An Assessment of Ecological Risk to Wild Salmon Systems from Large-scale Mining in the Nushagak and Kvichak Watersheds of the Bristol Bay Basin

October 2010

Developed for:

The Nature Conservancy

Developed by:

ecology and environment, inc.
International Specialists in the Environment

PREFACE

In 2003, The Nature Conservancy in Alaska identified the Bristol Bay Basin, and in particular the Nushagak and Kvichak watersheds, as a conservation priority under its Wild Salmon Ecosystems program. The Conservancy has been active in the region since the early 1990s.

The Bristol Bay Basin is an intact ecoregion with unimpeded natural ecological processes supporting healthy populations of terrestrial, avian, and aquatic species, including five species of anadromous Pacific salmon. Bristol Bay supports the largest runs of wild sockeye salmon on earth. Historically, the Kvichak River drainage is the world's single most productive sockeye salmon watershed. The Nushagak River watershed is the largest producer of Chinook in the Bristol Bay drainages. In short, these watersheds are the heart of the world's most productive wild salmon nursery.

In January of 2006, the Board of Trustees of The Nature Conservancy in Alaska issued a statement of concern regarding the Nushagak and Kvichak watersheds and the potential impact of mining projects in those watersheds and directed the staff to further evaluate these concerns in conjunction with the organization's conservation efforts in the region. In March 2008, as part of that process, the Salmon Working Group of the Board of Trustees held an internal workshop to better understand the severity, probability and duration of risks posed to the salmon systems of the Nushagak and Kvichak watersheds by large-scale mining operations in the region. Experts in large mine permitting, environmental engineering, salmon habitat, acid mine drainage and risk assessment, guided trustees and staff through a series of presentations, risk assessment exercises, and discussions. As a result, in April 2008 the Board directed staff to develop a risk framework and populate that framework with relevant information to more completely characterize the risks. The following assessment is the result.

As of the date of this assessment, no formal plans have been submitted for permitting large-scale mining in the Nushagak and Kvichak watersheds. Presently, exploration is underway and a preliminary plan for a mine in the Pebble prospect area was released by Northern Dynasty Minerals in 2006 as part of a water withdrawal application to the State of Alaska. For the sake of understanding potential risks of mining development, this analysis uses this preliminary plan as a *scenario* for evaluating risk to local fishery components (e.g., salmon). Use of the plan in this regard is only illustrative and is designed to facilitate assessment of risks from mining development in the region *regardless* of how the Pebble prospect may or may not be developed. Details of any mine plan may change prior to final permitting and a fully permitted mine may change significantly over its life. Risks identified in this scenario and found to be associated with mines regardless of their design (i.e., dewatering, alteration or loss of habitat, road construction, fugitive dust, chemical spills, pipeline spills, episodic and large scale pollution events, acid mine drainage and cumulative effects) apply to any large mining development in the region, whether it be at the Pebble prospect or any of a number of other mining claims currently identified and/or under exploration in the Nushagak and Kvichak watersheds.

The following assessment has been extensively peer reviewed and we would like to thank the following external reviewers, in particular, for their time and advice:

- Mindy Armstead, Ph.D., Potesta and Associates
- Douglas Beltman, Executive Vice President, Stratus Consulting
- John Hedgepeth, Project Manager/Fisheries, Tenera Environmental
- Bill Riley, retired, Environmental Protection Agency
- Thomas Quinn, Ph.D., School of Aquatic and Fishery Sciences, University of Washington

Many others assisted in review and advice at various junctures in the development of this assessment and we thank them for their time and contributions as well. We would also like to take this opportunity to thank the Gordon and Betty Moore Foundation, the Native Village of Ekwok, the Wallace Research Foundation, and the Bristol Bay Regional Seafood Development Association for the financial support that made this assessment possible.

It is important to stress that this risk assessment was designed to provide a science-based perspective of the nature of the potential risks to wild salmon systems and to initiate a greater dialogue about these risks. It is not intended to be exhaustive. For example, so little data is available on the area's groundwater systems that this assessment could not fully characterize risks associated with groundwater; hence, potential instream flow reductions are based solely on surface water data and do not reflect groundwater changes. In addition, this assessment was limited to ecological factors and does not incorporate social, health, economic or cultural considerations that might be relevant to understanding risks associated with large-scale mining in these watersheds. We welcome feedback and discussion about the methodology, assumptions and conclusions in this risk assessment and look forward to the larger public dialogue this may engender.

This assessment is only one component of the Conservancy's effort to understand the biological values in the Nushagak and Kvichak watersheds and the risks posed by mining development. In addition, the Conservancy has undertaken a range of field studies, including fish distribution surveys, water chemistry sampling, macroinvertebrate collection, and hydrologic analysis in and around the Pebble prospect. The results of these studies along with this risk assessment continue to inform the Conservancy's work to protect the biological diversity and abundance of the wild salmon ecosystems of the Bristol Bay region.

