively numerous (17.2 percent) in California streams. Most steelhead spawning takes place from late December through April, with peaks from January though March (Hallock *et al.* 1961). Steelhead spawn in cool, clear streams featuring suitable gravel size, depth, and current velocity, and may spawn in intermittent streams as well (Everest 1973; Barnhart 1986).

The length of the incubation period for steelhead eggs is dependent on water temperature, dissolved oxygen concentration, and substrate composition. In late spring and following yolk sac absorption, fry emerge from the gravel and actively begin feeding in shallow water along stream banks (Nickelson *et al.* 1992).

Steelhead rearing during the summer takes place primarily in higher velocity areas in pools, although young-of-the-year also are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Productive steelhead habitat is characterized by complexity, primarily in the form of large and small woody debris. Cover is an important habitat component for juvenile steelhead both as velocity refugia and as a means of avoiding predation (Shirvell 1990; Meehan and Bjornn 1991). Some older juveniles move downstream to rear in large tributaries and mainstem rivers (Nickelson *et al.* 1992). Juveniles feed on a wide variety of aquatic and terrestrial insects (Chapman and Bjornn 1969), and older juveniles sometimes prey upon emerging fry.

Steelhead generally spend two years in freshwater before emigrating downstream (Hallock *et al.* 1961; Hallock 1989). Rearing steelhead juveniles prefer water temperatures of 45˚ F to 58˚ F and have an upper lethal limit of 75˚ F. They can survive up to 81˚ F with saturated dissolved oxygen conditions and a plentiful food supply.

Juvenile steelhead emigrate episodically from natal streams during fall, winter, and spring high flows. Emigrating Central Valley steelhead use the lower reaches of the Sacramento River and the Delta for rearing and as a migration corridor to the ocean. Some may utilize tidal marsh areas, non-tidal freshwater marshes, and other shallow water areas in the Delta as rearing areas for short periods prior to their final emigration to the sea. Barnhart (1986) reported that steelhead smolts in California range in size from 140 to 210 mm (fork length). Hallock *et al.* (1961) found that juvenile steelhead in the Sacramento River Basin migrate downstream during most months of the year, but the peak period of emigration occurred in the spring, with a much smaller peak in the fall.

Risk of selenium exposure: Because steelhead are regarded as a life-history variant or "form" of the rainbow trout species, studies of the non-anadromous form of rainbow trout may provide a good indication of the risks of the exposure of steelhead to selenium. Such studies indicate that rainbow trout are among the more sensitive of fish to selenium. One of these studies examined the effects of selenium on fry of rainbow and brook trout exposed in streams in Alberta, Canada (Holm 2002, Holm *et al*. 2003). In summary, this study indicates that maternal selenium would result in 20 percent mortality of fry if female rainbow trout have a tissue selenium concentration of 2.93 µg/g wholebody dry weight (Figure 20). Among the swimup survivors, various deformities were also associated with elevated selenium in the eggs from which the fry hatched.

For example, both edema (Figure 21) craniofacial deformities (Figure 22) increase sharply above an egg selenium concentration of about 10 µg/g (wet weight), corresponding to a maternal tissue concentration of about 6.6 µg/g (whole body dry weight; for details, see USFWS 2005).

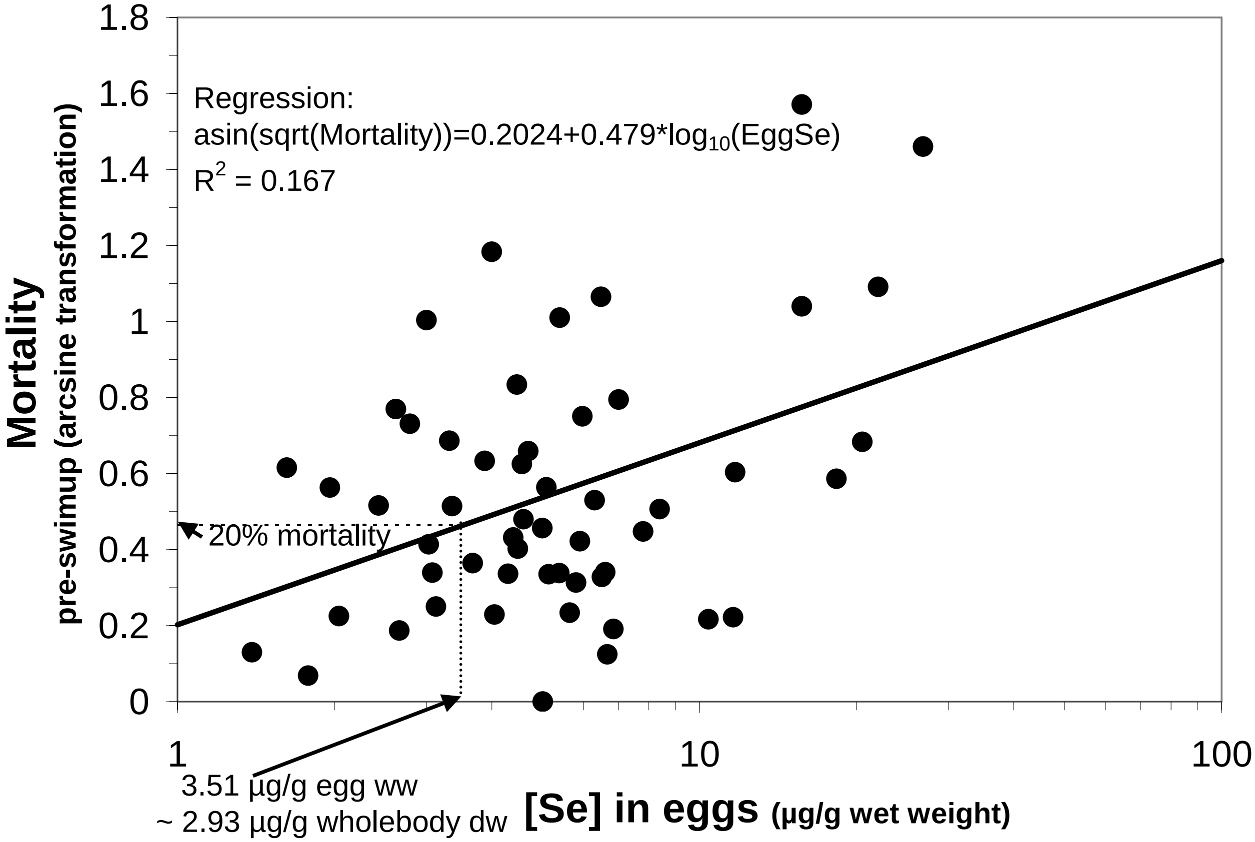

Figure 20. Relationship between selenium in rainbow trout eggs and mortality of eggs and fry by swimup stage. The arcsine transformation is applied to mortality data, as appropriate for linear regressions with percents or proportions (Sokol and Rohlf 1981). Data are from the years 2000-2002.

