Evaluation of Proposed New Point Source Discharges to a Special Resource Water and Mixing Zone Determinations: Thompson Creek Mine facility, Upper Salmon River Subbasin, Idaho

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EVALUATION OF PROPOSED NEW POINT SOURCE DISCHARGES TO A SPECIAL RESOURCE WATER
AND
MIXING ZONE DETERMINATIONS: THOMPSON CREEK MINE, UPPER SALMON RIVER SUBBASIN, IDAHO

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Summary

The purpose of this report is to evaluate the effects on aquatic life of, and to establish conditions for, ongoing and proposed discharges from the Thompson Creek Mine, operated by the Thompson Creek Mining Company ("the mine" or TCMC), in Custer County, Idaho. The evaluation and conditions are based upon compliance with Idaho's water quality standards, which include several narrative requirements, numeric criteria, and the protection of designated uses.

The mine currently discharges runoff into Thompson Creek and Squaw Creek. TCMC proposes a new discharge to Squaw Creek and the Salmon River. They have requested IDEQ authorize mixing zones, small areas where not all water quality standards are met as their effluents mix with the streams. Further, the Salmon River is designated as a "special resource water" for which no reduction in ambient water quality below the mixing zone may occur. This report sets an operating definition for determining a safe reduction in water quality that can be monitored. Based upon what changes in ambient water quality would be expected to have adverse effects and upon the limits of measurable changes, restrictions on increased discharges to the Salmon River were developed. For this situation, 25% of the difference between upstream metals concentrations, and the most stringent water quality standard (i.e. 25% of the assimilative capacity of the stream) is the limit on the combined increased discharges from the Thompson Creek Mine to the Salmon River. This difference remains below the threshold of adverse effects to aquatic life and is at the lower practical limit of measurable change.

Mixing zones are small areas where discharges mix with the receiving waters, and all water quality standards do not have to be met. Mixing zones are predicated on the assumption that the Clean Water Act and state requirements are intended to apply to streams, rather than to pipes. The practical effect of mixing zones is that without them, water quality standards would have to be met in discharge pipes, not after mixing with and being diluted by the receiving waters. This would result in discharge limits many times more stringent than would result if compliance were evaluated after the effluents were diluted with receiving water. Whether that scenario would be overprotective or appropriate depends upon the assimilative capacity of the waters that receive the discharges. If effects are limited, physical sizes are small, and the mixing zones do not jeopardize the integrity of the rest of the water body, as defined by Idaho Water Quality Standards (WQS) and EPA guidance, they will be considered acceptable. Otherwise, the discharges would need to be reduced.

The mixing zone analyses included evaluations of site and regional water and sediment chemistry, biological conditions in the receiving waters, whole effluent toxicity testing, potential fish avoidance around the mixing zones (zone of passage), risk of adverse bioaccumulative effects of mercury and selenium, relative flows of effluents and receiving waters, variations of flow by width and depth within the receiving waters, and extensive hydrodynamic modeling of effluent plume dispersion and dilution under varying flow and pollutant scenarios. The predicted areas and frequencies of potential adverse effects were compared with the overall sizes of the water bodies and expected habitat ranges of aquatic and semi-aquatic life.

After evaluating this information, mixing zone dimensions were determined that ranged from 3 – 10 meters downstream of the outfalls for the zones of initial dilution where acute water quality standards do not have to be met. The overall size of the mixing zones, below which all water quality standards must be met, were set at 50 – 200 meters downstream of the various outfalls. The fraction of critical flow conditions that may be used for calculating permit limits ranges from 0% to 66%, depending upon flow and the pollutant in question. Critical flow conditions would be exceeded 99.6% of the time in un-regulated runoff outfalls to Thompson Creek (e.g. snowmelt and rainfall), and critical flows would be exceeded 99.8% of the time in regulated effluents discharged through pipelines and diffusers into Squaw Creek and the Salmon River.

Existing and Proposed Discharges

The Thompson Creek Mine, operated by the Thompson Creek Mining Company, is a large open pit molybdenum mining operation located in the Salmon River Mountains, Custer County, Idaho. The operation accounts for about 8% of the world supply of molybdenum. Hydrologically, it is located in the Upper Salmon Hydrologic Unit Catalog (HUC) 17060201, also known as the Salmon River subbasin. The mine currently is permitted to discharge surface runoff and process water to three discharge locations; two additional discharge outfalls are proposed (Figure 1):

Existing Outfall 1 (Buckskin Creek) and existing Outfall 2 (Pat Hughes Creek) continuously drain natural runoff and seepage water downhill of large waste rock/overburden piles in their respective drainages into Thompson Creek.

Existing Outfall 3 (Bruno Creek) collects runoff from the mine access road and the diverted natural flow of upper Bruno Creek and discharges into Squaw Creek. No process water or mine runoff is received through Outfall 003 and these discharges are not analyzed further in this report.

Outfall 004 would be carried by a pipeline and discharged to Squaw Creek through a diffuser. Outfall 004 will consist of mostly uncontaminated spring water from the left abutment of the tailings dam, and a small amount of slightly contaminated water from the "pumpback station." The pumpback station is located downstream of the tailings dam and is used to pump seepage water that escapes the dam back to the dam, where it is re-used in the mill. However, if the mill does not operate for extended periods, this water will need to be released.

Outfall 005 will also be made up from left abutment water, pumpback station water, and water pumped from the open pit. TCMC recently amended their permit application to also discharge water from outfalls 001 and 002 through outfall 005. This option, if implemented, would entail building a pipeline along the Thompson Creek road, intercepting and diverting Buckskin and Pat Hughes Creeks into a pipeline, and discharging them through a diffuser into the Salmon River.

Regulatory Classification and Status of Receiving Waters

Protected uses designated for Thompson and Squaw Creeks include salmonid spawning, cold water biota, agricultural water supply, and secondary contact recreation. Protected uses for the Salmon River include these uses plus domestic water supply and primary contact recreation. Because of the sensitivity of the aquatic life uses to constituents in the discharges, the analysis is focused upon protecting these uses.

The Salmon River is further classified as a Special Resource Water. New or increased discharge of pollutants into Special Resource Waters is prohibited if pollutants significant to the designated beneficial uses will result in a reduction of the ambient water quality of the receiving water as measured immediately below the applicable mixing zone (IDAPA 16.01.02.130, 16.01.02.400[1]).

In the 1998 Idaho Water Quality Limited List, Thompson Creek and the Salmon River in the vicinity of the study area were listed as follows:

Water body	Boundaries	Listed Pollutant
Salmon River	Hellroaring Creek to East Fork Salmon R. (includes section in study area)	Sediment, temperature
Thompson Creek	Old Schellite Mill site to Salmon River (located about 1 mile upstream from the mouth, about 3 miles downstream of Outfall 002)	Metals, Sediment

Waters identified as water quality limited because of violation of Idaho water quality standards, or failure to fully support beneficial uses, require the development of total maximum daily loads (TMDLs) or equivalent processes to remedy the impairment. TMDLs are being developed for for each subbasin with water quality limited streams . The Upper Salmon subbasin, hydrologic unit code 17060201, is scheduled for TMDL development by 2001. In the interim, DEQ shall require changes in permitted point sources and nonpoint best management practices necessary to prevent further degradation of beneficial uses. This is referred to as a “no net increase policy” (WQS §054).

Critical habitats for threatened or endangered fish species and mixing zone determinations

The Salmon River, Squaw Creek, and Thompson Creek are all included in the definition of critical habitat for the protection of threatened Snake River spring/summer salmon populations (NOAA 1993) and steelhead trout (NOAA 2000), and are within bull trout key watersheds identified in Idaho’s Bull Trout Conservation Plan (State of Idaho 1996). These habitat protection programs require protection, or restoration, of necessary habitat features and water quality to protect these species.

According to EPA guidance, in no case may a mixing zone be granted that would likely jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of such species’ critical habitat (EPA et al. 1999). An

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[1] Henceforth, citations to water quality standards will be abbreviated “WQS 130”, WQS 400, and so on. All water quality standards are contained in chapter 16.01.02, which is henceforth dropped for brevity.

"adverse modification" is defined as a direct or indirect action that appreciably diminishes the value of critical habitat for both the survival and recovery of a listed species. The determination of jeopardy or adverse modification is based on the effects of the action on the continued existence of the entire population of the listed species or on a listed population, and/or the effect on critical habitat as designated in a final rulemaking. When multiple units of critical habitat are designated for particular purposes, these units may serve as the basis of the analysis if protection of different facets of the species' life cycle or its distribution is essential to both its survival and recovery. Adverse effects on individuals of a species or constituent elements or segments of critical habitat generally do not result in jeopardy or adverse modification determinations unless that loss, when added to the environmental baseline, is likely to result in significant adverse effects throughout the species' range, or appreciably diminishes the capability of the critical habitat to satisfy essential requirements of the species. Modification or destruction of designated critical habitat that does not reach this threshold is not prohibited by section 7 of the Endangered Species Act (USFWS and NOAA 1998).

Section 9 of the Endangered Species Act prohibits "taking" an endangered species, the definition of which includes harming individual organisms. "Harm" to a habitat means a significant modification or degradation which actually kills or injures wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, and sheltering (NOAA 2000).

In this analysis, the risk of behavioral disruption to migratory salmonids is the most significant potential for harm as a result of the proposed mixing zones, because the spatial extent of the potential effects is relatively large, and because access to habitats upstream of the zones could be affected, especially in the Salmon River. This risk is considered in detail in this report.

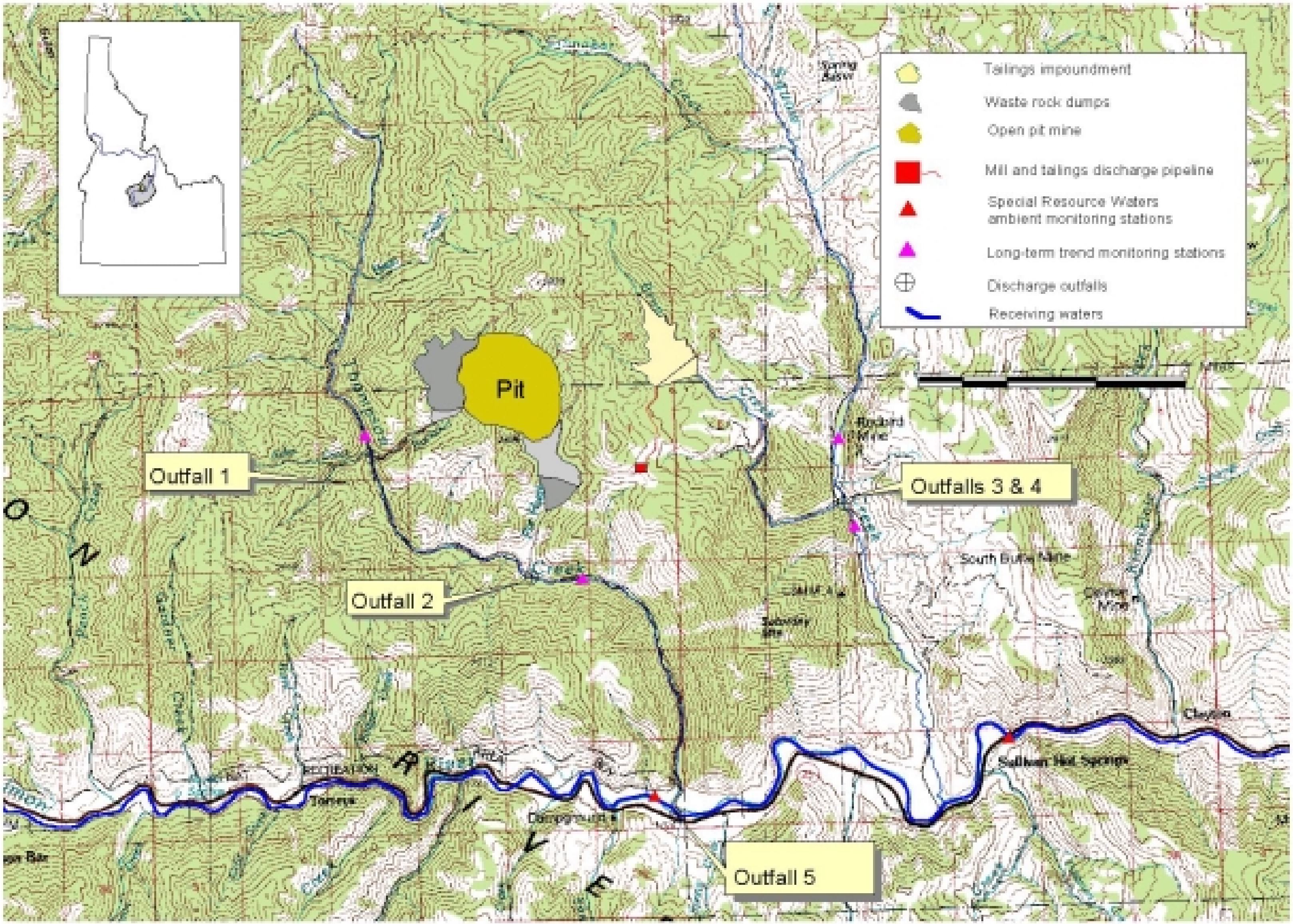

Figure 1. The Thompson Creek Mine facility, located in the Thompson and Squaw Creek watersheds of the Upper Salmon River subbasin, near Clayton, Idaho. Feature locations are approximate. Inset shows the Thompson and Squaw Creek watersheds, the Upper Salmon River subbasin, and the Salmon River.

New discharges to Special Resource Waters

Introduction

The Salmon River is designated as a "Special Resource Water" (WQS §056, 130). This designation imparts specific considerations to ensure protection of the Salmon River from new and increased point source discharges.

For Special Resource Waters, the Idaho Water Quality Standards §400.01b, generally require that:

> "…no new point source can discharge pollutants, and no existing point source can increase its discharge of pollutants above the design capacity of its existing wastewater treatment facility, to any water designated as a special resource water or to a tributary of, or to the upstream segment of a special resource water: if pollutants significant to the designated beneficial uses can or will result in a reduction of the ambient water quality of the receiving special resource water as measured immediately below the applicable mixing zone."

This requirement is not further defined in the Idaho WQS. However, other provisions in the WQS are used to determine the application of the SRW requirements. To implement this requirement in this permit, we have developed the following recommendations to ensure protection of this designated Special Resource Water. These requirements to monitor ambient water quality for practicably detectable changes in water quality are based upon the Idaho WQS, site conditions, precedence, and environmental science considerations. The latter include the actual capability to measure differences considering analytical precision of chemical measurements, natural variability, statistical probabilities, and the ecological risk associated with the discharges. In brief, we are including an explicit operational definition, for the purposes of this permit, of what constitutes a "reduction in ambient water quality" that is potentially significant to designated beneficial uses. The operational definitions are described in the following sections; their rationales follow the definitions.

Precedence for measures to protect Special Resource Waters

Relationship of Policies on Antidegradation and Special Resource Waters

Water quality standards are thresholds of protection of uses, not goals for high-quality waters that exceed these thresholds. The Idaho Antidegradation policy requires that in waters where ambient water quality exceeds water quality standards, that ambient water quality must be maintained (WQS section 051.02). Special Resource Waters are a related distinct, regulatory construct (WQS Section 056). A waterbody does not have to meet the definition of a "high quality" water to be designated a "special resource water" or vice versa.

"High quality waters" are those for which the water quality is better than that necessary to protect designated or existing beneficial uses. When evaluated on a parameter-by-parameter basis, the water quality in many waterbodies is better than the standards for those parameters, regardless of whether those waterbodies are designated as "special resource waters." Waters that do not exceed or even meet water quality standards and therefore could not meet the "high quality waters" definition may be designated as special resource waters because if their ecological significance is unique, if they were deemed threatened, or intensive protection of water quality was warranted (WQS section 056). There is no direct cross-referencing between sections 051 and 056, and somewhat different terminology and definitions are used. There are fundamental similarities though, as both sections address maintenance of existing water quality (although special resource waters also address restoration of degraded water quality). Because of the areas of similarity, to further define special resource waters we are borrowing some of the concepts from the antidegradation policy. The "special resource waters" of section 056 of the WQS and "high quality waters" of section 051 are not wholly interchangeable terms and meeting the definition of one category does not automatically apply the provisions of the other. Water quality on "high quality waters" that exceed standards may be lowered to those standards to accommodate important economic or social development. But there is no provision allowing water quality to be lowered for special resource waters.

Since the Salmon River is designated as a special resource water, and that designation triggers explicit restrictions on new or increased discharges, the following analysis is focused on defining and meeting special resource waters protections.

Specific recommendations to protect Special Resource Waters

Recommended requirements to assess protection of Special Resource Waters follow in this section. Further explanations and the rationale for their development is given in following sections.

> *To comply with Idaho WQS sections 056 and 400.01b, the concentrations of dissolved cadmium, copper, lead, mercury, selenium, silver, and zinc measured from samples collected at even increments from across the width of the river at the first bridge located downstream of the fully mixed confluence of Squaw Creek with the Salmon River, shall not be significantly different from concentrations measured in the same number of samples collected above the most upstream discharge to the Salmon River. The community structure of benthic macroinvertebrates collected from similar habitat types above and below the above locations should be similar.*

a. Working Definitions for Special Resource Waters Protections

Ambient concentration: The concentration of a chemical in a waterbody resulting from the addition of an incremental concentration to a background concentration (Suter 1993).

Ambient water quality: The phrase "ambient water quality," of which section 400.01b requires maintenance, is not defined in the Idaho Water Quality standards or in EPA (1991a, 1994). For the purposes of this permit, the definition of "ambient concentration" from an ecological risk reference text is used for chemical water quality (see above).

Assimilative capacity: The difference between the background concentration of a chemical and the concentration specified for the most stringent water quality criterion (Cairns 1977; EPA 1998).

Background: "The biological, chemical, or physical condition of waters measured at a point immediately upstream (up-gradient) of the influence of an individual point or nonpoint discharge" (WQS §003.06).

Laboratory analyses: Samples collected to assess protection of special resource waters need to be analyzed for cadmium, copper, lead, mercury, silver, zinc, calcium, and magnesium using analytical methods appropriate for detecting ambient concentrations of these metals. Calcium and magnesium measurements are required to calculate hardness, and thus the applicable metals criteria at the time of sampling.

Lower water quality: "A measurable adverse change in a chemical, physical, or biological parameter of water relevant to a beneficial use, and which can be expressed numerically. Measurable change is determined by a statistically significant difference between sample means using standard methods for analysis and statistical interpretation appropriate to the parameter. Unless otherwise defined for the parameter, statistical significance is defined as the 95% confidence interval when significance is not otherwise defined for the parameter in standard methods or practices." (WQS §003.56) Using standard statistical methods, statistical significance *is* otherwise defined for the parameters of interest below.

Monitoring locations: The first bridge located downstream of the fully mixed confluence of Squaw Creek with the Salmon River is the State Highway 75 bridge located about 2.5 miles below their confluence. The most upstream discharge to the Salmon River is proposed new Outfall 005, located just upstream of the confluence of Thompson Creek with the Salmon River. These locations correspond with the established monitoring stations SR1 and SR3 respectively. Monitoring of these paired stations should be synoptic, that is scheduled to approximately sample the same "parcel" of water as it moves downstream by the discharge locations.

Statistically significantly difference: Statistical significance is defined for the parameters of concern using the following tests with standard statistical methods: Upstream and downstream concentrations shall be considered significantly different if, with a 95% level of significance, the mean downstream concentration for a sampling event exceeds 25% of the assimilative capacity. In this case, this is the sum of the mean upstream concentration plus 25% of the difference between the upstream concentration and the numeric values for criterion continuous concentrations (CCC or "chronic" criterion). For example, if the mean upstream concentration of copper is 2 µg/l and the criteria were 8 µg/l, then the relative difference is 6 µg/l and the mean downstream criteria must not exceed 3.5 µg/l (2 µg/l + 0.25 X 6 µg/l = 2 µg/l + 1.5 µg/l = 3.5 µg/l).

The sampling program should allow these tests to be conducted using standard statistical methods according to the following statistical parameters: Type I error of 0.05 or better; and Type II error level of 0.25 or better with a minimum detectable difference for dissolved copper of at least ± 2 µg/l and dissolved zinc of ± 13 µg/l. The proposed minimum detectable differences are approximately 25% of the difference between background Salmon River concentrations and the chronic criteria. Further explanation of the statistical terminology and rationale for the definition of minimum detectable differences follow in the *Ambient Monitoring Data Needs* section below.

Rationale for Special Resource Waters Requirements

The preceding recommended requirements are based on the interpretation that for a reduction in ambient water quality to be significant to beneficial uses, and thus a lowering of water quality as defined in WQS § 003.56, it must be a practicably measurable adverse change.

WQS §400.01.b do not prohibit all new discharges to SRWs. They only prohibit new discharges when the discharge contains pollutants that are "significant to the designated beneficial uses" and when the discharge results in a "reduction of the ambient water quality." While "reduction of the ambient water quality" is not specifically defined in the WQS, the very similar term "lower water quality" is defined in section 003.56. This definition requires a measurable adverse change in a parameter that is relevant to a beneficial use. Using this definition with the language in 400.01.b a discharge to a SRW is prohibited if it (a) results in a measurable change in water quality; (b) results in an adverse change in water quality; and (c) involves a pollutant that is relevant and significant to a designated use for the receiving water body. The following sections describe how DEQ applies these concepts to the proposed discharger to the Salmon River.

a. Analytical variability and precision of chemical analyses near detection limits

The pollutants potentially significant to beneficial uses in this case are trace elements. Their typical concentrations in surface water and their criteria both are near their method detection limits for chemical analyses using routinely accepted methods for environmental samples. The reliability of a chemical measurement generally decreases as the contaminant concentration approaches its detection limit. Near the detection limit, the presence of the contaminant may be obscured by a complex mixture of chemicals or not distinguished from random electronic signals in the analytical instrument. Precision of approximately ± 30 to 50 relative percent difference between measurements (the random error of measurement) and bias of up to ± 50 percent of the true value (the systematic error measurement) are typical in analyses of samples at 5-10 times detection limits (EPA 1991). EPA data validation functional guidelines for evaluating inorganics analyses set an acceptable relative percent difference of ± 20 percent for laboratory duplicate analyses. Field replicate samples, which incorporate sampling technique and sample handling variability, in addition to the analytical variability, are expected to have higher inherent variability than laboratory duplicate analyses. This analytical variability increases as ambient concentrations approach detection limits.

Therefore, the inherent limitations and variability of laboratory analyses of water samples suggest that accurately distinguishing between results of chemical analyses is limited to about ± > 20 percent.

b. Sampling and natural variability in upstream concentrations

Due to the stochasticity (inherent randomness) of the crustal abundance of trace metals, and their resulting dissolution in water resulting from the weathering of rocks, there is a variability and uncertainty in ambient water quality which can be described and estimated but not reduced. For example, total zinc concentrations in Thompson Creek upstream of the mine discharges had an average coefficient of variation (CV)[2] of about 55%, based on 35 samples from 1993-1997.

Data from other locations or pollutants of concern at the Thompson Creek Mine site do not appear to have been sufficiently characterized to make these estimates. However, the natural background chemistry of Panther Creek, located about 30 miles North of the Thompson Creek Mine, was extensively characterized from 1993-1994. These analyses are likely reasonably representative for regional background chemistry for drainages in the Salmon River Mountains vicinity. Background concentrations of dissolved copper from Panther Creek had a coefficient of variability of about 85%, based on 38 samples collected from 1993-1994. Background values of dissolved copper were lower than those of total zinc, which may contribute to the differences (≈ 2 µg/l versus ≈ 20 µg/l zinc).

Therefore, the inherent variability of background trace elements in surface water limits the ability to distinguish small changes in ambient concentrations.

c. Significance of pollutants to beneficial uses

The restrictions on new or increased point source discharges to Special Resource Waters only apply to "pollutants significant to beneficial uses" (see section 2, above). For the proposed new or increased discharges, the following substances and pH were estimated to have a reasonable potential to exceed water quality standards in the Salmon River: cadmium, copper, lead, mercury, silver, zinc, total suspended solids (TSS), and pH. Selenium is not calculated to have the reasonable potential to exceed water quality standards, but based on a permit modification application that would discharge selenium containing discharges from outfalls 001 and 002, selenium should also be monitored unless EPA determines that there would be no reasonable potential to exceed water quality standards in the river. For all substances and pH, aquatic life criteria impose the most stringent criteria for effluent limits to comply with, rather than criteria that relate to other beneficial uses such as agriculture, drinking water, or recreation. Thus, only aquatic life beneficial uses will be considered in assessing the significance of pollutants.

Aquatic Life Beneficial Uses

The most direct way to determine significance of pollutants to aquatic life beneficial uses is to assess the beneficial use directly. Other ambient water quality protection measures such as numeric chemical criteria and whole-effluent toxicity (WET) testing can *predict* safe or harmful conditions for aquatic life, but cannot determine whether in fact those conditions actually occur.

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[2] CV, the standard deviation divided by the mean, is a standard statistical term used to describe how variable a sample group or population is.

Benthic macroinvertebrate community structures have been widely used as an integrative, in-situ biological sentinel in streams and rivers. The high-quality benthic macroinvertebrate trend data from Thompson and Squaw Creeks have been a persuasive line of evidence that adverse aquatic life effects from the mine discharges were unlikely. With the proposed expansion of mine discharges to the Salmon River, the ongoing benthic macroinvertebrate monitoring program should be expanded to include the Salmon River.

The fish community of the receiving waters is highly valued socially and is an important, sensitive, ecological component of the receiving waters. Like the benthic macroinvertebrate monitoring, the high-quality fish community trend data from Thompson and Squaw Creeks have been a persuasive line of evidence that adverse aquatic life effects from the mine discharges were unlikely. Likewise, with the proposed expansion of mine discharges to the Salmon River, the ongoing fish community monitoring program should be expanded to include the Salmon River. Together, the invertebrate and fish monitoring should provide the primary evidence of whether a reduction to ambient water quality significant to beneficial uses is occurring.

Total suspended solids

No numeric criteria apply to TSS; however, at sustained elevated concentrations, TSS may be harmful to fish. However, ubiquitous potential sources of sediment other than mine discharges along the Salmon River and in the Thompson and Squaw Creek watersheds would confound interpretation of TSS values above and below the mine discharges. TSS is therefore not recommended for inclusion in the ambient monitoring program for the purposes of compliance with WQS section 400.01b.

pH

Extremely high or low pH values can be harmful to aquatic life, especially in combination with certain trace metals. However, pH patterns in natural waters are strongly influenced by seasonal and daily patterns in various processes that can affect water quality, such as photosynthesis and respiration, dilution by snowmelt runoff, groundwater inputs, and microbial photoredox processes (Stumm and Morgan 1996). It is likely that different timing of snowmelt from the Thompson and Squaw Creek watersheds and the upper Salmon watersheds could cause measurable differences in pH above and below the new discharge points that have nothing to do with the discharges. pH is therefore not recommended for inclusion as a “pollutant” for the purposes of compliance with section 400.01b.

Metals

In aquatic systems the metals of greatest concern are copper, zinc, cadmium, mercury, and lead. These elements are toxic to organisms above specific threshold concentrations but many (e.g. copper and zinc) are essential for life at lower concentrations. Cadmium, lead, and mercury have no known biological function. Silver and other trace elements have been documented to cause adverse effects to aquatic life, albeit less frequently than the first group (Rand 1995). Since the draft permit lists cadmium, copper, lead, mercury, silver, and zinc as having a reasonable to exceed criteria, these are considered “pollutants significant to biological uses.”

Toxic thresholds for the metals above can be described. Above different thresholds for different organisms, they can be lethal or have sublethal adverse effects on aquatic life. At low concentrations, these constituents, which are found in all natural waters, are either essential for life or have no effect on aquatic life. Small changes in ambient concentrations well below thresholds of adverse effects are insignificant. Lethal and many sublethal effects for sensitive species are incorporated into the criterion maximum concentrations (CMC or "acute" criteria) and criterion continuous concentrations (CCC or "chronic") respectively (Table 1). Behavioral avoidance is a sublethal effect that is not incorporated into these criteria. For a river used as habitat and as a pathway for migratory salmonid fishes, behavioral avoidance is a sublethal effect of potential concern that is not incorporated into the criteria. Because of this, avoidance thresholds for sensitive salmonid species are developed in this report and are compared to potential ambient chemical concentrations to evaluate the potential for migratory disruption due to avoidance of constituents in the discharge. Under the restrictions imposed on the proposed discharge to the Salmon River, the concentrations of pollutants in the discharge would be below biological thresholds of concern (this report).

Therefore, small increases in very low concentrations of the trace elements of concern are unlikely to be significant to aquatic life and other beneficial uses.

Table 1. Pollutants significant to beneficial uses, their natural background concentrations, criteria, and detection limits

Dissolved Metal (µg/l)	Typical freshwater ambient concentrations in the U.S. (Note 1)			Upstream Salmon River concentrations (Note 2)	"Chronic" Criteria (CCC) (Note 3)	"Acute" Criteria (CMC) (Note 3)	Recent method detection limits (Note 4)
Cadmium	0.002	to	0.08	0.05	0.6	1.8	0.05
Copper	0.4	to	4	0.6	6.3	8.9	0.1
Lead	0.01	to	0.19	0.2	1.2	30	0.05
Mercury	0.001	-	0.020	<0.05	0.012	2.0	0.05 0.0005
Selenium	0.1		0.4	<1	5	20	1
Silver	0.01	to	0.5	<0.05	None	1.1	0.1
Zinc	0.03	to	5	3	58	64	1

Note 1: Mercury from Table 9, selenium from USDOI (1998), silver from Bell and Kramer (1999), others from Stephan et al. (1994)

Note 2: TCMC sampling results from all three Salmon River monitoring locations October 1998 to November 2000 (30 samples).

Note 3: Calculated for a hardness of 50 mg/l which is typical of the Salmon River, the median Salmon River hardness upstream of Thompson Creek from 1989-1998 was 55 mg/l.

Note 4: Lower of values from 40 CFR 136 or TCMC database. Mercury detection limits are from existing database using "clean" techniques, and newly promulgated EPA method using "ultra clean" techniques, 50 ng/l and 0.5 ng/l respectively

d. Ambient Monitoring Data Needs

Biological Monitoring

Macroinvertebrates: Benthic macroinvertebrate community data need to be collected above and below the proposed new and increased discharges at the monitoring locations described in section 3a above. Data should be comparable with IDEQ recommended protocols for collecting and interpreting macroinvertebrate data from large rivers. These protocols include stratifying sampling units to similar riffle habitats using a Slack sampler with 500 µm mesh, 3 replicates, and identifying organisms to the lowest practical taxonomic level. The macroinvertebrate river sampling period is limited to minimum, stable flows, from August to October (IDEQ 1998).

Fish: Logistically, monitoring fish communities in the Salmon River is more involved than in wadable streams, because boat electrofishing techniques are needed. Still, it is beneficial to characterize trends in the fish community in relation to the discharges, and in relationship to baseline (see Table 4 and related discussion). Fish monitoring should be conducted at least bi-annually or as as allowed with permits for scientific collection issued under section 10 of the Endangered Species Act.

Chemical monitoring

Chemical monitoring of pollutants potentially significant to beneficial uses needs to be sufficiently sensitive and reliable so that significant differences are detected. At the same time, occasional unrepresentative, anomalous high concentrations should not result in a conclusion that a reduction in water quality has occurred and trigger inappropriate management responses. The protection for both eventualities is an appropriate, statistically-based monitoring program. The detection limits listed in Table 1 are sufficiently sensitive to detect significant differences (with the possible exception of mercury for which few commercial laboratories can currently quantify criteria concentrations). Statistical considerations follow.

Statistical considerations in ambient monitoring

A fundamental question in ambient water quality monitoring is whether significant change has occurred. To comply with the Idaho Water Quality Standards, a monitoring plan for special resource waters must be likely to detect differences in ambient water quality if in fact they exist. Further, it must be unlikely to falsely indicate there is a difference when in fact there is none; that is, observed differences are just due to chance. These two needs involve statistical trade-offs. The answer to this dilemma depends on five interacting factors (Zar 1984):

1. Sample size: Larger sample size increases the ability to detect a difference between two groups of samples.

2. Variability: The more variable a measure, the less the ability to detect significant change.

3. Level of significance: This refers to the probability that an apparently significant difference is not real but simply due to chance. Convention has this referred to as α or a Type I error,

where the α value is set at 0.05 for most statistical tests which means there is only a 1 in 20 chance that an observed difference is due to chance, or a test is 95% "confident." The higher this confidence level is set at, the more likely the difference is real.

4. Power: The probability of detecting a difference when in fact one exists; designated (1-β). β or a "Type II" error, is the probability of incorrectly concluding that two groups of samples are the same when in fact they are different. In environmental sampling β is commonly set at 0.25 to 0.1; that is a test has a 75% to 90% probability of detecting a change if there is one. While higher probabilities would be desirable, because power function curves are logarithmic, as sample sizes increase, further increases in sample size make little improvement in a test's power. Tests with 90 to 95% statistical power would require huge sample sizes. Increasing the statistical power of a sampling plan reduces the likelihood of making a Type II error (failing to detect an actual difference), but at the same time <u>increases</u> the likelihood of making a Type I (concluding there is a difference when none exists).

5. Minimum detectable effect: Determining how much change is acceptable and thus needs to be detected in the ambient concentrations is a key factor in monitoring. Large differences are easily detected in environmental monitoring; subtle changes are difficult to detect. Therefore, for a monitoring program it is necessary to specify how much change is allowable before a beneficial use is impaired. Detecting "any change" is not a statistically acceptable answer because no monitoring program can detect an infinitesimal change (MacDonald et al. 1991).

   The following objectives for a minimum detectable change in ambient concentrations that the ambient water quality monitoring should be able to detect are proposed: 2 µg/l for copper and 13 µg/l for zinc. The basis for these values is that they are approximately equal to 25% of the difference between the chronic criteria and the upstream background concentrations for copper and zinc. Minimum detectable difference for the other metals of concern could not be meaningfully determined because they are seldom detected in the receiving waters and their detection limits are near criteria values. Assuming their occurrence in natural waters and source areas at the mine are proportional to copper and zinc, sample sizes adequate to detect differences in copper and zinc would also detect differences in the other metals.

<u>Sample Size Determination (Power Analysis) and Testing for Differences</u>

The 5 factors listed above can be used to answer two questions. To detect a specified difference, how many samples would be needed? Conversely, for a given number of samples, what will be (or, if done after the fact, what were) the minimum detectable differences for the sampling?

