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Alaska Juneau Gold Mine Project

Technical Assistance Report for the U.S. Army Corps of Engineers Alaska District

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SUMMARY

Project Description

The Alaska Juneau (AJ) Gold Mine project is a proposal by Echo Bay Alaska (Echo Bay) to reopen the historic AJ gold mine near Juneau in southeast Alaska. The proposal entails mining approximately 22,500 tons of ore per day and, after crushing and grinding the ore, recovering gold through the froth flotation and carbon-in-leach (CIL; also referred to as cyanide leach) processes. After destruction of residual cyanide in the CIL tailings using a sulfur dioxide/air process, the tailings would then be discharged in a slurry form to a tailings impoundment that would be created in Sheep Creek valley, four miles south of downtown Juneau.

Echo Bay proposes to construct a 345' high roller compacted concrete dam that would create a reservoir approximately 2.5 miles long in Sheep Creek valley. Tailings would be discharged into this impoundment via a barge mounted "elephant trunk" tailings discharge pipeline extending to the bottom of the reservoir. During operation, mine drainage water and process water, equivalent in volume to the net precipitation (after evaporation) over the impoundment, would be discharged from the impoundment to Gastineau Channel, approximately one mile to the west of the tailings impoundment.

The tailings impoundment would serve as a permanent disposal site for the AJ mine tailings. Echo Bay's proposal is to leave a minimum of twenty feet of water over the tailings and, after mining and tailings disposal cease, to allow the impoundment to serve as a recreational lake. A more complete project description can be found in the AJ Gold Mine Project Final Environmental Impact Statement (FEIS; BLM, 1992).

Regulatory Background

This report evaluates short and long-term water quality impacts from the project as well as long-term ecological consequences. Findings and recommendations have been developed to assist the Alaska District Corps of Engineers in determining whether the proposed project complies with the Clean Water Act (CWA) Section 404(b)(1) Guidelines.

A CWA Section 404 permit is required to place fill for construction of the tailings impoundment, which is intended to function as a wastewater treatment system. These permits are issued by the Corps of Engineers with the assistance of EPA. A CWA section 402 permit would be required during operation for the discharge of wastewater from the impoundment (as well as mine drainage) to Gastineau Channel. CWA section 402 permits are

subject to the provisions of the National Pollutant Discharge Elimination System (NPDES) administered by EPA.

EPA and the Corps of Engineers have agreed that if the construction of the impoundment is permitted under Section 404 of the CWA, via an individual permit, the tailings impoundment would be considered a non-jurisdictional waste treament system during operation. The waste treatment would occur as tailings settle to the bottom of the impoundment. As a non-jurisdictional waste treatment system, the impoundment would not have to meet State of Alaska water quality standards (WQS) during operation. Once mining ceases and the impoundment is no longer being used as a waste treatment system, EPA's position is that the reservoir would become a water of the U.S. that must meet state WQS, consistent with the goals of the CWA, within a reasonable timeframe after the discharge of tailings ceases.

Scope of the Report

This report addresses impacts from the proposed discharge of process wastes, both solid and liquid, from the AJ mine project. These impacts are analyzed with respect to risks associated with the potential release of contaminants into the aquatic environment and with respect to losses of aquatic habitat productivity (e.g., wetlands) from direct physical disturbance.

A fundamental question which this report addresses is whether or not there is a reasonable assurance that the impoundment would in fact provide adequate treatment such that EPA's applicable New Source Performance Standards (NSPS; see 40 CFR 440.104)) would be met at the point of discharge and that State of Alaska WQS would be met in the receiving waters of Gastineau Channel. Chapter VI addresses this question with respect to NSPS, relying heavily on water quality modeling analyses. Chapter VII addresses overall water quality impacts to Gastineau Channel and the likelihood of meeting WQS.

Another fundamental question relates to whether there is reasonable assurance that the proposed method for tailings disposal and long-term maintenance would prevent release of contaminants in harmful quantities. Chapter VIII presents an ecological risk assessment of post-operation conditions and reviews studies of Canadian lakes that have been used for tailings disposal.

Potential measures for mitigating short and long-term water quality impacts are addressed in Chapter IX.

A third key question is whether the significant impacts caused by construction and operation of the tailings impoundment can be mitigated to the point that overall impacts on aquatic resources are acceptable. Optional mitigation plans and strategies are reviewed in Chapter X.

Affected Environment

The Sheep Creek tailings disposal option would fill 420 acres of Sheep Creek valley. Waters that would be eliminated in association with the proposed fill are 2.5 miles of Sheep Creek above the tailings dam and associated wetlands for a total of 20.1 acres of aquatic habitat (COE, 1994). The flow of water in 1.1 additional miles of Sheep Creek downstream of the impoundment would be reduced significantly.

Mammals that inhabit the immediate area include black bear, mountain goat, Sitka black-tailed deer, beaver, marten, river otter, mink, ermine, and other mustelids, lynx, red fox, hoary marmot, porcupine and other small mammals (Holmberg, 1991). There are 131 species of birds that likely inhabit the project area for at least some time of the year. Among these are bald and golden eagles as well as more than a dozen neotropical migratory song birds and shore birds (Wilson and Comet, 1991).

The wetlands of Sheep Creek valley are an element of a diverse mosaic of vegetation communities which the FEIS describes as unique among the alternative tailings disposal sites considered. Due to this vegetation mosaic, the FEIS describes the Sheep Creek valley as having more species diversity than any other site accessible on the Juneau road system. The wetlands serve basic ecological functions within this mosaic typical of wetlands at other locations. However, the upper portion of Sheep Creek valley is unusual for the Juneau area because of the composition of its vegetative communities. Vegetation consists of coniferous forest, deciduous forest, tall shrub, upland meadow, shrub wetland and wet meadow. The deciduous forest, composed primarily of cottonwood, is uncommon in the greater Juneau area. Rough estimates suggest that Sheep Creek Valley contains 25% of the total area of cottonwoods between Taku Inlet and Berners Bay (BLM, 1992).

The mixture of cottonwood and wetland shrub communities apparently provides a high quality habitat for song birds. Song bird populations have been found to be "locally diverse and abundant" (Wilson and Comet, 1991). Sheep Creek valley had five times the song bird nest density and over 323% more successful nests than a nearby site with similar vegetation (Comet and Wilson, 1994). Habitats adjacent to the valley floor have notably fewer songbirds (Wilson and Comet, 1993). Many of the song bird species are of special interest. Five species whose abundance is thought to be in decline in Alaska breed in the valley. These are the fox sparrow, orange crowned warbler, blackpoll warbler, American robin, and varied thrush. Eight other species found in the valley may be increasing in abundance.

Fifteen of the 42 species of birds documented during formal censusing are of interest to the national program on neotropical birds (Wilson and Comet, 1991). One bird, the marbled murrelet, whose use of the valley has not been documented, is of concern because of population declines along the west coast of North America.

Aquatic resources of the upper portion of Sheep Creek include a local population of Dolly Vardon char. This population is isolated from other char populations by the impassible lower reach of Sheep Creek.

Gastineau Channel is a north-south oriented channel separating Douglas Island from the mainland. The shoreline for approximately 10 miles between Stephens Passage and Juneau is largely steep sided and rocky. Forty species of demersal fish, shellfish and other invertebrates have been reported from Gastineau Channel. Included among these are commercially important crab species. The commercial crab fishery was closed in 1978 but a popular personal use fishery continues (BLM, 1992).

Adequacy of Wastewater Treatment

Chapter VI addresses the projected performance of the tailings impoundment as a waste treatment system. Water quality models and information from existing mines are used to determine whether the discharge from the impoundment, when comingled with mine drainage, would be likely to meet EPA's New Source Performance Standards (NSPS) that would apply to the discharge as end-of-pipe effluent limits. Compliance with Alaska's water quality standards (WQS) is addressed in Chapter VII.

The chemical composition of the tailings slurry discharge is discussed. Concentrations of various chemical constituents of the influent to the tailings pond are presented. They represent the pollutant loadings to the tailings pond that are subsequently addressed in water quality models.

Two water quality models were applied to estimate the levels of pollutants that would be expected in the discharge. The EPA's Water Quality Simulation Program (WASP version 4.32) was used to simulate the levels of solids, cyanide and metals in the water and sediments of the proposed tailings pond during and after the operation of the mine.

The WASP software has been thoroughly tested and the program has been used in a wide variety of applications. The WASP water quality analysis characterizes the major processes affecting the distribution of solids, cyanide and certain metals in the tailings pond. These processes include turbulent mixing in the pond itself, settling of the tailings, leaching and partitioning of metals and effects of initial mixing associated with the discharge.

The WASP model generally reflects the current state of knowledge for simulation of both inorganic and organic toxic substances. It was used to predict concentrations of pollutants in the water column as well as in pore water within the tailings. During the preparation of this report, however, EPA consulted with various experts in the field of small particle transport, some of whom expressed concerns that the WASP4 model might not be capable of adequately evaluating the effects of the hydrodynamics of the reservoir on the settling of suspended solids. They recommended applying the CE-QUAL-W2 water quality model.

EPA therefore simulated the pollutant concentrations in the tailings impoundment discharge using the CE-QUAL-W2 model (version 2.04) in order to have a comparison with the earlier results from the WASP4 model. CE-QUAL-W2 has been applied successfully to estuarine systems as well as freshwater reservoir systems. CE-QUAL-W2 generally reflects the current state of knowledge for simulation of chemical, physical and biological state variables in estuarine or reservoir systems. While originally designed to model temperature stratification in Corps of Engineer managed reservoirs, it has been modified to predict pollutant concentrations in the water column.

For the WASP4 model analysis, twelve scenarios were developed that capture the potential variability among such factors as:

- vertical mixing due to wind stress, kinetic energy from sources that include Sheep Creek and the waste stream itself and potential energy associated with density differences
- hydrodynamic regime
- groundwater inflow
- mixing characteristics of sediment pore water

In the CE-QUAL-W2 model analysis, suspended solids and concentrations of dissolved metals and metals in suspended solids in the proposed tailings pond were simulated under the conditions representing natural variability of weather and inflow hydrology, as well as estimated variability in the discharge characteristics. Scenarios that were modeled are based on different assumptions regarding particle settling rates and groundwater inflow. Modeling results are presented as cumulative distribution functions that reflect the probability that each pollutant in the effluent would exceed a certain concentration.

In additon to water quality modeling, empirical data for total suspended solids (TSS) from other operating mines were reviewed for comparison with modeling results. TSS levels for the mine deemed to be most comparable to the AJ project, Island Copper, are in the range of results predicted for the AJ discharge.

The overall conclusions of the modeling effort are that the effluent would not meet projected end-of-pipe NSPS for total suspended solids (TSS) and copper.

Effects on Gastineau Channel

The WASP4 water quality model was also applied to model the effects of the tailings impoundment discharge on Gastineau Channel. Chapter VII discusses this effort, which examines both near-field and far-field (Channel-wide) dilution and dispersion of the effluent. The central question addressed in this chapter is whether there is adequate mixing in Gastineau Channel to dilute the pollutants in the impoundment and mine drainage effluent to ecologically safe levels in compliance with Alaska's WQS.

After a review of previous studies of currents in Gastineau Channel, the WASP4 model framework is presented. Model runs include a worst-case discharge scenario and an average case discharge scenario. The overall conclusion of the analysis is that WQS would likely be exceeded for cyanide, arsenic and copper under both average and worst-case scenarios in much of Gastineau Channel. This is due to the limited flushing within the Channel and to the ambient background levels of pollutants in the Channel that limit the Channel's assimilative capacity.

Potential Long-Term Contamination

Chapter VIII describes the impoundment setting and addresses the type of aquatic habitat that would likely develop in and around the impoundment. This is followed by an ecological risk analysis that examines the potential effects of contaminants on aquatic biota and wildlife likely to inhabit the area.

The impoundment would be an unusual aquatic feature insofar as it would be a shallow yet steep-sided reservoir. Shallow lakes generally have gradually sloping shorelines and steep sided lakes or reservoirs tend to be fairly deep. As such, no analagous reservoirs or lakes were found in the area upon which to base a comparison or predict food chains.

With an estimated sedimentation rate of only 1200 cubic yards per year, coverage of the tailings by sediments to form a more natural substrate for benthic (bottom dwelling) organisms would not be expected in the short term. After mine closure, fish populations would not be likely without a managed food base. Insect production and transport by the inflowing small streams would not provide enough food to sustain fish. In addition, the impoundment lacks the habitat characteristics required for species survival (e.g., cover and spawning areas). Construction of the upstream diversion dam would impede fish access to potential spawning areas above the impoundment.

The post-closure vegetation expected on the surrounding slopes and avalanche dissipators is likely to be alder and shrubs. It has been suggested that the impoundment's shoreline would create new wetland habitats, however, this has not been analyzed in any detail and, given the steep surrounding terrain, seems unlikely.

An ecological risk assessment evaluated the potential toxicity of contaminant concentrations in pore water (i.e., water trapped between tailings particles where bottom dwelling plants and animals would root or burrow), sediments, and the water column. In addition, three species were selected for an analysis of the effects of the heavy metals in the tailings on wildlife. The kingfisher, spotted sandpiper and river otter were evaluated based on the likelihood that they would inhabit the impoundment area and would ingest water, organisms that would bioaccumulate the metals, as well as the tailings themselves.

The results of this analysis indicate that aquatic biota (including wildlife) would be at substantial risk from the contaminants in the tailings. Water quality criteria would likely be exceeded at high levels in the pore water (up to 200 times the acute criterion for cyanide) and water column (2 times the acute criterion for copper); sediment concentrations would likely exceed benchmark comparison values (over 400 times the lowest effect level for cyanide); and wildlife are likely to be at substantial risk from their exposure to high levels of metals in their diets (exceeding draft Great Lakes criterion for mercury by over 200 times).

Canadian studies of lakes used for disposal of mine tailings were also reviewed. These studies examined to some degree the impacts of mine tailings disposal on the health of the aquatic systems in these lakes. The findings of these studies, however, do not alter EPA's conclusion that the Sheep Creek tailings impoundment would present substantial risks to wildlife in the long-term.

Mitigating Water Quality Impacts

In view of the findings of the two previous chapters, which conclude that the project as proposed would likely violate effluent limits during operation and, after closure, place wildlife at substantial risk, Chapter IX addresses potential measures for reducing water quality impacts to significantly lower levels. Measures that are addressed include secondary treatment of the effluent, measures for reducing total suspended solids, eliminating the cyanide leach circuit and potential means for isolating the tailings to minimize the risk of long-term contamination. The Powerline/Icy Gulch tailings disposal alternative is briefly reviewed with respect to its potential feasibility as a disposal site using a more conventional, sub-aerial tailings disposal method (i.e., tailings would not be discharged underwater), surface water diversion and conventional reclamation (tailings covered with soil rather than water).

No single measure or combination of measures are deemed to be adequate to reduce both short-term (during operation) and long-term (post-operation) water quality impacts to significantly lower levels that would clearly avoid significant degradation of waters of the U.S. If feasible, the Powerline/Icy Gulch alternative (with subaerial tailings deposition, surface water diversion, secondary wastewater treatment and conventional reclamation) would offer some significant advantages relative to the Sheep Creek alternative in terms of minimizing degradation of waters of the U.S. The feasibility of this alternative, as well as other potential measures such as elimination of the cyanide leach circuit, would require much more in depth evaluation by Echo Bay and resource agencies.

Mitigating Ecological Impacts

Chapter X addresses options for mitigating or off-setting the loss of aquatic habitats (wetlands and Sheep Creek) that would occur if the tailings impoundment was constructed as proposed. Due to the findings of Chapter VIII, i.e., that risks to wildlife from exposure to heavy metals would be high, the creation of the impoundment itself is not considered to in any way offset the loss of Sheep Creek and the wetlands of Sheep Creek Valley. These wetlands are part of a diverse mosaic of plant communities that support a diversity and abundance of birds and other wildlife. The wetlands and Sheep Creek itself contribute to the overall aesthetic value of the area which is a popular hiking destination.

Options examined for off-setting the loss of aquatic habitats include restoration of Lemon Creek Valley, enhancing three ponds in the Juneau area and performing certain habitat and recreational improvements at the U.S. Forest Service Mendenhall Glacier Visitor's Center.

All of these options have serious limitations and none are deemed capable of off-setting the unique values of Sheep Creek and the wetlands of Sheep Creek Valley, particularly the significant loss of high quality migratory bird habitat.

Conclusions

Based on the findings of this report, EPA concludes that there is a high potential for significant degradation of waters of the U.S. both within Gastineau Channel and within the tailings impoundment after closure, i.e., after it is no longer used for treatment of wastewater and disposal of mine tailings. The specific major findings that lead to this conclusion are as follows:

Finding #1:

During operation, the wastewater discharge from the impoundment co-mingled with mine drainage is likely to exceed EPA's New Source Performance Standards (end-of-pipe effluent limits) for total suspended solids, copper and possibly mercury (see Chapter VI).

Finding #2:

During operation, the wastewater discharge is likely to cause widespread exceedances of state of Alaska water quality standards for cyanide, arsenic, copper and possibly mercury and lead (see Chapter VII);

Finding #3:

After closure, indigenous wildlife that would likely inhabit the tailings impoundment would be at substantial risk due to contaminants that would likely persist in the impoundment. Water quality criteria would likely be exceeded at high levels in the pore water (up to 200 times the acute criterion for cyanide) and water column (2 times the acute criterion for copper); sediment concentrations would exceed benchmark comparison values (over 400 times the lowest effect level for cyanide); and wildlife are likely to be at substantial risk from their exposure to high levels of metals in their diets (exceeding draft Great Lakes criterion for mercury by over 200 times; see Chapter VIII).

Finding #4:

Unlike the Kensington Mine project, reliable measures (e.g., secondary treatment of the effluent, isolating the tailings) for reducing the anticipated water quality impacts described above to significantly lower levels do not appear to be feasible. Others, such as eliminating the cyanide leaching process or using subaerial tailings deposition and conventional reclamation at an alternative disposal site, would require much more detailed analysis to determine feasibility as well as overall environmental impacts (see Chapter IX).

Finding #5:

The loss of aquatic habitat (Sheep Creek and associated wetlands) would be significant due to their contribution to the unique diversity and productivity within the Juneau area, particularly in terms of migratory bird habitat and the aesthetic quality and recreational value of Sheep Creek valley. Potential measures identified to replace these values, including on-site and off-site measures, do not appear either feasible or adequate to prevent a significant loss of aquatic resources (see Chapter X).

I. INTRODUCTION

The Alaska Juneau (AJ) Gold Mine project is a proposal by Echo Bay Alaska (Echo Bay) to reopen the historic AJ mine near Juneau in southeast Alaska (see figure 1). The proposal entails mining approximately 22,500 tons of ore per day and, after crushing and grinding the ore, recovering gold through the froth flotation and carbon-in-leach (CIL; also referred to as cyanide leach) processes. After destruction of residual cyanide in the CIL tailings using a sulfur dioxide/air process, the tailings would then be discharged in a slurry form to a tailings impoundment that would be constructed in the Sheep Creek Valley. Mine drainage water and process water, equivalent in volume to the net precipitation (after evaporation) over the impoundment, would be discharged to Gastineau Channel, approximately one mile to the west of the tailings impoundment (see Figure 2).

This report evaluates short and long-term water quality impacts from the project as well as long-term ecological consequences. Findings and recommendations have been developed to assist the Alaska District Corps of Engineers in determining whether the proposed project complies with the Clean Water Act (CWA) Section 404(b)(1) Guidelines. A CWA Section 404 permit is required to place fill for construction of the tailings impoundment which is intended to function as a wastewater treatment system. These permits are issued by the Corps of Engineers with the assistance of EPA.

If permitted under CWA section 404, the impoundment would not be considered a jurisdictional water of the U.S. As such, a permit to discharge process wastewater, which includes tailings, into the impoundment would not be required but a CWA section 402 permit would be required for the discharge <u>from</u> the impoundment to Gastineau Channel. CWA section 402 permits are subject to the provisions of the National Pollutant Discharge Elimination System (NPDES) administered by EPA. The state of Alaska can require more stringent conditions if necessary to meet state water quality standards, in accordance with Section 401 of the Clean Water Act.

In summary, this report addresses the overall question of whether this project can be constructed and operated so as to comply with certain critical provisions of both CWA sections 404 and 402. These two permits are closely related. The primary purpose for the 404 permit, other than for permanent tailings disposal, is to construct a tailings impoundment wastewater treatment system that would ensure that discharges from the impoundment would meet the provisions of the CWA section 402 permit. Therefore a 404 permit should only be issued if it can be demonstrated that there is reasonable assurance that the 402 permit provisions would be attained.

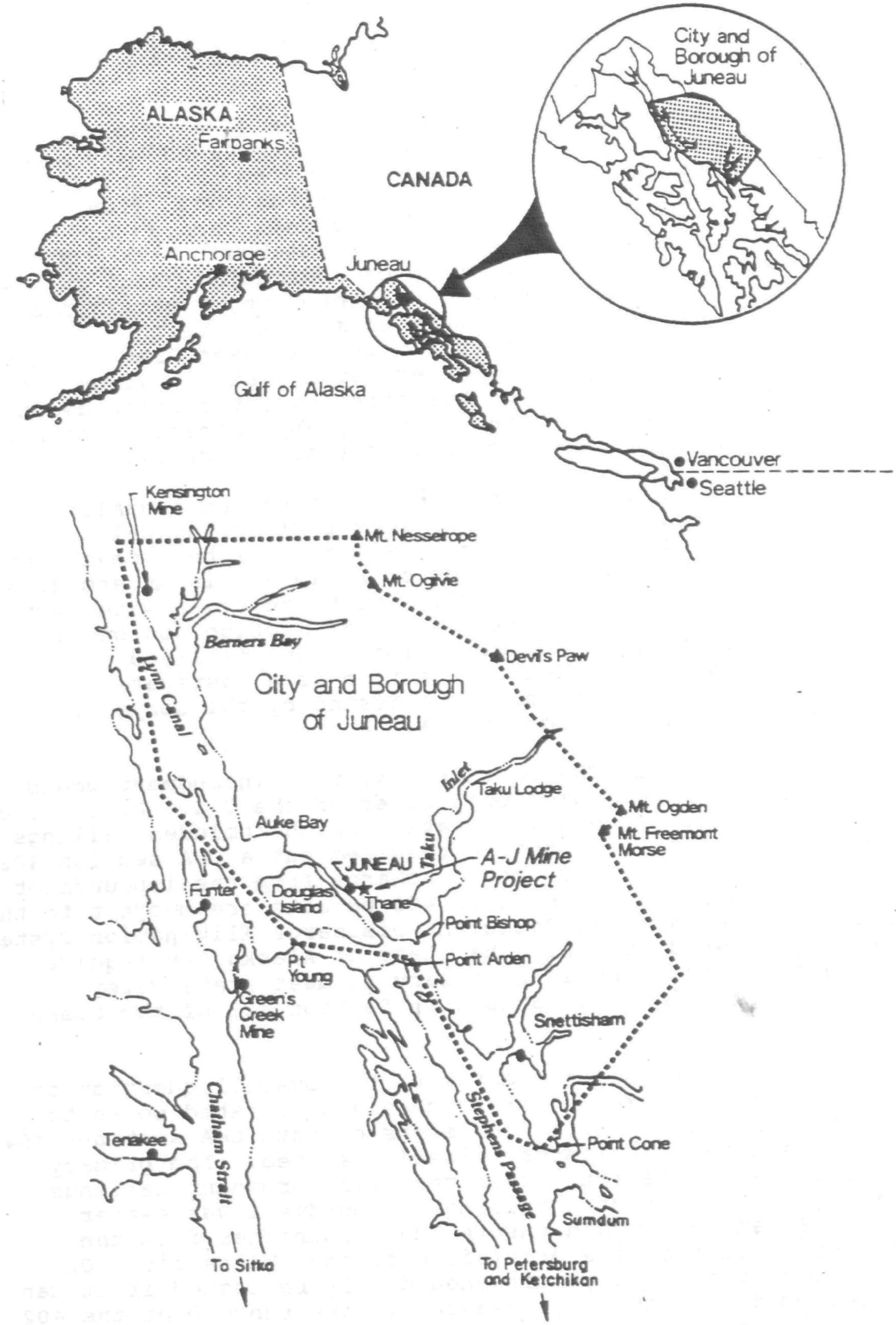


Figure 1

Location Map

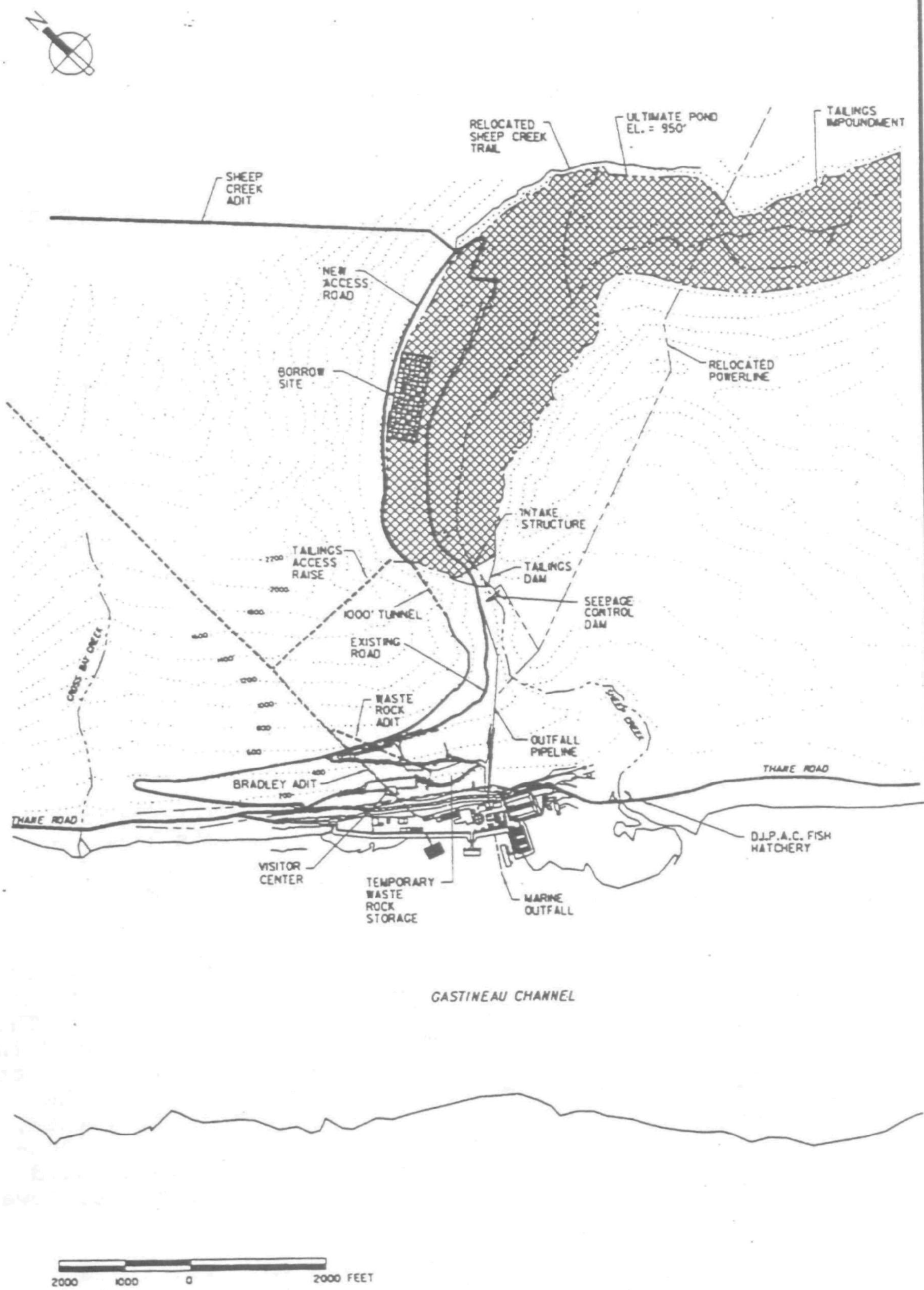


Figure 2
A-J Mine Project
Proposed Tailings Impoundment
Sheep Creek
Source: A-J Mine Project FEIS, BLM, 1992

II. SCOPE OF REPORT

This report addresses impacts from the proposed discharge of process wastes, both solid and liquid, from the AJ mine project. These impacts are analyzed with respect to risks associated with the potential release of contaminants into the aquatic environment and with respect to losses of aquatic habitat productivity (e.g., wetlands) from direct physical disturbance. A fundamental question which this report addresses is whether or not there is a reasonable assurance that the impoundment would in fact provide adequate treatment such that EPA's New Source Performance Standards (NSPS; see 40 CFR 440.104) would be met at the point of discharge to Gastineau Channel and that State of Alaska water quality standards (WQS) would be met in the receiving waters. Chapter VI addresses this question, relying heavily on water quality modeling analyses. Chapter VII addresses overall water quality impacts to Gastineau Channel.

Another fundamental question relates to whether there is reasonable assurance that the proposed method for tailings disposal and long-term maintenance would prevent release of contaminants in harmful quantities. Chapter VIII presents an ecological risk assessment of post-operation conditions and reviews studies of Canadian lakes that have been used for tailings disposal.

In view of the findings of the chapters VI, VII AND VIII, the report addresses potential measures for mitigating or reducing water quality impacts to significantly lower levels in Chapter IX.

A third key question is whether the significant impacts caused by construction and operation of the tailings impoundment can be mitigated to the point that overall impacts on aquatic resources are acceptable. Optional mitigation plans and strategies are reviewed in Chapter X.

This report addresses the AJ mine project design reflected in the CWA §404 permit application and in the Final Environmental Impact Statement prepared by the Bureau of Land Management. The only significant project modification which is not considered in the 404 application or FEIS but which is addressed in this report is a proposal by Echo Bay to construct a diversion dam at the headwaters of Sheep Creek Valley. According to Echo Bay, (EBA, 3/17/1994) this dam would allow diversion of approximately one-third of the flow on Sheep Creek through a pipeline that would float on the surface of the reservoir and then discharge to lower Sheep Creek (see Appendix E).

The information and analyses upon which this report is largely based were developed by Echo Bay and their consultants and the Bureau of Land Management and their third party contractor during the Environmental Impact Statement (EIS) process. Our analysis entailed a review of the Echo Bay submittals and EIS findings for completeness and technical validity, as well as a review of the available relevant technical literature. Much of the analysis is devoted to examining reasonably foreseeable "worst case" situations that could arise during the life of the project. Demonstrating that the project would meet environmental standards during extreme conditions is critical to determining whether there is reasonable assurance that such standards can be met at all times.

Additional information provided by Echo Bay in response to EPA concerns raised during the development of this report has also been relied upon. All of the above information is a part of the administrative record and is available for review at EPA's Region 10 Office in Seattle.

III. POLICY BACKGROUND

On October 2, 1992, EPA's Assistant Administrator for Water wrote a memorandum to Region 10's Water Division Director requesting that the Region provide technical assistance to the Corps of Engineers with respect to pending CWA §404 permits for the AJ and Kensington gold mine projects (see Appendix A). The EPA and the Corps of Engineers agreed, as reflected in this memo, that if the impoundments created by the discharge of fill material contemplated as part of the Kensington and AJ gold mine projects are each permitted under an individual CWA §404 permit for purposes of creating waste treatment systems, they would not be considered waters of the U.S. They would be considered non-jurisdictional waste treatment systems which, by definition (see 40 CFR 122.2 in Appendix A), are not waters of the U.S.

The October 2, 1992, memo reflects an agreement between EPA and the Corps concerning the respective roles of CWA §402 and CWA §404 permits for the mine tailings ponds at the AJ and Kensington projects. A CWA §402 permit, which EPA would issue, is a wastewater discharge permit under EPA's National Pollutant Discharge Elimination System (NPDES) program. It is required for wastewater discharges <u>from</u> the tailings impoundments. A CWA §404 permit, which the Corps would issue, is required to place fill material (the dams) to construct the tailings impoundments. Under the agreement, the CWA §404 permit evaluation must address the effects not only of placing fill for the dam but also the effects of the entire impoundment, including its use as a waste treatment system, on waters of the U.S. Since the purpose of creating the impoundment is to establish a waste treatment system, EPA agreed to provide technical assistance to the Corps with respect to water quality and mitigation as well as overall compliance with the CWA §404(b)(1) Guidelines. This Technical Assistance Report (TAR) is part of EPA's assistance in that regard.

The §404(b)(1) Guidelines (see Appendix A) promulgated by EPA are the substantive regulations that must be satisfied before a §404 permit can be issued. In accordance with current Region 10 policy, the Region will consider the following factors with respect to compliance with the §404(b)(1) Guidelines:

1) The proposed treatment system is determined by the Corps of Engineers and EPA to be the least environmentally damaging practicable alternative for meeting the applicant's project purpose of treating wastewater in order to meet all applicable federal, state and local wastewater discharge limits [see 40 CFR 230.10 (a)];

2) The proposed project does not cause or contribute to any violations of state water quality standards

(excluding discharges <u>into</u> the impoundment) and other provisions of 40 CFR 230.10(b);

3) The project would not cause or contribute to significant degradation of waters of the U.S., including waters located off-site [see 230.10(c)];

4) All appropriate and practicable measures will be taken to minimize impacts to waters of the U.S., [see 40 CFR 230.10(d)], including:

   a) contact between wastewater discharges and waters of the U.S. is minimized to the extent practicable through the application of best management practices (BMP's) such as diverting in-flowing streams and surface runoff around the impoundment, installing impermeable liners as appropriate, etc.;

   b) all practicable measures will be taken to minimize the amount and toxicity of pollutants entering the waste treatment system. No hazardous materials, as defined at 40 CFR 261.3, may be discharged to the treatment system;

   c) any unavoidable impacts to functions and values of waters of the U.S., including wetlands, resulting from construction and operation, are offset to the extent practicable through compensatory mitigation measures;

5) It can be demonstrated that, upon cessation of active use of the treatment system for treatment of the wastewater stream specified in the original CWA §404 permit application, measures will be taken to either:

   a) convert the treatment system to an ecologically productive upland that poses no significant threat to human health or the environment in general or

   b) the treatment system will be converted to a water of the U.S. that meets all Clean Water Act goals for fishable and swimmable waters as well as applicable state water quality standards within a reasonable time frame after cessation of treatment activities;

6) The individual CWA §404 permit for the project can be issued with specific enforceable permit conditions that provide assurances that the above criteria will be met.

IV. PROJECT DESCRIPTION

A. Mine Location and Processing Operations

The proposed AJ Mine project is located in southeast Alaska on the east side of Gastineau Channel about four miles south of downtown Juneau, and directly adjacent to the mouth of Sheep Creek at Thane. The proposed project is located entirely within the boundaries of the City and Borough of Juneau (CBJ). The orebody is situated under Mt. Roberts and the Silverbow Basin, and existing underground workings extend between Gold Creek and Sheep Creek.

The following description of the mine processing operations is summarized from the AJ Mine Project Final Environmental Impact Statement [FEIS, Bureau of Land Management (BLM), May 1992], and the Echo Bay AJ Project Description, Clarification for ACMP Phase I Review, Echo Bay Alaska, Inc., May, 1993. The latter reference identifies alterations to the proposed project stemming from the ongoing project permit reviews subsequent to issuance of the FEIS. The proposed development entails rehabilitation and expansion of the existing underground mine workings, and utilization of a mining method know as stoping, which would create large rectangular caverns. A 14,000 foot adit (the Bradley adit) would be constructed between the ore body and surface facilities proposed to be constructed at a 14.7 acre site along Gastineau Channel, near Thane. The gold ore would be initially processed by crushing, grinding, gravity separation, and flotation in an underground mill off the Bradley adit. Final processing of the gold concentrate would occur at the surface facilities.

Three separate underground ramps are proposed to be constructed off the existing Sheep Creek adit to emerge at the surface upslope of the Sheep Creek adit at the ultimate high water line of each of the three dam raises. These ramps would accommodate the transport of waste rock and tailings to Sheep Creek valley.

The surface facilities would consist of a process plant for leaching, cyanide destruction, and refining, an assay laboratory, liquid petroleum gas storage and generation facilities, electrical substation, offices, warehouse, dock, diesel fuel storage, sedimentation ponds, diked reagent storage, and related facilities. Approximately 12.7 acres of intertidal fill would be required to construct the surface facilities at a location previously used for tailings disposal prior to 1944.

Four streams would be produced by the gold milling process: gravity concentrate, flotation concentrate, flotation tailings, and carbon-in-leach (CIL) circuit tailings. Approximately 90 percent of the tailings, or 20,250 tons daily, would be produced as a by-product of the flotation process. The remaining 10 percent, about 2,250 tons daily, would be the by-product of both flotation and cyanidation. The CIL circuit dissolves gold in a weak solution of sodium cyanide in association with activated carbon granules. Tailings from the CIL circuit would be treated using a sulfur dioxide/air cyanide destruction process. End residual products from the mill processes would include flotation tailings slurry, and CIL circuit tailings slurry.

B. Proposed Tailings Disposal Plan

Tailings slurry would be pumped from the flotation thickener and cyanide destruction facility to the Sheep Creek valley for disposal in the proposed tailings impoundment.

The tailings impoundment would entail construction of a concrete dam across the Sheep Creek narrows (see Figure 2). The initial dam crest height would be 205 feet, at an elevation of 805 feet. Two additional dam raises (Stages 2 and 3) would be constructed during tailings disposal operations so that the ultimate crest height of the dam would be 332 feet, at an elevation of 932 feet. When mine operations start, the water depth would be from 80 to 100 feet at the dam. The impoundment would be designed to store 100 million tons of tailings with a minimum of 20 feet and a maximum of 30 feet of water maintained above the tailings (see Table 1).

Table 1

SHEEP CREEK
TAILINGS FACILITY SUMMARY

FEATURE	SIZE
Dam foundation elevation	600 feet
Dam final elevation	932 feet
Dam height:	
first stage	205 feet (elevation 805)
second stage	276 feet (elevation 876)
third stage	332 feet (elevation 932)
Downstream slope	0.75 - 1.00
Capacity	100 million tons (150+ million potential)
Final crest length	900 foot radius
Impoundment surface area	420 acres
Hydroelectric potential	4.9 Megawatts
Embankment volume	500,000 cubic yards
Estimated Construction cost	$48.4 million

Source: FEIS (BLM, 1992)

It would take from six to nine months, or longer, for the reservoir water to reach the initial dam outlet elevation. After that, the water would be kept near that level, within a range of plus or minus 10 feet. Minimum freeboard for the first two dam heights would be for a 20-year/24-hour storm event, and would be 10 feet for Stage 1 and 7 feet for Stage 2. At final operation, a minimum freeboard of 3 feet would be maintained (during the 10-year/24-hour storm event).

Plans for installing a hydro-electric generator, utilizing a 48 inch conduit from the dam impoundment, have been suspended. If used for hydroelectric power generation, the regulated impoundment storage would range from plus to minus 10 feet of minimum freeboard, but would approach the minimum near the end of mining operations. Floods greater than the 10-year/24-hour event would be discharged partially through the outlet and partially over the emergency overflow spillway. Excessive spillway flow energies would be structurally dissipated.

During dam construction, erosion, and sedimentation control measures would be used to maintain ambient Sheep Creek water quality. A sedimentation basin, along with other control facilities, would be constructed in the dam site area. Laydown areas would be constructed, as would a concrete batch plant for dam construction. Aggregate rock would be obtained from crushed waste rock or native talus slopes.

During operations, mill tailings would be deposited in the impoundment by a floating pipeline connected to either of two barges equipped with "elephant trunk" disposal lines. This type of line would place tailings at the bottom of the water column in order to minimize turbidity. The barges would be movable to allow placement of tailings throughout the impoundment. Provisions would be made to enable continued tailings disposal during the winter ice conditions.

A seepage return dam would be constructed about 150 feet downstream from the base of the main dam. The floor of the seepage return basin would be a slab of reinforced concrete which would be an integral part of the spillway of the primary dam. The seepage return dam would collect any water that seeps through the main dam or along its foundation. No tailings would be discharged within the seepage return basin. Monitoring wells would be constructed downstream of the seepage return dam to indicate if any seepage is occurring downstream of the seepage return dam.

During dam construction, Sheep Creek would be diverted through a pipe around the construction site. During operations, Sheep Creek stream flows would enter the impoundment, with the exception of the upper Sheep Creek flow that would be diverted via a proposed diversion dam. Diverted water, estimated at one-third the flow of Sheep Creek as measured at the narrows, would flow through a floating pipeline that would then discharge directly to lower Sheep Creek. This diversion has been proposed to address concerns of the Alaska Department of Fish and Game regarding minimum fish flows in lower Sheep Creek where the DIPAC fish hatchery is located. The remainder of the Sheep Creek flow, which would constitute the net precipitation over the impoundment's catchment area, would be discharged to Gastineau Channel along with mine drainage via a pipeline and diffuser placed at depth. The discharge would need to meet effluent limits imposed in the required NPDES wastewater discharge permit.

If the hydroelectric facility was eventually constructed, flows would terminate at a 49 megawatt (MW) hydroelectric generating facility. Excess penstock flows (above the 100 cubic feet per second hydraulic capacity of the turbine) would be routed to a flow bypass facility with energy dissipators. The total combined flow of excess tailings impoundment water would then be discharged via a tailrace and diffuser to the marine waters of Gastineau Channel. This discharge would also be subject to issuance of the National Pollutant Discharge Elimination System (NPDES) permit for the facility by the Environmental Protection Agency (EPA).

C. Other Tailings Disposal Options Considered

Tailings disposal alternatives were addressed in the FEIS as components of the overall project alternatives. The alternatives were compared and evaluated on the basis of technical or engineering feasibility, environmental impact, legality, and economics. Alternatives related to the disposal of tailings which were initially considered but, based on the above factors, determined to be unreasonable for the purposes of more detailed evaluations in the FEIS, included: alternative ore extraction and cyanide destruction methods, transporting concentrate to an existing mill, marine disposal of tailings, backfilling of mine excavations with tailings, separate treatment and disposal of cyanide-treated tailings, dry tailings disposal, tailings disposal at multiple upland sites, tailings disposal in the Carlson Creek valley, or in the Grindstone Creek valley, and a Sheep Creek hydroelectric project with another tailings impoundment site.

Alternative tailings disposal impoundment locations, in addition to the Sheep Creek location, which were evaluated in the FEIS included Powerline Gulch (also referred to as Icy Gulch), Sheep Fork Carlson Creek, and Rhine Creek. The no-action alternative was evaluated as no project, baseline conditions. The preferred alternative identified by the BLM in the draft EIS and FEIS, on the basis of the evaluation criteria, was the applicant's proposed action, which includes construction of a tailings impoundment in Sheep Creek valley. Additional analysis of tailings disposal options is included in Chapter IX.

V. AFFECTED ENVIRONMENT

A. Introduction

The affected environment of the AJ Mine project encompasses the greater Juneau area. This report, however, focuses on the impacts of the mine associated with the discharge of process wastewater. This waste stream includes the tailings slurry discharge to the proposed Sheep Creek tailings impoundment and the effluent discharge from the impoundment which would be comingled with mine drainage water. Therefore the affected environment considered here is Sheep Creek valley and Gastineau Channel. A more detailed description of the affected environment is contained in the Final Environmental Impact Statement (FEIS; BLM, 1992).

The Sheep Creek tailings disposal option of the AJ gold mine would fill 420 acres of Sheep Creek valley. Waters that would be eliminated in association with the proposed fill are 2.5 miles of Sheep Creek above the tailings dam and 8.1 acres of associated wetlands, for a total of 20.1 acres of aquatic habitat. The flow of water in 1.1 additional miles of Sheep Creek downstream of the impoundment would be significantly reduced. Marine fill would consist of 14.7 acres extending from the shoreline of Gastineau Channel to the -30 foot contour.

The climate of the Sheep Creek area is that of the cool northern temperate rainforest. The nearest source of climatic information is the weather station in downtown Juneau. Annual precipitation is 90 inches at downtown Juneau. Normal monthly temperatures range from 28 degrees Fahrenheit in January to 57 degrees Fahrenheit in July (BLM, 1992). Much of the precipitation falls as snow. The weather station averages 80 inches of snow annually (BLM, 1992). Average monthly flows in Sheep Creek, Gold Creek and Lawson Creek are shown in Table 2.

Human uses of lands in the area near the Sheep Creek watershed and Gastineau Channel include the urban and industrial centers of nearby Juneau. Residential dwellings dot the shore of Gastineau Channel. The Thane road parallels the shore of Gastineau Channel and crosses Sheep Creek at its mouth. A fish hatchery is located at the mouth of Sheep Creek. The mouth of Sheep Creek is an important recreational and personal use fishing site for the population of Juneau, in part due to the hatchery. The AJ mine was a major land user during operation until 1944, when it was closed due to conflicting national priorities during war time. The "rock dump," on which is located the Juneau sewage treatment plant and other industrial activities, is a product of the AJ Mine. A portion of Sheep Creek valley is now used as a staging area for exploration of the AJ ore body. Sheep Creek valley is also used by local residents for nature study, hiking and for its aesthetic beauty.

Table 2: Average Monthly Flow (cubic feet/sec)

Month	Gold Creek 1918-1982 Drainage Area: 9.76 mi^2	Sheep Creek 1919-1973 Drainage Area: 4.30 mi^2	Lawson Creek 1970-1971 Drainage Area: 2.98 mi^2
January	22.6	11.8	3.8
February	16.9	6.1	7.2
March	12.3	4.7	6.0
April	29.0	12.8	13.7
May	124.4	69.3	48.7
June	225.0	99.9	46.2
July	222.6	82.4	26.8
August	190.6	72.3	16.4
September	178.9	71.1	25.0
October	161.2	71.7	20.8
November	77.5	54.7	18.0
December	35.8	16.4	5.2

Source: Comprehensive Report (EBA, 1992)

Mammals of the immediate area include black bear, mountain goat, Sitka black-tailed deer, beaver, marten, river otter, mink, ermine, and other mustelids, lynx, red fox, hoary marmot, porcupine and other small mammals (U.S. Fish and Wildlife Service, 1991). There are 131 species of birds in the project area. Among these are bald and golden eagles, more than a dozen species of neotropical migratory song birds and several species of shore birds (Wilson and Comet, 1991).

Migratory peregrine falcons are the only terrestrial species listed under the Endangered Species Act, known to occur in Sheep CreekvValley, though a rare alpine sedge (*Carex plactocarpus*) is known from nearby Mount Roberts. The marbled murrlet is a sea bird thought to occur in Sheep Creek valley (its presence has not been documented). The U.S. Fish and Wildlife Service was petitioned to list the marbled murrlet but determined that it was premature to list the bird in Alaska. The ranges of two marine mammals include Gastineau Channel; Humpback whales, listed as endangered, and the Stellars sea lion, listed as threatened. It should be noted that the Southeast Alaska population of Stellar's sea lion is not considered at risk (BLM, 1992).

B. Sheep Creek Valley

Sheep Creek drains 5.9 square miles of mountainous terrain adjacent to and south of the underground workings of the AJ mine. The valley can be conceptually divided into two sections. The upper portion is a "U" shaped valley approximately 2.3 miles long. Valley slope is 2%. The valley floor has an area of about 540 acres. At the downstream end of this section the valley narrows and drops 600 feet to sea level over a distance of 1.1 miles.

Valley bottom vegetation is a mosaic of riparian and wetland vegetation types, all the result of the pattern in which water flows through the valley. This mosaic provides a diverse habitat that is unique for the Juneau area. Vegetation consists of coniferous forest, deciduous forest, shrub wetland, tall shrub, wet meadow and upland meadow. The deciduous forest is composed primarily of mature cottonwood trees which are dependent on the shallow ground water in the valley bottom. Such stands are uncommon in the greater Juneau area. Rough estimates suggest that Sheep Creek valley contains 25% of the total area of this type of plant community between Taku Inlet and Berners Bay (BLM, 1992). The FEIS describes Sheep Creek valley as having more species diversity than any other site accessible on the Juneau road system.

The richness in vegetative habitats supports an unusually diverse avifauna. Song bird populations have been found to be "locally diverse and abundant" (Wilson and Comet, 1991). Sheep Creek valley had five times the song bird nest density and over

323% more successful nests than a nearby site with similar vegetation (Comet and Wilson, 1994). Habitats on the valley walls adjacent to the valley floor have notably fewer songbirds (Wilson and Comet, 1993). Many of the song bird species found in Sheep Creek valley are of special interest. Five species whose abundance is thought to be in decline in Alaska breed in the valley. These are the fox sparrow, orange crowned warbler, blackpoll warbler, American robin, and varied thrush (Wilson and Comet, 1991). Eight other species found in the valley may be increasing in abundance. Fifteen of the 42 species of birds documented during formal censusing are of interest to the national program on neotropical birds (Wilson and Comet, 1991). One additional bird, the marbled murrlet, whose use of the valley has not been documented, is of concern because of population declines along the west coast of North America.

Aquatic resources of the upper portion of Sheep Creek include a local population of Dolly Vardon char. This population is isolated from immigration from other char populations by barriers to fish migration in the lower reach of Sheep Creek.

The lower 1.1 miles of the stream flows through a sequence of cascades and falls to tidewater at Gastineau Channel 3.5 miles south of downtown Juneau. A Douglas Island Pink and Chum (DIPAC) hatchery is located at the mouth of Sheep Creek. The hatchery produces pink and chum salmon for the recreation and commercial fisheries of the area. Before merging with Gastineau Channel, Sheep Creek crosses an alluvial fan formed by sediments carried by stream action from Sheep Creek valley above. Pink and chum salmon spawn in the Sheep Creek upstream as far as an impassible waterfall 440 feet above tidewater (BLM, 1992).

C. Gastineau Channel

Gastineau Channel is a north-south oriented channel separating Douglas Island from the mainland. The shoreline for approximately 10 miles between Stephens Passage and Juneau is largely steep sided and rocky. Sediments from Mendenhall Glacier have filled much of the channel to the north of downtown Juneau, so that a tidal channel exists in this area.

According to studies of nearby Auk Bay, much of the productivity of Gastineau Channel is likely tied to phytoplankton blooms that occur as environmental factors, such as day length, water temperature and nutrient levels, become more favorable in the spring. These primary producers support zooplankton which in turn support fisheries of commercial importance. The zooplankton also includes larval fish which may also be of commercial and ecological significance (BLM, 1992). At least 24 larval fish species have been collected in Auk Bay (BLM, 1992). Gastineau Channel is likely to contain similar numbers.

Planktonic productivity typically exceeds grazing rates of zooplankton, allowing much of the fixed carbon to settle to the bottom, feeding the benthic community (BLM, 1992). Forty species of demersal fish, shellfish and other invertebrates have been reported from Gastineau Channel. Included among these are commercially important crab species. The commercial crab fishery was closed in 1978 but a popular personal use fishery continues (BLM, 1992).

Intertidal and subtidal flora and fauna of Gastineau Channel is typical of protected shorelines of southeast Alaska. A mussel-barnacle-rockweed (*Mytilus-Balanus-Fucus*) assemblage is found on upper to mid-intertidal zones. Near the Sheep Creek delta the substrate can be covered by up to 100% mussels. Green sea urchin (*Strongylocentrotus drobachiensis*), and several sea star species were observed in this area during surveys associated with this mine project (BLM, 1992).

Much of the shore adjacent to the City of Juneau is composed of tailings deposited during previous mining of the AJ Mine. Surveys of these areas found reduced numbers of epifaunal and infaunal organisms compared to locations that were not composed of tailings (BLM, 1992).

VI. EVALUATION OF PROJECTED TAILINGS POND PERFORMANCE

A. Introduction

Echo Bay has proposed the tailings impoundment as the sole method of wastewater treatment for the AJ project. The impoundment would be operated to allow for the settling of solids that are delivered to the pond in the tailings slurry and surface runoff. At the AJ project, the tailings slurry would be comprised of approximately 55% solids by weight. This equates to 938,000 mg/l of total suspended solids (TSS) when the slurry is introduced to the tailings pond. EPA's New Source Performance Standards (NSPS) that apply to gold mining projects like AJ limit TSS in the discharge from the impoundment to a monthly average of 20 mg/l and a daily maximum of 30 mg/l. Therefore, the tailings pond would need to be capable of removing over 99.99% of the solids.

NSPS are minimum, technology-based effluent limits that apply to a single industrial category of "new source" wastewater dischargers. These limits must be met at the point of discharge (end-of-pipe). More stringent effluent limits may be imposed on a case-by-case basis if necessary to meet WQS. In theory, by meeting the TSS limits, NSPS effluent limits for metals would be met as well, since metals tend to be adsorbed on to the surface of suspended particles. However, this may not be sufficient metals removal for meeting water quality standards (WQS). Some natural degradation of cyanide may occur during summer months but for this analysis it is assumed that all cyanide destruction would take place prior to discharge to the impoundment.

Predictions of the origin and fate of materials entering the impoundment are critical for determining compliance with NPDES requirements and the risk of long term contamination. As shown in Figure 3, results of the impoundment analysis are subsequently used in evaluations of end-of-pipe performance standards, water quality impacts to Gastineau Channel, and the risk of long term contamination in the reservoir following cessation of mining activity.

