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Potential Effects Of Selenium Contamination On Federally-Listed Species Resulting From Delivery Of Federal Water To The San Luis Unit

U.S. Fish and Wildlife Service
Sacramento Fish and Wildlife Office
Environmental Contaminants Division

Artwork by Mirriam Morrill

For the U. S. Bureau of Reclamation
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Introduction

Federal water delivered to the San Luis Unit (the Project) is used principally for irrigated agriculture. Due to a nearly-impervious soil layer, irrigated agriculture in this area is unsustainable without subsurface drainage to keep the water table below the root zone of crops and to ameliorate the accumulation of salts in the soil. Therefore, an analysis of the effects of the delivery of federal water must include the effects of subsurface drainwater that may seep, be conveyed, or be carried by floodwaters downstream into sloughs and rivers and thence into the San Francisco Bay/Delta estuary.

Within the direct footprint of the project, consideration must be given to the effects of conveying and storing drainwater, as well as applying drainwater to irrigate salt-tolerant plants in reuse areas, and evaporating drainwater in evaporation ponds or solar evaporators. These are likely to be components of any long-term continuation of irrigated agriculture in the San Luis Unit. In this area, the subsurface drainage of irrigated lands mobilizes selenium that has been historically sequestered in the soil. Selenium concentrations in agricultural drainwater from this area reach levels that, when bioaccumulated through food chains, cause adverse effects on aquatic and aquatic-dependent wildlife. Where such drainwater is applied to uplands, as in reuse areas, strictly terrestrial wildlife may be impacted as well.

Downstream from the San Luis Unit, any drainwater from the Project area is diluted by relatively low-selenium water from rivers that drain the Sierra Nevada Mountains. However, as the San Joaquin River reaches the San Francisco Bay/Delta estuary, flow velocities decrease and salinity increases. In these slow-moving, saline waters, with abundant introduced filter-feeding invertebrates, ecosystems have developed that evidently are much more effective than riverine ecosystems at bioconcentrating water-borne selenium. Therefore, potential downstream effects must be considered.

Although selenium is the principle contaminant of concern in drainwater from this area, mercury in the soil may be similarly mobilized and bioconcentrated to toxic concentrations in food chains. However, less is known about mercury contamination in the San Luis Unit, and measures to minimize and mitigate selenium contamination could ameliorate the risk of mercury toxicity as well. The discussion below focuses on selenium and on the species that are most sensitive and most likely to be exposed to selenium as a result of the delivery of federal water to the San Luis Unit.

San Joaquin kit fox (*Vulpes macrotis mutica*)

Status: The San Joaquin kit fox has been federally listed as endangered throughout its range since 1967 (32 FR 4001). It is endemic to the western San Joaquin Valley in the vicinity of the San Luis Unit (Figure 1).

Life history summary: Studies of kit fox and their small mammal prey in the vicinity of Kesterson Reservoir indicate that kit foxes are likely to forage in drainwater reuse areas and around evaporation ponds where selenium concentrations in their prey are likely to be well above levels known to cause adverse effects in members of the canid family of carnivores to which kit fox belong.

Risk of selenium exposure: No toxicity tests have been performed on kit fox. The most closely related surrogate species for which toxicity data are available is the domestic dog (*Canis familiaris*), which is in the same family (Canidae) as the San Joaquin kit fox. Dogs exposed to 7.2 µg/g (dry weight) dietary (organic) selenium suffered adverse effects, including reduced appetite, subnormal growth, and poorly developed ovaries and testes (Rhian and Moxon 1943). The 7.2 µg/g concentration is a Lowest Observed Adverse Effect Concentration (LOAEC); the actual toxicity threshold for domestic dogs must be an unknown amount below this value. Further, any extrapolation of dog toxicity data to kit foxes must include an uncertainty factor to account for the risk that kit foxes may be more sensitive than dogs. Therefore, given available data, an appropriate selenium dietary toxicity threshold for San Joaquin kit fox diet must be well below 7.2 µg/g.

Areas of the San Luis Unit supplied directly with relatively good quality federal water are probably best represented by the small mammals collected by Clark (1989) on the Volta Wildlife Management Area in 1984. Clark did not report whole-body selenium analyses of these mammals, but his reported analyses of liver selenium indicate that selenium concentrations in the small mammal prey of San Joaquin kit foxes at Volta were as much as two orders of magnitude less than concentrations at the drainwater evaporation ponds of Kesterson Reservoir. For example, the California voles captured at Volta Pond 5 in May 1984 (n=5) had a mean liver selenium concentration of 0.228 µg/g; the same species collected at Kesterson pond 2 at the same time (n=5) had a mean (geometric) liver selenium concentration of 119 µg/g (Clark 1989). Since background selenium concentrations in mammal livers are about 1-10 µg/g

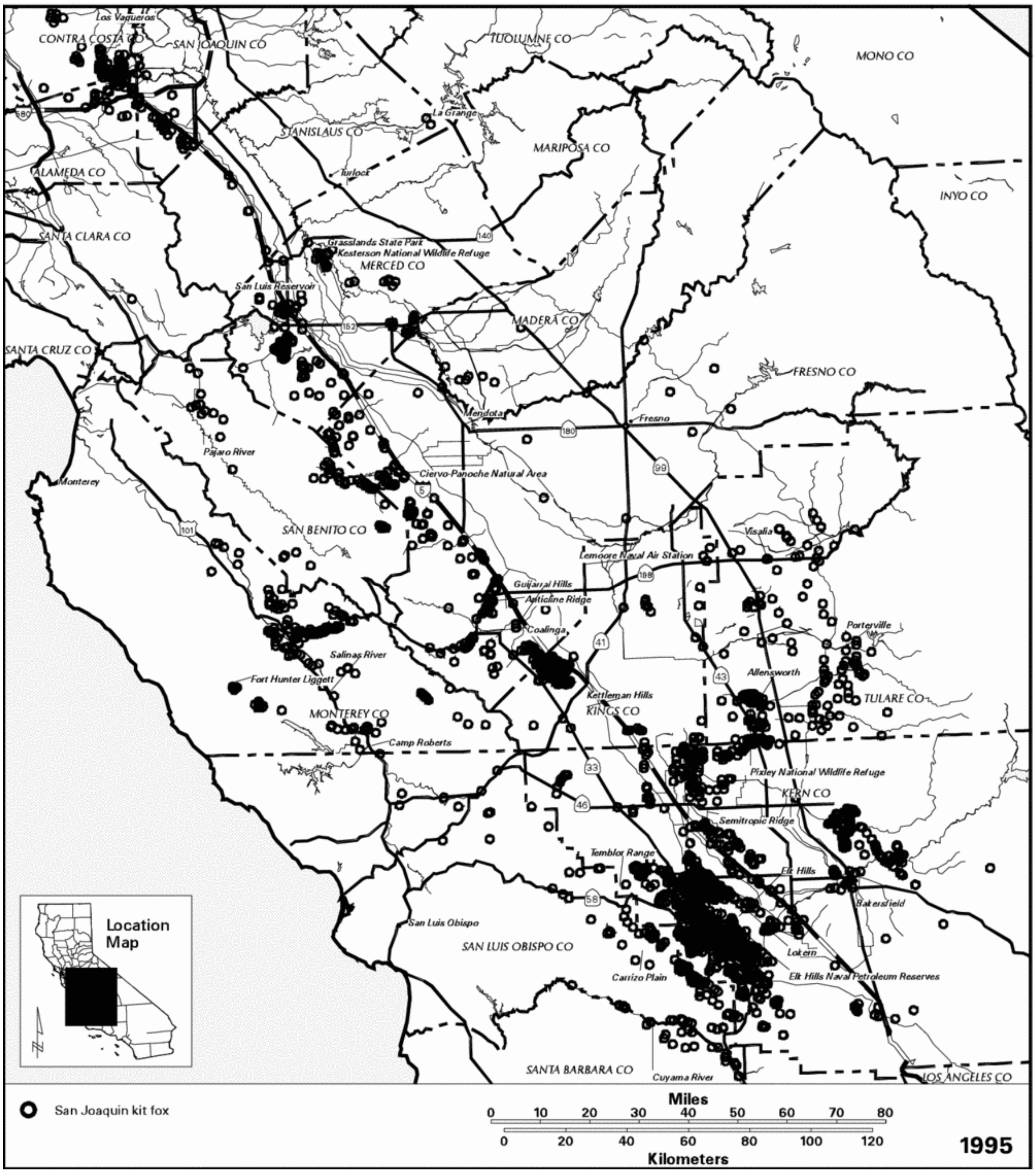

Figure 1. San Joaquin kit fox distributional records (Williams *et al*. 1998).

(NIWQP 1998), it seems likely that in portions of the Project area that are supplied with good quality water, selenium concentrations in prey pose no threat to the San Joaquin kit fox.

The San Luis Unit includes some localities that have (or are expected to have, as a consequence of application of federal water) elevated concentrations of selenium in soil and surface water or near-surface groundwater. Such localities include open ditches that convey subsurface drainwater, retired or fallowed seleniferous farm land, and drainwater reuse projects. Open drainwater conveyances are probably best represented by evaporation ponds of Kesterson Reservoir in the early 1980s.

The history of Kesterson Reservoir in the 1980s provides the best available information on potential exposure of the San Joaquin kit fox to contaminants due to the proposed action. Paveglio and Clifton (1988) sighted San Joaquin kit fox 39 times in 108 night surveys in the Kesterson Reservoir area between September 1986 and August 1988. They trapped and radio-tagged two kit fox within one mile of Kesterson Reservoir. They found that kit fox frequently used the San Luis Drain road, which formed the eastern boundary of Kesterson Reservoir. The California vole was the most important component of the diet of kit foxes in the Kesterson area (Paveglio and Clifton 1988). 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 prey items of kit fox collected 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

Because the mean concentrations of all San Joaquin kit fox prey items analyzed are about the level of the domestic dog LOAEC (7.2 µg/g, from above), it is likely that in any locations where San Joaquin kit fox range over upland portions of the Project area that may be contaminated with selenium (e.g. reuse areas), these foxes are potentially at risk from dietary intake of selenium. The average selenium concentration of each of the kit fox prey items sampled at Kesterson

Reservoir evaporation ponds was well above the dog LOAEC. Therefore, it is possible that selenium contamination in the small-mammal diet of kit foxes in the vicinity of Project evaporation ponds or solar evaporators may put San Joaquin kit foxes at risk.

If reuse areas and evaporation basins are fenced to exclude kit fox, or if other measures are taken to exclude kit fox from the project areas, recovery of remnant populations of kit fox may be impacted by loss of existing or potential habitat.

Kangaroo rats (*Dipodomys sp.*)
including:
Giant kangaroo rat (*Dipodomys ingens*)
Fresno kangaroo rat (*Dipodomys nitratoides exilis*)
Tipton kangaroo rat (*Dipodomys nitratoides nitratoides*)

Status: Three kangaroo rats in the vicinity of the San Luis Unit have been federally listed as endangered throughout their respective ranges: the Fresno kangaroo rat since 1985 (50 FR 4222-4226), the giant kangaroo rat since 1987 (52 FR 283-288), and the Tipton kangaroo rat since 1988 (53 FR 25608-25611). All three species are endemic to the San Joaquin Valley and found only in the vicinity of the San Luis Unit. The ranges of the giant and Tipton kangaroo rats extend farther south to the west side of the Tulare Basin (**Figure 2**).