For example, using regional background values for copper, in order to determine whether mean concentrations of two groups of samples differ by at least 2 µg/l with a 95% level of significance and with statistical power of 75%, about 9 samples would be required in each group (α = 0.05, β = 0.25). A sample size of 5 would be expected to detect a difference in copper values of about

2.6 µg/l and a sample size of 10 would detect a difference in copper values of about 1.7 µg/l. The power analysis used to make these estimates follows (Zar 1984).

$$n \geq \frac{2s_p^2}{\delta^2}(t_{\alpha(1),2(n-1)} + t_{\beta(1),2(n-1)})^2$$

Where: n = sample size required for each population

s_p^2 = estimated within population variance

δ^2 = Minimum detectable difference, $\delta = \mu_1 - \mu_2$

$t_{\alpha(1),2(n-1)}$ = Critical value associated with the α level (one tailed) of the test

$t_{\beta(1),2(n-1)}$ = Critical value of a t set at the desired level of a Type II error and 2(n-1) degrees of freedom B level (one tailed) of the test

These estimates were calculated using background dissolved copper data from the nearby Panther Creek watershed collected in 1993-1994 (mean 2.3 µg/l, S.D. 2.0 µg/l, n = 38). Similar calculations using total recoverable zinc values in Thompson Creek (TC4) gave similar results (n=35, mean 20 µg/l, S.D. 10.8 µg/l). Note that the power function is geometric. When sample sizes are small, increasing the sample size gives significant increases in power. When sample sizes are large, further increasing the sample size provides small increases in discriminatory power. This is illustrated in an appendix.

To test for and report whether sample groups are different for each constituent, the results of standard parametric t-tests or non-parametric ones such as the Mann-Whitney test should be reported, including the actual α and β levels achieved.

Mixing Zone Analysis

Introduction

The Clean Water Act establishes a policy that states are to protect and restore the physical, chemical, and biological integrity of their waters. Water quality standards and policies affecting their application are developed by the states to achieve this goal. Waste discharges from industrial, municipal, or other sources may be permitted so long as water quality of the water body receiving the discharges is still protected. However, it is not always necessary to meet all water criteria within the *discharge outfall* to protect the integrity of the water body as a whole. Sometimes it is appropriate to allow a mixing zone where ambient concentrations may exceed criteria for small areas near outfalls. EPA and Idaho Division of Environmental Quality (IDEQ) policies allow for mixing zones to be permitted after considering the physical, chemical, and biological characteristics of the discharge and the receiving system; the life history and behavior in the receiving system, and the desired beneficial uses of the waters (EPA 1993, WQS § 60).

The following is an analysis of the applicability of mixing zones in Thompson Creek, Squaw Creek and the Salmon River. These waters are the receiving waters for proposed discharges from the Thompson Creek Mine. This determination is based upon biological, chemical, and physical appraisals of both the receiving waters and the proposed discharges as required by Idaho's mixing zone policy (WQS § 60).

A mixing zone is a defined area or volume of the receiving water surrounding or adjacent to a wastewater discharge where the receiving water, as a result of the discharge, may not meet all applicable water quality criteria or standards. It is considered a place where wastewater mixes with receiving water and not as a place where effluents are treated.

Idaho's mixing zone policy states that after a biological, chemical, and physical appraisal of the receiving water and the proposed discharge, and after consultation with the persons responsible for the wastewater discharge, the Department will consider the applicability of a mixing zone and, if applicable, its size, configuration, and location. In defining a mixing zone, several principles are to be considered, including:

- avoiding interference with existing beneficial uses;
- water quality within a mixing zone may exceed chronic water quality criteria so long as chronic water quality criteria are met at the boundary of any approved mixing zone;
- acute water quality criteria may be exceeded within a zone of initial dilution inside the mixing zone;
- the mixing zone may not be acutely toxic to biota significant to the receiving water's aquatic community; and

- a presumption limiting the mixing zone to 25% of the width and volume of the stream to allow a zone of passage for aquatic life.

These principles are to be considered in the Department's determination of the applicability of mixing zones, and do not constitute stand-alone regulatory requirements. Neither is the Department limited to considering these principles in its exercise of discretion implementing the mixing zone policy. For example, narrative requirements that waters be free from toxic substances in amounts that impair beneficial uses also apply in mixing zones. These requirements are very broad and include prohibiting adverse effects such as behavioral abnormalities, and lethal or sublethal effects such as reproductive impairment as a result of food chain transfer (WQS §200, 003.105).

Mixing Zone Analysis Methods

In addition to the principles just described, EPA has developed an "integrated" water quality policy for the control of toxic discharges to public waters which involves three approaches: the use of chemical-by-chemical specific monitoring and permit limits; measurements of the toxicity of the whole-effluent samples, and the biocriteria-bioassessment approach (EPA 1991a). The following have been developed to implement this approach.

EPA has developed, and Idaho has adopted, chemical-specific criteria for many, but not all, potential toxic pollutants. To protect against the potential for toxic effects from pollutants for which criteria have not been developed, unmeasured chemical pollutants, or additive effects of different pollutants, EPA has developed whole effluent toxicity test procedures to protect designated uses. EPA has not developed national biocriteria approaches because of the diversity of aquatic ecosystems; however, Idaho has developed and is refining numeric bioassessment indices to aid in interpreting the condition of stream and river aquatic ecosystems (Barbour et al. 1999; IDEQ 1998, 1999, 2000).

The mixing zone analyses include evaluations of 1) site and regional water and sediment chemistry; 2) biological conditions in the receiving waters; 3) whole effluent toxicity testing; 4) potential fish avoidance around the mixing zones (zone of passage); 5) risk of adverse bioaccumulative effects of mercury and selenium; 6) relative flows of effluents and receiving waters; 7) variations of flow by width and depth within the receiving waters; and 8) extensive hydrodynamic modeling of effluent plume dispersion and dilution under varying flow and pollutant scenarios. The predicted areas and frequencies of potential adverse effects are compared with the overall sizes of the water bodies and expected habitat ranges of aquatic and semi-aquatic life. Further, conditions and monitoring necessary to protect water quality standards are described.

Chemical Evaluation

The Idaho mixing zone policy requires chemical appraisal of the receiving waters and the proposed discharge. The evaluation necessarily differs for discharges to Thompson Creek *(continuation of existing discharges)*, Squaw Creek, and the Salmon River *(proposed new discharges only)*. TCMC has been conducting water quality monitoring since the mine's inception; however, metals concentrations reported before labs and samplers followed “clean” techniques were biased high (Stephan et al. 1994). By 1998, Thompson Creek was following “clean” sampling and analysis techniques, and in 1999 Thompson Creek began specialized selenium analyses that provided low-concentration selenium speciation analyses. While there are remaining analytical problems with the low per billion trace element analyses that approach the limits of detection,[3] data that appears generally reliable is presented in the following figures and is compared to applicable chronic (most stringent) numeric criteria.

Recent baseline (pre-discharge) sampling results for the Salmon River upstream and downstream of Thompson and Squaw Creeks indicate that metals concentrations are generally quite low, and are often below detection limits (Figure 2). 3 of 18 lead results exceed numeric chronic criteria in the Salmon River either upstream and downstream of Thompson and Squaw Creeks. The possibility of elevated lead concentrations in the Salmon River upstream of the mine cannot be ruled out, but seems unlikely.

Metals concentrations in Thompson Creek and Squaw Creek are generally higher downstream of discharges from the Thompson Creek Mine. Except for selenium, concentrations were well below chronic criteria (Figures 4-6). Cadmium, selenium and zinc tended to be higher in Thompson Creek than Squaw Creek, whereas lead concentrations may be in Squaw Creek than Thompson Creek. These elevated lead concentrations may be associated with the Redbird Mine, which is located near the Squaw Creek monitoring site SQ-3 and produced Pb, Ag, Zn, Cu, Au, and fluorite.[4] Sediment lead concentrations from Squaw Creek upstream of Thompson Creek Mine discharges were extraordinarily elevated (Figure 9, discussed later). The sediment chemistry suggests an upstream lead source and the sediment-sorbed lead would presumably be released into the water column under some conditions. Copper concentrations were similar in Thompson Creek and Squaw Creek.

Selenium concentrations are elevated in Thompson Creek downstream of the Buckskin and Pat Hughes (001 and 002) drainages (Figure 6). This is a water quality concern that is addressed in more detail in the section “Potential for bioaccumulative effects….” Unlike the other metals for which criteria are expressed for the “dissolved” form, or more accurately 0.45 µm filtered, selenium criteria are expressed as total recoverable criteria. Low-level selenium analyses are difficult, and TCMC has submitted data indicating that selenium results in the low part per billion range are questionable, with routine analysis results averaging about 30% higher than split samples reported from high-resolution speciation analyses. Speciation analyses of water samples collected in December 1999 indicated that the Thompson Creek discharges consisted of 100% selenate (TCMC 2000). Concentrations in discharges were around 28 – 30 µg/l; concentrations in Thompson Creek ranged from <1µg/l upstream to 5 µg/l downstream of Buckskin Creek. Since the effluent may not have been fully mixed at that point, TCMC is proposing a mixing tracer study (TCMC 2000).

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[3] E.g. April 1999 lead concentrations were elevated in all samples whether up gradient or down gradient of the discharges, suggesting either systematic laboratory or sampling contamination.

[4] E. Modroo, P.G., IDEQ, Idaho Falls, personal communication (citing Ross 1937 USGS B-837).

Recent criteria research indicate that relative selenium toxicity varies by over two orders of magnitude based on the form of selenium, with selenate (Se VI) less toxic and selenite (Se IV) more toxic (Canton 1999, Maier et al. 1993, Ingersoll et al. 1990). Organically complexed selenium (SeMe) is highly bioavailable through the aquatic food chain, and is the form of selenium of most concern for chronic effects (discussed more later).

Water hardness values

Most metals criteria are hardness-dependent, with metals being more toxic to aquatic life in waters with low hardness. This is because calcium is one of three major factors that regulate metals toxicity (i.e. calcium has an antagonistic toxic effect)[5]. Hardness is composed of calcium hardness and magnesium hardness, and is expressed as mg/l $CaCO_3$ equivalents. Magnesium is not known to ameliorate metals toxicity; however, calcium hardness is dominant in most systems, including Thompson Creek. Hardness values have been shown to show significant change seasonally in and near the study area (Figure 2). In nearby Panther Creek, hardness values drop about 40% from base flow values to the peak of runoff (RCG/HB 1994). Assuming other factors are equal, the same metals concentrations would be more bioavailable during spring runoff than during base flow conditions. To account for this, EPA proposes to use the lowest 5th percentile of hardness values in order to calculate water quality based permit limits. This approach will result in conservative (protective) criteria values being used in permit calculations. In technical memoranda, TCMC has presented arguments that this approach is overly conservative, and that other approaches could be used that would be consistent with Idaho water quality. We have made no independent analysis of hardness regimes with site data. For consistency with EPA permit calculations, EPA-calculated hardness values are used in mixing zone modeling (described later). However, if data are presented which are sufficient to support different approaches, the hardness values used must at least account for seasonal changes in hardness with the spring runoff likely being the critical time period.

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[5] Factors affecting bioavailability of waterborne metals: Calcium hardness (competes with metals for binding to the gill surface), alkalinity (forms inorganic complexes with metals in the water which are less toxic forms), and dissolved organic carbon (organic complexes with metals that may make them less toxic or at least delay toxicity). Hardness and alkalinity usually co-vary in natural waters

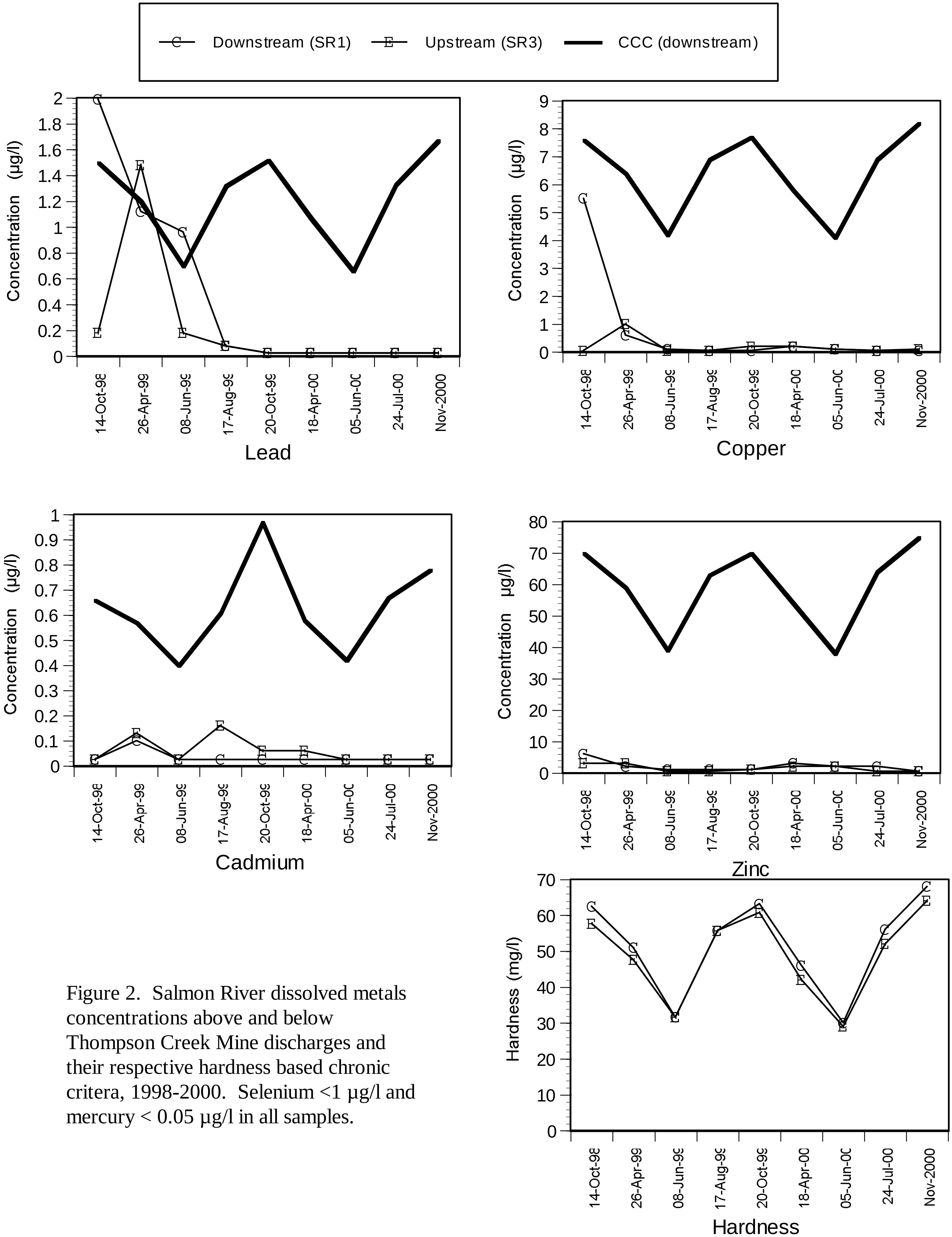

Figure 2. Salmon River dissolved metals concentrations above and below Thompson Creek Mine discharges and their respective hardness based chronic critera, 1998-2000. Selenium <1 µg/l and mercury < 0.05 µg/l in all samples.

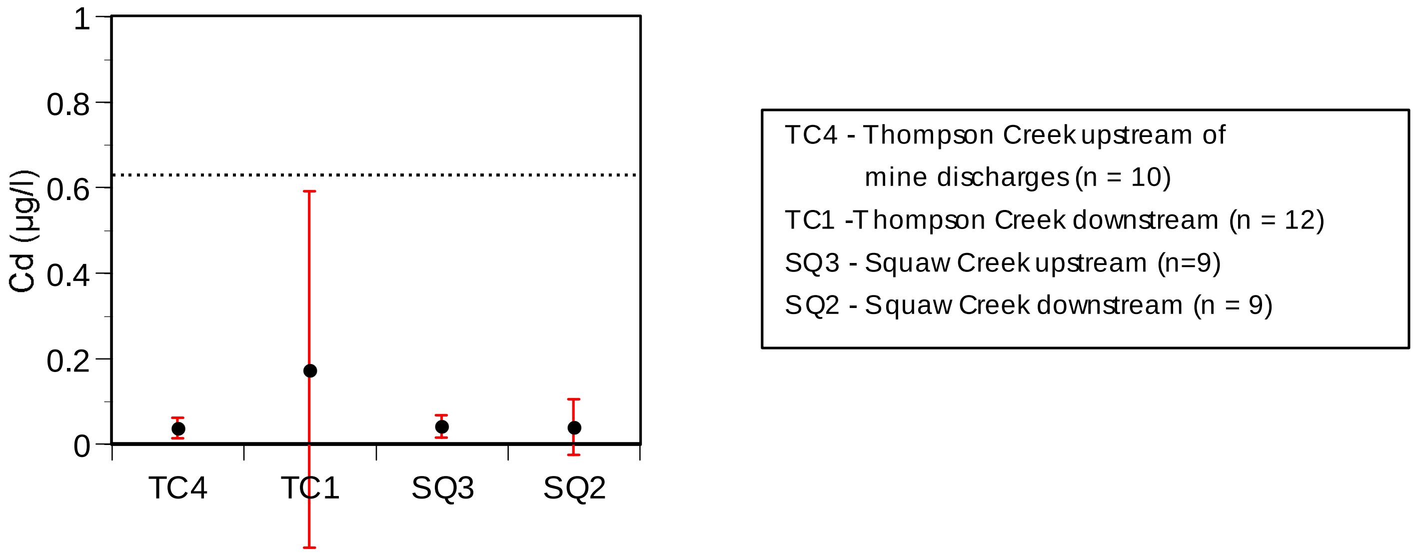

Figure 3. Total dissolved cadmium concentrations above and below Thompson Creek Mine discharges, 1998-2000. Error bars show +/- one standard deviation of the mean. Dashed line indicates chronic criteria calculated for a hardness of 50 mg/l).

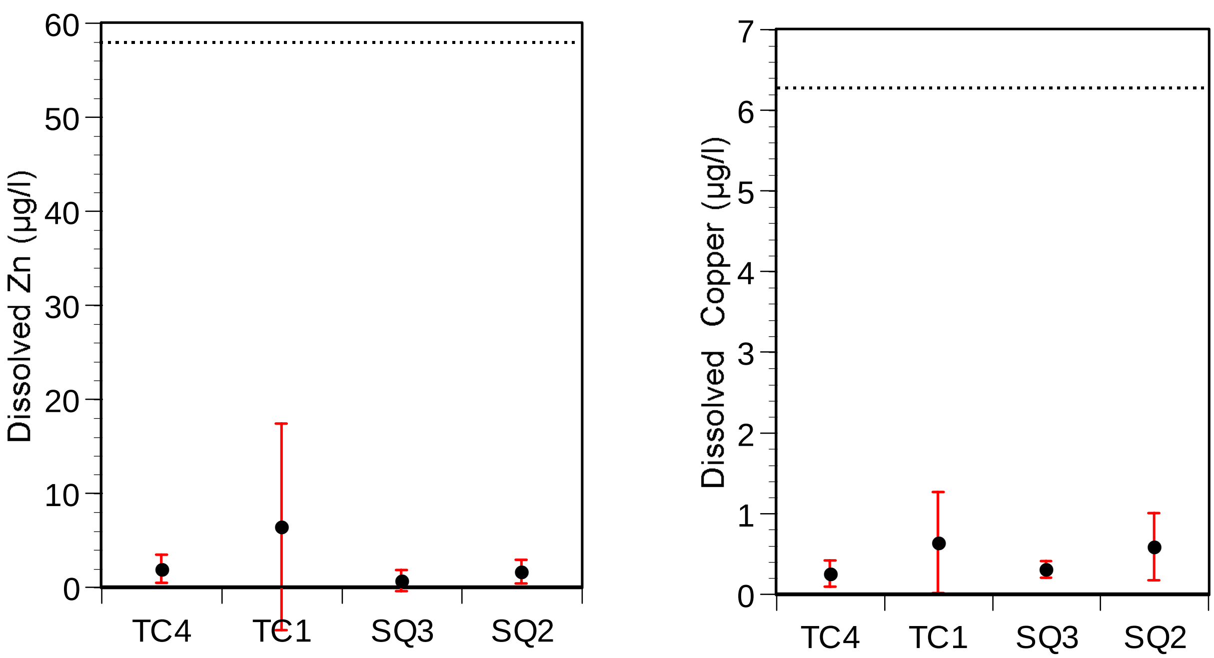

Figure 4. Dissolved zinc and copper concentrations from samples collected above and below Thompson Creek Mine discharges, 1998-2000. Error bars show +/- one standard deviation of the mean. Dashed line indicates chronic criteria calculated for a hardness of 50 mg/l).

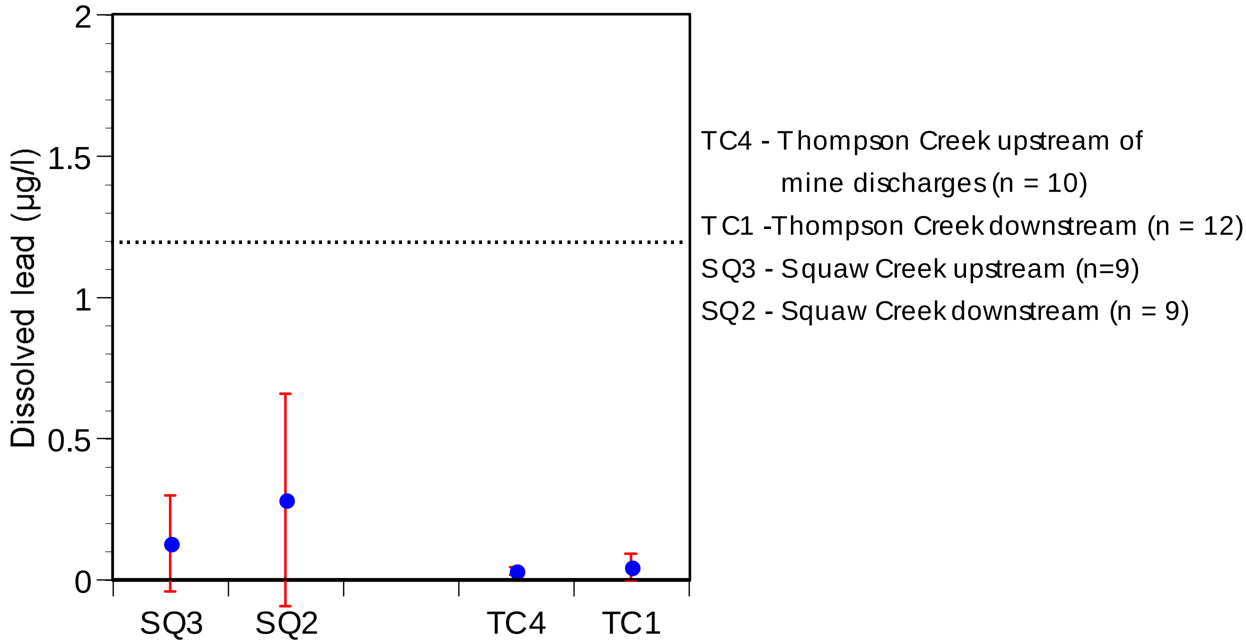

Figure 5. Dissolved lead concentrations from samples using collected above and below Thompson Creek Mine discharges, 1998-2000. Error bars show +/- one standard deviation of the mean. Dashed line indicates chronic criteria calculated for a hardness of 50 mg/l). Because of limited data, detection limit of 0.05 ploted for "less than" values.

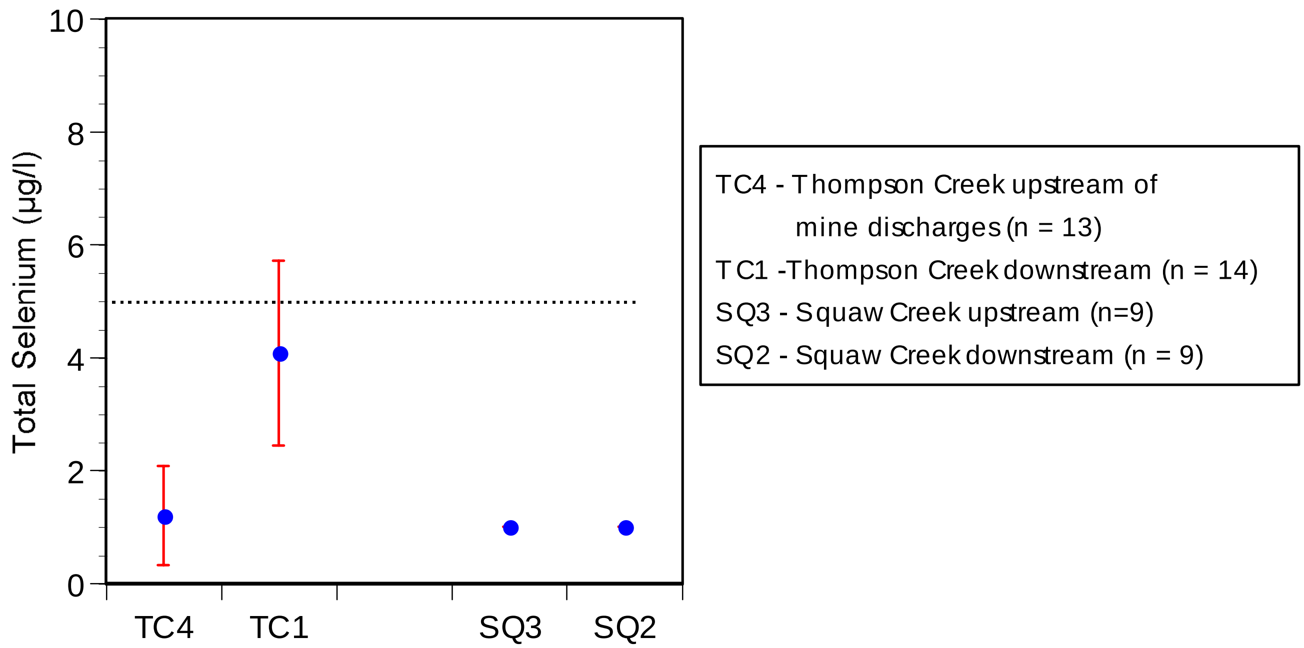

Figure 6. Total selenium concentrations from samples collected above and below Thompson Creek Mine discharges, 1998-2000. Error bars show +/- one standard deviation of the mean. Dashed line indicates chronic criteria (not hardness dependent). Plot restricted to recent data that used <1 ug/l detection limits. Because of limited data, detection limit was ploted for "less than" values (all Squaw Creek data).

Biological Evaluation

Data reviewed on biological conditions of the receiving waters included macroinvertebrate community composition, fish distribution, and toxicity test results of actual and proposed discharges. To determine whether a zone of passage for aquatic life would persist through the mixing zone, predicted chemical gradients in the mixing zones where compared to scientific literature on thresholds of chemical avoidance by fish. None of the data reviewed indicated that adverse biological effects were occurring in the vicinity of the existing discharges.

The study area is remarkable for the biological monitoring record in Thompson Creek and Squaw Creek. In most years since 1980[6], quantitative surveys above and below the current discharge points have been conducted twice annually for benthic macroinvertebrate community composition, and annually for fish community trends (Chadwick 1982, 1983, 1984, 1985, 1986, 1987, 1988, 1990, 1991, 1992, 1993, 1994, 1997, 1999, 2000c). The value of this record includes the obvious, evaluating whether the discharges are causing apparent effects to the aquatic invertebrate and fish communities (i.e. to the protected coldwater biota beneficial use). Further, the biomonitoring record provides insight into broader trends including climatic and flow phenomena such as the prolonged drought of 1987-1994, flood disturbances in 1997, and inter-specific dynamics such as the decline in local bull trout populations relative to rainbow trout populations.

In addition to this record, Thompson Creek, Squaw Creek, and to a lesser extent the Salmon River have been the focus of biological surveys by the USFS, IDFG and IDEQ. This is probably in part due to the large scale of the Thompson Creek Mine, and because of unfortunate lessons from other areas on the potential risk to aquatic ecosystems from mine discharges that were not adequately controlled (e.g. Clark Fork River, Coeur d'Alene River, Arkansas River, and Panther Creek watersheds). These data were also reviewed and compared with the annual trends data, and literature on the evaluation of effects of mine discharges on aquatic ecosystems.

a. Macroinvertebrate Community Analyses

Benthic macroinvertebrates are an essential component for energy cycling in aquatic ecosystems and are the primary food source for salmonids and sculpins. Field surveys of benthic macroinvertebrate communities are often used for ecological assessments of sediment and water quality monitoring. They have several features that make them significant for aquatic ecological assessments: Indigenous benthic macroinvertebrates are ecologically important as an intermediate trophic level between microorganisms and fish. They are abundant in most streams, and have either limited migration patterns or are sessile, which makes them suitable for site-specific impacts. Their life spans of several months to a few years allow them to be used as continuous

[6] Data actually go back to 1975 and are summarized in USFS (1980), but only data from 1980 on were available and reviewed.

indicators of sediment and water quality by integrating spatial and temporal variation, rather than a snapshot of conditions at one space in time (MacDonald et al. 1991).

Macroinvertebrate community structure analyses have been shown to be reliable and sensitive indicators of metals pollution in the water column. Shifts in benthic community structure commonly associated with adverse effects of metals include declines in the abundance of mayflies, reduced number of different mayfly species, reduced overall numbers of species, and increased dominance by midges, true flies, and worms. Declines in mayfly abundance and loss of mayfly taxa have consistently been reported as sensitive and reliable indicators of metals pollution, especially for copper and zinc (Winner et al. 1980, Clements and Kiffney 1994, Carlisle and Clements 1999; Richardson and Kiffney 2000, Mebane 2000).

b. Thompson and Squaw Creek conditions

Macroinvertebrates in Thompson and Squaw creeks upstream and downstream of mine discharges were sampled in 1999 for the 20th consecutive year. Diversity, as measured by taxa richness and Shannon-Weaver diversity index, were similar upstream and downstream, indicating the presence of balanced communities, with diverse, metals-sensitive species present. Mayfly density and taxa richness tended to be similar or higher at the locations downstream of the mine discharges during both July and October sampling events. The reasons for these patterns are not clear (e.g. possibly due to warmer downstream temperatures or higher photosynthesis rates), but clearly the mine discharges were not adversely affecting these continuous instream indicators of water quality. Similarly, the only long-term pattern apparent from the comparisons of density and taxa richness from 1980 to 1999 was that the upstream and downstream sites remained similar (Chadwick 2000c).

DEQ has also sampled macroinvertebrate communities several times each in the Salmon River, Thompson Creek, and Squaw Creek in the vicinity of the Thompson Creek Mine. To assess macroinvertebrate communities in wadable streams, we combined seven community metrics (measures) into an overall "macroinvertebrate biotic index" (MBI)[7]. Both the individual metrics and the overall MBI scores have been shown to generally decline with increasing fractions of fine-grained sediments and increasing metals concentrations above criteria values (Mebane 2000). DEQ has used this index to estimate the overall integrity of the benthic macroinvertebrate community. A score of 3.5 or greater out of 7.0 is considered to generally reflect an adequate community structure (IDEQ 1999). The component scores and the aggregate MBI have been shown to generally decline with increasing percentages of fine sediment or metals concentrations above criteria values (Mebane 2000). Scores in Thompson and Squaw creeks generally exceed 3.5, although one location in Squaw Creek above the Bruno Creek mine discharge (Outfall "003"), and one site in lower Thompson Creek on the Challis National Forest below an old schellite mill site were below this threshold (Table 3).

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[7] The component measures making up the MBI are(1) total number of taxa (taxa richness); (2) number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa (mayflies, stoneflies, and caddisflies); (3) percent EPT of total abundance; (4) Hilsenhoff Biotic Index; (5) Shannon-Weaver diversity; (6) percent dominance by the most common taxa; and (7) percent scrapers.

Bioassessment of macroinvertebrate communities in larger rivers used the invertebrate river index (IRI) developed by Royer and Minshall (1996). Salmon River macroinvertebrate community composition in the vicinity of the proposed new outfall and mixing zone has been sampled several times from 1996-1999. Community metrics used in a combined multimetric invertebrate river index (IRI) are shown in Table2. IRI scores ≥16 are considered to indicate good water quality, similar to that from reference sites (Royer and Minshall 1996). Salmon River results in the vicinity of the proposed outfall indicate excellent pre-discharge baseline water quality conditions.

Table 2. Salmon River macroinvertebrate composition

	Salmon River below Yankee Fork (ISU-1996)	Salmon River below Yankee Fork (IDEQ 1999RIDFP001)	Salmon River upstream of Thompson Creek (IDEQ 1999RIDFP005)	Salmon River below Squaw Creek (IDEQ 1999RIDFP002)	Salmon River upstream of Thompson Creek (IDEQ 1999)
Taxa richness	31	42	52	58	42
EPT taxa richness	20	21	19	20	19
% Dominance (single taxa)	19	24	33	21	16
% Riffle beetles (Elmidae)	9	3	2	4	5
% Predators	12	10	6	4	9.5
IRI Score (out of 23)	23	23	23	21	

Table 3. Macroinvertebrate biotic index (MBI) and habitat index (HI) and habitat features in waters in the vicinity of Thompson Creek Mine.

							Stable Banks (%)		Bank Vegetative Cover (%)	
Site ID	Stream	Elevation (feet)	HI	MBI	Fines (%)	Width/ Depth	Left Bank	Right Bank	Left Bank	Right Bank
95-A069	Squaw Creek	6440	78	4.69	35	23	84	76	66	58
94-41	Squaw Creek	6120	85	3.13	7	22	75	55	30	55
95-A070	Squaw Creek	5920	89	4.07	27	12	91	94	5	0
94-42	Squaw Creek	5680	82	4.55	10	44	85	70	20	35
94-39	Thompson Creek	7040	99	4.57	10	20	90	95	60	60
95-A104	Thompson Creek	7040	104	5.32	22	6	81	95	38	79
95-A105	Thompson Creek	5640	80	3.10	20	22	80	100	52	86
94-40	Thompson Creek	5560	89	4.06	2	33	95	100	15	10

c. Fish Populations

Fish populations are important both because of their value as a protected public resource, and for their value as indicators and integrators of water quality conditions. Fish populations in Squaw and Thompson Creek have been monitored periodically since 1975 above and below current mine discharges as part of environmental impact assessment for developing the mine operations monitoring plan. In addition to the series of data previously reported (e.g. Chadwick 1999), various unpublished monitoring data from 1992-1999 conducted by the IDFG and DEQ are reported below. (Tables 4-7).

Data from Chadwick (1999) were re-plotted to emphasize differences above and below the mine discharges (Figures 7 and 8). Throughout the 19-year period of record, fish species that are generally intolerant to elevated metals and sediment concentrations (rainbow trout, chinook salmon, and sculpin) have been well distributed throughout Thompson and Squaw Creeks, above and below discharge points. Bull trout and cutthroat trout have consistently been less common in lower Thompson and Squaw Creeks (Tables 5-7, Figures 7 and 8, Chadwick 1999). These differences are probably due to their preferences for the colder water occurring near the headwaters, and from competition from rainbow trout which thrive in the larger, warmer, lower elevation waters (Meehan and Bjornn 1991, Behnke 1992).

