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

Species at Risk from Selenium Exposure in California Inland Surface Waters, Enclosed Bays and Estuaries

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Table of Contents

Abbreviations

Body weight – bw
Centimeters – cm
Dry weight – dw
Food ingestion rate – FIR
Free-living metabolic rate – FMR
Fresh weight – fw
Grams – g
Kilograms – kg
Liter – L
Micrograms – µg
Milligrams – mg
Millimeters – mm
Parts per million – ppm
Wet weight – ww

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INTRODUCTION

Discharges of selenium from Bay Area oil refineries, subsurface drainwater from Central Valley farms, and natural sources reach surface waters where uptake by aquatic food chains is particularly efficient (Presser and Luoma 2006). Selenium concentrated through food chains may have a variety of adverse effects including reduced growth, failure to hatch, and embryonic deformities in fish and 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 California inland surface waters, enclosed bays and estuaries. This document is the contribution of the U. S. Fish and Wildlife Service (USFWS) to this interagency project. The role of the USFWS in this task is to review and compile existing pertinent data—diet, body weight, food ingestion rate (FIR), natural history, selenium exposure risk—on the species that are at greatest risk due to dietary selenium exposure in California.

SPECIES CONSIDERED

The 73 species considered for evaluation of selenium exposure risk in California's inland surface waters, enclosed bays and estuaries are listed in Table 1. To keep the initial list of species reasonably manageable but representative, most invertebrates, and most vertebrates that are members of strictly or mainly terrestrial food chains were not considered. Many species in the San Francisco Bay estuary are not included in this report as they are being address in a separate criteria development process (See USFWS 2008 and USFWS 2012c for species at risk of selenium in the San Francisco Estuary). In compiling this list, species protected by federal legislation were given priority; however, other species were considered if they were at a particularly high risk of exposure and/or are representative of other species or unique habitats in California.

Since the proposed criteria would cover the entire State and California ecosystems vary greatly, an effort was made to be certain that the initial species list developed covered as many of California ecosystems as possible. The ranges of the species on the initial list were evaluated and compared to the 7 ecoregions used for Strategic Habitat Conservation efforts within the U.S. Fish and Wildlife Service's Pacific-Southwest Region (Figure A1, Appendix A). These ecoregions are based upon Bailey (1995) ecoregion boundaries and were combined or sectioned to meet hydrologic or administrative needs. These ecoregions are similar to the Level III boundaries (Omernik 1987) utilized by the U.S. EPA (https://www.epa.gov/eco-research/ecoregions-north-america). Table A1 in Appendix A shows that the number of species in the initial list representing each of the 7 ecoregions ranged from 17 to 41 and the number of species in the final list for further evaluation ranged from 2 to 12. The Klamath Basin ecoregion had the fewest number of species (2) representing it in the final species list; however, the Klamath Basin is a subsection of a Bailey (1995) ecoregion and other species (e.g. salmonids) from ecoregions 2 and 4 will provide additional representation for the Basin.

SPECIES MOST AT RISK

Species considered most at risk ("Further Review" column in Table 1) were selected from among those in Table 1 using three general factors and best professional judgement: 1) Diet- dependence upon an aquatic food web especially a benthic diet; 2) Exposure- likelihood of exposure to selenium from surface waters (as regulated under this EPA action) as well as sensitivity to selenium; and 3) Status- generally endangered, threatened or of concern, especially population status. A high, medium, and low ranking was used for each factor. Some "medium" ranked species were not included for additional review if other species were selected that could well represent them.

For the species considered most at risk, information on the diet, body weight, and ingestion rate for the species was gathered. This information is summarized in Table 2. Additional information on documented exposure and sensitivity to selenium for each species is provided as well as appropriate surrogate information. For some species like salmonids and birds there is abundant information available; however, for others like amphibians and reptiles selenium exposure and toxicity information is limited. In this case, information from fish (for amphibians) and birds (for reptiles) are used with care because of the uncertainty of direct comparisons.

Table 1. List of species considered for evaluation of selenium exposure risk in California inland surface waters, enclosed bays and estuaries.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Mammals							
San Joaquin Kit Fox	*Vulpes macrotis mutica*	endangered	L	L	H	No	Although the kit fox is at risk to selenium exposure via reuse of subsurface agricultural drainage water on salt tolerant crops in the San Joaquin Valley or on irrigation impaired retired ag land, it is at low risk to selenium exposure via surface waters as regulated under this EPA action.
Buena Vista Lake Ornate Shrew	*Sorex ornatus relictus*	endangered	M	H	H	Yes	Small omnivorous mammal living in dense marshy and riparian vegetation along remaining wetland habitat in Tulare Basin. Extremely high metabolic rate. High potential for selenium exposure via surface waters or pumped groundwater in Tulare Basin.
Kangaroo Rats	*Dipodomys sp.*	endangered	L	L	H	No	Although kangaroo rats may be at risk to selenium exposure via reuse of subsurface agricultural drainage water on salt tolerant crops in the San Joaquin Valley or on irrigation impaired retired ag land, they are at low risk to selenium exposure via surface waters as regulated under this EPA action.
California Sea Otter	*Enhydra lutris nereis*	threatened	M	L	M	No	Although having a diet of marine benthic organisms the sea otters exposure potential to selenium is low.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Birds							
American Dipper	*Cinclus mexicanus*	MBTA	M	M	M	Yes	Highly aquatic dependent bird of mountain streams. Unique habitat and exposure. Selenium study available.
American White Pelican	*Pelecanus erythrorhynchos*	MBTA	M	M	L	No	Can be found on permanent inland waters as far north as Clear Lake (breeding) in Modoc County and as far south as the Salton Sea (winter). Preys on some bottom-feeding fish as well as schooling fish. Represented by brown pelican.
California Brown Pelican	*Pelecanus occidentalis californicus*	endangered	M	H	M	Yes	Primarily a coastal species; however, large populations winter in the Salton Sea where selenium levels can be elevated. Feeds mainly on surface-schooling fish.
White-faced Ibis	*Plegadis chihi*	concern	M	L	M	No	Breeds and winters in San Joaquin Valley. Inhabits mainly freshwater wetlands, but also estuarine wetlands. Eats aquatic and moist soil invertebrates.
Double-crested Cormorant	*Phalacrocorax auritus*	MBTA	M	L	L	No	Winters in Central Valley and SF Bay/Delta. Feeds on bottom-dwelling fish and invertebrates as well as schooling fish.
American Bittern	*Botaurus lentiginosus*	concern	M	M	M	No	Feeds mainly in freshwater marshes, eating mainly insects and small vertebrates. Represented by other avian species.
Western Least Bittern	*Ixobrychus exilis hesperis*	concern	M	M	M	No	Breeds in SF Delta. Feeds mainly in freshwater marshes, eating mainly insects and small vertebrates. Represented by other avian species.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Aleutian Canada Goose	*Branta canadensis leucopareia*	delisted, MBTA	L	L	L	No	Winters in California, feeding primarily in upland crops and fallow fields. Sensitive to selenium, but unlikely to be exposed in aquatic settings.
Greater Scaup	*Aythya marila*	MBTA	H	L	M	No	The scaup species are at high risk of selenium exposure in the Bay area due to their diet of non-native Asian clams and selenium sources, but that exposure drops dramatically outside the bay.
Lesser Scaup	*Aythya affinis*	MBTA	H	L	M	No	The scaup species are at high risk of selenium exposure in the Bay area due to their diet of non-native Asian clams and selenium sources, but that exposure drops dramatically outside the bay.
Black Scoter	*Melanitta nigra*	MBTA	H	L	M	No	The scoter species are at high risk of selenium exposure in the Bay area due to their diet of non-native Asian clams and selenium sources, but that exposure drops dramatically outside the bay.
White-winged Scoter	*Melanitta fusca*	MBTA	H	L	M	No	The scoter species are at high risk of selenium exposure in the Bay area due to their diet of non-native Asian clams and selenium sources, but that exposure drops dramatically outside the bay.
Surf Scoter	*Melanitta perspicillata*	MBTA	H	L	M	No	The scoter species are at high risk of selenium exposure in the Bay area due to their diet of non-native Asian clams and selenium sources, but that exposure drops dramatically outside the bay.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Osprey	*Pandion haliaetus*	MBTA	H	L	M	No	High trophic level piscivore; not at risk overall and exposure well represented by bald eagle. Protection of other avian species at risk will protect this species.
Bald Eagle	*Haliaeetus leucocephalus*	delisted, MBTA,BGEPA	H	M	H	Yes	Delisted but monitored for population status and contaminants. High trophic level piscivore.
American Peregrine Falcon	*Falco peregrinus anatum*	delisted, MBTA	M	L	M	No	High trophic level. May be exposed to selenium in aquatic food chain as predator on piscivorous birds, but exposure generally diluted by terrestrial component of diet; Protection of other avian species at risk will protect this species.
California Black Rail	*Laterallus jamaicensis coturniculus*	MBTA	M	M	H	Yes	Inhabits tidal marsh in SF Bay estuary and freshwater marshes of the lower Colorado River, Imperial Valley, and Northern Sierra foothills. Feeds on wide variety of terrestrial, and aquatic invertebrates, including snails, also seeds; the southern California and Bay populations would be at greatest risk of selenium exposure.
Ridgeway's Yuma Rail	*Rallus obsoletus yumanensis*	endangered	M	H	H	Yes	Endangered species using marsh habitat along lower Colorado River where elevated selenium levels are documented. Diet primarily crayfish.
Ridgeway's Light-footed Rail	*Rallus obsoletus levipes*	endangered	M	M	H	Yes	Consumes wide variety of terrestrial and aquatic invertebrates. S. California coastal marshes and lagoons. Elevated selenium levels in S. California coastal streams, Newport Bay.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Ridgeway's Rail	*Rallus obsoletus obsoletus*	endangered	H	M	H	Yes	Subspecies endangered and endemic to SF estuary; feeds on benthic invertebrates, including filter-feeders that bioaccumulate selenium.
Marbled Murrelet	*Brachyramphus marmoratus*	threatened	L	L	H	No	Forages in bays along Pacific coast in summer, but not recorded in SF Bay/Delta. Dives for pelagic food: schooling fish and euphausiids (krill).
California Least Tern	*Sternula antillarum browni*	endangered	M	M	H	Yes	Breeds primarily in Central San Francisco Bay and southern California coastal bays and lagoons. Feeds throughout bay/estuary systems, mainly on surface fish. Some locations with elevated selenium levels.
Black Tern	*Chlidonias niger*	concern	M	M	M	No	Breeds in Central Valley including Bay/Delta. Feeds on marine and freshwater surface fish and insects. Represented by least tern.
Caspian Tern	*Sterna caspia*	MBTA	M	M	L	No	Preys heavily on juvenile salmonids, but not endangered overall. Represented by least tern.
Western Snowy Plover	*Charadrius nivosus nivosus*	threatened	M	L	H	No	Terrestrial component of diet likely provides dietary dilution of aquatic system selenium exposures; has been shown to be tolerant of selenium exposure.
Mountain Plover	*Charadrius montanus*	concern	L	M	M	No	Winters in agricultural fields of Sacramento/San Joaquin Valley (non-breeding). Diet mainly terrestrial.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Tricolored Blackbird	*Agelaius tricolor*	concern	L	M	M	No	Nests in freshwater marshes and agricultural areas of Central Valley; however, feeds widely in grassland, ag fields and dairy lands on insects and grains. Protection of other avian species at risk will protect this species.
Reptiles							
Giant Garter Snake	*Thamnophis gigas*	threatened	M	M	H	Yes	Aquatic predator. West side San Joaquin Valley locations in selenium problem areas.
Blunt-nosed Leopard Lizard	*Gambelia sila*	endangered	L	L	H	No	Although leopard lizards may be exposed to selenium via reuse of subsurface agricultural drainage water on salt tolerant crops in the San Joaquin Valley or on irrigation impaired retired ag land, they would not be exposed to selenium via surface waters as regulated under this EPA action.
Fish							
Chinook Salmon	*Oncorhynchus tshawytscha*	endangered/ threatened	M	M	H	Yes	Salmonids sensitive to selenium; most sensitive life stages occur in rivers and estuary.
Steelhead	*Oncorhynchus mykiss*	threatened	M	M	H	Yes	Salmonids sensitive to selenium; most sensitive life stages occur in rivers and estuary.
Coho Salmon	*Oncorhynchus (=salmo) kisutch*	threatened	M	M	H	No	Salmonids sensitive to selenium; most sensitive life stages occur in rivers and estuary. Well represented by Chinook and Steelhead.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Little Kern Golden Trout	*Oncorhynchus aguabonita whitei*	threatened	M	M	H	Yes	Salmonids sensitive to selenium. Small isolated populations.
Lahontan Cutthroat Trout	*Oncorhynchus clarkii henshawi*	threatened	M	M	H	Yes	Salmonids sensitive to selenium. Small isolated populations.
Paiute Cutthroat Trout	*Oncorhynchus clarkii seleniris*	threatened	M	M	H	Yes	Salmonids sensitive to selenium. Small isolated populations.
Cui-ui	*Chasmistes cujus*	endangered	M	L	H	No	Primarily in Pyramid Lake, Nevada and lower Truckee River, NV.
Desert Pupfish	*Cyprinodon macularius*	endangered	M	M	H	Yes	Locations along Salton Sea known to have elevated selenium levels. Species is considered to be sensitive to selenium.
Owens Pupfish	*Cyprinodon radiosus*	endangered	M	M	H	Yes	Desert pupfish considered to be sensitive to selenium. Owens Lake area potential risk for selenium.
Mohave Tui Chub	*Gila bicolor ssp. mohavensis*	endangered	M	L	H	No	Small population; however, not likely to be exposed to selenium. Protection of other fish species will likely protect this species.
Owens Tui Chub	*Gila bicolor ssp. snyderi*	endangered	M	L	H	No	Small population; however, not likely to be exposed to selenium. Protection of other fish species will likely protect this species.
Bonytail Chub	*Gila elegans*	endangered	M	H	H	Yes	Like razorback sucker, high potential for exposure and risk to selenium.
Razorback Sucker	*Xyrauchen texanus*	endangered	M	H	H	Yes	Documented at risk to selenium exposure.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Santa Ana Sucker	*Catostomus santaanae*	threatened	M	M	H	Yes	Found in the Santa Ana River, the San Gabriel River, and Big Tujunga Creek. Potential for exposure to selenium.
Warner Sucker	*Catostomus warnerensis*	threatened	M	L	M	No	Found only in far NE corner of California. No critical habitat in CA. No known sources of selenium.
Shortnose Sucker	*Chasmistes brevirostris*	endangered	M	L	H	No	Endemic to the Klamath Basin. Drainwater studies in 1990s did not find selenium problems. Low exposure potential.
Lost River Sucker	*Deltistes luxatus*	endangered	M	L	H	No	Endemic to the Klamath Basin. Drainwater studies in 1990s did not find selenium problems. Low exposure potential.
Tidewater Goby	*Eucyclogobius newberryi*	endangered	M	M	H	Yes	Bottom-dwelling carnivore. Different watersheds present different potential risks, some high, some low. Those in southern coastal estuaries more likely at higher risk.
Unarmored Threespine Stickleback	*Gasterosteus aculeatus williamsoni*	endangered	M	M	H	Yes	Only found in four areas: the upper Santa Clara River and its tributaries; San Antonio Creek on Vandenberg Air Force Base; San Felipe Creek; and ponds in the Shay Creek vicinity. Elevated selenium levels documented in San Antonio Creek (CEDEN).
Delta Smelt	*Hypomesus transpacificus*	threatened	M	M	H	Yes	Endemic to the SF Bay/Delta estuary. Feeds on zooplankton, not a pathway of greatest exposure, but highly threatened species overall.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Longfin Smelt	*Spirinchus thaleichthys*	concern	L	L	H	No	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.
Sacramento Perch	*Archoplites interruptus*	concern	M	M	M	No	Fry feed primarily on bottom-dwelling crustaceans; adults on insect larvae, snails, and fish. Functionally extinct in the Central Valley. Isolated populations in Clear Lake, other reservoirs and ponds throughout western states. Protection of other fish species will afford protection for the perch.
Sacramento Splittail	*Pogonichthys macrolepidotus*	concern	H	H	M	Yes	Vulnerable to selenium as clam-eating bottom feeder in the SF estuary (as adult) and on San Joaquin River floodplain spawning areas (as juveniles) in San Joaquin Valley.
Amphibians							
California Red-legged Frog	*Rana draytonii*	threatened	M	M	H	Yes	Range includes areas where selenium exposure to natural selenium and elevated levels due to human disturbance (e.g. erosion, mining) is possible. Representative for other amphibians.
Mountain Yellow-legged Frog	*Rana muscosa*	endangered	M	L	H	No	Range does not generally include areas where selenium is considered a problem. Represented by CRLF.
Sierra Nevada Yellow-legged Frog	*Rana sierrae*	endangered	M	L	H	No	Range does not generally include areas where selenium is considered a problem. Represented by CRLF.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
Yosemite Toad	*Anaxyrus canorus*	threatened	M	L	H	No	Range does not generally include areas where selenium is considered a problem. Represented by CRLF.
Arroyo Toad	*Anaxyrus californicus*	endangered	M	M	H	Yes	Habitat in coastal and inland southern California may expose the toad to natural selenium and elevated levels due to human disturbance.
Oregon Spotted Frog	*Rana pretiosa*	threatened	M	L	H	No	Range does not generally include areas where selenium is considered a problem. No critical habitat in CA. Represented by CRLF.
Santa Cruz Long-toed Salamander	*Ambystoma macrodactylum croceum*	endangered	M	L	H	No	Range and occupied habitat does not generally include areas where selenium is considered a problem. Represented by CRLF.
Desert Slender Salamander	*Batrachoseps major aridus*	endangered	L	L	H	No	Potential diet and exposure are low for this rare species.
California Tiger Salamander	*Ambystoma californiense*	endangered	M	L	H	No	Some potential for exposure during aquatic phase at some locations but in general low risk. Represented by CRLF.
Invertebrates							
Conservancy Fairy Shrimp	*Branchinecta conservatio*	endangered	M	L	H	No	Vernal pool invertebrate species have low potential for exposure and invertebrates in general are not very sensitive to selenium.
Longhorn Fairy Shrimp	*Branchinecta longiantenna*	endangered	M	L	H	No	Vernal pool invertebrate species have low potential for exposure and invertebrates in general are not very sensitive to selenium.
Riverside Fairy Shrimp	*Streptocephalus woottoni*	endangered	M	L	H	No	Vernal pool invertebrate species have low potential for exposure and invertebrates in general are not very sensitive to selenium.

Common Name	Scientific Name	Federal Status	Diet	Exposure	Status	Further Review	Potential to be adversely affected by selenium
San Diego Fairy Shrimp	*Branchinecta sandiegonensis*	endangered	M	L	H	No	Vernal pool invertebrate species have low potential for exposure and invertebrates in general are not very sensitive to selenium.
Vernal Pool Fairy Shrimp	*Branchinecta lynchi*	threatened	M	L	H	No	Vernal pool invertebrate species have low potential for exposure and invertebrates in general are not very sensitive to selenium.
Vernal Pool Tadpole Shrimp	*Lepidurus packardi*	endangered	M	L	H	No	Vernal pool invertebrate species have low potential for exposure and invertebrates in general are not very sensitive to selenium.
California Freshwater Shrimp	*Syncaris pacifica*	endangered	M	L	H	No	Species has low potential for exposure and invertebrates in general are not very sensitive to selenium.
Shasta Crayfish	*Pacifastacus fortis*	endangered	M	L	H	No	Species has low potential for exposure and invertebrates in general are not very sensitive to selenium.

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.

Table 2. Species most at risk from selenium exposure in California inland surface waters, enclosed bays and estuaries: summary data.

Common Name	Scientific Name	Probable critical life stage for selenium effects[1]	Food ingestion rate (g wet weight per day)[2, 3]	Food ingestion rate (g dry weight per day)[3]	Body weight at critical life stage (g)[4]	Diet
Buena Vista Lake Ornate Shrew	*Sorex ornatus relictus*	Juvenile and adult	6.7 - 12.8	1.7 - 3.2	4 - 7.6	Terrestrial and aquatic invertebrates
American Dipper	*Cinclus mexicanus*	Adult female (egg laying)	33	9.1 (59)	55	Benthic macroinvertebrates, fish eggs, small fish
California Brown Pelican	*Pelecanus occidentalis californicus*	Adult female (egg laying)	500	228 (46)	2800	Anchovy, sardines, other surface-schooling fish
Bald Eagle	*Haliaeetus leucocephalus*	Adult female (egg laying)	644	249 (63)	5275	Fish, birds, mammals
Ridgeway's Rail	*Rallus obsoletus obsoletus*	Adult female (egg laying)	172	46.8 (39)	346	Mussels, spiders, clams, crabs, snails, cordgrass seeds
Ridgeway's Light-footed Rail	*Rallus obsoletus levipes*	Adult female (egg laying)	158	43.2 (39)	312	Mussels, spiders, clams, crabs, snails, marsh cordgrass seeds
Ridgeway's Yuma Rail	*Rallus obsoletus yumanensis*	Adult female (egg laying)	117	31.9 (39)	210	Crayfish, clams, and other aquatic invertebrates
California Black Rail	*Laterallus jamaicensis coturniculus*	Adult female (egg laying)	24.8	6.77 (39)	28	Aquatic invertebrates, seeds, clams
California Least Tern	*Sternula antillarum browni*	Adult female (egg laying)	33.9 (40)	9.25 (39)	42	Small, surface-schooling fish

Common Name	**Scientific Name**	**Probable critical life stage for selenium effects[1]**	**Food ingestion rate (g wet weight per day)[2, 3]**	**Food ingestion rate (g dry weight per day)[3]**	**Body weight at critical life stage (g)[4]**	**Diet**
Giant Garter Snake	*Thamnophis gigas*	Adult female	15.3	3.8	250	Fish, crayfish, other aquatic invertebrates, amphibians
California Red-legged Frog	*Rana draytonii*	Tadpole Adult female	0.35 2.89	0.035 0.61	5 T 72.2 A	Juv.-Algae, organic detritus Adult-Invertebrates, primarily terrestrial, fish, frogs, even mice
Arroyo Toad	*Anaxyrus californicus*	Tadpole Adult	0.037 1.2	0.0037 0.38	0.46 T 30.8 A	Algae, organic detritus, diatoms Terrestrial invertebrates
Chinook Salmon	*Oncorhynchus tshawytscha*	Migrating/rearing juvenile		0.012 - 0.41	0.5 -18 J	Insects, crustaceans, juvenile fish
Steelhead	*Oncorhynchus mykiss*	Migrating/rearing juvenile		0.03 - 0.45	1.3 – 22.6 J	Insects, annelids, *Daphnia*
Little Kern Golden Trout	*Oncorhynchus aguabonita whitei*	Juvenile Adult female		0.0056 0.73	0.7 J 91 A	Aquatic and terrestrial invertebrates
Lahontan Cutthroat Trout	*Oncorhynchus clarkii henshawi*	Juvenile Adult female		0.0036 2.1	0.45 J 258 A	Aquatic and terrestrial invertebrates
Paiute Cutthroat Trout	*Oncorhynchus clarkii seleniris*	Juvenile Adult female		0.0036 1.4	0.45 J 179 A	Aquatic and terrestrial invertebrates

Common Name	Scientific Name	Probable critical life stage for selenium effects[1]	Food ingestion rate (g wet weight per day)[2, 3]	Food ingestion rate (g dry weight per day)[3]	Body weight at critical life stage (g)[4]	Diet
Desert Pupfish	*Cyprinodon macularius*	Adult female		0.01 - 0.05	0.22	Insect larvae, detritus, aquatic vegetation, snails
Owens Pupfish	*Cyprinodon radiosus*	Adult female		0.003 - 0.02	0.76	Aquatic insects, detritus, aquatic vegetation
Bonytail Chub	*Gila elegans*	Adult female		0.9	422	Aquatic and terrestrial invertebrates, aquatic vegetation , detritus
Razorback Sucker	*Xyrauchen texanus*	Adult female		9.5	3000	Algae, planktonic crustaceans, aquatic insect, plants, detritus
Santa Ana Sucker	*Catostomus santaanae*	Adult female		0.04	12.7	Detritus, algae, diatoms, aquatic insects
Tidewater Goby	*Eucyclogobius newberryi*	Adult female		0.001	0.25	Crustaceans, aquatic insects, molluscs
Unarmored Threespine Stickleback	*Gasterosteus aculeatus williamsoni*	Adult female		0.03	1.3	Benthic invertebrates, crustaceans, larval insects, zooplankton
Delta Smelt	*Hypomesus transpacificus*	Juvenile Adult female		0.037 0.057	0.32 J 2.1 A	Copepods, cladocerans, amphipods, insect larvae
Sacramento Splittail	*Pogonichthys macrolepidotus*	Juvenile Adult female		0.007 4.1	0.11 J 121 A	Benthic detritus, mollusks, mysids

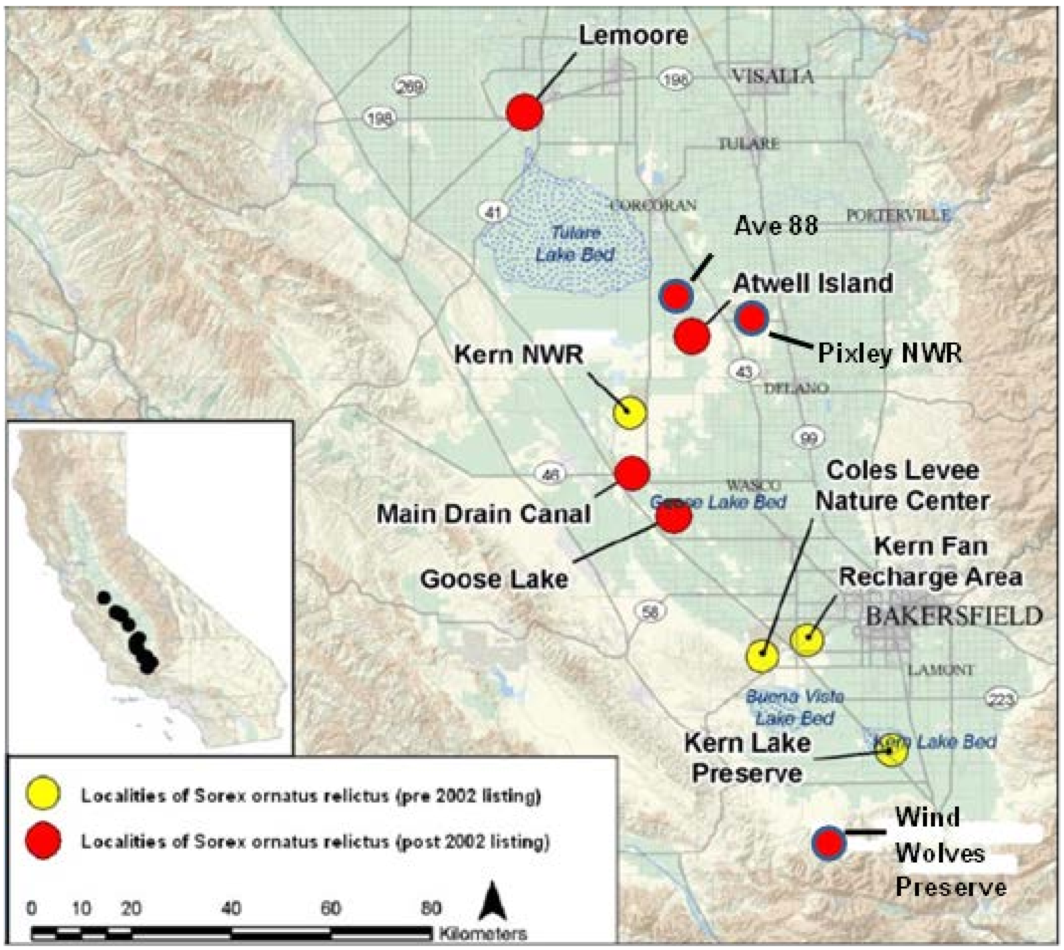
1 Juveniles are often the most sensitive life stage of an organism (Hutchinson et al. 1998). For Se, maternal transfer is a high risk factor for toxicity due to embryo exposure (Janz et al. 2010).
2 Food ingestion rates (at critical life stage) based on wet weight can be calculated from available parameters (Nagy 2001) for birds, mammals, reptiles and amphibians, but not, in general for fish.
3 For birds, the food ingestion rate (at critical life stage) 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 (specific equation parameters used from Table 3 (Nagy 2001) noted in cell).
4 For anadromous species, a range of body weights is given corresponding to the period juveniles spent rearing in freshwater and estuaries.

SPECIES INFORMATION

Buena Vista Lake Shrew (BVLS)
Sorex ornatus relictus

Status: Nine subspecies of ornate shrew, *Sorex ornatus*, exist, with the Buena Vista Lake Shrew subspecies, *Sorex ornatus relictus,* Federally listed as endangered in 2002 (67 FR 10101-10113). A 5-year review conducted in 2011 recommended no change to this listing.

Distribution: The Ornate shrew, *Sorex ornatus,* is distributed from the Sacramento area of California to Baja California in Mexico. Historically, the subspecies of Buena Vista Lake Shrew occupied marshlands throughout the southern end of the San Joaquin Valley from the Tulare Lake Basin to the vicinity of Bakersfield and Buena Vista Lake (BVLS Figure 1). Currently, there are known populations inhabiting at least 11 locations throughout the Tulare Basin at the south end of the San Joaquin Valley in California (USFWS 2011a).

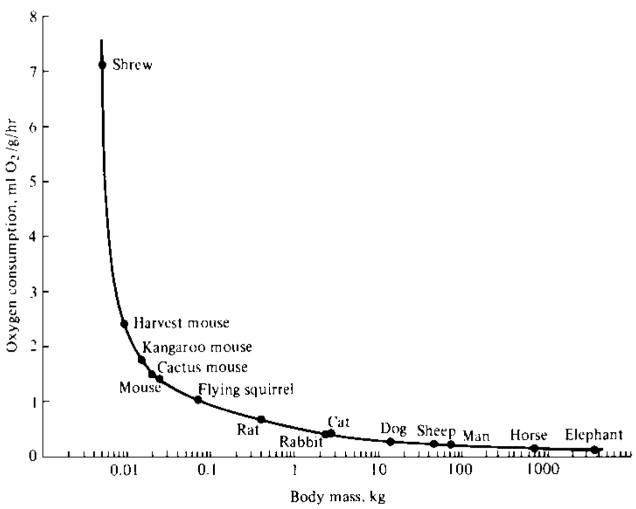


BVLS Figure 1. Buena Vista Lake Ornate Shrew Capture Locations 1999-2014 (adapted from USFWS 2011a).

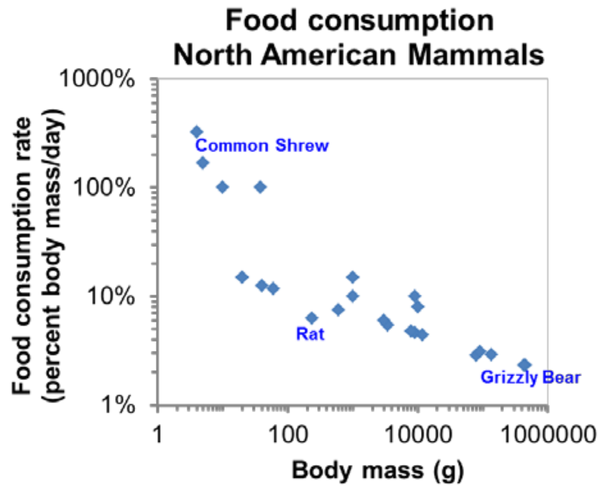
Size: Ranges of external measurements are: total length, 98 to 105 millimeters (3.86 to 4.13 inches); weights ranged from **4 to 7.6 grams** (Collins 1998).

Diet: The Buena Vista Lake Shrew is classified in the family *Soricidae* (shrews), whose members are mainly insectivorous and carnivorous, but some, including those of the genus *Sorex*, also eat some plant material; some members of the genus *Sorex* scavenge dead vertebrates (Walker 1975). Nothing more specific is known of the feeding habits of the Buena Vista Lake shrew, but it is likely that it feeds substantially on insects such as brine flies. Notable given that brine flies in turn feed in waters or marshy terrestrial habitats that may be contaminated by selenium in the Tulare Lake Basin.

Food Ingestion Rate: Shrews have an extremely high metabolic rate for their size (BVLS Figure 2), resulting in an extremely high rate of food consumption (Schmidt-Nielson, 1975). The vagrant shrew (*Sorex vagrans*), the closest shrew in size and taxonomy for which we have consumption data, consumes 168% of its body mass in food each day, 26.9 times greater than the food consumption rate of the rat (6.3% of body mass each day) according to data from the B.C. Environmental Protection Division (BVLS Figure 3; BC EPD 2001). For the BVLS 168% of its body mass (from above) would translate to **6.7 - 12.8 g/d ww**. Rust (1978) determined a food ingestion rate of 1.26 g/g/d ww for breeding shrews and 1.81 g/g/d ww for non-breeding shrews. For breeding BVLS this would translate to 5 - 9.6 g/d ww which overlaps the lower end of the range estimated above. Assuming 75% moisture a dry weight ingestion rate for the BVLS would be **1.7 – 3.2 g/d dw**.



BVLS Figure 2. Basal metabolic rate (measured as rate of oxygen consumption per unit of body mass) for a wide variety of mammals plotted against body mass (Schmidt-Nielson 1975).



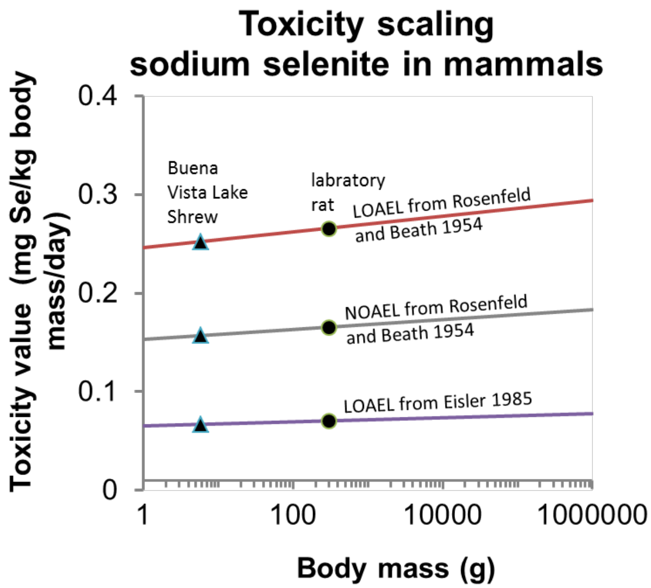
BVLS Figure 3. Food consumption rate as percent of body mass per day for North American mammals (data from Ministry of Environment, B.C., Canada (http://www.env.gov.bc.ca/wat/wq/reference/foodandwater.html#northamer)

Selenium toxicity: No literature was found to indicate shrews have yet been tested for selenium toxicity. Among mammals that have been tested, mice and rats are closest to shrews in size and habitat. Laboratory rats (presumably *Rattus norvegicus*) exposed to 2.5 mg/L of selenium as potassium selenite in drinking water suffered a 50% reduction (relative to controls) in survival of young produced by second generation mothers (Rosenfeld and Beath 1954). This concentration is the LOAEL (Lowest Observed Adverse Effect Level) for the experiment because the next lower treatment (1.5 mg/L of selenium as potassium selenite in drinking water) yielded no observed adverse effect. To be useful, the LOAEL concentration in drinking water must be translated into an overall dietary concentration of selenium. The rats in this experiment were fed Purina Cat Chow, which was evidently not analyzed for selenium. Our best estimate of the selenium content in the food is the minimum selenium concentration of 0.3 mg Se/kg in Purina Cat Chow currently published on the internet (https://www.catchow.com/products/complete-formula/). The best available estimate of the ratio of water to food consumption of the rat is 2:1 (http://web.jhu.edu/animalcare/procedures/rat.html). Therefore we estimate that each kg of food containing 0.3 mg selenium was accompanied by 2 kg of water containing 2 x 2.5 mg of Se, yielding an overall dietary (food and water) concentration of 1.7 mg/kg wet weight, or 6.02 mg/kg dry weight, assuming that the moisture content of the Purina Cat Chow was 12% (the maximum according to https://www.catchow.com/products/complete-formula/). This concentration corresponds to a LOAEL daily selenium load of 0.27 mg Se per kg of body mass per day, assuming the mass of the rat is 300 g and its daily food intake is 15 g (http://web.jhu.edu/animalcare/procedures/rat.html).

Eisler (1985) listed a lower LOAEL (dietary) of 1.4 mg/kg (dry weight) causing liver changes in rats fed natural selenium. Using the same rat mass and food consumption rate cited above, this corresponds to a LOAEL daily selenium load of 0.070 mg Se per kg of body mass per day.

Small rodents such as rats are considered to be comparatively sensitive to toxic effects of selenium (Eisler 1985). However, Sample and Arenal (1999) found the increase in sensitivity with decreasing size to be slight (allometric scaling factor of 1.013 for sodium selenite, the only form of selenium they tested; BVLS Figure 4). Using that scaling factor with their model, the rat LOAEL from Rosenfeld and Beath (1954), 0.27 mg Se per kg of body mass per day for the rat, translates into 0.25 mg Se per kg of body mass per day for the Buena Vista Lake Shrew (assuming the shrew average mass is 5.85 g, in the middle of the range of mass given in http://esrp.csustan.edu/speciesprofiles/profile.php?sp=soor for the Buena Vista Lake Shrew). Similarly, the LOAEL from Eisler (1985), 0.070 mg Se per kg of body mass per day (mg Se/kg-bw/d) for the rat, translates into 0.067 mg Se per kg of body mass per day (mg Se/kg-bw/d)for the Buena Vista Lake Shrew.

