UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 10
1200 Sixth Avenue
Seattle, WA 98101

Reply To
Attn Of: OW-131

August 4, 2000

Theodore F. Meyers
National Marine Fisheries Service
10215 W. Emerald, Suite 180
Boise, ID 83704

Robert G. Ruesink
Snake River Basin Office
U.S. Fish and Wildlife Service
1387 S. Vinnell, Suite 377/368
Boise, ID 83709

Re: Consultation on EPA Approval of State of Idaho Water Quality Standards for Numeric Criteria for Toxic Substances (FWS #1-4-00-F-187; File 600.1502)

On December 22, 1999 and January 7, 2000, the National Marine Fisheries Service and the U.S. Fish and Wildlife Service (the Services) received the Environmental Protection Agency Region 10's (EPA's) final biological assessment (BA) and request to initiate formal consultation on the State of Idaho's water quality standards for numeric criteria for toxic substances. The BA analyzed the effects of EPA's approval of Idaho's numeric criteria of toxic pollutants to listed threatened and endangered species and habitat. On May 2, EPA received notice from the Services that the BA did not contain all the information needed to initiate consultation. The Services provided a list of 12 topics the BA did not address or did not adequately address. This letter encloses a revised introduction to the BA that addresses the 12 areas as best we can given our current information. With this letter, we request initiation of formal consultation on the Idaho water quality standards for numeric criteria for toxic substances.

The revised introduction to the BA still does not provide complete information on all 12 topics; however, it represents what we know. As the introduction to the BA now points out, some of these areas are the topics of EPA research, and for others, no additional information is available. As allowed under Section 7 of the Endangered Species Act (ESA), the BA is to assess the current information, thereby recognizing that information may not be available. The revised introduction contains additional mitigation measures to compensate for the risk to the species from the information gaps about some of the impacts to the species and to critical habitat. My staff and I, and the Idaho Department of Environmental Quality, will be happy to discuss other potential mitigation measures that would address information gaps and reduce the risk to the species.

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As discussed in a conference call on July 6, 2000, I understand that the Services have been working on their Biological Opinions and expect to have drafts available for our review in September. I appreciate the cooperative approach displayed by your staff during this consultation.

If you have any questions, please do not hesitate to call me at (206) 553-1261 or my staff Paula vanHaagen at (206) 553-2857 or Lisa Macchio at (206) 553-1834.

Sincerely,

Randall F. Smith
Director
Office of Water

Enclosure

cc: Ed Murrell (NMFS)
Russ Strach (NMFS)
Michael McIntyre (IDEQ)

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I. INTRODUCTION

EPA's Water Quality Criteria

EPA's Water Quality Standards regulations require states to adopt water quality criteria that will protect the designated uses of a water body. These criteria must be based on sound scientific rationale and must contain sufficient parameters or constituents to protect the designated uses. The criteria are the best estimate using available data. These data must be of the highest quality and reproducibility.

Since 1980, EPA has been publishing criteria development guidelines and national criteria for numerous pollutants. EPA's criteria documents provide a toxicological evaluation of the chemical, tabulate the relevant acute and chronic toxicity information, and derive the acute and chronic criteria that EPA recommends for the protection of aquatic life resources. This original methodology is defined for criteria to protect aquatic organisms that inhabit the water column and the benthos. Exposure to chemicals is limited to passage of dissolved constituents through the gills. The criteria do not provide protection for ingestion of pollutants. They also do not account for site specific hydrological conditions, environmental chemistry of the medium or the organism tissue, extrapolation from laboratory data to field situations, water quality, temperature, dissolved oxygen, organic carbon, and relationships between species. Stresses by disease, parasites, predators, other pollutants, contaminated or insufficient food, and fluctuations and extreme conditions of flow are not included as factors which may alter toxicity or exposure. Also, degradation of pollutants to more toxic forms is not taken into account. The criteria are therefore not protective of all species at all times and places. It is expected that as the data become available, additional criteria will be developed for multiple types of exposure.

A criterion is determined from laboratory toxicity studies including acute and chronic tests for aquatic organisms, toxicity tests for aquatic plants, and bioaccumulation studies for chemicals with known tissue residue effects. Acute toxicity studies are bioassays which are completed in less than 96 hours. EPA derives acute criteria from 48- and 96-hour tests of lethality or immobilization. EPA derives chronic criteria from longer term (often greater than 28-day) tests that measure survival, growth, or reproduction.

The first step in deriving criteria is the calculation of the *Genus Mean Value*. This is based on the acute and chronic toxicity tests. The acute tests must include at least one species of freshwater animals in at least eight different families. These families must include 2 families from the class Osteichthyes along with 1 representative from each of the following : 1) the family Salmonidae, 2) the phylum Chordata, 3) a planktonic crustacean, 4) a benthic crustacean, 5) an insect, 6) a family in a phylum other than Arthropoda or Chordata, and 8) any phylum not represented. In addition, the acute to chronic ratios for species of aquatic animals must include at least three different families and including at least one fish, one invertebrate, and one acutely sensitive freshwater species. The *Final Acute Value* is derived from the cumulative distribution of the acute toxicity tests. The concentration at the cumulative probability of 0.05 is selected as the level which provides protection for a majority of species (95%).

Exposure

The criteria only account for one route of exposure: through the gills. They do not include exposures through ingestion. The variability in *magnitude*, *frequency*, and *duration* of exposure through the gills is included in the criteria.

Magnitude. The criteria contain two expressions of allowable magnitude: a *Criterion Maximum Concentration* to protect against acute (short-term) effects; and a C*riterion Continuous Concentration* to protect against chronic (long-term) effects.

Duration. The quality of an ambient water body typically varies in response to variations of effluent quality, stream flow, and other factors. Organisms in the water body do not typically receive constant, steady exposure but rather experience fluctuating exposures, including periods of high concentrations, which may have adverse effects. Thus, EPA's criteria indicate a time period over which exposure is to be averaged, as well as an upper limit on the average concentration, thereby limiting the duration of exposure to elevated concentrations. For acute criteria, EPA recommends an averaging period of 1 hour. That is, to protect against acute effects, the 1-hour average exposure should not exceed the criterion mean concentration. For chronic criteria, EPA recommends an averaging period of 4 days. That is, the 4-day average exposure should not exceed the criterion continuous concentration.

Frequency. To predict or ascertain the attainment of criteria, it is necessary to specify the allowable frequency for exceeding the criteria. This is because it is statistically impossible to project that criteria will never be exceeded. As ecological communities are naturally subjected to a series of stresses, the allowable frequency of pollutant stress may be set at a value that does not significantly increase the frequency or severity of all stresses combined.

EPA recommends the average frequency for excursions of both acute and chronic criteria not to exceed once in 3 years. In all cases, the recommended frequency applies to actual ambient concentrations and excludes the influence of measurement imprecision. EPA selected a 3-year average frequency of exceeding the criteria with the intent of providing for ecological recovery from a variety of severe stresses. This return interval is roughly equivalent to a 7 day average minimum flow expected every 10 years (7Q10) as a design flow condition. Because of the nature of the ecological recovery studies available, the severity of criteria excursions could not be rigorously related to the resulting ecological impacts. Nevertheless, EPA derives its criteria intending that a single marginal criteria excursion (i.e., a slight excursion over a 1-hour period for acute or over a 4-day period for chronic) would require little or no time for recovery. EPA thus expects the 3-year return interval to provide a very high degree of protection (EPA, 1994).

Idaho's Water Quality Standards

Section 303(c)(2)(E) of the Clean Water Act requires that all states adopt chemical-specific, numeric criteria for priority toxic pollutants. States may choose to adopt EPA's recommended criteria or modify these criteria to account for site-specific or other scientifically defensible factors. In 1992, the State of Idaho had not yet adopted such criteria. Therefore, on December 22, 1992, EPA promulgated such criteria for all waters in the State of Idaho as part of the National Toxics Rule (EPA, 1992). Idaho has since revised the Idaho Water Quality Standards to include the same criteria as EPA promulgated under the National Toxics Rule. Following completion of this consultation, EPA is proposing to recommend a federal action which would remove the State of Idaho from the National Toxics Rule, thus providing for the State's criteria to become effective.

The National Toxics Rule originally promulgated criteria for metals as total recoverable metals. Following EPA's promulgation of this rule, EPA issued a new policy for setting water quality criteria for metals. Therefore, on May 4, 1995, EPA issued a stay on the effectiveness of the metals criteria promulgated in the National Toxics Rule and promulgated revised criteria expressed in terms of dissolved metals (EPA, 1995). At this time, EPA also promulgated conversion factors for converting between dissolved and total recoverable criteria. States, when adopting criteria, may choose to adopt metals criteria measured as either dissolved or total recoverable. The metals criteria in the Idaho Water Quality Standards are expressed as dissolved metals.

In Idaho, both the aquatic life criteria and human health criteria apply to all surface waters of the State. Idaho's water quality standards contain a provision which states that when multiple criteria apply to a water body, the most stringent criterion is the applicable criterion. With regard to the numeric toxic criteria, most toxic pollutants have more stringent aquatic life criteria than human health criteria. Therefore, with regard to the majority of the toxic criteria, the aquatic life criteria are the applicable criteria for surface waters. An example of an exception to this generality is arsenic, where the human health criterion is lower (340 µg/L for aquatic life; 50 µg/L for human health) than the aquatic life criteria. Therefore, in all surface waters in Idaho, the applicable criterion for arsenic is the human health criterion.

All criteria in the Idaho Water Quality Standards, with the exception of the human health criterion for arsenic, are identical to the criteria promulgated by EPA under the

National Toxics Rule. These criteria were adopted by reference in IDAPA 16.01.02.250.07. The aquatic life criteria evaluated as part of this assessment are summarized in Table 250.07.a.1. For comparison purposes, this table provides metals criteria expressed as both dissolved and total recoverable.

Idaho's criteria for pentachlorophenol (PCP) is expressed as an equation dependent on pH, while seven of the criteria for metals are expressed as a function of water hardness. The PCP criteria in Table 250.07.a.1 were calculated at a pH of 7.8. In Table 250.07.a.1, EPA used a hardness of 100 mg/L $CaCO_3$ in order to present a value for the metals criteria. The equations used to derive these criteria are presented in the footnotes to Table 250.07.a.1. These equations include the use of Water Effect Ratios, the ratio between site water and laboratory water effect levels of metal toxicity (EPA, 1994). Water Effect Ratios default to 1, unless a state has done sufficient research to determine a ratio specific to a water body and adopted site specific criteria. Any adoption of site specific criteria must be approved by EPA and consulted on with the Services. Idaho's state standards currently apply the default Water Effect Ratios (see footnotes b and c in Table 250.07.a.1).

Table 250.07.a.1. Idaho Water Quality Standards General Aquatic Life Criteria (from 60FR22228)

Chemical Name	Criteria (g/L)		Total Recoverable Criteria (g/L)		Conversion Factor[a]	
	Acute	Chronic	Acute	Chronic	Acute	Chronic
Arsenic	360	190	360	190	1.000	1.000
Cadmium	3.7[b]	1.0[b]	3.9[c]	1.1[c]	0.944[d]	0.909[c]
Copper	17[b]	11[b]	18[c]	12[c]	0.960	0.960
Cyanide	22[e]	5.2[e]	N/A		N/A	
Endosulfan (a & b)	0.22	0.056	N/A		N/A	
Lead	65[b]	2.5[b]	82[c]	3.2[c]	0.791[d]	0.791[c]
Mercury	2.1	0.012	2.4	0.012	0.85	N/A
Selenium	20	5.0	N/A		N/A	
Zinc	110[b]	100[b]	120[c]	110[c]	0.978	0.986
Aldrin	3	N/A	N/A		N/A	
Chlordane	2.4	0.0043	N/A		N/A	
Chromium (III)	550[c]	180[c]	1,700[c]	210[c]	0.316	0.860
Chromium (VI)	15	10	16	11	0.982	0.962
4,4'-DDT	1.1	0.001	N/A			
Dieldrin	2.5	0.0019	N/A			
Endrin	0.18	0.0023	N/A			
Heptachlor	0.52	0.0038	N/A		N/A	
Lindane (gamma-BHC)	2	0.08	N/A		N/A	
Nickel	1,400[b]	160[b]	1,400[c]	160[c]	0.998	0.997
PCBs	N/A	0.014	N/A		N/A	
Pentachlorophenol	20[g]	13[g]	N/A		N/A	
Silver	3.4[b]	N/A	4.1	N/A	0 .85	N/A
Toxaphene	0.73	0.0002	N/A		N/A	

N/A - no applicable criteria

a - Conversion factors for translating between dissolved and total recoverable criteria.

b - Criteria for these metals are expressed as a function of total hardness (mg/L as $CaCo_3$), and the following formula:

Acute Criteria = WER $\exp\{m_A[\ln(\text{hardness})]+b_A\}$ x Acute Conversion Factor

Chronic Criteria = WER $\exp\{m_C[\ln(\text{hardness})]+b_C\}$ x Chronic Conversion Factor

where (see following page):

Metal	m_A^f	b_A^f	m_C^f	b_C^f
Cadmium	1.128	-3.828	0.7852	-3.490
Chromium (III)	0.8190	3.688	0.8190	1.561
Copper	0.9422	-1.464	0.8545	-1.465
Lead	1.273	-1.460	1.273	-4.705
Nickel	0.8460	3.3612	0.8460	1.1645
Silver	1.72	-6.52	N/A	N/A
Zinc	0.8473	0.8604	0.8473	0.7614

The term “exp” represents the base e exponential function.

c - For comparison purposes, the values displayed in this table correspond to a total hardness of 100 mg/l $CaCO_3$ and a WER of 1.0.
d - The conversion factors for cadmium and lead are hardness dependent. The values shown in the table correspond to a hardness of 100 mg/L $CaCO_3$. Conversion factors for any hardness may be calculated using the following equations:

Cadmium:

Acute- CF=1.136672-[(ln(hardness))x(0.041838)]

Chronic- CF=1.101672-[(ln(hardness))x(0.041838)]

Lead:

Acute and Chronic- CF=1.46203-[(ln(hardness))x(0.145712)]

e - Criteria expressed as Weak Acid Dissociable
f - m_A and m_c are the slopes of the relationship for hardness, while b_A and b_C are the Y-intercepts for these relationships
g - Criteria for pentachlorophenol is expressed as a function of pH and calculated as follows:

Acute Criteria = exp(1.005 (pH) - 4.830)

Chronic Criteria = exp(1.005 (pH) - 5.290)

Water Quality Condition of Idaho Waters

The analyses for the protectiveness of numeric criteria assume that the organisms are exposed to concentrations of pollutants at the water quality criteria levels, not the conditions which currently exist in Idaho’s waters. For waters that do not comply with the water quality standards, the State of Idaho and EPA are undertaking control actions to bring these waterbodies into compliance. However, due to the scale of the action that is the subject of this consultation and the temporal and spatial variability in water quality conditions throughout the state, this assessment will only analyze potential effects at the criteria concentrations. Where waters are not currently in attainment of the standards but where actions are in place to remedy current water quality problems, the analysis describes desired future conditions and thus underestimates potential current effects on the species of concern.

II. METHODS FOR DETERMINATIONS

Determinations regarding the potential for the criteria established by the Idaho Water Quality Standards to adversely affect threatened and endangered species were based on an analysis of the existing criteria documents and any new literature published after the criteria document publication. Acute criterion were compared to published toxicity data where exposure durations were less than or equal to 96 hours. Chronic criterion were compared to published toxicity data where exposure durations were greater than 96 hours. While the scientific community does not agree on precise definitions for the terms acute and chronic, the general approach used here can offer an adequate assessment of the criteria's potential effects on aquatic species.

For all aquatic species except sturgeon, a "may be likely to adversely affect" determination was made if 1) no information was available detailing the toxicity of the chemical with regard to the species of concern or a reasonable surrogate, or 2) the published toxicity data indicated adverse effects at concentrations at or below the established criteria. A "not likely to adversely affect" determination was made if the published toxicity data indicated adverse effects at concentrations above the established criteria. Adverse effects on species were divided into sublethal and lethal effects. Sublethal effects included any measurable or observable effect on a species, not including mortality, while lethal effects consisted only of mortality. Both lethal and sublethal effects were evaluated for each criterion. Generally, in an effort to refrain from duplicating previous work, reports reviewed for this document were published after the publication of EPA's criteria documents for the chemicals reviewed here. Most of the criteria documents were published between 1980 and 1985. In some cases, where information was lacking, we have included data published prior to the criteria documents.

Rather than taking the default approach and assigning a 'likely to adversely affect" determination for white sturgeon, we have chosen to evaluate the proposed standards by examining toxicity data for a variety of fish species, including cold water species (e.g. salmonids) and benthic species (e.g. catfish). If the proposed standards are protective of a variety of fish species, we can assume that the standards will also adequately protect white sturgeon for the following reasons: 1) the proposed standards are below the limits for other fish species and 2) the limited data available show that sturgeon have variable sensitivity compared to other species (i.e. they are not consistently more sensitive than other species).

Of the priority pollutants with Aquatic Life Criteria (see list below), it was jointly determined by EPA and the Services that some chemicals required a more detailed analysis. EPA examined the effects of nine chemicals: arsenic, cadmium, copper, lead, mercury, selenium, zinc, and cyanide, in more detail due to their prevalence in Idaho waters. Endosulfan was also addressed in more detail because of its current agricultural use in Idaho. Chromium III, chromium VI, nickel, silver, and Heptaclor/Heptachlor Epoxide were provided a minimal level of analysis because these chemicals do not occur in Idaho waters with the same regularity. The remaining 9 organic chemicals listed were also given a minimum level of analysis since their use has either been canceled or suspended. For those chemicals given a minimum level of analysis, EPA primarily relied upon information provided in EPA's water quality criteria guidance documents (1980-1985).

For each of the chemicals receiving a high level of analysis, the determination section is organized as described here: a preliminary description of the chemical and criteria followed by an evaluation of recent research on each of the species of concern or their surrogates. The species are considered together in phylogenetic groups such as invertebrates, fish, and birds. Within the evaluation for invertebrates and fish, sublethal and lethal effects are evaluated separately. Determinations for the chemicals that received a minimal level of analysis are grouped together at the end of this section. For each of these chemicals, some background information is provided along with an effects determination. For wildlife and plants, a more general analysis based on exposure is provided in the following sections. A summary of all determinations for all threatened and endangered species is presented in this section. The detailed analysis of effects to fish and invertebrates is presented in Chapter 3.

Priority Pollutants for Aquatic Life Criteria

Tier I - High level of analysis	Tier II - Low level of Analysis	
Arsenic	Chromium (III)	4-4'DDT
Cadmium	Chromium (VI)	Dieldrin
Copper	Nickel	Endrin
Lead	Silver	PCB Arochlors
Mercury	Heptachlor/Heptachlor	Toxaphene
Selenium	Pentachlorophenol	
Zinc	Aldrin	
Cyanide	gamma-BHC (Lindane)	
alpha and beta Endosulfan	Chlordane	

Biological Uptake, Bioaccumulation, Bioconcentration, Biomagnification

The following definitions are provided to explain EPA's determinations regarding biological uptake of chemicals. Bioaccumulation is defined by Rand (1995) as the "...process by which chemicals are taken up by aquatic organisms directly from water as well as through exposure through other routes, such as consumption of food and sediment containing the chemicals." Alternatively, Rand (1995) describes bioconcentration as the "... process by which there is a net accumulation of a chemical directly from water into aquatic organisms..." Since determining the source of chemical accumulation in tissues is difficult at best when reviewing literature, these terms are used somewhat interchangeably to mean an observed increase in tissue concentration of a substance in relation to the concentration in the water. In determining sublethal effects to invertebrates and fish, EPA has concluded that bioconcentration or bioaccumulation is an indicator of exposure to chemicals, but will not be classified as an effect. The concentration of chemicals in tissues of aquatic organisms can be an excellent indicator of environmental exposures, but bioconcentration alone does not constitute an effect to an organism. Effects may occur as a result of the bioconcentration. Where the studies reviewed for this document illustrated effects coincident with bioconcentration, we have included that information in the sections detailing effects to organisms. Otherwise, when the results of the studies reviewed included only bioconcentration of contaminants, information regarding those studies was described in the "Bioconcentration and Biomagnification" sections for each chemical.

Rand (1995) defines biomagnification as the "result of the processes of bioconcentration and bioaccumulation by which tissue concentrations of bioaccumulated chemicals increase as the chemical passes up through two or more trophic levels." Rand further states that the transfer of chemicals from food to consumer are efficient enough so that residue concentrations icrease systematically from one trophic level to the next. EPA considers biomagnification to increase the risk of adverse effects of waterborne chemicals, but demonstration of biomagnification alone is not classified as an effect to listed species.

III. ANALYSIS OF EFFECTS OF TOXIC POLLUTANTS TO WILDLIFE

Mammals

Woodland caribou, northern Idaho ground squirrels, and grizzly bears in Idaho are primarily vegetarians (Almack, 1985; FWS, 1994c). Gray wolves and lynx consume prey that are primarily vegetarian. These mammals should not be exposed to harmful concentrations of toxic pollutants as a result of exposure to contaminated aquatic organisms since they do not consume fish. Their primary route of exposure is through ingestion of water. Bald eagles and peregrine falcons do consume fish on a regular basis and may be exposed to aquatic contaminants through dietary exposure.

Water quality criteria for human health were considered to be protective of all threatened and endangered mammals. The human health criteria protect against long term health effects. These effects range from cancer to reproductive and neurological impairments. The toxicity endpoints are related to human health, however these endpoints are usually derived from laboratory studies of rats and mice. This interspecies extrapolation for all mammals is accounted for in the modifying factors used to derive the toxicity endpoints.

The exposure equation used to derive the criteria for non-carcinogenic effects for humans is:

$$C = \underline{(RfD \times WT) - (DT + In) \times WT}$$

WI = (FC X L x FM x BCF)

C =	updated water quality criterion (mg/L)
RfD =	oral reference dose (mg toxicant /kg human body weight/day)
WT =	weight of an average human adult (70kg)
DT =	dietary exposure (other than fish) mg toxicant/kg body weight/day)
IN =	inhalation exposure (mg toxicant/kg body weight/day)
WI =	average human adult water intake (2 l/day)
FC =	daily fish consumption (kg fish/day)
L =	ratio of lipid fraction of fish tissue consumed to 3%
FM =	food chain multiplier (from Table 3-1)
BCF=	bioconcentration factor (mg toxicant/kg fish divided by toxicant/L water) for fish with 3% lipid content.

While the exposure assumptions for developing the human health criteria used to estimate risks are based on human data, these assumptions should apply to any mammal with a body weight of 70 kg (as body weight decreases, exposure increases), a drinking water consumption rate of 2 liters per day, and a fish consumption rate of 6.5 g per day. The exposure duration for non-cancer endpoints will vary depending on the chemical effect and the condition of the population at risk. The exposure duration for carcinogens is 70 years. Since, the exposure assumptions for the mammals other than humans is unknown there is additional uncertainty which may increase or decrease the risk for these species.

The possibility of exposure to toxic pollutants via contamination of plant materials in aquatic systems is unlikely as well. Generally, the herbivorous species do not feed in or very near to aquatic habitats. From this information, EPA has determined that the approval of the **acute and chronic numeric criteria for toxic pollutants** established by the Idaho Water Quality Standards **is not likely to adversely affect the gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, and woodland caribou**.

Birds

Several models were examined to determine dietary levels of toxicants in organisms exposed to parameters at the adopted water quality criteria concentrations. Often, a model requires wildlife values that are unavailable for the species of concern, or the concentration of the chemical in the sediment is needed. For fish, even if a BCF or BAF is available for a particular species, the wildlife value may not be available. Also, the more complicated models require many assumptions that can cover a wide range. For example, feeding rates, amount of diet comprised of a "contaminated" food source, potential food source trophic levels, metabolic rates, and sensitivity factors can vary by orders of magnitude. The lowest tissue concentration of a chemical in the diet that will not cause adverse effects, the NOAEL, is also expressed as "wildlife value" or "body burden". These wildlife values can cover a large range for the same organism depending on the researcher's assumptions. Given the latitude in variables such as those mentioned above and the specific requirements of the food chain/wildlife models, a general approach was chosen to estimate effects on birds. The example at the end of this section shows this approach. To estimate the effects of an adopted water quality criterion on "higher" organisms, raptors were selected. Specifically, the bald eagle and peregrine falcon are species of concern. The bald eagle and peregrine falcon are both listed under the Endangered Species Act (ESA) for Idaho. For the higher priority chemicals, no wildlife-diet values are available for these bird species. Wildlife values for other bird species or alternately, general wildlife values are available. For many of the parameters of concern, BAFs/BCFs are available for fish, or more specifically, for trout. Since eagles may feed at least somewhat on fish, if a fish BAF is available for a particular parameter, then a general wildlife exposure to an eagle can be estimated for that parameter. BCFs in aquatic life allow for the general approach presented below (that is, substituting a BCF for lack of a BAF). In estimating the dietary exposure for birds, EPA made the assumption that the bird's diet consists only of fish and that all fish eaten were contaminated.

Equation to estimate toxicant exposure to birds through diet:

> toxicant (mg/L) X BCF or BAF (mg/kg in fish/ mg/L in water) = mg/kg in diet (assuming 100% fish diet)

IV. ANALYSIS OF EFFECTS OF TOXIC POLLUTANTS TO PLANTS

The four threatened or endangered plant species in Idaho do not exist in areas constantly inundated by water, therefore the effects of aquatic contaminant exposure should be minimal. The Ute ladies' tresses is a terrestrial orchid species that is only periodically exposed to surface waters. This species generally inhabits river shores where inundation occurs infrequently (Sheviak, 1984). McFarlane's four o'clock, also a terrestrial plant species, occurs in grassland habitats characterized by warm and dry conditions (FWS, 1997b). Exposure to surface water would generally occur in these areas only during rare flooding events when dilution of contaminants and length of exposure to contaminated water would minimize toxicity. Water howellia, an aquatic macrophyte, grows mostly in wetlands associated with temporary water bodies such as ephemeral glacial pothole ponds and former river oxbows (FWS, 1994b). This plant requires the seeds to dry out completely for germination to occur. The Spalding's catchfly primarily inhabits prairie or steppe grassland vegetation and does not tolerate extremely wet soils. Therefore, because of the lack of exposure to contaminants in aquatic systems, EPA has determined that the approval of the **acute and chronic numeric**

criteria for toxic pollutants established by the Idaho Water Quality Standards **is not likely to adversely affect the water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's Catchfly**.

V. CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, Tribal, local or private actions on endangered or threatened species or critical habitat that are reasonably certain to occur in the action area considered in this biological assessment. Future federal actions or actions on federal lands that are not related to the proposed action are not considered in this section .

Future anticipated non-Federal actions that may occur in or near surface waters in the State of Idaho include timber harvest, grazing, mining, agricultural practices, urban development, municipal and industrial wastewater discharges, road building, sand and gravel operations, introduction of nonnative fishes, off-road vehicle use, fishing, hiking, and camping. These non-Federal actions are likely to continue to adversely affect endangered and threatened species.

There are also non-Federal actions likely to occur in or near surface waters in the State of Idaho which are likely to have beneficial effects on the endangered and threatened species. These include implementation of riparian improvement measures, best management practices associated with timber harvest, grazing, agricultural activities, urban development, road building and abandonment and recreational activities, and other nonpoint source pollution controls.

VI. CRITICAL HABITAT

The only listed species with designated critical habitat in Idaho are the Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River sockeye salmon.

Description of Salmon Critical Habitat

NMFS has designated critical habitat in Idaho for Snake River spring/summer chinook salmon, Snake River fall chinook salmon, and Snake River sockeye salmon. As required by Section 7 of the ESA and the implementing regulations at 50 CFR Part 402, EPA has used the best available scientific data to determine whether the action is likely to "destroy or adversely modify the designated critical habitat of the listed species". The consultation regulations define the statutory term "destruction or adverse modification" of critical habitat to mean:

...a direct or indirect alteration that appreciably diminishes the value of critical

> *habitat for both the survival and recovery of a listed species. Such alterations include, but are not limited to, alterations adversely modifying any of those physical or biological features that were the basis for determining the habitat to be critical.*

The Federal Register (Vol 58 No. 247, December 28, 1993) final rule designates critical habitat and defines and describes habitat and its essential features as follows:

> *Essential Snake River salmon habitat for both chinook and sockeye consists of four components: 1) spawning and juvenile rearing areas, 2) juvenile migration corridors, 3) areas for growth and development to adulthood, and 4) adult migration corridors.*
>
> *<u>Spawning and rearing areas</u>:*
> *The essential features of the spawning and juvenile rearing areas of the designated critical habitat for Snake River sockeye salmon consist of adequate: 1) spawning gravel, 2) water quality, 3) water quantity, 4) water temperature, 5) food, 6) riparian vegetation, and 7) access.*
>
> *The essential features of the spawning and juvenile rearing areas of the designated critical habitat for Snake River spring/summer and fall chinook salmon are: 1) spawning gravel, 2) water quality, 3) water quantity, 4) water temperature, 5) instream cover/shelter, 6) food for juvenile salmon, 7) riparian vegetation, and 8) living space.*
>
> *<u>Migration corridors</u>:*
> *Essential features of the juvenile migration corridors for Snake River sockeye salmon and Snake River spring/summer and fall chinook salmon consist of adequate: 1) substrate, 2) water quality, 3) water quantity, 4) water temperature, 5) water velocity, 6) cover/shelter, 7) food, 8) riparian vegetation, 9) space, and 10) safe passage conditions.*
>
> *Essential features of the adult migration corridors for Snake River sockeye salmon and Snake River spring/summer and fall chinook salmon include adequate: 1) substrate, 2) water quality, 3) water quantity, 4) water temperature, 5) water velocity, 6) cover/shelter, 7) riparian vegetation, 8) space, and 9) safe passage conditions.*
>
> *<u>Growth and Development:</u>*
> *The areas in the Pacific Ocean that threatened and endangered salmon use for growth and development are not well understood; therefore, NMFS has not designated any essential areas and features for Snake River ocean habitat.*

Analysis of Effects of Criteria for Toxic Pollutants on Salmon Critical Habitat

To determine whether EPA's approval of Idaho's numeric criteria for toxic pollutants is likely to adversely affect critical habitat, EPA has identified possible threats to the essential features of habitat. In evaluating the effects of the action on critical habitat, EPA concluded that the water quality parameters considered in this consultation are an integral part of all the species' habitats. Chapter 3 of this document presents information describing the analysis of effects of specific water quality criteria to Snake River salmon.

Water quality standards for toxic chemicals characterize and define the conditions and quality of surface waters. EPA's approval of Idaho's water quality standards may directly and/or indirectly affect spawning gravels and food which are essential features of salmon habitat.

The concentration of toxic chemicals in the water column should not affect the following essential features of critical habitat: temperature, water quantity, riparian vegetation, access, instream cover/shelter, space, safe passage conditions, water velocity and substrate. Therefore, **EPA's approval of Idaho's numeric criteria for toxic pollutants addressed in this biological assessment is not likely to adversely affect these essential features of critical habitat of Snake River salmon.**

Spawning gravels. Toxic chemicals may sorb to sediments and accumulate in the benthic areas of water bodies. These can remain as potential sources or sinks for pollutants. EPA is in the process of developing sediment criteria for toxic chemicals. These criteria should provide additional protection for salmon habitat. In addition, criteria which limit the quantity of settleable solids will provide additional means for reducing exposure of fish to contaminated gravel beds. Gravel, being coarse and low in organic matter does not tend to accumulate either organic pollutants or metals.

Food sources. Based on the available information, this analysis indicates that the chronic mercury criterion and chronic selenium criterion may have the potential to adversely affect Snake River salmon. Because the criteria set the allowable concentrations of these pollutants in surface waters in Idaho, EPA has determined that the approval of these criteria may have the potential to affect food in critical habitat.

The effect of consuming contaminated food is discussed in the "Biomagnification and Bioaccumulation" section for each water quality criterion. The decline of prey due to exposure to toxic chemicals impacts growth, reproduction, and survival of prey species. The effect of the decline of individual prey species on food supply is unknown. Without this information, EPA is unable to determine whether this may have the potential to adversely affect food as an essential feature of critical habitat.

Research does document mercury and selenium biomagnification in aquatic food chains (Lemly and Smith, 1987; Lemly, 1985; Wren and MacCrimmon, 1986). Therefore, Snake River salmon may encounter harmful concentrations of mercury and selenium through biomagnification of these chemicals through prey. However, the efficiency of metal transfer through macroinvertebrates may not allow absorption of metal concentrations high enough to harm the fish (Reinfelder and Fisher, 1994). No evidence has been found describing effects to salmon through biomagnification of mercury and selenium in the food.

Determination

Although the above analysis indicates that Idaho's chronic criteria for mercury and selenium may have the potential to affect food as essential features of critical habitat, these effects alone would not be significant enough to appreciably diminish the value of critical habitat for both the survival and recovery of Snake River salmon.

Although the potential may exist for some elements of critical habitat to be adversely affected, other elements are not likely to be affected. Consequently, these effects are not likely to "result in significant adverse effects throughout the species' range or appreciably diminish the capability of the critical habitat to satisfy essential requirements of the species". Therefore, **EPA has determined that the approval of these provisions is not likely to destroy or cause an adverse modification to designated critical habitat of the Snake River sockeye, Snake River spring/summer chinook salmon, and Snake River fall chinook salmon.**

The analysis in Chapter 3, indicates that all remaining numeric toxic criteria which were evaluated were not likely to adversely affect Snake River salmon. **Therefore, these remaining criteria are not likely to adversely affect water quality or food as essential features of critical habitat of Snake River salmon.**

VII. SUMMARY OF DETERMINATIONS FOR INVERTEBRATES, FISH, WILDLIFE AND PLANTS

The following determinations of "not likely to adversely affect" were made:

Aldrin/Dieldrin, Chlordane, Chromium III and VI, DDT, Endrin, Heptachlor, Lindane, Nickel, PCBs, Pentachlorophenol, Silver, Toxaphene: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, whooping crane, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Arsenic Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Arsenic Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Cadmium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Copper Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Cyanide Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Endosulfan Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Lead Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, whooping crane, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Mercury Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Mercury Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Selenium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Selenium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Zinc Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

The following determinations of “likely to adversely affect” were made:

Chronic Mercury Criteria: peregrine falcon, bald eagle, and whooping crane.

Chronic Selenium Criteria: Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, peregrine falcon, bald eagle, and whooping crane.

VIII. UNCERTAINTY ANALYSIS

Water quality criteria are designed to provide protection at a large scale. They are not designed to fit all conditions and all species. Since these are generic rather than specific criteria they include a number of assumptions, defaults, and simplifications which results in some uncertainty in EPA’s determinations. These uncertainties are divided into 5 categories: generic criteria, surrogate species, sensitivity of different life stages, loss of prey species, dietary exposures, bioavailability, sediment exposures, chemical mixtures, and background water quality conditions including temperature, dissolved oxygen, alkalinity, conductivity, total dissolved solids, carbon, pH, and hardness.

Generic criteria

EPA’s use of all relevant data under a standard methodology is an attempt to reduce uncertainty in study design or results. However, this may result in the elimination of single studies which may identify critical pathways of exposure or toxicological endpoints not accounted for by the method of combining study results. Thus, in an attempt to assure high quality data are included in this combined approach, EPA’s method may eliminate the lowest effect concentration reported in the literature.

Surrogate species

The analysis of the potential effects of toxic pollutants on threatened and endangered species included the examination of research conducted primarily with surrogate species. The surrogate species were selected as the closest related organism for which information was available. The best surrogates would live in the same environment and consume the same food as the listed species. For example, little research exists describing the effects of toxic chemicals on chinook and sockeye salmon, but a wealth of information exists describing the effects of toxic chemicals on rainbow trout. Therefore, rainbow trout often served as a surrogate species to determine the effects of toxic pollutants on chinook and sockeye salmon.

Sensitivity of different life-stages

Sublethal effects of toxicant exposure on multiple life stages of salmonids have not been completely identified. For returning spawning adults, the potential effects on the population could be quite large and catastrophic. Some potential effects include disruption of reproductive cues or migration. The development of the criteria involved data on rainbow trout at a few life stages under acute exposures. The potential effects of some chemicals to different salmonid life stages have not been fully evaluated, and this lack of evaluation does limit the accuracy with which we may estimate the protection offered by the criteria. Further research into the effects of contaminants on all salmonid life stages is needed.

Loss of Prey Species

The analysis of the criteria did not address the effects of the criteria on prey items of individual species or on their habitat beyond the water column. Toxic chemicals may affect aquatic organisms via ingestion (of contaminated prey or sediment particles) or through absorption (from water or from sediment). Furthermore, prey populations may decrease as a result of chemical contamination, thus depriving a species of food sources. The development of the criteria included effects for many prey species and should adequately protect prey of the listed species examined in this document.

Dietary exposures

Many fish species are among the top consumers in aquatic ecosystems, and as a result, diet-borne pollutants can represent a unique hazard as they are transferred through the food chain. Exclusive use of water column criteria (either dissolved or total recoverable) may underestimate the toxicity of an aquatic system by excluding ingestion of particulates or ingestion of prey that consume particulates as a pathway for toxic chemical exposure. Evidence for ingestion of prey as an exposure pathway has been discussed by Kiffney and Clements (1993). Studies have correlated metal-contaminated diets with adverse effects on salmonids (Woodward et al., 1994; Farag et al., 1994; Woodward et al., 1995). Dallinger et al. (1987) also describes a "food chain effect," where metal-impacted systems may become dominated by metal-tolerant prey organisms, such as certain aquatic invertebrates. These invertebrates tolerate high metals concentrations by storing metals in vacuoles. Fish may be negatively affected by consuming the metal-rich prey. Evidence for the "food-chain effect" is provided by Woodward et al. (1994). The application of water column criteria is intended to protect water column organisms from exposure to metals from the water column. Little connection exists between the establishment of water column concentrations to protect against toxicity to aquatic organisms and the degree to which metals might accumulate in sediments and/or accumulate in benthic organisms that serve as prey for fish and other organisms. The existence and extent of metal accumulation in sediments is dependent on site-specific physical and chemical conditions. Accordingly, the degree of metal accumulation can not be inferred from water column criteria, whether total or dissolved. EPA recognizes that there is residual uncertainty regarding dietary metal exposure.

Research is needed to better understand the relative importance of food versus water in the transfer of metals to juvenile salmonids and in the development of toxic effects associated with uptake of metals. Other tools that could increase protection of endangered species from the threat of dietary exposure would be the development of sediment criteria, wildlife criteria, and bioaccumulation indicators.

Bioavailability of metals

Bioavailability of individual compounds was based on the likelihood of biological uptake from the water column. Metals in the water column will also partition into a solid or particulate phase depending on the sorption properties of the metal and particulate materials as well as the chemical condition (pH, etc) of the surrounding water. In a workshop hosted by the Society of Environmental Toxicology and chemistry in 1996, it was concluded that metals criteria should be expressed in terms of the dissolved fraction (Bergman and Doward-King 1997). There is a disagreement among scientists regarding the desorbtion of metals at the gill surface. Some scientists consider metal sorbed to sediments to be unavailable for biological uptake through the gills (Bergman and Dorward- King 1997). EPA scientists belived that gill uptake of particulate metal is generally insignificant.

As is the case with many scientific issues, EPA recognizes that it would be optimal to undertake additional study to better define the relative importance of particulate-bound metal. If such work were to indicate that the particulate pathway was significant compared to the dissolved pathway, the EPA would need to determine how to revise its procedures for deriving aquatic life criteria to account for this pathway. Currently, there is no scientific consensus on how to do this.

To improve the accuracy and reliability of its water quality criteria, EPA is developing a Biotic Ligand Model to evaluate aquatic life exposure to metals via membranes (i.e. gills) in contact with the water. Interaction with (including uptake through) such membranes apears to be the dominant mechanisms affecting the expression of toxicity of metals.

The Biotic Ligand Model is an inorganic geochemical equilibrium model coupled with a metal-organic matter complexation model and metal-biotic membrane complexation model. The biotic membrane is viewed as competing for uptake of the metal with other organic and inorganic ligands in the water, hence the term “biotic ligand”.

Sediment exposures

To protect the aquatic organisms against toxicity due to chemical contamination of sediments, EPA has developed an Equilibrium Partitioning Sediment Guideline. EPA is developing these sediment guidelines to:

1) determine the total extractable metal portion of the sediment which does not exceed the acid sulfide concentration in the sediment

2) deffierentiate between pore water chemical concentration and sediment bound concentrations

3) protect against chronic toxicity to benthic organisms from metals in sediment
4) apply to cadmium, copper, lead, nickel, silver, and zinc.

This approach is presently undergoing EPA's Science Advisory Board review.

Chemical Mixtures

The Idaho Water Quality Standards aquatic life criteria do not take into account the interactions between two or more chemicals which could be present in a water body. Some chemicals may interact resulting in more or less toxicity of one or more of the chemicals involved. Some metals such as cadmium and selenium exhibit antagonistic relationships with respect to toxicity (Furness and Rainbow, 1990). The literature provides little evidence to indicate synergistic interactions between metals (Furness and Rainbow, 1990). Synergism is defined as the interaction of toxicants resulting in greater toxicity than that predicted by the sum of the toxicities of each chemical. However, pollutant discharges such as those released by permitted dischargers are unique mixtures of elements. Research studies generally focus on the most abundant elements without reference to others present in a complex mixture. Synergistic, antagonistic, and additive biological effects are possible for fish exposed to mixtures. Categorizing elemental mixtures as synergistic, antagonistic, or additive depends on the element concentrations, solubility, and ratios to other elements, as well as the water hardness, measured parameters, species considered, and other factors (Sorenson, 1991).

One way to account for the interactions of contaminants is to use the Toxic Unit approach (see Pulley et al., 1998 and Wildhaber and Schmitt, 1998 for examples) or the Hazard Quotient method (US EPA, 1998). On a statewide basis, this approach would be neither practical nor relevant; however, on a site-specific basis, mixtures can be defined. At the present time EPA water quality criteria do not account for addivity of exposure for multiple contaminants.

Background water quality conditions

Toxicity of several pollutants for which criteria are included in the Idaho Water Quality Standards are either pH or hardness dependent. In these cases, the State's criteria are expressed as a function of pH or hardness. However, in many cases, the literature does not report the environmental conditions under which toxicology experiments have been performed, including pH and hardness. Where relevant, EPA's analysis took into account whether pH and hardness values were provided. Where pH and hardness values were not reported in the literature and the criteria are expressed as a function of pH or hardness, the results should be interpreted with caution.

EPA has considered hardness to represent not only calcium and magnesium, also to be a surrogate for two other parameters, alkalinity and pH, which co-vary with hardness in natural waters. Current thinking is that the hardness relationships work primarily through the combined effects of calcium, carbonate, and pH. However, until the development of the biotic ligand model, it has not be feasible to isolate the separate effects of these parameters. The biotic ligand model will allow more accurate prediction of toxicity in waters having unusual combinations of hardness, alkalinity, and pH.

Hardness cap for metals criteria. In the NTR, EPA described and required minimum and maximum hardness values (25 mg/L and 400 mg/L as $CaCO_3$, respectively) to be used when calculating hardness dependent freshwater metals criteria. Most of the data EPA used to develop the criteria formulas were in the hardness range of 25 to 400 mg/L. Therefore, EPA stated that the formulas were most accurate in that range. Using a hardness of 25 mg/L for calculating criteria, when the actual ambient hardness is less than 25 mg/L, could result in criteria that are not protective of aquatic life. The State has the option of using ambient hardness values outside the 25-400 mg/L range when calculating criteria for hardness dependent metals.

