Development of Site-Specific Water Quality Criteria for the South Fork Coeur d'Alene River, Idaho

COMPARISONS OF CADMIUM CRITERIA TO THE RESULTS TOXICITY TESTING WITH SPECIES RESIDENT TO THE SOUTH FORK COEUR D'ALENE RIVER

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February 28, 2003

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1) Introduction

The purpose of this report is to compare results of streamside cadmium toxicity testing using species native to the South Fork Coeur d'Alene River, Idaho, to potentialcadmium criteria values. A main objective is to evaluate whether the site-specific test data better fit Idaho's existing ambient water quality criteria for cadmium, EPA's 2001 updated criteria, or some different site-specific criteria. This subject was considered in an earlier report (Mebane 2001); this report analyzes the site-specific data and criteria in greater depth.

The South Fork Coeur d'Alene River (hereafter "South Fork") site-specific criteria development project was intended to develop aquatic life criteria that were more ecologically relevant to the cold water organisms and waters of the watershed than were criteria that applied to the entire nation. The "resident species approach" of testing a diverse variety of organisms resident to the watershed in site water was used to develop criteria. This approach is intended to concurrently account for potential differences between 1) the toxicity of metals in site waters and the laboratory waters that make up much of the data in the national criteria; and 2) differences in the sensitivity of a variety of aquatic organisms occurring in the waters of interest and the sensitivities of organisms used in the national criteria (Carlson et al. 1984).

From 1995 to 2000, toxicity testing with cadmium, lead, and zinc was conducted in water from near the headwaters of the South Fork Coeur d'Alene River ("South Fork") watershed, using native species collected from the watershed. Test facilities were constructed on site in space provided by the Hale Fish Hatchery, near Mullan, Idaho. The South Fork Coeur d'Alene River has been pervasively disturbed by over a century of mining activities, however the Hale Hatchery is located upstream of disturbed areas.

Windward (2002), summarizes six years of site-specific toxicity testing, and derives proposed site-specific ambient water quality criteria for lead and zinc. Cadmium was also studied, but in earlier interpretations of site-specific toxicity test data, results with the most sensitive resident species (cutthroat trout, *Oncorhynchus clarki*) seemed similar to results listed in the national criteria database for acutely sensitive species (Chinook salmon, *O. tshawytscha*), as published in the 1984 national cadmium criteria (EPA 1984). Because of the similar responses, it was anticipated that further testing would ultimately result in site-specific cadmium criteria similar to existing criteria. It was decided to focus project resources on further testing to develop and evaluate site-specific criteria that were different from the national criteria. Consequently, site-specific criteria for cadmium were not developed as part of Windward (2002).

By spring 2001, most toxicity testing to develop the site-specific criteria, data analyses and criteria derivation for lead and zinc, and extensive reviews had been completed. At about that time, EPA released their "2001 Update of the Ambient Water Quality Criteria for Cadmium" (EPA 2001a). Because these updated national criteria became available, the site-specific cadmium data were re-evaluated here in more detail to determine whether revisions of the present criteria are indicated.

The Idaho statewide ambient cadmium criteria are based upon EPA's 1992 National Toxics Rule "NTR" (40 CFR 131.36) which was in turn based on EPA (1984). Both the

EPA 1984 and 2001 cadmium criteria are log-linear equations which for a given total hardness value predict threshold concentrations for preventing unacceptable long-term and short-term effects to fish and benthic assemblages in rivers and stream following episodic, short-term (1-hour to 4-day) cadmium exposures. Comparative criteria values over the range of hardnesses occurring in the South Fork are given in Table 1 for criteria maximum concentrations (CMC or "acute" criteria) and criteria continuous concentrations (CCC or "chronic" criteria). The NTR specifies that the cadmium criteria equations should only use hardness values between 25 and 400 mg/L as $CaCO_3$ (hereafter the "as $CaCO_3$" is omitted). However, for the purposes of comparing the NTR and 2001 equation results, these constraints are ignored in Table 1.

Table 1. Equations (top) and sample results of the Idaho (NTR) and EPA 2001 ambient water quality criteria for dissolved cadmium:

Cd criteria		NTR and EPA 2001 Cd CMC "acute" and CCC "chronic" Equations (µg/L)
"acute"	NTR	$CMC=1.1367-(\ln(hardness) \times 0.041838) \times e^{(1.128 \times \ln(hardness)-3.828)}$
	2001	$CMC=1.1367-(\ln(hardness) \times 0.041838) \times e^{(1.0166 \times \ln(hardness)-3.924)}$
"chronic"	NTR	$CCC=1.101672-(\ln(hardness) \times 0.041838) \times e^{(0.7852*LN(hardness)-3.49)}$
	2001	$CCC=1.101672-(\ln(hardness) \times 0.041838) \times e^{(0.7409*LN(hardness)-4.719)}$

	CMC (µg/L)					CCC (µg/L)				
Hardness (mg/L):	5	10	25	50	100	5	10	25	50	100
NTR	0.13	0.29	0.82	1.7	3.7	0.11	0.18	0.37	0.62	1.0
2001	0.10	0.20	0.51	1.0	2.1	0.02	0.05	0.09	0.15	0.25

a) Site-specific criteria development

Two approaches were used here to compare the site-specific test data to the NTR and EPA 2001 criteria predictions. First, effects from site acute and chronic toxicity tests were analyzed and plotted against criteria concentrations to "eyeball" whether criteria concentrations appear acceptable. Second, site data were quantitatively taken through the criteria derivation calculations to see whether calculated site-specific criteria reasonably matched either the NTR or 2001 cadmium equations, or whether a separate site-specific South Fork cadmium criteria was warranted.

The summarized steps used in the project to develop criteria using the resident species approach generally included (Carlson et al. 1984; Stephan et al. 1985; Windward 2002):

1. Identifying resident species to test that are representative of the benthic invertebrate and fish assemblages of the South Fork.
2. Conducting toxicity tests with a diverse variety of wild fish and benthic invertebrates in rangefinding exposures to identify the most sensitive species by ranking their

acute values, LC50s[1]. Several aquatic invertebrates were tested and the only two native fish species present in the upper, unpolluted portions of the South Fork watershed were tested (i.e., shorthead sculpin and cutthroat trout). Cutthroat trout were the most acutely sensitive species to cadmium.

3. A broodstock of adult cutthroat trout captured from the upper South Fork was established. This allowed a source of cutthroat fry for testing that were of a known age, and risk of pre-exposure and acclimation to ambient metals, and avoided depleting an at risk population of native cutthroat trout.
4. Using the most sensitive species (i.e., cutthroat trout), more definitive acute toxicity tests were completed to determine mean acutely toxic values, referred to as the species mean acute value, or SMAV. The SMAV for the most sensitive species was used as the final acute value (FAV).
5. Tests were conducted at different times and with different water sources to develop hardness-toxicity relationships to and hardness-based criteria equations. Rainbow trout were used as a surrogate for resident westslope cutthroat trout in some of these tests of relative toxicity of different waters.
6. Using rainbow trout, *Oncorhynchus mykiss*, as a surrogate for cutthroat trout, paired acute and chronic tests to calculate acute/chronic ratios (ACR). Acute/chronic ratios are a means to relate the acute and chronic toxicities of a material. Because of limited availability of native cutthroat trout for testing, the closely related rainbow trout were used.
7. The derive the acute criteria that is safe for short term exposures, the FAV was divided by 2 to extrapolate from a value that on the average is lethal to 50% of a sensitive species a value that has few if any acute effects. To estimate a chronic value that is protective for long-term exposures, the FAV is divided by the average acute/chronic ratio.

More details on these steps are given in following sections.

2) Acute site-specific toxicity testing

Acute toxicity test results of cadmium in site water are plotted with the NTR and EPA 2001 criteria curves in Figure 1. 19 acute tests (96-hour) with eight native species were conducted. Rainbow trout were also tested as a surrogate for resident cutthroat trout to evaluate hardness-toxicity relationships, to test for spatial or temporal variability in toxicity, and to relate acute to chronic responses. Rainbow trout occur in the South Fork, but are considered an undesirable exotic species in the South Fork because they are invasive and displace or interbreed with native Westslope cutthroat trout. Thus they were not considered a "resident" species for criteria development (Windward 2002). A

[1] LC50 is the estimated test concentration that was lethal to 50% of the test organisms. All acute values from the site-specific testing would more precisely be called EC50s for the 50% effects concentration, since effects other than lethality such as immobilization were measured. However, LC50 is a more commonly used term, so it is used here too.

literature value (Stubblefield 1990) for mountain whitefish, *Prosopium williamsoni*, was also included in Figure 1 since mountain whitefish are a native species in the South Fork, but could not be tested because they were uncommon in the uncontaminated upper reaches where test organisms were collected. Also shown are the LC50s from testing the toxicity of cadmium + zinc mixtures at approximately equi-toxic ratios to cutthroat trout (Dillon and Mebane 2002).

The acute values for cutthroat trout, rainbow trout, and shorthead sculpin fall close to each other on the plot, suggesting generally similar sensitivities to cadmium for these species in stream water (Figure 1). As predicted by the criteria equations, a general pattern of increasing toxicity (lower LC50s) with decreasing hardness is apparent, and appears to hold to even very low hardnesses (7.5 mg/L). Two tests, one with hatchery cutthroat and one with rainbow trout at a hardness of 21 mg/L plot below the other cutthroat and rainbow trout data. These outlying results are from tests using fry from the Idaho Department of Fish and Game's Sandpoint Hatchery, a different source from all the other tests with rainbow trout which used fish from Mount Lassen Trout Farms. The resident cutthroat trout used in the other tests were progeny of the South Fork broodstock, or were field collected.

The toxicity of Cd+Zn mixtures to cutthroat trout, plotted as cadmium LC50 values follow a similar hardness-toxicity pattern as the tests with cadmium alone. This was also the case when these test results were plotted as zinc LC50s and compared with single metal zinc tests (data not shown). These observations are similar to other tests of cadmium and zinc mixtures. In tests of chronic toxicity of Cd+Zn mixtures to flagfish, the mixture toxicity was similar to the toxicity of zinc alone (cadmium was not tested singly) and uptake of one metal was not influenced by the presence of the other (Spehar et al. 1978b). The LC50s of cadmium and zinc in mixtures to rainbow trout and bull trout, *Salvelinus confluentus*, were also similar to LC50s of the metals individually (Hansen et al. 2002a).

The LC50s of the three resident cutthroat trout tests fall close to the NTR acute criterion values in Figure 1. In other words, criterion values that are intended to result in little if any toxicity to sensitive species instead resulted in unacceptable toxicity (up to 50% mortality). All LC50 values except the two tests using Sandpoint Hatchery fish plotted above the EPA 2001 acute criterion values.

Another way to estimate whether the NTR or the EPA 2001 criterion maximum concentrations achieve the intended balance between being under or over protective, is whether on the average, acute values for sensitive species such as cutthroat trout are ≥2X the criterion value. This comparison is used because in criteria development, a final acute value (FAV) is the LC50 value for sensitive species. However, a value lethal to 50% of a sensitive species cannot be considered protective of that species. Dividing the final acute value by two is intended to extrapolate from a concentration that is lethal to 50% of the test organisms to a concentration expected to kill few if any organisms (Stephan et al. 1985).