Life history summary: All three species of kangaroo rat are primarily seed eaters, but also eat insects as well as green plants. All three species are found in annual grassland and saltbush scrub in alkaline soils (Williams *et al*. 1998).

Risk of selenium exposure: We are not aware of any selenium toxicity studies with kangaroo rats. Sublethal liver changes have been found in laboratory rats (*Rattus norvegicus*) following lifetime exposure to natural selenium in the diet at a concentration of 1.4 µg/g (dry weight) and reduced longevity was found at 3 µg/g in the lifetime diet (Eisler 1985). Olson (1986) also reported reproductive selenosis in rats that consumed wheat with a concentration of 3 µg/g. Halverson *et al*. (1966) found a dietary selenium threshold of about 4.8 µg/g for growth retardation in rats.

All three species of kangaroo rat were probably displaced from historic scrub and grassland habitat that was converted into irrigated crop land in the San Luis Unit with the application of federal water. All three species are not likely to be impacted by selenium in high quality irrigation water delivered to primary fields because (1) such crop land habitat is not favored by kangaroo rats, and (2) this applied water generally has relatively low concentrations of selenium. However, in retired seleniferous land, along drainwater conveyances, near evaporation ponds, and especially in drainwater re-use areas, habitat that is attractive but toxic to kangaroo rats may occur, and individuals may attempt to recolonize the habitat.

Observers performing wildlife surveys at the Atwell Island Land Retirement Program pilot site found a population of the endangered Tipton's Kangaroo Rat (USBR, 2007). The mean selenium concentration in 20 species of plants collected from Atwell Island varied from less than

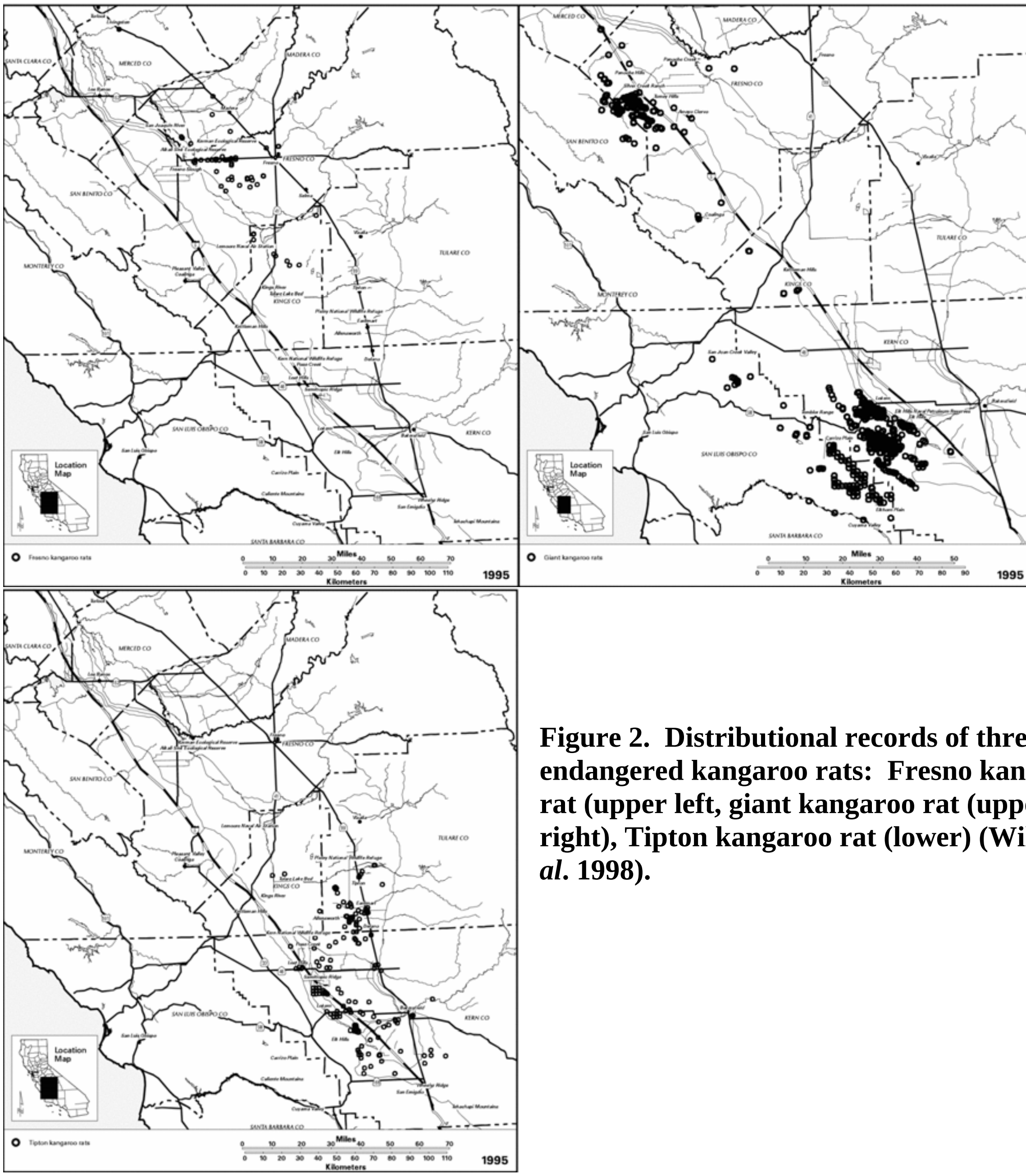

Figure 2. Distributional records of three endangered kangaroo rats: Fresno kangaroo rat (upper left, giant kangaroo rat (upper right), Tipton kangaroo rat (lower) (Williams *et al*. 1998).

0.17 to 0.5 mg/kg and none of the samples were above the 2 mg/kg threshold recommended for the project by the Service (USBR, 2005). There were no discernable differences in the selenium concentration between plant parts (whole, vegetation, fruits) at the Atwell Island site.

Agroforestry projects operated in the western San Joaquin Valley since the 1980's serve as pilot projects for the more extensive drainwater reuse areas that are likely to be established in the San Luis Unit to enable sustained irrigated agriculture there. Monitoring of agroforestry projects by the California Department of Fish and Game indicates that in reuse areas, selenium concentrations in dietary items of kangaroo rats are likely to exceed thresholds for adverse effects (Figure 3 and Figure 4).

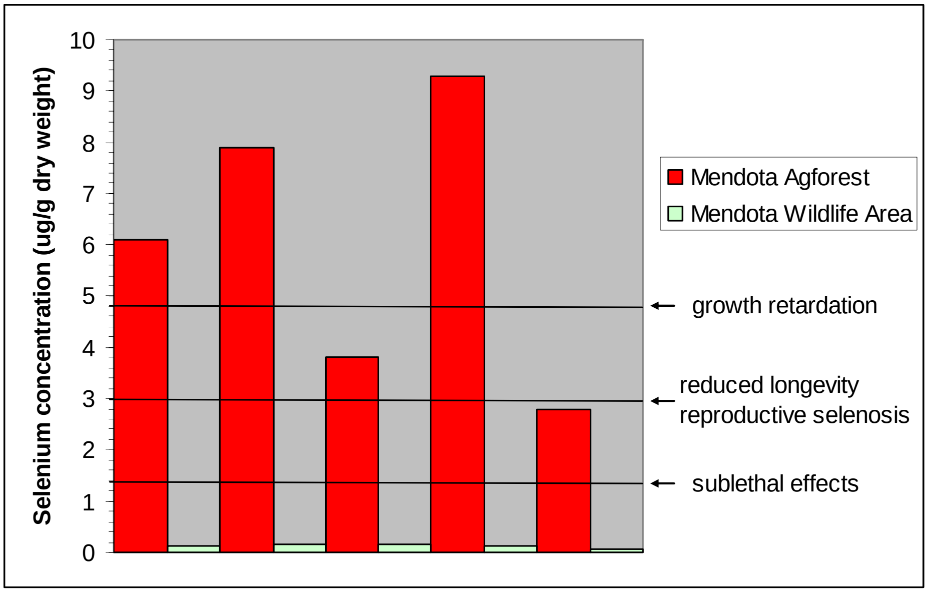

Figure 3. Selenium in rabbitfoot grass (*Polypogon monspeliensis*) collected in the Mendota agroforestry area and the Mendota Wildlife Area in May 1997 (Dunne pers. com.). Effect thresholds for rats (*Rattus norvegicus*) are from Eisler 1985, Olsen 1986, and Halverson *et al*. 1966 (See text).

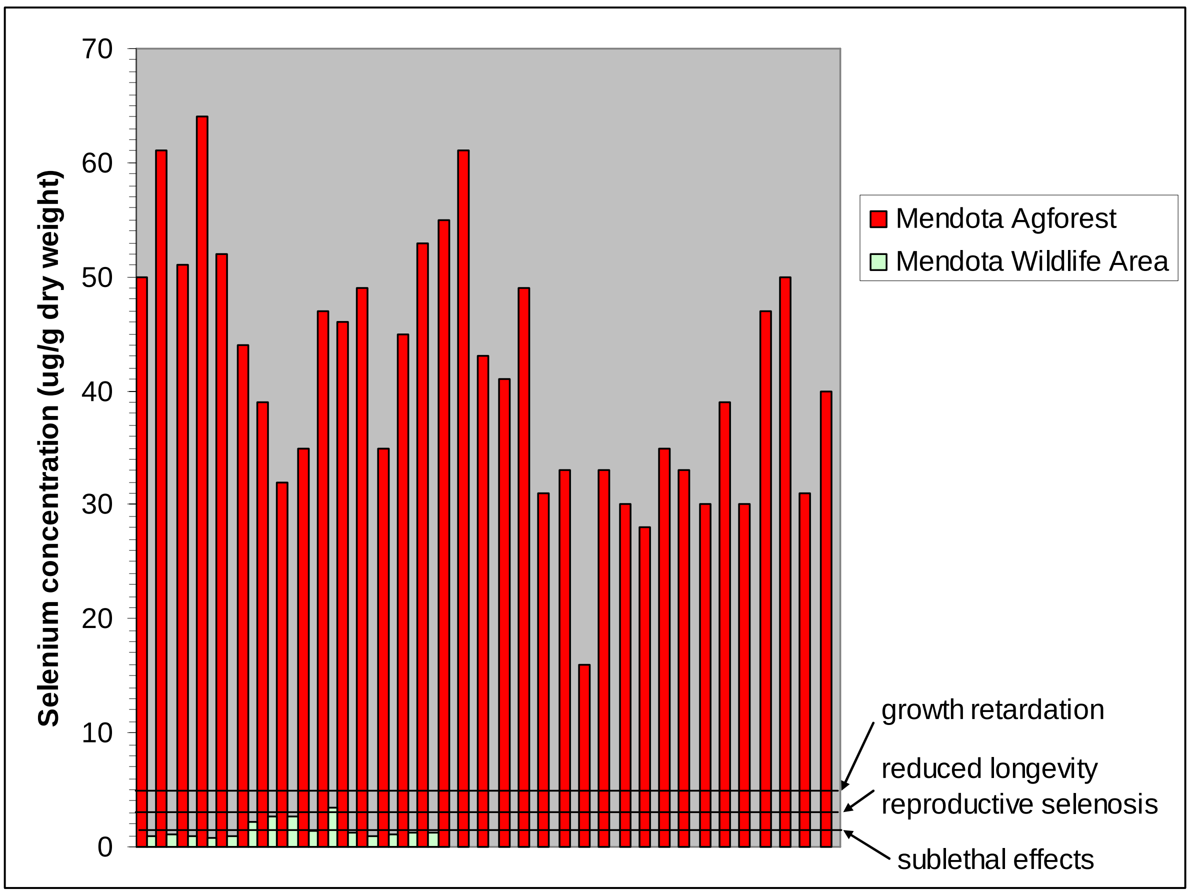

Figure 4. Selenium in sowbugs collected in the Mendota agroforestry area and the Mendota Wildlife Area in 1997 and 1998 (Dunne pers. com.) Effect thresholds for rats (*Rattus norvegicus*) are from Eisler 1985, Olsen 1986, and Halverson *et al*. 1966 (See text).