For reference, average, minimum, and maximum hardness measurements recorded in waters throughout the State of Idaho are presented in Appendix F. Hardness values observed throughout the State range from 14.07 mg/L in the Upper Selway River to 404 mg/L in the Lower Bear River, with an average of 103.8 mg/L. Literature describing the experiments referenced in this section did not always provide hardness values along with data. In cases where hardness values are lacking, comparisons of criteria to research results may not be reliable. For those metals which are hardness dependent, EPA Region 10 calculates NPDES permits limits and load allocations for TMDLs using the fifth percentile of the ambient and or effluent hardness values which are most often calculated from instantaneous data.

pH. The toxicity of several pollutants vary depending upon environmental conditions such as water hardness and pH. pH activity has a significant impact on the availability and toxicity of metals. The following is summarized from Elder (1988) and Baker et al. (1990) IN ODEQ (1995). Metal-hydroxide complexes tend to precipitate (i.e., reduced ability to remain suspended) and are quite insoluble under natural water pH conditions; thus, the metal is not able to exert a toxic effect. However, the solubility of these complexes increases sharply as pH decreases. pH activity also impacts the sensitivity of organisms to a given amount of metal. Each metal has its own range where pH and site-specific conditions become factors in the metal's bioavailability. Aluminum is the metal of greatest concern at low pH values. The effects of low pH are also more pronounced at low concentrations of calcium. No adverse effects to listed species due to pH-driven changes in metal toxicity (where the metals comply with the respective metals criteria) would occur in the range of Idaho's pH criteria. In summary, reductions in pH below natural levels will tend to increase metal availability and toxicity.

Temperature. No single pattern exists for the effects of temperature on the toxicity of pollutants on aquatic organisms. Temperature change in a given direction may increase, decrease, or cause no change in toxicity depending on many factors including the toxicant, species, or the experiment. Sprague (1985) demonstrates that the effects of temperature on acute toxicity are diverse, but for the most part are only small or moderate. Some evidence suggests that temperature may not have much effect at all on the chronic "no-effect" thresholds of pollutants. One study described that temperature may either increase or decrease the EC_{50}, but no general pattern was evident. The researchers concluded that temperature had no effect on the EC_{50} (Sprague, 1985).

pH and temperature effects on cyanide. The maximum temperature allowed by Idaho's water quality standards is 33°C (warm water biota), while the pH criterion requires that surface waters fall within a range of 6.5-9.0. Below a pH of 9.2 CN increasingly converts to HCN until, at a pH of 7.0, nearly all free cyanide exists as HCN. However, below pH of 8, only about 6% of total cyanide is present as free cyanide, thus any increase in cyanide toxicity due to free cyanide will be minimal (Stein, personal communication, 2000). Eisler (1991) also notes that the toxicity of simple cyanide complexes will not be measurably affected below pH 8.3. Acidification of dilute cyanide solutions (defined as milligrams per liter) will not initiate any greater release of HCN (the aquatic life criteria for cyanide are 22 and 5.2 µg/L). Temperature effects on the toxicity of cyanide reported in the literature vary with test species, life-stage of the species, concentration of cyanide, temperature range, and other conditions. Temperature decreases will increase toxicity of cyanide over long exposures to low concentrations (< 10 µg HCN/L); however, temperature increases will decrease cyanide toxicity at higher concentrations. Life stage of fish also affects the sensitivity to cyanide at varying temperatures. The LC_{50} for rainbow trout eggs increased with decreasing temperature; whereas the LC_{50} for juvenile rainbow trout decreased with decreasing temperature (Eisler, 1991). Additional studies with warm water fish (21.5°C-31.4°C), snails, insects, and plankton showed increasing toxicity associated with increasing temperature when cyanide levels ranged between 0.2-3.2 mg/L (Sarkar, 1990).

Dissolved Oxygen. Reductions in dissolved oxygen may increase the toxicity of aquatic pollutants, but are often not the major factors affecting toxicity. Most evidence suggests that tests conducted at low and high levels of dissolved oxygen may change toxicity by only a factor of 2 or less (low dissolved oxygen being generally in the range of 20% saturation). In studies where low dissolved oxygen significantly modified LC_{50}s, the same effect did not hold true for sublethal toxicity (i.e. growth). Low oxygen appears to be less important than might be expected as a modifier of sublethal toxicity. Sprague suggests that while the picture of the influence of dissolved oxygen on toxicity is incomplete, "the effects may be as small as, or even smaller, than the modest effects on acute lethality" (Sprague, 1985). From this information, it appears that when state waters comply with the dissolved oxygen standard (> 5 mg/L for warm water, > 6 mg/L for cold water), dissolved oxygen levels are unlikely to affect toxicity.

Dissolved Organic Carbon. Dissolved organic carbon can impact toxicity of some metals, such as copper. In the case of copper, as dissolved organic carbon decreases, copper toxicity increases (Sorenson, 1991). Research over the last 20 years indicates that dissolved organic carbon is important for determining metal toxicity and is especially important in rivers where dissolved organic carbon is very low (0-5 mg carbon/L). However, the studies used to develop EPA's criteria generally included water with low organic carbon, ideally representing a worst case (most toxic) scenario.

Recently, the Virginia Association of Municipal Water Agencies proposed a modification to the ambient water quality criteria for copper (Stein, personal communication, 2000). The equation is:

$$\text{Chronic criteria} = e^{(0.8545 \times \ln(\text{hardness}) + 1.27 \times \ln(\text{TOC}) - 2.903)}$$

TOC is defined as total organic carbon. This equation is based on research showing the effects of hardness and organic carbon on copper toxicity. In streams where the hardness and dissolved organic carbon are low, the copper criterion value will be very low. For example, in streams with a hardness of 20 ppm (as $CaCO_3$) and dissolved organic carbon levels of 2 mg/L, the chronic criteria would be 1.7 µg/L using the above equation. Hardness and dissolved organic carbon levels this low do occur in certain freshwater streams.

EPA criteria are developed from tests in waters with very low DOC or TOC. The Virginia equation will yield the same result as the 1995 update EPA equation if the TOC is set at approximately 2.5 mg/L. The Virginia equation was not designed to predict toxicity in waters having lower TOC, rather it was intended for waters with high TOC. BLM related work suggests that the acute tests on which EPA's criterion is based were perhaps in the range 0.5-1.0 mg/L DOC. EPA's criterion equation should be reliable to this level of DOC.

IX. STRATEGY FOR REDUCTION IN UNCERTAINTY OF WATER QUALITY CRITERIA FOR THE PROTECTION OF THREATENED AND ENDANGERED SPECIES

Bioavailability of metals and water quality conditions:

1. EPA has funded long-term research and modeling efforts to assess the speciation and toxicity of metals as they are affected by such factors as pH, dissolved organic carbon, and hardness. These efforts, known as the Biotic Ligand Model, are intended to more accurately predict the bioavailability of metals. Most of the data used to develop the Biotic Ligand Model involved copper. As part of the agreements negotiated under Section 7 of the Endangered Species Act for the consultation over the California Toxics Rule, EPA has agreed to continue development of the Biotic Ligand Model for other metals. At this time EPA does not have a definitive schedule for finalizing the biotic ligand model. It is still in the research phase. As of May 17, 2000, further validation work is currently being performed.

2. EPA in cooperation with the Services will issue a clarification to the *Interim Guidance on the Determination oand Use of Water-Effect Ratios for Metals* (EPA 1994) concerning the use of calcium-to-magnesium ratios in laboratory water.

3. Idaho DEQ and EPA Region 10 will work collaboratively to develop site specific determinations for adding a margin of safety at sites where there is a realistic reason for concern that particulate metal might contribute to toxicity to T&E species that are sensitive to the metal(s) of concern.

Species sensitivity and chemical specific uncertainties

4. EPA will revise its recommended 304(a) acute and chronic aquatic life criteria for *selenium* by January 2002. In revising these criteria, EPA Region 10 will cooperate with Region 9, EPA Headquarters', and the Services. Scientists will be invited to peer review documents and participate in discussion sessions.

5. EPA, Region 10 will review the *mercury* criterion developed by EPA Headquarters', Region 9, and the State of California with respect to federally listed species in Idaho. The Services and Region 10 will determine if the criterion is protective. need to clarify that EPA is developing a revised human health criteria for mercury by january 2002. EPA will work with Idaho to propose revised criteria in Idaho by January 2003.

6. EPA Region 10 will work with EPA Region 9 and EPA Headquarters' to review the chronic aquatic life criterion for *pentachlorophenol.* They will determine if the criterion is protective of federally listed species under varying abiotic conditions. By March 2001, EPA will review and if necessary revise its recommended 304(a) chronic aquatic life

criterion for PCP sufficient to protect federally listed speceis an/or their critical habitats. If EPA revises its recommended 304(a) criterion, EPA will then work with Idaho to propose the revised PCP criterion in Idaho by March 2002.

7. EPA Region 10 will work with EPA Region 9 and EPA Headquarters' to revise the chronic aquatic life criterion for *cadmium* so that it will be protective of salmonids by no later than January 2001.

8. EPA Region 10 will review the schedule and plan for updating the aquatic life criterion for *copper* by August 2000. The Services and EPA Region 10 will determine if the plan for updating the criteria will provide protection for salmonids.

Sediment exposure

9. By December 2000, EPA in cooperation with the Services, will develop sediment criteria guidelines for cadmium, copper, lead, nickel and zinc, and by December 2002, for chromium and silver. When the above guidance for cadmium, copper, lead, nickel and zinc is completed, Region 10, in cooperation with the Services and Idaho, will draft implementation guidelines for the State of Idaho to protect federally listed threatened and endangered species and critical habitat in Idaho

Site specific variability, dietary exposures, other routes of exposure

10. By June 2003, EPA , in cooperation with the Services , will develop a revised criteria calculation model based on best available science for deriving aquatic life criteria on the basis of hardness (calcium and magnesium), pH, alkalinity, and dissolved organic carbon (DOC) for metals. This will be done in conjunction with EPA intiating a process to develop a national methodology to derive site-specific criteria to protect federally listed threatened and endangered species, including wildlife, in accordance with the draft MOA betweeen EPA and the Services concerning section 7 consultations

Wildlife exposures and chemical mixtures

11. EPA will cooperate with HQ's and other regions to develop a national methodology to derive site specific criteria to protect federally listed threatened and endangered species, including wildlife. These methods will address exposure to multiple stressors, mixtures, and abiotic driving forces (pH, temperature, dissolved oxygen, dissolved organic carbon, hardness, etc).

12. The Services and EPA have agreed on the need for wildlife criteria research and methods. The strategy for completion of this effort will be done cooperatively with EPA, the Services, academia, and other interested individuals or groups. EPA will complete a Request for Assistance on wildlife assessments. This solicitation will be released to the public by May 2000.

Sensitive life stages and surrogate species

13. EPA's Office of Research and Development is developing a research strategy to evaluate the effects of toxic chemicals on the life stages of a variety of fish, invertebrates, plants, and wildlife (1999 Draft Wildlife Research Strategy).

Bioaccumulation

14. Based on peer review and public comment, EPA has revised the methodology for deriving national bioaccumulation factors. This methodology acknowledges three chemical classes for these factors (nonionic organics, ionic organics, and inorganic/organometallics).

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BIOLOGICAL ASSESSMENT OF THE IDAHO WATER QUALITY STANDARDS FOR NUMERIC WATER QUALITY CRITERIA FOR TOXIC POLLUTANTS

for the
U.S. FISH AND WILDLIFE SERVICE

and the
NATIONAL MARINE FISHERIES SERVICE

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EXECUTIVE SUMMARY

Section 303 of the Clean Water Act (CWA) mandates that States adopt Water Quality Standards (WQS) to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. Water quality standards consist of beneficial uses (i.e. salmonid spawning, cold water biota) designated for specific waterbodies and water quality criteria to protect uses. States have primary responsibility for developing appropriate beneficial uses for waterbodies in their State. States review and if appropriate, revise their water quality standards on a triennial basis in accordance with CWA §303(c). Also under CWA §303(c), EPA must review and approve or disapprove any revised or new standards. If EPA disapproves any portion of the state standards the state has 90 days to adopt the changes specified by EPA, after which time EPA must propose and promulgate such standards.

On June 25, 1996, EPA Region 10 completed a review of the Idaho Water Quality Standards adopted August 24, 1994. During this review EPA disapproved seven elements within the State's water quality standards. Most of these elements have since been revised by the State of Idaho and approved by EPA. These approvals are included as part of this consultation. The elements which were disapproved and not subsequently approved by EPA have been addressed through EPA promulgations and already undergone consultation and therefore are not included in this consultation.

The purpose of this Biological Assessment is to assess the potential effects of EPA's approval of Idaho's numeric water quality criteria for toxic pollutants on species and listed critical habitat listed under the Endangered Species Act (ESA). This assessment will be provided to the U.S. Fish and Wildlife Service (FWS) and the National Marine Fisheries Service (NMFS) under Section 7 of the ESA.

The following determinations of "not likely to adversely affect" were made:

> **Aldrin/Dieldrin, Chlordane, Chromium III and VI, DDT, Endrin, Heptachlor, Lindane, Nickel, PCBs, Pentachlorophenol, Silver, Toxaphene:** Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, whooping crane, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.
>
> **Acute Arsenic Criteria:** Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland

caribou, water howellia, MacFarlane's four o'clock,Ute ladies' tresses, and Spalding's catchfly.

Chronic Arsenic Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Cadmium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Copper Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Cyanide Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Endosulfan Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Lead Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, whooping crane, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Mercury Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Mercury Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Selenium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Selenium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Zinc Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

The following determinations of "likely to adversely affect" were made:

Chronic Mercury Criteria: Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, Kootenai River white sturgeon, peregrine falcon, bald eagle, and whooping crane.

Chronic Selenium Criteria: Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead and Kootenai River white sturgeon, peregrine falcon, bald eagle, and whooping crane

BIOLOGICAL ASSESSMENT OF THE IDAHO WATER QUALITY STANDARDS FOR NUMERIC WATER QUALITY CRITERIA FOR TOXIC POLLUTANTS

I. BACKGROUND INFORMATION

A. CONSULTATION HISTORY

The Idaho Division of Environmental Quality (IDEQ) conducted a Triennial Review of several of their water quality standards (standards) from 1992 to 1993, concluding in June 1994. IDEQ submitted their revised standards to the U.S. Environmental Protection Agency, Region 10 (EPA) in July 1994. In October 1995, EPA notified IDEQ that EPA had completed a review of the June 1994 standards and transmitted the results of the preliminary review with comments. By letter dated June 25, 1996, EPA approved and disapproved various provisions of Idaho's Water Quality Standards and stated that the approval is subject to completion of consultation as required under section 7 of the Endangered Species Act. Subsequently, EPA was sued by the Idaho Conservation League for failure to take timely action on Idaho's standards. On February 20, 1997, the District Court in ICL v. Browner held that EPA was obligated to promulgate standards to supersede all of those disapproved in the June 25, 1996 letter.

In March 1997, IDEQ adopted additional revisions to their standards, which were submitted to EPA for review and approval/disapproval. On May 27, 1997, EPA approved Idaho's antidegradation policy, numeric criteria for toxic substances and use designations for two waterbody segments. On June 25, 1997, IDEQ submitted additional revisions to their standards. This package was reviewed by EPA, and by letter date July 15, 1997, EPA approved use designations for undesignated surface waters, mixing zone policy, and use designations for 31 waterbody segments.

EPA had commenced consultation (with FWS and NMFS) in 1993. EPA submitted a request to FWS for a species list. On October 21, 1993, EPA received from FWS a species list for Idaho. On September 2, 1994, EPA received from NMFS a species list for Idaho. By letter dated July 9, 1996, under informal consultation, EPA transmitted a biological assessment to FWS (and NMFS) and requested concurrence. By letter dated August 14, 1996, FWS did not concur with EPA's determination.

EPA continued to work on developing a revised biological assessment and had discussions with FWS and NMFS on the consultation approach and analysis of effects to the species of concern. The three agencies met on numerous occasions to discuss the consultation, scope the species' lists, and issues of concern for the consultation. Decisions were made regarding listed species most likely to be affected by the standards. EPA was in frequent contact with the Services on the content and structure of the biological assessment during its preparation.

As a result of several meetings held in 1999 between the Services, EPA, and Idaho DEQ, all agencies agreed that EPA would develop two biological assessments. One would cover EPA's approval of Idaho's numeric toxic criteria and the other assessment would cover all remaining EPA approvals of Idaho's water quality standards, which would include numeric criteria for conventional pollutants, narrative criteria, designated beneficial uses, and antidegradation, mixing zone, and variance policies.

The following is a chronology of key events in this consultation:

<u>July 22, 1993</u> EPA to FWS

EPA reviewing revisions to Idaho's water quality standards, writing to ensure standards are protective of T&E species. EPA requested comments from FWS.

<u>September 2, 1993</u> NMFS to EPA

NMFS provided a statewide species list to EPA for the State of Idaho

<u>October 21,1993</u> FWS to EPA

FWS provided statewide species list to EPA

<u>March 29, 1994</u>

FWS/EPA joint Endangered Species Act Section 7 Consultation Meeting, Seattle, WA

<u>August 31- September 1, 1994</u>

FWS/EPA joint Water Quality Standards and Section 7 Consultation Meeting, Olympia, WA. Goal of meeting was to develop approach for consultation of EPA's approval of State and Tribal water quality standards for ID, WA, and OR.

<u>November 18, 1994</u> FWS to EPA

Boise FWS sent information to EPA to assist in the preparation of a BA for Idaho water quality standards. Included in this letter was a proposed approach for the level of analysis of effects to listed species.

<u>December 8, 1995</u> Meeting summary notes to File

Meeting between EPA, FWS, NMFS, and DEQ re: Idaho water quality standards and ESA issues.

<u>January 11, 1996</u> Conference Call between EPA and FWS

Discussion on the Biological Assessment being developed by EPA

<u>January 17, 1996</u> FWS to EPA

Statewide Species List provided to EPA

<u>May 31, 1996</u> EPA to FWS

Update of the water quality standards and section 7 consultation (informal consultation)

July 9, 1996 EPA to FWS
Request for consultation on approval of Idaho's revisions and concurrence of findings of effects to listed species

July 19, 1996 FWS to EPA
Updated Statewide Species List provided to EPA

August 14, 1996 FWS to EPA
Non-concurrence letter to EPA for determinations of effects to listed species

September 18, 1996 EPA to FWS
Request for Formal Consultation on EPA's approval of revisions to water quality standards for Idaho

October 21, 1996 FWS to EPA
FWS acknowledges receipt of request for formal consultation; will be initiated upon receipt of the Biological Assessment

December 23, 1996 EPA to FWS and NMFS
EPA develops a draft Concept Paper on Endangered Species Act Consultations for Water Quality Standards

February 1997 Meeting Notes - Consultation Workshop
Workshop held with EPA, FWS, NMFS to work out approach for consultation for water quality standards in Idaho, Washington, Oregon, and Tribes

April 28, 1997 EPA publishes proposed rule for promulgation of standards to Idaho for items disapproved during the triennial review.

June 17, 1997 FWS to EPA
Updated Statewide species list - EPA promulgation of standards to Idaho

July 8, 1997 NMFS to EPA
Updated Statewide species list

July 10, 1997 EPA to FWS and NMFS
EPA requests concurrence with their determination of effect for promulgation of standards

July 21, 1997 FWS and NMFS to EPA
FWS and NMFS concurrence with EPA's effect determinations

August 25, 1997 FWS/NMFS to EPA

Joint letter to EPA re: time frame for consultation on items approved by EPA from Idaho's triennial review; agencies commitments

July 29, 1998 FWS to EPA

Statewide species list

June 3, 1999 FWS, NMFS, EPA, and Idaho DEQ

Joint meeting to discuss FWS and NMFS comments on the draft biological assessment document

July 12, 1999 FWS, NMFS, EPA, and Idaho DEQ

Joint meeting to discuss addressing NMFS, FWS, and Idaho DEQ comments on the numeric toxic criteria section of the draft biological assessment document and EPA finalizing the draft biological assessment document

September 14-15 1999 FWS, NMFS, EPA, and Idaho DEQ

Joint meeting to discuss finalizing the draft biological assessment document for numeric toxic criteria and to agree on how to address the remaining water quality standards such as conventional pollutants, policies, and use designations.

November 5, 1999 FWS, NMFS, EPA, and Idaho DEQ

Joint meeting to discuss the comments EPA received from NMFS and FWS on the draft biological assessment on Idaho's numeric toxic criteria

B. EPA'S ACTION - APPROVAL OF IDAHO'S NUMERIC WATER QUALITY CRITERIA FOR TOXIC POLLUTANTS

Pursuant to Section 303(c) of the Clean Water Act (CWA), states are required to adopt water quality standards to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. These standards must be submitted to EPA for review and subsequent approval or disapproval. States are further required to review and revise (if appropriate) their standards every three years. This process is known as the triennial review.

The State of Idaho Department of Health and Welfare, Division of Environmental Quality (DEQ) conducted their most recent triennial review from 1992-1994. Subsequently, the State adopted the Idaho 1994 Water Quality Standards regulations on June 14, 1994. These revisions were certified by the Idaho Attorney General on July 8, 1994 and submitted to EPA for approval on July 11, 1994.

Prior to this adoption, on December 22, 1992, EPA promulgated the National Toxics Rule (NTR) which imposed toxics criteria (both aquatic life and human health) on states which had not yet adopted their own numeric criteria for toxics. Because Idaho had not yet adopted numeric toxics criteria, the National Toxics Rule covered Idaho. States were provided the option of adopting toxics criteria on their own and, once approved by EPA, the State could be removed from the National Toxics Rule. As part of this triennial review, Idaho chose to adopt the National Toxics Rule by reference into their water quality standards. Upon successful completion of this consultation/conference, EPA will initiate action to withdraw Idaho from the NTR, thereby allowing for the application of the State's toxics criteria.

During 1995, DEQ adopted three revisions to the Idaho 1994 Water Quality Standards. DEQ granted a variance to the aquatic life criteria for copper, selenium, and cyanide for the Kinross DeLamar Mine discharge (2/24/95), revised the human health criteria for arsenic (4/10/95), and revised the chronic ammonia criteria for warm water and cold water biota (4/14/95). These revisions were subsequently submitted to EPA for review and approval.

On June 25, 1996, EPA Region 10 completed a review of the Idaho 1994 Water Quality Standards. Based on this review, EPA approved, subject to successful conclusion of ESA consultation, Idaho's 1994 Water Quality Standards and subsequent submittals noted above with several exceptions (see Appendix B). The approval actions taken through this letter are subject to this consultation.

The State of Idaho has subsequently revised their water quality standards to address many of the items which EPA disapproved in the June 25, 1996 letter. EPA has reviewed and either approved or disapproved each of these revisions in letters dated May 27, 1997 and July 15, 1997 (Appendix B).

Subsequent to EPA's June 25, 1996 action, the State of Idaho has also adopted several revisions to their water quality standards which have not yet been submitted to EPA for our review and approval/disapproval under Section 303(c) of the CWA. The most-notable of these revisions is a site-specific temperature criteria for waters in which the bull trout reside. EPA is currently reviewing this revision. Because EPA has yet to propose approval or disapproval of this revision there is no EPA action and these criteria are not subject to consultation.

EPA has promulgated one rule pertaining to items disapproved in the June 25, 1996 approval/disapproval letter to DEQ. Prior to finalizing this rule, EPA completed consultation under Section 7 of the ESA with FWS and NMFS (see Appendix C for Biological Assessment and Concurrence Letters on these actions). As such, items addressed in the 1997 consultation are not subject to this consultation.

In conference with NMFS and FWS, it was agreed that EPA would consult on the numeric toxic criteria developed to protect aquatic life. During the development of this assessment, FWS and NMFS staff provided EPA with a priority ranking for the 23 aquatic life toxics criteria addressed in this consultation. EPA has utilized this ranking to guide the level of effort given each analysis in this assessment. In summary, the action for this consultation is EPA's approval of Idaho's Water Quality Standards pertaining to the aquatic life numeric criteria for toxic pollutants, per letters to DEQ dated June 25, 1996, May 27, 1997 and July 15,

1997. The following is a list of the numeric criteria for toxic pollutants addressed in this Biological Assessment and grouped by level of analysis:

Priority Pollutants for Aquatic Life Criteria

High Level of Analysis:
Arsenic
Cadmium
Copper
Lead
Mercury
Selenium
Zinc
Cyanide
alpha and beta Endosulfan

Lower Level of Analysis:
Chromium (III)
Chromium (VI)
Nickel
Silver
Heptachlor/Heptachlor Epoxide
Pentachlorophenol
Aldrin
gamma-BHC (Lindane)
Chlordane
4-4' DDT
Dieldrin
Endrin
PCBs (PCB-1242,1254, 1221, 1232, 1248, 1260, 1016)
Toxaphene

These criteria are currently in effect and are applicable to waters in the State of Idaho as put forth in Section 16 of the Idaho Administrative Procedures Act, Title 01, Chapter 02 (IDAPA 16.01.02). All of these approval actions were made subject to successful conclusion of ESA consultation. The assessment will cover species currently listed as threatened and endangered and listed critical habitat under the ESA and species currently proposed for listing under the Act.

C. OVERVIEW OF THE WATER QUALITY STANDARDS PROGRAM

A water quality standard defines the water quality goals of a waterbody by designating the use or uses to be made of the water, by setting criteria necessary to protect the uses, and by preventing degradation of water quality through antidegradation provisions. The CWA provides the statutory basis for the water quality standards program and defines water quality goals. For example, Section 101(a) states in part that wherever attainable, waters achieve a level of quality that provides for the protection and propagation of fish, shellfish and wildlife, and recreation in, and on the water ("fishable/swimmable).

In addition to adopting water quality standards, states are required to review and revise standards every three years. This public process, commonly referred to as the triennial review, allows for new technical and scientific data to be incorporated into the standards.

The regulatory requirements governing the program, the Water Quality Standards Regulation (published at 40 CFR 131) sets forth specifications for the water quality standards program as well as the minimum requirements for a State water quality standards submission.

The minimum requirements that must be included in the state standards are: designated uses, criteria to protect the uses, and an antidegradation policy to protect existing uses and high quality waters. In addition to these elements, the regulations allow states to adopt discretionary policies such as mixing zones and water quality standards variances. These policies are also subject to EPA review and approval.

State's have primary responsibility for developing appropriate designated uses. These uses either reflect a water quality goal for the waterbody or a use which is actually attained. The State then sets criteria which will provide for a level of water quality such that the designated uses can be attained and protected.

Section 303(c)(2)(B) of the CWA requires the State to adopt numeric criteria for all toxic pollutants listed pursuant to the CWA section 307(a)(1) for which criteria have been published under section 304(a). The majority of EPA's water quality criteria for toxic pollutants were derived between 1980 and 1985, and were based on the latest available scientific data at that time. EPA publishes criteria documents as guidance to states. States consider these criteria documents, along with the most recent scientific information, when adopting regulatory criteria.

Once the standards are officially adopted by the state, they are submitted to EPA for review and approval. EPA reviews the standards to determine whether the analyses performed are adequate and evaluates whether the designated uses and criteria are protective and compatible throughout the waterbody. EPA makes a determination whether the standards meet the requirements of the CWA and EPA's water quality standards regulations. EPA then formally notifies the state of these results. If EPA determines that any such revised or new water quality standard is not consistent with the applicable requirements of the CWA, EPA is required to specify the disapproved portions and the changes needed to meet the requirements. The State is then given an opportunity to make appropriate changes. If the State does not adopt the required changes, regulations require that EPA promulgate federal regulations to replace those disapproved portions.

D. OVERVIEW OF IDAHO'S WATER QUALITY STANDARDS

The Idaho Water Quality Standards are codified in Section 16, Title 01, Chapter 02 of the Idaho Administrative Procedures Act (IDAPA 16.01.02). Following is a brief overview of each section of the standards. Other policies and provisions of IDAPA 16.01.02 which go beyond the scope of EPA's approval authority under Section 303(c) of the CWA are not included in this assessment. A copy of the Idaho Water Quality Standards is included in Appendix D.

<u>051 Antidegradation Policy</u> This provision establishes a three-tiered approach to maintaining and protecting water quality and uses for Idaho's surface waters. The rule includes antidegradation provisions that provide protection for all waters (Tier 1), high quality waters (Tier 2), and outstanding resource waters (Tier 3). Protection under Tier 1 states that existing uses and the level of water quality necessary to protect existing uses shall be maintained and protected. Tier 2 provides protection for waters where existing water quality is better than applicable water quality standards and states that the existing water quality shall be maintained and protected. Tier 3 provides for the maintenance and protection of the existing water quality in waters that are classified as outstanding resource waters per procedures contained in Section 055. Activities covered by this provision include both point and nonpoint sources of pollution.

<u>060 Mixing Zone Policy</u> This provision provides for an exemption from meeting numeric water quality criteria within mixing zones established for point source discharges. The policy requires that a biological, chemical, and physical appraisal of the receiving water and the proposed discharge be completed prior to the granting of the mixing zone. The provision also sets forth limitations to be considered in defining the size, configuration, and location of the mixing zone.

<u>100 Surface Water Use Classifications</u> This provision defines the designated beneficial uses for which surface waters of the state are to be protected. Use classifications include water supply - agricultural, domestic and industrial; aquatic life - cold water biota, warm water biota, and salmonid spawning; recreation - primary and secondary contact; wildlife habitats; and aesthetics. Since the water supply and recreational uses are not directly associated with the protection of threatened and endangered species, they will not be further evaluated as part of this assessment. Specific waters to which these uses are applied are identified in Section 110 through 160. Numeric criteria applicable to each use designation are contained in Section 250.

<u>101.01 Use Designations for Surface Waters - Undesignated Surface Waters</u> This provision provides that the cold water biota and primary or secondary contact recreation criteria will apply to all waters not specifically designated in Sections 110 through 160. Numeric criteria applicable to these uses are contained in Section 250.

200 General Surface Water Quality Criteria (Narrative Criteria) This section sets forth narrative water quality criteria which apply to all surface waters of the state, regardless of use classification. These include seven "free from" statements which limit the concentrations or quantities of hazardous materials; toxic substances; deleterious materials; floating, suspended, and submerged matter; excess nutrients; oxygen-demanding materials; and sediment to levels which will not impair designated beneficial uses.

250 Surface Water Quality Criteria For Use Classifications (Numeric Criteria) This section establishes numeric criteria to be applied to each designated beneficial use identified in Section 100 and established under Sections 101 and 110 through 160. Aquatic life criteria are established in subsection 250.02 and criteria for wildlife habitat is established in subsection 250.04.

Section 250.02.a. establishes numeric criteria which are to be applied to all aquatic life uses. This subsection adopts by reference the criteria values set forth in the National Toxics Rule (40 CFR 131.36(b)(1), Columns B1, B2, and D2). These criteria are displayed in Table E-1, Appendix E. This consultation only addresses the criteria adopted for the protection of aquatic life (columns B1 and B2, Table E-1).

E. OVERVIEW OF IDAHO'S SURFACE WATER QUALITY PROGRAMS

Water quality standards are important for several environmental, programmatic, and legal reasons. Control of pollutants in surface waters is necessary to achieve the CWA's goals and objectives, including the protection of all species dependent upon the aquatic environment. Water quality standards provide the framework necessary to identify, protect, and restore the water quality in Idaho's surface waters.

Water quality standards are important to State and EPA efforts to address water quality problems. Clearly established water quality standards enhance the effectiveness of many of the state, local, and federal water quality programs including point source permit programs, nonpoint source control programs, development of total maximum daily load limitations (TMDLs), and ecological protection efforts.

Data acquired during chemical, physical, and biological monitoring studies is utilized in evaluating the quality of the State's waters and designing appropriate water quality controls. Waters identified as "water quality limited" are included on the 303(d) list and reported in the 305(b) report, both submitted to EPA biennially.

Idaho identified 960 waterbody segments on their 1996 303(d) list. These segments include approximately 10,020 stream miles or ten percent of the surface waters in the State. Pollutants identified on the 303(d) list fall into several major groups which include sediment, nutrients, metals, bacteria, oxygen demand, and toxic organics. Table I.E.1 summarizes the distribution of segments and parameters exceeding the Idaho Water Quality Standards by administrative basin and statewide.

Table I.E.1 Idaho 1996 303(d) List - Parameters by Basin

Basin Name (# segs)	Sed.	Nut.	Temp.	Bct.	FA	HA	DO	Metal	Other	Total
Bear (42)	41	23	0	0	9	2	0	1	1	77
Clearwater (225)	216	67	69	67	68	73	34	1	43	639
Panhandle (190)	158	30	26	17	15	49	15	35	19	364
Salmon (118)	105	27	5	3	10	5	0	9	11	175
Southwest Idaho (187)	175	37	41	13	41	4	18	3	18	350
Upper Snake (198)	182	96	44	41	68	12	51	0	26	520
Total (960)	877	280	185	142	211	145	118	49	118	2125

Sed - sediment
Nut - nutrients
Temp - temperature
Bct - bacteria
FA - flow alteration
HA - habitat alteration
DO - dissolved oxygen

As noted in the above table, sediment is the most prevalent parameter identified on the 1996 303(d) list (over 90% of all segments include listings for sediment). Sediment can play a major role in the fate and transport of nutrients, bacteria, metals, and toxics. In addition, land disturbance which affects the input of sediment into aquatic systems is also a key consideration relative to habitat alteration. As a result, development of sediment control measures will often address, to some degree, these other parameter categories and physical components of the stream network.

Human activities, such as timber harvesting, road building, stream channelization, farming, grazing,, and urbanization have resulted in the simplification of habitat and a reduction in aquatic system quality in many of the river basins in Idaho. These activities have caused or contributed to the loss of large woody debris, loss of riparian vegetation, loss of frequency, and depth of pools, increase in temperature,sedimentation, and other effects which have reduced the habitat quality. The system of dams in the Columbia Basin has altered water flows resulting in changes in water temperatures, timing, and level of peak flows, barriers to fish migration, reductions in riparian areas, and changes in the stream physical attributes. Habitat simplification and decreased quality leads to a decrease in the health and diversity of the anadromous salmonid populations. The composition, distribution, and status of fish within the Basin are different than they were historically. Habitat loss, fragmentation and isolation may place remaining populations at risk (Quigley et. al, 1996).

Surface Water Monitoring

Surface water monitoring activities in Idaho have focused on beneficial uses and ambient water quality trends. Data from this monitoring is used to document the existence of uses, the degree of use support, and reference conditions. This monitoring is made up of primarily the collection of biological and physical data. The ambient monitoring network is designed to document water quality trends at the river basin and watershed scales through the collection of mainly water column constituent data. Biological parameters are being added to this network as well. Fifty-six monitoring stations are currently sampled on a rotating basis.

Water body Assessment

The Idaho Division of Environmental Quality has started a water body identification project to facilitate water quality assessments, reporting, and standards updating. This project was initiated through an Environmental Protection Agency grant. The funds are being passed through to the Idaho Department of Water Resources to develop a geo-referenced database and numbering protocol.

The Division of Environmental Quality has published (Division of Environmental Quality 1996b) a water quality assessment guidance document. This document describes a water body assessment process that accounts for the beneficial uses and criteria currently required in the Idaho water quality standards. This assessment was used to prepare the Draft 1998 303(d) list.

IDAHO 1998 303(D) RESULTS AS OF 98 UPDATE

		%
Total number of segments on 1994 list	962	
Total number of segments removed from 1994 to 1998	335	-35
Total number of segments added on 1998	127	+17
Total number of segments for 1998	744	
% difference between 94/98		-23
Total number of miles on 1994 list	10,656	
Total number of miles removed from 1994 to 1998	3,542	-33
Total number of miles added for 1998 list	1,046	+13
Total number of miles on 1998 list	8,160	
% difference between 94/98		-23

TMDL's for those segments new to the 1998 list will be scheduled for 2006, after completion of the existing 8 year court agreed schedule.

SUMMARY OF 1998 303(d) LIST

Pollutant Specific Lists

Listed Pollutant	#New Miles	Total # Segments	Total Miles*
Bacteria	128	127	1,738
Channel Stability	0	2	7
Dissolved Oxygen	26	159	1,145
Flow Alteration	26	159	2,047
Habitat Alteration	9	113	1,224
Mercury	0	1	0
Metals	12	43	225
Ammonia	32	26	296
Nutrients	32	214	2,754
Oil or gas	0	15	174
Organics	0	7	121
Pesticides	0	12	138
pH	9	22	210
Salinity	0	1	42
Sediment	96	573	6,483
Total Dissolved Gas	0	6	80
Temperature	9	145	1,769
Unknown	832	109	1,078

*Rounded to whole miles.

For each “water quality limited” segment on the 303(d) list, DEQ develops a TMDL. That is, DEQ determines the total amount of a pollutant (load) that the receiving waters can assimilate while maintaining water quality standards and allocates these loads to the various sources. The CWA requires that all contributing sources, both point and nonpoint, be identified and addressed in this assessment, that seasonal variations be taken into account, that a margin of safety be established to account for uncertainties and that the establishment of a TMDL will lead to the attainment of applicable water quality standards. Idaho DEQ, in association with Watershed Advisory Committees, also develops an implementation plan for each TMDL. Per Court Ordered Schedule, Idaho must complete TMDLs for all waters on the 1996 303(d) list by the end of 2005.

One principal application of EPA's approved and/or promulgated water quality standards is the National Pollutant Discharge Elimination System (NPDES) permit program. The Idaho Water Quality Standards provide guidelines for NPDES permit writers to develop conditions and limits for inclusions in such permits to point source dischargers. NPDES permits in the state of Idaho are issued and enforced by EPA, Region 10.

DEQ is responsible for the overall coordination and implementation of Idaho’s nonpoint source program. Implementation of the program is accomplished through interagency coordination with local, state, and federal natural resource agencies. The nonpoint source program is implemented with assistance from public advisory committees which provide continuous feedback on direction and acceptability of the nonpoint source control strategy.

The nonpoint source control strategy is based on the feedback loop concept: site-specific management practices are applied and monitoring is used to evaluate their effectiveness. When receiving waterbodies do not support their beneficial uses after management practice implementation, changes are implemented. Monitoring continues to ensure the revised practices are adequate to restore impaired beneficial uses.

EPA provides funding and assistance for implementing nonpoint source controls through CWA Section 106, 305(b) and 319 grants. Assistance in water quality management plan development, funding, and implementation is also available through programs of numerous state and federal natural resource agencies including the Natural Resource Conservation Service (NRCS), Soil Conservation Districts, Idaho Department of Lands and Idaho Fish and Game. Significant funding is expected to become available for nonpoint source controls over the next several years through the Clean Water Action Plan (CWAP) and several NRCS programs including the Riparian Enhancement Initiative under the Conservation Reserve Enhancement Program.

F. DESCRIPTION OF ACTION AREA

The Idaho Water Quality Standards apply to all surface waters of the state, defined as all accumulations of water, natural and artificial, public and private, or parts thereof which are wholly or partially within, which flow through, or border upon the state (IDAPA 16.01.02.003.116). EPA’s approval action does not apply to and thus the action area of this consultation does not include, any waters within Indian Country (as defined in 18 USC 1151).

II. DESCRIPTION OF THE SPECIES

A. SPECIES OF CONCERN

Pursuant to lists provided by the U.S. Fish and Wildlife Service (FWS, 1998a) and the National Marine Fisheries Service (NMFS, 1998), the following threatened and endangered species will be considered in this assessment. This list contains all species currently listed and proposed for listing under the Endangered Species Act (ESA) which are known or suspected to occur in the State of Idaho.

Taylorconcha serpenticola	Bliss Rapids snail
Lanx	Banbury Springs lanx
Physa natricina	Snake River physa snail
Pyrgulopsis idahoensis	Idaho springsnail
Valvata utahensis	Utah valvata snail
Pyrgulopsis bruneauensis	Bruneau hot springsnail
Acipenser transmontanus	Kootenai River white sturgeon
Salvelinus confluentus	Bull trout
Oncorhynchus nerka	Snake River sockeye salmon
Oncorhynchus tshawytscha	Snake River spring/summer chinook salmon
Oncorhynchus tshawytscha	Snake River fall chinook salmon
Oncorhynchus mykiss	Snake River steelhead
Haliaeetus leucocephalus	Bald eagle
Falco peregrinus anatum	Peregrine falcon
Canis lupus	Gray wolf
Ursus arctos horribilus	Grizzly bear
Lynx canadensis	Lynx
Spermophilus brunneus brunneus	Northern Idaho ground squirrel
Grus americana	Whooping crane
Rangifer tarandus caribou	Woodland caribou
Howellia aquatilis	Water howellia
Mirabilis macfarlanei	MacFarlane's four o'clock
Spiranthes diluvialis	Ute ladies' tresses
Silene spaldingii	Spalding's Catchfly

B. SNAKE RIVER AQUATIC SNAILS

The U.S. Fish and Wildlife Service has determined that there are four aquatic snails present in the middle Snake River that are in endangered status as defined by the Endangered Species Act. One additional species of snail has been determined to be in threatened status.

1. Critical Habitat

There is no critical habitat designated for any of the aquatic snails.

2. Selected Life History and Habitat Data

a. Overview of Threatened and Endangered Snake River Freshwater Molluscs

There are 42 native molluscs, including 22 (FWS, 1994d) to 27 (Frest et al., in press) native snails presently found in Idaho. Many of the species are relics of Lake Idaho and Pleistocene lakes and rivers that formed after the waters of Lake Idaho were drained (Frest and Bowler, 1992). Eighteen of these species are considered cold water species.

The following species are listed under the Endangered Species Act as threatened or endangered:

<u>Threatened</u>
1. Bliss Rapids snail, *Taylorconcha serpenticola*

<u>Endangered</u>
1. Banbury Springs lanx (undescribed *Lanx* sp.)
   Historical record from three large springs in the Hagerman Reach.
2. Snake River physa snail, *Physa natricina*
3. Idaho springsnail, *Pyrgulopsis idahoensis*
   Historically from the mainstem Snake River from Weiser to Glens Ferry. The fossil record indicates occurrence in the Pliocene Glens Ferry Formation.
4. Utah valvata snail, *Valvata utahensis*

All of these listed species are considered cold water species and are found exclusively in the Middle Snake River and its associated springs and alcoves.

The cold water molluscs in the Middle Snake River are typically dependent on cold, well-oxygenated, swift-flowing, background water for survival (Frest and Johannes, 1991; FWS, 1995b). Preferred habitat is clear water, cobble and boulder substrate; less preferable habitat includes shallow water, soft-sediment habitats, and reducing conditions with subsurface methane gas generation. (Frest and Johannes, 1991). The surviving colonies of cold water snail taxa are most likely to be found adjacent to rapids, near springs, or near the mouth of major tributaries (Frest and Johannes, 1992).

b. Bliss Rapids snail (*Taylorconcha serpenticola*) - Threatened

This species was first identified in 1959 (Taylor, 1982a) and is endemic to the Middle Snake River and Lake Idaho. It is found on stable cobble-boulder substrate only in flowing waters and in a few spring alcove habitats. It avoids sediments and attached plants. They are found in areas associated with spring influences or rapids edge environments and tend to flank shorelines. They are found at varying depths if dissolved oxygen and temperature requirements persist and are found in shallow (<1 cm, 0.5 in) permanent, cold springs (Frest and Johannes, 1992). The species is photophobic and inhabits the undersides of rocks during daylight (Bowler, 1990). The snail will migrate to graze on perilithon on the upper surfaces of rocks during the night (Frest et al. in press). The Bliss Rapids snail occurs in fast-water riffles and in a few springs and alcoves and has two color variants (Frest et al., in press). The Bliss Rapids snail lacks both lungs and gills and is particularly sensitive to oxygen fluctuations (FWS, 1994d).