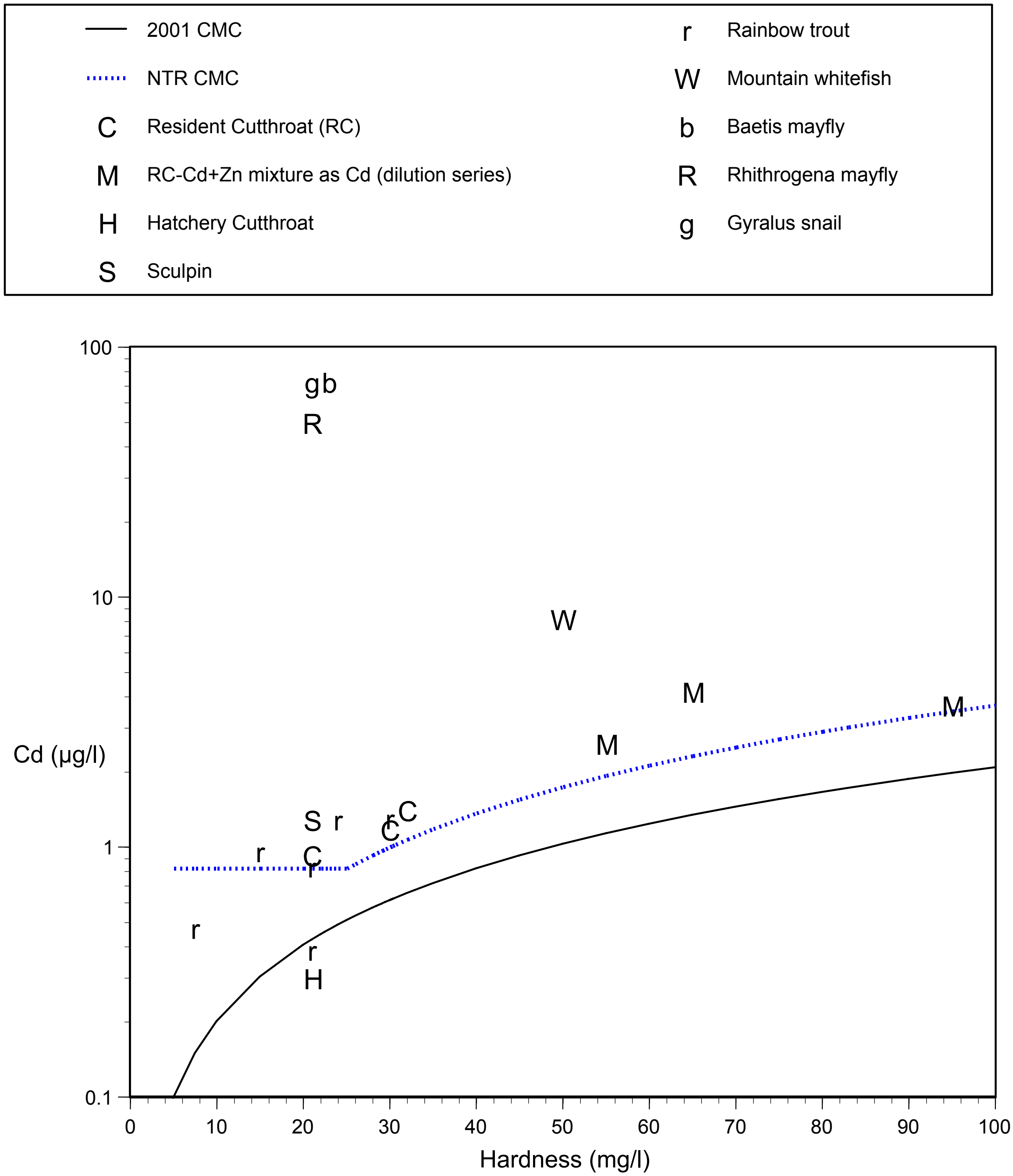

Figure 1. Acute values from site specific testing compared with the current Idaho acute cadmium criterion ("NTR CMC"), and EPA's 2001 acute cadmium criterion ("2001 CMC"). Data from Windward (2002), except for the literature value for mountain whitefish (Stubblefield 1990) which is shown since the species occurs in the study area and no site specific data are available.

Using this guide, the acute cadmium values from the site-specific testing with resident species were divided by the NTR and the EPA 2001 acute cadmium criteria (Table 2). Of the tests with South Fork-resident fish species, (i.e., sculpin and cutthroat trout), none of the four acute values were ≥2X the NTR acute cadmium criterion. In contrast, the ratios of the acute values for resident fish species divided by the EPA 2001 acute cadmium criterion ranged from 1.9 to 3.0. Hence the EPA 2001 acute cadmium criterion appears to be sufficiently protective for sensitive native fish, and does not overpredict toxicity (is not overprotective).

Table 2. Comparison of current Idaho (NTR) and 2001 acute cadmium criteria with site-specific acute toxicity test values. Concentrations less than ½ the LC50 concentration are expected to kill few if any individuals and are generally considered safe. Shaded values indicate LC50s of resident taxa that were <2X greater than the acute criteria. Rainbow trout and cutthroat trout from outside the South Fork drainage are not considered "resident" species for the purposes of establishing acute values. Test data are from Windward 2002.

Common Name	*Scientific Name*	Test Date	Test Hardness (mg/L as $CaCO_3$)	Test Acute LC50 (µg/L)	NTR CMC	LC50/NTR CMC	2001 CMC	LC50/2001 CMC
Caddisfly	*Arctopsyche* sp.	10/22/95	nm (~20)	>458	0.82	559	0.40	1145
Mayfly	*Rhithrogena* sp.	9/11/96	21	>50	0.82	61	0.42	118
Mayfly	*Baetis tricaudatus*	9/11/96	21	>73	0.82	89	0.42	172
Snail	*Gyraulus* sp.	9/11/96	21	>73	0.82	89	0.42	172
Stonefly	*Perlodidae sp*	10/22/95	nm (~20)	>5000	0.82	6098	0.40	12500
Stonefly	*Sweltsa* sp.	12/9/95	nm (~20)	>5,130	0.82	6235	0.40	17004
Shorthead sculpin	*Cottus confusus*	9/10/96	21	1.29	0.82	1.6	0.42	3.0
Rainbow trout (Mount Lassen hatchery)	*Oncorhynchus mykiss*	10/24/97	21	0.84	0.82	1.0	0.42	2.0
Rainbow trout (Mount Lassen hatchery)	*Oncorhynchus mykiss*	5/23/99	7.5	0.48	0.82	0.6	0.15	1.6
Rainbow trout (Mount Lassen hatchery)	*Oncorhynchus mykiss*	5/23/99	24	1.30	0.82	1.6	0.49	2.7
Rainbow trout (Mount Lassen hatchery)	*Oncorhynchus mykiss*	5/23/99	30	0.99	1.00	1.0	0.61	1.6
Rainbow trout (Mount Lassen hatchery)	*Oncorhynchus mykiss*	5/23/99	13.5	0.97	0.82	1.2	0.27	3.2
Rainbow trout (Mount Lassen hatchery)	*Oncorhynchus mykiss*	10/2/99	32	0.89	1.08	0.8	0.65	1.4
Rainbow trout (Mount Lassen hatchery)	*Oncorhynchus mykiss*	7/26/00	28.5	0.83	0.95	0.9	0.58	1.4
Rainbow trout (Sand Point hatchery)	*Oncorhynchus mykiss*	9/10/96	21	~0.39	0.82	0.5	0.42	0.9
Cutthroat trout (resident)	*Oncorhynchus clarki lewisi*	8/26/99	32	1.41	1.08	1.3	0.65	2.2
Cutthroat trout (resident)	*Oncorhynchus clarki lewisi*	8/9/00	30.5	1.18	1.02	1.2	0.62	1.9
Cutthroat trout (resident field-collected young-of-year)	*Oncorhynchus clarki lewisi*	9/10/96	21	0.93	0.82	1.1	0.42	2.2
Cutthroat trout (Sand Point hatchery)	*Oncorhynchus clarki lewisi*	9/10/96	21	~0.30	0.82	0.4	0.42	0.7

3) Acute site-specific criteria evaluations

The previous qualitative comparisons suggest that the site-specific test data would be congruent with the EPA 2001 acute cadmium criterion. The following is a more quantitative treatment of the site-specific data using EPA's national procedures for deriving numerical water quality criteria for the protection of aquatic organisms and modifications for site-specific criteria (Carlson et al. 1984; Stephan et al. 1985). These detailed toxicological and mathematical procedures provide a process to estimate thresholds of toxicity that would protect aquatic ecosystems from exposure to a substance. An important part of the process is the calculation of an overall acute value, or final acute value (FAV) that is derived from all the mean acute values (MAVs) available for different species that are available for a chemical. The MAVs could be calculated at any taxonomic level, but are usually expressed as species mean acute values (SMAVs), genus mean acute values (GMAVs), or family mean acute values (FMAVs). GMAVs are the specified taxonomic level for calculating FAVs because for a given substance, differences in sensitivities of species within a genus are usually smaller than are differences between genera The FAV is defined to be lower than all but a small fraction of the MAVs that are available for a chemical. The fraction was set at 0.05, i.e. the FAV lies at the fifth percentile of the statistical population represented by the set of MAVs available for a chemical. This fraction was selected because other fractions resulted in FAVs that were deemed too high or too low in comparison with the dataset from which they were obtained (Stephan et al. 1985). However, the procedures specify that if the set of MAVs contained a MAV for an important species that was lower than the calculated FAV, the FAV should be reduced to be equal to that MAV. For example, in EPA's 2001 acute cadmium criterion development, the FAV was calculated to be 2.763 µg/L, but was lowered to 2.108 µg/L to protect the commercially important rainbow trout (EPA 2001a; Stephan et al. 1985). While not explicitly stated, further rationale inferred for the selection of the 5^{th} percentile of available MAVs to set a FAV, was to predict toxicity thresholds that would protect the diversity of taxa in benthic macroinvertebrate, fish, and other aquatic assemblages, without over-predicting toxicity based on extreme values that might be in a set of GMAVs available for a substance.

The diversity of aquatic animals in North America is much larger than the diversity of organisms for which are represented in toxicity testing datasets. No overall estimates of North American aquatic diversity were reviewed, but since Idaho alone has >1200 stream macroinvertebrate taxa and 129 fish species records (IDEQ Beneficial Use Reconnaissance Program database), it seems likely that in North America somewhere in the order of >10,000 genera of aquatic fauna occur. In contrast, the number of GMAVs available in the EPA ambient water quality criteria documents is usually <50. In part because of this disparity, the guidelines for calculating a FAV also specify the minimal diversity needed in a dataset of eight families which contain representatives of certain types of taxa that are often important in freshwater ecosystems. These include at least one value for a member of each of the following families: 1) a salmonid, 2) a fish from a 2^{nd} family, 3) another vertebrate such as an amphibian or a 3^{rd} family of fish, 4) a planktonic crustacean, 5) a benthic crustacean such as a crayfish or amphipod, 6) an insect such as a mayfly, stonefly, caddisfly or midge, 7) a family from other than Chordata or Arthropoda such as a mollusc or annelid worm, and 8) another insect or animal from a phylum that is not already represented (Stephan et al. 1985). The calculation of a FAV from a small set of GMAVs meeting the above minimum diversity requires considering that the set is a random sample from a statistical population, and that the FAV is an estimate of the 5th percentile of that population. Erickson and Stephan (1988) analyzed available datasets and developed equations for estimating the 5th percentile of a GMAV "population" from relatively small datasets with minimal bias. They found that a triangular-shaped distribution of the "population" best fit the available data and agreed with ecotoxicology concepts. In a triangular distribution, mid-range toxicity values are most common, forming the peak in the triangular-shaped distribution, and values for highly resistant or highly sensitive taxa are rare, forming the bottom corners of the triangular distribution. A triangular distribution is similar in concept to the familiar bell-curve of a normal distribution, but instead of being described by curves, is described with straight lines which give a peak in the center of the distribution instead of a hump, and upper and lower limits to the distribution are given by definite corners, instead of tailing off. Depending upon the GMAV "sample size" and the distribution of the "sample" values, the 5th percentile estimate will either be interpolated between low values in the GMAV "sample" or be extrapolated below the lowest value in the "sample" (Erickson and Stephan 1988).

The guidelines for deriving a site-specific FAV using the resident species approach are similar to the national guidelines, except that if species from all eight of the specified groups could not be tested because they do not occur at a site, then other sensitive taxa should be tested until the eight family minimum is met. If all taxa at a site have been tested, and the eight family minimum is not met, then the most sensitive SMAV should be used as the FAV (Carlson et al. 1984). In the South Fork, the specified groups could not be tested because of low diversity. Upstream of significant mining pollution, only two families of fish are present, but neither a third vertebrate species, benthic crustaceans or planktonic crustaceans were present in sufficient numbers to be feasible to test. Members of two amphibian families, the tailed frog *Ascaphus truei* and the Pacific giant salamander *Dicamptodon ensatus* have been collected in the study area. However, the amphibians were scarce and available data on cadmium or other metals toxicity to amphibians indicated amphibians probably were among the more resistant taxa (EPA 2001a; EPA 2001b). At least 60 families of benthic macroinvertebrate families have been collected in the study area. Additional insect species were tested with cadmium to meet the eight resident-family minimum diversity. (EVS 1995; Windward 2002). The toxicity test results listed in Table 2 were normalized to estimate values at a hardness of 50 mg/L so that the resident taxa may be ranked by their relative acute sensitivity to cadmium (Table 3).