Present fish distribution in these creeks is similar to that prior to development of the mine. In surveys from 1975-1979 (before the mine was developed), cutthroat trout abundance in Thompson Creek increased in abundance and juvenile chinook decreased in abundance with distance upstream from the mouth. Juvenile chinook salmon were abundant at the mouths of both Thompson and Squaw creeks, but absent from the vicinities of the then-future mine discharges. Cutthroat and bull trout were captured from both streams, but in smaller numbers and fewer locations than rainbow trout, sculpins, or mountain whitefish. From 1975-1979, the number of fish taken from Squaw Creek was considerably lower than from Thompson Creek (USFS 1980). This latter trend is no longer present, with generally similar or higher densities of rainbow trout captured in Squaw Creek compared to Thompson (Figure 8). This may be related to Thompson Creek Mine's activities on lower Squaw Creek to eliminate diversions, restore perennial flows and connection to the Salmon River, and channel and riparian vegetation habitat restoration. These activities were undertaken in part to mitigate the loss of cutthroat trout habitat due to construction of the tailings impoundment in Bruno Creek (USFS 1980, 1999).

Bull trout populations in Thompson and Squaw Creek have steadily declined from 1980 to date and may be nearing a local extinction (Figure 7). Bull trout populations are depressed throughout much of the Northwest region (Rieman et al 1997), and the local Upper Salmon River Subbasin (Quigley and Arbelbide 1997). Competition from introduced rainbow trout and brook trout is considered a factor in the decline throughout their range, aggravated by habitat fragmentation, water quality impairment, and over-fishing (Behnke 1992, Quigley and Arbelbide 1997, Rieman et al 1997).

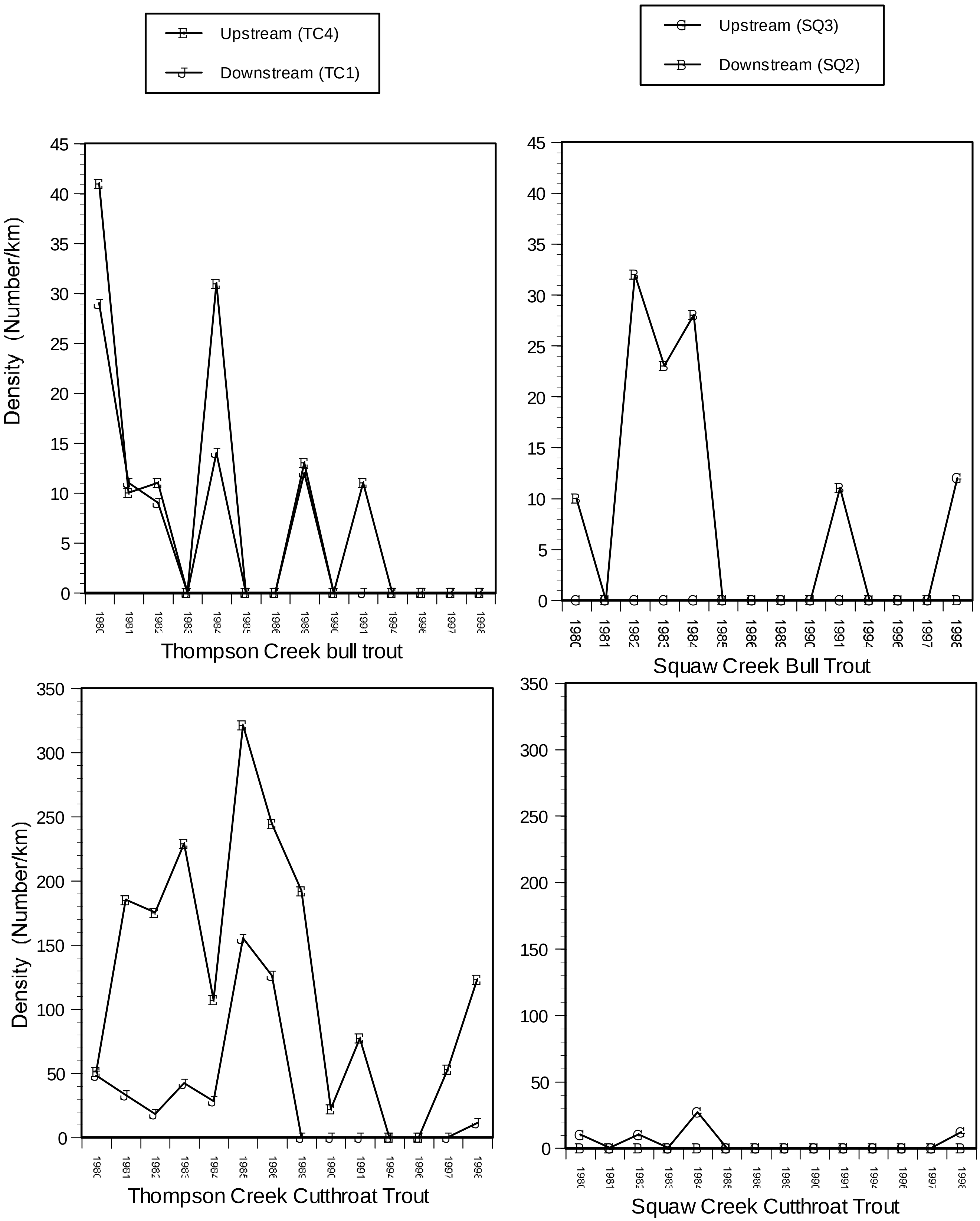

Figure 7. Bull trout and cutthroat trout population trends above and below the Thompson Creek Mine discharges. Data from Chadwick (1999).

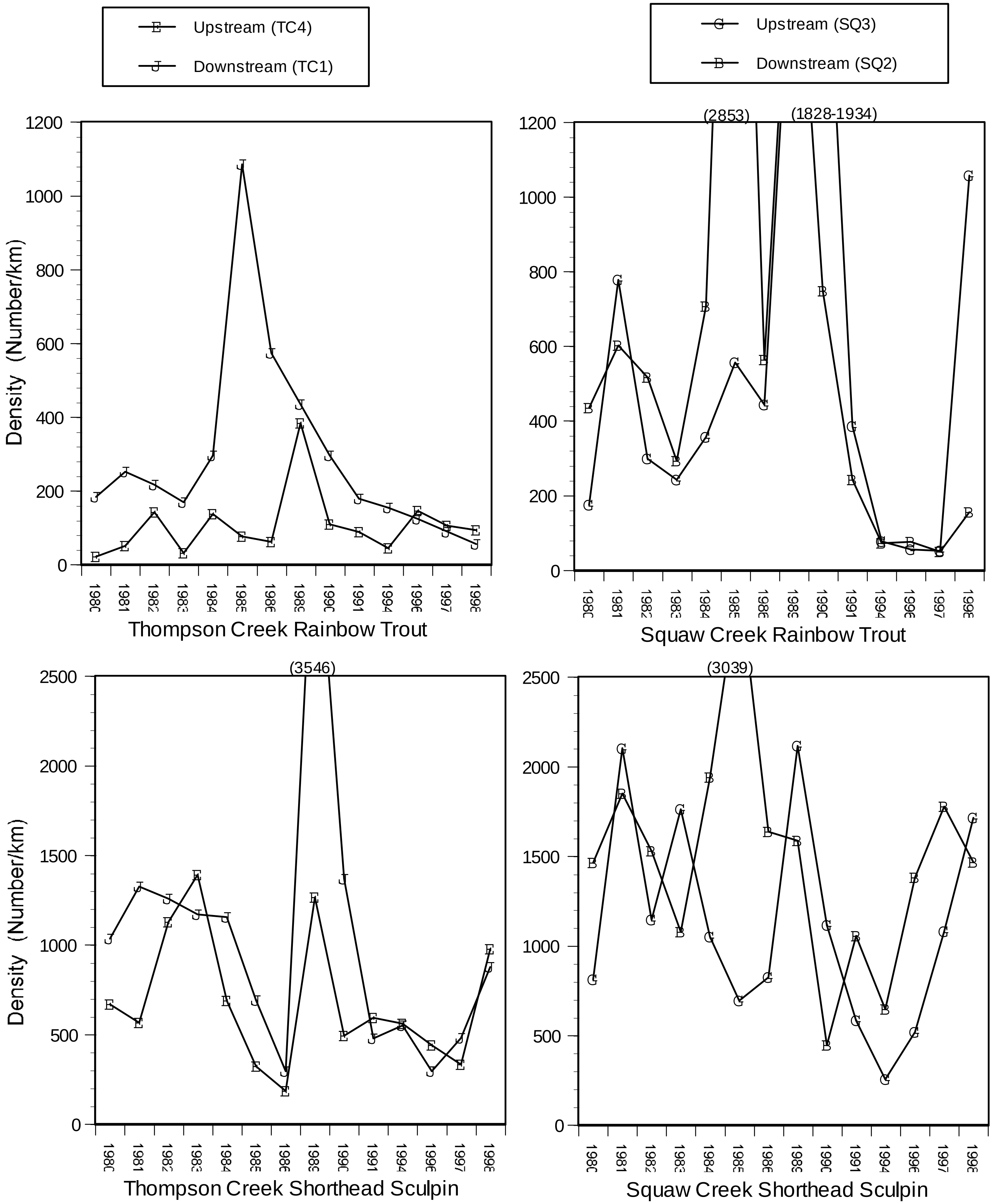

Figure 8. Rainbow trout and shorthead sculpin population trends above and below the Thompson Creek Mine discharges. Data from Chadwick (1999).

Shorthead sculpin are abundant throughout both Thompson and Squaw Creeks. Shorthead sculpin are particularly well suited as environmental indicators because they are comparably sessile fish that avoid excessive sediment (Mebane 2000), appear about as sensitive as trout to at least some metals (EVS 1996), and prefer cool temperatures. Continued monitoring is warranted to determine their population trends. The fish assemblage in the Salmon River in the vicinity of the proposed discharge was sampled in October 1999 by the U.S. Geological Survey (Table 4). This is an important component of the biological condition of the Salmon River in relation to proposed discharges, and continued monitoring would be appropriate.

Significant population changes have occurred during the monitoring period, but they seem more in line with fish temperature and habitat preferences rather than water quality. Bull trout and cutthroat trout have been more abundant upstream of the mine discharges; rainbow trout and shorthead sculpin have similar or at least overlapping abundances upstream and downstream of the discharges. Chinook salmon are typically found near the mouths of tributaries to the Salmon River, where they take refuge in cooler tributary waters (M. Larkin, IDFG, pers. comm.).

Juvenile *O. mykiss* cannot be distinguished from the resident rainbow or the anadromous steelhead form in field surveys, and are usually reported as rainbow/steelhead. IDFG has not stocked either Thompson Creek or Squaw Creek for many years; however the Salmon River is routinely stocked with adult rainbow trout. Juvenile *O. mykiss* that were captured in Thompson and Squaw Creek in 1998 were apparently the progeny of stocked rainbow trout, based on an examination of several specimens by Dr. R. J. Behnke, Colorado State University. From their morphology, they were more similar to coastal strains of rainbow trout than to residualized steelhead trout (Chadwick 1999).

Table 4. Fish species and relative abundance in the Salmon River, near proposed Outfall 005

Salmon River (5 km upstream of Thompson Creek)

	# Captured	% of total
Chinook salmon *(Oncorhynchus tshawytscha)*	8	7
Rainbow trout *(Oncorhynchus mykiss)*	3	3
Cutthroat trout (*Oncorhynchus clarki*)	0	0
Bull trout (*Salvelinus confluentus)*	0	0
Mountain whitefish *(Prosopium williamsoni)*	40	34
Mottled sculpin *(Cottus bairdi)*	2	2
Shorthead sculpin (*Cottus confusus*)	62	53
Total	115	100
Index of Biotic Integrity (IBI) score (out of 100)	94	

Source: USGS/IDEQ co-op sampling, October 1999
IBI from IDEQ 2000a

Table 5. Fish species composition in the vicinity of Thompson Creek Mine discharges, Thompson and Squaw Creeks (IDEQ electrofishing)

	Thompson Creek above mine discharge (1995SIDFA104)	Thompson Creek below mine discharge (1995SIDFA105)	Squaw Creek above mine discharge (1995SIDFA069)	Squaw Creek below mine discharge (1994SIDFA42)
Chinook salmon *(Oncorhynchus tshawytscha)*		15		
Rainbow/Steelhead Trout *(Oncorhynchus mykiss)*		6	5	8
Cutthroat trout (*Oncorhynchus clarki*)	6		2	
Bull trout (*Salvelinus confluentus)*	2	1		
Mountain whitefish *(Prosopium williamsoni)*				
Shorthead Sculpin (*Cottus confusus*)		67	14	2
Sampled by single pass electrofishing, sub-sample taxonomy by Dr. Richard Wallace, University of Idaho. Thompson Creek sampled 8-31-95, Squaw Creek sampled 9-02-98 (Source: DEQ Beneficial Use Reconnaissance Program)				

During a 1995 survey of spawning adults, anadromous steelhead trout were observed in both lower Squaw Creek and Thompson Creek by the Idaho Department of Fish and Game (USFS 1999). No adult salmon have ever been recorded in either Thompson Creek or Squaw Creek despite many years of fish sampling during the August/September time frame when, if present, adults would be expected to move into spawning streams. Steelhead trout can access more streams for spawning than chinook in part because higher flows and colder water occurs during their spring spawning.

Table 6. Fish species composition in the vicinity of Thompson Creek Mine discharges, IDFG sampling June, 1994

	Thompson Creek above mine discharge	Thompson Creek below mine discharge	Squaw Creek above mine discharge	Squaw Creek below mine discharge
Chinook salmon *(Oncorhynchus tshawytscha)*		6		
Rainbow/Steelhead Trout *(Oncorhynchus mykiss)*	1	7	5	1
Cutthroat trout (*Oncorhynchus clarki*)	5			
Bull trout (*Salvelinus confluentus)*				
Mountain whitefish *(Prosopium williamsoni)*				
Sculpin (*Cottus* spp.)	31	34	8	22

Sampled by two pass electrofishing, 6/10/94 (Squaw) and 6/12/94 (Thompson). (Liter and Lukens 1994)

Table 7. Salmonid composition in lower Thompson Creek from IDFG snorkel surveys 1992-1994

	6/25/92	7/16/93	7/19/94
Chinook salmon *(Oncorhynchus tshawytscha)*			34
Rainbow/Steelhead Trout *(Oncorhynchus mykiss)*	7	20	68
Cutthroat trout (*Oncorhynchus clarki*)			
Bull trout (*Salvelinus confluentus)*			3
Mountain whitefish *(Prosopium williamsoni)*		14	

Note: Hatchery rainbows excluded from table. Source: Liter et al. 1995

d. Toxicity Testing

Effluent samples have been tested for toxicity using rainbow trout, fathead minnows, a freshwater crustacean (the cladoceran *Ceriodaphnia dubia),* and green algae. Test responses have been variable. In 1994, samples of various ambient water samples that make up existing or proposed discharges (Squaw Creek and the Salmon River) were tested for acute lethality with rainbow trout and for sublethal effects with *Ceriodaphnia dubia*. There was 100% survival in all the 96-hour rainbow trout tests from the existing discharges at full concentrations of the effluent (no dilution). The sample of the proposed new discharge had 85% survival of the rainbow trout after 96-hours at full concentration, and 100% survival when diluted to 50% effluent. The reduced survival was not statistically significant with the standard study design used (Canton 1994).

Ambient samples (no dilution) from Outfall 001 (Buckskin Creek), 003 (Bruno Creek) and the Pumpback Station and Left Abutment (components of proposed effluents 004 and 005), had no reduction in *Ceriodaphnia* survival, but *Ceriodaphnia* biomass and reproduction (brood size) were reduced compared to controls (Canton 1994). The effluent proposed for discharge from Outfalls 004 and 005 will also include pit water, which was not included in the 1994 testing. 1998-1999 chemical analyses of the pit water show that chemicals of concern occurred in similar concentrations as in the waters tested, so it is likely that the toxicity results would have been similar with pit water included (i.e. no or limited acute lethality, significant sublethal effects of undiluted effluent). Further testing is needed before strong conclusions may be made about the toxic or nontoxic nature of the proposed effluent.

In 1999, samples from existing outfalls 001 and 002 were tested for acute and sublethal effects with *Ceriodaphnia*, fathead minnow, and green algae *Raphidocelis subcapitata* (formerly known as *Selenastrum capricornutum*). No significant toxicity was shown in the *Ceriodaphnia* and minnow acute and chronic tests. The green algae growth test for samples from outfall 002 resulted in an IC_{25} (growth inhibition concentration, considered the measurement threshold of effects) of 4.4% effluent and a no observed effects concentration (NOEC) of 12.5%. The reductions in algal growth were not consistent with increasing effluent concentrations (TCMC 2000).

In Thompson Creek, the rich instream biosurvey record provides strong evidence of the lack of adverse effects from outfalls 001 and 002. However, for the proposed discharges, the instream biological data provide an ecological baseline, but cannot be used to interpret effects of a discharge which has not happened yet. In this case, whole effluent toxicity testing of the waters that will make up the discharge will be useful to predict instream toxicity.

Significance of whole effluent toxicity results to beneficial uses

Whole effluent toxicity testing has been advanced as a practical approach to measuring compliance with the requirement that water bodies be free from toxic substances in concentrations that impair beneficial uses (WQS §200.02). Scientific literature on the development and validation of chronic WET tests was searched to evaluate their efficiency and reliability for predicting instream impairment. ***The literature reviewed amply supported the use of the fathead minnow and Ceriodaphnia chronic tests, but not the green algae growth test, to predict instream impairment and compliance with "free from toxic substances" water quality standards.***

For cost, logistical, and data-comparability reasons, standard laboratory bioassay organisms are used instead of resident species for whole effluent toxicity (WET) testing. For example, *Ceriodaphnia* (water fleas), are found in lentic environments, ponds and lakes, but not lotic environments (streams), and fathead minnows are alien west of the continental divide and do not occur in the study area. However, the fathead minnow and *Ceriodaphnia* results have been related to lotic benthic macroinvertebrate and fish assemblage composition (studies cited above). Fathead minnows also have been shown to be fairly responsive to metals, with ranges of responses similar to those of salmonids (Norberg-King 1984, Chapman 1978, Chakoumakos 1979).

Acute and sublethal (chronic) toxicity testing of ambient waters with fathead minnows and *Ceriodaphnia* has been related to biological impairment in streams in a wide variety of locations and stream types in the United States, Canada, and the U.K. (Birge et al. 1989, Eagleston et al. 1990, EPA 1991a; Dickson et al. 1992; Clements and Kiffney 1994; Grothe et al 1996; Sarakinos and Rasmussen 1998, LaPoint and Waller 2000, Diamond and Daley 2000, Maltby et al. 2000). One recent study was located relating *R. subcapitata* green algae growth response to the instream periphyton composition, and other phytotoxic responses associated with bleach kraft mill effluent. Although results were inconsistent, diluted effluent may result in growth stimulation, while higher concentrations resulted in growth inhibition (hormetic response). Instream diatom community composition changes were more sensitive and consistent than the algal growth or other WET tests (Culp et al. 2000).

In theory, if the effluent test concentrations correspond to ambient concentrations, the WET test results will be accurate. In practice, however, diluted whole effluent toxicity test results, as opposed to tests of ambient water samples, often do not relate well to instream biological conditions. This is probably because the WET dilution thresholds do not correspond to actual instream effluent concentrations, confounding the results (Diamond and Waller 2000). This is significant because the laboratory whole effluent toxicity test results are not a protected beneficial use; the instream biological condition is. The goal is to protect water bodies, not necessarily laboratory samples. Idaho's mixing zone policy implies that the mixing zone be free from acute toxicity to aquatic life, and waters below the mixing zone must be free from acute or chronic toxicity. The whole effluent toxicity test results completed to date suggest that the existing effluents would probably not be acutely toxic at full concentrations. The responses to the chronic tests have been variable. Further testing with the fathead minnow, *Ceriodaphnia,* and rainbow trout whole effluent tests would be useful.

Review of the accuracy of whole effluent toxicity testing for predicting instream effects

Test species — Where whole effluent toxicity tests are required to make decisions, EPA recommends the use of at least three test species, and further suggests that one of them be green algae[8] as a surrogate for the plant kingdom (EPA 1991a, Denton and Narvaez 1996). Idaho's regulatory language that waters be "free from toxic …substances in concentrations that impair beneficial uses" is interpreted here to call for the use of effluent tests that have been demonstrated

[8] (*Raphidocelis subcapitata,* formerly known as *Selenastrum capricornatum)*

to give a valid assessment of receiving water impacts on waters that support aquatic biota. So long as WET tests results are related to actual effluent concentrations in ambient waters, this relationship has been amply validated with the use of *Ceriodaphnia* and fathead minnows *(Biological Evaluation, Toxicity Testing)*. Curiously, despite the longstanding EPA recommendation to use the green algae growth test, until April 2000, no laboratory-to-field validation studies comparing the algal growth test to instream plant communities (periphyton) were made in the 15 WET-instream ecology studies reviewed, including EPA's Complex Effluent Toxicity Testing Program. The recent study results compared instream periphyton effects to algal growth WET from pulp mill effluent to the Fraser River. Periphyton community composition was a more sensitive and reliable response than WET results. WET results were hormetic (dilute concentrations stimulated growth, strong concentrations inhibited growth), which is difficult to interpret in a regulatory context (Culp et al. 2000).

While less significant for regulatory applications than the beneficial use validation, there are also significant unresolved protocol issues with the algae growth test. The test was originally designed to detect eutrophication, and algal growth is higher in most ambient waters than in the laboratory control water because algal growth is enhanced by the naturally occurring nutrients in many ambient waters. Poor growth in ambient samples does not necessarily indicate that a toxicant is present. Low algal growth in a sample can be due to low hardness or alkalinity or the absence of extra nutrients relative to the control or other sites (de Vlaming et al. 2000). Whether to exclude EDTA has been strongly debated in recent literature. EDTA is an important component of the algal growth medium that regulates trace elements' bioavailability, with Environment Canada and ASTM methods recommending its use to ensure adequate test performance, and EPA methods recommending against its use because excluding it increases the tests sensitivity to metals. Geis et al (2000) evaluated the use of the algal test as a regulatory assay, and recommended the use of EDTA to ensure adequate test performance.

Until *R. subcapita* growth-inhibition and stream periphyton validation studies are located or completed, and because of EPA's limitations on the use of alternate tests (EPA 1995a), predictions of chronic toxicity of effluents to receiving waters should be made using *Ceriodaphnia* and/or fathead minnows. The Pellston Workshop on whole effluent toxicity recommended field assessment approaches to compensate for the limitations of WET tests to predict phytotoxicity among other effects (Groethe et al 1996).

Toxicity triggers **—**For whole effluent toxicity testing to be a reliable, qualitative predictor of aquatic population impacts, testing needs to be done at effluent concentrations that approximate the effluent-receiving water exposure concentrations. To interpret whether the test results indicate the sample was toxic, test results need to be compared with benchmarks for effluent-receiving water concentrations. This raises the question, should WET tests be assumed *a priori* to be accurate predictors of instream responses? Or should they be assumed *a priori* to be biased to either underestimate or overestimate impairment to biological communities?

Recent investigations and expert opinion indicate that it would be reasonable to assume that WET test results are accurate predictors of ecological effects and are not systematically biased. My summaries of the conclusions of these reviews and studies are in bold below. Let us consider first recently published expert opinion. In 1995, the Society of Environmental Toxicology and

Chemistry (SETAC), with support from EPA and trade groups, convened a group of invited experts to a workshop in Pellston, Michigan on the technical adequacy and environmental relevance of WET regulation. The SETAC review concluded that currently available tools were acceptable to assess WET, but that a number of technical refinements were possible. One of the factors necessary for WET tests to be effective predictors of effluent effects on instream biota is the degree to which the actual exposure of instream biota to contaminants in the effluent is mimicked with the WET compliance (dilution) criteria. The results of bioassessments were judged to be useful in evaluating WET limits and margins of safety (Groethe et al 1996). ***Conclusion: Standard WET tests are not systematically biased and should mimic actual conditions.***

Next, let us review recent investigations which specifically explored these questions. de Vlaming and Norberg-King (1999) examined 77 studies in which the results of laboratory single-species toxicity tests were compared with biological community responses. In 74% of those studies the predictions were accurate; in 21% of the studies they underestimated instream biological responses. *C. dubia* in particular was a reliable predictor of instream response. ***Conclusion: Standard WET tests are accurate.*** Diamond and Daley (2000) compiled a database of 250 dischargers across the United States and matched WET test endpoints and benthic macroinvertebrate assessments. They reported when the effluent dominated the stream, and WET results were consistent, the WET results were accurate predictors of instream condition. The fish acute and chronic endpoints were most related to the instream condition. The accuracy of the WET test results depended on the degree to which this actual exposure was mimicked in standard WET tests. ***Conclusion: Standard WET tests overestimate instream toxicity.*** Sarakinos and Rasmussen (1998) attempted to quantitatively field-validate laboratory-derived toxicity thresholds. They quantified invertebrate community structure and density over a paper mill effluent gradient in a river and compared those data to effluent toxicity tested at two independent laboratories. They found that the most sensitive laboratory WET-derived thresholds were about 2X higher than the most sensitive response observed in the field. Effluents samples inhibited algal growth in one lab's tests (the most sensitive), but samples of the same effluent sent to the second lab stimulated algal growth. Sarakinos et al. (2000) also reported that toxicity for effluents containing metals were overpredicted by chemical criteria predictions of toxicity compared to WET. ***Conclusion: Standard WET tests underestimate instream toxicity, at least for pulp and paper facilities.*** Algal growth may not follow a "dose-response" curve. Maltby et al. (2000) compared the results of laboratory WET results (*Daphnia magna*) to *in situ* toxicity tests at different points in the effluent stream with an indigenous field-collected amphipod (*Gammarus)* and also compared the results to instream biological conditions. Actual receiving water toxicity and ecological degradation were consistent with results of WET tests for a corresponding range of dilutions. ***Conclusion: Standard WET tests are accurate.***

Overall Conclusion: Standard WET tests are not systematically biased and should mimic actual conditions. Because of the long bioassessment record on Thompson Creek, actual dilutions should be used as toxicity triggers. Despite the majority view that WET tests dilutions are not systematically biased, because less is known of the makeup and thus the biological effects of the proposed effluents, more conservative toxicity triggers based upon the rarely-occurring critical flow limits are recommended for "reasonable potential" testing. If results with the simulated proposed effluent show that there is a reasonable potential for receiving water toxicity, then WET permit limits should developed. In this case, using actual effluents instead of simulated effluents, WET testing should then be based on actual dilution measured at the time of WET sample collection.

e. Comparison of toxicity literature on sensitive species to existing and predicted chemical discharges

Potential adverse effects to sensitive fish species were estimated by comparing chemical concentrations occurring or predicted to occur in the discharges with toxicity literature. Chemicals of primary concern for aquatic life in the discharges into the mixing zones are heavy metals, and in particular, zinc (Section 6). Literature on the toxicity of heavy metals indicates that when closely-related salmonid fishes have been tested under similar test conditions, there was little difference between the species (Rand 1985, Chapman 1978). For example, the three *Oncorhynchus* species of concern in the Thompson Creek Mine area, chinook salmon, rainbow trout, and cutthroat trout, had generally similar effects thresholds reported in lethal and sublethal testing with copper and zinc (Chakoumakos 1979, Chapman 1978).

Chapman (1978) tested the relative sensitivities of juvenile steelhead trout and chinook salmon to cadmium, copper, and zinc. Steelhead were consistently more sensitive to these metals than were chinook salmon. Testing the relative toxicity of rainbow trout and brown trout to a metals mixture showed that rainbow trout were consistently the more sensitive species (Marr et al. 1995).

Native sculpin species, including the shorthead sculpin, have been shown through field and laboratory studies to be valuable environmental indicators of metals and other contamination in streams. Field surveys in the Panther Creek, South Fork Coeur d'Alene River, and Boise River systems have shown that native sculpins have been completely eliminated downstream of discharges when salmonids were still relatively abundant, or at least present (unpublished data). This is probably because the benthic sculpins are less motile than salmonids, and less apt to avoid and then re-occupy metals-contaminated habitats.

There is very little toxicological data on freshwater sculpins; what was located suggests that resident species are probably about as sensitive to, or slightly less sensitive to metals than most salmonids. In 96-hour survival tests shorthead sculpins were about as sensitive to cadmium as were cutthroat and rainbow trout, with significant mortality occurring at 0.75 µg/l cadmium (acute criteria was about 1.0 µg/l). They were much less sensitive to zinc and lead than the trout (EVS 1996). EPA lists mottled sculpin as having similar acute toxicity to silver as sensitive salmonids. No reports of toxicity testing with other metals were located for this understudied genus.

Relative sensitivities of bull trout and rainbow trout to copper, cadmium, and zinc under differing hardnesses and temperatures were recently determined (Table 8). Welsh et al. (1999) also completed chronic cadmium toxicity testing with bull trout in 55-day exposures in which lethality and growth were evaluated. The lowest observed effects concentration (LOEC) was 0.79 µg/l cadmium with reduced survival, length, and weight observed. This concentration is higher than the chronic criteria of 0.39 µg/l that would apply at the hardness at which the test was conducted. The no observed effects concentration (NOEC) was 0.38 µg/l cadmium, which is the same as the chronic criterion for cadmium concentrations at the hardness tested. In addition to the results shown here, Welsh et al. (1999) also noted that the acute toxicity of a cadmium/zinc mixture resulted in greater toxicity than cadmium alone to bull trout, and similar toxicity to cadmium alone to rainbow trout.

The data in Table 8 (summarized above) suggests that in soft water, acute lethality to rainbow trout could occur at cadmium continuous ("chronic") criteria concentrations. For zinc in low hardness water, lethality to both bull trout and rainbow trout could occur at continuous ("chronic") criteria concentrations. Overall, bull trout are generally either less sensitive or have similar sensitivity to cadmium, copper, and zinc toxicity as do rainbow trout under the range of conditions tested (Hansen et al. 1999; Welsh et al. 1999). In addition these data, the closely-related brook trout (*Salvelinus fontinalis*), had similar sensitivities reported to copper, cadmium, and lead as did the *Onchoryhchus* species (Rand 1985), although, Chapman (1978) reported that *Salvelinus* species were generally less sensitive to zinc than were *Oncorhynchus* species. In summary, all available information indicates that rainbow trout are generally as sensitive as, or more sensitive than, other salmonids to toxics.

The listed literature reviewed indicated that acute or chronic toxicity to salmonid fish species occurring in the vicinity of the mine discharges and mixing zones (chinook salmon, rainbow/steelhead, cutthroat and bull trout) would be unlikely. Maximum zinc concentrations reported in Thompson or Squaw Creeks (50 µg/l), are less than concentrations reported to affect these salmonids (> 84 µg/l) and are less than zinc criteria for the area (59-114 µg/l for a hardness range of 50-114). Rainbow trout are more abundant downstream of mine discharges than they are upstream, and they are more abundant than other salmonids. Because of this, and because rainbow trout have been reported to be at least as sensitive to metals as bull trout, cutthroat trout, sculpin, or chinook salmon, metals in Thompson Creek have probably not had adverse effects.

This review supports the conclusion that acutely or chronically toxic effects to chinook salmon, steelhead, rainbow, cutthroat, or bull trout are unlikely in Thompson or Squaw Creek under measured or predicted conditions. Additionally, comparisons of criteria for protection of aquatic life with the lowest effects concentrations reports relevant to the threatened or endangered salmonids listed above indicate that the criteria are protective at this site.

Table 8. Relative sensitivity of bull trout and rainbow trout to copper, cadmium, and zinc toxicity in comparison to applicable acute and chronic criteria. Data from Hansen et al. (1999) and Welsh et al. (1999)

Chemical	Hardness (mg/l)	Bull Trout LC_{50} (µg/l)	Rainbow Trout LC_{50} (µg/l)	Acute criteria in µg/l (Not to exceed 1-hour average concentration in a 3-year period)	Chronic criteria in µg/l (Not to exceed 96-hour average concentration in a 3-year period)
		120-hour LC_{50}	**120-hour LC_{50}**		
Copper	100	51 to 68 (2 tests)	36 to 83 (2 tests)	17	11
	220	204 to 218 (2 tests)	85	36	22
		96-hour LC_{50}	**96-hour LC_{50}**		
Cadmium	30	0.91 to 1.0 (3 tests)	0.38 to 0.71 (3 tests)	1.0	0.39
	90	6.06	2.85	3.3	0.87
Zinc	30	32 to 88 (3 tests)	27 to 60 (3 tests)	41	38
	90	315 to 452 (2 tests)	214 to 342 (2 tests)	105	95

Table notes: Results shown for tests with pH at 7.5 and 8º or 12ºC. Cd and Zn test results at lower pH (6.5) resulted in higher LC_{50} values (reduced sensitivity) of both species; Cu test results at higher temperature (16ºC) resulted in higher LC_{50} values (reduced sensitivity) for bull trout, 10 day exposure to high temperature alone (no copper) had no effect on bull trout. Copper LC_{50} values are estimates from presentation graphs. LC_{50} is the concentration lethal to 50% of the test organisms – it is not a safe threshold but a statistic used to compare relative toxicities.

f. Potential for bioaccumulative effects from mercury and selenium in the mixing zone

Careful consideration must be given to the appropriateness of a mixing zone where the constituents may be bioaccumulative or persistent. Concluding that these substances should be eliminated or minimized would be expedient from a regulatory view, ignoring complicating realities such as the facts that all elements are (by their very nature) persistent, and that some substances need to be bioaccumulated to sustain health of organisms. Factors such as the size of the mixing zone, concentration gradients within the zone, duration of exposure, and physical habitat are important in this consideration. Where unsafe fish tissue residues or other evidence indicates a lack of assimilative capacity in a particular water body for bioaccumulative pollutants, care should be taken in calculating discharge limits for these pollutants (EPA 1994). Further, Idaho WQS require that mixing zones be free from toxic chemicals in toxic amounts, which are defined to include substances causing adverse effects through food chain transfer. Of the discharge constituents of concern in this review, mercury and selenium are considered the pollutants of concern for bioaccumulative potential[9].

EPA recently proposed prohibiting the use of mixing zones for discharges of mercury and several persistent chlorinated organic chemicals into the Great Lakes system (EPA 1999b). The factors that EPA based that proposal on were considered for relevance here as follows. Mercury contamination is a significant problem in the Great Lakes system with health advisories in place in all states and provinces cautioning against eating salmonids, walleye, and other game fish. Bald eagles and other piscivorous birds and mammals may be at risk for reproductive impairment due to eating fish with enriched mercury (Table 11). The vast majority of mercury loading to the Great Lakes system is atmospheric. However, discharges to the Great Lakes would occur in the tributaries or nearshore areas of the lakes. The productivity of the nearshore areas, and their importance as spawning and nursery grounds is disproportionately higher than the pelagic areas. The higher productivity of the nearshore zones and circulation patterns of the lakes makes natural sinks for mercury to settle and enter the food chain (EPA 1999b). None of these factors are similar to the ambient conditions in the Thompson Creek study area, so the conceptual basis for a simple prohibition of mixing zones in the Great Lakes system for mercury would not seem to apply in the study area.