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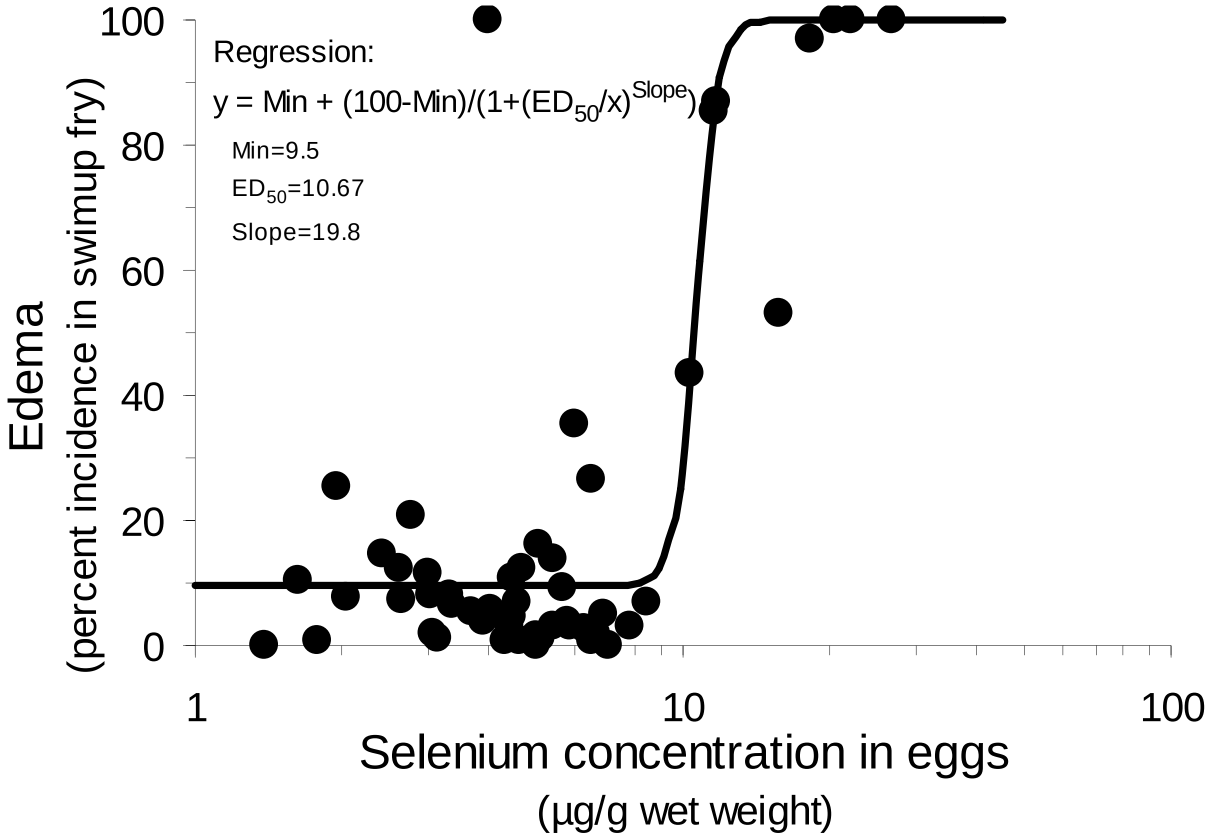

Figure 21. Relationship between selenium in rainbow trout eggs and edema in surviving swimup fry. The eggs were from rainbow trout collected in the McLeod River drainage, Alberta, Canada, 2000-2002 (Jodi Holm pers. com.).

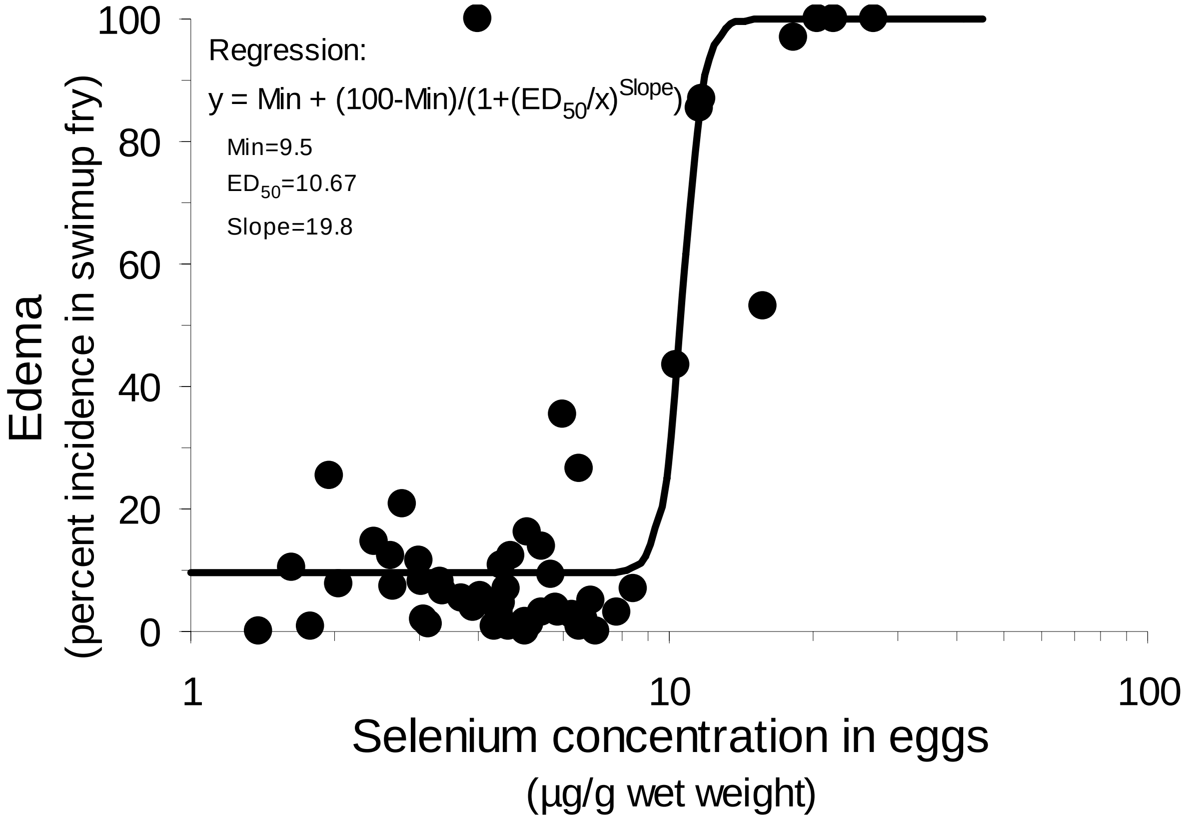

Figure 22. Relationship between selenium in rainbow trout eggs and craniofacial deformities in surviving swimup fry. The eggs were from rainbow trout collected in the McLeod River drainage, Alberta, Canada, 2000-2002 (Jodi Holm pers. com.).

Another laboratory experiment monitored the growth of juvenile rainbow trout exposed for 20 weeks to a diet spiked with selenium in the form of sodium selenite (Hilton *et al*. 1980). This experiment indicates that, relative to optimal selenium exposure, a weight reduction of 20 percent would be associated with a tissue selenium concentration of 2.15 µg/g (carcass dry weight) (Figure 23).

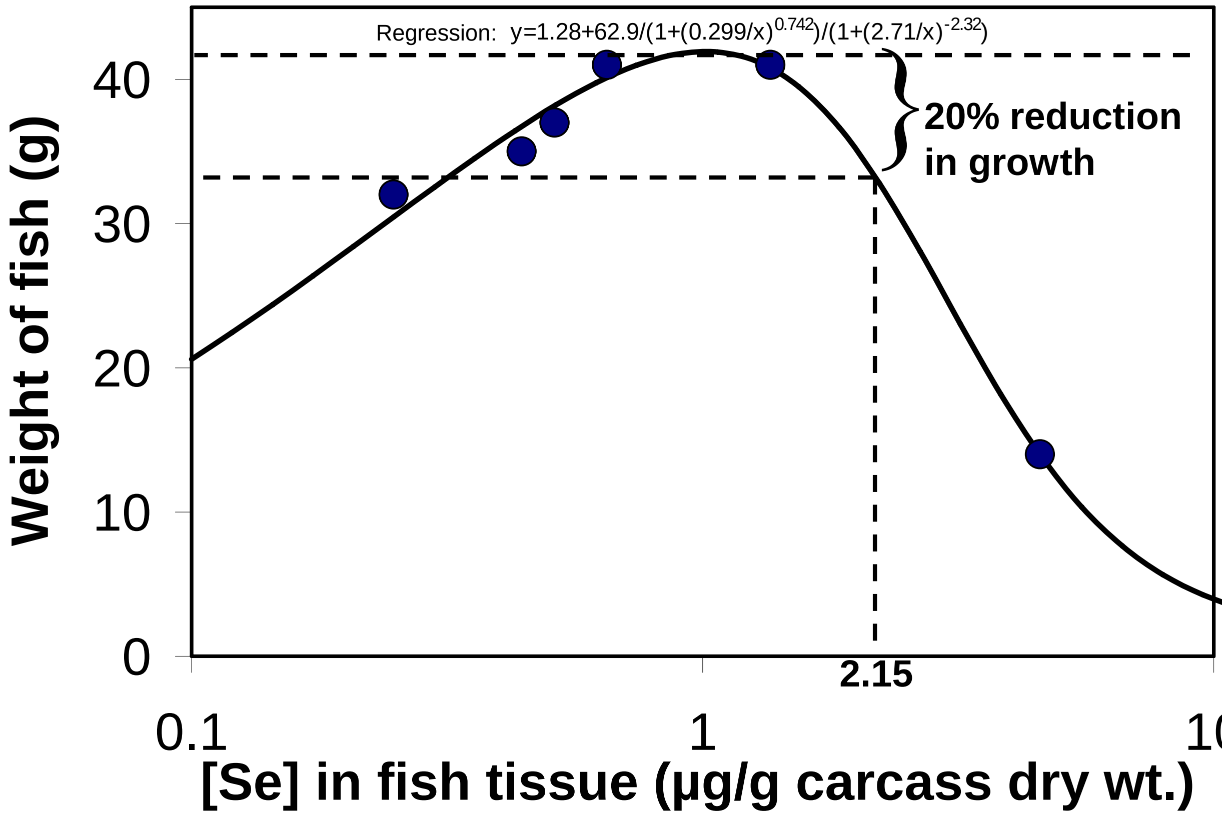

Figure 23. Average weights of juvenile rainbow trout after 20 weeks of exposure to diets spiked with sodium selenite (Hilton *et al*. 1980). The data were fitted with a biphasic model (Beckon *et al*. in prep.). In the model it was assumed that at extremely high and extremely low selenium concentrations, the fish would have failed to grow at all, i.e. they would have remained at the initial average weight of 1.28 g. Carcass concentrations are from Fig. 2 of Hilton *et al*. 1980.