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º C	degrees Celsius
ADEC	Alaska Department of Environmental Conservation
ADFG	Alaska Department of Fish and Game
ADNR	Alaska Department of Natural Resources
AMD	acid mine drainage
AWC	Anadromous Waters Catalog
AVM	average value method
BCF	bioconcentration factor
BMC	benchmark concentrations
BMP	best management practice
cfs	cubic feet per second
cm	centimeters
cm/s	centimeters per second
DEM	digital elevation model
DO	dissolved oxygen
DPS	Distinct Population Segment
E & E	Ecology and Environment, Inc.
EPA	United States Environmental Protection Agency
ERA	ecological risk assessment
fps	feet per second
GIS	geographic information system
HEP	Habitat Evaluation Procedure
HSI	habitat suitability index
ICMM	International Council on Mining and Metals
ILF	interactive limiting factor
LSI	lowest suitability index
m	meters
mg/L	milligrams per liter
mi^2	square miles
mm	millimeters

NDM	Northern Dynasty Minerals, Ltd.
NFK	North Fork Koktuli
NMFS	National Marine Fisheries Service
NRC	National Research Council
NTU	nephlometric turbidity units
RM	river mile
SFK	South Fork Koktuli
SI	suitability index
TMDL	total maximum daily load
TNC	The Nature Conservancy
TSF	tailings storage facility
TSS	total suspended solids
USACE	United States Army Corps of Engineers
USFS	United States Forest Service
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UTC	Upper Talarik Creek
WQC	water quality criteria

1.0 INTRODUCTION

Ecological risk assessment (ERA) has become an essential tool for determining impacts to biological receptors as a result of contamination from metal mining facilities (Brumbaugh *et al*. 1994, Canfield *et al*. 1994, Ingersoll *et al*. 1994, Kemble *et al*. 1994, Pascoe and DalSoglio 1994, Pascoe *et al*. 1994, Linkov *et al*. 2002). The United States Environmental Protection Agency's (EPA's) Risk Assessment Forum developed the *Framework for Metals Risk Assessment* (2007a), which is a science-based document that addresses the special attributes and behaviors of metals and metal compounds to be considered when assessing their human health and ecological risks.

To date, efforts have been designed to address the impacts or risks posed by metals' contamination subsequent to mining operations. Few, if any, ERAs have been directed at pre-mining impacts. Smith (2007) provided strategies to predict metal mobility at mining sites through evaluation of source characterization, geoenvironmental models, geoavailability, and metals speciation; controlling physicochemical attributes (e.g., solubility, pH, sorption) in aqueous environments are discussed relative to their potential to alter metals' bioavailability. The relevance of historical information on metals' contamination associated with other mine sites, along with the potential for acid mine drainage (AMD) and metals' release and exposure, based on review of the baseline data and geochemical characteristics at a site, have been used to develop both quantitative and qualitative predictions of risk.

The present ERA is designed to analyze and portray the potential risks to globally significant salmon resources of the Nushagak-Mulchatna, and Kvichak river drainages [proximal headwater areas] as a result of large-scale mining and associated facilities. These risks include both physical destruction and alteration of salmon habitat, in addition to probable effects from changes to water chemistry and other supporting habitat as a result of AMD and the influx of metals within the aquatic ecosystem from various sources. Although the ERA generally presents impacts to salmon from loss of food resources such as benthic invertebrates, it does not focus on specific effects to these fauna. Similarly, although risks to non-anadromous fish within potentially affected stream segments may be similar to salmon, these taxa are not addressed individually within the ERA.

1.1 Large-scale Mining Scenario Used for Assessment

To characterize risk associated with large-scale mining in the Nushagak-Mulchatna and Kvichak watersheds, this ERA employs a specific mining scenario. This scenario is based on the design of a large-scale mine detailed in permit applications submitted in 2006 by Northern Dynasty Minerals (NDM) to the Alaska Department of Natural Resources (ADNR). The proposed mine would be located in these watersheds on what is commonly referred to as "the Pebble prospect" and the applications are for water withdrawal to construct and operate the mine. Although this particular design may or may not form the basis for actual mine permitting in the future, this design does provide a conceivable scenario for how a large-scale mine might be constructed and operated for this prospect, and thus a suitable proxy for understanding the risks of such activity in these watersheds. In addition this ERA draws upon research performed on the Pebble prospect by a variety of interests as well as information publically available through the ADNR's Division of Mining, Land and

Water and from the websites of both NDM and the Pebble Limited Partnership, a successor in interest of NDM. For the remainder of this assessment this specific mining scenario will be referred to as "the proposed mine."

