RTD-194

APPENDIX E2

Selenium Ecological Risk Assessment

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TABLES

FIGURES

E.2.1 Selenium Ecological Risk Guidelines[1]

The assessment of the risks that selenium poses to fish and wildlife can be difficult due to the complex nature of selenium cycling in aquatic ecosystems (Lemly and Smith 1987). Early assessments developed avian risk thresholds through evaluating bird egg concentrations and relating those to levels of teratogenesis (developmental abnormalities) and reproductive impairment (Skorupa and Ohlendorf 1991). In 1993, to evaluate the risks of the Grassland Bypass Project (GBP) on biotic resources in Mud and Salt Sloughs, a set of Ecological Risk Guidelines based on selenium in water, sediment, and residues in several biotic tissues were developed by a subcommittee of the San Luis Drain Re-Use Technical Advisory Committee (CAST 1994, Engberg et.al. 1998). These guidelines (Table E.2-1) are based on a large number of laboratory and field studies, most of which are summarized in Skorupa et al. (1996) and Lemly (1993). In areas where the potential for selenium exposure to fish and wildlife resources exists, site-specific selenium risk guidelines can be used to trigger appropriate actions by resource managers, regulatory agencies, and dischargers. For the GBP the selenium risk guidelines have been divided into three levels: No Effect, Level of Concern, and Toxicity. In the No Effect range risks to sensitive species are not likely. As new information becomes available it should be evaluated to determine if the No Effect level should be adjusted. Since the potential for selenium exposure exists, periodic monitoring of water and biota is appropriate.

Table E.2-1 Recommended Ecological Risk Guidelines for Selenium Concentrations

Medium	Effects on	Units	No Effect	Level of Concern	Toxicity
Warmwater Fish (whole body)	fish growth/condition/survival	mg/kg (dry weight)	< 4	4-9	> 9
Vegetation (as diet)	bird reproduction	mg/kg (dry weight)	< 3	3-7	> 7
Invertebrates (as diet)	bird reproduction	mg/kg (dry weight)	< 3	3-7	> 7
Sediment	fish and bird reproduction	mg/kg (dry weight)	< 2	2-4	> 4
Water (total recoverable Se)	fish and bird reproduction (via foodchain)	µg/L	< 2	2-5	> 5
Avian egg	egg hatchability	mg/kg (dry weight)	< 6	6-10	> 10

Notes:

These guidelines, except those for avian eggs, are intended to be population based. Thus, trends in means over time should be evaluated. Guidelines for avian eggs are based on individual level response thresholds (e.g., Heinz 1996, Skorupa 1998)

A tiered approach is suggested with whole body fish being the most meaningful in assessment of ecological risk in a flowing system.

The warmwater fish (whole body) Level of Concern threshold is based on adverse effects on the survival of juvenile bluegill sunfish experimentally fed selenium enriched diets for 90 days (Cleveland et al. 1993). It is the geometric mean of the "no observable effect level" and the "lowest observable effect level."

The Toxicity threshold for warmwater fish (whole body) is the concentration at which 10% of juvenile fish are killed (DeForest et al. 1999).

The guidelines for vegetation and invertebrates are based on dietary effects on reproduction in chickens, quail and ducks (Wilber 1980, Martin 1988, Heinz, 1996).

If invertebrate selenium concentrations exceed 6 mg/kg then avian eggs should be monitored (Heinz et al. 1989, Stanley et al. 1996).

Within the Level of Concern range there may be risk to sensitive species, and contaminant concentrations in water, sediment, and biota should be monitored on a regular basis. Immediate actions to prevent selenium concentrations from increasing should be evaluated and implemented if appropriate. Long-term actions to reduce selenium risks should be developed and implemented. Research on effects on sensitive or listed species may be appropriate.

Within the Toxicity range, adverse affects are more likely across a broader range of species, and sensitive or listed species would be at greater risk. These conditions will warrant immediate

[1] The section was taken from the 2001 Grassland Bypass Project, Final EIS/EIR, Appendix E and the Grassland Bypass Project Report, 2004–2005, and references cited herein are contained in that report.

action to reduce selenium exposure through disruption of pathways, reduction of selenium loads, or other appropriate actions. More detailed monitoring, studies on site-specific effects, and studies of pathways of selenium contamination may be appropriate and necessary. Long-term actions to reduce selenium risks should be developed and implemented.

The guidelines (except those for avian eggs) are intended to be population based. Therefore they should be used for evaluating population means rather than contaminant concentrations in individuals.