The Bliss Rapids snail was known historically from the mainstem Middle Snake River and associated springs between Indian Cove Bridge (RKm 846, RM 525) and Twin Falls (RKm 982.9, RM 610.5) (Hershler 1994, cited in FWS 1997). Based on live collections, the species currently exists in discontinuous populations within its historic range. These colonies are primarily concentrated in the Hagerman reach, in tail waters of Bliss and Lower Salmon Falls Dams and several background springs including Thousand Springs, Banbury Springs, Box Canyon Springs, and Niagara Springs (FWS, 1997c).

c. Banbury Springs lanx (undescribed *Lanx* sp.) - Endangered

This lanx is a member of Lancidaea family of pulmonates endemic to western North America. At present, the Banbury Springs lanx is known to occur only in the largest, least disturbed spring habitats at Banbury Springs, Box Canyon Springs, and Thousand Springs. It was first discovered at Banbury Springs in 1988 and has not been formally described (FWS, 1994d). Other colonies have since been discovered in Box Canyon Springs (RKm 947, RM 588) and in the outflows of Thousand Springs (RKm 941, RM 584.6) (Pentec, 1991, cited in FWS, 1997c). It has only been found in spring run habitats with well oxygenated clear cold (15-16^{o}C) waters on boulder or cobble substrate. They are associated with swift currents on smooth basalt and avoid surfaces with large aquatic macrophytes or filamentous green algae. The lanx lacks both lungs and gills and respires through unusually heavy, vascularized mantles. It cannot withstand temporary episodes of poor water quality (FWS, 1995b). Localized decreases in dissolved oxygen can prove fatal for this species since respiration is accomplished only through the mantle; lungs, gills, and other specialized respiratory structures are lacking (Frest and Johannes, 1992).

d. Snake River physa snail (*Physa natricina*) - Endangered

The Snake River physa snail *(Physa natricina)* was named and described by Taylor (1988). The species occurs on the undersides of gravel to boulder substrate in swift current of the mainstem Snake River. Much of the habitat for this species is deep water (Taylor, 1982c). *Physa natricina* occurs on the undersides of rocks usually in swift water, but occasionally in more slowly flowing unimpounded reaches.

Taylor (1988, cited in FWS, 1997c) cites collections of this species from 1956 through 1985 and considers its "recent" range in the Snake River to extend from Grandview upstream through the Hagerman Reach (RKm 917, RM 573). Taylor (1988, cited in FWS, 1997c) stated that the Grandview sub-population was extirpated in the early 1980's "...as the native bottom fauna has been virtually eliminated in this segment of the Snake River." The Snake River physa was also recorded below Minidoka Dam (RKm 1086, RM 675) in 1987 (Pentec, 1991, cited in FWS, 1997c). However, recent comprehensive surveys in southeastern Idaho and northern Utah (Frest and Johannes, 1991, cited in FWS, 1997c) and in a free-flowing reach near Buhl (Frest and Johannes, 1992, cited in FWS, 1997c) failed to find live specimens. In 1997, two colonies were believed to remain in the Hagerman and King Hill reaches, with possibly a third colony immediately downstream of Minidoka Dam (FWS, 1997c).

e. Idaho springsnail *(Pyrgulopsis idahoensis)* - Endangered

The Idaho springsnail was first described by H.A. Pilsbry as *Amnicola idahoensis* (Pilsbry, 1933, cited in Taylor, 1982d). It is not found in tributaries or marginal springs (Taylor, 1982d). Respiration occurs through modified structures named ctenidium. The species is found only in permanent, flowing waters in the mainstem of the Snake River. The species occurs on mud or sand associated with gravel to boulder size substrate and is often attached to vegetation in riffles (FWS, 1995b). *P. idahoensis* occurs in sediments and beneath rocks in the flatter area of the gradient and is presently restricted to the Middle Snake River from the Bliss Dam to C.J. Strike Reservoir. This species has declined in numbers and remaining populations are small and fragmented (FWS, 1997c).

f. Utah valvata snail (*Valvata utahensis*) - Endangered

The Utah Valvata was first described as *Valvata sincera var utahensis* by Call in 1884 (Taylor, 1982b). *Valvata utahensis* lives in deep pools adjacent to rapids or in perennial flowing waters associated with large spring complexes (FWS, 1995b). The species avoids areas with strong currents or rapids. The snail prefers well oxygenated areas of non-reducing calcareous mud or mud-sand substrate among beds of submerged aquatic vegetation. It is absent from pure gravel boulder bottoms. *Valvata utahensis* lacks both lungs and gills and is particularly sensitive to oxygen fluctuations (FWS, 1995b)

Valvata utahensis occurred historically in Utah Lake in Utah and in the Snake River of southern Idaho (Taylor, 1988, cited in FWS, 1997c). Its modern range extended as far downstream as Grandview (RKm 783, RM 487) (Taylor, 1988, cited in FWS, 1997c).

In 1997, this species was known to occur in a few springs and mainstem Snake River sites in the Hagerman Valley (RKm 932, RM 579) (FWS, 1997c). Additional locations include a few sites immediately upstream and downstream of Minidoka Dam (RKm 1086, RM 675), near Eagle Rock dam site (RKm 1142, RM 709) and below American Falls Dam downstream to Burley (Taylor, 1988, cited in FWS, 1997c). Recent surveys at The Nature Conservancy's Thousand Springs Preserve revealed declines in the number and range of Utah valvata over a four-year period (Frest and Johannes, 1992; FWS, 1997c). In 1991, live colonies of this snail persisted in only two areas at the Preserve with a population estimate for each colony at or below 6,000 individuals.

3. Threats

The quality and quantity of free-flowing water environments required by the five Snake River cold water aquatic snail species have been decreased, continue to be impacted, and are vulnerable to continued habitat modification and deteriorating water quality. These past, present, and future threats are associated with one or more of the following: hydroelectric power generation, water withdrawals, and diversions, water pollution, inadequate regulatory mechanisms to address the sources of stress, and possible adverse effects from exotic species.

Water temperature, velocity, dissolved oxygen concentrations, and substrate type are all critical components of water quality that affect the survival of the five listed Snake River aquatic snails. These species require cold, clean, well-oxygenated, and rapidly flowing waters. They are intolerant of pollution and factors that cause oxygen depletion, siltation, or elevated water temperatures. Reduction of nutrient and sediment loading to the river and restoration of riverine conditions are needed to recover the listed species.

4. Recovery Plan Recommendations

A recovery plan for the five Snake River aquatic snails was prepared in 1995. Objectives of the Snake River Aquatic Species Recovery Plan are 1) preventing the extinction and/or further decline of extant colonies and habitat of the federally listed snails by eliminating or reducing known threats, and 2) collection of the basic information necessary to establish recovery criteria so that the listed species can be reclassified or delisted (FWS, 1995b). The five federally listed Snake River aquatic snails may be reclassified or recovered by implementing various conservation and recovery measures that preserve and restore the mainstem Snake River and spring habitats essential to their survival.

C. BRUNEAU HOT SPRINGSNAIL (*Pyrgulopsis bruneauensis*) - Endangered

1. Critical Habitat

There is no critical habitat designated for the Bruneau hot springsnail.

2. Selected Life History and Habitat Data

The Bruneau hot springsnail is a member of the family Hydrobiidae. Hydrobiids are gill-breathing, aquatic or semi-aquatic molluscs restricted to permanent or stable waters, particularly those that are spring-fed. They tend to occur as endemic species in single springs or spring systems.

The Bruneau hot springsnail was first collected in 1952. Hershler (1990) formally described the species from type specimens collected from the Indian Bathtub in Hot Creek, a tributary to the Bruneau River, naming it *Pyrgulopsis bruneauensis*. The species has been found in flowing thermal springs and seeps with temperatures ranging from 15.7ºC to 35.7ºC, with the highest densities of snails occurring in springs with higher temperatures (FWS, 1997c). The springsnails are found on exposed surfaces of various substrates, including rocks, gravel, sand, mud, and algal film.

The Bruneau hot springsnail occurs only in the remaining thermal spring flows entering Hot Creek and numerous, small, thermal springs, and seeps along an approximately 7.9 kilometer (4.85 mile) reach of the Bruneau River in southwestern Idaho (Mladenka, 1992; Mladenka, 1993; cited in FWS, 1997c). Surveys were conducted at a total of 201 thermal springs along the Bruneau River downstream and upstream of Hot Creek in 1993; 128 contained Bruneau hot springsnails (Mlandenka, 1993, cited in FWS, 1997c).

3. Threats

The major threat to the Bruneau hot springsnail is the reduction of water levels in thermal spring habitats due to groundwater withdrawals from the regional geothermal aquifer. Past cattle grazing also reduced some of the remaining springsnail habitats, especially those along Hot Creek. Cattle can affect snails by trampling instream substrate and snail habitats, causing direct springsnail mortality and displacement. Recreational access may also impact habitat of the Bruneau hot springsnail along the Bruneau River. Makeshift dams constructed by recreationists to form thermal pools for bathing can alter springsnail habitat and trap sediments (FWS, 1997c).

4. Recovery Plan Recommendations

No recovery plan has been prepared for the Bruneau hot springsnail.

D. KOOTENAI RIVER WHITE STURGEON (*Acipenser transmontanus*) - Endangered

1. Critical Habitat

There is no critical habitat designated for the Kootenai River population of the white sturgeon.

2. Selected Life History and Habitat Data

The Kootenai River population of the white sturgeon (*Acipenser transmontanus*) is restricted to approximately 270 km (168 river miles) in the Kootenai River. This reach extends from Kootenai Falls, Montana to Cora Linn Dam at the outflow from Kootenay Lake in British Columbia, Canada (FWS, 1994a). A natural barrier at Bonnington falls downstream of

Kootenai. The lake has isolated the white sturgeon in this system from other white sturgeon since the last glaciation (10,000 years ago). These fish have evolved life history characteristics that allowed them to thrive for centuries in large dynamic river systems containing diverse habitats with multiple food sources. These characteristics include opportunistic food habits, delayed maturation, longevity, high fecundity, and mobility (Beamesderfer & Farr, 1997). According to the best information available to FWS (1994a), this species has not had a successful spawning year class since 1974.

3. Threats

FWS (1994a) has identified lack of successful spawning as the single greatest threat to the species and has indicated that conditions for successful spawning include flows of adequate volume and duration, appropriate temperatures, and sufficient water velocities. Attainment of these conditions appear to be critical to the recovery of the species. Other factors identified as possibly impacting maintenance of a secure, self-sustaining population of Kootenai white sturgeon in the wild include elimination of rearing areas for juveniles, increased pollution in the form of metals, and a reduction in the overall productivity of the river system related to nutrients as a result of upstream dams (FWS, 1994a). Most of these factors, including one of the most critical factors - river volumes and timing of flows - do not fall within the scope of the Idaho Water Quality Standards.

The significant modification to the natural hydrograph in the Kootenai River caused by flow regulation at Libby Dam (in Montana) is considered the primary reason for the Kootenai River white sturgeon's declining numbers and continued lack of recruitment (Apperson and Anders, 1991). Since the Libby Dam began regulating flows in 1972, spring flows have been reduced by an average of 50% and winter flows have increased by 300% over normal flows in the Kootenai River (FWS, 1994a). As a result, the natural high spring flows required by white sturgeon for reproduction, rarely occur during the May to July spawning season when suitable temperature, water velocity, and photo period conditions exist. The alteration of the annual hydrograph by Libby Dam has modified the quality of water now entering Kootenay Lake. The modifications strip nutrients from the water in the river downstream from the dam and alter the time at which nutrients are supplied to the lake (FWS, 1994a).

Poor water quality and excessive nutrients in the Kootenai River were once considered major problems for the white sturgeon prior to the operation of Libby Dam. Graham (1981) concluded that poor water quality conditions in the 1950's and 1960's as a result of industrial and mine development, affected white sturgeon reproduction and recruitment. Significant improvements in Kootenai River water quality were noted in 1977, due in part to water control and treatment (FWS, 1994a). Today, many of these pollutants persist, primarily bound in sediments.

4. Recovery Plan Recommendations

A draft recovery plan has been prepared by the U.S. Fish and Wildlife Service but has never been finalized (FWS, 1998c). The recovery objectives of the draft plan are to prevent extinction, re-establish successful natural recruitment then delist the fish when the population is self sustaining and habitat is restored (FWS, no date).

E. BULL TROUT (*Salvelinus confluentus*) - Threatened

1. Critical Habitat

No critical habitat has been designated for the bull trout.

2. Selected Life History and Habitat Data

Cavendar (1978) identified bull trout *Salvelinus confluentus* as a distinct species of char, unique to western North America. Prior to the American Fisheries Society accepting the description of *Salvelinus confluentus* in 1980, biologists considered bull trout and Dolly Varden, *Salvelinus malma*, the same species (Pratt and Huston, 1993).

The U.S. Fish and Wildlife Service has determined two distinct population segments for bull trout in Idaho. They are the Jarbidge River distinct population segment and the Columbia River distinct population segment. The Jarbidge River, in southwest Idaho and northern Nevada, is a tributary in the Snake River basin and contains the southernmost habitat occupied by bull trout. This population segment is discrete because it is segregated from other bull trout in the Snake River basin by a large gap (greater than 240 km (150 mi)) in suitable habitat and several impassable dams on the mainstem Snake River. This distinct population segment is considered significant because its loss would result in a substantial modification of the species' range.

The Columbia River DPS occurs throughout the entire Columbia River basin within the United States and its tributaries, excluding bull trout found in the Jarbidge River. The Columbia River DPS is significant because the overall range of the species would be substantially reduced if this discrete population were lost.

Bull trout populations are known to exhibit two distinct life history forms (Idaho, 1996):

1) Resident - spend their entire life cycle in the same (or nearby) streams in which they were hatched.

2) Migratory -
   a) Fluvial - spawn in tributary streams where the young rear from one to four years before migrating to a river;

b) Adfluvial - spawn in tributary streams where the young rear from one to four years before migrating to lakes.

a. Spawning

Bull trout generally mature between 5 and 7 years of age (Fraley and Shepard, 1989; Goetz 1989; Leathe and Enk, 1985). Bull trout may spawn either yearly or in alternate years (Block, 1955; Fraley and Shepard, 1989; Pratt, 1985). Spawning occurs from August through November (Armstrong and Murrow, 1980; Brown, 1992; McPhail and Murray, 1979). Decreasing water temperatures influence the onset of spawning (Shepard et al., 1984; Weaver and White, 1985).

b. Eggs and Incubation

Embryos incubate over winter. McPhail and Murray (1979) found egg survival was highest at temperatures of 2 to 4 C. Hatching occurs in late winter or early spring (Weaver and White, 1985); the alevins may stay within the gravel for an extended period after they absorb the yolk, feeding and growing (McPhail and Murray, 1979). In laboratory tests, the quantity of fine sediment has been shown to reduce survival. Survival rates of 0% were measured with greater than 50% fines (<6.35 mm) to about 40% survival with zero fines (Shepard et al., 1984). Emergence has been observed over a relatively short period of time after a peak in stream discharge from early April through May (Rieman and McIntyre, 1993).

c. Growth and Juvenile Outmigration

Extensive migrations are characteristic of this species (Fraley and Shepard, 1989). Resident and migratory forms live together, but it is not known if they represent a single population or separate populations (Rieman and McIntyre, 1993). Growth differs little between forms during their first years of life in headwater streams, but diverges as migratory fish move into larger and more productive waters (Rieman and McIntyre, 1993).

Persistence of migratory life history forms and maintenance or re-establishment of stream migration corridors is crucial to the viability of bull trout populations (Rieman and McIntyre, 1993). Migratory bull trout ensure sufficient variability within populations by facilitating the interchange of genetic material between populations. Migratory forms also provide a mechanism for recolonizing local populations extirpated due to natural or anthropogenic effects.

d. Food

Juvenile bull trout have been found to feed on macroinvertebrates (Shepard et al., 1984; Boag, 1987). Adult bull trout are opportunistic fish eaters.

e. Rearing Habitat

Stream dwelling bull trout fry rear in low velocity water after hatching (McPhail and Murray, 1979). They hold in the substrate interstitial spaces, or within 0.03 m of the substrate, and are associated with cobble and boulders or submerged fine velocity where the water velocity is 0.09 m/s on average (Shepard et al., 1984). In streams, juvenile bull trout live close to in-channel wood, substrate, or undercut banks (Goetz, 1991; Pratt, 1984, 1992). Adult resident bull trout also closely associate with substrate but also select large cobble and boulder substrates, as well as pools and areas with complex woody debris and undercut banks (Graham et al., 1981; Pratt 1985; Shepard et al., 1984). Diel shifts in habitat use occur, bull trout often conceal themselves in cover (substrate and woody debris) during the day and move to near the substrate at night (Goetz, 1991).

Lake/River dwelling bull trout seek large deep pools with abundant cover in the autumn and winter (Jakober, 1995).

3. Threats

Bull trout growth, survival, and long-term population persistence are correlated with stream habitat conditions such as cover, channel stability, substrate composition, temperature, and migratory corridors (Rieman and McIntyre, 1993). These habitat features are impaired as the result of land management activities such as forest harvest, road building, hydropower development, irrigation diversions, mining, and grazing. Additional threats include hybridization and competition with introduced brook trout, predation, isolation, and over-utilization. Many of these factors are outside the scope of the Idaho Water Quality Standards. Below is a discussion of some of those factors that are, to some degree, related to water quality.

Salmonid habitat in the Columbia River Basin has been extensively affected by various land management activities. Timber harvest, road building, and livestock grazing near streams has removed riparian vegetation, changed stream channel morphology, and accelerated soil erosion. Sediment production due to land use practices has been accelerated in sensitive geomorphological formations. In Idaho, sediment loading has increased as a result of widespread logging, road building, and associated activities (Andrews, 1988; Fuller et al., 1985; Petrosky and Holubetz, 1986). Chapman et al. (1991) noted that livestock graze approximately 8 million acres of private and state lands within Idaho. More than 80% of the riparian areas managed by the US Bureau of Land Management (BLM) are in degraded conditions.

Damage to bull trout habitat due to mining has been documented in many drainages. Mining has altered stream channel morphology, increased sediment transport and deposition, decreased vegetative cover and is responsible for acidic water discharge and heavy metal water pollution (Chapman et al., 1991).

4. Recovery Plan Recommendations

A recovery plan has not yet been prepared for this species.

F. SNAKE RIVER SOCKEYE AND CHINOOK SALMON

1. Critical Habitat

The designated habitat for Snake River sockeye salmon consists of river reaches of the Columbia, Snake and Salmon Rivers, Alturas Lake Creek, Valley Creek and Stanley, Redfish, Yellow Belly, Pettit and Alturas Lakes (including their inlet and outlet creeks). The designated habitat for Snake River Spring/summer chinook salmon consists of river reaches of the Columbia, Snake and Salmon Rivers, and all tributaries of the Snake and Salmon rivers (except Clearwater River) presently or historically accessible to Snake River spring/summer chinook salmon (except reaches above impassable natural falls and Hells Canyon Dam). The designated habitat for Snake River fall chinook salmon consists of river reaches of the Columbia, Snake and Salmon Rivers, and all tributaries of the Snake and Salmon Rivers presently or historically accessible to Snake River fall chinook salmon (except reaches above impassable natural falls and Dworshak and Hells canyon Dams).

2. Selected Life History and Habitat Data

a. Historical Distribution and Abundance

Columbia River chinook populations were acknowledged at one time to be the largest in the world (Van Hyning, 1966). Prior to the 1960's, the Snake River was considered the most important drainage in the Columbia River system for the production of anadromous fishes.

The Snake River Basin encompasses an area of approximately 695 thousand square miles (1,118,255 square thousane km) in the states of Idaho, Oregon, and Washington. Historically, spring/summer chinook spawned in virtually all accessible and suitable habitat in the Snake River upstream from its confluence with the Columbia River (Evermann, 1896; Fulton, 1968). Evermann (1896) observed spring-run chinook spawning as far upstream as Rock Creek, a tributary of the Snake River just downstream from Auger Falls, Idaho.

Fall chinook were widely distributed in the main stem Snake River and the lower reaches of its major tributaries and ranged upstream to Shoshone Falls, Idaho (NMFS, 1995). The uppermost accessible reaches of the mainstem Snake River were the primary spawning areas of fall chinook.

Snake River sockeye were found in five lakes of the Stanley Basin and Big Payette Lake on the North Fork Payette River, in Idaho and in Wallowa Lake, Oregon (Evermann, 1896; Bjornn et al., 1968; Fulton, 1970).

b. Snake River Sockeye Salmon (*Oncorhynchus nerka*) - Endangered

Snake River sockeye salmon most commonly occur in two forms: an anadromous form referred to as sockeye and a nonanadromous (resident) freshwater form generally referred to as kokanee. Kokanee progeny occasionally migrate to sea and return as adults, however, there is only scattered evidence that these fish contribute to any sockeye population. A third form known as residual sockeye (residuals) often occur together with sockeye. Residuals are thought to be the progeny of sockeye but are nonanadromous (NMFS, 1995).

Snake River sockeye, now limited to a remnant population in Redfish Lake, represent the world's southernmost remaining natural sockeye population. These fish have a longer freshwater migration (approximately 900 miles) and reside at higher elevations (6,500 feet) than do sockeye anywhere else in the world. Sockeye, residuals, and kokanee each reside in Redfish Lake. Sockeye spawn along shoals of Redfish Lake in October and November. Residuals spawn in the same location and during the same period as sockeye, but are distinguishable by their smaller size (similar to kokanee) and reddish green coloration (in contrast to the red sockeye). In contrast, kokanee spawn in a tributary of Redfish Lake during August and September, indicating that they are reproductively isolated from sockeye and residuals (NMFS, 1991).

Snake River sockeye juveniles rear in a lake for one or sometimes two years. Sockeye smolts emigrate from freshwater rearing areas to the ocean in spring from April through June. Ocean residency is two to three years for sockeye. Sockeye arrive in the Columbia River in June and July. The only remaining Snake River sockeye return to Redfish Lake and spawn along the lake shoreline during October and November. A residual form of *O. nerka* that shares the spatial and temporal spawning distribution as sockeye and is genetically similar to the anadromous form also exists in Redfish Lake (Bevan et al., 1994).

Spawning and rearing: Snake river sockeye salmon spawning and rearing is currently limited to Redfish Lake. Other historical nursery areas that are essential to the conservation of the species include Alturas, Pettit, Stanley, and Yellow Belly Lakes (including their inlet creeks) (NMFS, 1993). Essential features of these areas include adequate spawning gravel, water quality, water quantity, water temperature, food, riparian vegetation, and access.

Juvenile migration: The sockeye juvenile migration corridors include the lakes above inlets and outlet creeks, Alturas Lake Creek, that portion of Valley Creek between Stanley Lake Creek and the Salmon River, the main fork of the Salmon River, the Snake River, and the Columbia River to the Pacific Ocean (NMFS, 1993). Essential features of the sockeye juvenile migration corridors include adequate substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe space passage conditions.

Adult migration: The adult migration corridors are the same areas in included in the juvenile migration corridors. Essential features for adult migration would include those required for the juvenile corridors, excluding adequate food.

c. Snake River Chinook Salmon (*Oncorhynchus tshawytscha*) - Threatened

Chinook salmon feature a diversity of juvenile and adult life history strategies that are used to categorize different stocks. These stocks are typically characterized according to the time of year that the adult enter freshwater to begin their spawning migrations. In the Columbia River Basin, adult salmon migrating past Bonneville Dam from February through May, June through July, and August through October are categorized as spring-, summer- and fall-run fish respectively (Burner, 1951). However, some adult chinook salmon passing Bonneville Dam in early June (summer chinook timing) return to Snake River Basin streams classified as "spring chinook streams;" conversely some chinook passing Bonneville Dam in late May (spring chinook timing) return to areas classified as "summer chinook streams." Thus, migration timing is not a definitive means for characterizing the different Snake River chinook stocks (Bevan et al., 1994).

Biological characterization of Snake River chinook (based on more extensive life history and genetic information) differentiates these fish into two primary aggregates or distinct population segments: spring/summer and fall chinook (NMFS, 1995).

Snake River spring/summer chinook salmon

Spring/summer chinook rear in headwater streams for one year. Spring/summer chinook smolts emigrate from freshwater rearing areas to the ocean in spring from April through June. Ocean residency varies but is generally one to four years. Spring chinook enter the Columbia River between February and May; summer chinook arrive in June and July (Bevan et al., 1994).

Spring/summer chinook spawn from August through September. Early arriving spring/summer chinook tend to spawn in the upstream reaches of tributary streams with the later arriving chinooks spawning progressively further downstream as the season advances. Snake River spring and summer chinook spawn and rear in high elevation tributaries before migrating to the ocean as yearling fish. In the upper Columbia River, summer chinook spawn in larger, lower elevation streams and subsequently emigrate as subyearlings (Bevan et al., 1994).

Spawning and rearing: Snake river spring/summer chinook salmon spawning and rearing is currently sparsely distributed throughout the Grande Ronde, Imnaha, Salmon, and Tucannon subbasins and Asotin, Granite, and Sheep creeks. However, the critical habitat designation includes all river reaches presently or historically accessible to this species except reaches above impassable natural falls and Dworshak and Hells Canyon Dams. Also, NMFS has proposed excluding the reach above Napias Creek Falls, as this barrier is considered an historical blockage to chinook access of upper Napias Creek (Federal Register, Vol. 64). Essential features of spawning and juveniles rearing areas include adequate spawning gravel, water quality, water quantity, water temperature, cover/shelter, food, riparian vegetation, and space.

Juvenile and adult migration corridors: The Snake River spring/summer chinook juvenile and adult migration corridors are the spawning and juvenile rearing areas plus the Snake River and the Columbia River to the Pacific Ocean. Essential features of the Snake River spring/summer chinook juvenile migration corridors include adequate substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe space passage conditions. Essential features for adult migration would include those is the juvenile corridors, excluding adequate food.

Snake River fall chinook salmon

Fall chinook begin their downstream migration a few months after emerging from the mainstem Snake River or the lower reaches of its major tributaries. Before the development of the Hells Canyon hydroelectric complex, the juvenile fall chinook out migration coincided with the latter stages (mid-May through June) of the sockeye and spring/summer chinook migration. Juvenile fall chinook now leave the Snake River from mid-June into August. Ocean residency varies but is generally one to four years for fall chinook.

Fall chinook enter the Columbia River between August and October. Fall chinook spawn through October and November in the mainstem Snake River, primarily between Lower Granite and Hells Canyon dams and in the lower reaches of major tributary streams. The lower elevation large tributaries and mainstem rivers are used for spawning. Juveniles migrate to the ocean soon after emergence (Bevan et al., 1994).

Spawning and rearing: Snake River fall chinook salmon spawning and rearing is currently limited to the Snake River below Hells Canyon Dam and within the Clearwater, Hells Canyon, Imnaha, Lower Grande Ronde, Lower North Fork Clearwater, Lower Salmon, Lower Snake, Lower Snake-Asotin, Lower Snake Tucannon, and Palouse hydrologic units (NMFS, 1993). This critical habitat designation includes all river reaches presently or historically accessible to this species (except reaches above impassable natural falls and Dworshak and Hells Canyon Dams). Essential features of spawning and rearing areas include adequate spawning gravel, water quality, water quantity, water temperature, cover/shelter, food, riparian vegetation, and space.

Juvenile and adult migration: Juvenile and adult migration corridors are the same areas as spawning and juvenile areas, plus the Columbia River to its mouth at the Pacific Ocean. Essential features of the Snake River fall chinook salmon juvenile migration corridors include adequate substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions. Essential features for adult migration include those essential for the juvenile corridors, excluding adequate food.

3. Threats

Factors contributing to the decline of the Snake River Salmon include: timber management, grazing, mining, water development, juvenile Snake River salmon passage, adult Snake River salmon passage, water withdrawal and storage, and Snake River salmon commercial, recreational and native ceremonial, and subsistence harvest (Bevan et al., 1994). Many of these factors are outside the scope of the Idaho Water Quality Standards. Below is a discussion of those factors that are, to some degree, related to water quality.

a. Timber Management and Grazing

Anadromous salmonid habitat in the Columbia River Basin has been extensively affected by various land management activities. More than 80% of the salmon in the Snake River Basin are produced on U.S. Forest Serve (USFS) and Bureau of Land Management (BLM) managed lands. Timber harvest, road building, and livestock grazing near streams has removed riparian vegetation, changed stream channel morphology, and accelerated soil erosion. Sediment production due to land use practices has been accelerated in sensitive geomorphological formations. In Idaho, sediment loading has increased as a result of widespread logging, road building, and associated activities (Andrews, 1988; Fuller et al., 1985; Petrosky and Holubetz, 1986). Chapman et al. (1991) noted that livestock graze approximately 8 million acres of private and state lands within Idaho and that more than 80% of the riparian areas managed by the BLM are in degraded conditions.

b. Mining

Damage to spring/summer chinook habitat due to mining has been documented in many drainages. Mining has altered stream channel morphology, increased sediment transport and deposition, decreased vegetative cover, and is responsible for acidic water discharge and heavy metal water pollution (Chapman et al., 1991).

c. Water Impounds for Development, Withdrawal, and Storage

Dams and reservoirs have resulted in substantial reductions in abundance of Columbia River Basin salmon and represent a significant factor affecting recovery. The Northwest Power Planning Council (NPPC) estimated that current annual salmon and steelhead production in the Columbia River Basin is 10 million fish below historical levels, with 8 million of this annual loss attributable to hydropower development and operation (NPPC, 1987).

It is widely acknowledged that juvenile and adult fish survival has been adversely affected by dams and reservoirs. Total dissolved gas is one issue associated with the operations of dams that relates directly to the Idaho Water Quality Standards. Dissolved gas supersaturation caused by large volumes of water spilling over dams can also result in injury or mortality for adult salmon. Since the 1960's, increased hydraulic capacity at powerhouses of mainstem projects, increased storage of water, and structural modifications to spillways have substantially reduced this problem (Bevan et al., 1984).

Diversion and storage of water within the Columbia River Basin has altered historical runoff patterns in the Snake and Columbia rivers. In addition, unscreened water withdrawals have often caused juvenile anadromous fishes to be diverted onto irrigated lands (Bevan, et al., 1994).

4. Recovery Plan Recommendations

A Draft Recovery Plan was proposed by NMFS for the Snake River salmon stocks in 1995 but was never adopted.

G. SNAKE RIVER STEELHEAD (*Oncorhynchus mykiss*) - Threatened

1. Critical Habitat

No critical habitat has yet been designated.

2. Selected Life History Data

Steelhead exhibit one of the most complex life histories of any salmonid species. Steelhead may exhibit anadromy or freshwater residency. Resident forms are usually referred to as “rainbow” or “redband” trout, while anadromous life forms are termed “steelhead”.

Steelhead typically migrate to marine waters after spending two years in freshwater. They then reside in marine waters for two or three years prior to returning to their natal stream to spawn as four- or five-year-olds. Depending on water temperature, steelhead eggs may incubate in redds for 1.5 to 4 months before hatching as alevins (larval stage dependent on yolk sac as food). Following yolk sac absorption, alevins emerge from the gravel as young juveniles (fry) and begin actively feeding. Juveniles rear in freshwater from one to four years, then migrate to the ocean as smolts.

Biologically, steelhead can be divided into two reproductive ecotypes, based on their state of sexual maturity at the time of river entry and the duration of their spawning migration. Stream maturing steelhead enter freshwater in a sexually immature condition and require several months to mature and spawn. Ocean maturing steelhead enter freshwater with well-developed gonads and spawn shortly after river entry. These two reproductive ecotypes are commonly referred to by their season of freshwater entry (e.g., summer and winter steelhead).

Two major genetic groups or “subspecies” of steelhead occur on the west coast of the United States: a coastal group and an inland group, separated on the Fraser and Columbia River Basins by the Cascade crest. Only inland steelhead occur in Idaho.

Historically, steelhead likely inhabited most coastal streams in Washington, Oregon, and California, as well as many inland streams in these states and Idaho. However, during this century, over 23 indigenous, naturally-reproducing stocks of steelhead are believed to have been extirpated and many more are thought to be in decline in numerous coastal and inland streams in Washington, Oregon, Idaho, and California (NMFS, 1996).

3. Threats

The NMFS has identified the destruction and modification of habitat, overutilization for recreational purposes, and natural and human-made factors as being the primary reasons for the decline of the west coast steelhead. Among the natural and human-made factors which have been identified as contributing to the decline of the species through the elimination, degradation, simplification, and fragmentation of habitat are forestry, agriculture, mining, urbanization, and water diversions (NMFS, 1996). A more detailed discussion of these threats is presented in the previous subsection outlining threats to the Snake River Sockeye and Chinook salmon.

4. Recovery Plan Recommendations

A recovery plan has not been developed for the Snake River steelhead.

H. BALD EAGLE (*Haliaeetus leucocephalus*) - Threatened

1. Critical Habitat

The FWS has not designated critical habitat in Idaho for the bald eagle.

2. Selected Life History and Habitat Data

The bald eagle is an endemic North American species. Little information exists on its longevity in the wild. However, longevity of captive eagles ranges from 15 to 47 years, with an estimated reproductive life of 20 to 30 years (Stalmaster, 1987). On average, bald eagles reach sexual maturity at 5 years of age following the fourth molt to adult plumage (Stalmaster, 1987). Sexual dimorphism is expressed in bald eagles by size differences between sexes, with females being larger. Pair bonds between breeding adults are believed to last over the life of the bird. Once sexually mature, eagles may exhibit considerable reproductive variation, likely in response to quality and quantity of food resources, although other factors such as human disturbance may preclude or interrupt nesting.

The bald eagle historically ranged throughout North America excluding extreme northern Alaska and Canada and central and southern Mexico. Current range in the lower 48 states includes five recovery populations: Chesapeake Bay, Pacific, Southeastern, Northern States, and Southwestern. In 1963, a National Audubon Society survey reported only 417 active nests in the lower 48 states. In 1994, about 4,450 occupied breeding areas were reported. There has been a 47 percent increase since 1990 in the number of occupied breeding territories, (FWS, 1995a, cited in FWS, 1997c).

The Pacific Recovery Region includes Idaho. Surveys during 1994 and 1995 in Idaho indicate that of 77 occupied nesting territories, 61 nests were active, and 75% were successful. Sixty-eight percent of the nest failures occurred along the Snake River in eastern Idaho (IDFG, 1995, cited in FWS, 1997c).

A significant population of bald eagles overwinter in Idaho and some are presumed to remain in the state year-round. In Idaho, bald eagle winter habitat includes the Coeur d'Alene Lake and River, Pend Oreille Lake and River, Snake River, Priest River, Clearwater River, and the American Falls Reservoir.

Eagles begin to appear at wintering sites in early November and concentrate at locations where there is open water during the colder months when smaller or slower moving waterbodies freeze (Spahr, 1990). Diet includes hatchery trout, other fish species including both salmonids and non-salmonids, mule deer, ground squirrels, rabbits, waterfowl, and other small mammals (Spahr, 1990). Consumption of fish relative to other species declines in the colder months as waterbodies freeze. This diet shift coincides with an increase in the availability of terrestrial species in the form of carrion. Water quality could potentially affect bald eagles through four avenues: prey displacement or quantitative decline, prey mortality, bioaccumulation in prey, or direct consumption.

3. Threats

In the Pacific Bald Eagle Recovery Plan (FWS, 1986a) habitat loss is identified as the most significant long-term threat to all bald eagle populations in the recovery area. Shooting continues to be the most frequently recorded cause of bald eagle mortality, though the rate appears to be declining. Bald eagle reproduction throughout the species range has improved since the registration of DDT and other organochloride pesticides was canceled in the early 1970's (Postupalsky, 1978). However, DDE (a derivative of DDT) and polychlorinated biphenyls (PCB's) are still present in bald eagles on the lower Columbia River and are associated with eggshell thinning and low breeding success. Secondary lead poisoning is a significant problems where eagles feed on crippled and poisoned waterfowl (FWS, 1986a). Many other environmental contaminants are potential threats to eagles. Dioxin, endrin, heptachlor epoxide, mercury, and PCB's are still detected in eagle food supplies (FWS, 1986a).

4. Recovery Plan Recommendations

The main steps outlined in the Recovery Plan (FWS, 1986a) are to: 1) provide secure habitat; 2) inventory, monitor, and research bald eagle habitat and populations to obtain adequate knowledge for developing and evaluating management programs; 3) develop and maintain law enforcement and public awareness programs; and 4) augment bald eagle populations through management and protection. One of the general recommendations for augmenting Pacific bald eagle populations is to reduce mortality through exposure to contaminants.

I. AMERICAN PEREGRINE FALCON (*Falco peregrinus anatum*) - Endangered

1. Critical Habitat

The FWS has not designated critical habitat for the American peregrine falcon.

2. Selected Life History and Habitat Data

Preferred peregrine nesting habitat is cliffs, or their equivalent, located near water. Peregrines primarily prey upon other birds, taking a broad range of species. The most common prey species are doves, pigeons, and other species which attract attention by their markings in flight or by conspicuous aerial courtship activities. When they are available, feral and domestic pigeons are the most preferred species and account for 20 to 60 percent of the number of individuals in the diet. These preferred prey species are all land grazing birds. Peregrines occasionally consume fish, frogs, and insects (FWS, 1982).

Peregrine falcons were extinct in Idaho by 1974 but were reintroduced to the state beginning in 1982. In 1993, 14 pairs of peregrine were observed to be nesting within Idaho (IDFG, 1993). Distribution is as follows: western Idaho, 3 pairs; central Idaho, 1 pair, eastern Idaho/Greater Yellowstone area, 10 pairs.

3. Threats

Peregrine falcon numbers declined rapidly beginning in the 1950's as a result of pervasive reproductive failure resulting from eggshell thinning and behavioral modifications inimical to successful reproduction. These were caused by exposure to organochloride pesticides such as DDT and dieldrin. Other possible causes for decline in peregrine falcons listed in the recovery plan are changes in climate, competition from prairie falcons, availability of nest sites, loss of foraging area (wooded areas, marshes, open grasslands, coastal strands, and bodies of water), transmission lines (collisions and electrocution), shooting, capture, disturbance, predators, and disease (FWS, 1982).

4. Recovery Plan Recommendations

A recovery plan for Pacific Coast peregrines exists (FWS, 1982) but it does not cover Idaho.

J. GRAY WOLF (*Canis lupus*) - Endangered

The Fish and Wildlife Service reintroduced wolves into Idaho in late 1994. The primary threats to wolf populations are human caused mortality. Potential threats to wolves from water quality impacts would be through direct drinking water exposure.

In general, wolves depend upon ungulates for food in the winter and supplement this diet during spring-fall with beaver and smaller mammals. In central Idaho, elk, mule deer, white-tailed deer, and moose where available, are the primary prey species. Columbian ground squirrels, snowshoe hare, and grouse are also available to wolves in central Idaho as an alternate prey source. These prey species are primarily vegetarian and as a result would be less prone to bioaccumulate toxics compared to carnivorous or piscivorus species.

K. GRIZZLY BEAR (*Ursus arctos horribilus*) - Threatened

Current grizzly bear habitat in Idaho is limited to the Selkirk Mountains in the northern panhandle; although there are occasional sightings in the Bitterroot National Forest near the Montana border and in the Greater Yellowstone area. The primary threat to grizzly bear survival is the penetration and fragmentation of habitat by roads and related mortality associated with human activity.

Primary exposure to toxics or other contaminants would be through direct drinking water exposure. The limited data available on grizzly bear diet in the Selkirk Mountains indicates that grizzly are primarily vegetarian (Almack, 1985). As a result, this population is not subject to the adverse effects from consumption of toxics through bioconcentration in prey species that may pose a threat to higher trophic level predators.

L. CANADA LYNX (*Lynx canadensis*) - Threatened (proposed)

The Canada lynx (*Lynx canadensis*), the only lynx in North America, is a secretive forest-dwelling cat of northern latitudes and high mountains. There it feeds primarily on small mammals and birds and is especially dependent on snowshoe hare for prey. It was historically found throughout much of Canada, the forests of northern tier States and subalpine forests of the central, and southern Rockies. The lynx is a medium-sized cat, similar to the bobcat, but appears somewhat larger. It has longer hind legs and very large well-furred paws, adaptations to the deep winter snows typical throughout its range. It also has unique long tufts on the ears and a short, black-tipped tail.

The following factors have been identified as threatening the lynx: (1) loss and/or modification of habitat; (2) past commercial harvest (trapping), which is partially responsible for the extremely small lynx population; (3) inadequate regulatory mechanisms to protect lynx and their habitat; and (4) other factors such as increased human access into suitable habitat and human-induced changes in habitat allowing other species (e.g., bobcats and coyotes) to move into lynx habitat and compete with them. Examples of human alteration of forests include loss of and conversion of forested habitats through urbanization, ski area, and other developments; packed snow trails (such as created by snowmobiles) that allow lynx competitors (bobcats, coyotes) into lynx habitat, fragmentation that leads to isolation of forested habitats by highways or other major construction; and certain timber harvesting practices and fire suppression measures (FWS, 1998a;1998b; 1998d).

M. NORTHERN IDAHO GROUND SQUIRREL (*Spermophilus brunneus brunneus*) - Threatened (proposed)

The Northern Idaho ground squirrel is a small terrestrial, burrowing mammal. This species is usually 15.24 - 20.32 cm in length, has a short, narrow tail, large conspicuous ears, and tan feet and ears. The subspecies appears dark, with reddish-brown spots and a dark undercoat. The ground squirrels emerge in late March or early April and cease above ground activity in late July or early August.. Adult (2 year old) males emerge first, followed by adult females, then yearlings. Ground squirrels are diurnally active. Their diet consists of forbs, grasses, seeds, and various green vegetation. (FWS, 1997c)

The habitat of the Northern Idaho ground squirrel is drier meadows surrounded by Ponderosa pine and Douglas-fir forests between 3,773 and 5,085 feet elevation. The Northern Idaho ground squirrel is endemic to west-central Idaho in Adams and Valley Counties. Occurrences are found on a tableland between Cuddy and Seven Devils Mountains, in the valleys to the east (Lost Valley Reservoir and Price Valley) and in Long Valley further east and south. The main concentration of the subspecies occurs in a large meadow complex near Bear (Adams County). (FWS, 1997c)

N. WHOOPING CRANE (*Grus americana*) - Endangered, experimental non-essential population

A population of four whooping cranes resides in the Grays Lake area during the period between April and September. During this period, birds also spend time in neighboring areas of Montana, Nevada, and Wyoming. In the remainder of the year the birds migrate to the southwestern United States. This population has declined to four birds and has been or is expected to shortly be delisted and reclassified an experimental population. Mortality has been the result of habitat loss, disease, and collisions with power lines. The Idaho population was foster reared by sandhill cranes and as a result of improper imprinting does not breed and is expected to become extinct with the demise of the remaining four birds.

Whooping cranes nest in marshy areas among bulrushes, cattails, and sedges. They are omnivorous feeders, but animal foods, especially blue crabs and clams, predominate in the winter diet (FWS, 1986b, cited in FWS, 1997c). Most foraging occurs in the brackish bays, marshes, and salt flats lying between the mainland and barrier islands. Occasionally, whooping cranes will fly to upland sites when attracted by foods such as acorns, snails, crayfish, and insects and then return to the marshes to roost. Uplands are particularly attractive when partially flooded by rainfall, burned to reduce plant cover, or when food is less available in the salt flats and marshes (FWS, 1986b, cited in FWS, 1997c).