Instead, the following methods were taken to consider the potential for bioaccumulative effects in the mixing zones: (1) recent reviews and primary literature on factors relating to bioaccumulative effects of mercury and selenium were surveyed; (2) surface sediments from above and below the existing and proposed discharges were sampled to provide information on sediment exposure to aquatic life and to compare with reference values; (3) ambient water, sediment, and fish tissue

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[9] This mercury literature review and sediment and tissue sampling and analysis were based on a misunderstanding that mercury concentrations in effluents had been measured at up to 2 µg/l, which is a concentration high enough to be of significant concern and would result in criteria exceedances. However, in fact it is EPA's practice to assume that 2 µg/l Hg is present in effluents at this type of mine, regardless of measured concentrations. Measured concentrations were <0.05 µg/l which would not exceed water quality criteria under most scenarios. Because of the considerable effort invested in food chain sampling and analyses, compiling and analyzing the information, it is included in the report even though mercury turned out to be more a of procedural contaminant, than an environmental contaminant of concern.

values were compiled and compared with ecological benchmark values; (4) the projected exposure area (mixing zone) was compared to ranges of potentially affected wildlife; and (5) the information was evaluated as a whole to conclude whether adverse bioaccumulative effects were likely in the mixing zone.

Mercury

Bioaccumulation in the environment is controlled by (1) the nature and duration of the exposure, including bioavailability of the chemical form; and (2) the nature and kinetics of processes that determine the rate, distribution, and magnitude of chemical accumulation in the organism (Spacie et al. 1995). These factors vary greatly depending upon the nature of the freshwater ecosystem.

Mercury primarily exists in two forms in aquatic ecosystems: inorganic ionic mercury and methylated organic methylmercury. Methylmercury is the form of mercury of particular concern in aquatic ecosystems for three reasons:

(1) All forms of mercury can be converted to methylmercury by natural processes in the environment;
(2) Methylmercury bioaccumulates and biomagnifies in aquatic food webs; and,
(3) Methylmercury is the most toxic form of mercury (EPA 1997b).

Nearly all (<90%) mercury in surface waters is inorganic mercury, yet nearly all (95-99%) of mercury in fish tissue is methylmercury. Diet contributes more than 90% of methylmercury in fish in natural waters (Weiner and Spry 1996). Thus, to evaluate the risk of mercury in a discharge, it is necessary to consider factors affecting its form.

The major source of methylmercury in natural waters is the methylation of mercury in sediments by anaerobic sulfur-reducing bacteria. The major source of methylation in aquatic systems is the sediment, but methylation can occur in the water column under certain conditions. Mercury methylation rates increase with low pH, and increase with increased microbial action, mercury loading, suspended sediment load, water column dissolved organic carbon (DOC), sediment total organic carbon (TOC), sediment redox conditions, and temperature. Highly productive lentic or wetland environments are favorable sites for high rates of mercury methylation (EPA 1997b; Zillioux et al. 1993). Curiously, selenium may interfere with mercury uptake by fish, and has actually been added to lakes to mitigate mercury bioaccumulation (Weiner and Spry 1996).

In contrast to these conditions, the great majority of the energy budget in mountain streams in this area is allochthonous; contributions from primary productivity are low (Minshall et al. 1992). In addition to the stream gradient, this lower productivity will limit organic detritus, reducing sediment conditions and microbial activity, and consequently limiting methylation. A further characteristic of this stream type is a frequent exchange of stream water with the water in the hyporheic zone. IDEQ sampling of the steams in the mountainous study area show that they have high gradients (0.5 -5%), coarse textured substrates, and co-occurring fauna that require well-oxygenated substrate interstitial conditions.

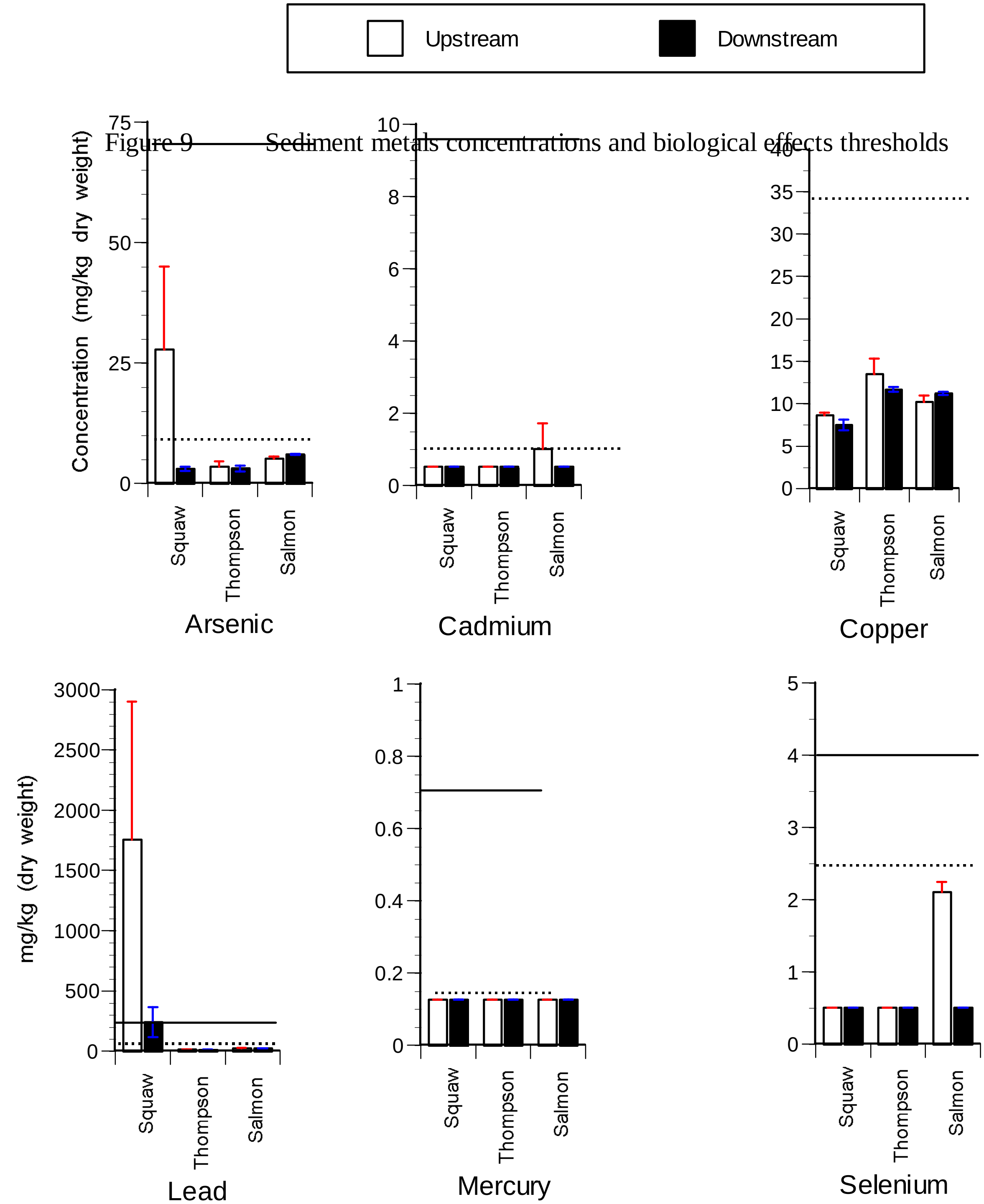
Mercury loading is also a limiting factor in mercury methylation and bioaccumulation. Tables 9-11 list mercury concentrations in water, sediment, and fish tissue from study area, regional, and broadscale studies. Rainwater and snow in the study area may contain inorganic mercury concentrations near the chronic aquatic life criterion, and greatly in excess of the EPA wildlife criterion (non-regulatory) (Table 9).

Mercury in Sediments

As noted above, sediments are the major source of methylmercury in aquatic systems. Sediment-sorbed contaminants can be an intermediate repository in aquatic ecosystems and can be directly toxic to aquatic life, or may be a source of contaminants for bioaccumulation in the food chain. Further, concentrations of trace elements in sediments may be several orders of magnitude higher than in the overlying water column (Ingersoll 1995). This is particularly useful because of the difficulty in quantifying mercury at ambient concentrations. Adverse effects to benthic communities or benthic-pelagic food chains can occur in areas where the overlying water quality criteria are not exceeded. For example, grossly contaminated Lake Ononodaga, NY has significant mercury contamination of its sediments with dietary exposure and associated adverse effects to fish and wildlife, yet water column mercury concentrations were below or near criteria levels, 7 – 19 ng/l (Weiner and Spry 1996).

Surface sediments were collected by DEQ in an effort to determine if there was a metals signal from the Thompson Creek discharges, and if sediment-sorbed metals concentrations exceeded sediment quality guidelines. Sediments were collected in December 1999 from depositional areas near established water chemistry monitoring stations upstream and downstream of ongoing and proposed discharges. Samples were analyzed for several metals (Figure 9). All sediment mercury concentrations were <0.2 mg/kg (dry weight). Assuming the true concentrations of mercury were the nondetect values, these values would be within or below ranges reported from uncontaminated systems and at or below the ecological thresholds of concern (Table 10). Mercury residues sampled in cutthroat trout tissue from the nearby Yankee Fork watershed were not elevated, although sediment mercury concentrations were elevated. Fish were not sampled from the Thompson Creek area; however, sediment mercury concentrations in the Thompson Creek vicinity were lower than those collected in the Yankee Fork drainage (Tables 10 and 11). It stands to reason that mercury residues in fish are not expected to be higher in the Thompson Creek system than in the Yankee Fork watershed. In conclusion, low level inorganic mercury discharges at concentrations within the range of those occurring in snow and rainfall in the region, or those concentrations occurring in natural waters not directly affected by anthropogenic sources, is unlikely to result in risk of bioaccumulation in the study area or downstream waters.

Figure 9 Sediment metals concentrations and biological effects thresholds



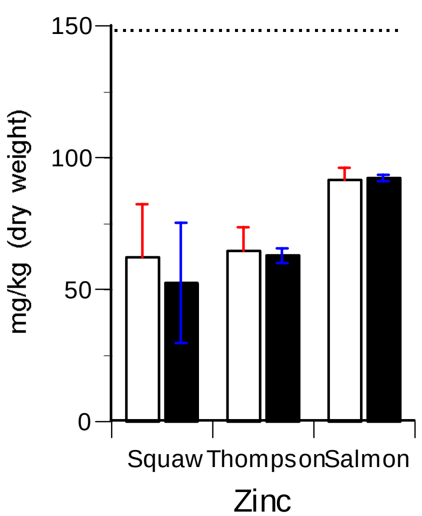


Figure 9. Potential biological effects thresholds and distribution of metals in surficial sediments from above and below the mixing zones for discharges from the Thompson Creek Mine. Solid lines indicate selected sediment quality guidelines ER-Ms, effects ranges-median. Adverse benthic impacts are probable at concentrations above ER-Ms, adverse effects are unlikely below the effects range-low ER-L, dashed lines (see text). Selenium apparent effects thresholds are based on protection from bioaccumulation. Error bars indicate standard deviation of laboratory replicates. Non-detected chemicals are plotted as 1/2 detection limits.

Table 9. Ranges of total dissolved mercury concentrations in natural waters and threshold concentrations of concern (in parts per trillion, nanograms/liter).

Situation	Water (ng/l) (ranges)			Reference
Mercury				
Thompson Creek, Squaw Creek, and Salmon River in vicinity of Thompson Creek Mine discharges			<50	TCMC database
Background in North American surface waters not directly affected by anthropogenic sources	0.1	-	20	EPA 1997a
Mercury in rainwater in areas not directly affected by anthropogenic sources (North American ranges)	3	-	<100	EPA 1997a
Mercury in snowfall in the central Idaho desert	1.9	-	5	D.D. Susong, USGS, personal communication
Mercury in snowfall in the Teton Mountains, Wyoming	1.8	-	4.6	
Canadian lakes and rivers not directly affected by anthropogenic sources	10	-	100	Moore et al. 1984
Greenland ice caps	2	-	19	Moore et al. 1984
Thresholds of concern				
Idaho acute criterion for protection of aquatic life			2000	WQS §250
EPA acute criterion for protection of aquatic life			1400	EPA 1999
Idaho chronic criterion for protection of aquatic life			12	WQS § 250
EPA chronic criterion for protection of aquatic life			770	EPA 1999
EPA lowest wildlife criterion for protection of piscivorous wildlife (belted kingfisher)			0.59	EPA 1997b
EPA wildlife criterion for protection of bald eagles			1.8	EPA 1997b

Table 10. Ranges of total mercury in stream sediments and threshold concentrations of concern.

Situation	Sediment (mg/kg dry wt)			Reference
Mercury				
Thompson Creek, Squaw Creek, and Salmon River in vicinity of Thompson Creek Mine discharges			<0.2	This study
Yankee Fork of the Salmon River in vicinity of current and historic mining activities	<0.19	-	0.57	IDEQ unpub. data July 1996
Northern Rockies – non-mining areas	0.02	-	0.1	USGS unpub. data
Northern Rockies –mining or mineralized areas	0.02	-	6	USGS unpub. data
Snake River basin	<0.02	-	0.13	Clark and Maret 1998
United States non urban indicator streambed baseline	0.01	-	0.26	Rice 1999
Unpolluted Canadian lakes	0.04		0.3	Moore et al 1984
Thresholds of concern				
Effects-range low (ER-L), below which adverse biological effects are unlikely (EC10)			0.15	Long et al. 1995
Effects-range median (ER-M), above which adverse biological effects are probable (EC50)			1.3	Long et al. 1995
Threshold to protect clapper rail, a benthic forager, (surrogate for American dipper)			0.2	USDOI 1998

Table 11. Ranges of mercury in aquatic tissue concentrations and thresholds of concern

Site	Matrix	Fish tissue (mg/kg wet wt)	Fish tissue (mg/kg dry wt)	Reference
Mercury				
Thompson Creek above and below mine outfalls, and from in Outfall 001 (*Cutthroat trout (14) and shorthead sculpin (15)*)	Whole body	<0.05	<0.2	Chadwick 2000d
Yankee Fork drainage – upstream of mined areas *(Cutthroat trout)*	Liver	<0.05	<0.2	USFWS unpub. data, July 1996
Yankee Fork drainage – downstream of mined areas *(Cutthroat trout)*	Liver	<0.05	<0.2	USFWS unpub. data July 1996
Yankee Fork drainage – upstream of mined areas *(Cutthroat trout)*	Muscle	<0.05	<0.2	USFWS unpub. data July 1996
Yankee Fork drainage – downstream of mined areas *(Cutthroat trout)*	Muscle	<0.05	<0.2	USFWS unpub. data July 1996
Salmon River at Whitebird *(Largescale sucker)*	Liver	0.01	0.4	Clark and Maret 1998
Upper Snake River basin (various species)	Muscle	0.08 – 0.29	0.4 – 1.4	Clark and Maret 1998
Upper Snake River basin (various species)	Liver	<0.025 – 0.22	<0.1 – 1.0	Clark and Maret 1998
National averages for piscivorous fish	Muscle	0.18 – 0.35	0.7 – 1.4	EPA 1997b
Thresholds of concern				
Critical tissue concentrations for adverse effects to salmonids	Muscle	10 - 20		Wiener and Spry 1996
Salmonid NOEC	Muscle	5	20	Wiener and Spry 1996
FDA action level	Muscle	1	4	
Fish tissue concentration resulting in reproductive impairment in piscivorous birds (common loon)	Whole body	0.3	1.2	USDOI 1998

<u>Selenium</u>

Runoff from the open pit waste rock dumps contains elevated concentrations of selenium, which requires careful consideration of its potential effects. Selenium contamination of aquatic systems is of widespread potential concern throughout much of the Western United States. In particular, research in two reservoir systems with severe adverse effects on aquatic life has influenced water criteria development and hazard assessment practices for selenium. Moderate (≈ 10 µg/l) selenium concentrations in Belews Lake, NC resulted in dietary exposures to fish of about 20-30 mg/kg dry weight, with severe impacts (loss of species) to the warmwater fish assemblage. Naturally-occurring selenium leached from soils due to irrigated agriculture in California bioaccumulated to over 100 mg/kg dry weight in prey items with severe reproductive impairment in aquatic birds (Lemly 1996, 1997; Peterson and Nebeker 1992). Further, declines in endemic fishes in the Colorado River system that were coincident with elevated selenium in irrigation drains by the 1930s, before major dam effects, implicates selenium as a factor inhibiting the fishes' recovery from their depressed populations (Hamilton 1999). The current chronic water quality criterion for selenium is 5 µg/l, which, unlike criteria for other trace elements, is expressed as total recoverable selenium rather than dissolved, because of the importance of the particulate-bound selenium in the food chain (EPA 1987).

There appears to be consensus in the literature that most of the selenium in fish tissues results from uptake through the diet rather than through water (Lemly 1996, Canton and Van Derveer 1997). There appears to be further consensus on there being two major routes of exposure: (1) strong bioconcentration of inorganic selenium by phytoplankton after which it is bioaccumulated by zooplankton and planktonic forage fish, and is recycled into the food chain from decaying plant matter; (2) particulate-sorbed selenium settles out, and is bioconcentrated by detrial microbial communities. In both mechanisms, settling of particulate-sorbed selenium and the uptake of selenium by bacteria, algae, and zooplankton can efficiently scavenge selenium from the water column, reducing water concentrations, which could give a confounding impression of selenium exposure and aquatic risk. In this situation, the concentrations in fish and wildlife tissues may be much higher than would be predicted solely on the basis of total waterborne selenium (Besser et al. 1993, Lemly 1996, Canton and Van Derveer 1997).

Apparently there is little or no trophic biomagnification of selenium in aquatic food chains, e.g. from zooplankton or aquatic insects to fish tissues (Besser et al. 1993; SDDENR 1996. Selenium is an essential micronutrient that may be regulated (i.e. either preferentially concentrated or depurated if scarce or elevated) by aquatic insects. Maier et al (1998) showed that when selenium was added to a selenium-deficient watershed by aerial spraying, aquatic insects from a slow-moving, low gradient stream quickly took up and retained the available selenium. Adams et al. (2000) review showed higher bioaccumulation factors (BAF) from concentrations in water to concentrations in fish occurred when selenium was scarce, than when enriched, as well as higher bioaccumulation from lentic than from lotic environments. Other important factors in the risk of selenium exposure to the environment are the substrate type and speciation. In uptake experiments, fine-organic rich sediments caused a more rapid flux of selenium into the substrate than coarse sand. In the water column, similar food-chain selenium residues resulted from 0.04 µg/l organic selenium, 10 µg/l inorganic selenite, and 100 µg/l inorganic selenate (Lemly et al. 1993). A significant portion of the selenium in natural waters can occur as organic selenium in some systems (Besser et al. 1993, Lemly et al. 1993). All selenium in several Thompson Creek discharges speciated in December 1999 was inorganic selenate (TCMC 2000). Some results have shown that sodium selenate is less toxic than sodium selenite in aqueous exposures but more toxic in dietary exposures (Besser et al.,1993, Jarvinen and Ankley 1999).

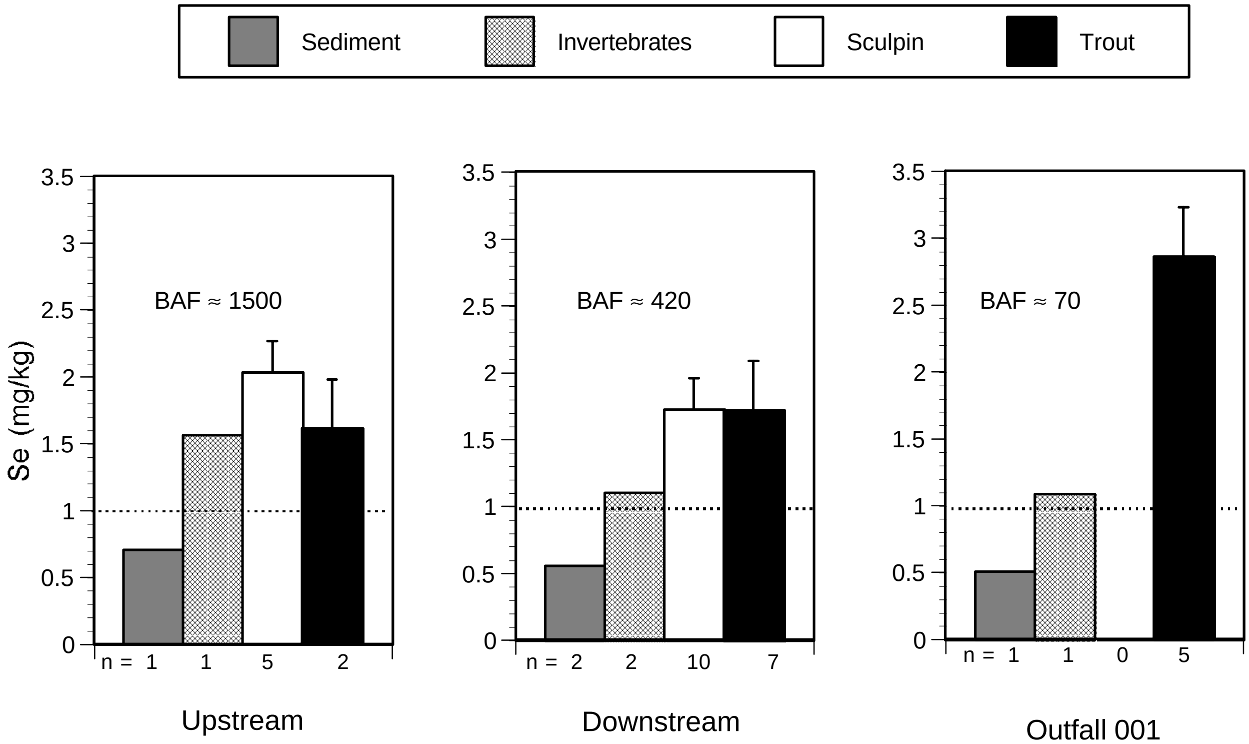


Figure 10. Mean selenium concentrations from sediment, invertebrates, sculpin, and rainbow and cutthroat trout collected from Thompson Creek and Outfall 001 (Buckskin Creek), in April 2000. Downstream data are combined from stations TC1 and TC3. Invertebrate samples are composites of the invertebrate community. Fish samples are whole body. Dashed lines show the lowest threshold of tissue concentrations that have been associated with reduced growth of salmonids in the literature (from Table 14). Sediment concentrations are dry weight, tissue concentrations are wet weight. Error bars show SD.

Bioaccumulation factors (BAF) are the ratio of average selenium tissue concentrations in fish and the selenium concentrations in the water (tissue µg/kg (wet wt.) to water µg/l). Sculpin and trout tissue concentrations were pooled since these insectivorous fish share habitats and are at the same trophic level. These tissue : water ratios were 1819 to 1.2, 1719 to 4.0, and 2860 to 40 for upstream of Outfall 001, downstream, and in Outfall 001 respectively.

Data from Chadwick 2000d.

Recent lentic-lotic and water-sediment selenium exposure pathway controversy

From an environmental management perspective, it would be nice if this consensus of scientific opinion continued. Unfortunately it has not. Adverse effects have been observed or predicted in reservoirs, marshes, irrigation drains, and other lentic or slow moving riverine and estuarine systems at 1-5 µg/l total selenium (Peterson and Nebeker 1992, Lemly et al. 1993). Other workers observed no apparent effects in lotic systems with 20 µg/l or greater selenium and argued that a sediment-based pathway would better describe aquatic selenium exposure than would a waterborne selenium pathway in western lotic systems. From a review of 27 selenium studies of western streams, they constructed an empirical model of predicted selenium effects as a function of sediment selenium and organic carbon (Canton and Van Derveer 1997; Van Derveer and Canton 1997). Using a range of stream organic carbon values, and backcalculating waterborne sediment concentrations, their model would predict chronic selenium concentrations ranging from <2 µg/l (low-gradient system with fine-grained sediments) to >30 µg/l (high-gradient systems with coarse-grained sediments). Hamilton and Lemly (1999) argued that the approach of determining apparent effects thresholds based upon field co-occurrence studies failed to demonstrate causality, and gave examples of toxic effects occurring below concentrations of 5 µg/l total selenium, among other criticisms. There was concurrence that the lower productivity of some lotic systems relative to lentic systems, would likely lower the bioaccumulation potential of selenium, and effects on lotic systems may not be apparent.

Hamilton and Lemly (1999) compared predicted "safe" sediment model results (30 µg/l) that were calculated for a mountain stream with 0.5% sediment organic carbon to experimental stream studies at Monticello, Minnesota, in which adverse effects to fathead minnows and bluegills occurred at 10 µg/l total selenium (Hermanutz et al. 1992). However, the streams are dissimilar. The Monticello streams were constructed with a low-gradient series of mud-bottomed pools, separated by gravelly riffles. While sediment organic carbon values were not reported, a stream with cladocerans, copepods, and rotifers abundant, is suitable habitat for bluegills, and has dissolved organic carbon ranging from 8 – 16 mg/l (Hermanutz et al. 1987), describes a near-lentic, low-gradient, slow-moving system, more like a pond, western slough, or backwater than a mountain stream. Assuming a typical sediment organic carbon value for a low-gradient stream of 2.5 – 7%, the Van Derveer and Canton model would predict a safe selenium criterion of 6 – 2 µg/l respectively. This modeled "safe" value would not conflict with the Hermanutz et al. (1992) findings of adverse effects to bluegills and minnows at 10 µg/l selenium.

To further evaluate the lotic-lentic selenium bioavailability question, in relation to estimating bioaccumulation risk in the study area, sediment model results were calculated for data sets from several western lotic systems: Sediment model results were calculated for Panther Creek, Idaho, a high gradient 3rd to 5th order stream with dynamics similar to the study area, which had been characterized over a 40 km length; the Clark Fork River, Montana, a large low-gradient western river; the Snake River, a large low- to moderate-gradient western river, and median values for 20 river basins nationwide. These results were compared with data from the study area, Thompson and Squaw Creeks. Model results for these river segments range from 3 – 8 µg/l (Table 12). The convergence, or at least lack of conflict, of these model results with other literature on selenium effects suggests that criticisms of the sediment-detrital model may be more conceptual than practical. Because of its relevance to lotic systems and simplicity, the Van Derveer and Canton (1997) sediment selenium model is one useful line of reasoning to evaluate potential for chronic effects on aquatic life from selenium. Using Thompson Creek, Squaw Creek, and nearby Panther Creek data, adverse selenium effects <u>could result</u> from average sediment Se concentrations >2.5 mg/kg and water Se >8 µg/l. Adverse effects would be probable if <u>sediment</u> Se concentrations >4.0 mg/kg and water Se >14 µg/l. Said another way, ambient waterborne concentrations of selenium up to the chronic criteria of 5 µg/l would be unlikely to have adverse effects in the site streams.

Table 12. Ranges of selenium concentrations in natural waters and threshold concentrations of concern (part per billion, micrograms/liter).

Situation	**Water (µg/l) (ranges)**			**Reference**
Selenium				
Thompson Creek upstream of TCMC discharges	<0.5		3	Figure 6; TCMC database
Thompson Creek downstream of TCMC discharges	3		6	Figure 6; TCMC database
Salmon River and Squaw Cr above and below mine discharges				TCMC database
Background in North American surface waters not directly affected by anthropogenic sources	0.1	-	0.4	USDOI 1998
Southeast Idaho phosphate mining region	<0.5	-	260	MW 1999
Kesterson Reservoir tributary drains			330	Hamilton et al 1986
Irrigation drainwaters in the Colorado River basin	55	-	>2000	Hamilton 1999
Thresholds of concern				
Idaho chronic and acute aquatic life criteria	5		20	WQS §210
Reduced growth and survival for juvenile chinook salmon (90d test)			≈50	Hamilton et al. 1986
Reduced survival with rainbow trout (90d selenite)			47	Lemly et al. 1993
Lowest wildlife criterion for protection of piscivorous wildlife (belted kingfisher), calculated for Kesterson Reservoir, CA			0.9	Peterson and Nebeker 1992
Wildlife criterion for protection of bald eagles, calculated for Kesterson Reservoir, CA			1.9	Peterson and Nebeker 1992
Necessary concentration specified for dilution waters used for chronic *Ceriodaphnia* WET tests	2		2	Lewis 1994
Threshold for significant chronic adverse effects to fish in reservoirs and ponds with pelagic foodwebs	2	-	5	Lemly 1996
Thresholds for hydrologic unit-specific adverse chronic effects in Western streams based on a sediment-detrital based foodweb, and limited primary productivity	2	-	30	Van Derveer and Canton 1997
Site-specific chronic Se threshold (possible effects – probable effects) model results calculated from McNeil sediment core samples from Thompson and Squaw Creeks, conditions (1.9% mean TOC, n=19)	8		13	Van Derveer and Canton 1997; TCMC data
Site -specific chronic Se threshold (possible effects – probable effects) model results calculated for Panther Creek, ID conditions (1.9% mean TOC, n=25)	8		14	Van Derveer and Canton 1997; Mebane 1994
Site -specific chronic Se threshold calculated for Clark Fork River, MT (3.5% mean TOC, n=6)			4	Van Derveer and Canton 1997, Brumbaugh et al. 1994
Site-specific chronic Se threshold calculated for Snake River, ID conditions (2.5% mean TOC, n=17)			6	Van Derveer and Canton 1997; Clark and Maret 1998
Range of model results calculated for 25th and 75th percentiles of TOC values from 541 stream sites across the US	3		8	Van Derveer and Canton 1997; Rice 1999

Table 13. Ranges of total selenium in stream sediments and threshold concentrations of concern.

Situation	Sediment (mg/kg dry wt)			Reference
Selenium				
Thompson Creek, Squaw Creek, and Salmon River in vicinity of Thompson Creek Mine discharges	<1	-	2.1	This study
Southeast Idaho phosphate mining region	<0.2	-	9.4	MW 1999
Northern Rockies – non-mining areas	0.1	-	0.9	USGS unpub. data
Northern Rockies –mining or mineralized areas	0.2	-	0.6	USGS unpub. data
Upper Snake River basin	0.3	-	2.5	Clark and Maret 1998
United States non urban indicator streambed baseline (range of 20 study area medians)	0.3	-	1.2	Rice 1999
Approximate background, normal freshwater environments	0.2	-	2.0	USDOI 1998
Thresholds of concern				
EC10 for fish and birds in 27 Western U.S. rivers and streams (population basis)			2.5	Van Derveer and Canton 1997
LOEC for fish and birds in 27 Western U.S. rivers and streams (population basis)			3.5	Van Derveer and Canton 1997
EC100 for fish and birds in 27 Western U.S. rivers and streams (population basis)			4.0	Van Derveer and Canton 1997

Table 14. Ranges of selenium aquatic tissue concentrations and thresholds of concern. See also Thompson Creek values in Figure 10.

Site	Matrix	Tissue (mg/kg wet wt)	Tissue (mg/kg dry wt)	Reference
Selenium				
Yankee Fork drainage – upstream of mined areas *(Cutthroat trout)*	Liver	0.81 – 1.03	3.5 – 4.1	USFWS unpub. data
Yankee Fork drainage – downstream of mined areas (lower Jordan Cr) *(Cutthroat trout)*	Liver	1.7	6.8	" "
Salmon River at Whitebird *(Largescale sucker)*	Liver	1.2	5.5	Clark and Maret 1998
Southeast Idaho phosphate mining region *(Cutthroat trout)*	Muscle	1.2 – 6	4.8 – 24	MW 1999
Upper Snake River basin (various species)	Muscle	0.14 – 0.44	0.7 – 2.1	Clark and Maret 1998
Upper Snake River basin (various species)	Liver	0.4 - 1.7	1.7 – 7.9	Clark and Maret 1998
Typical hepatic background	Liver	0.5 - 2	2 - 8	USDOI 1998
Thresholds of concern				
No reproductive or teratogenic effects in wild cutthroat trout collected from waters with ≈13 –28 µg/l selenium	Liver Eggs Muscle		18 – 114 9 – 58 7 - 41	Kennedy et al. 2000
Reduced cutthroat trout egg viability	Eggs		81	Kennedy et al. 2000
Smoltification and downstream migration disrupted in chinook salmon (6:1 selenate:selenite waterborne exposure, 30d test)	Whole body	2.1	9.6	Hamilton et al. 1986
Reduced growth or survival of juvenile chinook salmon (6:1 selenate:selenite mixture, 30d test)	Whole body	>4.9	>23	Hamilton et al. 1986
Reduced survival of juvenile chinook exposed to organic selenomethionine (SeMet) through diet (90d test)	Whole body	2.7	10.8	Hamilton et al. 1990
Reduced growth of juvenile chinook fed minnow-meal diet collected from Kesterson area drains which contained SeMet (5.3 mg/kg dw) (90d test)	Whole body	1.0	4.0	Hamilton et al. 1990
Reduced survival of rainbow trout exposed to 47 µg/l waterborne sodium selenite (NOEC = 12µg/l, 90d test)	Whole body	1.07 (LOEC) 0.4 – 0.9 (NOEC)	4.3 1.7 – 3.5	Hunn et al 1987
Effects threshold for larval salmonids, from critical literature review and regression models	Whole body	1.5	6	DeForest et al. 1999
Dietary threshold for larval salmonids, from critical literature review and regression models	Whole body		11	DeForest et al. 1999

In addition to this review, Chadwick (2000a, appended) recently reviewed fate and effects of mercury and selenium in stream environments. The context of their evaluation was bioavailability and potential effects from discharges into a similar, nearby environment (Yankee Fork of the Salmon River). Nothing in their review was specific to the Yankee Fork, as opposed to other oligotrophic mountain streams and is germane for the Thompson Creek study area. Some of their conclusions were similar to this report, with the exception of the application of recommended instream targets for compliance with water quality standards. Chadwick recommends the current EPA chronic mercury criterion of 0.77 µg/l and a site-specific chronic selenium criterion based upon an empirical sediment-based model as the most appropriate protective instream targets. We have not made any determination on adopting the new EPA criteria, and, site-specific criteria may only be applied in NPDES permitting if they have been formally adopted by the state, which these have not. These regulatory objections notwithstanding, no information in their review does not conflict with the conclusions that the more stringent (in this application) existing state-wide chronic criteria of 0.012 µg/l mercury and 5 µg/l selenium would therefore also be protective in this application of the standards.