E.2.1.1 Warmwater Fish

The warmwater fish guidelines (Table E.2-1) refer to concentrations of selenium in warmwater fish that adversely affect the fish themselves. The original 1993 fish guidelines have been replaced by explicitly "warmwater fish" guidelines in recognition of the evidence from the literature that coldwater fish (salmon and trout) are more sensitive to selenium than warmwater fish and that GBP monitoring data available is limited to warmwater fish. Although a coldwater fish guideline is not proposed here, a discussion of selenium effects on coldwater fish is provided in section E2.1.2 since the best information currently available happens to be very site-specific to the GBP.

The Level of Concern threshold for warmwater fish has been kept at about 4 mg/kg (all fish data are whole body, dry weight). Experimental data reported in the literature may be interpreted to support a range of thresholds around this value. In particular, bluegill sunfish dietary and waterborne toxicity data in Cleveland et al. (1993) can be used to support warmwater fish Level of Concern thresholds of 3.3 mg/kg, 3.4 mg/kg, 3.9 mg/kg, or 5.9 mg/kg. Bluegill sunfish are warmwater fish that are found in the sloughs in the GBP area, and the Cleveland et al. (1993) study yielded the best available data on warmwater fish toxicity applicable to GBP.

Cleveland et al. (1993) found no adverse effects after 59 days of exposure to concentrations of dietary selenium that resulted in a bluegill tissue concentration of 2.7 mg/kg (NOEC). Fifty nine days of exposure to dietary concentrations that resulted in tissue concentrations of 4.2 mg/kg (LOEC) caused a significant increase in mortality relative to controls. Following the USEPA method (Stephan et al. 1985) employed by DeForest et al. (1999), the tissue threshold is calculated as the geometric mean of the NOEC and the LOEC. Application of the USEPA procedure to these data yields a toxicity threshold of 3.4 mg/kg. A similar analysis of a water-borne selenium exposure experiment (Cleveland et al. 1993) yields a threshold value of 3.3 mg/kg.

Other data in Cleveland et al. (1993) may be interpreted to support a threshold closer to 4 mg/kg or a threshold of 5.9 mg/kg. The experiments of Cleveland et al. (1993) suggest that selenium concentrations in fish tissues do not reach equilibrium until at least 90 days of dietary exposure (see Figure 3 in Cleveland et al. 1993). This appears consistent with the finding, summarized below, that in the field, selenium concentrations in fish are best predicted by water concentrations averaged over the entire period of one to seven months prior to the date the fish is sampled. In deriving a tissue threshold, there then appears to be some support for using the relationship between dietary concentration and tissue concentration at 90 days rather than 59 days. After 90 days of dietary exposure bluegill with a tissue selenium concentration of 3.3 mg/kg did not exhibit adverse effects that were significantly greater than controls, but bluegill with a tissue concentration of 4.6 mg/kg experienced significantly increased mortality. Bluegill with a tissue concentration of 7.5 mg/kg had three times the mortality of controls, but that difference in mortality was not statistically significant at the 95% level of confidence (see Table 4 and Figure 3 in Cleveland et al. 1993). However, the condition factor (a measure of weight relative to length) of the fish at 7.5 mg/kg, was significantly worse than controls. Depending on whether or not the significant mortality at a tissue concentration of 4.7 mg/kg is treated as anomalous, the LOEC would be either 4.7 mg/kg or 7.5 mg/kg. Corresponding thresholds would be 3.9 mg/kg (geometric mean of 3.3 mg/kg and 4.6 mg/kg) or 5.9 mg/kg (geometric mean of 4.6 mg/kg and 7.5 mg/kg) respectively. Given the range of possible threshold values discussed above, the Level of Concern threshold of 4 mg/kg listed in Table E.2-1 was not changed from the original 1993 threshold. However, considering that these data do not include adverse effects on reproduction which may be affected at lower concentrations, this threshold may not be fully protective of sensitive warmwater fish species.

The Toxicity threshold for warmwater fish (whole body) of 9 mg/kg is recommended by DeForest et al. (1999). In the analysis of DeForest et al. (1999) the threshold represents an EC_{10}, that is, the concentration at which 10 percent of fish are affected. DeForest et al. (1999) excluded some toxicity data from their analysis that could support a lower threshold (Cleveland et al., 1993). Also, reproductive impairment may occur at lower selenium concentrations, but too few data are available to do a similar analysis on this effect. Therefore, this Toxicity threshold may not be fully protective of sensitive warmwater fish species.