Giant garter snake (*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 1999). A 5 year review completed in September 2006 recommended no change in the listing status for the snake (USFWS 2006a). 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).

Life history summary: Fish and amphibians (tadpoles and adults) are the primary food items of giant garter snakes (58 FR 54053-54066). 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 (USFWS 1999). 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 (USFWS 1999, 58 FR 54053-54066).

Risk of selenium exposure: 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*; viability 30-54%, egg selenium 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, Heinz 1996). Therefore, given the above data, an appropriate dietary selenium toxicity threshold for the giant garter snake is probably well below 10 µg/g.

Historical exposure: Open ditches in the Northerly Area of the San Luis Unit have in the past carried subsurface drainwater with elevated concentrations of selenium. Green sunfish (*Lepomis cyanellus*) in this drainwater have been found to have concentrations of selenium ranging from 12 to 23 µg/g (geometric mean: 17.3 µg/g) (Saiki 1998), within the range of concentrations associated with adverse effects on predatory aquatic reptiles (see above). Since 1996, subsurface drainwater has been discharged, via the Grassland Bypass Project, into lower Mud Slough North, where selenium concentrations in small fish, such as mosquitofish, inland silversides (*Menidia beryllina*), red shiners (*Cyprinella lutrensis*), and fathead minnows (*Pimephales promelas*), frequently reach 10-15 µg/g (Beckon *et al*. 2003). Most of the remaining water supply channels such as Salt Slough now have fish selenium levels that are below concern thresholds (Beckon *et al*. 2003).

Potential Project-related exposure: 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 (Miller and Hornaday 1999). 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 (*Pituophis melanoleucus*) 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.

Isolation of evaporation ponds from existing giant garter snake habitat may reduce the likelihood that the ponds could serve as attractive population sinks. Such isolation may be accomplished by positioning of drainwater treatment facilities in locations remote from existing habitat and by conveyance of Project drainwater exclusively through closed pipes rather than open ditches. However, it is not known how far giant garter snakes may disperse overland to new aquatic habitats.

Blunt-nosed leopard lizard (*Gambelia sila*)

Status: The Blunt-nosed leopard lizard was federally listed as endangered in 1967 (32 FR 4001). It is endemic to the San Joaquin Valley, and several remaining populations are found in the vicinity of the San Luis Unit (Figure 5).

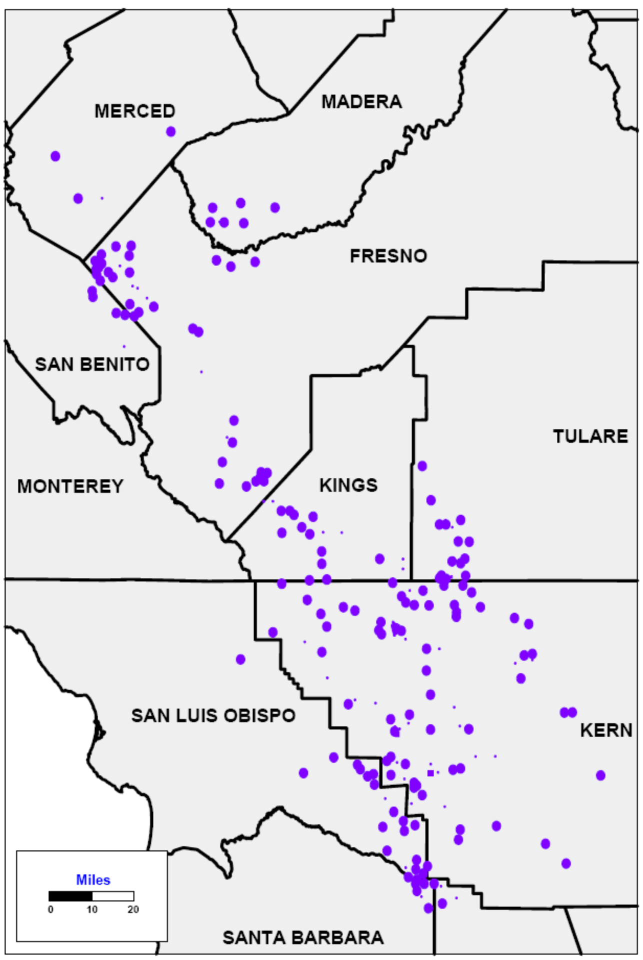

Figure 5. Currently occupied habitat of the blunt-nosed leopard lizard (http://www.cdpr.ca.gov/docs/es/espdfs/bnllall.pdf)

General life history: Blunt-nosed leopard lizards are most commonly found in open vegetated habitats dominated by non-native grasses or by low, alkali-tolerant shrubs of the family Chenopodiaceae, such as iodine bush, and seepweeds, which grow on saline and alkaline soils (Williams *et al*. 1998).

Risk of selenium exposure: Very little is known of the toxicity of selenium to reptiles (see giant garter snake discussion above); even less is known of the effects of selenium on lizards in particular. The effects of selenium on birds are better known, and birds are closely related to reptiles (Hedges 1994; Hedges and Poling 1999). Like birds, most other reptiles are oviparous (egg-laying); therefore, it is likely that in reptiles the maternal transfer of selenium to eggs is critical to the expression of selenium toxicity because the most selenium-sensitive life stage is the development of the embryo in the egg. Some of the mechanisms of maternal transfer of selenium to eggs in lizards are somewhat different from the mechanisms in birds (Unrine *et al*. 2006), but these mechanisms could be at least as efficient in moving selenium from the mother to her eggs. Roe *et al.* (2004) documented maternal transfer of selenium in alligators. Eggs from the contaminated sites had selenium concentrations ranging from 2.1 to 7.8 µg/g and lower viability (30-54 %) compared to reference sites (eggs, 1.4 to 2.3 µg/g: viability, 67 to 74 %). Alligator prey items at the contaminated sites ranged from 10 to 37 µg/g (Roe *et al.* 2004). Female western fence lizards bioaccumulated selenium in their gonads to a level (14.1 µg/g dry weight) that is toxic to bird reproduction after being fed crickets (15 µg/g Se dry weight) that had been fed on commercial feed spiked with seleno-D,L-methione (30 µg/g dry weight) (Hopkins *et al*. 2005). Therefore, lizards foraging in seleniferous habitats must be regarded as potentially at risk to selenium toxicity.

Blunt-nosed leopard lizards are likely to be exposed to selenium by feeding on insects in the vicinity of agricultural drainwater conveyances, evaporation ponds, retired seleniferous land, and re-use areas. At land retirement pilot project lands mean selenium concentrations in crickets ranged from 0.13 to 0.81 mg/kg; in beetles from 0.14 to 1.35 mg/kg; in spiders from 0.25 to 2.24 mg/kg; and in isopods 0.13 to 3.47 mg/kg (USBR 2005). These concentrations are generally within the range for terrestrial invertebrates found in non-seleniferous soils in the western United States (2.5 mg/kg, USDI 1998) although isopods at the Tranquillity site exceeded this range in most years. The selenium levels in all invertebrate groups collected from the land retirement sites are approximately an order of magnitude less than corresponding invertebrate groups collected between 1988 and 1992 in upland habitat at the closed Kesterson Reservoir (USBR 2005). The selenium exposure in invertebrates seen at the closed Kesterson Reservoir may be the best comparison data for drainwater reuse areas. Reuse areas used to grow salt-tolerant grasses and other salt-tolerant forage crops may provide habitat that is attractive to blunt-nosed leopard lizards but so enriched in selenium that it presents a risk of adverse effects.

Bald eagle (*Haliaeetus leucocephalus*)

Status: The bald eagle was federally listed as endangered on February 14, 1978 (43 FR 6233) in all of the conterminous United States except Minnesota, Wisconsin, Michigan, Oregon, and Washington, where it was classified as threatened. On August 15, 1995 (60 FR 36010), the bald eagle was down-listed to threatened throughout its range. On July 9, 2007 the Service, removed the bald eagle in the lower 48 States of the United States from the Federal List of Endangered and Threatened Wildlife (72 FR 37346). The bald eagle remains protected under the Bald and Golden Eagle Protection Act (BGEPA) and the Migratory Bird Treaty Act (MBTA) and a new permitting process will authorize limited take under BGEPA.

General life history: Breeds in coastal and aquatic habitat with forested shorelines or cliffs in North America, including the Pacific Northwest as far south as the northern Sierra Nevada Mountains in California. Wintering areas include coastal estuaries and river systems of northern California (Buehler 2000).

Risk of selenium exposure: Wintering bald eagles have been observed on occasion in the Project area and vicinity (USBR 1991). In addition, bald eagles forage for fish along waterways and the estuary downstream of the Project.

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

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

Even after considerable dilution, waters receiving agricultural drainwater from the west side of the San Joaquin Valley frequently exceed 1.4 µg/L selenium; however, bald eagle dietary exposure to fish from these waters is expected to be low.

California clapper rail (*Rallus longirostris obsoletus*)

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

General life history: The California clapper rail inhabits salt marshes surrounding the San Francisco Bay, California. Principal habitats are low portions of coastal wetlands dominated by cordgrass and pickleweed (USFWS 1984). Nesting habitat in San Francisco Bay is characterized by tidal sloughs, abundant invertebrate populations, pickleweed, gum plant, and wrack in upper zone. Individuals do not migrate far from the breeding grounds (Eddleman and Conway 1998).