Potential wildlife exposure to bioaccumulative substances in mixing zones

EPA guidance on evaluating exposure to bioaccumulative substances is limited to human health concerns, without regard to wildlife exposure (EPA 1991a). However, Idaho's water quality standards consider wildlife habitat to be a designated use for all waters. Generally, water quality standards to protect human health and recreational uses are presumed to adequately protect wildlife use of these habitats (EPA 1994). However, since a mixing zone is an area where numeric water quality standards do not apply, the standard that waters be "free from toxic substances in concentrations that impair beneficial uses" needs to be complied with. To evaluate this, the approximate size of the stream habitats that could be affected by the mixing zones (described later) was compared with the approximate foraging ranges of selected piscivorous animals (Table 15). In the calculations, it is assumed that the animals are centering their territory around the mixing zones, and in Thompson the territories straddle and include both mixing zones. This is admittedly an unlikely exposure scenario; similarly no attempt to reconcile referenced foraging ranges with regional conditions was made. For example, a pair of bald eagles could doubtfully sustain themselves along a 3 km reach of Thompson Creek. Thus risk of exposure would be worst case estimates to individual pairs of animals, not to populations. Each mixing zone is assumed to be 200m in length on Thompson Creek and 50m in length on Squaw Creek and the Salmon River. If concentrations of mercury and selenium exposure in the mixing zones were to result in insect or fish tissues exceeding dietary thresholds, then the exposed percentages of foraging ranges for these hypothetical individual animals range from trivial to about 40%.

Table 15. Potential wildlife exposure to bioaccumulative contaminants in mixing zones

Receptor	Foraging range (km shoreline)[1]	Potentially exposed habitat (stream km)		Worst case % of foraging range in exposed habitats
American dipper[2]	1 – 3	0.1	Thompson Creek	3 – 10
		0.05	Squaw Creek	2 – 5
		0.05	Salmon River	2 – 5
Bald eagle (Threatened)	3 – 26	0.1	Thompson Creek	0.4 – 3
		0.05	Squaw Creek	0.2 – 2
		0.05	Salmon River	0.2 – 2
Belted kingfisher[3]	5 – 9	0.1	Thompson Creek	1 – 2
		0.05	Squaw Creek	0.5 - 1
		0.05	Salmon River	0.5 - 1
Mink	1 – 5	0.1	Thompson Creek	2- 10
		0.05	Squaw Creek	1 – 5
		0.05	Salmon River	1 – 5
River otter	10 – 78	0.1	Thompson Creek	0.1 - 1
		0.05	Squaw Creek	0.05 - 0.5
		0.05	Salmon River	0.05 - 0.5

[1] McVey et al. 1993

[2] No information, range is for American robin, a terrestrial insectivorous passerine (dippers are semi-aquatic insectivorous passerines.

[3] No foraging range given, estimated from shoreline density values.

g. Zone of passage for migratory fish through the mixing zone

One of the factors to consider in applying a mixing zone is the need to preserve a zone of passage through or around the mixing zone for migrating fish or other organisms. Migratory fish species must be able to reach suitable spawning and living areas. Juveniles, and in some cases adults, must be assured a return route to their growing and living areas. Barriers or blocks that prevent or interfere with these types of essential transport and movement can be created by water with inadequate chemical or physical quality (EPA 1994). Idaho's water quality standards require that waters be "free from toxic substances in concentrations that impair beneficial uses." Toxic substances are defined to include substances which when discharged into waters of the state will cause behavioral abnormalities (WQS §003.105; §200.02). Hence, the potential of behavioral avoidance to substances included in the Thompson Creek Mine discharges is evaluated.

The purpose of this review is to relate literature on the avoidance behavior of salmonids to metals of potential concern in the mixing zones of the existing or proposed discharge outfalls. All salmonids of concern in the drainages (including chinook salmon, steelhead, bull, and cutthroat trout) are believed to migrate in and out of tributary streams at different parts of their life history. Fish rely on chemoreception in their homing behavior, pairing, and avoiding chemical stressors, including metals. Since in some cases fish have been shown to have their upstream passage blocked when encountering elevated metals concentrations, potential avoidance of the mixing zone by fish must be considered.

Anadromous salmon and steelhead trout are obvious examples of migratory fish that require unimpeded passage. The Salmon River is a critical migratory pathway for any remaining endangered Snake River sockeye salmon. Access to large amounts of critical habitat for threatened chinook salmon and steelhead trout is past the proposed Salmon River outfall.

Non-anadromous fish also migrate and require a zone of passage through or around mixing zones. Some cutthroat trout and bull trout also have highly migratory life strategies. These fluvial[10] fish may enter headwater streams to forage, take refuge from warm summer river temperatures, spawn and rear in headwater streams, but move downstream in the fall to overwinter in larger rivers. Seasonal migrations of up to 100 km have been documented in the Salmon River system. Bull, cutthroat, and rainbow trout also have stream resident (non-fluvial) life strategies. These fish remain in their natal streams their entire life. Resident fish will not reach as large of sizes as fluvial fish. For example, adult resident bull trout reach about 25 cm in length; fluvial bull trout may exceed 60 cm (Meehan and Bjornn 1991). No fluvial-sized trout were mentioned in the Thompson and Squaw Creek monitoring reports, suggesting that the trout there have primarily or exclusively a resident life history strategy. However, even "non-migratory" forms of trout undertake limited migrations that must not be prevented by a mixing zone. In Montana headwater streams that are similar to Thompson and Squaw, stream resident bull and cutthroat trout moved downstream >1 km in fall to deeper, quiet water to overwinter. Some bull trout made additional winter movements of up to 2 more km depending on ice conditions (Jakober et al. 1998). Winter is a very stressful time for trout, and additional stress from mixing zone avoidance could reduce survival. Bull trout in particular inhabit naturally fragmented habitat patches. Connections

[10] Fish that migrate between main rivers and tributaries.

between these patches are important for refugia from disturbances such as wildfires, and for recolonizing habitats. Resident trout populations in smaller, isolated patches may be at risk of extinction (Rieman et al. 1997, Dunham and Rieman 1999). Hence, to protect the integrity of these aquatic ecosystems, mixing zones cannot become a barrier to salmonid movements.

Mixing zone avoidance could be directly evaluated for existing discharges through biotelemetry, or, with less accuracy, mark and recapture, or distribution and abundance surveys. Further, the potential for avoidance may be estimated by comparing literature reports of fish behavioral avoidance of chemical concentrations with measured or predicted field concentrations. Controlled preference-avoidance studies with fish have repeatedly shown that many chemicals or effluent mixtures are avoided by fish. Several field observations of avoidance of natural waters containing waste discharges have also been reported (Giattina and Garton 1983, Atchison et al. 1987). (Sprague et al. 1965, Sutterlin and Gray 1973, Geckler, et al. 1976, Damkaer and Dey 1989, Gray 1990, Smith and Bailey 1990, Atland and Barluup 1995). However, due to the lack of experimental control in most of the field studies and the natural complexity of territorial, social, predatory, and reproductive behavior, field verification of experimental concentration-response relationships is difficult (Little 1990). Acknowledging these limitations, this review attempts to evaluate potential effects and probable safe conditions for migratory fishes in relation to proposed point source discharges.

Concentrations of Metals Shown to Cause Avoidance

Substances of potential concern in the proposed discharges include: arsenic, cadmium, chromium, copper, nickel, lead, mercury, selenium, silver, and zinc. Literature reports on behavioral changes by fish to these metals were reviewed and the CARL Uncover, EPA aquatic toxicity information retrieval (AQUIRE), and other databases were searched using the following effects keywords: Avoidance, behavior, detection, drift, equilibrium, food consumption, immobilization, locomotor behavior, migration, predatory behavior, predatory avoidance, and stress. Effects concentrations well outside the range of possible instream concentrations were not further investigated or listed here. A brief discussion of literature relevant to fish avoidance of each metal follows.

Arsenic

No reports of fish avoidance to arsenic were located. The estimated threshold for sublethal, chronic toxicity of arsenic (as arsenite) to rainbow trout *(Oncorhynchus mykiss)* is 4,900 µg/l (Rankin and Dixon 1994), while the current Idaho criteria for protecting human health is 50 µg/l. This exceeds anticipated potential discharge concentrations. Arsenic is unlikely to be a concern for the zone of passage of fish through the mixing zone.

Cadmium

Cadmium has been reported to be toxic at concentrations lower than fish can detect and avoid (Atchison et al. 1987). Woodward et al. (1997) reported no response by cutthroat trout *(Oncorhynchus clarki)* in avoidance testing with a cadmium concentration of 0.66µg/l. McNichol and Sherer (1991) reported that lake whitefish *(Coregonus clupeaformis)* avoided cadmium at

0.2µg/l; however, the avoidance was only significant at less than 1 and more than 8 µg/l. In further testing, lake whitefish showed a neutral response to cadmium at 0.2, 1, and 5 (McNichol and Sherer 1993). 8 µg/l is considered the avoidance threshold of concern for cadmium.

Chromium

Avoidance of chromium (VI) by rainbow trout has been reported from laboratory studies at concentrations ranging from 10 – 80 µg/l, and with golden shiners at 58 –95 µg/l (Anestis and Neufeld 1986, Hartwell et al. 1989 respectively). These laboratory thresholds are similar to the chronic criteria of 11 µg/l.

Copper

Copper has been well documented to cause avoidance with salmonids and other fishes in laboratory and field conditions. In laboratory tests, copper and zinc mixtures have been shown to act together to cause a lower threshold of avoidance than would result from either metal alone (Giattina and Garton 1983). Table 15 lists copper concentrations associated with avoidance responses in fish in laboratory testing. Based upon the internal consistency of responses reported within a study, consistency of responses reported between studies, similarity of test species to the species occurring in the study area, a preference for more recent study results over older studies when comparing similar studies, and whether actual test concentrations were reported to have been measured vice nominal concentrations, an avoidance threshold of 3 µg/l copper was selected.

Lead

Woodward et al. (1997) reported no response by cutthroat trout in avoidance testing with lead concentrations of 1.3µg/l. Adams (1975) reported that 14.3 µg/l zinc caused avoidance in brook trout, Sherer and McNicol (1998) reported 10 µg/l lead was avoided by lake whitefish, and Giattina and Garton (1983) reported rainbow trout avoided lead at 26 µg/l. These values are higher than chronic criteria of 0.6 – 2 µg/l depending upon which stream and season it is (variation in hardness). Lead concentrations in the discharge are unlikely to exceed avoidance thresholds, so lead is unlikely to be a concern for the zone of passage for fish through the mixing zone.

Mercury

There is little published information on the avoidance of fish to mercury. Atchison et al. (1987) list one older, unpublished study where fish were able to detect and were attracted to low concentrations (0.2 µg/l) of mercury. Behavioral avoidance of methylmercury chloride by threespine stickleback were reported at 0.5 µg/l (Evans 1973). Rehnberg and Schreck (1986) reported coho salmon avoided 20 µg/l and lost olfactory function to detect amino acids which are potent odors related to chemoreception, homing, and pairing. These potential effects levels are higher than the predicted concentrations in the mixing zones. Mercury is unlikely to be a concern for the zone of passage for fish through the mixing zone.

Nickel

There is relatively little published information on the avoidance of fish to nickel, despite its ubiquitousness as an urban or industrial pollutant of concern. Giattina et al. (1982) found an avoidance threshold for nickel in soft water of about 23.9 μg/l, regardless of whether the fish were exposed to an abrupt change in concentration or a gradual change. Nickel, copper, and zinc can have additive effects in lethality tests with fish (Rand 1995), but no behavioral effects of additivity have been verified.

Selenium

Cleveland et al. (1993) are cited as having reported behavioral abnormalities and locomotor impairment after 18-day and 60-day exposures of bluegills to selenium concentrations of ≥160 μg/l. This potential effects concentration is higher, and exposure duration much longer than would occur in the mixing zones. Watenpaugh and Beitinger (1985) reported a selenate avoidance threshold of 11,200 μg/l with fathead minnows. In other words, tested fish failed to avoid acutely lethal concentrations of selenium. Since the chronic criterion for selenium is set so much lower (5 μg/l), selenium is unlikely to cause avoidance responses or impede the zone of passage for fish through the mixing zone.

Silver

No references on behavioral effects of fish to silver were located in the databases and literature reviewed. This is despite a major recent research initiative into the fate and effects of silver in freshwater ecosystems, including the devotion of two entire issues of *Environmental Toxicology and Chemistry* in 1997 and 1999 solely to the subject. Because silver can have similar forms to copper and zinc, which elicit a strong avoidance response in fish, silver is assumed to have the potential to cause avoidance behavior above its acute criteria concentration of about 0.4 μg/l (silver does not have separate acute and chronic criteria). This assumption is admittedly overly general since like copper, zinc, and silver, cadmium is also a divalent metal and fish show little avoidance response to cadmium. Since data are insufficient to select an avoidance threshold for silver, compliance with criteria at the edge of the mixing zone is presumed to be protective.

Table 15. Copper concentrations associated with fish avoidance

Copper (µg/l)	Test Organism	Reference
0.1	Avoidance by rainbow trout under laboratory conditions[11]	Folmar 1976
0.7	Lowest observed avoidance effect concentration (LOEC) with chinook salmon *(Oncorhynchus tshawytscha)* under laboratory conditions[12]	Hansen et al. 1999b
1	Avoidance threshold with rainbow trout *(O. mykiss)* in presence of 12 µg/l zinc under laboratory conditions	Sprague et al. 1965
1	Avoidance threshold with lake whitefish when given no choice for light or shade	Sherer and McNicol 1998
1.6	Avoidance threshold with rainbow trout in presence of 14 µg/l zinc under laboratory conditions	Hansen et al. 1999a
2.4	Avoidance threshold with Atlantic salmon *(Salmo salar)* under laboratory conditions	Sprague et al. 1965
3	Avoidance threshold with chinook salmon under laboratory conditions	Hansen et al. 1999b
3	Avoidance threshold with rainbow trout under laboratory conditions	Hansen et al. 1999b
5	Avoidance threshold with goldfish *(Carassius auratus)* under laboratory conditions	Westlake et al. 1974
6.4	Avoidance threshold with rainbow trout under laboratory conditions	Giattina et al. 1982
6.4	Avoidance by coho salmon and loss of ability to detect potent odors related to chemoreception, homing, and pairing.	Rehnberg and Schreck 1986
6.4	Lowest field adjusted species mean avoidance threshold (SMAT)	Chadwick 2000b
6.5	Avoidance threshold with brown trout *(S. trutta*) in presence of 14 µg/l zinc under laboratory conditions	Woodward et al. 1995
7.4	Avoidance by cutthroat trout *(O. clarki)* under laboratory conditions	Woodward, et. al. 1997
10	Avoidance by non-acclimated juvenile steelhead trout (trout which were acclimated to 9 µg/l failed to avoid any copper concentration).	G.A. Chapman, written communication.

[11] Methods fail to specify how test concentrations were actually measured, thus results are assumed to indicate unmeasured nominal concentrations; further, a concentration of 0.1 µg/l was undetectable using methods routinely used at the time of the study. Study discounted for the purposes of estimating minimum avoidance thresholds.

[12] Chinook salmon showed statistically significant avoidance to 0.7 µg/l copper, but avoidance to the next highest concentration tested, 1.6 µg/l, was not significant. All concentrations between 3 µg/l and 22 µg/l were significantly avoided, therefore the avoidance threshold is 3 µg/l. Fish acclimated at 2 µg/l lost ability to avoid copper.

Copper (µg/l)	Test Organism	Reference
10 – 25	Possible interruption of upstream migration by spawning chinook salmon	Mebane 1994
20	Avoidance observed in the field with migrating adult Atlantic salmon in presence of 210 µg/l zinc (not a threshold)	Sprague et al 1965
≥25	Olfactory receptors damaged and homing ability reduced in chinook salmon and rainbow trout after 4-hours exposure.	Hansen et al. 1999c
44	90% reduction in home-water selection by returning adult salmon	Sutterlin and Gray 1973
≥44	Juvenile chinook salmon lose ability to detect and avoid (acutely lethal) concentrations	Hansen et al. 1999b
50	Avoidance threshold with lake whitefish *(Coregoneus clupeaformis)* under laboratory conditions	Macirowski et al. 1977
≥50	Olfactory receptors damaged and homing ability reduced in chinook salmon after 1-hours exposure.	Hansen et al. 1999c
≥50	Olfactory receptors damaged and homing ability reduced in rainbow trout after 1-hours exposure.	Hansen et al. 1999c
Bolded value (3 µg/l) is the lowest concentration reviewed which did not have significant confounding factors		

Zinc

Zinc has been well documented to cause trout and salmonid avoidance in laboratory and field conditions. Table 16 lists zinc effects concentrations associated with avoidance by salmonids. Based upon the internal consistency of responses reported within a study, consistency of responses reported between studies, similarity of test species to the species occurring in the study area, a preference for more recent study results over older studies when comparing similar studies, and whether actual test concentrations were reported to have been measured vice nominal concentrations, an avoidance threshold of 14 µg/l zinc was selected from the studies listed in Table 16.

<u>Comparisons of fish avoidance response to chemicals in lab and field conditions</u>

Few comparative lab and field studies of behavioral avoidance have been reported in the literature. Sprague et al (1965) reported adult spawning Atlantic salmon avoided zinc in field conditions at about 4X the threshold found in the lab with juvenile salmon. The upstream spawning migration of chinook salmon in Panther Creek, Idaho may have been interrupted when the fish encountered dissolved copper concentrations of about 10 to 25 µg/l, which are about 3X to 6X times higher than laboratory avoidance thresholds. In that case, the majority of spawning habitat and historical locations of chinook spawning were high in the watershed, upstream of copper discharges. However, chinook may have interrupted their upstream spawning migration in the vicinity of the first major diluting tributary, a point above which copper concentrations averaged about 10 to 25 µg/l during the late summer to early fall chinook spawning season. (Mebane 1994, RCG/Hagler

Bailly 1994). This concentration range is about 3X to 6X times higher than the avoidance threshold of 3 µg/l determined by Hansen et al. (1999b) with juvenile chinook in laboratory water with hardness, pH, and alkalinity that was specifically constituted to be similar to Panther Creek water.

Table 16. Zinc concentrations associated with fish avoidance

Zinc Concentration (µg/l)	Test Organism	Reference
5.6	Avoidance threshold with juvenile rainbow trout (in presence of 2 µg/l Cu in dilution water)	Sprague 1968
6.5	Avoidance threshold with coho salmon under laboratory conditions (concentrations not measured, nominal (unmeasured) concentration listed)	Rehnberg and Schreck 1986
10	Avoidance threshold with lake whitefish when given no choice for light or shade	Sherer and McNicol 1998
12	Avoidance threshold with juvenile rainbow trout (as a mixture with 1 µg/l copper)	Sprague et al 1965
14	Avoidance threshold with juvenile rainbow trout (as a mixture with 1.6 µg/l copper)	Hansen et al. 1999a
32	Avoidance threshold with juvenile brown trout (as a mixture with 6.5 µg/l copper)	Woodward 1995
41	Lowest field adjusted species mean avoidance threshold (SMAT)	Chadwick 2000b
47	94% avoidance by rainbow trout	Black and Birge 1980
52	Avoidance observed with juvenile cutthroat trout, in a mixture with 0.22 µg/l lead and 0.12 µg/l cadmium[13]	Woodward et al. 1997
53	Avoidance threshold with juvenile Atlantic salmon	Sprague et al 1965
210	Avoidance observed in the field with migrating adult Atlantic salmon (not a threshold)	Sprague et al 1965
284	Avoidance by male fathead minnows when zinc was the only variable	Korver and Sprague 1989
2200	Possible avoidance observed in the field with adult chinook salmon (not a threshold)	Goldstein et al 1999
Bolded value (14 µg/l) is the lowest concentration reviewed with the fewest confounding factors		

The most rigorous, controlled study reviewed comparing avoidance of metals in experimental natural streams and in laboratory countercurrent preference/avoidance chambers is that of

[13] 52 µg/l zinc was the measured test concentration, 28 µg/l (effects value listed in the paper's abstract) was the nominal concentration.

Hartwell et al (1987). They reported that schools of minnows were about twice as sensitive to a blend of Cu, Cr, Se, and As when tested in the lab rather than a natural stream. They speculated that in natural conditions the fish have many cues to respond to such as turbulence, turbidity, cover, presence of predators, and migratory instincts. In the lab, these other factors are eliminated. Due to the strong avoidance response of fish to copper, and the lack of avoidance response reported for Cr, Se, and As at the concentrations tested, the Hartwell results are likely similar to the results of copper alone.

Other studies compared the relative avoidance of metals in the laboratory with or without competing habitat behavioral cues such as shade. Korver and Sprague (1989) reported that breeding male fathead minnows avoided 284 μg/l zinc when zinc concentrations were the only variable in the tank. However, when the fathead minnow was allowed to establish a territory under a shelter within the contaminated side, 1830 μg/l were required to force the fish from the shelter. Thus, the avoidance threshold of minnows to zinc was raised by about 6X when the fish had a strong influence (shelter) to be in the zinc concentrations. Preference for shade, which is a form of shelter, can be stronger motivation than avoidance response to metals for some fish. Sherer and McNicol (1998) tested the avoidance of lake whitefish to metals in a countercurrent trough that was either uniformly illuminated, or shaded in one half. Fish preferred the shade when presented with a choice between shaded and illuminated. When metals were injected into the shaded, previously preferred area, avoidance of these ions was strongly suppressed, with the response to copper reduced 73X and the response to zinc reduced 100X.

The presence and interactions of other fishes is another major factor in fishes' behavior. Because of this, many preference/avoidance tests are conducted with single fish to avoid this additional variable. However, fish in field conditions always occur in the presence of other fish. Rainbow trout, tested singly, are very sensitive to and avoid low concentrations of many chemical pollutants. However, rainbow trout can be very territorial, and their territorial behavior instincts can overwhelm behavioral avoidance to odors from chemical contaminants. Gray (1990) reported that rainbow trout that were allowed to establish territories and were then exposed to complex waste mixtures defended their territories until death, driving off other trout. Other trout, having successfully "driven off" the dying, formerly dominant trout, then occupied the territory located within the acutely toxic plume until succumbing to the same fate. In some cases, dominant trout established territories in clean water portions of the tanks, relegating subordinate fish to contaminated portions.

A conclusion supported by comparisons of field and laboratory behavioral avoidance studies is that **no** field study has ever reported lower avoidance responses than laboratory studies using similar test organisms and chemicals. In biological testing where results are usually couched in cautious terms, such as "results suggest," "effects were associated with," or at best a probability of effects, this is significant. Although the scarcity of comparative studies prevents definite conclusions on the extent that the laboratory thresholds over-predict field responses, the range of relative responses reviewed is that the laboratory threshold values range from a factor 2X to 100X lower than field thresholds. ***Therefore, to select field avoidance thresholds to compare with potential chemical concentrations in the mixing zones, the field avoidance threshold is the lowest documented laboratory threshold multiplied by the lowest documented field to laboratory response ratio.***

Using zinc as an example, the field avoidance threshold equals 14 µg/l (lab threshold) X 2 (the lowest field to lab ratio) = 28 µg/l. If, for example, the upstream stream zinc concentration was 5 µg/l, if a significant portion of the stream width has concentrations less than 28 µg/l more than upstream (33 µg/l), then avoidance is unlikely. Because of the design of all avoidance/preference studies, the fish choose between relative differences between water #1 (low metals) or water #2 (higher metals). In contrast toxicity tests relate to absolute concentrations. Thus, avoidance thresholds are by relative concentrations rather than absolute concentrations.

Zone of Passage Thresholds Conclusions

To avoid risk of interrupting migration of fishes through the outfall discharge plume, with consequences of effective loss of upstream habitat or interrupting their life cycles, a significant portion of the width of the stream must be less than the following field avoidance threshold concentrations.

Table 17. Salmonid metals avoidance thresholds used for evaluating mixing zone metals concentrations (µg/l)

Selected Avoidance Thresholds	Cadmium	Copper	Chromium	Nickel	Lead	Mercury	Zinc
Lab	8	3	10	24	14	0.2	14
Field	***16***	***3***	***20***	***48***	***28***	***0.4***	***28***

Notes:

1. Lab avoidance thresholds from studies reviewed multiplied by 2, the lowest lab-to-field response ratio, to obtain field avoidance thresholds, except for copper

2. Because of ambiguity with the threshold avoidance response of juvenile chinook salmon to copper (Table 15, footnote), the recommended avoidance threshold is 3 µg/l, without multiplication by the field to lab response ratio.

Figures 10 and 11 illustrate the concept of fish avoidance of the discharge plumes until they disperse to below avoidance threshold concentrations. Figure 10 illustrates a plume from a center-channel instream diffuser such as would be employed in Squaw Creek and the Salmon River. Figure 11 illustrates a bank hugging plume that would be expected from the confluence of an open channel discharge, such as the Buckskin and Pat Hughes discharges into Thompson Creek (cover photo).

Figure 11a. Illustration of fish avoidance of the mixing zone of a discharge outfall, using a center channel diffuser.

To avoid risk to migratory fishes, the width of the plume which exceeds avoidance thresholds (stippled area), must leave sufficient space to provide a usable and desirable zone of passage for movement of fish. Criteria maximum concentrations (CMC) must be met at the edge of the zone of initial dilution (ZID), however concentrations expected to be acutely lethal may not occur within the ZID or anywhere. The criteria continuous concentrations (CCC) must be met at the edge of the regulatory mixing zone. The relative locations of the edge of the regulatory mixing zone and the avoidance plume may switch depending upon the chemical.

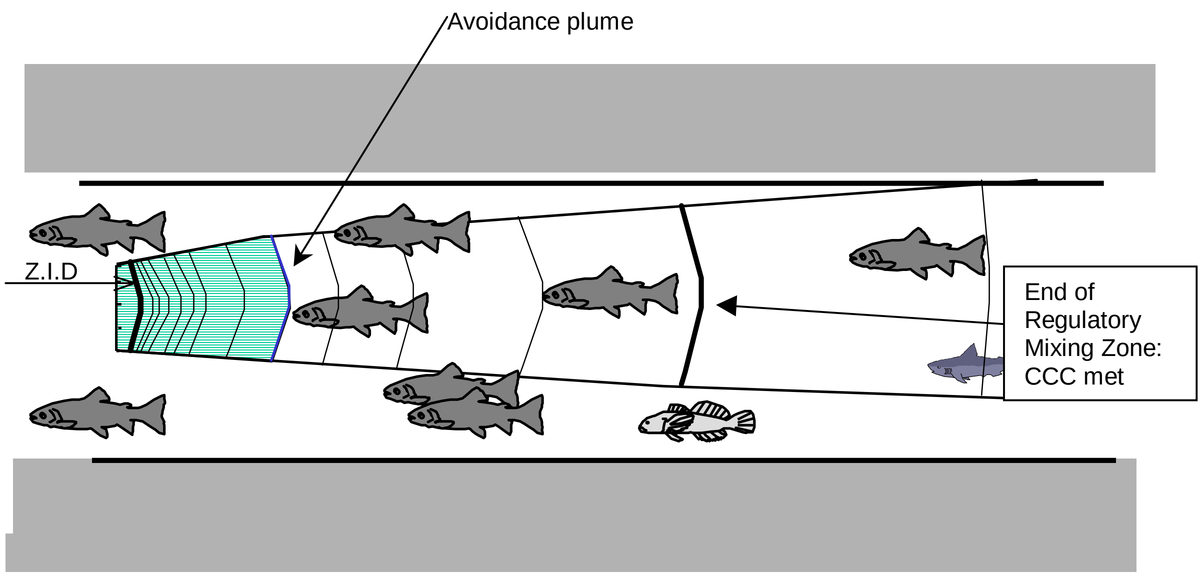
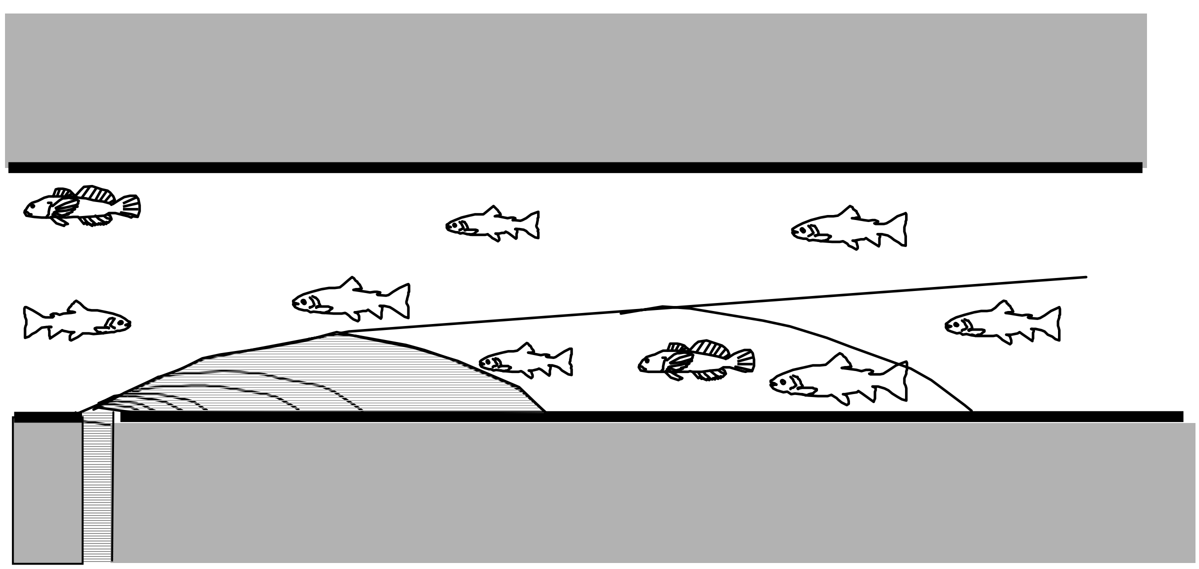


Figure 11b. Illustration of fish avoidance of the mixing zone of a discharge outfall, using an open channel confluence mixing zone configuration.



Other views on avoidance thresholds

There is presumably consensus that significant migratory disruption due to avoidance of mixing zones would be an unacceptable effect. However, short of conducting dosing and telemetry experiments in the study area streams, for which there would be some logistical hurdles to overcome, there is not consensus on how to define avoidance thresholds for mixing zones.

TCMC (2000b), commenting on the draft of this report, argued that unlike acute or chronic criteria, neither the EPA or State of Idaho have established a specific, scientifically defensible methodology for determination of fish avoidance threshold values for use in a regulatory, NPDES permit, setting. Our view is that since fish avoidance thresholds relate to their choosing between two waters as they mix, they are more relevant to mixing zone analyses than overall acute or chronic criteria. Additionally, the general requirement that waters be free from toxic substances in concentrations that impair beneficial uses, including behavioral abnormalities, involves comparing literature thresholds to expected site conditions. Somewhat, scientific defensibility may be in the eye of the beholder. However the approach used here included a careful review of the scientific literature, and that literature was interpreted in the context of site conditions (see following physical evaluation). Therefore, the approach used here was reasonable and can be defended based upon the available science.

Hillman (2000, appended) writing about a similar situation, takes a similar initial approach to my review, by selecting the lowest avoidance threshold of any salmonid to the constituents of concern. She presumably also used the notion that by considering the most sensitive responses of all salmonids, without regard to their actual occurrence at the site (e.g. Atlantic salmon), that would likely account for untested species (e.g. bull trout). By selective omission of literature differences between laboratory and field effects, she tacitly argues that field application adjustments of laboratory-determined avoidance thresholds should not be considered.

Chadwick (2000b, appended) argues that a more scientifically defensible way to calculate avoidance thresholds is to use the EPA concepts for deriving toxicological criteria. Using similar methods to those used to calculate water quality criteria, they calculated species mean avoidance thresholds (SMATs). Following the EPA criteria derivation approach, they calculated the SMATs through the geometric mean of published avoidance levels. They recommend this approach because it allows for use of all applicable, reliable data, instead of assuming that the lowest values are more significant than higher values. The SMAT approach excludes test results for species that do not occur in the study area, such as Atlantic salmon, lake whitefish, coho salmon, and minnows. Then, for those species that actually occur in the study area, they applied a lab-field adjustment to the laboratory determined SMATs. The lowest applicable field-adjusted SMAT values were 6.4 µg/l for copper and 41 µg/l for zinc, based upon chinook salmon and rainbow trout respectively.

In my view, the approach Chadwick (2000b) used is widely accepted for criteria derivation, but in this application, is limited by the relative scarcity of published avoidance values (in comparison to toxicity values). No avoidance data exist for bull trout, and none were located for any surrogate species in the *Salvelinus* genus, such as the Arctic char or brook trout. If *Salvelinus* avoidance data were located, or perhaps by taking an EC_{10} approach for laboratory avoidance values calculated for data from all the salmon, trout, and char species (those in the *Oncorhynchus*, *Salmo* and *Salvelinus* genera) then all the relevant data could be used, as Chadwick (2000b) advocates, instead of relying on the lowest values. In contrast, ignoring differences between laboratory and field conditions and relying solely on laboratory results, and their concomitant lower avoidance thresholds, is contrary to the body of work about fish behavioral responses to chemicals, and is clearly erroneous.

Necessary width for migratory stream fish passage

A quantitative determination of just how much stream width must be maintained below field avoidance thresholds is not possible without a major study program. Like Idaho, many states have a default fraction of the stream design flow to calculate assimilative capacity and provide for a zone of passage. Typically, under rapid and complete mixing conditions, the entire stream design flow is used as the basis for determining permit limits, that is, no spatial mixing zone is necessary. Under slow or incomplete mixing conditions where a mixing zone is necessary, fractions of a stream flow are used, unless a mixing zone analysis is performed to define site-specific mixing zones (EPA 1998).

The principles that 25% of stream width or volume should be the default fractions in Idaho's mixing zone policy, are derived from recommendations in the 1968 "Green Book." The "Green Book" committee recommended that there must be sufficient area, depth, and volume of flow to provide a usable and desirable passageway for the movement or drift of biota. Preferably, the passageway would contain 75% cross-sectional area and/or volume of flow of the estuary or river. Mixing zones should be as small as possible and mixing should occur as quickly as possible. The width of the zone, volume of flow, shape, and size of the mixing areas will vary with the character and size of the receiving water and should be established by proper administrative authority (FWPCA 1968).

A site-specific assumption in this mixing zone analysis is that small, high-gradient, turbulent, mountain streams have a different character from the larger rivers or estuaries that were evaluated with case studies (e.g. Jirka et al. 1996).

The closest scientific analogue to the question of how much stream width is needed for passage are studies on fish homing and detecting fractions of flows, especially to move around impassable water or obstructions. Sutterlin and Gray (1973) showed that hatchery salmon detected and preferred their home hatchery effluent, although it only made up 0.1% of the river flow (1/1000 volume). When 44 µg/l copper was introduced to the hatchery effluent, the fish no longer selected the small hatchery flow. Wild fish did not select the hatchery effluent at all, but instead selected their home upstream river flow. Damkaer and Dey (1989) evaluated the possible chemical disruption to salmon locating and ascending fish ladders at the John Day Dam, on the Columbia River. Salmon normally detect and ascend ladders with flows <1% of the Columbia River flow; however, during times when fluoride contamination in smelter outfalls was elevated, passage of the ladder may have been reduced. The NMFS has recommended at least 10% of stream flows be used as attractant flow in operating traps for adult salmon on spawning tributaries.