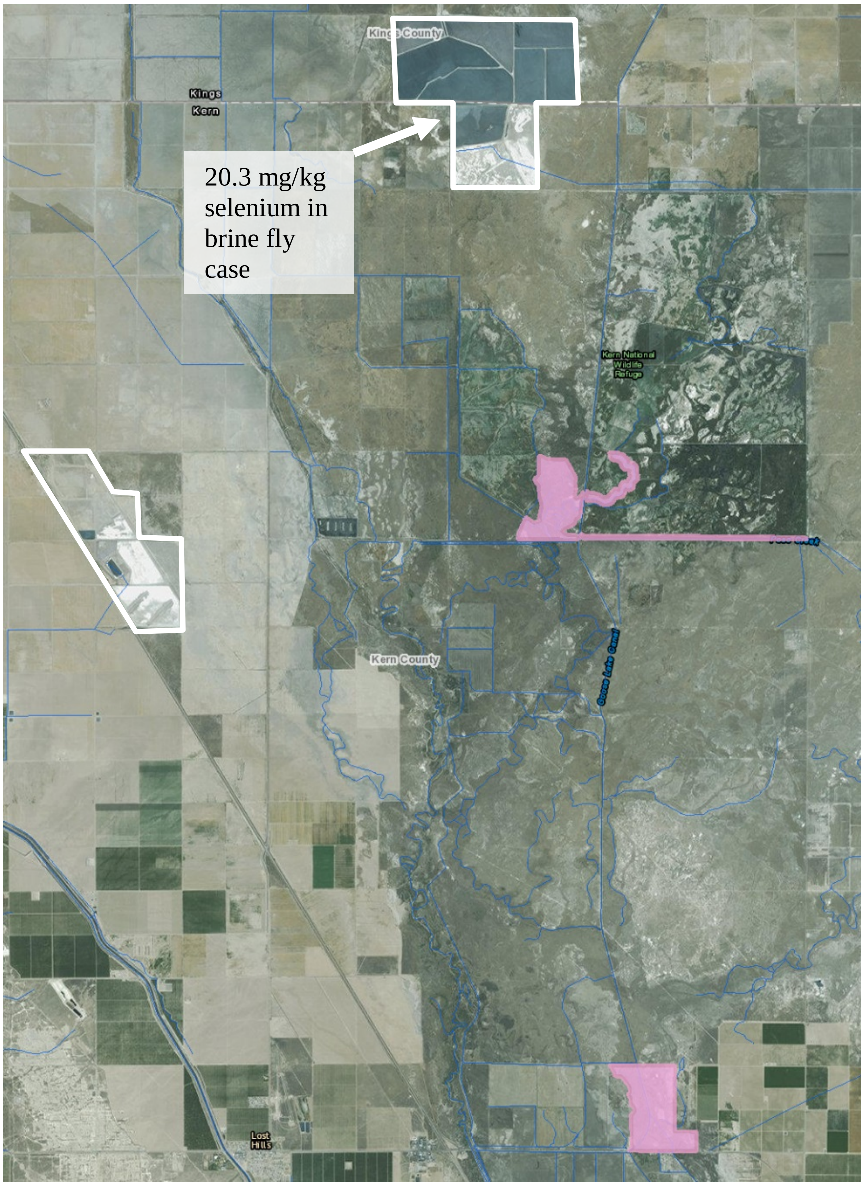
Assuming that assimilation of selenium is proportional to food intake rate, then the LOAEL for the Buena Vista Lake Shrew on a concentration basis, must be about 1/26.9 the LOAEL dietary concentration for the rat (1.4 mg/kg dry weight diet from Eisler 1985). This is a LOAEL of 0.049 mg Se/kg dry weight diet for the Buena Vista Lake Shrew (including allometric adjustment). The rat dietary concentration LOAEL from Rosenfeld and Beath (1954), 6.02 mg/kg dry weight diet, similarly translates into a LOAEL of 0.21 mg Se/kg dry weight diet for the Buena Vista Lake Shrew (including allometric adjustment).



BVLS Figure 4. Allometric scaling of toxicity of sodium selenite

Risk of selenium exposure: Selenium toxicity represents a serious threat to the shrew, due to its habitat requirements, trophic level, and extraordinarily high food consumption rate. The soils on the western side of the San Joaquin Valley have naturally elevated selenium concentrations. Due to extensive agricultural irrigation, selenium has been leached from the soils and concentrated in the shallow groundwater along the western side of the San Joaquin Valley. In areas where this groundwater reaches the surface or subsurface, selenium can accumulate in both plants and animals. Selenium can then enter the food chain of the shrew by becoming concentrated in insects that forage on the vegetation or reside in soils that concentrate these salts and result in adverse effects to growth, reproduction, and survival of the shrew (Saiki and Lowe 1987; Moore et al. 1989). Elevated concentrations of selenium in insects have been measured in many potential Buena Vista Lake shrew prey species such as brine flies (*Ephydridae*), damselflies (*Zygoptera*), midges (*Chironomidae*), and other insects collected at agricultural drainage evaporation ponds throughout the Tulare Basin, including ponds a few miles west of the Kern Preserve and along the northern border of the Kern NWR (Moore et al. 1989).

The preferred habitat of the BVLS is thought to be dense marshy and riparian vegetation along streams, canals, and sloughs, and around the edges of remaining ponded surface water in the Tulare Basin at the south end of the San Joaquin Valley (Collins 1998). The natural lakes (Buena Vista Lake and probably Tulare Lake) around which the shrew once occurred have been replaced to a large extent with facilities for conveying, storing, evaporating (“evaporation basins”) and treating subsurface agricultural drainwater and effluent from dairies (67 FR 10101-10113). These facilities result in surface water and groundwater containing elevated levels of selenium in or near the last remaining habitats of the Buena Vista Lake Shrew (BVLS Figure 5).



BVLS Figure 5. Evaporation basins near USFWS designated critical habitat (pink) of the Buena Vista Lake shrew (from California Department of Fish and Wildlife BIOS with additional labels)

In recent years, attention has been focused on monitoring selenium in bird eggs because it has been thought that bird egg viability is the most sensitive endpoint for adverse selenium effects on wildlife. Therefore, very little data has been collected on selenium concentrations in dietary items for the Buena Vista Lake shrew. The single brine fly case sampled from Pond 10 of the Tulare Lake Drainage District South evaporation basin in June, 1987, had a selenium concentration of 20.3 mg/kg (Moore et al. 1989), far above the dietary threshold levels (3.0 µg/g dw) for avian species (Lemly 1996a, 1996b; Ohlendorf and Heinz 2011, USDOI 1998) and orders of magnitude greater than thresholds calculated above for shrews (0.05 and 0.2 mg/kg). In recent years of severe drought in California, many of the ponds in the evaporation basins in the Tulare Basin have dried up; some of the remaining ponds have extremely elevated concentrations of selenium in the water (360 µg/l in Westlake Farms' evaporation basin Cell B North in October 2015; H.T. Harvey and Associates 2015).

Clark (1987, 1989) collected small mammals, including California voles at Kesterson Reservoir in 1984. He found selenium concentrations of 13 and 33 µg/g (mean 23.0 µg/g) in California voles collected at Pond 2 of Kesterson Reservoir. The average selenium concentration in all California voles collected at all ponds of the reservoir (n=5) was 10.4 µg/g. The average selenium concentrations in small mammals at Kesterson Reservoir while the ponds were operational was as follows:

Species	**Number Collected**	**Mean Selenium Concentration** (µg/g whole body dry wt.)
House mouse	5	18.5
Western harvest mouse	5	12.5
Ornate shrew	4	47.9
California vole	5	10.4

Seleniferous uplands that usually lack ponded water are best represented by data from Kesterson after it was closed and low-lying areas were filled (CH2MHILL 1999). This data is as follows:

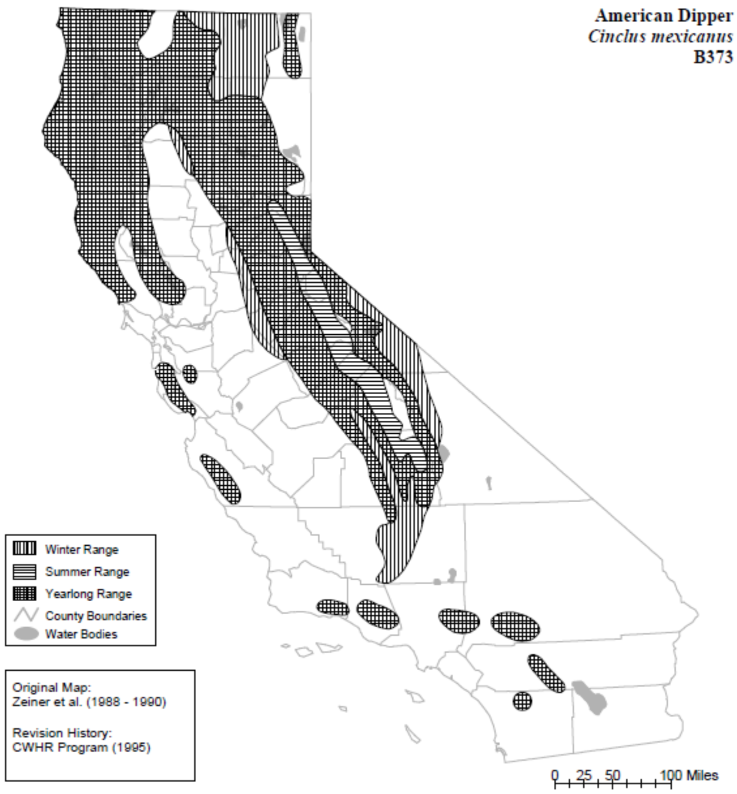
Species	**Number Collected**	**Mean Selenium Concentration** (µg/g whole body dry wt.)
House mouse	31	7.9
Western harvest mouse	17	7.7
Ornate shrew	1	7.5
Deer mouse	30	6.7
California vole	7	4.4

Water supplies to wetland areas with BVLS may include the California Aqueduct, groundwater pumping, and eastside streams such as Poso Creek. At Kern NWR water supplies primarily come from Goose Lake Canal which is supplied by water from the California Aqueduct and Poso Creek especially during wet years (Esralew et al. 2015). Monthly selenium concentrations in the California Aqueduct are usually 1 µg/L or below but can reach 2 µg/L and at least once 3 µg/L in recent years (California Department of Water Resources, State Water Project Operations and Maintenance, http://www.water.ca.gov/swp/waterquality/). Co-mingling wetland water supplies with groundwater, tailwater, or reuse water may increase selenium concentrations in water supplies to BVLS habitat.

American Dipper (AMDI)
Cinclus mexicanus

Status: The American dipper is protected under the Migratory Bird Treaty Act. Though dipper populations are difficult to count, numbers appear to be relatively stable. Partners in Flight (2012) estimate a global breeding population of 190,000 with 86% living in the U.S.

Distribution: The dipper can be found along mountain stream throughout the state of California, primarily within upper elevation watersheds of the Sierras and coastal ranges (AMDI Figure 1; Zeiner et al. 1990).



AMDI Figure 1. Distribution map of the American dipper (from: Zeiner et al. 1990).

Size: Adult American dippers grow to roughly 18 cm (7 in.) in length from beak to tail, and weighs on average 50 g (Robbins et al. 1966). Wilson and Kingery (2011) summarized dipper size information noting that males are 10% -16% larger than females with *C.m. unicolor* males weighing on average 61 g (57-66) and females weighing **55 g** (43-65).

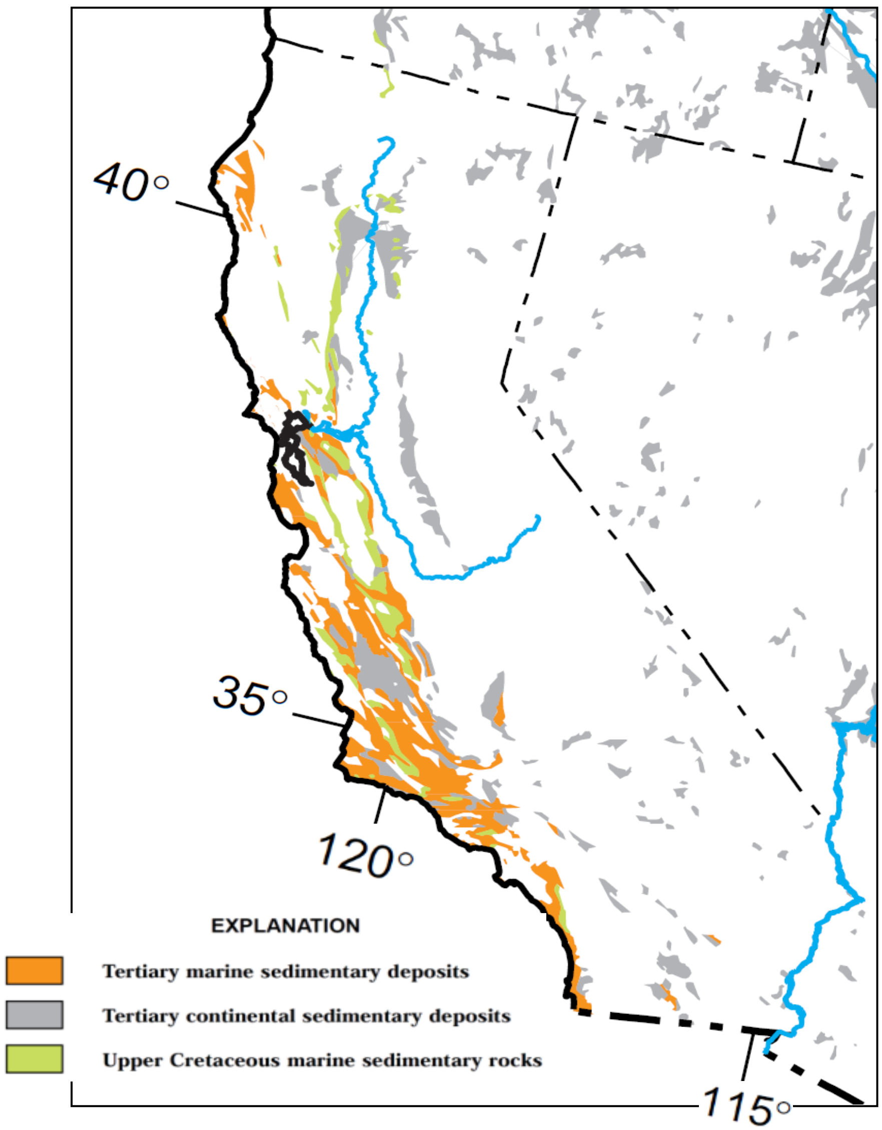
Diet: Aquatic insects, especially larvae attached to river bottoms, make up the majority of the American dipper's diet. They also eat other small aquatic creatures, including fish eggs and very small fish, and will feed at salmon spawning areas. The abundance of important dipper prey determines dipper presence (Feck 2002). Dippers primarily feed on aquatic insect larvae, or benthic macroinvertebrates (Price and Bock 1983). Mayflies (Ephemeroptera) and caddisflies (Trichoptera) dominate dipper diets, with stoneflies (Plecoptera) and Diptera also comprising a significant portion of the diets of American dippers (Tyler and Ormerod 1994). Morrissey et al. (2004) using isotopic signatures determined that resident dippers occupying river sites ate a higher percentage of fish (42% ± 7) than migrant dippers on the river tributaries (22% ± 6). This did not appear to make a difference in selenium exposure as dipper eggs from the river sites (2.96 µg/g dw) had nearly the same selenium concentrations as the tributary sites (2.67 µg/g dw). Morrissey et al. (2010, 2012) found that in American dippers, egg-laying females switched to feeding at a higher trophic level (i.e. consuming more fish and predatory invertebrates). Dipper diet varies geographically and locally (Wilson and Kingery 2011). Whether dippers in any given watershed eat significant numbers of salmon fry or eggs may depend on availability of salmon spawning habitat.

Food Ingestion Rate: No data has been found that specifically documents *Cinclus mexicanus* ingestion rates, but Nagy (2001) estimated a daily intake rate for the similarly sized *Cinclus cinclus*. Using the equation $y = a(\text{grams body mass})^b$ and Table 3 equation 59 (dw) and 60 (ww) parameters for insectivorous birds from Nagy (2001) results in ingestion rates of **27.5 g/d ww** and **9.1 g/d dw**.

General Life History: The American dipper is a specialized bird species that inhabits mountain streams in the western half of North America. The American dipper is described as the only true aquatic songbird and is noted for its dipping feeding behavior. Foraging activity of dippers is varied: diving, swimming, wading at the river margins, turning stones and leaves, fly catching, and gleaning stone surfaces (Wilson and Kingery 2011).

Risk of selenium exposure: Birds are particularly sensitive to excessive dietary selenium. Given their unique feeding methods, and habitat requirements, the American dipper can serve as a useful bioindicator of selenium contamination and other water quality conditions in mountainous, lotic ecosystems (Tyler and Ormerod 1994, Feck 2002, Morrissey et al. 2005). Elevated levels of selenium in water and aquatic biota downstream from two open-pit coal mines in the Rocky Mountain foothills of Alberta resulted in significant increases of selenium in dipper eggs, and at high enough concentrations to warrant developmental concerns (Wayland et al. 2006). In a follow up risk assessment, Wayland et al. (2007) modeled a 20% hatch failure rate for the dipper that was associated with diet and egg selenium concentrations of 6.4 and 17 µg/g dw. respectively. Harding et al. (2005) documented a 20% hatching failure in dippers from a coal mining area where the mean egg selenium concentration was 8.4 µg/g dw; however, the sample size was insufficient to show statistical significance. The reference sites had hatching failure of 6% and a mean selenium concentration of 7.4 µg/g dw. It should be noted that dipper eggs at the reference sites for Harding et al. (2005) had selenium concentrations 2.5 times higher than coastal dipper eggs. Spotted sandpipers in the same study did have a significant 22% hatch failure at a mean egg selenium concentration of 7.4 µg/g dw (Harding et al. 2005). Morrissey (2005) noted that invertebrates in dipper habitat from the Chilliwack watershed in British Columbia, Canada had selenium concentrations of around 6 µg/g dw while salmon fry from the area had 2.7 µg/g dw. In their risk assessment no matter what dietary percentage (invertebrates/fish) was considered the dippers were still exposed to selenium levels above a tolerable daily intake.

High selenium concentrations are found in phosphoritic sedimentary rock such as marine shales and sulfide ore bodies (Mayland et al. 1989). The dippers range in California includes areas with Tertiary and Cretaceous marine geology known to have elevated selenium (AMDI Figure 2; Presser et al. 1990, Seiler et al. 1998). Watersheds in these areas may have elevated selenium levels in water especially if human disturbances are high. Water samples from Indian Creek, a tributary to the Klamath River in northern California have elevated selenium levels as high as 126 µg /L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). This may be a result of copper mining activity in the watershed. Selenium is associated with copper mines and is recovered from refinery slime for commercial purposes (Mayland et al. 1989). Henderson et al. (2013) documented elevated selenium levels below Leviathan Mine in the East Walker River watershed. Selenium levels ranged from <1 to 6.6 µg/L below the mine site in 2001. In 1999, Thompson and Welsh (2000) documented selenium levels in streams below Leviathan Mine that ranged from <1 to 13 µg/L.

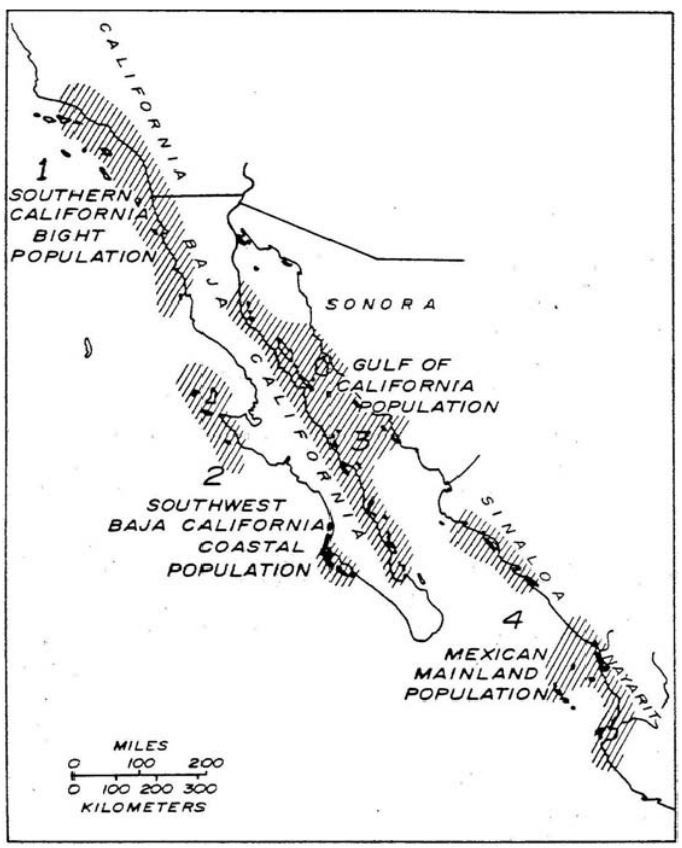


AMDI Figure 2. Areas of California with seleniferous marine geology. Modified from: Seiler et al. (1998).

California Brown Pelican (BRPE)
Pelecanus occidentalis californicus

Status: In 1970, USFWS listed the brown pelican as endangered. The California brown pelican was removed from Endangered Species Act (ESA) protections in 2009 based on overall population recovery (74 FR 59444). The brown pelican remains protected under the Migratory Bird Treaty Act (MBTA). The brown pelican was listed as endangered by the state in 1971 and delisted in 2009 (CDFW 2017).

Distribution: The most recent population estimate of the California brown pelican subspecies is approximately 70,680 nesting pairs, which equates to 141,360 breeding birds (Burkett et al. 2007). The only breeding colonies in California are in within the Southern California Bight, and within the Channel Islands National Park (West Anacapa and Santa Barbara islands). The Southern California Bight nesting population represents about 5% – 10% (6,000) of the California brown pelican population (USFWS 2007). Post-breeding adults and juveniles can be found roosting along the entire length of California's coast especially the central California coast where up to 75% of the population can be found in the fall (USFWS 2007). The Salton Sea is a major roosting area for non-breeding California brown pelicans (i.e., juveniles and sub-adults) during the post-breeding season. Currently 3,000-4,000 birds are regularly recorded in July and August (USFWS 2007).



BRPE Figure 1. Map of nesting populations of California brown pelican (USFWS 2007)

Size: Brown pelicans weigh an average of 2.4 to 4 kg with an overall length of 100 to 137 centimeters (Shields 2014). Males weigh 15% - 20% more than females (ave. **2.8 kg** for females to to 3.3 kg males) and generally have a longer bill than females (Shields 2014).

Diet: Along the California coast, the pelicans are highly dependent on small, surface-schooling fishes, including northern anchovy (*Engraulis mordax*) and Pacific sardines (*Sardinops sagax*) (Anderson et al. 1980; 1982). During their breeding season, 90% of the California brown pelican's diet consists of the northern anchovy. Fish available to brown pelican at the Salton Sea includes tilapia (*Oreochromis mossambicus*), orangemouth corvina (*Cynoscion xanthulus*), croaker (*Bairdiella icistia*), and sargo (*Anisotremus davidsoni*) (Reidel et al. 2001).

Food Ingestion Rate: No quantitative data on food intake of wild birds is found. Captive nestlings held indoors consumed average of 464–549 g/d (16.1%–17.3% of body mass; Schreiber 1976). Captive adult consumed average of 590 g fish/d, or 17.5% of daily body mass. Food requirements of wild birds are probably greater because of higher activity levels and cost of thermoregulation (Schreiber 1976). Twenty-three captive immatures and adults kept in outdoor pen in Florida consumed about **0.5 kg fish/d**; amount varied from 0.3 kg/d during summer to >0.8 kg/d on some winter days (Wolf et al. 1985). Using Table 3 and equation 46 parameters from Nagy (2001) results in an ingestion rate of **228 g/d dw**.

General Life History: In California, brown pelicans breed on isolated islands along the southern coast including West Anacapa Island and Santa Barbara Island (USFWS 2007). First breeding age is generally 3-5 years old while few birds live beyond 10 years (Shields 2014). Non-breeding pelicans and post-breeding adults and juveniles will move up and down the coast as far north as the Columbia River Estuary and east to the Salton Sea (Shields 2014, USFWS 2007). Weather and forage fish availability greatly impact nesting success especially during El Nino years (Jaques 2016).

Risk of Selenium Exposure: The Salton Sea is an important non-breeding roosting site for brown pelicans; however, die-offs of numerous bird species including thousands of brown pelicans have occurred in the Salton Sea (USFWS 2007). Selenium levels in livers of brown pelicans from a 1996 botulism die-off at the Salton Sea were at levels (16.9, range 13.0-34.3 µg/g dw.) known to cause immune system dysfunction and could have contributed to the die-off (Bruehler and de Peyster 1999; Fairbrother and Fowles 1990). White pelicans from the same die-off study had similar liver selenium levels. Control liver samples from the Bruehler and de Peyster (1999) study had a mean selenium level of 9.3 µg/g dw. (range, 4.4-13.1 µg/g dw.) Selenium levels in water and fish from the Salton Sea are known to have elevated levels. Tilapia composite filet samples in 2007 had selenium levels ranging from 10.6 to 17.1 µg/g dw. (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). Water selenium concentrations at the Salton Sea, tributaries, and drains are well above 2.0 µg/L with some higher than 30 µg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool).

Bald Eagle (BAEA)

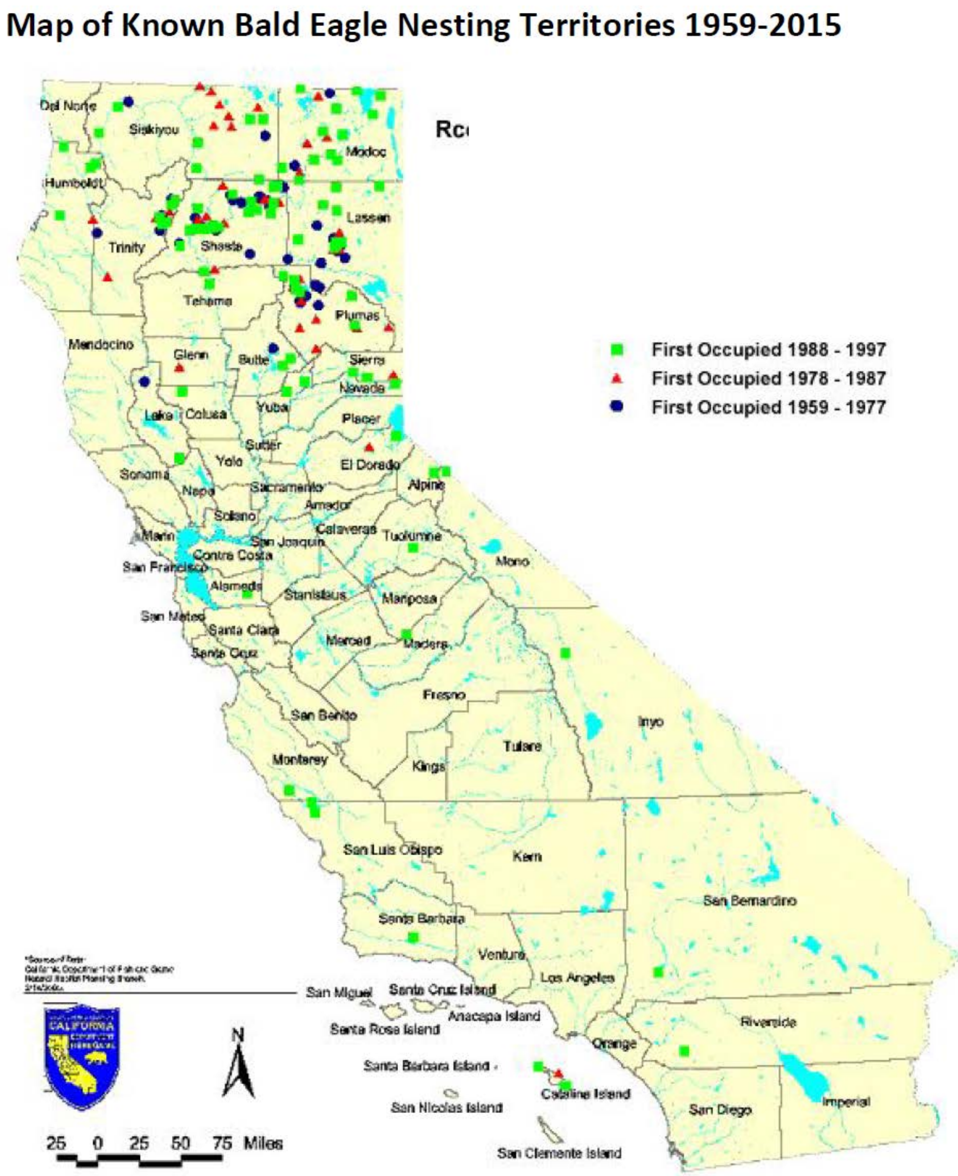
Haliaeetus leucocephalus

Status: The bald eagle was federally listed as endangered in 1978 (43 FR 6233) in all of the coterminous United States except Minnesota, Wisconsin, Michigan, Oregon, and Washington, where it was classified as threatened. In 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). In August 2007 the species was removed 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) remain in place after the species was delisted. California listed the bald eagle as endangered in 1971 and is still listed as endangered (CDFW 2017).

Distribution: Bald eagles in winter may be found throughout most of California at lakes, reservoirs, rivers, and some rangelands and coastal wetlands. The State's breeding habitats are mainly in mountain and foothill forests and woodlands near reservoirs, lakes, and rivers. Most breeding territories are in northern California, but the eagles also nest in scattered locations in the central and southern Sierra Nevada Mountains and foothills, in several locations from the central coast range to inland southern California, and on Santa Catalina Island (Jurek 1990, Sorenson et al. 2017). There are over 300 bald eagle breeding territories in California while over 1,000 birds are estimated to winter in California (BAEA Figures 1 and 2).

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% 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.

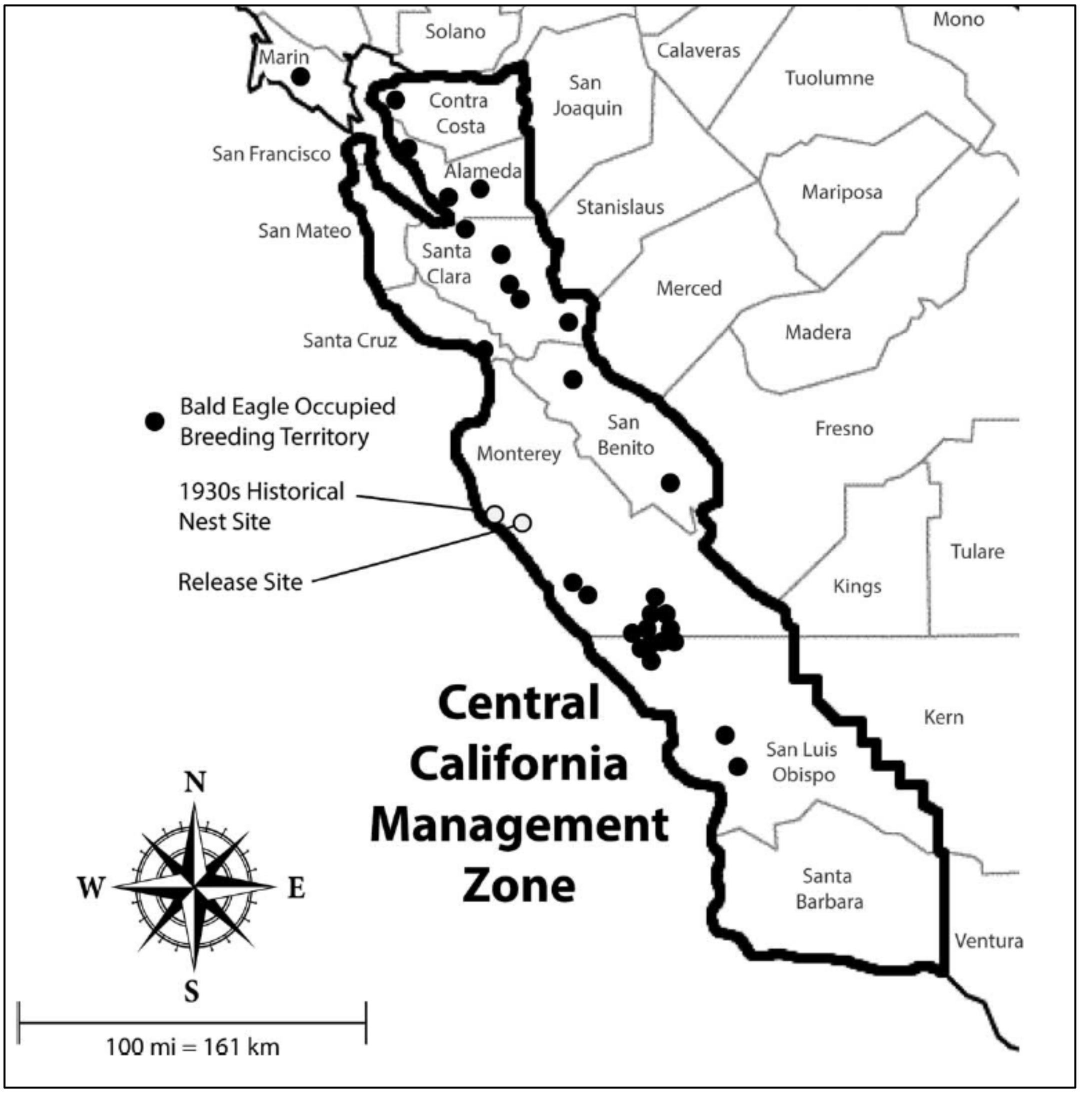
In the GLI, the USEPA presented an average body weight of 5.2 kg for female bald eagles. This value was based on the weights of 37 birds, taken from Snyder and Wiley (1976). Dunning (1993) presented an average female body weight of 5.35 kg, also based on the weights of 37 birds, taken from Palmer (1988).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



BAEA Figure 1. Bald eagle nesting sites in California (https://www.wildlife.ca.gov/Conservation/Birds/Bald-Eagle**).**

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).



BAEA Figure 2. Bald eagle breeding territories along the Central Coast of California (modified from Sorenson et al. 2017)

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 food ingestion rate (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 FIR using the generic allometric equation (64) 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% of the dietary biomass comprised fish and 8% 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% fish and 8% 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% fish, 22.8% birds, and 6% 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:

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

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

$$\begin{aligned} FIR &= FMR / ME_{avg} \\ &= 703.3/1.09 \\ &= \underline{\mathbf{644\ g/day}\ \text{wet weight}} \end{aligned}$$

The Nagy (2001) Table 3 equation 63 parameters (carnivorous bird) provides a dry weight food ingestion rate of **249 g/d dw**.

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, Sorenson et al. 2017). 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 (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. 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.

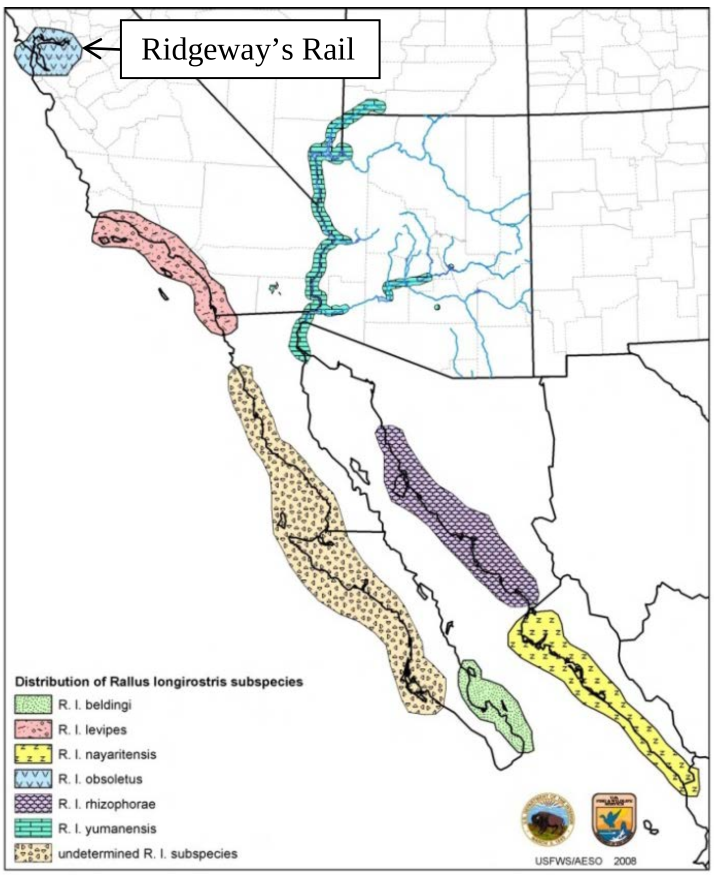
Bald eagles outside the San Francisco Estuary may be exposed to selenium if they are feeding in rivers, lakes or reservoirs where elevated selenium levels have been documented. The Bioaccumulatives Oversight Group (BOG) of the Surface Water Ambient Monitoring Program (SWAMP) has analyzed selenium in fish at over 250 river, lake, and reservoir locations in California from 2007-2011. Forty five fish samples of the 295 tested (15.3%) had selenium concentrations above 4 µg/g dw while 25 samples (8.5%) had selenium concentrations over 8 µg/g dw. (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). These sites were all lakes or reservoirs predominantly in Southern California (BAEA Figure 3). These are levels that can potentially cause problems for piscivorous birds.

BAEA Figure 3. Lakes and reservoirs in California where selenium levels in fish were greater than 4 and 8 µg/g dw.

Ridgeway's Rail (RIRA)
Rallus obsoletus obsoletus

Status: The Ridgeway's Rail (formerly the California Clapper Rail) was federally listed as endangered on October 13, 1970 (35 FR 16047-16048) and as state endangered in 1971 (CDFW 2017).

Distribution: Ridgeway's Rails are found throughout tidal marsh environments of the San Francisco Bay and Delta, where it is restricted to about 10% of its historic range (RIRA Figure 1). Population numbers reached an all-time low of about 500 birds in 1991, then rebounded somewhat. Results of an estuary-wide survey estimated a minimum average population between 2005 and 2008 of 1,425 rails; however, population numbers declined during that period at a per-year rate of 20%, as habitat was lost bay-wide, and are currently lower (Liu et al. 2012).



RIRA Figure 1. Subspecies ranges of Rallus rails in the western U.S. and Mexico. Ridgeway's rail are shown in blue, within the San Francisco Bay and Delta (modified from USFWS 2010a).

Size: In the only literature found for this particular subspecies that provided body weights, 19 female Ridgeway's 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 Ridgeway's 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% 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% 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% 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% 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 Ridgeway's 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 Ridgeway's rail diet, and the knowledge that Ridgeway's 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% vegetation and 90% animal matter.

Food Ingestion Rate: Ridgeway's 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, the 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 Ridgeway's rails was calculated using the following equation (Nagy 2001, Table 3, equation 40 parameters):

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**

Using Table 3 and the dry weight equation 39 parameters from Nagy (2001) results in an ingestion rate of **46.8 g/d dw**.