O. WOODLAND CARIBOU (*Rangifer tarandus caribou*) - Endangered

Since the 1960's woodland caribou habitat in the United States has been limited to the Selkirk Mountains in northeastern Washington and northern Idaho. The primary threats to caribou populations are habitat modification or fragmentation, predation, disease, and poaching. Caribou feed on arboreal lichens for half the year and on huckleberry leaves, Sitka valerian, boxwood, and smooth woodrush for the other half (FWS, 1994c). Identified recovery actions focus on habitat protection, reduction in accidental and intentional shootings, reduction in vehicle collisions, and research issues.

P. WATER HOWELLIA (*Howellia aquatilis*) - Threatened

Howellia aquatilis (water howellia) was described by Gray in 1879. It is an aquatic plant that grows 10-60 cm tall. Water howellia most frequently occurs in glacial pothole ponds and former river oxbows whose bottom surfaces are firm, consolidated clay and sediments. Water howellia has very narrow ecological requirements, and therefore even subtle changes in its habitat could be devastating to a population. The species does not appear to be capable of colonizing disturbed habitats (Shelly and Moseley, 1988).

The species is threatened by impacts from loss of wetland habitat and habitat changes due to timber harvesting, encroachment by an exotic grass, development, and grazing. Alterations of water quality and the composition of the wetland bottom and vegetation, may affect the viability of *Howellia aquatilis*. Idaho bottom land habitats have been altered by roads, development, conversion to agriculture, and pasture lands. Water howellia may be less able to adapt to environmental changes because of its lack of genetic variability (Lesica et al., 1988).

Q. MacFARLANE'S FOUR O'CLOCK (*Mirabilis macfarlanei)* - Threatened

The MacFarlane's four o'clock was originally listed as endangered in 1979. At the time of listing, only three populations were known from the Snake River and Salmon River canyons in Idaho and Oregon. Since 1979, six additional populations of this plant have been discovered in Idaho and Oregon and some populations have been actively monitored by the U.S. Forest Service and the Bureau of Land Management. As a result, the species was downlisted to threatened on March 15, 1996.

The MacFarlane's four o'clock is a long-lived herbaceous perennial with a deep-seated root and bright pink flowers. The species occurs in grassland habitats that are characterized by regionally warm and dry conditions. Sites are dry and generally open, although scattered scrubs may be present. Established plants generally start growth in early April with the timing and duration of flowering apparently linked to precipitation levels. Once established, individual plants may survive for decades.

Threats to the species include livestock grazing, herbicide use, road/trail construction and maintenance, exotic plant species, off-road vehicles, mining, fire suppression and rehabilitation efforts, trampling landslides, flood damage, exotic species and herbicide, and pesticide spraying (FWS, 1997b).

R. UTE LADIES' TRESSES (*Spiranthes diluvialis*) - Threatened

Ute ladies' tresses is a perennial, terrestrial orchid with three to fifteen small white or ivory flowers clustered into a spike arrangement at the top of the stem. This orchid is found in four general areas of the interior western United States including along the upper Snake River drainage in southeast Idaho. It was listed as threatened in January 1992 due to a variety of factors, including habitat loss and modification and hydrological modifications of existing and potential habitat areas.

Ute ladies' tresses is endemic to moist soils in mesic or wet meadows near springs, lakes, and perennial streams. The elevation range of known occurrences is 4,001 to 7,000 feet. Generally, this species occurs in areas where the vegetation is relatively open (e.g., grass and forb-dominated sites), but some populations are found in riparian woodlands.

The riparian and wetland habitats that support this species have been heavily impacted by urban development, stream channelization, water diversions and other watershed, and stream alterations that reduce the dynamics of stream systems. In addition, conversion of riparian/floodplain land to agricultural uses has destroyed habitat in many areas. Grazing could also potentially impact this species during critical periods such as flowering and fruit set. (FWS, 1997a).

S. SPALDING'S CATCHFLY (*Silene spaldingii*) - Proposed Threatened

Spalding's catchfly was first collected in the vicinity of the Clearwater River, Idaho, between 1836 and 1847, and was described by Watson (Watson 1875). A member of the pink or carnation family, spalding's catchfly is a long-lived perennial herb with four to seven pairs of lance-shaped leaves and a spirally arranged inflorescence (group of flowers) consisting of small greenish-white flowers. The foliage is lightly to densely covered with sticky hairs. Reproduction is by seed only.

Spalding's catchfly is typically associated with grasslands dominated by native perennial grasses. This species is primarily restricted to mesic (not extremely wet nor extremely dry) grasslands (prairie or steppe vegetation) that make up the Palouse region in southeastern Washington, northwestern Montana, and adjacent portions of Idaho and Oregon. This catchfly is currently known from a total of 52 populations. Seven populations occur in west-central Idaho much of the remaining habitat is fragmented.

Large-scale ecological changes in the Palouse region over the past several decades, including agricultural conversion, changes in fire frequency, and alterations of hydrology, have resulted in the decline of numerous sensitive plant species including *Silene spaldingii* (Tisdale 1961). More than 98 percent of the original Palouse prairie habitat has been lost or modified by agricultural conversion, grazing, invasion of non-native species, altered fire regimes, and urbanization (Noss et al., 1995). The U.S. Fish and Wildlife Service has not officially designated critical habitat.

III. ANALYSIS OF EFFECTS OF NUMERIC CRITERIA FOR TOXIC POLLUTANTS TO SPECIES OF CONCERN

A. Introduction

EPA's Water Quality Standards regulations require states to adopt water quality criteria that will protect the designated uses of a water body. These criteria must be based on sound scientific rationale and must contain sufficient parameters or constituents to protect the designated uses. Since 1980, EPA has been publishing criteria development guidelines and national criteria for numerous pollutants. EPA's criteria documents provide a toxicological evaluation of the chemical, tabulate the relevant acute and chronic toxicity information and derive the acute and chronic criteria that EPA recommends for the protection of aquatic life resources. States may choose to adopt EPA's recommended criteria or modify these criteria to account for site-specific or other scientifically defensible factors.

Water quality criteria for aquatic life contain two expressions of allowable magnitude: a criterion maximum concentration (CMC) to protect against acute (short-term) effects; and a criterion continuous concentration (CCC) to protect against chronic (long-term) effects. EPA derives acute criteria from 48- and 96-hour tests of lethality or immobilization. EPA derives chronic criteria from longer term (often greater than 28-day) tests that measure survival, growth, or reproduction.

The quality of an ambient water body typically varies in response to variations of effluent quality, stream flow, and other factors. Organisms in the water body are not typically receiving constant, steady exposure but rather are experiencing fluctuating exposures, including periods of high concentrations, which may have adverse effects. Thus, EPA's criteria indicate a time period over which exposure is to be averaged, as well as an upper limit on the average concentration, thereby limiting the duration of exposure to elevated concentrations. For acute criteria, EPA recommends an averaging period of 1 hour. That is, to protect against acute effects, the 1-hour average exposure should not exceed the CMC. For chronic criteria, EPA recommends an averaging period of 4 days. That is, the 4-day average exposure should not exceed the CCC.

To predict or ascertain the attainment of criteria, it is necessary to specify the allowable frequency for exceeding the criteria. This is because it is statistically impossible to project that criteria will never be exceeded. As ecological communities are naturally subjected to a series of stresses, the allowable frequency of pollutant stress may be set at a value that does not significantly increase the frequency or severity of all stresses combined.

EPA recommends an average frequency for excursions of both acute and chronic criteria not to exceed once in 3 years. In all cases, the recommended frequency applies to actual ambient concentrations, and excludes the influence of measurement imprecision. EPA selected a 3-year average frequency of criteria exceedence with the intent of providing for ecological recovery from a variety of severe stresses. This return interval is roughly equivalent to a 7Q10 design flow condition. Because of the nature of the ecological recovery studies available, the severity of criteria excursions could not be rigorously related to the resulting ecological impacts. Nevertheless, EPA derives its criteria intending that a single marginal criteria excursion (i.e., a slight excursion over a 1-hour period for acute or over a 4-day period for chronic) would require little or no time for recovery. If the frequency of marginal criteria excursions is not high, it can be shown that the frequency of severe stresses, requiring measurable recovery periods, would be extremely small. EPA thus expects the 3-year return interval to provide a very high degree of protection (EPA, 1994).

Section 303(c)(2)(E) of the Clean Water Act requires that all states adopt chemical-specific, numeric criteria for priority toxic pollutants. In 1992, the State of Idaho had not yet adopted such criteria. Therefore, on December 22, 1992 EPA promulgated such criteria for all waters in the State of Idaho as part of the National Toxics Rule (EPA, 1992). Idaho has since revised the Idaho Water Quality Standards to include the same criteria as EPA promulgated under the National Toxics Rule. Following completion of this consultation, EPA is proposing to recommend a federal action which would remove the State of Idaho from the National Toxics Rule, thus providing for the State's criteria to become effective.

The National Toxics Rule originally promulgated criteria for metals as total recoverable metals. Following EPA's promulgation of this rule, EPA issued a new policy for setting water quality criteria for metals. Therefore, on May 4, 1995 EPA issued a stay on the effectiveness of the metals criteria promulgated in the National Toxics Rule and promulgated revised criteria expressed in terms of dissolved metals (EPA, 1995). At this time, EPA also promulgated conversion factors for converting between dissolved and total recoverable criteria. States, when adopting criteria, may choose to adopt metals criteria measured as either dissolved or total recoverable. The metals criteria in the Idaho Water Quality Standards are expressed as dissolved metals.

In Idaho, both the aquatic life criteria and human health criteria apply to all surface waters of the State. Idaho's water quality standards contain a provision which states that when multiple criteria apply to a water body, the most stringent criterion is the applicable criterion. With regard to the numeric toxic criteria, all but several have more stringent aquatic life criteria than human health criteria. Therefore, with regard to the majority of the toxic criteria, the aquatic life criteria are the applicable criteria for surface waters. An example of an exception to this is arsenic, where the human health criterion is several orders of magnitude lower than the aquatic life

criteria. Therefore, in all surface waters in Idaho, the applicable criteria for arsenic is the human health criteria.

All criteria in the Idaho Water Quality Standards, with the exception of the human health criterion for arsenic, are identical to the criteria promulgated by EPA under the National Toxics Rule. These criteria were adopted by reference in IDAPA 16.01.02.250.07. The aquatic life criteria evaluated as part of this assessment are summarized in Table 250.07.a.1. For comparison purposes, this table provides metals criteria expressed as both dissolved and total recoverable.

Idaho's criteria for pentachlorophenol (PCP) is expressed as an equation dependent on pH and seven of the criteria for metals are expressed as a function of water hardness. The PCP criteria in Table 250.07.a.1 were calculated at a pH of 7.8. In the following table, rather than present the equation for the hardness-dependent metals, EPA used a hardness of 100 mg/L $CaCO_3$ in order to present a value for the metals criteria. Therefore, although the criteria value would be dependent of the particular hardness value for a waterbody, in the following table, criteria were calculated at a hardness of 100 mg/L $CaCO_3$.

In the NTR, EPA described and required minimum and maximum hardness values (25 mg/L and 400 mg/L as $CaCO_3$, respectively) to be used when calculating hardness dependent freshwater metals criteria. Most of the data EPA used to develop the hardness formulas were in the hardness range of 25 to 400 mg/L. Therefore, EPA stated that the formulas were most accurate in that range. Using a hardness of 25 mg/L for calculating criteria, when the actual ambient hardness is less than 25 mg/L, could result in criteria that are underprotective of aquatic life. Because the State of Idaho is still under the NTR, the lower and upper hardness cap values are applicable. Therefore until EPA withdraws the NTR from applying to Idaho the State is unable to use ambient hardness values outside this range for calculating hardness dependent metals criteria. When the NTR is withdrawn from applying to Idaho, the State will then have the option of using ambient hardness values outside the 25-400 mg/L range when calculating criteria for hardness dependent metals.

For reference, average, minimum, and maximum hardness measurements recorded in waters throughout the State of Idaho are presented in Appendix F. Hardness values observed throughout the State range from 14.07 mg/L in the Upper Selway River to 404 mg/L in the Lower Bear River, with an average of 103.8 mg/L. Literature describing the experiments referenced in this section did not always provide hardness values along with data. In cases where hardness values are lacking, comparisons of criteria to research results may not be reliable. For those metals which are hardness dependent, EPA Region 10 calculates NPDES permits limits and load allocations for TMDLs using the fifth percentile of the ambient and or effluent hardness values which are most often calculated from instantaneous data.

Table 250.07.a.1. Idaho Water Quality Standards General Aquatic Life Criteria (from 60FR22228)						
Chemical Name	Criteria (µg/L)		Total Recoverable Criteria (µg/L)		Conversion Factor[a]	
	Acute	Chronic	Acute	Chronic	Acute	Chronic
Arsenic	360	190	360	190	1.000	1.000
Cadmium	3.7[b]	1.0[b]	3.9[c]	1.1[c]	0.944[d]	0.909[c]
Copper	17[b]	11[b]	18[c]	12[c]	0.960	0.960
Cyanide	22[e]	5.2[e]	N/A		N/A	
Endosulfan (&)	0.22	0.056	N/A		N/A	
Lead	65[b]	2.5[b]	82[c]	3.2[c]	0.791[d]	0.791[c]
Mercury	2.1	0.012	2.4	0.012	0.85	N/A
Selenium	20	5.0	N/A		N/A	
Zinc	110[b]	100[b]	120[c]	110[c]	0.978	0.986
Aldrin	3	N/A	N/A		N/A	
Chlordane	2.4	0.0043	N/A		N/A	
Chromium (III)	550[c]	180[c]	1,700[c]	210[c]	0.316	0.860
Chromium (VI)	15	10	16	11	0.982	0.962
4,4'-DDT	1.1	0.001	N/A			
Dieldrin	2.5	0.0019	N/A			
Endrin	0.18	0.0023	N/A			
Heptachlor	0.52	0.0038	N/A		N/A	
Lindane (gamma-BHC)	2	0.08	N/A		N/A	
Nickel	1,400[b]	160[b]	1,400[c]	160[c]	0.998	0.997
PCBs	N/A	0.014	N/A		N/A	
Pentachlorophenol	20[g]	13[g]	N/A		N/A	
Silver	3.4[b]	N/A	4.1	N/A	0 .85	N/A
Toxaphene	0.73	0.0002	N/A		N/A	

N/A - no applicable criteria
a - Conversion factors for translating between dissolved and total recoverable criteria.
b - Criteria for these metals are expressed as a function of total hardness (mg/L as $CaCo_3$), and the following formula:

Acute Criteria = WER $\exp\{m_A[\ln(\text{hardness})]+b_A\}$ x Acute Conversion Factor
Chronic Criteria = WER $\exp\{m_C[\ln(\text{hardness})]+b_C\}$ x Chronic Conversion Factor

where:

Metal	m_A [f]	b_A [f]	m_C [f]	b_C [f]
Cadmium	1.128	-3.828	0.7852	-3.490
Chromium (III)	0.8190	3.688	0.8190	1.561
Copper	0.9422	-1.464	0.8545	-1.465
Lead	1.273	-1.460	1.273	-4.705
Nickel	0.8460	3.3612	0.8460	1.1645
Silver	1.72	-6.52	N/A	N/A
Zinc	0.8473	0.8604	0.8473	0.7614

The term "exp" represents the base e exponential function.

c - For comparison purposes, the values displayed in this table correspond to a total hardness of 100 mg/l $CaCO_3$ and a WER of 1.0.

d - The conversion factors for cadmium and lead are hardness dependent. The values shown in the table correspond to a hardness of 100 mg/L $CaCO_3$. Conversion factors for any hardness may be calculated using the following equations:

Cadmium:

Acute- CF=1.136672-[(ln(hardness))x(0.041838)]

Chronic- CF=1.101672-[(ln(hardness))x(0.041838)]

Lead:

Acute and Chronic- CF=1.46203-[(ln(hardness))x(0.145712)]

e - Criteria expressed as Weak Acid Dissociable

f - m_A and m_c are the slopes of the relationship for hardness, while b_A and b_C are the Y-intercepts for these relationships

g - Criteria for pentachlorophenol is expressed as a function of pH and calculated as follows:

Acute Criteria = exp(1.005 (pH) - 4.830)

Chronic Criteria = exp(1.005 (pH) - 5.290)

B. Analysis of Effects of Toxic Pollutants to Mammals and Plants

Mammals

Woodland caribou, Northern Idaho ground squirrels and grizzly bears in Idaho are primarily vegetarians (Almack, 1985; FWS, 1994c). Gray wolves and lynx consume prey that are primarily vegetarian. These mammals should not be exposed to harmful concentrations of the toxic pollutants as a result of exposure to contaminated aquatic organisms since they do not consume fish. Their primary route of exposure is through ingestion of water.

Water quality criteria for human health were considered to be protective of all threatened and endangered mammals. The human health criteria protect against long term health effects. These effects range from cancer to reproductive and neurological impairments. The toxicity endpoints are related to human health, however these endpoints are usually derived from laboratory studies of rats and mice. This interspecies extrapolation for all mammals is accounted for in the modifying factors used to derive the toxicity endpoints.

The exposure equation used to derive the criteria for non carcinogenic effects is:

$$C = \frac{\text{RfD x WT) - (DT + In) x WT}}{\text{WI = (FC X L x FM x BCF)}}$$

C = updated water quality criterion (mg/L)
RfD = oral reference dose (mg toxicant /kg human body weight/day)
WT = weight of an average human adult (70kg)
DT = dietary exposure (other than fish) mg toxicant/kg body weight/day)
IN = inhalation exposure (mg toxicant/kg body weight/day)
WI = average human adult water intake (2 l/day)
FC = daily fish consumption (kg fish/day)
L = ratio of lipid fraction of fish tissue consumed to 3%
FM = food chain multiplier (from Table 3-1)
BCF = bioconcentration factor (mg toxicant/kg fish divided by toxicant/L water) for fish with 3% lipid content.

While, the exposure assumptions used to estimate risks are based on human data, they should be protective of any mammal with a body weight of equal to or less than 70 kg, a drinking water consumption rate of 2 liters per day, and a fish consumption rate of 6.5 g per day. The exposure duration for non cancer endpoints will vary depending on the chemical effect and the condition of the population at risk. The exposure duration for carcinogens is 70 years. Since, the exposure assumptions for the mammals other than humans is unknown there is additional uncertainty which may increase or decrease the risk for these species.

The possibility of exposure to toxic pollutants via contamination of plant materials in aquatic systems is unlikely as well. Generally, the herbivorous species do not feed in or very near to aquatic habitats. The exposure of gray wolves and lynx to arsenic via this pathway would require that prey species consume enough contaminated vegetation to pass on a significant amount to their predators. Biomagnification through plants directly to mammals is uncommon. From this information, EPA has determined that the approval of the **acute and chronic numeric criteria for toxic pollutants** established by the Idaho Water Quality Standards **is not likely to adversely affect the gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, and woodland caribou**.

Plants

The four threatened or endangered plant species in Idaho do not exist in areas constantly inundated by water, therefore the effects of aquatic contaminant exposure should be minimal. The Ute ladies' tresses is a terrestrial orchid species that is only periodically exposed to surface waters. This species generally inhabits river shores where inundation occurs infrequently (Sheviak, 1984). McFarlane's four o'clock, also a terrestrial plant species, occurs in grassland habitats characterized by warm and dry conditions (FWS, 1997b). Exposure to surface water would generally occur in these areas only during rare flooding events when dilution of contaminants and length of exposure to contaminated water would minimize toxicity. Water howellia, an aquatic macrophyte, grows mostly in wetlands associated with temporary water bodies such as ephemeral glacial pothole ponds and former river oxbows (FWS, 1994b). This plant requires the seeds to dry out completely for germination to occur. The Spalding's catchfly primarily inhabits prairie or steppe grassland vegetation and does not tolerate extremely wet soils. Therefore, because of the lack of exposure to contaminants in aquatic systems, EPA has determined that the approval of the **acute and chronic numeric criteria for toxic pollutants** established by the Idaho Water Quality Standards **is not likely to adversely affect the water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's Catchfly**.

C. Level of Analysis for Determinations to Invertebrates, Fish, and Birds

Of the priority pollutants with Aquatic Life Criteria (see list below), it was jointly determined by EPA and the Services that some chemicals required a more detailed analysis. EPA examined the effects of nine chemicals: arsenic, cadmium, copper, lead, mercury, selenium, zinc, and cyanide, in more detail due to their prevalence in Idaho waters. Endosulfan was also addressed in more detail because of its current agricultural use in Idaho. Chromium III, chromium VI, nickel, silver, and Heptaclor/Heptachlor Epoxide were provided a minimal level of analysis because these chemicals do not occur in Idaho waters with the same regularity. The remaining 9 organic chemicals listed were also given a minimum level of analysis since their use has either been canceled or suspended.

For those chemicals given a minimum level of analysis, EPA primarily relied upon information provided in EPA's water quality criteria guidance documents (1980-1985).

Priority Pollutants for Aquatic Life Criteria

Tier I, High Level of Analysis:
Arsenic
Cadmium
Copper
Lead
Mercury
Selenium
Zinc
Cyanide
alpha and beta Endosulfan

Tier II, Low Level of Analysis:
Chromium (III)
Chromium (VI)
Nickel
Silver
Heptachlor/Heptachlor Epoxide
Pentachlorophenol
Aldrin
gamma-BHC (Lindane)
Chlordane
4-4' DDT
Dieldrin
Endrin
PCBs (PCB-1242,1254, 1221, 1232, 1248, 1260, 1016)
Toxaphene

1. Methods for Determinations

Determinations regarding the potential for the criteria established by the Idaho Water Quality Standards to adversely affect threatened and endangered species were made as follows. Acute criterion were compared to published toxicity data where exposure durations were less than or equal to 96 hours. Chronic criterion were compared to published toxicity data where exposure durations were greater than 96 hours. While the scientific community does not agree on precise definitions for the terms acute and chronic, the general approach used here can offer an adequate assessment of the criteria's potential effects on aquatic species.

For all aquatic species except sturgeon, a "may be likely to adversely affect" determination was made if 1) no information was available detailing the toxicity of the chemical with regard to the species of concern or a reasonable surrogate, or 2) the published toxicity data indicated adverse effects at concentrations at or below the established criteria. A "not likely to adversely affect" determination was made if the published toxicity data indicated adverse effects at concentrations above the established criteria. Adverse effects on species were divided into sublethal and lethal effects. Sublethal effects included any measurable or observable effect on a species, not including mortality, while lethal effects consisted only of mortality. Both lethal and sublethal effects were evaluated for each criterion. Generally, in an effort to refrain from duplicating previous work, reports reviewed for this document were published after the publication of EPA's criteria documents for the chemicals reviewed here. Most of the criteria documents were published between 1980 and 1985. In some cases, where information was lacking, we have included data published prior to the criteria documents.

Rather than taking the default approach and assigning a 'likely to adversely affect" determination for white sturgeon, we have chosen to evaluate the proposed standards by examining toxicity data for a variety of fish species, including cold water species (e.g. salmonids) and benthic species (e.g. catfish). If the proposed standards are protective of a variety of fish species, we can assume that the standards will also adequately protect white sturgeon for the following reasons: 1) the proposed standards are below the limits for other fish species and 2) the limited data available show that sturgeon have variable sensitivity compared to other species (i.e. they are not consistently more sensitive than other species).

Bioconcentration
In determining sublethal effects to invertebrates and fish, EPA has concluded that bioconcentration (an increase in concentration of a substance in relation to the concentration in the ambient environment) is an indicator of exposure to chemicals, but will not be classified as an effect. The concentration of chemicals in tissues of aquatic organisms can be an excellent indicator of environmental exposures, but bioconcentration alone does not constitute an effect to an organism. Effects may occur as a result of the bioconcentration, and where the studies reviewed for this document illustrated effects coincident with bioconcentration, we have included that information in the sections detailing effects to organisms. Otherwise, when the results of the studies reviewed included only bioconcentration of contaminants, information regarding those studies was described in the "Bioconcentration and Biomagnification" sections for each chemical.

Dietary exposure to birds
Several models were examined to determine dietary levels of toxicants in organisms exposed to parameters at the adopted water quality criteria concentrations. Information is inadequate to employ wildlife models for molluscs. Often, a model requires wildlife values that are unavailable for the species of concern, or the concentration of the chemical in the sediment is needed. For fish, even if a BCF or BAF is available for a particular species, the wildlife value may not be available. Also, the more complicated models require many assumptions that can cover a wide range. For example, feeding rates, amount of diet comprised of a "contaminated" food source, potential food source trophic levels, metabolic rates, and sensitivity factors, can vary by orders of magnitude. The lowest tissue concentration of a chemical in the diet that will not cause adverse effects, the NOAEL, is also expressed as "wildlife value" or "body burden". These wildlife values can cover a large range for the same organism depending on the researcher's assumptions. Given the latitude in variables such as those mentioned above, and the

specific requirements of the food chain/wildlife models, a general approach was chosen to estimate effects on wildlife. The example at the end of this section shows this approach. To estimate the effects of an adopted water quality criterion on "higher" organisms, raptors were selected. Specifically, the bald eagle and peregrine falcon are species of concern. The bald eagle and peregrine falcon are both listed under the Endangered Species Act (ESA) for Idaho. For higher priority chemicals, no wildlife-diet values are available for these bird species. Wildlife values for other bird species or alternately, general wildlife values are available. For many of the parameters of concern, BAFs/BCFs are available for fish, or more specific, for trout. Since eagles may feed at least somewhat on fish, if a fish BAF is available for a particular parameter, then a general wildlife exposure to an eagle can be estimated for that parameter. BCFs in aquatic life allow for the general approach presented below (that is, substituting a BCF for lack of a BAF). The conservative assumptions of a 100 percent fish diet and that all fish eaten were contaminated, were made.

Equation to estimate toxicant exposure to birds through diet:

> toxicant (mg/L) X BCF or BAF (mg/kg in fish/ mg/L in water) = mg/kg in diet (assuming 100% fish diet)

2. Effect of Abiotic Conditions on Toxicity

pH

The toxicity of several pollutants vary depending upon environmental conditions such as water hardness and pH. pH activity has a significant impact on the availability and toxicity of metals. The following is summarized from Elder (1988) and Baker et al. (1990) IN ODEQ (1995). Metal-hydroxide complexes tend to precipitate (i.e., reduced ability to remain suspended) and are quite insoluble under natural water pH conditions; thus, the metal is not able to exert a toxic effect. However, the solubility of these complexes increases sharply as pH decreases. pH activity also impacts the sensitivity of organisms to a given amount of metal. There are two types of metals: type I metals (e.g., cadmium, copper, and zinc), that are less toxic as the pH decreases; and type II metals (e.g., lead), that are more toxic at lower pH values. Each metal has its own range where pH and site-specific conditions become factors in the metal's bioavailability. Aluminum is the metal of greatest concern at low pH values. No adverse effects to listed species due to pH-driven changes in metal toxicity (where the metals comply with the respective metals criteria) would occur in the range of Idaho's pH criteria. Both the direct toxicity of pH and that of aluminum result in osmoregulatory failure. The effects of low pH are also more pronounced at low concentrations of calcium. In summary, reductions in pH below natural levels will tend to increase metal availability and toxicity.

Temperature

No single pattern exists for the effects of temperature on the toxicity of pollutants on aquatic organisms. Temperature change in a given direction may increase, decrease, or cause no change in toxicity depending on many factors including the toxicant, species or the experiment. Sprague (1985) demonstrates that the effects of temperature on acute toxicity are diverse, but for the most part are only small or moderate. Some evidence suggests that temperature may not have much effect at all on the chronic "no-effect" thresholds of pollutants. One study described that temperature may either increase or decrease the EC_{50}, but no general pattern was evident. The researchers concluded that temperature had no effect on the EC_{50} (Sprague, 1985).

Dissolved Oxygen

Reductions in dissolved oxygen may increase the toxicity of aquatic pollutants, but are often not the major factors affecting toxicity. Most evidence suggests that tests conducted at low and high levels of dissolved oxygen may change toxicity by only a factor of 2 or less (low dissolved oxygen being generally in the range of 20% saturation). In studies where low dissolved oxygen significantly modified LC_{50}s, the same effect did not hold true for sublethal toxicity (i.e. growth). Low oxygen appears to be less important than might be expected as a modifier of sublethal toxicity. Sprague suggests that while the picture of the influence of dissolved oxygen on toxicity is incomplete, "the effects may be as small as, or even smaller, than the modest effects on acute lethality" (Sprague, 1985).

3. Uncertainty Analysis

Concentrations of metals in the water column may be measured as either total recoverable or dissolved. The Idaho Water Quality Standards express metals concentrations as dissolved metals. Total recoverable analysis of metals allows an estimation of metal content of both the water and particulate matter. Since total recoverable methods take into account both dissolved and bound metal fractions, this method can sometimes overestimate the toxicity of an aquatic system. Dissolved analysis of metals estimates only the metal actually dissolved in the water column. This method may represent more closely the fraction of the metal that is bioavailable to aquatic organisms. However, it does not address metals bound to particulate matter and may underestimate the toxicity of an aquatic system by excluding ingestion of particulates or ingestion of prey that consume particulates as a pathway for toxic chemical exposure. In addition, in the laboratory, total recoverable methods are often used to determine metal concentrations.

Toxicity of several pollutants for which criteria are included in the Idaho Water Quality Standards are either pH or hardness dependent. In these cases, the State's criteria are expressed as a function of pH or hardness. However, in many cases the literature does not report the environmental conditions under which toxicology experiments have been performed, including pH and hardness. Where relevant, EPA's analysis took into account whether pH and hardness values were provided. Where pH and hardness values were not reported in the literature and the criteria are expressed as a function of pH or hardness, the results should be interpreted with caution.

Other factors may also limit the accuracy of the determinations on the effects of the Idaho Water Quality Standards aquatic life criteria. First, the analysis of the criteria did not address the effects of the criteria on prey items of individual species or on their habitat beyond the water column. Toxic chemicals may affect aquatic organisms via ingestion (of contaminated prey or sediment particles) or through absorption (from water or from sediment). Furthermore, prey populations may decrease as a result of chemical contamination, thus depriving a species of food sources. The development of the criteria included effects for many prey species and should adequately protect prey of the listed species examined in this document. Second, the Idaho Water Quality Standards aquatic life criteria do not take into account the interactions between two or more chemicals which could be present in a water body. Some chemicals may interact resulting in more or less toxicity of one or more of the chemicals involved. Some metals such as cadmium and selenium exhibit antagonistic relationships with respect to toxicity. The literature did not provide any evidence to indicate synergistic interactions between metals (Furness and Rainbow, 1990). Synergism is defined as the interaction of toxicants resulting in greater toxicity than that predicted by the sum of the toxicities of each chemical. Finally, the analysis of the potential effects of toxic pollutants on threatened and endangered species included the examination of research conducted primarily with surrogate species. The surrogate species were selected as the closest related organism for which information was available. For example, little research exists describing the effects of toxic chemicals on chinook and sockeye salmon, but a wealth of information exists describing the effects of toxic chemicals on rainbow trout. Therefore, rainbow trout often served as a surrogate species to determine the effects of toxic pollutants on chinook and sockeye salmon.

4. Organization of Toxic Pollutant Determinations for Invertebrates, Fish, & Birds

For each of the chemicals receiving a high level of analysis, the determination section is organized as described here: a preliminary description of the chemical and criterion followed by an evaluation of recent research on each of the species of concern or their surrogates. The species are considered together in phylogenetic groups such as invertebrates, fish, birds, mammals, and plants. Within the evaluation for invertebrates and fish, sublethal, and lethal effects are evaluated separately. Determinations for the chemicals that received a minimal level of analysis are grouped together at the end of this section. For each of these chemicals, some background information is provided along with an effects determination.

Chemical Analysis for Metals

Three methods for partitioning metals in surface waters have historically been important in the development of water quality criteria. These are total recoverable, dissolved, and acid soluble. For total recoverable metals, a procedure using nitric and hydrochloric acids is given in section 4.1.4 of "Methods for Chemical Analysis of Water and Wastes, 1979 and 1983" (EPA, 1983). Analysts should be cautioned, however, that this digestion may not be adequate for all samples. For dissolved metals, samples are passed through a 0.45 micron membrane filter prior to acid preservation, digestion, and analysis. For acid soluble metals, the procedure is given in method 200.1 of "Methods for the Determination of Metals in Environmental Samples, 1991" (EPA, 1991a). This method requires the sample pH to be adjusted to 1.75 ± 0.1, held for 16 hours and filtered through a 0.45 micron filter membrane prior to analysis. The acid soluble procedure is applicable for the determination of arsenic, cadmium, chromium, copper, and lead. Idaho's Water Quality Standards are measured as dissolved metals.

Where appropriate, the first paragraph of each subsection states the dissolved criterion (as adopted under Idaho's Water Quality Standards) and the corresponding total recoverable criterion. Laboratory testing most often uses total recoverable methods of determination for metals. Total recoverable criteria and dissolved criteria are related by a conversion factor promulgated by EPA on May 4, 1995 (EPA, 1995). These factors are listed in Table 250.07.a.1. To obtain a dissolved criterion from a total recoverable criterion, multiply the total recoverable criterion by the appropriate conversion factor.

D. Arsenic

The current Idaho Water Quality Standards specify criteria for dissolved acute and chronic arsenite, also known as trivalent arsenic (As), of 360 µg/L and 190 µg/L as acute and chronic criteria, respectively. The corresponding total recoverable criteria for arsenic are the same as the conversion factor for arsenic is 1.0. For pentavalent arsenic (arsenate), insufficient data is available to develop criteria, however the lowest-observed-effect levels (LOEL) measured as total recoverable for freshwater environments are 850 µg/L for acute exposures and 48 µg/L for long-term exposures (EPA, 1986b).

Arsenic occurs naturally in aquatic environments in trace amounts. Typical concentrations for background freshwater streams and rivers are less than 1 µg/L As (Moore and Ramamoorthy, 1984). The toxicity of arsenic can be altered by a number of factors including pH, Eh (redox potential), organic matter, phosphate content, suspended solids, presence of other toxicants, speciation of the chemical itself, and the duration of exposure to arsenic. Temperature has been shown to alter the toxicity of arsenic. In fish, tolerance of arsenic appears to increase with temperature; (McGeachy and Dixon, 1990) whereas in invertebrates the opposite is true (Bryant et al., 1985). Inorganic forms of arsenic are typically more toxic to aquatic species, particularly the more sensitive early life stages (Eisler, 1988a). While evidence does suggest that toxicity of arsenic can be altered by both temperature and phosphorus (two concerns for the mid-Snake River in Idaho), enough information to clearly characterize the relationship between arsenic toxicity and these two factors does not exist.

1. Bioconcentration and Biomagnification

Arsenic does not readily bioconcentrate (an increase in concentration of a substance in relation to the concentration in the ambient environment) in aquatic species. It is typically water soluble and does not combine with proteins. Planktivorous fish are more likely to concentrate arsenic than omnivorous or piscivorous fishes (Hunter, 1981). In 1995, Robinson et al. found no evidence of arsenic uptake or accumulation from water in both rainbow and brown trout.

Eisler (1988a) also found no evidence that biomagnification (a progressive increase in concentration from one trophic level to the next higher level) occurs in aquatic food chains. Aquatic invertebrates have been noted to accumulate arsenic more readily than fish, also an indication that biomagnification is unlikely (Spehar et al., 1980).

2. Invertebrates

Sublethal effects

Golding et al. (1997) studied non-lethal responses, such as avoidance and immobility, for the freshwater snail, *Potamopyrgus antipodarum*, when exposed to arsenic. The snails appeared to be more sensitive to arsenite (As(III)) than arsenate (As(V)), and those snails that had been exposed to arsenic prior to the experiment were more sensitive than those that had not. However, all concentrations that resulted in avoidance or immobility were greater than 15 mg/L As. Few other documented studies have examined the sublethal effects of arsenic on freshwater snails. While native snails are likely to be somewhat more sensitive than an invasive species such as *P. antipodarum*, this species was the only one for which toxicity information was available. Even with a difference in sensitivity, the effects levels were well above the criteria, indicating that snails will be protected from sublethal effects. From the information available, EPA has determined that both the acute and chronic arsenic criteria established in the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

At arsenic concentrations up to 973 µg/L, the snail species, *Stagnicola emarginata*, and *Helisoma campanulatum*, experienced less than 20% reduction in survival (Spehar et al., 1980). Bioconcentration factors for these species were also relatively low, between 2-5, depending on the form of the arsenic (EPA, 1985a). EPA (1985b) lists an LC_{50} (the concentration at which 50% of the test organisms die) of 24,500 µg/L for the freshwater snail, *Aplexa hyporum*. Arsenic concentrations that have been found to affect freshwater snails' survival are well above the current arsenic criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the acute and chronic arsenic criteria established in the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

From the information available, EPA has determined that the approval of the **acute and chronic arsenic criteria** (360 µg/L=acute, 190 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

Sublethal effects

Sublethal effects including anemia, gallbladder inflammation, and liver degeneration, were observed at aquatic concentrations of 9.64 mg/L and dietary concentrations of 43.1-60 µg/g (Cockell et al., 1992; Woodward et al., 1994; and Rankin and Dixon, 1994). Studies of the effects of long-term arsenate exposures (11 weeks) found that rainbow trout were more tolerant of arsenic concentrations ranging from 5-36 mg/L at higher temperatures, in this case, 15°C versus 5°C (McGeachy and Dixon, 1990). However, a previous study conducted by the same investigators found rainbow trout to be more tolerant of acute arsenate exposures (1.5-18 mg/L for 144 hours) at lower temperatures, again, 15°C versus 5°C (McGeachy and Dixon, 1989). Arsenate is the most stable inorganic form of arsenic in aquatic systems (Eisler, 1988a). Oladimeji et al. (1984) found that arsenic in dietary concentrations of 10-30 mg/kg impaired rainbow trout growth in a dose-dependent manner and caused an inversely related decreases in hemoglobin levels. Pre-exposed fish are more tolerant of arsenic unlike the decreased tolerance seen in invertebrates (Dixon and Sprague, 1981). In other studies of sublethal effects of arsenic, adult and juvenile coho salmon were exposed to 300 µg/L As for 5 and 6 months. The adult coho salmon were exposed for 5 months and experienced some physiological alterations (EPA, 1985a). Parr-smolt coho exposed to the same arsenic concentration for 6 months experienced delayed onset of plasma thyroxine, transient reduction of gill sodium and potassium ATPase activity, and reduced successful seaward migration (Nichols et al., 1984). Therefore, EPA has determined that the acute and chronic arsenic criteria established by the Idaho Water Quality Standards are not likely to adversely affect the health and behavior of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

Various studies estimate LC_{50} values for salmonids to be between 3,000 and 167,000 µg/L As (Hamilton and Buhl, 1990; EPA, 1985a). Estimates of LC_{50}'s for arctic grayling were above 8940 µg/L As (Hamilton and Buhl, 1990). For rainbow trout embryos, an LC_{50} as low as 550 µg/L As after a month long exposure and an LC_{10} (concentration at which 10% of test organisms are killed) of 134 µg/L As after a 28-day exposure, indicate that salmonids may be more sensitive in early life stages (Birge et al., 1980). Even though the LC_{10} concentration is below Idaho's chronic arsenite criteria, EPA previously evaluated this study when it determined the current aquatic life criteria for arsenic. Because of issues with the procedures used in this study, EPA did not consider the results of this study as an acceptable basis for lowering the chronic arsenic criteria (C. Stephan, pers. comm., 1999). Therefore, EPA has determined that the acute and chronic arsenic criteria established by the Idaho Water Quality Standards is not likely to adversely affect the survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

From the studies that investigated the effects of arsenic exposures on salmonids, it was determined that rainbow trout embryos may experience some mortality at arsenic concentrations less than those established by the chronic arsenic criteria established by the Idaho Water Quality Standards. Due to a lack of explanation of experimental procedures in the research reports, it is impossible to determine the quality of the results.

While EPA has determined that the approval of the acute and chronic aquatic life arsenic criteria established by the Idaho Water Quality Standards may have the potential to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout. An added level of protection is offered by the following:

> The human health arsenic criterion of 50 ug/l is the applicable arsenic criteria in all waters of Idaho. This criterion is significantly lower and more conservative than the acute and chronic aquatic life criteria.

If a recreational use is modified or removed from a waterbody and the criteria become less stringent than 50 ug/l, Idaho is required to submit this revision to EPA for approval/disapproval action. If EPA proposes to approve this revision, the agency will then reinitiate consultation on that approval action .

In light of these currently effective measures, EPA has determined that the approval of the **acute and chronic arsenic criteria** (360 µg/L=acute, 190 µg/L=chronic) established by the Idaho Water Quality Standards is **not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

The general effects of arsenic poisoning in birds include instability, drooped eyelids, huddled position, unkempt appearance, immobility, seizures, and death. These effects can be observed anywhere from 1 hour to 6 days after dietary exposure (Eisler, 1988a). Concentrations greater than 2,000 mg/kg of arsenic herbicide and greater than 47.6 mg/kg sodium arsenate were observed to be the LD_{50}s for avian species. (Hudson et al., 1984). In mallards, arsenic accumulates in adult livers and eggs. Arsenic exposures of 25-400 µg/kg resulted in reduced egg weight, eggshell thinning, and reduced duckling growth and reduced liver and body weight. These arsenic concentrations did not affect hatching success or increase duckling mortality, but did decrease overall duckling production (Stanley et al., 1994). Stanley et al. (1994) also found that arsenic alleviated selenium effects on hatching success and embryo deformities at dietary concentrations of 400 µg As/g. Dietary arsenic levels from 30-300 ppm (mg/kg) decreased growth rates in female mallards, but only levels of 300 ppm reduced growth rates in males (Camardese et al., 1990). In chickens, arsenite, and arsenate concentrations of 0.01-1.0 µg/embryo resulted in 34% and 8% survival, respectively. For both forms of arsenic, 0.03-3.0 µg/embryo was found to be the malformation threshold (Eisler, 1988a).

Table 250.07.a.2 illustrates a conversion of the maximum allowable water criteria under the Idaho Water Quality Standards to dietary concentrations for piscivorous birds using the maximum bioconcentration factor (BCF) for fish provided by Eisler (1988a). This allows an interpretation of the dietary concentrations referenced above in the context of the current Idaho Water Quality Standards. A dietary concentration of 25-370 mg As/ kg is deemed safe for

chickens (Eisler, 1988a). Even at the maximum allowable water criteria, birds would not consume a dietary arsenic concentration greater than that deemed safe by Eisler (see Table 250.07.a.2). Therefore, from the information available with regard to arsenic toxicity to avian species, EPA has determined that the approval of the **acute and chronic arsenic criteria** (360 µg/L=acute, 190 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

Table 250.07.a.2. Dietary Concentrations allowed by Idaho Water Quality Standards				
Dietary Concentrations		Arsenic Criteria		Maximum BCF for fish, as prey for birds (Eisler, 1988a)
Acute	Chronic	Acute	Chronic	
1.16 mg As/kg	3.23 mg As/kg	0.360 mg As/L	0.190 mg As/L	17

E. Cadmium

The current Idaho Water Quality Standards establish cadmium (Cd) criteria which are hardness dependent. At a hardness of 100 mg/L $CaCO_3$, the criteria are 3.7 µg/L and 1.0 µg/L for short-term and long-term exposures, respectively (See Table 250.07.a.3). Table 250.07.a.4 lists the criteria for hardness values used in the studies referenced in this report. The corresponding total recoverable criteria are 3.9 µg/L and 1.1 µg/L.