Green Sturgeon (*Acipenser medirostris*)

Status: The southern distinct population segment, or DPS, of north American green sturgeon was federally listed as threatened under the Endangered Species Act on Apr. 7, 2006 (71 FR 17757). The range of the southern DPS extends southward from the Eel River, in northern alifornia, and includes the green sturgeon inhabiting the San Francisco Bay/Delta estuary.

Size: Green sturgeon caught in the commercial fishery has declined in size over the years. In the 1960s, mean size of green sturgeon landed ranged between 17 and 19 kg (37-42 pounds), while since 1980, mean weight has usually been between 12 and 14 kg (26-31 pounds) (USFWS 1996). Here we use the midpoint (**13 kg**) of this more recent range of means.

Diet: Sturgeon feed primarily on benthic (bottom) organisms such as worms, mollusks, and crustaceans. Juveniles in the Sacramento-San Joaquin Delta feed on opossum shrimp

(*Neomysis inercedis*) and amphipods (*Corophium* sp.) (Table 7, Radtke 1966).

Table 7. Stomach contents of 74 green sturgeon caught in the Sacramento-San Joaquin Delta with gill nets and otter trawl from September 1963 through August 1964 (Radtke 1966).

	19-39 cm Green Sturgeon										40-57 cm Green Sturgeon									
	Fall		Winter		Spring		Summer		All year		Fall		Winter		Spring		Summer		All year	
Food Item	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.
Mysid shrimp (*Neomysis awatschensis*)	--	--	--	--	40.0	38.3	75.0	43.0	50.0	33.2	--	--	--	--	100.0	99.0	78.4	86.6	78.0	86.4
Amphipods (*Corophium*)	100.0	100.0	--	--	100.0	54.7	100.0	57.0	100.0	63.9	100.0	100.0	--	--	33.3	1.0	81.1	13.4	78.0	13.6
Unidentified shrimp	--	--	--	--	10.0	7.0	--	--	3.8	2.8	--	--	--	--	--	--	--	--	--	--
Stomachs examined	8		0		10		12		30		1		0		3		40		44	
Stomachs containing food	4		0		10		12		26		1		0		3		37		41	

Food ingestion rate: The food ingestion rate of green sturgeon in the wild is unknown, but 24 green sturgeon (average weight: 150 g) that were reared in captivity at 19˚C and fed to satiation on a diet of commercial Silvercup trout pellets ingested food at a rate of **2% of live fish body weight per day** (Mayfield and Cech 2004)

General life history: The ecology and life history of green sturgeon have received comparatively little study, evidently because of their generally low abundance and their low commercial and sport-fishing value in the past. The adults are more marine than white sturgeon, spending limited time in estuaries or fresh water.

Green sturgeon migrate up the Klamath River between late February and late July. The spawning period is March-July, with a peak from mid-April to mid-June (Emmett *et al.* 1991). Spawning times in the Sacramento River are probably similar, based on times when adult sturgeon have been caught there. Spawning takes place in deep, fast water. Female green sturgeon produce 60,000-140,000 eggs (Moyle 1976). Based on their presumed similarity to white sturgeon, green sturgeon eggs probably hatch around 196 hours (at 12.7 degrees Celsius [54.9 degrees Fahrenheit]) after spawning, and larvae should be 8-19 millimeters (0.3-0.7 inch) long. Juveniles likely range in size from 2.0-150 centimeters (1-59 inches) (Emmett *et al.* 1991). Juveniles migrate out to sea before 2 years of age, primarily during summer-fall (Emmett et al. 1991). Length-frequency analyses of sturgeon caught in the Klamath Estuary by beach seine indicate that most green sturgeon leave the system at lengths of 30-70 centimeters (12-28 inches), when they are to 4 years old, although a majority leave as yearlings (USFWS 1996). They remain near estuaries at first, but can migrate considerable distances as they grow larger (Emmett *et al.* 1991). Individuals tagged by DFG in San Pablo Bay (part of the San Francisco Bay system) have been recaptured off Santa Cruz, California, in Winchester Bay on the southern Oregon coast, at the mouth of the Columbia River and in Gray's Harbor, Washington (Chadwick 1959; Miller 1972). Most tags for green sturgeon in the San Francisco Bay system have been returned from outside that estuary (D. Kohlhorst, DEG, personal communication, cited in USFWS 1996).

Risk of selenium exposure: Little is known of the risk of selenium to green sturgeon, but white sturgeon (*Acipenser transmontanus*), a representative surrogate species for the green sturgeon, have been the subject of detailed studies within the San Francisco Bay estuary (e.g., Kohlhorst *et* *al*. 1991, Linares *et al*. 2004, Linville 2006). See the “Risk to selenium exposure” section for white sturgeon below.

White Sturgeon (*Acipenser transmontanus*)

Status: According to the World Conservation Union (Duke *et al*. 2004), in general the white sturgeon species is not threatened, but some subpopulations are endangered (Kootenai River and Upper Fraiser River) or critically endangered (Nechako River, Upper Columbia River). The Kootenai River population of the white sturgeon in Montana and Idaho was federally listed as endangered under the Endangered Species Act on September 6, 1994 (59 FR 45989). The California Department of Fish and Game (CDFG) established a daily bag and possession limit of one fish, which must be between 46 and 72 inches total length (CDFG 2007). Temporary (120 days) emergency regulations issued by the CDFG in March 2006 restricted fishing in California to individuals between 46 and 56 inches total length.

Size: The white sturgeon is the largest North American fish. A white sturgeon weighing approximately 682 kg was taken from the Snake River, Idaho in 1898. Individuals from landlocked subpopulations tend to be smaller (Duke *et al*. 2004). The average length of 390 white sturgeon captured in the Snake River in 2001 was 107 cm (range 42 to 307 cm; Everett and Tuell 2003). This corresponds to an average weight of 5.9 kg according to a weight-length regression equation ($W=6.5x10^{-7} L^{3.43}$ where W is weight in kg and L is total length in cm) for 278 white sturgeon from the Middle Snake River 1982-1984 (Lukens 1985 as reported in Beamesderfer 1993). The modal length of 2203 white sturgeon collected in the Sacramento-San Joaquin Estuary in 1965-1970 and 1973-1976 was 102 cm (Figure 24; Kohlhorst et al. 1980). This corresponds to a weight of **6.28 kg** according to a weight-length regression equation for 209 white sturgeon collected in the Sacramento-San Joaquin Estuary in 1965-1970 (Figure 25; Kohlhorst et al. 1980)