It is clear that salmonids can detect, select, and ascend small flows based on upstream odor. However, the goal in a mixing zone is to provide ample passage, not a minimal threshold of 0.1 – 10%. The 75% width recommendation from the Green Book committee is sufficient, but so may be other proportions, such as 40% or 90%. However, because of the rapid mixing that occurs in turbulent, high flows in Thompson Creek, the character of the stream and discharges, establishing a set, regulatory channel width which the outfall plumes cannot physically cross may not be hydrologically feasible. This is considered further in the next section.

Physical Evaluation

Physical features of the mixing zones that relate to Idaho water quality standards are those concerning aesthetics, recreational use, and habitat requirements for fish and other aquatic life. Aesthetics is a factor for designating special resource waters, and is a designated though undefined use for surface waters (WQS §56, 100). For the purposes of a mixing zone determination, so long as the proposed discharges do not discolor the receiving waters, diffusers and pipelines are as unobtrusive as possible, riparian and channel disturbance is avoided or minimized, pipes are submerged so that they do not create an obstacle to boaters during low flows, and the discharge jets do not break the surface of the water, then they will be considered not to interfere with aesthetic and recreational uses.

Physical evaluation of the mixing zones included site measurements of flows, channel geometry, sinuosity, and substrate roughness. These measures in turn were used in hydrodynamic modeling of outfall plume dispersal and dilution. The results of the plume dispersal and dilution modeling were subsequently compared to regulatory criteria, and to the results of the previous sections on fish avoidance and fish and wildlife exposure and risk of adverse effects.

Biological significance of physical features of the mixing zones

Because the discharges are unlikely to directly affect physical habitat features that are important to salmonids, salmonid habitat requirements are considered as in the context of potential constraints on fishes' movements through and around the mixing zone[14]. Areas in the stream channel downstream of the discharges would potentially be avoided by salmonids. Depending upon whether the effluents are discharged in the center of the stream channel or from the bank, these avoided areas would either be mid-channel, stream run-type habitat, or habitats along the stream banks (Figure 11). In relation to salmonid movements in streams, the major physical habitat differences between mixing zones centered in the stream channel, or along the bank, are depth, velocity, and cover.

Depth.—Fish obviously need a minimum water depth to swim. Were the effluent plume to force fish to extremely shallow stream margins, it would disrupt fish movement in the streams. Minimum depths that enable upstream migration of adult salmon and trout are about 12 cm for trout, 18 cm for large trout and steelhead, and 24 cm for spring and summer chinook salmon. For juvenile salmonids, the depth of water used depends on what is available, the amounts and type of cover present, and their perceived threat from predators and competitors. Young trout and salmon have been seen in water barely deep enough to cover them and in water more than a meter deep (Bjornn and Reiser 1991).

If fish have a preferred depth of water, Bjornn and Reiser (1991) believe it is readily subjugated to the needs for suitable velocities, access to food, and security from predators. In experimental laboratory streams, juvenile chinook salmon fry occupied all depths of the stream, including the

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[14] Principle physical habitat features are streamflow, cover, substrate, space, water depth, and velocity.

deepest pools (110 cm deep) when they were the only fish present. When yearling steelhead were also present, the chinook fry moved into shallow water (<6 cm deep) to avoid being eaten.

Velocity.—If stream velocities are unsuitable, no fish will be found. Natural streams contain a diversity of velocities and depths, some of which are suitable for most salmonids. The velocities required and used by salmonids vary with size of fish, and sometimes with species. Newly emerged fry (2 – 3.5 cm long) of salmon, trout, and char require velocities of less than 10 cm/s based on studies of sites selected by the fish in streams. Larger juvenile salmonids (4-18 cm long) usually occupy sites with velocities of up to 40 cm/s. In Idaho streams, young chinook and steelhead occupied deeper and faster water as they increased in size. By the end of summer, young chinook salmon were found in the full range of available stream depths, but in velocities that were on the low end of those available. Adult trout and steelhead and salmon can swim upstream against maximum velocities of up to about 120 to 240 cm/s respectively. Swimming speeds that salmonids can maintain for extended periods while migrating are usually called "cruising: speeds." For swimming through difficult areas for up to several minutes, fish have "sustained" speeds. Darting speeds for escape and feeding are about twice their sustained speeds (Bjornn and Reiser 1991). During winter, adult cutthroat and bull trout select pools with velocities of ≤2 cm/s, or if not available, leave the streams to find slow water habitats (Jakober et al. 1998). These requirements are summarized in the following table and compared to velocities expected in the mixing zones (Table 18, Figure 12). The comparison shows that juvenile salmonid movements would be limited to the stream margins, where the slowest water is located. Velocities would not limit movements by adult trout at all.

Table 18. (a) Approximate water velocities (cm/s) limiting salmonid movements (top) and (b) mixing zone velocities (from Figure 12).

Species	Adult migrating cruising speeds	Adult sustained speeds	Juvenile use
Chinook salmon	<104	104 – 329	9 – 25
Steelhead trout	<104	1.04 – 3.23	4 – 40
Bull trout	<61	61 – 195	9 - 12
Cutthroat trout	<61	61 – 195	10 – 22
Source: Bjornn and Reiser (1991)			

(b) Mixing zone velocities

Stream	Mid-stream velocities (center 25%)	Margins to mid-stream (remainder)
Thompson Creek 001	20 - 90	0 – 77
Thompson Creek 002	40 -120	0 – 70
Squaw Creek	35 – 150	0 – 140
Salmon River	80 – 140	0 – 80

Cover. —Cover is an important, but difficult to define, aspect of salmonid habitats in streams. Some of the features that may provide cover are water depth, water turbulence, large-particle substrates, overhanging or undercut banks, overhanging riparian vegetation, woody debris (brush, logs), shade, and aquatic vegetation. In experimental stream studies with juvenile salmonids, most studies have found that the fish were strongly attracted to shaded areas with instream or overhead cover (Bjornn and Reiser 1991). When availability of cover is introduced as a factor in chemical avoidance/preference testing, the preference of fish for cover greatly diminished their response to chemicals (as discussed earlier).

Mixing zone relevance. —The preference/avoidance of physical stream features interact and could either contribute to or counteract fishes' preference/avoidance response to chemicals in the effluent. The deepest water in riffles and runs of the channel would provide cover from overhead but not instream predators; this cover is found with the fastest water in the center of the channel. Foraging adult trout may use the fast water sections, but juveniles could not hold position there and would use shallower habitats near the banks. There, cover would need to be provided by woody debris or the substrates. Were an effluent plume to encompass the side of a stream section with cover, leaving a dangerous route around with no cover, but rather sunlit, shallow, and with a sandy substrate, movement of juvenile salmonids could be impeded.

Description of existing and proposed outfalls

Thompson Creek. In the vicinity of the outfalls, Thompson Creek runs through a narrow, steep mountain valley. Thompson Creek is a 2nd order[15] stream in the vicinity with a gradient of about 2.5 - 5% at the sites surveyed. Its geomorphic classification using the Rosgen (1996) system would be B2 with a Manning's *n* roughness coefficient of about 0.063, for a channel in a valley with moderate relief, with moderate entrenchment and gradient and a riffle-dominated channel with infrequently spaced pools, and with stable banks and profile. The valley floor is densely wooded, dominated by Douglas fir. The creek is mostly shaded by the trees; willows and other riparian understory shrubs are present, but provide sparse fish cover from overhanging brush, if any. No undercut banks were noted. Pools were mostly present as pocket water, that is small eddies behind boulders, and rubble surrounded by fast water. The confluences of both Buckskin and Pat Hughes Creeks are into riffle habitat, that is shallow sections with rapid current and a surface broken by rubble and boulders. Cover for fish would be provided by the cobble-rubble substrate, turbulence, and by the dappled shade from the forest canopy.

The "outfalls" from the mine to Thompson Creek are not constructed outfalls typically described in Idaho's mixing zone policy or in EPA permitting guidance documents. Instead they are the natural confluences of tributaries, Buckskin Creek "Outfall 001" and Pat Hughes Creek "Outfall 002." These discharges are not process discharges, rather they are the result of ambient precipitation and groundwater that is in contact with overburden (waste rock) from the open pit mining. The flow, mixing, and dilution of these tributaries with Thompson Creek are controlled by the natural channel factors. Where these "outfalls" cross the Forest Service road in Thompson

[15] First order streams are small headwater streams without tributaries. Second order streams form from the junction of two first order streams, and so on. Determined from USGS 1:100,000 hydrography.

Creek valley, they are unremarkable, small, clear flowing forest streams. Thus the physical features of the outfalls and mixing zone are not visible and would not detract from the aesthetics or recreation beneficial value of Thompson Creek.

Mixing of Pat Hughes Creek (Outfall 002) was observed with rhodamine red dye testing under low flow conditions in March 1988. By eye, the dye plume from Pat Hughes waters appeared fully mixed to observers about 35 meters below the confluence (Hopson 1988).

Squaw Creek. Near the confluence of Bruno Creek, the approximate location proposed for Outfall 004, Squaw Creek runs through a broad, open valley. Its geomorphic classification using the Rosgen (1996) system would probably be B4 with a Manning's *n* roughness coefficient of about 0.038, for a channel in an alluvial valley with gentle gradients, slightly entrenched, with riffle-dominated channel with infrequently spaced pools, and with stable banks and profile. The valley floor is vegetated by a mosaic of patches of aspen, grass and sagebrush, and Douglas fir. The streambanks are covered by a dense tangle of willows, with some cottonwoods present. Cover for fish would be provided by shade from the overhanging willows, the cobble-rubble substrate, and by turbulence. Squaw Creek is a 3rd order stream with a gradient of about 2 - 3% at the sites surveyed.

Thompson Creek Mine proposes to discharge by pipeline to an instream diffuser, with a year round, flow-tiered discharge. No location or proposed design for the instream diffuser was specified. For this evaluation, the location was assumed to be a short distance downstream of the Bruno Creek confluence, to coincide with IDEQ stream channel surveys that are conducted state-wide as part of the beneficial use reconnaissance program (BURP)(see Figure 1 and Table 3). Physical measures from the survey included flow, channel geometry, sinuosity, and substrate roughness, variables that are required as boundary conditions for hydrodynamic plume dispersal modeling. Since no diffuser design was provided, alternate configurations were considered based upon diffusers designed for the Salmon River (SRK 1993) and the Yankee Fork of the Salmon River (Kowaleski and Frechette 1999), and concepts described in EPA (1991a) and Jirka et al (1996). Single port, 3 port, and 5 port configurations were modeled; diffuser length, risers, and nozzles were limited in size to minimally protrude from the streambed under lowest flows, and to always be submerged. Coincidentally, the IDEQ 1994 sampling of Squaw Creek downstream of Bruno creek occurred at an extreme low flow time (3.96 cfs flow, which was equivalent to the 1Q10 critical low flow), so water depths and flows will rarely be below the evaluated scenario. Of these, a 1.2 meter diffuser, placed crosswise on the bottom of the stream, straddling the centerline, with 3 ports, each 6 cm in diameter provided rapid mixing, and limited the diffuser width to about 20% of the wetted channel at low stream flows. All further modeling used that configuration. While this diffuser design was adequate for completing the mixing zone analysis, this is by no means an optimized design that is suitable for construction. Before construction, an optimized diffuser design that will perform well under all anticipated flow conditions, and is sized and situated to avoid excessive depositional or scour areas in the stream, needs to be designed. The design and construction oversight will need to be conducted by qualified personnel with demonstrated success in designing instream diffusers that will perform adequately under severe environmental conditions and minimize instream construction disturbance in this sensitive environment. A Stream Channel Alteration Permit from the Idaho Department of Water Resources will probably be needed for instream diffuser construction.

Salmon River. Outfall 005 is proposed for the Salmon River just downstream of Thompson Creek. In this area, the Salmon runs through a narrow but gentle valley. The channel is broad and shallow, and is distinctly entrenched. The substrate is almost entirely boulder-cobble. The landuse is principally residential with livestock grazing. Cottonwoods are the primary riparian vegetation; willows are scarce, probably because the high banks extend beyond the rooting depth of riparian plants. Banks are rocky and stable. Cover for fish would be provided by water depth and the substrate. There is no shade or other overhanging or instream cover to speak of. Its geomorphic classification using the Rosgen (1996) system would probably be F3 with a Manning's *n* roughness coefficient of about 0.036, for a cobble- dominated, entrenched, meandering channel deeply incised in gentle terrain. At base flows in October 1999, the velocities in the Salmon River varied from 6 – 140 cm/s and depths from 0- 60cm.

SRK (1993) evaluated five options for a diffuser configuration for the proposed new Thompson Creek Mine outfall 005 to the Salmon River. The evaluation of the physical conditions at the proposed site included channel geometry, streambed roughness, and stream flows. Their recommended design involves burying and anchoring a 20 feet long diffuser with five ports rising up from the streambed about six inches. The ports would be submerged under all flow conditions. The diffuser would stretch across about 20% of the width of the river at low flows. Complete and rapid vertical mixing is predicted to occur in the water column very close to the diffuser (within a meter). SRK (1993) evaluated the performance of the design under very low flow conditions, a 10 year, 7-day flow condition (7Q10) of 323 cfs, mean annual flow conditions (1,240 cfs), and high runoff scenarios (5,040 cfs for a 2.33 year flood and 4,040 cfs for mean monthly flows for June).

Under the proposed design, the diffuser would remain sufficiently submerged to remain nearly invisible, and not obstruct shallow draft boats under minimum expected flow conditions.

Water velocity, channel width and salmonid requirements

By comparing the water velocities that limit salmonid movements with channel velocity and width profiles, salmonid usage and movements through different portions of the stream channels in the vicinity of the mixing zones can be estimated (Table 18, Figure 12). Only along the stream margins, where velocities are less than 40 cm/s, would juvenile salmonids be able to hold position in the stream or move upstream. Newly emerged fry could only hold position in the extreme margins, where the water is shallowest and slowest (<10 cm/s). The range of velocities measured in Thompson Creek, Squaw Creek, or the Salmon River would not limit movements of adult trout. Depth is unlikely to be a limiting factor for movements specifically around the mixing zone.

On Thompson Creek, the combined effect of the potential salmonid chemical avoidance behavior and velocity limitations could be to force salmonids to the river right side of the channel (Figures 11 and 12). The center channel would be unusable by juvenile salmonids because of the speed of the water; the left side could potentially be avoided because of chemical avoidance. The combination of potential avoidance of the confluences and fast water could theoretically have the effect of causing the fish to give a clear berth to the left half of the channel. If this potential situation is in fact occurring, it likely has been ongoing for about 15 years since the Buckskin and Pat Hughes waste rock dumps were built. Since salmonid distributions in Thompson Creek are unremarkable for streams of its size in the region, their movements appear unimpeded by this ongoing interaction of velocity, depth, cover, and potential chemical avoidance. It follows then, that for the Thompson Creek open-channel confluences (Figure 11), maintenance of around 50% of the channel width below potential chemical avoidance thresholds would be sufficient to allow the movement of salmonids up and downstream. This width would allow access to deep water in the center channel for adult trout movements, and access to the slow water habitat needed by juvenile fish. The default presumption that 75% of the channel be “reserved” for fish passage is physically unrealistic in a stream small enough to step across (as discussed in the hydrodynamic modeling section).

On Squaw Creek and the Salmon River, with the planned discharge through a diffuser buried in the center of the channel, the effects of the potential salmonid chemical avoidance behavior and velocity limitations would work together. Because of the speed of the water, juvenile salmonids could not use the center of the channel. Adding the potential chemical avoidance factor in the fast water portions of the channel would make little difference for juvenile salmonids (Figures 10 and 12). In these streams’ cases, the use of a diffuser would result in rapid dilution of the effluent plume to concentrations below avoidance thresholds. Movements of adult trout, and in the case of the Salmon River, adult salmon, would not be limited by water velocities encountered. They presumably would give a clear berth to the diffusers and their plumes and swim around them. These plumes could be expected to obstruct fish passage over approximately the center quarter of the channel width. The depths and velocities in the remaining channel width are suitable for passage by adult and juvenile salmon.

For the Squaw Creek and Salmon River discharges, which would be controlled through a center-channel diffuser in the deepest water, maintenance of 75% of the channel width below potential chemical avoidance thresholds would maintain access to slow water habitats in the stream margins for juvenile salmonids and deep, fast water habitat for adult salmonids. This would be sufficient allow the movement of salmonids upstream and downstream.

Flow volume and channel width relationships

Plume dispersal is affected by the channel form and flow distributions. Flow is a function of the stream’s cross sectional area and the water velocity. I attempted to use relationships between fractions of flow and fractions of stream width to define the volumes associated with the stream widths that provide zones of passage. Because of geometry and friction with the channel, velocities and flows are not evenly distributed in streams. Cross sectional area and higher velocities are disproportionally centered around the channel center – streams run fastest and deepest in the middle. In several channel surveys in Squaw and Thompson Creek, as well as other nearby streams with similar landform, valley, and channel shapes, the 25% of stream width centered on the stream thalweg accounted for about 50 – 58% of the stream volume (Figure 12). In Thompson Creek, where the discharges are open channel confluences, flow by width was considered from the stream edges. If the stream channel is mostly straight and symmetrical in the mixing zones, 50% of the width from bank to thalweg will account for about 50% of the flow. Measured values in the Thompson Creek mixing zones were around 30 – 79 % (Figure 12).

Figure 12.a. Thompson Creek main channel cross section depth, velocity, and flow profiles, below the confluence of Buckskin Creek (001), December 11, 1999, at 4.1 CFS. Cross section view is as looking downstream; river left is on the left side of the page. Overall channel wetted width (including the island) =3.5 m (11.5 ft). 50% of the channel width from left bank (including island) includes 29% of flow.

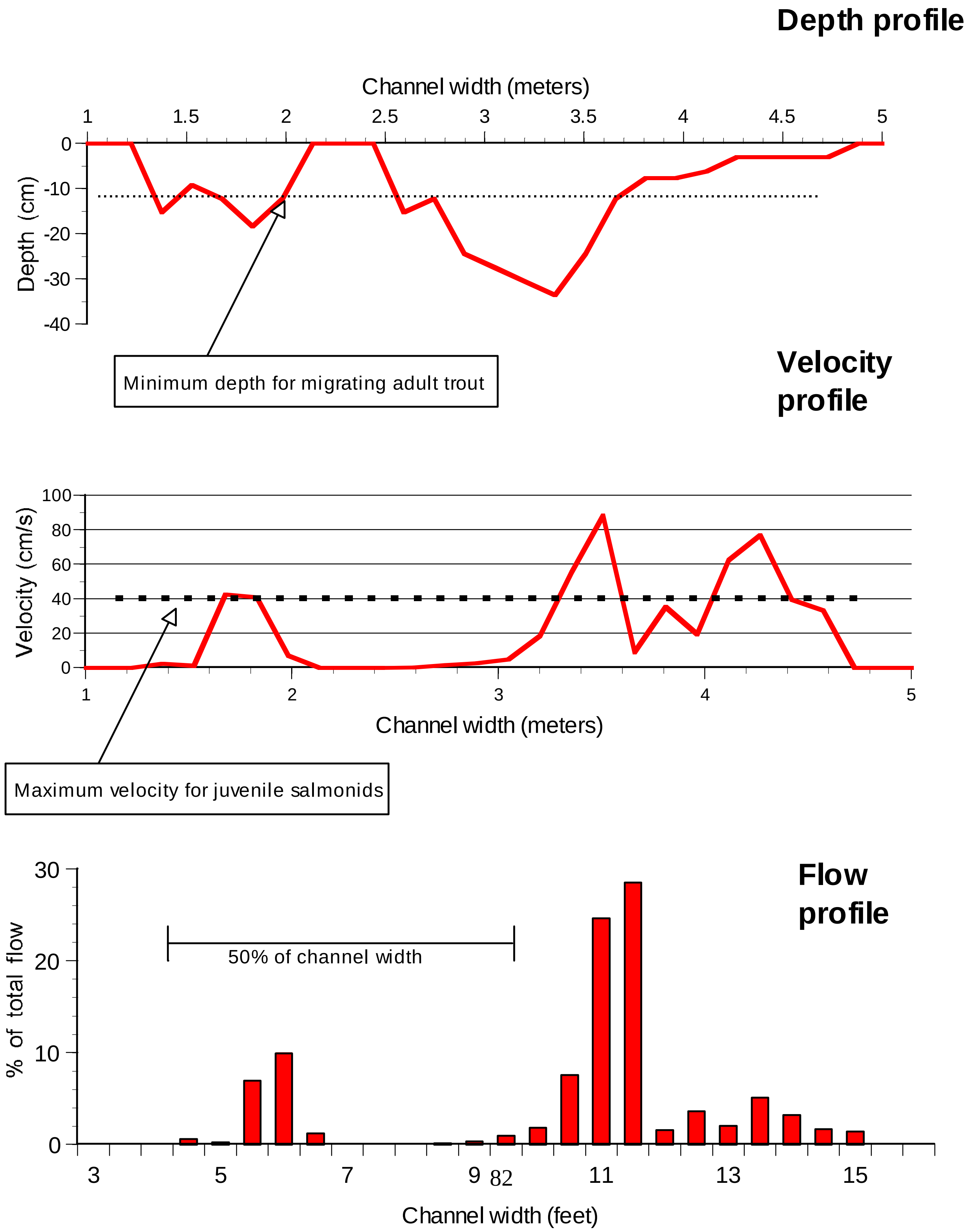


Figure 12.b. Thompson Creek cross section depth, velocity, and flow profile, below the confluence of Pat Hughes Creek, December 11, 1999. Main channel wetted width = 4.5 m (15 ft). 50% of width from the left bank encompassed 79% of the flow volume.

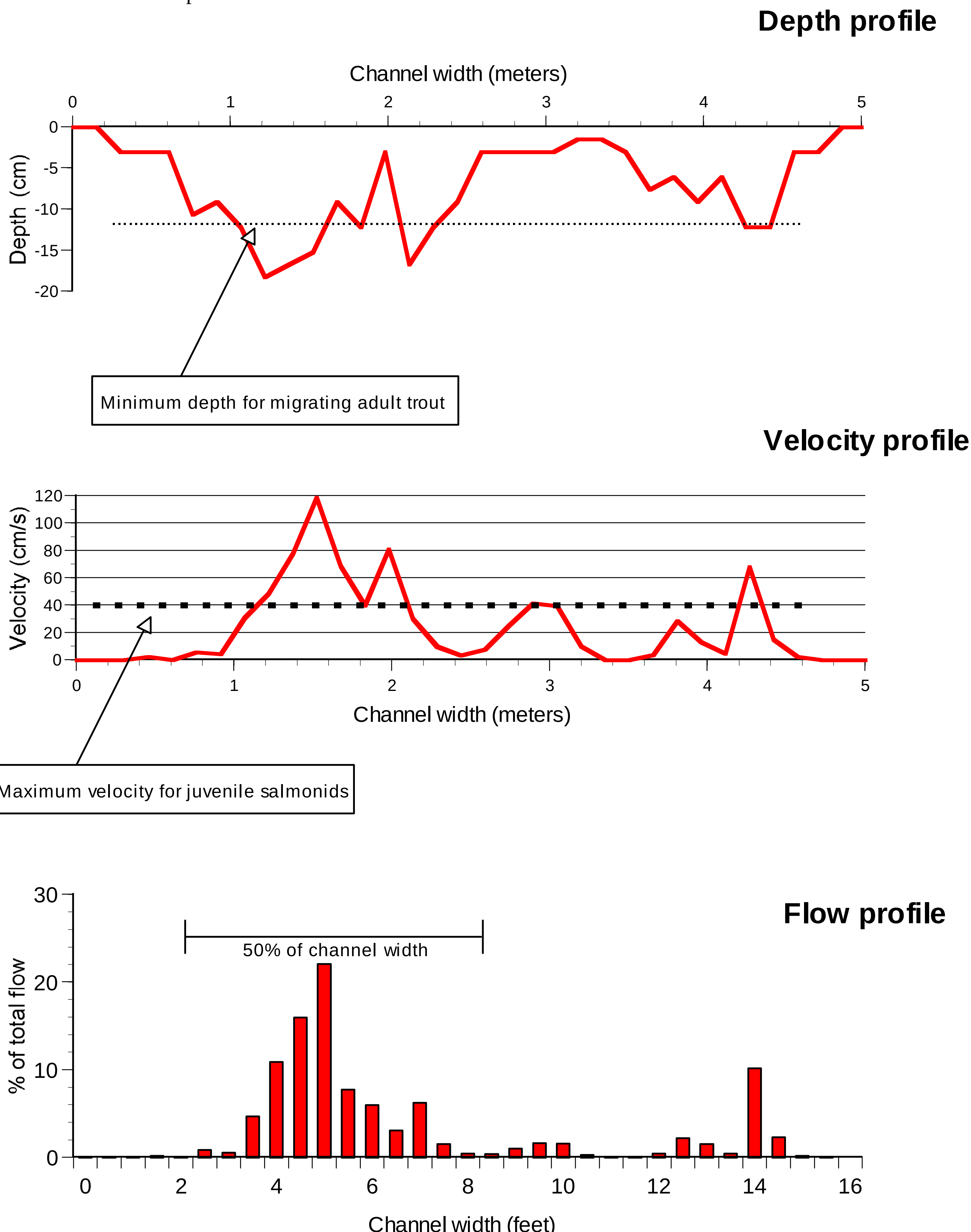


Figure 12.c. Squaw Creek channel cross sectioncross section depth, velocity, and flow profiles, downstream of Bruno Creek, at 4 CFS. Average bankfull channel width = 8 m, wetted width = 2.7m, 25% of wetted width = 0.7 m. The 25% of width centered upon the stream thalweg encompassed 58% of the flow volume. IDEQ Station 94EIRO042, July 21, 1994.

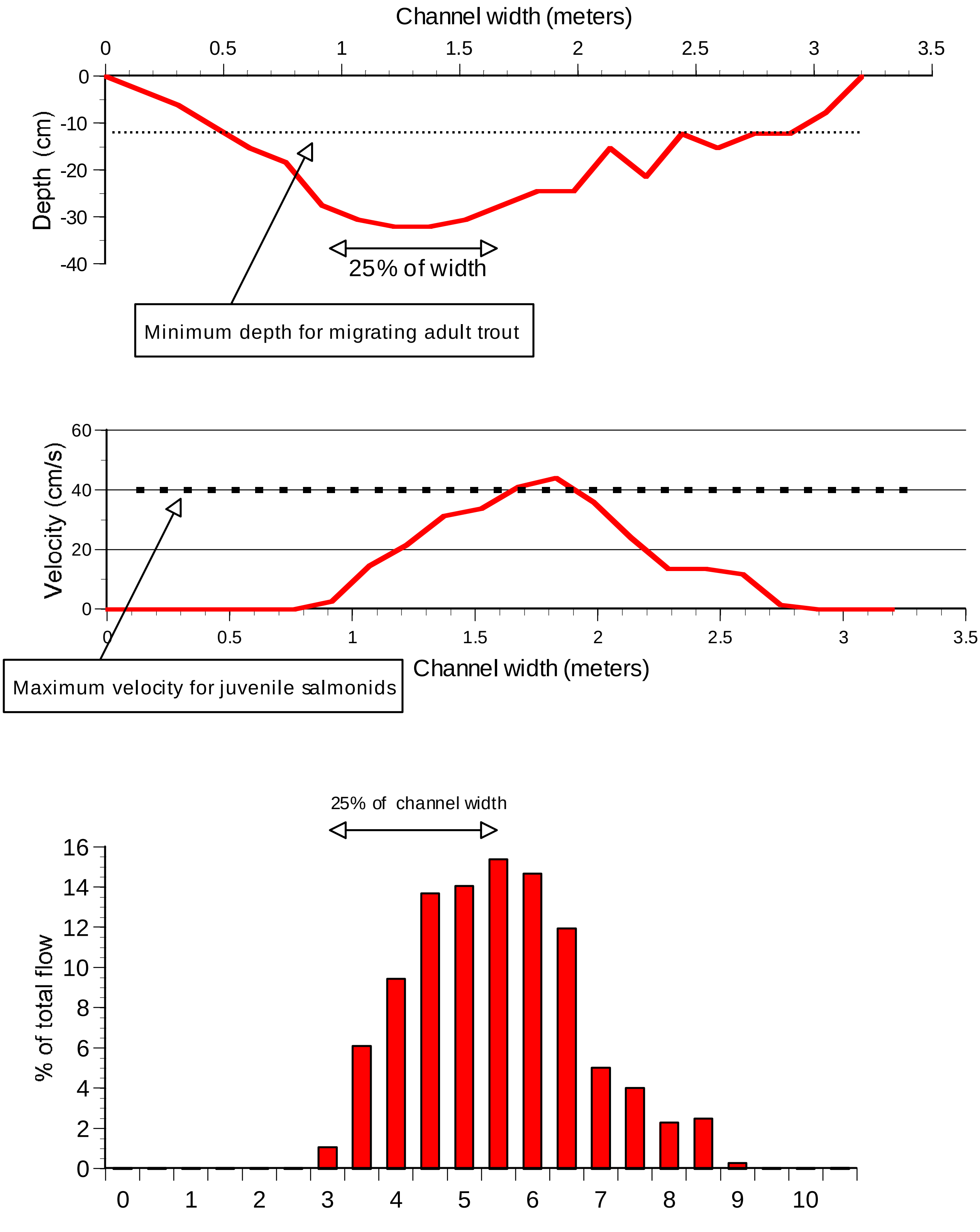


Figure 12.d. Squaw Creek channel cross section cross section depth, velocity, and flow profiles, upstream of Bruno Creek, at 28 CFS. Average bankfull width = 8m (26.2 ft). Wetted width = 4.9 (16 ft), 25% of wetted width = 1.2 m (4 ft). The 25% of width centered upon the stream thalweg encompassed 56% of the flow volume. IDEQ Station 95EIROA070, August 1, 1995,

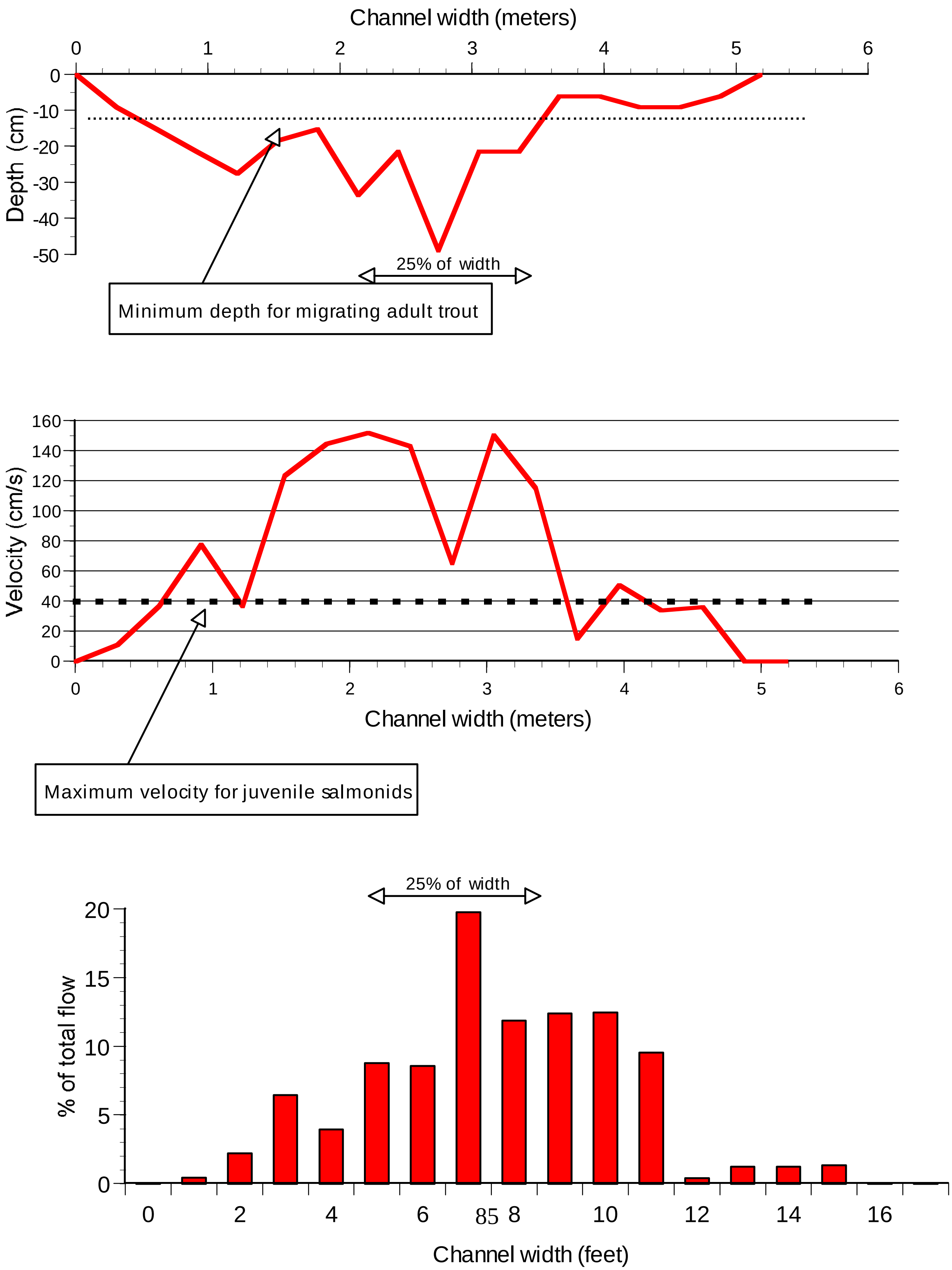


Figure 12.e. Squaw Creek channel cross section cross section depth, velocity, and flow profiles, below the confluence of Bruno Creek, December 11, 1999, at 14 CFS. Average wetted width = 6.5 m (21.5 ft), 25% of wetted width = 1.6m (5 ft). The 25% of width centered upon the stream thalweg encompassed 53% of the flow volume.