The following major features of the impoundment system would affect its performance in removing pollutants:

- Quality and quantity of tailings influent (slurry) from the froth flotation and cyanide leach circuits;

- Quality and quantity of runoff from the undiverted catchment area;

- Settling efficiency of the tailings impoundment under low flow, average flow and high flow conditions;

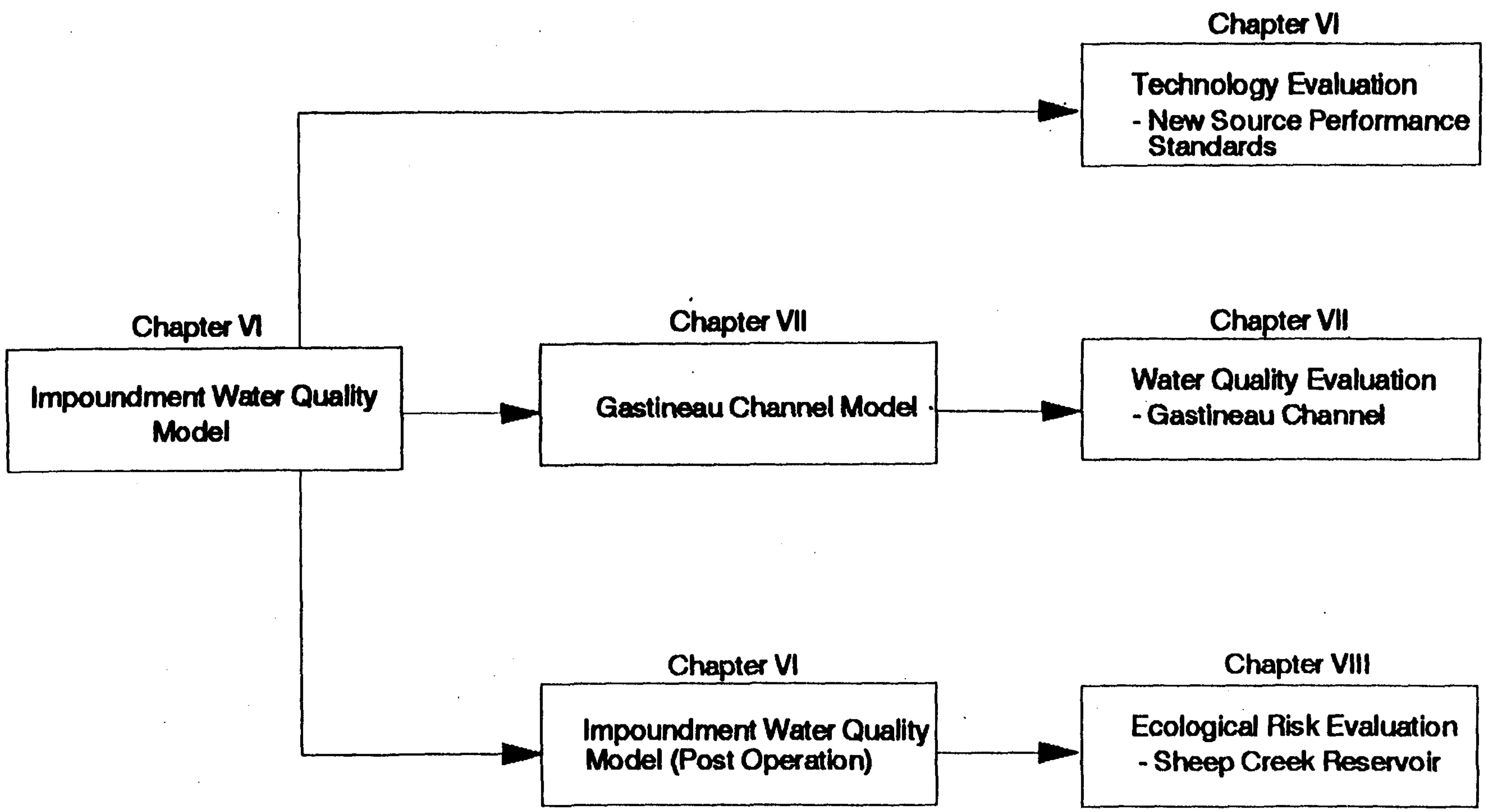


Figure 3 : Modeling and Assessment Linkages

- Size of the tailings pond and diversion structures in relation to anticipated high flows;

- Vulnerability of the impoundment to natural hazards such as avalanches, landslides and earthquakes.

There are two basic approaches for projections of impoundment performance. The first is to consider all of the inputs (influent) and outputs (effluent) to and from the tailings pond, respectively, and then, using water quality modeling, analyze the pond's effectiveness in terms of removing pollutants and thus estimate the projected composition of the effluent. This approach, using a simplistic model, was used during the EIS process to project the quality of the effluent. Our review entailed more sophisticated water quality modeling using the WASP4 and CE-Qual-W2 water quality models.

The second approach is to compare the AJ project to other mining operations that employ the same or similar technology. The reliability of this empirical approach hinges on the direct comparability between the AJ project and the other mines that are examined.

The next two sections of this chapter present brief overviews of the NPDES requirements for this project and the FEIS evaluation of the impoundment. The subsequent sections present EPA's analysis and projections of impoundment performance.

B. NPDES Requirements

An NPDES permit is required for the point source discharge from the tailings impoundment. Two sets of requirements must be met by the discharge from the proposed AJ impoundment. First, the discharge must meet the national New Source Performance Standards at the point of discharge from the impoundment (end-of-pipe). These limitations on metals, cyanide and suspended solids apply to all mining facilities using the froth flotation and cyanidation processes proposed for use at the AJ project (40 CFR 440.104(a) Subpart J). A comparison of the predicted discharge quality with these end-of-pipe limitations is provided in section I of this chapter.

The second set of NPDES requirements are the Alaska water quality standards for Gastineau Channel. Discharges from the AJ project must meet numeric water quality criteria for the protection of beneficial uses in Gastineau Channel. Specifically, the criteria are established to protect against toxicity to aquatic life from exposure to pollutants in the water column and effects to human health from consumption of contaminated fish and shellfish. These criteria must be met either at the end-of-pipe or after dilution in a mixing zone designated by the state of Alaska. An evaluation of the predicted discharge quality with respect to the water quality standards for Gastineau Channel is provided in Chapter VII of this report.

C. Review of the FEIS Water Quality Model

Water quality analyses of the proposed tailings pond, as described in the FEIS and errata to the FEIS (Tileston, 1992), provided a limited examination of the metals concentrations in the tailings pond using estimates from the literature and results of laboratory tests. Based on these initial estimates, the FEIS for this project predicted that "Suspended solids (TSS) content is likely to be at least 70 mg/l near the discharge, and may be as much as 220 mg/l throughout the entire impoundment." These values exceed the levels required by EPA's NSPS for mine tailings ponds. However, the methodology used in the FEIS to estimate levels of TSS and important chemical constituents in the proposed tailings pond was based on static, rather than dynamic conditions. In addition, the methods used in the FEIS to estimate the recovery time of the tailings pond after mining ceases gave results substantially different from recovery times based on average residence time of the tailings pond.

Due to the limited scope of the analysis performed in the FEIS, and results indicating that NSPS effluent limits for TSS would not be met, EPA Region 10 concluded that a more thorough analysis of water quality should be performed for purposes of the TAR. The approach used in the analysis described in this report represents an effort to clarify issues of mass balance as well as to expand the scope of the analysis to include issues of time dependence; sediment accumulation rates; inputs of kinetic energy from discharges, and winds. Characterizing the influent to the pond, discussed in the following section, is a critical first step in the analysis.

D. Characteristics of Process Influent to the Tailings Pond.

Water quality predictions for the tailings pond and discharge are based in part on estimates of the chemical composition of process materials that flow into the pond from the mill. A reevaluation of the approach, sources of data, and results of influent estimates provided in the FEIS and subsequent FEIS errata tables (Tileston, 1992) is provided here, along with improved estimates used for subsequent water quality modeling.

1. FEIS Approach

Estimates in the FEIS for the chemical composition of tailings pond influent from Echo Bay minerals processing were based on a variety of methods involving site specific field and lab data as well as assumed operating conditions. The FEIS approach uses what are termed "probable threshold" and "upper reasonable threshold" values for projected quality of process effluent (BLM, 1992, Table 4-8) which becomes the influent to the tailings pond.

Probable threshold is defined in the FEIS (Table 4-8; BLM, 1992) as the average water concentration resulting from dissimilar lab tests, including the following:

1. The sum of flow-weighted values from separate decant tests of flotation and cyanide-destruct (CIL) tailings. The FEIS notes that the values used from the decant tests are the maximum measured from four samples of water collected during a 99-day settling period. The flow weighting used in the FEIS assumes a proportion of about 23% of CIL process water, based on the water balance shown in the FEIS (Table 4-8 and Figure 2-17; BLM, 1992).

2. Maximum values from a decant test of combined flotation and CIL tailings. The mix used in the combined tailings contained 10% wet-weight (w/w) CIL tails. This test also provided samples at four settling times spanning 99 days.

3. Maximum values of weekly samples drawn during column leach tests conducted on combined flotation and CIL tailings. The leachant was Sheep Creek water. The proportion of CIL tailings was 10% w/w. The column tests spanned 105-140 days.

4. The FEIS limited the probable threshold values for some constituents based on the results of either a total digestion or slurry extract of combined tailings.

Upper reasonable threshold is defined in the FEIS (Table 4-8) as the maximum rather than the average of all of the values noted above, but not exceeding the results from the total digestion or slurry extract of tailings. Table 3 lists a summary of the types of lab tests that provide data useful for estimating contaminant concentrations in the tailings influent to the pond, as well as the risk of long-term contamination addressed in Section VIII-N. Table 4 lists information on the source and timing of lab tests, and Tables B-1 and B-2 in Appendix B provide a compilation of decant, column, digest and slurry results for tailings.

2. Validity of the FEIS Approach

Examination of available lab data for purposes of this report indicates that the FEIS approach probably yields neither most probable nor upper bound values for some of the more important constituents in the tailings pond influent. An initial concern was that FEIS values presented for digested tailings

Table 3. Types of Test Data Available for the A-J Project to Estimate the Composition of Tailings, Tailings Pore Water, Tailings Leachate, and Acid Generation Potential

Concentrations in Tailings Water

Decant Test
- Concentration of metals, cyanide, and other nonmetals in supernatant measured at 2-hour, 7-day, 30-day, and 99-day intervals.
- Material tested
  - Flotation Tailings
  - CIL Tailings
  - Physically Combined Tailings (10% w/w CIL tailings)

Concentrations in Tailings Solids

Digestion Analysis
- Concentration of metals in solid fraction of tailings
- Material tested
  - Physically Combined Tailings (10% w/w CIL tailings)

Slurry Extraction Analysis
- Concentration of cyanide and non-metals in extract of slurry of tailings and water (Note-conditions of this test are unknown)
- Material tested
  - Physically Combined Tailings (10% w/w CIL tailings)

Ore analyses (indirect estimate of tailings assuming similarity)
- Concentration of metals and sulfur in solid fraction of tailings
- Material tested
  - Bulk sample of ore
  - Core samples of ore
  - Grab samples of ore

Concentrations in Tailings Leachate

Column Leach Tests
- Concentration of metals, cyanide, and other non-metals at periodic intervals over 12-20 weeks.
- Material tested
  - Physically Combined Tailings (10% CIL tailings)

Batch Leach Tests
- EPTox (Extraction Procedure Toxicity) - Concentration of metals in 48-hour tumbling container. Previously used for comparison with criteria for hazardous waste designation
- TCLP (Toxicity Criteria Leaching Procedure) - Concentration of metals in 48-hour tumbling container. Used for comparison with criteria for hazardous waste designation
- Material tested
  - Physically Combined Tailings (10% CIL tailings)

Acid Generation Potential Tests

Acid-Base Accounting (ABA) Test
- Concentration of oxidizing components (sulfur) and neutralizing components (titratable reactants)
- Material tested
  - Waste rock
  - Ore
  - Flotation Tailings
  - CIL Tailings
  - Physically Combined Tailings (10% w/w CIL tailings)

BC Confirmation Column test
- pH trend in acidified, bacteria-inoculated sample
- Material tested
  - Flotation Tailings
  - CIL Tailings

Table 4. Source and timing of compositional data for ore and processed materials for the A-J mine project.

Material	Source	Sampled	Milled	Metals Analysis	Sulfur Analysis	ABA Analysis	Decant Test	Confirm Analysis	CN Analysis	Extract Analysis	Column Tests	EPTOX Analysis	TCLP Analysis	Reference
Rock - 18 samples	NOB-SOB GH-other	/87		/88?	missing									Bergstrom (1994a)
Waste rock stockpile - 8 samples	unknown	unknown		5/89	5/89	5/89								Ott (1989), Bergstrom (1994b), EBA (1994a)
Tailings - Flot	NOB-SOB	unknown	unknown	/88?	/88?		/88?							Lakefield Research (1988)
Ore - bulk sample	unknown	unknown	7/89?											Ott (1989)
Concentrate and Tailings-Flot	unknown	unknown	5/89?		5/89?									Lakefield Research (1989a)
Mill Feed	NOB-SOB	9/88-5/89			/89?									Lakefield Research (1989b)
Tailings - combo (5%cnd)	unknown	unknown									6/89			Ott (1989), Bergstrom (1994b), EBA (1994a)
Ore - bulk sample (144t)	NOB ("AJ", 4387SX)	9/88	7/90	7/90?	7/90?									Lakefield Research (1990) and Andrews (1991)
Tailings - Flot/CN/CND	NOB ("AJ", 4387SX)	9/88				9/90								Lakefield Research (1990), EBA (1994a)
Tailings - Flot/CND/combo(12%)	NOB ("AJ", 4387SX)	9/88					7/90-10/90							Lakefield Research (1990), Andrews (1991), and Bergstrom (1994b)
Tailings decant residue-combo	NOB ("AJ", 4387SX)	9/88				11/90								Bergstrom (1994b)
Tailings decant residue?-CN/CND	NOB ("AJ", 4387SX)	9/88						11/90						Bergstrom (1994b)
Tailings decant residue	NOB ("AJ", 4387SX)	9/88									3/91-7/91			Andrews (1991); Jokela (1991), and FEIS (1992)
Tailings decant residue	NOB ("AJ", 4387SX)	9/88		91?					missing	91?				Andrews (1991) and FEIS (1992)
Tailings column residue?	NOB ("AJ", 4387SX)	9/88										8/91-10/91	8/91	Bergstrom (1993)
Ore - 6 channel samples	NOB (4387SX)	9/88?		8/91	8/91									Frederickson (1992), EBA (1994a)
Waste rock - 24 samples	NOB-road	3/94			4/94	4/94								EBA (1994a)
Waste rock - 53 samples	NOB-SOB	5/94?			5/94	5/94								EBA (1994a)
Ore - 38 samples	NOB-SOB	3/94			3/94	3/94								EBA (1994a)
Ore - 101 crosscut samples	NOB-SOB	5/94			5/94	5/94								EBA (1994a)
Ore - 102 core samples	NOB-SOB	?			5/94	5/94								EBA (1994a)
Ore - 83 crosscut samples	NOB-SOB	?		5/94										EBA (1994b)

Footnotes:
NOB - north ore body; SOB - south ore body.
Some additional decant data for unmixed tails were obtained beyond the 99 days reported in the FEIS.

seemed unrealistically low. The impact was that these values were subsequently used as an upper bound for some of the FEIS-predicted constituents in the process effluent. After completion of the FEIS, an updated Table 8 which adjusted the digested tailings values upward was provided by the FEIS contractor (Tileston, 1992). The updated values are listed here in Table 5 under the heading FEIS.

Comparison of the updated FEIS Table 4-8 values with laboratory reports upon which they were based indicates that for several constituents the maximum decant values measured in the lab tests were not used for effluent quality projections in all cases. The discrepancy appears to result, at least in part, from use of total analytical values in the FEIS table, even though the lab reports show that in some cases total recoverable or dissolved values are greater. The impact of using all available decant data in a manner consistent with the approach described in the FEIS would be to increase somewhat the projected tailings concentrations for some constituents. Table 5 lists these revisions under the heading FEIS REV.

Other potential problems with the FEIS approach to effluent quality projections include incomplete information on the representativeness and variability of materials being tested, and the averaging of data from dissimilar tests without regard to the chemical reactions being tested.

With respect to representativeness of materials, Table 4 provides a list of the source and times of various tests conducted for the AJ project. Tailings tests used for the FEIS effluent projections are based on a single bulk sample collected from the 4387SX crosscut in the north ore body in 1988 (EBA, 1994a). In a recent report, the applicant contends that the single bulk ore sample is representative of average concentrations of material to be mined. A single tailings sample therefore may provide useful data for evaluating average conditions but no information on variability to be expected if the mill feed varies during production. Consequently, projections based on any of the various tests of the bulk sample products cannot necessarily be representative of worst or "upper reasonable threshold" conditions, because the sample is not necessarily worst case. Accordingly, the "upper reasonable threshold" conditions as listed in the FEIS should not be considered worst-case effluent quality.

The proportion of CIL tailings varies among the tests used in the FEIS projections. The only estimates based on a 23% mix of CIL tailings water as used in the FEIS water balance are the mathematically derived flow-weighted combinations of decant data from separately settled flotation and CIL tailings. Since much of the higher contaminant load is from the CIL tailings,

Table 5. Projections of tailings composition - aqueous and solid fractions.

	FEIS	FEIS REV		FEIS	FEIS REV		FEIS	FEIS REV		FEIS	FEIS		TAR	TAR	TAR
	Flow-wt combination			Probable Threshold			Upper Reasonable Threshold			Digest	Slurry Extract		aqueous	aqueous	solid
										mg/kg	mg/kg		basis	mg/L	mg/kg
Flow	3760														
pH	8.2			7.6			8.2								
TDS	3200			1500			3200						fwdec	3200	
TSS	14200			440			1400								
Turbidity	107			74			250								
Hardness	672			480			670						fwdec	670	
Alkalinity	209			160			210						fwdec	210	
Sulfate	4000			370	1517		4000				370		codec	1700	370
Ammonia		17.6		7.3	8.4		7.3	18			17		fwdec	18	17
Nitrate, N		20.4			8.19			20					fwdec	20	
Nitrite, N		0.69			0.28			0.69					fwdec	0.69	
TKN		80.1			41.9			80					fwdec	80	
Arsenic	0.002	0.005		0.0042	0.0047		0.014			9.78			l	0.014	9.8
Cadmium	0.008	0.01		0.0069	0.0091		0.012	0.024		11			fwdec	0.01	11
Chromium	0.001	0.004		0.007	0.0075		0.02			35.4			l	0.02	35
Copper	1.36			0.66	0.67		1.4			262			fwdec	1.3	260
Iron	0.137	0.188		3			14			45600			l	0.85	46000
Lead	0.05			0.063			0.16			197			l	0.02	200
Manganese	0.523	0.533		0.8	0.81		2.7			1560			l	2.7	1600
Mercury	0.005	0.005		0.0011			0.0054	0.005		0.274			fwdec	0.005	0.27
Nickel	0.011			0.014	0.021		0.022	0.03		25.7			l	0.03	26
Selenium	0.013	0.017		0.0056	0.0065		0.013	0.017		1.22			fwdec	0.016	1
Silver	0.002	0.002		0.01			0.05			3.06			l	0.05	3
Zinc	0.055			0.17	0.21		0.47	0.69		528			codec	0.087	530
Xanthate	0.927			0.6	0.66		0.6	0.93		0.6			fwdec	0.93	
Free Cyanide	0.774	0.781		0.47			1.1						codec	1.1	
Total Cyanide	1.29	1.75		1.1	1.2		2.5				42.4		l	2.5	
WAD Cyanide	0.82			0.19			0.82						cnf	1.1	
Cyanate	73.3			26			73						fwdec	73	
Thiocyanate	393			0.6			390				0.6		fwdec	390	

Footnotes:

FEIS - values from FEIS method, as listed in Corrected Table 4-8, letter from JMM to David Dorris, BLM, September 18, 1992 (Tileston, 1992).

FEIS REV - Values from FEIS method derived from data tables in FEIS supporting documents.

TAR - values recommended for use in the Technical Analysis Report.

fwdec - flow-weighted decant; codec - combined decant; l - leach; cnf - free cyanide

shading = less-than value

increasing the CIL proportion in any of the tests of combined tailings would probably increase the concentrations of contaminants in the tailings.

3. Alternative Approach

The FEIS approach considered dissolved and suspended concentrations together as a total process effluent concentration. An alternative approach used here is to consider the tailings pond influent from the mill as an aqueous phase and a solid phase, each with its own separate constituent concentrations. Table 5 lists these values under the heading TAR. Appendix B contains laboratory data listed in Tables B-1 and B-2 from which values in Table 5 are derived. Since these data are from the same limited tests as used for the FEIS, the limitations, as noted above, on being able to evaluate tailings variablity or worse-than-average tailings concentrations apply. Effluent quality based on tailings produced from milling an average bulk ore sample would not accurately project the total range of contaminant concentration to be expected. To offset to some degree the inherent underestimation of contaminant concentrations that arises when using average samples, the use of lab data described below emphasizies the higher values.

EPA's application of alternative influent values for this analysis relies on the greater of either decant or column leach data for the aqueous phase. Where decant data are used, the TAR values are from dissolved rather than total data. Mathematically flow-weighted decant values are used where greater because they represent the actual CIL proportions projected by the FEIS water balance. Comparison of flow-weighted values at a 23% mix of CIL water with the physically combined values of a mix of 10% CIL tailings indicates that the two sets of data would have similar initial decant results for many of the major parameters if the CIL proportions were the same.

Digest and slurry extract data from the FEIS are used for the solid phases in tailings. As with aqueous phase data, single digest and extract analyses of tailings as reported in the FEIS would not represent extreme values because of the reported (EBA, 1994b) average nature of the bulk sample.

E. Predictions of Effluent Quality: WASP4 Water Quality Model

1. Introduction

As described in the FEIS (BLM, 1992), the preferred alternative entails converting most of the Sheep Creek valley into a tailings disposal impoundment. While the proposed facility is in operation, this impoundment would function as a wastewater treatment facility for the tailings slurry discharge. The water from the impoundment would be combined with excess flow from the Gold Creek mine drainage and discharged through a diffuser into Gastineau Channel. This would continue after the mine has ceased operation unless natural processes are sufficient to dilute the water in the impoundment to levels which meet criteria established by the State of Alaska's water quality standards (WQS). This section of the report describes the results of using mathematical models of water quality, in conjunction with available information, to estimate total suspended solids (TSS), cyanide and metals concentrations in the sediments and water column in the proposed tailings impoundment. The available information includes data collected for and reported in the FEIS, data from the water quality analysis literature and parameter estimates from various experts in the field of water quality modeling and sediment transport. Since the tailings pond is viewed as a treatment facility or point source during the period which the mine would be in operation, estimates of water quality in the tailings pond are needed to assess the likelihood of compliance with effluent limitations imposed through the National Pollutant Discharge Elimination System (NPDES; see section VI.B.).

The application of mass balance methods requires the development or application of mathematical models for which certain parameters must be estimated. Whenever possible, estimates of these parameters were derived from the FEIS or from relevant existing reports. For those processes mentioned in the FEIS as ones with potential environmental impacts, and ones for which data were not available in the FEIS or in relevant existing reports, estimates based on best professional judgment have been used. In addition, EPA Region 10 has sought the advice and judgment of a number of experts in sediment transport in helping to reduce the uncertainty for certain important parameters related to the settling velocity of solids and the initial distribution of suspended solids resulting from the discharge.

The primary goal of this analysis is to use established water quality simulation methodologies as a framework for testing certain hypotheses about water quality impacts in the proposed tailings pond during and after the operation of the mine. Both the methodologies and the hypotheses have been developed using information consistent with the FEIS, related studies, and the best professional judgment of EPA's technical staff and selected technical experts in the field of sediment transport. This should provide a way of focusing on important issues and identifying the level of environmental risk associated with the construction and operation of this project.

2. Previous Modeling Efforts

Several water quality analysis methods were used to support the development of this TAR for the proposed AJ Mine project near Juneau, Alaska. The methods were used to provide support for decisions regarding whether or not the proposed wastewater treatment facility would provide adequate treatment and would support a healthy aquatic community in the tailings impoundment within a reasonable time-frame after mining/treatment operations cease. These analysis methods included:

i. Simple box model treating the tailings pond as a continuously stirred reactor with settling and leaching as the important processes (Yearsley, 1992). This model was used for estimating distributions of suspended solids and certain metals in the water column, only.

ii. One-dimensional (vertical), time-dependent model of the tailings pond based on the WASP4 (Ambrose et al, 1991) water quality simulation program. The application of WASP4 to estimating distributions of suspended solids and certain metals in the water column and sediments of the proposed AJ Mine tailings pond is described in Yearsley (1993).

iii. Two-dimensional (longitudinal, vertical), time-dependent model of the tailings pond water and sediment quality based on the WASP4 (Ambrose et al, 1991) water quality simulation program. The extension of the WASP4 methodology to two spatial dimensions was made for purposes of determining how the location of the discharge point affected the quality of the tailings pond effluent.

iv. Two-dimensional (longitudinal, vertical), time-dependent model of the tailings pond water quality based on the CE-QUAL-W2 (Cole and Chapman, 1994) computer model. CE-QUAL-W2 was included in the analysis to supplement the WASP4 simulations by incorporating a more accurate depiction of reservoir hydrodynamics than could be provided by the WASP4 methodology.

The first two applications ((i.) and (ii.)) were based on information provided in the FEIS, only. The results from (i.) and (ii.) are documented elsewhere (Yearsley, 1992; Yearsley, 1993). The primary objectives of these two applications were to provide the basis for a more systematic review of water quality issues, identify major processes affecting water quality in the proposed tailings pond, and evaluate the sensitivity of results to uncertainties in important parameters. It is often the case that simple models and environmentally conservative assumptions will be used to perform screening-level analyses of a complex problem of this type. That has been EPA's approach with respect to the proposed AJ tailings impoundment. The screening-level work set the stage for subsequent, more sophisticated analyses.

Subsequent review of the first two analyses by independent reviewers (Easton, Montgomery Watson) and by the applicant (Klone Leonoff, 1993; Woodward Clyde, 1993) led to a comprehensive review of model parameters and, ultimately, the decision to apply more sophisticated analysis methods in developing the TAR. The more sophisticated methods are represented by the model applications conducted in cases (iii.) and (iv.). Case (iv.) required data beyond that available in the FEIS.

Advanced mathematical modeling makes use of input data which may be incomplete or have a high degree of variability or uncertainty. Because of this, an important component of the report is to characterize the degree of variability or uncertainty in the available information.

3. WASP4 Conceptual Model

The toxic module (TOXIWASP) of EPA's Water Quality Simulation Program (WASP version 4.32) was used to simulate the levels of solids, cyanide and metals in the waters and sediments of the proposed tailings pond during and after the operation of the mine. The WASP software has been thoroughly tested and the program has been used in a wide variety of applications. The model kinetics in WASP generally reflect the current state of knowledge for simulation of both inorganic and organic toxic substances. The theoretical basis for the model and the user's manual can be found in Ambrose et al (1991).

This application of WASP4 evolved from the work reported in Yearsley (1993). It represents an effort to characterize the major processes affecting the distribution of solids and certain metals in the tailings pond. These processes include turbulent mixing processes associated in the pond itself, settling characteristics of the tailings, leaching and partitioning of metals and effects of initial mixing associated with discharge. It also represent a somewhat simplified approach, when compared to the CE-QUAL-W2 simulations described later in this report. This implementation of WASP4 is simplified in the sense that the variability in environmental factors such as pond geometry, inflow hydrology and local meteorology is not considered. Pond geometry is idealized so as to capture the overall dimensions of the tailings pond at Stage III (pond surface elevation approximately 920 feet above Mean Sea Level) and the annual average inflow hydrology of the Sheep Creek watershed. Effects of local meteorology, particularly high winds are included implicitly in other ways such as the coefficients of vertical eddy diffusivity.

The major difference between this application of WASP4 and the previous application (Yearsley, 1993) is in the treatment of the tailings pond geometry. For this application, the waters of the tailings impoundment have been conceptualized as a laterally well-mixed system which is compartmentalized vertically and horizontally into finite segments, each with the same surface area. Each of the segments is one meter in thickness, 350 meters long and 600 meters wide (see Figure 4). The discharge of the mine tailings is assumed to be distributed instantaneously over one or more of the water column compartments depending on the levels of available mixing energy. Once the tailings are distributed in the water column the levels of solids and associated dissolved and particulate concentrations of metals in the water column are determined by the following processes:

- Settling and resuspension rates of tailings solids
- Leaching rates of metals from tailings
- Partitioning between dissolved and adsorbed phases of metals
- Longitudinal mixing in the tailings pond due to various energy sources
- Vertical mixing in the tailings pond due to various energy sources
- Rates of inflow of surface water and groundwater
- Rates of discharge from the impoundment

The speciation between dissolved and adsorbed phases of the metals of interest are conceptualized in terms of the parent material (tailings) and leaching products (Figure 5). The parent material includes solids and associated metals. Two different classes of solids, S_1 and S_2, are considered, each class corresponding to a different median particle diameter. The metals system, C_1, associated with the i^{th} particle class (i=1,2) of solids, S_i, is transformed to the metals system, C_2, by leaching only, where the systems C_1 and C2 are actually the same metal (copper, for example). This is accomplished in the model by using a partition coefficient which is large enough to prevent the metals system, C_1, from going into solution except by leaching. The metals system, C_2, undergoes repartitioning to dissolved and adsorbed phases. The total concentration of the metal in a given compartment is the sum of the adsorbed and dissolved phases of C_1 and C_2.

The conceptual model includes a single sediment compartment associated with each vertical segment of the reservoir (Figure

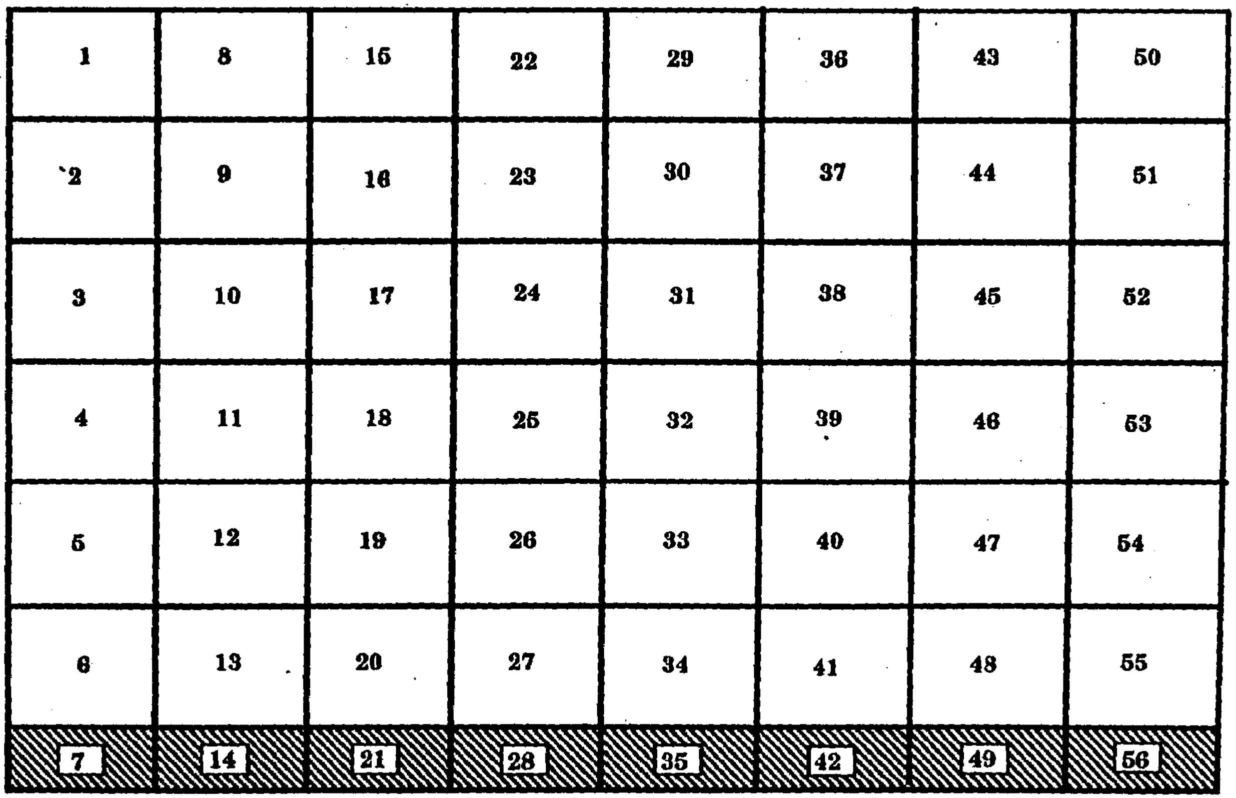


Figure 4. Two-dimensonal grid used for WASP4 simulations of the proposed AJ Mine tailings pond

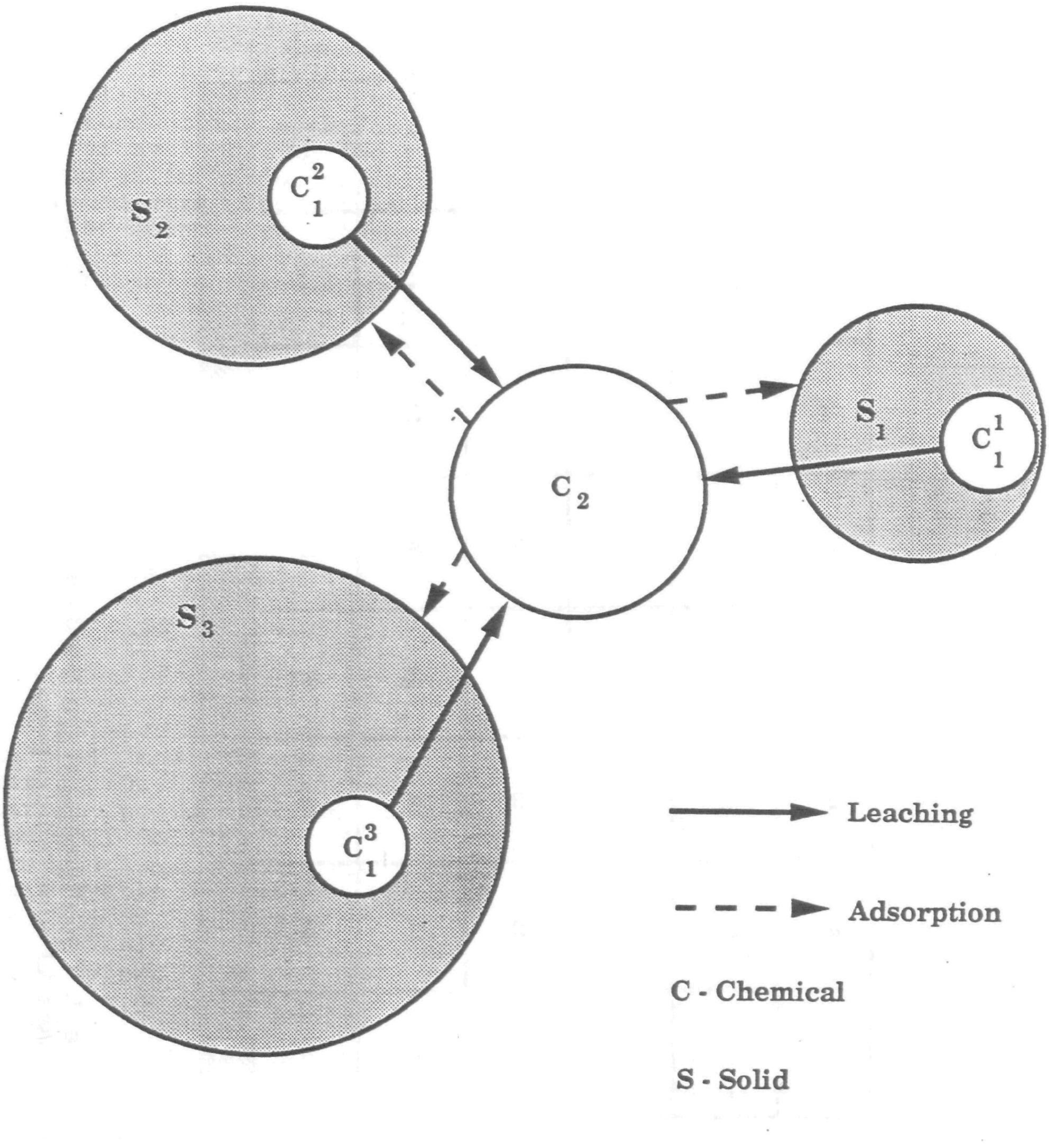


Figure 5. Conceptual model of geochemical processes for metals in the proposed AJ Mine tailings pond

4). Each of the sediment compartments has a variable volume which can account for accumulation of solids settling out of the tailings discharge. The sediment compartment also contains pore water within which dissolved species of chemicals can occur. Processes affecting levels of dissolved and adsorbed phases of metals in these sediments are:

- Settling and resuspension rates of tailings solids
- Leaching rates of metals from tailings
- Partitioning between dissolved and adsorbed phases of metals
- Vertical mixing in the tailings due to molecular diffusion in the pore water

4. Time and Length Scales

The waters of the proposed tailings pond are conceptualized as being comprised of a number of vertically and laterally well-mixed compartments the size of the project, or with overall length scales of 1000's of meters. The WASP 4 model formulation will resolve laterally-averaged concentration differences at scales of approximately the element size, which is one meter vertically and 350 meters longitudinally. The length scales are best professional estimates of what is needed to resolve details of the problem associated with deposition, resuspension and injection of the waste stream.

There are a number of time scales which may influence model results. The simulation time period (time step) for the numerical scheme used is of the order of minutes. The simulation time period does have an effect on simulation accuracy, but the fact that this time period is of the order of minutes does not mean the model is simulating processes in the tailings pond at this scale. The process time scales in the tailings pond and sediments which can be resolved by the simulation are determined by the rate of inflow of the waste stream, Sheep Creek and groundwater; the amount of storage in the tailings pond; the rates of vertical diffusion; the rate of leaching of the metals from solid to dissolved; and the rates of settling and resuspension.

5. Important Assumptions

Data from the FEIS and from other relevant sources were used to implement the conceptual model within the WASP framework. These sources provided the basis for defining important parameters used in the simulations. The fixed parameter values used in the simulations are given in Table 6. The implementation of WASP for this problem included the following assumptions:

Table 6. Important parameters and their values as used in the simulation of suspended solids and total and dissolved metals in the proposed AJ project tailings pond.

Parameter	Description	Value
ρ_t	Tailings density	2770kg/meter^3
ρ_f	Freshwater density	1000 kg/meter^3
μ	Dynamic viscosity of freshwater	1.52x10^{-3} kg/meter/second
W_t	Tailings loading	2.36x10^2 kg/second
A_{surf}	Pond surface area	1.70x10^6 meters^2
Q_{avg}	Average Sheep Creek discharge	1.382 meters^3/second
w_1	Settling rate of fine tailings (2.5%)	1.50x10^{-6} meters/secon
w_2	Settling rate of coarse tailings (97.5%)	1.00x10^{-4} meters/secon
K_l	Metals leaching rate	1.16x10^{-7} seconds^{-1}

- The surface level of the tailings pond is constant and pond size is approximately the same as the proposed tailings pond at the end of the project (surface level at 918 feet above Mean Sea Level)

- Settling rates of particles are determined from Stokes' Law

- The tailings slurry is made up of two particle classes, the first class comprising a very small fraction of the discharge which goes into suspension when the tailings are initially discharged, the second comprising the remainder of the discharge which settles out almost immediately. The particle size of each fraction is assumed to be equal to the particle size of the median fraction for each group.

- Settling velocities for each of the two particle classes can be estimated from Stokes' Law using the median diameter for each fraction, or the lowest reported value, in cases where the particle size of any fraction was not given in any reports provided by the applicant.

- Leaching rates of metals from solid to dissolved are characterized by first-order reactions

- Solubility of metals in freshwater is determined by ratios of dissolved to total metals analyzed by Frank (1994)

- Inflow rate to and discharge rate from the impoundment are both constant and equal to the annual average flow of Sheep Creek

- Dissolved levels of metals in Sheep Creek are below detection limits

6. Parameter Estimation

The density of the tailings, ρ_t, was obtained from results reported by Knight and Piesold (1989). The leaching rate, K_l, was chosen so as to have a time scale (100 days) approximately the same as the time scale found in the bench tests for iron, as presented in Appendix A of the FEIS. Shorter time scales for the leaching rate would result in higher concentrations of dissolved metals in the tailings pond.

The fraction of important metals in the digested slurry, for both solid and dissolved phases, were estimated from various data sources by Frank (1994). These estimates are given in Table 7.

Partitioning coefficients for determining the equilibrium concentrations of metals in the solid and dissolved phases were estimated from the ratios given in Table 7 using the following equation:

$$K_p = \frac{C_{solid}}{C_{diss} TSS} 10^6$$

where,

K_p= linear partition coefficient, l/kg

C_{solid} = concentration of constituent in solid phase, mg/l

C_{diss}= concentration of constituent in dissolved phase, mg/l

TSS= concentration of total suspended solids, mg/l

a. Vertical Mixing

Rates of vertical diffusion in the tailings pond were determined by the levels of available turbulent energy. According to the FEIS, the potential sources of this energy include wind stress, kinetic energy from advective sources including Sheep Creek and the waste stream itself and kinetic energy associated with avalanches. The FEIS (page 4-34) gives estimates of the coefficient of turbulent diffusion for stratified reservoirs as 1.00×10^{-4} meters^2/second and for unstratified reservoirs as 1.00×10^{-3} meters^2/second. The FEIS states

> "A sense of how much settling would occur can be gained by estimating a vertical diffusion time scale for the impoundment, and comparing it to the settling velocity of discrete particles. A maximum diffusivity value of D=0.0001 m^2/sec has been suggested for prediction of hypolimnetic diffusion in stratified reservoirs (Imberger and Patterson 1981; Harleman 1986). ---
>
> However, available data suggest that the impoundment would likely be unstratified, and vertical diffusivity would likely be much greater. Assuming a vertical diffusivity of D=0.001 m^2/sec. leads to a characteristic time scale of approximately one day.---"

In Woodward-Clyde's (1993) evaluation of the WASP4 analysis done by Yearsley (1993), coefficients of vertical eddy diffusivity ranged between 1×10^{-6} and 1×10^{-4} meters^2/second. The applicant asserted (Bergstrom, 1993) that for the WASP4 analysis of the tailings, "1.0×10^{-04} meters^2/second would characterize the

Table 7 Estimated solid and dissolved concentrations of certain metals in the waste stream discharged from the proposed AJ Mine milling process. Estimates of concentrations are derived from Section VI.D.

Metal	Concentration	
	Solid (mg/kg)	Dissolved (mg/l)
Arsenic	9.8	0.014
Cadmium	11.0	0.010
Chromium	35.0	0.020
Total CN	0.0	2.5
Copper	260.	1.300
Lead	200.	0.020
Mercury	0.27	0.005
Selenium	1.	0.016
Zinc	530	0.087

most conservative of high wind conditions, not characteristic of typical situations, and that $1.0x10^{-05}$ would be characteristic of general pond conditions". The range chosen for the coefficient of vertical eddy diffusivity in these simulations corresponds to the ranges described by Bergstrom (1993).

These coefficients are measures of the general levels of vertical mixing processes in reservoirs. For the purposes of this report, it has been assumed that levels of energy implied by these coefficients do not include energy sources unique to this project such as avalanches and energy transport in the waste stream itself. The energy associated with the waste stream must be dissipated in some way and this can happen by enhancing vertical mixing processes or by scouring sediment from the bottom. Data collected by Rescan Environmental Services Ltd (1983) in the vicinity of the Island Copper discharge show that vertical mixing associated with discharge of mine tailings can lead to high levels of suspended solids (Figures 6 and 7) in the water column.

b. Pore Water Diffusion Rates

Flux of dissolved metals from the sediment pore water to the overlying water column is of concern for water quality conditions in the tailings pond after the mining operation has ceased. This flux is determined by the geochemical processes which affect partitioning between solid and dissolved phases in the sediments as well as the rates of advection and diffusion within the pore water. Rates of advection are determined primarily by groundwater flow, which are discussed in a later section of this report. Diffusion rates within the pore water are generally at levels comparable to molecular diffusion (Schnoor et al., 1987). However, Schnoor et al. (1987) point out that bioturbation of the sediments by benthic fauna or fish can increase diffusion rates from the pore water to the overlying water by an order of magnitude. The metals concentrations in the pore water represent a serious threat to water quality in the proposed tailings pond after the mining operation is finished. To assess the effects of changes in the rate of diffusion from the pore water on the overlying water column, the pore water diffusion coefficient was varied between the molecular diffusion rate ($1.0x10^{-9}$ meters^2/second) and the diffusion rate for sediments subject to bioturbation ($1.0x10^{-8}$ meters^2/second).

c. Initial Mixing of Discharge

The FEIS (page 4-34) states that no methods of analysis are available to describe the initial mixing of the waste stream in the tailings pond. This is not entirely the case, as a model of the density flow for the proposed Quartz Hill project was used to estimate the plume characteristics of a discharge approximately

Figure 6. Suspended solids 300 meters from the Island Copper discharge (Rescan, 1983)

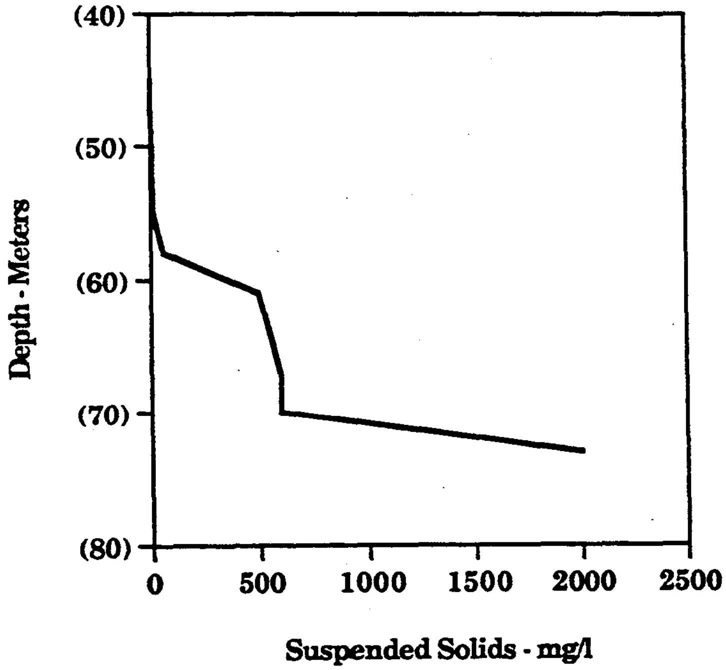
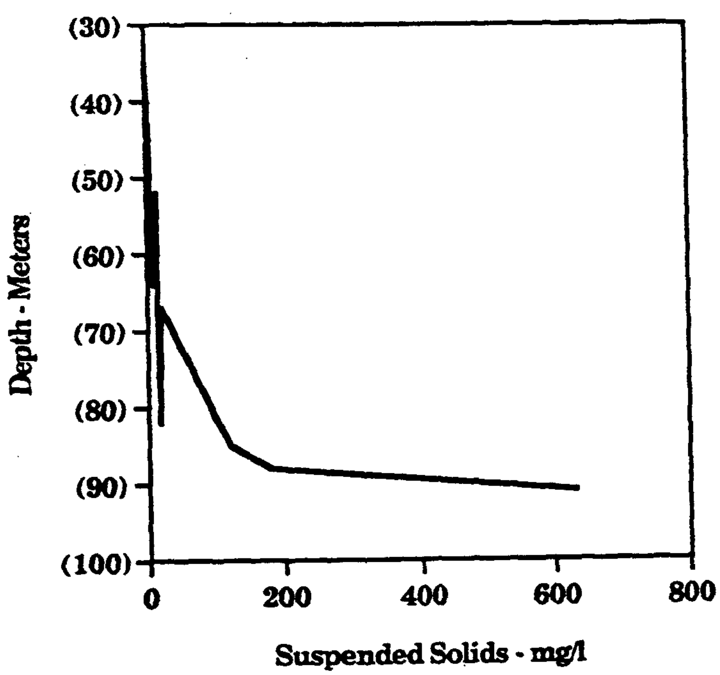


Figure 7. Suspended solids in Rupert Inlet 1200 meters from the Island Copper discharge (Rescan, 1983)



three times larger than the proposed AJ Mine discharge (Kowalik and Findikakis, 1985). The ocean disposal model of Koh and Chang (1973) which evolved into the Offshore Operators Committee drilling mud model (Brandsma et al, 1983) also provides a framework for examining density flows such as that associated with the proposed AJ Mine project. However, at the time the modeling process was begun by EPA Region 10, the only information available for evaluating the initial mixing characteristics of the discharge was limited to that provided in the FEIS. The analysis of initial mixing of the AJ Mine tailings discharge, as provided in the FEIS, was not sufficiently quantitative to provide inputs for simulation methods such as WASP4. The initial approach used to address this question was to survey the available literature; primarily that associated with the discharge of drilling muds and cutting from offshore drilling operations (Ayers et al, 1982, for example) as well as the discharge of tailings from other mining operations such Island Copper on Vancouver Island. In these studies, field data showed that 5-10% of the total load became suspended during the initial discharge phase.

The WASP4 simulations reported by Yearsley (1993) were based on initial mixing characteristics approximately the same as those described by Schubel et al (1978) for dredge materials, Brandsma et al (19783) for drilling muds discharges and from the evaluation of the tailings discharge at the Island Copper Mine on Vancouver Island. Assumptions regarding the initial mixing characteristics of the proposed discharge were reviewed by various consultants for the applicant and by a number of experts in the field of sediment transport. As a result of this review, EPA Region 10, in cooperation with Echo Bay, convened an *ad hoc* panel of experts to develop a best professional estimate of the characteristics of the initial discharge. Their recommendations are described, in part, below:

Mark Dortch, PhD, US Army Corps of Engineers, Waterways Experiment Station (1993):

"---- I recommend a mid-range compromise value of 2.5 percent, which should still be on the conservative side. ----- The 625 tons/day of suspension should be composed of particles from the 0 to 2.5 percent smallest size class, i.e., 97.5% of the particles by weight is (sic) larger than this class. Therefore, a median diameter corresponding to the 1.25 percentile should be used-------"

Earl Hayter, PhD, Clemson University, South Carolina Water Resources Research Institute (1993):

"--Based on the tailings size gradation, settling velocity analysis, and characteristics (e.g., specific gravity, mineralogy) of the tailings, my estimate is 10%---".

Ray B. Krone, PhD, Ray B. Krone & Associates (1993):

"--I would expect an upper limit of less than 3 percent of the total load of lowest decile particles and 2 percent of the second decile. In view of the dissimilarity between the finer sediments in the referred dredging studies and the proposed mine tailings, these estimates are probably high.---"

Based on these recommendations, the WASP4 simulations were performed assuming that 2.5% of the total load, comprising particles from the 0 to 2.5 percent smallest size class, were suspended initially at the discharge point and that the remainder (97.5%) settled quickly to the bottom of the tailings pond. The discharge was assumed to be distributed uniformly throughout the bottom element of the conceptualized two-dimensional grid, simulating a discharge confined to the bottom.

d. Settling, Deposition and Resuspension

Sediment transport mechanisms will play an important part in determining levels of suspended solids and metals in the tailings pond. Primary transport mechanisms which must be considered are settling, deposition and resuspension. While a great deal of research has been devoted to investigating mechanisms of sediment transport, there is still much uncertainty in quantifying these mechanisms. Information in the FEIS specific to this project is limited to a series of settling tests performed by Lakefield Research (1990). However, these tests were not designed to estimate settling velocities of particle classes.

Recently, Hartman Associates (1994) conducted tests on the mine tailings using the U.S. Army Corps of Engineers SETTLE methodology. Krone (1994), however, observed that the results of these particular tests could not be applied directly to estimating settling velocities of particle classes and recommended additional tests. The results of the additional tests recommended by Krone (1994) were not available at the time the modeling work described below was performed. Nor was it clear that these tests would address all the issues of uncertainty associated with the settling, deposition and resuspension of material in the proposed AJ Mine tailings pond. As a result, the WASP4 simulations do not include any of the test results. However, as described later in this report, a preliminary examination of these data was performed using CE-QUAL-W2.

Because of the limited information provided in the FEIS and by the applicant, it was, therefore, necessary to rely on the judgment of sediment transfer experts to provide input for the WASP4 simulations. None of the experts made specific recommendations regarding ways of computing the settling velocity, although Krone (1993) observed that "Absence of cohesion is assumed by the modelers, a conservative assumption." Given the lack of consensus from the panel, it was assumed that Stokes' Law settling was appropriate for this analysis.

Determination of settling velocities using Stokes' Law requires that a characteristic particle size be specified. Dortch (1993), as quoted above, is the most specific in regard to estimating particle sizes. Dortch (1993) recommended the use of the particle size associated with the 1.25 percentile. However, the tests on tailings characteristics performed by Knight and Piesold (1989) provide results which include particles associated with the 5.9 percentile, but no lower. Rather than extrapolate the Knight and Piesold (1989) results to the 1.25 percentile, the 5.9 percentile particle size diameter was chosen to represent the settling velocity of those particles which become suspended during initial discharge. As pointed out by Dortch (1993), this would lead to lower simulated levels of suspended material than if the true median diameter were used to estimate particle fall velocity.

According to the WASP4 manual (Ambrose et al, 1991) the probability of deposition depends on the shear stress on the benthic surface and the suspended sediment size and cohesiveness. For fine silts of 5 microns, or less, the manual states that deposition is not to be expected, even under quiescent conditions. Ariathurai (1985) presents results of Hjulstrom which show regions of erosion-transportation-sedimentation in the space characterized by mean velocity near the bed and particle diameter. These results show that particles of 100 microns or less remain in suspension at very low velocities (<0.001 meters/second). The FEIS gives a semi-quantitative discussion of resuspension of tailings pond sediments (page 4-34) as they are affected by cyclic changes in inflow-outflow characteristics and as a result of high winds. The panel of experts had a wide range of recommendations regarding actual values. This included values of the probability of deposition as low as 0.2 and as high as 1.0. Previous simulations using WASP4 (Yearsley, 1993) considering values as low as 0.2, also. However, for the simulations performed here, the probability of deposition was assumed to be 1.0 for all conditions. A lower probablility of deposition would result in higher concentrations of simulated suspended solids.

e. Geochemical Processes

Partitioning between dissolved and adsorbed phases will determine the importance of sediment transport in the mass balance of metals. The FEIS does provide the results of several leach column tests and these tests provide the basis for the estimate of first-order rate of leaching from solid to dissolved. The leaching rate used in the various scenarios was chosen so as to have a time scale (100 days) approximately the same as the time scale found in the bench tests for iron as present in Appendix A, Volume II of the FEIS. The partition coefficient, K_p, for each of the metals was estimated based on the ratio of total and dissolved concentrations listed in Table 7. The results for the equilibrium partition coefficients are given in Table 8.

f. Groundwater Flow

The FEIS (Dept. of Interior, 1992) estimated that groundwater flow was 24% of the total discharge at the "narrows". SRK (1994) estimated that the present level of groundwater flow to Sheep Creek is about 7 cfs. Although the SRK (1994) report is not specific regarding the source of the groundwater, others (Easton, 1993) have suggested it is probably associated with the alluvium covering the floor of Sheep Creek valley to thicknesses of as much as twenty feet (SRK, 1990). Groundwater flow in the igneous rocks beneath and surrounding Sheep Creek valley is estimated to be low. The previous application of WASP4 (Yearsley, 1993) considered the effects of groundwater flow, based on flow estimates given in the FEIS. Most of those who reviewed the previous WASP4 simulations or had visited Sheep Creek valley (Easton, 1993; Krone, 1994) felt that groundwater return to Sheep Creek would not be a factor once the tailings had been deposited. However, these are opinions, only, and have not included any quantitative assessment of:

(1) the relative transmissivity of alluvium, talus, basement rock and tailings. Data submitted by the applicant (SRK, 1990) show the extent of alluvium and talus, both of which have relatively high transmissivity compared to the basement rock;

(2) the piezometric head of the groundwater system;

(3) the rate of groundwater flow through the existing system of mine tunnels.