As of the date of this document, no comprehensive mine management plan (MMP) for mining in this area has been submitted for permitting or released to the public. Generally, a MMP is developed prior to mining commencement and includes identification and description of mining activities; particulars of the implementation of the management systems to address environmental issues; a plan and costing of closure activities; particulars of the organizational structure; and plans of current and proposed mine workings and infrastructure and other information as required. Detailed information on the mine's activities and management strategies, as provided in a MMP, may facilitate more precise estimates and characterization of risk. Although various details of a mine may change prior to final permitting, risks from various physical and chemical stressors are likely to be similar for any large-scale mines in this area.

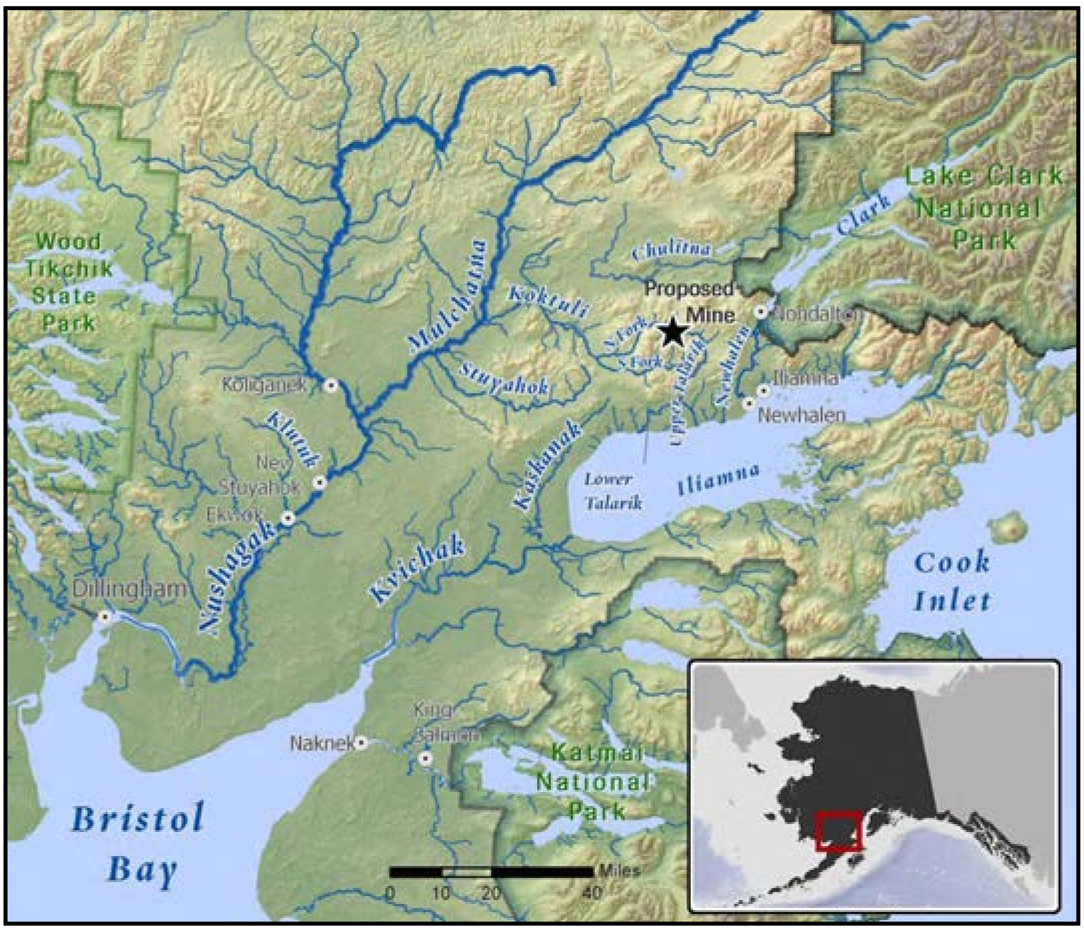

Figure 1. Site of Proposed Mine

1.1.1 Proposed Mine Characteristics

The Pebble prospect is located on 58,000 acres (90 square miles) of state land located approximately 25 miles north of Iliamna Lake in southwest Alaska (Figure 1). The area potentially ranks as the second largest copper mineral deposit in the world (NDM 2007). It is located within the Lake and Peninsula Borough.

The proposed mine would include:

- An open pit mine or an underground mine, or both. Long-term mining would result in an open pit at Pebble West up to 1,700 feet deep and cover about 2 square miles. [Pebble East has not been fully explored but appears to be of comparable size and underground block caving techniques might be used to mine to a depth of 5,000 feet];
- Various stream diversion channels, wells and devices to dewater the pit and extract water for mine processes;
- A mill to crush and process the ore;
- At least two tailings storage facilities (TSF) [ponds] with a combined surface area of approximately 10 square miles and storage capacity of 2.5 billion tons (NDM 2006c)]. The ponds would be created by five dams constructed of waste rock from the mine.
- Approximately 9 miles of dams would impound the reactive tailings ponds. The largest dam would be approximately 740 feet high;
- A deep-water port in Iniskin Bay, on the west side of Cook Inlet to load ore carriers;
- A new 104-mile road to connect the mine to the port;
- Two 104-mile long, 15-inch parallel pipelines to transport a slurry of copper ore concentrate from the mill to the port and return the water to the mine area; and
- A 300 megawatt power plant in the rail belt or on the Kenai Peninsula, and 135-mile transmission line from the Kenai Peninsula to the mine site.