Table 3. Ranked resident species mean acute values (SMAVs) normalized to a hardness of 50 mg/L using the hardness-toxicity slope from EPA (2001a).

Rank	Common Name	*Latin Name* or Family	SMAV (µG/L)
8	Stonefly	*Perlodidae sp.*	12,400
7	Stonefly	*Sweltsa* sp.	12,400
6	Caddisfly	*Arctopsyche* sp.	1106
5	Mayfly	*Baetis tricaudatus*	176
4	Snail	*Gyraulus* sp.	176
3	Mayfly	*Rhithrogena* sp.	120
2	Shorthead sculpin	*Cottus confusus*	3.1
1	Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	2.1

Using the FAV equation (Equation 1), the 5th percentile GMAV was estimated as 0.17 µg/L Cd at a hardness of 50 mg/L (Table4).

Equation 1. Final acute value (FAV) calculations (Erickson and Stephan 1988; Stephan et al. 1985)

$$P = R/(n+1)$$

$$S = \sqrt{\frac{\sum((\ln \text{GMAV})^2 - ((\sum(\ln \text{GMAV}))^2/4)}{\sum(P) - ((\sum(\sqrt{P}))^2/4)}}$$

$$A = S(\sqrt{0.05}) + L$$

$$FAV = e^A$$

Table 4. FAV equation calculation results using South Fork resident species GMAVs.

Rank	Resident Genera	GMAV – Genus mean chronic value (µg/L)	Cumulative Probability (P=R/(n+1)	ln GMAV	(ln GMAV)^2	SQRT P
4	Snail, *Gyraulus*	176	0.444	5.170	26.734	0.667
3	Mayfly, *Rhithrogena*	120	0.333	4.787	22.920	0.577
2	Sculpin, *Cottus*	3.1	0.222	1.131	1.280	0.471
1	Trout, *Oncorhynchus*	2.1	0.111	0.742	0.550	0.333
Sum:			1.111	11.831	51.485	2.049

Number of GMAVs = 8
S=4.316
L= -1.472
A= -0.5070
FAV= 0.17 µg/L

The SMAV for the most sensitive species tested, Westslope cutthroat trout, is 12X higher than the theoretical 5th percentile GMAV (SMAVs=GMAVs since only one species per genus was tested). All the resident fish species were tested, and there were large difference in sensitivity between the less sensitive fish and the next most sensitive taxa (~40X less sensitive). This contraindicates the assumption that the FAV equation estimates the 5th percentile GMAV of the population of resident taxa from the sample of resident taxa tested. Instead, the most sensitive species in the sample tested (cutthroat trout) probably represents the most sensitive species in the population, and that SMAV should be used as the site FAV. This is similar to the conclusions of EPA (2001a) in updating the national acute cadmium criterion, which passed over the calculated FAV results in favor of adjusting the criterion to match the rainbow trout SMAV. The resident cutthroat trout SMAV obtained from site-specific testing is almost identical to the rainbow trout SMAV that was used to adjust the FAV in the national cadmium criterion (Table 4). Rainbow trout were used in some South Fork site-specific criteria testing as a surrogate for the resident cutthroat trout. Rainbow trout mean acute values from testing in site water were also almost nearly identical to the national rainbow trout SMAV.

Table 5. Comparison of potential cadmium FAV results based on the most sensitive SMAV or the FAV equation in site-specific and national data sets. Site-specific values were adjusted to a hardness of 50 mg/l using EPA's (2001a) pooled hardness slope.

Species	Genus	SMAV	GMAV	FAV by equation	FAV from MSS SMAV
South Fork resident cutthroat trout mean acute values	*Oncorhynchus*	2.13	2.13		2.13
Mount Lassen rainbow trout mean acute values from site-specific testing	*Oncorhynchus*	2.17			
EPA 2001 Rainbow trout SMAV	*Oncorhynchus*	2.11			
EPA 2001 genus *Oncorhynchus* value (rainbow trout, chinook salmon, and coho salmon tests)	*Oncorhynchus*		3.84		2.11
FAV equation results – EPA 2001				2.76	
FAV equation results – Table 4				0.17	

Once the final acute value is determined, it is divided by two to extrapolate from a value that on the average is lethal to 50% of the individuals a population of the most sensitive species to a criterion maximum concentration (CMC) that is lethal to few if any individuals (Stephan et al. 1985). For the six acute test with resident cutthroat trout and cadmium or cadmium and zinc mixtures, dividing their LC50 values by two corresponded to <LC1 to LC15 values, calculated by Probit analysis (EPA 1992).

For metals, criteria are often expressed as hardness-dependent equations. Site-specific testing was conducted using different test waters and at different times of the year to include a range of hardnesses. The hardnesses tested, 7.5 – 32 mg/L (Table 2), included values that were beyond the midrange of hardness (20 – 200 mg/L) where the hardness-toxicity relationship is best explained aquatic chemistry – fish physiology mechanisms (Meyer 1999). The South Fork testing also included hardnesses lower than those previously evaluated (EPA 2001a). Regression analysis of the natural log acute values with Mount Lassen rainbow trout (dependent variable) with the natural log of hardness suggested the hardness-toxicity relationship continues in natural waters with low hardnesses (slope 0.400, r^2 0.48, P=0.08, n=7). This slope is lower than the slopes of ~1 that usually result from midrange hardness-toxicity regressions with divalent transition metals, supporting Meyer's (1999) theoretical argument that extrapolating hardness-toxicity relationships below midrange hardness values could drastically overpredict toxicity (Meyer 1999).

The range of hardnesses tested, 7.5 – 32 mg/L, reflects hardnesses commonly occurring in headwater areas of the South Fork watershed. However, hardnesses up to ~135 mg/L were measured in the lower South Fork Coeur d'Alene River (Dillon and Mebane 2002). At these higher hardnesses that occur in the lower South Fork watershed, the national acute cadmium criterion probably provides a better estimate of hardness-toxicity relationships than would extrapolating the relationship obtained from site testing at the low end of the hardness range. The cadmium toxicity-hardness relationship for the national acute criterion is based on 64 selected tests, the vast majority of which were from hardnesses of 20 – 200 mg/l, providing a slope of 1.01 (EPA 2001a). A site-specific cadmium criterion equation could be constructed as a hockey-stick shaped equation, with the lower slope at the low end of the hardness range (e.g. <≈ 25 mg/L hardnesses) and the higher slope at midrange hardness (e.g. 25 – 200 mg/L). However, this would be novel, and if regulated discharges are usually into receiving waters with midrange hardness, the overprediction of toxicity at low hardnesses may not be of much practical significance. Thus the simpler, national equation is probably sufficient for such cases.

In conclusion, the results of a quantitative treatment of the acute toxicity data following the framework of Carlson et al. (1984) and Stephan et al. (1985), lead to the same conclusion as did "eyeballing" the site-specific data and the criteria curves in Figure 1 – that the EPA 2001 acute criterion equation is a reasonable fit with the South Fork site-specific data.

4) Chronic site-specific cadmium testing

Two early-life stage (ELS) tests with rainbow trout in site water were conducted as part of this project (EVS 1997; Windward 2000). Rainbow trout were used in chronic testing as a surrogate for cutthroat trout because the numbers of resident cutthroat fry were limited. Rainbow and cutthroat trout had very similar acute sensitivity to cadmium (Table 4) and the species are so physiologically similar that they often interbreed (Behnke 1992). Thus the chronic toxicity of cadmium to rainbow and cutthroat trout is likewise probably similar. The objective of early-life stage testing was to expose the trout to cadmium continuous exposure, beginning before hatch, and ending after hatch, using the flow-through technique. Early-life stage tests are used in criteria development because they are generally useful estimates of comparable with life cycle tests with the same species (ASTM 1998; McKim 1985; Stephan et al. 1985). Acute tests were conducted concurrently with the same cohort of fish in the same dilution in order to estimate acute/chronic ratios from which chronic toxicity to other species might be estimated.

The 1997 test was conducted over a 69-day exposure from August 21 – October 27, 1997 using water from the Little North Fork of the South Fork Coeur d'Alene River (LNF). The test was initiated with six treatments with measured average concentrations ranging from 0.3 – 15 µg/L, plus a control, with three replicates per treatment. Two exposure upsets occurred during the test, otherwise the test followed ASTM (1998). On day 16 of the test, a dosing error resulted in overdosing all treatments by about 4X for about three days. A small increase in mortality (about 5 ~ 10 %) appeared to occur following the overdose (Figure 2). The fish were just beginning to hatch at the time. Eggs and newly hatched alevins have been found to be much more resistant to toxicity from cadmium or other metals than later juvenile forms of salmonids. The swim-up stage was more sensitive than earlier or later life stages (Chapman 1978a; Chapman 1978b; Chapman 1994). Because of this, the concentrations during that time were excluded from the time-weighted average exposures in the interpretation of later results (Figure 2, Table 5). Otherwise, the averages would be biased high, in comparison to the later concentrations experienced by the more sensitive swim-up fry.

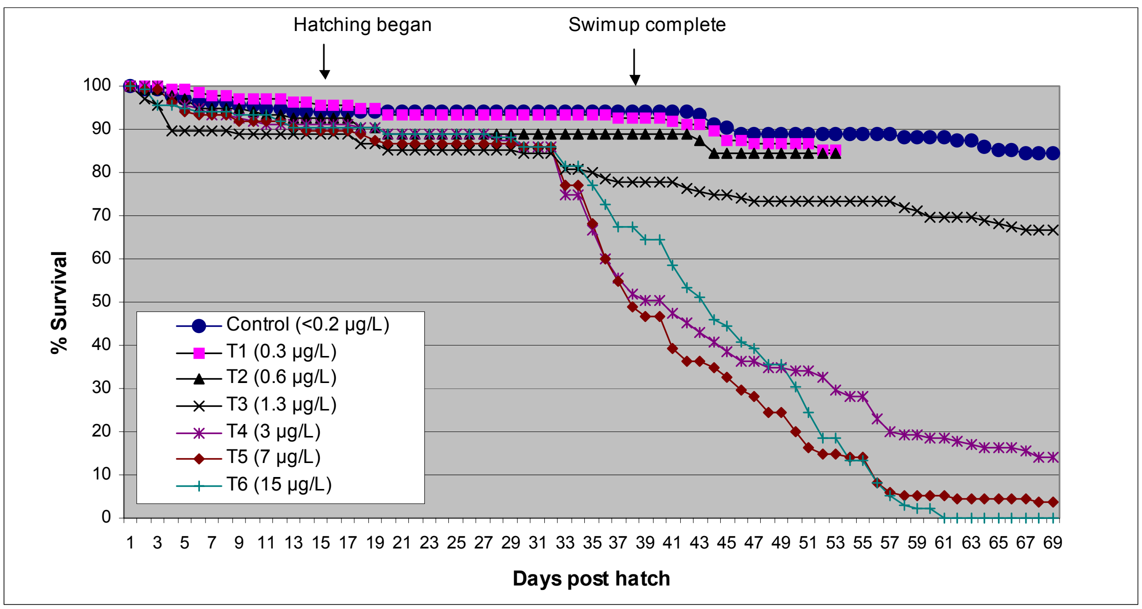

Figure 2. Progressive mortality of rainbow trout in early-life-stage testing with cadmium, August 21 - October 27, 1997. On day 53, the 0.3 µg/L and 0.6 µg/L treatments were overdosed resulting in complete mortality.

No appreciable mortality at any concentration occurred until day 32 of the test, as alevins began absorbing their yolk-sacs and becoming free-swimming. At this time, fry began to die off in the three high cadmium treatments, 3 – 15 µg/L. (Figure 2). A noticeable increase in mortality in the mid-range (1.3 µg/L) treatment was also apparent. No mortality occurred at that time in the control or the two low cadmium treatments (0.3 and 0.6 µg/L). By day 45, about seven days after swim-up was complete, mortality had plateaued in the 1.3 µg/L mid-range treatment. On day 53, a second upset resulted in overdosing the two low-cadmium treatments (0.3 and 0.6 µg/L treatments). At this stage of the test (~14-days past completion of swim-up), the fish were likely at their most sensitive stage (Figure 2). All fish in all replicates of these two treatments died within one-day of the overdose.