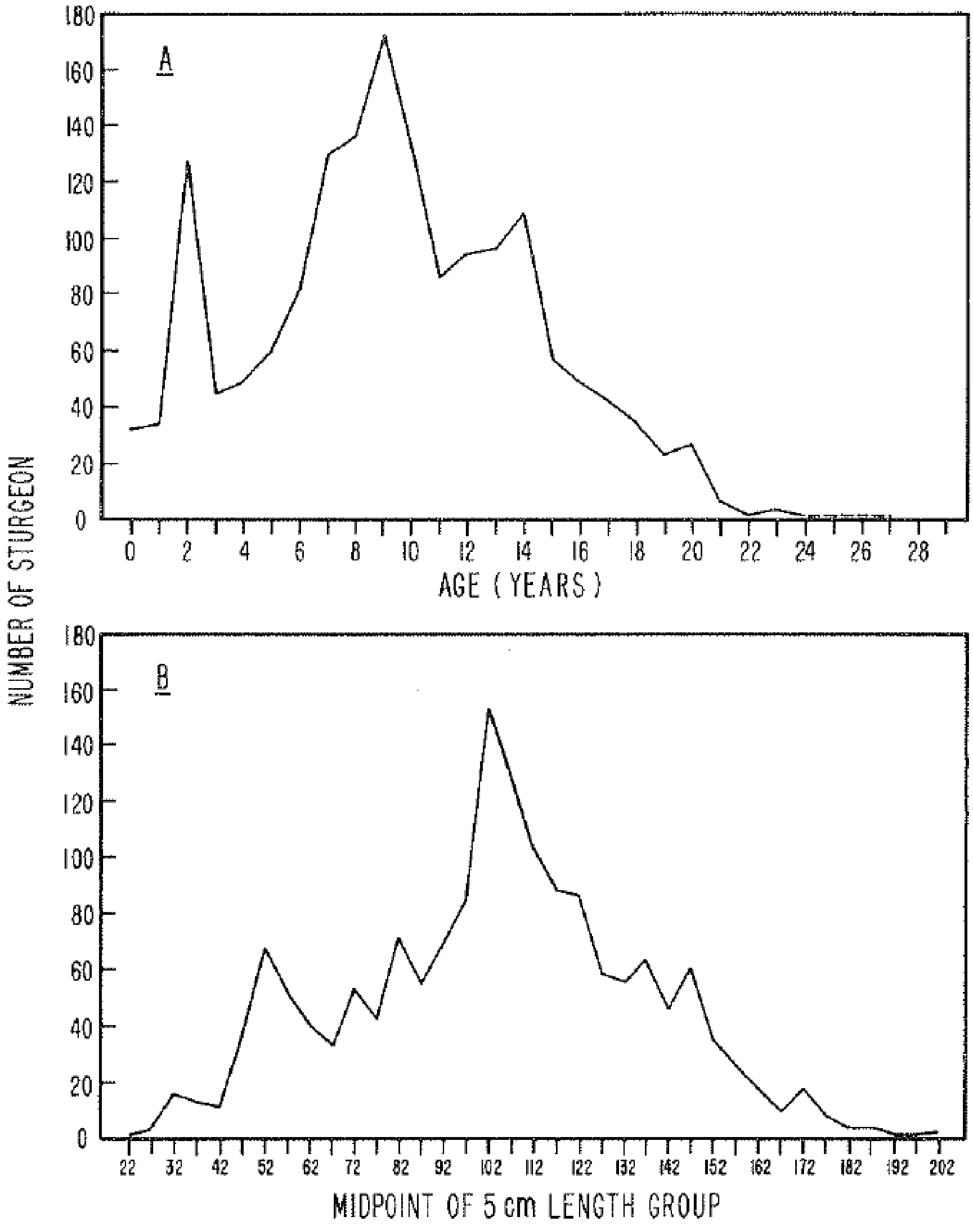

Figure 24. Age (A) and length frequency (B) distributions of white sturgeon collected in the Sacramento-San Joaquin Estuary in 1965-1970 and 1973-1976 (Kohlhorst et al. 1980).

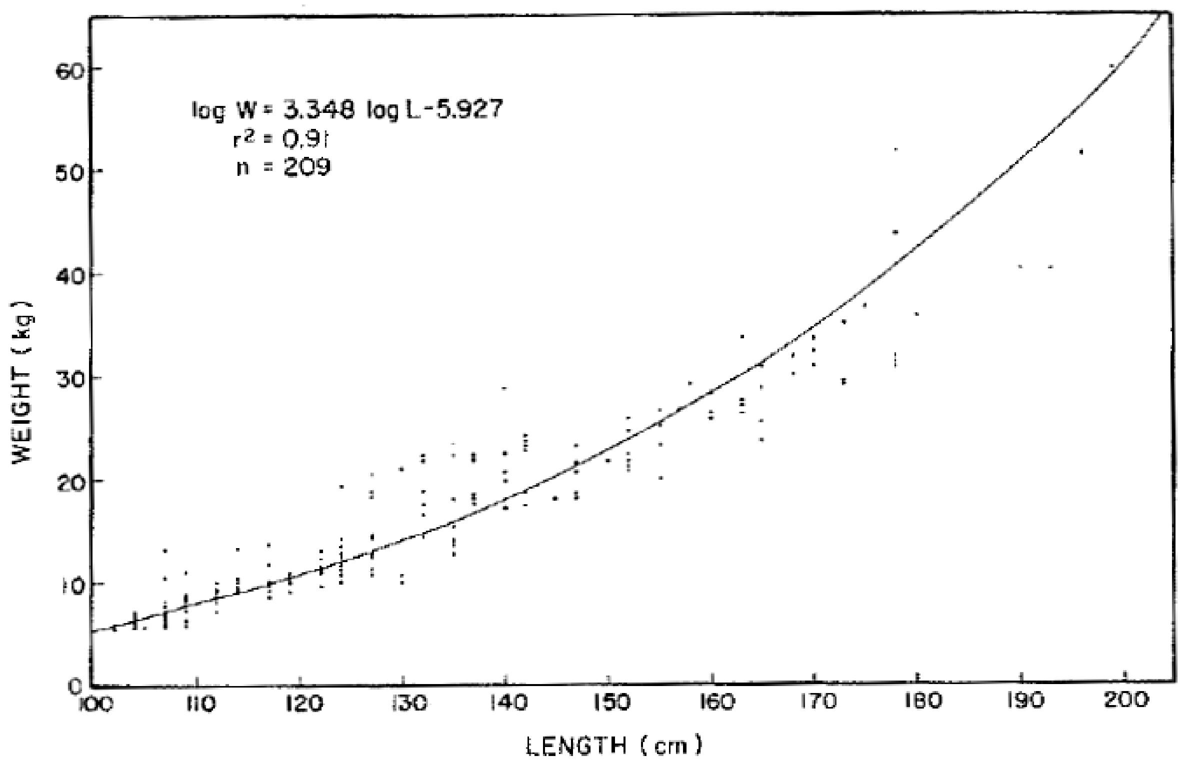

Figure 25. Length-weight relationship of white sturgeon collected in the Sacramento-San Joaquin Estuary from 1965 to 1970 (Kohlhorst et al. 1980).

Diet: Sturgeon feed primarily on benthic (bottom) organisms such as worms, mollusks, and crustaceans. Larger individuals are said to eat a greater proportion of fish (Scott and Crossman 1973). Mysid shrimp (*Neomysis awatschensis*) and amphipods (*Corophium* sp.) predominated in the stomach contents of 105 white sturgeon (ranging from 19 to 102 cm in total length) collected in the Sacramento-San Joaquin Delta in 1963-1964 (Table 8; Radtke 1966).

Table 8. Stomach contents of 105 white sturgeon caught in the Sacramento-San Joaquin Delta with gill nets and otter trawl from September 1963 through August 1964 (Radtke 1966).

	19-39 cm White Sturgeon										40-102 cm White Sturgeon									
	Fall		Winter		Spring		Summer		All year		Fall		Winter		Spring		Summer		All year	
Food Item	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.	Pct. Freq. Occ.	Pct. Tot. Vol.
Polychaetes	--	--	20.0	7.8	--	--	--	--	1.8	0.5	--	--	--	--	--	--	--	--	--	--
Mysid shrimp (*Neomysis awatschensis*)	17.2	41.2	60.0	35.1	61.5	33.6	100.0	92.9	44.6	45.9	80.0	96.4	100.0	76.7	85.7	79.5	100.0	71.6	92.0	83.7
Amphipods (*Corophium*)	96.6	58.8	100.0	57.1	76.9	51.0	44.4	7.1	83.9	50.9	--	--	25.0	23.3	71.4	20.5	22.2	28.4	32.0	14.7
Oriental shrimp (*Paleomon macrodactylus*)	--	--	--	--	--	--	--	--	--	--	20.0	3.5	--	--	--	--	--	--	4.0	1.6
Tendipedids	--	--	--	--	7.7	0.1	--	--	1.8	Tr.	--	--	--	--	--	--	--	--	--	--
Asiatic clam (*Corbicula fluminea*)	--	--	--	--	--	--	--	--	--	--	20.0	0.1	--	--	--	--	--	--	4.0	Tr.
American shad (*Alosa sapidissima*)	--	--	--	--	7.7	15.3	--	--	1.8	2.7	--	--	--	--	--	--	--	--	--	--
Stomachs examined	35		7		18		11		71		7		6		9		12		34	
Stomachs containing food	29		5		13		9		56		5		4		7		9		19	

Clams predominated in the esophageal and stomach contents of white sturgeon caught by anglers in San Pablo Bay (213 fish) and Suisun Bay/Carquinez Strait (142 fish) in 1965-1967 (Table 9 and Table 1; McKechnie and Fenner 1971). More recently with the change in the benthic food structure of the estuary (Feyrer *et al*. 2003) white sturgeon may depend more on the introduced Asian clam, *Potamocorbula amurensis*, which is an extraordinarily efficient bioaccumulator of selenium (Stewart *et al*. 2004).