General life history: The Ridgeway's 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: The Ridgeway's 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. Ridgeway's 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 Ridgeway's rails collected from the north San Francisco Bay in 1987 contained up to 7.4 μg/g selenium (North Bay mean 4.00 μg/g, range 1.60–7.40 μg/g). Water data from this time and location are not available. Based, in part, on the data for Ridgeway's rails, staff technical reports prepared for the San Francisco Bay Regional Water Quality Control Board recommend decreasing selenium loading to the estuary by 50% or more (Taylor et al. 1992; 1993). More recent data on Ridgeway's rail failed to hatch eggs have shown a decline of 50% in selenium (North Bay mean 1.98 μg/g, range 1.12–3.13 μg/g) since implementation of discharge controls (Schwarzbach et al. 2006).

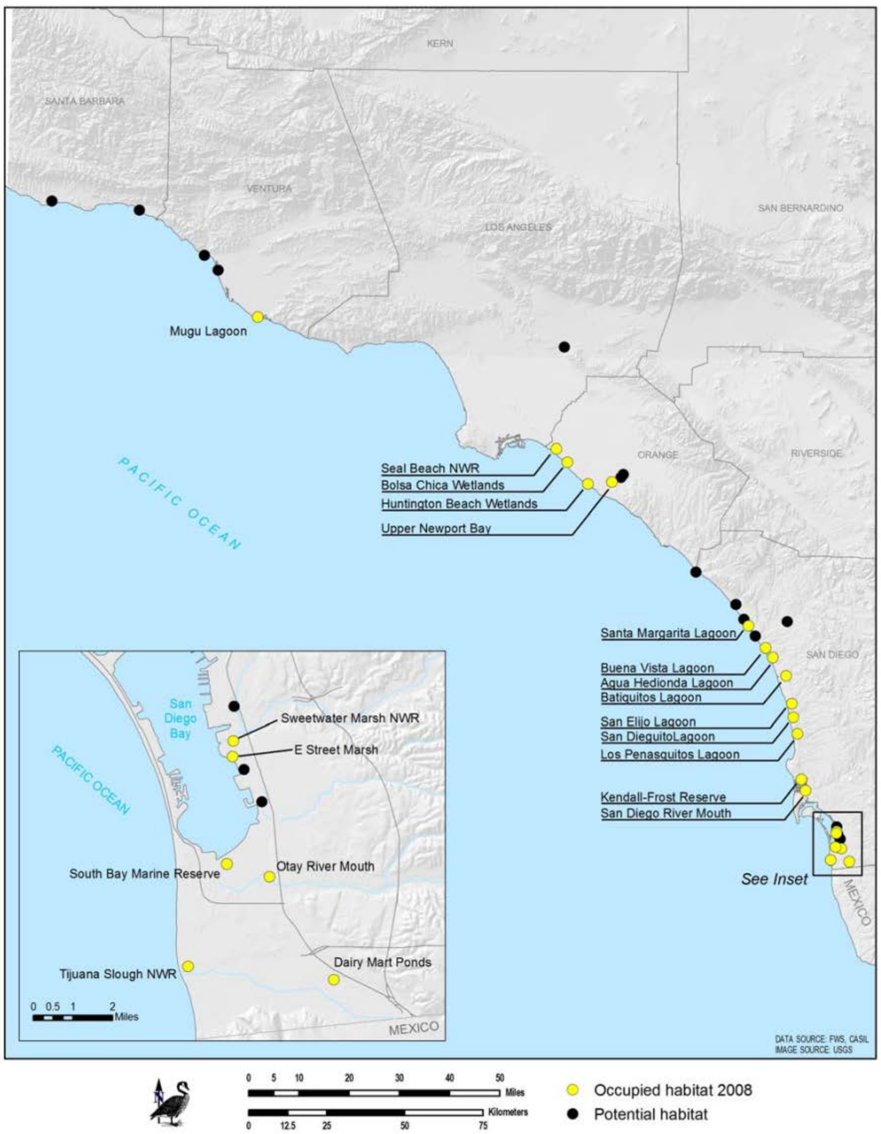
The *in ovo* threshold for selenium exposure that causes toxic effects on embryos of Ridgeway's rails is unknown. For another benthic-foraging marsh bird, the black-necked stilt, the *in ovo* threshold for embryotoxicity is 6 μg/g selenium (Skorupa 1998). Skorupa reanalyzed his black-necked stilt egg data for the Santa Ana Regional Water Quality Control Board (SARWQCB) using EPA's Toxicity Relationship Analysis Program (TRAP). The results showed a no effect concentration of 5.8 μg Se/g dw and a lower 95% confidence bound on the estimated EC10 of 10.2 μg Se/g dw (SARWQCB 2017).

Beckon et al. (2008) applied a biphasic model to Heinz et al. (1989) data from laboratory experiments with mallard reproduction. Their analysis indicates that in mallard eggs there was a 10% reduction in hatchability at 7.7 μg/g dw. These results are in line with Skorupa's black-necked stilt analyses.

Ridgeway's Light-footed Rail (RLRA)
Rallus obsoletus levipes

Status: Ridgeway's Light-footed Rails (formerly Light-footed Clapper Rail) were listed as endangered in 1970 (35 FR 16047). California listed the RLRA as endangered in 1971 (CDFW 2017).

Distribution: Ridgeway's Light-footed Rails are resident in coastal wetlands, as well as inhabiting coastal marshes, lagoons, and their maritime environs. Found exclusively in southern California from Mugu Lagoon (Santa Barbara County) to Tijuana Slough NWR (San Diego County; RLRA Figure 1; see also RIRA Figure 1).



RLRA Figure 1. Map of known locations of Ridgeway's Light-footed Rail. From USFWS (2009b) 5-yr Review.

Size: The light-footed rail is a hen-sized bird, averaging approximately 36 centimeters in length, and weighing approximately 227-398 g (mean **312g**; Thelander and Crabtree, 1994).

Diet: Light-footed clapper rails are omnivorous and opportunistic foragers, which rely mostly on salt marsh invertebrates such as mussels, snails, fiddler and hermit crabs, fish, crayfish, isopods, and beetles (USFWS 1985; Zembal and Fancher 1988).

Food Ingestion Rate: Using the same methodology as described above for Ridgeway's Rail (Nagy 2001, Table 3, equation 40 parameters), and an average body weight of 312g:

FIR (wet weight) = $1.914 \times (\text{body weight in g})^{0.769}$
FIR = $1.914 \times 312^{0.769}$
FIR = **158 g/day wet weight**

Using Table 3 and equation 39 parameters from Nagy (2001) results in an ingestion rate of **43.2 g/d dw**.

General Life History: The Ridgeway's light-footed rail inhabits coastal salt marshes and lagoons, nesting in habitat including: dense cordgrass (*Spartina foliosa*) and pickleweed (*Salicornia virginica*), wrack deposits and hummocks of high marsh (Massey et al. 1984). Fringing areas of high marsh serve as refugia during high tides (Zembal et al. 1989). Although less common, light-footed clapper rails have also been observed to reside and nest in freshwater marshes (Thelander and Crabtree 1994). They require shallow water and mudflats for foraging, with adjacent higher vegetation for cover during high water (Zeiner et al. 1990). They forage in all parts of the salt marsh, concentrating their efforts in the lower marsh when the tide is out, and moving into the higher marsh as the tide advances (USFWS 2009b).

Risk of Selenium Exposure: There are similar risks to selenium exposure between all Ridgeway's rails; however, specific food item differences and site-specific selenium sources may impact overall risks for each subspecies. Ridgeway's light-footed rail is likely very similar to Ridgeway's rail (see discussion above for RIRA). Selenium water concentrations in some coastal streams of Southern California are documented to be well above 5 µg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool).

Allen et al. (2004) documented elevated selenium levels in small forage fish from Upper Newport Bay in Southern California. Composite samples of arrow goby, California killifish, and topsmelt were well above the 3 µg/g dw screening level (7.95, 9.8, and 7.98 µg/g dw respectively). All fish samples from across Newport Bay ranged from 0.92-9.5 µg/g dw.

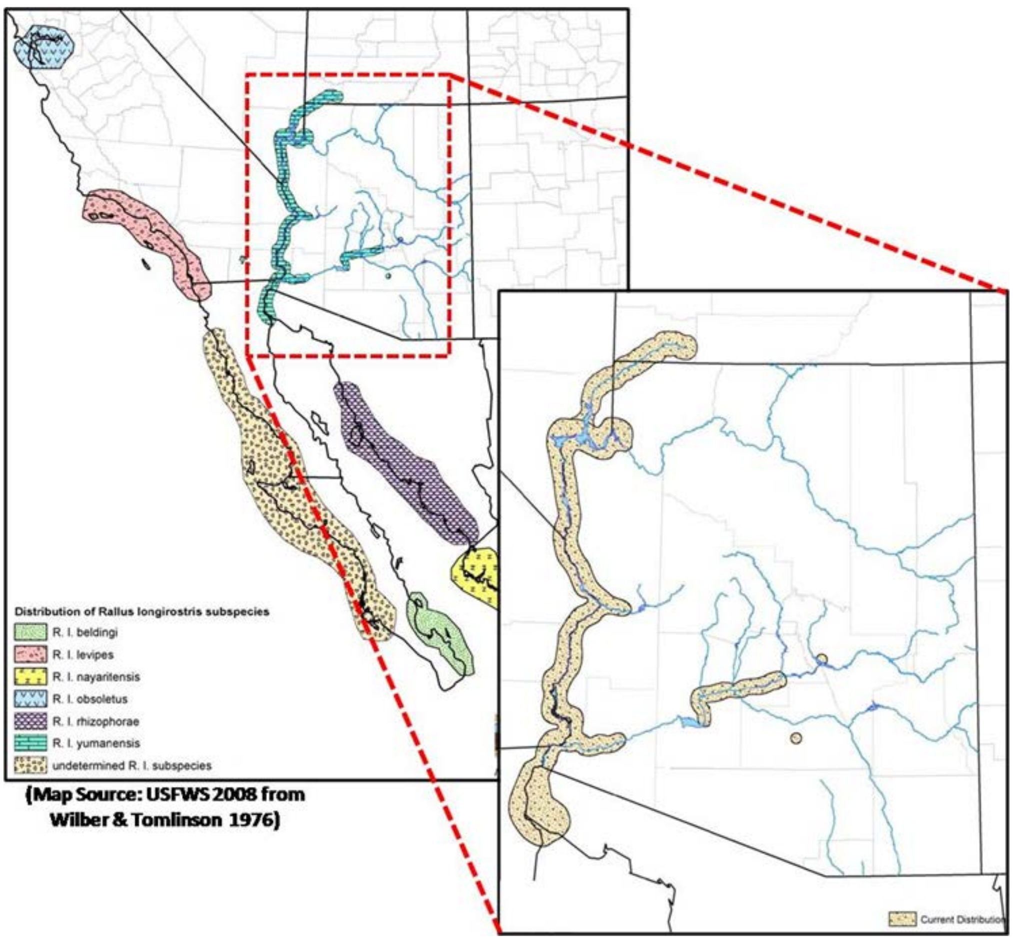
The *in ovo* threshold for selenium exposure that causes toxic effects on embryos of Ridgeway's light-footed rails is unknown. For another benthic-foraging marsh bird, the black-necked stilt, the *in ovo* threshold for embryotoxicity is 6 µg/g selenium (Skorupa 1998). Skorupa reanalyzed his black-necked stilt egg data for the Santa Ana Regional Water Quality Control Board (SARWQCB) using EPA's Toxicity Relationship Analysis Program (TRAP). The results showed a no effect concentration of 5.8 µg Se/g dw and a lower 95% confidence bound on the estimated EC10 of 10.2 µg Se/g dw (SARWQCB 2017). The SARWQCB developed selenium targets for fish (as diet) and bird eggs for the Upper Newport Bay watershed which is habitat for RLRA (SARWQCB 2017). Final site-specific targets were 5 µg/g dw in fish and 8 µg/g dw in bird eggs.

Beckon et al. (2008) applied a biphasic model to Heinz et al. (1989) data from laboratory experiments with mallard reproduction. Their analysis indicates that in mallards there was a 10% reduction in hatchability at 7.7 µg/g dw. These results are in line with Skorupa's black-necked stilt analyses.

Ridgeway's Yuma Rail (RYRA)
Rallus obsoletus yumanensis

Status: Ridgeway's Yuma rails (formerly Yuma clapper rail) were listed as endangered on March 11, 1967 (32 FR 4001). California originally listed Ridgeway's Yuma rails as endangered in 1971, and currently lists them as threatened (CDFW 2017). The species was considered stable in 2002 (USFWS 2006a) at 500-600 birds. Surveys from 2000-2008 showed the numbers have fluctuated between 503 and 890 (USFWS 2010a).

Distribution: The Ridgeway's Yuma rail is a marsh bird found in dense cattail or cattail-bulrush marshes along the lower Colorado River and the Imperial Valley near and around the Salton Sea in California (RYRA Figure 1).



RYRA Figure 1. Map showing distribution of Ridgeway's Yuma rail in southern California and northern Mexico (modified from USFWS 2010a).

Size: The Ridgeway's Yuma rail is one of the smaller subspecies of Ridgeway's rails with male mass ranging from 194 to 347 g (avg. 269 g) and female mass ranging from 160 to 310 g (ave. **210 g**). Males are typically 20% larger than females (Eddleman and Conway 1998).

Diet: The principal prey of the Ridgeway's Yuma rail is two species of introduced crayfish that occur in the area (Inman et al. 1998). Ohmart and Tomlinson (1977) found that about 95% of the stomach contents of two Ridgeway's Yuma rail specimens were crayfish, leading them to suggest that the expansion of the Ridgeway's Yuma rail may be related to the introduction and spread of the crayfish. Other prey items taken by Yuma clapper rail include small fish, insects, amphibian larvae, clams, and other aquatic invertebrates (AGFD 2006, USFWS 2010a).

Food Ingestion Rate: Using the same methodology as described above for Ridgeway's rail (Nagy 2001, Table 3, equation 40 parameters), and an average female body weight of 210g:

FIR (wet weight) = $1.914 \times (\text{body weight in g})^{0.769}$
FIR = $1.914 \times 210^{0.769}$
FIR = **117 g/day wet weight**

Using Table 3 and equation 39 parameters from Nagy (2001) results in an ingestion rate of **31.9 g/d dw**.

General Life History: Ridgeway's Yuma rails are generally found in freshwater dominated by stands of emergent vegetation interspersed with areas of open water and drier, upland benches (USFWS 2010a). This subspecies prefers mature marsh stands along margins of shallow ponds with stable water levels. Nest sites selected by this subspecies are near upland areas in shallow sites dominated by mature vegetation, often in the base of a shrub. Ridgeway's Yuma rails move into different cover types in winter, showing a preference for denser cover than in summer.

Risk of Selenium Exposure: Bioaccumulation of selenium by the Ridgeway's Yuma rail has been documented. Lonzarich et al. (1992) noted that W. Kepner (USFWS, pers. comm.) documented selenium in Yuma rail eggs of 12.5 μg/g dw. This is above threshold effect levels (6-8 μg/g dw) in avian eggs (see discussion in Ridgeway's rail and Ridgeway's light-footed rail).

King et al. (2000) noted liver samples of four Ridgeway's Yuma rails collected in 1987 from Topock Gorge averaged 25.1 μg/g dw (8.6-38.9). Radtke et al. (1988) documented a selenium concentration of 26.0 μg/g dw in a liver sample from a Ridgeway's Yuma rail at Mittry Lake. These levels are known to cause immune system dysfunction in birds (Bruehler and de Peyster 1999; Fairbrother and Fowles 1990). Background hepatic tissue concentrations in nonmarine birds is generally <10 μg/g dw (Ohlendorf and Heinz 2011, USDOI 1998). Concentrations of selenium between 10 and 20 μg/g dw in the liver can be considered elevated and potentially harmful to the adult bird and reproduction while concentrations above 20 μg/g dw can be directly toxic to adults and result in embryonic deformities (Ohlendorf and Heinz 2011).

King et al. (2000) collected Yuma rail prey items along the Lower Colorado River and summarized data from other studies (RYRA Table 1). Selenium in crayfish ranged from 1.4-11.3 μg/g dw (RYRA Table 1). Ninety-five% of the King et al. (2000) samples (36/38) had selenium concentrations >3.0 μg/g dw. The mean selenium concentration in crayfish from U.S. sites was 8.91 μg/g dw (range, 5.78-15.5). Crayfish from the Ciénega de Santa Clara marsh in Mexico contained 4.21 ppm selenium, a level above the dietary concern threshold (King et al. 2000). Selenium in mosquitofish ranged from 2.85-16.6 μg/g dw (RYRA Table 1). These levels are well above dietary threshold levels (3.0 µg/g dw) for avian species (Lemly 1996a, 1996b; Ohlendorf and Heinz 2011, USDOI 1998).

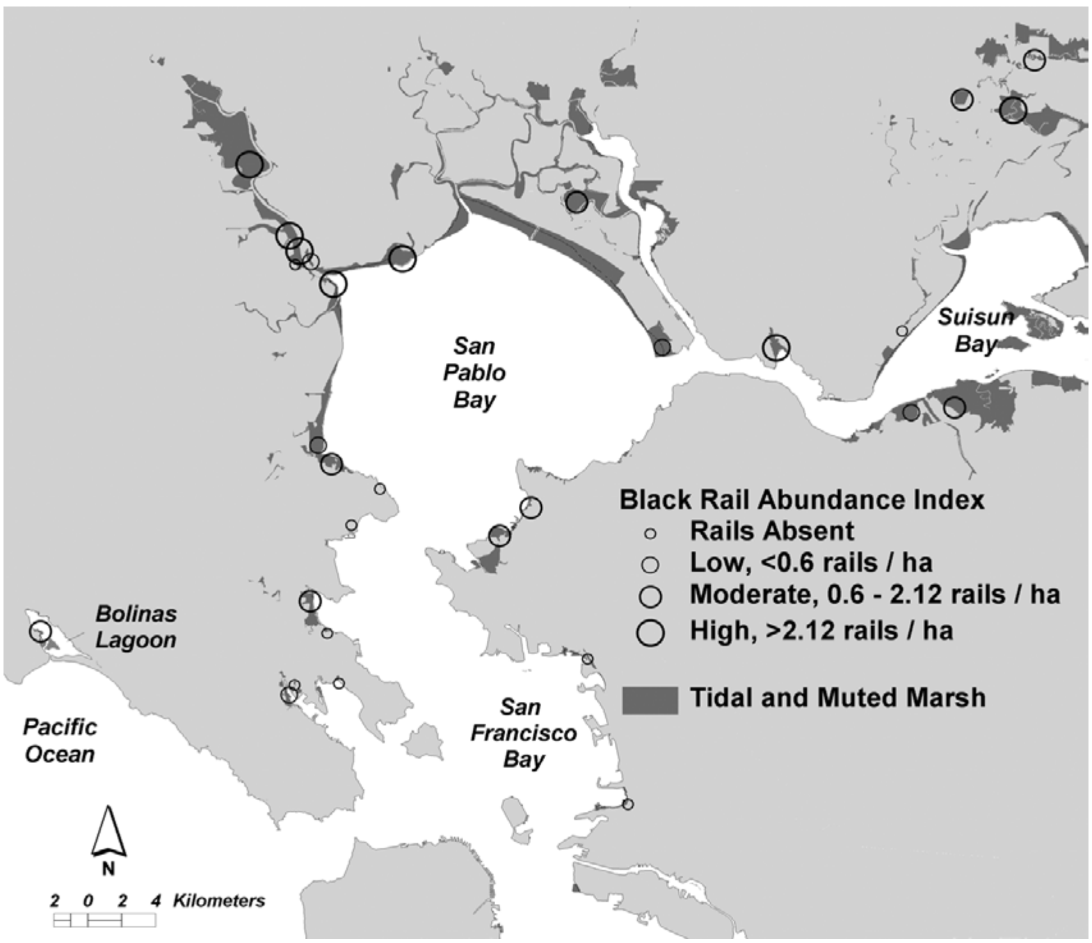
RYRA Table 1. Selenium concentrations in crayfish and mosquitofish collected from the lower Colorado River: a comparison among studies (modified from: King et al. 2000)

Sample	Area	Year	Selenium (ppm dw) Mean (Range)	Study
Crayfish	Cibola Lake	1989	2.60 (1.4 - 4.6)	Welsh and Maughan (1992)
	Cibola Lake	1990	2.80 (NA)	Rusk (1991)
	Cibola Lake	1999	15.5 (NA)	King et al. 2000
	Mittry Lake	1987	4.60 (NA)	Kepner unpub. data
	Mittry Lake	1990	1.76 (NA)	Rusk (1991)
	Mittry Lake	1999	8.03 (NA)	King et al. 2000
	McAllister Lake	1991	4.28 (3.22 - 5.10)	Lusk (1993)
	McAllister Lake	1999	5.78 (NA)	King et al. 2000
	Butler Lake	1991	1.63 (1.47 - 1.77)	Lusk (1993)
	Butler Lake	1999	6.67 (NA)	King et al. 2000
	Topock Gorge	1987	3.70 (NA)	Kepner unpub. data
	Topock Gorge	1999	11.3 (NA)	King et al. 2000
Mosquitofish	McAllister Lake	1991	4.98 (3.94 - 6.36)	Lusk (1993)
	McAllister Lake	1999	16.6 (NA)	King et al. 2000
	Butler Lake	1991	3.97 (2.85 - 5.73)	Lusk (1993)
	Butler Lake	1999	4.15 (NA)	King et al. 2000
Rail Liver	Topock Gorge	1987	25.1 (8.6- 38.9)	USFWS, unpub. data
	Mittry Lake	1987	26.0 (NA)	Radtke et al. 1988

California Black Rail (CBRA)
Laterallus jamaicensis coturniculus

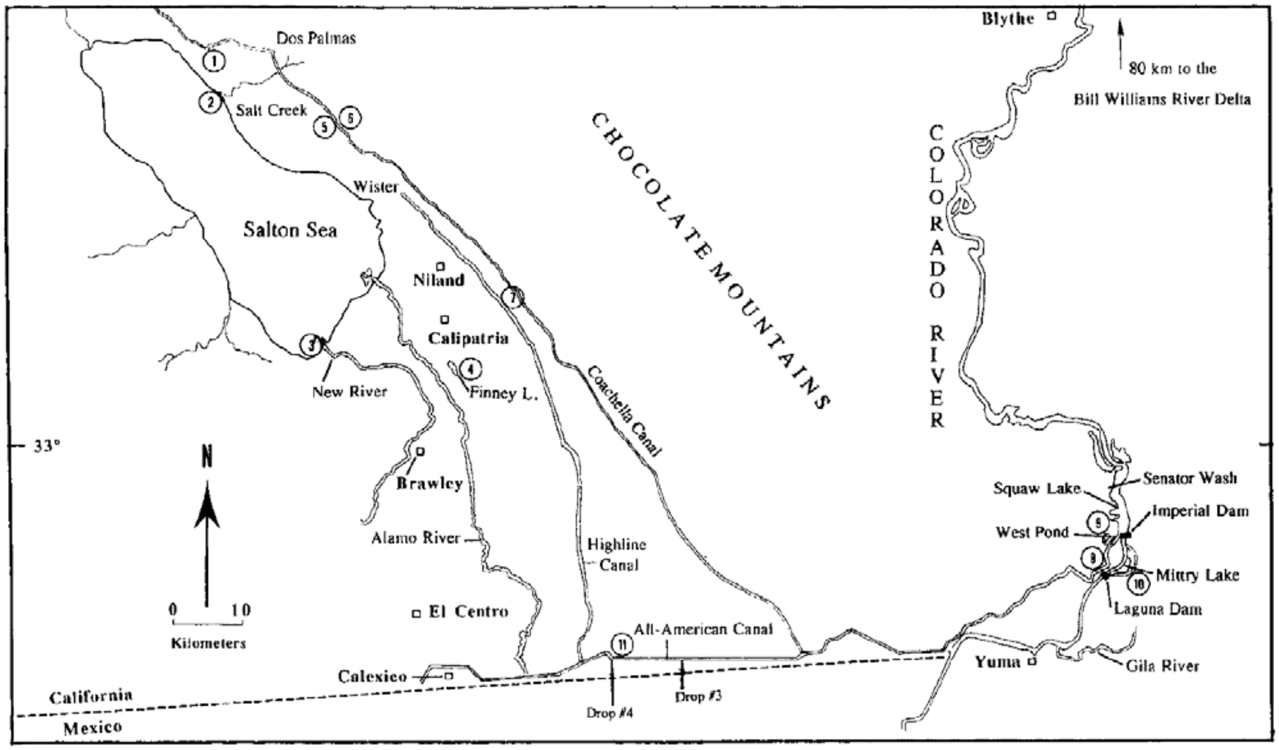
Status: The USFWS lists the California black rail as a migratory nongame bird of special concern, but was listed as threatened by California in 1971 (CDFW 2017).

Distribution: The majority of California black rails (>90%) are found in the tidal salt marshes of the northern San Francisco Bay region, primarily in San Pablo and Suisun Bays (Evens et al. 1991, Manolis 1978, Spautz et al. 2005; CBRA Figure 1). Smaller populations occur in the outer coast of Marin County, freshwater marshes in the foothills of the Sierra Nevada, and in the Colorado River Area (CBRA Figure 2; Evens et al. 1991, Trulio and Evens 2000).



CBRA Figure 1. California Black Rail survey sites and abundance indices, 2001 (from: Spautz et al. 2005).

Size: Mass (g) of black rails during the nesting (males only) and winter season (both sexes) ranged from 26.9 to 29.0 g (**avg. 28g**; Eddleman et al. 1994).



CBRA Figure 2. Distribution of black rail detections in the lower Colorado River and Salton Sea (from: Evans et al. 1991).

Diet: Information is limited, but black rails are likely an opportunistic forager of both invertebrates and seeds. Eddleman et al. (1994) summarized the dietary items of black rail from Southern California and Arizona. A nesting bird ate predaceous diving, ground, and other beetles (73% of volume); earwigs (14%); bulrush seeds (13%); and trace amounts of cattail. During late summer and autumn black rails fed on grasshoppers (59% of volume); beetles (30%), ants (3%); earwigs (3%); spiders (2%); snails (1%); bulrush seeds (1%); and trace amounts of various other insects. During winter months black rails ate bulrush seeds (47% of volume), earwigs (25%); beetles (25%); ants (3%); and trace amounts of cattail seeds. The change in diet is likely related to lower availability of insects in winter.

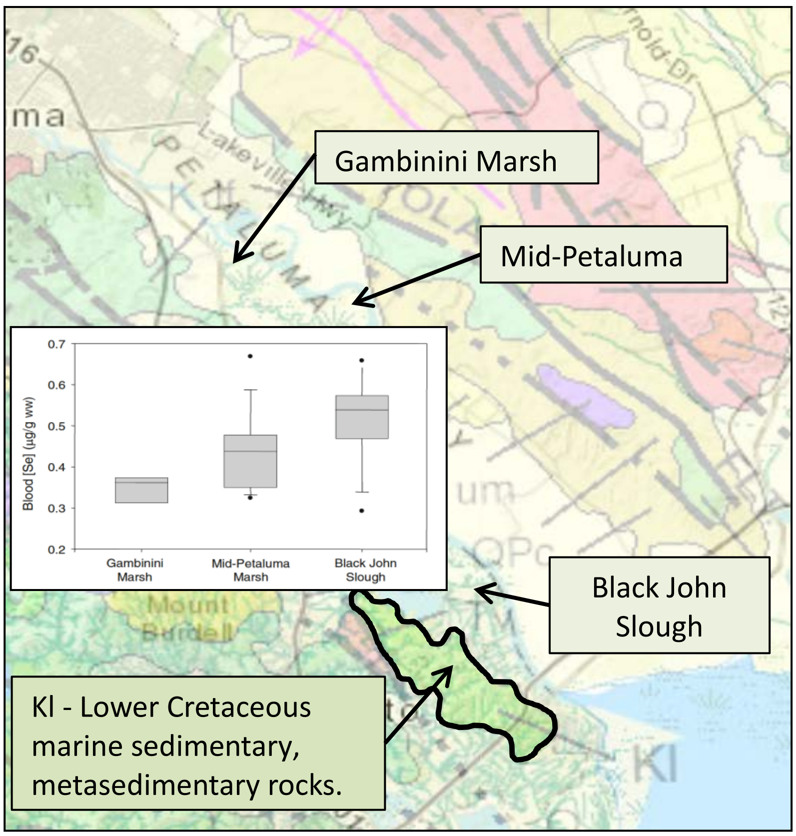
Food Ingestion Rate: Using the same methodology as described above for Ridgeway's rail (Nagy 2001, Table 3, equation 40 parameters), and an average body weight of 28g:

FIR (wet weight) = $1.914 \times (\text{body weight in g})^{0.769}$
FIR = $1.914 \times 28^{0.769}$
FIR = **24.8 g/day wet weight**

Using Table 3 and equation 39 parameters from Nagy (2001) results in an ingestion rate of **6.77 g/d dw**.

Risk of Selenium Exposure: Mean selenium concentrations for black rail blood and liver from the San Francisco Bay area were 0.42 µg/g ww and 2.76 µg/g dw, respectively (Tsao et al. 2009).

These liver and blood concentrations are below levels known to cause problems (Fairbrother and Fowles 1990, USDOI 1998). Of interest is the difference between the three marsh sites studied by Tsao et al. (2009). Black rails from the site furthest downstream, Black John Slough, had higher blood Se concentrations (0.51 µg/g; BLRA Figure 3) than those at Mid-Petaluma (0.42 µg/g) and the marsh farthest upstream Gambinini Marsh (0.35 µg/g). At first one might consider the proximity of Black John Slough to San Pablo Bay as a source of selenium since North San Francisco Bay in considered selenium impaired due to agricultural and oil refinery discharges (Baginska 2015; Presser and Luoma 2006; Taylor et al. 1992, 1993). However, the current driver of selenium exposure in the North Bay/Suisun Marsh area is the benthic overbite clam (*Potamocorbula amurensis*) and the black rail diet would not expose the rail to this part of the food web. A closer look at local geology may offer a more reasonable explanation since the Black Point hills consist of Cretaceous marine deposits (BLRA Figure 3) which are known to contain elevated levels of selenium (Presser et al. 1990, Seiler et al. 1998).



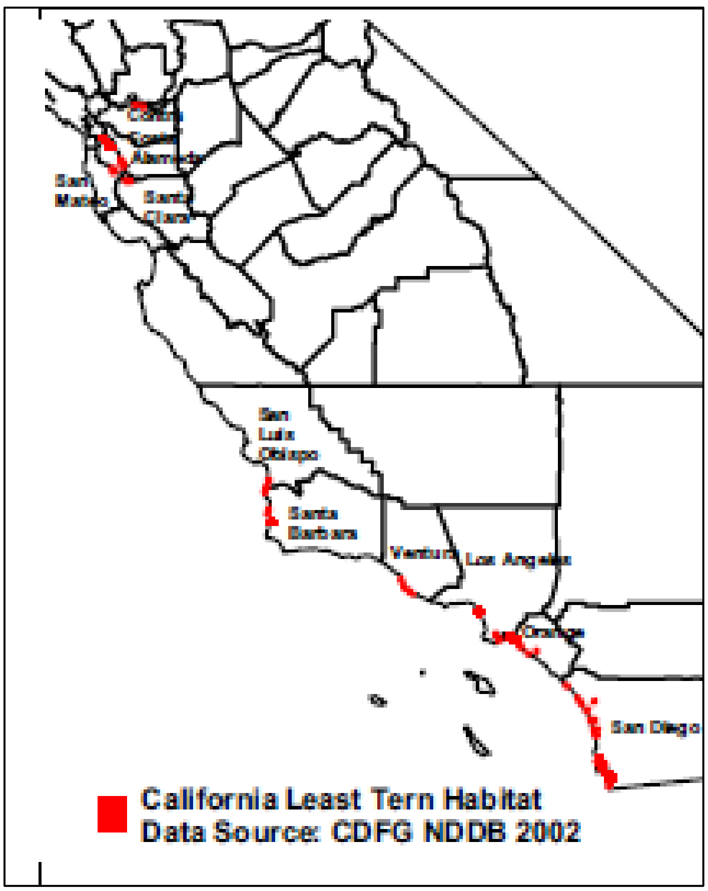
BLRA Figure 3. Lower Cretaceous marine deposits proximity to Black John Slough may explain the higher selenium levels in black rail blood (data insert from Tsao et al. 2009; map from California Department of Conservation, Geologic map of California 2010 http://maps.conservation.ca.gov/cgs/gmc/).

The black rails of the Salton Sea and Lower Colorado River are likely at greater risk to elevated selenium levels in potential dietary items than the San Francisco Bay population. Given similar regional distribution and habitat requirements, it is likely that the Southern California population of black rails is exposed to many of the same selenium risks as the Ridgeway's Yuma rail (See discussion for the Ridgeway's Yuma rail).

California Least Tern (CALT)
Sternula antillarum browni

Status: The California least tern was originally listed as endangered in 1970 (35 FR 8491-8498, 35 FR 16047-16048). The subspecies was listed as endangered in 1971 by the California Department of Fish and Wildlife (CDFW 2017). A 5-year review in 2005 recommended the tern be down listed to threatened due to a significant population increase (USFWS 2006b).

Distribution: A 2009 breeding season survey revealed the highest number of breeding pairs observed in California since 1969, with an estimated 7,130-7,352 pairs occupying 51 sites (Marschalek 2010). Similar breeding population levels were observed during the previous 6 years, with the California breeding population consistently estimated at well over 6,000 pairs. In 2011, Marshalek (2012) estimated between 4,800-6,100 California least tern breeding pairs at 49 documented locations throughout California, ranging from Alameda County south to San Diego County (CALT Figure 1). Nests found at Camp Pendleton, Naval Base Coronado, Batiquitos Lagoon Ecological Reserve, Huntington Beach, Pt. Mugu and Alameda Point represented 79% of the breeding pairs.



CALT Figure 1. Locations of California least tern nesting sites (CNDDB 2002).

Size: The least tern is the smallest of North American terns, averaging 21-23 cm (8¼ to 9 inches) long, with a wingspan of 48-53 cm (19 to 21 inches). Little information on California least tern mass was available but adult interior least terns mass ranges from 36 to 50 g with an average of around **42 g** (Thompson et al. 1997).

Diet: Least terns primarily eat small fish species that are less than 8 cm in length and small young-of-year fish of larger species. Fish species include northern anchovy (*Engraulis mordax*), top smelt (*Atherinops affinis*), silversides (*Atherinopsidae*), herring (*Clupeidae*) and yellowfin goby (*Acanthogobius flavimanus*). Up to 50 species of fish have been documented in their diet (USFWS 1985a).

Food Ingestion Rate: Nagy (2001) lists common tern, much larger spp. (127g), as eating 77g fresh weight food daily (~61% of body weight). Using this same proportion, the least tern would consume approximately 26 g ww food daily (using avg. body weight of 42g). Using Table 3 and equation 40 ww parameters for Charadriiformes from Nagy (2001) results in an ingestion rate of 33.9 g/d ww while using the dw parameters for equation 39 results in an ingestion rate of **9.25 g/d dw**.

General Life History: California least terns are migratory, wintering along the southern coast of Mexico (Thompson et al. 1997). Currently, breeding colonies of California least tern are confined to scattered, isolated locations on vegetation free beaches along the coast of California and in the San Francisco estuary, where they feed on surface fish in adjacent waters (USFWS 2006b). Breeding birds forage within 3.2 km (2 mi.) of the colony (USFWS 1985a). Females clutch size is 2 – 3 eggs but they may renest if the first attempt failure is early in the season. Terns reach breeding areas in mid-April and breeding season lasts from mid-May to early August (USFWS 1985a).

Risk of selenium exposure: Selenium water concentrations in some coastal streams of Southern California are documented to be well above 5 µg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). These water concentrations could lead to elevated selenium in prey items in estuaries, marshes, or lagoons within these watersheds.

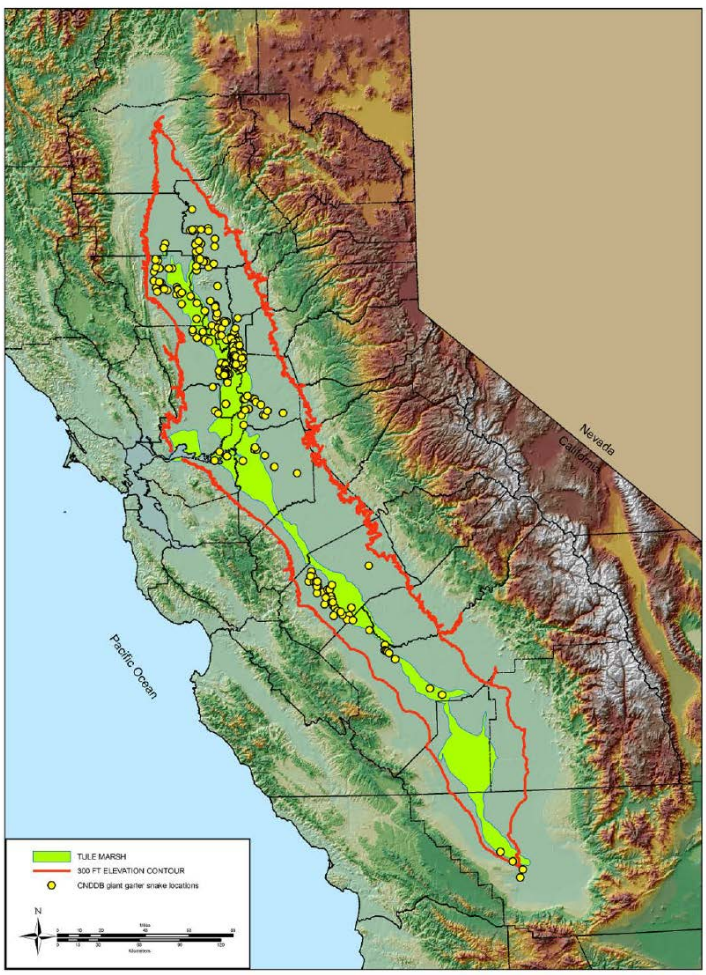
Ackerman and Eagles-Smith (2009) found selenium levels in livers from adult Forster's terns in San Francisco Bay ranged from 3.73 to 14.50 µg/g dw (7.13+/-0.38), and 4.77 to 14.40 µg/g dw in Caspian terns (6.73 +/- 0.78). There was a weak negative correlation between selenium levels and body condition for the Forster's tern but not the Caspian tern and two shorebirds. Hoffman et al. (2011) did not find any correlations with similar selenium levels and biomarkers in Caspian and Forster's terns. Normal liver concentrations in birds are generally less than 10 µg/g dw although concentrations above 10 µg/g dw does not necessarily indicate a selenium problem and marine birds can have selenium liver levels > 20 µg/g dw (Ohlendorf and Heinz 2011, USDOI 1998).