The test results were analyzed two ways – after 53 days cadmium exposure with all treatments, and after 69 days exposure with the remaining four treatments (Table 6). In the 1.3 µg/L mid-range cadmium treatment, mortalities were similar after 53 and 69 days exposure, 16% and 18% higher than control mortality, respectively (P=0.4 by Student's one-sided t-test). It seems reasonable to assume the mortality in the two lower treatments also would have remained proportional to the control mortality. Thus, despite the interruption of the planned exposures, the 53 days exposure seems sufficient to estimate effects of cadmium on an early life stage growth and survival of rainbow trout. This provided ~20 days exposure beyond the rapid onset of mortality that happened as most swim-up occurred, and 14-days exposure after all swim-up was complete. Mortality had plateaued in the low and mid-range cadmium treatments about 8 days before the low treatments were lost, suggesting ample exposure time had occurred to evaluate early-life stage toxicity.

These observations are supported by a recent 55-day juvenile growth test of bull trout swim-up fry with cadmium (Hansen et al. 2002b). Mortality from cadmium was essentially ended within the first 5 days of cadmium exposure, and the observed mortality was similar to that observed in concurrent acute toxicity tests with cadmium and swim-up bull trout fry. Similarly, in 8-day exposures, Chapman (1978) found little or no increases in mortality from exposures of juvenile steelhead or chinook salmon beyond 4-days exposure to cadmium.

Table 6. Responses of rainbow trout exposed to cadmium in early life stage testing, 1997, 69 day test (see note), averages of three replicates. Underlined values were significantly lower than control means at P <0.05, by Dunnetts's test.

Cadmium treatments, mean dissolved cadmium µg/L (± SD)	Control (< 0.2)	T1 0.3 (0.05)	T2 0.6 (0.12)	T3 1.3 (0.20)	T4 2.9 (0.47)	T5 6.9 (0.77) b	T6 15 (2.7)	Minimum detectable difference from control (% of control)
Mortality – 53 days (%)	11.1	14.8	15.6	**<u>26.7</u>**	**<u>70.4</u>**	**<u>85.2</u>**	**<u>81.5</u>**	+15.5% (17.5%)
Mortality – 69 days (%)	15.6	a	a	**<u>33.3</u>**	**<u>86</u>**	**<u>96</u>**	**<u>100</u>**	+13.4% (15.8%)
Wet weight – 53 days (g)	0.218	0.208	0.219	a	a	a	a	-0.048 g (22.0%)
Wet weight – 69 days (g)	0.395	a	a	0.342	**<u>0.222</u>**	**<u>0.146</u>**	All dead	-0.092 g (23.3%)
Biomass – 69 days (g)	13.27	a	a	**<u>10.2</u>**	**<u>1.3</u>**	**<u>0.049</u>**	All dead	-0.93 g (7.0%)
Length – 69 days (mm)	33.8	a	a	**<u>31.7</u>**	**<u>28.36</u>**	**<u>25</u>**	All dead	-1.27 mm (3.8%)

Mean hardness 21 (range 20-21), mean pH 6.76 ± 0.34 SD

a – No data. Exposures 1 and 2 (0.3 and 0.6 µg/L treatments) were overdosed on day 53, causing complete mortality in those treatments.
b – Only one fish surviving at 69 days, so treatment excluded from Dunnett's test.

A second early-life stage test with rainbow trout was conducted with more treatments at low cadmium concentrations and used a different water source. The 1999 test was conducted over a 62-days from July 23 – September 22, 1999 using water from the Hale Fish Hatchery intake from the South Fork Coeur d'Alene River. The test was initiated with five treatments with average measured concentrations of 0.2 – 2.5 µg/L, plus a control, with three replicates per treatment (Windward 2000).

Noticeable mortality in the highest treatment began about day 23 of the test. As with the 1997 test, this was a few days before swim-up was complete as most of the alevins absorbed their yolks and became free swimming. Mortality at the highest treatment essentially ended about 5 days after swim-up was complete (~day 32). Mortality at lower exposures (0.2 to 1.0 µg/L) were indistinguishable from control mortalities (Figure 3).

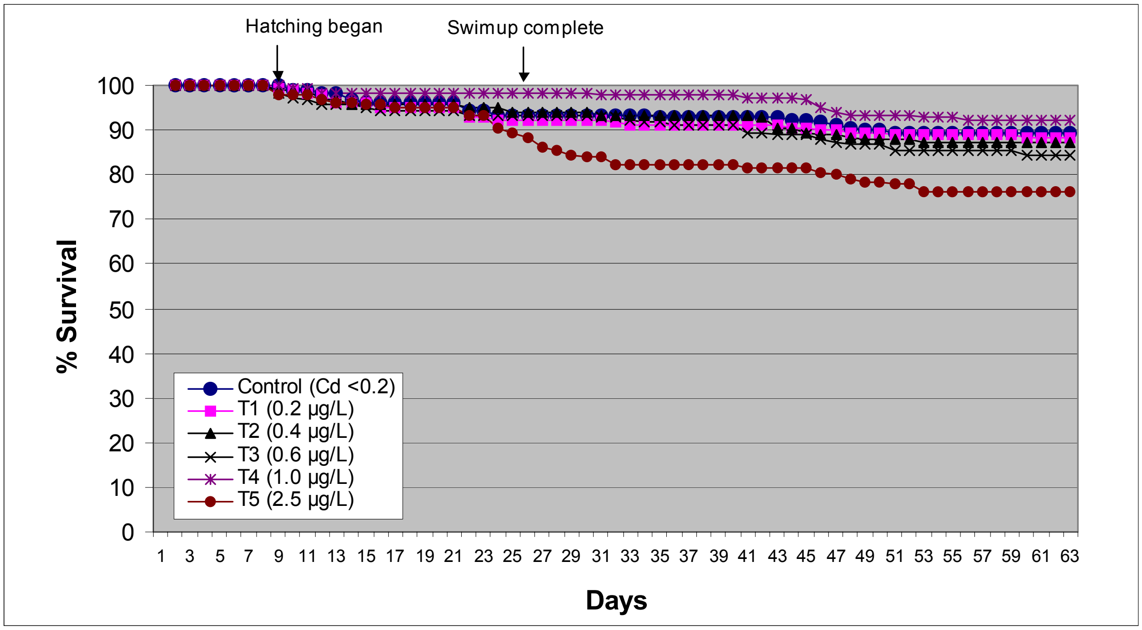

Figure 3. Progressive mortality of rainbow trout in early-life-stage testing with cadmium, July 22 - September 22, 1999.

On day 43 of the test a problem with the dosing developed and cadmium concentrations decreased in all treatments. This occurred about 16 days after the most sensitive life stage had been exposed (swim-up fry) and about 10 days after mortalities ended (Figure 3). The mean treatment exposures listed in Table 7 were calculated as time-weighted averages over the entire test. For example, the time-weighted average concentration in treatment 5 from day 1 to day 43 was 3.5 µg/L, whereas the time-weighted average for the entire 62 days was 2.5 µg/L. This is a conservative interpretation of the test concentrations that the fish experienced during the critical swim-up stage when most mortality occurs.

Increased mortality resulted from the highest treatment (T5). No clear concentration-response was apparent with the growth endpoints. Biomass was significantly lower in treatment 5 than controls, but that was due to the reduced density of fish, not growth of surviving fish.

Table 7. Responses of rainbow trout exposed to cadmium in early life stage testing, 1999, 62 day test. Underlined values were significantly lower than control means at P <0.05, by Dunnetts's test.

Mean dissolved cadmium µg/L (± SD)	Control (< 0.2)	T1 0.19 (0.05)	T2 0.38 (0.17)	T3 0.6 (0.38)	T4 1.0 (0.69)	T5 2.5 (1.6)	Minimum detectable difference from control (% of control)
Mortality (%)	17.1	19.4	13.9	17.2	12.8	**<u>36.7</u>**	+14% (16.7%)
Wet weight (g)	0.448	0.386	0.387	0.407	**<u>0.384</u>**	0.405	-0.0623 g (14%)
Biomass (g)	22.55	**<u>18.58</u>**	20.02	20.10	20.10	**<u>15.30</u>**	-3.19 g (14%)
Length (mm)	35.70	**<u>34.06</u>**	34.43	34.03	**<u>34.03</u>**	34.58	-1.53 mm (4.3%)
Mean hardness 26 (range 21-32), mean pH 7.22 ± 0.15 SD							

Mortality and growth responses to cadmium are graphed in Figure 4. In the 1997 test with the higher exposures, growth was apparently reduced in the fry that did not succumb (top). At the lower cadmium exposures that were interrupted on day 53 of the 1997 test, fish weights were similar to controls at that time (Table 6). In the 1999 chronic test, no concentration-growth response was apparent, even at the highest treatment (Figure 4 – bottom). Mean fish weights were slightly less than controls (10 – 14%) in all treatments, however the reduced weights did not correspond to increasing concentrations and most differences were not statistically significant.

Other long-term exposures of salmonids to cadmium have resulted in no or ambiguous effects on growth. Similarly to our 1997 test, Benoit et al. (1976) observed reduced growth in brook trout exposed to concentrations that also reduced survival and no growth reductions at non-lethal concentrations. Chapman (1982) observed no growth effects to juvenile chinook salmon even in cadmium concentrations that killed some fish. Similarly to our 1999 test, Hansen et al (2002b) observed slight reductions in weight in all treatments, but there was no obvious concentration-response between the treatments. These studies are summarized later in this report (Table 8).

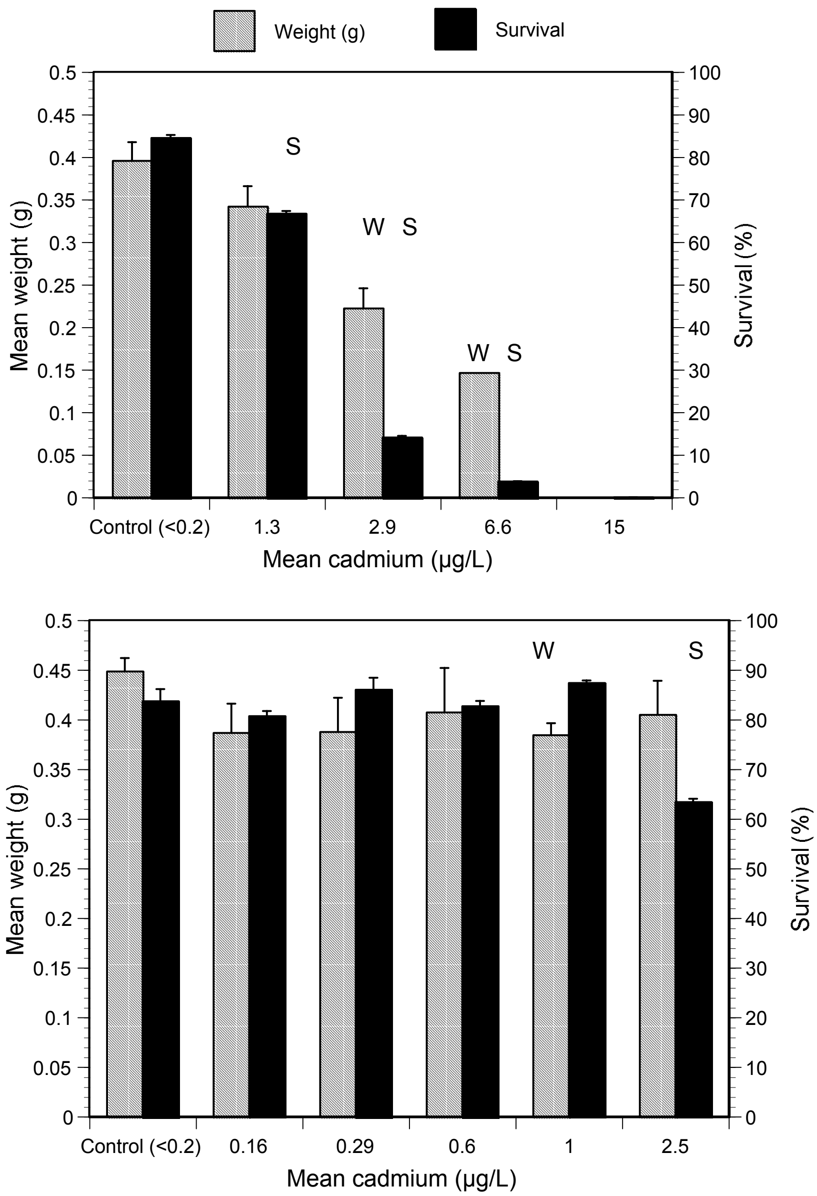

Figure 4. Rainbow trout mean wet weight and survival following early-life-stage testing in site water in 1997 (top) and 1999 (bottom). Letters "W" or "S" indicate weight or survival were significantly lower than control values at $P < 0.05$ by Dunnett's test. Error bars show standard deviation of the mean.

a) What does a "significant effect" in chronic testing have to do with wild fish populations?