Table 9. Esophageal and stomach contents of white sturgeon caught by anglers in San Pablo Bay from April 1965 to November 1967 (McKechnie and Fenner 1971).

Food item	Winter (49 fish)		Spring (90 fish)		Summer (35 fish)		Fall (39 fish)	
	Volume percentage	Frequency percentage	Volume percentage	Frequency percentage	Volume percentage	Frequency percentage	Volume percentage	Frequency percentage
Crustaceans								
Shrimp								
Crago sp.	3.1	30.6	3.1	38.0	6.8	40.0	6.9	28.2
Palaemon macrodactylus	2.0	12.2	--	--	0.2	5.7	0.1	2.6
Unidentified	3.4	4.1	--	--	--	--	1.5	2.6
Isopods *Synidotea* sp.	3.1	6.1	0.3	11.1	4.6	34.3	*T	5.1
Amphipods (unidentified)	0.1	2.0	0.1	6.7	--	--	7.7	7.7
Barnacles *Balanus* sp.	3.4	4.1	0.9	4.4	44.1	25.7	--	--
Crabs								
Rhithropanopeus harrisii	28.6	51.0	0.8	8.9	0.4	11.4	12.1	30.8
Cancer magister	0.9	2.0	--	--	1.5	2.8	31.7	12.8
Hermit (unidentified)	--	--	--	--	--	--	0.4	2.6
Annelids								
Polychaete *Nereis* sp.	1.7	2.0	--	--	--	--	--	--
Nematode	--	--	T	2.2	--	--	--	--
Molluscs								
Clams *Gemma gemma, Macoma* sp., *Tapes semidecussata* unidentified remains	14.4	34.7	11.8	78.9	40.9	57.1	32.9	61.5
Mussel *Mytilus* sp.	0.4	4.1	2.6	3.3	0.2	8.6	--	--
Snail (unidentified)	0.1	2.0	0.3	6.7	1.1	11.4	0.1	2.6
Fish								
Striped bass *Morone saxatilis*	3.4	4.1	--	--	--	--	--	--
Starry flounder *Platichthys stellatus*	8.9	6.1	1.1	2.2	--	--	--	--
Goby	3.4	2.0	--	--	--	--	0.4	5.1
Herring *Clupea harengus pallasi*	--	--	T	1.1	--	--	--	--
Unidentified fish remains	2.5	4.1	0.1	2.2	--	--	6.0	2.6
Herring eggs *Clupea harengus pallasi*	20.5	18.4	78.9	46.7	--	--	--	--
Plant material	--	--	T	1.1	--	--	--	--
Empty	--	14.3	--	5.5	--	5.7	--	12.8
Total	99.9	--	100.0	--	99.8	--	99.8	--

* T = Trace.

Table 10. Esophageal and stomach contents of white sturgeon caught by anglers in Suisun Bay and Carquinez Strait from April 1965 to November 1967 (McKechnie and Fenner 1971).

Food item	Winter (15 fish)		Spring (59 fish)		Summer (27 fish)		Fall (41 fish)	
	Volume percentage	Frequency percentage	Volume percentage	Frequency percentage	Volume percentage	Frequency percentage	Volume percentage	Frequency percentage
Crustaceans								
Shrimp								
Crago sp.	4.9	26.7	6.9	47.5	7.6	31.5	3.2	17.1
Palaemon macrodactylus	7.5	13.3	0.3	5.1	1.1	6.3	1.7	9.8
Neomysis sp.	--	--	*T	1.7	--	--	--	--
Unidentified	2.6	6.7	--	--	--	--	--	--
Isopods *Synidotea* sp.	1.5	20.0	1.6	22.0	9.5	47.2	0.5	14.6
Amphipods (unidentified)	--	--	T	1.7	T	1.6	--	--
Barnacles *Balanus* sp.	T	6.7	10.9	13.5	29.1	21.2	5.4	12.2
Crabs *Rhithropanopeus harrisii*	27.4	40.0	10.3	16.9	0.4	4.7	3.1	4.9
Annelids								
Polychaete *Nereis* sp.	--	--	T	3.4	--	--	--	--
Molluscs								
Clams *Gemma gemma*, *Macoma* sp., *Tapes semidecussata* unidentified remains	44.3	53.3	21.1	50.8	40.7	46.5	77.0	75.6
Mussel *Mytilus* sp.	0.6	13.3	--	--	--	--	--	--
Fish								
Striped bass *Morone saxatilis*	--	--	17.5	1.7	--	--	2.3	2.0
Starry flounder *Platichthys stellatus*	--	--	11.4	1.7	--	--	6.8	2.0
Anchovy *Engraulis mordax*	--	--	4.0	1.7	0.2	0.8	--	--
Midshipman *Porichthys notatus*	--	--	--	--	8.2	0.8	--	--
Staghorn sculpin *Leptocottus armatus*	--	--	0.1	1.7	--	--	--	--
Unidentified fish remains	3.8	20.0	12.9	5.1	3.2	4.7	--	--
Herring eggs *Clupea harengus pallasi*	7.5	6.7	2.9	1.7	--	--	--	--
Plant material	--	--	--	--	T	0.8	--	--
Empty	--	6.7	--	15.2	--	7.1	--	1.7
Total	100.0	--	99.9	--	100.0	--	100.0	--

*T = Trace.

Food ingestion rate: The food ingestion rate of white sturgeon in the wild is unknown, but feeding experiments in captivity may provide some indication of food ingestion rates. A feeding experiment with 240 sub-yearling white sturgeon (initial average weight: 250.5 g) that were reared in captivity at 18˚C suggested that the optimal feeding rate for growth is between **1.5 and 2 % (dry weight) of live fish body weight per day** (Hung *et al*. 1989). In another experiment, 30-g-size white sturgeon maintained at 20˚C also showed maximum growth when fed between **1.5 and 2 % (dry weight) of live fish body weight per day** (Hung and Lutes 1987). Hung *et al.* (1993) found that 30-g-size white sturgeon maintained at 23˚C and 26˚C grew optimally when fed 2.0-2.5% and 2.5-3.0%, respectively, of wet weight of fish per day, but temperatures of 18˚C and 20˚C are probably closer to temperatures generally experienced by white sturgeon in their natural habitats.

General life history: Like green sturgeon, white sturgeon are anadromous, but the adults are less marine than green sturgeon, spending more time in estuaries or fresh water. At sea, white sturgeon have been found from Ensenada, Baja California (Mexico) to the Gulf of Alaska (Fry 1973). The majority of white sturgeon rear in the Columbia-Snake River and Sacramento-San Joaquin basins (Duke *et al*. 2004). White sturgeon have been the subject of detailed studies within the San Francisco Bay estuary (e.g., Kohlhorst *et al*. 1991, Linares *et al*. 2004, Linville 2006). White sturgeon are long-lived, large-bodied, and demersal (bottom-dwelling) fish. For most species of sturgeon, females require several years for eggs to mature between spawnings (Conte *et al*. 1988). White sturgeon in the San Francisco Bay estuary congregate in Suisun and San Pablo Bays where they remain year-round except for a small fraction of the population that moves up the Sacramento River, and to a lesser extent the San Joaquin River, to spawn in late winter and early spring (Kohlhorst *et al*. 1991).