A preliminary analysis of groundwater is given in Appendix C. The preliminary analysis described in Appendix C supports the hypothesis that groundwater will flow through the tailings into the tailings pond. However, the analysis is based on very limited data. Until these issues are specifically addressed, impacts of the groundwater cannot be ignored, particularly as it could potentially affect water quality after mining has ended and the tailings pond reverts to waters of the United States. For purposes of quantifying this potential, some simulations were performed assuming a groundwater flow of 7 cfs, as assumed by SRK (1994).

Table 8. Equilibrium partition coefficients, K_p, for various metals in the proposed AJ Mine tailings discharge, computed using data provided by the applicant. Coefficients computed from the data are compared to estimates given in Mills et al (1985)

Element	FEIS Digest (mg/kg)	Decant (mg/l)	Total Solids (kg/l)	Total (mg/l)	K_p (data) (l/kg)	K_p (Mills et al,1985) (l/kg)
Arsenic	9.8	0.014	0.637	6.26	700	278
Cadmim	1.1	0.010	0.637	7.02	1100	16
Chromium	35	0.020	0.637	22.3	1750	59000
Copoper	260	1.300	0.637	167.	200	17
Mercury	0.27	0.005	0.637	0.18	54	0.3
Lead	200	0.020	0.637	127.	10000	1700
Selenium	1	0.016	0.637	0.66	63	-----
Zinc	530	0.087	0.637	338.	6100	380

g. Avalanches

The previous simulations using WASP4 (Yearsley, 1993) considered the potential impacts of avalanches on water quality, primarily as they might affect levels of suspended solids. The simulations were based on the FEIS estimates of a maximum probable energy of 10^7 joules (page 4-33) for a single event. This kinetic energy was converted into potential energy by assuming the effect of the avalanche was to resuspend tailings deposited on the bottom of the pond during operation of the mine. The tailings disturbed by the avalanche were distributed uniformly throughout the pond, with the center of gravity of this mass of material at a distance equal to half the depth of the reservoir above the bottom of the pond, It was further assumed that the process was 100% efficient in converting the kinetic energy to potential energy.

One reviewer of this avalanche analysis felt that avalanche energy could well be orders of magnitude higher, while the applicant believes that the avalanche issue has been resolved based on work reported by Mears (1993). The range of uncertainty associated with this question was substantially greater than for any of the others. Until this uncertainty can be reduced to a level commensurate with other issues, the avalanche issue should be considered qualitatively as an additional risk factor, but one which will not be included in WASP4 simulations.

h. Reservoir Hydrodynamics

WASP4 is constructed as a framework for performing mass balances of heavy metals and other toxic substances. It does not simulate the hydrodynamics of surface or groundwater systems. Rather, it relies on reservoir hydrodynamic results being supplied externally by other methods of analysis. For the simulations described in this report, three different modes of reservoir hydrodynamics were considered. These modes were formulated to capture flow characteristics which have been observed in other reservoir systems or were suggested as possible outcomes by those who reviewed the previous application of WASP4 to the AJ Mine tailings pond. The three modes of reservoir hydrodynamics were 1) surface flow; 2) uniform flow; and 3) underflow. In the first mode, the inflow from Sheep Creek is confined to the surface two meters (Figure 8), much as in the analysis done by Woodward-Clyde (1993). In the second mode, the inflow is distributed uniformly from top to bottom (Figure 9) and in the third mode, the inflow is confined to the bottom two meters (Figure 10). In all cases, inflow and outflow are assumed to be constant and equal to the annual average flow of Sheep Creek.

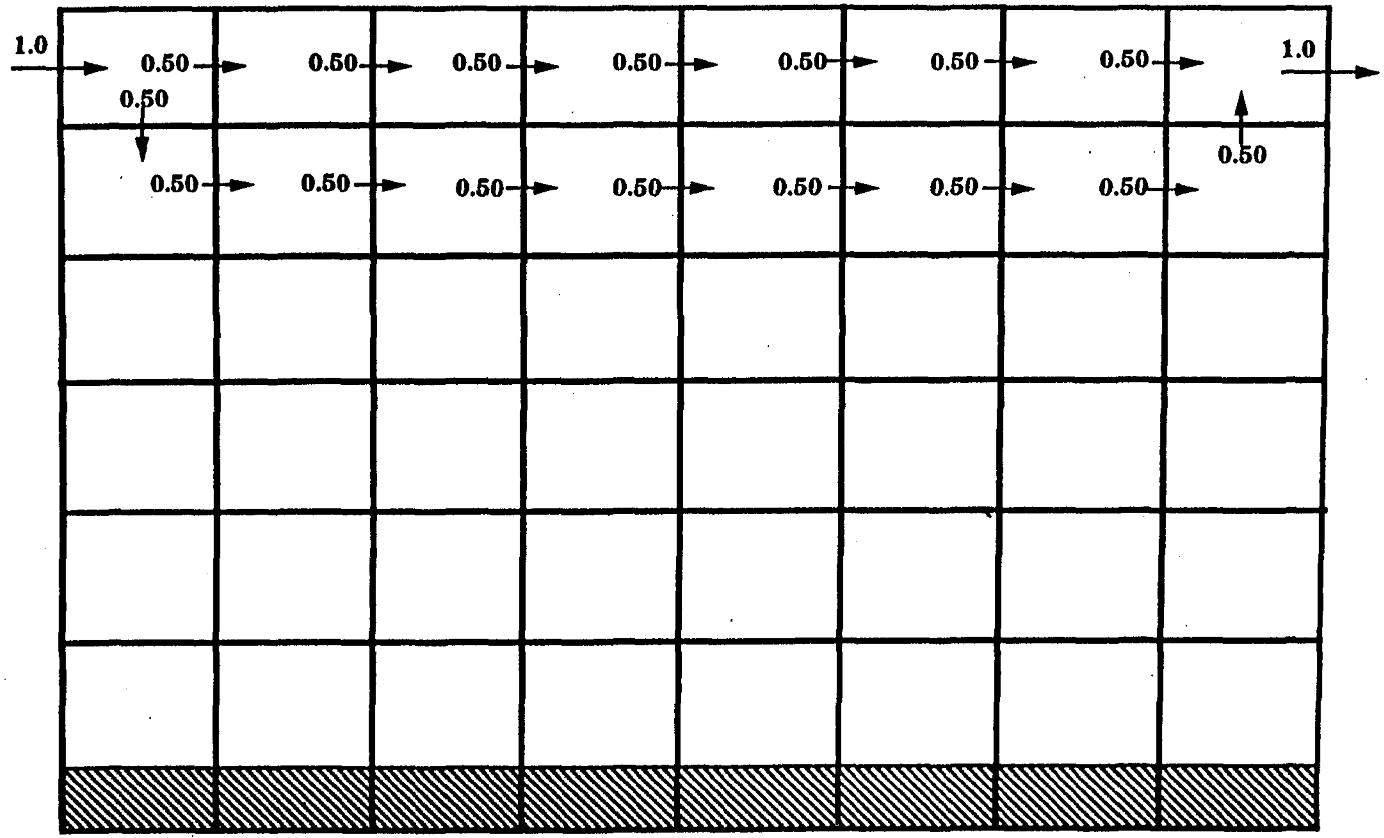


Figure 8. Velocity profiles used for surface flow scenarios in the WASP4 analysis of the proposed AJ Mine tailings pond. Numbers by arrows indicate fraction of total flow.

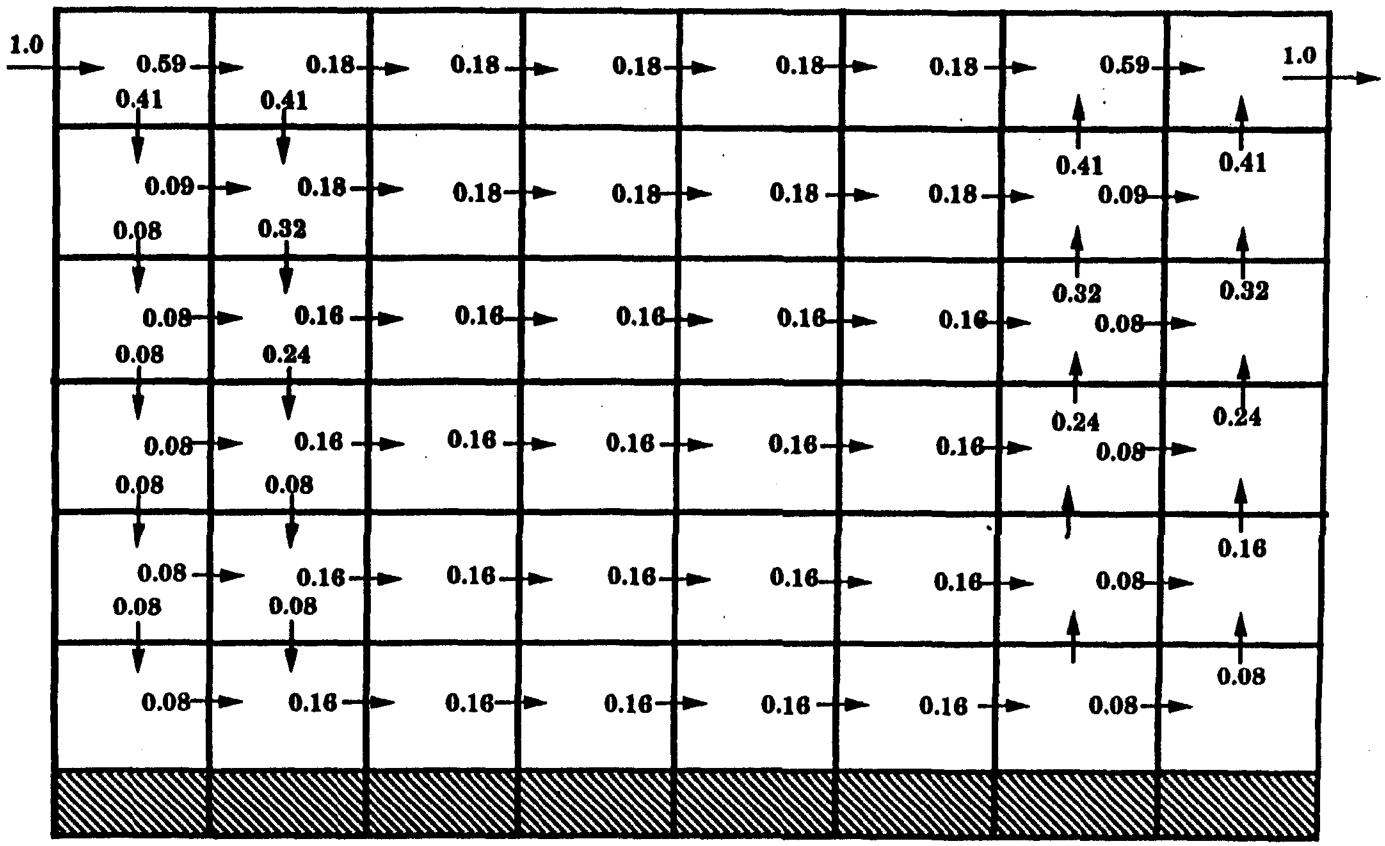


Figure 9. Velocity profiles used for uniform flow scenarios in the WASP4 analysis of the proposed AJ Mine tailings pond. Numbers by arrows indicate fraction of total flow.

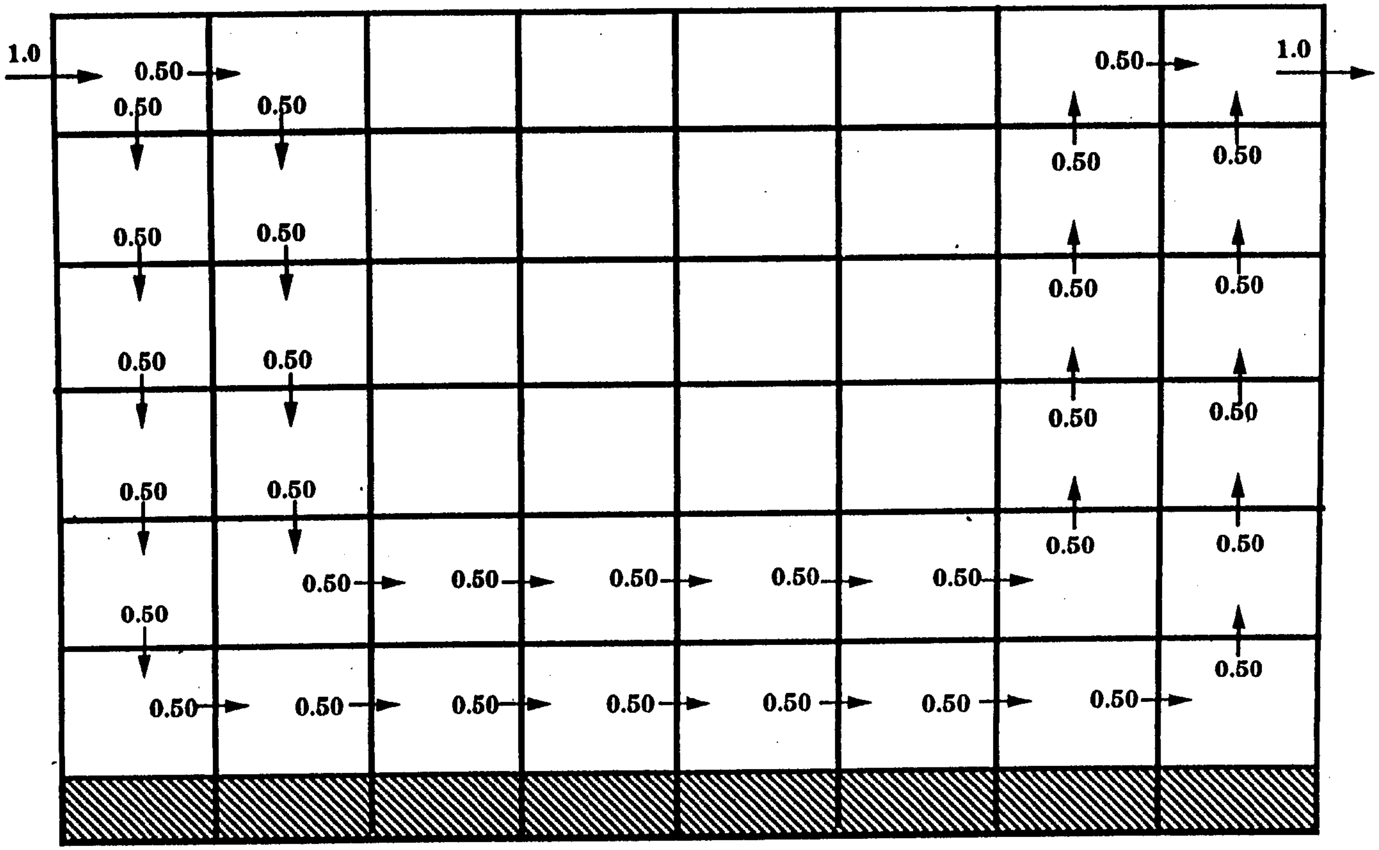


Figure 10. Velocity profiles used for underflow scenarios in the WASP4 analysis of the proposed AJ Mine tailings pond. Numbers by arrows indicate fraction of total flow.

i. Discharge Location:

The applicant has proposed moving the discharge point periodically so that the tailings will be distributed more or less uniformly over the bottom of the pond. Furthermore, they have indicated the discharge point will be no closer to the proposed dam than 0.5 miles and no farther upstream than 0.5 miles from the Sheep Creek Portal. To simulate the range of outcomes, the discharge point was located at the most downstream point indicated by the applicant for certain scenarios and at the most upstream point for other scenarios. In all cases, discharge was assumed to be at the bottom of the tailings pond.

7. Simulations

Suspended solids and concentrations of dissolved metals and metals in suspended solids in the proposed tailings pond and in the sediments beneath the table were simulated under the following conditions:

Operation of Mine

1. Initialize the reservoir on January 1 at a depth of approximately 20 ft (6.0 meters).

2. Begin simulation on January 1 of a fictitious year assuming inflow from Sheep Creek is constant at a rate equal to the annual average flow given in the FEIS.

3. Begin discharge of tailings on January 1 at the rate of 22,500 tons/day (236 kg/second) and discharge continuously for a period sufficiently long for the system to reach a steady state (one year appears to be adequate).

4. Discharge water from the tailings pond at a rate equal to the average annual discharge of Sheep Creek (approximately 49 cfs).

Post-Operational Period

1. Cease discharging tailings to the tailings pond on December 31 of the first fictitious year. From January 1 of the second fictitious year until the end of the simulation, the hydrology of Sheep Creek is assumed to be equal to the estimated annual average flow.

2. Discharge water from the tailings pond at a rate equal to the average annual discharge of Sheep Creek for a period of three years.

Simulations of TSS, cyanide and various metals were performed for several different scenarios. Scenarios were designed to provide a means of assessing the sensitivity of WASP4 simulations to uncertainty in knowledge of important processes. These processes included:

- vertical mixing due to wind stress, kinetic energy from advective sources including Sheep Creek and the waste stream itself and potential energy associated with density differences

- hydrodynamic regime

- groundwater inflow

- mixing characteristics of sediment pore water

Model parameters which characterize processes for the various scenarios, and their respective values, are given in Table 9.

An example of a WASP4 input data set, based on Scenario #1 is given in Appendix C.

8. Results

Simulated steady-state values of total solids and solid and dissolved phases of certain heavy metals associated with the mining process are given in Tables 10 - 13. During the period when tailings are being discharged (Tables 10 and 11), simulated steady-state levels of total solids and heavy metals in the effluent from the pond are more sensitive to changes in the coefficient of vertical eddy diffusivity than in changes to the hydrodynamic environment. With the exception of total CN, total concentrations of the chemicals in the discharge are closely related to the level of TSS in the discharge. Since total CN is considered only in the dissolved form and is assumed to have an equilibrium partition coefficient, K_p, equal to zero, it is dependent only on the amount of dilution provided by the inflowing freshwater. As is clear from Tables 10 and 11, this estimated discharge is independent of such parameters as vertical mixing rate, settling velocity of solids and hydrodynamic regime type.

When tailings are discharged at the most upstream point (Scenarios 1-6), estimated steady-state levels of both total solids and heavy metals during conditions of high turbulence (coefficient of vertical eddy diffusivity = 1.0×10^{-4} meters^2/sec) are approximately 50% higher than estimates for conditions of low turbulence (coefficient of vertical eddy diffusivity = 1.0×10^{-5} meters^2/sec) . Corresponding variability between estimates when

Table 9. Description of scenarios used to assess sensitivity of WASP4 simulations

Scenario	Hydrodynamic Flow Regime	Discharge Location	Groundwater Flow (meters^3/sec)	Coefficient of Vertical Eddy Diffusivity (meters^2/sec)	Coefficient of Molecular Diffusion (meters^2/sec)	Settling Velocity (meters/sec)	
						Solids 1	Solids 2
1	Underflow	Upstream	0.0	1.00×10^{-5}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
2	Underfow	Upstream	0.0	1.00×10^{-4}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
3	Uniform Flow	Upstream	0.0	1.00×10^{-5}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
4	Uniform Flow	Upstream	0.0	1.00×10^{-4}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
5	Surface flow	Upstream	0.0	1.00×10^{-5}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
6	Surface Flow	Upstream	0.0	1.00×10^{-4}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
7	Underflow	Downstream	0.0	1.00×10^{-5}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
8	Underfow	Downstream	0.0	1.00×10^{-4}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
9	Uniform Flow	Downstream	0.0	1.00×10^{-5}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
10	Uniform Flow	Downstream	0.0	1.00×10^{-4}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
11	Surface flow	Downstream	0.0	1.00×10^{-5}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
12	Surface Flow	Downstream	0.0	1.00×10^{-4}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
13	Uniform Flow	----------------	0.0	1.00×10^{-5}	1.00×10^{-8}	1.50×10^{-6}	3.50×10^{-3}
14	Uniform Flow	----------------	0.198	1.00×10^{-5}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}
15	Uniform Flow	----------------	0.198	1.00×10^{-4}	1.00×10^{-9}	1.50×10^{-6}	3.50×10^{-3}

Table 10. Steady-state values of total solids, arsenic, cadmium chromium and copper in the discharge from the proposed AJ Mine tailings pond during operation of the proposed mine as simulated by WASP4. Parameters associated with each of the Scenarios are given in Table 9.

Scenario #	Total Solids (mg/l)	Arsenic (µg/l)		Cadmium (µg/l)		Chromium (µg/l)		Copper (µg/l)	
		Total	Dissolved	Total	Dissolved	Total	Dissolved	Total	Dissolved
1	850	9	1.5	9.9	1.3	31	2.8	260	70
2	1200	13	1.4	14.0	1.2	44	2.5	360	73
3	790	8	1.5	9.3	1.3	29	2.9	250	72
4	1200	13	1.4	14.1	1.2	44	2.5	360	74
5	760	8	1.5	9.0	1.3	28	2.9	240	71
6	1200	13	1.4	14.1	1.2	44	2.6	360	74
7	1200	12	0.8	13.9	0.7	44	1.4	340	43
8	1700	17	0.4	19.3	0.3	61	0.6	470	24
9	1100	11	0.9	12.1	0.7	38	1.5	300	44
10	1700	17	0.3	19.0	0.3	61	0.6	460	25
11	940	10	0.9	10.8	0.7	34	1.6	270	45
12	1800	17	0.3	19.6	0.3	62	0.5	480	24

Table 11. Steady-state values of mercury, lead, selenium and zinc in the discharge from the proposed AJ Mine tailings pond during operation of the proposed mine as simulated by WASP4. Parameters associated with each of the Scenarios are given in Table 9.

Scenario #	Mercury (μg/l)		Lead (μg/l)		Selenium (μg/l)		Zinc (μg/l)		Cyanide (μg/l)
	Total	Dissolved	Total	Dissolved	Total	Dissolved	Total	Dissolved	Dissolved
1	0.30	0.11	170	3.7	1.1	0.4	460	16	430
2	0.41	0.13	250	3.2	1.5	0.5	660	13	430
3	0.29	0.11	160	3.8	1.1	0.4	430	16	430
4	0.42	0.13	250	3.2	1.5	0.5	660	13	430
5	0.28	0.11	150	3.7	1.0	0.4	410	16	430
6	0.42	0.13	250	3.2	1.5	0.5	660	13	430
7	0.38	0.08	250	1.8	1.4	0.3	650	7	430
8	0.52	0.06	340	0.6	1.9	0.2	920	2	430
9	0.35	0.08	220	1.9	1.3	0.3	570	8	430
10	0.52	0.06	360	0.6	1.9	0.2	910	3	430
11	0.31	0.08	190	2.1	1.1	0.3	500	9	430
12	0.53	0.06	350	0.6	1.9	0.2	940	2	430

Table 12. Steady-state values of total solids, arsenic, cadmium chromium and copper in the discharge from the proposed AJ Mine taiings pond after operation of the proposed mine as simulated by WASP4. Parameters associated with each of the scenarios are given in Table 9.

Scenario #	Arsenic (μg/l)		Cadmium (μg/l)		Chromium (μg/l)		Copper (μg/l)	
	Total	Dissolved	Total	Dissolved	Total	Dissolved	Total	Dissolved
1-12	0.06	0.06	0.05	0.05	0.09	0.09	5.9	5.9
13	0.61	0.61	0.43	0.43	0.84	0.84	56	56
14	2.0	2.0	1.4	1.4	2.8	2.8	190	190
15	2.0	2.0	1.4	1.4	2.8	2.8	190	190

Table 13. Steady-state values of mercury, lead, selenium and zinc in the discharge from the proposed AJ Mine taiings pond after operation of the proposed mine as simulated by WASP4. Parameters associated with each of the Scenarios are given in Table 9.

Scenario #	Mercury (µg/l)		Lead (µg/l)		Selenum (µg/l)		Zinc (µg/l)	
	Total	Dissolved	Total	Dissolved	Total	Dissolved	Total	Dissolved
1-12	0.022	0.022	0.084	0.084	0.072	0.072	0.387	0.387
13-	0.215	0.215	0.844	0.844	0.686	0.686	3.70	3.70
14	0.71	0.71	2.65	2.65	2.29	2.29	11.9	11.9
15	0.71	0.71	2.65	2.65	2.29	2.29	11.9	11.9

vertical turbulence is held constant is very small for the high turbulent condition and of the order of 15% for the low level of turbulence. For discharge at the downstream point (Scenarios 7-12), variability of results associated with changes in level of turbulence is similar to that for upstream discharge. However, variability of results associated with changes in the hydrodynamic flow regime is much greater for the downstream discharge than for the upstream.

Simulated results for the sediment pore water and solids are not reported. The reason for this is that since the values reported by Frank (1994) for the effluent solid and dissolved phases of metals (Table 7) were used to estimate partition coefficients, simulated results for the sediment concentrations of metals were the same as the dissolved and solids concentrations for each of the heavy metals as given in Table 7.

The high metals concentrations in the pore water of the tailings pond sediment do not have a noticeable impact on discharge concentrations during the period the mine is in operation. However, once the mine has stopped discharging tailings to the pond, the sediment pore water, for the assumptions made in this application of WASP4, is the primary source for heavy metals in the overlying water column. For the base conditions (Scenarios 1-12), the estimated concentrations of all metals are generally low. Bioturbation of the sediments (Scenario 13) leads to an increase in discharge concentrations of metals of an order of magnitude. Inflow of the maximum amount groundwater (Scenarios 14 and 15) results in a threefold increase over estimates from the bioturbation conditions.

For the parameters considered in the scenarios (Table 9), the simulated results for TSS and metals during operation of the mine were most sensitive to the coefficient of vertical eddy diffusivity and to the location of the discharge point. As pointed out previously, the simulated concentration of total CN was a function of the ratio of freshwater inflow to the rate of effluent discharge, only. After mining operations cease, the concentration of metals in the water of the tailings pond is sensitive to the rate of bioturbation and to the quantity of groundwater flow through the sediments.

F. Effluent Quality Predictions: CE-QUAL-W2 Water Quality Model

1. Conceptual Model

Version 2.04 of CE-QUAL-W2 (Cole and Chapman, 1994) was used to simulate tailings pond hydrodynamics, tailings pond water temperature, suspended solids and level of dissolved metals. CE-QUAL-W2 has been applied to a wide variety of water bodies which have small lateral variability and for which density plays an important role in the hydrodynamics. This includes estuarine systems, as well as freshwater reservoir systems. The model kinetics in CE-QUAL-W2 generally reflect the current state of knowledge for simulation of chemical, physical and biological state variables in estuarine or reservoir systems. The theoretical basis for the model and the user's manual can be found in Cole and Buchak (1993).

CE-QUAL-W2 was applied to the AJ Mine tailings pond after a number of reviewers recommended that a model be used which had the potential for simulating underflows associated with differences between inflow density and ambient density structure of the pond. CE-QUAL-W2 was chosen based on its availability and widespread usage. CE-QUAL-W2 was not developed to simulate the kinetics of metals partitioning. Nor was CE-QUAL-W2 developed to simulate dissolved and solids concentrations in the sediments accumulating beneath the pond and the interactions between metals in the sediments and surface water. CE-QUAL-W2 does, however, simulate suspended solids and conservative constituents in laterally-averaged surface waters including lakes, reservoirs and estuaries. The conceptualization of the tailings pond system for purposes of applying CE-QUAL-W2 was done so as to accommodate this limitation without making substantive changes to what is a complex system of software. The conceptual model was designed to assess effects of density flows, variability in environmental conditions, but to aggregate geochemical processes in a manner consistent with the existing capabilities of CE-QUAL-W2. Since CE-QUAL-W2 was not developed to simulate the interchange of metals between sediments and the overlying water column, there was little to be gained by using CE-QUAL-W2 to assess water quality conditions after mining operations cease. Therefore, CE-QUAL-W2 was used to simulate conditions only while the mine was in full operation.

In this application of CE-QUAL-W2, the waters of the tailings impoundment have been conceptualized as a laterally well-mixed system which is compartmentalized vertically and horizontally into finite segments for two different operating levels of the proposed impoundment. These operating levels correspond to the proposed Stage I level when the impoundment first begins discharging and the proposed Stage III level when the impoundment is at maximum level. These two stages correspond to nominal surface levels of 765 feet above Mean Sea level and 918 feet above Mean Sea Level, respectively. Each of the segments in both conceptualization is one meter in thickness. Segment lengths are variable, but on the order of 200-400 meters, while the widths of each of the segments were determined from topographical maps provided by the applicant. Longitudinal sections of each conceptualization are shown in Figures 11 and 12.

The discharge of the mine tailings is assumed to be distributed instantaneously over the bottom segment at one of six

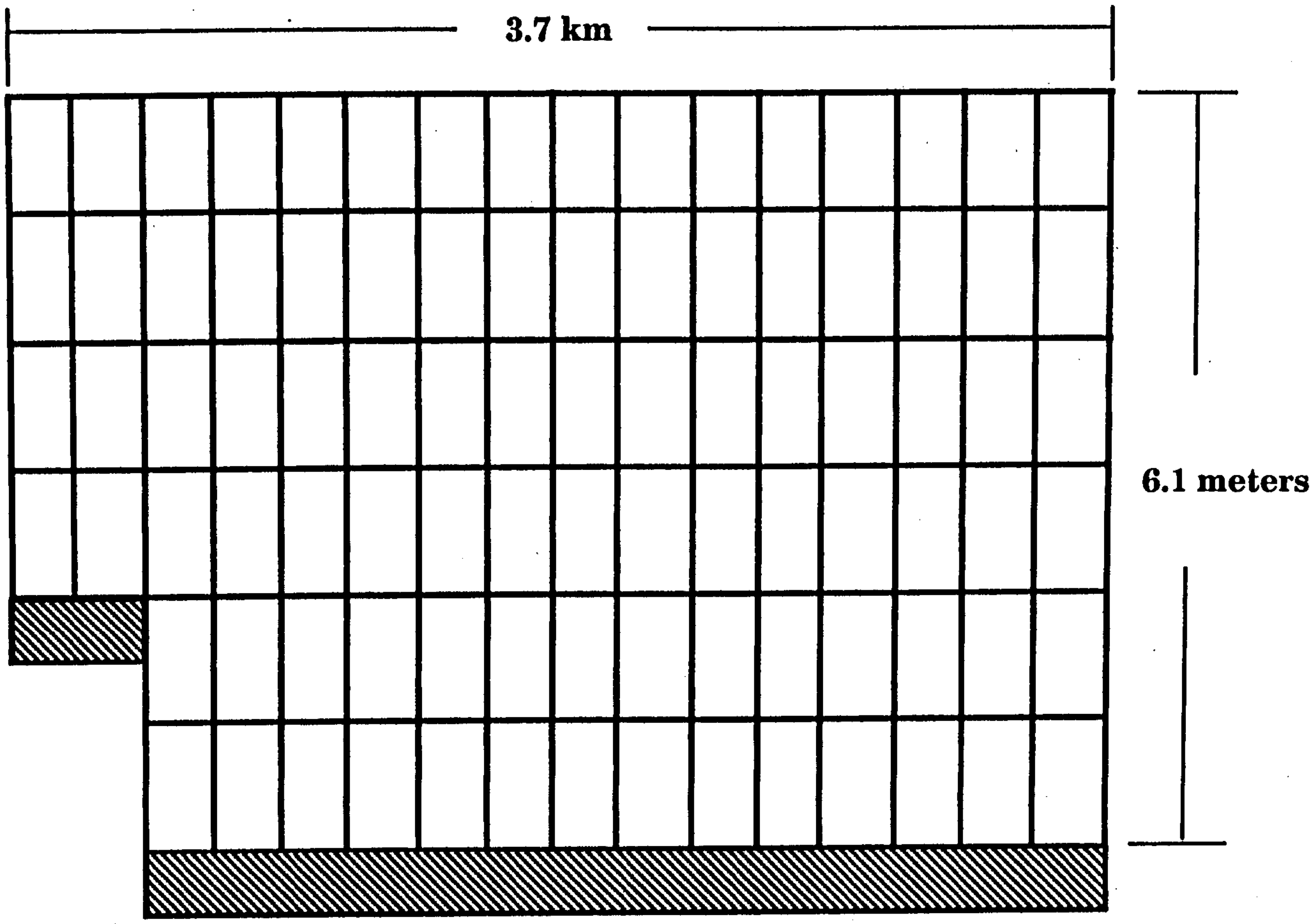


Figure 11. Two-dimensonal grid used for CE-QUAL-W2 simulations for tailings pond in Stage I. Nominal surface elevation of the pond is 233.0 meters above sea level.

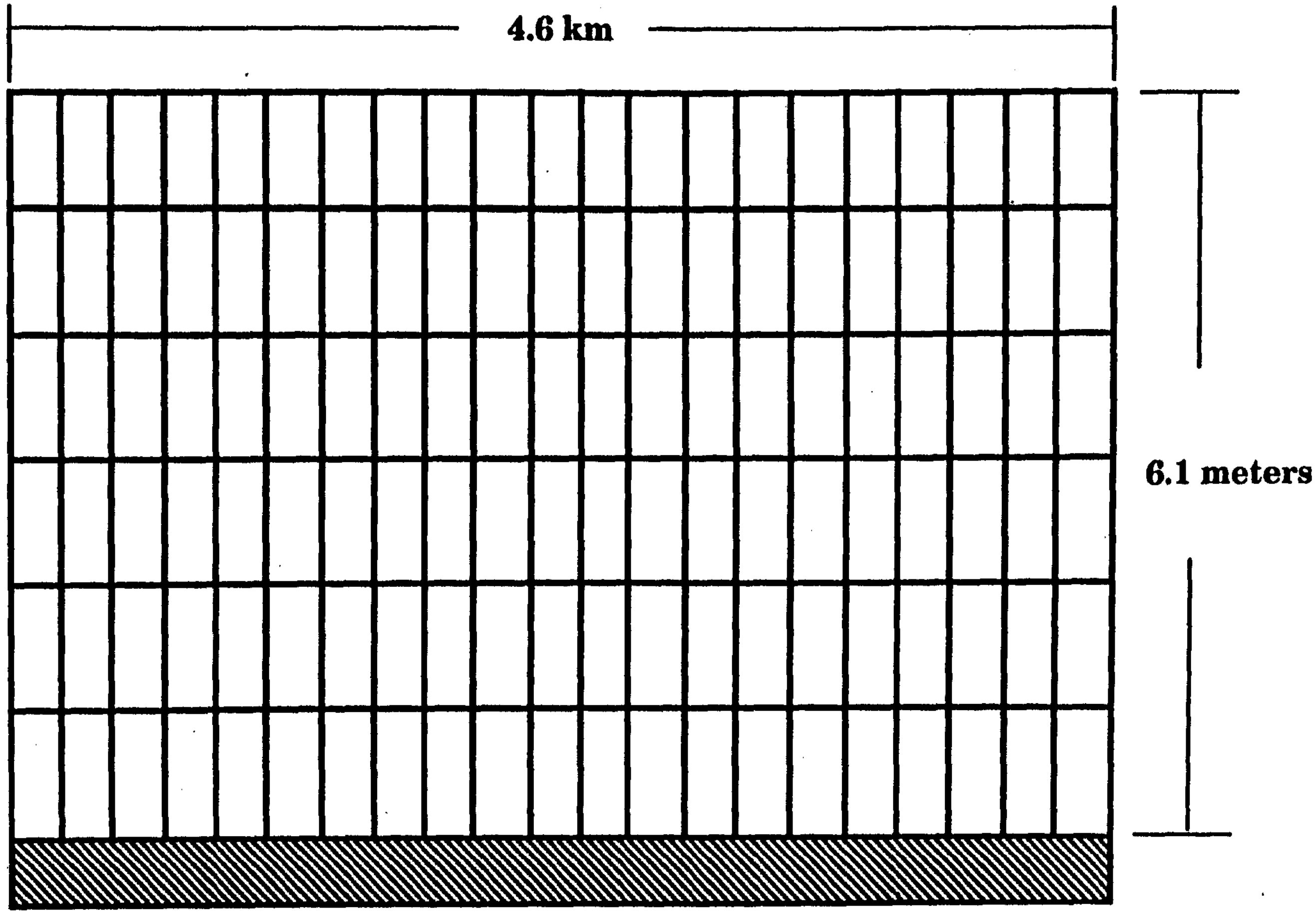


Figure 12. Two-dimensonal grid used for CE-QUAL-W2 simulations for tailings pond in Stage III. Nominal surface elevation of the pond is 279.5 meters above sea level.

locations in the tailings pond. The six locations were chosen to represent approximately evenly distributed points spanning the maximum excursion of the elephant trunk barge as indicated by the applicant (Bergstrom, 1994). Once the tailings are distributed in the water column the levels of solids and associated dissolved and particulate concentrations of metals in the water column are determined by the following processes:

- Settling rates of a single class of tailings solids
- Longitudinal mixing in the tailings pond due to various energy sources
- Vertical mixing in the tailings pond due to various energy sources including wind stress and kinetic energy
- Rates of inflow surface water, groundwater and effluent from the mining process
- Rates of outflow of recycle water from the tailings pond
- Rates of discharge from the impoundment
- Potential energy developed in the impoundment due to solar heating

2. Time and Length Scales

The waters of the proposed tailings pond are conceptualized as being comprised of a number of vertically and laterally well-mixed compartments the size of the project, or with overall length scales corresponding to the size of the tailings pond at Stage I (765 feet above Mean Sea Level) and Stage III (918 feet above Mean Sea Level). The CE-QUAL-W2 model formulation will resolve laterally-averaged concentration differences at scales of approximately the element size, which is one meter vertically and 200-400 meters longitudinally. The length scales are best professional estimates of what is needed to resolve details of the problem associated with deposition, resuspension and injection of the waste stream.

There are a number of time scales which may influence model results. The simulation time period (time step) for the numerical scheme used is of the order of minutes. The simulation time period does have an effect on simulation accuracy, but the fact that this time period is of the order of minutes does not mean the model is simulating processes in the tailings pond at this scale. The process time scales in the tailings pond and sediments which can be resolved by the simulation are determined by the rate of inflow of the waste stream, Sheep Creek and groundwater; the amount of storage in the tailings pond; the rates of vertical diffusion; the response time of the system to imposed wind stresses; and the seasonal input of energy from solar radiation.

3. Important Assumptions

As in the case of the WASP4 simulations, data from the EIS and from other relevant sources were used to implement the conceptual model. These sources provided the basis for defining important parameters used in the simulations. The fixed parameter values used in the simulations are given in Table 6. The implementation of WASP for this problem included the following assumptions:

- The surface level of the tailings pond is determined by the water balance and the geometry of the tailings pond. As described previously, two tailings pond geometry's were considered; one corresponding to the lowest level of operation, the other to the maximum level of operation.
- The bottom of the impoundment remains at a constant level, approximately 20 feet below the water surface.
- The water balance, including effluent flow rates, groundwater flow rates, outflow rates and recycle flow rates, was similar to that used by SRK (1994).
- Settling rates of particles are determined from Stokes' Law
- The tailings slurry comprises only one particle class. The particle size of this fraction is assumed to be equal to the minimum particle size reported by Knight and Piesold (1989).
- Settling velocities for the particle class can be estimated from Stokes' Law using the median diameter for each fraction, or the lowest reported value, in cases where the particle size of any fraction was not given in any reports provided by the applicant.
- Dissolved metals can be treated as conservative constituents with no leaching or partitioning between solid and dissolved.
- Dissolved levels of metals in Sheep Creek are constant.

4. Parameter Estimation

The density of the tailings, r_t, was obtained from results reported by Knight and Piesold (1989).

The fraction of important metals in the digested slurry, for both solid and dissolved phases were estimated from various data sources as described in Section VI.D. These estimates are given in Table 7.

a. Vertical Mixing

Rates of vertical diffusion in the tailings pond will be determined by the levels of available turbulent energy. In CE-QUAL-W2, these levels are estimated using a turbulent closure scheme based on the Richardson number. Coefficients of vertical eddy diffusivity are calculated by CE-QUAL-W2 using the simulated vertical density structure and the simulated vertical shear. These parameters are, of course, influenced by buoyancy flux from external sources, heat exchange between the water body and the atmosphere and wind speed and fetch. The closure scheme includes a minimum level of vertical mixing such that turbulence unaccounted for by the Richardson number closure scheme can be included. Based on discussions with the applicant (Bergstrom, 1993) and results from shallow lakes (Bowie et al, 1985), the minimum level for the coefficient of vertical eddy diffusivity for all constituents was chosen to be $1.0x10^{-5}$ meters^2/second. The net result is that the CE-QUAL-W2 simulation software will compute the vertical eddy diffusivity based on the Richardson number, but will use the computed value only if it exceeds the minimum value of $1.0x10^{-5}$ meters^2/second. If the computed value does not exceed the minimum, the minimum value is used as the estimate.

b. Initial Mixing of Discharge

The initial mixing characteristics of the discharge were assumed to be the same for the CE-QUAL-W2 base case simulations as for the WASP4 simulations described previously. That is 2.5% of the total loading, as represented by the smallest 2.5th percentile of particles were assumed to be suspended in the lowest one meter at the time of initial discharge.

c. Settling

The settling velocity for the base case conditions, (2.5% of the material initially suspended in the water column in the segment to which discharge is made) was assumed to be the same as that described for the WASP4 simulations.

The settling velocity for those scenarios designed to evaluate the SETTLE results were based on the zone settling velocity reported by Hartman Associates (1994).

d. Groundwater

Groundwater inflow was assumed to be constant and equal to 7 cfs as reported by SRK (1994).

e. Avalanches

Impacts of avalanches on the impoundment water quality were not considered, based on the rationale described previously.

f. Discharge Location

The applicant has proposed moving the discharge point periodically so that the tailings will be distributed more or less uniformly over the bottom of the pond. Furthermore, they have indicated the discharge point will move no farther than one-half mile downstream from the Sheep Creek Adit and no farther than one-half upstream from the Sheep Creek Adit. To simulate the range of outcomes, the discharge point was located for a period of two months during each year at each of the six locations representative of barge positioning strategies proposed by the applicant.

g. Effluent Characteristics

Estimates of the variability in effluent characteristics were based on the discussion in Bergstrom (1994). A lag-one Markov model was used to describe the daily loading from the mining operation according to:

$$Q(n+1) = \overline{Q(n+1)} + P_Q(Q(n) - \overline{Q}(n)) + V_Q \sigma_Q \sqrt{1-p_Q^2}$$

where,

$Q(n)$ = simulated effluent flow rate from the proposed facility on the n^{th} day of the year, $meters^3/second$,

$\overline{Q}(n)$ = average effluent flow rate from the proposed facility on the n^{th} day of the year, $meters^3$/second,

ρ_Q = correlation coefficient between effluent flow rate on the n^{th} and $n+1^{th}$ days,

V_Q = random component distributed as N(0,1),

σ_Q = variance of random fluctuations in effluent flow rate

Effluent concentrations of suspended solids and metals in the dissolved and solid phases of the discharge were kept constant using the values given in Table 7.

Stream Inflow Characteristics: Inflow temperatures for stream temperatures in Sheep Creek were generated from the lag-one Markov model

$$T(n+1) = \overline{T(n+1)} + \rho_T(T(n) - \overline{T}(n) + V_T\sigma_T\sqrt{1-\rho_T^2}$$

where,

T(n) = simulated water temperature in Sheep Creek on the n^{th} day of the year, °C,

$\overline{T}(n)$ = average water temperature in Sheep Creek on the n^{th}day of the year based on available data, °C,

= T° + DT sin(2p(n - n0)/365.)

To = annual average water temperature in Sheep Creek, °C,

DT = one-half the annual variation in water temperature in water temperature in Sheep Creek, °C,

n_o = the first day in the year when the daily average water temperature is equal to the annual average water temperature, °C,

ρ_T = correlation coefficient between temperatures on the n^{th} day and $n+1^{th}$ day,

V_T = random component distributed as N(0,1),

σ_T = variance of random fluctuations in daily averaged water temperature in Sheep Creek

Parameter values for both Markov models are in Table 14.

Daily averaged total suspended solids in the waters of both Sheep Creek and Gold Creek were treated as independent random variables distributed lognormally with means and variances in logarithmic space as given in Table 14. Daily averaged values of all metals in both Sheep Creek and Gold Creek were assumed to be below detection limits.

5. Simulations

Suspended solids and concentrations of dissolved metals and metals in suspended solids in the proposed tailings pond were simulated under the conditions representing natural variability of weather and inflow hydrology, as well as estimated variability in the discharge characteristics. The natural variability in the inflow hydrology was derived from the 30-year record of streamflow measurements at the USGS gage in Sheep Creek. The natural variability in weather data was derived from two sources. Daily averaged values of dry-bulb temperature, relative humidity, and wind speed for a three-year period were obtained from the meteorological station maintained by the applicant during the period 1991-1993. Daily averaged cloud cover data were obtained from data collected at the Juneau airport during the five years, 1987, 1989-1992.

Based on the recommendations of the panel of experts, as described previously under the WASP4 simulations, two basic scenarios were considered (Scenarios C1 and C2 in Table 15). The difference between these two scenarios was in the configuration of the pond, only. Scenario C1 uses the geometry of the pond when discharge from the pond first begins (Stage I). Scenario C2 uses the geometry corresponding to the pond configuration in the final year of operation of the mining facility. All other parameters, including coefficients of eddy diffusivity, settling rates and percentage of fines suspended initially in the water column are based on the recommendation of experts. An example of the input to CE-QUAL-W2, based on Scenario C1, is given in Appendix B.

Two additional scenarios, C3 and C4, were added after the expert panel had convened and after much of both the WASP4 and CE-QUAL-W2 modeling had been completed. These two scenarios were added in an effort to incorporate additional, but incomplete data provided by the applicant on settling rates. The data are from results obtained by Hartman Associates (1994). Krone (1994) stated that these data were not adequate to estimate settling velocities for the solids. However, in the interests of

Table 14. Important parameters and their values as used in the lag-one Markov models to simulate temperature and suspended solids in the tailings effluent and in Sheep Creek and Gold Creek.

Parameter	Description	Value
ρ_Q	Effluent flow correlation coefficient	0.6
σ_Q	Standard deviation of effluent flow	0.4 cfs
ρ_T	Correlation coefficient of water temperature	0.6
σ_T	Standard deviation of water temperatüre	1.0 °C
n_0	Temperature lag	135 days
ΔT	Variation in water temperature	2.4 °C
T_0	Annual average Sheep Creek temperature	4.9 °C
μ_{SC}	log mean of TSS in Sheep Creek	0.43558
σ_{SC}	log standard deviation of TSS in Sheep Creek	1.48
μ_{GC}	log mean of TSS in Gold Creek	1.07304
σ_{GC}	og standard deviation of TSS in Gold Creek	1.04042

evaluating all available information, an effort was made to incorporate this data into modeling scenarios. Analysis of the referenced measurements by Hartman Associates (1994), using the US Army Corps of Engineers (USACE, 1987) SETTLE model, gave results of zone settling rates for the effluent in a static test column. According to the USACE (1987), zone settling is used to determine the area of the settling pond and not for estimating the concentration of solids in the discharge from the pond. Since this was the only new analysis available at the time CE-QUAL-W2 was applied to the problem, two scenarios were developed based on zone settling rates. This was done by assuming that 100% of the material was injected into the pond at the bottom and that it settled at the rate of zone settling computed by Hartman Associates (1994). This rate was 1.77 meters/day or 2.0×10^{-5} meters/second. As shown in Table 15, all parameters and conditions other than the settling rate and the percent of material suspended in the water column, is the same as for Scenarios C1 and C2.

For all four scenarios, a 30-year simulation was performed in which the inflow hydrology was taken from the existing 30-year record of flows at the USGS gaging station. Outflow from the impoundment was based on the rule curve provided by SRK (1994). For each year of streamflow, a year of cloud cover was chosen at random from the five years of data collected at the Juneau airport and a year of dry-bulb temperature, relative humidity, and wind speed was chosen at random from the three-year record obtained by the applicant at the meteorological station in Sheep Creek valley.

Using the environmental inputs described above, CE-QUAL-W2 was used to simulate laterally-averaged vertical and longitudinal velocities, water temperature, suspended solids and a hypothetical conservative substance. The hypothetical conservative substance was used to represent concentrations of dissolved metals. The concentration of the hypothetical conservative substance in the tailings discharge was assumed to be 100 units. The concentration of dissolved metals in the outflow from the impoundment, C_{metal}, for those metals listed in Table 7, was estimated from

$$C_{metal}^{out}=C_{metal}^{effluent}X\frac{C_{conservative}^{out}}{C_{conservative}^{effluent}}=C_{metal}^{effluent}X\frac{C_{conservative}^{out}}{100}$$

Concentrations of metals in the solid phase were calculated using the ratio of metals to solids given in Table 7.

The resulting concentrations of metals in solid and dissolved phases, obtained from CE-QUAL-W2 for the 30-year simulations, were combined with a similar 30-year hydrologic

Table 15. Description of scenarios used in CE-QUAL-W2 simulations

Scenario	Pond Stage	% Fines Suspended in Water Column	Groundwater Flow (meters^3/sec)	Minimum Vertical Eddy Diffusivity (meters^2/sec)	Settling Velocity (meters/sec) Solids 1
C-1	Stage I	2.5	0.198	1.00×10^{-5}	1.50×10^{-6}
C-2	Stage III	2.5	0.198	1.00×10^{-5}	1.50×10^{-6}
C-3	Stage I	100	0.198	1.00×10^{-5}	1.50×10^{-6}
C--4	Stage III	100	0.198	1.00×10^{-5}	1.50×10^{-6}

record from Gold Creek. The fraction of flow captured from Gold Creek and diverted through the diffuser with Sheep Creek impoundment outflow was calculated using the capture ratios provided by SRK (1994). The metals concentrations estimated from the flow-weighted values of simulated Sheep Creek impoundment metals and Gold Creek water diverted to the discharge pipe diffuser in Gastineau Channel were used to develop cumulative distribution functions (CDF's) for each scenario.

This analysis resulted in daily estimates of total suspended solids and total metals in the discharge for a 30-year period. Since the analysis was developed from a conceptual model which included the natural variability in streamflow and meteorology and projected variability in the effluent characteristics, it was assumed that this 30-year sample could be used to describe the statistical properties of the simulated effluent concentrations. This was done by ordering the 30-year record of simulated results from lowest to highest and then computing the plotting position of each simulated concentration using the method proposed by Blom (1958). That is,

$$PP = (i-0.375)/(N+0.25)$$

where

PP = the plotting position
i = the rank or order of the simulated result
N= the total number of simulated results

For example, if a simulated concentration of 100 was determined to be ranked 365th, that is there were 364 results lower than 100, its plotting position in sample of 30x365 or 10950 simulated results would be:

$$PP=(365-0.375)/(10950.25)=0.033$$

Plotting positions are calculated for each of the 10950 simulated results and used to develop the empirical cumulative distribution function (CDF) for TSS and total metals in the effluent. The CDF gives the estimated probability that the effluent concentration of TSS or total metals will be no greater than some certain value. For the example given above, the estimated probability that the concentration would never be any greater than 100 would be 0.033.

6. Results

Simulated cumulative distribution functions (CDF's) for the scenarios, C1 and C2, which were based on the experts recommendations are shown in Figures 13-22. Figures 23 and 24 are the CDF's for TSS and total CN from the scenarios, C3 and C4, derived from the limited data provided by Hartman Associates

(1994). For scenarios C1 and C2, the results, though stated somewhat differently, are similar to those provided by the steady-state WASP4 simulations. This implies that the processes controlling the quality of the effluent in these scenarios are percent fines suspended in the water column, settling rate of the solids, turbulent mixing rates within the pond and the residence time of the pond.

With respect to scenarios C3 and C4 derived from the limited column settling tests, TSS levels are two orders of magnitude less than the TSS levels simulated in scenarios C1 and C2. Corresponding metals concentrations were not computed for scenarios C3 and C4. Since the total metals concentrations are highly dependent on the concentration of solids, the estimated CDF's for scenarios C3 and C4 will be less in approximately the same proportion as the CDF's for TSS.

As mentioned earlier in the discussion of the WASP4 results, the concentration of total CN does not depend on the level of solids. For this reason, the estimated CDF's for total CN are similar for all four simulations using CE-QUAL-W2 and are of the same order as the results predicted by WASP4.

G. Discussion of the WASP4 and CE-QUAL-W2 Results

Simulations of water quality in the proposed AJ Mine tailings pond were obtained using two different mathematical models, WASP4 (Ambrose et al, 1991) and CE-QUAL-W2 (Cole and Buchak, 1993). While these are complex and sophisticated models based on principles of conservation of matter and energy, model results are highly dependent on estimates of those parameters which characterize major processes of the system being modeled, such as the proportion of tailings that would become suspended and the probability that these particles would settle out. In the case of the proposed AJ Mine tailings pond, data to estimate these parameters for the specific site were incomplete or not available. It was, therefore, necessary to obtain parameter estimates from recognized experts, from the scientific literature or from environments for which there was some basis for extrapolation. The resulting estimates of expected discharge concentrations should be considered in light of what they are, which is predictions of water quality.