Risk of selenium exposure: California clapper rails feed largely on benthic invertebrates, including filter-feeding mussels and clams (Moffitt 1941), a well-documented pathway for bioaccumulation of selenium (Pease *et al*. 1992, Stewart *et al*. 2004). Lonzarich *et al*. (1992) reported that eggs of California clapper rails collected from the north bay in 1987 contained up to 7.4 µg/g selenium. Water data from this time and location are not available. The *in ovo* threshold for selenium exposure that causes toxic effects on embryos of California clapper rails is unknown. For another benthic-foraging marsh bird, the black-necked stilt, the *in ovo* threshold for embryotoxicity is 6 µg/g selenium (Skorupa 1998). The most widely-used biphasic model (Brain and Cousens 1989) applied to Heinz *et al*. (1989) data from laboratory experiments with mallard reproduction indicates that in mallards, a selenium concentration of 7.4 µg/g (dry weight) in the eggs would be associated with a 32 percent reduction in hatchability of the eggs (Figure 6).

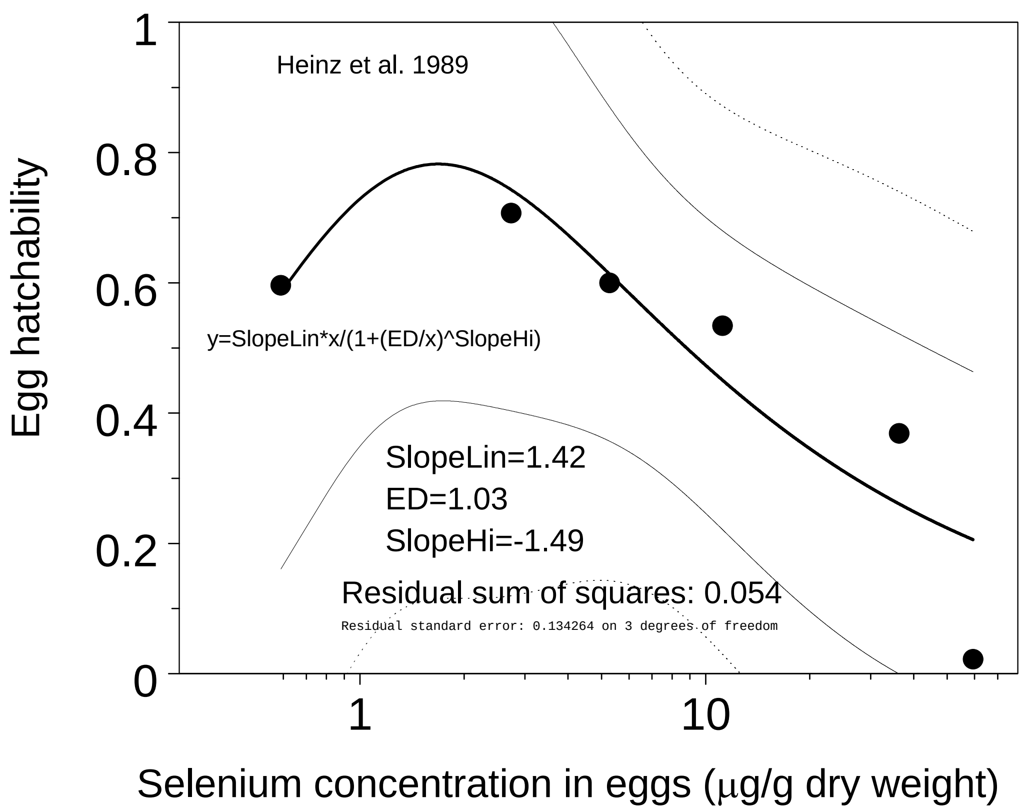

Figure 6. The hatching success of mallard eggs as a function of selenium concentration in the eggs, with the Brain-Cousens biphasic model fitted by least squares regression. Confidence intervals of 95% and 99% are shown.

It has been demonstrated for mallard ducks that interactive effects of selenium and mercury can be super-toxic with regard to embryotoxic effects (Heinz and Hoffman 1998). Lonzarich *et al*. (1992) also reported potentially embryotoxic concentrations of mercury in eggs of California clapper rails. Abnormally high numbers of nonviable eggs, 13.7-22.9 percent (Schwarzbach 1994) and 31 percent (Schwarzbach *et al.* 2006), have also been reported for the California clapper rail.

Based, in part, on the data for California clapper rails, staff technical reports prepared for the San Francisco Bay Regional Water Quality Control Board recommend decreasing current selenium loading to the estuary by 50 percent or more (Taylor *et al*. 1992, Taylor *et al*. 1993). The California clapper rail is particularly vulnerable to any locally elevated effluent concentrations of selenium as the rail generally occupies small home ranges of only a few acres. As selenium loads to the San Joaquin River and hence to the estuary are reduced over time due to implementation of selenium total maximum daily load limits and the Grassland Bypass Project, potential impacts to clapper rails due to delivery of water to the San Luis Unit will diminish.

California least tern (*Sterna antillarum browni*)

Status: The California least tern has been federally listed as endangered throughout its range since 1970 (35 FR 8491-8498, 35 FR 16047-16048). Distributed along the Pacific coast from the San Francisco Bay to Baja California, it is widely separated from the four other subspecies of least tern (Thompson *et al*. 1997). A 5-year review was completed in 2006 which recommended down listing the species to threatened (USFWS 2006b).

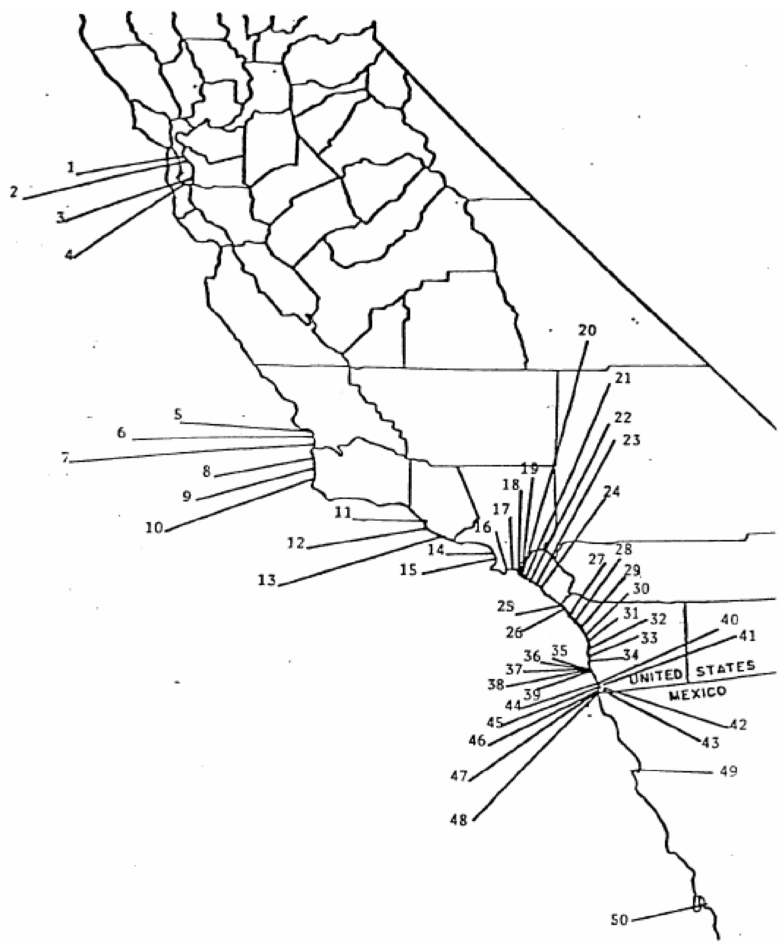

Figure 7. Nesting sites of the California least tern recorded since 1970 (USFWS 1985).

Life history summary: California least terns are migratory, wintering along the southern coast of Mexico (Thompson *et al*. 1997). The primary nesting site in San Francisco Bay is located at the former Alameda Naval Air Station. 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*), and yellowfin goby (*Acanthogobius flavimanus*). Up to 50 species of fish have been documented in their diet (USFWS 1985).

Risk of selenium exposure: Currently, breeding colonies of California least tern are confined to scattered, isolated locations on beaches along the coast of California and in the San Francisco estuary, where they feed on surface fish in adjacent waters. In these locations any agricultural drainwater from the San Luis Unit is well diluted. Therefore, the current risk of selenium to this bird is probably *de minimis*. However, it is possible that the creation of evaporation ponds for disposal of agricultural drainwater from the San Luis Unit could provide habitat attractive to California least terns. 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. Concentrations of selenium in evaporation pond invertebrates are likely to be sufficiently elevated to cause reproductive impacts in least terns. 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). Methods of configuring evaporation ponds to discourage shorebird usage (deepening and steepening sides) will be ineffective in deterring foraging by least terns.

Chinook Salmon (*Oncorhynchus tshawytscha*)

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

Life history summary: Chinook salmon are anadromous and semelparous. That is, as adults they migrate from a marine environment into the fresh water streams and rivers of their birth (anadromous) where they spawn only once and die (semelparous). Juvenile Chinook may spend from 3 months to 2 years in freshwater after emergence before migrating to estuarine areas as smolts, and then into the ocean to feed and mature. The timing and duration of the migratory movements of Chinook salmon are important in assessing their exposure to selenium and estimating consequent risks. Natal streams and estuary rearing habitat vary seasonally in selenium concentration and the salmon evidently vary in sensitivity to selenium across stages in their life histories. A more detailed life history discussion is provided for salmon in order to more clearly define the selenium exposure risks to the various ESUs and to identify the ones at greatest risk to selenium exposure resulting from irrigation deliveries to the San Luis Unit.

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

Use of estuarine habitat: On their migration downstream, many Chinook salmon fry take up residence in the river estuary where they rear to smolt size (about 70 mm fork length) before resuming their migration to the ocean. The proportion of fry that rear in the estuary is not known. On Vancouver Island, BC, about 30 percent of the estimated downstream migrants could be accounted for in the estuary; the fate of the remaining 70 percent 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 1983, Healey 1991). "Stream-type" Chinook salmon, enter freshwater months before spawning and reside in freshwater for a year or more following emergence, whereas "ocean-type" Chinook salmon spawn soon after entering freshwater and migrate to the ocean as fry or parr within their first year. Spring-run Chinook salmon exhibit a stream-type life history. Adults enter freshwater in the spring, hold over summer, spawn in fall, and the juveniles typically spend a year or more in freshwater before emigrating. Winter-run Chinook salmon are somewhat anomalous in that they have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter freshwater in winter or early spring, and delay spawning until spring or early summer (stream-type). However, juvenile winter-run Chinook salmon migrate to sea after only four to seven months of river life (ocean-type). Adequate instream flows and cool water temperatures are more critical for the survival of Chinook salmon exhibiting a stream-type life history due to over summering by adults and/or juveniles. The stream-type life history also increases selenium exposure risks during the critical egg development stage of the adult and the growth stage of 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 Red Bluff Diversion Dam (RBDD) on the Sacramento River may begin as early as mid-July, typically peaks in September, and can continue through March in dry years (Vogel and Marine 1991; NMFS 1997). From 1995 to 1999, all winter-run Chinook salmon outmigrating as fry passed RBDD by October, and all outmigrating pre-smolts and smolts passed RBDD by March (Martin *et al.* 2001). The emigration timing of Central Valley spring-run Chinook salmon is highly variable (CDFG 1998). Some fish may begin emigrating soon after emergence from the gravel, whereas others over summer and emigrate as yearlings with the onset of intense fall storms (CDFG 1998). The emigration period for spring-run Chinook salmon extends from November to early May, with up to 69 percent of the young-of-the-year fish outmigrating through the lower Sacramento River and Delta during this period (CDFG 1998).