Risk of selenium exposure: Many individuals of this species remain year-round in San Pablo Bay, the part of the San Francisco Bay estuary with the highest selenium concentrations (up to 2.7 µg/L). Stewart *et al*. (2004) found the median concentration of selenium in Asian clams (*Potamocorbula amurensis*) from San Pablo Bay to be above 10 µg Se/g. Based on histopathological alterations in the kidney, Tashjian *et al.* (2006) estimated that for juvenile white sturgeon a threshold dietary selenium toxicity concentration lies between 10 and 20 µg Se/g. It is uncertain at what point in their life white sturgeon begin feeding on Asian clams. Linares *et al*. (2004) found concentrations of selenium as high as 46.7 µg/g in gonads of 39 white sturgeon captured in the San Francisco Bay. Kroll and Doroshov (1991) reported that developing ovaries of white sturgeon from San Francisco Bay contained as much as 71.8 µg/g selenium or 7-times the threshold for reproductive toxicity in fish (Lemly 1996a, 1996b) of 10 µg/g. The threshold specifically in white sturgeon has been confirmed to be between 9 µg/g and about 16 µg/g in experiments in which seleno-L-methionine was injected into yolk sac larvae of white sturgeon (Linares *et al*. 2004). Linville (2006) showed that significant developmental defects and mortality occurred in white sturgeon eggs at a threshold of around 11–15 µg/g selenium. A hazard threshold of around 3–8 µg/g in developing white sturgeon was suggested by Linville (2006). Sampling of pallid sturgeon (*Scaphirhynchus al*bus) in the Missouri River system suggests that normal selenium levels in sturgeon eggs are 2-3 µg/g (Ruelle and Keenlyne 1993) as has been found for many other fish species (see review in Skorupa *et al*. 1996 and in USDI-BOR/FWS/GS/BIA 1998). Thus, white sturgeon in the San Francisco Bay estuary are producing eggs with as much as 35-times normal selenium content. Based on studies regarding toxicity response functions for avian and fish eggs (e.g., Lemly 1996a, 1996b; Skorupa et al. 1996; USDI-BOR/FWS/GS/BIA 1998) and assuming that sturgeon are as sensitive to selenium as birds and other fish, it is highly probable that these fish are reproductively impaired due to selenium exposure. For example, bluegill embryos resulting from ovaries containing 38.6 µg/g selenium exhibited 65 percent mortality (Gillespie and Bauman 1986).

Delta Smelt (*Hypomesus transpacificus*)

Status: Delta smelt were federally listed as a threatened species on March 5, 1993, (58 FR 12854). In recent years the population of delta smelt has declined dramatically, dropping to the lowest levels on record, and placing the species at increased risk of extinction.

Size: Delta smelt are slender-bodied fish that typically reach 60-70 mm standard length (measured from tip of the snout to origin of the caudal fin), although a few may reach 120 mm standard length. The median fork length of 113 delta smelt was about 62 mm (Kimmerer *et al.* 2005). This corresponds to a weight of 2.1 g, using the regression in Kimmerer *et al.* (2005): $W = 0.0000018\ L^{3.38}$ where W is weight in grams and L is fork length in mm. In tow-net surveys in June-August 2000 (Figure 26) 785 Delta smelt young-of-the-year had an average length of 35.7 mm (Gartz 2001) corresponding to an average weight of **0.32 g**, using the above regression. Young-of-the-year were defined as less than 69 mm fork length; only six Delta smelt larger than this were caught.

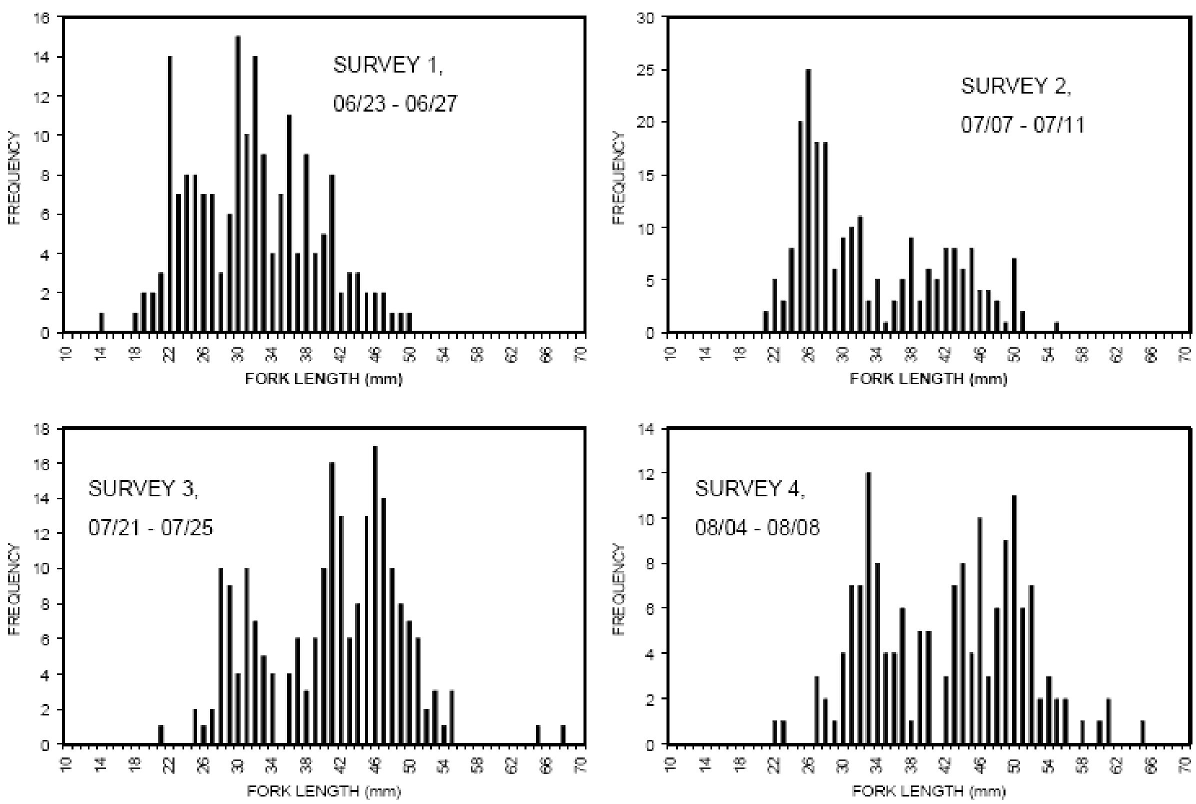

Figure 26. Length-frequency plots for young-of-the-year delta smelt for townet surveys in 2000 in the Sacramento and San Joaquin Rivers and Bay/Delta system (Figure 2 in Gartz 2001).

Diet: Delta smelt feed primarily on planktonic copepods, cladocerans, and amphipods. To a lesser extent, they feed on insect larvae. Larger fish may also feed on the opossum shrimp, *Neomysis mercedis*. The most important food organism for all sizes seems to be the euryhaline copepod, *Eurytemora affinis*, although by 1992 the exotic species, *Pseudodiaptomus forbesi*, had become a major part of the diet (Moyle *et al.* 1992).

Food ingestion rate: Food ingestion rate for wild delta smelt in the estuary may be estimated using measured growth rates in the estuary combined with an estimate of food conversion efficiency. By tracking the length-frequency distributions of delta smelt in the Sacramento and San Joaquin Rivers and Bay/Delta system, Gartz (2001) calculated growth rates to be 0.11 mm/day from late June to early July, 0.56 mm/day from early to late June, and 0.07 mm/day from late June to early August. The average of these growth rates is 0.247 mm/day, which corresponds to mass growth rate of 0.0075 g/day, calculated by differentiating the regression $W = 0.0000018\ L^{3.38}$ where W is weight in grams and L is fork length in mm (Kimmerer *et al.* 2005). The closest (in size and taxonomic relationship) available food conversion efficiency (growth in weight / weight of food ingested) is 0.202 for 3 g sea trout (*Salmo trutta trutta*) in Norway (Neveu 1980, as seen in Worldfish Center 2000). Dividing mass growth rate by this food conversion efficiency yields an estimated food ingestion rate of 0.037 g /day (dry weight) or **11.4 %** of body wet weight per day for Delta smelt with a fork length of 36 mm weighing 0.32 g.