Due to the lack of existing prototypes, evaluation of the uncertainty in the results must be based on knowledge of the uncertainty in those parameters and those processes which are believed to have greatest influence on the predictions. In the simulations described in this report, certain parameters such as settling velocity, percent solids suspended in the water column and coefficients of eddy diffusivity were fixed or kept within a range that did not include the known variability. Instead, they

Figure 13. Estimated CDF's for Total Suspended Solids in the proposed AJ tailings pond discharge for pond Stage I and Stage III

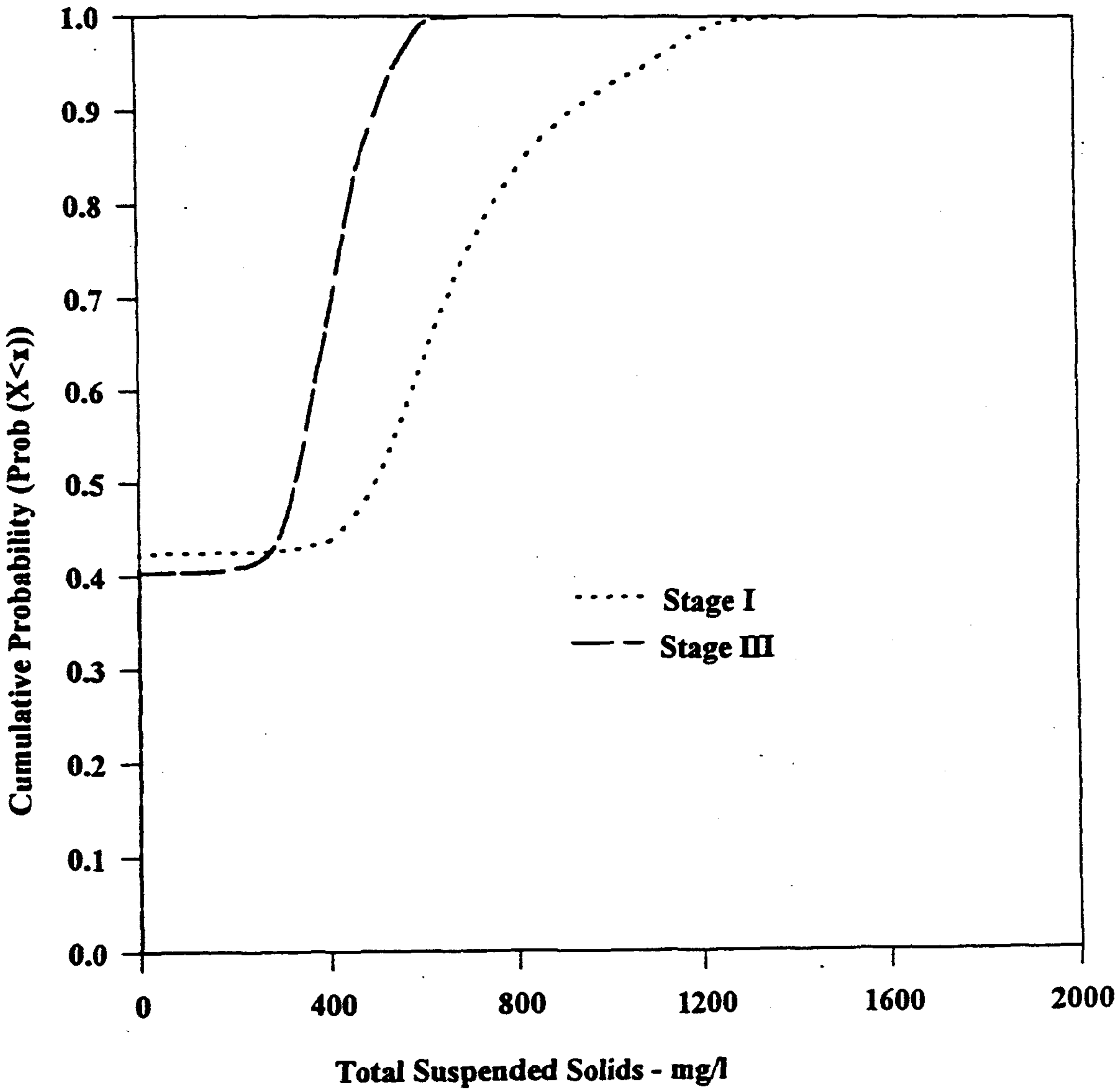


Figure 14. Estimated CDF's for Total As in the proposed AJ tailings pond discharge for pond Stage I and Stage III

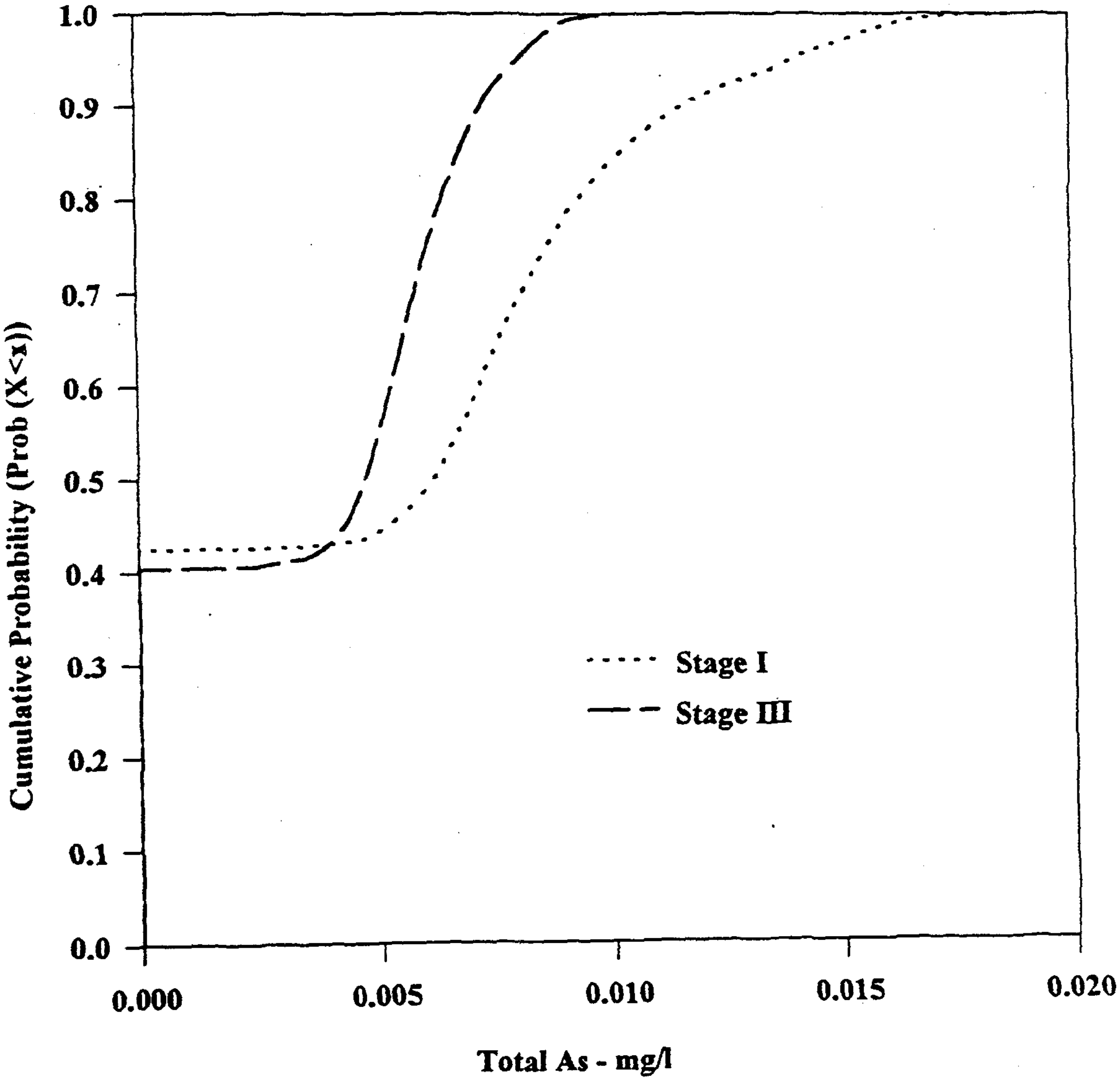


Figure 15. Estimated CDF's for Total Cd in the proposed AJ tailings pond discharge for pond Stage I and Stage III

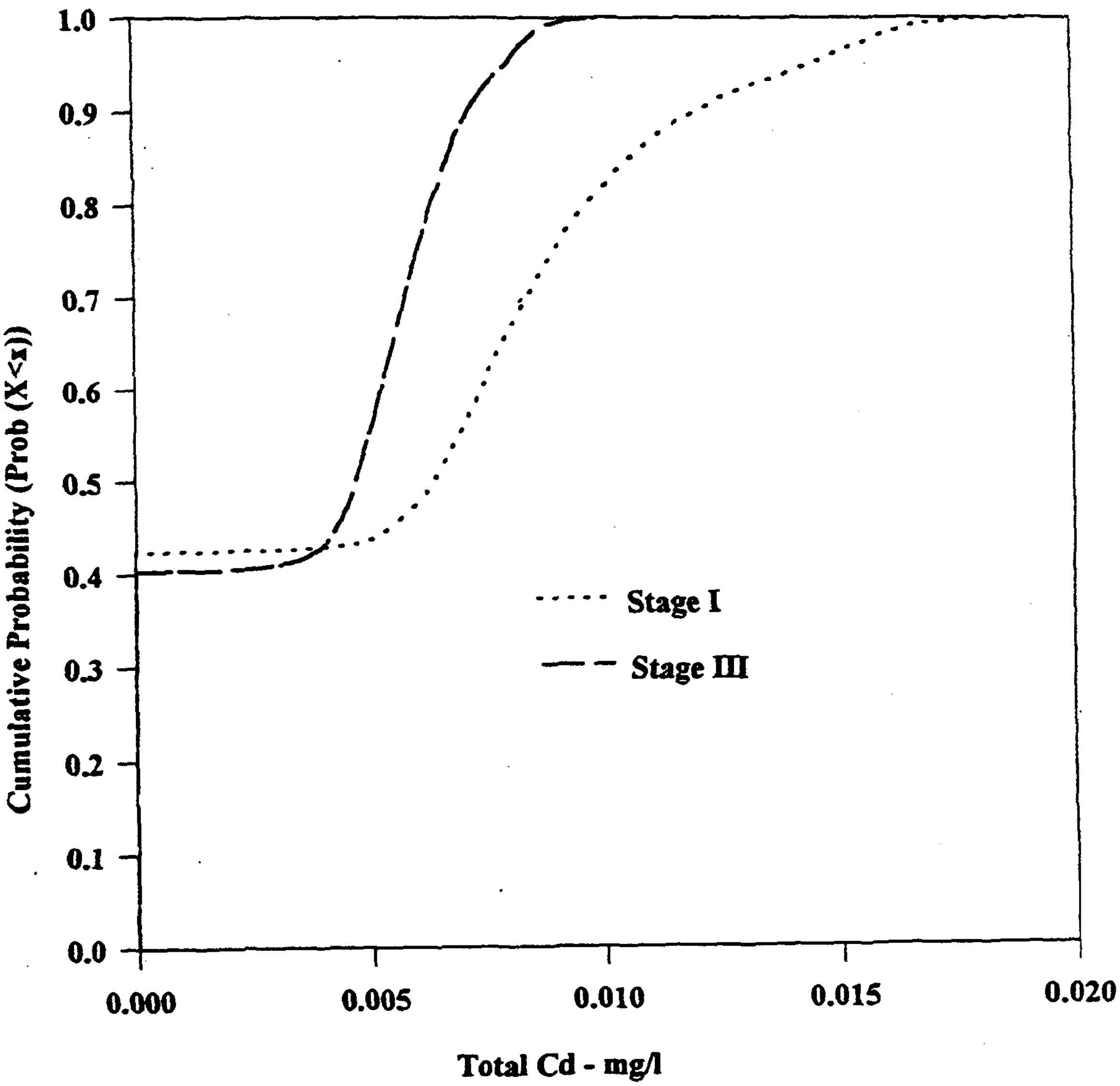


Figure 16. Estimated CDF's for Total Cr in the proposed AJ tailings pond discharge for pond Stage I and Stage III

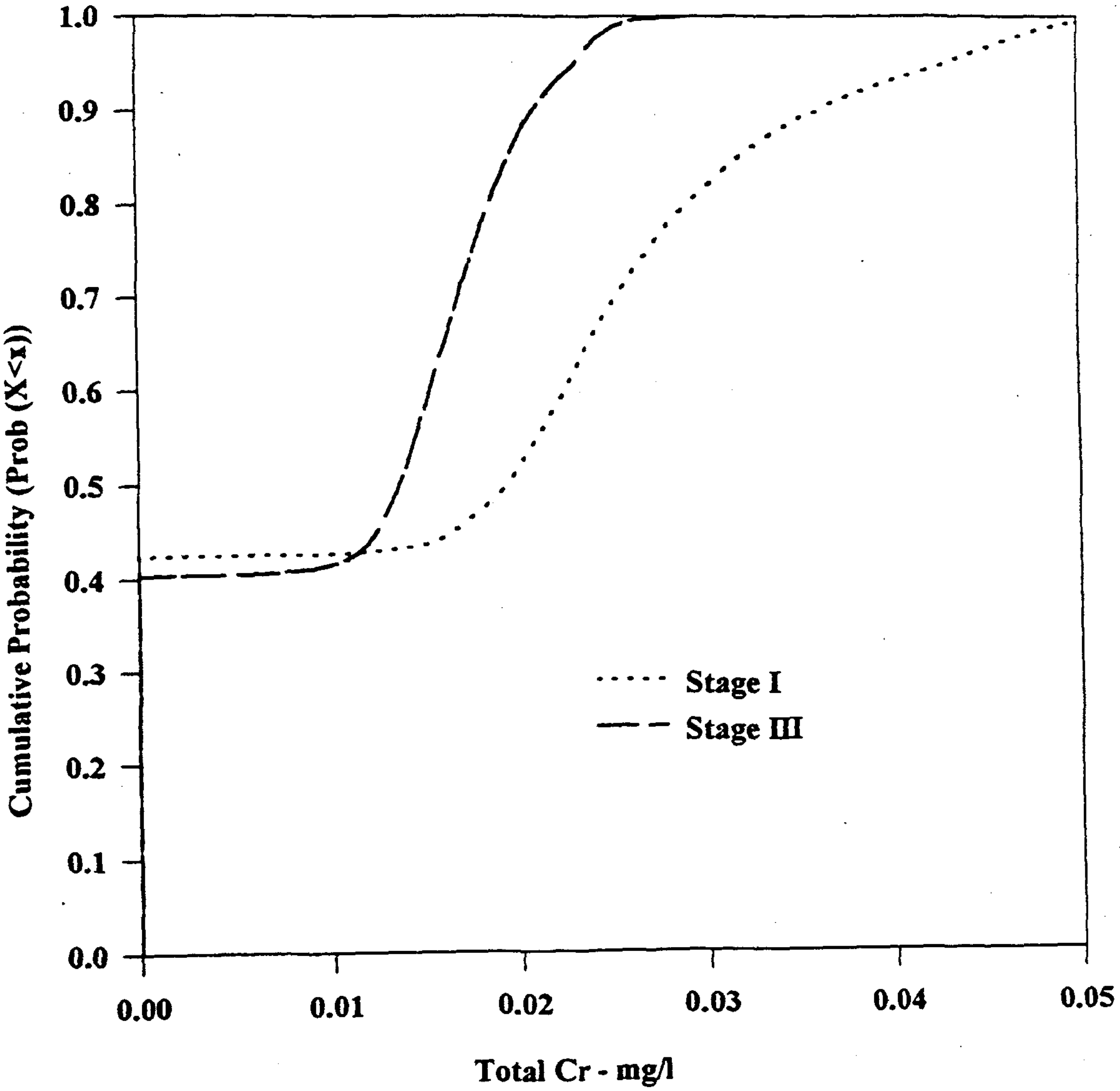


Figure 17. Estimated CDF's for Total Cu in the proposed AJ tailings pond discharge for pond Stage I and Stage III

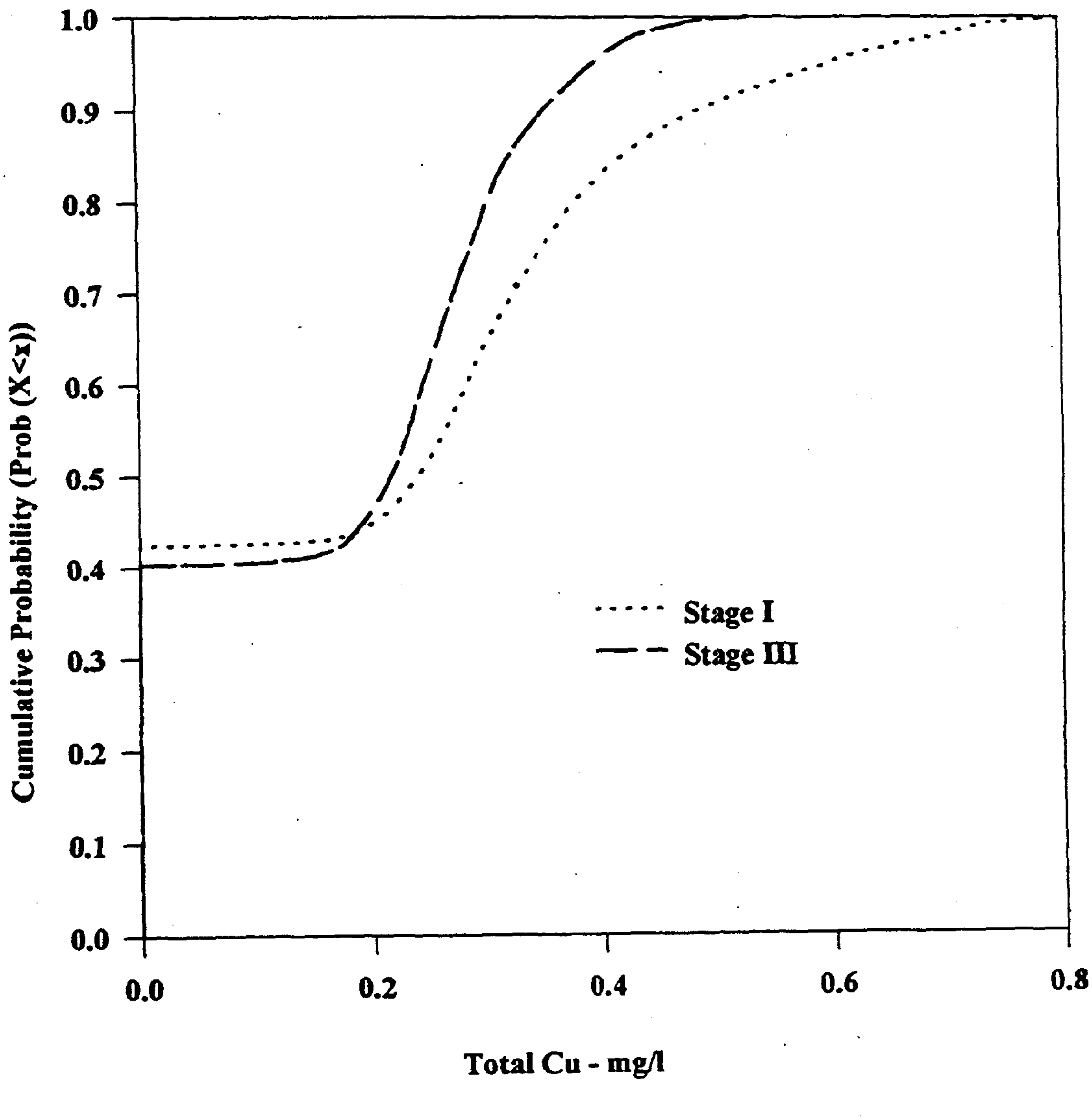


Figure 18. Estimated CDF's for Total Hg in the proposed AJ tailings pond discharge for pond Stage I and Stage III

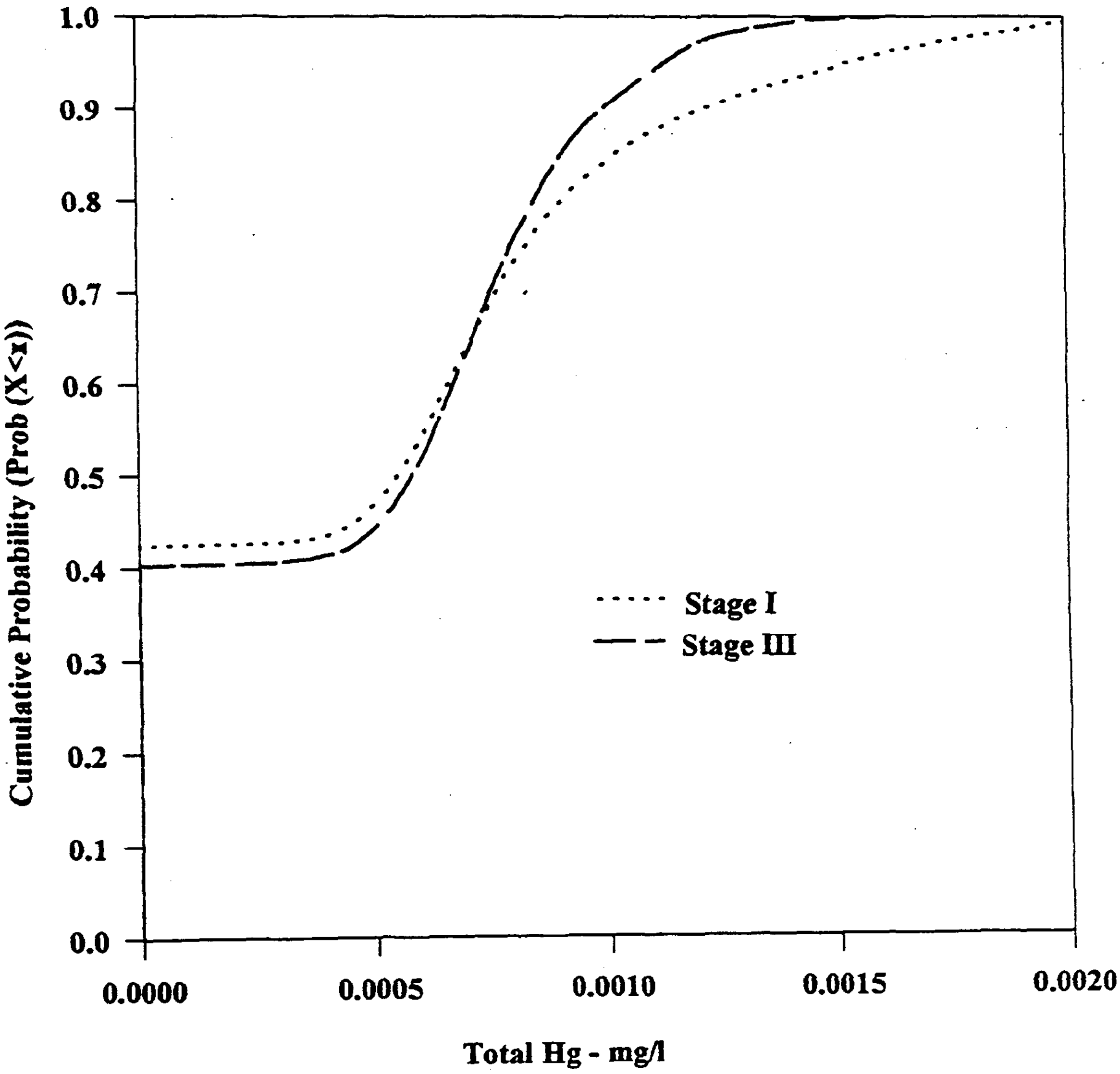


Figure 19. Estimated CDF's for Total Pb in the proposed AJ tailings pond discharge for pond Stage I and Stage III

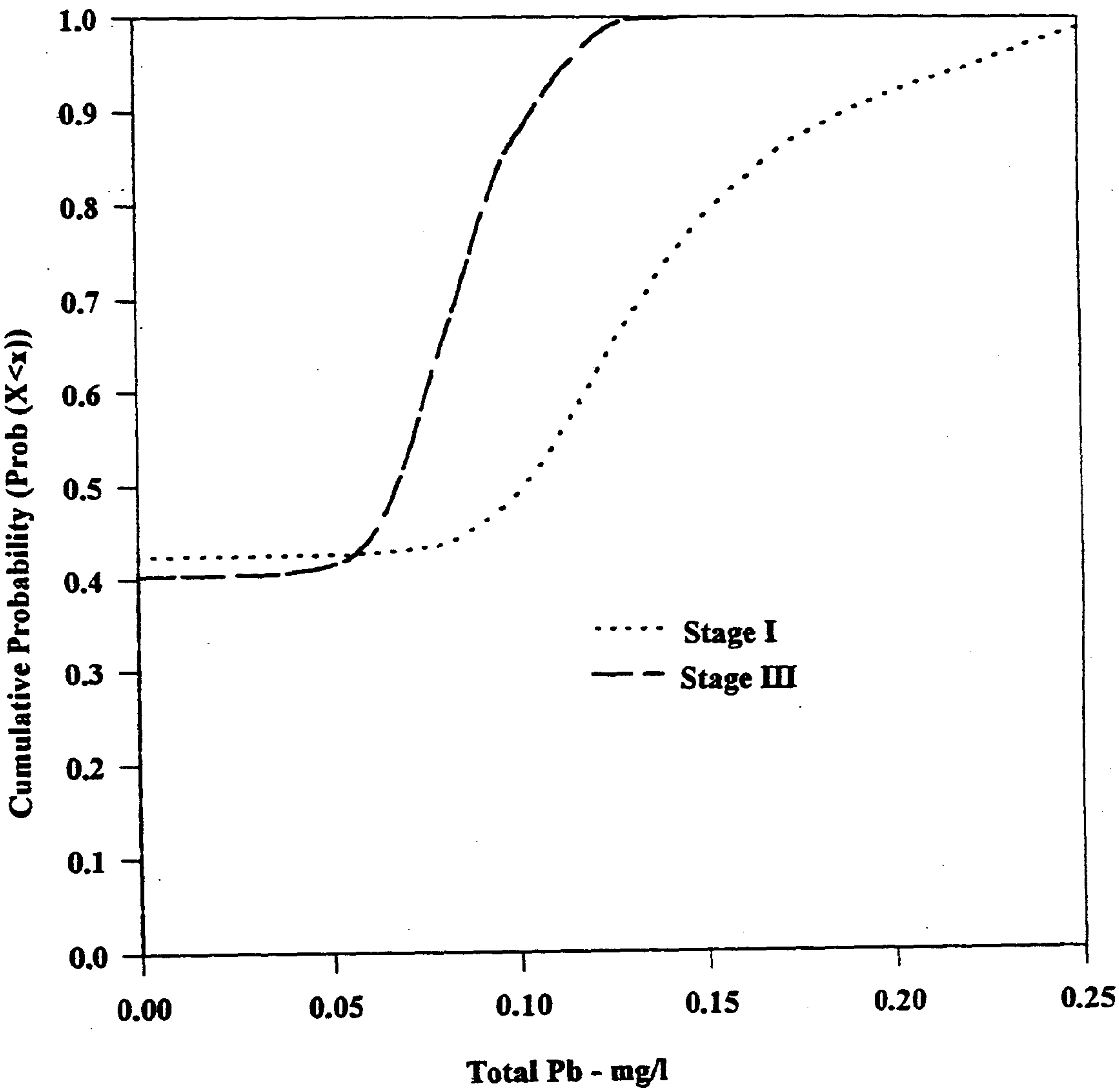


Figure 20. Estimated CDF's for Total Se in the proposed AJ tailings pond discharge for pond Stage I and Stage III

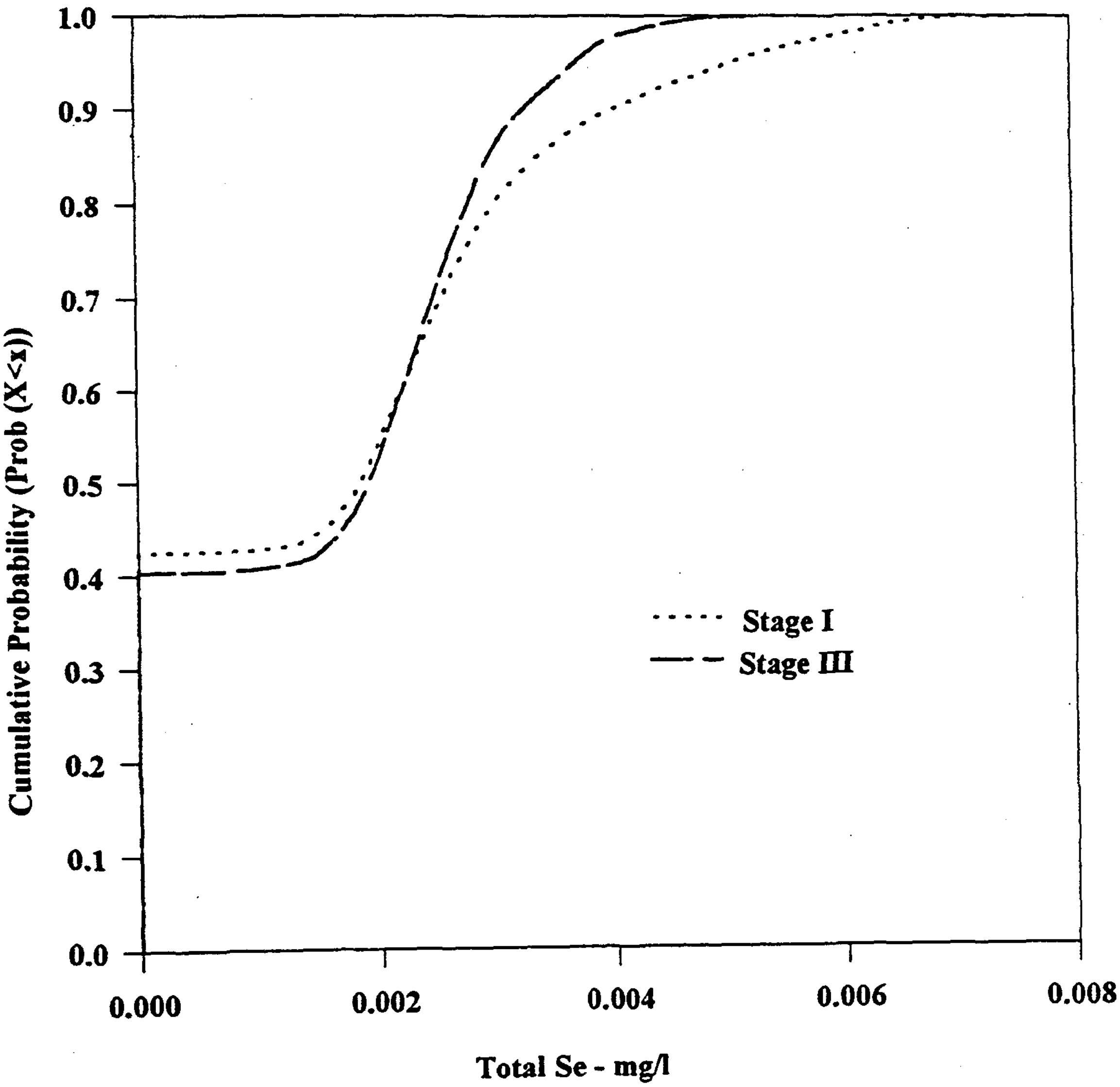


Figure 21. Estimated CDF's for Total Zn in the proposed AJ tailings pond discharge for pond Stage I and Stage III

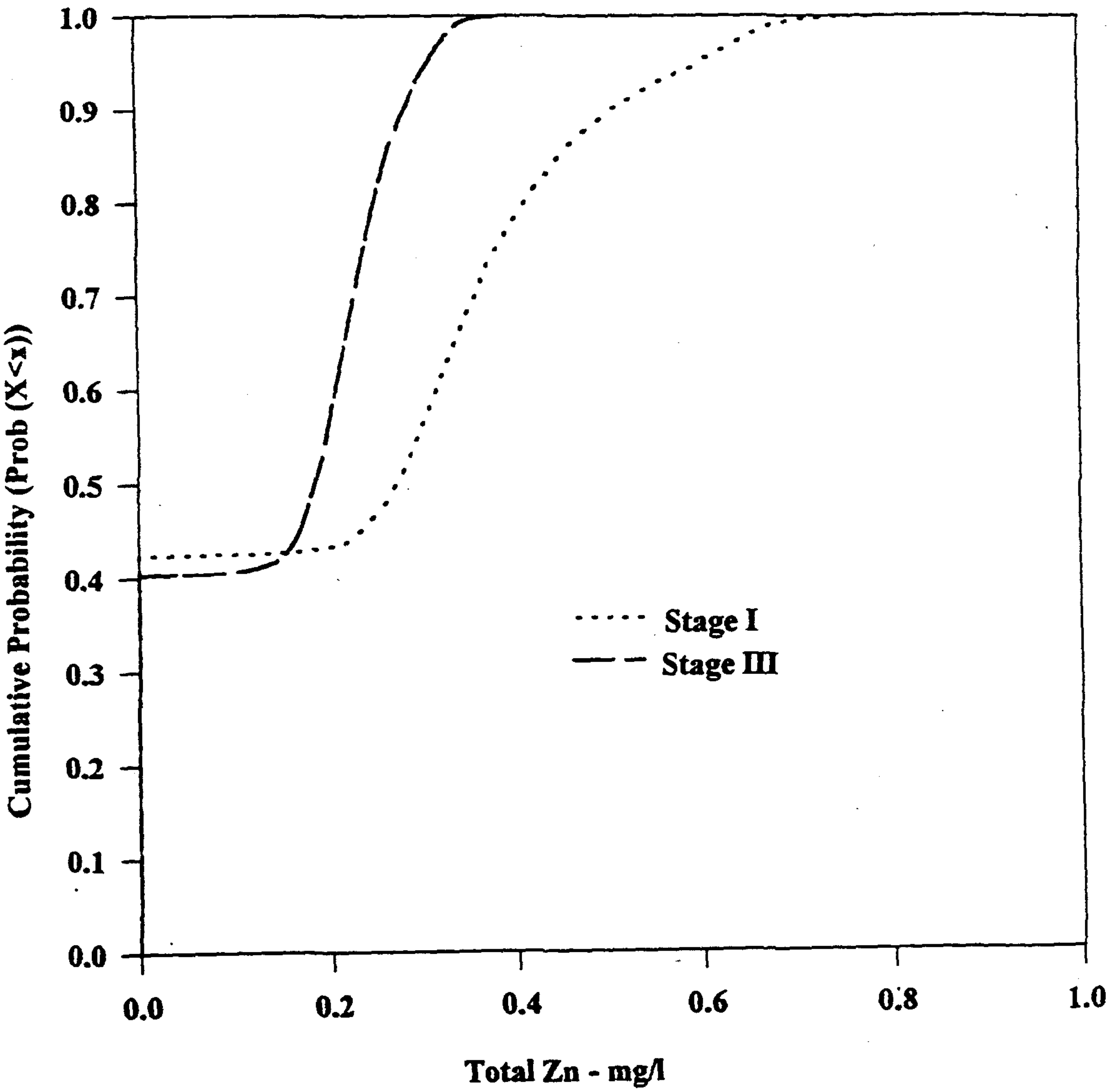


Figure 22. Estimated CDF's for Total CN in the proposed AJ tailings pond discharge for pond Stage I and Stage III

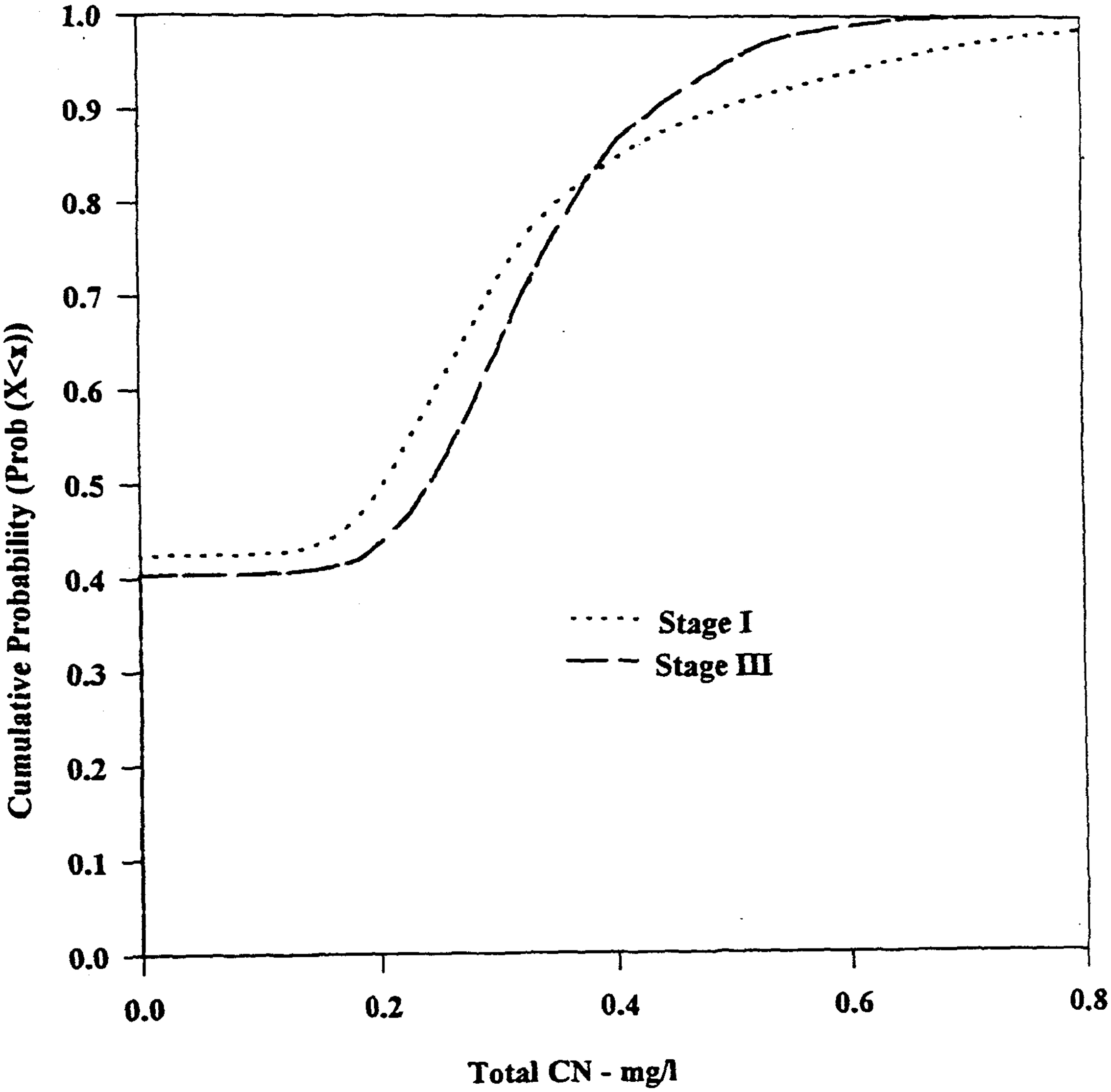


Figure 23. Estimated CDF's for Total Suspended Solids in the proposed AJ tailings pond discharge for pond Stage I and Stage III. Settling rates are derived from zone settling results reported by Hartman & Associates (1994)

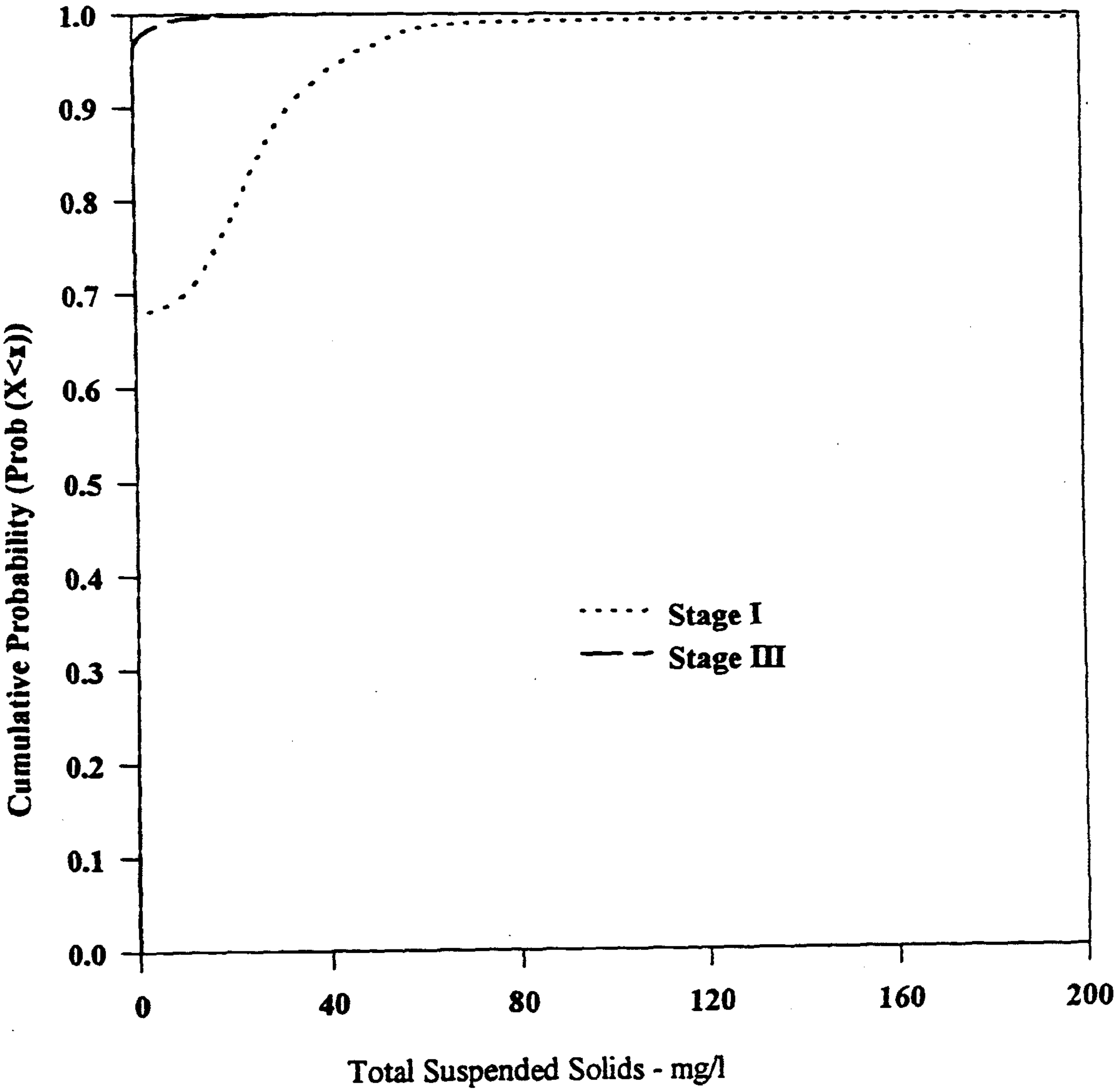
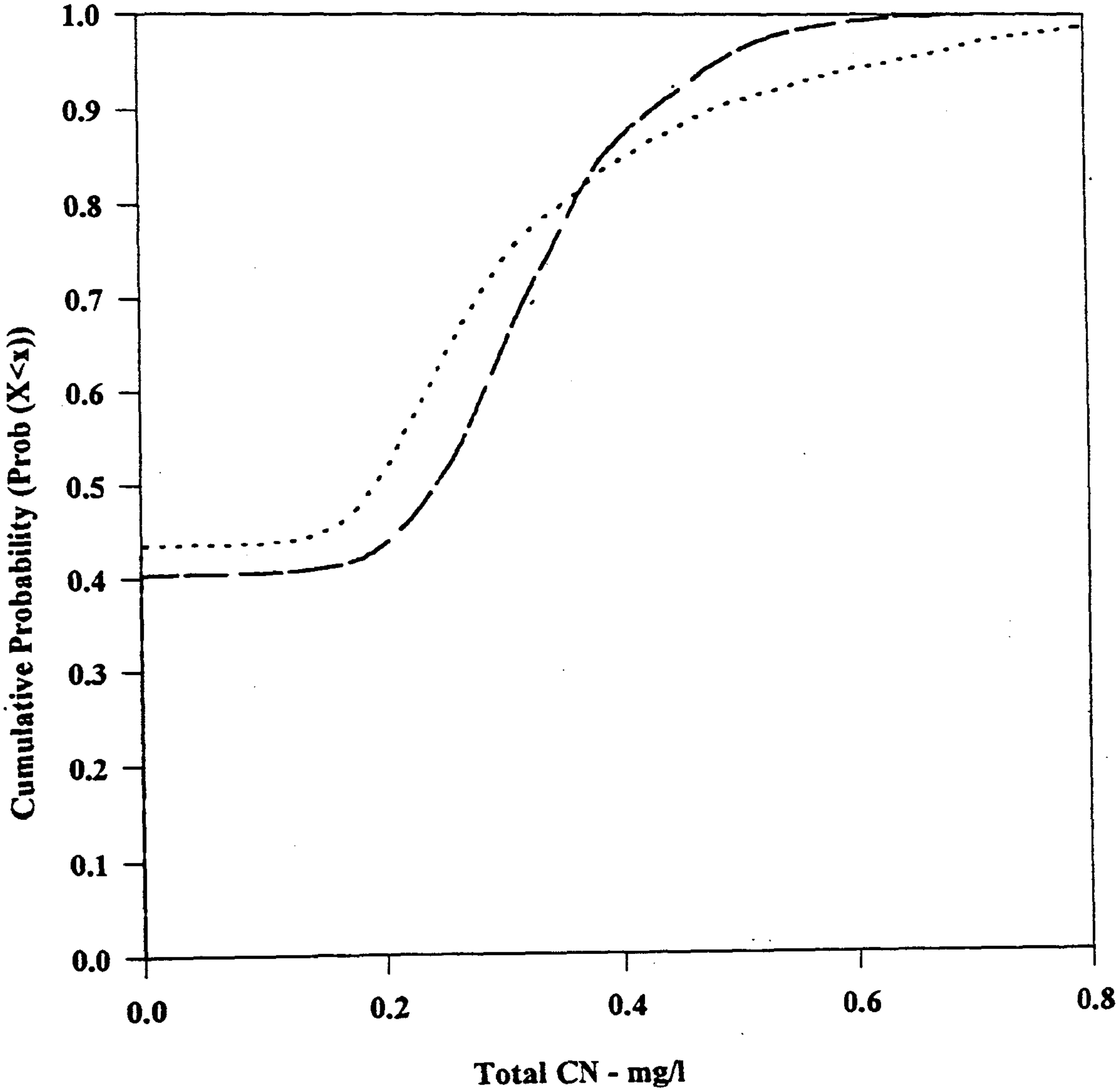


Figure 24. Estimated CDF's for Total CN in the proposed AJ tailings pond discharge for pond Stage I and Stage III. Settling rates for solids are derived from zone settling results reported by Hartman & Associates (1994)



were limited to the best available estimates from experts or from other studies. However, the results of earlier model studies by Yearsley (1992, 1993) and Woodward-Clyde Consultants (1993a) show that the results are quite sensitive to changes in these parameters. Use of different parameter values could lead to outcomes considerably different from those described in this report. It is interesting to note, that in this case, the results were not particularly sensitive to the use of a particular model as was originally expected.

While the work described here, as well as that done previously (Yearsley, 1992; Yearsley; 1993; Woodward-Clyde Consultants, 1993a) represents a thorough sensitivity analysis of certain major processes including settling rates of solids, percent solids suspended in the water column, turbulent mixing and hydrologic variability, there are certain other processes to which limited attention has been given. These include deposition and resuspension of solids, geochemistry of the pond-effluent mixture, partitioning characteristics of the discharged metals between solid and dissolved phases, hydrodynamics of the subaqueous discharge, and effects of groundwater flow through the sediments.

The deposition and resuspension issue was discussed briefly in Yearsley (1993) and Woodward-Clyde Consultants (1993a), but only in terms of limited information provided by Ambrose et al (1991). The panel of experts provided some discussion, but the discussion was not quantitative in terms of environmental conditions characteristic of this site. Deposition and resuspension are likely to be affected by the energy dissipated by the discharge itself, by the extremely high Taku winds which occur in this valley, and by slumping of the sediments as waste piles build up on the bottom and become unstable. The slumping of sediments could be an important factor in resuspending sediments near the downstream end of the proposed pond where nominal bottom depths are 20 feet and drop to 45 to 60 feet within half a mile of the proposed dam (Andrews, 1994).

The geochemistry of the effluent-pond mixture does not appear to be an important factor while the mine is in operation, since the results from the geochemically-based WASP4 simulations are not greatly different from the dilution-based results based on the CE-QUAL-W2 results. However, it does appear to be important after the mine has ceased operation, when the concentration of dissolved metals in the sediment pore water can have a significant impact on the overlying waters.

Groundwater flow through the sediments is also a process which represents a significant potential impact on the water quality of the tailings pond after mining has ceased. While many experts have discounted the possibility of groundwater flow, none of this has been supported by quantitative analysis. Based on an analysis of limited existing data (see Appendix C), the hypothesis that groundwater flow will reach the tailings pond cannot be rejected. Since the water quality of the tailings pond is quite sensitive to groundwater input after mining operations have ended, uncertainty in its ultimate impact can only be resolved by thorough analysis.

Uncertainties in processes can be invoked to justify choice of parameters which could lead to both greater or smaller estimates of the impact of this project during and after operation of the mine. The parameter ranges chosen for the simulations described here represent a year-long effort on the part of EPA Region 10 and the applicant to obtain best professional judgments on appropriate values.

H. Comparison with Other Tailings Ponds

Since this project is unique in terms of the method of discharge of tailings, characterizing the accuracy of the prediction, based on previous observations or prototypical systems, is difficult. Klohn Leonoff (1993) have, for example, reviewed the discharge water quality of a number of tailings basins in Canada and concluded that effluent quality from these can be compared favorably to the proposed AJ Mine tailings ponds. Although details provided by Klohn Leonoff (1993) are incomplete, it would appear that none of the ponds reviewed use subaqueous disposal, rather they all use spigoted or end spilling methods of discharge. This could lead to very different conditions arising from density flows and slumping which are very likely to occur under conditions similar to those proposed for the AJ Mine tailings pond.

Of equal importance in the case of the Klohn Leonoff data is the lack of quality control associated with the referenced data. There is little indication of the method of sampling, frequency of sampling or of environmental conditions prevailing during the period which samples were collected. While quality control and assurance may have been a part of these studies, it is not clear from the review conducted by Klohn Leonoff (1993) that the data were of adequate quality.

The only other tailings disposal project discussed in connection with this proposed facility is the Island Copper discharge on Vancouver Island. This is a subaqueous discharge, but the discharge is to a much larger body of water than the proposed AJ Mine tailings pond. Slumping and density flows have been observed at this discharge, but bottom slopes are much greater in the case of Island Copper than they would be in the proposed AJ Mine tailings pond. Observed levels of TSS in the receiving water at this site (Figures 6 and 7) are of the same order as those predicted for the AJ Mine pond using WASP4 and CE-QUAL-W2.

I. Conclusions Regarding Adequacy of Wastewater Treatment with Respect to Meeting New Source Performance Standards

The overall conclusion with respect to the adequacy of wastewater treatment is that the tailings impoundment alone would not provide treatment that would be adequate to meet EPA's New Source Performance Standards at the point of discharge to Gastineau Channel.

Projections of effluent quality are based in large part on concentrations of chemical parameters in the influent to the tailings impoundment. The influent values were derived from analyses of AJ mine tailings produced during pilot scale milling. Two water quality models, WASP4 and CE-QUAL-W2, were used to simulate the processes that would occur in the tailings impoundment as suspended particles settle and as metals are partitioned between solid and dissolved phases.

Table 16 compares projected effluent quality from WASP4 and CE-QUAL-W2 water quality modeling with EPA's New Source Performance Standards that would apply to the discharge from this project. The water quality model results are presented as a range for both the WASP4 and CE-QUAL-W2 simulations. The ranges reflect different assumptions used in the scenarios that were modeled.

Bold numbers indicate those NSPS parameters that are predicted to be exceeded in the discharge. These include copper and TSS and possibly mercury, which the CE-QUAL-W2 model indicates would be exceeded approximately 5% of the time. Average monthly copper limits would be exceeded under all scenarios as would TSS when particle settling velocities are determined according to Stokes' Law (scenarios 1-12 for the WASP4 model and scenarios C-1 and C-2 for the CE-QUAL-W2 simulations). When zone settling velocities are used (CE-QUAL-W2 scenarios C-3 and C-4), TSS is still predicted to exceed limits during Stage I operations at least 5% of the time.

Table 16: Comparison of Projected Effluent Quality and New Source Performance Standards

Parameter	WASP4 (ug/l except TSS))		CE-QUAL-W2 (ug/l except TSS))		New Source Peformance Standards	
	LOW	HIGH	50%	95%	Maximum Daily Limit (ug/l except TSS)	Average Monthly Limit (ug/l except TSS)
Cadmium	9.0	19.6	4.7/6.4	8.0/14.4	100	50
Copper	**240**	**480**	**218/242**	**395/599**	300	150
Lead	150	940	67.9/102.0	114/223	600	300
Mercury	0.28	0.53	.581/.550	**1.13/1.53**	2	1.0
Zinc	410	940	184/273	309/600	1500	750
TSS (mg/l)	**760**	**1800**	**328/429**	**552/1090**	30 mg/l	20 mg/l
TSS[1] (mg/l)			0/0	0/**44**	30 mg/l	20 mg/l

[1] CE-QUAL-W2 projections using zone settling numbers

VII. POTENTIAL EFFECTS OF THE DISCHARGE ON WATER QUALITY IN GASTINEAU CHANNEL

A. Introduction

This portion of the TAR focuses on the potential impacts to water quality in Gastineau Channel due to wastewater discharges from the AJ mine and tailings impoundment. The central question is whether there is adequate mixing in Gastineau Channel to dilute the pollutants in the impoundment effluent to ecologically safe levels. In this chapter, EPA reviews the available information related to mixing properties in Gastineau Channel and constructs a model to predict impacts to the channel from the proposed impoundment discharges.