E.2.1.2 Coldwater Fish

Testing fall run Chinook salmon from the Merced River, Hamilton et al. (1990) found that salmon fry growth was significantly reduced compared to controls after 30 and 60 days of being fed a diet (containing mosquitofish from the SLD) having a selenium concentration of 3.2 mg/kg dry weight. After 90 days of that diet, the selenium concentration in the salmon fry averaged 2.7 mg/kg whole body, dry weight. This fish tissue concentration was the lowest observable effect concentration (LOEC). The no observable effect concentration (NOEC) in salmon fry tissue was 0.8 mg/kg. Following the USEPA method (Stephan et al. 1985) employed by DeForest et al. (1999), the tissue threshold is calculated as the geometric mean of the NOEC and the LOEC. This procedure applied to the Hamilton et al. (1990) SLD data yields a threshold of 1.5 mg/kg (geometric mean of 0.8 and 2.7 mg/kg). It should be noted that this threshold may incorporate the interacting effects of other toxic constituents of drainwater that may have been assimilated by the SLD mosquitofish that were used as feed in the Hamilton, et al. (1990) experiments. Furthermore, at the time of these experiments (1985), the SLD held agricultural drainwater from the Westlands, an area adjacent to the Grasslands area. Therefore, although these are the most site-specific selenium toxicity data available, these data may not perfectly match the current risk of toxicity to coldwater fish in the San Joaquin River due to agricultural drainwater from the GBP. Although the sloughs affected by the GBP have coldwater beneficial uses designated by the Central Valley Regional Water Quality Control Board, the fish community principally consists of warmwater species. A temporary barrier is installed seasonally across the San Joaquin River to exclude Chinook salmon (a coldwater species) from these sloughs and from the San Joaquin River upstream of its confluence with the Merced River. Additionally, any application of the coldwater fish risk guidelines should take into account the fact that many coldwater fish are anadromous, and therefore feed in the selenium-contaminated portion of the San Joaquin River for a limited period of time-- a brief period in their juvenile stage as they migrate downstream to the ocean.

A Toxicity threshold for coldwater fish (whole body) of 9 mg/kg has been recommended by DeForest et al. (1999). In the analysis by DeForest et al. (1999) the toxicity threshold represents an EC_{10}, that is, the concentration at which 10 percent of fish are affected. DeForest et al. (1999) excluded site-specific and longer term data (Hamilton et al. 1990) which could support lower thresholds. For example, to derive their toxicity threshold for coldwater fish, DeForest et al. (1999) used only the 60 day growth data in Hamilton et al. (1999); they disregarded the 90 day mortality data in Hamilton et al. (1999) that would have yielded a toxicity threshold (corresponding to 10% mortality) of 1.7 mg/kg. In addition, the DeForest et al. (1999) analysis focused on growth and mortality. Reproductive impairment may occur at lower selenium concentrations, but too few data are available to do a similar analysis on this effect. Therefore, this threshold may not fully protect sensitive coldwater fish species.

E.2.1.3 Vegetation and Invertebrates

The guidelines for vegetation (as diet) and invertebrates (as diet) refer to selenium concentrations in plants and invertebrates affecting birds that eat these items. These guidelines are mainly based on experiments in which seleniferous grain or artificial diets spiked with selenomethionine were fed to chickens, quail or ducks resulting in reproductive impairment (Wilber 1980, Martin 1988, Heinz 1996). The Level of Concern threshold for vegetation is 3 mg/kg (dry weight) and the Toxicity threshold is 7 mg/kg. The invertebrate Level of Concern threshold and Toxicity threshold are the same as those for vegetation.

E.2.1.4 Water

Fish and wildlife are much more sensitive to selenium through dietary exposure from the aquatic food chain than by direct waterborne exposure. Therefore the guidelines for water reflect water concentrations associated with threshold levels of food chain exposure (Hermanutz et al. 1990, Maier and Knight 1994), rather than concentrations of selenium in water that directly affect fish and wildlife. The Level of Concern threshold is 2 µg/L and the Toxicity threshold is 5 µg/L.

E.2.1.5 Sediment

As with water, the principal risk of sediment to fish and wildlife is via the aquatic food chain. Therefore the sediment guidelines are based on sediment concentrations as predictors of adverse biological effects through the food chain (USFWS 1990, Van Derveer and Canton 1997). The Level of Concern threshold for sediment (dry weight) is 2 mg/kg and the Toxicity threshold is 4 mg/kg.

E.2.1.6 Bird Eggs

Bird eggs are particularly good indicators of selenium contamination in local ecosystems (Heinz 1996). However, the interpretation of selenium concentrations in bird eggs in the GBP area is complicated by the proximity of contaminated and uncontaminated sites and by the variation in foraging ranges among bird species. Relative to the guidelines originally used for the GBP, the guidelines used in the 2001 EIR/EIS and here for bird eggs have been revised upward based on recent studies of hatchability of ibis, mallard, and stilt eggs (Henny and Herron 1989, Heinz 1996, USDI-BOR/FWS/GS/BIA 1998). The Level of Concern threshold has been raised from 3 to 6 mg/kg dry weight, and the Toxicity threshold has been raised from 8 to 10 mg/kg dry weight.