Distribution: Delta smelt are endemic to the upper Sacramento-San Joaquin estuary. They occur 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 January to July) 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, and, (5) Montezuma Slough, (6) Suisun Bay, (7) Suisun Marsh, (8) Carquinez Strait, (9) Napa River, and (10) San Pablo Bay. It is not known if delta smelt in San Pablo Bay are a permanent population or if they are washed into the Bay during high outflow periods. Since 1982, 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 (adult, larvae, and juveniles) of delta smelt have the potential to be present in Suisun Bay (DWR and USDI 1994; Moyle 1976; and Wang 1991).

Life History: Delta smelt of all sizes are found in the main channels of the Delta and Suisun Marsh and the open waters of Suisun Bay where the waters are well oxygenated and temperatures relatively cool, usually less than 20˚-22˚ C in summer. When not spawning, they tend to be concentrated near the zone where incoming salt water mixes with out flowing freshwater (mixing zone). This area has the highest primary productivity and is where zooplankton populations (on which delta smelt feed) are usually most dense (Knutson and Orsi 1983; Orsi and Mecum 1986). At all life stages delta smelt are found in greatest abundance in the top two meters of the water column and usually not in close association with the shoreline.

Delta smelt inhabit open, surface waters of the Delta and Suisun Bay. In most years, spawning occurs in shallow water habitats in the Delta. Shortly before spawning, adult smelt migrate upstream from the brackish-water habitat associated with the mixing zone to disperse widely into river channels and tidally-influenced backwater sloughs (Radtke 1966; Moyle 1976, 2002; Wang 1991). Some spawning probably occurs in shallow water habitats in Suisun Bay and Suisun Marsh during wetter years (Sweetnam 1999 and Wang 1991). Spawning has also been recorded in Montezuma Slough near Suisun Bay (Wang 1986) and also may occur in Suisun Slough in Suisun Marsh (P. Moyle, UCD, unpublished data).

The spawning season varies from year to year, and may occur from late winter (December) to early summer (July). Pre-spawning adults are found in Suisun Bay and the western delta as early as September (DWR and USDI 1994). Moyle (1976, 2002) collected gravid adults from December to April, although ripe delta smelt were common in February and March. In 1989 and 1990, Wang (1991) estimated that spawning had taken place from mid-February to late June or early July, with peak spawning occurring in late April and early May.

Delta smelt spawn in shallow, fresh, or slightly brackish water upstream of the mixing zone (Wang 1991). Most spawning occurs in tidally-influenced backwater sloughs and channel edgewaters (Moyle 1976, 2002; Wang 1986, 1991; Moyle *et al*. 1992). Laboratory observations have indicated that delta smelt are broadcast spawners (DWR and USDI 1994) and eggs are demersal (sink to the bottom) and adhesive, sticking to hard substrates such as: rock, gravel, tree roots or submerged branches, and submerged vegetation (Moyle 1976, 2002; Wang 1986). Growth of newly-hatched delta smelt is rapid and juvenile fish are 40-50 mm long by early August (Erkkila *et al.* 1950; Ganssle 1966; Radtke 1966). By this time, young-of-year fish dominate trawl catches of delta smelt, and adults become rare. Delta smelt reach 55-70 mm standard length in 7-9 months (Moyle 1976, 2002). Growth during the next 3 months slows down considerably (only 3-9 mm total), presumably because most of the energy ingested is being directed towards gonadal development (Erkkila *et al*. 1950; Radtke 1966). There is no correlation between size and fecundity, and females between 59-70 mm standard lengths lay 1,200 to 2,600 eggs (Moyle *et al.* 1992). The abrupt change from a single-age, adult cohort during spawning in spring to a population dominated by juveniles in summer suggests strongly that most adults die after they spawn (Radtke 1966 and Moyle 1976, 2002). However, in El Nino years when temperatures rise above 18˚ C before all adults have spawned, some fraction of the unspawned population may also hold over as two-year-old fish and spawn in the subsequent year. These two-year-old adults may enhance reproductive success in years following El Nino events.

In a near-annual fish like delta smelt, a strong relationship would be expected between number of spawners present in one year and number of recruits to the population the following year. Instead, the stock-recruit relationship for delta smelt is weak, accounting for about a quarter of the variability in recruitment (Sweetnam and Stevens 1993). This relationship does indicate, however, that factors affecting numbers of spawning adults (*e.g.*, entrainment, toxics, and predation) can have an effect on delta smelt numbers the following year.

Risk of selenium exposure: The Recovery Plan for the Sacramento/San Joaquin Delta Native Fishes (USFWS 1996) states that delta smelt are ecologically similar to larval and juvenile striped bass (*Morone saxitilis*). Saiki and Palawski (1990) sampled juvenile striped bass in the San Joaquin River system including three sites in the San Francisco Bay estuary. Striped bass from the estuary contained up to 3.3 µg/g whole-body selenium, a value just below Lemly's 4 µg/g toxicity threshold, even though waterborne selenium typically averages <1 µg/L (ppb) and has been measured no higher than 2.7 µg/L (ppb) within the estuary (Pease *et al*. 1992). Striped bass collected from Mud Slough in 1986, when the annual median selenium concentration in water was 8 µg/L (ppb) (Steensen *et al*. 1997), contained up to 7.9 µg/g whole-body selenium and averaged 6.9 µg/g whole-body selenium.

Delta smelt spawning sites are almost entirely restricted to the north-Delta channels associated with the selenium-normal Sacramento River and are nearly absent from the south-delta channels associated with the selenium-contaminated San Joaquin River (USFWS 1996). Delta smelt (n=41) salvaged from CDFG annual abundance surveys primarily around Chipps Island in 1993 and 1994 had a mean of 1.5 µg/g whole-body selenium dry weight (range, 0.7-2.3 µg/g) (Bennet *et al.* 2001). Three composite samples of delta smelt eggs also had less than 2 µg/g selenium dry weight (USFWS unpublished data).

Sacramento splittail (*Pogonichthys macrolepidotus*)

Status: The Sacramento splittail was listed as threatened on February 8, 1999 (FR 64:5963). The listing was challenged in Federal District Court, and rescinded on September 22, 2003 (FR 68:55139); however, they remain a species of concern.

Sacramento splittail are endemic to certain waterways in California's Central Valley, where they were once widely distributed (Moyle 1976, Moyle 2002). Sacramento splittail currently occur in Suisun Bay, Suisun Marsh, the San Francisco Bay-Sacramento-San Joaquin River Estuary (Estuary), the Estuary's tributaries (primarily the Sacramento and San Joaquin rivers), the Cosumnes River, the Napa River and Marsh, and the Petaluma River and Marsh.

Size: The average fork length of 14 adult Sacramento splittail captured in a fyke trap on their upstream migration in the Yolo Bypass Toe Drain in February 2001 was 320 mm with a standard deviation of 31 mm (Sommer *et al*. 2002). Fork lengths of 83 juvenile and adult splittail collected in newly-restored tidal and flood plain habitats in the Napa River/Napa Creek Flood Control Project area in 2001-2002 ranged from about 25 to 250 mm with a modal fork length of 200-224 mm (Dietl *et al*. 2003) (Figure 27). The center of this modal range is 212 mm. Using the length-weight regression of $W = 0.000003\ L^{3.27}$, where W is weight in grams and L is fork length in mm (Kimmerer *et al*. 2005), a modal Sacramento splittail with a fork length of 212 mm would be expected to have a weight of $0.000003 \times 212^{3.27}$ = **121 g**.