B. Area Description

1. Physical Characteristics

Gastineau Channel is a coastal plain estuary which is partially stratified in the summer and well-mixed during the winter.

The area is located in southeast Alaska between 58°20' and 58°10' latitude and 134°15' and 134°24'. The channel is a tidal inlet and is an extension of Stephens Passage. Juneau is located at the head of the channel (Figure 25). The entrance to Gastineau Channel is 4 kilometers west of the convergence of Taku Inlet and Stephens Passage. At the upstream end of the estuary, a small navigation channel, passable only at high tide, connects the tide flats of Gastineau Channel to those of Fitz Cove. Meltwater from Mendenhall Glacier and Lemon Creek flows through these tidal flats into Gastineau Channel, to the southeast, and Fitz Cove to the west.

The channel is approximately 14 kilometers long from the mouth at Stephens Passage to the Juneau-Douglas bridge. The width varies from a minimum of 182 meters at the Juneau bridge to a maximum of 1,300 meters. The width of over 90% of the channel is approximately 1,200 meters. The bottom of the channel is relatively flat, the depth varies from 70 meters at the mouth to 20 meters near Juneau. Over 90% of the channel the depth varies only slightly from the average channel depth of 38 meters.

Topography surrounding the channel is characterized as steep, glacially- carved valleys to the east, with elevations ranging from sea level at the channel to 1,200 meters in the glacial areas. Glaciers surrounding the channel provide a continual freshwater source due to meltwater during summer months. The steep hills of Douglas Island border the channel to the west.

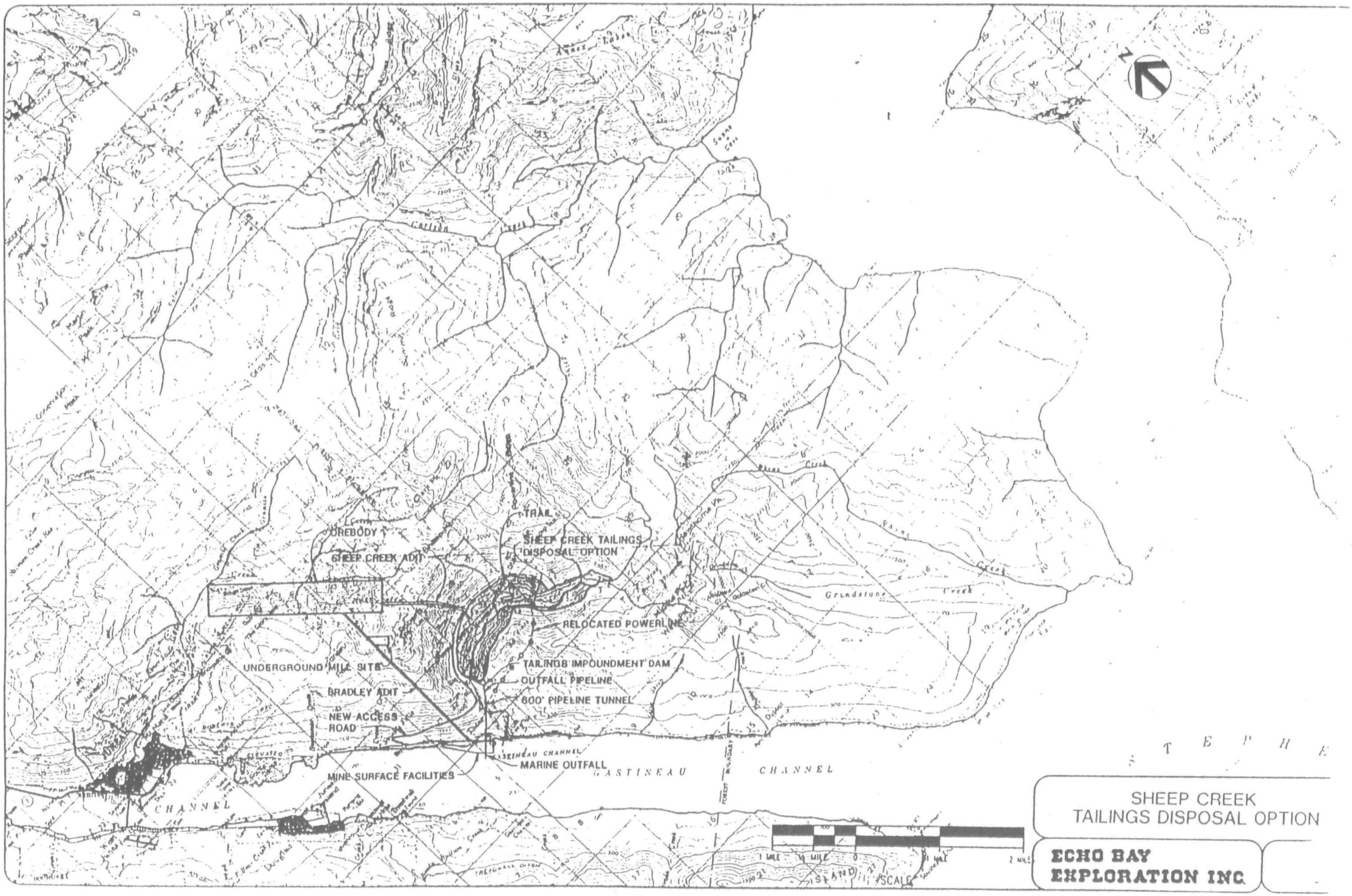


Figure 25 : Project Area

Source: Echo Bay Alaska, Inc.

2. Meteorology

Juneau is located in a coastal zone region with maritime climate. The climate is affected by storms which originate in or cross the Gulf of Alaska in a general westerly direction. Due to the moderating effects of the maritime climate, temperatures are mild for this latitude.

There is moderate variation in daily and seasonal temperatures in the Juneau area. Monthly average temperatures range from 28°F in January to 57°F in July.

This area receives an abundant amount of precipitation with a total annual average of 90 inches. The lowest monthly precipitation occurs in June, with an average of 4 inches at the downtown station. The maximum average monthly precipitation occurs in October (13 inches).

Due to the orographic effects precipitation varies greatly over short distances. The quantity of precipitation increases sharply with elevation to the east of Juneau. At Juneau Ice Field, 30 miles east of Juneau, the estimated annual precipitation is 200 inches.

Snowfall at the downtown station averages over 79 inches annually. Semi-permanent snowfields are found at elevations of 3.300 feet on the north-facing slopes of Roberts Peak and Sheep Mountain.

Surface winds are strongly channeled by the steep terrain. Gastineau Channel is oriented northwest-southeast, the same direction as the surface winds. Wind observations from the downtown station indicate that southeast winds occur 65 percent of the time and northwest winds occur 15 percent of the time. The mean annual wind speed from the airport station records indicate an annual mean wind speed of 3.7 meters per second. The monthly means range from 3.3 m/s in August to 4.3 m/s in October. There is less channelization of wind at the airport (northeast of town) than downtown (near the channel). Short periods of strong gusty northerly or easterly winds occur during the winter months.

3. Freshwater Sources

A number of creeks provide freshwater to the surface layer of Gastineau Channel. U.S. Geological Service (USGS) streamflow records are available for Gold Creek, Sheep Creek and Lawson Creek. Gold Creek has been gauged from 1918-1982 and 1985-present, Sheep Creek for the period of 1919-1973, and Lawson Creek 1970-1971.

To estimate the total freshwater discharge into the channel, Echo Bay used information from the gauged streams to estimate the total freshwater discharge into the channel (Andrews, 1991). To determine the total flows into the channel, discharge from the areas not represented by historic USGS gauged stations were estimated.

To obtain the estimates for ungauged areas, a discharge rate per unit drainage area was determined for each area based on USGS flow and drainage area records. These values were then multiplied by the ungauged areas with similar topography. The Gold Creek drainage value was used to determine the discharge for the Salmon Creek and Snowslide/Cross Bay area, the Sheep Creek value for Little Sheep/Dupont area, and the Lawson Creek value for the Douglas Island area.

Relative discharge contributions for each creek to the total flow are summarized below.

Table 17: Basin Contributions

Basin	Percent Contribution to Channel
Gold Creek	27%
Salmon Creek	25%
Cross Bay/Snowslide Creek	12%
Sheep Creek	16%
Little Sheep/Dupont Creek	16%
Douglas Island	4%

Flowrates for the above basins vary significantly over the seasons. Generally, low flows occur in winter months (November - April), while higher flows occur in the summer months (June - October). For example, average July flows in Gold Creek are ten times higher than average March flows (Comprehensive Report, Table 3-4; EBA, 1992).

4. Tidal Influence

Gastineau Channel is not connected directly to the open marine waters of the Pacific Ocean. Rather, the channel is influenced by adjoining channels, which provide a source of freshwater to the channel. The tides in this area are semidiurnal unequal, with a period of approximately 12.4 hours. The mean tidal height is 2.5 meters with an average wave height of 3.5 meters.

C. Previous Studies

Several field studies have been conducted to assess the capacity of the channel to provide dispersion and dilution of waste discharges. The following discussion and analysis briefly summarize data presented in these field reports.

1. Sewage Outfall Study 1965

One of the first studies conducted in the channel was conducted in August 1965. The purpose of this study was to evaluate the placement of the city's sewage outfall. Investigations were conducted mainly in the area near the Juneau-Douglas Island Bridge. This site was investigated because observed turbulence in this region suggested that it might provide additional mixing capacity. Measurements of temperature, salinity, dissolved oxygen and pH were obtained. A plot of the longitudinal salinity distribution near high water slack is shown in Figure 26.

Current float studies were conducted at various depths and locations during both flood and ebb tides. A rhodamine dye tracer was released at the water surface under the bridge during a flood tide on August 17 and during an ebb tide on August 20. Several rip tides were observed in the vicinity of the bridge. These tides were observed above the bridge during the flood tide and below the bridge during the ebb. Surface dye was observed sinking at the rip tide area.

2. Seawater Monitoring 1989 - 1990

Salinity is an important element in the analysis of estuaries, because salt can be treated as a conservative tracer, descriptive of the mixing characteristics of the waterbody. Salinity levels also describe a waterbody's vertical density stratification, which is an important element of discharge plume dynamics. Echo Bay conducted a seawater monitoring program during the months of May 1989, September 1989, March 1990, August 1990, and September 1990. Salinity and temperature profiles were obtained during single excursions for a given month. Salinity data for September 1989 and March 1990 are included in Appendix D1.

3. Channel Current Survey 1990

During the summer months, Echo Bay conducted a current survey near Thane and the Rock Dump Site (see Appendix D3). The mooring at the Thane site was deployed in 30 meters of water, with current meters at 12, 19 and 26 meter depths. This meter was operated from June 29, 1990, to July 21, 1990. The mooring at the Rock Dump site was deployed in 24 meters of water and operated from June 29, 1990 to August 28, 1990, with current

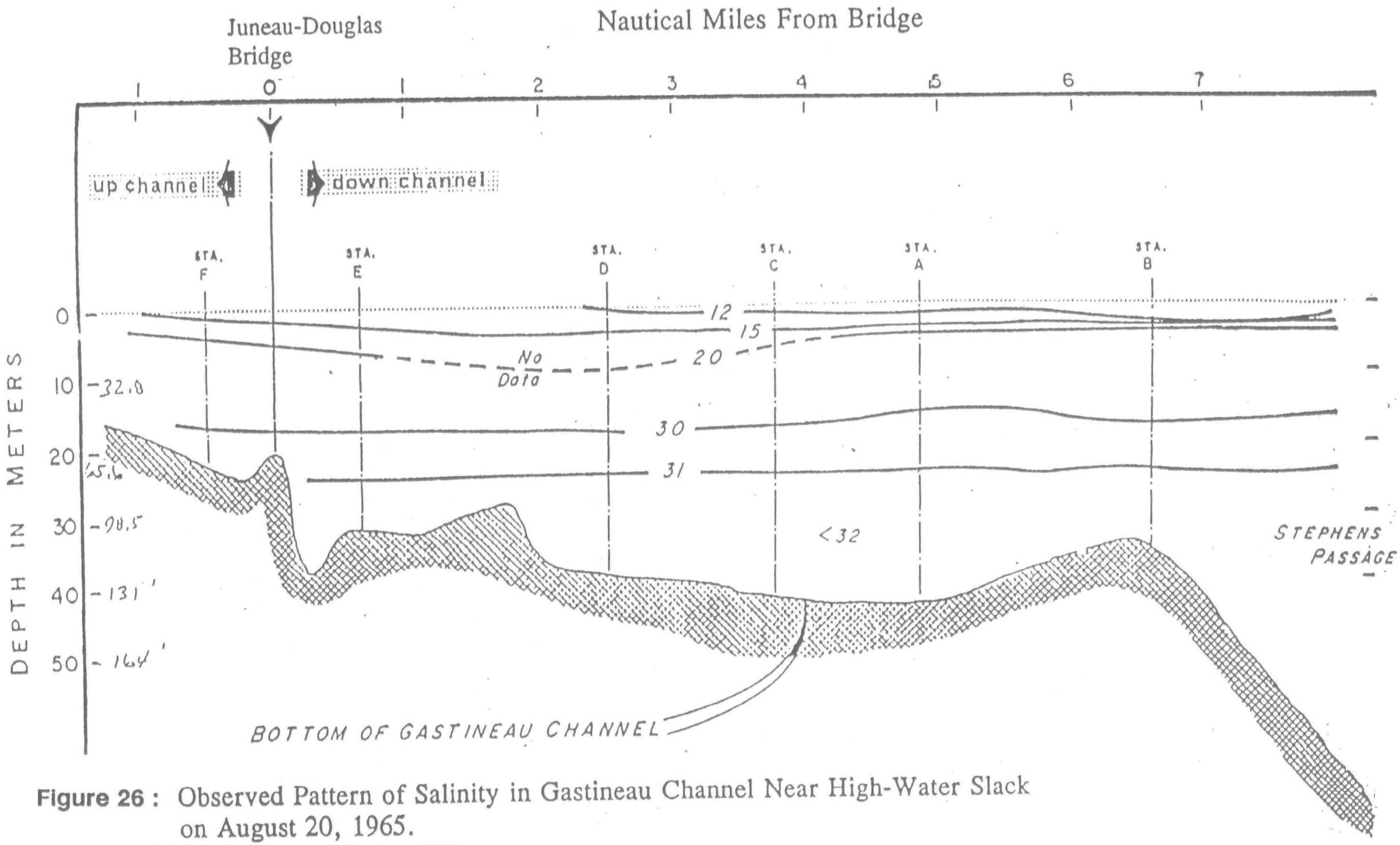


Figure 26 : Observed Pattern of Salinity in Gastineau Channel Near High-Water Slack on August 20, 1965.
(U.S. Department of Interior, 1966)

meters located at 13, 17 and 22 meter depths. Temperature, conductivity, current speed and direction were measured at 15 minute intervals. During the period of August 27 to August 30, drogued buoys were deployed near the Thane site during both flood and ebb condition. The buoys were drogued at a depth of 1.5 meters, except on August 30 when the drogue was changed to a 5 meter depth.

4. Thane Current Survey 1992

One current meter was deployed at the Thane site (see Appendix D3) for the period of February 18 through April 24, 1992. Current meters were located at depths of 5, 10, 20 and 30 meters. In addition to measuring current speed, temperature and salinity information were obtained. Drift buoys drogued at 1.5 and 3 meters and surface drifters were also deployed on two days during this survey (see Appendix D9).

5. Drift Card Study 1992

A drift card study was conducted by the National Marine Fisheries Service (NMFS) Auke Bay Laboratory in May 1992. The purpose of this study was to evaluate salmon fry migration over Mendenhall bar. Two thousand drift cards were released at three different locations in Gastineau Channel. The cards were released on ebb and flood tides on two different days. Appendix D9 contains the location and percent recovered for corresponding release stations.

6. Study Summaries and Comparison

a. Surface Flow Characteristics

i. Winter

NMFS' drift card study was conducted in May 1992. A majority of the cards released in Gastineau Channel remained in the channel, indicating a limited degree of communication between the surface of Gastineau Channel and Fitz Cove to the north.

Echo Bay's drift surveys in 1992 indicate some influence of winds on surface flows. Under steady north winds during an ebb tide on March 31, buoys were driven to the Douglas Island shoreline and then slightly up-channel to the northwest. Meanwhile, buoys at 1.5 and 10 meter depths travelled southeast in accordance with the expected tidal current.

Under southeast winds during a flood tide on April 23, bouys at all depths travelled northwest in the direction of the tidal current. The travel distance for buoys at the 10 meter depth was relatively short, while the surface buoys travelled as far as the rock dump area.

ii. Summer

Five deployments were made during ebb and flood tide during the period of August 27 to August 30, 1990. All buoys recovered during the August 1990 deployment were recovered further up-channel from the location they started. For these deployments, transport was up-channel during high winds (10-15 knots) and followed the expected tidal current during calm periods.

b. Channel Flow Properties

i. Winter

Mean current speeds in March 1992 at Thane are summarized in Table 18. Residual, along-channel velocities for the period are presented graphically in Figure 27. The depth at which the velocity profile changes direction is approximately 10 meters below the surface. Based on Table 19 the average residual velocity in the upper 10 meters is approximately .8 cm/sec, in the down-channel direction. The average residual velocity for the lower layer 20 meters is approximately .5 cm/sec, in the up-channel direction.

Table 18: Average Current Speeds at Thane in March 1992

Depth (m)	Avg Velocity (cm/sec)
5	11.9
10	9.1
20	7.4
30	7.6

ii. Summer

Average current velocities in March 1992 at Thane are summarized in Table 19. The average residual velocities for the June to August 1990 investigation are listed in Table 19. The residual data for Station A would suggest that there is a net flow down-channel at all depths; however, the surface layer of the estuary was not monitored. The up-channel flow at the shallowest meter monitored (Station B, 11.5 meters) suggests that measurements were generally obtained below a down-channel flow in the surface layer.

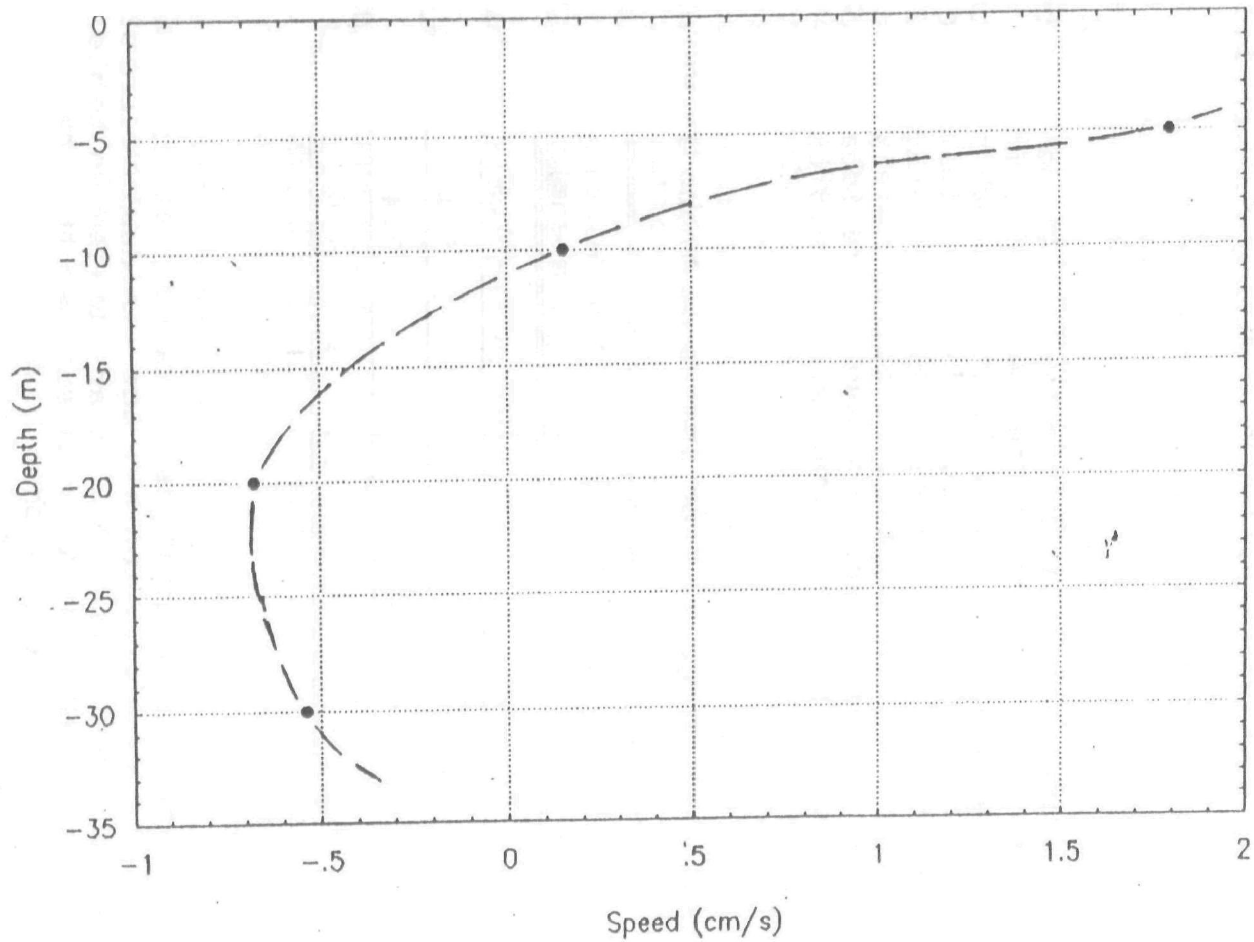


Figure 27 : **Residual along-channel velocity profile for period 18 February to 24 April 1992**

Source: Comprehensive Report

Table 19 Average Current Speeds for June - August 1990

Station A Depth (m)	Velocity cm/sec	Station B Depth (m)	Velocity cm/sec
13.5	5.3	11.5	8.7
17.0	5.0	19.0	4.4
22.1	2.2	26.5	4.7

Table 20 Average Residual Current Velocity June - August 1990

Station A Depth (m)	Velocity cm/sec	Station B Depth (m)	Velocity cm/sec
13.5	2.1	11.5	-1.8
17.0	1.6	19.0	0.5
22.1	0.7	26.5	0.8

(negative values represent down channel flow)

c. Salinity Distributions

i. Winter

Observations of salinity reported in the seawater monitoring report indicate a minimal amount of stratification for the period between November and March. For March 1991, the average salinity for stations near the mouth was 31 ppt. The average salinity for stations up channel was 29 ppt. The March 1990 records indicate an average salinity of 29 ppt, with this value relatively uniform up-channel. The November profile indicates an average salinity of 31 ppt near the mouth, and an average up-hannel salinity of 29 ppt.

Salinity profiles obtained during the February-April 1992 investigation indicate a relatively uniform profile for February with an average salinity of 30.2 ppt. The March and April profiles indicate a stratified layer with increased surface temperatures. It appears that over the data collection period of February 18 - April 24, 1992, stratification was increasing. This increasing degree of stratification was probably due to an increase in freshwater flows. The monthly average freshwater flow for Gold Creek for March 1992 was 68 cfs, which is over 5 times greater than the long-term average March flow.

ii. Summer

Profiles for the summer period were available for August 1990 and September 1989. The August profiles show a distinct stratification extending to a depth of approximately 10 meters. The minimum salinity observed in the surface layer was 9 ppt. Salinity in the surface layer appears to increase in the up-channel direction. This phenomenon was also observed in salinity data from the 1965 study. The September 1989 profiles show a similar trend, with a stratified layer extending to a depth of 10 meters. The September data exhibits the same increase in the surface salinity with up-channel distance.

Salinity data from the June-August 1990 current mooring were obtained at depths generally below the stratified layer. The shallowest meter was located at a depth of 12 meters.

7. Physical Description Summary & Interpretation

In summary, there appear to be two distinct periods and mixing conditions in Gastineau Channel. The winter period between November and March is characterized by relatively low freshwater tributary input, colder surface layer, and well mixed vertical conditions.

The summer period, between April and September, may be characterized by a five-fold increase in tributary inflow, a stratified upper layer consisting of less saline and warmer water, and an increase in salinity in the upper layer in the up-channel direction. The period of maximum stratification appears to coincide with the maximum glacier melt and subsequent runoff. Based on the pattern of salinity increasing in the up-channel direction, it appears a significant portion of the freshwater entering the channel during the summer period originates in Stephens Passage.

The general circulation observed during both winter and summer periods is directed up-channel (towards Juneau) at depth and down-channel in the upper layer. During periods of high winds, surface currents are enhanced either in the down- or up-channel surface flows. The predominant direction would appear to be up-channel influence due to the frequency of the up-channel wind occurrences (65%). This effect may explain the March 1992 drift buoys moving up-channel at the surface and down-channel below 1.5 meters.

The primary mixing mechanisms for an estuarine system identified by Ippen (1966) include: a) the effect of the tide; b) the effect of gravitational forces due to density variations between freshwater from upland sources and saline water from the sea; c) the gravitational force needed to produce a net seaward transport of freshwater; and d) the Coriolis forces and centrifugal forces inducing transverse fluid motion due to the rotation of the earth and any curvature of the estuary. From the above physical description of the Gastineau Channel system, the primary mixing mechanisms at work are the tides, the seaward transport of freshwater, and wind-driven advection at the surface.

D. Screening Analysis for Water Quality Impacts

1. Analysis by Echo Bay Alaska

Analyses provided by Echo Bay have employed the use of a single cell or "box" model to assert that long-term (or far-field) mixing is not significantly limited. A diagram of the single cell concept is shown in Figure 28. If this far-field mixing is assured, near-field dilution modeling would be useful in ascertaining the dimensions and dilution in any mixing zone (Comprehensive Report; EBA, 1992).

For a number of reasons, the box model described in the Comprehensive Report does not represent a worst-case view of the far-field impacts. First, this model does not address the potential for stratification to trap the discharge plume in the lower waters of the channel, which tend to migrate up-channel. Another problem stems from the use of average current speeds, artificially set in one direction, to move water into the channel. Residual currents are a more appropriate measure of minimum net transport into and out of a waterbody over time, because residual currents represent the net current after the flow reversal effect of tidal currents is filtered out of the data. Finally, the box model assumes that all waters within the box are mixed instantaneously, which neglects the potential for relatively higher impacts in the upper channel than in the lower channel due to relative distances from the tidal source (Stephens Passage).

2. Alternative Screening Analysis

As background to conducting a more detailed analysis of mixing in Gastineau Channel, EPA used the residual current speed documented in the Comprehensive Report (Figure 27) to estimate the available dilution flows at a given site in the channel. In this case, EPA compared the projected effluent flow from the AJ mine with the estimated cross-sectional flow in Gastineau Channel, calculated as the net flux passing through the channel cross-section by virtue of the residual current (Figure 29).

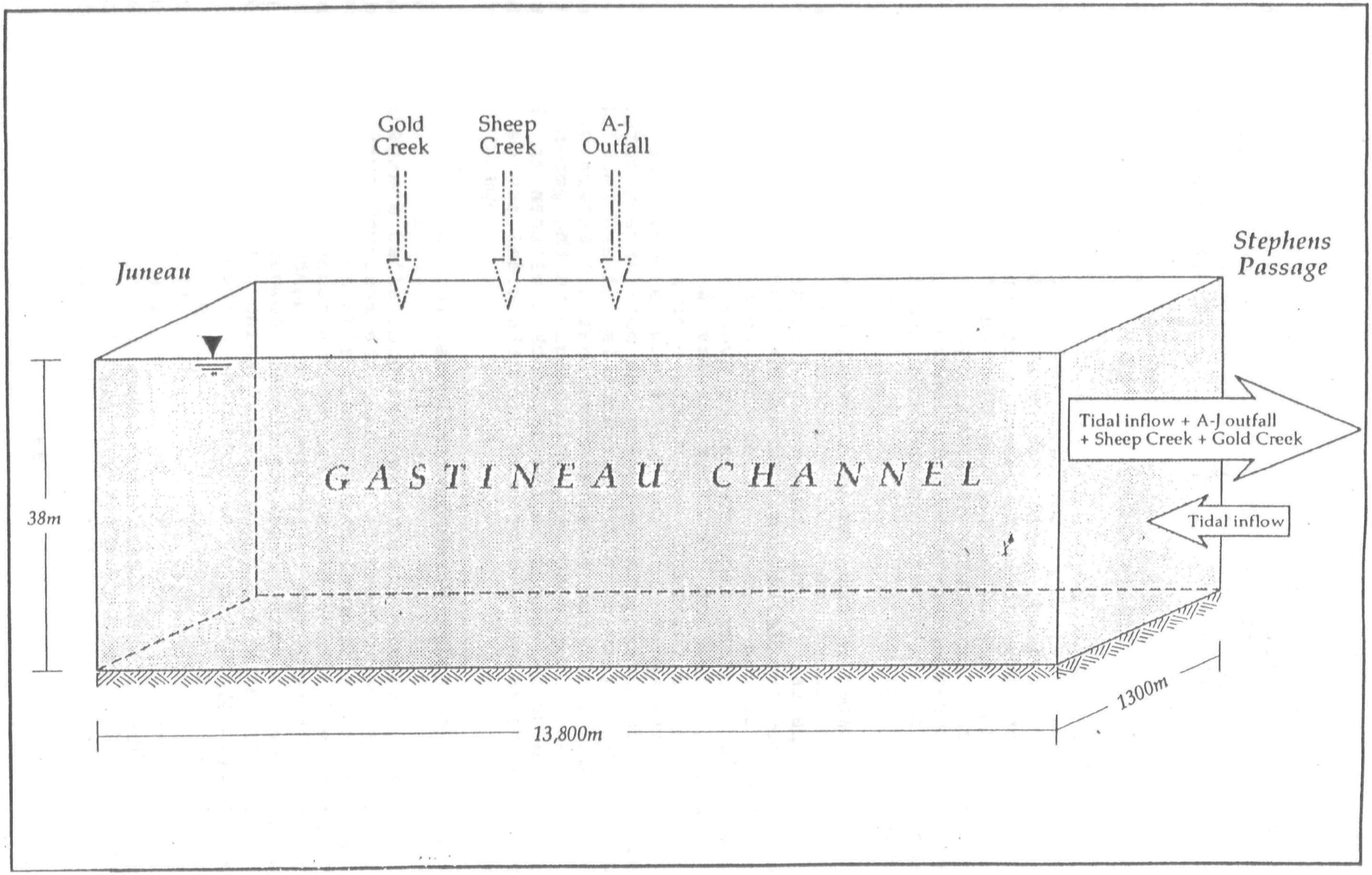


Figure 28 : Conceptual Diagram of Single Cell Model

Source: Comprehensive Report

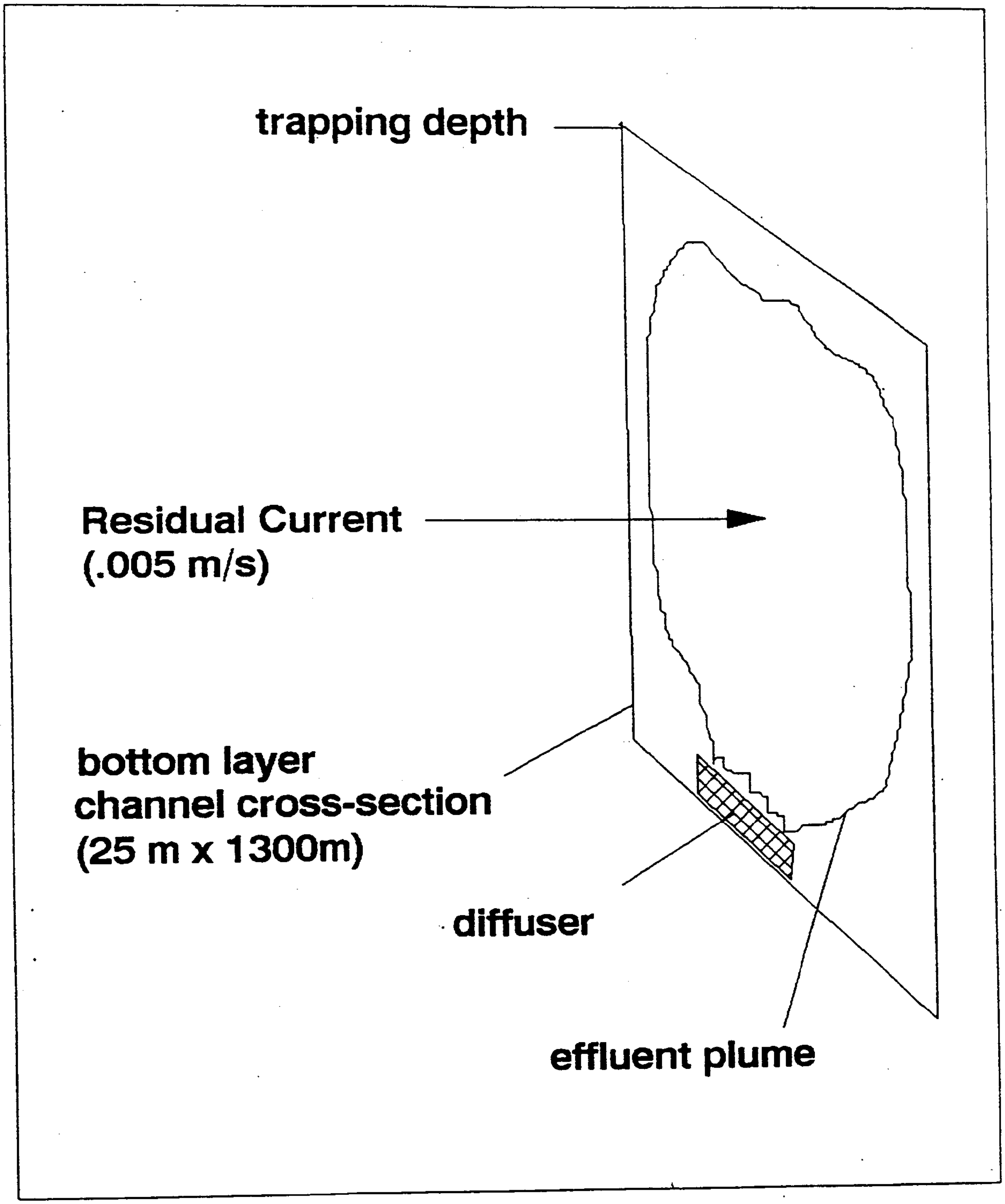


Figure 29 : Conceptual Framework for Volume Flux Approach

The volume flux is calculated as follows:

AJ Maximum Discharge Flow = 200 cfs = 18.6 m^3/sec

Channel Cross Section (below 10 m) = 25 m x 1300 m = 32,500 m^2

Avg Residual Current (below 10 m) = .5 cm/sec = .005 m/s

Volume Flux = (Cross section) x (avg residual velocity)
= 32500 x .005 = 162.5 m^3/sec

Dilution at Max Flow = (Volume Flux)/(AJ Discharge Flow)
= 162/18.6 = 9

These calculations suggest that flows through the channel at slack tide (residual currents only) would dilute the effluent by a factor of 9, significantly less than the estimated initial dilution of approximately 100:1 described in Echo Bay's Comprehensive Report. While the tides would be expected to provide additional dilution in the system, this contribution would be expected to dwindle with increased distance from the channel entrance.

The above screening calculations, coupled with concerns about the assumptions of the Echo Bay box model, leads EPA to conclude that the analyses to date do not adequately evaluate the potential for accumulation of pollutants in the upper part of the channel. As a result, EPA pursued a more detailed analysis of the problem using the WASP4 model.

E. Analysis using WASP4 Framework

1. Introduction

The Water Quality Analysis Simulation Program Version 4.32 (Ambrose et al., 1991) is a dynamic compartment modeling system that has been widely used in the United States to predict water quality responses to pollution. It simulates transport and transformation of conventional and toxic pollutants in surface waters using finite difference techniques. The model is maintained and supported by the USEPA Center for Exposure Assessment Modeling in Athens, Georgia.

The WASP4 Model developed for Gastineau Channel was calibrated using salinity to determine reasonable mixing conditions. Using these mixing parameters, the model was then run using average and worst-case pollutant loadings from the impoundment to estimate pollutant concentrations in the channel. These pollutant concentrations are then compared to applicable water quality standards for the channel.

2. Model Structure

a. Spatial Scale

The WASP4 parameters for the Gastineau Channel problem were estimated on the basis of available monitoring information, most of which has been submitted by Echo Bay Alaska, Inc. In this analysis, the channel is divided into 8 longitudinal sections. Salinity data and a residual tidal velocity graph from Echo Bay's Comprehensive Report indicate that the channel is stratified to approximately 10 meters depth. To capture this feature, the model structure includes two layers of cells in the vertical, one from the surface to 10 meters and the second from 10 to 35 meters.

The model contains 16 (2 X 8) cells or parcels of water linked together in the generalized shape of Gastineau Channel. A diagram of the model grid is shown in Figure 30.

WASP4 is designed such that effects on a scale smaller than the cell dimensions are not predicted. For example, localized impacts around the diffuser are underpredicted by the model, because loadings are assumed to completely mix in the cell volume, which is significantly larger in scale than the diffuser dimensions. A more refined cell grid would be required to focus on smaller areas of concern.

b. Time Scales

The characteristic time scales for this problem are estimated below. The basis for the listed parameters is described later in this report.

Horizonal Diffusion (head of channel):
$T_{diff} = L^2/k = (1300 \text{ meters})^2/(.16 \text{ m}^2/\text{sec}) = 122 \text{ days}$

Horizonal Diffusion (mouth of channel):
$T_{diff} = L^2/k = (1300 \text{ meters})^2/(1.0 \text{ m}^2/\text{sec}) = 20 \text{ days}$

Vertical Diffusion (March):
$T_{diff} = L^2/k = (12.5 \text{ meters})^2/(.00001 \text{ m}^2/\text{sec}) = 180 \text{ days}$

Vertical Diffusion (September):
$T_{diff} = L^2/k = (12.5 \text{ meters})^2/(.000001 \text{ m}^2/\text{sec}) = 1800 \text{ days}$

Advection (based on tidal height):
T_{advect} = mean depth/tidal depth = (35 m)/(5 m/day) = 7 days

Advection (based on longest path & residual tidal velocity):
T_{advect} = 13,800 m/.005 m/s + 13,800 m/.008 m/s = 52 days

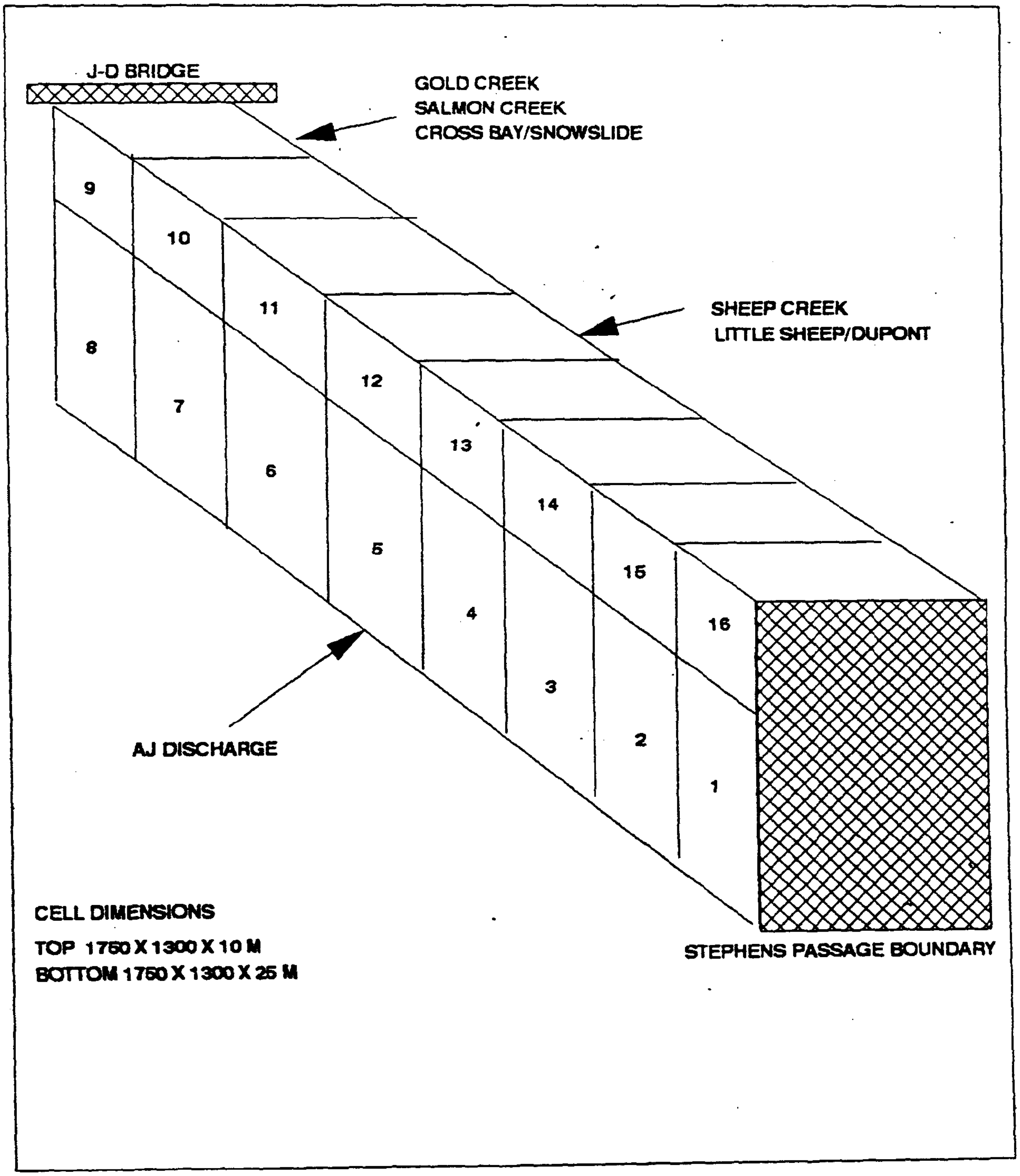


Figure 30 : WASP4 GRID AND BOUNDARIES FOR GASTINEAU CHANNEL MODEL

The time scales above offer a picture of the relative importance of a given parameter or factor on mixing. For example, the relatively short time scales for transport of bay waters through tidal advection and residual advection (gravitational), compared to the larger horizontal diffusion time scale, support the notion that advection is the predominant mixing process in Gastineau Channel in late summer. Also, reduced vertical mixing would be expected in September due to a stronger density gradient, resulting in a longer time scale for vertical mixing.

3. Model Assumptions

The following is a list of major assumptions of this modeling assessment:

1. Tidal flows decrease linearly in the up-channel direction, based on continuity and the speed of the tidal wave.

2. Tidal flows are uniformly distributed in the vertical dimension.

3. Residual flows move up-channel at depth and down-channel at the surface. There is no residual flow in late winter.

4. Wastewater is trapped below 10 meters in summer and reaches the surface in winter.

5. Wastewater is completely mixed in the first cell upon discharge. Inflows to each cell are completely (and instantaneously) mixed in both horizontal and vertical dimensions.

6. Mixing beyond the Juneau-Douglas Bridge is not significant. Freshwater inflows at the bridge boundary are included.

7. Horizontal diffusion decreases from the mouth of the channel (Stephens Passage) to the head (Juneau-Douglas Bridge) in proportion to the square of the distance from the mouth.

4. Solution Approach

The model was calibrated to salinity data observed in September 1989 and March 1990 to represent stratified and non-stratified conditions. Salinity levels were predicted using estimates of seasonal flows to the system. Using fixed tidal and residual advective flows, certain model parameters were adjusted to most closely approximate actual salinity gradients by varying (1) horizontal and vertical diffusion coefficients and (2) residual current flows. Using the best-fit parameters derived from the two calibrations, the model was then run with the proposed discharge as an input. This method presumes that the conditions of September 1989 and March 1990 are representative of seasonal extremes (high and low runoff, respectively).

To the extent possible, available monitoring data from the study area were used to estimate the important parameters affecting mixing. Due to the limited data, the complexity of the problem and number of variables involved, there is no single, unique solution to the problem. The uncertainty in parameter values necessitated using an approach of fixing parameters for which data was available, and varying those for which limited data was available until a reasonable solution was obtained.

Because tidal advection is included, a steady state solution was not obtained. However, the model was run with a time step and duration sufficient to obtain a dynamic, steady-state solution. Time series plots are included in Appendix D7.

Finally, because WASP4 is a finite difference program used for far-field analysis, it provides only average parameter values for a given cell. Because of this, the values calculated by the model do not represent local maxima within cells. This should be considered in particular when reviewing the results for the cell receiving the discharge loading, because local pollutant concentrations around the discharge would be higher than the average value calculated by this model.

5. Parameter Estimation

a. Characterization of Discharges

Echo Bay Alaska, Inc. originally proposed to discharge an average of 49 cfs and maximum of 100 cfs of wastewater from the tailings impoundment into Gastineau Channel near Thane (FEIS). A subsequent analysis, which described the operating rule for controlled effluent releases from the impoundment, revised the maximum flowrate to 200 cfs (SRK, April 1994). This operating rule and the 30-year record of flows at the USGS station form the basis for predicting daily outflows. Predictions of metals concentrations in the discharge are described elsewhere in this report (see Chapter VI).

The Gastineau Channel analysis is performed using fixed discharge values designed to represent average and worst-case conditions. The average condition is defined as the 95th percentile value of running 180-day averages of the impoundment loadings predicted in Chapter VI (Stage I of the project). The 180-day time frame was chosen on the basis of the approximate response time of the estuary to the pollutant discharge (see Appendix D7). The worst-case condition assumes a continuous discharge from the impoundment at maximum flow (200 cfs) and 95th percentile maximum pollutant concentrations.

The AJ impoundment discharge is input to the model as a loading (mass/time). Based on the predicted concentration of each constituent above (for average and worst-case conditions) and the average and maximum discharge flow rates, the following total loadings (combined dissolved and particulate) in the impoundment discharge are assumed.

Table 21: Impoundment Discharge Loading

Parameter	Avg Discharge (lbs/day)	Max Discharge (lbs/day)
Mercury	.079	1.61
Cyanide	138	678
Arsenic	2.9	15.1
Copper	76	646
Zinc	155	648
Lead	59	241

Initial dilution modeling by EPA (see Appendix D5) supports predictions in the Comprehensive Report that the discharge plume would be trapped beneath the surface by the density gradient under stratified conditions. In the summer analysis, the entire discharge loading was introduced to the bottom cell corresponding to the proposed outfall location in Gastineau Channel.

Dilution modeling also indicates the potential for a surfacing wastewater plume under unstratified conditions. In the winter analysis, the discharge is uniformly distributed to the bottom and top layer cells corresponding to the outfall location to evaluate the impacts from a surfacing plume.

b. Mixing Dynamics - Salinity Calibration

The amount of mixing occurring in Gastineau Channel is a function of processes driven by tides, freshwater discharges, and wind. In this analysis, the dynamics associated with the tide and residual estuarine circulation were estimated by assuming salinity is a conservative tracer. Wind was not incorporated into the quantitative analyis, though its influence on surface flow is discussed qualitatively in this report.

c. Advection

WASP4 is capable of calculating the effects of both advection and dispersion on water quality in the Channel. In the absence of hydrodynamic model results, WASP4 accepts user-defined flow fields and exchange fields. For this problem, several advective flows are included. They include freshwater inflows, residual flows, and tidal flows. Freshwater inflows and residual flows were fixed, while a time varying function was used for tidal flows. Figure 31 describes the advection routing pictorially.

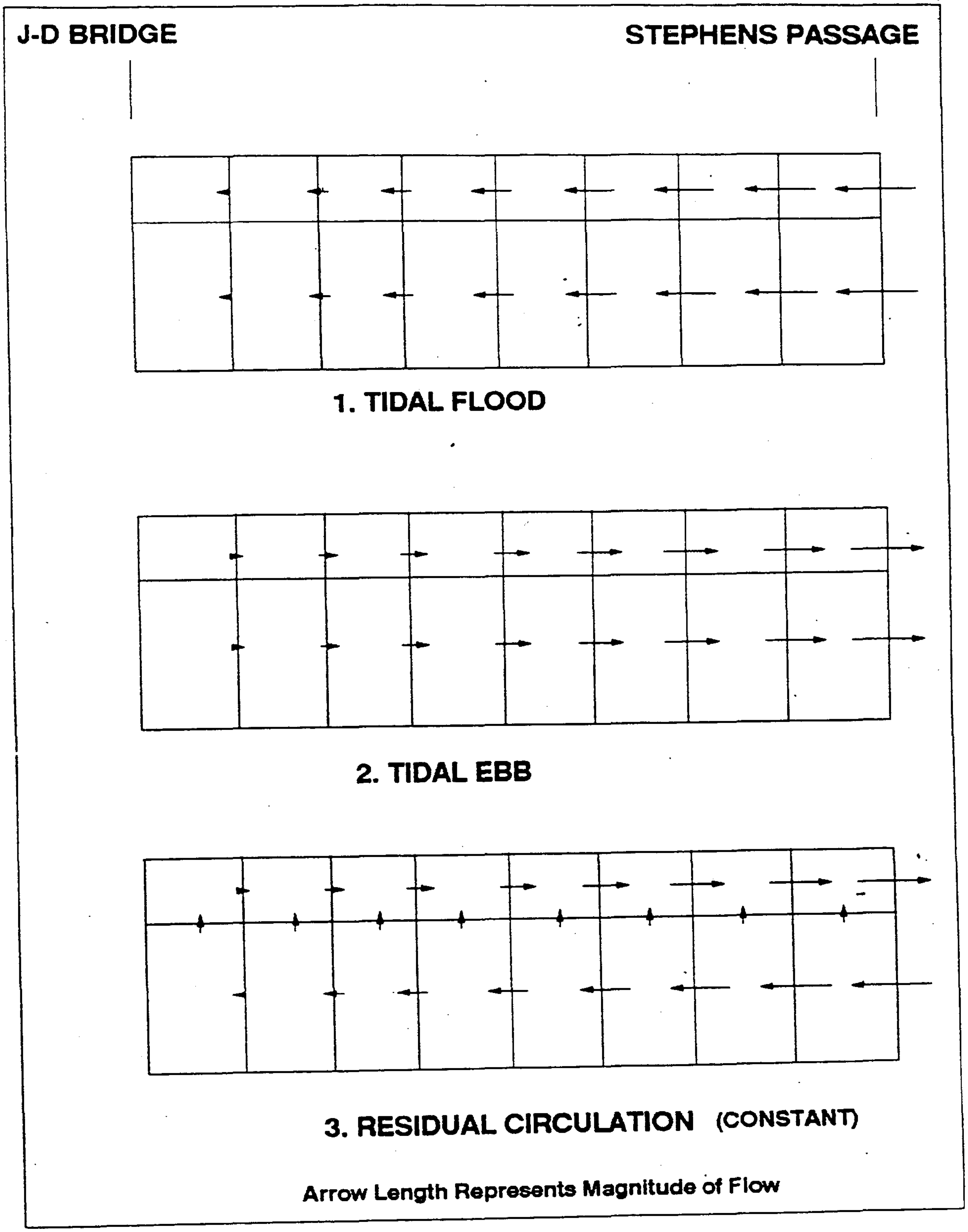
Average residual velocities from Figure 27 were coupled with cell dimensions to obtain the general range for residual flow at the Stephens Passage boundary, with uniform vertical upwelling from the lower layer assumed. The average residual velocity (.005 m/sec) for the Thane current meter would suggest an average inflow greater than 160 m3/sec in the lower layer at the Stephens Passage boundary. However, using the salt mass balance methodology described by Pritchard (1955), the lower layer flow for September 1989 is 27 m3/sec. The flowrate at the boundary was adjusted in the salinity calibration. The best fit for September 1989 salinity data was found at a residual flow of approximately 60 m3/sec, within the range described above. The residual flow for March (low stratification) was assumed to be zero as a worst case assumption.

The tidal chart for August 29, 1990 (from Attachment 6 to the Comprehensive Report) was input as a time variable flow function which repeats itself each day until the model run is complete.

The flows in the model correspond to current velocities that are conservative in comparison to average conditions shown in the current meter results for the 1990 and 1992 current meter surveys. For example, the average tidal current at WASP4 station #7 is approximately 1 cm/sec, while the corresponding average current for station A in the 1990 current survey was approximately 2 cm/sec.

d. Dispersion

Dispersion in estuaries also results from small-scale turbulent diffusion and velocity shear. These factors result from interactions within the tidal flow as well as between the flow and the channel bottom. Velocity shear effects are created by density structures due to freshwater inflow and also interactions between tidal flows and the channel bottom.



ADVECTION ROUTING

Figure 31

Turbulent diffusion in the horizontal and vertical dimension and velocity shear combine to cause dispersion of particles or pollutant plumes. The mechanics describing the aggregate effect are considered analogous to molecular diffusion; as a result, the Fickian molecular diffusion equations are used to estimate turbulent diffusion in a number of predictive tools, including the WASP model framework. These estimates are driven by the selection of horizontal dispersion coefficients and vertical coefficients of eddy diffusivity.

No site-specific data are available to provide estimates of the dispersion coefficient (K). Therefore, this parameter was treated as a variable; coefficients were adjusted to fit the salinity data. The horizontal dispersion coefficient was varied as the square of the distance to the tidal source in accordance with available literature (Bowden, 1967). The assumed increase in horizontal dispersion down-channel is reasonable given the proximity to Stephens Passage. The use of 0.16 m^2/sec for the uppermost cell, increasing to 10 m^2/sec for the lowermost cell obtained reasonable fits to the salinity data. Of note, this order of magnitude of horizontal dispersion resulted in model results that matched the pattern shown by the aggregate salinity data (higher salinity at station 12), whereas increasing the values by one order of magnitude removed this pattern. The chosen values fall within literature ranges for this parameter (Kowalik, 1984).