This ERA does not, however, address risk relevant to processes or activities associated with a deep-water port, a power plant, or a transmission line.

1.2 Ecological Risk Assessment Technical Approach

The following sections present the approach and information used to determine the potential risks that may be present to salmon from the various stressors expected to occur during construction, operation and post-operation of the proposed mine. Section 2.0 – Problem Formulation, presents general information on stressors of concern and the biological resources that are at risk. In Section 3.0 – Risk Analysis, the magnitude and/or extent of each stressor is presented, followed by an in-depth evaluation of the overall expected impacts to salmonid populations. The Risk Analysis uses literature, historical and/or theoretical data/information to predict what the impacts to salmon would be based on construction requirements and operational activities, and considers potential mining duration and effects after closure. Finally, in Section 4.0 – Risk Summary and Conclusion we characterize each stressor's expected impacts on salmon, the cumulative risk expected from all stressors combined, the potential loss of salmon production from mine development, and lastly, the uncertainty associated with the ERA's findings, and an overall conclusion. Literature citations used for this ERA are provided in Section 5.0.

2.0 PROBLEM FORMULATION

The mining and transport of ores carries with it a number of potential direct and indirect environmental risks. From a direct perspective, various pre-operational, operational, and post-operational mining activities can result in impacts that may reduce or alter fishery habitats or populations in affected watersheds. These include both physical and chemical stressors such as:

1. Loss or alteration of habitat (including flow or temperature modifications) from mining activities (e.g., mine creation and expansion, tailings ponds, road construction, water diversion and dewatering) and subsequent reduction in fisheries' populations and genetic diversity;
2. Instream impacts to fish and habitat from road construction;
3. Dust containing metals released from mining activities to be deposited in the adjacent watersheds and then readily transported into rivers and streams;
4. Accidental release of ore concentrates from pipelines;
5. Chemical spills during transport, storage, and/or use; and
6. Acid mine drainage and release as a result of fractured ore body, tailings ponds infiltration; and
7. Episodic spill events and/or dam failures.

Indirect effects can result from each of these sources via benthic community structure shifts, and degradation, loss and/or contamination of benthic and other food resources. Again, the magnitude of these effects is a function of a stressor's physical/chemical characteristics and relative to its intensity, duration, frequency, timing, and scale.

2.1 Physical Stressors

Physical stressors include permanent removal of waterways that either directly support fisheries resources or provide necessary flow for species and population viability in downstream reaches. Similarly, stream crossing impacts may limit upstream migration and reduce reproductive potential for many of the populations affected. Reduced down-gradient stream water quality and quantity, and subsequent secondary effects to fisheries from sedimentation, could be expected from changes to groundwater and surface water flow, and from fugitive dust emissions, as a result of mine activities. Figure 2 presents a conceptual site model for pathways and exposures to relevant salmonid receptors from the physical stressors described.