As Chinook salmon fry and fingerlings mature, they prefer to rear further downstream where ambient salinity is up to 1.5 to 2.5 parts per thousand (Healey 1980, 1982; Levings *et al.* 1986). Juvenile winter-run Chinook salmon occur in the Delta from October through early May based on data collected from trawls, beach seines, and salvage records at the Central Valley Project (CVP) and State Water Project (SWP) pumping facilities (CDFG 1998). The peak of listed juvenile salmon arrivals in the Delta generally occurs from January to April, but may extend into June. Upon arrival in the Delta, winter-run Chinook salmon spend the first two months rearing in the more upstream, freshwater portions of the Delta (Kjelson *et al.* 1981, Kjelson *et al.* 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: Due to water diversions and consequent loss of breeding and migrating habitat, California Central Valley Chinook salmon have been effectively extirpated from the San Joaquin River above the confluence of the Merced River. Planning is underway to restore salmon to this river by increasing flows and restoring habitat. However, seepage and flood flows carrying agricultural drainwater from the San Luis Unit into the San Joaquin River may impact salmon and could impair efforts to restore them to this river.

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

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

The effects of selenium on animals (including fish) are well known to be biphasic (beneficial at low doses; toxic at high doses; see, for example, Beckon *et al*. 2008), and in the Hamilton *et al*. (1990) experiment, the 90-day survival data appear to confirm a biphasic dose-response relationship with respect to the survival endpoint (Figure 8). Therefore, we fitted a biphasic model (Brain and Cousens 1989) to the data by least squares regression. This regression provides a weight-of-evidence estimate of the maximum survival rate (0.7, or 70 percent) of young salmon under these experimental conditions at the estimated optimal selenium concentration in the fish (about 1 µg/g whole body dry weight). It also provides an estimate of the survival rate at any given selenium concentration above the optimum. Any such survival rate estimate can be compared to the maximum survival rate to yield an estimate of the mortality (inverse of survival) specifically attributable to selenium. For example, at a fish tissue concentration of 7.9 µg/g (whole body dry weight) the regression curve predicts a survival of 0.29 (29 percent). As a proportion of the maximum survival this is 0.29/0.7 = 0.41, or 41 percent. Therefore our best weight-of-evidence estimate of the mortality due to selenium toxicity at a tissue concentration of 7.9 µg/g is the inverse of 0.41, which is 0.59, or 59 percent. Similarly, the model predicts that fish with a selenium concentration of 2.45 µg/g (whole body dry weight) after 90 days of exposure would experience 20 percent mortality due to selenium (Figure 8 lower graph).

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

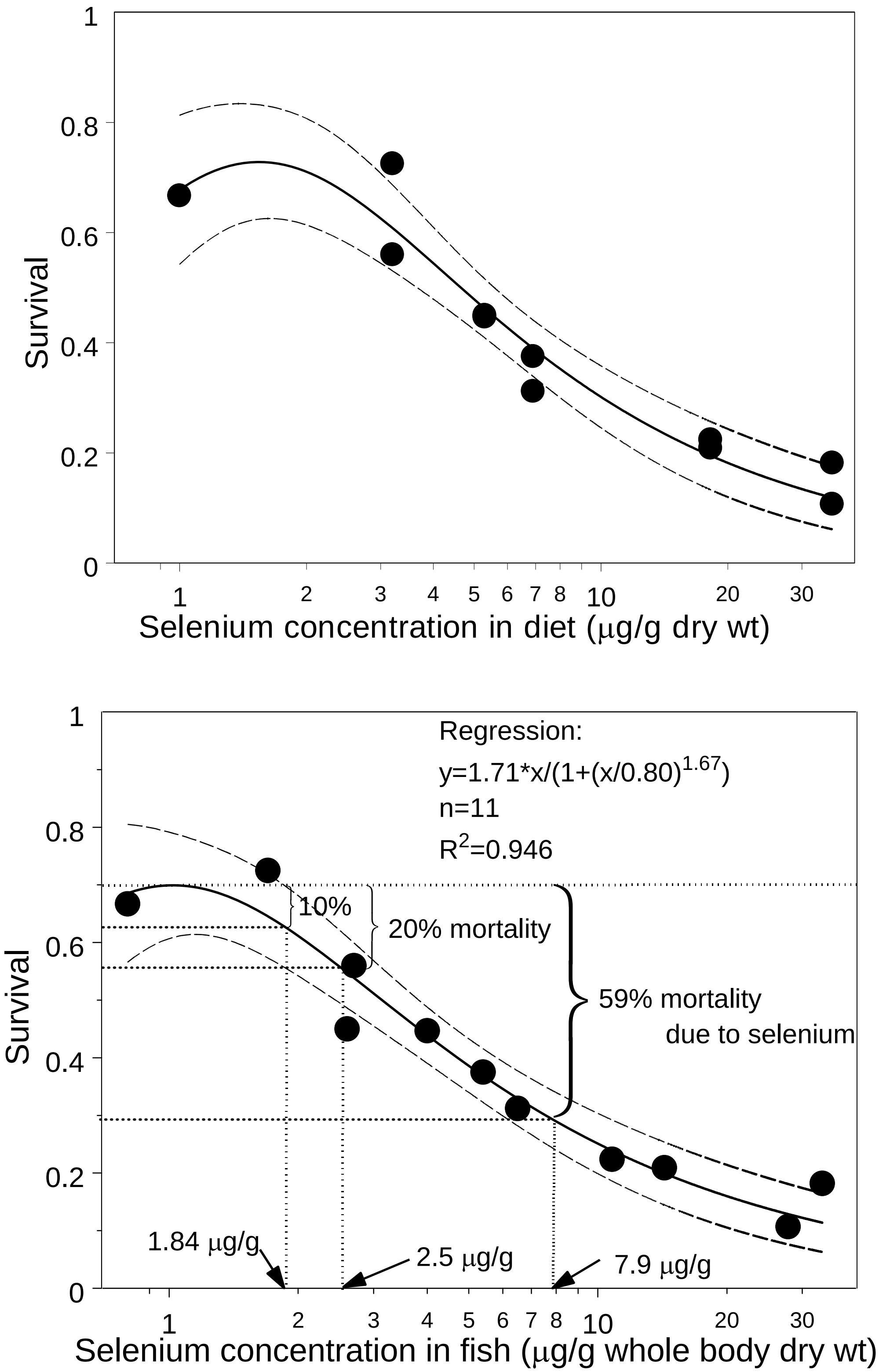

Figure 8. Survival as a function of selenium concentration in diet (above) and tissue (below) of juvenile Chinook salmon after 90 days of exposure to dietary selenium. A biphasic model (Brain and Cousens 1989) was fitted by least squares regression. Dashed lines indicate 95% confidence bands around the regressions.

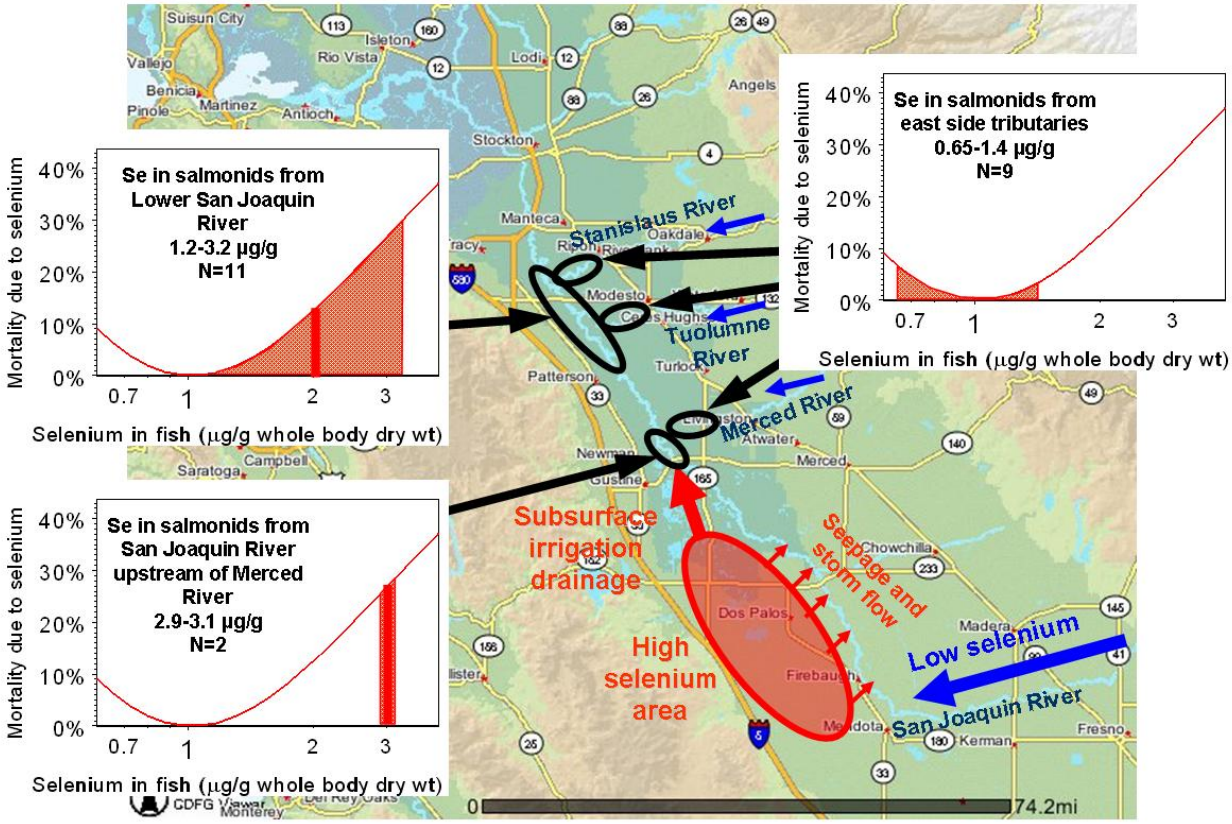

Figure 9. Risk of mortality to juvenile Chinook salmon based on selenium measured in the salmon (Saiki, *et al*. 1991) and the toxicity data shown in Figure 8 (presented here as mortality). Solid red bars represent the geometric mean selenium concentration in sampled fish at each location or cluster of locations. The stippled red areas span the ranges of concentrations in fish at the respective locations.

commonly reach levels (Beckon *et al*. 2003) that could kill a substantial portion of young salmon (Figure 8 upper graph) if the salmon, on their downstream migration, are exposed to those selenium-laden food items for long enough for the salmon themselves to bioaccumulate selenium to toxic levels.