Least terns in North Carolina and the Caribbean are known to eat invertebrates, including shrimp (review in Thompson et al. 1997). Although unlikely, California least terns could learn to feed opportunistically on abundant brine shrimp and other invertebrates in evaporation ponds. Fish may also be available in subsurface drainage canals. Concentrations of selenium in evaporation pond invertebrates and fish from drainage canals may be sufficiently elevated to cause reproductive impacts in least terns. Mosquitofish were the only fish species that survived in the ponds of Kesterson Reservoir after September 1983 (Saiki 1986). Concentrations of selenium ranged up to 366 µg/g in samples of mosquitofish collected from the San Luis Drain and up to 293 µg/g in the ponds of Kesterson Reservoir in May and August, 1983. Forster's tern eggs from San Joaquin Valley nests at evaporation ponds had an average of 7.1 µg/g dw of selenium (n=10, range 2.6 to 12 µg/g) while Caspian tern eggs averaged 2.4 µg/g (n=7, range 1.9 to 3.3 µg/g) (USFWS unpublished data). Effect thresholds for bird eggs range from 6 to 8 µg/g (See discussion for Ridgeway's light-footed rail).

Giant Garter Snake (GGSN)
(*Thamnophis gigas*)

Status: The giant garter snake was listed as threatened in 1993 (58 FR 54053-54066). It is endemic to the wetlands of the Central Valley from Butte County in the north to Kern County in the south (USFWS 2017a). A 5-year review completed in 2006 recommended no change in the listing status for the snake (USFWS 2006).

Distribution: The giant garter snake is endemic to the wetlands of the Sacramento and San Joaquin Valleys of California, inhibiting the tule marshes and seasonal wetlands created by overbank flooding of the rivers and streams of the Central Valley (Fitch 1940). Most populations of giant garter snakes are found in the Sacramento Valley while small isolated populations are found in northern San Joaquin Valley primarily Merced County and western Fresno County (GGSN Figure 1; Halstead et al. 2015, USFWS 2017a). Today only about 5% of its historical wetland habitat acreage remains.



GGSN Figure 1. Historic location of tule marsh and giant garter snake observations (from USFWS 2017a).

Size: Females tend to be slightly longer and proportionately heavier than males. Wylie et al. (2010) documented female mean Snout/Vent Length of 692 mm (95% CI = 377–1168 mm) while male SVL was 581 mm (95% CI = 387–839 mm). Female mean mass was **250 g** (95% CI = 24–1030 g) with a male mean mass of 101 g (95% CI = 26–274). Although growth rates are variable, young typically more than double in size within the first year (58 FR 54053-54066).

Diet: Fish and amphibians (tadpoles and adults) are the primary food items of giant garter snakes (58 FR 54053-54066, USFWS 2017). Adult giant garter snakes feed primarily on a wide variety of native and non-native aquatic prey such as fish and amphibians, capturing all their food in the water (Halstead et al. 2015, Hansen 1980). Giant garter snakes have been observed actively hunting for and capturing small fish in the wild (Fitch 1941; Hansen 1980), and have been observed feeding on mosquito fish confined to small pools of water (Hansen 1980).

Food Ingestion Rate: No specific data was found regarding food ingestion rate for the giant garter snake. Peterson et al. (1998) determined the similar, but smaller-sized, common garter snakes (*Thamnophis sirtalis*) had an ingestion rate averaging 14% of body weight. For a 250 g female giant garter snake this would be 35 g per day ww. On a dry weight basis assuming 75% moisture this would be 8.8 g/d dw. Using the Nagy (2001) Table 4 equation (73) for Scleroglossan lizards, which includes snakes, and 250 g for the GGSN results in an ingestion rate of 2.0 g/d dw. Utilizing data from the EPA's Wildlife Exposure Handbook, for the northern water snake (*Nerodia sipedon sipedo*), a similar species in size and prey selection, feeding rates were determined in a laboratory setting. Juveniles (up to 1 yr.) consumed at a rate of 0.088 g/g-d ww, juveniles (2-3 yrs.) at a rate of 0.043 g/g-d ww, while adults (>5yrs) averaged 0.061 g/g-d ww (EPA 1993). For a 250 g adult female giant garter snake the 0.061 g/g-d converts to **15.3 g/d ww** or **3.8 g/d dw** using 75% moisture—a value mid-range between the other two methods.

General Life History: Giant garter snakes prefer marshes, sloughs, ponds, small lakes, and low gradient streams. Currently agricultural wetlands such as irrigation and drainage canals and rice fields provide key habitat for the snake (Halstead et al. 2015, USFWS 2017a). These wetland habitats must include sufficient water through the summer; emergent vegetation for escape cover; grassy banks and openings for basking; and higher elevation uplands for cover and refuge from flood waters (Halstead et al. 2015, USFWS 2017a). Home range estimates from several studies averaged from 42 to 109 acres with individuals not traveling more than 1,000 feet from their initial capture point (USFWS 2017a). Giant garter snakes move into protective underground areas above flooding zones (mammal burrows, crevasses) in early October and emerge in March when mating begins (USFWS 2017a). Adult males reach maturity in 3 years while females reach maturity in 5 years (58 FR 54053-54066).

Risk of selenium exposure: The potential for giant garter snake exposure to selenium is greatest in the San Joaquin Valley. Selenium in giant garter snake liver samples from Sacramento Valley averaged 0.77 µg/g ww while livers of gopher snakes (*Pituophis melanoleucus*) from the selenium contaminated Kesterson Reservoir averaged 2.61 µg/g ww and banded water snake (*Nerodia fasciata*) liver samples from a coal ash settling pond averaged 35.5 µg/g ww (Wylie et al. 2009).

Very little research has been done on the toxicity of selenium to reptiles (Hopkins 2000); no such studies have been done on giant garter snakes or on any other species of garter snake (Campbell and Campbell, 2001). Hopkins et al. (2002) found that in another species of aquatic snake, the banded water snake (*Nerodia fasciata*), bioaccumulation of dietary selenium was most notable (greatly exceeding toxicity thresholds that have been established for other vertebrates) compared to other elevated trace elements at a site contaminated with coal ash. At the same selenium-contaminated site, Roe et al. (2004) found clutch viability to be reduced in alligators (*Alligator mississippiensis*) with viability of 30% -54% at egg selenium levels of 2.1-7.8 µg/g dry weight compared to a reference site (viability 67% -74%, egg selenium 1.4-2.3 µg/g). Average selenium concentrations in common prey items of alligators (fish and frogs) in the contaminated site ranged from 10 to 27 µg/g (dry weight), with an average concentration of 14.3 µg/g in mosquitofish (*Gambusia affinis*). Average concentrations in the same prey items from the reference site ranged from 1.12 to 3.43 µg/g, with an average concentration of 1.82 µg/g in mosquitofish (Hopkins et al. 1999). Other contaminant in prey species varied between the sites, so the role of selenium in reduced clutch viability is not unequivocal.

These data suggest that dietary selenium concentrations of 10 to 27 µg/g may have a negative impact on reptiles that are dependent on an aquatic food chain. It should be noted that interpretation of these field data is confounded by the co-occurrence of other contaminants that could also affect egg viability. However, in such coal ash-contaminated sites, as in subsurface drainwater-contaminated sites, selenium has been implicated as the chief cause of toxicity to wildlife. If, as is most likely, selenium is the principal cause of reduced clutch viability, then the corresponding selenium concentration in prey items must be treated as a dietary LOAEC for a single effect on a single species of aquatic reptile. The actual toxicity threshold for alligators is an unknown amount below this LOAEC value (10 µg/g). Further, any extrapolation of alligator toxicity data to giant garter snakes must include an uncertainty factor to account for the risk that giant garter snakes may be more sensitive than alligators. This accords with findings by a study of dietary selenium effects on the brown house snake (*Lamprophis fulginosus*), a common terrestrial snake found in southern Africa. Female snakes exposed to a diet containing 10 µg/g seleno-D, L-methionine produced about half as many eggs as control females exposed to 1 µg/g (Hopkins et al. 2004). Also, the dietary selenium toxicity threshold for the avian descendants of reptiles is about 3 to 7 µg/g (dry weight; Wilber 1980, Martin 1988, Ohlendorf and Heinz 2011). Therefore, given the above data, an appropriate dietary selenium toxicity threshold for the giant garter snake is probably well below 10 µg/g.

Dietary uptake is the principle route of toxic exposure to selenium in wildlife, including giant garter snakes. Giant garter snakes feed primarily on aquatic prey such as fish and amphibians (USFWS 2017). The extent to which they may take aquatic invertebrates is unknown. Open drainwater ditches may constitute risks of exposure of giant garter snakes to selenium in the aquatic food chain. In addition, these conveyances could provide routes of dispersal of giant garter snakes from existing habitat to evaporation ponds. The drainwater conveyances and ponds of Kesterson Reservoir in the early 1980s serve as the best available prototype for estimation of the effects on giant garter snakes of selenium contamination associated with water deliveries to the San Luis Unit. Mosquitofish were the only fish species that survived in the ponds of Kesterson Reservoir after September 1983 (Saiki 1986). Concentrations of selenium ranged up to 366 µg/g in samples of mosquitofish collected from the San Luis Drain and up to 293 µg/g in the ponds of Kesterson Reservoir in May and August, 1983; aquatic insects collected in these localities had selenium concentrations of up to 326 and 295 µg/g respectively (Saiki 1986). These concentrations are far above dietary selenium concentrations associated with adverse effects in aquatic reptiles (see above).

Gopher snakes collected at Kesterson Reservoir in April-June 1984 and April-July 1985 had liver selenium concentrations ranging from 8.2 to 19 µg/g (dry weight; geometric mean 10.9; Ohlendorf et al. 1988). Such a range of liver concentrations corresponds to a selenium concentration range of about 7 to 20 µg/g in eggs in the brown house snake (*Lamprophis fuliginosus*) (Hopkins et al. 2005), the closest relative of the giant garter snake for which data are available linking liver and egg concentrations. Therefore the eggs of gopher snakes at Kesterson Reservoir were probably within or above the range (2.1-7.8 µg/g) associated with adverse effects in reptiles (see above). Gopher snakes have a more terrestrial diet than giant garter snakes, but the gopher snake data provide an additional indication that reptiles in an agricultural drainwater evaporation pond environment may be at risk.

Hansen et al. (2015) assessed the exposure effects on giant garter snakes of groundwater pumping as an additional water source for the Volta Wildlife Management Area (WMA) in the San Joaquin Valley. Concentrations of selenium in mosquitofish ranged from about 0.5 to 1.7 µg/g dry weight and did not appear to have any effects on giant garter snake blood levels or health parameters. Blood selenium levels were all well below 1.2 µg/g wet weight and are considered to be background levels (USDOI 1998). Water selenium concentrations at Volta WMA during the 5 year study were consistently below 1.0 µg/L (U.S. Bureau of Reclamation, unpublished data).

California Red-legged Frog (CRLF)
Rana draytonii

Status: The California red-legged frog (CRLF) was federally listed as threatened in 1996 (59 FR 4888-4895). Critical habitat was designated in 2006 and updated in 2010. There are eight recovery units identified for the red-legged frog.

Distribution: The CRLF once ranged from Shasta County south into Baja California at elevations from sea level up to 5,200 feet although most observations are below 3,500 feet (USFWS 2002a). The current range of the CRLF is limited to coastal counties and a few isolated populations in counties on the west side of the Sierra Nevada and Cascade Mountains (CRLF Figure 1; USFWS 2002a, Barry and Fellers 2013).

CRLF Figure 1. Critical habitat (beige) and core recovery areas (blue) for the California red-legged frog.

Size: Tadpole length ranges from 14–80 mm (USFWS 2002a). Hayes and Miyamoto (1984) found that adult female red-legged frogs range from 78–138 mm in length. Fellers (pers. comm., in USEPA 2007) documented lengths of adult and juvenile female red-legged frogs ranging from 50–131 mm and weights ranging from 8.7–238 g. Fellers (pers. comm., in USEPA 2007) also provided a length-weight equation for red-legged frogs:

$$BW = 0.7291 + 0.0981*L^3$$

Where BW = body weight (g) and L= length (cm)*
*Note: The original reference has mm for length but based on the results it appears this should have been cm.

Assuming a more likely minimum length of 9 cm (90 mm) for a breeding **adult female** the equation provides a weight of **72.2 g.** Tadpole weights are likely less than 10 g in weight based on the lowest terrestrial weight of juvenile frogs noted above, thus a **5 g tadpole** is selected as a mid-range weight.

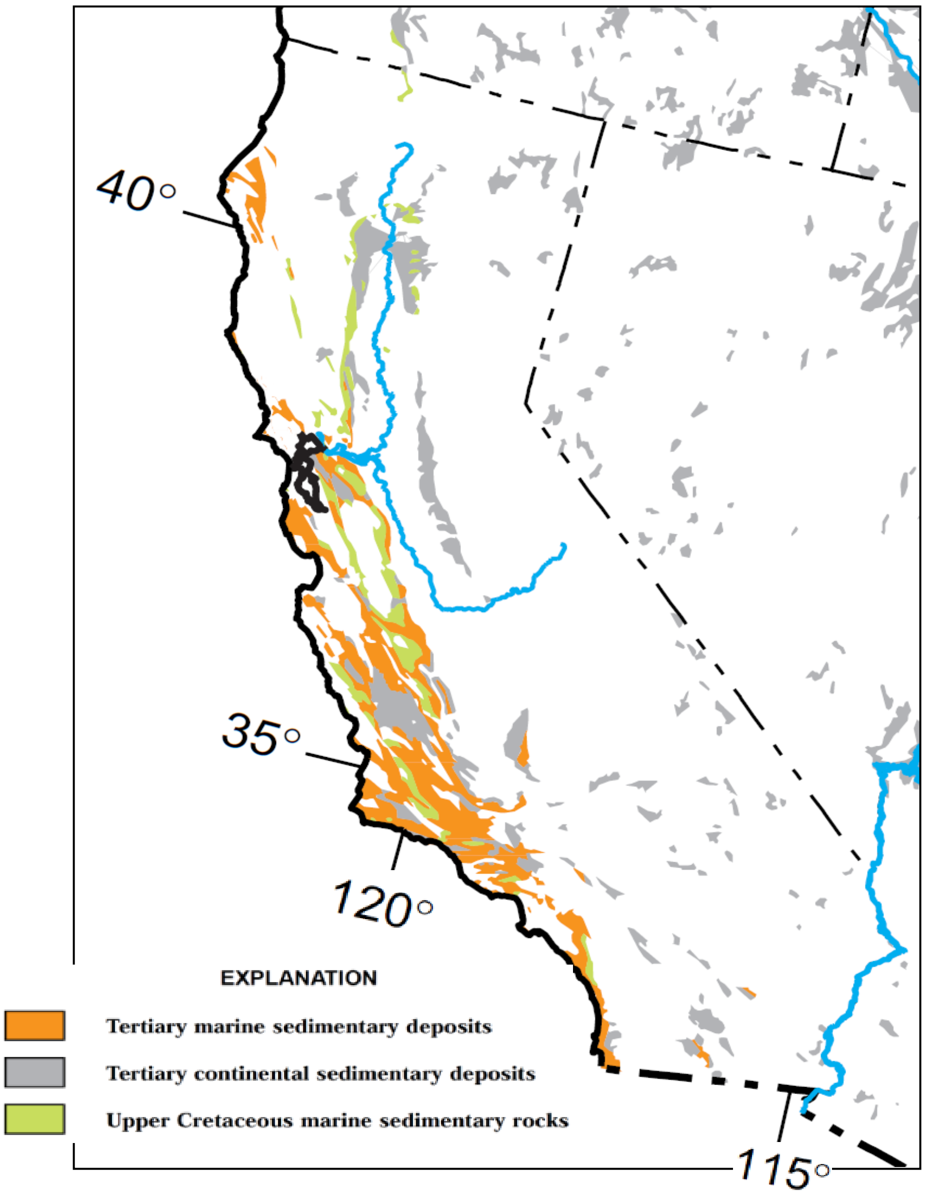
Diet: The diet of the larvae is not well studied, but is likely similar to that of other ranid frogs, feeding on algae, diatoms, and detritus by grazing on the surface of rocks and vegetation (Fellers 2005; Kupferberg 1996a, 1996b, 1997). Hayes and Tennant (1985) analyzed the diets of adult California red-legged frogs from Santa Barbara County during winter months and found invertebrates to be the most common prey item consumed (beetles, water striders, lycosid spiders, larval alderflies, pillbugs, and snails). Larger frogs were recorded to have preyed on Pacific chorus frogs, threespine stickleback, and even California mice, which were abundant at the study site (Hayes and Tennant 1985, Fellers 2005). Larger vertebrate prey represented over half of the prey mass eaten by larger frogs suggesting that such prey may play an energetically important role in their diets (Hayes and Tennant 1985). Bishop et al. (2014) found that a terrestrial diet was most important to red-legged frogs with 90% of their diet being terrestrial in either wet or dry season.

Food Ingestion Rate: There is little information on amphibian food ingestion rates, particularly at the larval stage. The USEPA (2008) developed a herptile (amphibian and reptile) pesticide exposure model based on its avian T-Rex model but using an iguanid food ingestion rate because of the differences in metabolic needs between birds and herptiles. The T-Herps guidance document specifically focuses on the CRLF. They found the iguanid food ingestion rates of 3% - 5% bw/d were in a similar range as those determined for juvenile bullfrogs 3% -7% bw/day. The 3% -7% range is from Modzelewski and Culley (1974) who show an ingestion rate for a 28-77 g bull frog as 0.04 g/g/d ww. The ingestion rate for an adult 72.2 g frog would then be **2.89 g/d ww**. Using an average 79% moisture for the diet used in Modzelewski and Culley (1974) provides a dry weight value of **0.61 g/d dw**. For a 5g tadpole using the higher ingestion rate of 7% of body weight as suggested for the smaller size frogs in Modzelewski and Culley (1974) provides a value of **0.35 g ww**. Assuming a 90% moisture for the diet (algae, organic detritus) results in an ingestion rate of **0.035 g/d** dw for an average tadpole.

General Life History: Breeding habitat includes natural or manmade freshwater ponds, slow moving streams, pools in streams and other ephemeral or permanent water bodies (USFWS 2010b). Permanent waterbodies affords less ideal breeding habitat as predators such as fish and bull frogs can utilize these habitats. Ephemeral waterbodies should hold water for at least 20 weeks to allow for tadpole metamorphosis (USFWS 2010b). The breeding season generally runs from November through April with most mating occurring in February and March.

Adult frogs utilize a wide variety of upland habitat and may move up to 2 miles from breeding habitat although most remain within 1 mile (USFWS 2010b). Upland habitat must provide shelter, forage, and predator avoidance.

Risk of Selenium Exposure: CRLF in the Coastal Range of California are more likely to be exposed to selenium due to the extensive marine sedimentary geology of the range (CRLF Figure 2) which contains elevated selenium concentrations (Presser et al 1990). Central and southern coastal streams have shown elevated selenium levels well above 5 µg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). Areas where human activity can mobilize selenium such as mining, road work, and urban development can increase exposure potential for the frog.



CRLF Figure 2. Areas of California with seleniferous marine geology. Modified from: Seiler et al. (1998).

The toxicity of selenium to amphibians is limited and field data is often confounded by other contaminants (Janz et al. 2010, Sparling et al. 2010, USDOI 1998). Amphibians with elevated selenium levels show signs of impairment (e.g. axial malformations, behavior alteration, and reduced survival) similar to those seen in fish (Janz et al. 2010, USDOI 1998). Hopkins et al. (2006) documented maternal transfer of selenium to eggs in narrow-mouth toads (*Gastrophryne carolinensis*) from a site contaminated with coal fly ash. Water selenium concentrations at the contaminated site averaged 4 µg/L while soil averaged 8 µg/g. This led to an average whole body selenium concentration in females of 42 µg/g dw and an average egg concentration of 44 µg/g dw. Impacts observed were: reduced hatching success, developmental abnormalities in hatchlings, craniofacial abnormalities, and impaired swimming behavior. Rowe et al. (2010) fed enriched selenium diets to larvae of the gray treefrog (*Hyla versicolor*) and documented selenium concentrations in the larvae, pre-metamorphs, and metamorphs. At dry weight dietary concentrations of 1 µg/g (control), 7.5 µg/g (low) and 32.7 µg/g (high) larval treefrogs accumulated selenium concentrations of 8.1 µg/g, 15 µg/g, and 28 µg/g, respectively. However, Rowe et al. (2010) did not document any negative effects noting variable selenium sensitivity of various organisms.

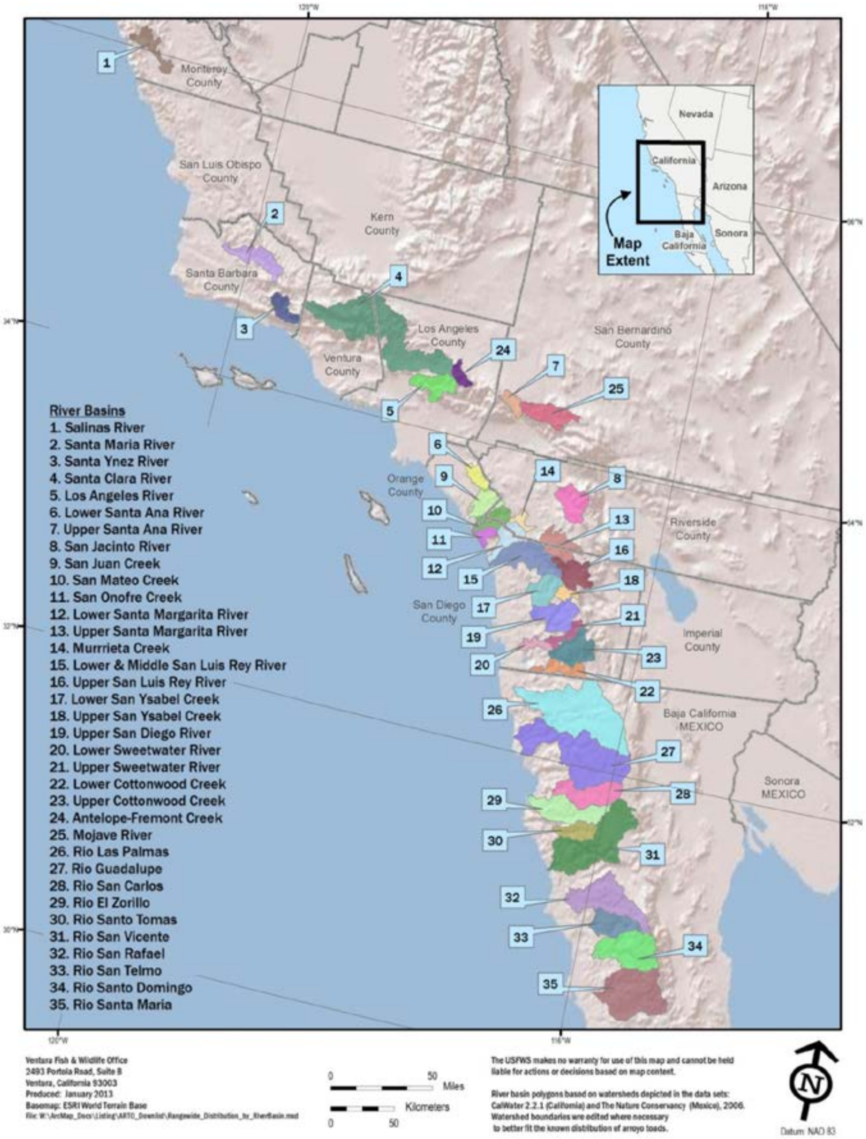
All things considered selenium levels that are protective of fish will likely be protective of amphibians also.

Arroyo Toad (ARTO)

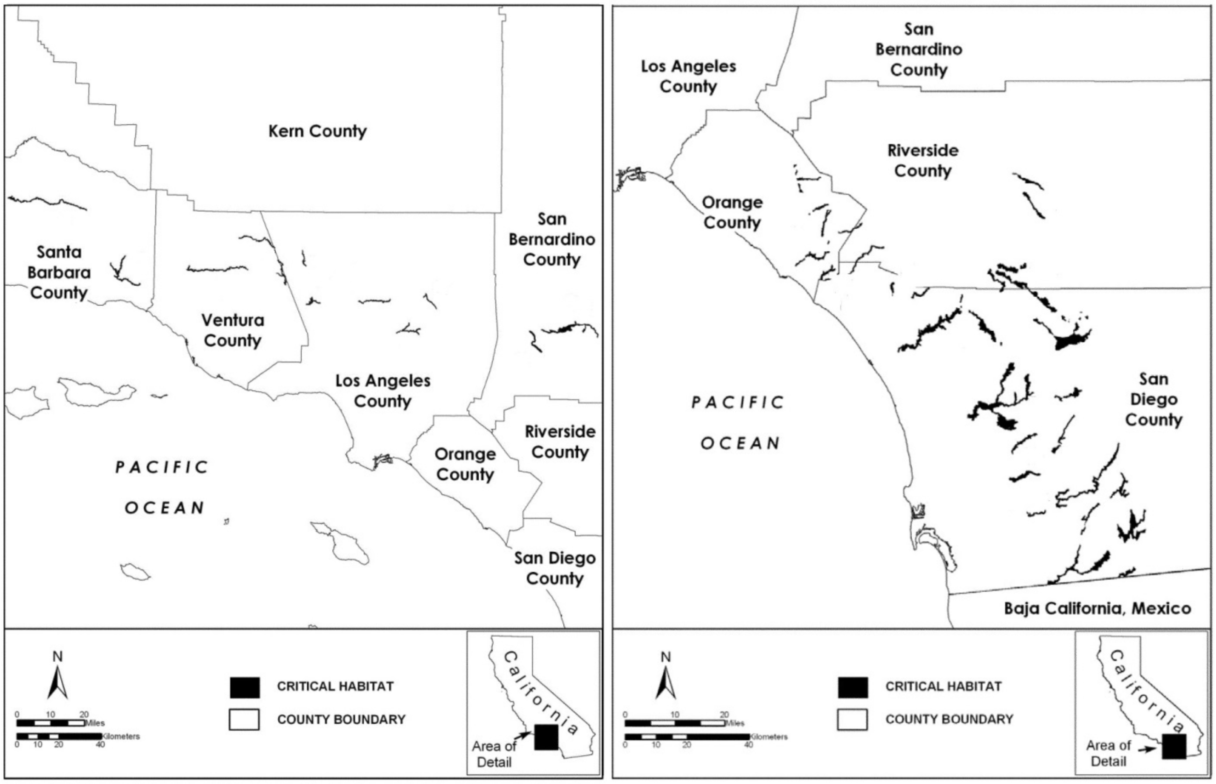
Anaxyrus californicus

Status: The Arroyo Toad (originally the Arroyo Southwestern Toad) was listed as endangered in 1994 (59 FR 64859). A 5–year review (USFWS 2009a) recommended down-listing the toad to threatened; however, a proposal to down-list the toad to threatened was withdrawn in 2015 since the threats to the toad remain (80 FR 79805).

Distribution: Historically, arroyo toads occurred in the upper Salinas River, Monterey County, south through the Santa Ynez, Santa Clara, and Los Angeles River Basins, as well as the coastal drainages of Orange, Riverside, and San Diego Counties (ARTO Figure 1; Campbell et al. 1996). Critical habitat designations were established in 2011 (ARTO Figure 2).



ARTO Figure 1. Current river basin occurrences of arroyo toads in the U.S. and Baja California, Mexico (from: USFWS 2014).



ARTO Figure 2. Critical habitat for the arroyo toad (76 FR 7246).

Size: The arroyo toad is a relatively small toad with adults measuring 4.6 - 8.6 cm from snout to vent (Stebbins and McGinnis 2012). Mitrovich et al. (2011) documented arroyo toad lengths and weights for a habitat use study. Male toad mean length was 5.6 cm with a mean weight of 20.6 g. **Female toad** mean length was 6.1 cm with a **mean weight of 30.8 g**. No information was found regarding specific weights of juvenile arroyo toads; however, the closely related western toad (*Anaxyrus boreas*, formerly *Bufo boreas*) has been documented in both field and laboratory settings. Sivulva et al. (1972) measured western toad **tadpoles** attaining weights of **0.46 g** prior to the non-feeding metamorphosis stage. Newly morphed and 1 year old western toads weighed an average of 2.5 g (range 1.2 - 4.6 g) with the 1 year old toads roughly 62% larger suggesting that the newly morphed toads weighed $\leq$ 2 g (Lillywhite et al. 1973).

Diet: Arroyo toad tadpoles feed on loose organic material such as interstitial algae, bacteria, and diatoms. They do not forage on macroscopic vegetation (Sweet 1992, Jennings and Hayes 1994). The diet of juvenile arroyo toads is exclusively ants (Sweet 1992). Adult arroyo toads consume a wide variety of insects and arthropods including ants, beetles, spiders, larvae, and caterpillars.

Food Ingestion Rate: No ingestion rate information is available for the Arroyo toad; however, the Lillywhite et al. (1973) feeding study using juvenile western toads is available. Using the information at 27^{o}C in Figure 6 the youngest toads at an average size of 2.5 g had an ingestion rate of 0.08 g/g/d ww or 8% of bw/d. At the end of the 8 week feeding study the ingestion rate was 0.039 g/g/d ww or 3.9% bw/d. These rates are similar to the iguanid food ingestion rates of

3% -5% bw/d from EPA (2008) and the 3% -7% bw/d range from Modzelewski and Culley (1974).

A tadpole at 0.46 g and assuming an 8% ingestion rate would eat **0.037 g/d ww**. At 90% moisture this converts to **0.0037 g/d dw.** For a 30.8 g adult toad using the 3.9% ingestion rate results in a value of **1.2 g/d ww**. The percent moisture used for the red-legged frog is likely too high for the toad since the red-legged frog diet included fish. Finke (2002) studied the nutrient composition of invertebrates used as food for insectivores. The average percent moisture of mealworms (larvae and adult), waxworm (larvae), crickets (adult and nymph), silkworm, caterpillar, and earthworm was 68.4% (range 57.9% – 83.6%). Using 68.4% provides a final adult ingestion rate of **0.38 g/d dw**.

General Life History: The elevational range for the arroyo toad extends from near sea level to about 2,440 meters (8,000 feet) in Baja California (Welsh 1988, Beaman et al. 1995*).* Currently, most arroyo toad populations in the northern and central parts of the range are restricted to elevations of 300 to 1,400 meters (1,000 to 4,600 feet), perhaps due to widespread habitat loss at lower elevations (USFWS 1999b).

Adult arroyo toads begin breeding in late March in the northern portion of the range (Sweet 1992) and as early as January in the coastal areas of southern California. The larval period for arroyo toads lasts about 65 to 85 days, depending on water temperatures (Sweet 1992). Juvenile arroyo toads remain in the saturated substrate at the edges of breeding pools for 1 to 3 weeks (USFWS 1999b).

Risk of Selenium Exposure: Arroyo toads in the southern Coastal Range of California are likely to be exposed to selenium due to the extensive marine sedimentary geology of the range (See CRLF Figure 2). Southern coastal streams have shown elevated selenium levels above 5 µg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). Areas where human activity can mobilize selenium such as mining, road work, and urban development can increase exposure potential for the frog.

The toxicity of selenium to amphibians is limited and field data is often confounded by other contaminants (Janz et al. 2010, Sparling et al. 2010, USDOI 1998). Refer to the discussion on risk of selenium exposure for the red-legged frog. All things considered selenium levels that are protective of fish will likely be protective of amphibians also.

Chinook Salmon (CHSA)
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 in 1994 (59 FR 440). In 1999, NMFS listed the Central Valley spring-run ESU as threatened (64 FR 50394).

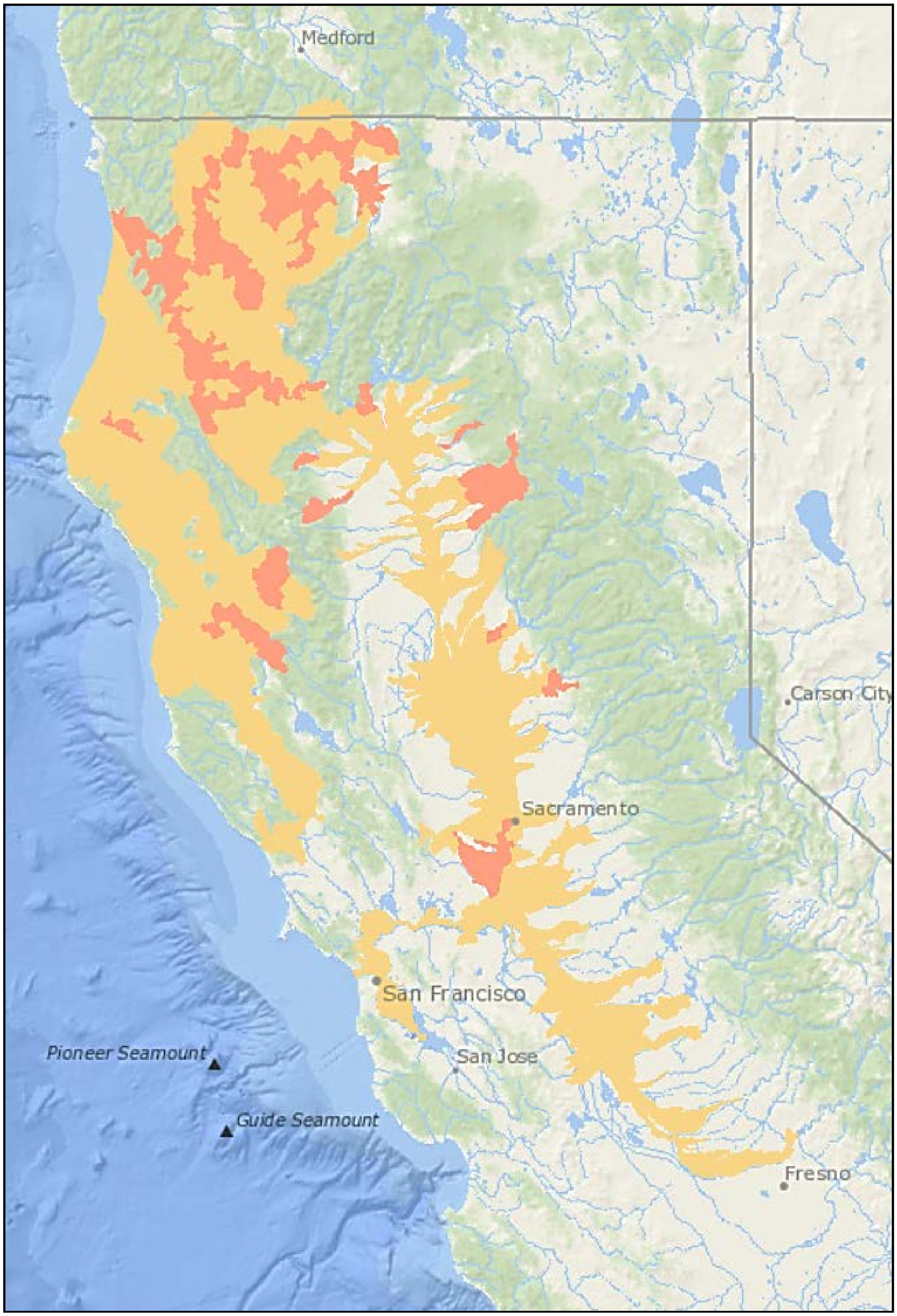
Distribution: California supports the southern-most Chinook salmon runs on the west coast (CHSA Figure 1). Chinook salmon are anadromous fish, thus the time spent in the ocean and freshwater varies among the various runs. The Chinook salmon runs in California have been classified into six major groups:

- Southern Oregon and Northern California Coastal
- California Coastal
- Upper Klamath - Trinity River
- Central Valley Fall and Late Fall-run
- Central Valley Spring-run
- Sacramento River Winter-run

Size: Chinook salmon fry remain in the estuarine nursery areas as they grow from about **0.5 g to about 18 g** (CHSA Figure 2). They migrate seaward from the estuary when they reach about 70 mm fork length (Healy 1991).

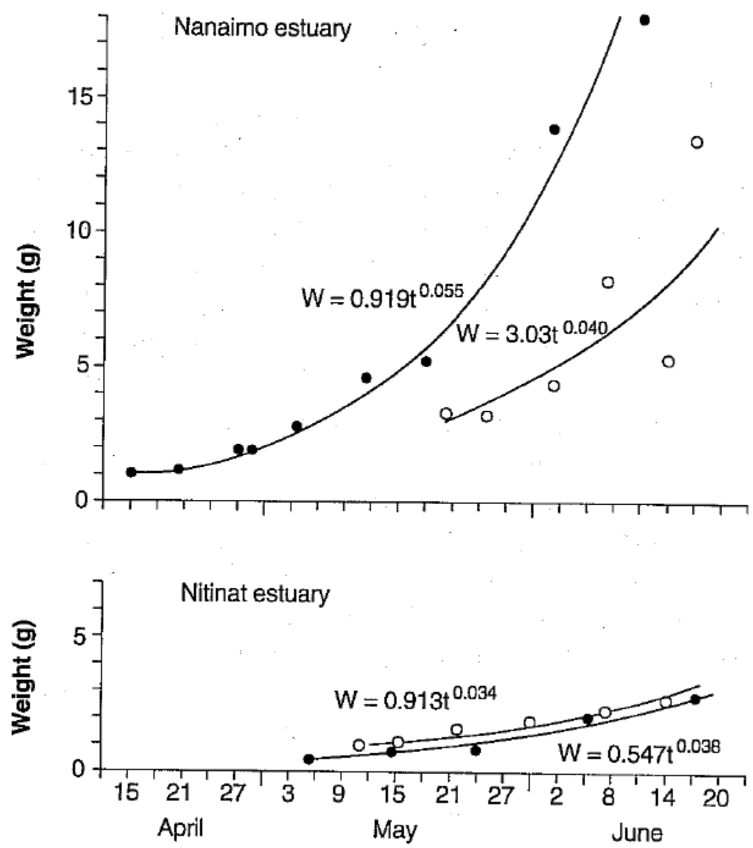
Diet: Young Chinook salmon in the Sacramento-San Joaquin River Delta feed primarily on insects (especially chironomid larvae and dipterans) and crustacea (especially Neomysids (opossum “shrimp”), amphipods (mud “shrimp”), copepods, and cladocerans (water fleas)) (Sasaki 1966, Kjelson et al. 1982). Chinook salmon fry appear to be opportunistic feeders; hence their diet varies with locale, season, and available prey (see CHSA Figures 3-6). 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 Fate: 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, CHSA Figure 2). Based on the average FCE (1.37) from the Cowlitz River, 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.