Figure 2
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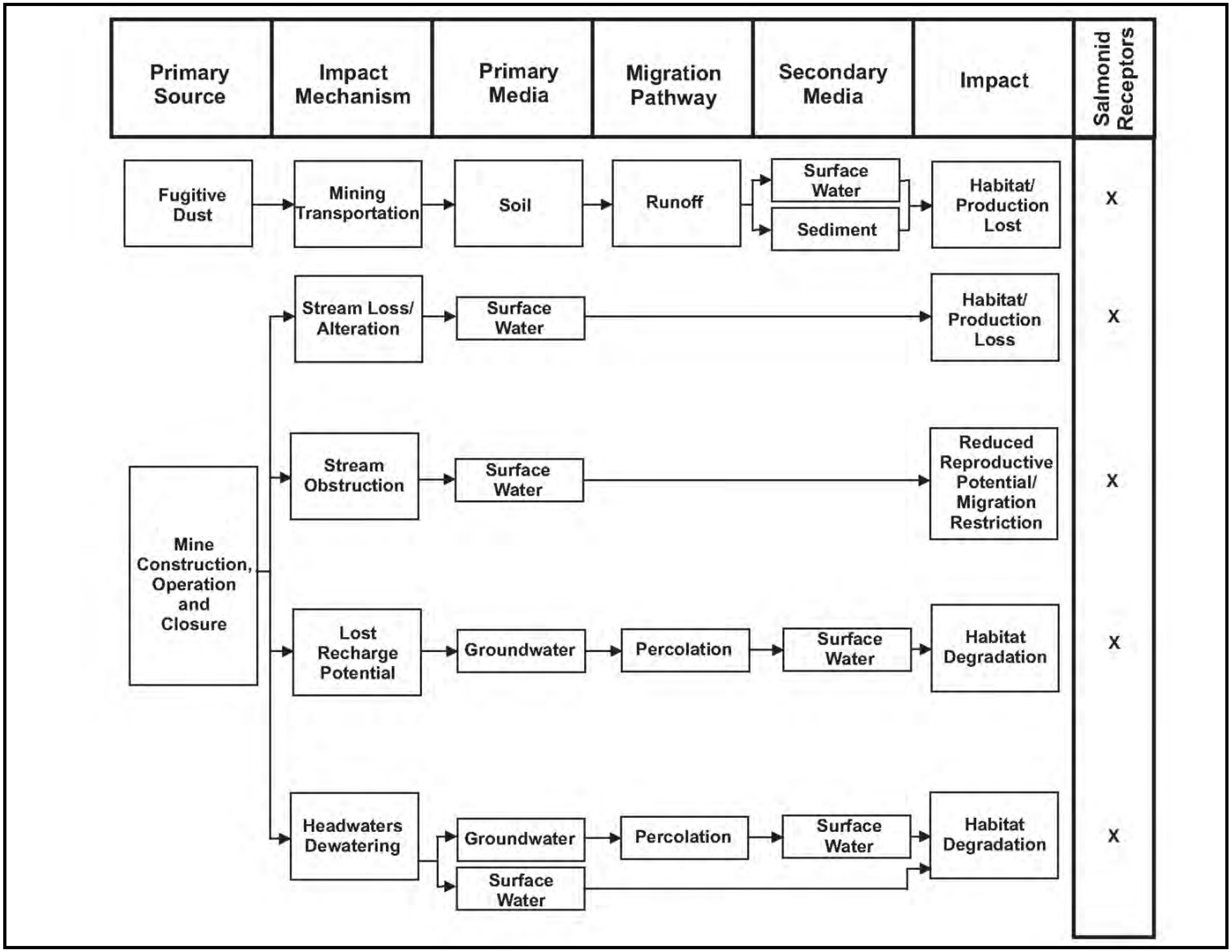


Figure 2. Conceptual Site Model – Physical Stressors

2.2 Chemical Stressors

Discharges of heavy metals (copper, nickel, zinc, etc.) via AMD and metals leaching from mine workings, tailings pond leakage, and waste rock piles, especially after mine closure, can impact fishery resources, both acutely and chronically, in affected water bodies. Short-term episodic events could be expected during high rainfall or snowmelt events, resulting in lowered instream pH values and subsequent bioavailability of metals to fish. Similarly, chronic impacts are expected over the life of the project from an infusion of metals in dust generated by the mining process and from inadvertent pipeline slurry spills within watersheds. Large-scale pollution events such as tailings dam failures can result in both short and long-term impacts to fishery resource in affected areas. Figure 3 presents a conceptual site model for pathways and exposures to relevant salmonid receptors from the chemical stressors described.

2.3 Resources at Risk

The Alaska Peninsula and Bristol Bay Basin are considered to be intact ecoregions with unimpeded natural ecological processes supporting healthy populations of terrestrial, avian, and marine species, including five species of anadromous Pacific salmon (Table 1). The region supports the largest runs of sockeye salmon on earth (Ruggerone et al. 2010).

Table 1. Fish Known to Occur in the Nushagak-Mulchatna and Kvichak River Watersheds, Bristol Bay, Alaska (Source: Woody 2009b)

Common name	Scientific name
Anadromous Salmon	
sockeye salmon	*Oncorhynchus nerka*
Chinook salmon	*Oncorhynchus tshawytscha*
coho salmon	*Oncorhynchus kisutch*
pink salmon	*Oncorhynchus gorbuscha*
chum salmon	*Oncorhynchus keta*
Resident Fishes	
northern pike	*Esox lucius*
least cisco	*Coregonus sardinella*
broad whitefish	*Coregonus nasus*
humpback whitefish	*Coregonus pidschian*
round whitefish	*Prosopium cylindraceum*
Arctic grayling	*Thymallus arcticus*
lake trout	*Salvelinus namaycush*
Arctic char	*Salvelinus alpinus*
Dolly Varden	*Salvelinus malma*
rainbow trout	*Oncorhynchus mykiss*
burbot	*Lota lota*
threespine stickleback	*Gasterosteus aculeatus*
ninespine stickleback	*Pungitius pungitius*
slimy sculpin	*Cottus cognatus*