E.2.2 Selenium Environmental Impacts Modeling

Estimation of the effects of changing water quality on fish and wildlife is especially difficult in flowing systems where fish and wildlife may move between waters with widely different concentrations of contaminants. For example, fish seasonal migration, dispersal and diurnal foraging movements may result in a poor correlation between the concentrations of selenium in water and in fish collected at the same location. Nonetheless, a broad relationship does emerge from the large body of data on contaminant levels in water and in biota that have been collected since 1991 by the agencies participating in Grassland Bypass Project Monitoring Program (Figure E.2-1 and Figure E.2-2) (Regional Board 2008; SFEI 2008; Beckon, Eacock, and Westman, written comm. 2008). The average concentration (whole body dry weight) of selenium in all species of fish sampled generally follows trends in selenium concentration in water, with a lag period.

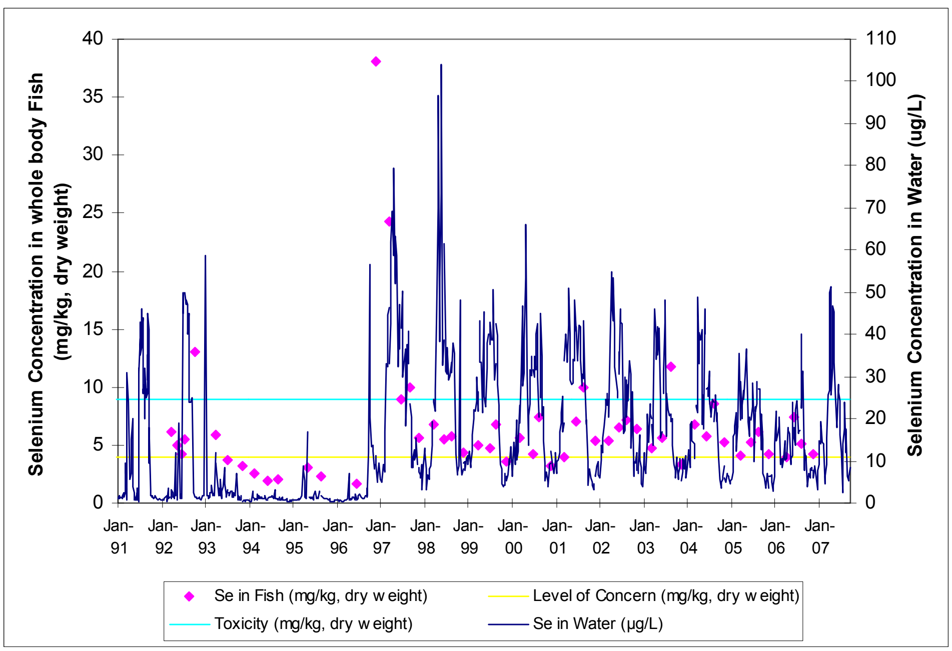

Figure E.2-1 **Selenium in Water and Fish (average of all species) in Mud Slough just below the outfall of the San Luis Drain**