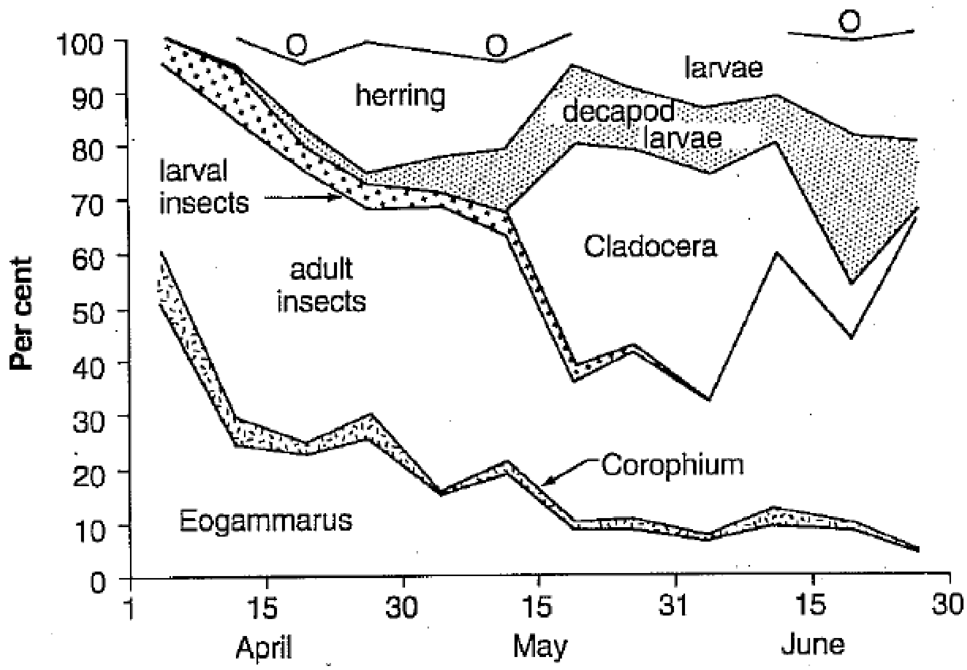


CHSA Figure 1. Observed and extant range of California Chinook salmon (Santos et al. 2014; https://pisces.ucdavis.edu/map)

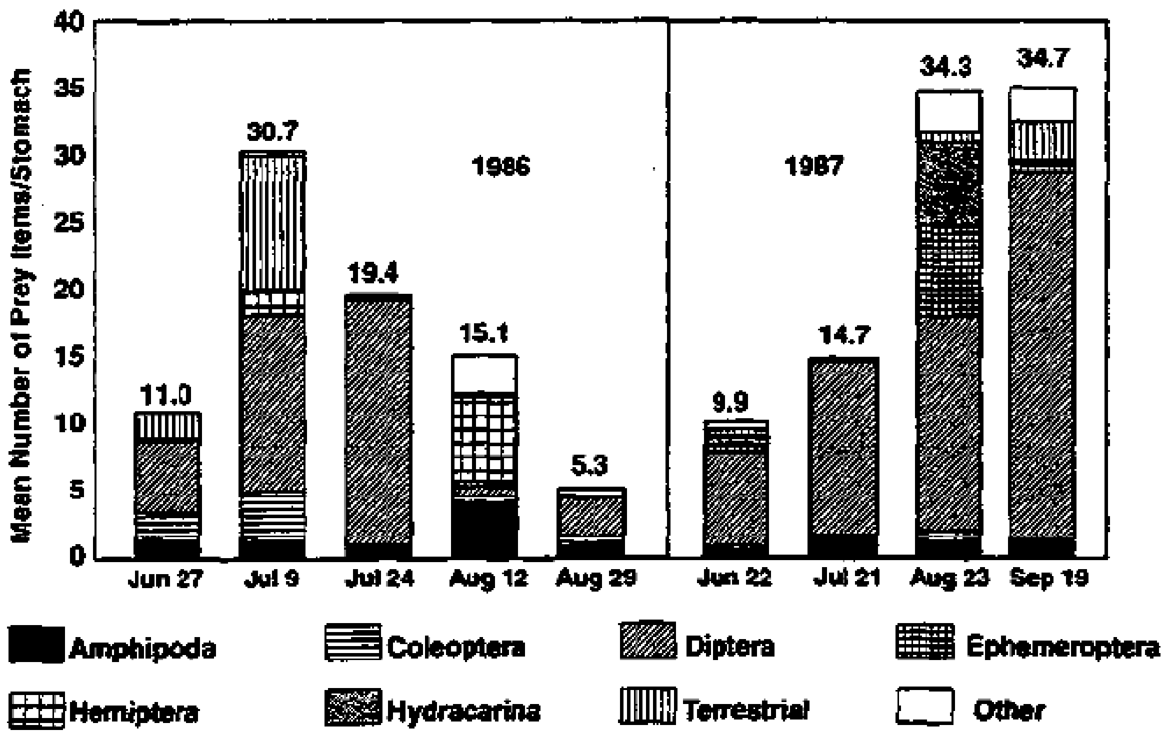
MacFarlane and Norton (2002) estimated the growth rate of Chinook salmon in the San Francisco estuary to be about 0.02 g per day for young averaging about 7 g in weight (about 0.3% body weight per day). However, this estimate is not based on recaptures of tagged individuals but by comparing averages 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 (CHSA Figure 7) is combined with seemingly good data from MacFarlane and Norton (2002) relating weight to fork length (best fit equation of CHSA Figure 8(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 River food conversion efficiency of 1.37 yields a food (dry weight) ingestion rate of 2.33% of live body weight per day. Using the larval weights range above provides an ingestion rate range of **0.012 – 0.41 g/d dw**.



CHSA Figure 2. 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).



CHSA Figure 3. 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).



CHSA Figure 4. 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

CHSA Figure 5. 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).

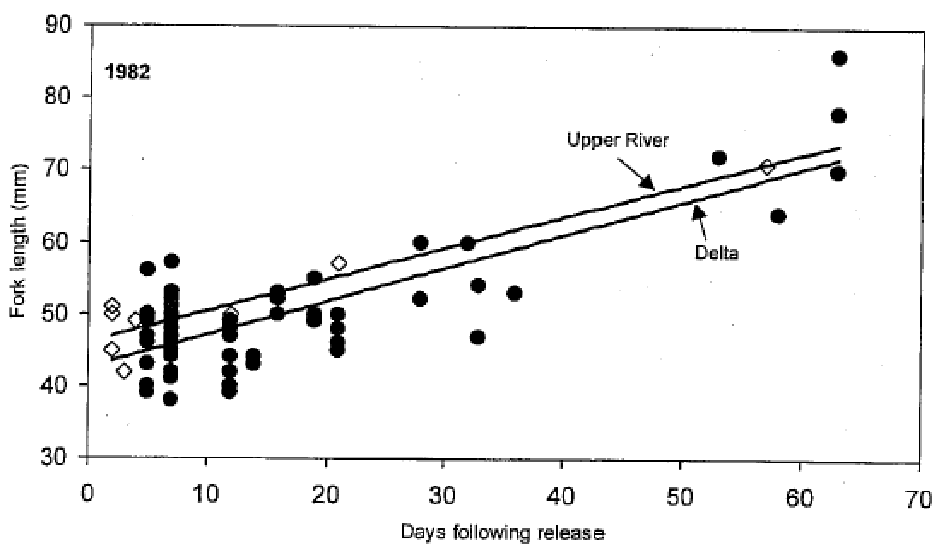
Location and prey species	%*N*	%*V*	%*FO*	*IRI*
km 68, Chipps Island (*n*=21)				
Malacostraca				
Decapoda				
Caridean shrimp	2.6	5.7	4.8	39.3
Crab megalopae	7.7	10.0	19.0	336.3
Mysidacea				
Unidentified	7.7	4.6	14.3	175.9
Amphipoda				
Gammaridea	2.6	5.7	4.8	39.8
Corophium spp.	30.8	26.4	33.3	1905
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

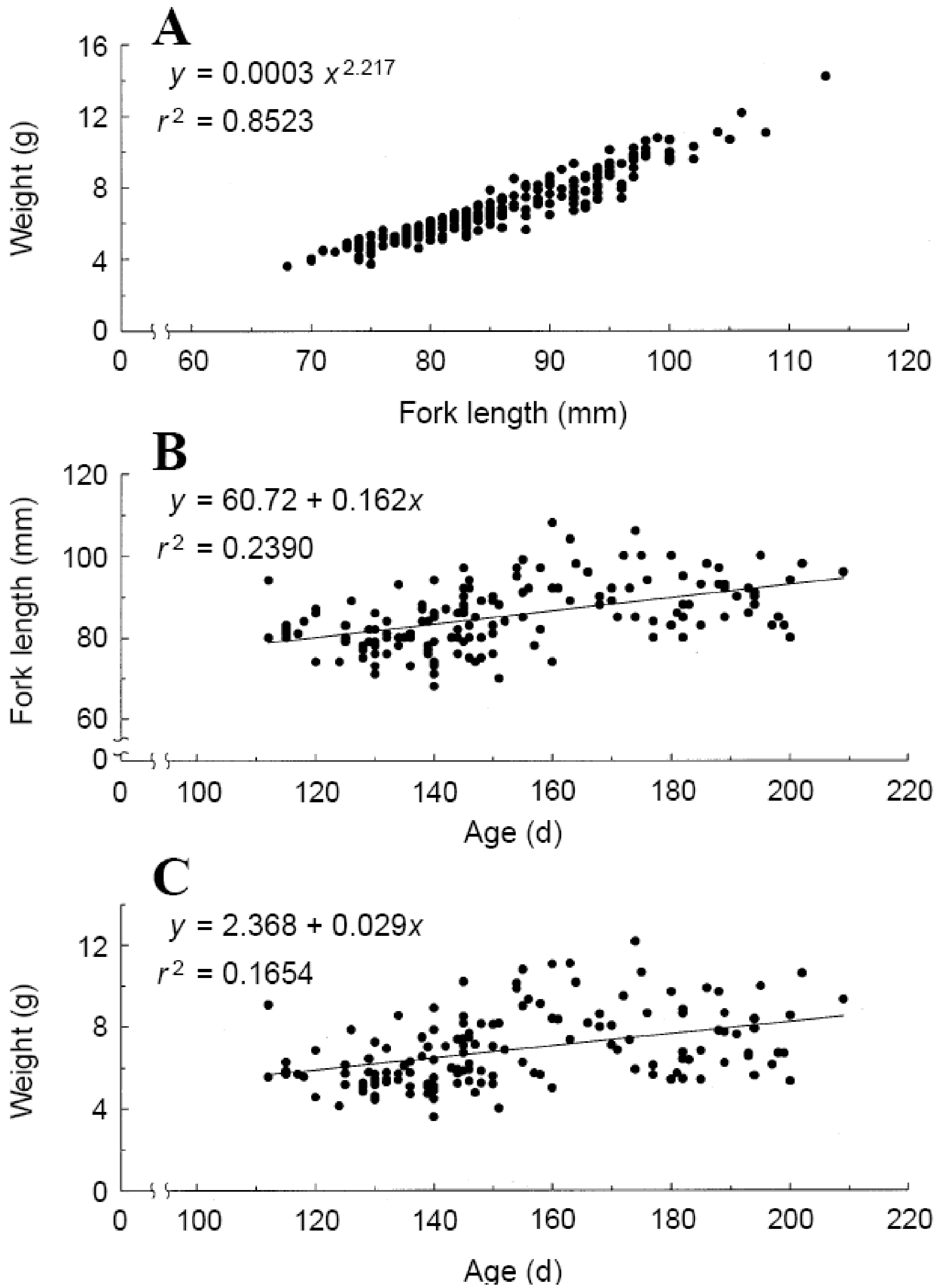
CHSA Figure 6. Stomach contents of juvenile Chinook salmon in the San Francisco estuary and the Gulf of the Farallones (Table 2 in MacFarlane and Norton 2002). %N is the numerical percentage; %V is the percent relative volume; %FO is the frequency of occurrence percentage; and IRI is the index of relative 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

CHSA Figure 6, continued. 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.



CHSA Figure 7. 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).



CHSA Figure 8. 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: 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 2 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 4 to 7 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 the Red Bluff Diversion Dam (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% of the young-of-the-year fish outmigrating through the lower Sacramento River and Delta during this period (CDFG 1998).

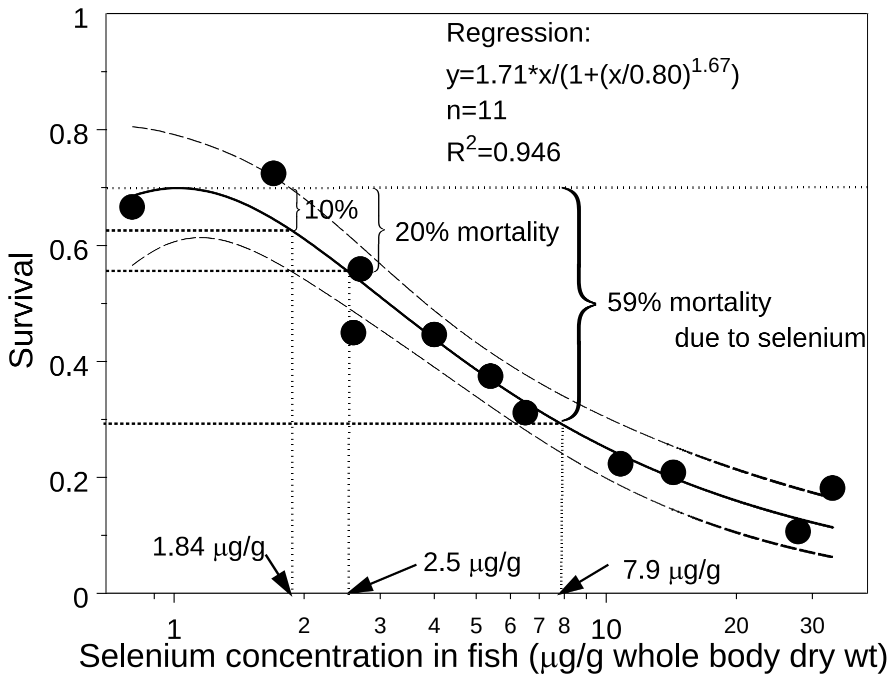
As Chinook salmon fry and fingerlings mature, they prefer to rear further downstream where ambient salinity is up to 1.5 to 2.5 parts per thousand (Healy 1980, 1982; Levings et al. 1986). Juvenile winter-run Chinook salmon occur in the Delta from October through early May based on data collected from 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 2 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.

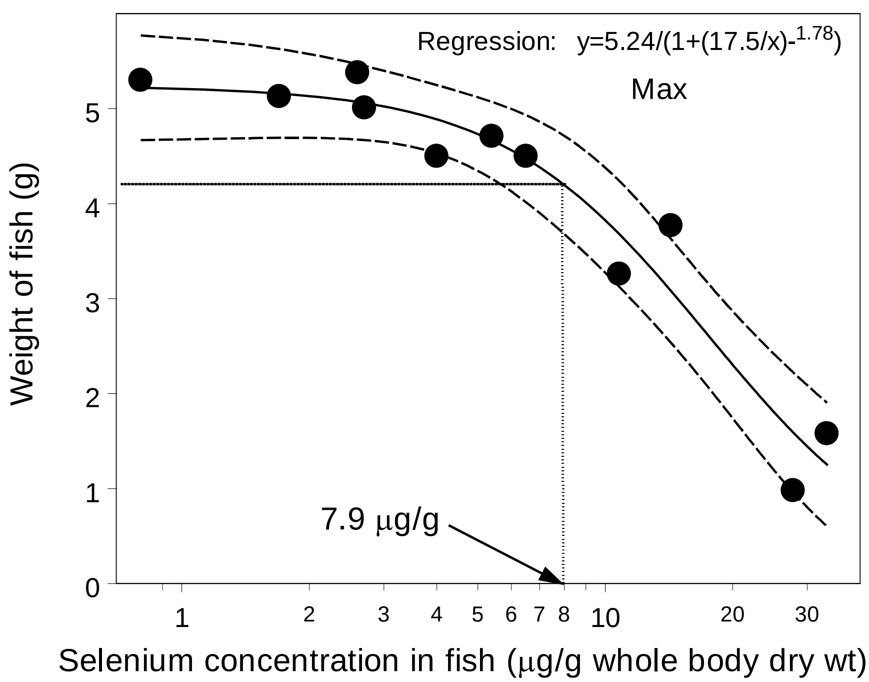
Risk of selenium exposure: California Central Valley Chinook salmon evidently are among the most sensitive of fish and wildlife to selenium. They are especially vulnerable during juvenile life stages when they migrate and rear in selenium-contaminated Central Valley rivers and the San Francisco Bay/Delta 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 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 (CHSA Figure 9). 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%) 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%). As a proportion of the maximum survival this is 0.29/0.7 = 0.41, or 41%. 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%. 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% mortality due to selenium (CHSA Figure 9).



CHSA Figure 9. 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.



CHSA Figure 10. Juvenile fall run Chinook salmon weight 90 days after swim up, in fresh water with dietary exposure to selenium.

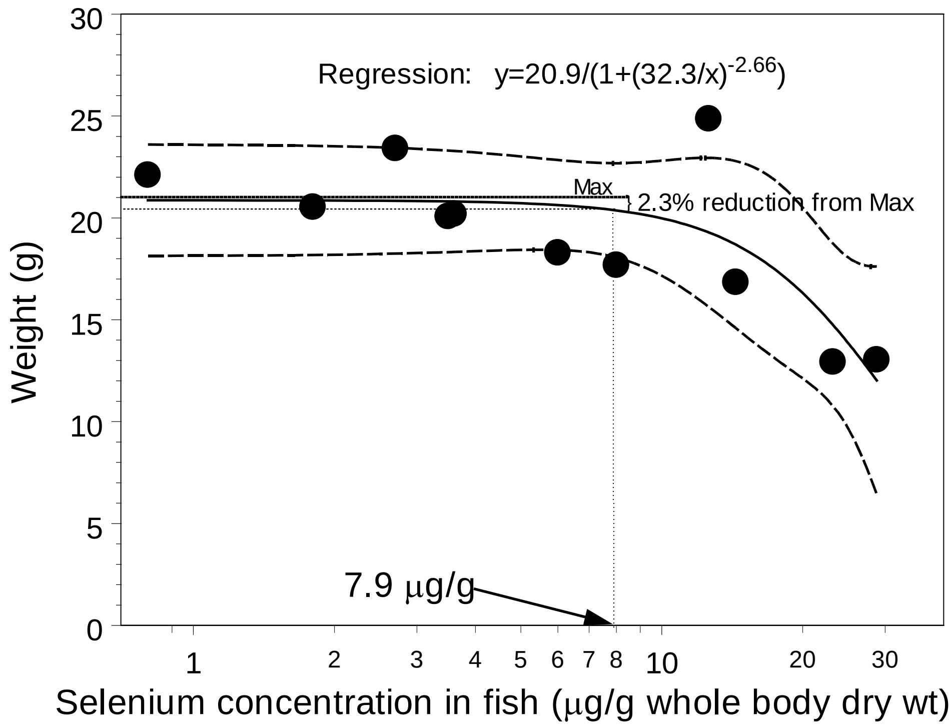
In the Hamilton et al. (1990) experiment, the weight measurements after 90 days of exposure do not exhibit clear evidence of a biphasic dose-response relationship (CHSA Figure 10). This suggests that none of the dietary treatments in this experiment was low enough in selenium to be substantially deficient with respect to weight gain. This is consistent with the results of an experiment in which rainbow trout juveniles were 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% reduction from the maximum weight of 5.24 g (CHSA Figure 10).

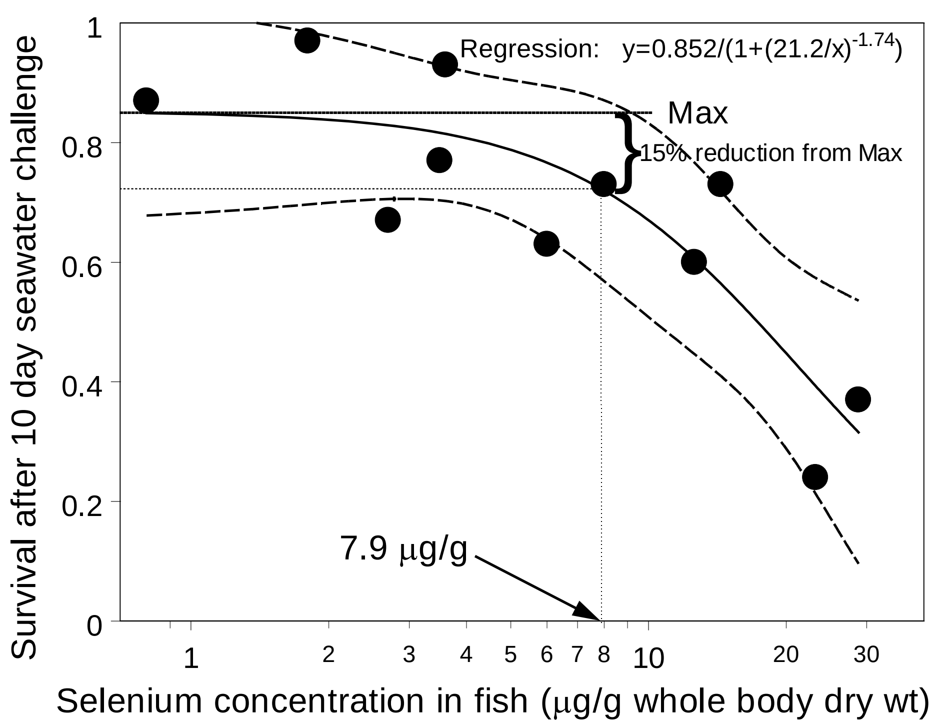
We conclude that, 90 days after swimup, Chinook salmon juveniles that bioaccumulate selenium to a concentration of 7.9 µg/g could suffer up to 59% mortality due to selenium (CHSA Figure 9, and the survivors may be reduced in weight by 20% due to selenium (CHSA Figure 10).

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% reduction in growth due to selenium (CHSA Figure 11).

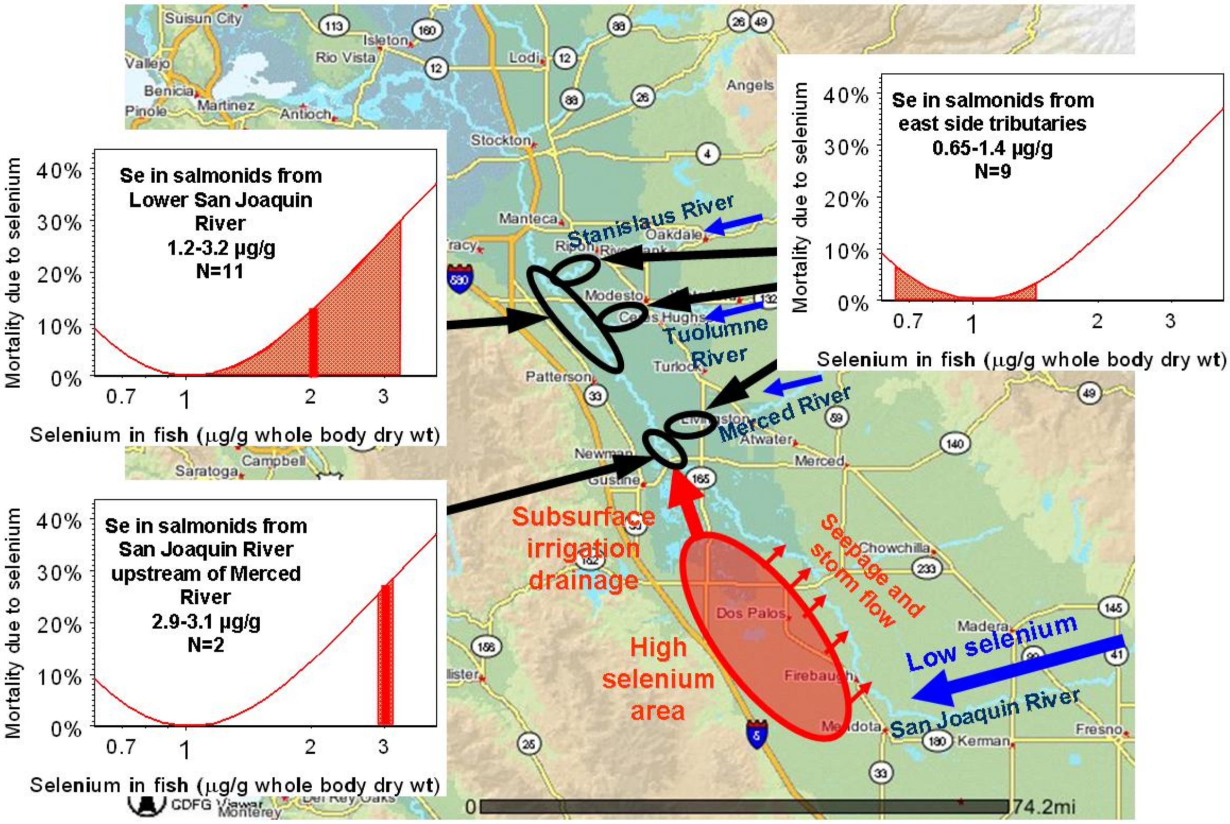
After rearing the young salmon in brackish water, Hamilton et al. (1990) simulated the passage of the salmon from the estuary into the ocean by challenging them with 10 days of emersion in reconstituted seawater. The results of this final phase of the experiment suggest that upon entering the ocean, young salmon with a tissue concentration of 7.9 µg/g could experience 15% mortality within 10 days, due to selenium (CHSA Figure 12).



CHSA Figure 11. Juvenile fall run Chinook salmon weight after 120 days of rearing in brackish water with dietary exposure to selenium.

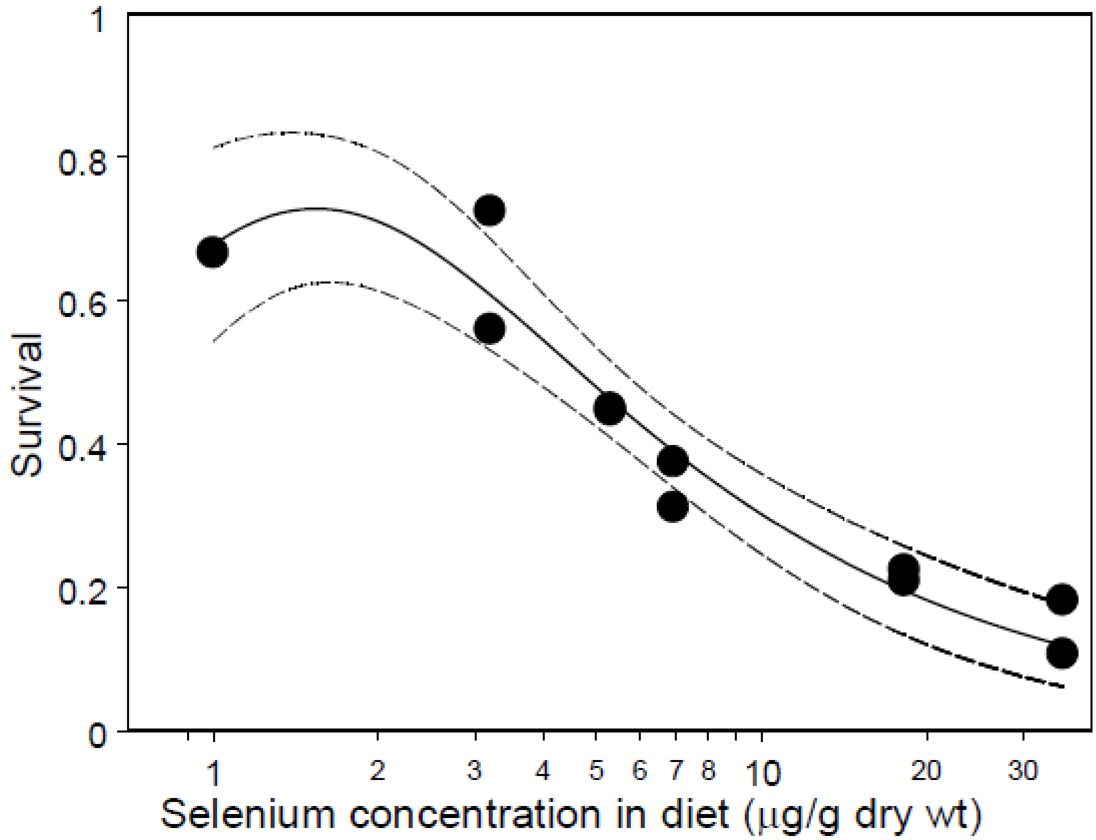


CHSA Figure 12. Survival of juvenile fall run Chinook salmon after 10 day seawater challenge following rearing for 120 days in brackish water with dietary exposure to selenium.



CHSA Figure 13. Risk of mortality to juvenile Chinook salmon based on selenium measured in the salmon (Saiki, et al. 1991) and the toxicity data shown in Figure 9 (presented here as mortality). The stippled red areas span the ranges of concentrations in fish at the respective locations.

Available data (Saiki et al. 1991) confirm that young salmon migrating down the San Joaquin River in 1987 bioaccumulated selenium to levels (about 3 µg/g whole body dry wt.) that has the potential to kill more than 25% of those juveniles spending sufficient time in this area of the river system(CHSA Figure 13). In the Hamilton et al. (1990) experiment, the concentrations of selenium in the food that was provided to the salmon were about the same as the concentrations reached by the salmon themselves. This experiment indicates that, in sloughs that carry agricultural drainwater, concentrations of selenium in invertebrates, small (prey) fish, and larger predatory fish commonly reach levels (Beckon 2015, Gordus and McNeal 2015) that could kill a substantial portion of young salmon (CHSA Figure 13) if the salmon, on their downstream migration, are exposed to those selenium-laden food items long enough for the salmon themselves to bioaccumulate selenium to toxic levels. However, selenium concentrations in the San Joaquin River have slowly declined as the Grassland Bypass Project continues to reduce selenium loads to the river (SFEI 2015, CEDEN 2017).

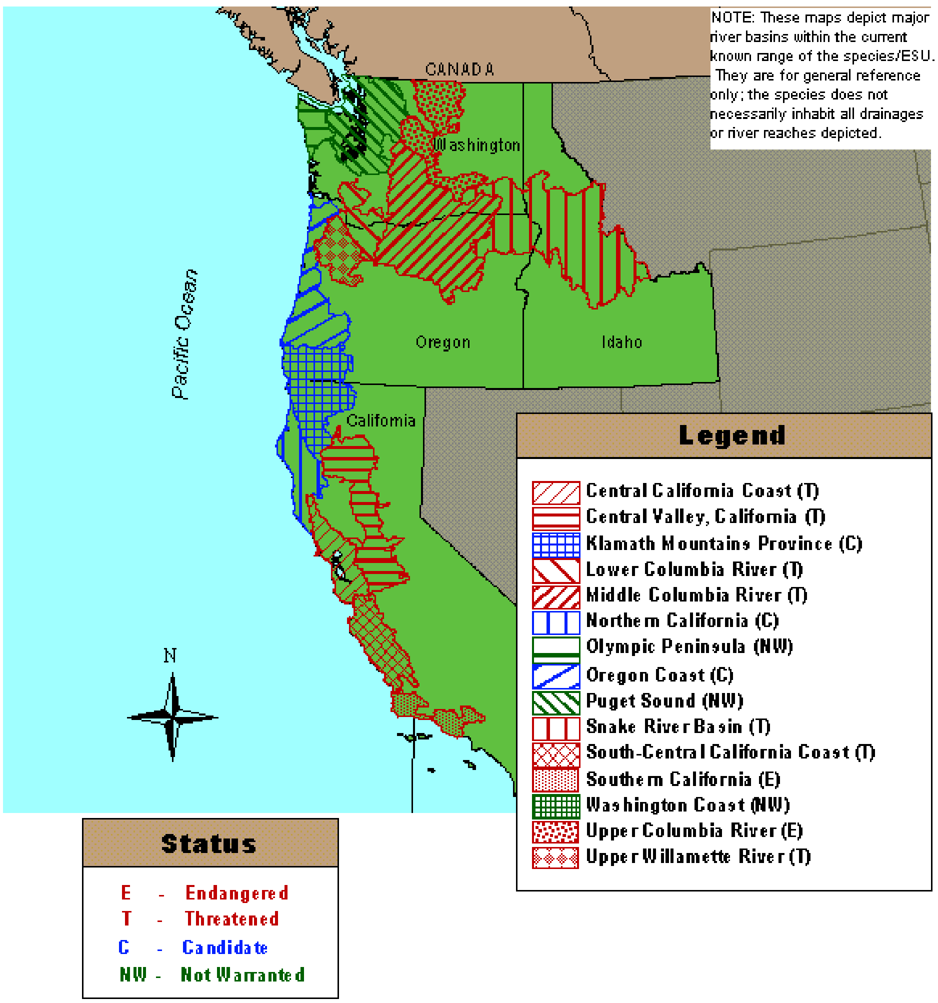


CHSA Figure 14. Survival as a function of selenium concentration in diet of juvenile Chinook salmon after 90 days of exposure to dietary selenium. A biphasic model (Brain and Cousens 1989) was fitted by least squares regression. Dashed lines indicate 95% confidence boundaries.

Steelhead Trout (STTR)

Oncorhynchus mykiss

Status: Steelhead trout are the anadromous form of the rainbow trout species. Central California Coast steelhead were listed as threatened in 1997 (62 FR 43937). This Evolutionarily Significant Unit (ESU) includes steelhead in San Francisco and San Pablo Bays and their tributaries (STTR Figure 1). Central Valley steelhead were listed as threatened in 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 (STTR Figure 1).



STTR Figure 1. Status of west coast steelhead (http://swr.nmfs.noaa.gov/psd/stlesu.htm).

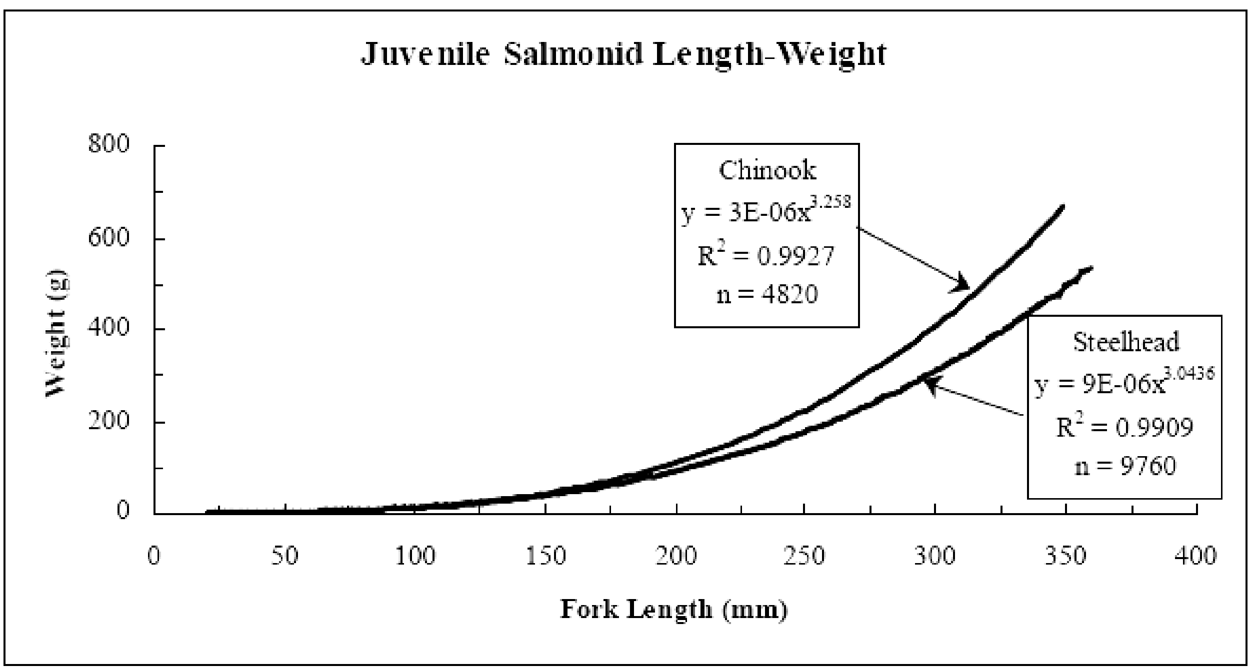
The breeding of wild steelhead in the Central Valley is mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks and a few wild steelhead are produced in the American and Feather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

At one time steelhead were thought to be extirpated from the San Joaquin River system. 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).