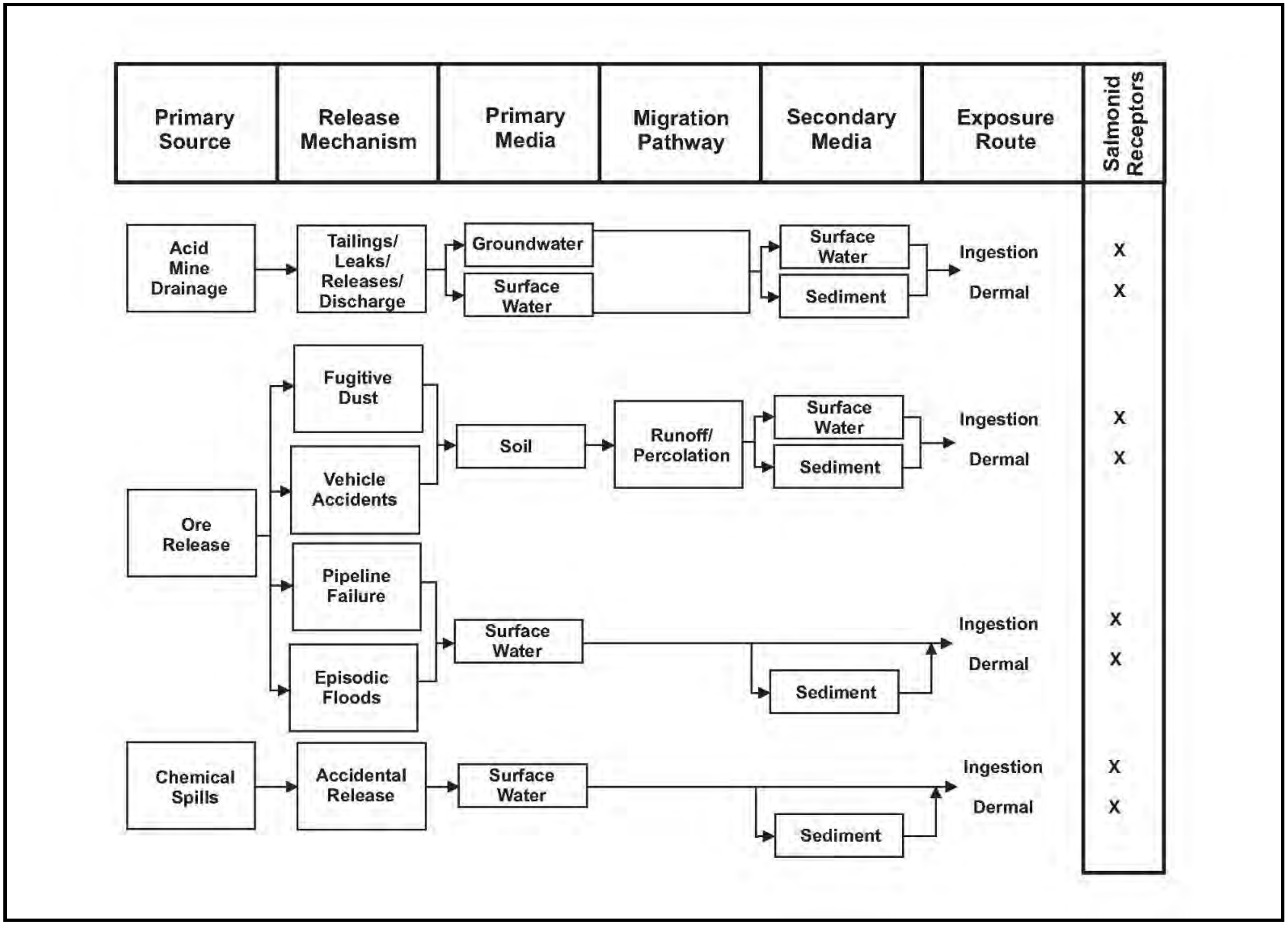


Figure 3. Conceptual Site Model – Chemical Stressors

Salmon species present in the Nushagak-Mulchatna and Kvichak river drainages which would be directly affected by the proposed mine include Chinook, sockeye, coho, chum and pinks. The North Fork Koktuli (NFK) and South Fork Koktuli (SFK) rivers and Upper Talarik Creek (UTC), which are the stream segments closest to the proposed mine site, also support large numbers of other high value resident fish species, including Arctic grayling, Arctic char, rainbow trout, Dolly Varden, and northern pike. The Koktuli River and Upper Talarik Creek provide sport fishing opportunities for rainbow trout, coho and Chinook salmon. Five species of Alaska salmon spawn into the Nushagak River: Chinook, coho, sockeye, chum and pink. Together the Kvichak and Nushagak River drainages have historically been the biggest producers of sockeye and other species of salmon in Bristol Bay. Hauser (2007) provided a summary of anadromous salmonids' freshwater habitat preference for spawning, rearing and overwintering for Bristol Bay (Table 2).