Table 250.07.a.3. Idaho Cadmium Water Quality Criteria				
Hardness	Acute Criteria		Chronic Criteria	
	Total	Dissolved	Total	Dissolved
100 mg/L $CaCO_3$	3.9 µg/L	3.7 µg/L	1.1 µg/L	1.0 µg/L

Table 250.07.a.4. Idaho Water Quality Criteria for Cadmium Calculated for Referenced Hardness Values and Total Recoverable Analysis		
Hardness	Acute Criteria (total recoverable)	Chronic Criteria (total recoverable)
9.2 mg/L $CaCO_3$	0.27 µg Cd/L	0.17 µg Cd/L
19.5 mg/L $CaCO_3$	0.62 µg Cd/L	0.31 µg Cd/L
23 mg/L $CaCO_3$	0.75 µg Cd/L	0.36 µg Cd/L
120 mg/L $CaCO_3$	4.8 µg Cd/L	1.3 µg Cd/L
165 mg/L $CaCO_3$	6.9 µg Cd/L	1.7 µg Cd/L

Cadmium naturally occurs in the aquatic environment, but is of no known biological use and is considered one of the most toxic metals. Concentrations of cadmium associated with background freshwater systems are estimated to range between 0.05-0.2 µg/L (Korte, 1983). While cadmium is released through natural processes, anthropogenic cadmium emissions have greatly increased its presence in the environment. In aquatic systems, cadmium quickly partitions to sediment, but is readily remobilized through a variety of chemical and biological processes (Currie et al., 1997). Toxicity of cadmium to aquatic organisms varies with the type and life stage of organisms, presence of other toxicants and the duration of exposure. Hardness affects cadmium toxicity as well. Møller et al. (1994) discovered that the toxicity of cadmium increases with increasing temperature (5-20°C) for one freshwater snail species at concentrations of 1-4 mg/L Cd. Currie et al. (1997) also found that cadmium can be transported from aquatic to terrestrial food webs by emerging insects. Cadmium removal from aquatic systems by aquatic insects has been shown to be significant: 1.3-3.9 g Cd/ year removed by insects out of a total 0.26-19.5 g Cd/ year removed. Pip (1992) found that cadmium concentration is negatively correlated with percent organic matter in natural environments. The presence of zinc and selenium have been shown to antagonize the toxic effects of cadmium. Other metals do not appear to compete with cadmium for receptors in aquatic organisms nor is there evidence for synergistic toxicity (Furness and Rainbow, 1990).

1. Bioconcentration and Biomagnification

Cadmium does not bioconcentrate (an increase in concentration of a substance in relation to the concentration in the ambient environment) significantly in fish species, but does tend to accumulate more readily in invertebrates. Pip (1992) found that snails accumulated a significantly higher level of cadmium when compared with the surrounding habitat. In a study of the uptake and depuration of cadmium in *Lymnea stagnalis*, Presing and Salanki (1993) found that the shells did not uptake cadmium, but the soft body tissues were saturated at concentrations of 200 µg Cd/g. The study exposed the snails to 0.1 mg Cd/L for four weeks then allowed an 8 week depuration period in clean freshwater. Even after 8 weeks in freshwater, Presing et al. still found significant cadmium concentrations in tissues. The average hardness for this experiment was 19.5 mg/L $CaCO_3$. Omnivorous and insectivorous predators tend to accumulate cadmium in their tissues more than piscivorous predators (Scheuhammer, 1991). After 7, 15, and 30 day exposures to the Po River in Italy, rainbow trout were found to have cadmium residues in the kidney, spleen, gills, muscle, and bone tissues ranging from 0.01-0.38 µg/g. The cadmium concentrations in the tissues increased with longer exposure durations. Cadmium concentrations characteristic of the Po River range from 0.07-0.26 µg/L (Camusso and Balestrini, 1995). Concentrations of cadmium in fish tissues reflect the bioavailability of cadmium in the water. It is unknown what effects may be associated with high tissue cadmium concentrations. No hardness information was reported for the Po River. The nature of this study as a field sample also prohibits the ability to determine whether accumulation of cadmium resulted from exposure to the waters sampled during the study. Saiki et al. (1995) found no evidence of biomagnification (a progressive increase in concentration from one trophic level to the next higher level) in steelhead on the Upper Sacramento River. Eisler (1985a) also maintains that evidence for cadmium biomagnification suggests that only the lower trophic levels exhibit biomagnification.

Cadmium tends to form stable complexes with metallothionein (a sulfhydryl-rich protein). The resulting cadmium complexes have long half-lives and a tendency to accumulate with age in exposed organisms. As such, long lived species tend to be at a higher risk from chronic low-level dietary cadmium exposure.

2. Invertebrates

Availability of cadmium to predators depends on the cell structure of the prey organism. In snails, cadmium is more available to predators than other metals because it is bound to soluble proteins in the snail's digestive glands, as illustrated by Nott and Nicolaidou's (1994) study of *Littorina littorea*. When predators of snails consume cadmium contaminated tissues, the metal may be remobilized and absorbed by the predator, thus causing retention of cadmium along the food chain (Nott and Nicolaidou, 1994).

Sublethal effects

In a study on two terrestrial snails (*Helix aspersa aspersa* and *H. aspersa maxima*), Gomot (1997) found that growth was affected at dietary concentrations of 100 µg Cd/g. Gomot did not observe differences in feeding rate up to concentrations of 400-800 µg Cd/g depending on the snail species. Growth studies on marine snails determined a significant reduction in growth at cadmium concentrations of 100-200 µg/L and a relationship between cadmium toxicity and salinity (Forbes, 1991).

The sublethal effects outlined above occurred at cadmium concentrations well above the cadmium criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the acute and chronic cadmium criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

Allah et al. (1997) exposed both healthy and schistosome infected freshwater snails (*Biomphalaria glabrata*) to waterborne cadmium for six weeks. At the end of two weeks, the LC_{25} (the concentration that is lethal to 25% of the population) was estimated to be 0.22 µM (24.73 µg Cd/L). The concentration ranged from 0.075-0.25 µM (8.43-28.1 µg/L); no hardness was given for this experiment.

Acute toxicity tests for aquatic snails conducted by Møller et al. (1994) and Allah et al. (1997) resulted in LC_{50} and LC_{25} estimates for two snail species, *Potomopyrgus antipodarum* and *Biomphalaria glabrata*. The LC_{50} for *P. antipodarum* was calculated as being between 1-4 mg Cd/ L, but water hardness was not included in the description of this research (Møller et al., 1994). For *B. glabrata*, LC_{25}'s were given as 0.22 µg Cd/ L and 0.08 µg Cd/ L for healthy and schistosome infected animals, respectively, but again, no hardness was given for this experiment (MacInnis and Voge, 1970). The hardness of the test waters would need to be below 20 mg/L for these results to be below the cadmium criteria. If the research was conducted in waters with such low hardness, it is unlikely the research would have intended to evaluate the effects of low hardness on toxicity. For other snail species, LC_{50} values were greater than 10 µg Cd/L at hardness values ranging from 44-58 mg $CaCO_3$. While native snails are likely to be somewhat more sensitive than an invasive species such as *P. antipodarum*, this species was the only one for which toxicity information was available. Even with a difference in sensitivity, the effects levels were well above the criteria, indicating that snails will be protected from sublethal effects. From this evidence, EPA has determined that the acute and chronic cadmium criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

From the information available, EPA has determined that the approval of the **acute and chronic cadmium criteria** (3.7 µg/L=acute, 1.0 µg/L= chronic, with hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

Sublethal effects

Hontela et al. (1996) exposed juvenile steelhead to cadmium concentrations of 400-2,400 µg/L for durations ranging from 2 hours to 1 week. After only 2-4 hour exposures, the fish experienced a significant increase in plasma thyroxine levels. Plasma cortisol levels significantly increased after 96 hours. Both plasma thyroxine and glucose, as well as liver glucose were significantly lower after one week. The hardness for this experiment was 110 mg/L.

While the obviously adverse sublethal effects recorded by the previously described studies occurred at concentrations higher than the acute and chronic cadmium criteria established by the Idaho Water Quality Standards.

Based on the above information, EPA has determined that the acute and chronic cadmium criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

In chinook salmon, EPA (1985c) reported LC_{50} values for 96 hour exposures ranged from 1.1-3.5 µg Cd/L dependent on the life stage tested (hardness=23). Sastry and Shukla (1994) found an LC_{50} value of 11.2 mg/L for the freshwater fish, *Channa punctatus,* when exposed to this concentration for 96 hours at a hardness of 165 mg/L.

The reported lethal cadmium concentrations are above the cadmium criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the acute and chronic cadmium criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

Some reviewers of this document have expressed concern for the accumulation of cadmium in salmonid prey. However, cadmium has not been shown to accumulate significantly in benthic invertebrates in field studies (Farag et al., 1998; Woodward et al., 1994). From the available information, EPA has determined that the approval of the **acute and chronic cadmium criteria** (3.7 µg/L=acute, 1.0 µg/L= chronic, with hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.**

4. Birds

In birds, most of the body burden of cadmium is localized in the liver and kidneys. Michot et al. (1994) found an average of 0.91 µg Cd/ g dry weight in livers of waterfowl (redheads, *Aythyta americana*). The range of concentrations was 0.8-5.41 µg/g with a background measurement of 2 µg/g. Baron et al. (1997) found cadmium in feathers, eggshells, and organs of kingfishers (*Ceryle alcyon*), but the levels were below those expected to cause acute toxicity or reproductive impairment in birds. In healthy birds from unpolluted areas, cadmium concentrations ranged from 0.1-32 mg Cd/ kg in livers and 0.3-137 mg Cd/ kg in kidneys. Scheuhammer (1997) has shown that increasing the dietary calcium can reduce kidney and liver cadmium concentrations in birds.

Table 250.07.a.5 illustrates a conversion of the acute and chronic cadmium criteria under the Idaho Water Quality Standards to dietary concentrations for piscivorous birds using the maximum bioconcentration factor (BCF) for fish provided by Eisler (1985a). Eisler (1985a) gives a general wildlife No Observable Adverse Effect Level (NOAEL) of 0.1 mg Cd/ kg diet. At high hardness values, this amount is exceeded by the acute cadmium criterion established by the Idaho Water Quality Standards (see Table 250.07.a.5, Hardness=100). However, the accumulation of metals from food generally occurs over a longer period of time than that for which the acute criterion is applied. For this reason, the accumulation potential was evaluated based on exposure to the chronic criteria. At the highest recorded hardness value, 404 mg/L and the highest known BCF value, 540, the dietary concentration of cadmium to which birds would be exposed is 1.83 mg/kg. From this information, EPA has determined that the approval of the **acute and chronic cadmium criteria** (3.7 µg/L=acute, 1.0 µg/L= chronic, with hardness of 100 mg/L $CaCO_3$) established by Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

Table 250.07.a.5. Dietary Concentrations allowed by Idaho Water Quality Standards					
Dietary Concentrations		Cadmium Criteria (total recoverable)		Maximum BCF for fish, as prey for birds (Eisler, 1985a)	Hardness
Acute	Chronic	Acute	Chronic		
.037 mg Cd/kg	.018 mg Cd/kg	.00067 mg Cd/L	.00033 mg Cd/L	55	21 mg/L $CaCO_3$
.083 mg Cd/kg	.033 mg Cd/kg	.00155 mg Cd/L	.00060 mg Cd/L	55	44 mg/L $CaCO_3$
.215 mg Cd/kg	.061 mg Cd/kg	.0039 mg Cd/L	.0011 mg Cd/L	55	100 mg/L $CaCO_3$

5. Proposed actions for revisions to the chronic cadmium criterion for the protection of threatened and endangered species:

Although EPA has determined that the chronic cadmium criterion was not likely to adversely effect the species of concern in this biological assessment, as the result of the California Toxic Rule consultation, EPA is proposing to revise the chronic aquatic life criterion for cadmium . EPA will develop a revision to its recommended 304 (a) chronic aquatic life criterion for cadmium by January 2001 to ensure the protection of federally listed species and/or critical habitats. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' Opinion, to ensure that the revised criterion will adequately protect federally listed species. If the revised criterion is less stringent than that proposed by the Services, EPA will provide the Services with a biological evaluation/assessment on the revised criterion by the time of the proposal to allow the Services to complete a biological opinion on the proposed cadmium criterion before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate a final criterion as soon as possible, but no later than 18 months, after proposal. If indicated by the results of this revision, EPA will collaborate with Idaho to propose revised criterion in Idaho by January 2002.

F. Copper

The current Idaho Water Quality Standards establish hardness dependent criteria. At a water hardness of 100 mg/L $CaCO_3$, the copper (Cu) criteria are 17 µg/L for short-term exposures and 11 µg/L for long-term exposures (See Table 250.07.a.6). Corresponding total recoverable criteria are 18.0 µg/L and 12.0 µg/L for short-term and long-term exposures, respectively. Table 250.07.a.7 lists the criteria calculated for hardness values used in the studies referenced in this report.

Table 250.07.a.6. Idaho Copper Water Quality Criteria

Hardness	Acute Criteria		Chronic Criteria	
	Total	Dissolved	Total	Dissolved
100 mg/L $CaCO_3$	18.0 µg Cu/L	17.28 µg/L	12.0 µg Cu/L	11.52 µg/L

Table 250.07.a.7. Idaho Water Quality Criteria for Copper
Calculated for Referenced Hardness Values and Total Recoverable Analysis

Hardness	Acute Criteria (Total Recoverable)	Chronic Criteria (Total Recoverable)
12 mg/L $CaCO_3$	2.4 µg/L	1.9 µg/L
13-13.3 mg/L $CaCO_3$	2.6 µg/L	2.1 µg/L
20 mg/L $CaCO_3$	3.9 µg/L	3.0 µg/L
25-30 mg/L $CaCO_3$	4.8-5.7 µg/L	3.6-4.2 µg/L
28.4 mg/L $CaCO_3$	5.4 µg/L	4.0 µg/L

Table 250.07.a.7. Idaho Water Quality Criteria for Copper Calculated for Referenced Hardness Values and Total Recoverable Analysis		
29.69-32.72 mg/L $CaCO_3$	5.6-6.2 µg/L	4.2-4.6 µg/L
35-55 mg/L $CaCO_3$	6.6-10.0 µg/L	4.8-7.1 µg/L
41.3 mg/L $CaCO_3$	7.7 µg/L	5.6 µg/L
46-47 mg/L $CaCO_3$	8.5-8.7 µg/L	6.1-6.2 µg/L
50 mg/L $CaCO_3$	9.2 µg/L	6.5 µg/L
100 mg/L $CaCO_3$	18.0 µg/L	12.0 µg/L

Copper occurs naturally in the environment and is an essential element for most organisms as a component of some oxidative enzymes. Concentrations of copper associated with background freshwater systems are estimated to range between 0.5-1.0 µg/L (Moore and Ramamoorthy, 1984; Groth, 1971). While copper may form complexes with suspended organic matter, it will ultimately settle out of the water column and be deposited in the sediment (EPA, 1984). The toxicity of copper to aquatic organisms is dependent on the speciation of the chemical itself, water hardness and the type and life stage of the exposed organisms. Total organic content in the aquatic system may also decrease copper toxicity, while temperature may affect copper toxicity, although the relationship has yet to be clearly defined. The distinction between deficiency and toxicity for copper is small for organisms that do not have effective mechanisms to control the absorption of copper (e.g. fungi, algae, and invertebrates).

1. Bioconcentration and Biomagnification

Copper is not strongly bioconcentrated (an increase in concentration of a substance in relation to the concentration in the ambient environment) in vertebrates, but is more strongly bioconcentrated in invertebrates. Bioconcentration factors (BCF's) reported in the EPA water quality criteria for copper (EPA, 1984) ranged from zero in bluegill (*Lepomis macrochirus*) to 22,600 in asiatic clams (*Corbicula fluminea*). The concentration of copper in the tissues of aquatic invertebrates is well-documented. In the Mediterranean snail, *Murex trunculus*, Nott and Nicolaidou (1994) found copper to be progressively accumulated with age in the visceral mass of the snail. Copper is bound strongly to sulfur complexes within the snail tissues and is thus less bioavailable to snail predators. Ying et al. (1993) found that the concentration of copper in the tissues of snails (*Polinices sordidus*) differed between organisms of the same species exposed to the same spiked sediment. The total body burden of copper increases with size and weight, and copper is capable of concentrating in the shell of the snail, *Lymnea stagnalis* (Pip, 1992).

In salmonids the accumulation of copper in muscle, kidney, and spleen tissues occurred at copper concentrations ranging from 0.52-3 µg/L in both seawater and freshwater (freshwater hardness=46-47 mg/L; Camusso and Balestrini, 1995; Peterson et al., 1991; Saiki et al., 1995).

The concentrations of copper in fish tissues reflect the amount of bioavailable copper in the environment.

There is little information available concerning biomagnification (a progressive increase in concentration from one trophic level to the next higher level) of copper in aquatic food chains. Also, since the literature describing the effects of copper on birds or mammals are minimal, there is little information from which to quantify the biomagnification of copper. Baudo (1983), Wren et al. (1983) and Mance (1987) have all concluded that copper, along with zinc and cadmium do not biomagnify in the aquatic environment.

2. Invertebrates

Sublethal effects

Copper affects invertebrates in a variety of ways. Kitching et al. (1987) demonstrated a decline in activity for the estuarine snail, *Polinices incei*, as salinity decreased, and copper concentrations increased (range=0.25-1 mg/L). Pipe and Coles (1995) found that previous exposure of marine mussels to copper (0.4 mg/L) resulted in compromised immune systems, as demonstrated by an increased susceptibility to *Vibrio tubiashi*, a bacteria that causes vibriosis or bacillary necrosis.

The speciation of copper can affect its toxicity. Copper sulfate was found to be harmful to freshwater snails (*Bulinus tropicus*). After 6 hour exposures in air to 1 ppt (mg/kg) $CuSo_4$, vascular connective tissues and epithelial cells on the rectal ridge were swollen and enlarged (Wolmorans et al., 1986).

The copper concentrations observed to adversely affect freshwater snails as described above are much greater than the acute and chronic criteria established for copper by the Idaho Water Quality Standards. From this information, EPA has determined that both the acute and chronic copper criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata.

Lethal effects

Lethal effects of copper were observed by Nebeker et al. (1986) for the two snail species, *Juga plicifera* and *Lithoglyphus virens*. LC_{50}'s (concentration at which 50% of the animals die) for 96 hour exposures were calculated as 0.015 mg/L for *J. plicifera* and 0.008 mg/L for *L. virens*. All LC_{50} values were normalized to a hardness of 20 mg/L (see Table 250.07.a.7 for converted criteria). These values are uncommonly low when viewed in reference to other LC_{50}s, but the author attributes that to many factors including the variable water hardness associated with the experiment, the experiment's flow-through setup and the use of operculate versus non-operculate snails. For *Physella cubensis*, a freshwater snail, copper concentrations found to affect the survival, food consumption, growth, egg mass deposition, and embryogenesis were found to be well above the copper criteria established by the Idaho Water Quality Standards. Hart et al. (1993) found that concentrations from 0.01-0.05% (50-253 mg/L) caused complete mortality of the juvenile and adult snails within 3 days of initial exposure. At copper concentrations of

0.001-0.005% (5-25 mg/L), food consumption, growth, egg mass deposition, and embryogenesis were all reduced. No hardness was given for this study. EPA (1984) gives LC_{50} values of 39µg/L for *Physa integra* (hardness=35-55) and 69 µg/L (hardness=100) for *Physa heterostropha*, both freshwater snail species. The LC_{50} values determined for three freshwater snails are 108 µg/L (hardness=100) for *Gyraulus circumstriatus*, 1,700 µg/L (hardness=35-55) for *Campeloma decisum* and 9,300 µg/L and 900 µg/L respectively for embryo and adult *Amnicola* sp. (hardness=50).

The research described here shows that copper concentrations found to cause mortalities in freshwater snails are well above the acute and chronic copper criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the acute and chronic copper criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

Lethal and sublethal effects of copper were observed at copper concentrations above those criteria established by the Idaho Water Quality Standards. From this evidence, EPA has determined that the approval of the **acute and chronic copper criteria** (17 µg/L=acute, 11 µg/L=chronic, hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

Sublethal effects

In fish, the toxicity of copper appears to be inversely related to the tendency of the metal to bind with the external gill surface via ionic interactions. In other words, a lower affinity of the gill surface to copper leads to a greater likelihood of disruption of intracellular processes, which may lead to gill dysfunction (Reid and McDonald, 1991). Some studies have examined the disruption of gill processes by copper. For example, gill Na^+, K^+- ATPase activity in chinook parr was unaffected after an 18 hour exposure to stream water with elevated copper levels of 48 µg/L (hardness=13.3). With the same exposure, significant inhibition of gill Na^+, K^+- ATPase activity was observed in smolts. Significant increases in hematocrit and plasma glucose were also observed in both parr and smolts resulting from the same 18 hour exposure (Beckman and Zaugg, 1988). Divalent copper (Cu^{2+}) totally suppressed gill Na^+, K^+- ATPase activity and produced significant cell damage, edema, mucus production, smoothing of apical membranes, swelling of tubular system and destruction of mitochondria in rainbow trout at concentrations of 0.1 and 1 mM $CuCl_2$, also 13.5 and 134.5 mg/L (Sola et al., 1995). A hardness value was not included in the description of this study. The investigators concluded from this study that bioavailable copper, such as divalent copper, immediately damages the hydromineral balance of rainbow trout and causes morphological modifications that are irreversible.

Carbello et al. (1995) also found rainbow trout to be more susceptible to the microbial parasite, *Saprolegnia parasitica*, at copper levels of 0.25 mg/L (hardness= 28.4 mg/L). Rainbow trout growth was significantly reduced and whole body copper concentrations elevated in fry after 20 days of exposure to copper levels of 4.6 µg/L; whereas 90 µg Cu/L caused a 45% reduction in mean weight after 40 days which was sustained through the end of the experiment at day 60 (hardness=25-30 mg/L; Marr et al., 1996). In another rainbow trout study, Munoz et al. (1991) observed rapid elevations of plasma cortisol, an indicator of stress, after a one hour exposure to 185 ng Cu/L (hardness=12 mg/L). The elevated plasma cortisol levels were maintained throughout the experiment's duration of 21 days.

It has been shown that copper concentrations at or below the established criteria of the Idaho Water Quality Standards may elevate plasma corisol in rainbow trout. However, elevated cortisol levels are only an indicator of physiological stress. No corresponding adverse physiological effects were observed along with the elevated cortisol levels. Therefore, EPA has determined that the acute and chronic copper criteria established by the Idaho Water Quality Standards is not likely to adversely affect the general health and behavior of Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

For adult chinook, an LC_{50} value was determined as 10 µg Cu/L at a hardness of 13 mg/L (EPA, 1984). In steelhead smolts, Chapman (1978) found an LC_{10} of 7 µg Cu/L (hardness=22-25 mg/L). Buhl and Hamilton (1990) also examined copper effects on rainbow trout and calculated a 96-hour LC_{50} of 13.8 µg/L (average hardness=41.3 mg/L). Brook trout were exposed to copper for 24 hours by Drummond et al. (1973), resulting in an LC_{50} calculation of 9 µg/L (hardness = 44-46 mg/L).

From this evidence, EPA has determined that the acute and chronic copper criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

Elevated plasma cortisol was observed in salmonids and sturgeon after exposure to copper concentrations below the criteria established by the Idaho Water Quality Standards. However, it is important to note that adverse physiological effects were not observed along with these results. Therefore, EPA has determined that the approval of the **acute and chronic copper criteria** (17 µg/L=acute, 11 µg/L=chronic, hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

The literature is particularly sparse when it comes to the effects of copper on birds. Wenzel et al. (1996) concluded from a study on kittiwakes (*Rissa tridactyla*) that since copper is regulated metabolically in birds, an increase in copper body burden is not directly proportional to bioavailability. She also observed that concentrations of copper in liver and kidney tissues were low in hatchlings, but increased with age of nestlings, thus indicating the effect of the ingestion of contaminated food during growth. The highest copper concentrations were found in adults. In another study evaluating copper's effects on birds, male broiler chickens were fed from hatching to 42 days of age with a diet containing 250 mg Cu/kg. The birds fed the copper supplemented diet experienced hypocholesterolemia and decreased breast muscle cholesterol (Bakalli et al., 1995).

Table 250.07.a.8 illustrates a conversion of the copper criteria established by the Idaho Water Quality Standards to dietary concentrations for piscivorous birds using the maximum bioconcentration factor (BCF) for fish provided by EPA (1984). This allows an interpretation of the dietary concentrations referenced above in the context of the current Idaho Water Quality Standards. The maximum dietary concentrations for birds with a 100% fish diet that would occur with the acute and chronic copper criteria established by the Idaho Water Quality Standards, are still well below the dietary concentration resulting in adverse effects in chickens. EPA has determined that the approval of the **acute and chronic copper criteria** (17 µg/L=acute, 11 µg/L=chronic, hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

Table 250.07.a.8. Dietary Concentrations allowed by Idaho Water Quality Standards

Dietary Concentrations		Copper Criteria (Total Recoverable)		Maximum BCF for fish, as prey for birds (EPA, 1984)	Hardness
Acute	Chronic	Acute	Chronic		
1.29 mg Cu/kg	0.977 mg Cu/kg	0.00444 mg Cu/L	0.00337 mg Cu/L	290	23 mg/L $CaCO_3$
2.43 mg Cu/kg	1.73 mg Cu/kg	0.00837 mg Cu/L	0.00598 mg Cu/L	290	45 mg/L $CaCO_3$
5.22 mg Cu/kg	3.48 mg Cu/kg	0.018 mg Cu/L	0.012 mg Cu/L	290	100 mg/L $CaCO_3$

G. Cyanide

The current Idaho Water Quality Standards cyanide criteria establish acute and chronic cyanide (CN) criteria of 22.0 µg/L and 5.2 µg/L, respectively.

The Idaho Water Quality Standards are measured as weak acid dissociable, or free cyanide. Free cyanide measurements are a more reliable indicator of toxicity to aquatic life than total cyanide because the latter measurement includes the relatively stable organic cyanides and metallocyanides. The dissociation of metallocyanides is dependent on pH, therefore a measurement of free cyanide at the lowest occurring pH, plus total cyanide measurements should be adequate to monitor freshwater systems. If total cyanide is much higher than free cyanide, the importance of the dissociation of metallocyanide compounds should be given special consideration (EPA, 1985c).

Cyanide occurs naturally in the environment via production by a variety of plant species. In background freshwater systems, the average cyanide concentration is 0.9 µg/L (Eisler, 1991). It is toxic to most living organisms and primarily occurs in aquatic environments as free cyanide (the concentration of HCN and CN). Relative concentrations of hydrocyanide (the more toxic form) and cyanide ion (CN) are dependent on pH and temperature, thus the toxicity of cyanide may increase with decreasing pH and temperature. Other forms of cyanide that may occur are simple cyanides and metallocyanide complexes. Accumulation of metallocyanide complexes in sediment is not likely because dissociation occurs easily at pH values lower than 8. The mechanism of cyanide toxicity involves inhibiting cytochrome oxidase, the terminal oxidative enzyme of the mitochondrial electron transport chain, thus blocking aerobic adenosine triphosphate (ATP) synthesis. The result of this mechanism of toxicity is that cyanide is a rapid and potent asphyxiant (Eisler, 1991). Sarkar (1990) observed that the toxicity of cyanide increased with increasing temperature for fish, molluscs, insects, and plankton, although this relationship was least strong for the mollusc species.

1. Bioconcentration and Biomagnification

Bioconcentration (an increased concentration of a substance in relation to the concentration in the ambient environment) of cyanide is considered to be negligent in fish because the compound is easily metabolized. However, after 16 weeks exposed to thiocyanate (SCN) at 720 or 89 µg/L or CN at 0.98 or 0.32 µg/L, rainbow trout experienced significantly elevated plasma SCN concentrations. The authors concluded that the fish could not “avoid” the contaminated water, because of the contained laboratory setting, and thus could not easily depurate the cyanide (Lanno and Dixon, 1996). In nature, fish may have the ability to move to less contaminated habitat.

As reported by EPA (1985e) the existing literature does not provide evidence for cyanide biomagnification (a progressive increase in concentration from one trophic level to the next higher level). This is likely due to the fact that vertebrate species, such as fish, may readily metabolize cyanide, thus removing the cyanide from the food chain at that level.

2. Invertebrates

Sublethal effects

Little information is available regarding the sublethal effects of cyanide for freshwater snails. In the presence of 26.02 mg CN/L, the active transport of glucose was inhibited on the mucosal and serosal side of the intestine of the freshwater snail, *Biomphalaria glabrata* (El-Shaikh et al., 1993). From this information, it appears that the concentration of cyanide that affects freshwater snails is well above that established by the cyanide criteria of the Idaho Water Quality Standards. With the evidence available, EPA has determined that the acute and chronic cyanide criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

After 96-hour exposures of three snail species, *Lymnaea leuteola*, *Valvata bengalensis*, and *Pila globosa*, to NaCN, Sarkar (1990) calculated LC_{50} values (the concentration at which 50% of the test organisms die) ranging from 1.68-2.97 mg/L. The Sarkar study was the only available recent information concerning the lethal effects of cyanide on freshwater snails. From this information, it appears that the concentration of cyanide that affects the survival of aquatic snails is well above the Idaho Water Quality Standards. Therefore, from the available information, EPA has determined that the acute and chronic cyanide criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

From the information available, EPA has determined that the approval of the **acute and chronic cyanide criteria** (22.0 µg/L=acute, 5.2 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

The toxicity of cyanide may act through the direct action of the cyanide compound itself or through the metabolism of cyanide compounds to thiocyanate (SCN) and the subsequent action of SCN (Lanno and Dixon, 1996). The toxicity of SCN can be influenced by the level of activity maintained by the affected fish. Short bouts of strenuous swimming have been shown to be particularly influential on the toxicity of thiocyanate. However, the mechanism by which the toxicity is increased is not understood (Heming and Blumhagen, 1989). Chlorine concentration in ambient water has also been observed to inhibit the uptake of SCN through the gills in rainbow trout (Heming et al., 1985).

Sublethal effects

At concentrations ranging from 205-670 µg/L, Sarkar (1990) observed fish secreting mucus over their entire bodies, exhibiting respiratory distress, sluggishness, hyperexcitability, loss of equilibrium, hemorrhaging from the gills, and jumping out of the test water. Heming et al. (1985) exposed rainbow trout to 10-518 mg SCN /L and observed a sudden death syndrome that included convulsions, loss of equilibrium and buoyancy, extreme rigor, and death. In rainbow trout, Sawyer and Heath (1988) observed changes in heart rate, ventilatory activity, and oxygen consumption over the course of a 7-hour exposure to 0.02-0.14 mg CN/L (0.02 increase per hour starting at 0 mg CN/L). An increase in respiration rate was also observed in rainbow trout after 96-hour exposure to 9.6 µg CN/L (EPA, 1985c).

After exposure to potassium thiocynate (KSCN) for three hours at concentrations ranging from 85-3,000 mg/L, the incidence of alevin deformities in rainbow trout eggs, both before and after water hardening, increased significantly. In a similar experiment, only exposure concentrations of sodium thiocyanate (NaSCN) of 3,000 mg/L caused a significant increase in the incidence of hatched alevin deformities. Fertilization success was not affected by either chemical at any of the concentrations tested (Kevan and Dixon, 1991). Potassium cyanide (KCN) was not found to be toxic to ova or sperm alone at concentrations up to 1 mg/L. However, rainbow trout fertilization was negatively affected at concentrations as low as 0.001 mg/L when ova and sperm were mixed together in a solution spiked with cyanide (Billard and Rouboud, 1985). Ruby et al. (1993a) observed an absence of an inverse correlation between Type I granular basophils in the pituitary gland and the number of spermatocyte cysts in the testes in rainbow trout exposed for 12 days to 10 µg/L of hydrocyanic acid (HCN). The relationship had been observed previously in untreated rainbow trout. At the same concentration, the number of spermatogonial cysts was found to be statistically higher than that seen in the control fish.

Mean diameters of oocytes were also found to be significantly reduced in ovaries of vitellogenic females exposed for 12 days to 0.01 mg HCN/L (Szabo et al., 1991). Again, after a 12-day exposure to 0.01 mg HCN/L, Ruby et al. (1993b) observed decreases in the gonadosomatic index, plasma vitellogenin levels, plasma 17 - estradiol levels, plasma thyroxine levels, and mean diameter of oocytes of rainbow trout. Similarly, after 7 days exposed to 0.01, 0.02, and 0.03 mg (HCN)/L, rainbow trout exhibited reductions in serum calcium levels at concentrations up to 0.02 mg/L and as well as a decline in hepatosomatic indices (DaCosta and Ruby, 1984). All of the evidence provided in these studies suggest that cyanide can affect reproductive mechanisms in salmonids via the hypophyseal- gonad axis.

Other sublethal effects of cyanide in fish include rapid elevations of plasma cortisol within one hour of the initiation of a 21-day exposure to 0.05 mg KCN/L. The elevated plasma cortisol levels were maintained through day 15 of the experiment, then began decreasing by the termination of the experiment on day 21 (Munoz et al., 1991). Reduced biomass (64-day exposure to 20 µg/L), reduced weight gain, liver damage, higher relative body-water concentration of cyanide (18-day exposure to 9.6 µg/L), reduction in fat content (18-day exposure to 19 µg/L), reduction in swimming ability (21-day exposure to 19 µg/L), and abnormal oocyte development (20-day exposure at 9.6 µg/L) were reported as sublethal effects on rainbow trout. In coho salmon, a reduction in swimming speed was observed after 2 hours exposed to 10 µg/L (EPA, 1985c).

The above evidence illustrates that the behavior of individual salmonids and the success of egg fertilization may be affected by cyanide concentrations below those established by both the acute and chronic cyanide criteria in the Idaho Water Quality Standards. However, it is important to note that the form of cyanide used in this experiment was KCN, which will dissociate into ionic cyanide. Idaho's cyanide criteria call for measurement of cyanide using weak acid dissociable (WAD) cyanide. WAD cyanide analysis reports ionic cyanide as well as cyanide weakly bound to metals such as copper, nickel, and zinc. As a rule, cyanide complexed with metal is less toxic than free cyanide, and in the ambient stream environment, cyanide is more likely to be complexed with metals. The cyanide measured by WAD analysis will encompass the most toxic forms along with those that are less toxic and therefore may overestimate toxicity by assuming all cyanide is in the ionic form. In this experiment, 0.001mg/L KCN would likely be greater than 0.001mg/L CN measured by WAD methods in Idaho. The effects on salmonid gametes in the Billard and Roubaud experiment occurred only when the gamete solutions were diluted first. This situation is not entirely realistic as salmonids spawn in redds where the gametes, especially the ova are highly concentrated. The gametes that are more dilute are less likely to successfully fertilize with or without the cyanide toxicity due to the nature of salmonid fertilization in the wild.

Based on the above information, EPA has determined that the acute and chronic cyanide criteria established by the Idaho Water Quality Standards are not likely to adversely affect thr general health and behavior of Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

Documented LC_{50} values for rainbow trout differ depending on the life stage and condition of the animal. For rainbow trout fry, the reported LC_{50} is 90 µg/L. Similarly, for juvenile rainbow trout calculated 24-hour LC_{50}s ranged from 90-98 µg/L dependent on the temperature during the exposure (EPA, 1985c). However, for a 96 hour exposure, the LC_{50}s for juvenile rainbow trout ranged from 43-52 µg/L (McGeachy and Leduc, 1988). Heming and Blumhagen (1989) calculated the 96-hour LC_{50} for unstressed rainbow trout exposed to SCN at 94 mg/L. However, when the same fish were forced to endure a single 30 second bout of strenuous exercise, the fish experienced "sudden death" at SCN concentrations above 25 mg/L.

Based on the available information, EPA has determined that the acute and chronic cyanide criteria established by the Idaho Water Quality Standards are not likely to adversely affect survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

Sublethal effects of cyanide, such as behavior of individuals and reduced fertilization of rainbow trout eggs may occur at concentrations below both the acute and chronic cyanide criteria established by the Idaho Water Quality Standards. As stated previously, the results of the fertilization study may be questioned due to the procedures used during the research. Also, effects on individual fish are not considered compelling evidence to prompt lowering of the criteria (Stephen, 1999). Therefore, EPA has determined that the approval of the **acute and chronic cyanide criteria** (22.0 µg/L=acute, 5.2 µg/L=chronic) established by the Idaho Water Quality Standards **are not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

The sensitivity of bird species to cyanide is related to diet. Birds that feed on flesh are more sensitive to cyanide toxicity than those that are herbivorous, with the exception of the mallard. Also, metallocyanide complexes may be dissociated by stomach acids, thus causing birds to exhibit a delayed response to cyanide poisoning (Eisler, 1991).

In mallards, mitochondrial respiratory control ratios decreased in heart, liver, and brain tissues 2 hours after exposure to drinking water contaminated with 20 ppm (mg/L) CN . A decrease was also observed in liver and brain ATP (Pritsos and Ma, 1997). Doses of 1.27 mg/kg resulted in a 33% reduction of survival in mallards, whereas 1.43-2.7 mg/kg were calculated as the LD_{50} values for mallards. In American Kestrel (*Falco sparverius*), LD_{50}s were calculated as 2.12-4.0 mg NaCn/kg (Eisler, 1991; Henny et al., 1994).

Bioconcentration factors (BCF) were unavailable for cyanide in fish. From the evidence available, it seems unlikely that cyanide will bioconcentrate in fish tissues, making it also unlikely that piscivorous birds will ingest cyanide through their diet. The other remaining cyanide exposure pathways for birds include ingestion of plants that naturally produce cyanide or ingestion of cyanide contaminated drinking water. The effects observed by Pritsos and Ma (1997) in mallards that had ingested water contaminated with 20 ppm (mg/L) CN were sublethal and occurred at concentrations well above the acute and chronic cyanide criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the approval of the **acute and chronic cyanide criteria** (22.0 µg/L=acute, 5.2 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

H. Endosulfan

The current Idaho Water Quality Standards establish acute and chronic criteria for both -endosulfan and -endosulfan, the two isomers that make up approximately 70% and 30%, respectively, of technical endosulfan. For both isomers, the acute criterion is specified as 0.22 µg/L and the chronic criterion is set as 0.056 µg/L. Endosulfan does not occur in background freshwater systems.

Endosulfan is a man-made, chlorinated cyclodiene insecticide that is not persistent in water (Sunderam et al., 1992). It is easily metabolized by many organisms to endosulfan sulfate, a compound that can be just as toxic as the parent isomers. Fish can further metabolize endosulfan sulfate to less toxic compounds such as endosulfan diol, endosulfan ether, and endosulfan hydroxyether (Nowak et al., 1995). Endosulfan is used as a wood preservative, but is primarily used on a wide variety of food crops, including tea, coffee, fruits and vegetables, as well as on rice, cereals, maize, sorghum or other grains. This pesticide is compatible with many other pesticides and may be found in formulations with dimethoate, malathion, methomyl, monocrotophos, prirmicarb, treazophos, fenoprop, parathion, petroleum oils, and oxine-copper (Oregon State University, 1996). The toxicity of individual endosulfan isomers is variable and little studied. We do know, however, that toxicity varies by type of isomer and the type of organism affected. Some evidence exists showing the differential ability of fishes to metabolize endosulfan. Striped mullet were found with endosulfan sulfate concentrated in their tissues. After removal of the fish to clean water, endosulfan sulfate was no longer detectable. In sheepshead minnow juveniles, the endosulfan in tissues existed mostly as the parent form. This indicates that some fish species or life stages may be able to metabolize endosulfan more easily than others (EPA, 1980e).

Seasonal variations in toxicity have also been observed in invertebrates. For two freshwater bivalve species, *Lamellidens corrianus* and *L. marginalis*, increased sensitivity to endosulfan was observed during summer months at concentrations ranging from 0.001-0.05 mg/L. Summer months were characterized by increased temperature, pH, and carbonate content. In summer testing, concentrations from 0.016-0.02 mg/L resulted in mortality of the bivalves after 60 hours exposure. In winter months, mortalities were not observed until exposure to concentrations of 0.028-0.05 mg/L lasted at least 60 hours (Mane and Muley, 1984). Temperature may affect endosulfan's toxicity by altering the activity and inducibility of enzymes. If studies do not take the effect of temperature on enzyme activity into account, the impact of endosulfan may be overestimated (Sunderam et al., 1992).

Endosulfan may also affect estrogenic processes particularly when endosulfan is combined with other pesticides. The potency of endosulfan alone is much lower than natural estradiol. However, in the environment, endosulfan tends to occur with other pesticides such as dieldrin and toxaphene (Arnold, 1996; Heufelder and Hofbrauer, 1996).

Endosulfan as specified by the Idaho Water Quality Standards is measured as total recoverable endosulfan. The accepted method of determining the total recoverable endosulfan is liquid-liquid extraction using gas chromatography. Endosulfan is a relatively stable compound as is its major degradation product, endosulfan sulfate. Due to the stability and toxicity of endosulfan and endosulfan sulfate, the presence of these chemicals in aquatic systems should be monitored closely (Franson et al., 1989).

1. Bioconcentration and Biomagnification

Both endosulfan and endosulfan sulfate are known to bioconcentrate (an increase in concentration of a substance in relation to the concentration in the ambient environment) in fish and aquatic invertebrates. Bioconcentration factors (BCF) differ between the and endosulfan isomers. For snails, BCFs range from 1,336-5,763 for -endosulfan and 8,174-39,457 for -endosulfan. Fish do not bioconcentrate endosulfan to the same high degree. BCF's for fish range from 30-304 for -endosulfan and 90-388 for -endosulfan (Callahan et al., 1979).

Biomagnification of endosulfan has not been well addressed in the literature. However, endosulfan is a highly lipid-soluble insecticide similar to DDT. It is therefore assumed to follow the biomagnification pattern of DDT, which, in birds, includes sequestration and elimination of the compound via eggs (Hudson et al., 1984).

2. Invertebrates

Sublethal effects

Information in the literature regarding the sublethal effects of endosulfan on freshwater invertebrates is lacking. At concentrations ranging from 0.18-1.8 mg/L endosulfan, Rambabu and Rao (1994) observed adverse effects on glucose, glycogen, lipid, and protein contents of the viscera, mantle, and foot in the freshwater snail, *Bellamya dissimilis*. Endosulfan was more potent than the other organophosphate pesticides tested, including Methyl Parathion, Quinalphos, and Nuvan. Results of the experiment also included the secretion of a mucus film covering the snail body wall. Anoxic stress and domination of anaerobic metabolism followed, possibly due to inefficient gas exchange through the mucus film. Also in *B. dissimilis*, atrophied and, in some cases, hypertrophied lumen of the digestive gland tubule, plus vacuolations were observed after a 96 hour exposure to 1.8 mg/L endosulfan. Disruption of the central connective tissue and columnar muscle fibers were also observed (Jonnalagadda and Rao, 1996). From this information, it appears that the concentration of endosulfan that affects freshwater snails is well above the endosulfan criteria established by the Idaho Water Quality Standards. Therefore, based on available evidence, EPA has determined that the acute and chronic endosulfan criteria established by the Idaho Water Quality Standards are not likely to adversely and sublethally affect the general health and behavior Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

Few studies detail the lethal effects of endosulfan on freshwater invertebrates. Mane and Muley (1984) exposed freshwater bivalves, *L. corrianus*, and *L. marginalis* to endosulfan for 96 hours. The LC_{50} values varied with the season during which the exposure occurred. For *L. corrianus*, the summer LC_{50} was 0.017 mg/L, while the monsoon and winter LC_{50}s were 0.04-0.044 mg/L. When *L. marginalis* was tested, the summer LC_{50} was 0.006 mg/L; however, the monsoon and winter LC_{50}s were much higher at 0.036-0.04 mg/L. Even though the toxicity of endosulfan does appear to increase with temperature, the concentration resulting in bivalve mortality are much greater than those specified by the acute and chronic criteria. For the freshwater snail, *B. dissimilis*, the calculated 96-hour LC_{50} value is 1.8 mg/L (Jonnalagadda and Rao, 1996). The 96-hour LC_{50} calculated for the Atlantic oyster, *Crassostrea virginica*, in 0 ppt salinity water is 65 µg/L (WHO, 1984). From the information above, it appears that the concentrations of endosulfan that result in lethal effects on freshwater invertebrates are well above the acute and chronic endosulfan criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the acute and chronic endosulfan criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

Based on the available information, EPA has determined that the approval of the **acute and chronic endosulfan criteria** (0.22 µg/L=acute, 0.056 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

Rainbow trout (*Oncorhynchus mykiss*), a fish species commonly used to determine the toxicity of aquatic pollutants, has been shown to be highly sensitive among fish species to the effects of endosulfan (EPA, 1980e).