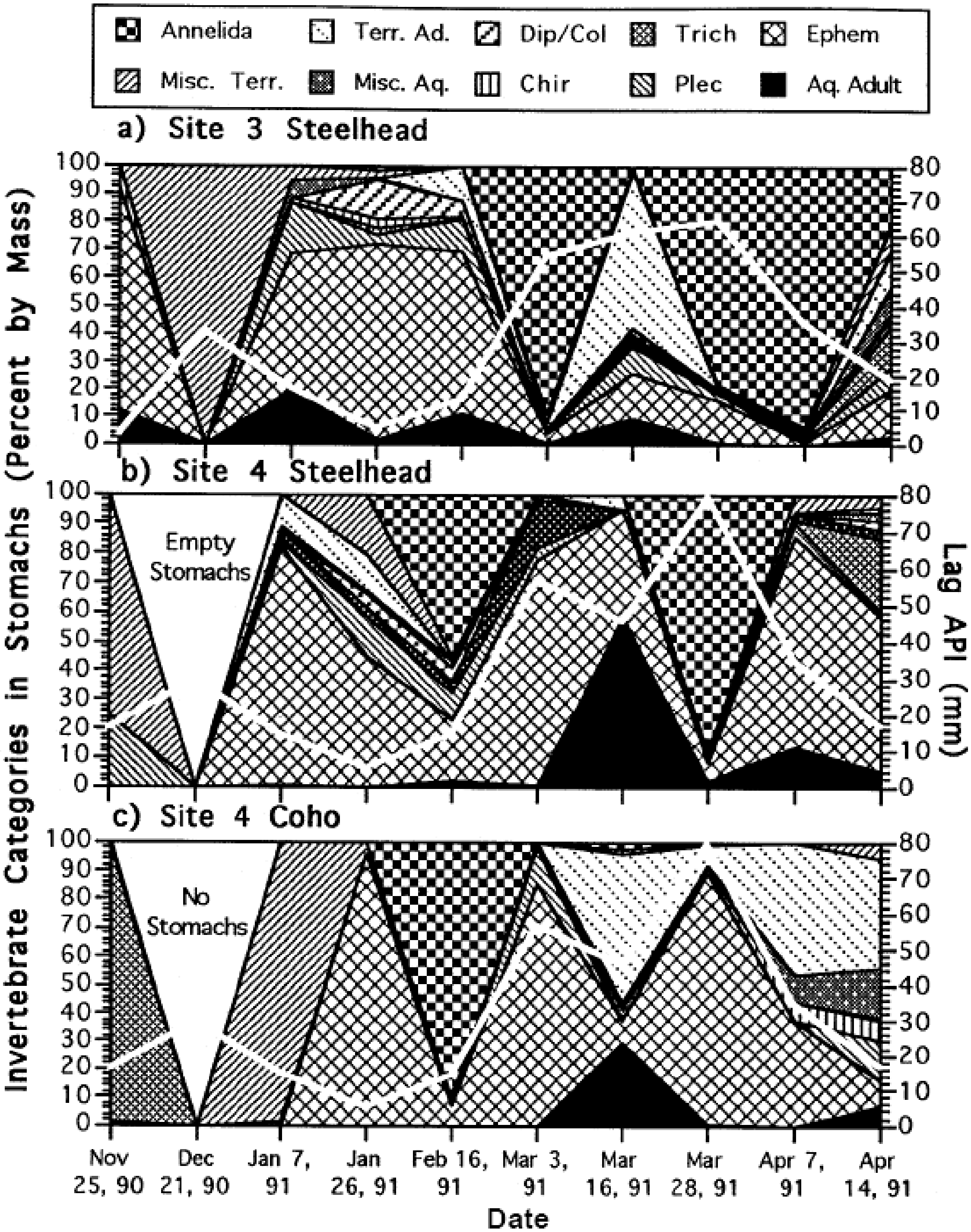
Historically, steelhead were abundant in San Francisco Bay area streams, such as the Napa River and its tributaries; however, populations have declined. The South-Central California Coast steelhead ESU critical habitat includes all river reaches and estuarine areas accessible to listed steelhead in coastal 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 STTR Figure 2 for central California coastal steelhead. A mark and recapture study from June through August on the Feather River in 2002 (CDWR 2003) had juvenile weights range from **1.3 - 22.6 g**.

Diet: Juvenile steelhead feed on a wide variety of aquatic and terrestrial insects (STTR Figure 3 and STTR Table 1) and emerging fry are sometimes preyed upon by older juveniles.



STTR Figure 2. 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)



STTR Figure 3. Diet composition, shown as percent 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.

STTR Table 1. Number (No.), percent abundance (Abund.), percent weight (Wt.), and percent occurrence frequency (Occur.) of prey items from stomachs of 3 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 (STTR Table 2) 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

$$SpecificGrowthRate = 1 - e^b$$

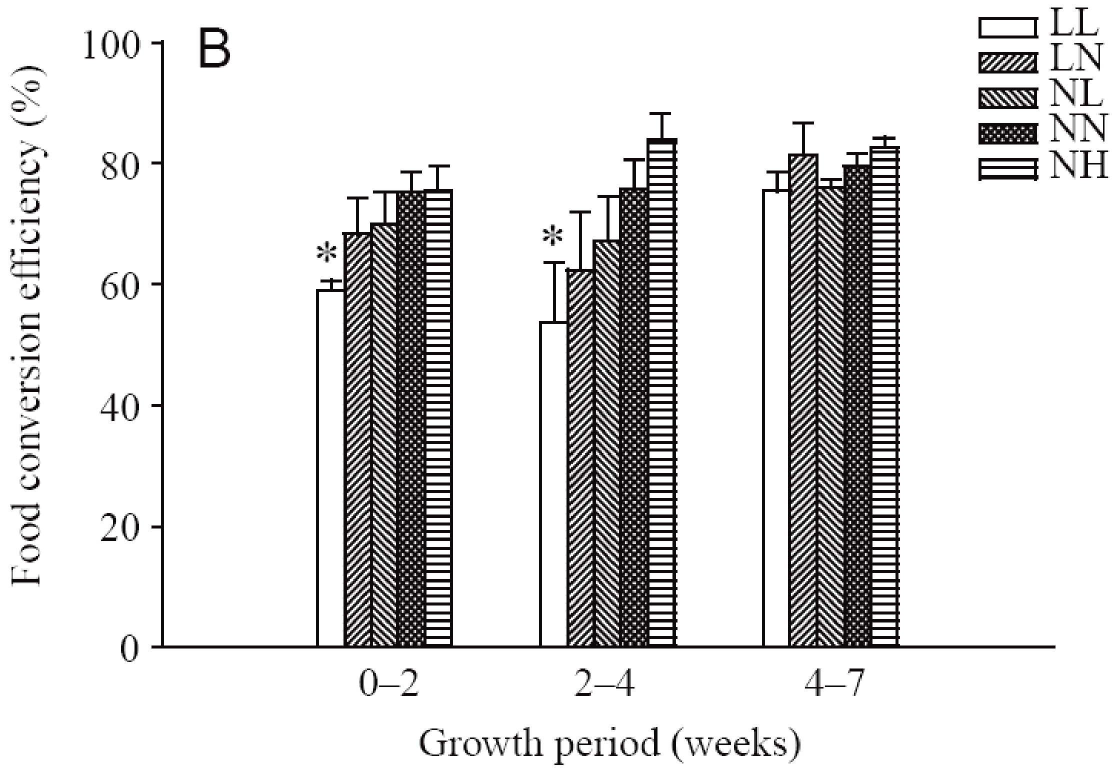
STTR Table 2. Measurements of all steelhead marked and recaptured in the Feather River in 2002 (CDWR 2003). Specific locations are BP=Bedrock Park, HD=Hatchery Ditch, SR=Steep Riffle.

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 STTR Table 2 yields the specific growth rates listed in STTR Table 3, with an average specific growth rate of 2.29%.

STTR Table 3. Specific growth rates calculated from the data in STTR Table 2.

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%



STTR Figure 4. 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, STTR Figure 4). 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 (STTR Table 4), a water temperature commonly encountered in the shallow waters of the Delta.

If we use the food conversion efficiency (FCE) of 0.80 and the average specific growth rate (SGR) of 2.29% (STTR Table 3) 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)

STTR Table 4. 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).

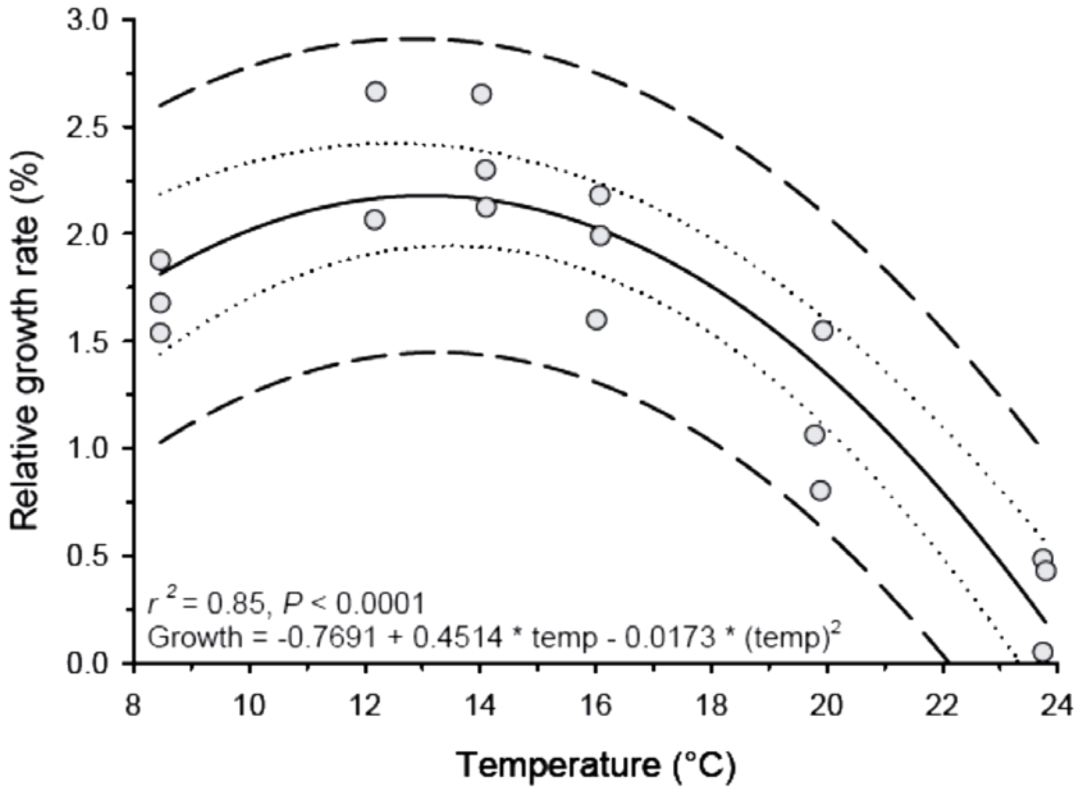
Temperature	Feed consumption (% body weight)		Feed conversion efficiency (g growth / g consumed)	
°C	WCT	RBT	WCT	RBT
8	0.73(0.04) A	0.86(0.07) A	1.27(0.05) A	1.17(0.03) A
12	0.87(0.04) A	1.03(0.05) A	1.33(0.06) A	1.15(0.04) A
14	0.99(0.03) A	1.08(0.05) A	1.27(0.03) A	1.15(0.02) A
16	1.13(0.06) B	0.92(0.10)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

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, STTR Figure 5, see Life History below), then the Bear (2005) experiment indicates that food conversion efficiency would be about 1.15 (STTR Table 4). 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, STTR Table 4) 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. Using the weight information above, a food ingestion rate range for juvenile steelhead is **0.03 – 0.45 g/d dw**.



STTR Figure 5. 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 (percent) 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.

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 ^{o}F to 52 ^{o}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%) 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).

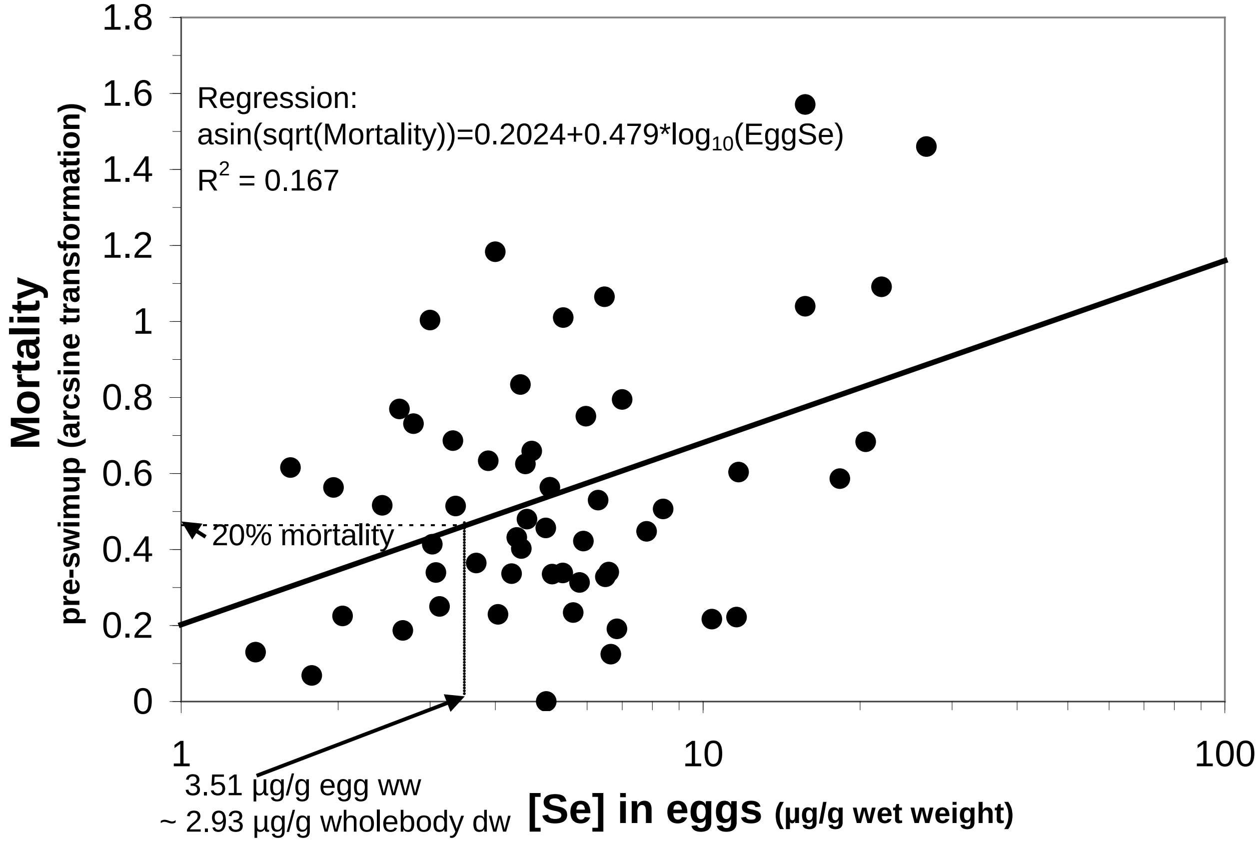
Steelhead rearing during the summer takes place primarily in higher velocity areas in pools, although young-of-the-year also are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Productive steelhead habitat is 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 2 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.

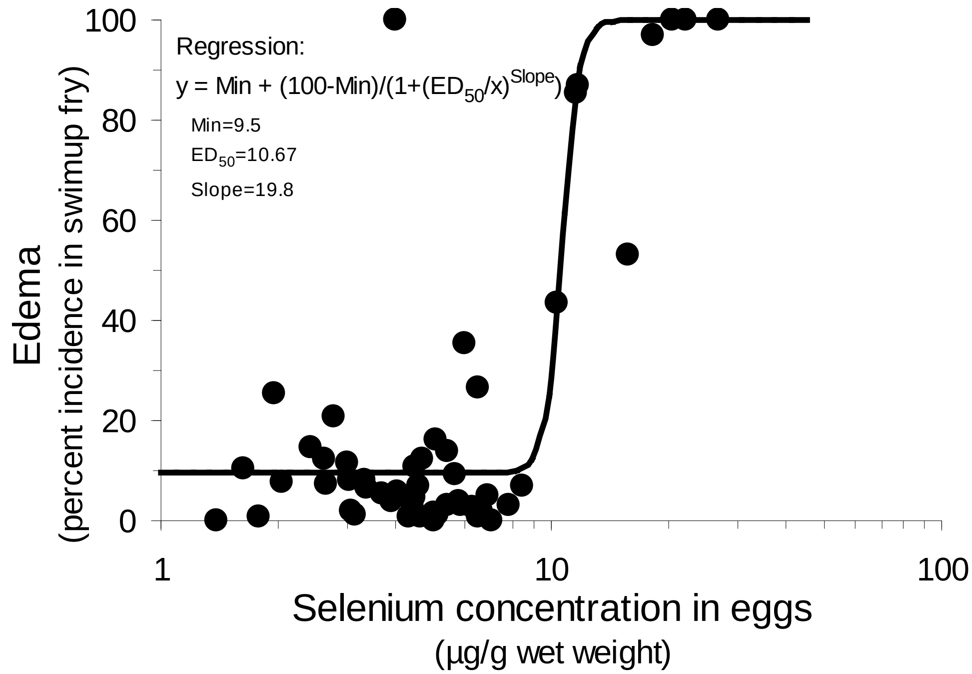
Juvenile steelhead emigrate episodically from natal streams during fall, winter, and spring high flows. Emigrating Central Valley steelhead use the lower reaches of the Sacramento River and the Delta for rearing and as a migration corridor to the ocean. Some may utilize tidal marsh areas, non-tidal freshwater 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% mortality of fry if female rainbow trout have a tissue selenium concentration of 2.93 µg/g whole body dry weight (STTR Figure 6). Among the swim up survivors, various deformities were also associated with elevated selenium in the eggs from which the fry hatched. For example, both edema (STTR Figure 7) craniofacial deformities (STTR Figure 8) increase sharply above an egg selenium concentration of about 10 µg/g ww. This egg threshold corresponds to a maternal tissue concentration of about 6.6 µg/g (whole body dry weight; for details, see USFWS 2005).

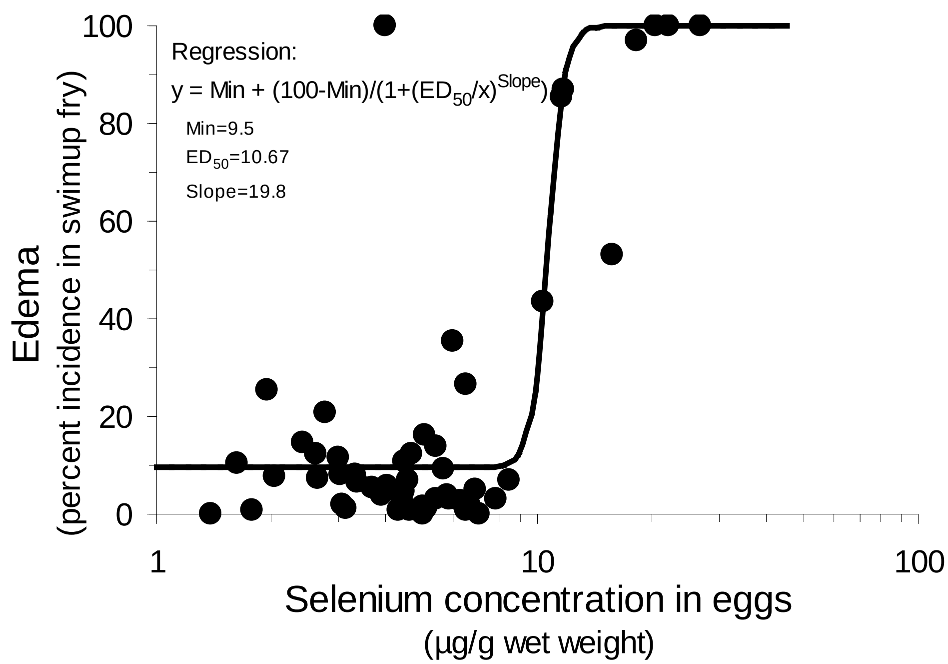
Another laboratory experiment monitored the growth of juvenile rainbow trout exposed for 20 weeks to a diet spiked with selenium in the form of sodium selenite (Hilton et al. 1980). This experiment indicates that, relative to optimal selenium exposure, a weight reduction of 20% would be associated with a tissue selenium concentration of 2.15 µg/g (carcass dry weight) (STTR Figure 9).



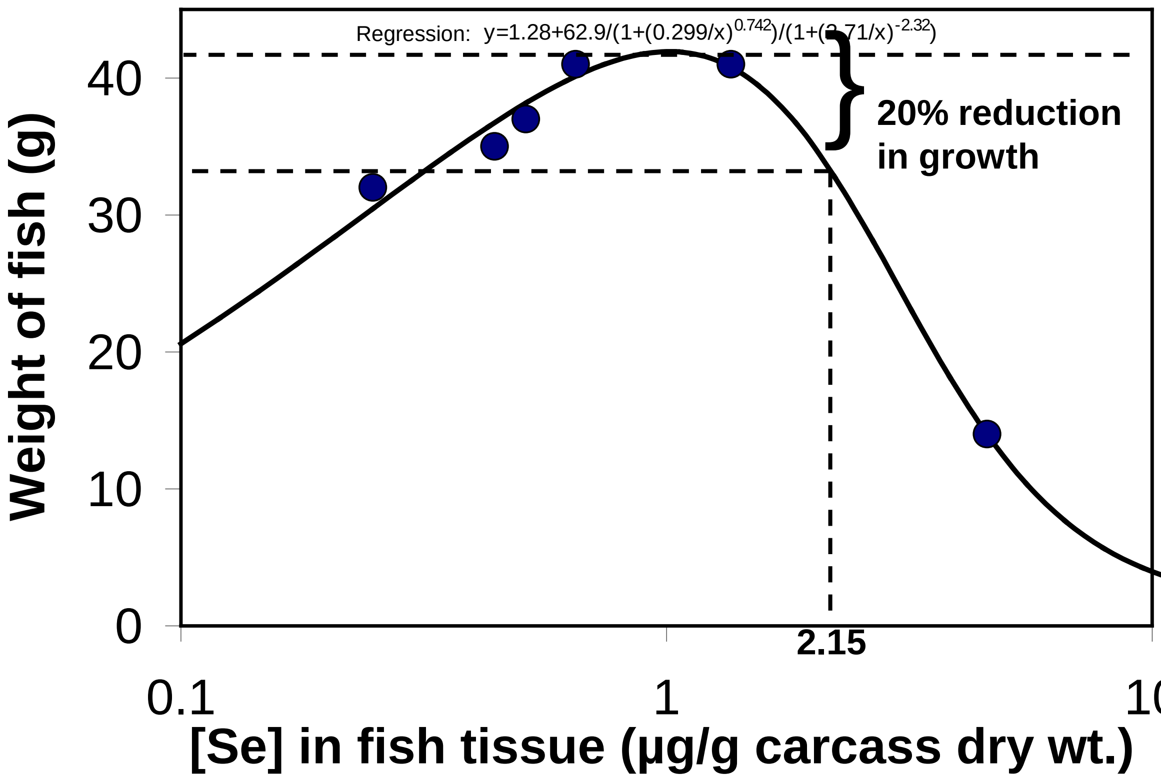
STTR Figure 6. Relationship between selenium in rainbow trout eggs and mortality of eggs and fry by swimup stage. The arcsine transformation is applied to mortality data, as appropriate for linear regressions with percents or proportions (Sokol and Rohlf 1981). Data are from the years 2000-2002.



STTR Figure 7. 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.).



STTR Figure 8. 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.).



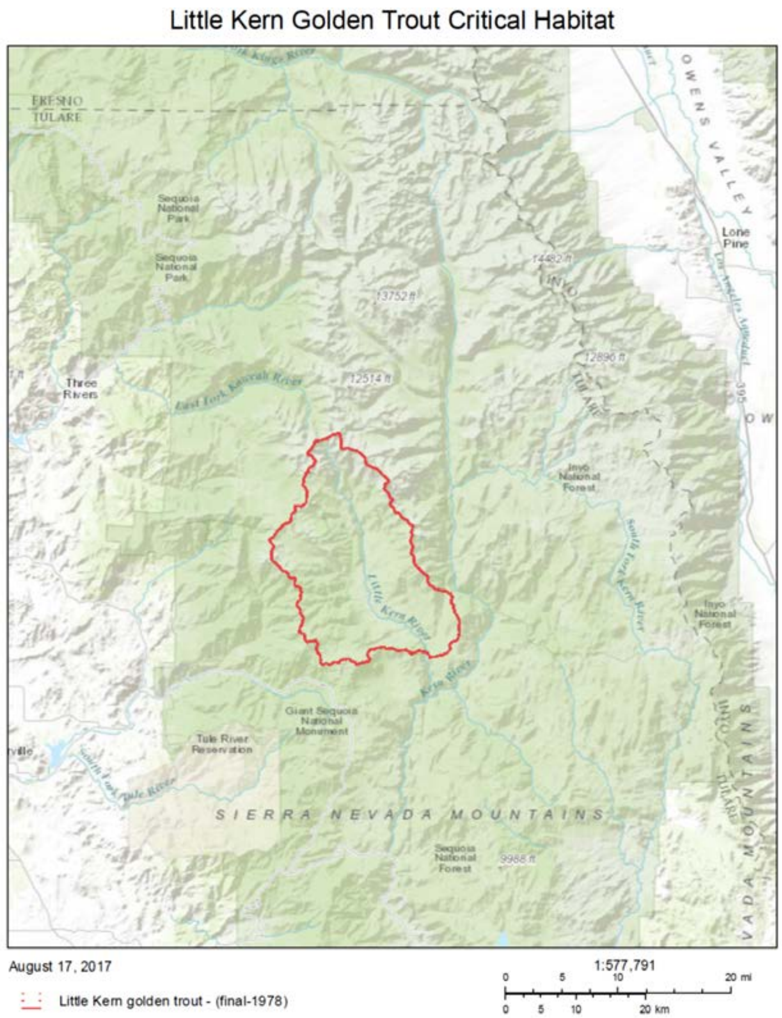
STTR Figure 9. Average weights of juvenile rainbow trout after 20 weeks of exposure to diets spiked with sodium selenite (Hilton et al. 1980). The data were fitted with a biphasic model (Beckon et al. 2008). In the model it was assumed that at extremely high and extremely low selenium concentrations, 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.

Little Kern Golden Trout (LKGT)

Oncorhynchus aguabonita whitei

Status: The Little Kern golden trout was listed as threatened under the Endangered Species Act in 1978 (43 FR 15427-15429). Its cause for decline is due to hybridization with introduced coastal rainbow trout, habitat loss and degradation (Moyle 2002). It was estimated that less than 5,000 Little Kern golden trout individuals occurred at the time of listing with only pure individuals located in short reaches of headwater streams with barriers (USFWS 2011). The current abundance of the Little Kern golden trout cannot be determined due to inconsistent methodology and incorrect assumptions concerning the level of hybridization.

Distribution: The Little Kern golden trout is only found in the Little Kern River drainage of Tulare County (USFWS 2011). Critical habitat was designated in 1978 and includes the Little Kern River main channel and all tributaries to the Little Kern River above the barrier falls located on the Little Kern River one mile below the mouth of Trout Meadows Creek (LKGT Figure 1).



LKGT Figure 1. Little Kern golden trout critical habitat map.

Size: Golden trout species have slow growth rates due to the low productivity habitat and short growing seasons but can live up to 9 years (Moyle 2002). Golden trout will reach up to 4 cm in length (standard length [SL], tip of snout to end of vertebral column) their first year, 10-11 cm SL by the end of their third summer, and then grow 1-2 cm per year up to a maximum 20 cm SL (Moyle 2002). This corresponds to a weight range of **0.7 g** at end of first year, to a maximum **91 g** at adulthood, according to the length-weight regression equation shown in STTR Figure 2 for central California coastal steelhead.

Diet: Little is known about the specific diet of the Little Kern golden trout but golden trout, in general, eat primarily larval and adult aquatic invertebrates and some terrestrial invertebrates (Moyle 2002).

Food Ingestion Rate: There is no specific information about the food ingestion rate of Little Kern golden trout, but is likely low due to the general low productivity within its small stream habitats and their slow growth rates. A detailed discussion of food ingestion rates for juvenile steelhead trout (*Oncorhynchus mykiss*), the anadromous form of rainbow trout, is provided above. Based on that discussion an appropriate food ingestion rate for the Little Kern golden trout would be in the lower range of 0.8% to 1.0% (dry weight percent of live body weight per day). Using the weight range above provides an ingestion rate range of **0.0056 – 0.73 g/d dw**.

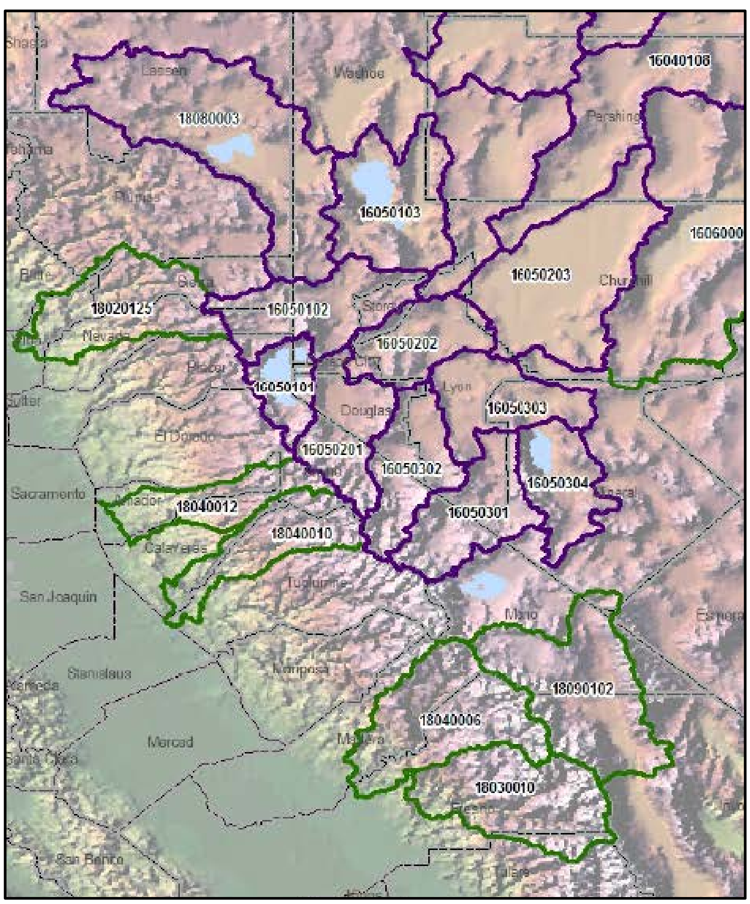
General Life History: Little Kern golden trout require cold, flowing waters with pool-tail crests of 5 to 15 cm deep for spawning (USFWS 2011). Spawning temperatures range from 10 to 15^{o}C (Moyle 2002) and substrate size is between 5 to 10 mm. Maturity is usually reached in 3 to 4 years (Moyle 2002) and females produce relatively few eggs (41–65) compared to other trout species (USFWS 2011). Average home range for Little Kern golden trout is 16.5 m (USFWS 2011).

Risk of Selenium Exposure: As a subspecies of rainbow trout the general sensitivity of golden trout to selenium are likely similar. The discussions of selenium sensitivity of steelhead trout in this report apply to the Little Kern golden trout; however, specifically for the Little Kern golden trout, direct exposure to selenium is expected to be relatively low due to the apparent lack of selenium sources in its range. A review of the U.S. Geological Survey's Mineral Resources Data System (https://mrdata.usgs.gov/mrds/map-us.html) shows a single mine (Pine Tree Mine-tungsten) in the Little Kern golden trout watershed, but potential impacts from that mine are unknown. Some mining activity especially for copper and sulfur may be sources of selenium. Henderson et al. (2013) documented elevated selenium levels below Leviathan Mine in the East Walker River watershed. Selenium levels ranged from <1 to 6.6 µg/L below the mine site in 2001. In 1999, Thompson and Welsh (2000) documented selenium levels in streams below Leviathan Mine that ranged from <1 to 13 µg/L.

Lahontan Cutthroat Trout (LCTR)
Oncorhynchus clarkii henshawi

Status: The Lahontan cutthroat trout (LC trout) was listed as endangered in 1970 (40 FR 16047-16048) and subsequently down-listed to threatened in 1975 (40 FR 29863-29864). Habitat fragmentation and isolation pose a substantial threat to the LC trout range wide since 72% of the LC trout populations are completely isolated in short (< 8 km) stream reaches (USFWS 2009d). Year to year abundance is highly variable and depends greatly on stream flows in spring and early summer (USFWS 2009d). Most occupied habitat (83%) has fewer than 94 fish per kilometer with the greatest densities >250 occurring in only 3% of occupied habitat (USFWS 2009d). Two watersheds in California had more than 250 LC trout per kilometer—West Fork Walker River and the Upper Yuba River.

Distribution: LC trout historically occupied lakes, streams, and rivers of the Lahontan Basin of northern Nevada, eastern California, and southern Oregon. In California this includes the Truckee, Carson, Walker, and Susan River watersheds (LCTR Figure 1). There are additional out-of-basin populations in 6 other California watersheds—Upper Yuba, Upper King, Upper San Joaquin, Upper Mokelumne, and Crowley Lake (LCTR Figure 1). The LC trout 5-year review has more detailed maps of currently occupied habitat in each watershed.



LCTR Figure 1. Lahontan cutthroat trout watersheds in California. Historic watersheds in purple with out-of-basin transplanted populations in green watersheds. Modified from: USFWS 2009d.

Size: Growth rates of LC trout vary depending on habitat (USFWS 2009d). Lacustrine LC trout can grow rapidly (60-100 mm/yr.) while fluvial LC trout grow slower (25-89 mm/yr.) as estimated from Table 1 (USFWS, 2009d). Juvenile LC are likely similarly sized as the Paiute cutthroat Trout, described below, averaging approximately 35 mm as fry. Average size of adult fluvial LC trout ranges from 251-282 mm while lacustrine LC trout can grow to well over 400 mm (Table 1, USFWS 2009d) with a record LC trout of over 990 mm from Pyramid Lake, NV (Moyle 2002). This corresponds to a weight range of **0.45 g** for fry to **258 g** for fluvial adults, according to the length-weight regression equation shown in STTR Figure 2 for central California coastal steelhead.

Diet: Like other trout species in stream habitats the LC trout depends upon terrestrial and aquatic insects (Moyle 2002). Lacustrine LC trout eat insects when young but begin eating fish as they reach larger sizes. In Pyramid Lake, 200mm lacustrine LC trout begin to feed on fish, at 300mm 50% of the diet is fish, and at 500mm their diet approaches 100% fish (USFWS 2009d). Since it appears that LC trout in California are fluvial populations their diet of terrestrial and aquatic insects would be most appropriate to consider for assessing selenium exposure.

Food Ingestion Rate: There is no specific information about the food ingestion rate of LC trout. It is likely low for stream dwelling LC trout due to the general low productivity within small stream habitats and their slow growth rates. A detailed discussion of food ingestion rates for steelhead trout (*Oncorhynchus mykiss*), the anadromous form of rainbow trout, is provided above. Based on that discussion an appropriate food ingestion rate for the fluvial LC trout would be in the lower range of 0.8% to 1.0% (dry weight percent of live body weight per day). Lacustrine LC trout likely have a higher ingestion rate and could be as high as 2% (dry weight percent of live body weight per day). Using 0.8% and the size range above provides an ingestion rate range of **0.0036 – 2.1 g/d dw**.

General Life History: LC trout live in both lakes and streams but must spawn in streams. They have been documented traveling up to 160 km to reach spawning areas (USFWS 2009d). Depending on stream flows spawning occurs from April to July at temperatures from 5 to 16^{o}C. Repeat spawning of lacustrine LC trout (not fluvial) is known to occur 1 to 3 years later at variable percentage rates depending on location (USFWS 2009d). Lacustrine LC trout produce 600 to 8,000 eggs while stream fish produce less than 300 eggs.

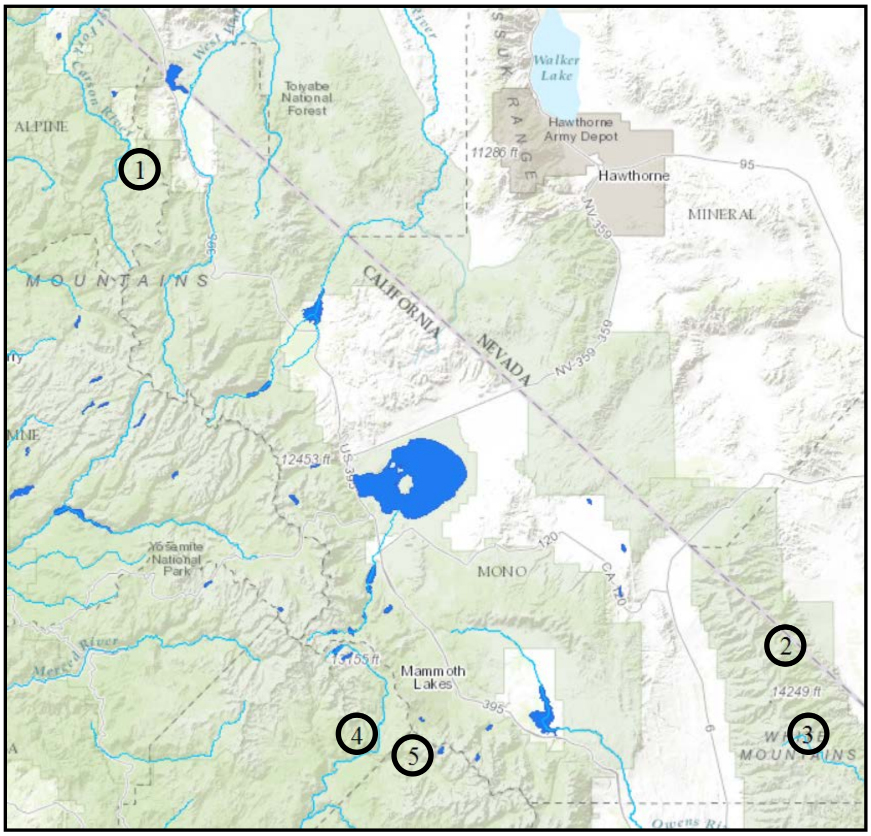
Risk of Selenium Exposure: As a species within the Oncorhynchus genus the general sensitivity of LC trout to selenium are likely similar to rainbow trout. The discussions of selenium sensitivity of steelhead trout in this report apply to the LC trout. Direct exposure to selenium, specifically for the LC trout in California, is expected to be relatively low due to the lack of selenium sources in its range; however, some mining activity especially for copper and sulfur may be sources of selenium. Therefore, fluvial LC trout are likely the most at risk for Se exposure. Henderson et al. (2013) documented elevated selenium levels below Leviathan Mine in the East Walker River watershed. Selenium levels ranged from <1 to 6.6 µg/L below the mine site in 2001. In 1999, Thompson and Welsh (2000) documented selenium levels in streams below Leviathan Mine that ranged from <1 to 13 µg/L.

Paiute Cutthroat Trout (PCTR)

Oncorhynchus clarkii seleneris

Status: The Paiute cutthroat trout (Paiute trout) was listed as endangered in 1967 (FR 32 4001) with a revised listing to threatened in 1975 (40 FR 29863–29864). Numbers of Paiute trout can vary dramatically from year-to-year (<100-1,800) within a given watershed due to natural conditions (flooding, thick ice) and human activities (water treatments, transplanting). Overall, estimates suggest the Paiute trout populations are stable (USFWS 2013).

Distribution: Paiute trout historically occupied approximately 17.8 km of stream habitat within the Silver King Creek drainage, Alpine County, California. They currently occupy about 37.8 km of stream habitat in five widely distributed drainages outside of their historical range (PCTR Figure 1). Existing Paiute trout populations are found in: Silver King Creek, Humboldt-Toiyabe



PCTR Figure 1. Location of existing Paiute cutthroat trout populations in the Sierra Nevada Mountain Range: 1) Silver King Creek, Humboldt-Toiyabe National Forest, Alpine County, California; 2) Cabin Creek, Inyo National Forest, Mono County, California; 3) North Fork Cottonwood Creek, Inyo National Forest, Mono County, California; 4) Stairway Creek, Sierra National Forest, Madera County, California; and 5) Sharktooth Creek, Sierra National Forest, Fresno County, California (Based on: USFWS 2013).