Table 22: Horizontal Dispersion Coefficients

WASP Cells	Distance (m)	K (m2/sec)
1, 16	1725	.16
2, 15	3450	.60
3, 14	5175	1.4
4, 13	6900	2.5
5, 12	8625	3.9
6, 11	10350	5.6
7, 10	12075	7.6
8, 9	13800	10.0

For vertical coefficients of eddy diffusivity, no data are available to provide estimates. Therefore, this parameter was treated as a variable; it was adjusted to fit the salinity data and account for relative stratification. The use of 0.00001

m^2/sec for March and 0.00001 m^2/sec for September obtained relatively good fit solutions to the problem. The increased diffusion in March is reasonable given the weak stratification at that time, and both values fall within literature ranges for this parameter (Kowalik, 1984).

e. Salinity Calibration

i. Freshwater Inflows

Freshwater flows into Gastineau Channel are important for the salinity calibration. They were estimated using the values from Echo Bay's "Gastineau Channel Freshwater Flushing Study" (September 1991). In this report, flows were calculated for individual drainage basins based on discharge rates and relative basin size for the creeks that have been gauged in the past (Gold Creek 1918-1982, Lawson Creek 1970-1971, and Sheep Creek 1919-1973). Using the average Gold Creek flow for February 1990 and September 1989 and the relative fraction of total flows for August from the Flushing Study, the flows for the other creeks were estimated for February 1990 and September 1989.

Due to dimensional constraints of the WASP4 code, flows were aggregated and introduced to cells in the model that correspond to the largest creeks (cells 9 and 12) (See Figure 30). For predictive model runs, the flows into cell 12 associated with Sheep Creek were reduced based on project predictions for flows after dam construction.

ii. Ambient Salinity Data

As indicated in Figure 32, seawater monitoring stations established by Echo Bay provide limited longitudinal coverage of Gastineau Channel. To reduce potential for local influences to mask salinity on a larger scale, Echo Bay stations were aggregated for comparison with WASP4 predictions. Monitoring stations were established in three areas of the channel: near Stephens Passage, Thane, and Douglas. These locations correspond to six WASP4 stations (one top layer and one bottom layer cell for each location).

In order to translate salinity profiles into average cell salinities for comparison to WASP4 predictions, weighted averages were calculated for salinities measured from the surface to 10 meters, and from 10 meters to 35 meters (see Appendix D1).

iii Boundary Conditions - Salinity

Monitoring information for Stephens Passage and Taku Inlet is not available for the time period of Gastineau Channel monitoring (February 1990 and September 1989). In absence of data, EPA used September 1988 information to establish boundary values for the September 1989 calibration. For March 1990, the longitudinal variation in salinity was minimal, and boundary values for the surface layer were set approximately equal to the those at the station nearest to Stephens Passage. Unfortunately, the March 1990 sampling was conducted to depths greater than 15 meters; in this case, a value from the March 1992 Thane current meter was used for lower layer salinity.

iv Salinity Simulation

A comparison of WASP4 predictions and seawater monitoring data is depicted in Figure 33.

f. Boundary Conditions for Pollutants of Concern

Three boundaries were established in the model: one at the mouth of the bay, and one at the entrance of Gold Creek and Sheep Creek. It is assumed that no mixing occurs beyond the Juneau-Douglas Bridge, though inflow from Gold Creek and Salmon Creek is included.

EPA reviewed 1990 seawater monitoring data collected by Echo Bay in Gastineau Channel to establish boundary conditions for the model. Gastineau Channel values are used as boundary values at the entrance to Stephens Passage and creek inflows. Mean values (excluding non-detects) are used in the model. All data are drawn from the Comprehensive Report (Document #2; EBA, 1992). The table below summarizes the values used in the model (all metals values are in "total recoverable").

Table 23: Gastineau Channel Background Water Quality

Parameter	Conc. (ug/l)
Mercury	.009
Cyanide	NA(0)
Arsenic	1.4
Copper	0.8
Zinc	2.3
Lead	0.2

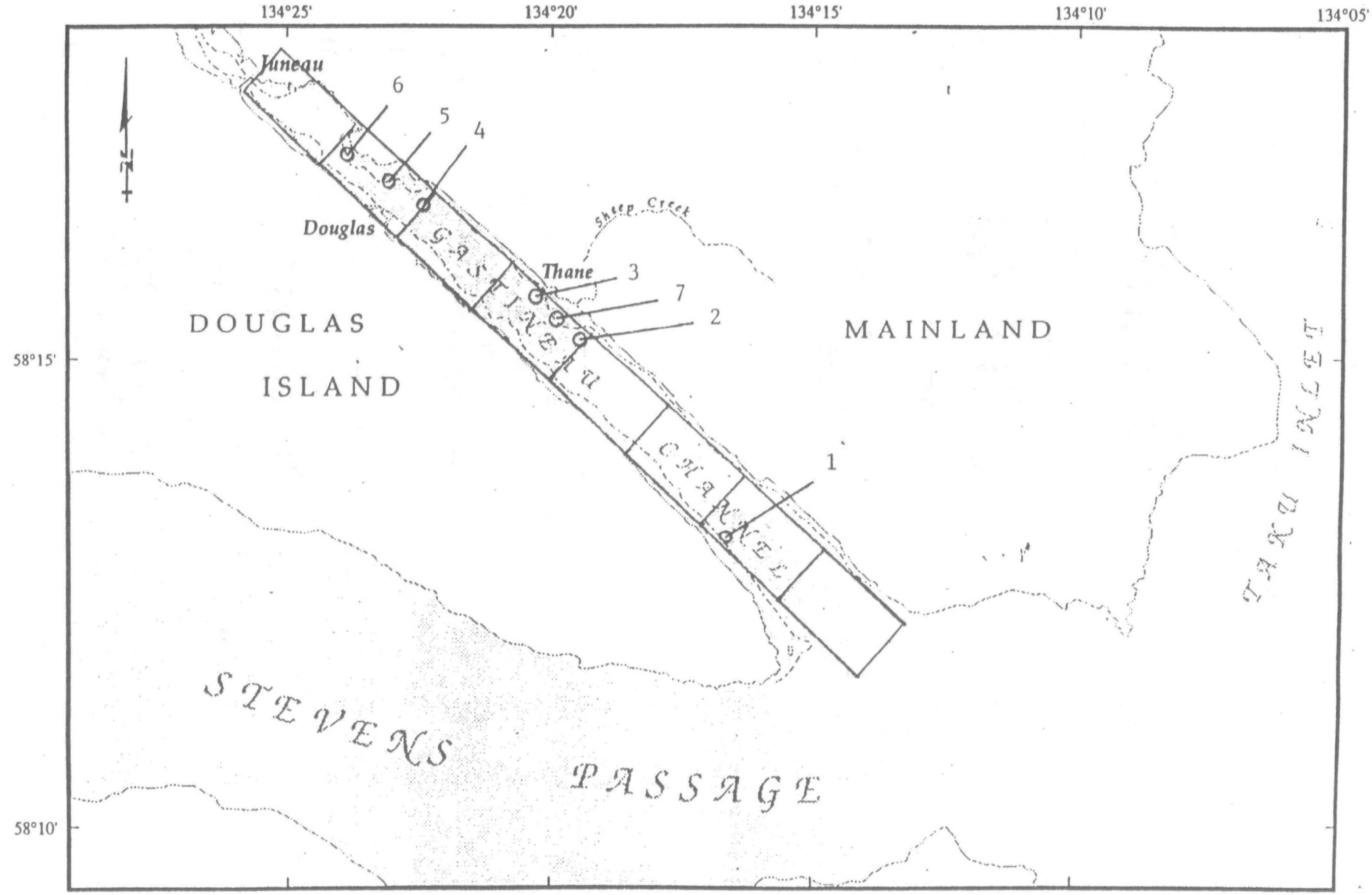


Figure 32 : Salinity Stations in Relation to WASP4 Grid

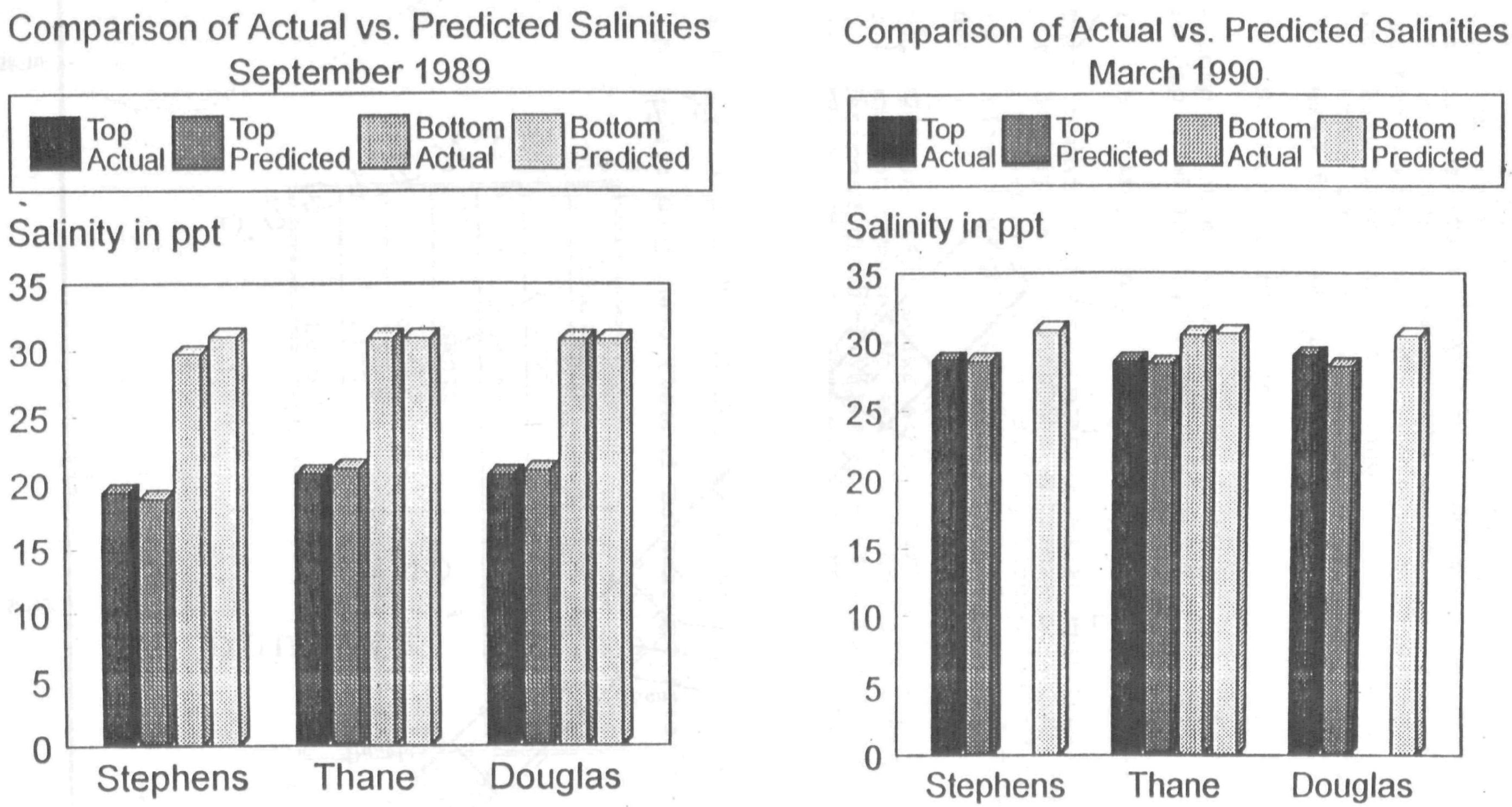


Figure 33 : Salinity Calibration Results

6. Projected Impacts

a. Applicable Water Quality Standards

The state of Alaska water quality standards are codified in 18 AAC 70. The regulation establishes beneficial uses of Alaskan waters as well as the water quality criteria that must be met to assure protection of a given use.

b. Beneficial Uses Affected

Gastineau Channel is protected under the Alaska Water Quality Standards for the following beneficial uses:

- Water supply for aquaculture, seafood processing, and industrial uses.
- Contact and secondary recreation.
- Growth and propagation of fish, shellfish, other aquatic life, and wildlife.
- Harvesting for consumption of raw mollusks or other raw aquatic life.

c. Water Quality Criteria

The State of Alaska water quality standards establish the following criteria for total recoverable metals and cyanide in marine waters, designed to protect against chronic toxicity to aquatic organisms (for arsenic only, the limiting criterion is for protection against human health effects from fish ingestion):

Table 24: Water Quality Criteria

Parameter	Applicable Criterion (ug/l)
Mercury	.025
Cyanide	1.0
Arsenic	1.4
Copper	2.9
Lead	8.5
Zinc	86.0

d. Model Predictions

Figures 34 through 36 depict comparisons of the water quality standard concentration and the predicted average and maximum pollutant concentrations from the WASP4 model analysis. For each case (top/bottom layer, summer/winter), predictions for the entire channel length are shown, with the Juneau-Douglas bridge at the left of each graph and the entrance to Stephens Passage at the right. Results in tabular form are included in Appendix D8.

For the winter case, one set of plots represents concentrations in both the top and bottom layer. This is because concentrations in the two layers are approximately equal for the winter case, due to the assumption that a surfacing discharge plume would be uniformly distributed in the water column during the winter.

7. Conclusions

The analysis above indicates that far-field mixing in Gastineau Channel is significantly limited for discharges of the magnitude proposed for the AJ impoundment. As a result, impacts from the impoundment discharge are predicted to occur on a channel-wide scale. This contradicts the conclusion in the Comprehensive Report (EBA, 1992) that sufficient dilution of the proposed discharge would be achieved in a confined mixing zone around the diffuser.

Based on the model assumptions described herein, the predicted average discharges from the AJ Mine are projected to violate the Alaska water quality standards for arsenic, copper, and cyanide in Gastineau Channel on a channel-wide scale. Under worst-case assumptions, mercury and lead standards would be violated as well.

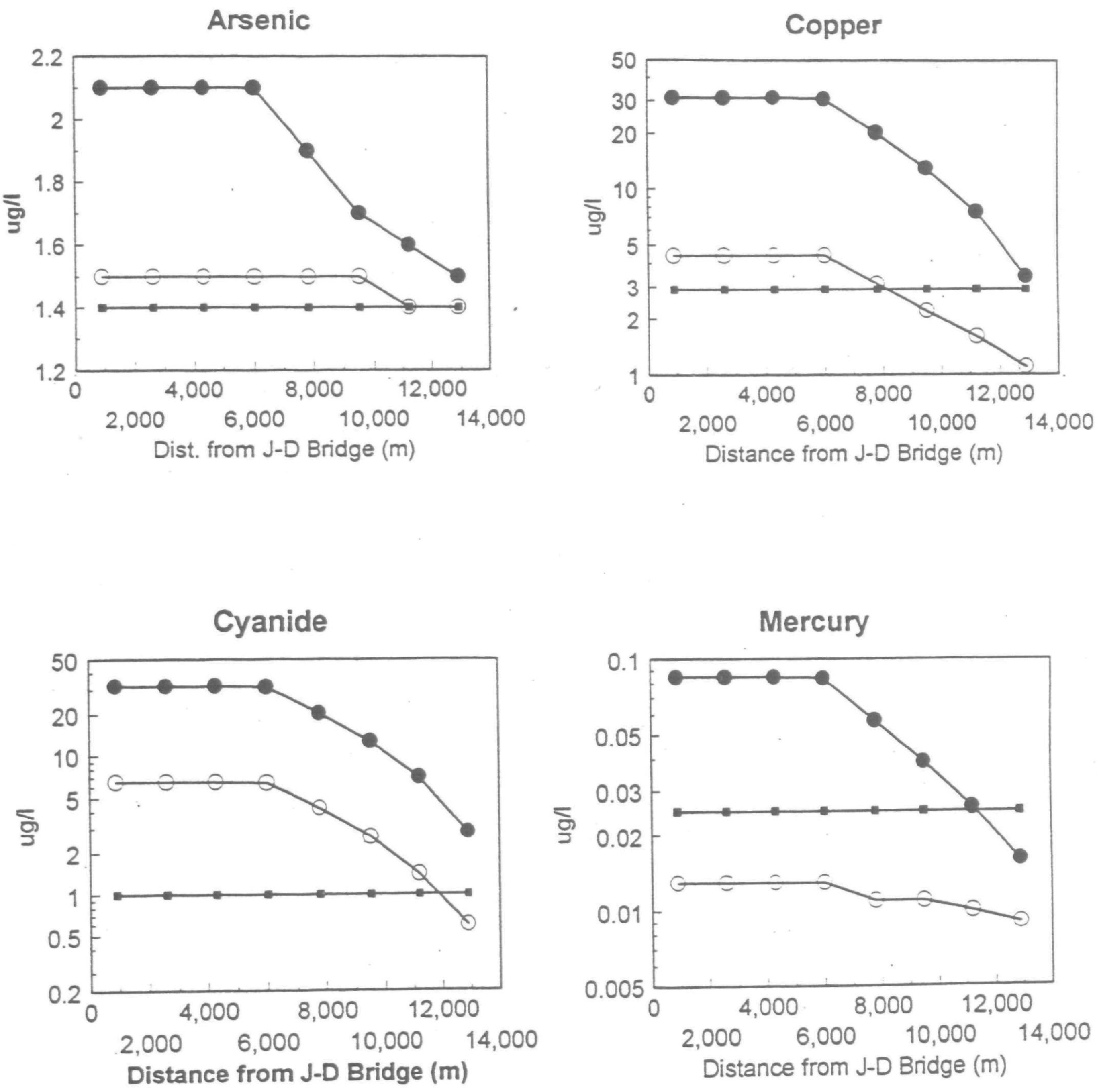
8. Uncertainty

a. Wind

Wind-driven circulation is not considered in this assessment, though it is acknowledged as one of the important factors affecting circulation. Southeast winds would serve to counter the flushing mechanisms due to tidal and gravitational advection, generally restricting or reversing the migration of surface waters out of the channel and reducing the dilution of pollutants. Local, episodic impacts to surface waters, particularly with a surfacing plume, are not addressed by this report.

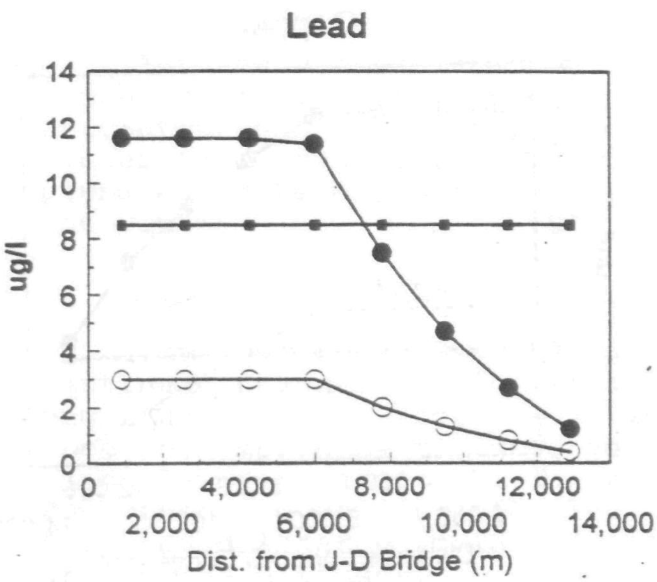
Figure 34
WASP Model Results - Summer
Bottom Layer

(solid dot = max, clear dot = avg, square = standard)



WASP Model Results - Summer
Bottom Layer

(solid dot = max, clear dot = avg, square = standard)



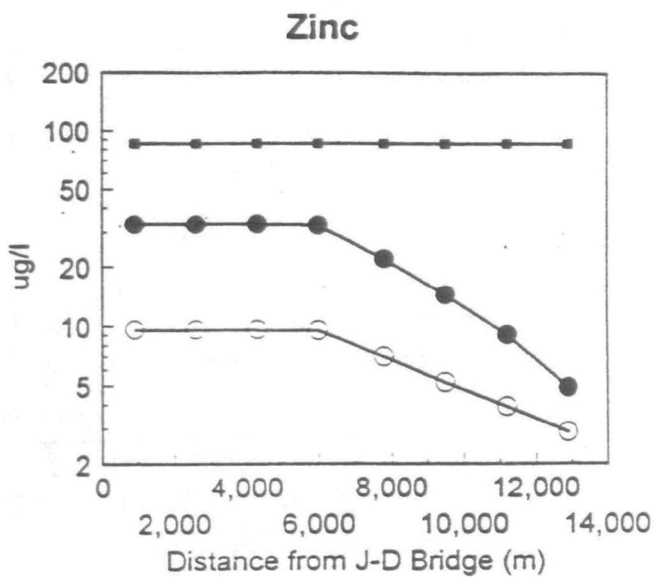
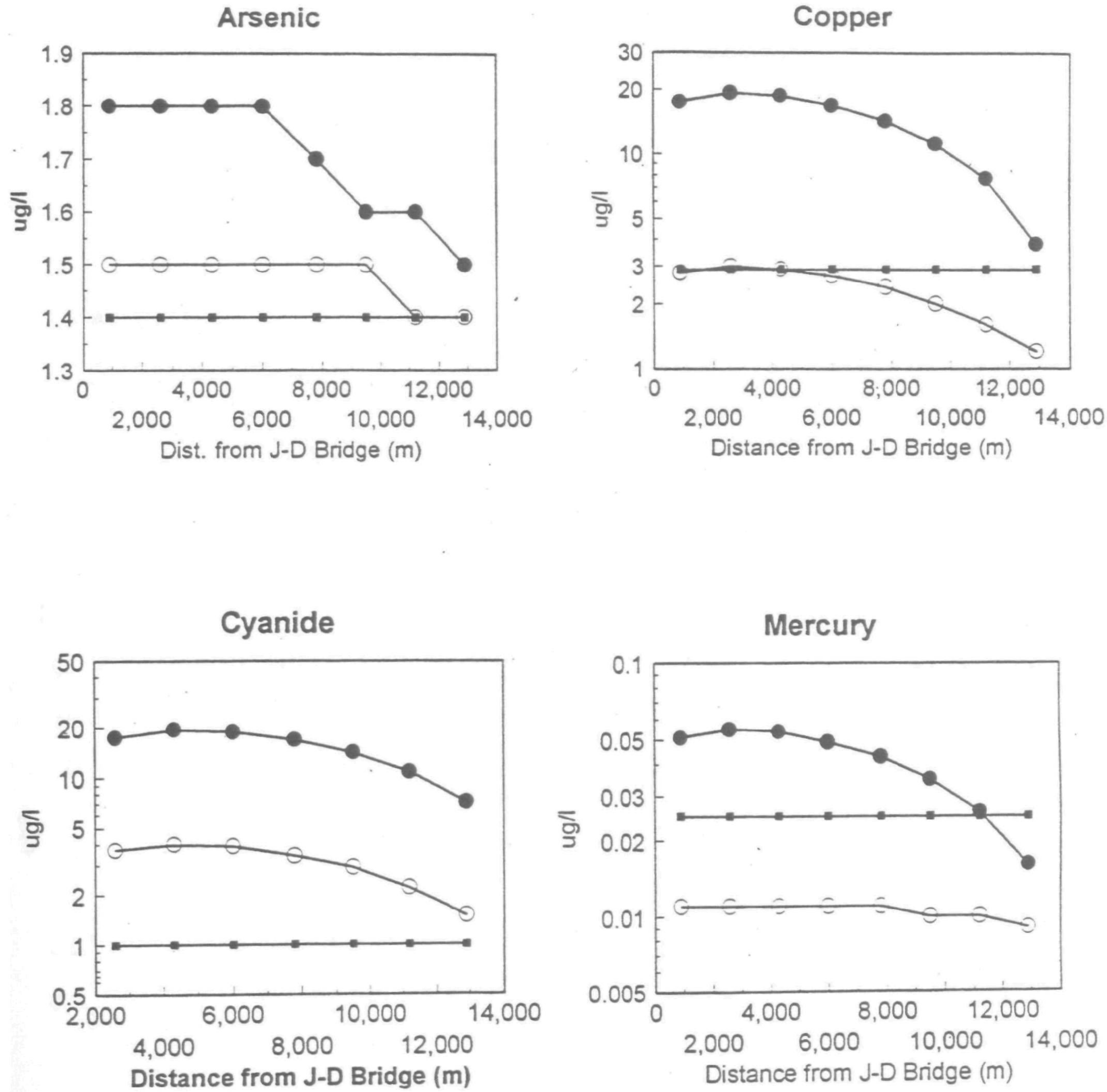


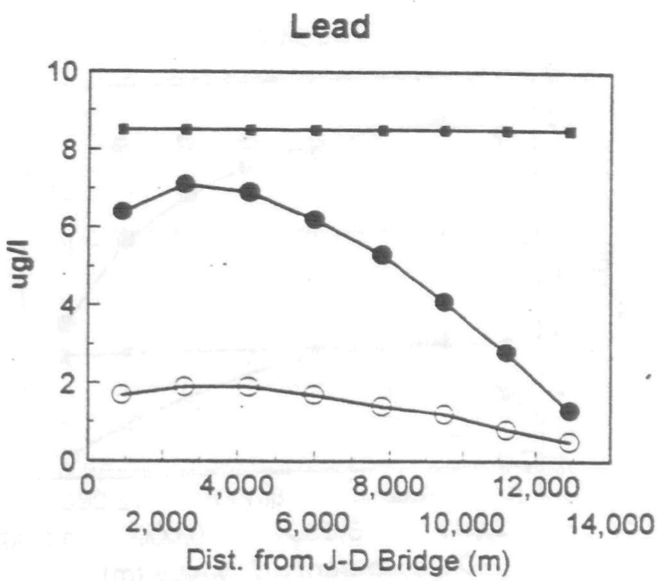
Figure 35
WASP Model Results - Summer
Top Layer

(solid dot = max, clear dot = avg, square = standard)



WASP Model Results - Summer
Top Layer

(solid dot = max, clear dot = avg, square = standard)



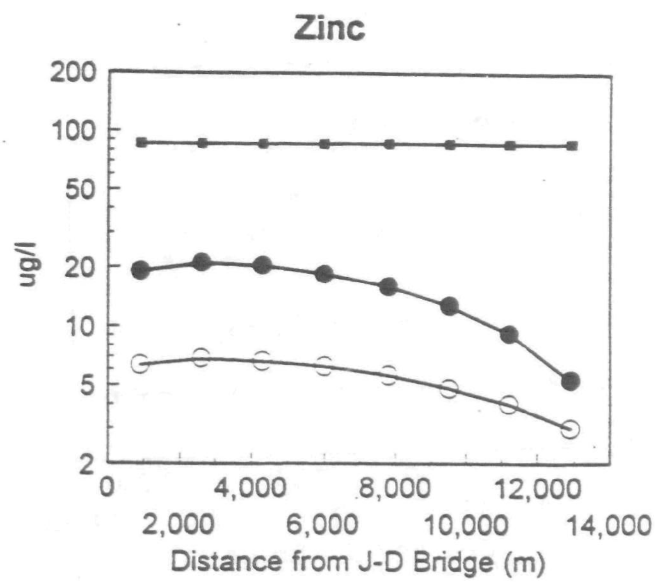
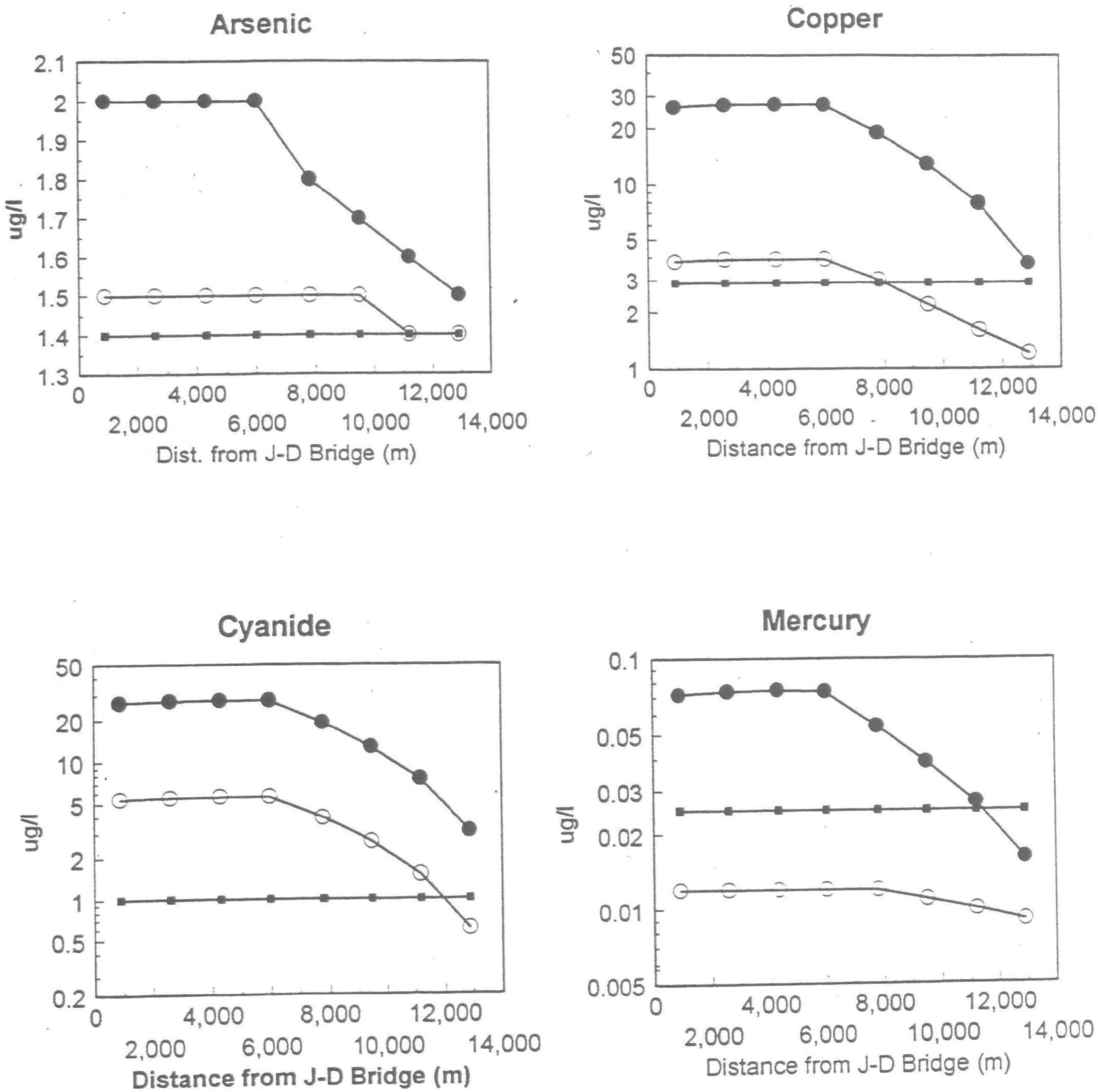


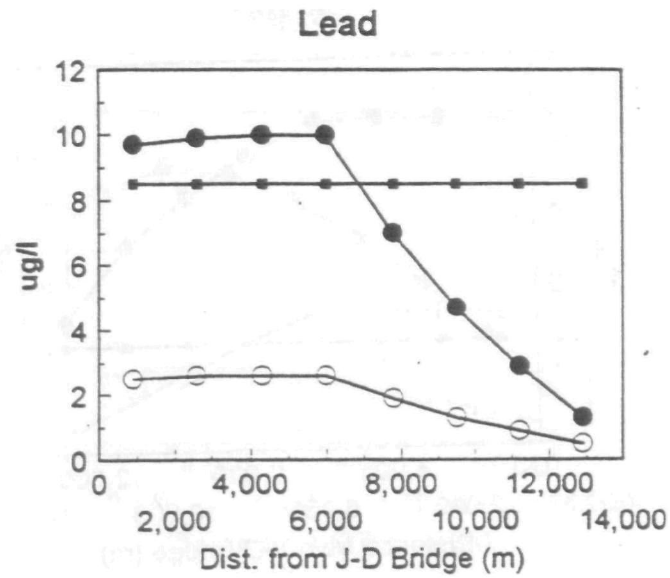
Figure 36
WASP Model Results - Winter
Top and Bottom Layer

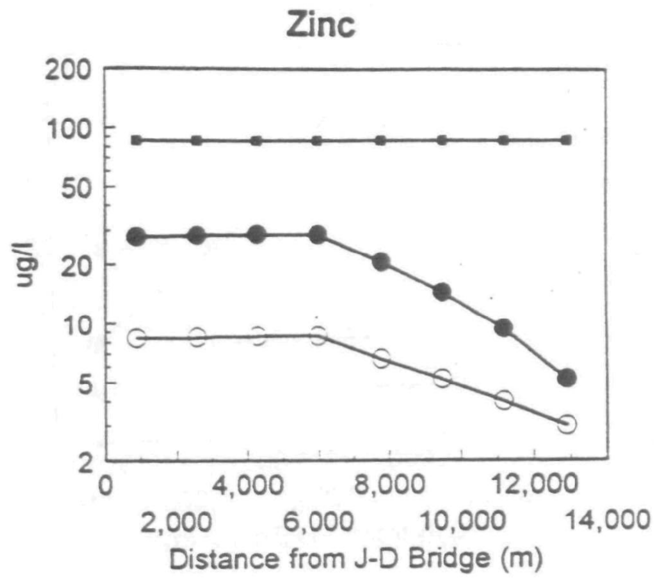
(solid dot = max, clear dot = avg, square = standard)



WASP Model Results - Winter
Top and Bottom Layer

(solid dot = max, clear dot = avg, square = standard)





b. Local Effects

This assessment provides only average pollutant concentrations for a given cell. Because of this, the values calculated by the model do not represent local maxima within cells. This should be considered in reviewing the results for the cell receiving the discharge loading, because local pollutant concentrations around the discharge would be higher than the average value calculated by this model.

c. Discharge Quality

Because this report relies on the impoundment assessment for estimates of the quality of the discharge, the predictions and conclusions herein carried forward the uncertainties associated with those estimates.

d. Discharge Quantity

As stated earlier in the report, the quantity of the discharge is somewhat uncertain. A significant change (from 100 to 200 cfs) has been made to maximum flow volume estimates in the period since the EIS was finalized. This was due in part to changes in project designs and flow routing. Because of the importance to loadings from the discharge, this indicates the potential for inaccuracies in any predictions of impacts to Gastineau Channel from this proposed facility.

e. Potential Impacts Beyond the Juneau Douglas Bridge

As stated earlier in the chapter, the drift card studies have indicated the potential for surface waters and pollutants to migrate beyond the Juneau-Douglas bridge and reach the tide flats around the Mendenhall River and Fitz Cove. This highlights the potential for impacts beyond the boundaries of the modeled system. Persistent southeast winds and a surfacing plume (winter) could restrict flushing in the surface and impact these locations above the bridge.

VIII. RISK OF LONG-TERM CONTAMINATION

A. Introduction

The goal of this analysis is to determine the likelihood that a healthy aquatic food chain and habitat system would develop and persist after the cessation of mining and the active use of the impoundment for tailings disposal. The physical and chemical characteristics of the impoundment and surrounding area would dictate the expected food chain and habitats.

The majority of the chemical impact analysis is contained in an ecological risk analysis presented later in this chapter. It is based on modeling of predicted residual chemical levels in the impoundment. A review of relevant literature on subaqueous disposal of mine tailings is included as well. This is followed by an assessment of the potential for the tailings to generate acid and mobilize metals in the tailings.

The following is a description of impoundment geomorphology, and its setting from the watershed and physical standpoint. This analysis will be carried through to discuss the expected biological community that would be expected to develop in the impoundment in the absence of any chemical contamination.

B. Watershed/Physical Setting

Following closure of the AJ Mine, the resulting tailings impoundment would rest in a typical U-shaped glacial valley with steep side slopes. Elevation would be approximately 932' MLLW. The long narrow body of water would be 20 feet deep at a minimum, and 2.5 miles long, with a surface area of 420 acres (FEIS p. 4-20). When there is a weakly developed thermocline (late summer), the impoundment would have an approximate surface temperature of 11 degrees Celsius and a bottom temperature of 7 degrees Celsius, otherwise the impoundment would be relatively isothermal (HDR, 1990). The impoundment would generally freeze from November through March.

Due to the steep valley walls, the impoundment would be avalanche prone. The applicant proposes to construct flat bench areas out of waste rock from the mine. The benches would be constructed along the impoundment shoreline in avalanche chute areas for avalanche dissipation (BLM, 1992). Following mine closure, the shoreline sites would be recontoured to provide relatively flat beach areas (BLM, 1992).

ADFG has required as mitigation diversion of an average annual flow of 16.3 cfs from the upper watershed to Sheep Creek below the impoundment dam and above the hatchery. This would require diversion structures at the inlet creeks above the impoundment, and changes in water flows and other inputs (e.g., sediments) from the upper watershed to the impoundment (Letter from ADFG, Lana Shea to Frank Bergstrom dated February 2, 1994).

Average annual discharge from the Sheep Creek watershed above the proposed dam is 49 cfs. During mine operation 16 cfs would be diverted, via upstream diversion structures, around the impoundment to lower Sheep Creek. Monthly stream flows from the watershed vary seasonally from lows of 6 cfs in February up to nearly 100 cfs in June (see Table 2). Residence time in the impoundment (the amount of time it takes for a particle of water to enter and then leave the impoundment) is calculated to be 4 months in the summer and 7.2 months in February (p. H-8). Natural sediment loading into the impoundment would be low, with total annual sediment transport into the impoundment of 1200 cubic yards (p. 4-89). The addition of upstream diversion structures would decrease sediment transport.

Following use as a tailings treatment facility, the bottom of the impoundment would be covered with well-graded silty-sand tailings. These tailings could be periodically resuspended by large avalanche events. At a minimum, 20 feet of water would cover mine tailings. It is planned that water level variations should result only in increased depth and not decreased depth over the tailings.

C. Uncertainty Factors

The above description includes information that is relatively certain from the project's supporting documentation. However, the following are some important dynamic "unknowns" that could have a large impact on the resulting biological community. Resuspension of sediments by Taku winds (FEIS V1-3-2: fastest one-minute wind Nov 68 = 58 mph) could have an impact on the impoundment's biological community, including smothering, toxicity, and turbidity.

Avalanches would also play a role in resuspension within the impoundment, depending on the season, and in the development of vegetation on the avalanche dissipators.

Water level fluctuations in the impoundment would affect riparian and nearshore vegetation, depending on the degree and frequency of inundation. Water level fluctuation extremes are a relative unknown though theoretically the impoundment level can be managed as desired by the resource agencies. Reservoirs whose surface level fluctuates according to unnatural cycles generally result in stressed communities of macrophytes in the zone that is repeatedly inundated and then dewatered, especially if this occurs during the growing season.

In addition, the impoundment as proposed to date is an unusual feature in southeast Alaska. A steep-sided fjord-like impoundment with relatively shallow water depth is not frequently found in this landscape. A query to Alaska Fish and Game discovered no representative natural lake from which to base predictions of biological activity for the proposed design (Letter Dr. Dana Schmidt, Alaska Fish and Game to Justine Barton, dated 12/ 13/93).

D. Predicted Community Components.

1. Benthos

With a sedimentation rate of only 1200 cubic yards/year, impoundment tailings coverage and natural substrate for benthos is not expected in the short term. The coarseness of the tailings as well as chemical inputs would play an important role in determining whether a benthic community could establish on the tailings themselves. No analysis to date has focussed on the grain size requirements of a healthy benthic community in this environment. Additionally the impoundment lacks a food web base due to expected low input of organic material, and lacks an existing lake benthic community to serve as a source for recruitment to the impoundment.

2. Plankton

An analysis of anticipated phytoplankton and zooplankton populations is required, especially considering the lake's shallowness and planktonic migrations based on depth. It is highly unlikely that mining discharges will increase the productivity of the impoundment compared to a similar impoundment with no discharges of tailings. An analysis of nutrients and their anticipated concentrations in the tailings pond water is needed, including an analysis of any break down products. (Enclosure A from Letter Lana C. Shea, Alaska Fish and Game to Bill Riley, dated 1/10/94).

3. Fish

Fish populations are not anticipated in the impoundment during mine operation. Subsequent to mine closure, fish populations are unlikely without a managed food base. Insect production and transport by the inflowing small streams will not provide enough food to sustain fish, and the diversion dams will exacerbate this. In addition, the impoundment lacks the habitat characteristics required for potential species survival (e.g. cover, spawning areas). The habitat constructed for fish would depend on the species desired and whether the impoundment is to be managed for fish, based on anticipated chemical levels in tissues (e.g. kokanee, cutthroat). As stated by ADFG (1/10/94 ltr-pA3) "We do not know if the stream resident char currently found in upper Sheep Creek could adapt and survive in the proposed tailings pond (independent of toxicity issues)", and "it is not clear how robust the resulting population might be" since they require spawning habitat and suitable aquatic insect fauna.

4. Macrophytes

The Mine Environment Neutral Drainage (MEND) studies, conducted by a consortium of Canadian government agencies and industry representatives, examined lakes in Canada that have been used in the past for subaqueous disposal of mine tailings. Four potentially relevant Canadian lakes studied included Buttle Lake, Anderson Lake, Mandy Lake and Benson Lake. The MEND case studies at Mandy and Benson lakes show that macrophytes can take up metals when grown directly on tailings.

The argument for subaqueous disposal is that all tailings will be covered by water such that no contact with rooted macrophytes is likely. However, the potential magnitide of resuspension events (e.g. Taku winds, avalanches) and resuspension of tailings into shallow areas limits the analysis of expected species. This is especially of concern on the littoral and fringe zones of the impoundment where dabbling waterfowl and macrophytes are anticipated, and where other routes of exposure to the tailings besides consumption of leafy portions of the macrophytes (e.g. sediment, roots) could be a factor.

5. Littoral/Riparian (fringe) zone and Vegetation

The post-closure vegetation expected on the surrounding slopes and avalanche dissipators is likely to be alder and shrubby vegetation. It has been suggested that the impoundment's shoreline would create new wetland habitats, however, this has not been analyzed, and based on steep surrounding terrain seems unlikely. Obviously any shoreline vegetation would be greatly influenced by the impoundment's operation with respect to how the surface water level fluctuates. Any vegetation growing on the avalanche dissipators would be affected by avalanche events.

6. Wildlife

It is likely that the tailings impoundment will be attractive to birds, even during mine construction and operation, and certainly following mine closure. This includes migratory shorebirds and waterfowl, and resident breeding birds. (ADFG 1/10/94) Many passerines and other species of shorebirds will use the impoundment as a source of drinking water. If vegetation remains or recolonizes along the edge, other species including warblers, sparrows and thrushes, might continue to use the area for nesting, feeding, and water. (USFW Comments on Final EIS, Memo from USFW Region 7 to BLM dated July 3, 1992) Some known or likely breeding birds in the Sheep Creek area include Harlequin duck, Common merganser, Great blue heron, Golden eagle, Blue grouse, Rock ptarmigan, White-tailed ptarmigan, Spotted sandpiper, Common snipe, Marbled murrelet, Rufous hummingbird, Belted kingfisher, Pacific slope flycatcher, Tree swallow, Barn swallow, Steller's jay, Northwest crow, Raven, and other creepers including wrens, vireos, sparrows, finches, pipits, kinglets, & dippers. (FEIS V-1, 3-56). Sensitive species include: spotted and rock sandpipers, dunlin, marbled murrelets, bald eagles, sharp-shinned hawks, great horned owls, three-toed woodpeckers, belted kingfishers, olive-sided and Pacific slope flycatchers, western wood peewees, and dozens of other locally-breeding species of passerines. (Fish and Wildlife Coordination Act letter from Nevin Holmberg to Colonel William Kakel dated March 28, 1991).

In addition to birds, other common species found in the area include black bear, mountain goat, Sitka black-tailed deer, beaver, marten, river otter, mink, ermine and other mustelids, lynx, red fox, hoary marmot, porcupine, and a number of other small mammals. (Fish and Wildlife Coordination Act letter from Nevin Holmberg to Colonel William Kakel dated March 28, 1991).

Bird and other species may or may not remain in the area depending on disturbance from human activity, and cover and foraging provided at the impoundment's margins. Certainly the lack of deciduous trees following project construction means that fewer passerines would nest in the valley. More and various types of waterfowl would likely be attracted to the impoundment.

E. Ecological Risk Analysis

Analysis of potential effects of the post-operation impoundment on local biota were based on estimates of contaminant concentrations in the impoundment sediment, pore water (i.e., water in the pore spaces within the tailings), and water column as provided in Chapter VI of this report.

A preponderance-of-evidence approach was used. Adverse effects on biota from contaminants in pore water and the water column were estimated by comparing concentrations with Ambient Water Quality Criteria (adjusted for hardness, pH, and temperature where appropriate; 40 CFR Part 131, 1992) and aquatic toxicity data. Effects of contaminants in sediments on benthic biota were estimated by comparing sediment concentrations with Canadian Provincial sediment quality guidelines (Ecology 1991, Jaagumagi, 1993), Wisconsin Department of Natural Resources guidelines (Ecology 1991), and draft Great Lakes Effects Range Medians (pers. comm., Ingersoll 1994). Potential effects on wildlife from contaminants in the pore water, sediment, and water column were estimated by constructing models to compare the exposure of sandpipers, river otters, and kingfishers with the potential toxicity from cadmium, lead, mercury, and selenium. These metals were selected following discussion with EPA ORD (pers. comm., Norton 1994) because integrative assessments of their toxicity and exposure pathways relevant to wildlife exist (Eisler 1985a, 1985b, 1987, 1988) and because these have the potential to bioaccumulate. The Wildlife Exposure Factors Handbook (EPA 1993b) was used to derive many of the wildlife exposure parameters. Additional information was used to evaluate mercury (EPA 1993a) and selenium (Saiki 1986). The data used are considered reasonable and representative.

F. Evaluation of Contaminants in Pore Water

Predicted pore water concentrations for various contaminants are shown in Tables 25 and 26. Contaminants of particular concern included metals, cyanide, the breakdown products of cyanide (thiocyanate, cyanate, and ammonia), and the xanthates (a collecting agent added to the milling process; Hawley, 1977). It is recognized here that water quality criteria are generally designed to be protective of water column species and that benthic organisms may or may not be similarly protected. It is not unrealistic to assume, however, that certain species of fish will be exposed to pore water or that fish or other water column species can be useful indicators of pore water toxicity. Therefore, in this analysis all data encountered for freshwater organisms are included in the evaluation of potential toxicity of the pore water in the tailings impoundment.

1. Pore Water Evaluated Using Water Quality Criteria

Water Quality Criteria were available for most metals, cyanide, and ammonia. Several of the **metals** criteria were dependent on water hardness. The hardness value used (670 mg/L $CaCO3$) was that predicted from the decant studies as described in Chapter VI. **Cyanide** criteria are based on measurements of free cyanide because this form is a much more reliable index of toxicity to aquatic life than total cyanide since the latter can include nitriles and more stable metallocyanide complexes. However, because these complexes can dissociate as a function of pH in the range that commonly occurs in many water bodies, total cyanide is recommended for evaluation with the criteria especially if only a few measurements are made on a water body (EPA, 1985; 440/5-84-028). For this analysis, the cyanide criteria were applied to both free and total cyanide. It should be noted that weak cyanide concentrations are more toxic to fish at temperatures near freezing since HCN will not evaporate from the surface of a waterbody when it is covered with ice; winter

Table 25. Comparison of projected pore water concentrations with ambient water quality criteria. Concentrations exceeding acute or chronic criteria are enclosed in boxes with an A or C respectively. Ratios that exceed unity are also enclosed in boxes.

Hardness	670	mg/L
pH units	8.2	

Parameter	Pore Water Concentration *ug/L*	Water Quality Criterion (a) Acute	Water Quality Criterion (a) Chronic	Ratio of Parameter to Criterion Concentration Acute	Ratio of Parameter to Criterion Concentration Chronic
Ag	50	107		0.5	
As(III)	14	360	190	0.0	0.1
Cd	10 C	34	5	0.3	2.0
Cr(III)	20	8246	1003	0.0	0.0
Cr(VI)	20 A	16	11	1.3	1.8
Cu	1300 A	106	60	12	22
Fe	850		1000		0.9
Hg	5 A	2.4	0.012	2.1	417
Mn	2700				
Ni	30	7089	788	0.0	0.0
Pb	20	919	36	0.0	0.6
Se	16 C	20	5	0.8	3.2
Zn	87	586	531	0.1	0.2

Parameter	*mg/L*	Acute	Chronic	Ratio Acute	Ratio Chronic
CNfree	1.1 A	0.02	0.0052	50	212
CNtot	2.5	0.02	0.0052	114	481

Parameter	Concentration	oC	Acute pH 7.4	Acute pH 8.2	Chronic pH 7.4	Chronic pH 8.2	Ratio Acute 7.4	Ratio Acute 8.2	Ratio Chronic 7.4	Ratio Chronic 8.2
NH3	18 A	*1*	16.0	4.1	2.1	0.8	1.1	4.4	8.8	23
		4	15.3	3.9	2.0	0.8	1.2	4.6	9.1	24
		7	14.8	3.8	1.9	0.7	1.2	4.7	9.5	24

(a) Based on hardness of 670 and Water Effects Ratio of 1, following EPA 1992; As(III), Cr(VI), Hg, Fe, Se, CN, and NH3 are not dependent on hardness; see discussion in text on application of Water Quality Criteria to CNfree, CNtot, and NH3.

Table 26. Comparison of projected pore water concentrations with aquatic toxicity data. Parameter concentrations exceeding aquatic values of concern are enclosed in boxes. Ratios that exceed unity are also enclosed in boxes.

Parameter	Pore Water Concentration *mg/L*	Concentration Range of Concern (a) High	Concentration Range of Concern (a) Low	Ratio of Parameter to Concentration of Concern High	Ratio of Parameter to Concentration of Concern Low
CNS Thiocyanate	[390]	24	1	[16]	[390]
CNO Cyanate	[73]	95	18	0.8	[4.1]
Xanthates	[0.93]	5.6	0.01	0.2	[93]

(a) See discussion in text on estimation of high and low concentrations of concern for CNS, CNO, and Xanthates. Note that the concentration ranges of concern would be much lower if based on sublethal, chronic effects.

conditions can increase the risk of fish kills by cyanides (Palmes, 1993). Weak acid dissociable cyanide (CNwad) is a measure of free cyanide plus cyanide bound to other metals, especially copper, nickel, and zinc. CNwad is a measure of those forms of cyanide considered to be toxicologically significant (Smith and Mudder, 1991). For the pore water, the predicted CNwad is equal to the predicted free cyanide, indicating that there will not be much metal-bound cyanide and that evaluation of free cyanide will be sufficient to evaluate CNwad. **Ammonia** criteria depend on both pH and temperature. The pH values used here ranged from 7.4 (which was reported as a background value; Lakefield 1990, No. 3980, Table 7) to 8.2 (a value projected for pore water; see Table 25). The temperatures used here ranged from 1 oC (the impoundment is expected to freeze in the winter; HDR) to 7 oC (expected in the bottom water in summer; HDR, 1990). The estimated pore water concentrations were compared with both acute and chronic criteria.

2. Pore Water Evaluated Using Aquatic Toxicity Data

Palmes (1993) review reported **thiocyanate** toxicity in fish occurs over a range from 24 mg/L to 5,000 mg/L. In addition, if fish are stressed (e.g., chased), mortality can increase after exposure to as little as 5.5-7.7 mg/L. Heming et al. (1985) found 10 to 100% mortality in stressed versus non-stressed brook trout exposed to these lower concentrations. They also found that, at 1.0 mg/L, rainbow trout accumulated thiocyanate at a rate that would put 50% of the exposed population at risk of lethal effects following a one to two month exposure. More recently, Kevan and Dixon (1991) reported that exposures of 85 mg/L produce increased deformities in developing embryos of rainbow trout. Green et al (1990) reported 50% reduction in feeding by freshwater snails exposed to 5 mg/L. Smith and Mudder (1991) reported a range of LC50s for fish from 50-200 mg/L. For this analysis, 1.0 mg/L is used as a low (approximately "chronic") concentration of concern and 24 mg/L as a higher concentration. A no effect level of thiocyanate is likely to be lower than 1.0 mg/L.

The toxicity of the predicted **cyanate** concentration was evaluated using several sources. Cyanate is reported to be 1/1,000 as toxic as free cyanide (Response to comments 94.21; Kensington project EIS) which suggests levels of 5.2 and 22 mg/L (based on Water Quality Criteria for cyanide) might be protective. Palmes (1993) review of cyanate toxicity suggests a lower limit of 75 mg/L for fish (lethal to creek chub) which is expected to be even lower for salmonids, based on salmonid sensitivity to cyanide. Crustaceans experience mortality at 18 mg/L (Dauchy et al., 1980). Although cyanate is considered more toxic than thiocyanate (Smith and Mudder, 1991), its toxicity is affected by hardness (reports exist of both dramatic decreases and moderate increases with hardness). LC50s for fish range from

13-82 mg/L, but concentrations of 85 mg/L at hardness greater than 200 mg/L have shown no mortality, even though another study found an LC50 at 24 mg/L for hardness of 250 mg/L and an LC50 of 95 mg/L for hardness of 200 mg/L. The low to high range chosen for this analysis is from 18 mg/L (for crustaceans) to 95 mg/L (high hardness, and one of the highest LC50s).