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 were likely to kill more than 25% (**Figure 9**).

Concentrations of selenium in the San Joaquin River have been reduced since juvenile Chinook salmon were sampled in 1987 (Saiki *et al*. 1991). However, the relationship between selenium in water and in young salmon in 1987 (Figure 10) indicates that there remains a substantial ongoing risk to migrating juvenile Chinook salmon in the San Joaquin River (Figure 11).

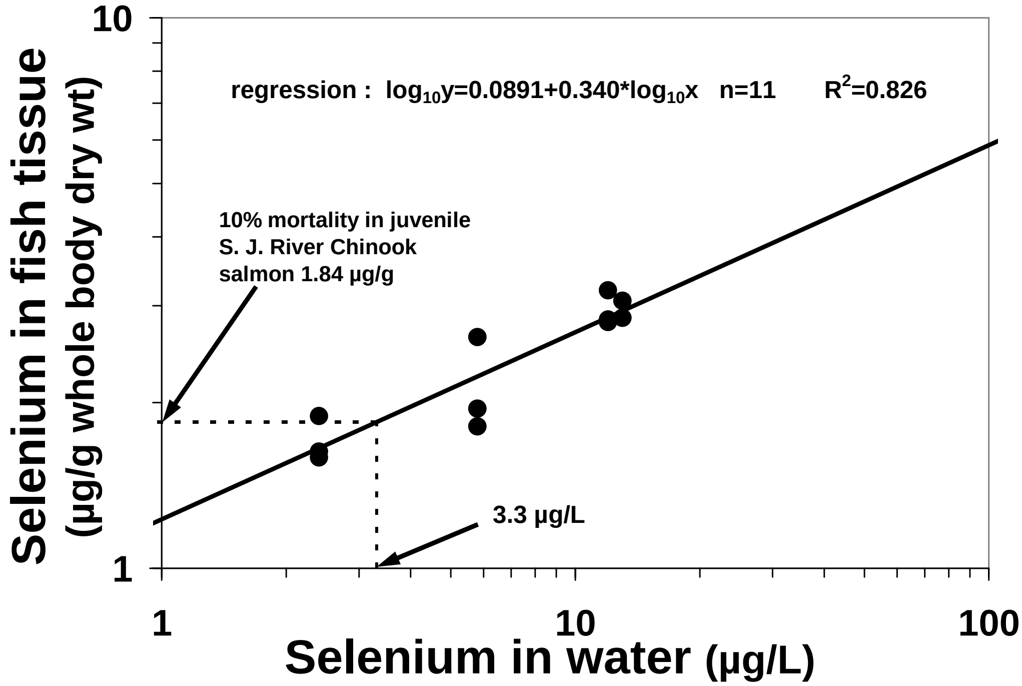

Figure 10. Relationship between selenium in juvenile Chinook salmon (Saiki *et al*. 1991, Saiki pers. com.) and water (Central Valley Regional Water Quality Control Board "Flat File") in the San Joaquin River and its tributaries.

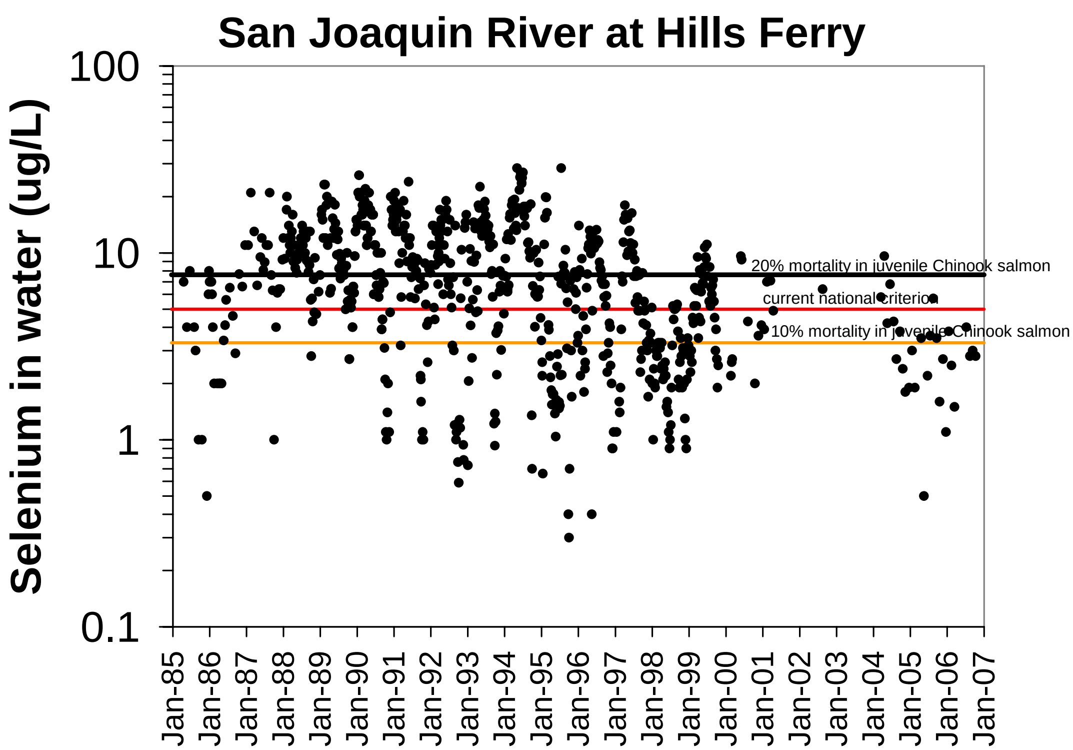

Figure 11. Selenium concentrations measured in the San Joaquin River at Hills Ferry, just upstream of the confluence of the Merced River. The data are from the Central Valley Regional Water Quality Control Board.

Steelhead Trout (*Oncorhynchus mykiss***)**

Status: Steelhead trout are the anadromous form of the rainbow trout species. Central Valley steelhead were listed as threatened under the ESA on March 19, 1998 (63 FR 13347). This ESU consists of steelhead populations in the Sacramento and San Joaquin River (inclusive of and downstream of the Merced River) basins in California's Central Valley.

The breeding of wild steelhead in the Central Valley is mostly confined to the 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).

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).

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

Winter steelhead generally leave the ocean from August through April, and spawn between December and May (Busby *et al.* 1996). Timing of upstream migration is correlated with higher flow events 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 percent) in California streams. Most steelhead spawning takes place from late December through April, with peaks from January though March (Hallock *et al.* 1961). Steelhead spawn in cool, clear streams featuring suitable gravel size, depth, and current velocity, and may spawn in intermittent streams as well (Everest 1973; Barnhart 1986).

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

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

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

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

Risk of selenium exposure: Planning is underway to restore salmon to the San Joaquin River by increasing flows and restoring habitat. Such restoration efforts would likely improve the small steelhead population in the San Joaquin Valley. However, as with salmon, seepage and flood flows carrying agricultural drainwater from the San Luis Unit into the San Joaquin River may impact steelhead and may confound efforts to restore them to this river.

Because steelhead are regarded as a life-history variant or “form” of the rainbow trout species, studies of the non-anadromous form of rainbow trout may provide a good indication of the risks of the exposure of steelhead to selenium. Such studies indicate that rainbow trout are among the more sensitive of fish to selenium. One of these studies examined the effects of selenium on fry of rainbow and brook trout exposed in streams in Alberta, Canada (Holm 2002, Holm *et al.* 2003). In summary, this study indicates that maternal selenium would result in 20 percent mortality of fry if female rainbow trout have a tissue selenium concentration of 2.93 µg/g wholebody dry weight (Figure 12). The USEPA (2004) has proposed that a fish tissue chronic criterion of 7.9 µg/g selenium (wholebody) would be protective. However, female rainbow trout in the wild with a concentration of about 8 µg/g selenium in their (wholebody) tissue would produce eggs that suffer 44.2 percent mortality by swimup stage (Figure 12). Among the swimup survivors, 96 percent would suffer edema (Figure 13) and 42 percent would have craniofacial deformities (Figure 14) (for details, see USFWS 2005).

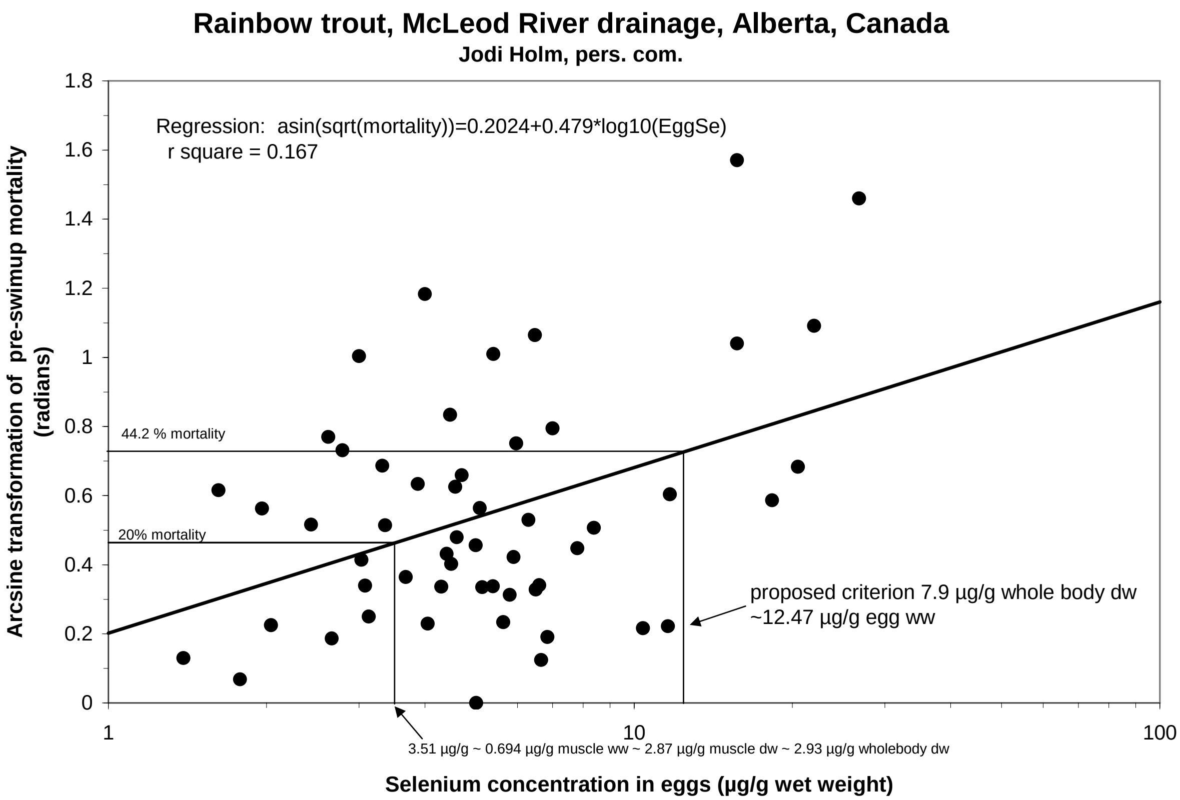

Figure 12. Relationship between selenium in rainbow trout eggs and mortality of eggs and fry by swimup stage. The arcsine transformation is applied to mortality data, as appropriate for linear regressions with percents or proportions (Sokol and Rohlf 1981). Data are from the years 2000-2002.