The fundamental purpose of chronic testing is to predict no-adverse-effect levels (or nearly so) of a substance in aquatic environments in sustained exposures. The following discussion seeks to relate what chronic test statistics would be biologically significant, and might best protect aquatic populations in the wild.

The chronic responses of rainbow trout to cadmium in Tables 6 and 7 were considered "significant" if statistical hypothesis testing indicated there was less than a 5% likelihood that apparent differences were simply due to chance. However, it is possible that biologically important adverse effects may not be statistically significant due to high variability, small sample sizes, or other statistical vagaries. Likewise, biologically trivial effects might be statistically significant if variation is low and sample sizes are large.

Early-life stage and other chronic testing has customarily been statistically evaluated through hypothesis testing to estimate chronic values for developing water quality criteria. In this approach, the treatments responses are compared to the control responses by ANOVA to test the null hypothesis that they are the same. If the null hypothesis is rejected, then Dunnett's multiple comparison test is used to determine which treatments differ from the control. The no-observed-effect concentration (NOEC) is the highest test concentration that resulted in responses that are not statistically different from the controls. The lowest-observed-effect concentration (LOEC) is the lowest test concentration that is statistically different from controls. These values are commonly used as "chronic limits," between them is a hypothetical MATC which is assumed to be a threshold for toxic effects. Its point estimate is the geometric mean of the NOEC and the LOEC. This point estimate is also called the chronic value (ASTM 1998; McKim 1985; Stephan et al. 1985).

This use of multiple comparison testing has been roundly criticized as being inferior to regression based approaches (Chapman et al. 1995; Denton and Norberg-King 1996; Hoekstra and Van Ewijk 1993; Moore and Caux 1997; Norberg-King 1993; Suter et al. 1987). The criticisms are for a variety of reasons, but center on the fact that multiple comparison test methods were not designed for, and are undesirable for predicting effects at other than concentrations tested. These include (1) the likelihood of observing an effects concentration has as much to do with the experimental design as it does the contaminant's toxicity or the sensitivity of the organism, (2) an effect at a the "true" no-effect concentration may be lower than the NOEC, but the test had inadequate statistical power to detect low toxic effects; (3) NOECs and LOECs can be ambiguous. For example, in Table 7 with *"% Mortality,"* treatment 4 is clearly the NOEC and treatment 5 the LOEC. However for "*Biomass*," either Treatment 1 or Treatment 5 could be considered the LOEC; and (4) NOECs and LOECs do not necessarily reflect biologically relevant thresholds or acceptable/unacceptable effects levels. For example, in both the 1997 and 1999 cadmium early-life stage tests, a 4% reduction in length was statistically significant, while increased mortality was not statistically significant until about a 16% increase. A 4% reduction in length and a 16% increase in death rate probably are not biologically equivalent endpoints even though they are roughly equally statistically significant. On this point, Stephan et al. (1985) caution that judgement is required to determine what an adverse chronic effect is, considering the magnitude of effects rather than rote reliance on statistical significance.

In lieu of NOECs and LOECs, the authors in the previous paragraph recommended using effects percentages (ECp) from regression-based concentration-response curves. However, this begs the question – *how does one specify what percentage ECp value is biologically important in aquatic environments?* The authors cited earlier on this point examined statistical considerations, not ecological acceptability to wild populations. For example, confidence limits, model-dependence or comparisons of the NOECs, MATCs, or LOECs back to various ECp percentages were tested. Relating effects levels from chronic toxicity test endpoints (e.g. mortality and growth) to potential effects to wild populations may be more pertinent to the purpose for conducting the tests in the first place, even if uncertain. To consider this question, literature relevant to young-of-year fish growth and survival in streams were reviewed.

i) What percentage of early-life stage mortality (LCx) puts wild fish populations at risk?

The primary adverse effects studied in an early-life stage toxicity test with fish are reduced growth and survival (ASTM 1998). The most comparable responses in wild fish populations in streams are probably young-of-year mortality growth and survival. If an increased die-off of young-of-year fish translated directly into decrease in populations of adult fish, predicting population effects from contaminant-induced mortality would be straightforward. It does not translate largely because the survival of juvenile fish cohorts in nature is often density dependent. That is, when densities are high, food and space become limiting, growth is stunted, and survival is low. When densities are low, food and space are abundant, growth is higher, and survival is higher (Scheffer et al. 1995). In streams, fish populations are often limited by food availability, and for territorial species such as salmonids, space (Chapman 1966). As fish grow, their per capita space and energy requirements increase and the number of individuals that can be supported in equilibrium decrease. For successful growth, maturation, and reproduction, dense fish populations need to "self-thin" their numbers through die-off or emigration (Dunham et al. 2000; Dunham and Vinyard 1997). A die-off of smaller young-of-year fish is usually followed by a compensatory increase in growth and survival rates by the survivors. (Nordwall and Lundberg 2000; Scheffer et al. 1995).

If a proportion of young-of-year fish died off because of contaminant-induced toxicity, then the survivors would benefit from increased food and increased growth, increasing their chances of surviving to adulthood. Yet this density-dependent compensation is not limitless – at some point young-of-year mortality has to become excessive which would result in year-class failures, decreasing spawner recruitment, and increasing the extinction risk of the population. Further, if stream populations of fish are depressed due to density-independent factors (e.g. food and habitat space are not limiting), then less or no compensatory increased survival would be predicted.

The question remains: what percentage of young-of-year mortality is generally sustainable in natural stream fish populations? Examples with stable or depressed populations provide some bits of evidence. Only one published account was located that explicitly evaluated the additional risks of contaminant-induced mortalities to exploited fish populations (Barnthouse et al. 1990). Two examples of modeling increased harvest-related mortality to stable or depressed salmonid populations were reviewed (Nordwall and Lundberg 2000; Post et al. 2003). These examples give some insight on the potential consequences of different rates of human caused young-of-year mortality that should be pertinent to a contaminant caused LCx effect.

Barnthouse et al. (1990) evaluated the vulnerability of fish populations with differing life history models to contaminant-induced stress. They used toxicity test data and fish population models to evaluate potential effects to two well studied fish populations that differed in life histories and population status, the Gulf of Mexico menhaden and the Chesapeake Bay striped bass populations. The Gulf menhaden life history was considered robust to stress and the population was considered stable, while the Chesapeake Bay striped bass was considered vulnerable to adverse changes in their environment due to low reproductive potential and long life. The Chesapeake Bay striped bass population was further depressed due to overexploitation and habitat degradation. Barnthouse et al. (1990) modeled the 100-year extinction risk to both species under scenarios of 8%, 20%, or 30% contaminant-induced annual young-of-year mortality, and low, medium, or high uncertainty in various extrapolations of toxicity data (e.g. extrapolating from a species with toxicity data for a contaminant to a different species of interest, extrapolating from acute LC50s to life-cycle tests (high uncertainty) or extrapolating from a early-life stage test to a life-cycle test (lower uncertainty). No extinctions to the vulnerable species were predicted at 20% annual mortality at low and medium toxicity extrapolation uncertainties; about a 10% frequency of extinction was predicted at 20% annual mortality and high uncertainties. At 30% annual mortality, predicted extinction frequencies for the overexploited striped bass ranged from about 2 to 25% at low to high toxicity uncertainty. At 8% mortality, no extinctions were predicted regardless of the toxicity extrapolation uncertainty. The extinction risks for the stable menhaden population were much lower.

Nordwall and Lundberg (2000) simulated human-mortality caused population changes to a well studied, unexploited, representative coldwater stream trout population. In their unexploited study stream, natural young-of-year survival rates averaged 33%, and overall young-of-year to adult (age 4) survival averaged 1%. Using a simulated harvest regime that did not use minimum size limits to protect the youngest age-classes, they found that at >40% annual harvest-induced mortality rates to young-of-year and 1-year old trout, the stream population could not be sustained.

Similarly, Post et al. (2003) simulated human-mortality caused population changes to an adfluvial bull trout population that was considered vulnerable to overexploitation. Species such as bull trout, that have relatively slow growth, late age at maturity, low fecundity, and high catchability have low sustainable levels of fishing effort and harvest. Post et al. estimated whether a minimum viable population would be sustainable if bull trout were subjected to catch-and-release fishing, where killing and keeping caught bull trout of any size were prohibited. Using estimates of 10% hook-and-release adult mortality, and 10% non-compliance (poaching) adult mortality, under high angler effort, even these low mortalities incident to catch-and-release fishing were not sustainable.

Post et al.'s (2003) high angler effort scenario exposed a larger proportion of the population to angler-induced mortality than did lower efforts. This high angler effort scenario seems pertinent to a contaminant induced mortality scenario. The assumption made with contaminant-induced mortalities is that most of the fish population would be exposed for at least part of their life cycle.

In summary, the studies reviewed suggest that for fish population that are reasonably stable, not under high existing levels or stress, or for populations that have life history traits such as high reproductive potential or short life cycles, 30 – 40% contaminant-induced mortality to young-of-year would be sustainable. In contrast, population wide 20 – 30% contaminant-induced young-of-year mortalities could be too high for more vulnerable bull trout or striped bass populations, respectively. If the simulated mortalities for the bull trout population were halved (equivalent to reducing the 20% angling mortality to 10% human-induced mortality to adult fish), the population would be sustainable (Post et al. 2003). Admittedly, the angling mortality assumed in their simulations would mostly affect sexually mature fish, and comparable young-of-year mortalities might have a smaller effect on the reproductive potential of the population. Still, Post et al.'s simulations caution that a small, vulnerable fish population may not be able to sustain much increased mortality.

Aquatic insects commonly have much shorter life cycles than fish, have high dispersal abilities, and generally high reproductive potential. Aquatic insects commonly have annual life cycles in the north temperate zone, although they range from less than two weeks for some mayflies and midges to several years for some dragonflies and beetles (Wallace and Anderson 1996). This suggests the premise that aquatic insect populations would commonly be less vulnerable to contaminant-induced mortality than more long-lived fish populations. The EC25 seems to becoming a commonly reported endpoint in chronic toxicity tests with invertebrates (Lewis et al. 1994). Based on factors considered here, this seems like an appropriate chronic endpoint for insect populations.

In the South Fork Coeur d'Alene River study area, westslope cutthroat trout probably share some of the vulnerability factors that make bull trout populations vulnerable to extinction risk (Post et al. 2003; Shepard et al. 1997). Thus, similarly to the Chesapeake Bay striped bass or the adfluvial bull trout population, the South Fork population of westslope cutthroat trout might only be able to compensate for low rates (10 – <30%) of additional, sustained mortalities. *This suggests a potential contaminant-induced young-of-year mortality of up to about 10% would be sustainable in South Fork fish populations. This corresponds to an LC10 as the measurement endpoint for a lethality threshold in early-life stage testing with salmonids.*

ii) What percentage of early-life stage growth reduction (ECx) likely puts wild fish populations at risk?

In addition to mortality, the other primary endpoints measured in an early-life stage test with fish are reduced weight or length. If as argued earlier, statistical hypothesis testing of these endpoints is not necessarily biologically meaningful, what amount of growth reduction in early-life stage fish is biologically meaningful? Research on size-dependent overwinter survival of fish provides some clues about potential effects to wild fish populations from growth reductions.

Survival during cold winters has been directly related to the size achieved before winter in a number of salmonids and nonsalmonids Growth can be limited by food availability, cohort density, habitat complexity and stream geomorphology. Even when food is abundant during their first winter, young-of-year fish may be unable to assimilate it A size-dependent overwinter survival relationship appears especially strong for populations at the northernmost or most upstream portions of their ranges (Connolly and Petersen 2003). Shuter and Post (1990) argued that in cold, temperate climates, equilibrium population abundance is dependent on winter survival of young-of-year fish. Whether young-of-year fish survive their first winter is closely related to their growth during their first summer. Winter is often a critical period in the life of stream salmonids and other temperate fish, especially for the youngest age classes. Stream dwelling trout often suffer a metabolic deficit in early winter where their feeding and digestion cannot keep up with their metabolism, resulting in starvation. Smaller fish tend to be less resistant to starvation conditions because they exhaust their energy stores sooner because their basal metabolism increases as size decreases (Cunjak and Power 1987; Shuter and Post 1990).