Table 2. Habitat Requirements for Select Bristol Bay Salmonids

Species	Spawning	Rearing	Overwintering
Sockeye Salmon	Stream/river riffles Beaches and upwelling in lakes	Lakes primarily River and ponds	Lakes primarily River and ponds
Chinook Salmon	River Deep riffles	Slow water along stream banks	Deep pools – between rocks
Coho Salmon	Headwater streams	Beaver ponds, Sloughs Small stream estuaries	Ponds and sloughs
Pink Salmon	Lower stream riffles Intertidal areas	Estuary	Marine
Chum Salmon	Upwelling areas in stream and sloughs	Estuary	Marine

Source: Hauser 2007

Information on distributions and life histories of anadromous salmon species is well documented in literature and has been summarized by the United States Fish and Wildlife Service (USFWS) in its *Species Profile Series* (Beauchamp *et al*. 1983, Pauley *et al.* 1989, Bonar *et al*. 1989, Laufle *et al*. 1986, Pauley *et al*. 1988). Historical information and potential mining impacts on Bristol Bay-specific salmon resources have also been addressed by Hauser (2007). The following sections provide life history information and environmental limitations for the five salmon species addressed within this ERA.

Coho Salmon (*Oncorhynchus kisutch*)

Koski (2009) has described the nomadic life-history of coho – adult coho salmon typically enter small coastal streams or tributaries of larger rivers and ascend to the headwaters to spawn, enabling their progeny to fill habitats throughout the system. Generally, in southwest Alaska, coho begin escapement into freshwater streams in late summer to fall (September – October) as flow increases (Drucker 1972; Crone and Bond 1976). Spawning occurs primarily in moderate-sized stream and tributaries of larger rivers (Morrow 1980). Coho do not use main channels of large rivers for spawning as heavily as do

Chinook, or intertidal reaches as heavily as do chum (Scott and Crossman 1973). Egg incubation period varies inversely with temperature and usually lasts 35 to 50 days (Shapovalov and Taft 1954). Fry generally emerge within 20 to 25 days after hatching (Mason 1976). Coho fry emerge from gravel from early March through mid-May. Newly emerged fry aggregate along stream margins, in shallow pools, and in backwaters and eddies (Lister and Genoe 1970) and gradually move into deeper pools. First year emigrants (< 40 millimeters [mm]) often makeup a major portion of seaward migrants, but their return probability is extremely low due to their poor physiological adaptation for surviving in high salinities. In Alaska, parr freshwater residence lasts from 2 to 4 years (Drucker 1972; Crone and Bond 1976).

Many processes and environmental factors initiate, control and affect parr-smolt transformation (smoltification) in coho and other anadromous salmon. Among these factors photoperiod, temperature and flow are especially critical (Parry 1960; Hoar 1965; Clark *et al*. 1978; Clarke and Shelbourn 1980; Wedemeyer *et al*. 1980). Smoltification and seaward migration occurs in the spring and often follows periods of rapid temperature warming (Shapovalov and Taft 1954), typically peaking in mid-June in southeast Alaska (Crone and Bond 1976).

Each of these life stages has environmental factors relevant to subsequent life stage development, maturation or viability. For adults, the accessibility of spawning stream and water quality appear to be major factors affecting coho during upstream migration. Thompson (1972) has recommended a minimum depth of 0.18 m and a maximum velocity of 244 cm/sec (~8 fps) as criteria for successful upstream migration of adult coho. Temperatures in excess of 25.5º C are lethal to migrating adults (Bell 1973). Sublethal temperatures may result in major pre-spawning mortalities through activation of latent infections. Temperatures ≤13º C have been recommended to minimize pre-spawning mortality of coho during upstream migration. Dissolved oxygen (DO) level > 6.3 mg/l are recommended for successful upstream migration of salmonids (Davis 1975). Lower DO concentrations adversely affect upstream migration by reducing the swimming ability of migrants and by eliciting avoidance responses. Also, the maximum sustained swimming speed of coho is sharply reduced at DO levels < 6.5 mg/l at all temperatures (Davis *et al*. 1963).

Redd construction occurs in swift, shallow areas at the head of riffles (Burner 1951; Briggs 1953). Preferred sites in riffle areas have velocities of 0.69 to 2.3 fps and minimum depths ≥ 15 cm (Smith 1973). Gravel and small rubble substrate with low amounts of fine sediments is optimum for survival, growth and development of embryos and alevins and for later emergence of fry (Platts *et al*. 1979). Although specific composition of substrates for high survival of embryos and alevins has not been established, Reiser and Bjornn (1979) estimated that redds with 1.3 to 10.2 cm diameter substrate sizes and a low percentage of fines resulted in high survival of embryos. In all studies, emergence of coho fry was high when fines were less than 5%, but dropped sharply when fines were greater than 15%.