National Forest, Alpine County, California (upstream of historic habitat); Cabin Creek, Inyo National Forest, Mono County, California; North Fork Cottonwood Creek, Inyo National Forest, Mono County, California; Stairway Creek, Sierra National Forest, Madera County, California; and Sharktooth Creek, Sierra National Forest, Fresno County, California. More detailed location maps are available in the most recent 5-year review (USFWS 2013).

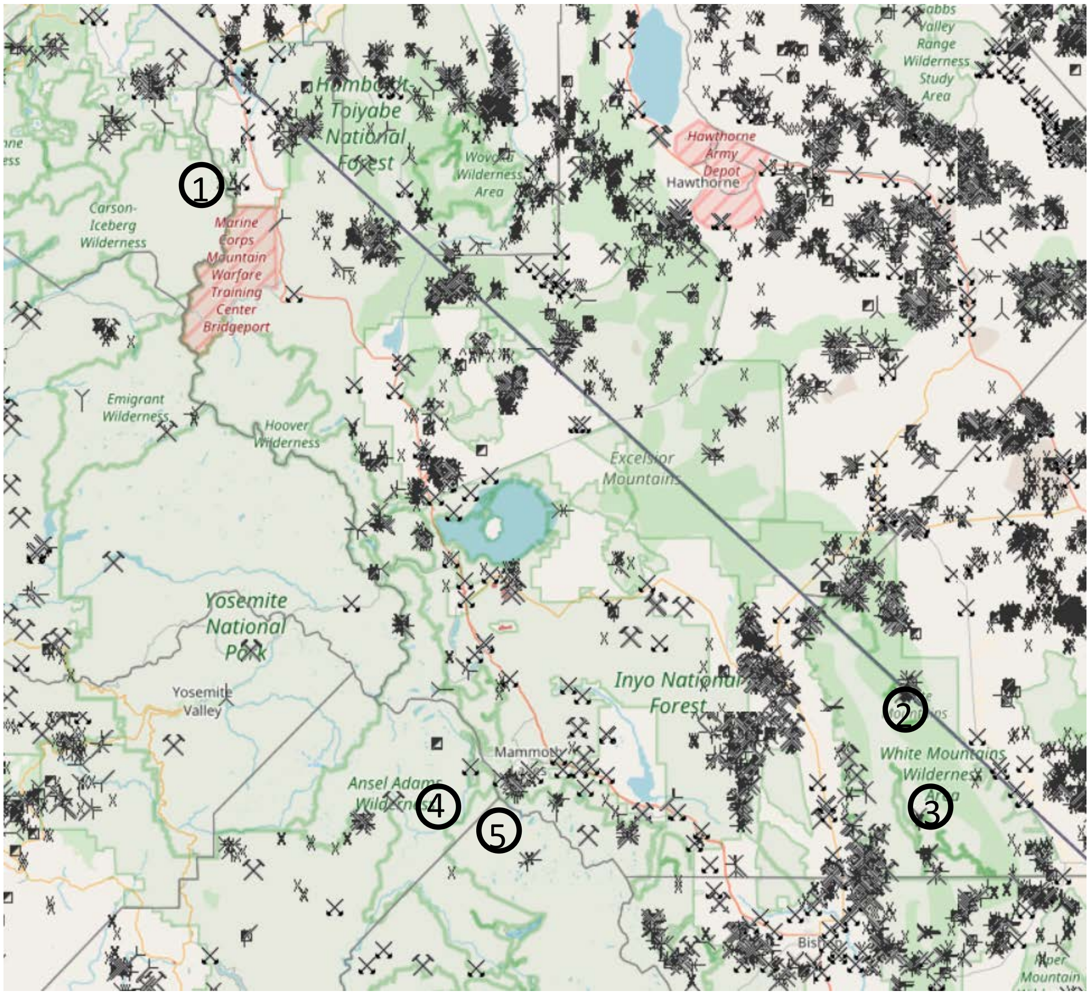
Size: Adult Paiute trout seldom reach sizes in excess of 250 mm total length but have reached 342 millimeters in Silver King Creek (USFWS 2004). Fry have grown to at least 35 mm by early fall. These lengths correlate to weights of **0.45 g** for fry and **179 g** for adults according to the length-weight regression equation shown in STTR Figure 2 for central California coastal steelhead.

Diet: Like other trout species in stream habitats the Paiute trout depends upon terrestrial and aquatic insects (Moyle 2002).

Food Ingestion Rate: There is no specific information about the food ingestion rate of Paiute trout. It is likely low due to the general low productivity within small, cold stream habitats and their slow growth rates. A detailed discussion of food ingestion rates for steelhead trout (*Oncorhynchus mykiss*), the anadromous form of rainbow trout, is provided above. Based on that discussion an appropriate food ingestion rate for the Paiute trout would be in the lower range of 0.8% to 1.0% (dry weight percent of live body weight per day). Using the size range above and 0.8% provides an ingestion rate range of **0.0036 – 1.4 g/d dw**.

General Life History: Paiute trout spawn from May through July (USFWS 2013). Spawning habitat is typical of stream-spawning salmonids requiring flowing waters with clean gravel substrates. They usually become sexually mature in year two depositing 250-400 eggs. Some have been known to spawn again (USFWS 2013); however, few Paiute trout live beyond the age of 3 years in the wild (USFWS 2004). Paiute trout generally do not travel far but have been documented moving over 1 km (USFWS 2013).

Risk of Selenium Exposure: As a species within the Oncorhynchus genus the general sensitivity of Paiute cutthroat trout to selenium are likely similar to rainbow trout. The discussions of selenium sensitivity of steelhead trout in this report apply to the Paiute cutthroat trout. Direct exposure to selenium, specifically for the Paiute cutthroat trout, is expected to be relatively low due to the lack of selenium sources in its range; however, some mining activity especially for copper and sulfur may be sources of selenium. For example, Henderson et al. (2013) documented elevated selenium levels below Leviathan Mine in the East Walker River watershed. Selenium levels ranged from <1 to 6.6 ug/L below the mine site in 2001. In 1999, Thompson and Welsh (2000) documented selenium levels in streams below Leviathan Mine that ranged from <1 to 13 ug/L. However; a review of the U.S. Geological Survey's Mineral Resources Data System (https://mrdata.usgs.gov/mrds/map-us.html) shows no known mining activity in the individual Paiute trout watersheds (PCTR Figure 2).



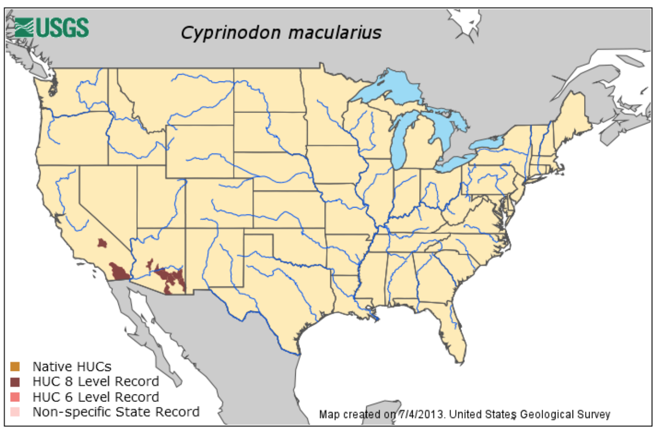
PCTR Figure 2. Location of existing Paiute cutthroat trout populations in the Sierra Nevada Mountain Range and known mining operations (https://mrdata.usgs.gov/mrds/map-us.html): 1) Silver King Creek; 2) Cabin Creek; 3) North Fork Cottonwood Creek; 4) Stairway Creek; and 5) Sharktooth Creek.

Desert Pupfish (DEPU)

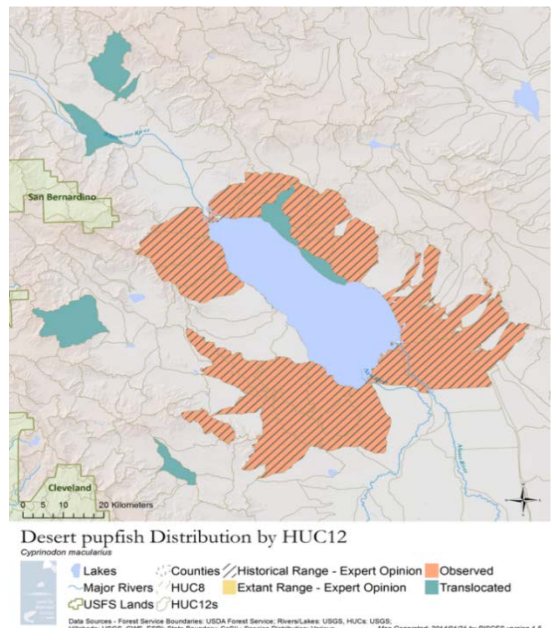
Cyprinodon macularius

Status: The desert pupfish is a federally listed endangered species 1986 (51 FR 10842). A 5-year review conducted by the USFWS in 2010 recommended no change to this designation (USFWS 2010c).

Distribution: Inhabit irrigation drains in the Imperial Valley agricultural area of Southern California. Historic distribution confined to shoreline and tributaries of the Salton Sea. Naturally occurring populations of desert pupfish are now restricted in the United States to two streams tributary to, and in shoreline pools and irrigation drains of, the Salton Sea in California (Lau and Boehm 1991).



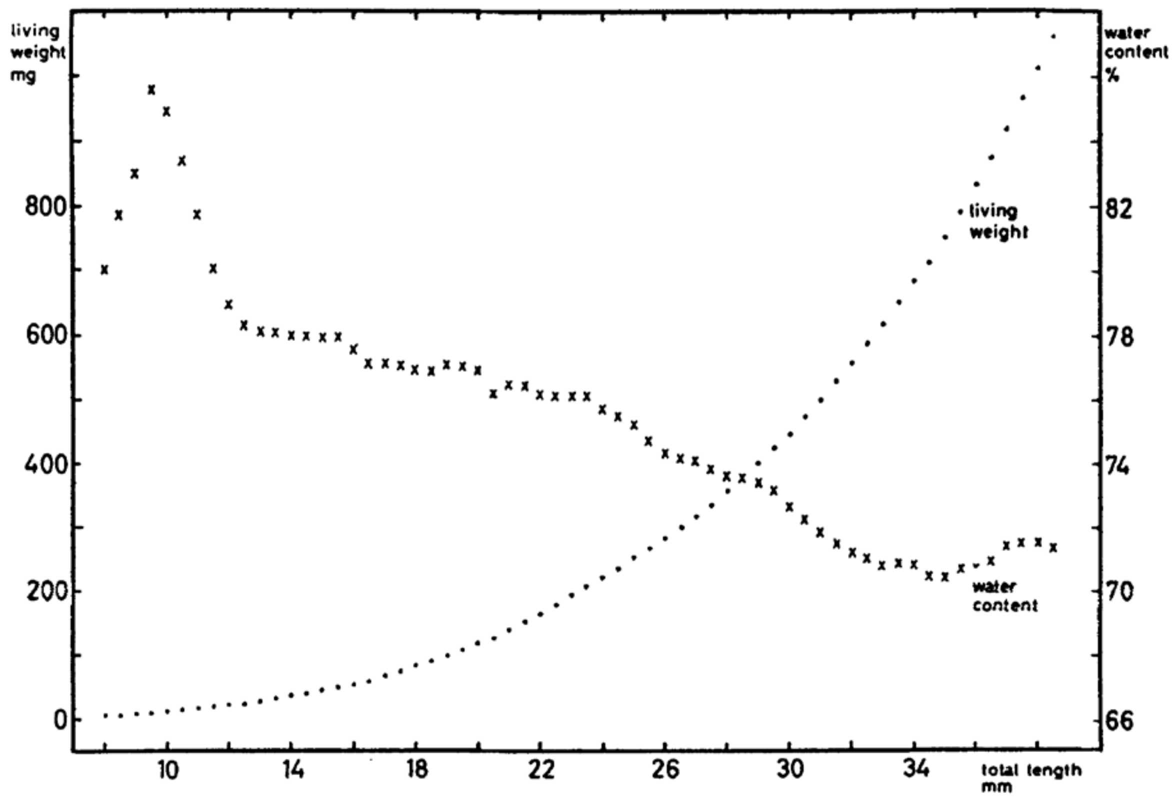
DEPU Figure 1. Distribution of desert pupfish in the United States, including transplanted and reintroduced populations.



DEPU Figure 2. Distribution of desert pupfish in California, from PISCES, University of California at Davis (Santos et al. 2014).

Size: Desert pupfish from the Salton Sea hatch at total length of 5.3 mm and may double in length within the first 8 weeks of life (Kinne 1960). Salton Sea adults commonly reach a length of 20 – 40 mm (Kinne 1960). Lengths of 25 to 30 mm (0.3 – 0.5 g) are attained in the laboratory by 26 weeks (Kinne 1960). Moyle (1976) found maximum lengths of up to 75 mm may be attained by the second summer. Pupfish growth rate is dependent upon age, habitat and environmental conditions, and population density. In the laboratory at the optimum temperature of 30^{o}C, pupfish reached 24 mm at 16 weeks by which time they will have reached maturity (Kinne 1960). This length translates to about **0.22 g** using the Kinne (1960) length-weight relationship (DEPU Figure 3).

Diet: Desert pupfish generally eat insect larvae, detritus, aquatic vegetation, snails, and occasionally their own eggs and young. They are known to feed heavily on mosquito and Chironomid midge larvae (Legner et al, 1975), and have been used to control these pest species.



DEPU Figure 3. Body weight (mg) and body water content (percent of body weight) by total body length of desert pupfish (Kinne, 1960)

The diet of the desert pupfish varies depending on availability, as well as seasonally. Naiman (1979) found stomach contents comprised of approximately half animal matter vs half algae in consecutive samplings.

Food Ingestion Rate: Kinne (1960) measured growth and food intake in desert pupfish, at a variety of temperatures and salinity levels. Fish were fed *Enchytraeus* white worms (avg 7.1 mg ww, or 1.48 mg dw). Pupfish (20-22 weeks old) on a regulated maintenance diet (maintained at 30^0C in 35ppt salinity) ate an average of 7 worms per day (Table 15 in Kinne 1960). This results in a food ingestion rate of approximately 0.05 g/d ww or **0.01 g/d dw**. Under an unrestricted diet the pupfish ate an average of 36.9 worms per day or **0.05 g/d dw** (Table 17 in Kinne 1960). Under colder temperatures the pupfish would eat significantly less.

General Life History: The desert pupfish tolerates an extreme range of environmental conditions: salinities from freshwater to nearly twice that of seawater, water temperatures ranging from 36°F to 113°F, and oxygen levels down to 0.1 parts per million (ppm). Desert pupfish can also survive rapid changes in salinity and daily water temperature fluctuations of 72°F to 80°F (Saiki et al, 2011). Sexual maturity is reached in 3 – 4 months (Crear and Haydock

1971, Kinne 1960). Life span in the wild appears highly variable, ranging from less than a year (Minckley 1973) to 3 years (Kynard and Garrett 1979).

Risk of selenium exposure: Desert pupfish are evidently particularly sensitive to toxic effects of selenium on their reproduction. A laboratory study by Besser et al. (2012) determined that desert pupfish egg production is reduced at selenium concentrations (7.3 µg/g dietary Se, 3.4 µg/g in wholebody dry-weight pupfish tissue, and 4.4 µg/g in pupfish eggs) well below reproductive toxicity thresholds published for other fish (Janz et al. 2010). This contrasts with relatively insensitivity of desert pupfish to selenium effects on growth and survival from larval through adult stages (Besser et al. 2012).

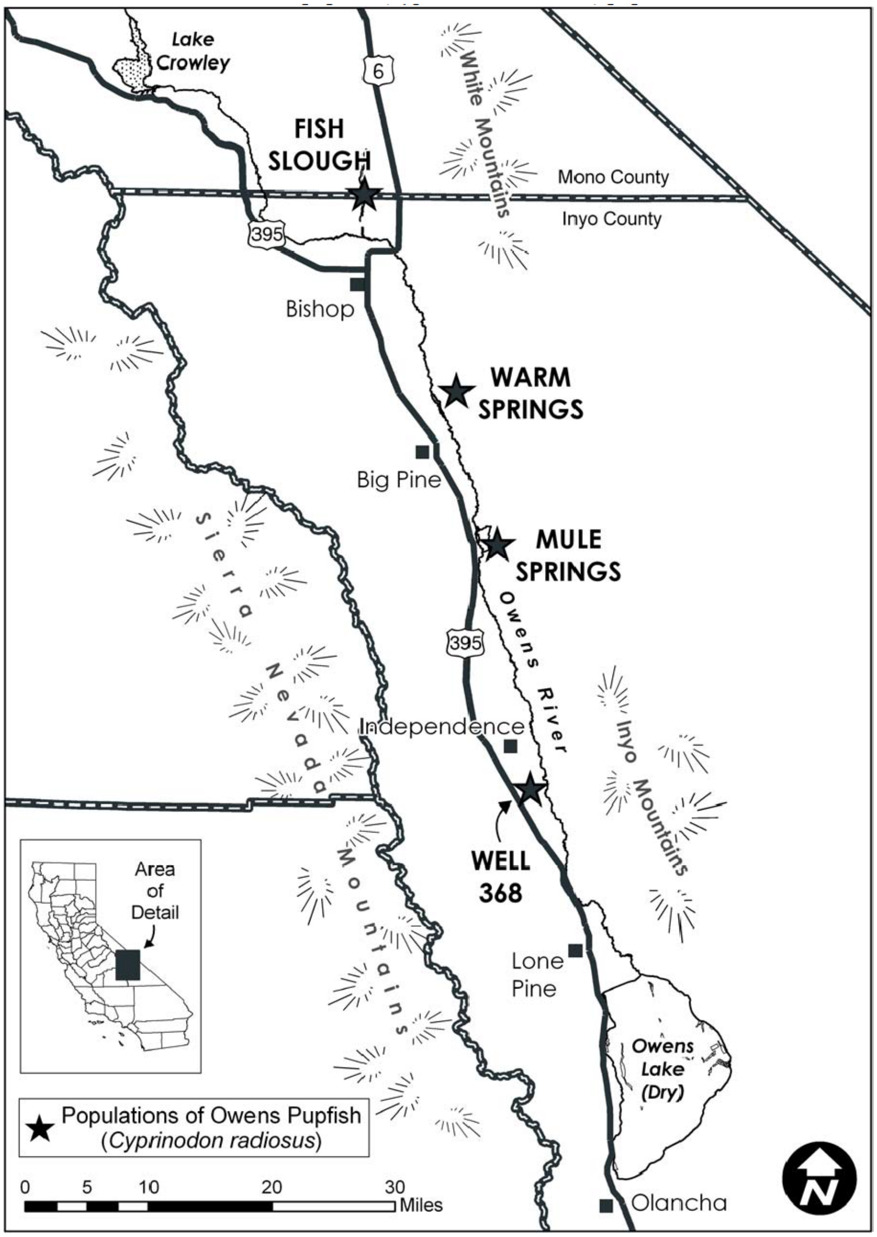
In the habitat of remaining natural populations of the desert pupfish in the Imperial Valley, dietary items of desert pupfish and the tissues of pupfish themselves frequently exceed toxicity thresholds associated with reduced egg production. Dietary items (midges) exceeded the toxicity threshold (7.3 µg/g) for desert pupfish egg production at two of seven sampled sites. Mean concentrations of selenium in pupfish (whole-body) exceeded the toxicity threshold (3.4 µg/g) in each of all seven sampled sites (Saiki et al. 2010; Saiki et al. 2012).

Considering the extremely restricted distributions of species and subspecies of pupfish in California, it seems likely that the principal threats to their survival relate to potential loss of remaining habitat. Nonetheless, the possibility of contaminant threats, including elevated selenium exposure cannot be discarded. If the critically important aquifers may be depleted by regional agricultural and urban groundwater pumping, then those aquifers may also be vulnerable to contamination.

Owens Pupfish (OWPU)
Cyprinodon radiosus

Status: The Owens pupfish was federally listed as endangered in 1967 (32 FR 4001), and state (CA) listed as endangered in 1971 (CDFW 2017).

Distribution: Historically the Owens pupfish occurred in the Owens River and spring pools, sloughs, irrigation ditches and swamps in the Owens Valley, (Mono and Inyo Counties). Currently this fish is confined to four populations in the Owens Valley (OWPU Figure 1; USFWS 2009e).



OWPU Figure 1. Distribution of Owens pupfish (USFWS 2009e).

Size: The Owens pupfish is a small, deep bodied fish with a total length that rarely exceeds 60 mm (USFWS 2009). They can reach at least 35 mm in their first growing season and breed before they reach 1 year old (Moyle 2002). Assuming they can breed in their first season like the desert pupfish, 35 mm is used for the critical length. Using DEPU Figure 6, the length-weight relationship for desert pupfish in Kinne (1960), provides an estimated weight of **0.76 g**.

Diet: Owens pupfish congregate in small schools and feed mostly on aquatic insects, detritus, and aquatic vegetation (USFWS 2009e).

Food Ingestion Rate: No feeding rate information for the Owens pupfish was available; however, Kinne (1960) measured growth and food intake in desert pupfish, at a variety of temperatures and salinity levels. A freshwater ingestion rate would be most appropriate for the Owens pupfish. Fish were fed *Enchytraeus* white worms (avg. 7.1 mg ww, or 1.48 mg dw). Juvenile (20-22 week old) desert pupfish on a regulated maintenance diet (maintained at 30^0C in freshwater) ate an average of 2.1 worms per day (Table 15 in Kinne 1960). This results in a food ingestion rate of approximately **0.003 g dw**. Under an unrestricted diet the pupfish ate an average of 14.6 worms per day or **0.02 g/d dw** (Table 17 in Kinne 1960). Under colder temperatures the pupfish eats significantly less.

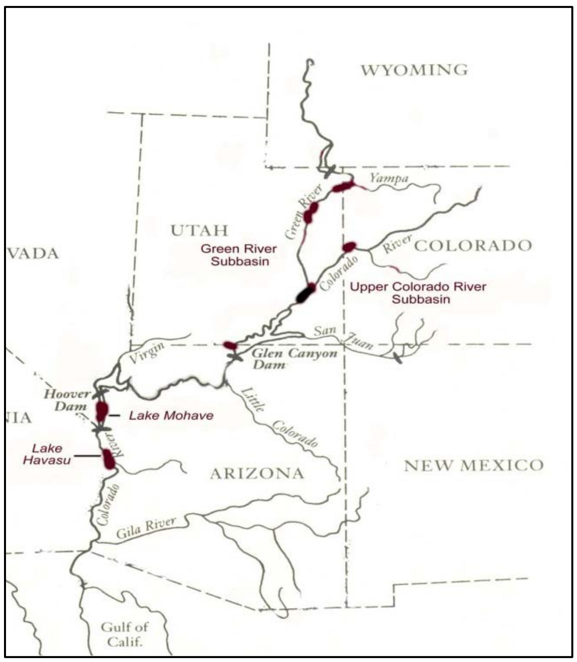
General Life History: The species is sexually dimorphic. Juvenile pupfish grow rapidly to sexual maturity in 3 to 4 months (Barlow 1961). They are usually able to spawn before their first winter, and lifespan is rarely greater than 1 year (Soltz and Naiman 1978), though Mire (1993) observed them living as long as 3 years in refuge habitats.

Risk of Selenium Exposure: In the Owens Valley, agricultural drainwater could possibly contaminate pupfish habitat. A risk analysis for contaminants in soil/sediment on Owens Lake rated the risk of selenium to birds and mammals as "probable impact." No analysis was done on risk to fish (Kleinfelder East Inc. 2007).

Bonytail Chub (BOCH)
Gila elegans

Status: Listed as endangered by USFWS in1980 (45 FR 27710). Listed as threatened by California in 1971 and as endangered in 1974 (CDFW 2017). No remaining wild population is self-sustaining and it is functionally extinct. Its survival currently relies on release of hatchery-produced fish (USFWS 2012a).

Distribution: This fish species experienced the most abrupt decline of any of the long-lived fishes native to the main-stems of the Colorado River system and, because no young individuals have been found in recent years, has been called functionally extinct. The chub was one of the first fish species to reflect the changes that occurred in the Colorado River basin after the construction of Hoover Dam; the fish was extirpated from the lower basin between 1926 and 1950 (USFWS 1994).



BOCH Figure 1. Recent distribution of wild bonytail chub in the Colorado River Basin (from USFWS 2002b)

Size: Adults of the species likely attain a maximum size of about 550 mm total length (TL; Bozek et al. 1984) and 1.1 kg in weight (Vanicek 1967). Bonytail reach maturity in 4 – 5 years at lengths of 300 – 400 mm (Moyle 2002). Vanicek (1967) aged 67 bonytail using scales, and found the largest to be a 7 year old adult at a length of 338 mm and weight of **422 g**.

Diet: Bonytail chub diet is comprised of a wide variety of aquatic and terrestrial insects, worms, algae, plankton, and plant debris (Marsh et al. 2013). Juvenile bonytail, <30mm, feed on aquatic invertebrates (Moyle 2002). Vanicek (1967) reported that adult *"Colorado chubs"* fed mainly on terrestrial insects (mostly adult beetles and grasshoppers), plant debris, leaves, stems, and woody fragments.

Food Ingestion Rate: Little is known about the feeding rate of the chub in the wild; however, Hansen et al. (2006) maintained bonytail chub in the lab, feeding at the rate of 0.5% – 1.5% body mass. For a 422 g bonytail at 1% this would translate to 4.2 g/d ww and at a 21% dry weight conversion for aquatic invertebrates (based on percent moisture of benthic invertebrate samples, USFWS, unpub.) provides a dry weight ingestion rate range of **0.9 g/d**.

General Life History: Little is known about the specific habitat requirements of the bonytail because the species was extirpated from most of its historic range prior to extensive fishery surveys. The bonytail chub is adapted to mainstem rivers where it has been observed in pools and eddies (USFWS 2002b). A study conducted by Pimentel and Bulkley (1983) suggest that bonytail chub, when given the opportunity, may tend to select water with high levels of total dissolved solids. They were able to persist in water with a total dissolved solids concentration of 4,700 milligrams per liter, the highest tolerance reported for any species of Colorado River *Gila*, suggesting an ability to persist despite anthropogenic water quality and habitat degradation (BOR 2016). The species is long-lived. Ulmer (1983) used otoliths to determine that two Lake Mohave bonytail were 32 and 39 years old, while Rinne (1976) estimated four Lake Mohave fish to be between 34 and 49 years old.

Risk of Selenium Exposure: The Lower Colorado River does not contain local sources of selenium; however, the Upper Colorado River Basin (Utah, Wyoming and Colorado) picks up selenium from the seleniferous soils via return flows of irrigation water and transports it to the Lower Colorado Basin. Selenium is concentrated in the water through evaporation, and then becomes deposited into the sediments and is accumulated by vegetation, invertebrates, and into fish (USFWS 2006a).

The majority of lower Colorado River mosquitofish samples contained selenium in excess of 3 µg/g dry weight (BOCH Table 1). In a 1991 study (Lusk 1993), 94% of whole fishes and invertebrates (n=185) collected contained concentrations of selenium that exceeded 3 µg/g dry weight selenium (see RYRA Table 1), with mosquitofish averaging between 4 and 5 µg/g dry weight selenium. Samples collected in 1999 (King et al. 2000) ranged between 4.15 and 16.6 µg/g.

See the discussion of selenium sensitivity of the razorback sucker for potential impacts to the bonytail chub.

BOCH Table 1. Selenium concentrations in mosquitofish collected from the lower Colorado River. (King et al. 2000 and Lusk 1993)

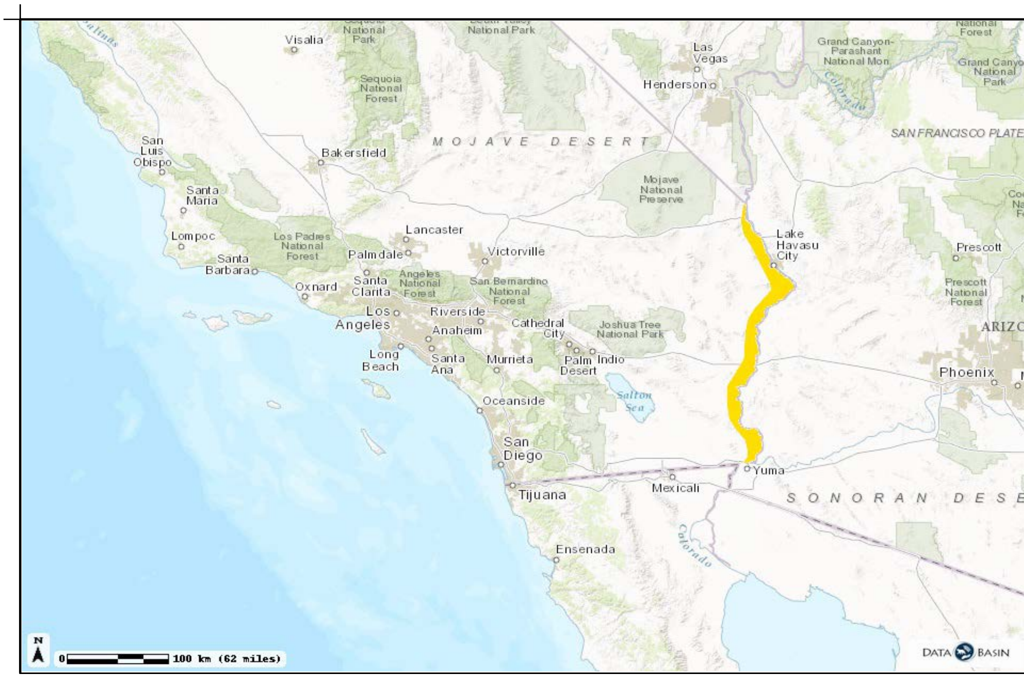
Sample	Area	Year	Selenium (µg/g dw) Mean (Range)	Study
Mosquitofish	McAllister Lake McAllister Lake	1991 1999	4.98 (3.94 - 6.36) 16.6 (NA)	Lusk (1993) King et al. 2000
	Butler Lake Butler Lake	1991 1999	3.97 (2.85 - 5.73) 4.15 (NA)	Lusk (1993) King et al. 2000

Razorback Sucker (RASU)

Xyrauchen texanus

Status: The razorback sucker was listed as endangered by USFWS in 1991 (56 FR 54957). It was listed as threatened by California in 1971 and as endangered in 1974 (CDFW 2017).

Distribution: The razorback sucker was once abundant throughout the Colorado River basin. It is near extinction in the in the lower Colorado River basin. In the lower basin, the only substantial population remaining is in Lake Mohave in Nevada, and suckers are rare in all other areas, with small numbers in Lake Havasu along the California/Nevada border (USFWS 2012b). At Lake Mohave the population is maintained by collecting larvae and growing them out for restocking as well as the capture and spawning of adults (USFWS 2012b).



RASU Figure 1. Razorback sucker distribution in California as of 2002 (from CBI 2011).

Size: LCRMCP (2008) provides a good summary of razorback sucker growth and size. At hatch they are 7–9 mm long and can reach lengths of over 23 mm in 2 months. In optimal pond conditions razorback suckers can grow 55–307 mm in 6 months. After 6 years growth becomes minimal (2 mm per year or less). Razorback suckers are typically found within the 400 - 700 mm TL range and weigh less than **3 kg** (LCRMCP 2008).

Diet: Razorbacks eat algae, planktonic crustaceans, aquatic insect larvae, plants, and detritus. The razorback sucker's diet composition is highly dependent upon life stage, habitat, and food availability. Larvae feed mainly on phytoplankton and small zooplankton while juveniles and adults feed on chironomids and other benthic insects (LCRMCP 2008, USFWS 1998).

Food Ingestion Rate: Little published information is available on the feeding rates or rearing methods of razorback sucker. A knowledge assessment by the Arizona Game and Fish Department (Ward et al. 2007) collected details from a variety of hatcheries, on maintenance of the sucker. Feeding rates varied from 1.0% – 7.0% body weight per day, with the higher rates reduced to 1% – 2% of body weight by the time they are approximately 300-mm TL size. At 3 kg and 1.5% this is 0.045 kg/d or 45 g/d ww. Based on a benthic invertebrate dry weight conversion of 21% (USFWS, unpub.) this translates to **9.5 g/d dw**.

General Life History: Razorbacks are long lived (> 40 years). Both males and females mature at age four. The majority of spawning is generally carried out from January through April (Mueller and Marsh 1998; Albrecht et al. 2013) when water temperatures are typically within the range of 10–15 °C (Bestgen 1990). Razorback suckers inhabit a diversity of areas from mainstream channels to backwaters of medium and large streams or rivers as well as artificial reservoirs (LCRMCP 2008, USFWS 1998). They prefer to live over sand, mud, or gravel bottoms.

Risk of Selenium Exposure: As noted in the discussion on the bonytail chub, above, the Colorado River in the Upper Basin (Utah, Wyoming and Colorado) picks up selenium from the seleniferous soils and transports it to the Lower Colorado River. Several studies have been conducted under the Recovery Implementation Program for endangered fish species in the Upper Colorado River (a partnership of private organizations, and state and federal agencies). Those studies investigating the interaction of razorback suckers and selenium included Hamilton et al. (2005a-d) and Beyers and Sodergren (2002). Hamilton exposed adult suckers for 9 months to various Colorado River waters (in selenium concentrations from 2.2 to 9.5 µg/L). Adults exposed to water from the most contaminated site accumulated body burdens of up to 16.6 µg/g selenium (Hamilton et al. 2005a). Eggs collected from these same experimental adults were found to contain from 6.0 to 46 µg/g selenium, with increased larval deformations found in the highest exposure groups (Hamilton et al. 2005b). Similarly, sucker larvae, maintained for 30 days in the same site waters (selenium from 0.9 to 10.7 µg/L), showed increased deformities and decreased survival from the same high selenium exposure sites (Hamilton et al. 2005c). Further, Hamilton et al. (2005d) demonstrated that even relatively low selenium concentrations in the food chain, in combination with lower selenium water concentrations resulted in low or no survival of larval suckers. Somewhat conflicting conclusions were reached by Beyers and Sodergren (2002) studies, which generally concluded selenium was not a problem for larval razorback sucker survival exposed to waters as high as 20.3 µg/L selenium. Hamilton (2005); however, noted several issues with the exposure regimen, laboratory practices, and subject age which likely influenced the findings in the Beyers and Sodergren studies.

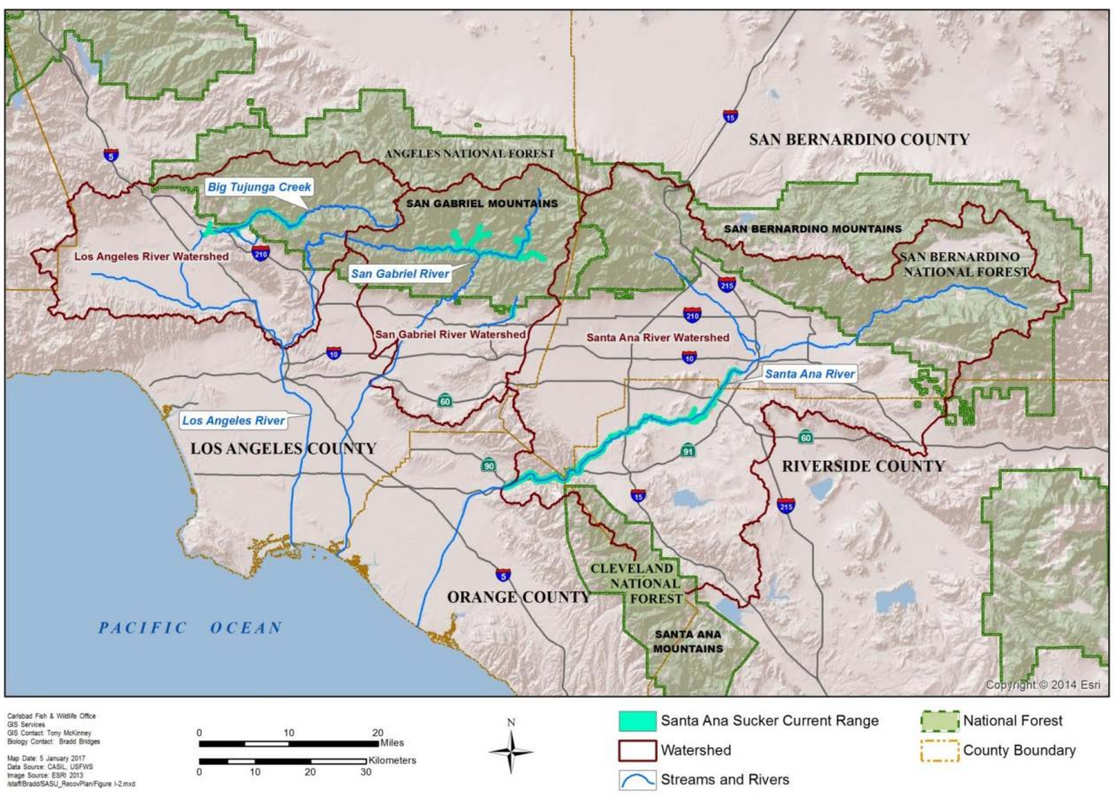
See the discussion above for the bonytail chub regarding selenium concentrations in potential razorback sucker prey items in the Lower Colorado River.

Santa Ana Sucker (SASU)

Catostomus santaanae

Status: The Santa Ana sucker was listed as threatened by USFWS in 2000 (65 FR 19686), and though not listed by the State of California, it is considered a species of special concern.

Distribution: The Santa Ana sucker is native to the Los Angeles and Santa Ana River basins in Southern California. Today it is restricted to three geographically separate populations in three different stream systems: the lower and middle Santa Ana River; east, west, and north forks of the San Gabriel River; and the lower Big Tujunga Creek. A population also occurs in the Santa Clara River (USFWS 2011b). Critical habitat includes three units within the Santa Ana River, the San Gabriel River, and Big Tujunga Creek (75 FR 77962).



SASU Figure 1. Current Range of the Santa Ana Sucker (from: USFWS 2017).

Size: Santa Ana suckers are generally less than 160 mm in length; however, some have been collected at lengths up to 200 mm (Russell 2010). Males and females appear to grow at equivalent rates. Growth studies in the Santa Clara River indicate that by the first year the suckers average 44 mm; by the second year (when they reach maturity), 77-110 mm; and by the third, 141-153 mm SL (Greenfield et al. 1970; Moyle 2002). Using an average length during the second year of 94 mm, this corresponds to a weight of **12.7 g**, using the regression for the Sacramento sucker in Kimmerer et al. (2005): $W = 0.0146\ L^{3.01}$, where W is weight in milligrams and L is fork length in mm.