Sublethal effects

Endosulfan in combination with disulfoton reduced the activity of specific cytosolic acetylcholinesterase and microsomal unspecific esterase in rainbow trout hepatocytes after prolonged (18 or 34 days) exposure to 50 ng/L endosulfan in combination with 0, 1, 5, or 10 µg/L disulfoton (Arnold et al., 1995). Acetylcholinesterase catalyzes the hydrolysis of the neurotransmitter, acetylcholine. Rainbow trout enzyme activity was altered after 14 days of exposure to 8.3 µg/L endosulfan. Jensen et al. (1991) concluded that this alteration in rainbow trout enzyme activity indicates that endosulfan may contribute to the environmental induction of enzymes used in the metabolism of xenobiotics in fish. This may also detrimentally affect physiological parameters involved in sexual maturation and reproduction in rainbow trout. Endosulfan has been shown to impair vitellogenin synthesis production in catfish (*Clarias batrachus*) exposed to endosulfan at concentrations of 1.5 µg/L for 16 days. The mechanism by which endosulfan inhibited vitellogenesis is hypothesized by the authors to be either interference with ovarian estrogen production or prevention of the action of estradiol on the liver (Chakravorty et al., 1992). Also in freshwater catfish (*C. batrachus*), exposure to 8 µg/L endosulfan for 96 hours resulted in a significant increase in thyroxine and decrease in triiodothyronine levels in serum and pharyngeal thyroid follicles concurrent with peroxidase activity. The intensity of the effects varied depending on the reproductive status of the fish (Sinha et al., 1991).

Calcium deposition in the scales of snakehead fish (*Channa punctatus*) was adversely affected by exposure to endosulfan at concentrations of 2.2 µg/L. The investigators determined that since the overall growth of fish directly depends on calcium metabolism, an alteration of the deposition of calcium in the hard parts of fish indicates an overall deterioration of the health of the fish (Johal and Dua, 1995). Blood dyscrasia (disease) was observed in the freshwater fish, *Barbus conchonius*, after 4 weeks exposure to 6.72 µg/L endosulfan. The blood disease effects were mostly abated after recovery in clean water, with the exception of thrombocytosis, or an increase in blood platelets (Gill et al., 1991). Nowak (1992) observed edema (swelling) with lifting and hyperplasia (an increase in the number of normal cells in tissues) of the lamellar epithelium in the gills of catfish (*Tandanus tandanus*) exposed to 1µg/L endosulfan for 24 hours. The intensity of the effects on the gills was correlated with the amount and isomer composition of endosulfan residues detected in the gills of the fish. In catfish exposed to 0.1 and 1 µg/L endosulfan, residues of 1-82 µg/kg were found in liver tissues, along with some structural changes, directly related to the amount of endosulfan to which the fish were exposed (Nowak, 1996). Researchers did not observe any negative effects associated with these structural changes. Results of other studies also indicate that these negative effects may abate in a manner similar to other cellular effects of endosulfan that have been shown to abate after exposure to endosulfan ceased (Gill et al., 1991).

In most studies referenced here, the concentrations of endosulfan shown to adversely affect fish are above the acute and chronic endosulfan standards 0.22 µ/L and 0.056 µ/L established by the Idaho Water Quality Standards. In those cases where effects were seen at concentrations below the criteria, confounding factors such as combination with other pesticides or the potential recovery from effects occurring below the acute criterion indicate that these studies do not provide compelling evidence for revision of the aquatic life criteria. Therefore, based on this information, EPA has determined that the acute and chronic endosulfan criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal Effects

Little information is available on the lethal effects of endosulfan on fish. For juvenile rainbow trout, calculated 96 hour LC_{50}s ranged from 0.17-2.43 µg/L in static tests and 0.17-0.86 µg/L in flow through experiments (EPA, 1980e; WHO, 1984; Sunderam et al., 1992). These toxicity values are above the endosulfan criteria established by the Idaho Water Quality Standards and most were considered by EPA for the previous evaluation of the aquatic life criteria. Therefore, based on the available information, EPA has determined, that the acute and chronic endosulfan criteria established by the Idaho Water Quality Standards are not likely to adversely affect the Snake River sockeye and chinook salmon, survival of Snake River steelhead, and bull trout.

Summary

Sublethal effects of endosulfan, such as reduced enzyme activity, cellular structural changes and accumulation of endosulfan, may occur at concentrations below those allowed by the acute and chronic endosulfan criteria. However, (in those studies referenced), enzyme disruption occurred only where endosulfan was combined with another pesticide. Also, fish recovered from toxic effects after removal from contaminated water. These facts cast doubt on the ability of endosulfan alone to cause sublethal toxic effects at concentrations at or below the criteria. Therefore, EPA has determined that the approval of the **acute and chronic endosulfan criteria** (0.22 µg/L=acute, 0.056 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

Endosulfan is suspected to follow the pattern of DDT in birds: sequestration and elimination in eggs. Hudson et al. (1984) found 1 year old pheasants to be less sensitive to endosulfan than mallards and other pheasants that were not laying or had not recently lain eggs. The differences in sensitivity may also be due to maturity of the birds and their ability to detoxify the chemical. Signs of endosulfan poisoning include: ataxia, slowness, high carriage, jerkiness, wings crossed high over back, dyspnea (labored respiration), tremors, wing shivers, and falling. These effects were observed in as little as 10 minutes in mallards and 1 hour in pheasants. Mortalities were observed 0.5-2 hours after exposure in mallards and 2-4 hours after exposure in pheasants (Hudson et al., 1984).

Endosulfan also has been shown to affect survival and reproductive success in mourning doves, (*Zenaida macroura*), and American robins, (*Turdus migratorius*) (Fleutsch and Sparling, 1994). Acute oral LD_{50}s found for ducks exposed to endosulfan for 36 hours, 7 days, 30 days, and 6 months were 27.8 mg/kg, 6.47 mg/kg, 7.89 mg/kg, and 34.4 mg/kg, respectively (WHO, 1984). In pheasants, LD_{50} estimates ranged from 80-190 mg/kg for 3-4 month old birds and were estimated to be greater than 320 mg/kg for 1 year old egg laying hens. Unlike pheasants, mallards did not show a difference in sensitivity to endosulfan based on age and reproductive status. The LD_{50} estimated for 3 month old birds was 33 mg/kg and for that 1 year olds ranged between 31.2-45 mg/kg, with 1 year old females being more sensitive than males (Hudson et al., 1984).

Table 250.07.a.8 illustrates a conversion of the maximum allowable water criteria under the Idaho Water Quality Standards to dietary concentrations for piscivorous birds using the maximum bioconcentration factor (BCF) for fish provided by Callahan et al. (1979). This allows an interpretation of the dietary concentrations referenced above in the context of the current Idaho Water Quality Standards. The dietary concentrations shown to affect bird species are well above the maximum dietary concentration that would result from the endosulfan concentrations allowed by the acute and chronic criteria established by the Water Quality Standards. Therefore, from the information available with regard to endosulfan toxicity to avian species, EPA has determined that the approval of the **acute and chronic endosulfan criteria** (0.22 µg/L=acute, 0.056 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

Table 250.07.a.9. Dietary Concentrations allowed by Idaho Water Quality Standards				
Dietary Concentrations		Endosulfan Criteria (Total Recoverable)		Maximum BCF for fish, as prey for birds (Callahan et al., 1979)
Acute	Chronic	Acute	Chronic	
for both and isomers separately	for both and isomers separately	for both and isomers separately	for both and isomers separately	
0.08536 mg/kg	0.021728 mg/kg	0.00022 mg/L	0.000056 mg/L	388

I. Lead

The current Idaho Water Quality Standards establish hardness dependent lead criteria. At a water hardness of 100 mg/L CaCO3, the dissolved lead (Pb) criteria are 65 µg/L and 2.5 µg/L for short-term and long-term exposures, respectively (See Table 250.07.a.10). Corresponding total recoverable lead criteria are 62 µg/L and 3.2 µg/L for short-term and long-term exposures, respectively. Table 250.07.a.11 lists criteria calculated for the hardness values used in the studies referenced in this report.

Table 250.07.a.10. Idaho Lead Water Quality Criteria				
Hardness	Acute Criteria		Chronic Criteria	
	Total	Dissolved	Total	Dissolved
100 mg/L $CaCO_3$	65 µg/L	49.04 µg/L	3.2 µg/L	2.53 µg/L

Table 250.07.a.11. Idaho Water Quality Criteria for Lead Calculated for Referenced Hardness Values and Total Recoverable Analysis		
Hardness	Acute Criteria	Chronic Criteria
8 mg/L $CaCO_3$	3.28 µg/L	0.13 µg/L
10-20 mg/L $CaCO_3$	4.35-10.52 µg/L	0.17-0.40 µg/L
26-31 mg/L $CaCO_3$	14.70-18.38 µg/L	0.56-0.70 µg/L
28 mg/L $CaCO_3$	16.15 µg/L	0.62 µg/L
35 mg/L $CaCO_3$	21.46 µg/L	0.82 µg/L

Table 250.07.a.11. Idaho Water Quality Criteria for Lead Calculated for Referenced Hardness Values and Total Recoverable Analysis		
42 mg/L $CaCO_3$	27.06 µg/L	1.03 µg/L
46 mg/L $CaCO_3$	30.38 µg/L	1.15 µg/L
61 mg/L $CaCO_3$	43.52 µg/L	1.65 µg/L
100-106 mg/L $CaCO_3$	81.65-87.93 µg/L	3.08-3.32 µg/L
101 mg/L $CaCO_3$	82.69 µg/L	3.12 µg/L
128 mg/L $CaCO_3$	111.79 µg/L	4.21 µg/L
135 mg/L $CaCO_3$	119.63 µg/L	4.51 µg/L
139 mg/L $CaCO_3$	124.16 µg/L	4.67 µg/L
290 mg/L $CaCO_3$	316.64 µg/L	11.86 µg/L
353 mg/L $CaCO_3$	406.67 µg/L	15.21 µg/L

Lead is a naturally occurring, ubiquitous compound that can be found in rocks, soils, water, plants, animals, and air. Concentrations of lead associated with background freshwater systems are estimated to be <3.0 µg/L (Moore and Ramamoorthy, 1984). It is soluble in water and its bioavailability increases in environments with low pH, low organic content, and low metal salt content (Eisler, 1988b). Lead is most often precipitated to sediments in aqueous environments. The toxicity of lead varies with water hardness. As hardness increases, lead precipitates, and becomes less bioavailable to aquatic organisms. Adsorption of lead by aquatic animals is affected by the age, gender and diet of the organism, as well as the particle size, chemical species and presence of other compounds in the water (Eisler, 1988b; Hamir et al., 1982). Aquatic organisms are sensitive to lead are affected more strongly by dissolved rather than total lead. Likewise, the toxicity of lead is increased when it forms organolead compounds and when environmental conditions consist of high temperature and low pH. Animals are also more sensitive at younger life stages and when exposure durations are greater.

1. Bioconcentration and Biomagnification

Lead has been shown to bioconcentrate (an increase in concentration in relation to the ambient concentration) in aquatic species. Invertebrates tend to have higher bioconcentration factors (BCF) than vertebrates. For example, the BCF for the freshwater snail, *Lymnaea palustris*, is 1,700 and the BCF for the blue mussel, *Mytilus edulis*, is 2,570. In the freshwater snail, *Physa integra*, tissue concentration changes were correlated with changes in dissolved lead in the water column, but not with changes in the amount of lead found in substrate. Similarly, *Campeloma decisum* (sub-tropical freshwater snail) had lower tissue concentrations than the substrate even though the organism was associated closely with contaminated sediments. Lead was found to accumulate in the ganglia of freshwater snails (*Lymnaea stagnalis*). In vertebrates, such as brook trout embryos, the BCF is 42 (Eisler, 1988b). Inorganic lead is poorly accumulated in fish. Organic lead compounds such as tetraalkyllead are more toxic than smaller compounds such as trialkyllead. This may be due to the rapid accumulation of tetraalkyllead by fish (Hodson et al., 1984). BCFs decrease as waterborne lead concentrations increase, thus suggesting accelerated depuration or saturation of uptake mechanisms (Hodson et al., 1984). Exposures of rainbow trout to 3.5-51 µg/L (hardness = 135) tetramethyllead from 7 days to two weeks resulted in rapid accumulation of lead. However, once the fish were removed to clean water, lead was initially removed rapidly from organs followed by a slower release until base levels were reached. An increase in dietary calcium of 0-8.4 mg/kg (hardness=8 mg/L) reduced the uptake of waterborne lead in coho salmon, possibly due to interactions with gill membrane permeability (Hodson et al., 1984).

In vertebrates, lead concentrations tend to increase with age and localize in hard tissues such as bone or teeth. Lead residues have been shown to be greater in older birds, sexually mature females, and in birds that have ingested lead shot pellets. While lead has been shown to concentrate in aquatic species, there is little evidence for biomagnification (a progressive increase in concentration from one trophic level to the next higher level; Eisler, 1988b).

2. Invertebrates

Sublethal effects

The toxicity of alkyllead and alkyllead salts was found to be sex-dependent in salt marsh snails (*Littorina irrorata*), with males accumulating higher residues than females (Krishnan et al., 1988). At lead concentrations of 1, 2, 5, 10, and 20 mg/L, lead caused (no hardness given, interpret with caution) morphological and biochemical dysfunction (Bolognani-Fantin et al., 1985) and at concentrations of 0.278-278 mg/L (hardness not given, $[Ca^{2+}]$=80 mg/L, $[Mg^{2+}]$=3.16 mg/L), lead modified the excitability and chemosensitivity of *L. stagnalis* neurons in vitro (Rozsa and Salanki, 1990). Stimulation of hyperactivity has also been caused in *L. stagnalis* by lead at concentrations of 50 and 200 µg/L for 19 hours. Snails exposed to 200 µg/L maintained their hyperactive state for 50 days, however movement was significantly reduced after 1 year (hardness not given, $[Ca^{2+}]$=92 mg/L, $[Mg^{2+}]$=32 mg/L). The investigators concluded that these results suggest acclimation to lead contamination. Effects resulting from exposure to 50 µg/L did not persist beyond the exposure (Truscott et al., 1995). Neuronal cytolosis (rupture of nerve cells) occurred in the freshwater snail, *Viviparus ater*, when exposed to 1000 mg/L (no hardness given) for 7 days (Bolognani-Fantin et al., 1985).

Stimulation of hyperactivity in snails occurred at concentrations less than the acute criterion established by the Idaho Water Quality Standards. However, long-term exposure to these high concentrations were required before corresponding observable adverse effects were seen. Therefore, EPA has determined that the current acute and chronic lead criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

At concentrations as low as 19 µg/L (hardness=139) survival of *L. palustris* was significantly reduced, while growth and reproduction were unaffected. The No Observed Effect level (NOEL) determined from this chronic (120 days) experiment was 12 µg/L (EPA, 1985d). For the aquatic snail species, *L. emarginata*, and *Goniobasis livescens*, 48-hour LC_{50}s were calculated at 14,000 and 71,000 µg/L, respectively (hardness not given). After 28 days exposed to 265 µg Pb/L (hardness=46), the survival of *Physa integra* was not affected (EPA, 1985d). A 96-hour LC_{50} of 117 mg/L (no hardness given) was determined for *Viviparus ater* (Bolognani-Fantin et al., 1985). For *Aplexa hypnorum*, an LC_{50} of 1,340 µg/L (hardness=61) was determined (EPA, 1985d).

Information regarding the lethal effects of lead show that lethal concentrations of lead are much greater than those allowed by the Idaho Water Quality Standards' lead criteria. Therefore, EPA has determined, based on the information available, that the acute and chronic lead criteria established by the Idaho Water Quality Standards are not likely to affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

There is a great deal of uncertainty regarding sublethal effects. The research regarding sublethal effects is sparse and often does not reference the water hardness in the experiment, thus making results difficult to interpret. However, based on the information available, EPA has determined that the approval of the **acute and chronic lead criteria** (65 µg/L=acute, 2.5 µg/L=chronic, hardness of 100 mg/L CaCO3) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

Sublethal effects

Adult trout exposed to lead as part of their diet (0.86-1.77 µg/g) for 21 days experienced increased scale loss and accumulation of lead in their guts. When exposed to lead for the same length of time through the water column (4.3-6.4 µg/L, hardness=100-106), trout experienced scale loss, reduced survival, and accumulation in gill and kidney tissues. A combination of dietary and water-borne lead exposure at the same concentrations resulted in lipid peroxidation in kidneys of adults and a decrease in the whole body potassium of juveniles (Farag et al., 1994). Other documented sublethal responses include hematological, neurological, teratogenic, growth, and histological effects at lead concentrations of 8-119 µg/L and >1000 µg/L (hardness=42-353) during exposures from 3-16 weeks (Hodson et al., 1984).

Concentrations of lead >10 µg/L (hardness=135) caused long-term effects such as: spinal curvature; anemia; caudal chromatophore degeneration (black tail); caudal fin degeneration; destruction of spinal neurons; ALAD inhibition in blood cells, spleen, liver, and renal tissues; reduced swimming ability; destruction of respiratory epithelium; elevated lead in blood, bone and kidney; muscular atrophy and paralysis; inhibition of growth; retardation of maturity; changes in blood chemistry; testicular and ovarian histopathology; and even death (EPA, 1985d). The effects of lead increase under rapid growth conditions as illustrated by the increase of the rate of intoxication by lead increased with growth rate, but not fish size (Hodson et al., 1982). In sexually maturing male rainbow trout exposed to 10 µg/L (hardness=128) for 12 days during spermatogenesis, spermatogonial cysts increased, spermatocytes declined, and the sensitivity of the reproductive cycle was expressed as the transformation of spermatogonia to spermatocytes decreased (Ruby et al., 1993a). In whitefish (*Coregonus* sp.) from contaminated lakes (0.5-4.5 µg Pb/L, hardness=10-20) -aminolevulinic acid (ALAD) activity was inhibited up to 88% when compared to fish from uncontaminated lakes. Inhibition of ALAD activity leads to problems with hemoglobin synthesis that can result in anemia. Higher blood glucose levels and lower plasma sodium content were also found in fish taken from lead contaminated lakes (Haux et al., 1986).

Spinal deformities in rainbow trout resulted from exposure to lead concentrations of 18.9 and 101.8 µg/L (hardness=28 and 35, respectively). In juvenile rainbow trout, ALAD activity was inhibited. Red blood cells and blood iron content were also affected after 28 days exposed to lead levels of 13 µg/L (hardness=135). At 120 µg Pb/L (hardness=135) for 32 weeks, 30% of juvenile rainbow trout exposed had black tails caused by degeneration of caudal chromophores (EPA, 1985d).

Based on the information presented on this report, EPA has determined that the acute and chronic lead criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

At lethal concentrations (543 µg/L; H=135 mg/L), lead can cause increased mucus formation in rainbow trout where the excess mucus coagulates over the fish's entire body, most prominently the gills. The mucus interferes with respiratory function and results in the death of the fish by anoxia (Hodson et al., 1982).

Many studies have determined LC_{50} values for various life stages of rainbow trout. The 32-week LC_{50} value for embryo/larval stages was 220 µg/L (hardness=101). Two month old fry had an LC_{50} of 8,000 µg/L (hardness=82-132 mg/L). For juvenile rainbow trout, the 21-day LC_{50} value was calculated as 2,400 µg/L (hardness=135). Finally, in adults, LC_{50} values ranged from 1,170 µg/L to 471,000 µg/L to 542,000 µg/L depending upon the hardness values: 28, 353, 290, respectively (EPA, 1985d).

All of the above studies found that survival of rainbow trout was not affected at levels allowable under the lead criteria established by the Idaho Water Quality Standards. Therefore, from the information available, EPA has determined that the acute and chronic lead criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

From the available information, EPA has determined that the approval of the **acute and chronic lead criteria** (65 µg/L=acute, 2.5 µg/L=chronic, hardness of 100 mg/L CaCO3) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

The age and gender of birds contribute significantly to differences in lead accumulation (Gochfield et al., 1996). Birds can rid their bodies of heavy metals, such as lead, through both excretion and deposition in feathers. Females may also eliminate metals through the contents and shells of eggs (Burger, 1994a).

The effects of lead exposure in birds are well documented due largely to the fact that birds eat lead shot pellets and suffer the toxic effects of lead poisoning from that dietary exposure. The pellets may be retained in the gizzard for weeks before they are reduced both chemically and mechanically into soluble toxic salts. These salts can cause characteristic signs of lead poisoning such as lethargy and emaciation (Eisler, 1988b). Lead poisoning through the ingestion of lead shot embedded in prey has been a significant mortality factor for many species including bald eagles (Pain and Miard-Triquet, 1993). Secondary poisoning has been documented in more than 5 raptor species that eat prey containing lead shot (Pattee and Hennes, 1983). Lead in the form of shotgun pellets and fishing sinkers accounted for approximately 20% of known mortality of trumpeter swans in Idaho, Montana, and Wyoming from 1975-1989 (Blus et al., 1989). Trust et al. (1990) found that ingested lead affected antibody production in mallards (*Anas platyrhynchos*). Similar effects were seen in wild mallards (Trust et al., 1990) and birds of prey (Reiser and Temple, 1981).

Via other exposure routes, lead can still affect the health and survival of bird species. Adult and nestling ospreys (*Pandon haliaetus*) living along the Coeur d'Alene River (contaminated from mine activities in the area; sediment concentrations as large as 4,600 mg/kg) had higher blood lead levels than ospreys residing at Flathead Lake (a reference area). Inhibition of ALAD activity and elevation of protoporphyrin correlated with high lead levels in blood. No mortality, behavioral abnormalities or reduced productivity was observed in the affected birds (Henny et al., 1991).

Burger and Gochfield (1994) injected herring gull chicks with lead (100 mg/kg) then observed the chicks in their natural environment. The treated chicks experienced lower survival and were less healthy as measured by observing the chicks begging and walking and by recording the number of times the chicks stumbled while walking. In 1996, Burger and Gochfield again looked at herring gull chicks injected with lead. The investigators observed the compensatory measures displayed by the parents of lead contaminated chicks. In the chicks, significant lead-induced differences existing in righting responses, locomotion, thermoregulation, and begging and feeding behaviors. The treated chicks were less able to compete with their siblings for food which resulted in significant decreases in body weight. In nests where differences in body weight of chicks was large, the parents divided feeding responsibilities thus ensuring that the treated chicks were fed. This extra parental care resulted in increased survival for lead-injected chicks and a decrease in the differences in body weight between untreated and treated chicks.

In laboratory studies, American kestrels (*Falco sparverius*) were not affected by exposures to lead at 10 and 50 ppm (mg/kg) over 5 months (Franson et al., 1983). Kestrels were also observed to have elevated tissue levels of lead after exposure to dietary lead at concentrations of 16-87 mg/kg for 60 days (Stendell, 1980), 50 mg/kg for 5 months (Franson et al. 1983), 10 and 50 mg/kg for 6 months (Pattee, 1984), 25, 125, and 625 mg/kg for 10 days (Hoffman et al. 1985a, 1985b), and 448 mg/kg for 60 days (Custer et al., 1984). Effects of these concentrations included reduced blood ALAD activity following exposure to 50 mg/kg for 5 months (Franson et al., 1983) and 25, 125, and 625 mg/kg for 10 days (Hoffman et al., 1985a, 1985b). Reduced growth, kidney and liver weight and abnormal skeletal development were also observed at 125 and 625 mg/kg for 10 days (Hoffman et al. 1985a, 1985b).

A large pool of information has been obtained from examination of dead birds thought to have been exposed to lead through various sources. Lead residues in raptors thought to have died of lead poisoning ranged from 17-38 ppm (Stendell, 1980). Necropsies of urban peregrine falcons revealed that the birds died of *Pseudomonas* infection of the pharynx perhaps indirectly caused by lead poisoning. Concentration of lead in the liver was 0.74 ppm and 1.40 ppm in the kidney. It was hypothesized that the exposure to lead may have resulted from ingestion of contaminated rock doves (DeMent et al., 1986). In eagles found dead, there was also evidence for lead's contribution to the bird's mortality. A golden eagle was found to contain 6.3 ppm in its liver. Lead may have contributed to the birds death by necrotic colitis. Three dead eagles were also found to have elevated blood lead levels and exhibited lead poisoning symptoms (Craig, 1990).

Table 250.07.a.12 illustrates a conversion of the maximum allowable water criteria under the Idaho Water Quality Standards to dietary concentrations for piscivorous animals using the maximum bioconcentration factor (BCF) for fish provided by Eisler (1988b). This allows an interpretation of the dietary concentrations referenced above in the context of the current Idaho Water Quality Standards. The maximum dietary concentrations for the acute and chronic criteria established by the Idaho Water Quality Standards are in most cases well below those seen to adversely affect bird species. In cases where the effective concentrations are near those allowed by the criteria, the effect observed was elevated tissue concentration with no associated adverse physiological effect. Therefore, EPA has determined that the approval of the **acute and chronic lead criteria** (65 µg/L=acute, 2.5 µg/L=chronic, hardness of 100 mg/L CaCO3) established by the Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

Table 250.07.a.12. Dietary Concentrations allowed by Idaho Water Quality Standards

Dietary Concentrations		Lead Criteria (Total Recoverable)		Maximum BCF for fish, as prey for birds and mammals (Eisler, 1988b)	Hardness
Acute	Chronic	Acute	Chronic		
11.72 mg Pb/kg	0.45 mg Pb/kg	0.01615 mg Pb/L	0.00062 mg Pb/L	726	28 mg/L $CaCO_3$

Table 250.07.a.12. Dietary Concentrations allowed by Idaho Water Quality Standards					
14.45 mg Pb/kg	0.57 mg Pb/kg	0.01991 mg Pb/L	0.00078 mg Pb/L	726	33 mg/L $CaCO_3$
21.45 mg Pb/kg	0.83 mg Pb/kg	0.02954 mg Pb/L	0.00115 mg Pb/L	726	45 mg/L $CaCO_3$

J. Mercury

The current Idaho Water Quality Standards establish an acute criterion for dissolved mercury as 2.1 µg/L. The chronic criterion established for dissolved mercury is 0.012 µg/L. The total recoverable acute and chronic total mercury criteria are 2.4 µg/L and 0.012 µg/L, respectively.

Mercury is cycled through the environment through an atmospheric-oceanic exchange. This cycling is facilitated by the volatility of the metallic form of mercury. Natural bacterial transformation of mercury results in stable, lipid soluble alkylated compounds such as methylmercury (Beijer and Jennelov, 1979). Methylmercury is highly toxic to mammals and can interfere with thiol metabolism resulting in mitotic disturbances. This compound can also irreversibly destroy the neurons of the central nervous system (Clarkson et al., 1984). While mercury does occur naturally in small amounts in aquatic environments, the cycling of mercury prolongs the influence of man-made mercury compounds (Hudson et al., 1995). In sediments, mercury is usually found in its inorganic forms, but aquatic environments are a major source of methylmercury (EPA, 1985e). In background freshwater systems, mercury occurs naturally at concentrations of 0.02-0.1 µg/L (Moore and Ramamoorthy, 1984).

1. Bioconcentration and Biomagnification

Mercury has been shown to bioconcentrate (an increase in the concentration of a substance in relation to the concentration in the ambient environment) in a variety of aquatic organisms. Fish have been shown to concentrate mercury as methylmercury even when they are exposed to inorganic mercury. Aquatic predators face the greatest danger of bioconcentrating mercury, and thus their tissue concentrations best reflect the amount of mercury available to aquatic organisms in the environment. Fish, such as rainbow trout, have been found to accumulate mercury in the form of methylmercury at aquatic concentrations as low as 1.38 ng/L (Ponce and Bloom, 1991). Temperature has been shown to affect the magnitude of bioconcentration factors (BCF) in aquatic snails. In the freshwater gastropod, *Viviparus georgianus*, BCFs were observed to increase with temperature in snails from three different age classes. Similar effects were also observed for the medium sized pelecypods, *Elliptia complanata*. For animals between 74-86 mm in length, BCF increased with increasing temperature (Tessier et al., 1994).

Some evidence supports the biomagnification (a progressive increase in concentration from one trophic level to the next higher level) of mercury in aquatic food chains. In a comparison of benthic feeding fish and fish that feed on plankton, invertebrates and vertebrates, the greatest mercury concentrations were found in piscivorous fishes. The authors of this study concluded that mercury content in fish increased with higher trophic levels (Wren and MacCrimmon, 1986).

2. Invertebrates

Sublethal effects

Little information is available regarding the sublethal effects of mercury on freshwater invertebrates. In marine molluscs, specifically blue mussels (*Mytilus edulis*), 24-hour exposures to 32 and 400 µg/L resulted in abnormal development and reduced feeding rate, respectively. These mercury concentrations are much higher than the criteria levels established by the Idaho Water Quality Standards. Therefore, from the information available, EPA has determined that the acute and chronic criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

Aquatic LC_{50} values reported by EPA (1985g) for mercury range from 1.5-2,100 µg/L for aquatic invertebrates. Daphnia were more sensitive to the lethal effects of mercury. Snail embryos were the most resistant to the lethal effects of mercury of the organisms tested. LC_{50}s for two snail species, *Amnicola* sp. and *Aplexa hypnorum*, ranged from 80-2,100 µg/L (EPA, 1985e). Thain (1984) estimated 96-hour LC_{50}s of 60 µg/L for slipper limpet (*Crepidula fornicata*) larvae of 60 µg/L and 330 µg/L for adult slipper limpets. In most cases, especially for the aquatic snails, the LC_{50}s were well above the mercury criteria established by the Idaho Water Quality Standards. Thus, EPA has determined that the acute and chronic mercury criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

Little information is available detailing the effects of mercury on freshwater snails. However, the information available indicates that snails are sensitive to mercury concentrations much higher than those allowed by the Idaho Water Quality Standards and are among the more tolerant of aquatic invertebrates to mercury contamination. Therefore, EPA has determined that the approval of the **acute and chronic mercury criteria** (for dissolved mercury, 2.1 µg/L=acute, 0.012 µg/L=chronic) established by the Idaho Water Quality Standards is **not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

The effects of exposure to mercury have been studied extensively in fish. The uptake of mercury is proportional to the concentration of mercury in water. However, the uptake of methylmercury in fish increases with increased water temperature, exposure concentration, size and age of the fish, breeding status, and food ingestion rate. Decreases in pH have also been correlated with increasing methylmercury uptake (Wren and MacCrimmon, 1986; Ponce and Bloom, 1991).

Sublethal effects

Long term dietary exposure to mercury has been shown to cause instability, inability to feed and diminished responsiveness. The central nervous system is the site of the most extensive damage due to mercury exposure. Dietary exposures of 16-48 µg/g over a period of 84-270 days adversely affected growth, skin color, weight, and behavior in rainbow trout. As little as 7.9 µg/g affected the survival and behavior of walleye. Long-term exposures to waterborne concentrations of mercury ranging from 0.1-0.2 µg/L also affected behavior, reproduction and survival of fish, specifically fathead minnows (Weiner and Spry, 1996).

EPA has determined that the acute and chronic mercury criteria established by the Idaho Water Quality Standards is not likely to adversely affect the general health and behavior of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

EPA (1985g) reported LC_{50} values for fish exposed to inorganic mercury which ranged between 150-420 µg/L. For organic mercury, LC_{50}s range from 24-84 µg/L. In both cases, the LC_{50}s reported by EPA (1985g) were determined under flow-through conditions. In a study of the chronic toxicity of mercury chloride ($HgCl_2$) to rainbow trout, Niimi and Kissoon (1994) exposed subadults to 64 µg/L $HgCl_2$ until the fish died. The average time to death was 58 days at this concentration. At 426 µg/L, the mean time to death was 1 day. Niimi and Kissoon (1994) also conducted a similar experiment using methyl mercury chloride exposures. The investigators found that fish lived more than 100 days when exposed to 4 µg/L, but lived an average of only 2 days when exposed to 34 µg/L methylmercury chloride. The toxicity of methylmercury was also examined by Devlin and Mottet (1992). Coho salmon embryos were exposed to methylmercury at concentrations of 6, 13, 29, 62, and 139 µg/L methylmercury at 10°C for 48 days. The resulting LC_{50} values ranged from 54-71 µg/L.

All of the mercury concentrations found to affect the survival of fish are well above the mercury criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the acute mercury criterion established by the Idaho Water Quality Standards is not likely to adversely affect the survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

From the information available regarding the effects of mercury exposure on both the health and survival of fish species, EPA has determined that the approval of the **acute and chronic mercury criteria** (2.1 µg/L=acute, 0.012 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

Many studies have been conducted to determine the effects of mercury on avian species. These studies fall into four categories of bioaccumulation, deposition in feathers, reproductive effects, and mortality.

Sublethal Effects

In studies to determine the amount of mercury found in the blood of bald eagles exposed to mercury, Wood et al. (1996) found that even captive animals intended for use as baseline information had accumulated an average of 0.23 ppm mercury in their blood. In a comparison of eagles living on the shores of Lake Superior to those inhabiting the inland areas of Wisconsin, shoreline eagles were found to accumulate higher mercury concentrations than inland eagles, a trend that continued from 1983-1988 (Kozie and Anderson, 1991).

Bald eagles and other birds also accumulate mercury within their eggs. An analysis of the contents of 19 eggs taken from the Columbia estuary between 1985-1987 found an average concentration of 0.20 ppm mercury with a range of 0.13-0.36 ppm (Anthony et al., 1993). In a study of birds inhabiting the Agassiz National Wildlife Refuge in Minnesota, the concentration of mercury in eggs was found to increase with increasing trophic levels of the species examined (Burger and Gochfeld, 1996). In herring gulls, chicks were found to accumulate higher mercury concentrations than eggs (Becker and Sperveslage, 1989).

The analysis of feathers to determine mercury exposure is well established (Burger et al., 1993; Burger, 1994b; and Monteiro and Furness, 1997). Seabirds have been shown to excrete dietary methylmercury into plumage during feather growth (Lewis and Furness, 1991) and the analysis of mercury in old feather specimens reveals past dietary exposures (Thompson et al., 1992). However, the deposition of mercury into feathers can vary with bird species. Feathers in Agassiz gulls and geese contained lower concentrations of mercury than the median feather concentration (2,100 ppb) reported in over 180 studies on feathers (Burger, 1994b).

Dietary concentrations (1-2 µg/g) of methylmercury that produced significant reproductive effects in adult birds were found to be approximately a fifth of those seen to produce observable neurological effects. Mercury concentrations of 1-2 µg/g have been observed to impact loon reproduction (Scheuhammer, 1995). Mallards exposed to mercury over 3 generations experienced adverse reproductive effects at concentrations as low as 0.078 mg/kg/day (Eisler, 1987).

Analysis of mercury in eggs has been used to determine the amount of mercury that has been passed on from the female of a species to its young plus the amount of mercury the young are exposed to at birth. Mercury concentrations of 2-5 µg/g reduced reproductive success in ring doves (*Streptopelia resoria*), mallard ducks (*Anas platyrhynchos*), and pheasants (*Phasianus colchicus*; Scheuhammer, 1987). Eggs of bald eagles that contained 0.5 µg/g or more mercury resulted in adverse effects on reproduction (Wiemeyer et al., 1993). In white leghorn chickens, unbounded LOEL (Lowest observable effect level) levels for reproductive effects was estimated to be 10 ppm. An unbounded LOEL for ring-necked pheasants was estimated to be 4.2 ppm

(EPA, 1985e).

Lethal Effects

The LOEL and NOEL (no-observed-effect level) for mortality of chickens consuming methylmercury was 0.86 mg/kg/day and 0.57 mg/kg/day, respectively. The unbounded LOEL for growth in chicks was estimated to be 0.29 mg/kg/day. In ring-necked pheasants, the LOEL and NOEL for mortality were 12.5 ppm and 4.2 ppm, respectively (EPA, 1985e). For mallards, the LOEL and NOEL resulting in mortality and neurological impairment were 0.18 mg/kg/day and 0.030 mg/kg/day after 1.5 years of exposure to mercury (Eisler, 1987). Hudson et al. (1984) also described LD_{50} values for methyl and ethylmercury exposures. The LD_{50} for methylmercury was estimated to be between 2.2-2.4 mg/kg, while the LD_{50} for ethyl mercury was much higher at 76 mg/kg.

Summary

Table 250.07.a.13 illustrates a conversion of the maximum allowable water criteria under the Idaho Water Quality Standards to dietary concentrations for piscivorous birds using the maximum bioconcentration factor (BCF) for fish provided by EPA (1993). This allows an interpretation of the dietary concentrations referenced above in the context of the current Idaho Water Quality Standards. A dietary concentration of 100 µg Hg/kg is deemed safe for chickens (Eisler, 1987). At the maximum allowable water criteria, birds would consume a dietary mercury concentration greater than deemed safe by Eisler (1987). Additionally, many of the adverse effects listed above occur at concentrations less than those that would occur at the chronic criterion levels. However, accumulation of metals from food generally occurs over a longer period of time that for which the acute criterion is applied. For this reason, the accumulation potential was evaluated based on exposure to chronic criteria. Therefore, from the information available with regard to mercury toxicity to avian species, EPA has determined that the approval of the **chronic mercury criterion** (0.012 µg/L=chronic) established by the Idaho Water Quality Standards **may be likely to adversely affect the peregrine falcon, bald eagle, and whooping crane**. However, EPA's approval of the **acute mercury criterion** (2.1 µg/L=acute), established by the Idaho Water Quality Standards **is not likely to adversely affect the peregrine falcon, bald eagle, and whooping crane.** EPA has begun to prepare a schedule for updating aquatic life criteria for mercury. The updated criteria will consider recent data as well as methodologies revised with regard to bioavailability. The findings of this evaluation will be provided to the State of Idaho and, if necessary, EPA will recommend that the State make appropriate changes to their criteria.

Table 250.07.a.13. Dietary Concentrations allowed by Idaho Water Quality Standards

Dietary Concentrations		Mercury Criteria (Total Recoverable)		Maximum BCF for fish, as prey for birds and mammals (Eisler, 1987)
Acute	Chronic	Acute	Chronic	
66.96 mg Hg/kg	0.03348 mg Hg/kg	0.0024 mg Hg/L	0.0000012 mg Hg/L	27,900

5. Proposed revisions to the mercury criteria and other actions for the protection of threatened and endangered species

A. EPA will revise its recommended 304(a) human health criteria for mercury by January 2002. These criteria should be sufficient to protect federally listed aquatic and aquatic-dependent wildlife species. The revised criteria will be derived using a new national Bioaccumulation Factor (BAF) derivation methodology and more recent bioaccumulation data. Although the revised criteria can not be precisely predicted it is not unreasonable to expect that the revised criterion will be in the range of 1-5 ng/L. EPA will work in close cooperation with the Services to evaluate the degree of protection afforded to federally listed species by the revised criteria. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' Opinion, to ensure that the revised criteria will adequately protect federally listed species. If the revised criteria are less stringent than those proposed by the Services in the Opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criteria by the time of the proposal to allow the Services to complete a biological opinion on the proposed mercury criteria before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. If indicated by the results of this revision, EPA will collaborate with Idaho to propose revised criteria for mercury by January 2003.

B. EPA will utilize existing information to identify water bodies impaired by mercury in the State of Idaho. Impaired is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect federally listed species are not met. Pursuant to Section 303(d) of the CWA, EPA will work, in cooperation with the Services, and the State of Idaho to promote and develop strategies to identify sources of mercury contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce mercury loading into their habitat. (See also "Other Actions B." below.)

C. EPA promulgated a new more sensitive analytical method for measuring mercury (see 40 CFR Part 136).

Other Actions:

A. EPA will initiate a process to develop a national methodology to derive site-specific criteria to protect federally listed threatened and endangered species in accordance with the draft MOA between EPA and the Services concerning section 7 consultations.

B. EPA will use existing information to identify water bodies impaired by mercury and selenium in the State of Idaho. "Impaired" is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect the above species are not met. Pursuant to Section 303(d) of the CWA, EPA will work with the State of Idaho to promote and develop strategies to identify sources of selenium and mercury contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce selenium and/or mercury loading into their habitat.

K. Selenium

The current Idaho Water Quality Standards establish an acute criterion of 20 µg/L and a chronic criterion of 5.0 µg/L for total recoverable selenium. Selenium is not measured as dissolved under the Idaho Water Quality Standards.

Selenium occurs naturally in aquatic environments in trace amounts. While selenium is ubiquitous in the earth's crust, only trace levels occur in aquatic environments. Selenium enters aquatic habitats from a number of anthropogenic and natural sources. Elevated levels in aquatic systems are found in regions where soil is selenium-rich or where soils are extensively irrigated (Dobbs et al., 1996). As an essential micronutrient, selenium is used by animals for normal cell functions. However, the difference between useful amounts of selenium and toxic amounts is small. The toxic effects of selenium range from physical malformations during embryonic development to sterility and death (Lemly and Smith, 1987). Selenium has also been shown to protect some species from the toxicity of other chemicals. For example, the toxicity of cadmium in freshwater snails is inhibited by selenium and antagonizes mercury toxicity in rainbow trout (Eisler, 1985b).

The behavior of selenium in biological systems is complex. Selenium is a metalloid that exists in three oxidation states in water: selenide (-2), selenite (+4) and selenate (+6). The toxicity of selenium varies with its chemical species. Organic and reduced forms of selenium (e.g. seleno-methionine and selenite) are generally more toxic and will bioaccumulate (Besser et al., 1993; Kiffney and Knight, 1990). Toxicity also varies with the species exposed. Species at higher trophic levels, such as piscivorous fish and birds, are affected by the lowest concentrations of selenium. It appears that long term, low level exposures from water or food have the greatest effect on aquatic organisms (Lemly, 1985).

1. Bioconcentration and Biomagnification

Bioconcentration of selenium may be modified by water temperature, age of receptor organism, organ, and tissue specificity and mode of administration (Eisler, 1985b). Fish bioconcentrate selenium in their tissues with particularly high concentrations observed in ovaries when compared to muscle tissues (Lemly, 1985; Hamilton et al., 1990) and milt (Hamilton and Weddall, 1994). Reproductive failure is often associated with bioaccumulation of selenium in ovaries and offspring (Hamilton et al., 1990). Selenium that is bioconcentrated appears to occur in its most harmful concentrations in predator species such as mallard ducks or chinook salmon (Hamilton et al., 1990). At concentrations greater than 0.002-0.005 mg/L in water, selenium can be bioconcentrated and cause significant toxicity and reproductive failure in fish (Hermanutz et al., 1992). Bioconcentration factors (BCFs) in rainbow trout range from 2-20 after exposure to 220-410 µg/L selenium. The magnitude of the BCFs appeared to be inversely related to exposure concentration (Adams and Johnson, 1981). The transformation of selenium to organoselenium increases the bioconcentration of the compounds in fish ovaries resulting in significant pathology and reproductive failure (Srivastava and Srivastava, 1994; Sorenson and Bauer, 1983; Baumann and Gillespie, 1986).