Additionally, winter survival of juvenile trout may be dependent on concealment cover such as cobble-boulders, woody debris, or undercut banks which may provide refuge from predation and downstream displacement. Competition for suitable cover under cobble may result in aggressive encounters between juvenile salmonids of the same or different species. Larger fish will win out in aggressive encounters and may exclude smaller fish from necessary cover, which may contribute to higher mortality rates of smaller fish (Griffith 1988; Smith and Griffith 1994). Size-dependent mortality may be more of a factor when environmental conditions are more severe, such as ice-scour, lower temperatures, or less suitable winter habitat (Meyer and Griffith 1997).

Three studies addressing size dependent mortality with trout and salmon in the genus *Oncorhynchus* were reviewed. Miyakoshi et al. (2003) reported a linear releationship between in overwinter survival of hatchery reared masu salmon *Oncorhynchus masou* and late-fall fish size in a stream in northern Hokkaido, Japan (Survival = -0.547+1.66 (weight (g), r^2 =0.94, P<0.05). For a salmon parr weighing 20 g at the onset of winter, a 20% reduction in weight would predict about 25% increased mortality at the end of winter, and for a 10 g fish, a 20% weight reduction would predict about a 33% increase in mortality. Overwinter survival rates for all sizes of parr were low, only 9 – 17% (Miyakoshi et al. 2003).

Smith and Griffith (1994) reported a strong size-dependent overwinter mortality in young-of-year rainbow trout held in cages in the Henry's Fork of the Snake River, Idaho. Rather than a linear pattern, plotting their data suggested a sigmoid relationship, much like toxicity data (Figure 5). Among the larger young-of-year fish, these were little differences in overwinter survival. Yet no fish survived that was <60% of the length of the largest fish. The survival data were converted to percentages in Figure 5 because Smith and Griffith noted that the rainbow trout they captured and used were larger than their counterparts in many streams and larger than members of other salmonid species.

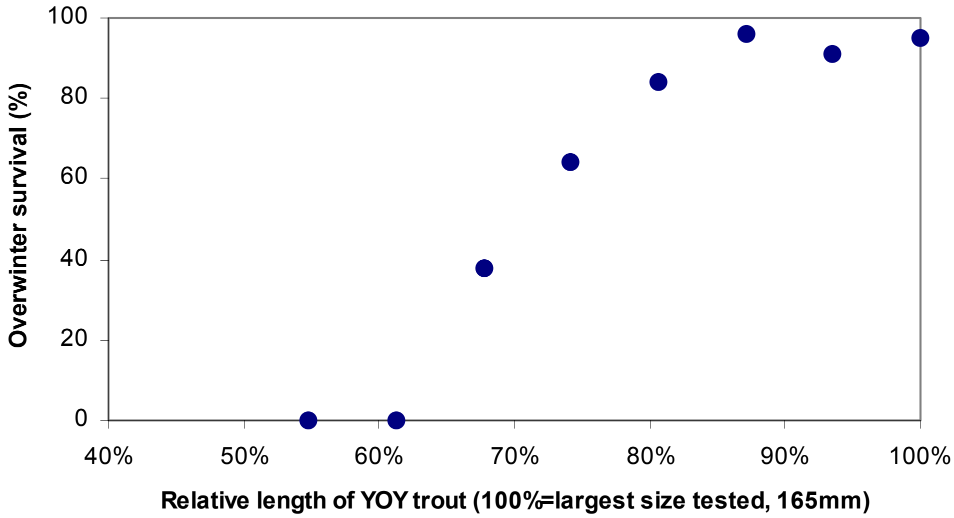

Figure 5. Size-dependent pattern of overwinter die-off of young-of-year rainbow trout. Data from Smith and Griffith (1994).

Meyer and Griffith (1997) reported a similar study using large and small young-of-year rainbow and brook trout in “cold” and “warmer” river sites, which had average January water temperatures of 1.6 and 3.1ºC respectively. At the “warmer” site, there was no size-dependent mortality, whereas at the “cold” site all large rainbow trout juveniles survived, but only 1 of 5 small rainbow trout survived. Brook trout survival was consistently lower than rainbow trout survival which the authors attributed to the available boulder-cobble cover being more suitable habitat to rainbow trout than brook trout. The Meyer and Griffith (1997) young-of-year rainbow trout averaged about 40mm smaller than the Smith and Griffith (1994) young-of-year rainbow trout, even though the fish for both studies were captured from the same locations at the same time of year. Meyer and Griffith suggested that the smaller fish in the later study were likely the result of a cooler summer growing season. These two studies suggest that relative sizes among fish cohorts rather than actual sizes may be more important in some cases when fish compete for limited suitable habitat. No comparable studies of size-dependent overwinter mortality with cutthroat trout are known of, but young-of-year cutthroat occupy similar winter habitat as young-of-year rainbow trout, and their life histories are mostly similar (Behnke 1992; Griffith and Smith 1993).

Generally, this review found that summer mortality of stream-living salmonids is generally low compared to winter. In contrast, early-life stage tests with salmonids probably reflects a benign, summertime stream environment with moderate temperatures, ample food, low velocities, and the absence of predators. In such a test, the only stressors are the tested contaminant and the lack of in-tank visual shelter. This suggests that wild fish exposed to contaminants in cold climates might encounter more severe effects than in laboratory tests, although other mitigating factors such as acclimation or avoidance might be present in the wild (Chapman 1983).

Further, size-dependent mortality of young fish during the onset of their first winter may only be significant when environmental conditions are severe. Under less severe conditions, size differences might be less important (Meyer and Griffith 1997) or the size advantage could be reversed. Laboratory experiments by Connolly and Petersen (2003) suggested that exceptionally mild winters could deplete the energy stores of *large* young-of-year fish, and that cold winters could decrease the energy stores of *small* young-of-year fish. The different size-mortality patterns observed in the few quantitative studies reviewed here do not seem to support any particular response pattern (e.g. linear, logarithmic, sigmoid, etc.). The most reasonable generalization may simply be that size-reductions in young-of-year salmonids over a range from <10% to 30% could result proportional winter mortality. This in turn suggests that sustainable contaminant-induced growth reductions in young-of-year fish would be about the same as sustainable lethality rates. Expressed as a measurement endpoint in an early-life stage toxicity test, contaminant-induced reductions in length or weight of about 20% (i.e. EC20) should be sustainable in fish populations that are reasonably stable or environmental conditions are not otherwise severe. *In more vulnerable populations, or in high elevation or high latitude locations with severe winters, an EC10 for growth endpoints would likely be safer.* Lower ECx values such as EC5s or lower have high statistical uncertainty (Moore and Caux 1997) and do not appear necessary to avoid unacceptable effects to even vulnerable fish populations.

5) Comparison of site-specific and other chronic cadmium effects data

Much less is known on the effects of long-term cadmium exposures to stream-resident aquatic life than for short-term effects. Because of this, results of long-term exposures from both site-specific testing and selected other testing that might be relevant to the species occurring in the South Fork watershed were evaluated. Non-site-specific tests were sought and reviewed that reported long-term cadmium exposures to salmonids and other aquatic species that are close relatives to those occurring in the South Fork. EPA (2001a) was the primary source used to locate reports for detailed reviews, tests that were conducted in very high hardness test waters (>200 mg/l) were ignored. Results of the site-specific testing with rainbow trout and 13 other tests are summarized in Table 8. The 1997 rainbow trout early-life stage test is reported both as a 53-day test with six treatments and as a 69-day test with four treatments, as discussed earlier. One study of cadmium toxicity to several relevant invertebrate taxa only reported their results graphically, so no statistical analyses were possible (Spehar et al. 1978a). However, since Spehar et al. reported the only known long-term exposures of cadmium to coldwater stream macroinvertebrates, their results seemed important to consider as well.

From the conclusions of previous section, the EC10 was considered the most appropriate measurement endpoint for evacuating low toxic effects.[2] None of the studies summarized

[2] Some authorities (e.g. ASTM 1998) specify effects concentrations for dichotomous data such as dead or alive animals as “ECx” values to distinguish them from more continuous data such as egg counts, lengths,

in Table 8 reported EC10 values, so EC10 values were calculated from the original data using the Probit regression model for mortality data and linear interpolation for growth data (EPA 1992; Norberg-King 1993). For tests with multiple endpoints such as mortality, length, or weight, the EC10 value listed is the lowest endpoint obtained for that test. In all cases, the mortality endpoints were lower or similar to other endpoints.

In all but one case, the EC10 values were usually lower than or similar to MATC values for the studies reviewed (Table 8). The one exception was the 1997 site-specific test when evaluated as a 69-day, four treatment test. That MATC value is a very uncertain estimate because adverse effects occurred at the lowest treatments, so no true NOEC was obtained. The MATC was estimated by taking the geometric mean of the ½ the detection limit of controls and the lowest tested concentration. Simply by using different possible choices of estimates for the below-the-detection-limit cadmium concentrations in the controls ranging from ½ the detection limit of low -level analyses of background cadmium in the vicinity (non-detected at <0.038 µg/L, T.R. Maret, USGS, personal communication) up to the actual detection limit of the control water (0.2 µg/L), calculated "MATC" values would vary by 3X from 0.15 to 0.5 µg/L. EC10 values were similar (0.84 – 0.89 µg/L) for the 1997 cadmium early-life stage test whether calculated as a 69-day, four treatment or 53-day, six treatment test.

Chronic tests with coldwater stream insects are difficult to conduct because few taxa are amenable to being artificially cultured as test organisms (ASTM 1997). No site specific chronic testing of invertebrates with cadmium was attempted. The Spehar et al. (1978a) tests were the most relevant studies located, even though some results were uncertain and none were true chronic tests as defined by Stephan et al. (1985). In Spehar et al.'s tests, lowest effects concentrations obtained after 28-days exposure of mayflies to cadmium were similar to those reported for salmonids. Effects concentrations obtained for snails were about 2X higher, and were >100X higher for stonefly and caddisfly species.

Most of chronic responses with different salmonid species in soft water (20 – 45 mg/L) were similar, with EC10s of about 0.8 to 1.7 µg/L and NOECs around 1 µg/L. An exception was a 55-day subchronic test with bull trout, which had response values of about ½ those of most other tests (Table 8). The bull trout test differs from the other tests in that it was a subchronic growth and mortality test, not an early-life stage or life cycle test as are the other tests. EPA chronic criteria development guidelines call for the exposure of embryo and early-life stages of fish through life cycle or early-life stage tests (Stephan et al. 1985). The bull trout test did not expose the embryo or early-life stages, but instead began exposures at the swim-up fry life stage (Hansen et al. 2002b).

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or weights which are called "ICx" or inhibition concentrations. For simplicity, effects concentrations (ECx) are used here for either data type.

Table 8. Site-specific rainbow trout chronic toxicity test values and selected literature chronic toxicity test values.