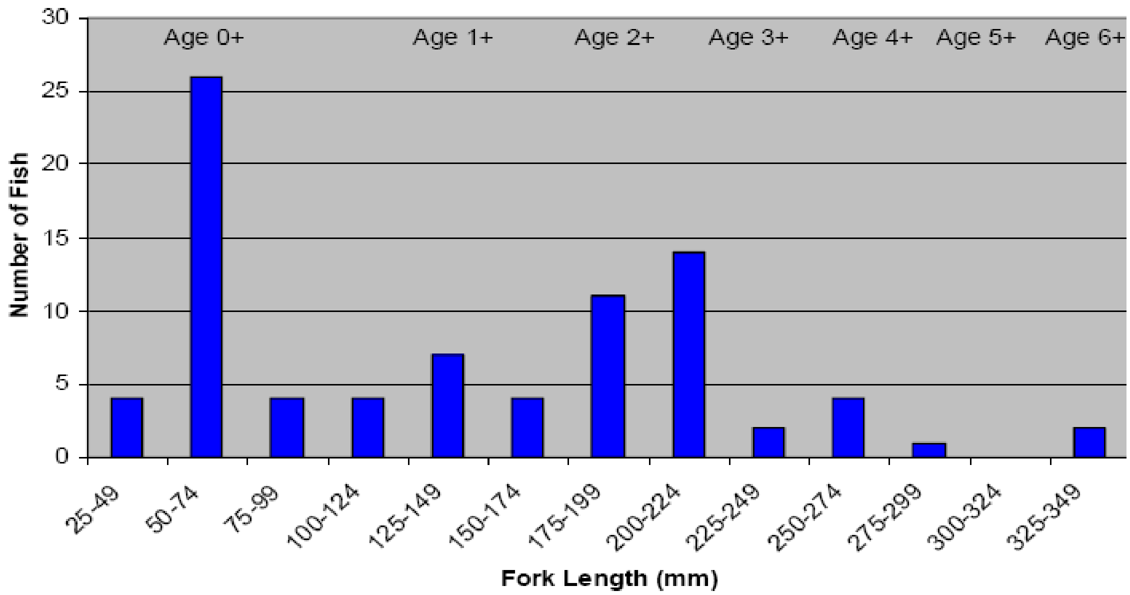

Figure 27. Lengths of captured Sacramento splittail in newly-restored tidal and flood plain habitats in the Napa River/Napa Creek Flood Control Project area in 2001-2002 (Dietl *et al*. 2003).

Diet: Splittail are benthic (bottom) feeders. Stomach contents from 70 splittail collected in Suisun Marsh between March 1998 and January 1999 indicate that their diet consisted mainly (43%) of unidentified material, probably detritus. The overbite clam (Asian clam), *Potamocorbula amurensis*, and other mollusks constituted 34% of the diet (Feyrer and Matern 2000, Feyrer *et al*. 2003).

Food ingestion rate: Food ingestion rate for wild splittail in the estuary may be estimated using measured growth rates in the estuary combined with an estimate of food conversion efficiency. A cohort of 2100 splittail with a median standard length of about 134 mm grew at an average rate of about 0.281 mm/day (standard length) from February to September 1980 (Figure 28) (Moyle et al. 2004). These data correspond to a median fork length of 154 mm, weight of 42.6 g and mass growth rate of 0.292 g/day using the relationship $L = (SL + 0.2657)/ 0.8722$ where L is fork length in mm and SL is standard length in mm (Randall Baxter pers. com.), and the length-weight regression of $W = 0.000003\ L^{3.27}$, where W is weight in grams and L is fork length in mm (Kimmerer *et al*. 2005). The closest (in size and taxonomic relationship) available food conversion efficiency (growth in weight / weight of food ingested) is 0.202 for 3 g sea trout (*Salmo trutta trutta*) in Norway (Neveu 1980, as seen in Worldfish Center 2000). Dividing mass growth rate by this food conversion efficiency yields an estimated food ingestion rate of 1.45 g/day (dry weight) or **3.37 %** of body wet weight per day for splittail with a fork length of 154 mm weighing 42.6 g.

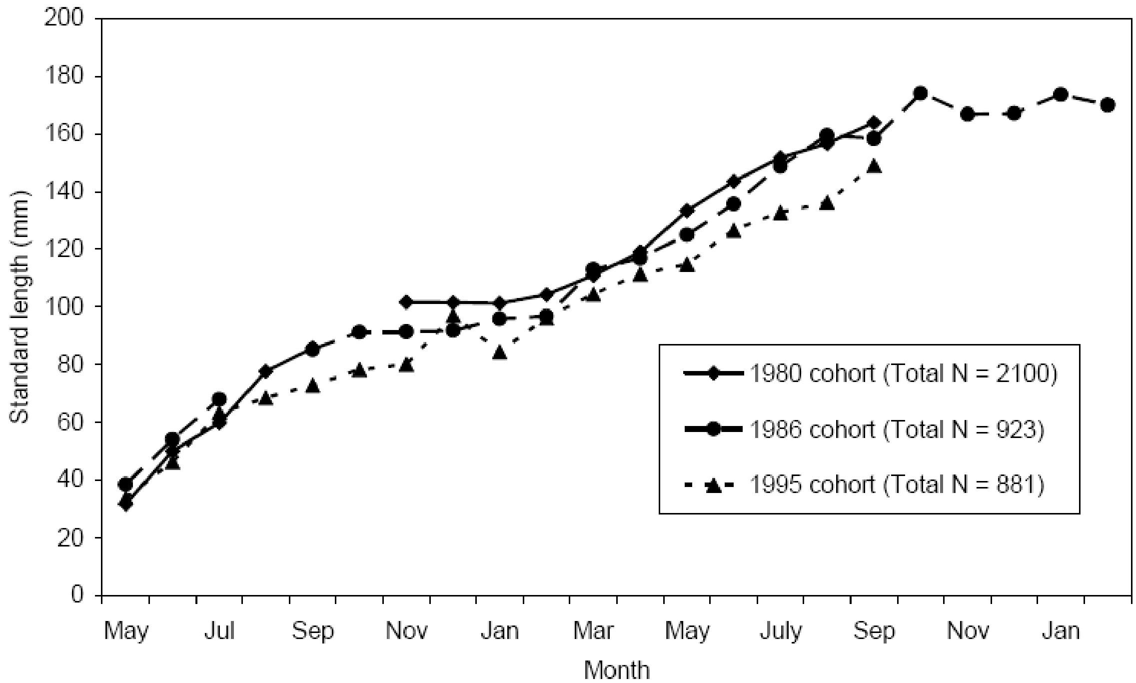

Figure 28. Mean lengths of monthly samples of three splittail cohorts in Suisun Marsh (Figure 11 in Moyle et al. 2004).

General life history: Splittail are relatively long-lived (about 5-7 years) and are highly fecund (up to 100,000 eggs per female). Their populations fluctuate on an annual basis depending on spawning success and strength of the year class (Daniels and Moyle 1983). Both male and female splittail mature by the end of their second year (Daniels and Moyle 1983), although occasionally males may mature by the end of their first year and females by the end of their third year (Caywood 1974). Fish are about 180-200 millimeters (7-8 inches) standard length when they attain sexual maturity (Daniels and Moyle 1983), and the sex ratio among mature individuals is 1:1 (Caywood 1974).

There is some variability in the reproductive period, with older fish reproducing first, followed by younger fish that tend to reproduce later in the season (Caywood 1974). Generally, gonadal development is initiated by fall, with a concomitant decrease in somatic growth (Daniels and Moyle 1983). By April, ovaries reach peak maturity and account for approximately 18 percent of the body weight. The onset of spawning seems to be associated with increasing water temperature and day length and occurs between early March and May in the upper Delta (Caywood 1974). However, Wang (1986) found that in the tidal freshwater and euryhaline habitats of the Sacramento-San Joaquin estuary, spawning occurs by late January and early February and continues through July. Spawning times are also indicated by the salvage records from the State Water Project pumps. Adults are captured most frequently in January through April, when they are presumably engaged in spawning movements, while young-of-year are captured most abundantly in May through July (Meng 1993). These records indicate most spawning takes place from February through April.

Splittail spawn on submerged vegetation in flooded areas. Spawning occurs in the lower reaches of rivers (Caywood 1974), dead-end sloughs (Moyle 1976) and in the larger sloughs such as Montezuma Slough (Wang 1986). Larvae remain in the shallow, weedy areas inshore in close proximity to the spawning sites and move into the deeper offshore habitat as they mature (Wang 1986).

Strong year classes have been produced even when adult numbers are low, if outflow is high in early spring (e.g., 1982, 1986). Since 1988, recruitment has been consistently lower than expected, suggesting this relationship may be breaking down (Meng 1993). For example, both 1978 and 1993 were wet years following drought years, yet the young-of-year abundance in 1993 was only 2 percent of the abundance in 1978.

Risk of selenium exposure: Splittail are likely to be relatively vulnerable to selenium contamination because of their estuarine habitat and bottom-feeding habits. Splittail feed primarily on bivalves including the overbite clam, *Potamocorbula amurensis*, which are efficient selenium bioaccumulators (Stewart *et al*. 2004). Teh *et al*. (2004) found that juvenile splittail are adversely affected (liver lesions) by chronic exposure (nine months) to a diet of 6.6 μg/g selenium. Deformities typical of Se exposure have been seen in splittail collected from Suisun Bay (Stewart *et al*. 2004).

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