A search of EPA's AQUIRE (Aquatic toxicity information retrieval) database on 2/15/94 yielded information on six different **xanthates**. The crustacean *Daphnia magna* was generally the most sensitive organism. Immobility in this species was reported at concentrations from 0.35 to 3.7 mg/L (for sodium ethyl and sodium isopropyl xanthate, respectively). Rainbow trout exhibited mortality over a concentration range from 0.3 to 180 mg/L for sodium isopropyl xanthate, but LC50s ranged from 13 to 320 mg/L for other xanthates. Hawley (1977) summarized effects of a variety of xanthates on *Daphnia*, a shiner, and fathead minnow. *Daphnia* was consistently most susceptible; xanthates were generally described as having high toxicity eliciting effects in the range from 0.1 to 1.0 mg/L or from 0.56 to 10 mg/L. The shiner's susceptibility varied depending on the specific xanthate. Sodium isopropyl, sodium ethyl, and potassium ethyl xanthate had high toxicity (0.01 to 0.1 mg/L), whereas sodium isobutyl and potassium amyl xanthate had moderate toxicity (10 to 100 mg/L). The fathead minnow had the widest range of susceptibility. Susceptibility was greatest for sodium isopropyl, sodium ethyl, and potassium ethyl xanthate (approximately 0.1 to 1.0 or 0.18 to 1.8 mg/L) and least for potassium hexyl xanthate (100 to 1000 mg/L). The range selected for analysis here is from 0.01 (effects on shiner by three xanthates) to 5.6 (median of upper ranges of effects reported by Hawley (1977). This range encompasses much of the AQUIRE data for sensitive species.

G. Results of Pore Water Evaluation

The pore water can be characterized as potentially highly toxic based on the analysis presented here (note how exceedances for Hg and cyanide are more than two orders of magnitude above chronic water quality criteria). Tables 25 and 26 summarize the results of comparing projected pore water concentrations with water quality criteria and aquatic toxicity data. These results indicate that several of the metals (Cu, Hg, and Cr if it is present exclusively as Cr(VI)), cyanide, and ammonia would exceed applicable acute water quality criteria and that thiocyanate would occur at a level likely to be associated with aquatic toxicity. In addition, Cd, Se, cyanate, and xanthates are of concern due to exceedance of chronic water quality criteria or more conservative concentrations associated with aquatic toxicity.

H. Evaluation of Contaminants in Sediments

Contaminant concentrations in sediment (see Table 27) are expressed as mg-contaminant/kg-sediment dry weight. Potential toxicity of metals and cyanide in the impoundment sediments was evaluated using available "benchmarks" also based on dry weight. Each of these benchmarks has limitations, but when taken together, they provide an indication of which contaminants are likely to exert adverse effects on any kind of benthic community that may develop on and in the impoundment sediments. Two sets of benchmarks were taken from the summary provided in Ecology (1991). First, were the Provincial Sediment Quality Guidelines (both a lowest effect and a severe effect, with benchmarks for silver and cyanide based on dredged disposal criteria) and, second, the Wisconsin Department of Natural Resources benchmark numbers (Ecology, 1991, discusses the basis and limitations of these benchmarks). A third set of draft numbers was provided by the Great Lakes Program (C. Ingersoll, pers. comm.). These preliminary numbers are the effects range medians (ERMs) for associations between total metals (dry weight) in whole sediment samples from the Great Lakes and other regions and toxicity to the freshwater amphipod, *Hyallela azteca*. These numbers are based on approximately 60 to 70 samples and are used here only for those metals for which the ERM could correctly classify samples as having effects or not, 70 to 80 per cent of the time. It must be noted that the numbers used here are only preliminary and are subject to some final adjustment.

Results of comparing predicted sediment concentrations of metals and cyanide with three sets of benchmarks for freshwater sediments are presented in Table 27. All parameters are expected to exceed at least one of the benchmarks. Contaminants of particular concern include cyanide, copper, and cadmium, which exceeded benchmarks by over an order of magnitude (consistently so for cadmium). Lead, too, exceeded all three benchmarks. In general, these comparisons indicate moderate to very high levels of toxicity will be present in the tailings sediments (note that all parameters evaluated exceeded the Provincial SQ Guidelines for a lowest-effect).

I. Evaluation of Metals in the Water Column

Metals concentrations in the water column were predicted using the WASP4 model as described in Chapter VI. The same adjustments for hardness were made as done previously for the pore water (for cadmium, copper, lead, and zinc).

The modeling results predicted virtually identical values for dissolved and total metals in the water column (Table 28), suggesting very little metal would be bound to suspended sediments. It should be noted, however, that this analysis

Table 27. Comparison of projected sediment concentrations of metals and cyanide with several potential freshwater sediment benchmarks (Provincial Sediment Quality Guidelines, Wisconsin DNR benchmarks, and draft Great Lakes Effects Range Median for a freshwater amphipod). Concentrations exceeding at least one benchmark are enclosed in boxes. Ratios (concentration/benchmark) from 1-10 are enclosed in lightly outlined boxes; ratios greater than 10 are enclosed in heavily outlined boxes.

Parameter	Impoundment Sediments *mg/kg*	Provincial SQ Guidelines (a) Lowest-Effect		Provincial SQ Guidelines (a) Severe-Effect		WI DNR(b)		Draft Great Lakes ERM(c)	
		mg/kg	*Ratio*	*mg/kg*	*Ratio*	*mg/kg*	*Ratio*	*mg/kg*	*Ratio*
Ag	3	0.5	6	-		-		-	
As	9.8	6	1.6	33	0.3	10	1.0	48	0.2
Cd	11	0.6	18	10	1.1	1	11	3.9	2.8
Cr	35	26	1.3	110	0.3	100	0.4	274	0.1
Cu	260	16	16	110	2.4	100	2.6	187	1.4
Fe	46,000	20,000	2.3	40,000	1.2	-		-	
Hg	0.27	0.2	1.4	2	0.1	0.1	2.7	-	
Mn	1600	460	3.5	1100	1.5	-		-	
Ni	26	16	1.6	75	0.3	100	0.3	-	
Pb	200	31	6.5	250	0.8	50	4	99	2.0
Se	1	-		-		1	1	-	
Zn	530	120	4.4	820	0.6	100	5.3	-	
CNtot	42.4	0.1	424	-		-		-	

(a) - Dry weight basis. Taken from Jaagumagi, 1993, and Ecology, 1991. Ag & CN numbers from Dredged Disposal Criteria

(b) - Taken from Ecology, 1991. Based on background

(c) - Preliminary Effects Range Medians for associations between total metals (dry wt) in whole sediment samples from the Great Lakes and other regions and toxicity to Hyallela azteca. Based on approximately 60-70 samples. Results for contaminants for which the ERM could correctly classify samples 70-80% of the time (C. Ingersoll, pers. comm. 05/11/94. Note that the numbers presented here are only preliminary and subject to some final adjustment).

Table 28. Comparison of projected water column concentrations (from Tables 12 & 13) with ambient water quality criteria. Concentrations exceeding acute or chronic criteria and ratios exceeding unity are enclosed in boxes.

	Impoundment Water Column Concentration		Water Quality Criterion (a)		Ratio of Parameter to Criterion Concentration	
Parameter	Dissolved *ug/L*	Total *ug/L*	Acute *ug/L*	Chronic *ug/L*	Acute	Chronic
Scenarios 1-12						
As(III)	0.06	0.06	360	190	0.0	0.0
Cd	0.05	0.05	34	5	0.0	0.0
Cr(III)	0.09	0.09	8246	1003	0.0	0.0
Cr(VI)	0.09	0.09	16	11	0.0	0.0
Cu	5.9	5.9	106	60	0.1	0.1
Hg	[0.022]	[0.022]	2.4	0.012	0.0	[1.8]
Pb	0.084	0.084	919	36	0.0	0.0
Se	0.072	0.072	20	5	0.0	0.0
Zn	0.387	0.387	586	531	0.0	0.0
Scenario 13 - Includes bioturbation						
As(III)	0.61	0.61	360	190	0.0	0.0
Cd	0.43	0.43	34	5	0.0	0.1
Cr(III)	0.84	0.84	8246	1003	0.0	0.0
Cr(VI)	0.84	0.84	16	11	0.1	0.1
Cu	56	56	106	60	0.5	0.9
Hg	[0.215]	[0.215]	2.4	0.012	0.1	[18]
Pb	0.844	0.844	919	36	0.0	0.0
Se	0.686	0.686	20	5	0.0	0.1
Zn	3.7	3.7	586	531	0.0	0.0
Scenarios 14 & 15 - Includes groundwater moving up through pore water into water column						
As(III)	2	2	360	190	0.0	0.0
Cd	1.4	1.4	34	5	0.0	0.3
Cr(III)	2.8	2.8	8246	1003	0.0	0.0
Cr(VI)	2.8	2.8	16	11	0.2	0.3
Cu	[190]	[190]	106	60	[1.8]	[3.2]
Hg	[0.71]	[0.71]	2.4	0.012	0.3	[59]
Pb	2.650	2.650	919	36	0.0	0.1
Se	2.29	2.29	20	5	0.1	0.5
Zn	11.9	11.9	586	531	0.0	0.0

(a) Based on hardness of 670 and Water Effects Ratio of 1, following EPA 1992; As(III), Cr(VI), Hg, Fe, and Se are not dependent on hardness; see discussion in text on application of Water Quality Criteria.

assumed no resuspension of bottom sediments. Potential effects of resuspended sediments on wildlife is evaluated in a later section. The results from comparing modeled concentrations of contaminants in the water column with ambient water quality criteria (Table 28) show that mercury is predicted to exceed the chronic criterion. Furthermore these results show that when the influence of bioturbation and groundwater flow (Scenarios 13-15) are included in the modeling, copper concentrations are likely to exceed the acute criterion as well.

J. Evaluation of Potential Effects on Wildlife

1. Selection of Contaminants

Cadmium, mercury, selenium, and lead were selected, in consultation with EPA-ORD (S. Norton, pers. comm.), as contaminants of concern because of their potential to bioaccumulate; bioconcentration factors (BCFs; concentration in tissue [wet-weight] divided by concentration in water) for these metals are generally greater than 1000 (see Table 29 below). **Mercury** in particular, is of high concern. The mercury concentration in pore water is projected to exceed the acute water quality criterion (Table 25); sediment concentrations of mercury are predicted to exceed two freshwater sediment benchmarks (Table 27); and mercury in the water column is projected to exceed the chronic water quality criterion (Table 28). **Cadmium** is expected to exceed all the freshwater sediment benchmarks (Table 27) as well as the chronic water quality criterion applied to pore water (Table 25). **Selenium** is predicted to be at toxic levels in the pore water and sediment (Tables 25 and 27) and **Lead** is expected to exceed several of the freshwater sediment benchmarks (Table 27). These four metals represent a range in level of concern for potential effects on wildlife, with mercury of highest, selenium and lead lowest, and cadmium in between. In addition, wildlife toxicity data for these metals are available (Eisler, 1985a, 1985b, 1987, 1988).

2. Selection of Species

Three species, the kingfisher, spotted sandpiper, and river otter, were selected in consultation with EPA-ORD (S. Norton, pers. comm.). These species are expected to or are known to inhabit the general Sheep Creek area (see Section D.6). Information on conducting exposure assessments is available for these three species (EPA, 1993b). The kingfisher was chosen to represent a bird with a relatively small home range that feeds primarily on fish. The sandpiper was chosen to represent a bird that also forages along shorelines but has a high rate of incidental ingestion of sediment. The river otter represents a

Table 29. Bioconcentration factors for accumulation of mercury, cadmium, lead, and selenium from water into fish, invertebrates, and amphibians.

Organism	BCF	Source
		Mercury
Fish	95,000	EPA 1993a, p 2-7, methylHg in fish tissue is related to total Hg in water column, lists 60,000 and 130,000 depending on fish trophic status; average of 95,000 used here.
Invertebrates	95,000	assume same as for fish; freshwater BCFs in EPA 1985 range from 300 to 8000, with marine BCFs up to 350,000.
Amphibians	95,000	assume same as for fish
		Cadmium
Fish	1,500	Eisler, 1985a, p 27, for trout exposed for 3 months to 10 ug/L; mean of measurements made on various tissues.
Invertebrates	1,200	Ibid., p 28; mean of 6 values for various species.
Amphibians	1,500	assume same as for fish
		Selenium
Fish	1,000	Saiki, 1986, p 30, mean BCF for Kesterson was >1000. Eisler, 1985, reports 460 and 3,300 for fish exposed to Se at up to 3.3 ug/L and 40 ug/L.
Invertebrates	322	AQUIRE search, 5/5/94, for *Daphnia* exposed to 5 ug/L.
Amphibians	1,000	assume same as for fish
		Lead
Fish	726	Eisler, 1988, p 61, for trout exposed for 7 days to 3.5 ug/L
Invertebrates	5,000	Ibid., p 59; range from 1000 to 9000
Amphibians	726	assume same as for fish

mammal that may feed primarily on fish. The river otter is also characterized as a "noteworthy indicator of bioaccumulative pollution in aquatic ecosystems" and it is known to ingest mud and other debris while probing and feeding in bottom sediments (EPA, 1993b).

3. Selection of Exposure Pathways

Exposure of these three species to the four metals was evaluated for the ingestion pathway which generally included: (1) drinking the surface water including resuspended bottom sediments at a concentration of 0.385 mg-sed-dry-wt/L (see Table 32; TSS), (2) eating fish, invertebrates, and amphibians, and, where appropriate, (3) incidental ingestion of sediments.

The kingfisher is expected to forage in relatively shallow water, so that uptake from fish, invertebrates, and amphibians is based on bioconcentration from contaminants in the water column. The kingfisher is not expected to ingest any sediment other than sediment present in its drinking water. An additional scenario evaluated assumes the invertebrates consumed by the kingfisher would come from the bottom of the impoundment where they would bioaccumulate metals from the pore water (see Table 30).

The **sandpiper** would be expected to ingest sediments that are resuspended and deposited in shallow areas along the shoreline. It is assumed that these shoreline sediments would all come from tailings. The diet of the sandpiper is assumed to be exclusively invertebrates (see Table 33). Two scenarios are evaluated, first, assuming invertebrates would not be exposed to pore water and then assuming they would be exposed.

The **river otter** is expected to forage in deeper water, so that uptake from invertebrates is based on bioconcentration from contaminants in the pore water, whereas uptake from fish and amphibians is based on bioconcentration of contaminants in the water column. The river otter is also expected to ingest bottom sediments. A second scenario assumes half of the invertebrates consumed are from shallow areas and are not exposed to pore water.

Dermal absorption and inhalation pathways were not evaluated because they are expected to have a low contribution to the total exposure to the metals in the proposed impoundment.

Relevant data on exposure were taken from EPA (1993b) for each of the species as discussed below.

4. Exposure of Wildlife to Metals

The general approach to evaluating exposure to the metals was driven somewhat by the toxicity data available for comparison. The toxicity data were of three kinds, doses (mg-contaminant/kg-bw/day), dietary concentrations (mg-contaminant/kg-food-wet-wt), and water concentrations (mg-contaminant/L). Corresponding exposure data were developed for the dose and dietary concentration scenarios as described below; water concentrations considered safe to wildlife were compared directly with the predicted water column concentrations. This section refers repeatedly to the analyses done for the three species (Tables 27-37) in terms of defining parameters and the equations used to calculate exposure.

a. Ingested Doses

As discussed in detail in this section, consumption rates were developed for water, food, and sediment ingestion on a body weight basis. Parameters used in exposure calculations are found in Tables 30, 33, 34, 36, and 39; data developed for the river otter are used here to provide an example of the approach used. Body weights (BW) and water ingestion rates (IRW) were taken directly from EPA (1993b). Food ingestion rates (on a wet weight basis, IRFW) were either taken directly from EPA (1993b), or estimated using the approach outlined in Figure 4-7 of that document (this allows one to estimate food ingestion rate based on free-living metabolic rate and dietary composition). Sediment ingestion rates were estimated by multiplying the food ingestion (wet weight) rate (IRFW) by the ratio of sediment to food (STF) in the diet, as explained below.

i. Water Ingestion Rate and Dose

Water ingestion rates were available in EPA (1993b). They were multiplied by water column concentrations to obtain the dose from drinking water (DW; mg-contaminant/kg-bw/day). The ingested water will have suspended sediments in it and this contribution is included in the present analysis. The separate estimate of sediment ingestion is based on the amount of sediment measured in scat samples and presumably integrates over all routes of incidental sediment ingestion. Although the overall sediment ingestion estimate may already include suspended sediment in the drinking water, the separate estimates presented here allow a comparison of the relative influence of this pathway (suspended sediments in drinking water).

ii. Food Ingestion Rate and Dose

The overall **food ingestion rate** was taken either directly from EPA (1993b), or, as in the case of the river otter, calculated following the example calculation in Figure 4-7 of that document. Estimation of the food ingestion rate (IRFW; wet-wt basis) for the river otter is shown in Table 36, which calculates the rate as follows:

Table 30. Source of input values for parameters used to estimate kingfisher exposure.

Parameter	Variable	Value	Units	Source(EPA1993a)
Body weight	BW	0.147	kg/bw	p 2-176; ave of 3 means for adults
Water intake	IRW	0.11	L/kg-bw/day	p 2-176; adult rate
Food ingestion	IRFW	0.5	kg-food-wet-wt/kg-bw/day	p 2-176
Sediment ingestion				Assumed no incidental sediment ingestion other than sediment in drinking water
Proportion of diet = fish	PF	0.46	kg-fish/kg-food	p 2-177; first entry; sum of trout and non-trout
Proportion of diet = invertebrates	PI	0.24	kg-inv/kg-food	p 2-177; first entry; sum of insects and crustacea
Proportion of diet = amphibians	PA	0.27	kg-amph/kg-food	p 2-177; first entry
Home range	HR	0.39	km of shoreline	p 2-178; non-breeding

Table 31. Source of wildlife toxicity data used to estimate doses, dietary concentrations, and drinking water concentrations of metals non-toxic to kingfisher and sandpiper (same value used except where indicated by KF and SP).

	Variable	Value	Units	Source
Hg	NTOXdose	0.032	mg-Hg/kg-bw/day	EPA 1993a, p 2-7; proposed avian NOAEL (note: a factor of 0.1 is recommended for application to kingfisher).
	NTOXdiet	0.05	mg-Hg/kg-diet	Eisler 1987, Table 11, p 71; no effect value for birds ranges from 0.05 to <0.1.
	NTOXwater	$1x10^{-7}$	mg-Hg/L	KF: EPA 1993a, p 2-8; proposed no effect value for kingfisher.
		$1.8x10^{-7}$	mg-Hg/L	SP: EPA 1993a, p 2-8; proposed avian value.
Cd	NTOXdose	42	mg-Cd/kg-bw/day	Eisler 1985a, p 34; mallard dietary value of 200 mg-Cd/kg-diet considered here to be a LOAEL (divided by 2 to estimate NOAEL). Converted to dose (see text). No species extrapolation factor used.
	NTOXdiet	0.1	mg-Cd/kg-diet	Eisler 1985a, p 34; wildlife dietary levels above this value should be viewed with caution.
	NTOXwater	0.003	mg-Cd/L	Eisler 1985a, p iii; above this level, adverse effects are pronounced or probable.
Se	NTOXdose	2.1	mg-Se/kg-bw/day	Eisler 1985b, p 39; 5 ppm NOAEL; no effect on mallard and progeny; converted to dose (see text). No species extrapolation factor used.
	NTOXdiet	5	mg-Se/kg-bw/day	Eisler 1985b, p 29; birds are sensitive to Se in diet at concentrations above 6 ppm. 5ppm is a NOAEL for the mallard. No species extrapolation factor used.
Pb	NTOXdose	2.8	mg-Pb/kg-bw/day	Eisler 1988, Table 7, p 84; no effect on starlings, effects on other species above this level of organolead. No species extrapolation factor used.
	NTOXdiet	500	mg-Pb/kg-diet	Eisler 1988, p86; precocial birds NOAEL (growth) at 500 mg/kg.
	NTOXwater	0.1	mg-Pb/L	Eisler 1988, Table 7, p 83; NOAEL for egg production in turtle-dove. No species extrapolation factor used.

Table 32. Predicted toxicity of impoundment metals to the belted kingfisher (Ceryle alcyon). Calculated variables are italicized. Sources of exposure and toxicity data are given in Tables 25 and 27-31.

EXPOSURE PARAMETERS:

KINGFISHER	Units	Variable	Value
Body weight =	kg-bw	BW	0.147
Ingestion rate: water =	L-water/kg-bw/day	IRW	0.11
Ingestion rate: food (wet-wt)=	kg-food-wet-wt/kg-bw/day	IRFW	0.5
Proportion of diet that is Fish =	kg-fish-wet-wt/kg-food-wet-wt	PF	0.46
Proportion of diet that is Invertebrates =	kg-inv-wet-wt/kg-food-wet-wt	PI	0.24
Proportion of diet that is Amphibians =	kg-amph-wet-wt/kg-food-wet-wt	PA	0.27
Home range =	km shoreline used	HR	0.39
IMPOUNDMENT			
Impoundment Shoreline =	km shoreline available	SL	4
Suspended Sediment =	kg-sed-dry-wt/L	SS	0.000385

			CONTAMINANT			
CONTAMINANT			Hg	Cd	Se	Pb
Concentration in SEDIMENT =	mg-cont/kg-sed-dry-wt	CSED	0.27	11	1	200
Concentration in WATER COL =	mg-cont/L	CW	2.25E-05	4.50E-05	7.20E-05	8.3E-05
BCF-water into fish =	(mg-cont/kg-fish)/(mg-cont/L)	BCF-F	95,000	1500	1000	726
BCF-water into inverts =	(mg-cont/kg-inv)/(mg-cont/L)	BCF-I	95,000	1200	322	5000
BCF-water into amphibians =	(mg-cont/kg-amph)/(mg-cont/L)	BCF-A	95,000	1500	1000	726

EXPOSURE CALCULATIONS:

			Hg	Cd	Se	Pb
VIA DIET						
Concentration in Fish: CW*BCF-F=	(mg-cont)/kg fish (wet wt)	*CF*	2.14	0.0675	0.072	0.060
Concentration in Invertebrates: CW*BCF-I=	(mg-cont)/kg inv (wet wt)	*CI*	2.14	0.054	0.023	0.415
Concentration in Amphibians: CW*BCF-A=	(mg-cont)/kg amphibs (wet wt)	*CA*	2.14	0.0675	0.072	0.060
Dose from Fish: CF * IRFW*PF =	(mg-cont)/(kg-bw)/day	*DF*	0.492	0.0155	0.0166	0.014
Dose from Benthic Inverts: CBI * IRFW*PI =	(mg-cont)/(kg-bw)/day	*DI*	0.257	0.006	0.003	0.050
Dose from Amphibs: CA * IRFW*PA =	(mg-cont)/(kg-bw)/day	*DA*	0.289	0.0091	0.010	0.0081
VIA DRINKING WATER						
Dose from Water: CW*IRW =	(mg-cont)/(kg-bw)/day	*DW*	2.48E-06	4.95E-06	7.92E-06	9.13E-06
Dose from Sed in Water: SS*IRW*CSED =	(mg-cont)/(kg-bw)/day	*DSW*	1.14E-05	4.66E-04	4.24E-05	8.47E-03

Table 32 (continued). Predicted toxicity of impoundment metals to the belted kingfisher.

			CONTAMINANT			
SUM ACROSS PATHWAYS			**Hg**	**Cd**	**Se**	**Pb**
DF + DI + DA + DW + DSW=	(mg-cont)/(kg-bw)/day	*EXPdose*	1.04	0.032	0.029	0.080
CF*PF + CI*PI + CA*PA =	(mg-cont)/(kg-diet)	*EXPdiet*	2.07	0.062	0.058	0.144
CW =	(mg-cont)/L	*EXPwater*	2.25E-05	4.5E-05	7.2E-05	8.3E-05
ADJUSTMENT FOR HOME RANGE						
Home Range adjustment (if >1) =	HR/SL	*HRA*	--	--	--	--
TOXICITY PARAMETERS:						
Dose Non-Toxic to kingfisher =	(mg-cont)/(kg-bw/day)	NTOXdose	0.032	42	2.1	2.8
Diet Non-Toxic to kingfisher =	(mg-cont)/kg-diet	NTOXdiet	0.050	0.1	5	500
Water Column conc non-toxic to kingfisher =	(mg-cont)/L	NTOXwater	1E-07	0.003	--	0.10
HAZARD INDEX CALCULATION:			**Hg**	**Cd**	**Se**	**Pb**
Exposure/Toxicity = EXP/(NTOXdose)		*HIdose*	**32**	**0.0**	**0.0**	**0.0**
Or = EXPdiet/NTOXdiet		*HIdiet*	**41**	**0.6**	**0.0**	**0.0**
Or = EXPwater/NTOXwater		*HIwater*	**225**	**0.0**	--	**0.0**

IF BENTHIC INVERTEBRATES EXPOSED TO PORE WATER CONCENTRATIONS ARE CONSUMED:

			Hg	Cd	Se	Pb
Concentration in PORE water =	mg-cont/L	CPW	0.005	0.01	0.016	0.02
EXPOSURE VIA DIET						
Concentration in Benthic Invertebrates: CPW*BCF-I=	(mg-cont)/kg inv (wet wt)	*CIPW*	475	12	5.15	100
Dose from Benthic Inverts: CIPW * IRFW*PI =	(mg-cont)/(kg-bw)/day	*DIPW*	57	1.44	0.618	12
SUM ACROSS PATHWAYS						
DF + DI + DA + DW + DSW=	(mg-cont)/(kg-bw)/day	*EXPdose*	57.8	1.47	0.645	12.0
CF*PF + CI*PI + CA*PA =	(mg-cont)/(kg-diet)	*EXPdiet*	116	2.93	1.29	24.0
HAZARD INDEX CALCULATION:			**Hg**	**Cd**	**Se**	**Pb**
Exposure/Toxicity = EXP/(NTOXdose)		*HIdose*	**1806**	**0.0**	**0.3**	**4.3**
Or = EXPdiet/NTOXdiet		*HIdiet*	**2311**	**29**	**0.3**	**0.0**

Table 33. Estimating food ingestion rate for the sandpiper based on free-living metabolic rate and diet of invertebrates; input values and method of calculation from EPA 1993 (see p4-17 for estimating ME).

		Units	Variable	Value
1. Estimate Field Metabolic Rate Normalized to Body Weight				
		kcal/kg-bw/day	NFMRkg	448
2. Normalize to Body Weight in g				
	NFMRkg*1kg/1000g =	kcal/g-bw/day	NFMR	0.448
3. Estimate Average Metabolizable Energy				
	GE x AE =	kcal-assim/g-inv-wet-wt	ME	0.87

Dietary Item	*Gross Energy* GE kcal-gross/g-inv-wet-wt	*Assimil. Efficiency* AE kcal-assim/kcal-gross	*Metabolizable Energy* ME=GE x AE kcal-assim/g-inv-wet-wt
Invertebrates	1	0.87	0.87

		Units	Variable	Value
4. Estimate Total Normalized Ingestion Rate				
	NFMR/ME =	g-inv-wet-wt/g-bw-day	IRFW	0.515

Source of Input Parameters (in EPA 1993)

NFMRkg	p 2-152; average of means for adult free-living males and females.
GE	p 4-13, Table 4-1; median number for invertebrates.
AE	p 4-15, Table 4-3; assimilation efficiency of invertebrates.

Table 34. Source of input values for parameters used to estimate sandpiper exposure.

Parameter	Variable	Value	Units	Source(EPA1993b; spotted sandpiper)
Body weight	BW	0.0425	kg/bw	p 2-152; ave of means for adult males and females.
Water intake	IRW	0.165	L/kg-bw/day	Ibid.
Food ingestion	IRFW	0.515	kg-food-wet-wt/ kg-bw/day	calculated in Table 33.
Proportion of diet = sediment (dry wt)	PDS	0.18	kg-sed-dry-wt/ kg-(sed+food)-dry-wt	p 4-20; value for western sandpiper (median of 4 species).
Proportion of diet = invertebrates	PI	1	kg-inv/kg-food	p 2-152; dietary composition.
Home range	HR	2.5	km of shoreline	p 2-152; territory of 0.25 hectares; assume shoreline width of 1m, the HR = (0.25)*(10,000m2)/1m = 2.5 km.
Proportion of invertebrates = water	PWI	0.78	(wet-wt - dry-wt)/ wet-wt	p 4-13; median of 5 values for invertebrates.

Table 35. Predicted toxicity of impoundment metals to the sandpiper (Actitis macularia). Sources of exposure and toxicity data are given in Tables 25, 27-29, 31, 33, and 34. Calculated variables are italicized.

EXPOSURE PARAMETERS:

SANDPIPER	**Units**	**Variable**	**Value**			
Body weight =	kg-bw	BW	0.0425			
Ingestion rate: water =	L-water/kg-bw/day	IRW	0.165			
Ingestion rate: food (wet-wt) =	kg-food-wet-wt/kg-bw/day	IRFW	0.515			
Proportion of diet that is Invertebrates =	kg-inv/kg-food	PI	1			
Proportion of Invertebrates that is water =	(wet-wt - dry-wt)/wet-wt	PWI	0.78			
Proportion of diet that is sediment =	kg-sed-dry-wt/kg-(sed+food)-dry-wt	PDS	0.18			
Home range =	km shoreline	HR	2.5			
IMPOUNDMENT						
Impoundment Shoreline =	km shoreline	SL	4			
Suspended Sediment =	kg sed/L	SS	0.000385			
			CONTAMINANT			
CONTAMINANT			**Hg**	**Cd**	**Se**	**Pb**
Concentration in SEDIMENT =	mg-cont/kg-sed	CSED	0.27	11	1	200
Concentration in WATER COL =	mg-cont/L	CW	2.25E-05	4.50E-05	7.20E-05	8.30E-05
BCF-water into inverts =	(mg-cont/kg-inv)/(mg-cont/L)	BCF-I	95,000	1200	322	5000

EXPOSURE CALCULATIONS:

SEDIMENT INGESTION						
Proportion of diet in dry weight = (1-PWI) =	kg-food-dry-wt/kg-food-wet-wt	*PDD*	0.22			
Ratio of sediment to food (dry-wt)= PDS/(1-PDS) =	kg-sediment-dry-wt/kg-food-dry-wt	*STFdry*	0.22			
Ratio of sediment to food (wet-wt)= STFdry*PDD =	kg-sediment-dry-wt/kg-food-wet-wt	*STFwet*	0.0483			
Ingestion rate: sediment = IRFW*STFwet =	kg-sed-dry-wt/kg-bw/day	*IRS*	0.0249			
VIA DIET						
Concentration in Invertebrates: CW*BCF-I=	(mg-cont)/kg inv (wet wt)	*CI*	2.14	0.0540	0.0232	0.415
Dose from Benthic Inverts: CI * IRFW =	(mg-cont)/(kg-bw)/day	*DI*	1.10	0.0278	0.0119	0.214
Dose from Sediment: CSED*IRS =	(mg-cont)/(kg-bw)/day	*DS*	0.00671	0.274	0.0249	4.97

Table 35 (continued). Predicted toxicity of impoundment metals to the sandpiper.

VIA DRINKING WATER						
Dose from Water: CW*IRW =	(mg-cont)/(kg-bw)/day	*DW*	3.71E-06	7.43E-06	1.19E-05	1.37E-05
Dose from Sed in Water: SS*IRW*CSED =	(mg-cont)/(kg-bw)/day	*DSW*	1.72E-05	6.99E-04	6.35E-05	0.0127
SUM ACROSS PATHWAYS						
DI + DS + DW + DSW =	(mg-cont)/(kg-bw)/day	*EXPdose*	1.11	0.302	0.0369	5.20
CI + STFwet*CSED =	(mg-cont)/(kg-food)	*EXPdiet*	2.15	0.585	0.0715	10.1
CW =	(mg-cont)/L	*EXPwater*	2.25E-05	4.50E-05	7.20E-05	8.30E-05
ADJUSTMENT FOR HOME RANGE:						
Home Range adjustment (if >1) =	HR/SL	HRA	–	–	–	–
TOXICITY PARAMETERS:						
Dose Non-Toxic to sandpiper =	(mg-cont)/(kg-bw/day)	NTOXdose	0.032	42	2.1	2.8
Diet Non-Toxic to sandpiper =	(mg-cont)/kg-diet	NTOXdiet	0.05	0.1	5	500
Water Column conc non-toxic to sandpiper =	(mg-cont)/L	NTOXwater	1.80E-07	0.003	–	0.01
HAZARD INDEX CALCULATION:			**Hg**	**Cd**	**Se**	**Pb**
Exposure/Toxicity = EXPdose/NTOXdose		*HIdose*	**35**	**0.0**	**0.0**	**1.9**
Or = EXPdiet/NTOXdiet		*HIdiet*	**43**	**5.9**	**0.0**	**0.0**
Or = EXPwater/NTOXwater		*HIwater*	**125**	**0.0**	**–**	**0.0**
IF INVERTEBRATES EXPOSED TO PORE WATER ARE CONSUMED:						
Concentration in PORE water =	mg-cont/L	CPW	0.005	0.01	0.016	0.02
EXPOSURE VIA DIET						
Concentration in Benthic Invertebrates: CPW*BCF-I=	(mg-cont)/kg inv (wet wt)	*CIPW*	475	12.0	5.15	100
Dose from Benthic Inverts: CIPW* IRFW*PI =	(mg-cont)/(kg-bw)/day	*DIPW*	245	6.18	2.65	51.5
SUM ACROSS PATHWAYS						
DIPW + DS + DSW =	(mg-cont)/(kg-bw)/day	*EXPdose*	245	6.45	2.68	56.5
CIPW + STFwet*CSED =	(mg-cont)/(kg-food)	*EXPdiet*	475	12.5	5.20	110
HAZARD INDEX CALCULATION:			**Hg**	**Cd**	**Se**	**Pb**
Exposure/Toxicity = EXP/(NTOXdose)		*HIdose*	**7644**	**0.2**	**1.3**	**20**
Or = EXPdiet/NTOXdiet		*HIdiet*	**9500**	**125**	**1.0**	**0**

Table 36. Estimating food ingestion rate for the river otter based on free-living metabolic rate and dietary composition. Input values and method of calculation from EPA 1993 (see p4-17 for estimating ME).

	Units	Variable	Value
1. Estimate Field Metabolic Rate Normalized to Body Weight			
	kcal/kg-bw/day	NFMRkg	180
2. Normalize to Body Weight in g			
NFMRkg*1kg/1000g =	kcal/g-bw/day	NFMR	0.18
3. Estimate Average Metabolizable Energy			
Sum (P x AE)=	kcal-assim/g-inv-wet-wt	AveME	0.98

Dietary Item	Proportion of Diet P	*Gross Energy* GE kcal-gross/g-inv-wet-wt	*Assimil. Efficiency* AE kcal-assim/kcal-gross	*Metabolizable Energy* ME=GE x AE kcal-assim/g-inv-wet-wt	*Weighted ME* P x ME kcal-assim/g-inv-wet-wt
Fish	0.53	1.2	0.91	1.09	0.579
Invertebrates	0.26	1	0.87	0.87	0.226
Amphibians	0.16	1.2	0.91	1.09	0.175

	Units	Variable	Value
4. Estimate Total Normalized Ingestion Rate			
NFMR/AveME =	g-inv-wet-wt/g-bw-day	IRFW	0.184

Source of Input Parameters (in EPA 1993)

NFMRkg	p 2-264; means for free-living otters.
GE	p 4-13, Table 4-1; median number for invertebrates.
AE	p 4-15, Table 4-3; assimilation efficiency of invertebrates.

Table 37. Source of input values for parameters used to estimate river otter exposure.

Parameter	Variable	Value	Units	Source(EPA1993b;river otter)
Body weight	BW	10	kg/bw	p 2-264; midpoint of range for adult males and females.
Ingestion rate: water	IRW	0.081	L/kg-bw/day	p 2-264; average of means for adult males and females.
Ingestion rate: food	IRFW	0.184	kg-food-wet-wt/ kg-bw/day	calculated in Table 36.
Proportion of diet = sediment (dry wt)	PDS	0.094	kg-sed-dry-wt/ kg-(sed+food)-dry-wt	p 4-20; value for raccoon.
Proportion of diet = fish	PF	0.53	kg-fish/kg-food	p 2-266, assuming % vol=% wt; combining 3 entries.
" = invertebrates	PI	0.258	kg-inv/kg-food	p 2-266, " .
" = amphibians	PA	0.16	kg-amph/kg-food	p 2-266, " .
Home range	HR	28	km of shoreline	p 2-266; midpoint of range (10-78 km) for adult and young, male and female.
Proportion of fish = water	PWF	0.72	(wet-wt - dry-wt)/ wet-wt	p 4-13; average of 2 entries for fish.
" invertebrates = water	PWI	0.78	(wet-wt - dry-wt)/ wet-wt	p 4-13; median of 5 values for invertebrates.
" amphibians = water	PWA	0.85		p 4-13; value for frogs and toads.

Table 38. Source of wildlife toxicity data used to estimate doses, dietary concentrations, and drinking water concentrations of metals non-toxic to the river otter.

	Variable	Value	Units	Source
Hg	NTOXdose	0.016	mg-Hg/kg-bw/day	EPA 1993a, p 2-2; proposed mammalian NOAEL.
	NTOXdiet	1.1	mg-Hg/kg-diet	Eisler 1987, Table 11, p 71; no effect value for mammals.
	NTOXwater	1.5×10^{-6}	mg-Hg/L	EPA 1993a, p 2-3; proposed value for the otter.
Cd	NTOXdose	0.9	mg-Cd/kg-bw/day	Eisler 1985a, p 24; 1.8 ppm in oysters fed over 28 days to young mice is a possible chronic LOAEL; factor of 2 used to extrapolate to NOAEL. No species extrapolation factor used.
	NTOXdiet	0.1	mg-Cd/kg-diet	Eisler 1985a, p 34; wildlife dietary levels above this value should be viewed with caution.
	NTOXwater	0.003	mg-Cd/L	Eisler 1985a, p iii; above this level, adverse effects are pronounced or probable.
Se	NTOXdose	0.25	mg-Se/kg-bw/day	Eisler 1985b, p 29; chronic selenosis in cattle at 0.5 mg-Se/kg-bw, possible chronic LOAEL; lethal doses in other mammals occur above 1.5 mg-Se/kg-bw (p 28); factor of 2 used to extrapolate to NOAEL. No species extrapolation factor used.
	NTOXdiet	--	mg-Se/kg-diet	Eisler 1985b, no dietary estimate considered protective of wildlife or mammals.
	NTOXwater	0.05	mg-Se/L	Eisler 1985b, p 41; 50 ppb is drinking water level protective of livestock. No species extrapolation factor used.
Pb	NTOXdose	0.05	mg-Pb/kg-bw/day	Eisler 1988, Table 8, p 92; irreversible inhibition of ALAD activity in mouse at 0.05 mg-Pb/kg-bw/day; also, p 108, behavioral impairment in monkeys given 50 to 100 ug Pb/kg bw over 200 days. No species extrapolation factor used.
	NTOXdiet	--	mg-Pb/kg-diet	Eisler 1988, no dietary estimate considered protective of wildlife or mammals.
	NTOXwater	1.5	mg-Pb/L	Eisler 1988, Table 8, pp. 92-97; wide range of no effects, from 20 to 2000 mg/L; effects in the rat as low as 1.5 mg/L (possible NOAEL). No species extrapolation factor used.

Table 39. Predicted toxicity of impoundment metals to the river otter (Lutra canadensis). Calculated variables are in italics. Sources of exposure and toxicity data are in Tables 25, 27-29, 36, and 37.

EXPOSURE PARAMETERS:

RIVER OTTER	Units	Variable	Value
Body weight =	kg-bw	BW	10.0000
Ingestion rate: water =	L-water/kg-bw/day	IRW	0.081
Ingestion rate: food (wet-wt) =	kg-food-wet-wt/kg-bw/day	IRFW	0.184
Proportion diet that is fish=	kg-fish/kg-food	PF	0.53
Proportion of diet that is invertebrates =	kg-invertebrates/kg-food	PI	0.26
Proportion of diet that is amphibians =	kg-amphibians/kg-food	PA	0.16
Proportion of water in fish =	(wet-wt - dry-wt)/wet-wt	PWF	0.72
Proportion of water in invertebrates =	(wet-wt - dry-wt)/wet-wt	PWI	0.78
Proportion of water in amphibians =	(wet-wt - dry-wt)/wet-wt	PWA	0.85
Proportion of diet that is sediment =	kg-sed-dry-wt/kg-(sed+food)-dry-wt	PDS	0.094
Home range =	km shoreline	HR	28
IMPOUNDMENT			
Impoundment Shoreline =	km shoreline	SL	4
Suspended Sediment =	kg sed/L	SS	0.000385

			CONTAMINANT			
CONTAMINANT			Hg	Cd	Se	Pb
Concentration in SEDIMENT =	mg-cont/kg-sed	CSED	0.27	11	1	200
Concentration in PORE water =	mg-cont/L	CPW	0.005	0.01	0.016	0.02
Concentration in WATER COL =	mg-cont/L	CW	2.25E-05	4.50E-05	7.20E-05	8.30E-05
BCF-water into fish =	(mg-cont/kg-fish)/(mg-cont/L)	BCF-F	95000	1500	1000	726
BCF-water into inverts =	(mg-cont/kg-inv)/(mg-cont/L)	BCF-I	95000	1200	322	5000
BCF-water into amphibians =	(mg-cont/kg-amph)/(mg-cont/L)	BCF-A	95000	1500	1000	726

Table 39 (continued). Predicted toxicity of impoundment metals to the river otter

EXPOSURE CALCULATIONS:

SEDIMENT INGESTION						
Proportion of diet which is dry weight = (1-PWF)*PF + (1-PWI)*PI + (1-PWA)*PA	kg-food-dry-wt/kg-food-wet-wt	PDD	0.23			
Ratio of sediment to food (dry-wt)= PDS/(1-PDS) =	kg-sediment-dry-wt/kg-food-dry-wt	*STFdry*	0.10			
Ratio of sediment to food (wet-wt)= STFdry*PDD =	kg-sediment-dry-wt/kg-food-wet-wt	*STFwet*	0.0238			
Ingestion rate: sediment = IRFW*STFwet =	kg-sed-dry-wt/kg-food-dry-wt	*IRS*	0.0044			
			CONTAMINANT			
VIA DIET			**Hg**	**Cd**	**Se**	**Pb**
Concentration in Fish: CW*BCF-F=	(mg-cont)/kg fish (wet wt)	*CF*	2.14	0.0675	0.0720	0.0603
Concentration in Invertebrates: CPW*BCF-I=	(mg-cont)/kg inv (wet wt)	*CI*	475	12	5.15	100
Concentration in Amphibians: CW*BCF-A=	(mg-cont)/kg amph (wet wt)	*CA*	2.14	0.0675	0.0720	0.0603
Dose from Fish: CF * IRFW*PF =	(mg-cont)/(kg-bw)/day	*DF*	0.208	0.00657	0.00701	0.00587
Dose from Inverts: CI * IRFW*PI =	(mg-cont)/(kg-bw)/day	*DI*	22.7	0.573	0.246	4.78
Dose from Amphibs: CA * IRFW*PA =	(mg-cont)/(kg-bw)/day	*DA*	0.0628	0.00198	0.00212	0.00177
Dose from Sediment: CSED*IRS =	(mg-cont)/(kg-bw)/day	*DS*	0.00118	0.0481	0.00438	0.875
VIA DRINKING WATER						
Dose from Water: CW*IRW =	(mg-cont)/(kg-bw)/day	*DW*	1.82E-06	3.65E-06	5.83E-06	6.72E-06
Dose from Sed in Water: SS*IRW*CSED =	(mg-cont)/(kg-bw)/day	*DSW*	8.42E-06	3.43E-04	3.12E-05	0.0062
SUM ACROSS PATHWAYS						
DF + DI + DA + DS + DW + DSW =	(mg-cont)/(kg-bw)/day	*EXPdose*	23.0	0.63	0.26	5.7
CF + CI + CA + STFwet*CSED =	(mg-cont)/(kg-food)	*EXPdiet*	479	12.4	5.32	105
CW =	(mg-cont)/L	*EXPwater*	2.25E-05	4.50E-05	7.20E-05	8.30E-05
ADJUSTMENT FOR HOME RANGE:						
Home Range adjustment (if >1) =	HR/SL	*HRA*	7			

Table 39 (continued). Predicted toxicity of impoundment metals to the river otter

			CONTAMINANT			
TOXICITY PARAMETERS:			**Hg**	**Cd**	**Se**	**Pb**
Dose Non-Toxic to river otter =	(mg-cont)/(kg-bw/day)	NTOXdose	0.016	0.9	0.25	0.05
Diet Non-Toxic to river otter =	(mg-cont)/kg-diet	NTOXdiet	1.1	0.1	--	--
Water Column conc non-toxic to river otter =	(mg-cont)/L	NTOXwater	1.50E-06	0.003	0.05	1.50
HAZARD INDEX CALCULATION:			**Hg**	**Cd**	**Se**	**Pb**
Exposure/Toxicity = EXPdose/NTOXdose/HRA		***HIdose***	**205**	**0.1**	**0.1**	**16.2**
Or = EXPdiet/NTOXdiet/HRA		***HIdiet***	**62**	**18**	--	--
Or = EXPwater/NTOXwater/HRA		***HIwater***	**2**	**0.0**	**0.0**	**0.0**
IF 50% OF INVERTEBRATES CONSUMED ARE EXPOSED TO PORE WATER:						
EXPOSURE VIA DIET						
Concentration in Invertebrates (ave): = 0.5*(CPW+CW)*BCF-I=	(mg-cont)/kg inv (wet wt)	*CIPW*	239	6.03	2.59	50.2
Dose from Benthic Inverts: CIPW* IRFW*PI =	(mg-cont)/(kg-bw)/day	*DIPW*	11.4	0.288	0.124	2.40
SUM ACROSS PATHWAYS						
DF + DIPW + DA + DS + DW + DSW =	(mg-cont)/(kg-bw)/day	*EXPdose*	11.7	0.345	0.137	3.29
CF + CIPW + CA + STFwet*CSED =	(mg-cont)/(kg-food)	*EXPdiet*	243	6.42	2.76	55.1
HAZARD INDEX CALCULATION:			**Hg**	**Cd**	**Se**	**Pb**
Exposure/Toxicity = EXP/(NTOXdose)/HRA		***HIdose***	**104**	**0.1**	**0.1**	**9.4**
Or = EXPdiet/NTOXdiet/HRA		***HIdiet***	**32**	**9.2**	--	--

The field metabolic rate (NFMRkg, in kcal/kg-bw/day, which is the metabolic need of the otter, taken from EPA [1993b], p2-264, the mean for free-living otters) was normalized to a per gram body weight (kcal/g-bw/day). This was divided by the average kcal available in the diet (MEavg, in kcal/g-diet-wet-wt) and the result expressed as kg-food-wet-wt/kg-bw/day. Proportions of dietary components, gross energy of each component (GE) and assimilation efficiency (AE) of each component were used to calculate the average kcal in the diet. The proportions of diet that are fish (PF), invertebrates (PI), and amphibians (PA) were based on percentages taken from EPA (1993b; also see Table 35, here). The gross energy (GE) and assimilation efficiencies (AE) were taken from EPA (1993) Figure 4-7, and Table 4-1 (median value for invertebrates was used for GE).

The ingested **dose** of contaminants in food was estimated by first calculating the dose from each food component. In general, three components were evaluated, fish, invertebrates, and amphibians. As mentioned just above, the proportions of each of these in the diet were obtained. The proportions were multiplied by the ingestion rate to get component-specific ingestion rates which were then multiplied by the contaminant concentration in the component. This tissue concentration was estimated by multiplying the appropriate medium concentration by a bioaccumulation factor. For example, for an organism exposed to mercury (see Table 37), the dose from fish (DF; mg-Hg/kg-bw/day) is given by:

(1) DF = CF * IRFW * PF

where:

CF is the concentration in fish (mg-Hg/kg-fish-wet-wt)
IRFW is the food ingestion rate (kg-food-wet-wt/kg-bw/day)
PF is the proportion of food that is fish (kg-fish-wet-wt/kg-food-wet-wt)

And CF is given by:

(2) CF = CW * BCF-F

where:

CW is the concentration in water (mg-Hg/L)
BCF-F is the bioconcentration factor for water into fish ([mg-Hg/kg-fish-wet-wt]/[mg-Hg/L])

Using these parameters and equations, contaminant doses were estimated for each category then summed, along with sediment ingestion and drinking water doses.

Bioconcentration factors for mercury, cadmium, selenium, and lead are shown in Table 29.

iii. Sediment Ingestion Rate and Dose.

Estimates of the sediment **ingestion rates** (IRS; kg-sed-dry-wt/kg-bw/day) were calculated as follows (refer also to Tables 35 and 39):

(1) IRS = IRFW * STFwet

where:

IRFW is the food ingestion rate (kg-food-wet-wt/kg-bw/day), and is obtained as described previously.

STFwet is the ratio of sediment to food in the diet on a wet weight basis (kg-sed-dry-wt/kg-food-wet-wt).

And, since

(2) STFwet = STFdry * PDD

where:

STFdry is the ratio of sediment to food in the diet on a dry weight basis (kg-sed-dry-wt/kg-food-dry-wt).

PDD converts from wet weight to dry weight (kg-food-dry-wt/kg-food-wet-wt)

STFdry and PDD are obtained as follows:

(3) STFdry = PDS/(1-PDS)

where:

PDS is the proportion of total ingested material that is sediment on a dry weight basis (kg-sed-dry-wt/kg-food+sediment-dry-wt) and is obtained from EPA (1993b) listings of percent sediment in diet. For example, a PDS of 0.094 was chosen for the otter (EPA [1993b] p 4-20, using the value listed for the raccoon).

and:

(4) PDD = (1-PWF)*(PF) + (1-PWI)*(PI) + (1-PWA)*(PA)

where:

PDD converts ingestion rates from wet to dry weight (kg-food-dry-wt/kg-food-wet-wt).

PWF, PWI, and PWA are the proportions of fish, invertebrates, and amphibians that are water (kg-water/kg-food; or [kg-food-wet-wt - kg-food-dry-wt]/kg-food-wet-wt).

PF, PI, and PA are the proportions of the diet that are fish, invertebrates, and amphibians.

PDD was estimated from data in Table 4-1 of EPA (1993b) as follows: 0.72 (mean of two values) was used as the proportion of fish that is water (PWF); 0.78 (the median of five values, after computing the midpoints of ranges) was used as the proportion of invertebrates that is water (PWI); and 0.85 (single value) was used for amphibians (PWA). These proportions were converted to proportion dry matter by subtracting from 1, yielding units of dry-wt/wet-wt. These new proportions (dry-wt/wet-wt) were then multiplied by the proportion of that dietary item in the total diet to convert ingestion rates to a dry weight basis.

For example, the river otter ingests 0.184 kg-food-wet-wt/kg-bw/day, 0.53 of which is fish (PF) at 72% water (PWF), 0.26 of which is invertebrates (PI) at 78% water, and 0.16 of which is amphibians (PA) at 85% water. Therefore, PDD would be:

0.53 kg-fish-wet-wt/kg-food-wet-wt x
(100-72)%, or 0.28 kg-fish-dry-wt/kg-fish-wet-wt
plus
0.26 kg-inv-wet-wt/kg-food-wet-wt x
0.22 kg-inv-dry-wt/kg-inv-wet-wt
plus
0.16 kg-amph-wet-wt/kg-food-wet-wt x
0.15 kg-amph-dry-wt/kg-amph-wet-wt

which totals to 0.23 kg-food-dry-wt/kg-food-wet-wt.

The **dose** (mg-sed associated contaminant/kg bwt/day) was calculated by multiplying the ingestion rate (IRS) by the contaminant concentration in the sediment.

b. Comparisons between exposure and toxicity.

Three approaches were used. First, the doses were summed (food, water, sediment in drinking water, other incidental sediment ingestion) to produce a total dose (mg-contaminant/kg-body weight/day) and this was compared with literature toxicity data. The assumption made here was that all of the contaminant ingested would be at least as bioavailable as the form of the contaminant used in the toxicity studies. The main comparison of interest was to compare the ingested dose with a NOAEL (no adverse effects level) dose. In general, NOAELs were obtained from the literature or else doses characterized as non-toxic to wildlife were used. Similarly, dietary concentrations characterized as non-toxic to wildlife of some general subgoup of interest (such as birds) were also used. Dietary concentrations for individual species (e.g., mallard) were not used, because scaling from that species to the species of interest (e.g., kingfisher) involved computing the dose first. There was no need to derive a dietary concentration if a suitable dose was available.

If no suitable dose was available, however, it was estimated from dietary exposure if the dietary concentration was available. For example, given a dietary NOAEL concentration of 100 mg-Cd/kg-food or 5 mg-Se/kg-food for the mallard, the NOAEL can be multiplied by the mallard ingestion rate to yield a daily acceptable dose. The mallard ingestion rate can be derived from the mallard body weight of 1225 g (EPA 1993b, p2-43) using equation 3-5 of EPA (1993b; p3-5) which expresses ingestion as g-food-dry-wt/day using the formula:

food ingestion (g-food-dry-wt/day) = 0.495 Body wt (g) ^ 0.704

This ingestion rate can then be converted to a wet-weight basis by multiplying by 7 g-food-wet-wt/g-food-dry-wt (Eisler reports conversion factors of 4 and 10 in 1985b, p 37 and 1988 p 71, respectively). Based on the example presented above, mallard food intake is calculated to be 73.9 g-food-dry-wt/day, which converts to 0.517 kg-food-wet-wt/day. This can be scaled to body weight to yield 0.42 kd-food-wet-wt/kg-bw/day. Multiplying this by the non-toxic NOAELs of 100 mg-Cd/kg-food and 5 mg-Se/kg-food yields 42 mg-Cd/kg-bw/day and 2.1 mg-Se/kg-bw-day.

Second, an attempt was made to estimate the dietary exposure in terms of mg-contaminant/kg-diet and compare this exposure with toxicity data similarly expressed. This was done by multiplying tissue concentrations for fish, amphibians, and invertebrates by their relative percent contributions to the diet, then summing. To this sum was added an estimate of the dietary contribution by incidental sediment ingestion. This component was estimated by multiplying STFwet (the ratio of sediment-dry-weight to food-wet-weight) by sediment contaminant concentrations. For example, for the river otter ingesting mercury:

STFwet = 0.0218 kg-sed-dry-wt/kg-food-wet-wt

And, multiplying by the mercury concentration of 0.27 mg-cont/kg-sed-dry-wt, yields:

0.0059 mg-Hg/kg-food-wet-wt (incidental sediment ingestion)

Note how this adds only a small amount to the total dietary concentration of mercury intake which is estimated at 33.3 mg-Hg/kg-food-wet-wt (Table 39, EXPdiet).