COMMON NAME	SPECIES NAME	TEST DATE	DURATION	TEST TYPE	NOEC	LOEC	MATC	EC10	AVERAGE HARDNESS	NOTES AND ORIGINAL DATA SOURCE
Site Specific Testing										
Rainbow trout	*Oncorhynchus mykiss*	8/22/1997	69 days	ELS	<0.2	1.3	0.36	0.84	21	Survival and growth reduced with increasing concentrations, low cadmium treatments lost at day 53 due to dosing error (EVS 1997). LOEC is lowest treatment, no true NOEC exists. MATC calculated using ½ detection limit as NOEC, so is a very uncertain and low-biased value.
Rainbow trout	*Oncorhynchus mykiss*	8/22/1997	53 days	ELS	0.6	1.3	0.88	0.89	21	Values calculated before dosing error (Table 6; EVS 1997)
Rainbow trout	*Oncorhynchus mykiss*	07/23/99	62 days	ELS	1.0	2.5	1.58	1.61	26	No concentration-response with growth endpoints (Table 7, Windward 2000)
Other Chronic Test Results with Salmonids and genera resident to the study area										
Chinook salmon	*Oncorhynchus tshawytscha*		120 days	ELS	1.3	1.9	1.56	1.23	25	No growth reductions at any treatment (Chapman 1982)
Bull trout	*Salvelinus confluentus*		55 days	SC	0.38	0.78	0.55	0.49	30	Growth reduced only with treatments which also had reduced survival (Hansen et al. 1999; Hansen et al. 2002b). Test water had a low Ca/Mg ratio, which may have made the test water's Ca hardness effectively similar to that of a South Fork total water hardness of about 22 mg/L.
Brook trout	*Salvelinus fontinalis*		3 years	LC	1.7	3.4	2.4	1.7	44	EC10 estimated at NOEC because no partial responses were apparent. Complete mortality occurred at the LOEC, and no concentration-response below LOEC. (Benoit et al. 1976)
Brook trout	*Salvelinus fontinalis*		60 days	ELS	1.1	3.8	2.0	nc	44	(Eaton 1978)
Brook trout	*Salvelinus fontinalis*		~60 days	ELS	1	3	1.7	1.7	37	EC10 calculated from authors' data (Sauter et al. 1976)
Brook trout	*Salvelinus fontinalis*		~60 days	ELS	3	7	4.5	2.9	188	Statistics recalculated from original data (Sauter et al. 1976)
Mountain whitefish	*Prosopium williamsoni*		14 days	EA	4.0	8.2	5.7	1.7	48	Not a test of chronic sublethal effects (growth or reproduction) but 14 day test duration is long enough to give some indication of effects of extended Cd exposures. (Stubblefield 1990) Species occurs in the South Fork
White sucker	*Catostomus commersoni*		60 days	ELS	4.2	12.0	7.1	nc	44	Other *Catostomus* species occur in South Fork (Eaton 1978)
Mayfly	*Ephemerella sp.*		28 days	EA	<0.05	3.0	1.5	nc	45	LC50 ≈3 µg/l, adjusted for 31% control mortality, higher treatments resulted in 84-94% mortality although not consistent with concentration. Almost all control and treatment mortality occurred after day 21 (Spehar et al. 1978a). Treatment spacing and lack of concentration-response precludes calculating statistics, LC50/2 is MATC best guess.
Snail	*Physa integra*		21 days	EA	3.0	8.3		5	45	High control survival and concentration-response through day 21, afterwards controls and all treatments suffered high mortalities that did not correspond to concentrations (Spehar et al. 1978a)
Stonefly	*Pteronarcys dorsata*		28 days	EA	>238				45	(Spehar et al. 1978a). Genus occurs in South Fork
Caddisfly	*Hydropsyche betteni*		28 days	EA	>238				45	(Spehar et al. 1978a). Genus occurs in South Fork
Caddisfly	*Brachycentrus sp.*		28 days	EA	>238				45	(Spehar et al. 1978a). Genus occurs in South Fork

Table notes: All values are in µg/l, 0.45 µm filtered (site-specific) or unfiltered (all others) concentrations are assumed to be similar. **NOEC** – no observed effects concentration; **LOEC** – Lowest observed effects concentration; **MATC** – maximum acceptable toxicant concentration, geometric mean of NOEC and LOEC. **EC10** – concentration causing an adverse effect to 10% of the test organisms relative to controls. **ELS** – Early life stage test; **LC** – Full life cycle test; **SC** – Sub-chronic test of growth and survival in juvenile fish; **EA** – Extended acute test, test duration much longer than usual 96-hour acute test, but lethality was the only measured endpoint of adverse effects; **nc** – not calculable from the reported data.

One can argue, however, that in watersheds such as the South Fork, the subchronic exposures tested by Hansen et al. are at least as ecologically relevant to salmonid populations as are continuous life cycle or early-life cycle exposures. It is conceivable that long-term metals exposures that begin at the metals-resistant egg life stage could induce an acclimation response that make the fry more resistant to later metals exposure post hatching. However in field conditions, the patterns of fish development and movement and pollution patterns combine so that metals exposures might be expected to begin at the juvenile, not egg stage. Cutthroat trout and some other salmonids tend to spawn high in stream headwaters, and after emerging from spawning gravels, young-of-year fry move downstream into larger waters over a period of days to months. There they may remain as they mature or they may continue to make seasonal upstream-downstream migrations to avoid stressful warm summer temperatures or cold winter temperatures and icing (Hilderbrand and Kershner 2000; Jakober et al. 1998; Knight et al. 1999). Less migratory species such as sculpins would additionally be exposed to metals in their spawning and egg life stages. In the South Fork watershed, stream headwaters have generally low metals concentrations. Metals concentrations increase from upstream to downstream as the watershed drains areas disturbed by mining related activities (Dillon and Mebane 2002). These patterns of cutthroat trout life-cycles and metals contamination in the watershed work together so that cutthroat eggs, embryos, and newly emerged fry would not be exposed to elevated metals, but as they move downstream as juveniles they could encounter elevated metals concentrations. Pre-exposure of the egg and embryo life stages to metals might result in an acclimation response and increased resistance, compared to acute tests with swim-up fry. This argument that exposure of the egg stage may increase metals resistance, is not necessarily supported by field observations. In the study area the non-migratory sculpins would have been exposed at the egg and all life stages, yet they appear to be more severely affected by cadmium and zinc than were salmonids (Dillon and Mebane 2002; Maret and MacCoy 2002).

Hence, the chronic cadmium effects values listed in Table 8 for bull trout that are lower than all the other chronic effects values obtained with salmonids, including other *Salvelinus* values, could reflect testing differences as much or more than inherent differences in species sensitivity. In side-by-side acute testing with rainbow trout and bull trout, rainbow trout were mostly more sensitive to cadmium than bull trout (Hansen et al. 2002a). Because a growth and survival subchronic test that begins with swim-up fry could provide lower effects values than would a test that begins exposure to more resistant life stages, and because there is an argument for the relevance of the subchronic test to fish populations, it follows that subchronic growth and survival need not be discounted in favor of life cycle or early-life stage tests.

6) Chronic site-specific criteria evaluations

To evaluate a site-specific chronic cadmium criteria appropriate to site conditions, based on the site-specific and other relevant data, data were considered in three ways: qualitatively, acute/chronic ratios were calculated, and calculations similar to those used to evaluate the final acute value were made. Guidance for developing site-specific chronic criteria through the resident species approach relies heavily upon using acute/chronic ratios (Carlson et al. 1984). However, as explained in the following discussions, there may be limitations to the acute/chronic ratio approach with cadmium.

Because of this, the available data were evaluated in different ways to try to estimate chronic thresholds that would neither over-predict or under-predict toxicity to resident organisms. These evaluations were intended to be in the spirit of the criteria derivation guidelines, i.e. to make the best use of the available data to derive the most appropriate criteria (Carlson et al. 1984; Stephan et al. 1985).

a) Qualitative evaluation

Site-specific and selected other data on the effects of long-term cadmium exposures to salmonids are plotted with the NTR and EPA 2001 chronic cadmium criteria curves in Figure 6. The 1997 rainbow trout early-life stage test is plotted with its EC10 calculated both as a 69-day, 4-treatment test or a 53-day, six treatment test. Differences between these two interpretations of the test are small, compared to chronic criteria values. The plot shows that all test results are above the NTR chronic criterion, and all but the bull trout value are well above the criterion. As discussed earlier, it seems most plausible that this difference probably reflects methodological differences more than inherent biological differences between the bull trout, other *Salvelinus*, rainbow trout, and other salmonids. If that is the case, it is possible that if tests with other species were initiated with the most sensitive swim-up fry instead of the egg life stage, lower effects values could be obtained, and vice versa. This suggests that even though bull trout are not currently believed to occur in the South Fork watershed, the lower threshold of effects value obtained in that test by exposing swim-up fry to long-term cadmium exposures could be relevant to other species that are similarly acutely sensitive to cadmium such as cutthroat trout or shorthead sculpin (Table 3; Dillon and Mebane 2002).

The 2001 chronic cadmium criterion is much lower than any measured effects of cadmium to salmonids or other species that are closely related to resident species (Figure 6, Table 8). This suggests that the 2001 chronic criterion equation would overpredict chronic effects to resident species.

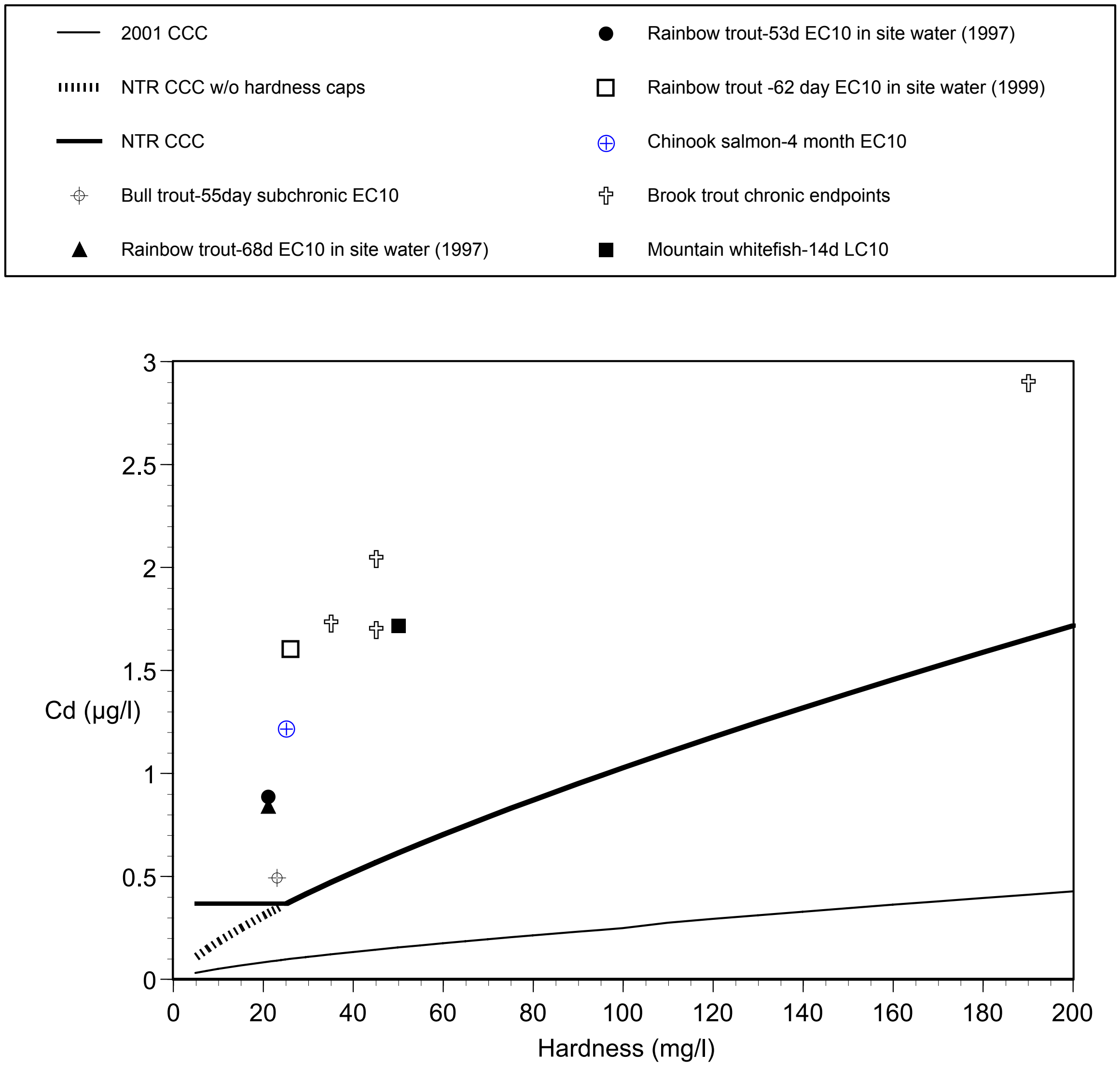

Figure 6. Chronic threshold of effects values (EC10s) for salmonids compared with the current Idaho chronic cadmium criteria ("NTR CCC"), and EPA's 2001 recommended chronic cadmium criterion ("2001 CCC").

Acute/chronic ratio approach to deriving a chronic criterion

Because chronic effects data are much more difficult to obtain than data on acute effects of contaminants to organisms, chronic values are always more scarce than acute values. Acute/chronic ratios are often used in water quality criteria derivation to estimate chronic effects values for species without chronic data. Site-specific and other data that seemed relevant to resident species were reviewed and acute/chronic ratios were calculated.