Biomagnification (a progressive increase in concentration from one trophic level to the next higher level) of selenium has also been well documented. The magnitude of the biomagnification ranges from 2-6 times between producers and lower consumers (Lemly and Smith, 1987). Piscivorous fish accumulate the highest levels of selenium and are generally one of the first organisms affected by selenium exposure, followed by planktivores and omnivores (Lemly, 1985).

2. Invertebrates

Sublethal effects

No recent information was available documenting the sublethal effects of selenium on freshwater snails. However, EPA (1980i) has previously evaluated sublethal effects of selenium on aquatic invertebrates as part of the development of the most recent criteria and found the criteria not likely to adversely affect these organisms. Sublethal effects for aquatic invertebrates documented by EPA include EC_{50}s determined for *Daphnia magna* that ranged from 9.9-2,500 µg/L after acute exposures. Under long-term exposure periods, scientists determined an EC_{50} of 430 µg/L for *D. magna* (EPA, 1980i). Therefore, EPA has determined that the acute and chronic selenium criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

In the marine Pacific oyster, *Crassostrea gigas*, the 48-hour LC_{50} was estimated to be greater than 10,000 µg/L. For freshwater snails of the genus, *Physa*, the LC_{50} was determined to be 24,100 µg/L (EPA, 1980i) and greater than 10,000 µg/L (Eisler, 1985b) by different researchers. In other freshwater snail species such as *Aplexa hypnorum*, the 7.5-day LC_{50} was determined to be 193,000 µg/L. The 7.5-day LC_{50} determined for *Lymnaea stagnalis* was 3,000 µg/L (EPA, 1980i). In all cases, the LC_{50} values determined were well above both the acute and chronic criteria established by the Idaho Water Quality Standards. EPA has determined that the acute and chronic selenium criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

No recent information was available detailing the sublethal effects of selenium on listed aquatic snail species. However, EPA (1980i) addressed the potential for sublethal effects of selenium on aquatic invertebrates during the recent development of the criteria. Additionally, all lethal effects were well above both the acute and chronic criteria established by the Idaho Water Quality Standards. Therefore, EPA has determined that the approval of the **acute and chronic selenium criteria** (20 µg/L=acute, 5.0 µg/L=chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

Studies have shown that selenium negatively affects aquatic organisms at concentrations between 10-100 µg/L (EPA, 1980i). Fish appear to be sensitive to selenium toxicity under conditions of long-term exposure from both water and dietary sources. Waterborne selenium is depurated in fish via a passive excretion pathway, while dietary selenium is excreted more actively. The half-life of selenium is inversely proportional to dietary loading. Inorganic selenium absorbed from water is stored in fish as inorganic selenium. However, inorganic selenium absorbed from the diet is transformed by the liver to an organic form that is more toxic, but can be excreted easily (Hodson et al., 1984b). Selenium taken up from water is absorbed across the gills and taken directly to all tissues except the liver. The liver receives its blood supply via a portal system from the gut. Dietary selenium is taken up through the gut, thus passing through the liver first. The tissue distribution of selenium within fish is a function of the loading rate, but not the source of selenium (Hodson and Hilton, 1983).

Due to the sensitivity of fish to long-term low concentration exposures of selenium, the indications of relative sensitivity to waterborne selenium may become reversed when comparing acute and chronic studies. For this reason, comparisons of acute and chronic sensitivities of fish to selenium should be interpreted with caution (Lemly, 1985). Hermanutz et al. (1992) also suggest that the estimation of effects using studies of waterborne exposure exclusively may underestimate the danger of selenium exposure to fish. The optimum dietary selenium level in rainbow trout is estimated to be between 0.15-0.38 µg/g by Hilton et al. (1980). However, trout appear to be able to accommodate excess dietary selenium in the short term using both behavioral and physiological adaptations.

Sublethal effects

Studies have shown that exposure to selenium can reduce fish growth particularly weight and, to a lesser extent, length (Albers et al., 1996; Green and Albers, 1997; Hamilton et al., 1990). At selenium concentrations of 250 ppb in water, rainbow trout fry growth was reduced following a 21-day exposure (Eisler, 1985b). Weight was reduced by 29-70% in fall-run chinook fed greater than 18.2 µg/g for 90 days (Hamilton et al., 1990). Concentrations of 35.4 µg/L for 60-days and 9.6 µg/L for 90-days reduced chinook salmon body weight and survival (Hamilton et al., 1986).

Selenium exposures can also reduce red blood cell volumes and cellular blood iron content in rainbow trout juveniles at concentrations greater than or equal to 53 and 16 µg/L, respectively, after 44 weeks. Hatchability of eggs was affected at concentrations as low as 16 µg/L in the same experiment. A slight decrease in the time to hatch was observed at 4.4 µg/L, however the results were not statistically significant when compared to controls (Hodson et al., 1980). Selenium also affects the immune responses of fish by influencing the activity of glutathione peroxidase (GPX). GPX is an antioxidant that protects cellular membranes and organelles from peroxidative damage that may be caused by superoxide radicals (Felton et al., 1990). Selenium concentrations of 13 µg/L for 6 weeks reduced smolting success of chinook salmon (Hamilton et al., 1986).

At concentrations of 47-50 ppb (µg/L) in water, selenium exposures were associated with anemia and reduced hatch of rainbow trout (Eisler, 1985b). At 47 µg/L over 41 days, investigators observed reduced hatch of eyed embryos of rainbow trout (EPA, 1980i). Significant deformities resulted from exposure of rainbow trout eggs to 80 µg/L selenium (Lemly and Smith, 1987).

Due to the ability of fish and invertebrates to bioconcentrate selenium, fish can be exposed to harmful concentrations of selenium via diet even when water concentrations are low. In chinook salmon, specifically, swim-up larvae and fingerlings, 3.2 µg/g selenium in the diet adversely affected growth. Using a bioaccumulation factor of 1,800 for aquatic invertebrates (Pease et al., 1992), it would be possible to obtain a dietary concentration of 3.2 µg/g at a water concentration as low as 1.8 µg/L. Lemly (1996) set forth a limit of 2 µg/L on a chronic basis as hazardous to the health and survival of fish. Selenium concentrations at low levels near this limit would primarily act through bioaccumulation.

The results of research examining the sublethal effects of selenium on trout indicate that fish are adversely affected by selenium concentrations in water that are above both the acute and chronic selenium criteria established by the Idaho Water Quality Standards. It is possible however, that water concentrations lower than the chronic selenium criteria may result in dietary concentrations of selenium that may be harmful to fish species.

Therefore, due to the potential adverse effects due to bioaccumulative exposures to selenium, EPA has determined that the chronic selenium criterion established by the Idaho Water Quality Standards is likely to adversely affect the general health and behavior of the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, and Kootenai River white sturgeon. From the available information, EPA has determined that the acute selenium criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

Dietary concentrations as low as 13 µg/g caused elevated mortality, reduced feeding, slower growth, higher feed-to-weight gain ratios and liver paleness in trout within 4 weeks. Dying fish were reported to swim in uncoordinated spirals and were noted as being oblivious to physical obstacles (Hilton et al., 1980). In rainbow trout, 96-hour and 9-day LC_{50}s were determined to be 8.1 mg/L and 6.5 mg/L, respectively. After 44 weeks, significant mortality was observed in rainbow trout eyed eggs at concentrations greater than or equal to 25 µg/L (Hodson et al., 1980). In fall-run chinook salmon, reduced survival was observed at 35.4 µg/g dietary selenium for 60 days and greater than 9.6 µg/g dietary selenium for 90 days (Hamilton et al., 1990). Long-term exposures (44 weeks) to 130 µg/L selenium caused elevated mortality rates in rainbow trout along with increased incidence of deformities at concentrations as low as 60 µg/L (Hodson et al., 1984b).

Lethal effects of selenium can vary among and within species. For example, when Puget Sound wild and hatchery reared coho salmon were compared, wild fish survival rates were found to be 1.5-2.0 times higher than those of hatchery reared fish exposed to the same selenium contaminated water. Selenium residues were also higher in wild fish versus hatchery reared fish (Felton et al., 1990). In chinook salmon fry, exposures to 17 µg/L for 30 days caused a significant increase in mortality (Hamilton et al., 1986). The 43-day LC_{50} for chinook larvae and the 48-day LC_{50} for chinook fry was 160 µg/L (Eisler, 1985b; Lemly and Smith, 1987). In rainbow trout, the 9-day LC_{50} was estimated to range between 5,400-7,000 µg/L (EPA, 1980i). The 48-day LC_{50} for rainbow trout larvae was determined to be 500 µg/L and significant mortality was observed at 80 µg/L over a 12 month exposure (Lemly and Smith, 1987). In bull trout, the LC_{50} was estimated to be 10,200 µg/L (EPA, 1980i).

From the information presented regarding the lethal effects of selenium on salmon species, EPA has determined that the acute and chronic selenium criteria established by the Idaho Water Quality Standards are not likely to adversely affect survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

EPA has determined that the approval of the **acute selenium criterion** (20 µg/L=acute) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**. Furthermore, EPA has determined that the approval of the **chronic selenium criterion** (5.0 µg/L=chronic) established by the Idaho Water Quality Standards **may be likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

The results of investigations examining the effects of selenium on avian species indicate that the ability to eliminate or attenuate selenium in feathers and the length of exposure to selenium can vary greatly between avian receptors, sometimes even within a genus (Burger et al., 1992). The domestic chicken is considered, however, to be one of the more sensitive of bird species to selenium toxicity. Reduced hatching success of chicken eggs occurred at dietary concentrations of only 7-9 ppm (mg/kg) and similar results occurred in Japanese Quail at 6-12 ppm (Eisler, 1985b).

The dietary concentration of selenium that may be likely to adversely affect avian reproductive systems is typically much lower than those concentrations that cause mortality. For example, 40 mg/kg dietary selenium caused mortality in mallard ducklings, whereas only 8 mg/kg fed to adults impaired reproduction (Heinz et al., 1988; Heinz et al., 1989). The dietary NOEL for reproductive effects in mallards is 4 mg/kg for both organic and inorganic selenium (Peterson and Nebeker, 1992). However, in chickens, decreased hatchability of fertile eggs was associated with dietary concentrations of 5 mg/kg and mortality was significant when chicks were fed 40 mg/kg. Selenium concentrations of 100 ppm fed to adult mallards were fatal within 1 month. Similarly, survival of mallards was high when fed 25 ppm selenium for 3 months, but poor egg hatchability was recorded at the same time. A dietary concentration of only 10 ppm reduced productivity and duckling survival in adult mallards (Heinz et al., 1987). Hatching success was also reduced in adults fed 10 ppm selenomethionine (Eisler, 1985b). Heinz (1996) set forth a threshold for reproductive effects of 3 ppm in eggs. Mallards fed 4 ppm selenium in their diet were shown to have accumulated 3.4 ppm selenium in their eggs.

In a study of birds in the San Joaquin Valley, 40.6% of the nests studied had at least one dead embryo and 19.6% had at least one embryo or chick with an obvious external defect. Defects included missing or abnormal eyes, beaks, wings, legs and feet, plus brain, heart liver, and skeletal anomalies. The mean concentration of selenium in plants, invertebrates, and fish in this area was 22-175 ppm. Bird eggs and livers also had elevated selenium concentrations: 2.2-110 ppm in eggs and 19-130 ppm in livers (Ohlendorf et al., 1986).

There appears to be a clear relationship between egg selenium concentration and measures of embryo toxicity and teratogenesis in avian species. Mean egg selenium levels of 13-24 mg/kg increase overt embryo deformity, while the threshold for reduced hatchability can occur at egg concentrations greater than 8 mg/kg. Waterborne selenium concentrations were positively correlated with selenium concentrations in aquatic organisms.

Albers et al. (1996) concluded from a 16 week study on ducks exposed to a range of dietary selenium concentrations from 0-80 ppm, that selenium accumulated in tissues proportionally to dietary concentrations. All ducks exposed to the highest selenium concentration, 80 ppm, died by the conclusion of the test. Those animals exposed to 40 and 80 ppm selenium consumed less feed than others and their body weights subsequently declined. The post breeding molt was also delayed in ducks surviving the 40 ppm exposure. In all cases of mortality, abnormalities and histological lesions were observed. A smaller number of abnormalities and lesions occurred in surviving ducks. The weights of organs in those ducks that died were generally lower than those of ducks euthanized at the end of the test, with the exception of the kidneys (Albers et al., 1996; Green and Albers, 1997).

Table 250.07.a.14 illustrates a conversion of the maximum allowable water criteria under the Idaho Water Quality Standards to dietary concentrations for piscivorous birds using the maximum bioaccumulation factor (BAF) for fish provided by Eisler (1985b). This allows an interpretation of the dietary concentrations referenced above in the context of the current Idaho Water Quality Standards. A dietary concentration of 5 mg Se/kg is deemed safe for mallards (Eisler, 1985b). The accumulation of metals from food generally occurs over a longer period of time than that for which the acute criterion is applied. For this reason, the accumulation potential was evaluated based on exposure to the chronic criteria. At the maximum allowable chronic water criteria, birds would consume a dietary selenium concentration greater than deemed safe by Eisler (1985b). Many of the adverse effects listed above occur at concentrations less than those that would occur at the maximum allowable water criteria levels as well. Therefore from the information available with regard to selenium toxicity to avian species, EPA has determined that the approval of the **acute selenium criterion** (20 µg/L=acute) established by the Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**. However, EPA has determined that the approval of the **chronic selenium criterion** (5.0 µg/L=chronic)established by the Idaho Water Quality Standards **may be likely to adversely affect the bald eagle, peregrine falcon, and whooping crane.**

Table 250.07.a.14. Avian Dietary Concentrations allowed by Idaho Water Quality Standards

Dietary Concentrations		Selenium Criteria (Total Recoverable)		Maximum BAF for fish, as prey for birds and mammals (Eisler, 1985b)
Acute	Chronic	Acute	Chronic	
125.1-136 mg Se/kg	31.3-34 mg Se/kg	0.020 mg Se/L	0.005 mg Se/L	6,800

5. Proposed revisions to the selenium criteria and other actions for the protection of threatened and endangered species

A. EPA will revise its recommended 304(a) acute and chronic aquatic life criteria for selenium by January 2002. EPA will work in close cooperation with the Services to evaluate the degree of protection afforded to listed species by the revisions to these criteria. EPA will solicit public comment on the proposed criteria as part of its rulemaking process, and will take into account all available information, including the information contained in the Services' Opinion, to ensure that the revised criteria will adequately protect federally listed species. If the revised criteria are less stringent than those proposed by the Services in the Opinion, EPA will provide the Services with a biological evaluation/assessment on the revised criteria by the time of the proposal to allow the Services to complete a biological opinion on the proposed selenium criteria before promulgating final criteria. EPA will provide the Services with updates regarding the status of EPA's revision of the criterion and any draft biological evaluation/assessment associated with the revision. EPA will promulgate final criteria as soon as possible, but no later than 18 months, after proposal. If indicated by the results of this revision, EPA will collaborate with Idaho to propose revised criteria by January 2003.

B. EPA will utilize existing information to identify water bodies impaired by selenium in the State of Idaho. Impaired is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect federally listed species are not met. Pursuant to Section 303(d) of the CWA, EPA will work, in cooperation with the Services, and the State of Idaho to promote and develop strategies to identify sources of selenium contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce selenium loading into their habitat. (See also "Other Actions B." below.)

Other Actions:

A. EPA will initiate a process to develop a national methodology to derive site-specific criteria to protect federally listed threatened and endangered species, including wildlife, in accordance with the draft MOA between EPA and the Services concerning section 7 consultations.

B. EPA will use existing information to identify water bodies impaired by mercury and selenium in the State of Idaho. "Impaired" is defined as water bodies for which fish or waterfowl consumption advisories exist or where water quality criteria necessary to protect the above species are not met. Pursuant to Section 303(d) of the CWA, EPA will work with the State of Idaho to promote and develop strategies to identify sources of selenium and mercury contamination to the impaired water bodies where federally listed species exist, and use existing authorities and resources to identify, promote, and implement measures to reduce selenium and/or mercury loading into their habitat.

L. Zinc

The current Idaho Water Quality Standards establish hardness dependent zinc criteria . At a hardness of 100 mg/L $CaCO_3$, the acute and chronic criteria are 110 µg/L and 100 µg/L, respectively. The corresponding total recoverable criteria are 120µg/L and 110 µg/L at a water hardness of 100 mg/L $CaCO_3$ for short-term and long-term exposures, respectively (Table 250.07.a.15). Table 250.07.a.16 lists criteria calculated at the hardness levels used in the studies referenced in this report.

Table 250.07.a.15. Idaho Zinc Water Quality Criteria				
Hardness	Acute Criteria		Chronic Criteria	
	Total	Dissolved	Total	Dissolved
100 mg/L $CaCO_3$	120 µg/L	117.36 µg/L	110 µg/L	108.46 µg/L

Table 250.07.a.16. Idaho Water Quality Criteria for Zinc Calculated for Referenced Hardness Values and Total Recoverable Analysis		
Hardness	Acute Criteria (Total Recoverable)	Chronic Criteria (Total Recoverable)
2 mg/L $CaCO_3$	4.3 µg/L	3.9 µg/L
2.7 mg/L $CaCO_3$	5.5 µg/L	5.0 µg/L
15 mg/L $CaCO_3$	23 µg/L	21 µg/L
20 mg/L $CaCO_3$	30 µg/L	27 µg/L
41.3 mg/L $CaCO_3$	55 µg/L	50 µg/L
60 mg/L $CaCO_3$	76 µg/L	69 µg/L
100 mg/L $CaCO_3$	120 µg/L	110 µg/L
170 mg/L $CaCO_3$	180 µg/L	170 µg/L

Zinc is naturally introduced into aquatic systems, usually via leaching from igneous rocks. Concentrations of zinc associated with background freshwater systems are estimated to range between 0.5-15 µg/L (Moore and Ramamoorthy, 1984; Groth, 1971). Most of this naturally introduced zinc is adsorbed to sediments, however a small amount remains in the water, predominantly in the form of the free Zn^{2+} ion. Release of zinc from sediment is enhanced by the combination of high dissolved oxygen, low salinity, and low pH (Eisler, 1993). All life forms require zinc as an essential element, however aquatic animals tend to accumulate excess zinc which can result in growth retardation, hyperchromic anemia, and defective bone mineralization. Zinc primarily affects zinc-dependent enzymes regulating RNA and DNA. Zinc also increases the numbers of metallothioneins, low molecular weight proteins involved in zinc homeostasis. In mammals and birds, the pancreas and bone seem to be the primary targets of zinc toxicity, whereas in fish, it is the gill epithelium (Eisler, 1993). Toxicity of zinc to aquatic organisms is dependent upon the type and life stage of organism as well as the concentrations of other chemicals in the water. Substances such as calcium and magnesium can reduce zinc toxicity. Other compounds such as cadmium, copper, iron, and molybdenum also interact antagonistically with zinc (Hammond and Beliles, 1980). Zinc ions and other toxic species affect aquatic organisms most severely in environments characterized by low pH, low alkalinity, low dissolved oxygen and elevated temperature (Eisler, 1993). However, there is some evidence that fish acclimate to elevated temperature are more tolerant of zinc toxicity. An increase in temperature during exposure to zinc appears to cause increased sensitivity to zinc as a result of temperature stress, while fish that have acclimated to higher temperatures (no temperature stress) are less sensitive to zinc (Hodson and Sprague, 1975).

1. Bioconcentration and Biomagnification

Because zinc combines with biomolecules in target species and most of these species accumulate more than they need for normal metabolism, data showing bioconcentration factors for target receptors may be misleading. Bioconcentration (an increase concentration of a substance in relation to the concentration in the ambient environment) is also dependent on the target organism of interest. Bioconcentration factors (BCF's) reported in the EPA water quality criteria for zinc (EPA, 1987b) ranged from 51 in Atlantic salmon (*Salmo salar*) to 1,130 for the mayfly (*Ephemerella grandis*).

Little to no evidence exists indicating the successive biomagnification (a progressive increase in concentration from one trophic level to the next higher level) of zinc in tissues of fish and avian receptors. This assumption is based on several factors. First, existing BCF data (EPA, 1987b) shows that the greatest BCF was seen in mayflies while the least was found in Atlantic salmon. This trend was also seen in Elder and Collins (1991) who showed that molluscs accumulated more zinc than the fish who feed off of these molluscs. Furthermore, the existing zinc toxicity data for birds is predominantly based on force feeding studies of zinc shot or dietary supplements (Eisler, 1993).

2. Invertebrates

Sublethal effects

Willis (1988) evaluated the effects of zinc on the reproduction of the snail, *Ancylys fluviatilis*. Concentrations of 100 µg/L Zn were lethal to newly hatched organisms in artificial streams with a hardness of 15 mg/L $CaCO_3$. Zinc concentrations greater than 50 µg/L affected cellulolytic enzyme activity of freshwater asiatic clams (*Corbicula fluminea*) in artificial streams with a hardness of 60 mg/L (Farris et al., 1994). However, effects on cellulolytic enzyme activity take place on a cellular and individual level and were not associated with any observable adverse effects on the clams. Additionally, a twelve week growth study by Dorgelo et al. (1995) found 75 µg/L Zn to be the lowest concentration to significantly suppress growth of *Potamopyrgus jenkinsi* (a freshwater snail) in an eight week experiment where zinc was provided via lake water enriched with $ZnCl_2$; however, no hardness was given for this experiment.

Based on this information, EPA has determined that both the acute and chronic zinc criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Lethal effects

Lethal effects for freshwater snails and clams have been detailed in EPA (1987b). The lowest documented LC_{50} value is 241 µg/L for the freshwater snail, *Physa heterostropha*, after a 96 hour exposure at a hardness of 100 mg/L. For other aquatic snails, LC_{50} values ranged from 658-20,000 µg/L at hardness values ranging from 20-170 mg/L. Based on this information, EPA has determined that the acute and chronic zinc criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.

Summary

Farris et al. (1994) found that the cellulolytic enzyme activity of asiatic clams was adversely affected at zinc concentrations below both the acute and chronic criteria established by the Idaho Water Quality Standards. However, these effects occur at the cellular level and no adverse effects were observed at the organism or species levels. Therefore, EPA has determined that the approval of the **acute and chronic zinc criteria** (110 µg/L=acute, 100 µg/L=chronic, hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail**.

3. Fish

Sublethal effects

Coho salmon and cutthroat trout fry were observed to avoid water contaminated with zinc at nominal concentrations ranging from 6.54-28 µg/L at hardnesses of 15-100 mg/L $CaCO_3$ (Rehnberg and Schreck, 1986; Woodward et al., 1997). However, the significance of the zinc avoidance in the Rehnberg and Schreck study may have been due to small sample size as higher zinc concentrations did not deter juvenile coho salmon. In the Woodward study, it should be noted that the measured zinc concentrations in waters avoided by cutthroat trout ranged from 66-74 µg/L, much higher than the nominal concentrations (Hardness = 15-25 mg/L $CaCO_3$). Therefore, EPA has determined that the acute and chronic zinc criteria established by the Idaho Water Quality Standards are not likely to adversely affect the general health and behavior of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Lethal effects

Sayer et al. (1989) saw 75-95% mortality for brown trout yolk sac fry at exposures to concentrations of 4.9-19.6 µg/L Zn for 30 days. This mortality occurred in waters with a hardness of only 2 mg/L $CaCO_3$ and low pH, not typical of waters in Idaho. For steelhead, Buhl and Hamilton (1990) observed LC_{50}s of 169-215 µg/L Zn at a hardness of 41.3 mg/L $CaCO_3$. Similarly, Buhl and Hamilton found LC_{50} values ranging between 112-168 µg/L Zn at a hardness of 41.3 mg/L $CaCO_3$ for arctic grayling juveniles, (*Thymallus arcticus*). An LC_{54} was obtained when rainbow trout larvae and alevins were exposed to 10 µg/L zinc for 28 days (hardness = 2.7 mg/L; Affleck, 1952). When the Idaho Water Quality Standards for zinc recalculated for comparable hardness values (see Table 250.07.a.16), concentrations of zinc that cause lethal effects in salmonid species are above those allowed by the Idaho Water Quality Standards. Therefore, EPA has determined that both the acute and chronic zinc criteria established by the Idaho Water Quality Standards are not likely to adversely affect the survival of the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Summary

EPA has determined that the approval of the **acute and chronic zinc criteria** (110 µg/L=acute, 100 µg/L=chronic, hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

4. Birds

Little information is available regarding the toxicity of zinc to bald eagles, peregrine falcons, and whooping cranes. The lowest concentration of zinc found to affect any avian species is a dietary concentration of 100 mg Zn/kg in the domestic chicken, *Gallus sp.* This dietary concentration of zinc caused pancreas histopathology in chicks under conditions of selenium deficiency (Eisler, 1993).

The acute and chronic criteria for zinc established by the Idaho Water Quality Standards have been converted to dietary concentrations representative of what would be consumed by a piscivorous bird consuming fish that concentrated zinc at the highest known BCF (Table 250.07.a.17). To prevent marginal negative effects in chickens, feed should contain <178 mg Zn/kg. Even at the maximum allowable zinc concentrations, birds would not consume a dietary concentration greater than 100 mg Zn/kg. Therefore, EPA has determined that the approval of the **acute and chronic zinc criteria** (110 µg/L=acute, 100 µg/L=chronic, hardness of 100 mg/L $CaCO_3$) established by the Idaho Water Quality Standards **is not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

Table 250.07.a.17. Dietary Concentrations allowed by Idaho Water Quality Standards					
Dietary Concentrations		Zinc Criteria (Total Recoverable)		Maximum BCF for fish, as prey for birds and mammals (Eisler, 1993)	Hardness
Acute	Chronic	Acute	Chronic		
28.08 mg Zn/kg	25.5 mg Zn/kg	0.065 mg Zn/L	0.059 mg Zn/L	432	50 mg/L $CaCO_3$
51.84 mg Zn/kg	47.5 mg Zn/kg	0.120 mg Zn/L	0.110 mg Zn/L	432	100 mg/L $CaCO_3$
90.72 mg Zn/kg	82.1 mg Zn/kg	0.210 mg Zn/L	0.190 mg Zn/L	432	200 mg/L $CaCO_3$

M. ANALYSIS OF EFFECTS OF NUMERIC CRITERIA FOR TOXIC POLLUTANTS TO STURGEON

1. Kootenai River White Sturgeon

A literature search yielded very limited information on effects of toxicants to white sturgeon. An accepted practice in this situation is to use a species for which there is adequate toxicity information as a surrogate for the species in question (EPA, 1995). It is not difficult to state the liklihood of adverse effects with relatively good certainty if we apply interspecies correlation models (e.g., rainbow trout vs. shortnose sturgeon; see below) as an estimate of toxicity to the sturgeon family. However, the available cold water species surrogate, rainbow trout, is very different from white sturgeon in terms of life history, habitat use, and feeding strategy. For example, the long lives of adult sturgeon may result in bioaccumulation of persistent toxicants that could be passed to offspring (Bennett and Farrell, 1998). However, other species of fish, such as catfish, have more similar life histories, but they occur in warm water. Used in combination, this data can offer a good estimate of toxicity of the criteria to sturgeon.

Limited studies that have been conducted on sturgeon species suggests that they have some resistance to certain toxicological effects that is variable compared to other species. Bennett and Farrell (1998) concluded that juvenile white sturgeon lie within the sensitivity range of other juvenile fish for chlorinated phenols. But, white sturgeon fry appear to have greater sensitivity to didecyldimethylammonium chloride than other fish species. In a study of early growth of coho and Masu salmon and Siberian sturgeon related to toxic conditions, all three species exhibited similar growth impairment disturbance (Glubokov, 1990). Of the three species, Masu salmon were the most sensitive to copper and phenol. Variation in the toxicoresistance among aquatic species and among chemicals, with no single species always being the most sensitive has been well documented (Mayer and Ellersieck, 1986). However, Mayer et al. (1987) found that interspecies correlation models for acute toxicity were highly dependable in estimating toxicity for species with unknown sensitivity to chemicals from acute toxicity values for common test fishes (rainbow trout, fathead minnows, bluegills). Correlations are good among many phyletic families and a variety of chemicals (Doherty, 1983), but with pesticides, correlations are best within families or closely related families (e.g., fishes) as reported by Kenaga (1978), LeBlanc (1984), Mayer et. al. (1987), and Suter and Vaughan (1985).

Rather than taking the default approach and assigning a "likely to adversely affect" determination for white sturgeon, we have chosen to evaluate the proposed standards by examining toxicity data for a variety of fish species, including cold water species (e.g. salmonids) and benthic species (e.g. catfish). If the proposed standards are protective of a variety of fish species, we can assume that the standards will also adequately protect white sturgeon for the following reasons: 1) the proposed standards are below the limits for other fish species and 2) the limited data available show that sturgeon have variable sensitivity compared to other species (i.e. they are not consistently more sensitive than other species). Thus, standards that protect other fish species will adequately protect white sturgeon. This has recently been supported in research led by F.L. Mayer (Dwyer et al. 1995, Dwyer et al. 1999a, 1999b, Mayer et al. 2000). Acute toxicity tests with five chemicals (carbaryl, copper, 4-nonylphenol, pentachlorophenol, permethrin) and 19 fish species (rainbow trout, fathead minnows, sheepshead minnows, and 16 endangered fishes) indicated that salmonid data are generally protective of sturgeons (shortnose sturgeon). Also, interspecies correlations with rainbow trout or fathead minnows are highly predictive for acute toxicity with the shortnose sturgeon.

a. Arsenic

The discussion of arsenic effects on salmonids is discussed in the acute and chronic cadmium criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that approval of Idaho's acute arsenic criterion is not likely to adversely affect endangered salmonids. Some lethal effects may occur at arsenic levels permitted by the chronic criterion, however, the human health arsenic criterion of 50 µg/L will apply in all Idaho surface waters. This number has been shown to be protective of endangered salmonid species.

Sublethal effects

In catfish, sublethal effects including impaired growth and altered histopathology occur at arsenic concentrations of 1,500-15,000 µg/L (Clemens and Sneed, 1959; Gupta and Chakrabarti, 1993; Shukla et al., 1985; Shukla et al. 1987). Green sunfish experienced sublethal effects such as bioaccumulation of arsenic and histopathological changes when exposed to 31,700-62,500 µg/L arsenic (Sorensen, 1976).

Lethal effects

Catfish experience increased mortalities at arsenic concentrations between 10,900-100,000 µg/L (Clemens and Sneed, 1959; Gupta and Chakrabarti, 1993; Shukla et al., 1987). Other fish species, such as asiatic knifefish and goldfish, experience lethal effects when arsenic concentrations reach 490-30,930 µg/L (Birge et al., 1979; Ghosh and Chakrabarti, 1990).

b. Cadmium

The discussion of cadmium effects on salmonids is discussed in the acute and chronic cadmium criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that the acute and chronic cadmium is not likely to adversely affect endangered salmonids. Effects to other fish species are described in the following sections.

Sublethal effects

Sublethal effects such as altered enzyme activity, physiology and histopathology occur in catfish exposed to cadmium concentrations ranging from 300-400,000 µg/L (Bhattacharya et al., 1987; Bhattacharya et al. 1989; Dalal, 1989; Dalal and Bhattacharya, 1991; Dalal and Bhattacharya, 1994; Dalwani et al., 1985; Ghosh and Bhattacharya, 1992; Ghosh and Jana, 1988; Gupta and Rajbanshi, 1982; Gupta and Rajbanshi, 1988; Jana and Sahana, 1988; Jana and Sahana, 1989; Katti and Sathyanesan, 1984a; Katti and Sathyanesan, 1984b; Katti and Sathyanesan, 1985; Saksena and Agarwal, 1986; Sastry and Subhadra, 1984; Sastry and Subhadra, 1985; Sastry et al., 1997; Smith et al., 1976). Whitefish appear to preferentially select water with 5µg/L cadmium over control waters in avoidance testing (McNicol and Scherer, 1993). Sublethal hemaetological effects on greenfish occur at concentrations ranging from 300-20,000 µg/L (Kuroshima, 1992), while goldfish experienced similar effects at 445 µg/L (Houston and Keen, 1984).

Lethal effects

For the Siberian sturgeon, Blubokov (1990) found that for early fry, exposure to cadmium concentrations of 5, 50, and 500 *u*g/L resulted in 11.5%, 6%, and 100% mortality respectively after a 16 day exposure. A hardness value for the test water was not given in this report.

Increased mortality occurs in catfish exposed to cadmium at levels ranging from 338.3-405,000 µg/L (Birge et al. 1985; Dalal and Bhattacharya, 1994; Chakrabarti and Ghosh, 1990; Das and Benerjee, 1980; Ghosh and Chakrabarti, 1993; Gupta, 1988; Gupta and Rajbanshi, 1982; Gupta and Rajbanshi, 1988; Gupta and Rajbanshi, 1991; Mitra, 1991; Phipps and Holcombe, 1985; Rausina et al., 1975; Sastry et al., 1997; Saxena and Parashari, 1983; Saxena et al., 1993; Spehar and Carlson, 1984). Squawfish experience significant mortalities at cadmium concentrations of 78-10,000µg/L (EPA, 1985b; Buhl, 1997). Scientists measured lethal effects in goldfish at 170µg/L (Birge et al., 1979). In a comparison study, researchers found LC_{50}s for bonytail and razorback sucker to be 148-168 µg/L and 139-160 µg/L, respectively, at a hardness of 199 mg/L $CaCO_3$ (Buhl, 1997). For the Siberian sturgeon, Glubokov (1990) found that for early fry, exposure to cadmium concentrations of 5, 50, and 500µg/L resulted in 11.5%, 6%, and 100% mortality respectively after a 16 day exposure. A hardness value for the test water was not given in this report.

c. Copper

The discussion of copper effects on salmonids is discussed in the acute and chronic copper criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that the acute and chronic copper is not likely to adversely affect endangered salmonids. Effects to other fish species are described in the following sections.

Sublethal effects

Apperson (1992) found 1.18-3.2 µg Cu/kg in white sturgeon oocytes in the Kootenai River whereas copper levels in the Kootenai River range from 2-12 µg/L. She concluded that the chronic effects of copper on wild sturgeon spawned in polluted waters and reared in contaminated sediments pose a severe threat on reproductive success. The average hardness for the Kootenai River ranges from 29.69-32.72 mg/L $CaCO_3$. However, it is important to note that not enough information was provided in this study to determine which ambient concentrations resulted in bioaccumulation of copper in sturgeon oocytes.

In catfish, sublethal effects such as altered enzyme levels, hemaetological parameters, histopathology, growth, and physiology occur at copper levels from 50-200,000 µg/L (Ansari, 1987; Asztalos, 1986; Bakshi, 1991; Benerjee and Homechaudhuri, 1990; Bhattacharya and Mukherjee, 1976; El-Domiaty, 1987; EPA, 1984; Ghosh and Jana, 1988; Gupta and Rajbanshi, 1979; James et al., 1995; James and Sampath, 1995; Jana and Sahana, 1988; Jana and Sahana, 1989; Khangarot et al., 1988; Khangarot, 1992; Mukherjee and Bhattacharya, 1974; Mukherjee and Bhattacharya, 1975; Mukherjee and Bhattacharya, 1977; Nemcsok et al, 1991; Perkins et al., 1997; Rajbanshi and Gupta, 1988; Sastry and Sachdeva, 1994; Sastry et al., 1997; Shaffi, 1978; Shaffi and Jeelani, 1985; Srivastava and Pandey, 1982; Sultana and Devi, 1995; Wurts and Perschbacher, 1994).

Lethal effects

Squawfish mortality increases at copper levels of 363-10,000 µg/L (EPA, 1984; Buhl and Hamilton, 1996). Lethal effects occur when killifish encounter waters with concentrations of copper measuring 330-1,300 µg/L (EPA, 1984).

d. Cyanide

The discussion of cynide effects on salmonids is discussed in the acute and chronic cyanide criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that the acute and chronic cyanide criteria is not likely to adversely affect endangered salmonids. Effects to other fish species are described in the following sections.

Sublethal effects

In a search for documented sublethal effects of cyanide on fish species, goldfish avoided waters contaminated with 260-2,000 µg/L (Berry, 1976; Costa, 1965). Green sunfish avoided waters containing 1,000-5,000 µg/L cyanide (Summerfelt and Lewis, 1967).

Lethal effects

Decreased survival of catfish occurred at concentrations of 161-310 µg/L (EPA, 1985c), while studies with killifish found decreased survival for that species at 370-420 µg/L (Schaut, 1939). Squawfish mortality increased at 4,000 µg/L (EPA, 1985c). LC_{50}s for goldfish ranged between 261-1134 µg/L (EPA, 1985c).

e. Endosulfan

The discussion of endosulfan effects on salmonids is discussed in the acute and chronic endosulfan criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that the acute and chronic endosulfan criteria is not likely to adversely affect endangered salmonids. Effects to other fish species are described in the following sections.

Sublethal effects

Catfish concentrated endosulfan when exposed to 0.05 µg/L -endosulfan and 3 µg/L -endosulfan (Murty and Devi, 1982). Hawkfish, a carp species, experienced some immunological effects at 1.3 µg/L -endosulfan and 8.8 µg/L -endosulfan (Swarup et al., 1981).

Lethal effects

Researchers found LC_{50}s for catfish to be 0.16 µg/L for -endosulfan and 6.6 µg/L for -endosulfan (Devi et al., 1981).

f. Lead

The discussion of lead effects on salmonids is discussed in the acute and chronic lead criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that the acute and chronic lead criteria is not likely to adversely affect endangered salmonids. Effects to other fish species are described in the following sections.

Sublethal effects

Lead concentrations ranging from 2,300-145,720 µg/L caused sublethal effects such as bioaccumulation, altered enzyme levels and hemaetology and histopathological effects in catfish (Abdelhamid and El-Ayouty, 1991; Chaurasia et al., 1996; Jana et al., 1986; Jha and Pandey, 1989; Jha, 1991; Katti and Sathyanesan, 1983; Katti and Sathyanesan, 1985; Katti and Sathyanesan, 1986a; Katti and Sathyanesan, 1986b; Katti and Sathyanesan, 1987a; Katti and Sathyanesan, 1987b; Mishra and Singh, 1997; Sastry and Gupta, 1978a; Sastry and Gupta, 1978b; Sastry and Gupta, 1979; Sastry and Gupta, 1980; Shaffi and Jeelani, 1985; Sharma et al., 1985). Hawkfish bioaccumulated lead at levels of 250,000-1,000,000 µg/L (Shakoori et al., 1992). In goldfish, sublethal effects such as cellular, enzyme, histopathological, and other physiological effects occur at lead levels of 400-5,000 µg/L (Bolognani Fantin et al., 1992; Bolognoni Fantin et al., 1993; EPA, 1985d; Franchini et al., 1991).

Lethal effects

LC_{50}s for catfish ranged from 16,600-38,000 µg/L (Saxena and Parashari, 1983), while the LC_{50} determined for mosquitofish was greater than 56,000,000 µg/L (EPA, 1985d). In goldfish, LC_{50}s ranged from 1660-40,000 µg/L (Birge et al., 1979; Bolognani Fantin et al., 1992).

g. Mercury

The discussion of mercury effects on salmonids is discussed in the acute and chronic mercury criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that approval of the chronic mercury criterion is likely to adversely affect salmonids. We have determined that there is sufficient data in the Biological Assessment to conclude that the approval of the chronic mercury criterion is likely to adversely affect Kootenai River white sturgeon as well. Effects of the acute mercury criterion to non-salmonid species are described in the following sections.

Sublethal effects

Catfish experience adverse effects from mercury concentrations ranging from 12-12,000 µg/L. The sublethal effects include bioaccumulation, altered enzyme activity and histopathological and physiological effects (Kendall, 1975; Kendall, 1977).

Lethal effects

The LC_{50}s determined for catfish fall between 340-50,000 µg/L (Clemens and Sneed, 1958; Kirubagaran and Joy, 1988). In killifish, scientists found LC_{50} values between 110-270 µg/L (EPA, 1985e).

h. Selenium

The discussion of selenium effects on salmonids is discussed in the acute and chronic selenium criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has determined that approval of the chronic selenium criterion is likely to adversely affect salmonids. We have determined that there is sufficient data in the Biological Assessment to conclude that the approval of the chronic selenium criterion is likely to adversely affect Kootenai River white sturgeon as well. Effects of the acute selenium criterion to non-salmonid species are described in the following sections.

Sublethal effects

In goldfish and flagfish, behavior and growth were affected by selenium concentrations between 250-33,200 µg/L (EPA, 1980i; Weir and Hine, 1970).

Lethal effects

Catfish LC_{50}s ranged from 19,100-46,700 µg/L, while LC_{50}s for goldfish were determined to fall between 8,800-110,000 µg/L (EPA, 1980i).

i. Zinc

The discussion of zine effects on salmonids is discussed in the acute and chronic zinc criteria section of this Biological Assessment. Based on EPA's review of the literature, the Agency has already determined that approval of Idaho's acute and chronic zinc criteria is not likely to adversely affect endangered salmonids. Effects to other fish species are described in the following sections.

Sublethal effects

Effects on catfish ranging from altered enzyme levels, bioaccumulation, and hemaetological, and histopathological effects occurred at concentrations of zinc ranging from 500-130,000 µg/L (Banerjee, 1993; Banerjee, 1998; Banerjee and Banerjee, 1988; Dalal and Bhattacharya, 1991; Dalal and Bhattacharya, 1994; Hemalatha and Dalal, 1989; Jeelani, 1989; Khangarot et al., 1981a; Khangarot, 1982b; Khangarot, 1984; Nemcsok and Boross, 1981; Shaffi, 1980; Shandilya and Banerjee, 1989; Shukla and Pandey, 1986a; Shukla and Pandey, 1986b; Sultana and Devi, 1995).

Lethal effects

In killifish, LC_{50}s ranged from 840-22,600 µg/L (EPA, 1987b; Rehwoldt et al., 1971), while LC_{50}s determined for squawfish occurred at 1,660-40,000 µg/L (Andros and Garton, 1980; Buhl and Hamilton, 1996; Hamilton, 1995). Researchers determined LC_{50}s for catfish to be between 1,700-12,000 µg/L (Banerjee, 1998; Hemalatha and Banerjee, 1993; Hilmy et al., 1987; Khangarot et al., 1981a; Khangarot, 1981b; Khangarot, 1982a; Khangarot and Durve, 1982; Reed et al., 1980; Saxena and Parashari, 1983; Saxena et al., 1993).

j. Summary

With the information regarding the toxicity of these seven chemicals to a variety of fish species, we have determined that EPA's approval of the **acute and chronic criteria for arsenic, cadmium, copper, cyanide, endosulfan, lead, and zinc and the acute criteria for mercury and selenium is not likely to adversely affect Kootenai River white sturgeon.**

We have also determined that EPA's approval of the **chronic criteria for mercury and selenium is likely to adversely affect Kootenai River white sturgeon.**

IV. LOWER PRIORITY POLLUTANTS

A. Effect of Lower Priority Pollutants on Threatened and Endangered Snails

A literature search resulted in no references revealing the toxicity of the following chemicals on snails of any species or the more general category of molluscs. During the development of the criteria, EPA considered toxicity data detailing the effects of these chemicals on other invertebrate species that may be used as surrogates for snails, including one of the most sensitive invertebrates, *Daphnia magna*. EPA has determined that the approval of **the acute and chronic criteria for aldrin/dieldrin, chlordane, chromium, DDT, Heptachlor, Lindane, PCBs, Nickel, Pentachlorophenol, Silver, and Toxaphene** established by the Idaho Water Quality Standards is **not likely to adversely affect the Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, and Utah valvata snail.**

B. Effect of Lower Priority Pollutants on Threatened and Endangered Fish, Including Sturgeon, and Birds

1. Aldrin/Dieldrin

Background

An acute criterion of 3 µg/L has been established by the Idaho Water Quality Standards for aldrin. EPA determined (EPA, 1980a) that the available data did not support the determination of a chronic toxicity criteria for aldrin. For dieldrin, the acute and chronic criteria are 2.5 µg/L and

0.0019 µg/L, respectively.