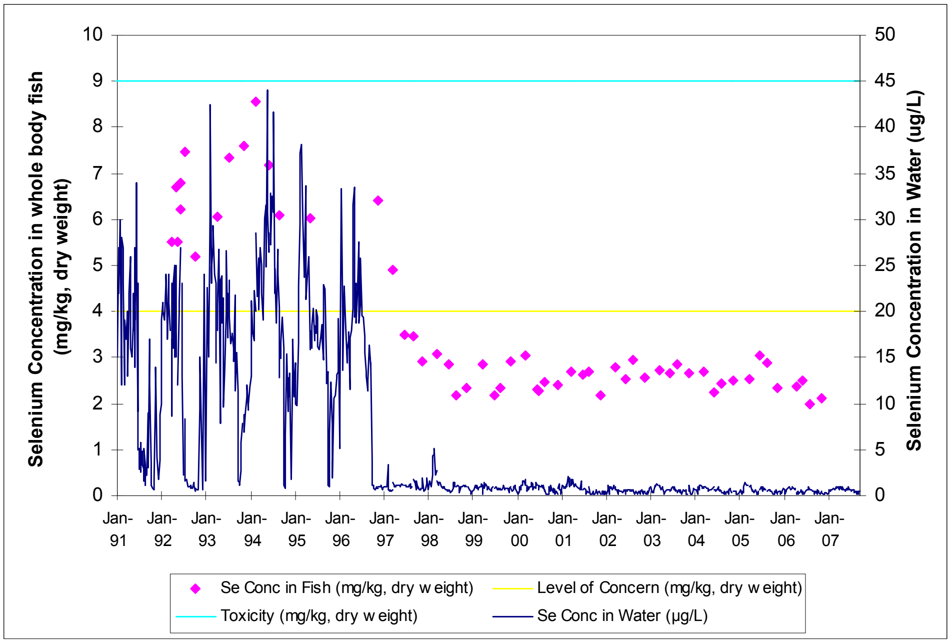

Figure E.2-2 Selenium in Water and Fish (average of all species) in Salt Slough

E.2.2.1 Fish Model

The lag in the response of fish selenium concentrations to water selenium concentrations probably results from biogeochemical processes as well as reflecting the time it takes for selenium to be assimilated and depurated (eliminated) through successive links of the food chain. The lag effect is particularly evident in the Salt Slough data (see Figure E.2-2) after the beginning of the GBP in September 1996 when the selenium concentration in water dropped abruptly, but concentrations in fish declined more gradually over a period of several months. The overall lag period is effectively a composite of the individual lag times characteristic of different fish species at various trophic levels. For example, some small fish feed directly on algae, whereas other species, such as largemouth bass, eat smaller fish that in turn feed on invertebrates that feed on algae. Catfish may feed on clams that filter detritus from previous generations of organisms.

The average lag time for all fish can be estimated by comparing the fit of linear regressions of average water selenium concentrations versus fish selenium concentrations using a variety of water averaging periods involving candidate lag times. This procedure has been used with 1991-2006 GBP data to compare several potential lag times and averaging intervals (Table E.2-2). Two data points for whole body fish collected at Station D were considered outliers. These occurred during the first few months subsequent to the first flush of the San Luis Drain and it was hypothesized that theses samples may have contained Mosquitofish which were previously living in the San Luis Drain. These outliers were removed during the 2001 and 2008 analysis.

The maximum selenium concentration in water during the prior month was also regressed against average fish selenium concentration to provide a test of an assumption underlying an ecosystem risk index for selenium developed by Lemly (1993, 1995). To evaluate ecosystem risk, Lemly's protocol uses maximum concentrations of selenium rather than averages. However, the GBP data indicate that the maximum concentration of selenium in water during the prior month is worse than any averaging interval tested as a predictor of fish selenium concentration (Table E.2-2).

Table E.2-2 Results of linear regressions of average selenium concentration in all species of fish vs. selenium concentrations in water using various averaging periods for water (n=62)

Independent variable	Dependant variable	Proportion of variance explained by linear regression (r2)
Maximum water concentration 1-30 days prior to fish sample	Average fish concentration	0.45
Average water concentration 1-30 days prior to fish sample	Average fish concentration	0.48
Average water concentration 30-60 days prior to fish sample	Average fish concentration	0.54
Average water concentration 0-3 months prior to fish sample	Average fish concentration	0.59
Average water concentration 1-4 months prior to fish sample	Average fish concentration	0.61
Average water concentration 1-7 months prior to fish sample	Average fish concentration	0.59
Log_{10} of average water concentration 1-7 months prior to fish sample	Log_{10} of average fish concentration	0.76

The best prediction of fish selenium concentrations is provided by the logarithmic transformation of selenium concentrations in water averaged over the period one to seven months prior to collection of the fish sample (Figure E.2-3). That averaging period may be used not only to predict average tissue concentrations for all fish, but also to predict the proportion of individual composite samples of fish that fall into each of the ecological risk classes: No Effect, Level of Concern, and Toxicity (see Table E.2-1). To make such predictions, logistic models are fitted to the existing (1992–2006) data on proportions of samples that have fallen into each of the risk classes (Figure E.2-4 and Figure E.2-5). These models may be combined to provide estimates of the expected effects on fish resulting from projected selenium concentrations in water in the waterways potentially affected by the GBP alternatives (Figure E.2-6).

For example, if the six-month average concentration of selenium in water is 5 µg/L, one month after the end of that averaging period, the expected average concentration of selenium (whole body dry weight) in all fish sampled at the same site would be 4.1 mg/kg (see Figure E.2-3), which is above the Level of Concern threshold for warmwater fish (4 mg/kg). However, of the composite fish samples collected at that time, 53% would be expected to have selenium concentrations below the Level of Concern threshold, 45% would be expected to be within the Level of Concern class (4-9 mg/kg), and 2% would be expected to be above the Toxicity threshold (9 mg/kg). The modeling is based on selenium analyses of composite samples, each sample usually consisting of 5 to 50 individual fish. Therefore, predictions of the models must be understood in the same terms, i.e., the model does not predict the distribution of individual fish into risk classes, but rather, the distribution of composite samples into risk classes.