In four long-term studies, exposures of salmonids to cadmium were concurrent with acute (96-hour) exposures of the swim-up fry of the same cohort of fish in the same test water. These matched tests provide acute/chronic response ratios (ACRs), which can be used to estimate chronic response thresholds to other, untested species. Using the 96-hour LC50s as acute values with the available EC10s values from Table 8 as chronic values ACR estimates (acute LC50/chronic EC10) follow in Table 9. The EPA guidelines for deriving water quality criteria advise that for invertebrates, only results of toxicity tests that expose groups of individuals throughout a life cycle should be used to obtain chronic values. For stream resident macroinvertebrates such as mayflies, stoneflies, or caddisflies which are hard to culture in laboratories, life-cycle tests would be very difficult to complete. However, effects concentrations obtained in Spehar et al's (1978a) 28-day exposures are probably reasonable estimates of chronic toxicity to at least their tested mayflies and snails. Effects concentrations with those species were near those obtained in life-cycle tests with salmonids, and were lower than apparent field responses of macroinvertebrate communities associated with cadmium in the study area (Dillon and Mebane 2002). Thus the ratio of acute LC50/28-day quasi-MATCs probably gives the best ACR estimates available of stream resident macroinvertebrates.

Table 9. Acute/chronic ratio (ACR) estimates

SPECIES	96-HOUR ACUTE VALUE (µG/L)	CHRONIC VALUE (µG/L)	ACR	SOURCE AND NOTES
Rainbow trout (1997)	0.84	0.84	1.0	Table 2, Table 8, EVS 1997
Rainbow trout (1999)	0.89	1.61	0.55	Table 2, Table 8, EVS 1997
Chinook salmon	1.41	1.23	1.1	Table 8, Chapman 1982
Bull trout	0.95	0.49	1.9	Acute value average of 4 tests conducted with similar conditions to chronic test (Table 8, Hansen et al. 1999)
Ephemerella mayfly	>238	1.5	>158	Best guess chronic value (Table 8, Spehar et al 1978a)
Physa snail	>238	5	>47	7-day LC50 was 114 µg/L, if used instead of 4-day value would result in an ACR of 23. Best guess chronic value (Table 8, Spehar et al 1978a)

Most (¾) of the acute/chronic ratios with salmonids were about 1, and all were less than 2.0. Estimated acute/chronic ratios with macroinvertebrates were much higher, because the acute values were much higher than were acute values with salmonids, yet the chronic values were not as different. EPA guidelines for deriving water quality criteria advise that in cases where the acute/chronic ratio seems to be related to the species mean acute values, the final acute/chronic ratio should be calculated as the geometric mean of the acute/chronic ratios for species whose species mean acute values are close to the Final Acute Value (Stephan et al. 1985). These species are rainbow trout, chinook salmon, brook trout, and bull trout (Table 5, Table 9). The geometric mean of their species mean acute/chronic ratios is 1.2. In cases where acute effects are greater than those obtained from the early-life stage continuous exposures, acclimation probably occurred during the resistant egg or embryo stage. In this case, since continuous exposure and acclimation cannot be assumed in field conditions, the acute/chronic ratios for the most appropriate species should be assumed to be 2, so that the chronic value is equal to the criterion maximum concentration (Stephan et al. 1985).

b) Recalculation approach to deriving a chronic criterion

If chronic values are available for species in eight diverse families as described earlier in section 3, a final chronic value (FCV) should be calculated using the same procedures for calculating a FAV (Stephan et al. 1985). Although site-specific chronic values were only available for rainbow trout, other estimated chronic cadmium values for genera that are resident to the study area, or are closely related to resident taxa, were available. These quasi-resident values were used with the FAV procedures as another approach to estimate a potential chronic criterion value. Carlson et al. (1984) specify a "recalculation approach" for modifying national criteria for a material by eliminating non-resident values from the national acute criterion dataset and adding values for resident taxa. The calculations are adapted from the concepts behind that approach, but do not strictly follow those procedures. For example, Carlson et al.'s (1984) recalculation guidance is intended for acute criteria, where chronic criteria would use existing acute/chronic ratios, and some of relevant non-site-specific long-term cadmium effects data for stream macroinvertebrates could not strictly be called "chronic" data as defined by Stephan et al. (1985). Nevertheless, analyzing the available data through different approaches seemed like a useful way to evaluate potential chronic cadmium criteria. The data summarized in Table 8 seemed to be the best available, and the use of the FAV equation as a tool to estimate a reasonable level of ecosystem protection has a statistical and empirical basis (Section 3).

Where available, EC10 values were used in the calculations based on the earlier arguments that for vulnerable salmonid populations, EC10s seemed the most appropriate endpoint. For tests where no EC10 values could be calculated, estimated MATC values were used.

Chronic values used in the FCV calculation were selected from Table 8 and from EPA 2001's Table 2a. Chronic values from Table 8 were "normalized" to a hardness of 50 mg/L using the hardness-toxicity slopes from EPA (2001a) so that the organisms can be ranked by their hardness-normalized values. Only values for resident genera were used, with the following exception. The midge *Chironomus*, Oligochaete *Alesoma*, and snail *Aplexa* values from the national dataset were retained. Chironomids, Oligochaetes, and snails are commonly collected in the study area, even though the specific genera tested have not been reported from stream macroinvertebrate samples from the study area

(Dillon and Mebane 2002; EVS 1995). In FCV calculations, the values for these genera are assumed to be representative of the Chironomids, Oligochaetes, and snail genera that are resident to the South Fork. The genus *Oncorhynchus* values used were the geometric mean of the lowest EC10s from both of the rainbow trout early-life stage tests. Although the Chinook salmon and rainbow trout EC10 values were similar (Table 8), the rainbow trout values were preferable to the Chinook salmon values because the rainbow trout values were obtained in site water.

Using Equation 2, chronic cadmium values for 10 genera provide a calculated FCV of 1.0 µg/L at a hardness of 50 mg/L (Table 10). Coincidentally this is the same as the FCV obtained using the acute/chronic ratio approach, which at a hardness of 50 mg/L is the same as the 2001 EPA acute criterion for cadmium.

Equation 2. Final chronic value (FCV) calculations, modified from the FAV approach (Equation 1).

$$P = R/(n+1)$$

$$S = \sqrt{\frac{\sum((\ln \mathrm{GMCV})^2 - ((\sum(\ln \mathrm{GMCV}))^2/4)}{\sum(P) - ((\sum(\sqrt{P}))^2/4)}}$$

$$A = S(\sqrt{0.05}) + L$$

$$FCV = e^A$$

Table 10. FCV recalculation using site-specific chronic data and non-site-specific data for resident genera, or closely related genera

Rank	Resident Genera	GMCV – Genus mean chronic value (µg/L)	Cumulative Probability (P=R/(n+1)	ln GMCV	(ln GMCV)^2	SQRT P
10	Caddisfly, *Hydropsyche*	>238				
9	Caddisfly, *Brachycentrus*	>238				
8	Stonefly, *Pteronarcys*	>238				
7	Oligochaete, *Aeolosoma*	20.7				
6	Sucker, *Catostomus*	7.8				
5	Snail, *Physa*	5				
4	Snail, *Aplexa*	4.8	0.3636	1.5686	2.4606	0.6030
3	Midge, *Chironomus*	2.8	0.2727	1.0310	1.0631	0.5222
2	Trout, *Oncorhynchus*	2.0	0.1818	0.6931	0.4805	0.4264
1	Mayfly, *Ephemerella*	1.5	0.0909	0.4055	0.1644	0.3015
Sum:			0.909	3.6983	4.1685	1.8532

Number of GMCVs = 10, S=3.8504, L= -0.8593, A= -0.0017
FCV= 1.00 µg/L

The use of Equation 2 provides a degree of structure and objectivity to calculating a FCV. Still, a number of judgements were necessary, and at least five different choices were made that influenced calculated FCV values. First, was the use the quasi-chronic invertebrate values from Table 8. Because the values were from relevant taxa, no better data were available, and because the values ranged from the most resistant to about the most sensitive long-term values, the values seemed reasonable to use. Still, any use of quasi-chronic data is likely to be debatable. Second, if counter to the EPA criteria development guidelines, subchronic salmonid tests that began with swim-up fry were considered more biologically relevant than early-life stage or life-cycle tests that began with more resistant stages (discussed in section 5), then perhaps the bull trout subchronic value should be included in the FCV calculation, as a surrogate for similarly obtained values for resident westslope cutthroat trout. If that test were included without averaging the result with the *Salvelinus* early-life stage or life cycle values, the calculated FCV would be lower than that calculated in Table 10: 0.70 µg/L at a hardness of 50 mg/L, using 11 genera. Third, EC10 values were used in calculations. If other commonly used endpoints such as an EC20 or MATC were used, a different calculated FCV would result. Fourth, the hardness-effects normalization for long-term cadmium exposures is uncertain. The hardness normalization used the ln(hardness) versus ln(chronic effects) slope of 0.74 from EPA (2001a). The limited available data suggests that that slope may be too high, at least for salmonids. The brook trout EC10s calculated from Sauter et al's (1976) data at hardnesses of 37 and 188 mg/L yields a slope of 0.37. The only slope calculated for a salmonid in EPA (2001a) was 0.52 with brown trout. This is similar to the slope of 0.57 (r^2 0.89, P=0.2) calculated from the rainbow trout MATC best estimates of 0.88 µg/L at hardness 21 mg/L, 1.61 µg/L at hardness 26 mg/l and 4.31 µg/L at hardness 250 mg/L (Table 8, Table 2a of EPA 2001a). If the brook trout and rainbow trout slopes were pooled with the Daphnia, fathead minnow, and brown trout slopes used in EPA (2001a) the all-species hardness-effects slope would be about 0.60, instead of 0.74. If a lower slope were used to normalize site-specific salmonid values, a lower calculated FCV would result. Fifth, the basis for the use of the FAV equation with chronic data sets is not as well supported as it is with acute datasets. Erickson and Stephan (1988) developed the FAV equation based largely on statistical analyses of available acute datasets. No similar analysis of chronic datasets is known of. Chronic datasets are usually much smaller than acute datasets, and are probably biased toward species that are readily cultured, or economically important species.

These uncertainties are mentioned to emphasize that even with well defined, quantitative procedures for establishing criteria values, much judgement is still needed. This is also the case in any criteria derivation or other processes to establish targets or benchmarks.

7) Conclusions

Review of the South Fork Coeur d'Alene River site-specific acute data support the conclusion that the EPA (2001a) acute cadmium criterion is functionally identical to a criterion developed solely from site-data. Therefore, the 2001 acute criterion should be applied to the South Fork (Table 1).

Interpreting the relevant chronic cadmium data was more equivocal. The acute/chronic ratio approach and the final acute value (FAV) equation approach to estimating a final chronic value (FCV) both yielded similar results. The acute/chronic ratio of 2 would result in the chronic criterion equaling the acute criterion, which is 1.0 µg/L at a hardness of 50 mg/L (Table 9). Calculating a FCV with selected chronic values normalized to a hardness of 50 mg/L also yielded a value of 1.0 µg/L at a hardness of 50 mg/L (Table 10). This value is higher than either the currently applicable chronic criterion of 0.62 µg/L or EPA's 2001 recommended criterion of 0.15 µg/L (Table 1).

If the acute/chronic approach were used to set criteria, the slope of the criterion equation would be the same as that in the acute criterion equation (Carlson et al. 1984). However, even the lower slope from the EPA (2001a) chronic equation may be too high. If so, at hardnesses above 50 mg/L, using that slope with the FCV from Table 10 could result in a chronic criterion that was less protective than intended by the national guidelines. Before deriving a chronic criterion with either the acute/chronic ratio approach or the FAV equation, the hardness-toxicity extrapolation should be carefully evaluated.

Because of the uncertainties with the two quantitative approaches to setting a chronic criterion, a qualitative evaluation may be as suitable. A simple inspection of the plot of selected thresholds chronic effects data and chronic criteria shows that the existing Idaho chronic criterion is lower than any effects data reviewed, and that the 2001 chronic criterion values are considerably lower than any adverse effects reviewed for species relevant to the South Fork Coeur d'Alene River watershed (Figure 6). The existing Idaho, or "NTR" chronic criterion seems most suitable of the alternatives evaluated.

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