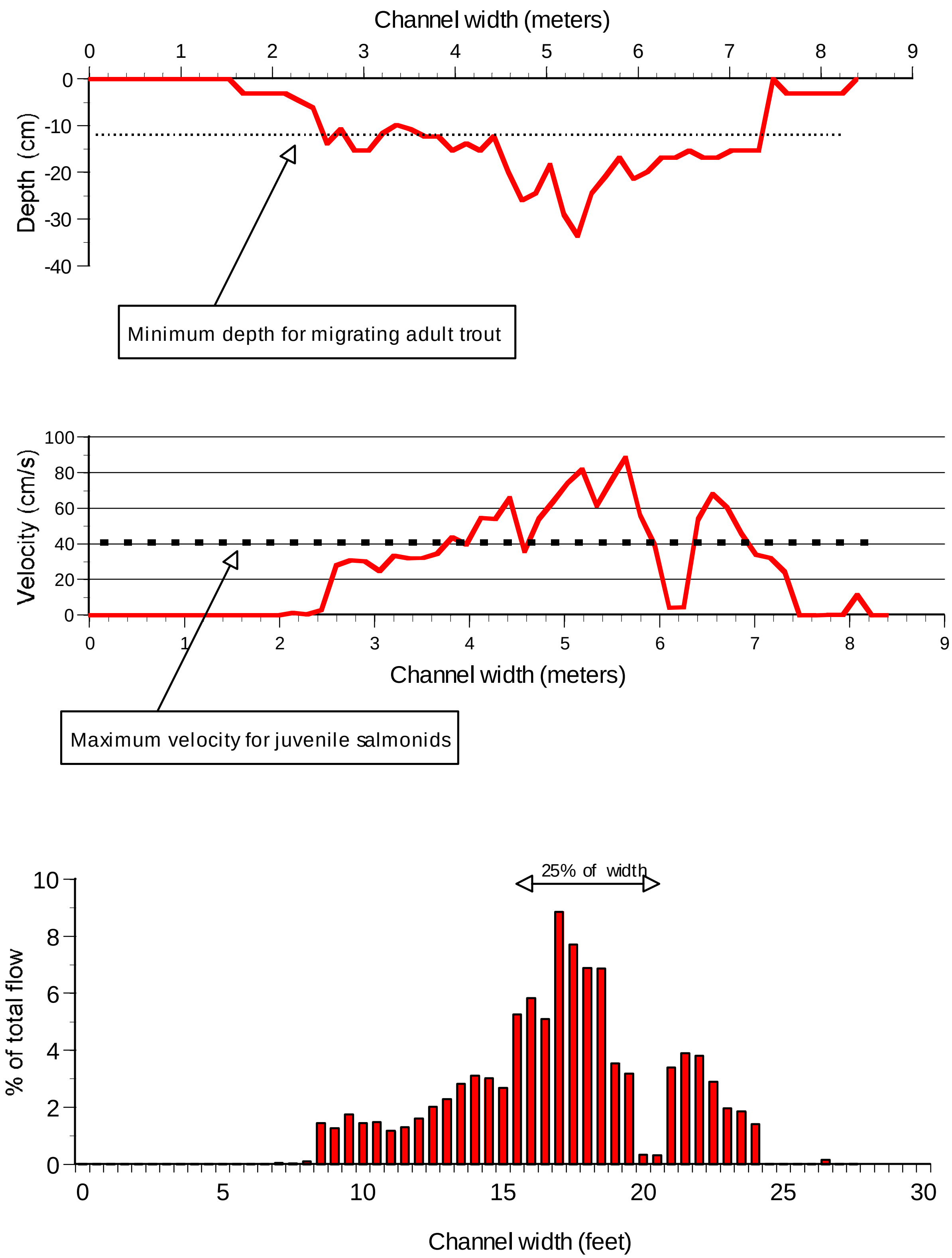
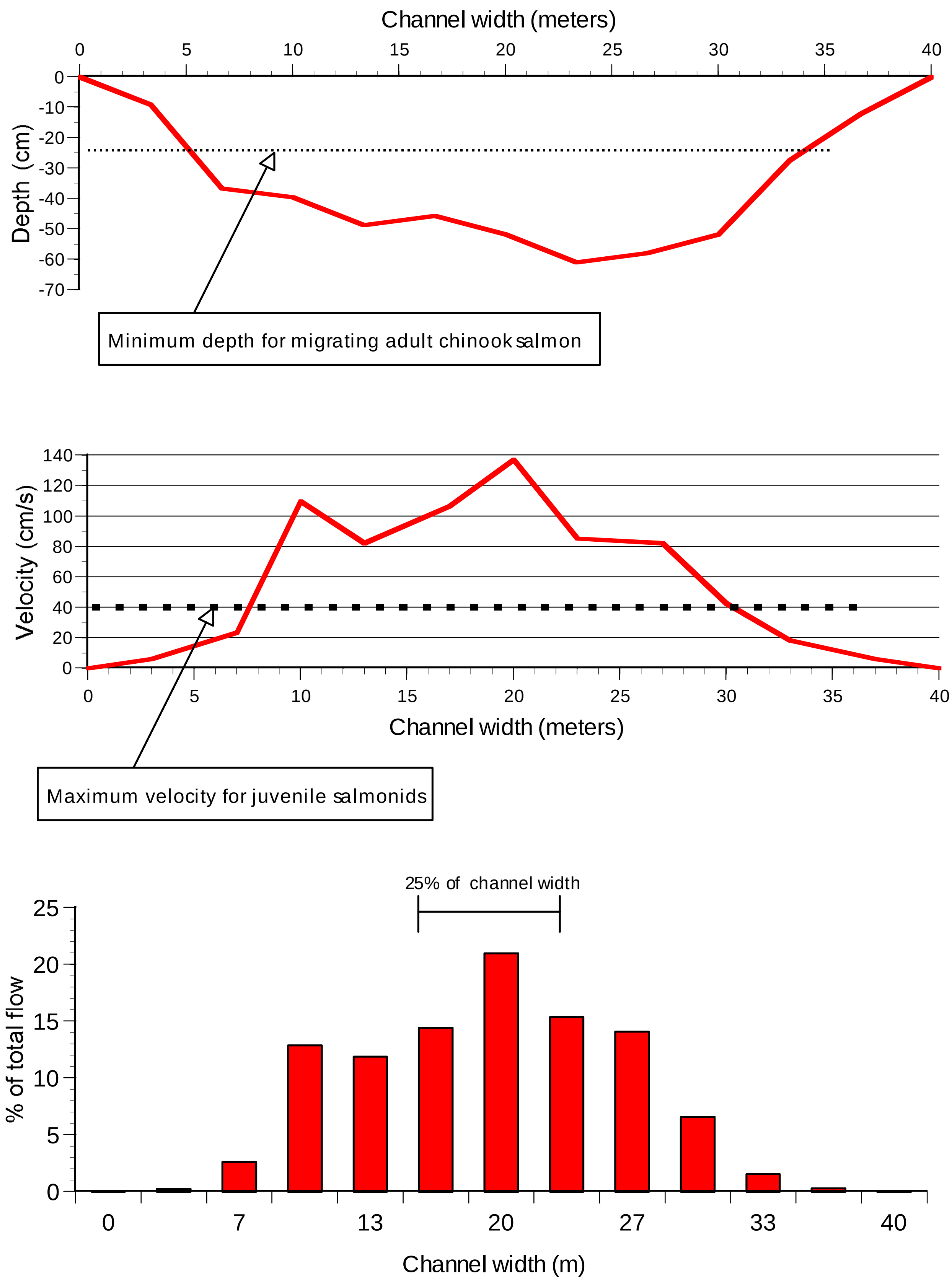


Figure 12.f. Salmon River channel cross section depth, velocity, and flow profile, upstream of Thompson Creek, IDEQ Station 1999RIDF001, October 5, 1999, at 410 CFS. Average channel wetted width = 40m (131 ft).



Flow regimes

Accurate understanding of discharge and receiving water flows is required in order to calculate an appropriate mixing zone and effluent limits that do not harm the aquatic life in the receiving waters. Flows for calculating permit limits were calculated differently for Thompson Creek than for the Salmon River and Squaw Creeks. Thompson Creek receives unregulated natural runoff and drainage from the Buckskin and Pat Hughes waste rock dumps. Squaw Creek and the Salmon River will receive controlled flows through a pipeline.

EPA recommends using dynamic modeling for developing design conditions that correspond with aquatic life criteria. If dynamic modeling is unavailable, then steady state modeling is an alternative. Steady state modeling uses simplifying assumptions is less complex, and less realistic than dynamic modeling. Steady state modeling requires calculating design flows in order to appropriately allocate wasteloads. EPA supports the use of either hydrologically based or biologically based design flows, although hydrologically based design flows have the disadvantage of being independent of biological conditions. The biologically based method calculates the design flow directly from national or site-specific duration and frequency, it always provides the maximum allowable number of excursions and never provides more excursions than allowed (EPA 1991a, Appendix D). For calculating dilution ratios of Outfalls 001 and 002 to Thompson Creek, the frequencies and durations of the chronic criteria were used, that is the lowest ratio expected to occur in a consecutive 4-day period once every 3 years. This value is nicknamed the "4B3" here.

Thompson Creek

TCMC has consistently measured flows in outfalls 001, 002, and Thompson Creek on at least a daily basis since 1983. This period of record includes extremely high flood flows (spring 1997) and extremely low drought flows (1992 and 1994). In other words, this is a comprehensive record. Flow measurements before 1983 were sporadic, and were not used in evaluations. Since the effluent flows are unregulated, dilution ratios of effluent to receiving water were used instead of fixed flows to define flow regimes for permit limits. TCMC reduced their actual flow measurements to a more manageable daily summary of 5,753 matched daily effluent to stream values from 1983-1998 (Figure 13). From these, daily dilution values were calculated by subtracting the daily effluent volumes from the Thompson Creek gage volumes to obtain the receiving water volume upstream of the discharges, and then calculating an upstream receiving water and effluent volume ratios[16]. The two steps were needed since the gage on Thompson Creek is located downstream of the effluents, and EPA (1991a) recommends use of upstream receiving water flows. The dilution ratio results were split into two groups: those occurring when Thompson Creek flows were < 7cfs and ≥ 7 cfs. These groups were examined to define critical flows, or more correctly critical dilution ratios. Critical flows are the flows used for modeling and discharge calculations. The discharges are calculated to comply with water quality standards at the critical flow values. Discharges below the critical flows that would likely not comply; above the critical flows they would always comply with standards.

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[16] $Q_{u1} = Q_d - (Q_{e1} + Q_{e2})$ and $Q_{u2} = Q_d - Q_{e2}$ where Q_u is upstream flow, Q_e effluent flows, and Q_d is downstream receiving water at the gage, for outfalls 1 and 2

The selected critical dilution ratio corresponds with the biologically-based chronic water quality criteria for pollutants: the lowest ratio expected to occur in a 4-day period once every 3 years. This was calculated by taking a 4-day rolling average of the daily average outfall flows divided by the estimated daily average flows upstream of each outfall. The lowest value was the 4B3 for that 3-year period. This was done for each of the 5 3-year periods of record, and the average of these lowest 3-year values were the dilution ratios used for the design flows in Thompson Creek. These values are between the 98th and 99th highest percentiles of all daily dilution values for the period of record (Table 19). In criteria derivation, infrequent, brief excursions of toxics criteria are not expected to have significant effects on aquatic populations. These recurrence intervals for design flows are consistent with that rationale.

Squaw Creek

Effluent discharges to Squaw Creek will be controlled by valve and pipeline and limited to 1.3 cfs (TCMC 2000). Flows at the Squaw Creek USGS gage have ranged from 3.3 – 771 cfs during the period that the mine has operated. For low flow conditions, plume dispersion in the mixing zone was calculated with 1.3 cfs of effluent discharged into the 1Q10 and 7Q10 flow conditions of 4.05 and 4.56 cfs respectively. At all Squaw Creek flows ≥50 cfs, the discharge of 1.3 cfs was evaluated as if only 50 cfs were available.

Salmon River:

The estimated streamflow in the Salmon River below Thompson Creek was estimated as approximately 323 cfs for the 7-day 10-year low flow condition (7Q10). SRK (1993) was provided a computer analysis of streamflow characteristics by U.S. Geological Survey for gage 13296500, which operated from 1922-1991 at the Salmon River below the Yankee Fork, near Sunbeam. The calculated 7Q10 for the gage was multiplied by the drainage basin ratio between the gage and the diffuser location to obtain the 7Q10 for the diffuser location (SRK 1993). The 1Q10 of 295 cfs at the diffuser was estimated from the gage:diffuser location proportion (263 cfs : 323 cfs). These flow calculations were checked for reasonableness by checking measured flows on major tributaries in this section with their calculations. IDEQ beneficial use reconnaissance program (BURP) crews have measured late-summer base flows of 49 cfs and 113 cfs respectively on Slate Creek and Warm Springs Creek, the two major tributaries that enter the Salmon River between the gage and diffuser locations. With ≈160 cfs from these two drainages during annual low flow period during a typical water year (August 1998), a 60 cfs difference at 7Q10 low flows seems reasonable.

TCMC (2000) calculated dilution ratios for prospective effluents into the Salmon River in a similar manner as for Thompson Creek outfalls. The measured flows from the effluent sources were calculated and plotted against Salmon River flows. Above 2000 cfs in the Salmon River, potential effluent flows seldom made up more than 6.6 cfs, or a dilution ration of 1 : 303 effluent to Salmon River flows. This ratio was enough different from the Salmon River low flow ratio of 1 : 120 to warrant calculating permit limits, assimilative capacities, and modeling the effluent dispersion and dilution separately from the low flow design condition.

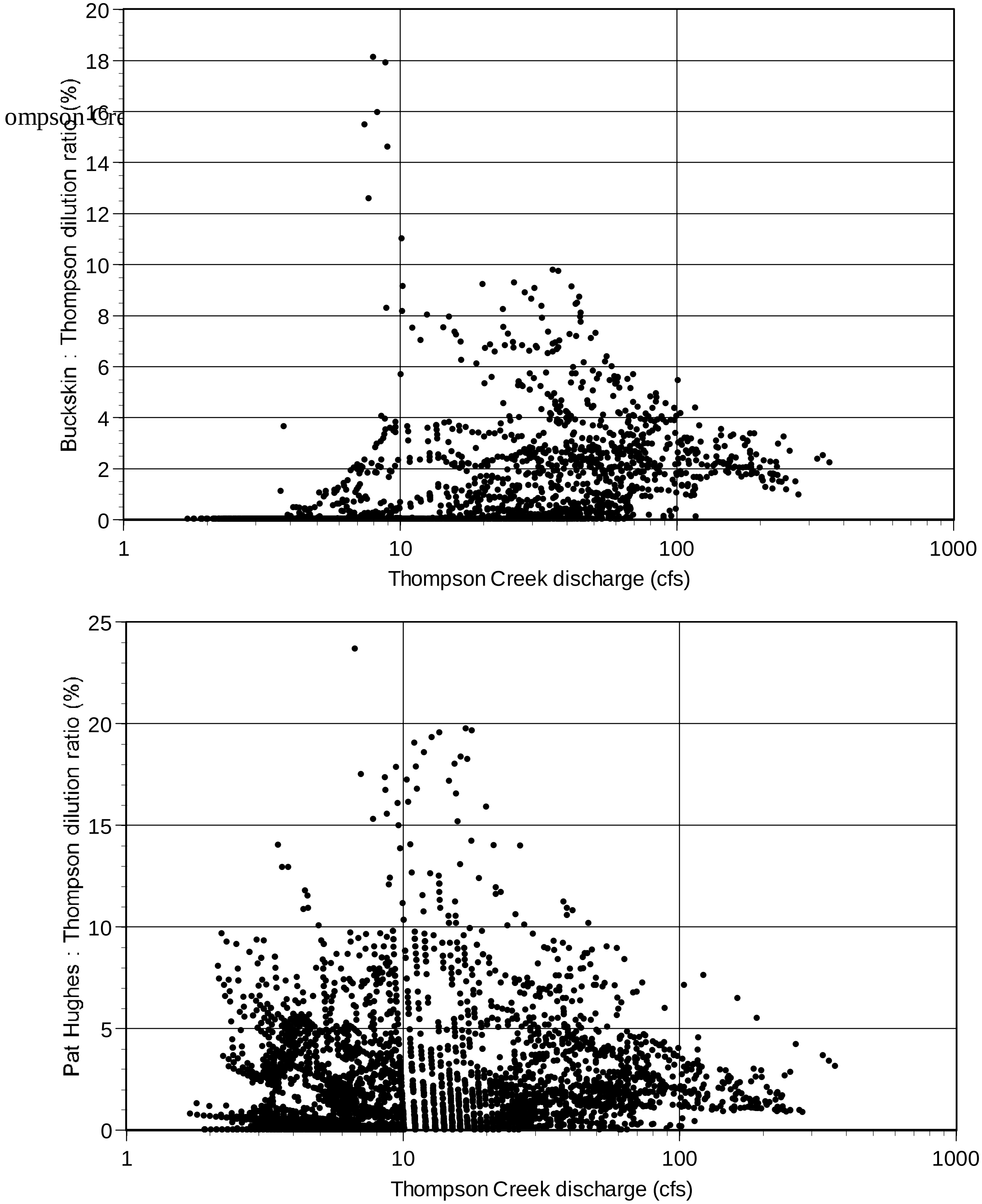


Figure 13. Relative daily flows of Buckskin Creek (Outfall 001) and Pat Hughes Creek (Outfalls 002) into Thompson Creek, 1983-1998. n = 5,753

Hydrodynamic modeling and field studies of the dispersion and dilution of the effluent plumes

The physical stream measurements and flow regimes were then used in hydrodynamic modeling of the dispersion and dilution of the effluent plumes. Extensive modeling of the discharge plume behaviors to determine likely downstream and across-channel behavior under differing ambient flow and discharge volume combinations was completed as part of the present and previous analyses (SRK 1993, Canonie 1994, 1994a, Benowitz 1997). The Cornell Mixing Zone Expert System (CORMIX) version 4.1 was used for modeling. CORMIX consists of a series of software subsystems for the analysis, prediction, and design of aqueous discharges into watercourses, with emphasis on the geometry and dilution characteristics of the initial mixing zone, including the evaluation of regulatory requirements (Jirka et al. 1996). CORMIX is probably the most widely used of several dilution models for effluent dispersion that EPA has supported. Hydrodynamic modeling by any known technique is not an exact science. However, extensive comparison with field and laboratory data has shown that the CORMIX predictions on dilutions and concentrations (with associated plume geometries) are reliable for the majority of cases and are accurate to within about ±50% the standard deviation of model predictions. In addition, CORMIX displays a high degree of flexibility in predicting a wide variety of flow possibilities, including various flow patterns and boundary interactions. (Jirka et al. 1996, Jones et al 1996, Baumgartner et al. 1994).

Regulatory requirements notwithstanding, in nature, mixing of discharges into waterbodies occurs in two major stages, discharge-induced mixing and ambient-induced mixing. The first stage of mixing is controlled by the momentum of the discharge jet relative to the momentum of the receiving water. The higher the velocity of the discharge jet, the more rapidly mixing occurs in this "near-field region" (Baumgartner et al. 1994; Jirka et al. 1996). This scenario describes the proposed discharges to Squaw and the Salmon well, which will discharge effluent jets through nozzles. With Thompson Creek, the effluent "jet" is discharged through the confluence of natural channels. Beyond the near-field region where momentum-induced mixing predominates over ambient mixing, mixing is controlled by ambient turbulence. In nature, mixing results from ambient turbulence and currents in the stream. These will be higher with cross-section irregularity from flowing over rocks and gravels in riffles and eddies, and from curves in the channel as the main current shifts sides of the channel. These complex factors are not supported by the CORMIX model, or any of the other EPA dispersion models (Jirka et al. 1996, Baumgartner et al. 1994; EPA 1991a). Instead the far-field model is based on passive diffusion, just as tea from a teabag diffuses without stirring. This characteristic results in an underestimation of dilution in the far-field mixing region. This underestimation is likely most significant on Thompson Creek, because of its higher gradient and more irregular channel than Squaw Creek or the Salmon River. The CORMIX model uses the roughness of the channel bed and channel irregularity as factors to predict far field mixing.

EnviroNet (2000) conducted a mixing zone field study to better understand mixing in Thompson Creek and compare with modeled predictions. The study was conducted at what were expected to be the annual worst case dilution ratio conditions in mid-April. At that time of the year, snowmelt and precipitation in the Pat Hughes and Buckskin drainages are usually in full runoff, whereas Thompson Creek runoff is expected to lag as much of its upper watershed have not begun significant melting. Results showed that the plume shapes and overall dilution were very similar to CORMIX results for the same condition. However, within the first 100m downstream, the field tests showed that mixing occurred at about 2X earlier than predicted by CORMIX (Figure 14). CORMIX has a Far-Field Locator post-processing tool to improve far field plume predictions, however it was not used in this comparison due to data limitations. Since the field study conditions represented severe high flow conditions (high effluent flows into low dilution flows), it follows that during most high flow conditions mixing would occur as soon or sooner than the tested conditions, the field mixing zone test results were used to predict mixing for discharges into Thompson Creek when it is running at >7cfs. CORMIX was used to evaluate all other discharge scenarios. Although far-field mixing may underestimated, and downstream distance to meet criteria through mixing are exaggerated, the CORMIX far-field results are useful to estimate the *relative* downstream distances to achieve water quality criteria for different pollutants from the same outfall.

Boundary conditions for the model, such as stream channel geometries at differing ambient and discharge flow conditions, were developed through surveys for the SRK and Canonie/Benowitz reports, IDEQ stream surveys, and recent verification measurements by TCMC personnel at the request of IDEQ. Bed resistance was estimated from Rosgen's (1996) compilation of Mannings *n* values for the corresponding Rosgen channel types. The model was run through many different iterations for each flow scenario and for each pollutant of concern, to calculate plume concentrations. These were then compared to instream criteria, and predicted zone of passage avoidance thresholds. Flow and water quality boundary conditions used in the modeling are summarized in Table 20. Receiving water hardnesses, which determine the applicable criteria, were calculated by EPA based upon the lowest 5th percentile of measured hardnesses in Thompson Creek. For Squaw Creek and the Salmon River, hardnesses were estimated by mass balance of the 5th percentile values of the waters which will make up the effluent, diluted into the receiving waters, at the 5th percentile of the receiving water hardnesses, using the relative flows given in Table 20 (EPA 2000b). EPA uses the extremes of the ranges of site conditions to ensure that the permit limits will protect beneficial uses over the expected ranges of conditions, not just average conditions. Thus, the criteria calculated in this manner, are minimum expected criteria. The calculations, while quite conservative and stringent for the discharger, are fully protective of the aquatic environment.

Instream pollutant concentrations after mixing for Outfalls 001, 004, and 005 were calculated by adding the measured average upstream pollutant concentrations to the CORMIX outputs (Table 20). Upstream pollutant concentrations were used twice, in this evaluation. First, EPA used the upper 95th percentile of the upstream pollutant concentrations in the calculation of "reasonable potential", and daily, and average monthly limits (EPA 2000b, Tables C-8 through C-12). Second, for modeling purposes, to make realistic predictions of instream pollutant concentrations, mean ambient concentrations were added to the increases that would result downstream from the discharges. This was necessary because the CORMIX model assumes upstream ambient pollutant concentrations are zero, which is not true for the naturally occurring elements of concern at this site. For the modeling, conservation of mass was assumed for all substances. Outfall 002 to Thompson Creek was treated differently because it is located downstream from Outfall 001 and its upstream pollutant concentrations include both natural background plus the pollutants from

Outfall 001 (Figure 1). For predicting the pollutant concentrations downstream of Outfall 002, its upstream concentrations were assumed to include the measured mean background concentrations upstream of 001 plus the resulting increase of 001's discharges at permit limits and at critical flow conditions. These values were used instead of measured concentrations upstream of 002 because (1) instream measured concentrations may not represent maximum conditions, and (2) the mixing zone field study showed that at monitoring station TC3 the 001 discharges were not fully mixed.

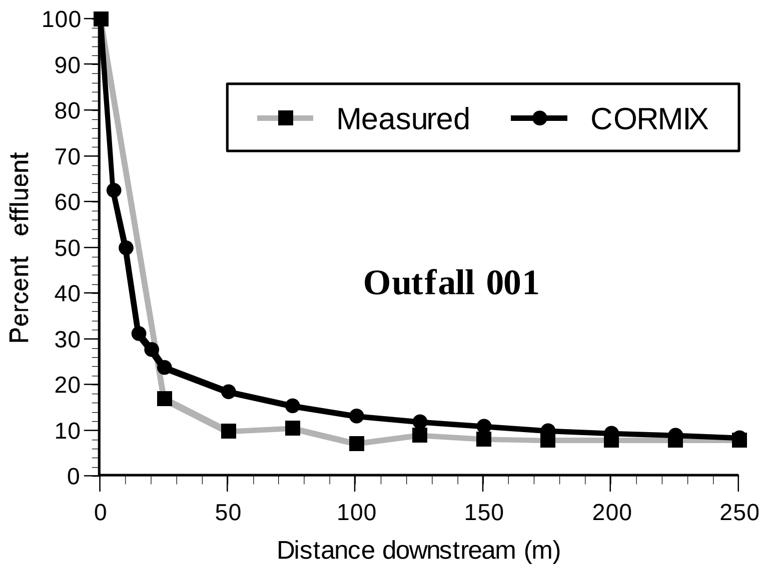
Changing the regulatory dilution volume percentage (% of actual ambient flow) has the effect of adjusting the permit limits and the resultant instream concentrations. Different mixing zone percentages were iteratively entered in the EPA permit limits derivation spreadsheets, and then the resulting permit limits, modeled with either CORMIX or, for Thompson Creek at <7cfs, with the empirical Thompson Creek mixing zone results. Iterations were repeated until mixing zone percentages provided permit limits that resulted in instream concentrations that met all of the following at design conditions:

> (1) at 50% of the width of Thompson Creek concentrations of copper and zinc were less than zone of passage avoidance thresholds (effluents discharged from bank).
>
> (2) at 25% of the width of Squaw Creek and the Salmon River, concentrations of copper and zinc were less than zone of passage avoidance thresholds (effluents discharged from mid-channel diffuser).
>
> (3) Resulting instream concentrations met chronic criteria within a reasonably short distance downstream of the outfalls.
>
> (4) For the Salmon River, increases in instream concentrations would not be more than 25% of the assimilative capacity, which is difference between ambient concentrations and the most stringent criteria.
>
> (5) Acute criteria would be met a short distance below the outfalls and the amount of time was low that organisms drifting through a mixing zone would be exposed to concentrations above acute criteria.

Additionally, for Thompson Creek with two sequential outfalls discharging the same pollutants of concern, an allocation of waste loads between the outfalls is needed. For substances with markedly different proportions of existing loads, mixing zones were adjusted according to existing discharges. An increase in allocation to one outfall results in a corresponding decrease for the other. Maximum effluent concentrations from 1998-2000 were used to judge the relative balance between the outfalls. The majority of available dilution for selenium was allocated to Outfall 001, and the majority of available dilution for cadmium, lead, and zinc was allocated to Outfall 002.

Results are summarized in Table 21. The column titled "Effluent Concentration" (3rd column from the left) in Table 21 is the prospective average monthly limit (AML) calculated using EPA Region 10 procedures with the mixing zone volumes for the 2nd column. Initially, maximum daily effluent concentrations were calculated and also modeled into the 1Q10 flows to account for short-term "acute" conditions. However, the plume modeling under these rules gave dispersion distances that were similar to applying the lower AML concentrations to the slightly higher 7Q10 flows to estimate limiting chronic criteria conditions. Henceforth, only AML pollutant concentrations were modeled into the 7Q10 flows.

The mixing zone field study (Environet 2000) and selected CORMIX outputs are appended.



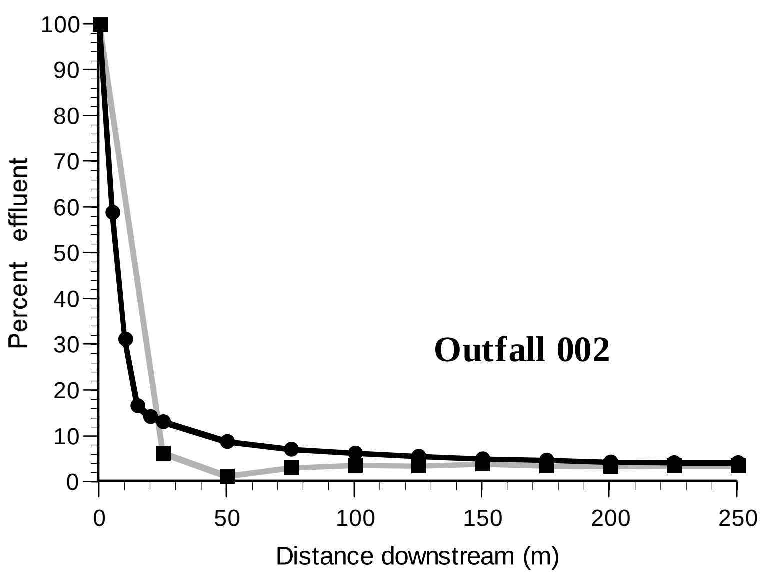


Figure 14. Modeled vs. measured mixing of effluents into receiving water. Both results showed an overall similar pattern and magnitude of mixing, except for the one anomalously dilute patch 50m downstream of 002. However measured dilutions showed that beyond initial near-field zones, mixing occurred more rapidly than predicted by CORMIX modeling. Measured results are for the stream centerline, modeled results for the plume centerline.

Table 19. Thompson Creek relative effluent and upstream receiving water flows and dilution ratios.

Relative flows (% of Thompson flows)	**n**	**Percent rank of 4B3 dilution out of all daily dilution ratios**	**4B3 expressed as a ratio of upstream receiving water to effluent** (1 : x)	**4B3** (average highest consecutive 4-day flow in a 3-year period) [1]	**Maximum**[1]	**95th percentile** [1]	**75th percentile** [1]	**Median** [1]
Tier: Thompson >7 cfs *								
Buckskin:Thompson	2694	98.0%	16.2	6.14	17.8	3.9	1.1	0
Pat Hughes:Thompson	2694	98.8%	8.0	12.47	23.6	8.6	3.5	1.6
Tier: Thompson <7 cfs								
Buckskin:Thompson	3138	99.2%	212	0.47	3.6	0	0	0
Pat Hughes:Thompson	3139	98.9%	12.55	7.97	14.0	5.2	2.1	0.3

[1] Percentage effluent into upstream Thompson Creek

Table 20. Summary of upstream receiving water and effluent flow boundary conditions used in hydrodynamic modeling

Receiving Water	Receiving Water Upstream Flow	Effluent Flow	Dilution Ratio	Receiving Water Hardness	Mean upstream ambient concentrations (dissolved µg/l)							
	cfs	cfs		mg/L	Cd	CrVI	Cu	Pb	Hg	Se	Ag	Zn
Thompson Creek (001) @ 7Q10	2.05	0.0096	1 : 212	85	0.04		0.24	0.03	0	1.2		3.0
Thompson Creek (001) @ >7cfs	7	0.43	1 : 16.2	55	0.04		0.24	0.03	0	1.2		3.0
Thompson Creek (002) @ 7Q10	2.05	0.16	1 : 12.5	93	0.07		0.56	0.06	0	2		2.8
Thompson Creek (002) @ >7cfs	7	0.87	1 : 8.0	72	0.07		0.56	0.06	0	2.7		2.8
Squaw Creek @ 7Q10	4.56	1.3	1 : 3.5	290	0.15	0	0.6	0.48	0	0		9.5
Squaw Creek @ >50cfs	50	1.3	1 : 38	67	0.15		0.6	0.48	0			9.5
Salmon River @ 7Q10	323	2.7	1 : 120	27	0.05		0.61	0.2	0		0	3.1
Salmon River @ >2000 cfs	2000	6.6	1 : 303	27	0.05		0.61	0.2	0		0	3.1

Table 21. Mixing of effluents with receiving waters: effluent dispersion and dilution modeling results. All concentrations are in μg/l and distances are in meters.

Thompson Creek 001: Flow tier <7 cfs: **212 : 1 (0.47%)** ambient Water to Effluent Dilution Ratios and ambient flows. Hydrodynamic modeling results of 0.01 cfs effluent into 2.05 cfs ambient flow (7Q10)

Parameter	Regulatory dilution volume (% of actual stream flow)	Effluent concentration (i.e. average monthly limits)	Downstream distance to meet acute criteria (CMC) (m)	Travel time for organisms drifting through acute mixing zone (min)	Downstream distance to meet CCC (m)	Potential resulting instream concentration (ambient + increase)	CCC- Applicable chronic criteria	Maximum Δ plume concentration at 50% of width	Zone of passage field avoidance thresholds
Cadmium	25	50	<1	<1	2	0.28	0.91		16
Copper	25	150	<1	<1	<1	0.95	9.9	1.4	3
Lead	12	70 (57)[7]	<1	<1	<1	0.30	2.1		28
Mercury	25	0.7	0	0	2	0.003	0.012		0.4
Selenium	25	236	1	1	7	2.3	5		NA
Zinc	25	750	1	<1	1	6.5	91	7	28

Thompson Creek 001: Flow tier >7 cfs: **16.2 : 1 (6.1%)** ambient Water to Effluent Dilution Ratios and ambient flows. Calculated results of 0.43 cfs effluent into 7 cfs ambient flow using mixing zone field study results (EnviroNet 2000).

Parameter	Regulatory dilution volume (% of actual stream flow)	Effluent concentration (i.e. average monthly limit)	Downstream distance to meet acute criteria (CMC) (m)	Travel time for organisms drifting through acute mixing zone (min)	Downstream distance to meet CCC (m)	Potential resulting instream concentration (ambient + increase)	CCC- Applicable chronic criteria	Maximum Δ plume concentration at 50% of width	Zone of passage field avoidance thresholds
Cadmium	5	1.4	0	0	<25	0.12	0.66		16
Copper	12.5	20	<25 (11)[7]	<1	<25	1.5	6.8	3	3
Lead	15	5 (4)[7]	0	<1	25-50	0.28	1.3		28
Mercury	25	0.07	0	0	<25	0.004	0.012		0.4
Selenium	47.5	40	<25 (10)[7]	<1	25-50	3.7	5		NA
Zinc	15	158	<25 (11)[7]	<1	<25	13	63	27	28

Table 21. Mixing of effluents with receiving waters: effluent dispersion and dilution modeling results. All concentrations are in µg/l and distances are in meters.