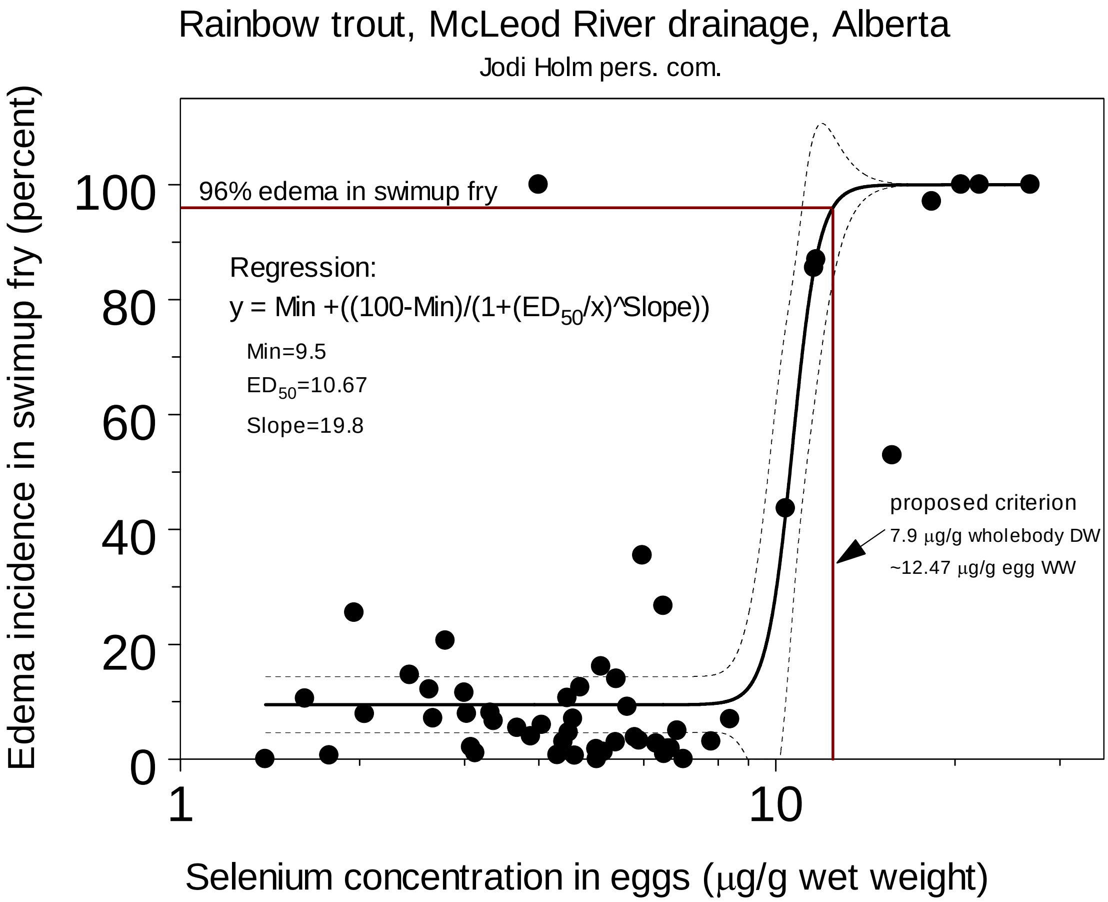

Figure 13. Relationship between selenium in rainbow trout eggs and edema in surviving swimup fry. Data from the years 2000-2002.

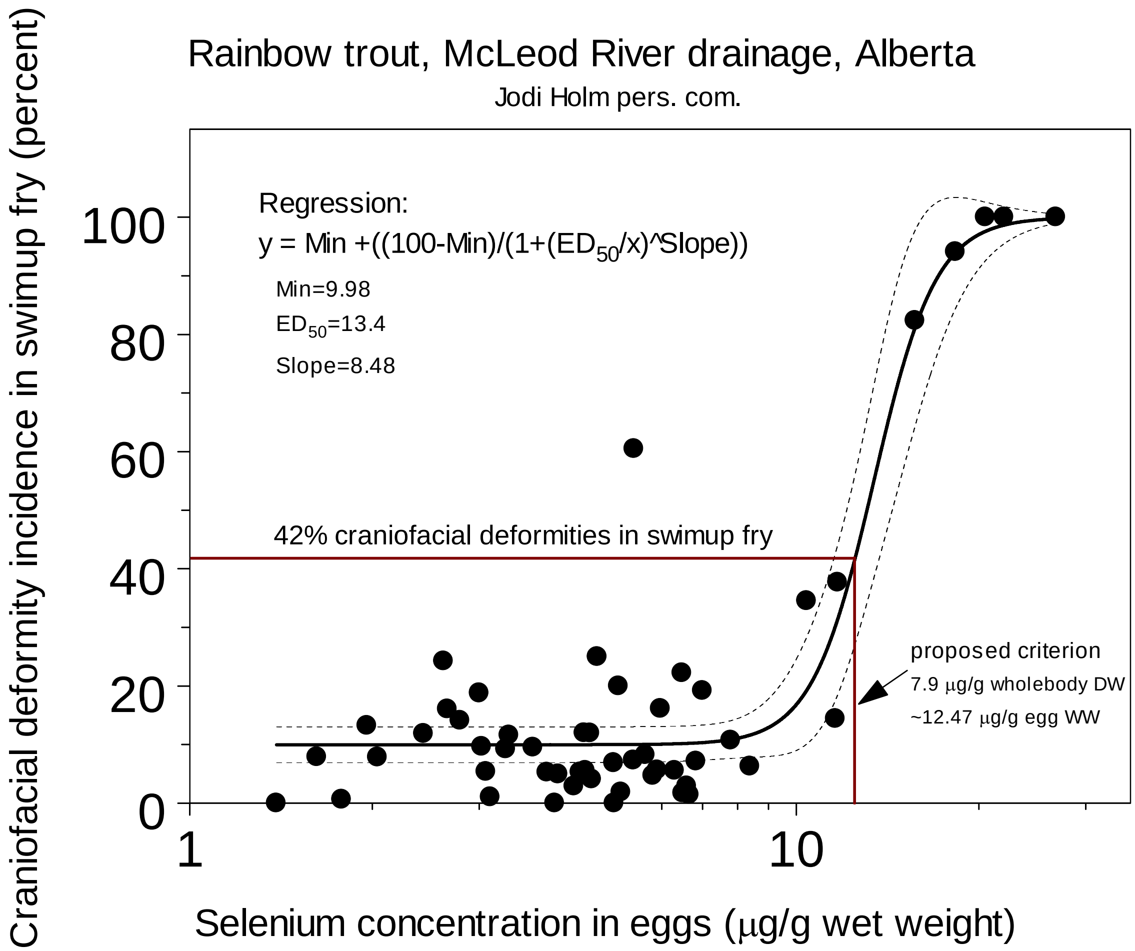

Figure 14. Relationship between selenium in rainbow trout eggs and craniofacial deformities in surviving swimup fry. Data from the years 2000-2002.

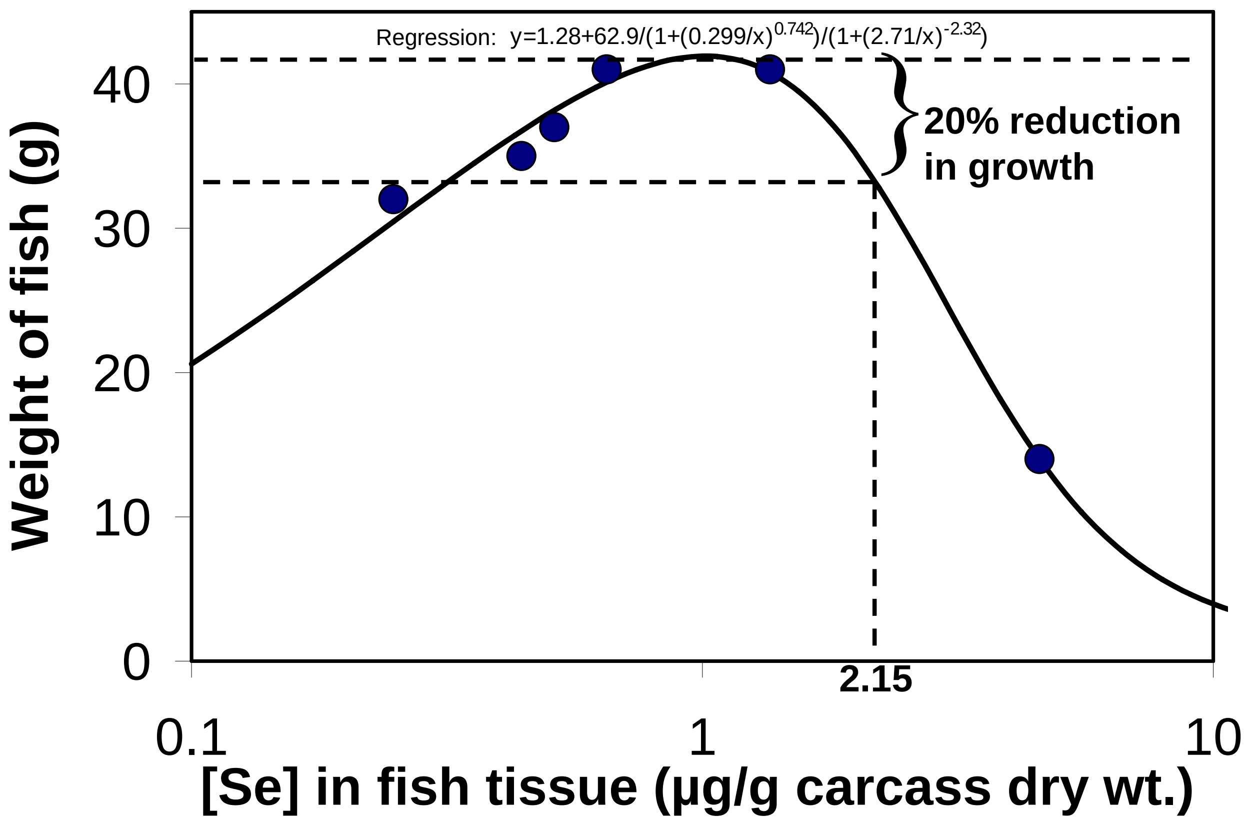

Figure 15. Average weights of juvenile rainbow trout after 20 weeks of exposure to diets spiked with sodium selenite (Hilton *et al*. 1980). The data were fitted with a biphasic model (Beckon *et al*. 2008). In the model it was assumed that at extremely high and extremely low selenium concentrations, 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.