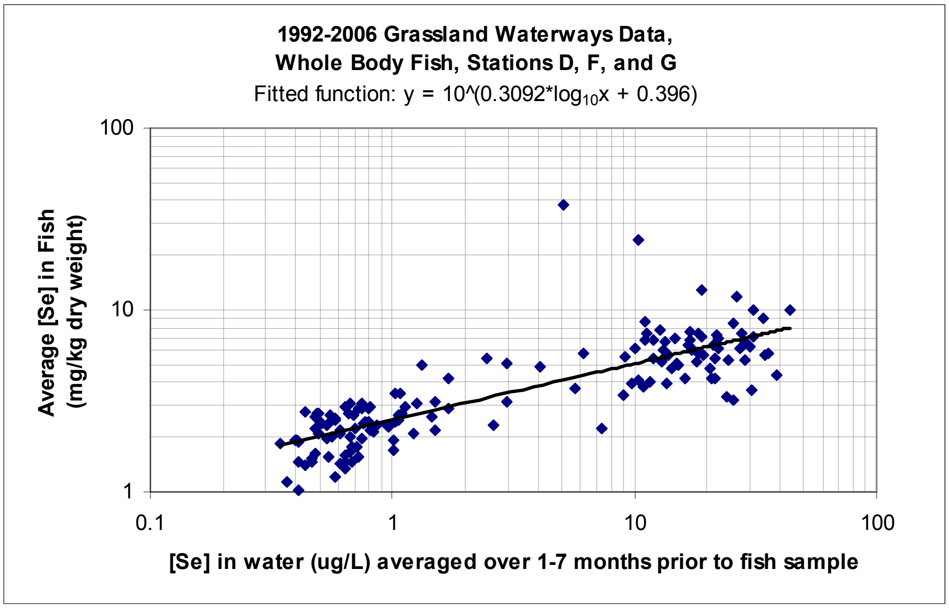

Figure E.2-3 Bioaccumulation of selenium in fish in Grassland area waterways

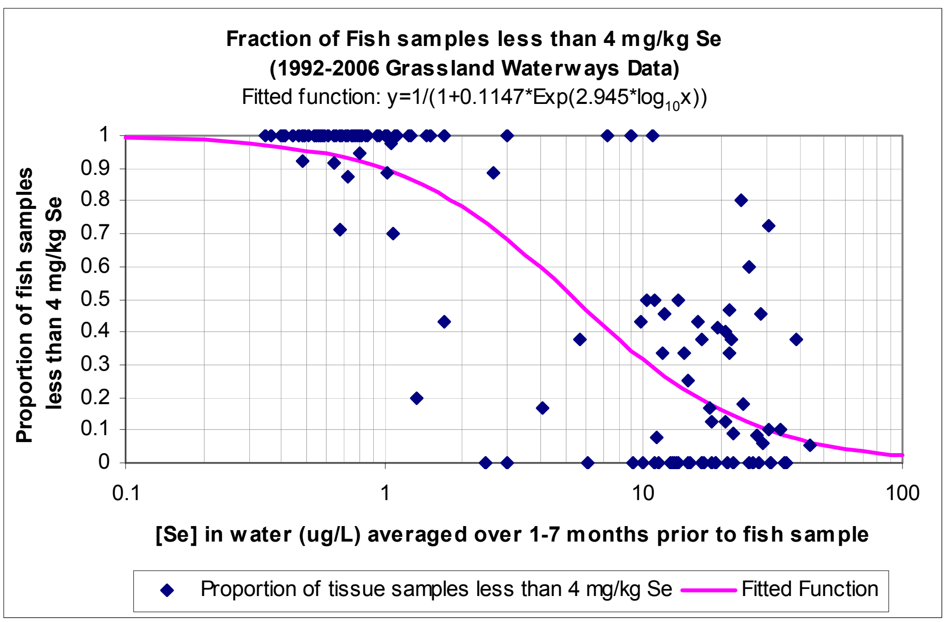

Figure E.2-4 The effect of selenium in water on the proportion of fish samples with selenium concentrations below the threshold of concern