Diet: Greenfield et al. (1970) found that 97% of the stomach contents of Santa Ana sucker consisted of detritus, algae, and diatoms; while aquatic insect larvae, fish scales, and fish eggs constituted 3%. They also found that larger fish usually had a higher percentage of insect material in their diets.

Food Ingestion Rate: Given similarities in diet, the feeding rate of juvenile razorback sucker would be appropriate. In hatchery conditions juvenile razorbacks are provided feed at a rate of 1% – 7% of body weight while the rate is reduced to 1% – 2% by the time they reach 300 mm in length (Ward et al. 2007). Greenfield et al. (1970) noted that Santa Ana suckers from ponds grew at faster rates than the river fish so it would seem an ingestion rate in the 1% – 2% range is most appropriate for the Santa Ana sucker. Using 1.5% and the average weight at maturity of 12.7 g provides an ingestion rate of 0.19 g/d ww. Based on a benthic invertebrate dry weight conversion of 21% (USFWS, unpub.) this translates to **0.04 g/d dw**.

General Life History: The only extensive study documenting the life history of the Santa Ana sucker is Greenfield et al. (1970). Santa Ana suckers produce from 4,423 to 16,151eggs with spawning beginning as early as late April and ending as late as July with peak spawning occurring from late May through June. Santa Ana suckers spawn at age 1 while few suckers survive to 3+ years of age. The high fecundity as well as the ability to spawn one year after hatching allows the sucker to quickly recover from population declines due to flooding events (Greenfield et al. 1970). Santa Ana sucker are most abundant in unpolluted, clear water, at temperatures that are typically less than 72 ºF (22 ºC) (Moyle 2002).

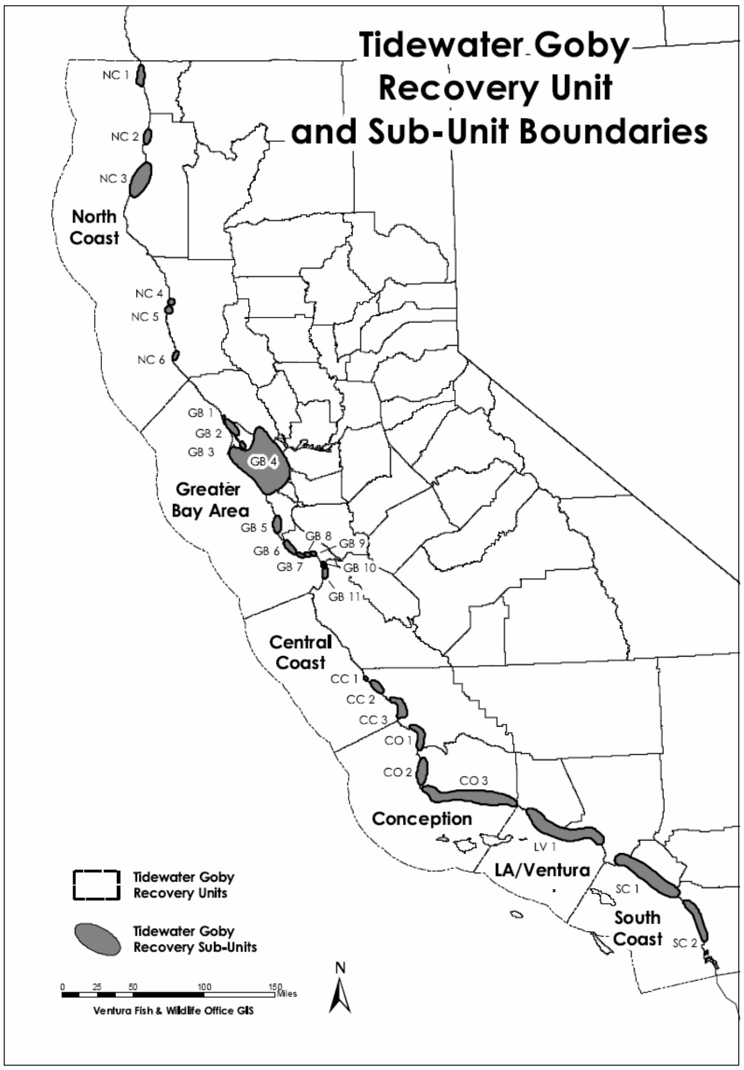
Risk of Selenium Exposure: Santa Ana sucker in the southern Coastal Range of California are likely to be exposed to selenium due to the extensive marine sedimentary geology of the range (See AMDI Figure 2). Southern coastal streams have shown elevated selenium levels well above 5 μg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). Areas where human activity can mobilize selenium such as mining, road work, and urban development can increase exposure potential for the fish.

Greenfield et al. (1970) observed a vertebral deformity in up to 3.47% of the Santa Ana sucker individuals in the monthly samples collected at two sites in the Santa Clara River from 1968 to 1969. They considered whether pesticide exposure and high temperatures could be possible causes; however, selenium was not considered. Selenium is known to cause spinal and other deformities in fish (Eisler 1985, Hamilton et al. 2005b, Stewart et al. 2004, USDOI 1998) and could be a possible cause of the deformities seen in the suckers. Greenfield et al. (1970) also noted these suckers were age 1+ and had survived recent flooding events. Although Greenfield et al. (1970) spawned some females in the lab to observe embryo development they did not note whether there were any deformities in that group. See the discussion for the razorback sucker for potential risks of selenium to the Santa Ana sucker.

Tidewater Goby (TIGO)
Eucyclogobius newberryi

Status: The tidewater goby was listed by the USFWS as endangered in 1974 (59 FR 5494) but was proposed for down-listing in 2014 (79 FR 14340). Critical Habitat was designated for the species in 2013 (78 FR 8745 8819). A recovery plan is in effect (USFWS 2005a).

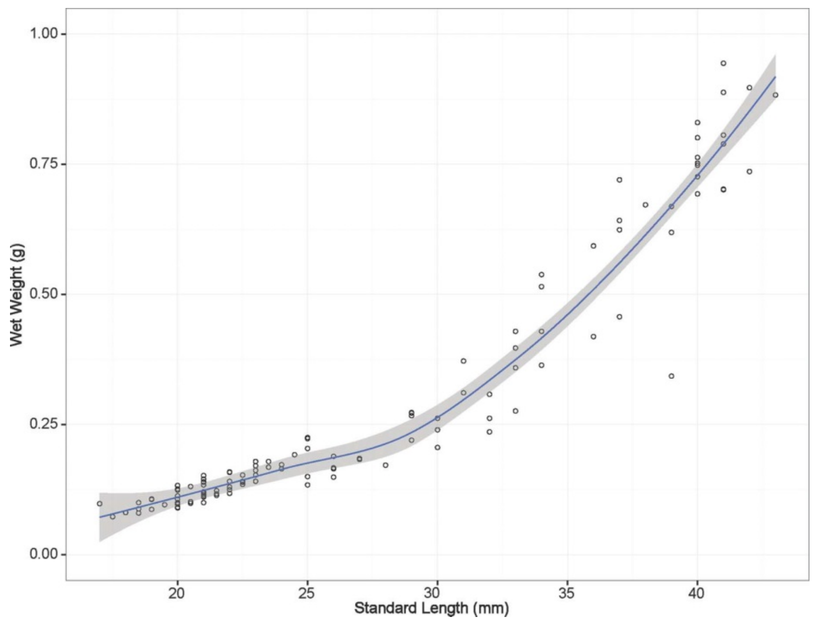
Distribution: Found primarily in waters of coastal lagoons, estuaries, and marshes, tidewater gobies historically ranged from mouth of the Smith River (Del Norte County) to Agua Hedionda Lagoon (San Diego County). They currently are found throughout their known, historic range, but reside at fewer locations than historically occurred (USFWS 2005a). The tidewater goby currently can be found in only about 96 of those historic locations, and only about 54 of those populations are thought to be secure at this time (TIGO Figure 1).



TIGO Figure 1. Tidewater goby range wide distribution with recovery unit (and sub-unit) boundaries (USFWS 2005a).

Size: The tidewater goby is a small, elongate, benthic fish that rarely exceeds 45 mm SL (TWGO Figure 2). Swift et al. (1989) found the goby varied seasonally and regionally, with the majority ranging from 25 mm to 35 mm. Weights are relative and variable, with adults rarely exceeding 1g. Using an average length of 30 mm and TWGO Figure 2 provides an average weight of **0.25 g**.

Diet: The diet consists mostly of small crustaceans (i.e., mysid shrimp, ostracods, amphipods), aquatic insects (i.e., chironomid larvae, diptera larvae), and molluscs (Wang 1982, Irwin and Soltz 1984, Swift et al. 1989). Inorganic material consistently found in the guts indicates a benthic foraging mode. Swenson and McCray (1996) found that overall, diet variability was low, but that diet would vary across seasons (April, August and November) and habitats (lagoon, creek and marsh).



TWGO Figure 2. Standard length (mm) and wet weight (g) of Tidewater Goby surveyed at Salmon Creek Lagoon, California. (Chase and Todgham 2016)

Food Ingestion Rate: No species specific information was found regarding feeding rates. Juvenile goby (~0.1 g) maintained in a laboratory setting were satiated with approximately 10 *Daphnia pulex* daily (Chase et al. 2016). Using an approximate average *Daphnia* dry weight of 7.5 µg/individual (Jeppesen et al. 2004) results in a feeding rate of 75 µg/d or 0.075 g/d dw. This value seems high in that it represents at least a 75% bw/d ingestion rate (up to 300% based on ww). It could be high for at least two reasons. 1) Chase et al. (2016) maintained their own *Daphnia pulex* cultures, which are generally a smaller species, and it is possible they were not feeding fully grown *D. pulex* to the goby. 2) The size range of *Daphnia* in Jeppesen et al. (2004) is wide (<1 µg - 54 µg) and they do not specify which species (may be more than one) were collected. Also, since the diet for Chase et al. (2016) was for juvenile goby it is reasonable to assume that an adult would have a lower feeding rate. Therefore, we use an ingestion rate of 2%

bw/d that is within the range for similar small sized adult fish discussed previously. Using the 0.25 g adult weight and 2% bw/d ingestion rate results in 0.005 g/d ww. Based on a benthic invertebrate dry weight conversion of 21% (USFWS, unpub.) this translates to **0.001 g/d dw**.

General Life History: The tidewater goby is a benthic species that inhabits shallow lagoons and the lower reaches of coastal streams, and is almost unique among fishes along the U.S. Pacific coast in its restriction to low-salinity waters in California's coastal wetlands. It differs from other species of gobies in California in that it is able to complete its entire life cycle in fresh or brackish water (Wang 1982, Irwin and Soltz 1984, Swift et al. 1989).

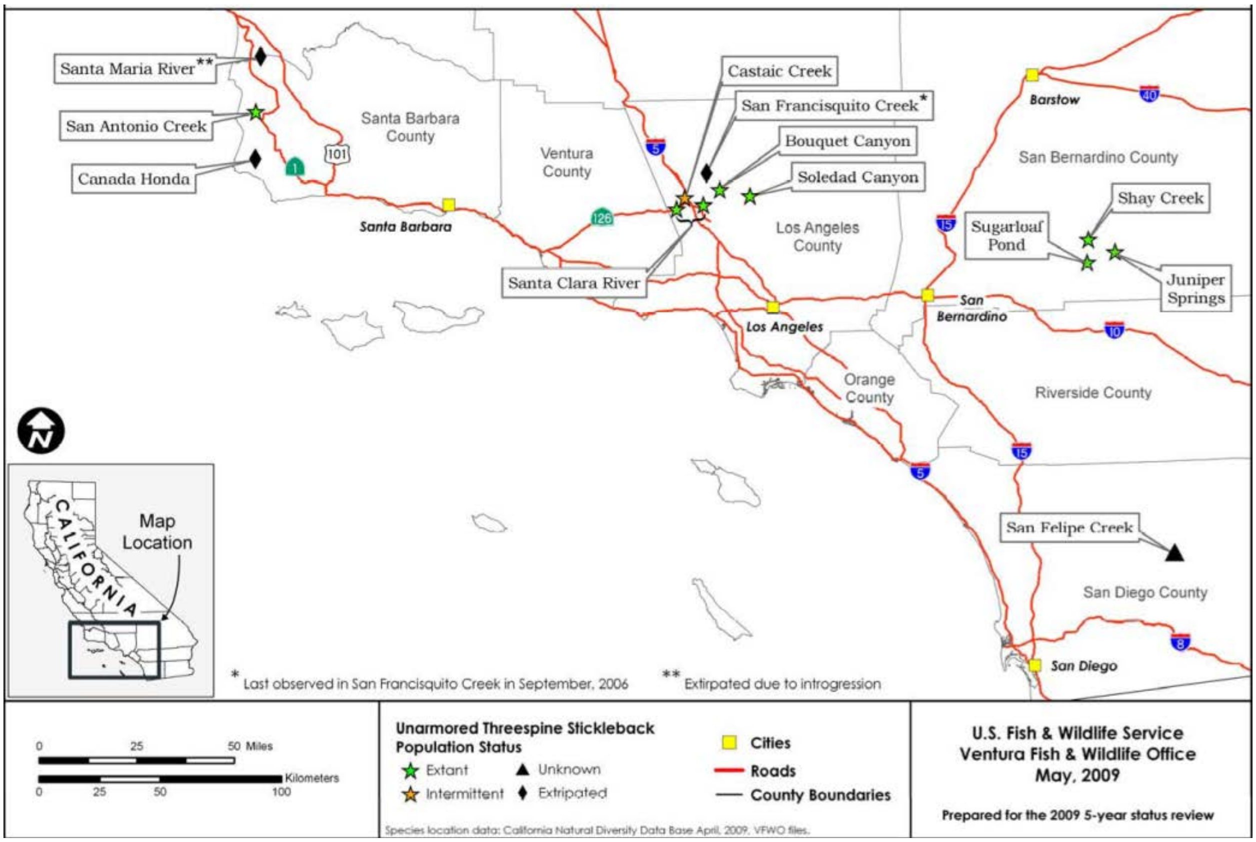
Tidewater gobies generally live for only 1 year, with few individuals living longer than a year (Moyle 2002). Reproduction occurs at all times of the year, as indicated by female tidewater gobies in various stages of ovarian development (Swenson 1999). The peak of spawning activity occurs during the spring and then again in the late-summer.

Risk of Selenium Exposure: Tidewater goby in the central and southern Coastal Range of California are likely to be exposed to selenium due to the extensive marine sedimentary geology of the range (See AMDI Figure 2). Southern coastal streams have shown elevated selenium levels well above 5 μg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). Areas where human activity can mobilize selenium such as mining, road work, and urban development can increase exposure potential for the fish. Selenium levels protective of other fish species will likely protect the goby.

Unarmored Threespine Stickleback (UTST)
Gasterosteus aculeatus williamsoni

Status: The unarmored threespine stickleback (UTST) was listed as endangered by USFWS in 1970 (35 FR 16047), and endangered in the State of California in 1971.

Distribution: The UTST is currently restricted to the upper Santa Clara River drainage in Los Angeles and Ventura Counties, San Antonio Creek on Vandenburg Air Force Base, San Luis Obispo County, and an isolated population in San Felipe Creek in San Diego County. The San Felipe Creek population is both introduced and outside the historic range of this species. A remnant population of stickleback exists in Shay Creek and Sugarloaf Pond, San Bernardino County. (USFWS 2009c).



UTST Figure 1. Distribution of unarmored threespine stickleback since listing (USFWS 2009c).

Size: Small, streamlined fish, generally not greater than 60 mm, average of 50 mm (Spilseth and Simenstad 2011, USFWS 2009c). This corresponds to a weight of **1.3 g**, using the regression for the threespine stickleback in Kimmerer et al. (2005): $W = 0.0086\ L^{3.04}$, where W is weight in milligrams and L is fork length in mm.

Diet: The UTST is a generalist carnivore feeding on benthic invertebrates and epibenthic prey such as copepods, ostracods, and epibenthic crustacea (Spilseth and Simenstad 2011). They may also eat snails, flatworms, nematodes, and terrestrial insects (USFWS 2009c).

Food Ingestion Rate: No specific ingestion rate data was found for the UTST; however, Spilseth and Simenstad (2011) determined feeding rates for the threespine stickleback (*Gasterosteus aculeatus*) in the Columbia River Estuary. Feeding rates varied from 0.09 – 0.18 g/d ww in the 40 – 55 mm size range. Based on a benthic invertebrate dry weight conversion of 21% (USFWS, unpub.) this translates to an ingestion rate of 0.019 - 0.039 g/d dw and a mid-range value of **0.03 g/d dw**.

General Life History: UTST are found in shallow pools and riffles with abundant cover including aquatic vegetation. UTST live only one year but breed year round with most breeding occurring from February to September (USFWS 2009c). Miller and Hubbs (1969) reported the stocking of armored threespine stickleback (*G. a. microcephalus*) into the native range of the UTST in certain California drainages. One consequence has been extensive hybridization between the two subspecies.

Risk of Selenium Exposure: Threespine sticklebacks in the southern Coastal Range of California are likely to be exposed to selenium due to the extensive marine sedimentary geology of the range (See AMDI Figure 2). Southern coastal streams have shown elevated selenium levels well above 5 μg/L (CEDEN, http://ceden.waterboards.ca.gov/AdvancedQueryTool). Areas where human activity can mobilize selenium such as mining, road work, and urban development can increase exposure potential for the fish.

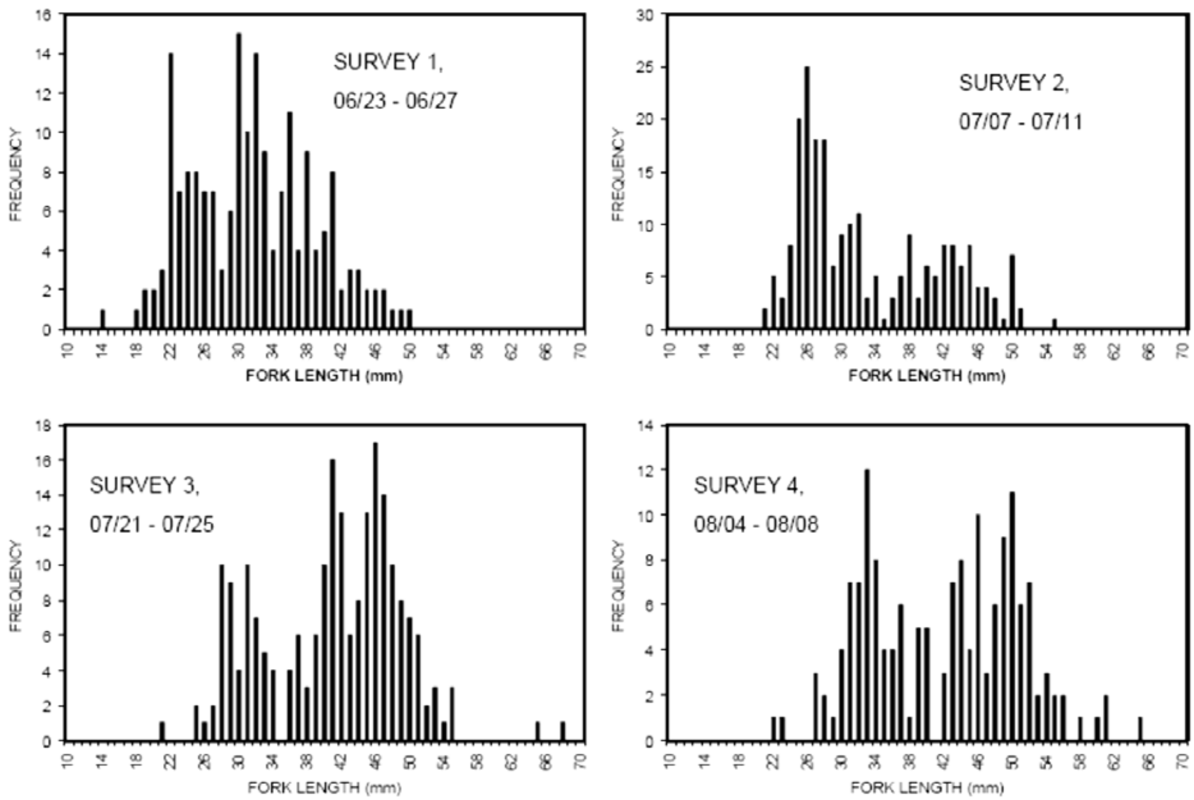
Delta Smelt (DESM)
Hypomesus transpacificus

Status: Delta smelt were federally listed as a threatened species in 1993 (58 FR 12854). In recent years the population of delta smelt has dropped to the lowest levels on record placing the species at increased risk of extinction. In 2010 the Service found that the delta smelt should be reclassified as endangered, but was precluded by other higher priority actions (75 FR 17667). The State of California listed the delta smelt as threatened in 1993 and as endangered in 2010 (CDFW 2017).

Distribution: Delta smelt are endemic to the upper Sacramento-San Joaquin River estuary (DESM Figure 1). 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 (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. 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, 2002; and Wang 1991).

DESM Figure 1. Delta smelt critical habitat.

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 (USFWS 1996). The median fork length of 113 delta smelt was about 62 mm (Kimmerer et al. 2005). This corresponds to an **adult weight of 2.1 g**, using the regression in Kimmerer et al. (2005): $W = 0.0018\ L^{3.38}$, where W is weight in milligrams and L is fork length in mm. In tow-net surveys in June-August 2000 (DESM Figure 2) 785 delta smelt young-of-the-year had an average length of 35.7 mm (Gartz 2001) corresponding to an average **juvenile weight of 0.32 g**, using the above regression.



DESM Figure 2. 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, Moyle 2002).

Food Ingestion Rate: The 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 $W = 0.0018\ L^{3.38}$ where W is weight in milligrams 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/d (dw)** or 11.4% of body weight per day for a juvenile delta smelt weighing 0.32 g. In a laboratory settings where adult Delta smelt are bred, a commercial diet of 1% - 3% bw was provided (Hung et al. 2014, LaCava et al. 2015). Applying the 3% to an adult smelt at 2.1 g gives an ingestion rate of 0.063 g/d ww. Assuming the commercial diet had no more that 10% moisture results in **0.057 g/d dw.**

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 without 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 2 meters of the water column and usually not in close association with the shoreline.

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 (Wang 1991, Sweetnam 1999). Spawning has also been recorded in Montezuma Slough near Suisun Bay (Wang 1986). 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). Delta smelt are broadcast spawners (DWR and USDI 1994) and eggs sink to the bottom and stick to hard substrates such as rock, gravel, tree roots, and submerged vegetation (Moyle 1976, 2002; Wang 1986). Growth of newly-hatched delta smelt is rapid. Juvenile fish are 40-50 mm long by early August (Erkkila et al. 1950; Ganssle 1966; Radtke 1966). Delta smelt reach 55-70 mm standard length in 7-9 months (Moyle 1976, 2002). Growth during the next 3 months slows down considerably because most of the energy ingested is being directed towards gonadal development (Erkkila et al. 1950; Radtke 1966). Females 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). Some fraction of the unspawned population may 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.

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 indicates that factors affecting the number 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 (Lemly 2002), 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. However, selenium loads to the estuary have decreased significantly since 1990 (Baginska 2015, SFEI 2015).

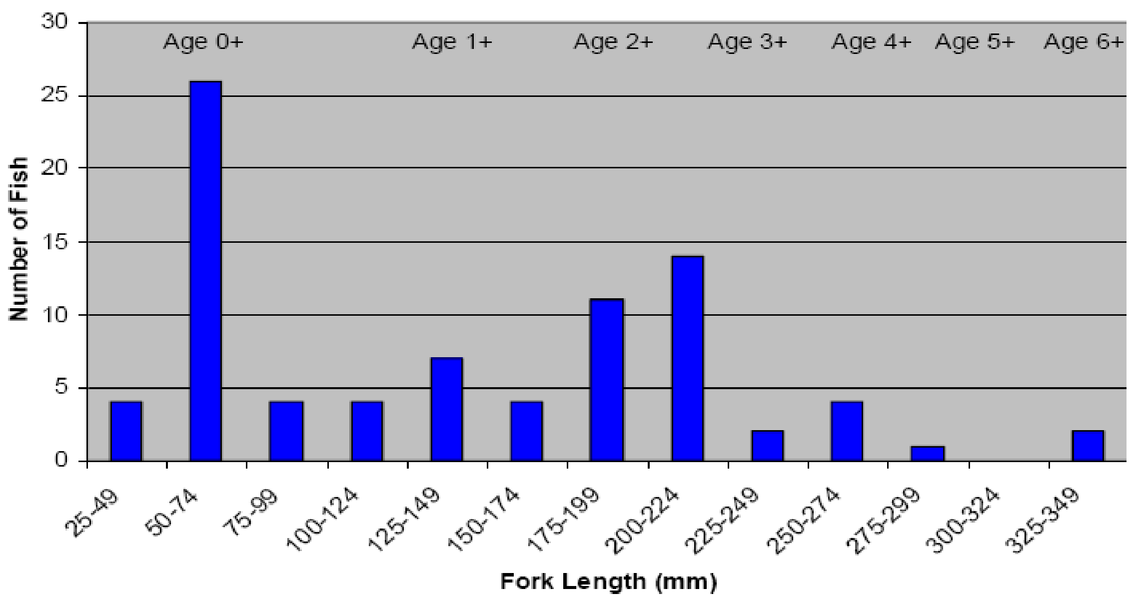
Delta smelt spawning sites are almost entirely restricted to the north-Delta channels associated with the selenium-normal Sacramento River and are nearly absent from the south-delta channels associated with the selenium-contaminated San Joaquin River (USFWS 1996). Delta smelt (n=41) salvaged from CDFW annual abundance surveys primarily around Chipps Island in 1993 and 1994 (well before significant selenium load reductions to the estuary occurred) 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). Delta smelt are not within the estuarine benthic food web pathway that is most problematic for selenium exposure (Stewart et al. 2004).

Sacramento Splittail (SASP)
Pogonichthys macrolepidotus

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

Distribution: Sacramento splittail are endemic to 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 (Moyle 2002, Moyle et al. 2004).

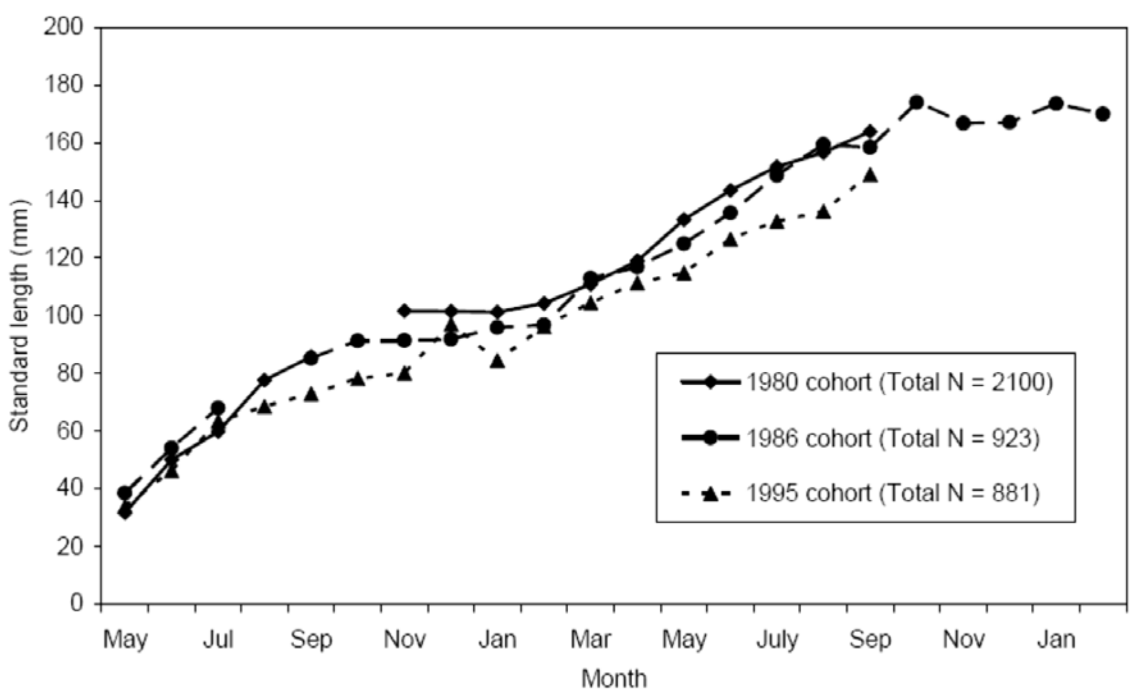
Size: The average fork length of 14 adult Sacramento splittail captured in a fyke trap on their upstream migration in the Yolo Bypass Toe Drain in February 2001 was 320 mm with a standard deviation of 31 mm (Sommer et al. 2002). Fork lengths of 83 juvenile and adult splittail collected in newly-restored 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) (SASP Figure 1). The center of this modal range is 212 mm. Using the length-weight regression for splittail of $W = 0.000003\ L^{3.27}$, where W is weight in grams and L is fork length in mm (Kimmerer et al. 2005), an adult Sacramento splittail with a fork length of 212 mm would be expected to have a weight of **121 g**. A juvenile splittail at 25 mm would weigh **0.11 g**.



SASP Figure1. 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 (*Potamocorbula* *amurensis*) and other mollusks constituted 34% of the diet (Feyrer and Matern 2000, Feyrer et al. 2003).

Food Ingestion Rate: Acuna et al. (2012) fed juvenile splittail (avg. 12.5 cm, 12.6 g) a diet of 2% body weight as dry weight for experimental purposes. For a 12 g splittail this would be about 0.25 g/d dw. The 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 2,100 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 (SASP Figure 2) (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 for splittail 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 weight per day for splittail with a fork length of 154 mm weighing 42.6 g. Using an adult weight of 121 g at 3.37% of bw yields **4.1 g/d dw**. Assuming a juvenile ingestion rate of at least twice the adult rate, a rate similar to those noted for other juvenile fish in discussions above, results in an ingestion rate of **0.007 g/d dw**.



SAST Figure 2. 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 7-9 years) and are highly fecund producing up to 100,000 eggs per female (Moyle 2002, Moyle et al. 2004). 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, Moyle et al. 2004), 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% 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 where adults are captured most frequently in January through April (Moyle 2002) 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 for 10 to 14 days (Moyle 2002) and move into the deeper offshore habitat as they mature (Wang 1986). Juveniles will leave floodplain habitats when they are in the 25 – 40 mm length range (Moyle et al. 2004).

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% 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 which are efficient selenium bioaccumulators (Stewart et al. 2004). Teh et al. (2004) found that juvenile splittail are adversely affected (liver histopathology and deformity) by chronic exposure (9 months) to a diet of 6.6 µg/g selenium. A reanalysis of the Teh et al. (2004) data derived selenium EC_{10} values of 0.9 µg/g dw in feed, 7.9 µg/g dw in muscle, and 18.6 µg/g dw in liver for juvenile splittail (Rigby et al. 2010).

Deformities typical of selenium exposure have been seen in splittail collected from Suisun Bay (Stewart et al. 2004). Juvenile splittail have been caught at San Joaquin River tributary monitoring sites for the Grassland Bypass Project after above normal wet years. Whole body selenium concentrations in juvenile splittail from Mud Slough where subsurface irrigation drainwater has been discharged ranged from 4 to 10 µg/g dw while splittail from Salt Slough that is free of selenium contamination ranged from 3 to 4 µg/g dw. (Beckon et al. 2013).

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Appendix A

Figure A1. Ecoregions of the Pacific Southwest Region of the U.S. Fish and Wildlife Service. Based upon Bailey (1995).

Table A1. Ecosystems in which each species evaluated are found. Shaded species are those selected for further review.

		Ecoregion						
Common Name	Scientific Name	1	2	3	4	5	6	7
Mammals								
San Joaquin Kit Fox	*Vulpes macrotis mutica*		*X*	*X*	*X*			
Buena Vista Lake Ornate Shrew	*Sorex ornatus relictus*			*X*				
Kangaroo Rats	*Dipodomys sp.*		*X*	*X*				X
California Sea Otter	*Enhydra lutris nereis*		*X*					
Birds								
American Dipper	*Cinclus mexicanus*	*B*	*B*		*B*	*B*		B
American White Pelican	*Pelecanus erythrorhynchos*	*B*	*X*	*X*	*B*	*B*	X	X
California Brown Pelican	*Pelecanus occidentalis californicus*		*B*				X	B
White-faced Ibis	*Plegadis chihi*	*B*	*X*	*B*	*B*	*B*	B	B
Double-crested Cormorant	*Phalacrocorax auritus*	*B*	*X*	*X*	*X*	*B*	B	B
American Bittern	*Botaurus lentiginosus*	*B*	*B*	*B*	*X*	*B*	X	X
Western Least Bittern	*Ixobrychus exilis hesperis*	*B*		*B*			B	B
Aleutian Canada Goose	*Branta canadensis leucopareia*		*X*	*X*				
Greater Scaup	*Aythya marila*	*X*	*X*	*X*	*X*		X	X
Lesser Scaup	*Aythya affinis*	*B*	*X*	*X*	*X*	*X*	X	X
Black Scoter	*Melanitta nigra*		*X*					
White-winged Scoter	*Melanitta fusca*		*X*					
Surf Scoter	*Melanitta perspicillata*		*X*					
Osprey	*Pandion haliaetus*	*B*	*B*	*B*	*B*	*B*	X	X
Bald Eagle	*Haliaeetus leucocephalus*	*B*	*B*	*B*	*B*	*B*	X	B
American Peregrine Falcon	*Falco peregrinus anatum*	*B*	*B*	*B*	*B*	*B*	B	B
California Black Rail	*Laterallus jamaicensis coturniculus*		*B*				B	

Common Name	Scientific Name	Ecoregion 1	2	3	4	5	6	7
Ridgeway's Yuma Rail	*Rallus obsoletus yumanensis*						B	
Ridgeway's Light-footed Rail	*Rallus obsoletus levipes*							B
Ridgeway's Rail	*Rallus obsoletus obsoletus*		*B*					
Marbled Murrelet	*Brachyramphus marmoratus*		*B*					
California Least Tern	*Sternula antillarum browni*		*B*	*X*				B
Black Tern	*Chlidonias niger*	*X*	*X*	*B*	*X*	*X*	X	X
Caspian Tern	*Sterna caspia*	*B*	*B*	*X*	*B*	*B*	X	X
Western Snowy Plover (Pacific)	*Charadrius nivosus nivosus*		*B*	*X*	*X*		X	B
Mountain Plover	*Charadrius montanus*			*X*			X	X
Tricolored Blackbird	*Agelaius tricolor*	*B*	*B*	*B*	*B*	*B*	B	B
Reptiles								
Giant Garter Snake	*Thamnophis gigas*			*X*				
Blunt-nosed Leopard Lizard	*Gambelia sila*		*X*	*X*				
Fish								
Chinook Salmon	*Oncorhynchus tshawytscha*	*X*	*X*	*X*	*X*			
Steelhead	*Oncorhynchus mykiss*	*X*	*X*	*X*	*X*			X
Coho Salmon	*Oncorhynchus (=salmo) kisutch*	*X*	*X*					
Little Kern Golden Trout	*Oncorhynchus aguabonita whitei*				*X*			
Lahontan Cutthroat Trout	*Oncorhynchus clarkii henshawi*				*X*	*X*		
Paiute Cutthroat Trout	*Oncorhynchus clarkii seleniris*				*X*	*X*		
Cui-Ui	*Chasmistes cujus*					*X*		
Desert Pupfish	*Cyprinodon macularius*						X	
Owens Pupfish	*Cyprinodon radiosus*						X	

		Ecoregion						
Common Name	Scientific Name	1	2	3	4	5	6	7
Mohave Tui Chub	*Gila bicolor ssp. mohavensis*						X	
Owens Tui Chub	*Gila bicolor ssp. snyderi*					*X*	X	
Bonytail Chub	*Gila elegans*						X	
Razorback Sucker	*Xyrauchen texanus*						X	
Santa Ana Sucker	*Catostomus santaanae*							X
Warner Sucker	*Catostomus warnerensis*					*X*		
Shortnose Sucker	*Chasmistes brevirostris*	*X*						
Lost River Sucker	*Deltistes luxatus*	*X*						
Tidewater Goby	*Eucyclogobius newberryi*		*X*					X
Unarmored Threespine Stickleback	*Gasterosteus aculeatus williamsoni*							X
Delta Smelt	*Hypomesus transpacificus*		*X*	*X*				
Longfin Smelt	*Spirinchus thaleichthys*		*X*	*X*				
Sacramento Perch	*Archoplites interruptus*		*X*	*X*				
Sacramento Splittail	*Pogonichthys macrolepidotus*		*X*	*X*				
Amphibians								
California Red-legged Frog	*Rana draytonii*		*X*	*X*	*X*			X
Mountain Yellow-legged Frog	*Rana muscosa*				*X*			X
Sierra Nevada Yellow-legged Frog	*Rana sierrae*				*X*			
Yosemite Toad	*Anaxyrus canorus*				*X*			
Arroyo Toad	*Anaxyrus californicus*							X
Oregon Spotted Frog	*Rana pretiosa*	*X*						
Santa Cruz Long-toed Salamander	*Ambystoma macrodactylum croceum*		*X*					

Common Name	Scientific Name	Ecoregion 1	2	3	4	5	6	7
Desert Slender Salamander	*Batrachoseps major aridus*						X	X
California Tiger Salamander	*Ambystoma californiense*		*X*	*X*				
Invertebrates								
Conservancy Fairy Shrimp	*Branchinecta conservatio*			*X*				X
Longhorn Fairy Shrimp	*Branchinecta longiantenna*		*X*	*X*				
Riverside Fairy Shrimp	*Streptocephalus woottoni*							X
San Diego Fairy Shrimp	*Branchinecta sandiegonensis*							X
Vernal Pool Fairy Shrimp	*Branchinecta lynchi*		*X*	*X*				X
Vernal Pool Tadpole Shrimp	*Lepidurus packardi*			*X*				
California Freshwater Shrimp	*Syncaris pacifica*		*X*					
Shasta Crayfish	*Pacifastacus fortis*			*X*	*X*			
Species Evaluated		20	41	35	25	17	25	32
Species Selected for Review		4	12	9	8	4	8	11

For Avian Species:

X-non-breeding

B-breeding

<u>B</u>-primary breeding if isolated breeding in other ecosystems

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