Thompson Creek 002: Flow tier <7 cfs: **12.5 : 1 (8.0%) a**mbient water to effluent dilution ratios and ambient flows. Hydrodynamic modeling results of 0.16 cfs into 2.05 cfs stream flow (7Q10)

Parameter	Regulatory dilution volume (% of actual ambient flow)	Effluent concentration (i.e. average monthly limit)	Downstream distance to meet acute criteria (CMC) (m)	Travel time for organisms drifting through acute mixing zone (min)	Downstream distance to meet CCC (m)	Potential resulting instream concentration (ambient + increase)	CCC- Applicable chronic criteria	Maximum Δ plume concentration at 50% of width	Zone of passage field avoidance thresholds
Cadmium	50	8.5	6	1	42	0.75	0.98		16
Copper	10	22	<1	<1	2	2.3	11	3	3
Lead	50	23 (18) [7]	0	<1	34	1.8	2.3		28
Mercury	25	0.06	0	0	7	0.008	0.012		0.4
Selenium	50	25	<1	<1	21	4.3	5		NA
Zinc	22	194	4	<1	4	18	98	28	28

Thompson Creek 002: Flow tier >7 cfs: **8 : 1 (12.5%) a**mbient to effluent flow dilution ratios. Modeling results of 0.87 cfs into7 cfs ambient flow ambient flow using mixing zone field study results (EnviroNet 2000)

Parameter	Regulatory dilution volume (% of actual stream flow)	Effluent concentration (i.e. average monthly limit)	Downstream distance to meet CMC	Travel time for organisms drifting through acute mixing zone (min)	Downstream distance to meet CCC (m)	Potential resulting instream concentration (ambient + increase)	CCC- Applicable chronic criteria	Maximum Δ plume concentration at 50% of width	Zone of passage field avoidance thresholds
Cadmium	62	5.5	<25 (9) [7]	<1	<25	0.81	0.81		16
Copper	25	26	<25 (9) [7]	<1	<25	3.8	8.6	1.6	3
Lead	60	12.5 (11) [7]	0	<1	<25	1.7	1.8		28
Mercury	40	0.07	0	0	<25	0.012	0.012		0.4
Selenium	25	11	0	<1	<25	5.0	5		NA
Zinc	75	300	<25 (11) [7]	<1	<25	50	79	19	28

Table 21. Mixing of effluents with receiving waters: effluent dispersion and dilution modeling results. All concentrations are in µg/l and distances are in meters.

Squaw Creek 004: Flow tier <50 cfs: **3.5 : 1 (28%) a**mbient to effluent flow dilution ratios. Modeling results of 1.3 cfs into 4.56 cfs ambient flow

Parameter	Regulatory dilution volume (% of actual stream flow)	Effluent concentration (i.e. average monthly limit)	Downstream distance to meet acute criteria (CMC) (m)	Travel time for organisms drifting through acute mixing zone (min)	Downstream distance to meet CCC (m)	Potential resulting instream concentration (ambient + increase)	CCC-Applicable chronic criteria	Maximum Δ plume concentration at 25% of width	Zone of passage field avoidance thresholds
Cadmium	50	5.7	<1	<1	<1	2.2	2.3		16
Chromium VI	50	20	<1	<1	<1	7.1	11		20
Copper	0	24	0	<1	0	9.2	28	2.5	3
Lead	50	26	<1	<1	<1	6.9	7.9		28
Mercury	100	0.04	0	0	<1	0.011	0.012		0.4
Selenium	50	11	<1	<1	<1	3.9	5		NA
Zinc	25	254	0	<1	0	61	257	27	28

Squaw Creek 004: Flow tier ≥ 50 cfs: **38:1 (3%)** ambient to effluent flow dilution ratios. Modeling results of 1.3 cfs into 50 cfs ambient flow

Parameter	Regulatory dilution volume (% of actual stream flow)	Effluent concentration (i.e. average monthly limit)	Downstream distance to meet acute criteria (CMC) (m)	Travel time for organisms drifting through acute mixing zone (min)	Downstream distance to meet CCC (m)	Potential resulting instream concentration (ambient + increase)	CCC-Applicable chronic criteria	Maximum Δ plume concentration at 25% of width	Zone of passage field avoidance thresholds
Cadmium	50	12	<1	<1	<1	0.47	0.77		16
Copper	15	37	<1	<1	<1	2.1	8	3	3
Lead	50	19	<1	<1	<1	0.74	1.6		28
Mercury	25	0.1	0	0	<1	0.003	0.012		0.4
Silver	25	11	<1	<1	<1	0.29	1.7 [CMC]		NA
Zinc	20	350	<1	<1	<1	21	74	26	28

Table 21. Mixing of effluents with receiving waters: effluent dispersion and dilution modeling results. All concentrations are in µg/l and distances are in meters.

Salmon River at 7Q10: 120:1 (0.8%) ambient to effluent flow dilution ratios. Modeling results of 2.7 cfs into 323 cfs ambient flow with a regulatory mixing dilution volume of 25% of actual stream flow.

Parameter	Effluent conc. (i.e. average monthly limit)	Downstream distance to meet acute criteria (CMC) (m)	Downstream distance to meet CCC (m)	Increased concentration after mixing	25% of assimilative capacity [4]	Potential resulting instream concentration (ambient + increase)	CCC- Applicable chronic criteria (hardness adjusted)	Maximum pollutant concentration at 25% of width	Zone of passage field avoidance Thresholds
Cadmium	6.2	<1	1.7	0.05	0.085	0.10	0.39		16
Copper	59	1	1.7	0.49	0.77	1.1	3.7	1.9	3
Lead	10	<1	2	0.08	0.098	0.28	0.59		28
Mercury	0.2	0	2	0.002	0.003	0.002	0.012		0.4
Silver	6	1.6	<1	0.05	0.12	0.05	0.36		NA
Zinc	500	1	1.4	4.2	7.7	7.4	34	17	28

Salmon River at 2000 cfs: 303:1 (0.33%) ambient to effluent flow dilution ratios. Modeling results of 6.6 cfs into 2000 cfs ambient flow with a regulatory mixing zone dilution volume of 25% of actual stream flow, except for mercury which is 6.25%.

Parameter	Effluent conc. (i.e. average monthly limit)	Downstream distance to meet acute criteria (CMC) (m)	Downstream distance to meet CCC (m)	Increased concentration after mixing	25% of assimilative capacity [4]	Potential resulting instream concentration (ambient + increase)	CCC- Applicable chronic criteria (hardness adjusted)	Increased pollutant concentration at 25% of channel width	Zone of passage field avoidance Thresholds
Cadmium	15	<1	2	0.050	0.098	0.010	0.39		16
Copper	150	2	8	0.50	0.90	1.1	3.7	2.2	3
Lead	25	<1	13	0.11	0.125	0.28	0.59		28
Mercury	0.2	0	2	0.0007	0.003	0.0007	0.012		0.4
Silver	16	20		0.073	0.12	0.057	0.36		NA
Zinc	500	<1	<1	1.6	9.2	4.8	34	7	28

Table 21. Mixing of effluents with receiving waters: effluent dispersion and dilution modeling results. All concentrations are in µg/l and distances are in meters.

Table 21 Notes:

[1] The 2nd column from the left "Effluent concentration in the effluent" corresponds with the monthly average effluent limit that would result from the given mixing zone.

[2] Mercury limits for this site are required through a regulatory quirk, not by environmental conditions, i.e. whether mercury is present in effluents at concentrations likely to exceed water quality criteria. To determine the "reasonable potential" of selected pollutants to exceed water quality standards, it is EPA's practice to use either measured maximum effluent concentrations or technology based effluent limits in their calculations, whichever is higher (EPA 2000b, p. C17, footnote 3). "Reasonable potential" to exceed mercury criteria is present only for one outfall (004@7Q10) when calculated with the actual maximum effluent concentrations (<0.05 µg/l) and assuming 25% by volume mixing zones, but would occur when calculated with the 40X higher technology based effluent limits (2 µg/l), which have never occurred. The consequence of setting mixing zones for a pollutant even though it is not present in concentrations of environmental concern is that it allows effluent limits that are high enough to be measured with conventional analytical techniques, rather than by comparatively costly sub-nanogram level techniques. While acknowledging that this is a regulatory fiction, it was continued here in the dispersion and dilution modeling of permitted mercury concentrations, which were modeled like the other constituents were. In the scenario of the Salmon River @ 2000cfs, a 25% volume mixing zone would result in an average monthly limit of 0.75 µg/l, which CORMIX modeling predicted that the chronic criterion would not be met until 87m below the outfall. Lest this fiction be repeated in future reports, and since the length of the mixing zone is otherwise limited to 50m, the mixing zone was limited to a volume that resulted in a 0.2 µg/l limit. This is the same limit as at low flow conditions for that outfall, for which the chronic criterion would be modeled to be met a short distance below the diffuser.

[3] Increased pollutant concentration at a given channel width was modeled only for copper and zinc, since these are the only pollutants which could potentially cause avoidance reactions in migrating fish at sub-criteria concentrations.

[4] Salmon River – For the Salmon River 25% of the assimilative capacity (difference between most stringent criterion and average upstream ambient concentrations) is used as a guide to ensure that lower water quality does not result (i.e. changes to water quality that are measurable and adverse to beneficial uses) from new or increased discharges to special resource waters. All modeled travel times for organisms drifting through the acute mixing zone are <1 minute.

[5] The column "Potential resulting instream concentrations" gives the estimated concentration that would result from discharging at permit limits at the specified dilution ratio limits into the average measured upstream concentrations in the receiving water (Table 20). These results from outfall 001 into Thompson Creek were then used as the upstream concentrations for outfall 002 into Thompson Creek in Table 21. The great majority of the time, the increased concentrations would be less than the Table 21 values, otherwise permit exceedances would result.

[6] Effluent concentrations are expressed in µg/l as total recoverable concentrations for all metals, however some of the metals criteria are expressed as dissolved metals. To relate total recoverable to dissolved concentrations, EPA recommends using either site-specific or default "translators" (EPA 2000b, p. C-7). The default translators are the total-dissolved conversion factors from Idaho WQS. For most metals of concern except for lead and cadmium, the translator values are all >0.9, close enough to 1.0 that their use in modeling dispersion and dilution would have little consequence. Lead default translator values ranged from 0.8 to 0.9 in Thompson Creek, 0.6 – 0.8 in Squaw Creek, and 0.98 in the Salmon River. Lead values were "translated" with the default factors in modeling Thompson Creek only, since results would not be affected much for the other outfalls. The measured dissolved/total lead fraction for all values in Thompson Creek that exceeded the 0.05 µg/l detection limits (n=10) was 0.91. Lead values in parenthesis are the "translated values." Cadmium "translator" values range from 0.4 to 0.9, but since modeling at the higher, total values shows criteria would quickly be met, there was no point in also modeling lower concentrations.

[7] In the mixing zone field study, the first sampling station was 25m downstream of the outfall. For pollutants that met the CMC between 0 and 25m, CORMIX was used to estimate intermediate distances at which CMC would be met (in parentheses). CORMIX predicted that acute criteria would be met for all by about 11m or less. Since at 25m, measured concentrations were about 50% modeled concentrations, these results are rounded down to 10m, and the acute mixing zone is set at 10m.

[8] Since the basis for imposing limitations on mixing zone size limitations is to allow fish passage, regulatory mixing zone volumes for whole effluent toxicity and recreation criteria are actual (100%) receiving water volumes.

Physical sizes of Mixing Zones

Cumulative Effects of Mixing Zones

Because of the under-prediction of far-field mixing in these waters (*Hydrodynamic modeling* section), rather than allowing the entire CORMIX far-field region as a regulatory mixing zone, mixing zone limits are imposed at 100 meters downstream of the discharges to Thompson Creek, and 50 meters downstream for the discharges to Squaw Creek and the Salmon River. These values were qualitatively selected by considering visual mixing observations and the predicted values from Table 21. These dimensions are summarized below.

Table 22. Physical sizes of mixing zones for the Thompson Creek Mine discharges.

		Size of allowable mixing Zone in the receiving waters		
Discharge point	Receiving water	Percent of volume (Table 21)	Zone of initial dilution (meters downstream)	Longitudinal downstream extent of mixing (meters)
Outfall 1 (Buckskin Creek)	Thompson Creek	Varies	10	50
Outfall 2 (Pat Hughes Creek)	Thompson Creek	Varies	10	50
Outfall 4 (pipeline and diffuser)	Squaw Creek	Varies	3	50
Outfall 5 (pipeline and diffuser)	Salmon River	Varies	3	50

The combined areas of the mixing zones must be as small as practicable to avoid interfering with designated uses or the established aquatic life communities. Whereas the acute mixing zone, or zone of initial dilution, is sized to prevent lethality to passing organisms, the larger chronic mixing zone needs to be sized to protect the ecology of the water body as a whole. If the total area affected by elevated concentrations is small compared with the total area of a river segment, then the mixing zones are likely to have little effect on the integrity of the water body as a whole, provided they do not affect unique or critical habitats (EPA 1994).

The mixing zone analysis used the 99.6th percentiles of the most severe flow situations to define the critical design flow for the waterbodies. In other words, concentrations of pollutants in and below the mixing zones will be less than the Table 21 instream concentrations 99.6% of the time. Using the effluent concentrations in Table 21, but calculating resulting instream concentrations at the 95th percentile flow conditions (Table 19) shows that 95% of the time, instream pollutant concentrations are approximately half the concentrations listed in Table 21. Within the acute zone of initial dilution, these concentrations fall below the chronic criteria. Hence, about 19/20 of the time, chronic criteria would be met within the acute mixing zone. In the remaining 1/20 of the time, chronic criteria would not be met throughout the chronic mixing zone, but would be met before the edge of the chronic mixing zone. The significance of this varies for different organisms. Adult salmonids can be quite motile, and would likely move into and out of the mixing zone as chemical concentrations dictate. Juvenile salmonids are much less motile, and can spend months within a 10 to 100 meter stream reach (Chapman and Bjornn 1969; Jakober er al. 1998). So for adult salmonids, the habitat losses in space and time would only be a fraction of the spatial habitat losses. For more sessile juvenile salmonids, sculpins, and invertebrates, habitat losses within the mixing zones are assumed to exist 100% of the time. These assumed losses range from a maximum of 0.5% habitat loss for sessile organisms in Thompson Creek to 0.0007% for adult fluvial salmonids (Table 23). These assumed losses are have been occurring in Thompson Creek since the mid 1980s, and for the present analysis, are considered part of the baseline condition. The assumed losses in Squaw Creek and the Salmon River are prospective. This comparison of the size of habitats that are potentially affected by the mixing zones indicates that the available habitat losses to the water bodies on the whole are small, and are likely to have little effect on the integrity of the water body as a whole.

Table 23. Cumulative impingement of chronic mixing zones on ambient waters

	Total Habitat Length (km)	Cumulative Mixing Zone Lengths (km)	Fraction of Total Habitat Length	Fraction of time chronic criteria could be exceeded in at least part of the mixing zones	Approximate habitat loss in space and time (Fraction)
Thompson Creek	20	0.1	$\frac{1}{200}$	$\frac{1}{20}$	$\frac{1}{4{,}000}$
Squaw Creek	42	0.05	$\frac{1}{850}$	$\frac{1}{20}$	$\frac{1}{17{,}000}$
Salmon River	360	0.05	$\frac{1}{7{,}200}$	$\frac{1}{20}$	$\frac{1}{144{,}000}$

Estimated from USGS 1:250,000 and 1:100,000 hydrography. Assumes that all streams at 1:250,000 scale are mostly perennial and provide habitat, streams shown only at 1:100,000 have no habitat value. Assumes river habitat only occurs in the mainstems of largest upper Salmon subbasin tributaries: Salmon River, E. Fork, Valley Cr, and Yankee Fork (i.e. for this exercise, the many Squaw Creek-sized and smaller streams are assumed to have no habitat value to fluvial fish)

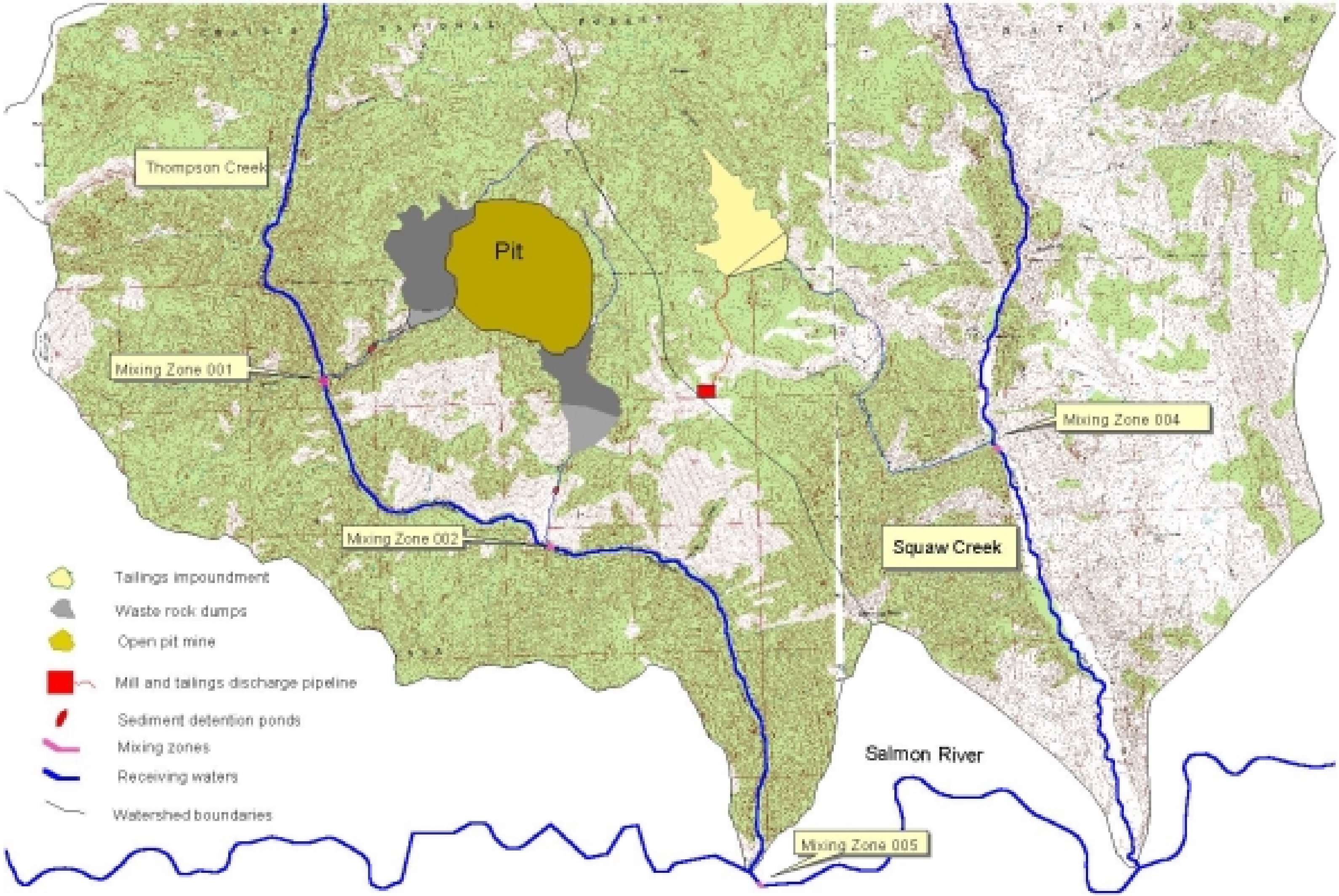


Figure 15. Mixing zones for discharges from the Thompson Creek Mine facility.

Monitoring to evaluate compliance with Idaho water quality standards

Ambient chemical monitoring in receiving waters

Special resource water (SRW) monitoring – During baseline pre-discharge operating conditions, the pollutants significant to designated beneficial uses listed in section 3 should be sampled at least 4-times annually at monitoring stations SR-1 and SR-3 (Figure 1)[17]. Single grab samples are sufficient at this phase of SRW monitoring. These sampling results will be used characterize background concentrations and temporal variability. When actually discharging to Outfalls 4 or 5, sampling frequency at SR-1 and SR-3 shall increase to at least monthly. At least 4X annually when discharging, SRW monitoring will include adequate sample replication with an objective to detect a 25% change in the assimilative capacity with statistical Type I error (α) no greater than 0.05 and a statistical type II error (β) no greater than 0.25. The necessary number of replicates to meet this test will be assumed to be at least 8 unless otherwise demonstrated using a statistical sample power calculation described in section 3 using actual sample variability and the objectives listed above.

Follow up sampling if criteria concentrations exceeded (for all receiving water monitoring) – Numeric toxics criteria are defined by concentrations, and durations and frequencies of exceedances. Acute criteria are considered the highest concentration of a pollutant to which aquatic life can be exposed for a short period of time (1-hour average) without deleterious effects. Chronic criteria are considered the highest concentration of a pollutant to which aquatic life can be exposed for an extended period of time (4-day average) without deleterious effects. DEQ considers single grab samples to adequately represent 1-hour average concentrations for criteria exceedance purposes. However, a single grab sample may not always represent the 4-day average concentrations applicable to chronic criteria.

If monitoring results show that any chronic criteria concentrations are exceeded, then at the next scheduled monitoring, sampling and analysis for at least those pollutants at that station, shall be expanded to include 4-day average concentrations. The 4-day average concentrations shall include at least one grab sample per day for 4 consecutive days. If the 4-day average concentration also exceeds criteria, then all further monitoring at that station for those pollutants shall include 4-consecutive day samples instead of single-time grab samples, until otherwise notified by DEQ. If the original criteria concentration exceedance occurred on the last scheduled hydrograph based sampling date of the year, then the station should be re-sampled as soon as practicable, which we assume will be no later than 1-week after TCM receives the sampling results. For example, if the mine is on an April, June, August, and October sampling rotation, if the October results exceed criteria, then the station should be promptly re-sampled, rather than waiting until the following April.

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[17] TCMC currently follows a hydrograph weighted schedule rather than a straight quarterly schedule to their ambient water quality monitoring. They weight their sampling frequency to the high-flow period based on the assumption that water quality is more variable during high-flow periods. They currently sample receiving waters in April, June, August, and October.

Salmon River flow data

Accurate flow information is necessary for the flow-based limits of new and increased discharges to the Salmon River. A gaging station was maintained at the Salmon River downstream of the Yankee Fork near Sunbeam a short distance upstream of Thompson Creek from 1922-1991 (USGS station 13296500). There are two major drainages (Warm Springs Creek and Slate Creek) between this site and the location of proposed outfall 005, the flows from which would not be captured at this location. However, the contribution of flow from these drainages can be estimated from their proportional drainage areas. USGS has published regional rating curves for estimated discharges from un-gaged basins. Therefore, because of the period of record, re-establishing this station would be preferable to establishing a new station closer to the point of discharge, which would have no period of record.

Field bioassessments

Field bioassessments of instream biota (benthic macroinvertebrate, fish assemblages, and periphyton assemblages) is necessary to monitor protection of aquatic life beneficial uses. Based upon the analyses in this report, and experiences from an extensive state-wide bioassessment network the following biological monitoring is recommended.

Water samples collected for metals analysis or toxicity testing may reflect a snapshot of ephemeral conditions at the time of sampling. In contrast, periphyton, benthic macroinvertebrates, and fish assemblages integrate changes in exposure conditions over time and provide a continuous monitor of water quality. Because of this, the recommended approach is to evaluate and take regulatory action based upon chemical data, whole-effluent toxicity, and field biosurveys. The regulatory interpretation of receiving water chemical and whole effluent toxicity data have been treated in detail in national regulatory and scientific guidance; however, decision making based upon biosurvey results is only generally discussed (EPA 1991a, Groethe et. al. 1996). Recommendations for decision making using field assessment data at this site follow.

The long-term trend biomonitoring that the Thompson Creek Mine has conducted on Thompson Creek and Squaw Creek should be continued in a similar manner to the program to date. This effort has produced an outstanding data set of macroinvertebrate and fish population dynamics in the affected streams. Only minor modifications are recommended as follows:

Periphyton monitoring

Primary producers can be very sensitive to toxicants in effluent discharges. Changes in algal assemblage may result from metals stress, because some metals depress cell division and photosynthesis in some algal species more than in others. Algal communities adapted to heavy metal stress may be more resistant to further metals stress (LaPoint and Waller 2000).

Idaho, along with the majority of those states with active bioassessment programs, has primarily based interpretations on benthic macroinvertebrate assemblages. However, to supplement the invertebrate and fish community assessments, a multimetric diatom index of biotic integrity (D-IBI) is under development for diatom algae in rivers. The D-IBI is composed of 10 measures of tolerance/intolerance to pollution, autecological guild, morphometric guild, and individual condition (IDEQ 2000a). The 10 metrics are scored and added together to make a D-IBI score, similarly to multimetric macroinvertebrate and fish indices of biotic integrity that have been developed throughout the country. In support of this, periphyton was sampled from the Salmon River in the vicinity of the Thompson Creek Mine in 1998 and 1999 (IDEQ 1998, 2000a). Periphyton sampling should be conducted annually during base flow conditions in the vicinity of established monitoring sites SR-1 and SR-3.

Macroinvertebrate monitoring

The current macroinvertebrate monitoring on Thompson and Squaw Creeks has been rigorous and data interpretation has been appropriate for the constituents of concern in the effluents (metals). Baseline and ongoing macroinvertebrate monitoring in the Salmon River should be initiated on an annual basis during base flow conditions. IDEQ has established sampling protocols for macroinvertebrates in rivers. The protocols are scaled up from stream sampling protocols wherein riffle or cobble-substrate areas of the river margins are sampled with a Slack sampler (IDEQ 1998; Cuffney et al. 1993).

Fish monitoring

The current fish monitoring on Thompson and Squaw Creeks has been rigorous and data interpretation has been appropriate. We understand that NMFS would prefer less rigorous trend data on Thompson and Squaw Creeks and have limited data collection to tri-annual sampling under ESA §10 scientific collection permits. This monitoring frequency is incompatible with the purpose of monitoring, which includes annual trends analysis. The permittee or their consultant is encouraged to re-apply for at least bi-annual sampling. Baseline and ongoing fish monitoring in the Salmon River should be initiated as practicable under ESA §10. The USGS has established procedures for sampling fish assemblages in rivers (Meador et al. 1993 Maret 1997). Trends monitoring of fish assemblage composition in Idaho rivers is a routine part of the USGS/DEQ statewide water quality trends network (O'Dell et al. 1998), although there is currently no site established near the mine.

Data interpretation and decision making

The interpretation of macroinvertebrate community composition reported in the annual monitoring reports has been appropriate. Additionally, IDEQ is developing multimetric indices for macroinvertebrates, periphyton, and fish assemblages that will be used to interpret aquatic life beneficial use attainment (*Biological Evaluation*, this report; IDEQ 2000a). These are currently under revision or development, but as available, these should be calculated and reported. Further, the abundance and diversity of mayflies has consistently been shown in field and lab experiments to be a sensitive measure of metals (*Biological Evaluation*). This measurement endpoint is more specifically suited to evaluate the discharges from a mining site than the general purpose multimetric indices.

Table 26. Receiving water bioassessment measures and analyses

Assemblage	Receiving Waters	Endpoint	Analysis
Benthic macroinvertebrates	Salmon River	Invertebrate River Index (Multimetric index)	Calculate scores
Benthic macroinvertebrates	Salmon River	Abundance and taxa richness of Ephemeroptera	Hypothesis and trends testing (e.g. Chadwick 1999)
Periphyton	Salmon River	Diatom IBI (Multimetric index)	Calculate scores
Fish	Salmon River	Presence and relative abundance of species; multimetric index	Hypothesis and trends testing (e.g. Chadwick 1999), calculate scores
Benthic macroinvertebrates	Thompson and Squaw	Stream macroinvertebrate index (Multimetric index)	Calculate scores
Benthic macroinvertebrates	Thompson and Squaw	Abundance and taxa richness of Ephemeroptera	Hypothesis and trends testing (e.g. Chadwick 1999)
Fish	Thompson and Squaw	Presence and relative abundance of species; multimetric index as available	Hypothesis and trends testing (e.g. Chadwick 1999)

Interpretation and follow-up actions to be taken based upon monitoring results

Macroinvertebrates: abundance or taxa richness of mayflies – If upstream-downstream sampling sites have similar substrates, stream size, aspect, and other habitat features, abundance or taxa richness of mayflies would be expected to be similar. If hypothesis tests indicate downstream differences or declining trends of abundance or taxa richness of mayflies occurs compared to upstream reference stations, then the causes shall be investigated. Investigations should consider more frequent chemical sampling to better define waterborne potential exposure routes, exposure through sediments or *awfuchs, in situ* toxicity testing, or sediment toxicity testing. Other actions to considered include increasing the frequency of WET testing to 4X annually with both *Ceriodaphnia* and fatheads. If WET testing was ongoing, but not showing toxicity despite declining mayfly taxa richness or abundance, then receiving water trigger concentrations should be re-evaluated, or additional safety factors applied.

Macroinvertebrates: multimetric scores – If scores are lower downstream than upstream, the component metrics should be considered, and the components causing the reduced scores should be evaluated. If the evaluation indicates water quality is depressing the scores, then further investigations to identify and remedy the causes should be undertaken.

Periphyton – If multimetric scores are lower downstream than upstream, then the component metrics should be considered, and the components causing the reduced scores should be evaluated. If the evaluation indicates water quality is depressing the scores, then further investigations to identify and remedy the causes should be undertaken. Patterns discerned through descriptive and exploratory statistics should be interpreted.

Fish – As has been the practice with the Chadwick annual monitoring report series, results of hypothesis tests and trends assessment should be interpreted, explained, and if necessary investigated further. If multimetric scores are lower downstream than upstream, then the component metrics should be considered, and the components causing the reduced scores should be evaluated. If the evaluation indicates water quality is depressing any portions of the assemblage, then further investigations to identify and remedy the causes should be undertaken. If, for example, trout densities are lower below the discharges than above, and environmental co-variates such as physical habitat features or temperature differences cannot not fully explain the differences, then IDEQ will presume that the apparent effects are due to the discharges. Additional investigation to identify and reduce the toxicity should then be undertaken.

Bioaccumulation study

Available information indicates that bioaccumulation of potential pollutants at levels harmful to aquatic life are unlikely (e.g. sediment chemistry, absence of apparent fish population effects, similar tissue concentrations in aquatic insects and fish upstream and downstream of discharges). However, to definitively resolve whether selenium from TCMC discharges results in risk to aquatic life in Thompson Creek, a focused field assessment to assess whether exposure to selenium through the food chain poses a risk of adverse effects to aquatic life. The goal of the bioaccumulation study includes establishing a threshold for preventing risk to Thompson Creek fish populations from selenosis. While the thresholds could be developed for various media, they should be able to be related to a waterborne concentration. The bioaccumulation threshold should account for aqueous selenium concentrations low enough to prevent accumulation in fish food organisms, which in turn would result in the accumulation of selenium to high enough levels in parental fish to cause reproductive impairment, or other adverse effects. Adverse effects are those toxicological endpoints with clear relevance to population effects such as reproduction, survival, growth, and teratogenisis.

Test questions of the study would also include whether selenium concentrations in environmental media and the food chain are elevated above reference conditions and if these elevated concentrations were biologically meaningful. These questions could be met by answering two major testable questions:

1) *Statistical significance* – Are selenium concentrations in water, sediment, *aufwuchs,* macroinvertebrates, sculpin, or trout downstream of mine discharges to Thompson Creek statistically higher than concentrations upstream of mine discharges?

2) *Biological meaningfulness* –Do dry weight concentrations of selenium in the food chain exceed the following biological screening levels?

Media	Selenium concentrations mg/kg (dry weight)
Sediment	3.5
Aufwuchs (periphyton and abiotic material embedded in the periphyton)	4.0
Macroinvertebrates (community composite)	4.0
Forage fish (sculpins) (Whole body)	4.0
Salmonid (Whole body)	4.0

Data interpretation and other recommendations.

Conclusions whether concentrations are statistically higher should be based upon hypothesis testing between reference and test sites using a t-test, Mann-Whitney test, or other appropriate statistical test, with $\alpha = 0.05$. The actual statistical power of the test should also be reported.

If for all of the media or thresholds listed in test questions #1 and #2 above, the answers to *either* test questions #1 and #2 are *no*, then the conclusion will be that adverse biological effects from bioaccumulative chemicals from the mine's discharges are unlikely in Thompson Creek. If for any of the media or thresholds listed in test questions #1 and #2 above, the answers to *both* test questions #1 and #2 are *yes*, then further analysis would be appropriate.

What further analyses, if any, are appropriate will need to be determined by the principal investigator, depending on the results. Based on the selenium review in this report, a few examples of potential further analyses follow; these examples are not directive or limiting. For selenium, if fish tissues exceed the screening threshold, analyses of selenium concentrations in ovaries, or histopathological examinations of gill tissue may provide evidence whether effects are occurring the field (Lemly 1993, 1996). Lemly (1996) considered reproductive success to be probably the most sensitive indicator of selenium impacts to fish, and recommends the evaluation of gravid ovaries when designing aquatic monitoring studies for selenium. If ovaries have elevated selenium concentrations, collecting eggs and milt from field exposed fish, and comparing hatching success and embryonic deformities among exposed and controls would likely be the definitive test for reproductive effects (Lemly, written communication). Other possibly useful tests include short-term teratogenesis bioassays since teratogenic effects to fish or amphibians from selenium have been documented (Bantle 1995, Lemly 1997).

Surface sediment samples should be collected whole, rather than field sieved, to mimic biota exposure. Samples should be collected from about the top 2 cm of depositional areas in pools or margins with fine-grained surface sediments. Lab analyses of sediment conventionals (e.g. grain size and total organic carbon) may help interpret results. *Aufwuchs* are suggested in the media because with copper and arsenic at least, invertebrate tissue residues have been more strongly correlated with *aufwuchs* than water or sediment (Beltman et al. 1999)..

Routine follow-up bioaccumulation monitoring during fish monitoring – The study described above is anticipated as a one-time effort. However, limited follow-up tissue residue sampling and analysis should be incorporated into the monitoring program. Because of exposure of aquatic life to potentially bioaccumulating metals in the mixing zone, trend monitoring of selenium tissue residues is appropriate to ensure general water quality criteria are met, e.g. avoidance of food-chain effects. Based upon the study findings, the principal investigator should include recommendations for ongoing monitoring in the study report. Retaining a sub-sample of sculpins captured during fish population sampling for whole-body tissue analyses is suggested. Sculpins are suggested due to their intermediate trophic level, limited motility, and abundance in the study area. Much more information will be available after the bioaccumulation study is completed, and other target organisms or media could be appropriate.

Conclusions

The information reviewed supports the following conclusions:

1. Existing discharges have not caused and are unlikely to cause measurable biological harm to the receiving waters.

   a. Abundances of metal-sensitive macroinvertebrates (Ephermeropterans) are similar above and below existing discharges.

   b. Salmonids and sculpins, fishes that are known to be sensitive to constituents that are regulated in the mine's discharges, are widely distributed in Thompson and Squaw Creeks above and below existing discharges

2. Toxicity testing showed the discharges were not expected to be toxic to laboratory test organisms at concentrations expected in the zone of initial dilution of the discharges.

3. Idaho's water quality standards for metals, as implemented here, are protective of all aquatic life species likely to occur in the vicinity, including threatened species occurring in the receiving waters (chinook salmon, and steelhead and bull trout), as well as for cutthroat trout and shorthead sculpin.

4. The existing and proposed discharges are unlikely to block zones of passage for migratory fish species through the mixing zones.

5. Chemical conditions from existing discharges comply with applicable water quality standards, with the possible exception of selenium, for which instream concentrations approach and may exceed the chronic criterion. Further evaluation of whether selenium is elevated in the aquatic food chain is required. Chemical conditions from proposed discharges are projected to, and must comply with, applicable water quality standards.

6. The existing and proposed physical configurations of the discharges are unlikely to interfere with aquatic life or recreation uses in the receiving waters.

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