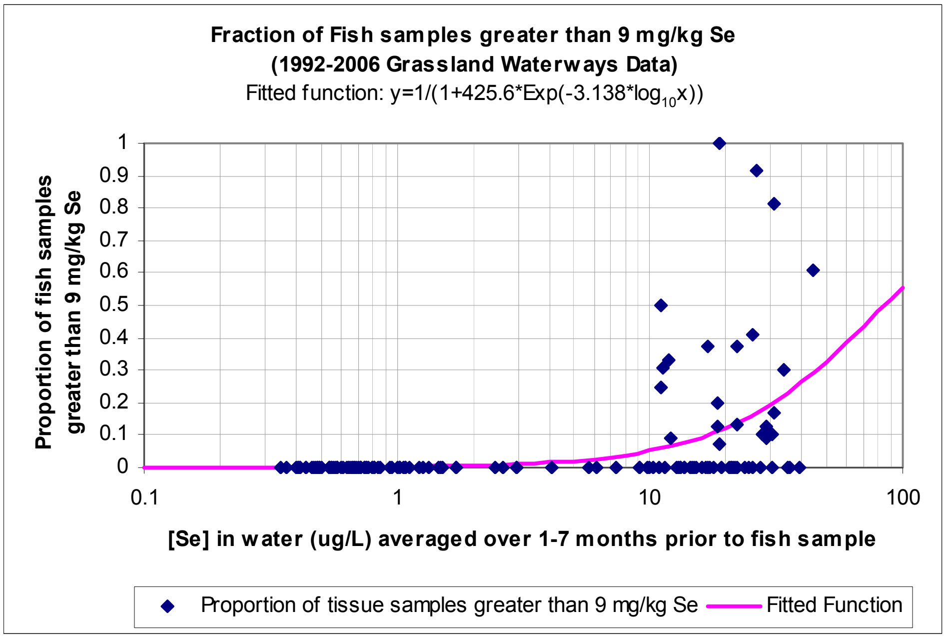

Figure E.2-5 **The effect of selenium in water on the proportion of fish samples with selenium concentrations above the threshold of toxicity**

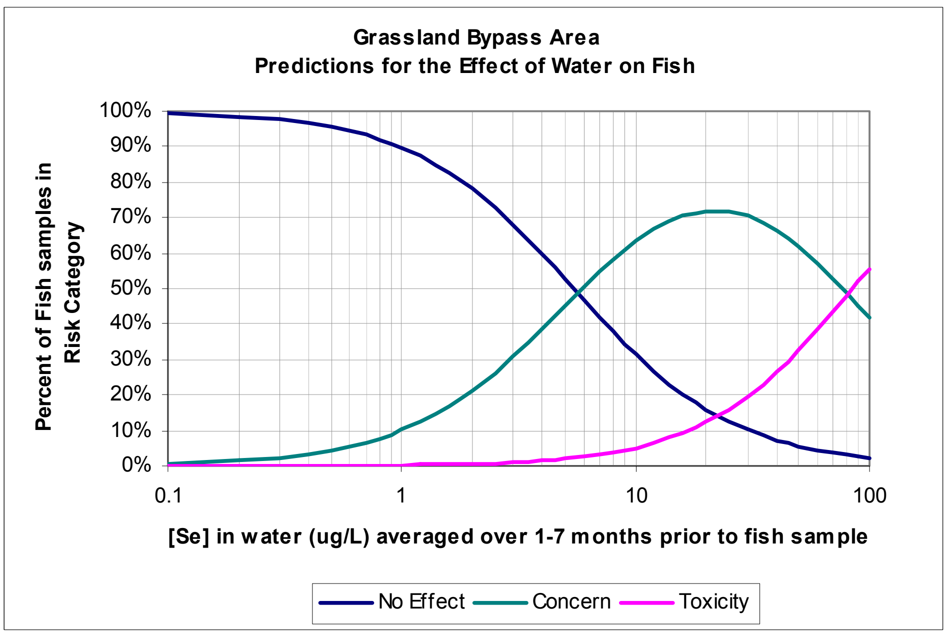

Figure E.2-6 **The project effects of selenium in water on proportions of composite samples of warmwater fish falling into fish classes, based on 1992 to 1999 data from Grassland area waterways**

E.2.2.2 Invertebrate Model

Grassland Bypass Project Monitoring Program data (1992–2006) suggest that the selenium levels in aquatic invertebrates as well as fish are broadly correlated with selenium concentrations in water (Figure E.2-7 and Figure E.2-8).