A laboratory experiment monitored the growth of juvenile rainbow trout exposed to a diet spiked with selenium in the form of sodium selenite (Hilton *et al*. 1980). This experiment indicates that juvenile rainbow trout that reach a selenium concentration of about 8 µg/g (carcass dry weight) by exposure for 20 weeks to dietary selenium in the form of sodium selenite will experience at least an 86 percent reduction in weight relative to the weight they would gain if their exposure to dietary sodium selenate were optimal (Figure 15). A weight reduction of 20 percent would be associated with a tissue selenium concentration of 2.15 µg/g (carcass dry weight).

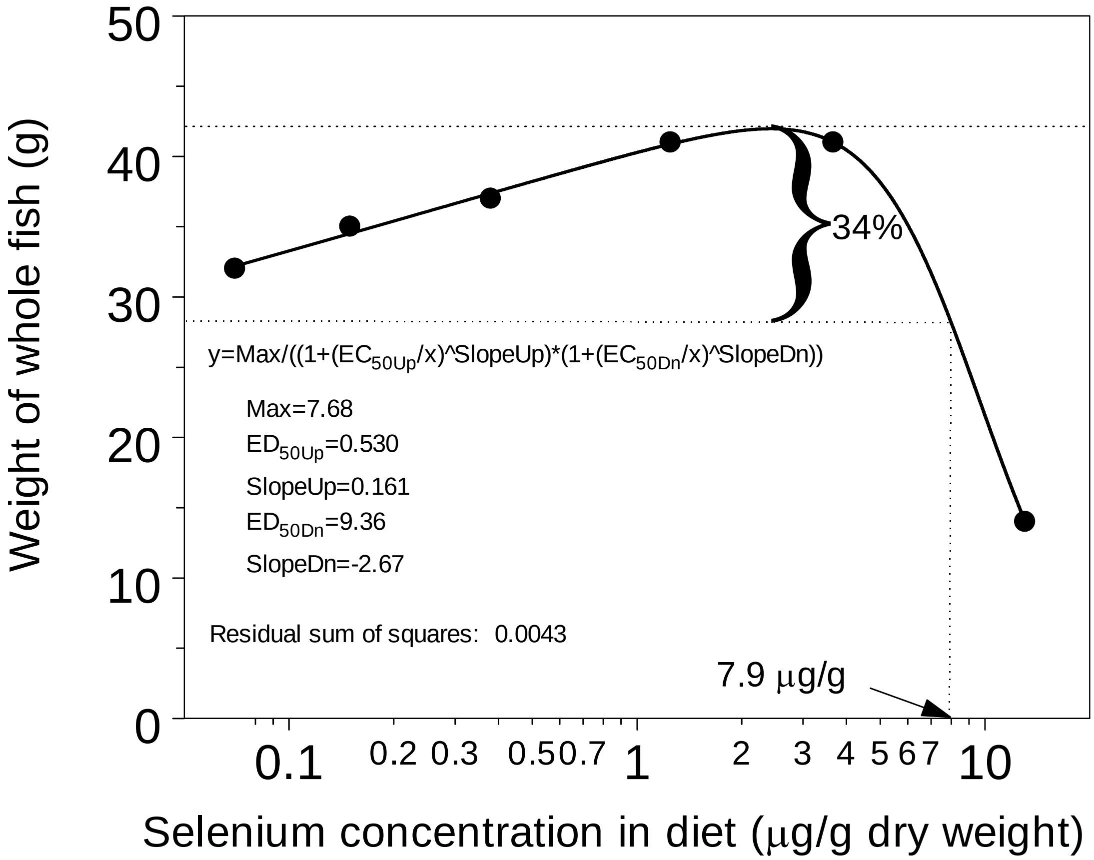

Figure 16. Average weights of juvenile rainbow trout after 20 weeks dietary exposure to sodium selenite (Hilton *et al*. 1980). A biphasic model (Beckon *et al*. 2008) is fitted to the data by least squares non-linear regression.

This experiment also indicates that if young rainbow trout feed on tissue that has a selenium concentration of about 8 µg/g (in the form of sodium selenite) they will suffer a reduction in growth of about 34 percent (Figure 16). Because the form of selenium administered to the fish in this experiment was sodium selenite, this analysis may yield an underestimate of the adverse effects of the more bioavailable organic forms of selenium that fish consume in the wild.

The experiments summarized above indicate that the larval survival and the health and growth of young steelhead trout would be impaired by a concentration of selenium (about 8 µg/g) commonly exceeded in invertebrates, small (prey) fish, and larger predatory fish in waterways that carry agricultural drainwater in the vicinity of the San Luis Unit (Beckon *et al*. 2003).

Green sturgeon (*Acipenser medirostris*)

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

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

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

Risk of selenium exposure: Little is known of the risk of selenium to green sturgeon, but white sturgeon (*Acipenser transmontanus*), a representative surrogate species for the green sturgeon, have been the subject of detailed studies within the San Francisco Bay estuary. See the discussion for white sturgeon below.

White Sturgeon (*Acipenser transmontanus*)

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

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

Risk of selenium exposure: Many individuals of this species remain year-round in San Pablo Bay, the part of the San Francisco Bay estuary with the highest selenium concentrations (up to 2.7 µg/L). Clams predominated in the esophageal and stomach contents of white sturgeon caught by anglers in San Pablo Bay (213 fish) and Suisun Bay/Carquinez Strait (142 fish) in 1965-1967 (McKechnie and Fenner 1971). More recently with the change in the benthic food structure of the estuary (Feyrer *et al*. 2003) white sturgeon may depend more on the introduced Asian clam, *Potamocorbula amurensis*, which is an extraordinarily efficient bioaccumulator of selenium (Stewart *et al*. 2004). The median concentration of selenium in Asian clams from San Pablo Bay was found to be above 10 µg Se/g (Stewart *et al*. 2004). Based on histopathological alterations in the kidney, Tashjian *et al.* (2006) estimated that for juvenile white sturgeon a threshold dietary selenium toxicity concentration lies between 10 and 20 µg Se/g. It is uncertain at what point in their life white sturgeon begin feeding on Asian clams.

Linares *et al*. (2004) found concentrations of selenium as high as 46.7 µg/g in gonads of 39 white sturgeon captured in the San Francisco Bay. Kroll and Doroshov (1991) reported that developing ovaries of white sturgeon from San Francisco Bay contained as much as 71.8 µg/g selenium or 7-times the threshold for reproductive toxicity in fish (Lemly 1996a, 1996b) of 10 µg/g. An effect threshold in white sturgeon eggs has been estimated to be between 9 µg/g and about 16 µg/g in experiments in which seleno-L-methionine was injected into yolk sac larvae of white sturgeon (Linares *et al*. 2004). Linville (2006) showed that significant developmental defects and mortality occurred in white sturgeon eggs at a threshold of around 11–15 µg/g selenium. A hazard threshold of around 3–8 µg/g in developing white sturgeon was suggested by Linville (2006).

Sampling of pallid sturgeon (*Scaphirhynchus al*bus) in the Missouri River system suggests that normal selenium levels in sturgeon eggs are 2-3 µg/g (Ruelle and Keenlyne 1993) as has been found for many other fish species (see review in Skorupa *et al*. 1996 and in USDI-BOR/FWS/GS/BIA 1998). Thus, white sturgeon in the San Francisco Bay estuary are producing eggs with as much as 35-times normal selenium content. Based on studies regarding toxicity response functions for avian and fish eggs (e.g., Lemly 1996a, 1996b; Skorupa *et al.* 1996;

USDI-BOR/FWS/GS/BIA 1998) and assuming that sturgeon are as sensitive to selenium as birds and other fish, it is highly probable that these fish are reproductively impaired due to selenium exposure. For example, bluegill embryos resulting from ovaries containing 38.6 µg/g selenium exhibited 65 percent mortality (Gillespie and Bauman 1986).

Considering the high bioaccumulation efficiency of Asian clams and their importance in the diet of white sturgeon any selenium reaching the estuary from upstream sources likely contributes to the exposure risk of white sturgeon. As selenium loads to the San Joaquin River and hence to the estuary are reduced over time due to implementation of selenium total maximum daily load limits and the Grassland Bypass Project, potential impacts to sturgeon due to delivery of water to the San Luis Unit should diminish.

Delta smelt (*Hypomesus transpacificus*)

Status: Delta smelt were federally listed as a threatened species on March 5, 1993, (58 FR 12854). The Service completed a 5-year review in March 2003 (USFWS 2003) and recommended no change in its listing status; however, there has been a recent dramatic decline in Delta smelt numbers since 2005.

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

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

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

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

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

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

Delta smelt, salvaged from the Chipps Island area during the springs of 1993 and 1994, had whole-body selenium concentrations of 1.5 µg/g dw (n=41, range 0.7 - 2.3 µg/g) (Bennett *et al.* 2001). 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). Therefore, Delta smelt would appear to be at low risk to selenium exposure.

Sacramento splittail (*Pogonichthys macrolepidotus*)

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

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

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

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

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

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

Risk of selenium exposure: Like white sturgeon, splittail are likely to be relatively vulnerable to selenium contamination because of their estuarine habitat, bottom-feeding habits, and high bioaccumulation rates of Asian clams. The Asian clam and other mollusks constituted 34 percent of the splittail diet (Feyrer and Matern 2000, Feyrer *et al*. 2003).

The median selenium liver level in splittail from the Suisun Bay area of the estuary was about 13 µg/g dw (Stewart *et al*. 2004) while background liver concentrations in fish are generally less than 5 µg/g (USDI-BOR/FWS/GS/BIA 1998). Deformities typical of Se exposure have been seen in splittail collected from Suisun Bay (Stewart *et al*. 2004). Teh *et al*. (2004) found that juvenile splittail are impacted (liver lesions) by chronic exposure (nine months) to a diet of 6.6 µg/g selenium.

In 1998, an above normal rainfall year type, splittail were collected from Mud and Salt Sloughs within the San Luis National Wildlife Refuge during quarterly fish sampling for the Grassland Bypass Project (GBP)(Beckon *et al*. 1999). This was the only time in the 14 year life of the project (1993-2007) that splittail were documented in these two sloughs. Selenium levels in splittail composite whole-body samples at the three Mud Slough sites were all above the GBP concern threshold of 4 µg/g dw with the site immediately downstream of the San Luis Drain having 7.1 µg/g dw (Beckon *et al*. 1999). At Salt Slough where drainwater no longer is discharged into the slough the splittail whole-body composite concentration was 3.1 µg/g dw (Beckon *et al*. 1999).

Considering the high bioaccumulation efficiency of Asian clams and their importance in the diet of splittail any selenium reaching the estuary from upstream sources likely contributes to the exposure risk of splittail. As selenium loads to the San Joaquin River and hence to the estuary are reduced over time due to implementation of selenium total maximum daily load limits and the Grassland Bypass Project, potential impacts to splittail due to delivery of water to the San Luis Unit should diminish.

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