Aldrin and dieldrin are considered together because aldrin transforms to dieldrin in the environment or, metabolically, within organisms (Gakstatter, 1968). Both aldrin and dieldrin are chlorinated hydrocarbons and are two of the most widely used domestic pesticides. Aldrin is used more, but degrades into dieldrin (EPA, 1980a). The use of both these pesticides has now been restricted. Dieldrin is the most stable of the cyclodienes (a group that also includes endrin, heptachlor, endosulfan, and chlordane) and has a high affinity for lipids. This affinity results in rapid accumulation in aquatic food chains that may result in an organism accumulating enough dieldrin to exceed the lethal limit for a consumer or predator (EPA, 1980a).

Effects

Acute sensitivity to dieldrin has been shown to occur at 1.1 µg/L for rainbow trout. For cutthroat trout, 1-4-day LC_{50}s were estimated to be 11.1-19.6 µg/L (Swedburg, 1969). Aldrin is less toxic to fish as evidenced by the 1 day LC_{50} of 90 µg/L (Khan et al., 1973, Georgacakis et al., 1971). Based on this information some potential may exist for the acute criterion for dieldrin and aldrin to pose a threat to salmonids, but effects at low concentrations have only been shown in the referenced study. **Therefore, EPA has determined that the approval of Idaho's acute dieldrin and aldrin criteria is likely to adversely affect Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

While EPA has determined that the approval of the acute aquatic life criteria for aldrin/dieldrin established by the Idaho Water Quality Standards may have the potential to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout, an added level of protection for these species is offered by the following:

> The human health criteria for aldrin is 0.00013 µg/L and 0.00014 µg/L. The human health criteria for dieldrin is 0.00014 *u*g/L. These are the applicable aldrin and dieldrin criteria in all surface waters of Idaho. These criteria are significantly lower and more conservative than the acute and chronic aquatic life criteria.

If a recreational use is modified or removed from a waterbody and the criteria become less stringent than 0.00013 or 0.00014 *u*g/L for aldrin and dieldrin respectively, Idaho is required to submit this revision to EPA for approval/disapproval action. If EPA proposes to approve this revision, the Agency will then reinitiate consultation on that approval action .

In light of the information and measures described above and the effective human health criteria, EPA has determined that the approval of the **acute and chronic aldrin and dieldrin criteria** (3µg/L= acute aldrin, 2.5 µg/L = acute dieldrin, 0.0019 *u*g/L = chronic dieldrin) established by the Idaho Water Quality Standards is **not likely to adversely affect the Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

No threshold values of safety have been established for raptors, but the US Food and Drug Administration action level for maximum permissible tissue concentration of dieldrin in fish is 0.3

mg/kg. This tissue concentration is not expected to occur at concentrations allowable under the criteria (calculated by using the highest BCF available for fish). Based on this information, the criteria for dieldrin and aldrin are not likely to have an effect on birds which feed on fish. Therefore, EPA has determined that the approval of Idaho's **acute and chronic dieldrin and aldrin criteria are not likely to adversely affect bald eagle, peregrine falcon and whooping crane**.

2. Chlordane

Background

The acute and chronic criteria established by the Idaho Water Quality Standards for chlordane are 2.4 µg/L and 0.0043 µg/L, respectively. Chlordane is a broad spectrum, cyclodiene insecticide that has been used extensively for termite control, as a home and garden insecticide and an agricultural insecticide. However, the registration of chlordane for use as a home and agriculture insecticide has been suspended by EPA since 1983 (EPA, 1980b). Due to evidence of carcinogenicity, it is permitted to be used only to control underground termites.

Effects

The toxicity of chlordane can vary with temperature, sediment loading, age, condition, and nutritional history of the exposed organism and the formulation and isomer of the chemical. Lethal effects of chlordane have been observed at concentrations of 3 µg/L in carp and bass, 7.1 µg/L in bluegill and 25-115 µg/L in fathead minnows. Concentrations as low as 0.32 µg/L have been observed to cause adverse effects in brook trout over long-term exposure periods. Eisler (1990) states that the acute criteria may not be protective since 0.2-3 µg/L chlordane can be harmful to sensitive fish. However, these values reflect long-term exposures and are thus above the chronic criteria. **Therefore, EPA has determined that the approval of Idaho's chronic chlordane criterion is not likely to adversely affect Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout and EPA's approval of Idaho's acute chlordane criteria may have the potential to adversely affect Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

While EPA has determined that the approval of the acute aquatic life criteria for chlordane established by the Idaho Water Quality Standards may have the potential to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout, an added level of protection for these species is offered by the following:

> The human health chlordane criteria of 0.00057 *u*g/L and 0.00059 *u*g/l are the applicable chlordane criteria in all waters of Idaho. These criteria is significantly lower and more conservative than the acute and chronic aquatic life criteria.

If a recreational use is modified or removed from a waterbody and the criteria become less stringent than 0.00057 *u*g/l, Idaho is required to submit this revision to EPA for approval/disapproval action. If EPA proposes to approve this revision, the agency will then reinitiate consultation on that approval action .

In light of the above information and the currently effective measures, EPA has determined that the approval of the **acute and chronic chlordane criteria** (2.4 µg/L = acute, 0.0043 µg/L=chronic) established by the Idaho Water Quality Standards is **not likely to adversely affect the Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout**.

The highest BCF for fish referenced by Eiser (1990) is 18,700, which would result in a dietary exposure for birds of 0.08 mg/kg. The lowest dietary level of chlordane shown to effects birds is 10 mg/kg (Eisler, 1990). **Therefore EPA has determined that the approval of Idaho's acute and chronic chlordane criteria is not likely to adversely affect bald eagle, peregrine falcon, and whooping crane.**

3. Chromium (III)

Background

The toxicity of chromium is affected by water hardness. Therefore, the acute and chronic chromium (III) criteria established by the Idaho Water Quality Standards change with hardness values. Idaho Water Quality Standards include dissolved acute and chronic criteria of 550 µg/L and 180 µg/L, respectively, at a hardness of 100 mg/L $CaCO_3$. At the same hardness, the corresponding total recoverable acute criteria is 1,700 µg/L and the corresponding total recoverable chronic criteria is 210 µg/L.

Sources of chromium in aquatic systems include electroplating and metal finishing industries, publicly owned treatment plants, iron and steel foundries, inorganic chemical plants, tanneries, textile manufacturing, and runoff from urban and residential areas (Towill et al., 1978, Eisler, 1986a). In freshwater environments, hydrolysis and precipitation are the most important processes in determining the environmental fate of chromium, while absorption and bioaccumulation are considered minor (Ecological Analysts, 1981).

Effects

Chromium (III) is considered to be much less toxic than chromium (VI) (see next subsection). Data taken from the AQUIRE database shows effects levels for rainbow trout ranged from 2125-4625 µg/L for acute exposures and 277-922 µg/L for long-term exposures (Stevens and Chapman, 1984). Chromium (no oxidation state listed) was found to affect *Amnicola* sp. (a freshwater snail) at concentrations of 8,400-15,200 µg/L for acute exposures (Rehwoldt et al., 1973). Eisler (1986a) references a study where adverse effects in steelhead trout were observed at chromium levels as low as 30 µg/L. However, upon further inspection of the original study (Stevens and Chapman, 1984), it was found that the hardness value in the study was 25 mg/L. Therefore, effects from long-term exposures of steelhead embryos occurred at concentrations greater than 89 µg/L (Hardness=25 mg/L; chronic criteria= 57.2 µg/L at this hardness). These studies indicate that toxic effects occur at concentrations higher than those established by the Idaho Water Quality Standards.

From the above information, EPA has determined that the approval of the **acute and**

chronic chromium (III) criteria (550 µg/L=acute, 180 µg/L=chronic, hardness of 100 mg/L $CaCO_3$) **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

4. Chromium (VI)

Background

The dissolved acute and chronic chromium (VI) criteria established by the Idaho Water Quality Standards are 15 µg/L and 10 µg/L, respectively. Corresponding total recoverable criteria for short-term and long-term exposures are 16 µg/L and 11 µg/L, respectively. Sources of chromium in aquatic systems include electroplating and metal finishing industries, publicly owned treatment plants, iron and steel foundries, inorganic chemical plants, tanneries, textile manufacturing and runoff from urban and residential areas (Towill et al., 1978; Eisler, 1986a). In freshwater environments, hydrolysis and precipitation are the most important processes in determining the environmental fate of chromium, while absorption and bioaccumulation are considered minor. Chromium (VI) is highly soluble in water and thus very mobile in aquatic systems (Ecological Analysts, 1981).

Effects

Younger life stages of aquatic biota tend to be more sensitive to the toxic effects of chromium (VI). These effects include abnormal enzyme activities, altered blood chemistry, lowered resistance to disease, behavioral modifications, disrupted feeding, histopathology and osmoregulatory upset. For the freshwater mussel, *Anodonta imbecilis*, the 96-hour LC_{50} was found to be 39 µg/L (Keller and Zam, 1991). In rainbow trout, the 96-hour LC_{50} was 7,600 µg/L (Van der Putte, 1981). The more sensitive salmon fingerlings had a 12-week LC_{50} of 200 µg/L (Steven et al., 1976). An even lower chromium concentration of 16-21 µg/L reduced growth of both rainbow trout and chinook salmon fingerlings after 14-16 weeks (EPA, 1980c). Juvenile coho experienced a reduction in disease resistance and serum agglutinin production after only 2 weeks exposed to 0.5 mg/L. In seaward migrating coho, tolerance of salinity and serum osmolality were impaired during exposure to 0.4 mg/L for 4 weeks (Sugatt, 1980a, 1980b). EPA gives chronic data for rainbow trout that indicate chromium (VI) effects at a concentrations of 68.63 µg/L.

Many acute toxicity studies have been conducted at concentrations in excess of the proposed criterion (Eisler 1986). In acute toxicity tests in rainbow trout, the LC_{50} 96hr test ranged from 3,400 to 12,200 chromium concentration in juveniles (0.2 g weight) at pH of 6.5 to 7.8. For larger rainbow (25 g), LC_{50} 96hr test ranged from 20,200 to 65,500 chromium concentration at pH of 6.5 to 7.8. In salmon fingerlings, concentrations of 200ug/l resulted in LC 53 after 12 weeks.

In sublethal tests, concentrations of 16 to 21ug/l (ppb) chromium VI resulted in reduced growth in rainbow trout and chinook fingerlings after 14 to 16 weeks of exposure. Altered plasma cortisol methabolism occurred after seven days. In juvenile coho salmon, disease resistance and serum agglutinin production both decreased after two weeks in water with concentrations of 0.5 ppm chromium (Sugatt 1980b as reviewed in Eisler 1986). In coho smolts, salinity tolerance and serum osmolality were impaired during exposure to 0.23 ppm chromium VI for 4 weeks (Sugatt 1980a as reviewed in Eisler 1986). Studies on other endangered fish species have found these criteria to be protective (Buhl 1997). Two studies of sublethal effects conducted at concentrations lower than those proposed found behavioral avoidance as the response (Hartwell et al. 1989, Anestis and Neufeld 1986). After long-term (180 days) exposure to 0.2 ppm, rainbow trout had elevated chromium VI levels in kidneys (Review in Eisler 1986). At higher chromium VI concentrations (>2.0 ppm) chromium VI levels were highest in gill, liver, kidney, and digestive tract tissues of rainbow trout (Review in Eisler 1986).

Toxicity tests on salmonid species indicate that toxic effects occur at concentrations higher than those established by the Idaho Water Quality Standards. Therefore, EPA has determined that the approval of the **acute and chronic chromium (VI) criteria** (15 µg/L=acute, 10 µg/L=chronic) **is not likely to adversely affect Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

5. DDT

Background

The criteria established by the Idaho Water Quality Standards for DDT are 1.1 µg/L for acute exposures and 0.001 µg/L for chronic exposures. The legal use of this pesticide has been banned in the United States since 1972, however the chemical and its derivatives are persistent in both the environment and in organisms. The chemical may accumulate in fish tissues and be consumed by avian species. Birds will sequester DDT in their eggs which can result in shell thinning that is potentially lethal to their offspring. While DDT is still present in the environment, many species, such as bald eagles, have experienced a significant recovery.

Effects

Fish can accumulate DDT in their tissues and be affected by exposure to DDT in water and soils. Cutthroat trout are more sensitive to DDT than other salmonids, however much of the research on DDT has focused on other species, such as fathead minnows. Birds have been studied much more thoroughly with regards to the effects of DDT (EPA, 1980d).

Egg shell thinning has been reported for screech owls exposed to 2.8 mg/kg DDT in their diets, while dietary concentrations of 3.0 mg/kg caused egg shell thinning in mallards, black ducks, and sparrow hawks. As little as 0.5 mg/kg has been reported to cause thinning in brown pelican eggs. DDT at 200 mg/kg in diet caused death in mallards. Some variation in sensitivity has been observed between avian species, however, avian reproduction has been shown to be sensitive to DDT exposure (EPA 1980d).

It appears from the information available regarding the concentration levels of DDT that affect fish and birds, the acute criterion is not likely to adversely affect fish and birds. However, the chronic criterion may adversely affect threatened and endangered fish and bird species.

While EPA has determined that the approval of the chronic aquatic life criteria for DDT established by the Idaho Water Quality Standards may have the potential to adversely affect Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon and whooping crane, an added level of protection is offered by the following:

> The human health criteria for DDT is 0.00059 *u*g/L. This is the applicable DDT criterion in all surface waters of Idaho. These criteria are significantly lower and more conservative than the acute and chronic aquatic life criteria.

If a recreational use is modified or removed from a waterbody and the criterion become less stringent than 0.00059 *u*g/l for DDT, Idaho is required to submit this revision to EPA for approval/disapproval action. If EPA proposes to approve this revision, the agency will then reinitiate consultation on that approval action .

In light of these effective measures, EPA has determined that the approval of the **acute and chronic DDT criteria** (1.1 µg/L= acute, 0.001 µg/L = chronic) established by the Idaho Water Quality Standards is **not likely to adversely affect Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon and whooping crane.**

6. Endrin

Background

The acute and chronic criteria established by the Idaho Water Quality Standards for endrin are 0.18 µg/L and 0.0023 µg/L, respectively. Endrin is a pesticide that may be used against birds, rodents and insects. Its largest use is as an insecticide for cotton crops in southeast Mississippi (EPA, 1980f). The use of endrin has been restricted by EPA since the late 1970s.

Effects

Invertebrates tend to be more tolerant of endrin than fishes. For aquatic snails, LC_{50}s range from 73-12,000 µg/L (Hashimoto and Nishiuchi, 1981; Nishiuchi and Yashida, 1972; Trnkova, 1977), while LC_{50}s for salmonids range from 0.113-343.4 µg/L (Post and Schroeder, 1971; Katz and Chadwick, 1961; Katz, 1961; Bennett and Wolke, 1987a, 1987b; Wohlgemuth, 1977; Cope, 1965). The lower of these concentrations were nominal concentrations only, not measured concentrations. Therefore, the accuracy of the estimate of the concentration is not assured. A number of other studies performed at the same time and more recently show effects at concentrations much higher than the criteria. When food contaminated with endrin is fed to aquatic species, the toxicity of waterborne endrin is greater than its toxicity when food items are uncontaminated. The contribution of food-borne endrin to the total body burden is only 10-15% with the rest contributed by waterborne endrin. Residues contributed by food-borne endrin are also additive to those contributed by water (Jarvinen and Tyo, 1978). Bioconcentration factors (BCF) for fish range from 1,640-15,000 and rapid equilibrium with water concentrations has been demonstrated (EPA, 1980f). Some acute effects have been shown to occur near the criteria, but the interaction of dietary toxicity makes these studies hard to interpret. Since no other evidence shows endrin toxicity at levels higher than the criteria EPA has determined that the approval of the **acute and chronic endrin criteria** (0.18 µg/L = acute, 0.0023 µg/L = chronic) **is not likely to adverse affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

7. Heptachlor

Background

The acute and chronic criteria established by the Idaho Water Quality Standards for heptachlor are 0.52 µg/L and 0.0038 µg/L, respectively. Heptachlor is a broad spectrum insecticide that was used commonly for crop pest control until 1976, when it was prohibited from home and agricultural use. Commercial applications to control insects, such as termites, continued. Chemically, heptachlor is part of the cyclodiene insecticides. Its principal metabolite is heptachlor epoxide, which may be formed from metabolism in soil, water, and plant and animal tissues, and heptachlor epoxide is not known to be more toxic than heptachlor. Heptachlor is persistent in aquatic systems and accumulates in plant and animal tissues (EPA, 1980g).

Effects

Sublethal effects of heptachlor have only been evaluated in a few species. Inhibition of ATP-ase activity (Na and Mg) have been found to occur in rainbow trout at concentrations between 3,735-37,350 µg/L after 15 minutes. After 171 days of dietary exposure to heptachlor at 5-25 mg/kg/day, bluegill experienced a decrease in growth. Bioconcentration factors found in fathead minnows were 9,500-14,400 (EPA, 1980g).

Lethal effects are documented for many salmonids, but not for freshwater gastropods. In coho salmon, an LC_{50} of 81.9 µg/L was found, while the LC_{50} for chinook salmon was 24.0 µg/L. The LC_{50} for rainbow trout was determined to be 10-26.9 µg/L. A chronic study that exposed bluegill to 69.4 µg/L heptachlor for 171 days resulted in >90% mortality, but no effects on growth or reproduction in those fish that survived (EPA, 1980g).

From the small amount of data available detailing the effects of heptachlor on aquatic species, and the evaluation in the criteria document (EPA 1980s) it appears that concentrations much higher than the acute and chronic criteria established by the Idaho Water Quality Standards are required to adversely affect aquatic organisms. Therefore, from the information available, EPA has determined that the approval of the **acute and chronic heptachlor criteria** (0.52 µg/L = acute, 0.0038 µg/L = chronic) **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

8. Lindane (gamma-BHC)

Background

The acute and chronic criteria established by the Idaho Water Quality Standards for lindane are 2 µg/L and 0.08 µg/L, respectively. Lindane is one of the few chlorinated hydrocarbon insecticides still in use for agricultural purposes. This chemical is relatively persistent and experiences significant degradation only under anaerobic conditions (Brooks, 1972; Nash and Woolson, 1967). However, biological accumulation and persistence of lindane is low when compared to compounds such as DDT or dieldrin (Wilson, 1965; Gakstatter and Weiss, 1967).

Effects

In rainbow trout, the 96-hour LC_{50} was 22 µg/L; however, in brook trout, effect levels were much higher. After 261 days exposed to 16.6 µg/L lindane, brook trout survival was not affected, but a reduction in weight and length was observed. Some disruption in reproductive activity was also recorded during the same experiment (Macek et al., 1976). From the available information, EPA has determined that the approval of the **acute and chronic lindane criteria** (2 µg/L = acute, 0.08 µg/L chronic) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

9. Nickel

Background

The toxicity of nickel depends on water hardness, therefore the criteria for nickel established by the Idaho Water Quality Standards is also dependent upon water hardness. At a hardness of 100 mg/L, the acute criterion for nickel is 1,400 µg/L, and the chronic criterion is 160 µg/L. The corresponding total recoverable criteria are the same.

Nickel occurs naturally in rocks and soils and can leach into aquatic environments. However, weathering of nickel-containing substrates results in only small amounts of nickel entering into aquatic systems. Manmade sources of nickel include mining, combustion of coal, petroleum and tobacco, manufacture of cement and asbestos, food processing, textile and fur fabrication, laundries, and car washes (EPA, 1981).

Effects

Invertebrates have been affected by long-term exposure to nickel at concentrations as low as 0.5 mg/L, while chronic effects for fish have been reported in soft water at 2 mg/L or higher. However, aquatic species exposed to nickel in ambient waters are typically at low risk. Short term exposures do not appear to be harmful to aquatic organisms (EPA, 1981). Therefore, EPA has determined that the approval of the **acute and chronic nickel criteria** (1,400 µg/L = acute, 160 µg/L = chronic, hardness of 100 mg/L) established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

10. PCBs

Background

The criterion for PCBs established by the Idaho Water Quality Standards is 0.014 µg/L for chronic exposures. There is no chronic criterion for PCBs. PCBs are halogenated aromatic hydrocarbons that are generally used in products such as heat transfer agents, dielectric agents, flame retardants, plasticizers, and waterproofing materials (Roberts et al., 1978). Environmental contamination with PCBs has resulted from industrial and domestic discharges, landfills, equipment dumps, and through atmospheric transport of incompletely incinerated PCBs. These chemicals are no longer produced in the United States since a 1979 ban on the manufacture, processing and distribution in commerce and use of PCBs except in a totally enclosed system. Under environmental conditions, PCBs are extremely stable and slow to chemically degrade (Eisler, 1986b). Therefore, even though the chemicals have been banned from production, problems still occur due to historical discharges and contaminated sediments, not from current permitted discharges.

Effects

Fish exhibit great interspecies differences in their responses to PCBs. Concentrations of 1.4 ppm in the gonads of striped bass have been associated with poor reproductive success (Ray et al., 1984), while 2.8 ppm in the eggs of rainbow trout resulted in heavy fry mortality (Rohrer et al., 1982). Sublethal effects for fish include skin lesions, immunotoxicity, reproductive toxicity, genotoxic and epigenetic effects and hepatomegaly and related liver damage. Fish tend to accumulate PCBs from their diet and retain them for long periods of time. Measurable sublethal effects have been observed at concentrations ranging from 0.4-15 µg/L (EPA, 1980h).

Therefore, EPA has determined that the approval of Idaho's chronic criterion (0.014 µg/L) for PCBs is not likely to adversely affect Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

Dietary exposure to PCBs causes a variety of adverse effects in avian species as well. Concentrations as low as 1 mg/kg can cause diarrhea and liver histopathology in chickens after 8 weeks of exposure, while 50 mg/kg can result in a 50% reduction in egg hatch. Japanese quail have experienced mortality after 5 days exposed to 3,850 mg/kg, but survival was not affected after exposure to 3,100 mg/kg. Eisler contends that any concentration greater than 1.0 mg/kg is evidence of environmental contamination (Eisler, 1986b). Due to the highly bioaccumulative nature of PCBs, it is likely that dietary concentrations that birds might encounter in fish would exceed Eisler's (1986b) recommended safe dietary levels for avian species. Therefore, EPA has determined that the chronic criteria for PCBs is likely to adversely affect bald eagle, peregrine falcon and whooping crane.

While EPA has determined that the chronic criteria may have the potential to adversely affect bald eagle, peregrine falcon, and whooping crane, an added level of protection for these species is offered by the following:

> The human health criteria for PCB's is 0.000044 µg/L and 0.000045 µg/L. These are the applicable PCB criteria in all surface waters of Idaho. These criteria are significantly lower and more conservative than the acute and chronic aquatic life criteria.

If a recreational use is modified or removed from a waterbody and the criteria become less stringent than 0.000044 or 0.000045 *u*g/L for PCB's, Idaho is required to submit this revision to EPA for approval/disapproval action. If EPA proposes to approve this revision, the Agency will then reinitiate consultation on that approval action .

In light of the information and measures described above and the effective human health criteria, EPA has determined that the approval of the **chronic PCB criterion** (0.014 µg/L) established by the Idaho Water Quality Standards is **not likely to adversely affect the bald eagle, peregrine falcon, and whooping crane**.

11. Pentachlorophenol

Background

The criteria for pentachlorophenol established by the Idaho Water Quality Standards are pH dependent. At a pH of 7.8, the criteria are 20 µg/L for acute exposures and 13 µg/L for chronic exposures. The toxicity of PCP increases with decreasing pH. However, since PCP is rarely present in pure form, accurate measurement is difficult. This, in turn, raises questions regarding PCP toxicity tests and the criteria.

Pentachlorophenol (PCP) is a synthetic organochlorine compound used primarily as a wood preservative, but also secondarily as an herbicide, insecticide, fungicide, molluscicide, and bactericide (Eisler, 1989). PCP can affect metabolism in animals and plants by impairing the production of adenosine triphosphate (ATP) and altering liver enzymes. One response to this impairment is increased basal metabolism, resulting in increased oxygen consumption and high fat utilization. The effects of PCP may reduce the availability of energy for maintenance and growth, thus reducing survival of larval fish and ability of prey to escape from a predator (Johansen et al., 1987; Brown et al., 1985; Eisler, 1989).

Effects

Eisler (1989) reviewed the effects of PCP on invertebrates' growth, survival, and reproduction at levels of 3-100 µg/L. Fish are affected at concentrations from 1-68 µg/L, while birds are affected at dietary concentrations greater than 3,580 mg/kg. Chronic values for rainbow trout are 5.67-14.46 µg/L at pH values of 6.5-7.4. However, concentrations as low as 0.035-1 µg/L have been correlated with elevated tissue residues in rainbow trout. A 96-hour LC_{50} was determined for carp larvae at 9.5 µg/L at a pH of 7.2 (Eisler, 1989).

Due to the effect of pH on the toxicity of PCP, it is difficult to compare the effects levels from research studies. A review of the literature with the criteria converted for the pH values, reveals that some sublethal effects may occur during long-term exposures below the criteria (EPA 1986a). Juvenile sockeye salmon experienced decreased growth rates and conversion efficiencies at PCP concentrations of 1.74-1.8 µg/L at pH conditions (between 7.0-9.0) where the chronic criterion was 4.7 µg/L (Webb and Brett, 1973). Hodson and Blunt (1981) also observed reduced weight, growth rate, and biomass in rainbow trout exposed over 4 weeks from embryo to fry stages. Some mortality of rainbow trout eggs has also been observed at levels below the chronic criterion when dissolved oxygen fell to low levels of 3-5 mg/L (Chapman and Shumway, 1978). Since mortality occurred at dissolved oxygen levels that would not be present in waters in attainment of the Idaho Water Quality Standards, these lethal effects should not be seen at conditions put forth under this action. In the previous evaluation of the aquatic criteria, EPA considered the studies cited here where PCP affected fish sublethally at concentrations below the chronic criterion.

Based upon the studies described above, EPA has determined that the acute aquatic life criterion (20 µg/L=acute) for PCPs is not likely to adversely affect the Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout. However, the chronic aquatic life criterion for PCP is likely to adversely affect the Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, and bull trout.

The highest BCF for fish referenced by Eiser (1989) is 1,000, which would result in a dietary exposure for birds of 13 mg/kg. The lowest dietary level of pentachlorophenol shown to effects birds is 1 mg/kg (Eisler, 1990). Therefore, EPA has determined that the acute criterion for PCP is not likely to adversely affect bald eagle, peregrine falcon and whooping crane. However, the chronic criterion is likely to adversely affect bald eagle, peregrine falcon and whooping crane.

While EPA has determined that the approval of the chronic aquatic life criteria for PCP's established by the Idaho Water Quality Standards may have the potential to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon and whooping crane, an added level of protection for these species is offered by the following:

> The human health criteria for PCP's is 0.28 µg/L and 8.2 µg/L. These are the applicable PCP criteria in all surface waters of Idaho. These criteria are significantly lower and more conservative than the acute and chronic aquatic life criteria.

If a recreational use is modified or removed from a waterbody and the criteria become less stringent than 0.28 and 8.2 *u*g/L for PCP's, Idaho is required to submit this revision to EPA for approval/disapproval action. If EPA proposes to approve this revision, the Agency will then reinitiate consultation on that approval action .

In light of the information and measures described above and the effective human health criteria, **EPA has determined that the approval of the acute and chronic PCP criteria (20 µg/L = acute, 13 µg/L = chronic, pH of 7.8) established by the Idaho Water Quality Standards is not likely to adversely affect the Kootenai River white sturgeon, Snake River sockeye and chinook salmon, Snake River steelhead, bull trout bald eagle, peregrine falcon and whooping crane.**

12. Silver

Background

The dissolved acute criterion established by the Idaho Water Quality Standards for silver is 3.4 µg/L, at a hardness of 100 mg/L $CaCO_3$. The corresponding total recoverable criterion is 2.4 µg/L for acute exposures. No chronic criterion is set. The toxicity of silver is affected minimally by hardness.

Anthropogenic sources of silver in surface waters include industrial and smelting wastes, wastes in jewelry manufacture, or electrical supply and, primarily, the production and disposal of photographic material (EPA, 1987a).

Effects

LC_{50} values for rainbow trout larvae range from 11.8-280 µg/L due to hardness differences. For juvenile rainbow trout, LC_{50}s range 8.5-84.4 µg/L (EPA, 1987a). Ionic silver is highly toxic to fish at very low concentrations (Hogstrand and Wood 1998). Acute toxicity, caused by the interference of ionic silver with Na^+ and Cl^- transport at the gills, appears to be exclusively from ionic silver (Hogstrand and Wood 1998). Most of the acute toxicity studies have used silver nitrate ($AgNO_3$) which is a highly soluble and therefore a highly toxic form of silver. These tests show acute toxicity at low concentrations (96hr LC_{50} is 5 to 70 µg/l total silver) based on review by Hogstrand and Wood (1998). Because ionic silver is rare in the environment, these tests have questionable relevance (Hogstrand and Wood 1998). The more common forms of silver, silver thionsulfate, and silver chloride, are bioavailable but do not appear to contribute to acute toxicity, (low to moderate toxicity). In bioassays based on Ag^+, the 168-hr LC_{50} was 3.2 µg/l regardless of total silver quantity (Hogstrand and Wood 1998).

In a review of the chronic toxicity literature for silver, the maximum acceptable concentrations were <0.5 µg/l (Hogstrand and Wood 1998). Again, these tests used $AgNO_{3,}$ yielding low toxicity thresholds. As with acute toxicity, the presence of sulfide and thiosulfate complexation with silver reduced silver toxicity.

Silver toxicity to fish is affected by the amount of reducing agents available. The reduction of silver by chloride, dissolved organic carbon, and sulfide are important for reducing silver toxicity. Although water hardness does affect the toxicity of silver, the degree of protection from hardness is minor. From this information, EPA has determined that the approval of the **acute and chronic silver criteria** (acute = 3.4 µg/L, hardness of 100 mg/L $CaCO_3$)established by the Idaho Water Quality Standards **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

13. Toxaphene

Background

The acute and chronic criteria established by the Idaho Water Quality Standards for toxaphene are 0.73 µg/L and 0.0002 µg/L, respectively. Toxaphene is a broad spectrum insecticide whose registration was canceled by EPA in 1982. While toxaphene degrades more rapidly than other chlorinated pesticides (Matsumura, 1978), biomagnification in aquatic systems has been demonstrated even when toxaphene was not detectable in water or sediment (Niethammer et al., 1984). The bioconcentration factor (BCF) measured for brook trout is 10,000 (Eisler and Jacknow, 1985).

Effects

Acute toxicity studies on fish show effects levels well above the criteria. For example, acute effects were observed at 2 µg/L in bass (Johnson and Finley, 1980), 2.4-29 µg/L in bluegill (Johnson and Finley, 1980; EPA, 1980j; Isensee et al., 1979), 3.1 µg/L in brown trout (Johnson and Finley, 1980) and 18.0 µg/L in fathead minnows (Johnson and Finley, 1980). Sublethal effects such as reduced reproduction (Sanders, 1980; Mayer et al., 1975), growth inhibition (Mayer and

Mehrle, 1977) and histopathology of the kidney and intestinal tract (Pollock and Kilgore, 1978) have been observed in fish at concentrations as low as 0.054 µg/L.

Avian species can readily metabolize and excrete toxaphene with little accumulation in the tissues (Eisler and Jacknow, 1985). In a long term study (19 months), American black ducks were fed 10-50 mg/kg toxaphene with no significant effects on survival, egg production, fertility, hatchability, eggshell thickness, or growth and survival of young (Eisler and Jacknow, 1985). In a lifetime study, chicken were not affected by toxaphene in their diet at concentrations as high as 3.8-5 mg/kg/day (Eisler and Jacknow, 1985). Eisler and Jacknow (1985) estimates that 3 mg/kg is biologically insignificant to fish-eating birds.

Therefore, based on the information available and the fact that EPA has canceled the registration for toxaphene, EPA has determined that the approval of the **acute and chronic toxaphene criteria** (0.73 µg/L = acute, 0.0002 µg/L = chronic) **is not likely to adversely affect the Snake River sockeye and chinook salmon, Snake River steelhead, bull trout, Kootenai River white sturgeon**, **bald eagle, peregrine falcon, and whooping crane**.

V. SUMMARY OF DETERMINATIONS

A. Background

The analyses for the protectiveness of numeric criteria assume that the organisms are exposed to concentrations of pollutants at the water quality criterion, not the conditions which currently exist in Idaho's waters. As discussed previously (Overview of Idaho's Water Quality Programs), approximately 10% of the surface waters in Idaho are listed on the 1996 303(d) list for not being in attainment of the Idaho Water Quality Standards. The other 90% of the waters are either in attainment of the standards or have not been recently monitored. For waters in non-attainment, the State of Idaho and EPA are undertaking control actions to bring the waterbodies into compliance with the standards. However, due to the scale of this action and the temporal and spacial variability in water quality conditions throughout the state, this assessment will only analyze potential effects at the criteria concentrations. EPA realizes that the analysis was conservative on the side of the species for the majority of the state's waters which contain pollutant concentrations well below the criterion level and, where waters are not currently in attainment but where actions are in place to remedy current water quality problems, the analysis described desired future conditions and thus underestimated potential current effects on the species of concern.

B. Determinations

The following determinations of "not likely to adversely affect" were made:

Aldrin/Dieldrin, Chlordane, Chromium III and VI, DDT, Endrin, Heptachlor, Lindane, Nickel, PCBs, Pentachlorophenol, Silver, Toxaphene: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, whooping crane, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Arsenic Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Arsenic Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bull trout, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Cadmium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Copper Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Cyanide Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Endosulfan Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Lead Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, whooping crane, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Mercury Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Mercury Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute Selenium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Chronic Selenium Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

Acute and Chronic Zinc Criteria: Bliss Rapids snail, Banbury Springs lanx, Snake River physa snail, Idaho springsnail, Bruneau hot springsnail, Utah valvata snail, Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, bald eagle, peregrine falcon, whooping crane, gray wolf, grizzly bear, lynx, Northern Idaho ground squirrel, woodland caribou, water howellia, MacFarlane's four o'clock, Ute ladies' tresses, and Spalding's catchfly.

The following determinations of "likely to adversely affect" were made:

Chronic Mercury Criteria: Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead, Kootenai River white sturgeon, peregrine falcon, bald eagle, and whooping crane.

Chronic Selenium Criteria: Kootenai River white sturgeon, bull trout, Snake River sockeye salmon, Snake River spring/summer chinook salmon, Snake River fall chinook salmon, Snake River steelhead and Kootenai River white sturgeon, peregrine falcon, bald eagle, and whooping crane.

VI. ANALYSIS OF EFFECTS OF TOXIC POLLUTANTS TO CHINOOK AND SOCKEYE SALMON CRITICAL HABITAT

A. Description of Salmon Critical Habitat

NMFS has designated critical habitat in Idaho for Snake River spring/summer chinook salmon, Snake River fall chinook salmon and Snake River sockeye salmon. As required by Section 7 of the ESA and the implementing regulations at 50 CFR Part 402, EPA has used the best available scientific data to determine whether the action is likely to "destroy or adversely modify the designated critical habitat of the listed species". The consultation regulations define the statutory term "destruction or adverse modification" of critical habitat to mean:

...a direct or indirect alteration that appreciably diminishes the value of critical habitat for both the survival and recovery of a listed species. Such alterations include, but are not limited to, alterations adversely modifying any of those physical or biological features that were the basis for determining the habitat to be critical.

The Federal Register (Vol 58 No. 247, December 28, 1993) final rule designates critical habitat and defines and describes habitat and its essential features as follows:

Essential Snake River salmon habitat for both chinook and sockeye consists of four components: 1) spawning and juvenile rearing areas, 2) juvenile migration corridors, 3) areas for growth and development to adulthood, and 4) adult migration corridors.

Spawning and rearing areas:

The essential features of the spawning and juvenile rearing areas of the designated critical habitat for Snake River sockeye salmon consist of adequate: 1) spawning gravel, 2) *water quality, 3) water quantity, 4) water temperature, 5) food, 6) riparian vegetation, and 7) access.*

The essential features of the spawning and juvenile rearing areas of the designated critical habitat for Snake River spring/summer and fall chinook salmon are: 1) spawning gravel, 2) water quality, 3) water quantity, 4) water temperature, 5) instream cover/shelter, 6) food for juvenile salmon, 7) riparian vegetation, and 8) living space.

Migration corridors:

Essential features of the juvenile migration corridors for Snake River sockeye salmon and Snake River spring/summer and fall chinook salmon consist of adequate: 1) substrate, 2) water quality, 3) water quantity, 4) water temperature, 5) water velocity, 6) cover/shelter, 7) food, 8) riparian vegetation, 9) space, and 10) safe passage conditions.

Essential features of the adult migration corridors for Snake River sockeye salmon and Snake River spring/summer and fall chinook salmon include adequate: 1) substrate, 2) water quality, 3) water quantity, 4) water temperature, 5) water velocity, 6) cover/shelter, 7) riparian vegetation, 8) space, and 9) safe passage conditions.

Growth and Development:

The areas in the Pacific Ocean that threatened and endangered salmon use for growth and development are not well understood; therefore, NMFS has not designated any essential areas and features for Snake River ocean habitat.

B. Analysis of Effects of Numeric Criteria for Toxic Pollutants to Listed Critical Habitat

To determine whether EPA's approval of Idaho's numeric criteria for toxic pollutants is likely to adversely affect critical habitat, EPA has identified possible threats to the essential features of habitat. In evaluating the effects of the action on critical habitat, EPA concluded that the water quality parameters considered in this consultation are an integral part of all the species' habitats. Therefore, the analysis of effects to the species relates directly to their habitats. Section III. of this document presents information describing the analysis of effects of specific water quality criteria to Snake River salmon.

Water quality standards characterize and define the conditions and quality of surface waters. Because there are essential features of salmon critical habitat which are related to the conditions of the aquatic environment, EPA's approval of Idaho's water quality standards may directly and/or indirectly affect water quality related essential features of salmon habitat. Water quality may affect the following essential features of critical habitat: spawning gravel, water quality, water temperature, and food. EPA evaluated whether the water quality criteria may affect the condition/quality of the essential features and/or whether the water quality criteria may affect the presence/absence of these essential features of habitat.

Water quality should not affect the following essential features of critical habitat: water quantity, riparian vegetation, access, instream cover/shelter, space, safe passage conditions, water velocity and substrate. Therefore, **EPA's approval of Idaho's numeric criteria for toxic pollutants addressed in this biological assessment is not likely to adversely affect these essential features of critical habitat of Snake River salmon.**

Based on the available information, this analysis indicated that the chronic mercury criterion and chronic selenium criterion may have the potential to adversely affect Snake River salmon. Because these criteria set the allowable concentrations of these pollutants in surface waters in Idaho, EPA has determined that the approval of these criteria may have the potential to affect water quality and food in critical habitat.

The effect of consuming contaminated food is discussed is the biomagnification/bioaccumulation section for each water quality criteria. The decline of prey due to exposure to toxic chemicals results in an impact in the growth, reproduction and survival of prey species. The effect of the decline on individual prey species on food resources is unknown. Without this information EPA is unable to determine whether this may have the potential to adversely affect food as an essential feature of critical habitat.

Research does document mercury and selenium biomagnification in aquatic food chains (Lemly and Smith, 1987; Lemly, 1985; Wren and MacCrimmon, 1986). Therefore, Snake River salmon may encounter harmful concentrations of mercury and selenium through biomagnification of these chemicals through prey. However, the efficiency of metal transfer through macroinvertebrates may not allow absorption of metal concentrations high enough to harm the fish (Reinfelder and Fisher, 1994). No evidence has been found describing effects to salmon through biomagnification of mercury and selenium in the food.

Effects of water quality on food may also include toxic effects of pollutants on prey species. The analysis included in this assessment (see Section III.) has determined that the concentrations of toxic pollutants allowed by the Idaho Water Quality Standards are not sufficient to threaten the prey base for salmonids. Therefore EPA's approval of Idaho's numeric toxic criteria is not likely to adversely affect the quality and/or availability of food as an essential feature of juvenile salmon spawning, rearing and migration corridors. Consequently, **EPA has determined that the approval of the chronic criterion for mercury and chronic criterion for selenium is not likely to adversely affect the designated critical habitat of the Snake River sockeye salmon, Snake River spring/summer chinook salmon, and Snake River fall chinook salmon.**

Although the above analysis indicates that Idaho's chronic criteria for mercury and selenium may have the potential to affect water quality and food as essential features of critical habitat, these effects alone would not be significant enough to appreciably diminish the value of critical habitat for both the survival and recovery of Snake River salmon.

The analysis in Section III., indicated that all remaining numeric toxic criteria which were evaluated were not likely to adversely affect Snake River salmon. **Therefore, these remaining criteria are not likely to adversely affect water quality or food as essential features of critical habitat of Snake River salmon.**

C. Summary of Determination of Effects to Listed Critical Habitat

While the above analysis indicates EPA's approval of the chronic criteria for mercury and selenium of these provisions may have the potential to have adverse effects on Snake River sockeye salmon, Snake River spring/summer chinook salmon, and Snake River fall chinook habitat, the constituent elements of critical habitat likely will not be altered or destroyed to the extent that the survival and recovery of the species would be appreciably reduced. Although the potential may exist for some elements of critical habitat to be adversely affected, other elements are not likely to be affected. Consequently, these effects are not likely to "result in significant adverse effects throughout the species' range or appreciably diminish the capability of the critical habitat to satisfy essential requirements of the species". Therefore, **EPA has determined that the approval of these provisions is not likely to destroy or cause an adverse modification to designated critical habitat of the Snake River sockeye, Snake River spring/summer chinook salmon, and Snake River fall chinook salmon.**

VII. CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, Tribal, local or private actions on endangered or threatened species or critical habitat that are reasonably certain to occur in the action area considered in this biological assessment. Future federal actions or actions on federal lands that are not related to the proposed action are not considered in this section .

Future anticipated non-Federal actions that may occur in or near surface waters in the State of Idaho include timber harvest, grazing, mining, agricultural practices, urban development, municipal and industrial wastewater discharges, road building, sand and gravel operations, introduction of nonnative fishes, off-road vehicle use, fishing, hiking, and camping. These non-Federal actions are likely to continue having adverse effects on the endangered and threatened species.

There are also non-Federal actions likely to occur in or near surface waters in the State of Idaho which are likely to have beneficial effects on the endangered and threatened species. These include implementation of riparian improvement measures, best management practices associated with timber harvest, grazing, agricultural activities, urban development, road building and abandonment and recreational activities, and other nonpoint source pollution controls.

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IX. APPENDICES