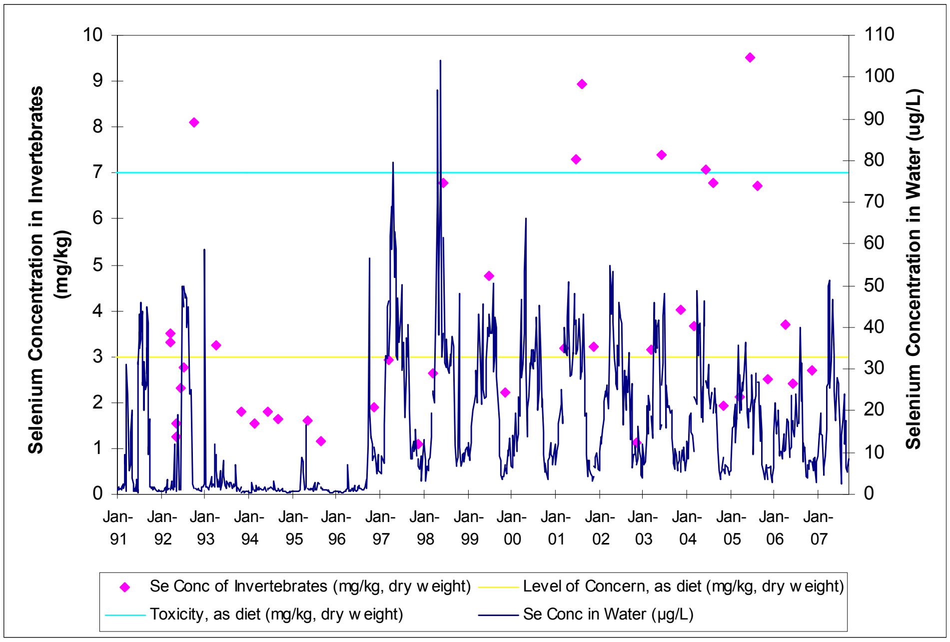

Figure E.2-7 Selenium in water and invertebrates (average of all species, mainly waterboatmen and red crayfish) in Mud Slough just below the outfall of the San Luis Drain

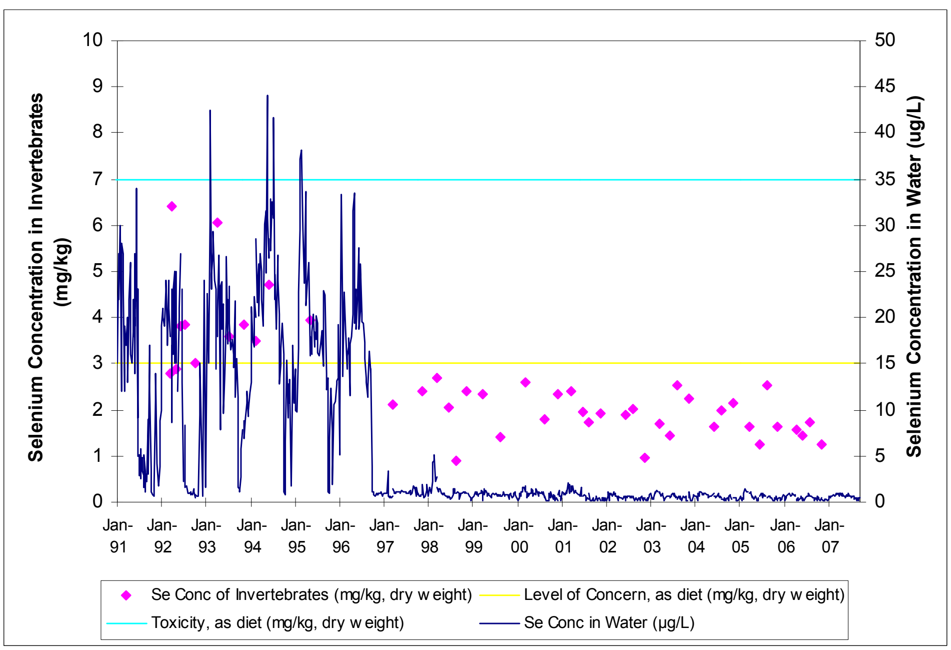

Figure E.2-8 Selenium in water and invertebrates (average of all species, mainly waterboatmen and red crayfish) in Salt Slough

As with fish, some lag time would be expected in the relationship. Generally, aquatic invertebrates are lower on food chains and have shorter life cycles than fish. Therefore, selenium concentrations in invertebrates would be expected to respond more rapidly than in fish to changes in water concentrations. In fact, an analysis similar to that done on fish (see above) confirms this expectation. Linear regressions were performed on average invertebrate selenium concentration versus water selenium concentration, successively using a selection of different water averaging time periods (Table E.2-3). Of the averaging time periods tested, the best predictor of invertebrate selenium concentration was 30 to 60 days prior to the time of collection of the invertebrate samples (Table E.2-3) in contrast to the one-to-seven-month water averaging time that best predicts selenium concentrations in fish.

Table E.2-3 Results of linear regressions of average selenium concentration in all species of invertebrates (mainly backswimmers and red crayfish) vs. selenium concentrations in water using various averaging periods for water (n=43)

Independent variable	Dependant variable	Proportion of variance explained by linear regression (r2)
Maximum water concentration 1-30 days prior to invertebrate sample	Average invertebrate concentration	0.47
Average water concentration 1-30 days prior to invertebrate sample	Average invertebrate concentration	0.52
Average water concentration 30-60 days prior to invertebrate sample	Average invertebrate concentration	0.68
Log_{10} of average water concentration 30-60 days prior to invertebrate sample	Log_{10} of average invertebrate concentration	0.52
Average water concentration 1-7 months prior to invertebrate sample	Average invertebrate concentration	0.42
Log_{10} of average water concentration 1-7 months prior to invertebrate sample	Log_{10} of average invertebrate concentration	0.48

E.2.3 Literature Cited

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