Third, where possible, water column concentrations were compared with proposed water quality criteria protective of wildlife. These three comparisons were also adjusted for the influence of home range relative to the size of the impoundment (assumed to have a shoreline length of 4 km).

5. Kingfisher - Exposure, Toxicity, and Effects

The sources of the input values used to estimate kingfisher exposure are shown in Table 30. Based on the reported density of 0.15 to 0.6 pairs of kingfishers per km of shoreline (EPA 1993b, p 2-178), the proposed impoundment could support a pair. The sources of the toxicity data selected to evaluate potential effects to kingfishers that might utilize the impoundment are presented in Table 31.

Results of exposure and toxicity calculations for the kingfisher are shown in Table 32. Cadmium is not expected to present a problem to kingfishers. However, mercury in particular as well as lead and selenium may be at concentrations above a level of no effect. Although bioavailability of these metals may be overestimated for ingestion of sediments in drinking water, this component has only a minor contribution. In addition, if species extrapolation factors are used, the hazard indices would increase by an order of magnitude.

6. Exposure and Toxicity to the Sandpiper

The sources of the input values used to estimate sandpiper exposure are shown in Tables 32 and 33. Based on the reported density of about 12 per hectare in summer (EPA 1993b, p 2-153) and assuming a shoreline width of 1 m, the proposed impoundment (4 km of shoreline) could support approximately 5 birds (i.e., [4,000 m shoreline length x 1 m width] x [0.0001 hectare/m^2 x 12 birds/hectare] = 4.8 birds). The sources of the toxicity data selected to evaluate potential effects to sandpipers that might utilize the impoundment are presented in Table 31.

Results of exposure and toxicity calculations for the sandpiper are shown in Table 35. Based on the assumptions and values used selenium is not expected to be present in concentrations toxic to sandpipers. Mercury in particular may be at concentrations above a level of no effect as might cadmium and lead to a lesser extent. Based on comparisons of pathways, most of the exposure to mercury comes from ingestion of invertebrates, whereas the pathway of concern for lead and cadmium is sediment ingestion (estimated separately from ingestion of sediment in drinking water which is minor). Again, based on the approach used here, ingestion of water and sediments in drinking water are likely to be minor contributions to the total ingested dose. A second, more conservative scenario where the invertebrate prey are exposed to pore water concentrations, suggests that all of the metals would be of concern, particularly mercury, followed by cadmium and lead. It is important to note that if species extrapolation factors are used, the hazard indices would increase by an order of magnitude.

7. Exposure and Toxicity to the River Otter

The sources of the input values used to estimate river otter exposure are shown in Tables 38 and 39. Based on the reported density of about one otter per 28 km of shoreline (EPA 1993b, p 2-266), the proposed impoundment (4 km of shoreline) could be important in supporting an otter. The sources of the toxicity data selected to evaluate potential effects to river otters that might utilize the impoundment are presented in Table 38.

Results of exposure and toxicity calculations for the river otter are shown in Table 39. Based on the assumptions and values used and adjusting for home range, selenium is not expected to be present in toxic concentrations. Mercury in particular may be at concentrations above a level of no effect as might cadmium and lead to a lesser extent. As was found in the analyses for the two bird species, ingestion of water and sediments in drinking water are likely to be minor contributions to the total ingested dose. A second, less conservative scenario where half rather than all the invertebrate prey are exposed to pore water concentrations, does not change the above results appreciably.

Based on comparisons of pathways, most of the exposure to the metals comes from ingestion of invertebrates with additional doses contributed from fish or sediment ingestion depending on the metal. As pointed out for the bird species, it is important to note again that if species extrapolation factors are used, the hazard indices would increase by an order of magnitude.

K. Uncertainty Factors

This analysis builds on data supplied from other sections of this report and so incorporates the uncertainties in those sections by reference. For the most part, sediment, pore water, and water column toxicity were evaluated by comparison with benchmarks, and the uncertainties in each of those benchmarks (water quality criteria and sediment benchmarks) are similarly incorporated by reference. However, the analysis of potential toxicity to wildlife has several types of uncertainty that can only be reduced with further site-specific exposure and toxicity information.

Several assumptions were made concerning exposure of the selected species. The animals are expected to ingest resuspended sediments along with their drinking water, and these sediments are assumed to come entirely from tailings. Resuspension of tailings and their subsequent settlement along the shoreline has not been modeled, nor is it understood whether such a scenario would result in elevated levels of metals in near-shore pore water as compared to the water column. In general, it was assumed that ingested metals were as bioavailable as the form of the contaminants used in the toxicity tests that form the basis for comparison. While this may be an overestimate for sediments, tissue concentrations are likely to be highly bioavailable and these doses seemed to contribute the most contaminants to the overall dose.

Similarly, selecting from a range of available toxicity data is not straightforward. Ideally, one would like to find several studies that generate NOAEL's (no adverse affect levels) for dose or dietary concentration that are based on chronic studies using endpoints that are relevant to populations or individuals of the selected species. Because these data are generally not available to this level of desired detail, selections are made and extrapolation factors are used. Here, extrapolation factors were used only to estimate a NOAEL from a LOAEL (lowest adverse effect level), where a factor of two was used. It could be argued for most of the wildlife comparisons that an additional factor of at least 10 be used to reflect the uncertainty of applying data from acute studies on species other than the one of concern.

Nevertheless, despite these and other sources of uncertainty, the attempt here was made to select organisms that could reasonably represent a range of wildlife that would use the impoundment area. In addition, the following guidelines were used to avoid creating an unrealistic or overconservative analysis: (1) average wildlife exposure parameters were used (vs selecting maximum ingestion rates, smallest body weights, juveniles, etc.); (2) reasonable estimates were used for most parameters. For example, BCFs were taken from those used in other analyses of wildlife (EPA 1993a), and where possible, non-toxic doses and dietary concentrations were taken from Eisler's summaries (1985a, 1985b, 1987); and (3) species sensitivity factors were not used, but could be factored in. Also, the sensitivity of the models used here were examined for the response of the hazard indices to assumptions about the exposure of invertebrates to pore water.

Based on the approach used here, the results are useful in: (1) indicating the general expected toxicity of impoundment sediments, pore water, and the potential for bioaccumulation; and (2) describing the uncertainties of the proposed project in terms of the potential of the impoundment to affect aquatic and benthic biota as well as wildlife upon cessation of the discharge.

L. SUMMARY OF EXCEEDANCES OF CRITERIA AND BENCHMARKS

PORE WATER (Tables 25 and 26)

Exceedance of AWQC

- Acute:
Cu (12x), Hg (2x), NH_3 (5x), Cr (2x)*, **CN** (200x)**
- Chronic: Cd (2x), Se (3x)

*If Cr(VI); **CN criterion applied to CNfree

Exceedance of aquatic values of concern

- High end of concentration range: **CNS** (16x)
- Low end of concentration range: CNO (4x), **Xanthate** (93x)

SEDIMENT (Table 27)

Exceedance of Provincial Sediment Quality Guidelines

- Severe Effect: Cd (1x), Cu (2x), Fe (1x), Mn (4x)
- Lowest Effect:
Ag (6x), As (2x), Cr (1x), Hg (1x), Ni (2x), Pb (7x), Zn (4x), **CNtot** (420x)

Exceedance of Wisconsin DNR Guidelines:
Cd (11x), Cu (3x), Hg (3x), Pb (4x), Zn (5x), As & Se (1x)

Exceedance of Draft Great Lakes ERMs:
Cd (3x), Cu (1x), Pb (2x)

WATER COLUMN (Table 28)

Exceedance of AWQC
- Acute: Cu (2x)
- Chronic: **Hg** (60x)

WILDLIFE (Cd, Pb, Hg, Se) (Tables 32, 35, and 39)

Exceedance of estimated nontoxic dose:
Kingfisher: **Hg** (32x)
Sandpiper: **Hg** (35x), Pb (2x)
River Otter: **Hg** (205x), **Pb** (16x)

Exceedance of estimated nontoxic dietary concentration:
Kingfisher: **Hg** (41x)
Sandpiper: **Hg** (43x), Cd (6x)

River Otter: **Hg** (62x), **Cd** (18x) (NA for Se, Pb)

Exceedance of draft Great Lakes Water criterion for **Hg:**
Kingfisher: Yes 225 x
Sandpiper: Yes 125 x
River Otter: Yes 2 x

If benthic invertebrates are exposed to pore water and are then consumed, the following additional contaminants are of concern to the kingfisher and sandpiper:

Exceedance of estimated nontoxic dose:
Kingfisher: Pb (4x)
Sandpiper: Se (1x)

Exceedance of estimated nontoxic dietary concentration:
Kingfisher: **Cd** (29x)
Sandpiper: Se (1x)

M. Review of Literature on Subaqueous Disposal of Mine Tailings

The Mine Environment Neutral Drainage (MEND) studies, conducted by a consortium of Canadian government agencies and industry representatives, examined lakes in Canada that have been used in the past for subaqueous disposal of mine tailings. Four potentially relevant Canadian lakes studied included Buttle Lake, Anderson Lake, Mandy Lake and Benson Lake, with results presented in a number of reports. These studies are of some use in predicting what could occur in the AJ tailings impoundment in terms of uptake of metals, and were subsequently reviewed by the Rawson Academy in terms of the utility of the subaqueous disposal technique. A major difference between the lakes reviewed and the proposed impoundment is that the lakes reviewed have very different characteristics in terms of sediment inputs and existing biological communities than might be expected in the Sheep Creek tailings impoundment.

The Rawson Academy's review of the studies conducted on three of these lakes (Buttle, Anderson and Mandy) is summarized in a report entitled A Critical Review of MEND Studies Conducted to 1991 on Subaqueous Disposal of Tailings RAAS 7/92 SRT Report. The review is not an endorsement of the disposal technique in all cases. For example,

p. 2 "The process of tailings disposal is potentially highly disruptive of lake ecosystems and normally it would take several decades (possibly centuries in some lakes) before natural sedimentation provided sufficient cover to insulate the lake ecosystem from the influence of the tailings (both the metals flux and substrate effects)."

p. 2. "The background limnologies of the MEND case study lakes (Anderson, Buttle, Mandy) are suitable only for gross comparisons. They do not support clear interpretation of cause and effect associated with tailings disposal."

p. 3. "..the data are neither useful to address effects which occur during the process of disposal nor long term ecosystem adjustments to the introductions of tailings."

p. 9. Concerns to be addressed: Biological and physical reworking of tailings underwater, burial of benthic biota (and modification of food web structure), effects of solids ingestion on metal uptake by organisms.

p. 10. "The application of a single "generic" approach to subaqueous tailings disposal does not seem practical, considerations ... should be site-specific."

p. 12. "..the background limnologies are inadequate to establish anything other than the most obvious trends in lake conditions."

Subsequent to the Rawson Academy review, the MEND program conducted a preliminary biological and geological assessment of Benson Lake. Of the lakes studied to date, the Benson Lake setting appears to be most applicable to the AJ mine project with some caveats. Benson Lake is described as a small, deep, oligotrophic coastal mountain lake situated in the coastal western hemlock biogeoclimatic zone on the northern end of Vancouver Island. Benson Lake was used as a tailings repository for approximately eleven years by the Benson Lake Coast Copper Mine operated by Cominco Ltd. Operations ceased in 1973. This lake is not comparable in terms of some site specifics, especially the fact that Benson Lake is an existing lake with supporting aquatic and riparian habitats, populations, and relatively large stream inflows allowing foraging and recruitment. The regeneration of the communities associated with this lake are not applicable to the AJ project.

Aquatic vegetation was well established in the littoral zone of the lake and compared to a control lake, aquatic vegetation in Benson Lake was found to contain elevated levels of arsenic and copper. Arsenic accumulated in the tops and roots of horsetail (*Equisetum* sp.) and pond weed (*Potamogeton* sp.). The roots of both species had very high levels of arsenic ranging from 100 to 320 ug/g and 330 to 2100 ug/g respectively. Copper accumulated in the roots of horsetail and in the tops and roots of pond weed. Copper in horsetail roots ranged from 29 ug/g to 380 ug/g. Roots typically accumulated more metal than the tops (MEND, 1991).

Concentrations of metals in the flesh of fish from Benson Lake were lower than for the control lake. However, concentrations of arsenic, copper, cobalt, and cadmium were significantly higher in the livers of fish from Benson Lake, suggesting metals bioaccumulation.

These data indicate that metals have accumulated in vegetation and fish from Benson Lake when compared to a control lake.

N. Laboratory Tests Relevant to the Long-Term Behavior of Metals and Other Constituents.

The ecological risk analysis presented earlier in this chapter is based in part on analyses of the tailings. A principal concern is the bioavailability of metals to organisms that could inhabit the impoundment which is in utrn related to the mobility of the metals.

Two types of information, leach tests and indicators of acid generation potential, provide an empirical basis to evaluate the mobility of metals from the solid phase into water (see Table 3). As with samples discussed earlier for decant tests, a major concern with analytical results for leach tests and acid generation potential of tailings material has been a lack of information on the representativeness of the materials tested with respect to the material expected to come out of the mill. As noted above, data intended to demonstrate the representativeness of the average nature of bulk ore samples used in test work have recently been provided by Echo Bay (EBA, 1994a,b). The availability of a single tailings sample, if representative, provides at best average values for test results rather than reasonable maximum values.

1. Leach Tests.

Column leach tests were conducted in 1989 (Ott, 1989) and 1991 (Andrews, 1991) on tailings material. Little information has been available on the source of material and methods used in the 1989 tests and they are not discussed further here. The 1991 column tests were done on tailings produced from a bulk sample of ore collected from the 4387SX crosscut in the North Ore Body. Tailings from this sample have undergone a variety of tests and have been the primary focus of testing for the FEIS (Table 4).

The column tests which extended up to about 240 days indicated an early release of many constituents, such as cyanide, thought be be related to the tailings water. The results were used in conjunction with decant data (both in the FEIS and in this report in Table 5) to develop estimates of tailings water composition for projections of pond pore-water quality. A tabulation of column test and decant results is included Appendix B (Tables B-1 and B-2).

Batch-type leach tests (EP - extraction procedure toxicity and TCLP - toxicity characteristic leaching procedure) of tailings believed to have been previously leached in the 1991 columns (Table 4 and Bergstrom, 1993) were conducted to provide a comparison with the hazardous waste toxicity characteristic under the Resources Conservation and Recovery Act (RCRA). Because of the previous column leaching, the batch tests are not useful for hazardous waste characterization of the raw tailings. Furthermore, the tests are not necessarily appropriate for determining the mobility of contaminants from mine wastes under environmental conditions at the tailings impoundment because of the organic acid leachant and the limited analyte list.

2. Acid Generation Potential.

The oxidation of sulfur in the presence of water can produce sulfuric acid which can in turn liberate metals at concentrations well above what might be released under near-neutral pH conditions. The presence of calcium carbonate and other alkaline minerals, however, can neutralize acid if the minerals are available in sufficient quantity and dispersed such that neutralizing reactions can occur at rates exceeding the acid generating reactions. Two means of assessing the potential for acid drainage from the mine workings, waste rock, and tailings are available for the AJ project:

* Historical drainage characteristics from old mine workings and tailings piles.

* Laboratory tests of acid generation potential of samples from the mine sites.

Several historical mine workings and waste dumps occur near Juneau (Bureau of Land Management, 1992). Reports of acid drainage have not been found for these localities. However, the use of new processing techniques and methods of waste disposal for AJ would argue against reliance on historical information alone as the sole basis for evaluating the potential for acid generation.

The FEIS notes that Lakefield Research tests indicate flotation tailings would have a net potential to consume acid, and cyanided tailings might be acid producing. The FEIS concludes that combined tailings are "... expected to provide a relatively strong potential for acid consumption." (BLM, 1992, p.4-42). Actual data to support the FEIS statement were sparse at best because of lack of information on the source and variability of samples. Extensive additional data on sulfur content and acid-base accounting tests for ore and waste rock were provided in 1994 by Echo Bay (EBA, 1994b). These results demonstrate that the sulfur content of ore and waste rock is sufficiently low enough and the neutralization potential high enough that these materials are not likely to produce acid drainage.

Few results are available, on the other hand, for acid-base accounting of tailings samples (Tables 4, 40). The distribution of data for sulfur content in ore as presented by Echo Bay (EBA, 1994a) demonstrates that the bulk sample used to produce tailings for acid generation testing is reasonably representative of the average material to be mined with respect to the parameters used to determine acid generation potential. However, the single bulk sample and the single test result available at present for combined tailings is clearly not sufficient to demonstrate no potential for acid generation. For example, a single test provides no allowance for ore variablity, test variability or for variability related to the proportion of CIL tailings in the combined flotation-CIL mix (Table 40).

O. Conclusions Regarding Potential Long-Term Contamination

The Sheep Creek tailings impoundment, after closure, would be an unusual aquatic feature insofar as it would be a steep sided, shallow reservoir. No comparable lakes or reservoirs were found from which to infer the type of aquatic biological community that might develop in the impoundment over time. The impoundment would likely provide relatively poor habitat for fish due to lack of cover, lack of spawning areas, and low organic input (e.g., natural sediment and insects) from the small streams that would enter the impoundment. Regardless, it is assumed that aquatic plants and benthic (bottom dwelling) organisms would colonize the impoundment to some extent, providing potential pathways for bioaccumulation of metals.

An ecological risk analysis was performed based on predicted pore water quality, contaminants in the tailings (sediments) and predicted pollutant concentrations in the water column. The results of this analysis indicate that aquatic biota (including wildlife) would be at substantial risk from the contaminants in the tailings. Water quality criteria would likely be exceeded at high levels in the pore water (up to 200 times the acute criterion for cyanide) and water column (2 times the acute criterion for copper); sediment concentrations would exceed benchmark comparison values (over 400 times the lowest effect level for cyanide); and wildlife are likely to be at substantial risk from their exposure to high levels of metals in their diets (exceeding draft Great Lakes criterion for mercury by over 200 times).

Canadian studies of lakes used for disposal of mine tailings were also reviewed. These studies examined to some degree the impacts of mine tailings disposal on the health of the aquatic systems in these lakes. The findings of these studies, however, do not alter EPA's conclusion that the Sheep Creek tailings

Table 40. Data for evaluation of acid drainage potential for AJ mine waste.

Material	Source	Sulfur %	AP kg/t	NP kg/t	NNP kg/t	Reference
head analysis	north ore body	0.7				LR 3980 - Lakefield Research, 1990, p. 4.
head analysis	north ore body	0.81				LR 3586-001, progress report 8, referenced in Lakefield Research, 1990, p. 4.
ore - 38 samples	north ore body-average	0.42	12.9	139	126	EBA (1994a)
ore - 84 samples	north ore body-average	0.58	17.7	143	125	EBA (1994a)
ore - 6 xcut samples	north ore body-average	0.61	18.7	129	110	EBA (1994a)
ore - 18 samples	south ore body-avg	1.02	31.3	230	199	EBA (1994a)
ore - 6 xcut samples	south ore body-avg	0.5	15.4	189	174	EBA (1994a)
ore - 101 samples	NOB - SOB averaged	0.56	17.1	160	143	EBA (1994a)
flotation tails	north ore body	0.04	1.22	82.3	81.1	LR 3980 - Lakefield Research, 1990, p. 9.
CN tails	north ore body	4.76	146	83.8	-61.9	LR 3980 - Lakefield Research, 1990, p. 9.
CND tails	north ore body	4.64	142	81.1	-60.9	LR 3980 - Lakefield Research, 1990, p. 9.
Combo flot/CND tails	north ore body	0.41	12.55	70.32	57.77	Bergstrom (1994b)
tailings (5% cnd)?	10% N-69% S-21% GH?	0.21				LR 3586 - Lakefield Research, 1988, p. 25.
waste rock 1	unknown	0.48	14.7	86.7	72	LR 3586 - Ott, 1989.
waste rock 2	unknown	0.82	25.1	162.7	137.6	LR 3586 - Ott, 1989.
waste rock 3	unknown	0.47	14.4	135.2	120.8	LR 3586 - Ott, 1989.
waste rock 4	unknown	0.47	14.4	103.9	89.5	LR 3586 - Ott, 1989.
waste rock 5	unknown	0.36	11	144	133	LR 3586 - Ott, 1989.
waste rock 6	unknown	0.24	7.3	89.2	81.9	LR 3586 - Ott, 1989.
waste rock-53 samples	NOB - SOB averaged	0.4	12.5	215	208	EBA (1994a)

Notes:
1. AP - acidification potential; NP - neutralization potential; NNP - net neutralization potential.
2. Units for AP, NP, and NNP are kg H2SO4 per 1000 kg waste.
3. NOB-north ore body; SOB-south ore body.

impoundment would present substantial risks to wildlife in the long-term.

IX. POTENTIAL MEASURES FOR MITIGATING WATER QUALITY IMPACTS

A. Introduction

As described in the preceeding chapters, there are significant obstacles to permitting the proposed tailings disposal site in Sheep Creek valley. The discharge from the impoundment during operation is projected to exceed effluent limits and lead to wide-spread violations of Alaska's water quality standards in Gastineau Channel. After tailings disposal ceases, the impoundment would present high risks to local wildlife species due to the expected levels of contaminants in the tailings and in the pore water within the tailings.

During the course of preparing this report, various potential measures for reducing the water quality impacts of the project have been identified. The following sections review measures that have been identified in the FEIS and elsewhere (see Appendix E) to potentially reduce the projected water quality impacts to significantly lower levels.

B Secondary Wastewater Treatment

The FEIS (BLM, 1992) briefly reviewed secondary wastewater treatment options to further reduce suspended solids, metals and cyanide in the discharge to Gastineau Channel. These options were eliminated from detailed consideration due to "major cost increases without significant environmental benefit."

For cost comparison, the FEIS described the Eklutna Water Project in Anchorage. This facility, which cost nearly $40 million and has a design flow of 54 cfs, provides flocculation, sedimentation and filtration. The system costs $875,000 per year to operate and consumes 250 kW of energy.

Given that the peak discharge from the Sheep Creek impoundment would be on the order of 200 cfs, costs for constructing a secondary treatment plant for the impoundment discharge could be considerably higher than the cost of the Eklutna facility. Such a facility could also require a large area of fill in Gastineau Channel, potentially as large as 30 to 40 acres. This would clearly add significantly to project costs and to the overall loss of intertidal habitat in Gastineau Channel.

Such a facility would likely be capable of reducing suspended solids to levels that would comply with EPA's effluent limits, i.e., to less than 20 mg/l of TSS. The AJ facility, however, would require treatment processes for other pollutants of concern, including cyanide and possibly dissolved metals. The FEIS indicates that additional cyanide destruction could be accomplished by dewatering the tailings slurry and washing the dried tailings to rinse away complexed cyanide and trace metals adsorbed to the particulates. This process would add significantly to the treatment costs.

In conclusion, EPA concurs with the FEIS findings that secondary treatment of the wastewater stream from the Sheep Creek tailings impoundment would likely be prohibitively expensive, primarily because of the high flows that would require treatment.

C. Potential Measures for Reducing TSS

Three potential means for reducing TSS concentrations in the discharge have been identified by Echo Bay (see Appendix E). These include storing water in the tailings impoundment during periods of high runoff (e.g., in late spring), adding flocculants or coagulants (settling aids) to the tailings waste stream and installing a diffuser on the tailings discharge line to dissipate energy. No studies have been performed with respect to the AJ mine tailings in terms of how effective such measures would be and specifically whether they would likely reduce TSS concentrations below 20 mg/l, the average monthly limit.

Regardless, TSS is only one of several pollutants of concern. Even if such measures could reduce TSS to below 20 mg/l, they would not affect pollutants such as cyanide which would be in a dissolved state, nor is it clear whether such measures would reduce arsenic, copper and mercury levels so as to meet effluent limits and WQS. Furthermore, such measures would not reduce the potential for long-term contamination after the tailings discharge ceases.

D. Isolating the Tailings

In response to EPA concerns regarding the potential for long-term contamination within the impoundment after closure, Echo Bay submitted preliminary designs for isolating the tailings. Two proposals were submitted (see Appendix E). The first proposal entailed mixing cement with the tailings slurry during the last years of tailings discharges. According to Echo Bay, the tailings would form a concrete cap a foot or two in thickness.

EPA does not believe this approach would be feasible. The concrete layer, if it formed, would eventually weather and crack, exposing the tailings. Also, concrete is porous and would not prevent leaching of contaminants. Furthermore, it would provide essentially no habitat for bottom dwelling organisms.

A second option would be to cover the tailings with waste rock. While this approach would not prevent flux of contaminants from the tailings to the water column, it would help to isolate the tailings from direct ingestion by larger aquatic organisms foraging along the bottom of the impoundment. While it would provide habitat for bottom dwelling organisms and rooted aquatic vegetation, these would in turn provide pathways for bioaccumulation of metals due to the flux of contaminants through the waste rock. Also, placement of the waste rock would likely cause resuspension of large amounts of tailings. For these reasons, EPA does not consider isolating the tailings underwater to be feasible.

E. Eliminating the Cyanide Leach Circuit

Echo Bay has discussed the possibility of altering the project to eliminate the use of cyanide to recover gold. Instead of producing gold, the project would produce a "concentrate" from the froth flotation mill that would contain most of the metals. This concentrate, estimated at four barge loads per day, would then be shipped elsewhere (e.g., Japan) for processing to recover the gold.

There is not adequate information available to properly evaluate this potential project modification. Short and long-term water quality impacts would likely be reduced with the elimination of the cyanide circuit, but an in-depth analysis would be needed to determine the impacts of such a significant modification to the project.

F. Alternative Tailings Disposal Sites

Various tailings disposal alternatives were reviewed during the EIS process. Some, such as the Powerline Gulch alternative (also known as Icy Gulch), were eliminated at least in part because they were located in smaller watersheds that would not provide adequate "dilution water" needed to meet effluent limits. Given the finding of this report that effluent from the Sheep Creek alternative would not meet effluent limits, this point is moot.

In view of the findings of this report, EPA believes it may be prudent to re-examine some alternatives to the Sheep Creek tailings impoundment. For example, the proponent of the Powerline/Icy Gulch alternative asserts that it would be feasible to construct a diversion ditch around the north side of the valley capable of diverting about 80% of the surface runoff (AGDC, 1992). If feasible, this would significantly reduce the volume of effluent from a tailings impoundment located at this site, improving the feasibility of secondary treatment of the effluent.

All of the tailings disposal alternatives, including the Sheep Creek alternative, were viewed in terms of subaqueous (underwater) disposal of tailings. Reclamation for all the alternatives consisted of maintaining a reservoir of water over the tailings. All would therefore present long-term contamination risks similar to those for the Sheep Creek alternative.

None of the alternatives were examined from a more conventional subaerial tailings deposition approach similar to the proposed Kensington project. This approach involves depositing the tailings on dry land to form a tailings beach that can be covered with soil and reclaimed after cessation of mining. Water from the tailings slurry would collect in a relativley small, shallow pond at one end of the impoundment.

If feasible, this approach would reduce the potential risks of long-term contamination since aquatic organisms would not be directly exposed to tailings. However, diverting surface water around such an impoundment would be necessary. This would not be feasible in Sheep Creek valley due to the steep slopes. While substantial surface water diversion may be feasible in Powerline/Icy Gulch. there are many perceived problems with the Powerline/Icy Gulch alternative. These include access, cost, availability of construction material and a limited construction season. While ecological values of the site are much lower than Sheep Creek valley, it is located at a higher elevation with greater potential for erosion of the dam and tailings.

X. POTENTIAL MEASURES TO MITIGATE ECOLOGICAL IMPACTS

A. Introduction

This chapter addresses measures that have been proposed to mitigate the physical loss of aquatic resources. The proposed tailings impoundment of the AJ Mine would fill 420 acres of the Sheep Creek valley. Waters that would be eliminated by this fill are 2.5 miles of Sheep Creek and 8.1 acres of wetlands (total of 20.1 acres of aquatic habitat; COE, 1994). The flow of 1.1 additional miles of Sheep Creek downstream of the impoundment would be significantly reduced. Marine fill consists of 14.7 acres extending from the shoreline of Gastineau Channel to the -30 foot contour.

The value of the wetlands and associated stream channel within Sheep Creek valley stems from the part they play in a mosaic of habitat types within the valley. The vegetative assemblage of the valley is essentially a riparian assemblage, a result of the surface and ground water regime of the valley. The mixture of cottonwood and wetland shrub communities provides habitat for an unusual assemblage of song birds, as well as other animals (see Chapter V, Affected Environment). Among the total area of cottonwood forest in the greater Juneau area, Sheep Creek valley contains 25%. In addition, the Sheep Creek valley has much greater nesting success than nearby areas with the same assemblage of vegetation. It is the context of Sheep Creek valley in which wetland values are considered. EPA considers the loss of these values to be a significant adverse impact.

Measures to mitigate the functions and values of aquatic resources that would be lost due to construction of the proposed tailings impoundment are described in the FEIS (BLM, 1992). Among these are:

1. Measures to maintain the flow of water in the reach of Sheep Creek used by anadromous fish:

2. Development and implementation of a fishery enhancement project at another site;

3. Restocking of the tailings impoundment with fish after closure;

4. Enhancement of fishery values of the impoundment;

5. Enhancement of wildlife values of the impoundment, targeting waterfowl in particular;

6. Develop and implement a wetlands enhancement project at another location.

B. On-site Mitigation Proposals

The long term aquatic resource potential of the tailings impoundment has been evaluated. If, within a reasonable amount of time, the impoundment were to reach a state of water quality and productivity such that a healthy aquatic ecosystem could be maintained, then long term resource losses might be reduced. However, as concluded in the Chapter VIII, subaqueous disposal of the AJ Mine tailings in the Sheep Creek valley would place indigenous wildlife at substantial risk. EPA therefore concludes that no "mitigation credit" can be ascribed to the tailings impoundment and that efforts to offset these significant losses should be directed toward off-site mitigation options.

C. Off-site Mitigation Proposals

EPA suggested restoration of degraded wetlands of Lemon Creek valley and development of access for the public as potential mitigation. EPA's rationale was that enhancement of values in a nearby degraded valley could potentially offset the losses in habitat and aesthetics of the Sheep Creek valley. Restoration of wetland values in the lower part of Lemon Creek, while clearly not an in-kind replacement of the high quality habitat of Sheep Creek valley, would provide quality waterfowl habitat and some songbird habitat. In addition, improved access to Lemon Creek valley would provide an alternative for aesthetic values lost in Sheep Creek valley, with similar proximity to downtown Juneau.

This proposal was ultimately determined to be infeasible (R&M Engineering, Inc., 1993) because much of the disturbed land in Lemon Creek valley is unavailable as a mitigation project for any or all of the following reasons:

1. It is private property and not for sale;

2. It is currently under industrial development. Gravel mines are the primary use.

3. It is under the authority of active Corps 404 permits that require reclamation to mitigate adverse environmental impacts on site.

Echo Bay outlined an alternative project that identified three ponds in the Juneau area for enhancement to mitigate the loss of values from the project. They used a numerical evaluation of Sheep Creek and the Sheep Creek valley wetlands to balance the loss with an assessed numerical gain in values at the ponds. Interpretive signs and other visitor facilities would also be a part of the proposed enhancement project.

This project was not deemed adequate mitigation by EPA because it did not take into account the unique setting of the Sheep Creek valley, both in terms of habitat value and aesthetics. The value of wetlands in the Sheep Creek valley is tied to the larger landscape of which it is a part. The interaction of soils, hydrology and other factors in the valley result in a diversity of vegetation that supports a diversity and productivity of avifuana unique in the Juneau area. Also, this proposal would not replace the aesthetic value of Sheep Creek valley, a high, secluded valley of substantial aesthetic value, close to downtown Juneau.

Echo Bay has outlined a second proposal to enhance waters in the Mendenhall Glacier Recreation Area, managed by the U.S Forest Service. This project was intended to take into account the setting, in terms of access, and scale of the waterbodies to be enhanced. Among the enhancement measures proposed are:

1. Fisheries improvements to attempt to provide improved spawning and rearing habitat for salmon;

2. Trail improvements, including footbridges to improve access and gates to eliminate off-road vehicles;

3. Interpretive signs and picnic area development;

4. Small mammal habitat improvements;

5. Avian nest structure installation and waterfowl habitat improvements;

6. Revegetation of selected sites.

EPA does not support this proposal, in part because the U.S. Forest Service has already proposed to make these improvements. The site is managed by the U.S Forest Service and is part of their long-term plans for enhancement when funding becomes available. While Echo Bay would be accelerating the rate of enhancement at this site, over the long term there would still be loss of values from Sheep Creek valley that would not be replaced.

Most importantly, there is no indication that the unique diversity and productivity of avian habitat of the Sheep Creek valley would be mitigated by this project. Nesting boxes and platforms do not provide self-sustaining habitat and more importantly there would be a significant net loss of foraging (feeding) habitat for birds.

D. Mitigation for Lost Values in Marine Waters

No mitigation has been proposed to offset the loss of aquatic resource values in the 14.7 acres of marine and intertidal waters of Gastineau Channel that would be filled to create dry land for the surface facility.

E. Conclusion

On review of mitigation opportunities presented it is unlikely that the loss of aquatic resource values in the Sheep Creek valley can be offset. The values derived from Sheep Creek and associated wetlands are particular to the setting in which they are found. Resources that are generally associated with wetlands can conceivably be improved at other sites. In many instances, where the aquatic resources lost are of lower value, this sort of trade off is common. However, as previously stated, the aquatic resources of the Sheep Creek valley, as part of a larger landscape, are uniquely diverse and productive for the project area. It is unlikely that these values can be recreated off-site.

XI. CONCLUSIONS

Based on the findings of this report, EPA concludes that there is a high potential for significant degradation of waters of the U.S. both within Gastineau Channel and within the tailings impoundment after closure, i.e., after it is no longer used for treatment of wastewater and disposal of mine tailings. The specific major findings that lead to this conclusion are as follows:

Finding #1:

During operation, the wastewater discharge from the impoundment co-mingled with mine drainage is likely to exceed EPA's New Source Performance Standards (end-of-pipe effluent limits) for total suspended solids, copper and possibly mercury (see Chapter VI).

Finding #2:

During operation, the wastewater discharge is likely to cause widespread exceedances of state of Alaska water quality standards for cyanide, arsenic, copper and possibly mercury and lead (see Chapter VII);

Finding #3:

After closure, indigenous wildlife that would likely inhabit the tailings impoundment would be at substantial risk due to contaminants that would likely persist in the impoundment. Water quality criteria would likely be exceeded at high levels in the pore water (up to 200 times the acute criterion for cyanide) and water column (2 times the acute criterion for copper); sediment concentrations would likely exceed benchmark comparison values (over 400 times the lowest effect level for cyanide); and wildlife are likely to be at substantial risk from their exposure to high levels of metals in their diets (exceeding draft Great Lakes criterion for mercury by over 200 times; see Chapter VIII).

Finding #4:

Unlike the Kensington Mine project, reliable measures (e.g., secondary treatment of the effluent, isolating the tailings) for reducing the anticipated water quality impacts described above to significantly lower levels do not appear to be feasible. Others, such as eliminating the cyanide leaching process or using subaerial tailings deposition and conventional reclamation at an alternative disposal site, would require much more detailed analysis to determine feasibility as well as overall environmental impacts (see Chapter IX).

Finding #5:

The loss of aquatic habitat (Sheep Creek and associated wetlands) would be significant due to their contribution to the unique diversity and productivity within the Juneau area, particularly in terms of migratory bird habitat and the aesthetic quality and recreational value of Sheep Creek valley. Potential measures identified to replace these values, including on-site and off-site measures, do not appear either feasible or adequate to prevent a significant loss of aquatic resources (see Chapter X).

XII. REFERENCES

Alaska-Gastineau Development Company, 1992, Letter from Robert Craig to Ben Pollard, Chief Engineer, City and Borough of Juneau re. AJ Mine Project, Icy Gulch Option Water Balance, September 28, 1992.

Ambrose, R.B., J.L. Martin, J.P. Connolly and R.W. Schanz, 1991, WASP4, A hydrodynamic and water quality model - model theory, user's manual and programmer's guide. Environ-mental Research Laboratory, U.S. Environmental Protection Agency, Athens, Georgia, 324 pp.

Andrews, Gene, 1991a, letter with lab reports from Gene Andrews, Echo Bay Mines, to David Dorris, USBLM, on "AJ Tailings and Leachate Study": April 17, 1991.

Andrews, G., 1991b, Gastineau Channel Freshwater Flushing Study, September 1991.

Andrews, G. 1993, Memo to Frank Bergstrom, Echo Bay Alaska, Inc., dated November 11, 1993.

Andrews, G. 1994, AJ Tailing Pond Operations. Memo to Tom Wang, dated September 29, 1994.

Ariathurai, R., 1985, Fundamentals of sediment transport, Presented at the Hydrologic Engineering Center, U.S. Army Corps of Engineers, Davis, California. 47 pp.

Batelle Northwest, 1970, Effect of Hanford Plant Operations on the Temperature of the Columbia River: 1964 to present, November 1970.

Bengtsson, L. and T. Hellstrom, 1992, Wind-induced resuspension in a small shallow lake. Hydrobiologia 241: 163-172.

Bergstrom, Frank W. 1993a, Letter with lab reports from Frank Bergstrom EBM, to Sharmon Stambaugh, ADEC, on "Trans-mittal of Hazardous Waste Characterization of the A-J Tailings": March 5, 1993.

Bergstrom, Frank W. 1993b, Letter to Mr. Paul Rusanowski, Office of the Governor, Division of Governmental Coordination, Juneau, Alaska, dated August 23, 1993.

Bergstrom, Frank W. 1993c, Personal communication, August 24, 1993.

Bergstrom, Frank W., 1994a, Letter with lab reports from Frank Bergstrom, EBA, to Bill Riley, USEPA, on "Information relevant to a November 10, 1993 Memorandum by Stephen Hoffman, Chief Mining Waste Section, "regarding AJ Mine Tailings": February 15, 1994.

Bergstrom, Frank W., 1994b, Letter with lab reports from Frank Bergstrom, EBA, to Bill Riley, USEPA, relative to ARD: March 25, 1994.

Bergstrom, F.W., 1994c, Memo to Bill Riley, US EPA, Region 10, dated May 5, 1994.

Bowden, K.F., 1967, Circulation and Diffusion, Estuaries, American Association for the Advancement of Science, Publication No. 83, Washington, DC, 1967, pp. 15-37.

Bowie, G.L. , W.B. Mills, D.B. Porcella, C.L. Campbell, J.R. Pagenkopf, G.L. Rupp, K.M. Johnson, P.W.H. Chan and S.A. Gherini, 1985. Rates, constants, and kinetics formulations in surface water quality modeling (second edition), EPA/600/3-85/040. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, Georgia.

Brandsma, M.G., T.C. Sauer, Jr. and R.C. Ayers, Jr., 1983. Mud discharge model report and user's guide, prepared for the Offshore Operators' Committee.

Cole, T.M. and E.M. Buchak, 1993, CE-QUAL-W2: A two-dimensional, laterally averaged, hydrodynamic and water quality model, User Manual. Instruction Report ITL-93-? (Draft), U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MI.

Comet, T. A. and M. F. Wilson, 1994, Letter of October 18, 1994 from TallChief A. Comet, Forestry Sciences Laboratory, Juneau, to Laurie Ferguson Craig, Alaskans for Juneau, 2 pp.

Dauchy, J.W., W.T. Waller, and M.D. Piwoni, 1980, Acute toxicity of cyanate to *Daphnia magna*. Bulletin of Environmental Contamination and Toxicology 25, pp. 194-196.

Dortch, M., 1993, Letter to William Riley, EPA Region 10, dated November 16, 1993.

Easton, D., 1993, Letter to John Yearsley, PhD., EPA Region 10, dated February 8, 1993.

Echo Bay Alaska, Inc., 1991, Seawater Monitoring Data, May 1989 through March 1991.

Echo Bay Alaska, Inc., 1992, A-J. Mine Water Discharge Comprehensive Report, August 1992.

Echo Bay Alaska, 1994a, Assessment of the sulfur content and potential for acid generation from ore and waste materials: AJ

Mine Project: Echo Bay Alaska, Inc., and Kea Pacific Holdings, Inc., June 1994.

Echo Bay Alaska, 1994b, Comparison of multi-element data from the rocks of the AJ deposit, southeast Alaska: Echo Bay Alaska, October 1994.

Ecology, Department of, 1991, Summary of criteria and guidelines for contaminated freshwater sediments. Washington State Department of Ecology, Sediment Management Unit, WDOE, Olympia, WA.

Eisler, R., 1985a, Cadmium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wild-life Service, Biological Report. 85 (1.2), 46 pp.

Eisler, R., 1985b, Selenium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wild-life Service, Biological Report, 85 (1.5), 57 pp.

Eisler, R., 1987, Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wild-life Service, Biological Report, 85 (1.10), 90 pp.

Eisler, R., 1988, Lead hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wild-life Service, Biolical Report, 85 (1.14), 134 pp.

Fischer, H.B., E.J. List, R.C.Y. Koh, J. Imberger and N.H. Brooks, 1979, Mixing in inland and coastal waters. Academic Press, Orlando, Florida.

Frank, D, 1994, Draft memorandum describing estimated influent composition of tailings.

Fredericksen, Rick, 1992, Analysis of ore samples from bulk sample crosscut: Memo from Rick Fredericksen, Echo Bay Mines, to Frank Bergstrom, January 27, 1992.

Fredericksen, Rick S., 1993, Progress report on the A-J Project in Conference Juneau 1993: Alaska Miners Association, proceedings, pp. 27-28.

Green, T.L R.S. Boswell, and D.W. Aldridge, 1990, The effect of thiocyanate on feeding, oxygen uptake, and nitrogenous excretion in the freshwater snail *Elimia proxima* (Say). American Zoologist, 30 (4).

Hawley, J.R., 1977, The use, characteristics and toxicity of mine-mill reagents in the province of Ontario. OMOE, Toronto.

Hayter, E. J., 1993, Letter to Dr. John Yearsley, EPA Region 10, dated October 14, 1993.

HDR Engineering, Inc., 1990, Alaska-Juneau mine Sheep Creek tailings impoundment overturning study. Bellevue, WA., November 9, 1990.

Heming, T.M., R.V. Thurston, E.L. Meyn, R.K. Zajdel, 1985, Acute toxicity of thiocyanate to trout. Trans. American Fish. Soc., 114: 895-905.

Hill, Steven, 1991 Echo Bay Alaska Projects. Gastineau Channel Current Meter Moorings, Raw Data Report (Re-Issue), October 1991.

Holmberg, Nevin, 1991, Letter to colonel William W. Kakel dated March 28, 1991.

Ingersoll, C., 1994, Personal communication, May 11, 1994, regarding development of draft effect range median values for Hyallela azteca.

Ippen, A.T., 1966, Estuary and Coastline Hydrodynamics, McGraw-Hill, 1966.

Jaagumagi, R., 1993, Development of the Ontario provincial sediment quality guidelines for arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, and zinc. ISBN 0-7729-9249-5. OMOE, Toronto, 10 pp.

Jokela, Brett, 1991, Memo from Brett Jokela, JMM, to David Dorris, BLM, on "AJ Mine FEIS, leachate Column Tests": July 18, 1991.

Kevan, S.D. and D.G. Dixon, 1991, The acute toxicity of pulse-dosed thiocyanate (as KSCN and NaSCN) to rainbow trout (*Oncorhynchus mykiss*) eggs before and after water hardening. Aquatic Toxicology, 19: pp.113-122.

Knight and Piesold, 1989, Tailings material characteristics. Unpublished data.

Koh, R., and Y. Chang, 1973, Mathematical model for barged ocean disposal of wastes, U.S. EPA report, EPA-6602-73-029.

Kowalik, Z., 1984, Physical Oceanography of Boca de Quadra; Marine Environmental Studies in Boca de Quadra; Numerical Modeling of the Boca de Quadra Fjord, June 1984.

Kowalik, Z. and Findikakis, A.N., 1985, The numerical modeling of fjord circulation and submarine disposal of mine tailings in

Wilson Arm/Smeaton Bay, southeast Alaska. Prepared for U.S. Borax & Chemical Co., 74 pp.

Krone, R. B., 1993, Letter to William M. Riley, EPA Region 10, dated October 7, 1993.

Lakefield Research, 1988, An investigation of the recovery of gold from AJ ore samples submitted by Echo Bay Mines Ltd.: Progress Report No. 3, Project No. L.R. 3586, November 7, 1988.

Lakefield Research, 1989a, An investigation of semi-autogenous grinding and the recovery of gold from "AJ" Project Samples submitted by Echo Bay Mines Limited: Progress Report No. 6, Project No. L.R. 3586, May 16, 1989.

Lakefield Research, 1989b, An investigation of the recovery of gold from A-J ore samples submitted by Echo Bay Mines Ltd.: Progress Report No. 8, Project No. L.R. 3586-001, November 13, 1989.

Lakefield Research, 1990, An investigation of tailing characteristics and water quality of A-J Project tailings requested by Echo Bay Mines Ltd.: Progress Report No. 1, Project No. L.R. 3980, October 11, 1990.

Lawrence, G.A., P.R.B. Ward, and M.D. MacKinnon, 1991, Wind-wave-induced suspension of mine tailings in disposal ponds - case study. Canadian Journal of Civil Engineering, Vol. 18, No. 6, pp. 1047-1053.

Lick, W., 1993, Fax to John Yearsley and Ben Cope, EPA Region 10, dated November 1, 1993.

Luettich, R.A., Jr., D.R.F. Harleman and L. Somlyody, 1990, Dynamic behavior of suspended sediment concentrations in a shallow lake perturbed by episodic wind events. Limnology Oceanography, 35(5), pp. 1050-1067.

Mears, A.I., 1993, Performance specifications for avalanche dissipators Sheep Creek tailings. Prepared for Echo Bay Mines, 55 pp.

Meyer, L., 1994, Mixing Properties in the Gastineau Channel, (Independent Masters Degree Project for the University of Washington Engineering Dept.), March 1994.

Mills, W.B., D.B. Porcella, M.J. Ungs, S.A. Gherini, K.V. Summers, L. Mok, G.L. Rupp, G.L. Bowie, and D.A. Haith, 1985. Water quality assessment: A screening procedure for toxic and conventional pollutants in surface and ground water, EPA/600/6-85/002. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, Georgia.

Mine Environment Neutral Drainage (MEND) Program, Ministry of Energy, Mines, and Petroleum Resources, and CANMET, Environment Canada, 1991, A preliminary Biological and Geological Assessment of Subaqueous Tailings Disposal in Benson Lake, British Columbia dated Mated March 1991.

Norton, S., 1994, Personal communication regarding bioaccumulative contaminants and selection of wildlife species for evaluation.

Ott Engineering, Inc., 1989, Column leach tests and waste rock acid generation potential - A.J. Mine Project: TechnicalMemorandum 3, Job No. S1129.01/3, September 14, 1989.

Palmes, J., 1993, Cyanide: A review of literature on the toxicity of cyanide to fish and wildlife related to its use in mining. In: (T. Brock, Ed.) Proceedings of Watershed 1991 soil, air, and water stewardship conference, Juneau, AK, pp. 47-57, USDA R10-MB-217.

Pritchard, D., 1955, Estuarine Circulation Patterns, Proceedings of the American Society of Civil Engineers, Volume 81, pp. 717/1-717/11.

Quadra Consultants, 1986, Salmon Creek reservoir penstock intake inspection, letter report.

Quadra Consultants, 1987, Salmon Creek monitoring, summary report.

R&M Engineering, Inc., 1993, Land Use Reviews Lemon Creek Area, R&M Project No. 921384, March 5, 1993.

Rawson Academy of Aquatic Science, 1992, A Critical Review of MEND Studies Conducted to 1991 on Subaqueous Disposal of Tailings dated July 21, 1992.

Rescan Environmental Services, Ltd., 1983, Suspended solids survey of Rupert Inlet, B.C. conducted on behalf of U.S. Borax and Chemical Corporation.

Resource Consultants & Engineers, Inc., 1993, Fine particle settling, Telecopy to Jan Hastings, August 17, 1993, 2 pp.

Ritcey, Gordon M., 1989, Tailings management, problems and solutions in the mining industry: New York, Elsevier, 970 pp.

Saiki, M.K., 1986, Concentrations of selenium in aquatic food-chain organisms and fish exposed to agricultural tile drainage water. In: Selenium and agricultural drainage: Implications for San Francisco Bay and the California environment, Proceedings of the Second Selenium Symposium, Berkeley, CA. ISBN: 0-937995-00-2.

Schmidt, Dana, 1993, Letter to Justine Barton, EPA Region 10 dated December 13, 1993.

Schnoor, J.L., C. Sato, D. McKechnie, and D. Sahoo, 1987. Processes, coefficients, and models for simulating toxic organics and heavy metals in surface waters. EPA/600/3-87/015, Environmental Research Laboratory, ORD, U.S. Environmental Protection Agency, Athens, Georgia, 303 pp.

Schubel, J.R., H.H. Carter, R.E. Wilson, W.M. Wise, M.G. Heaton and M.G. Gross, 1978, Field investigations of the nature, degree and extent of turbidity generated by open-water pipeline disposal operations. Technical Report D-78-30. U.S. Army Engineer Waterways Experiment Station Environ-mental Laboratory, Vicksburg, Mississippi, 245 pp.

Shea, Lana, 1994a, Letter to Frank Bergstrom, Echo Bay Alaska, Inc., dated Febraury 2, 1994.

Shea, Lana C., 1994b, Letter and attachments to William Riley, EPA Region 10 dated January 10, 1994.

Sheng, P.Y. and W. Lick, 1979, The transport and resuspension of sediments in a shallow lake, Journal of Geophysical Research, Vol. 84, No. C45, 1809-1826.

Smith, A. and T. Mudder, 1991, The chemistry and treatment of cyanidation wastes, Chapter 5, The toxicity of cyanide and related compounds, Mining Journal Books,, Ltd, London. ISBN: 0-900117-51-6.

Steffen, Robertson & Kirsten, 1989, Estimation of seepage from the Sheep Creek tailings impoundment, Project 65304B for Echo Bay Resources, 13 pp.

Steffen, Robertson & Kirsten, 1990, Report 65304, AJ project Sheep Creek tailings dam report on geologic, geotechnical and hydrologic investigations. Prepared for Echo Bay Mines. March 1990.

Steffen, Robertson & Kirsten, 1994, Sheep Creek tailings dam 1959-1971 water years, LOTUS-123 spreadsheet output.

Sturdevant, Molly V., 1994, National Marine Fisheries Service, Memorandum addressed "To Whom It May Concern." Preliminary Drift Card Study Results, February 1994.

Tileston, J.V., 1992, Correction in the A-J FEIS Water Quality Analysis, Memorandum to David Dorris dated September 18, 1992, JMM Consulting Engineers, Anchorage, Alaska.

U.S. Army Corps of Engineers, 1961, General Design memorandum Gold Creek and tributaries, Alaska, flood control channel improvement, Supplement No. 2.

U.S. Army Corps of Engineers, 1985, Interim guidance for predicting quality of effluent discharged from confined dredged material disposal areas. Technical Notes EEDP-04-1,-2,-3,-4. U.S. Army Engineer Waterways Experiment Station, Environmental Laboratory, Vicksburg, Mississippi.

U.S. Armay Corps of Engineers, 1987, Confined Disposal of Dredged Material. Users Manual. EM 1110-2-5027.

U.S. Army Corps of Engineers, 1994, Memorandum for Record: Verification and Review of the Wetlands Delineations by Dames and Moore and Three Parameters Plus (3P+) for the AJ Mine Project Proposal, September 7, 1994.

U.S. Bureau of Land Management, 1992, A-J mining project, Final environmental impact statement: Bureau of Land Management and James M. Montgomery, Inc., Volumes 1 and 2, BLM-AK-ES-92-028-2800-980, May 1992.

USEPA, 1984, Ambient water quality criteria for cyanide. EPA 440/5-84-028. National Technical Information Service, Springfield, VA.

USEPA, 1985, Ambient water quality criteria for mercury. EPA 440/5-84-026. National Technical Information Service, Springfield, VA.

USEPA, 1985, Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water, September 1985.

USEPA, 1991a, WASP4, A Hydrodynamic and Water Quality Model: Model Theory, User's Manual, and Programmer's Guide, July 1991.

USEPA, 1991b, Technical Support Document for Water Quality-Based Toxics Control, EPA 505/2-90-001, March 1991.

USEPA, 1992, Water quality standards; establishment of numeric criteria for priority toxic pollutants; states' compliance final rule. 40 CFR Part 131, Federal Register Vol. 57, No 246, 60848-60923.

USEPA, 1993a, Great Lakes water quality initiative criteria documents for the protection of wildlife (proposed). EPA 822-R-93-007.

USEPA, 1993b, Wildlife exposure factors handbook. EPA/600/R-93/187a.

USFW Region 7, 1992, Memo to Bureau of Land managment dated July 3, 1992.

Wilson, M. F. and T. A. Comet, 1991, Birds of Sheep Creek Valley, Report for 1991, USDA Forest Service, Forestry Sciences Lab., Juneau, Alaska, 17 pp.

Wilson, M. F. and T. A. Comet, 1993, Birds of Sheep Creek Valley, Report for 1993. USDA Forest Service, Forestry Sciences Lab., Juneau, Alaska. 9 pp.

Woodward-Clyde Consultants, 1993a. WASP4 modeling of the proposed Sheep Creek tailings pond, AJ Mine. Draft report prepared for AJ Mines.

Woodward-Clyde Consultants, 1993b, List of additional water quality model simulations of Echo Bay's AJ Mine tailings pond, July 13, 1993.

Yearsley, J.R., 1992, A simple model for metals in the proposed AJ Mine tailings pond DRAFT, EPA Region 10, Seattle, WA.

Yearsley, J.R., 1993, Applications of WASP4 to the A-J Mine tailings pond, EPA910/R-93-005, EPA Region 10, Seattle, Washington, 22 pp.

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