Emergence and survival of embryos and alevins is greatly influenced by dissolved oxygen (DO) supply within the redd (Mason 1976). DO concentrations ≥ 8 milligrams per liter (mg/L) are required for high survival and emergence of fry. Coble (1961) and other researchers showed that embryo survival drops significantly at levels ≤6.5 mg/L; concentrations < 3 mg/L are lethal. DO supply/availability in redds relates to gravel permeability, water velocity, and DO concentrations. When any of these factors, alone or in combination, reduces intragravel DO supply below saturation, hypoxial stress occurs. Hypoxial stress results in delayed hatching and emergence, smaller size of emerging fry, and increased incidence of developmental abnormalities (Alderice *et al*. 1958, Coble 1961, Silver *et al*. 1963, Shumway *et al*. 1964, Mason 1976). Reiser and Bjornn (1979) recommend DO criteria concentrations at or near saturation, with temporary reductions no less than 5 mg/L, for successful reproduction. Temperatures in the 4.4 to 13.3º C range are considered optimum for embryo incubation; survival decreases if these thresholds are exceeded (Bell 1973, Reiser and Bjornn 1979).

Coho parr require an abundance of food and cover to sustain fast growth rates, avoid predation, and have successful freshwater rearing and development into smolts, in order to avoid premature displacement downstream to the ocean. Low levels of food result in larger and fewer territories per unit area, increased emigration of resident fry, and a slower growth rate of remaining fish. Substrate composition, riffles, and riparian vegetation appear to be the most important factors influencing production of aquatic and terrestrial insects as food for coho (Mundie 1969, Giger 1973, Reiser and Bjornn 1979). Highest aquatic invertebrate production was observed in substrates comprised of gravel and rubble (Giger 1973). Because substrate size is a function of water velocity, larger substrate sizes are associated with faster currents; food production is, thus, also higher in riffles (Ruggles 1966). Increased fines in riffles have been shown to reduce benthic food production (Phillips 1971). Embeddedness has been shown to correlate with lowest coho production (Crouse *et al.* 1981). Finally, riparian vegetation along coho streams act as habitat for terrestrial insects, as well as a source of leaf litter utilized by stream invertebrates as food (Chapman 1966).

Coho parr were shown to be most abundant in large, deep pools ([> 0.30 meters [m]). A pool-to-riffle ratio of 1:1 provides optimum food and cover conditions for coho parr. A shift in this ratio resulted in lower numbers of fry remaining in stream channels. As water temperatures decrease below 9º C, coho fry become less active and seek deep (≥ 45 centimeters [cm]), slow (< 0.5 fps) water in or very near dense cover of roots, logs, and flooded brush (Hartman 1965, Bustard and Narver 1975). Beaver ponds and quiet backwater areas, often some distance from the main channel and dry during summer low flow periods, are also utilized as winter habitat (Narver 1978). Several studies indicate that the amount of suitable winter habitat may be a major factor limiting coho production (Chapman 1966, Mason 1976, Chapman and Knudsen 1980). Winter cover is critical because as water temperatures drop swimming abilities of coho decrease.

Several studies have shown a positive relationship between stream coho carrying capacity and stream flow (Scarnecchia 1981). Lowest returns of adult coho coincide with low summer flows coupled with high winter floods. Burns (1971) found that the highest mortality of coho in the summer occurred during periods of lowest flows. Higher stream flows during rearing appear to provide more suitable habitat for growth and survival through increased production of stream invertebrates and availability of cover (Chapman 1966). Growth rate and food conversion efficiency of coho fry is optimum at DO concentrations above 5 mg/L. Below this concentration, growth and food conversion rapidly decrease.

Also, swimming speed decreases below 6 mg/L. Significant decreases in swimming speeds occur at temperatures > 20º C (Bell 1973).

Streamside vegetation is important for regulating temperature in rearing streams. Cooler winter water temperatures may occur if the stream canopy is absent or reduced. Stream canopy reduction during summer periods can result in increased water temperatures (>20º C) and increase fry mortality from disease (Hall and Lantz 1969).

The radical physiological and behavioral changes that occur during smoltification make this stage particularly vulnerable to environmental stressors. Elevated water temperatures can accelerate the onset of smoltification and shorten the smolting period, resulting in seaward migration at a time when conditions are unfavorable (Wedemeyer *et al.* 1980). Specifically, temperatures should not exceed 12º C during smolting and seaward migration in the spring to prevent shortened duration of smolting and premature onset of desmoltification, and to reduce the risk of infection from pathogens. DO requirements for smolts are unknown, but are probably similar to those for parr (McMahon 1983).

Chinook salmon (*Oncorhynchus tshawytscha*)

Chinook salmon spawning in the Nushagak River constitutes one of the major populations of North America. There is a large amount of diversity within the Chinook salmon species. The races of Chinook salmon reduce in number for each river system from south to north within their Pacific Coastal range. Freshwater entry and spawning timing are generally thought to be related to local temperature and water flow regimes (Miller and Brannon 1982). Only one run timing for Chinook salmon is found in most rivers in Alaska and northern British Columbia, where summers are short and water temperatures are cold (Burger *et al*. 1985).

The productive potential of the river system is most important to juveniles who may spend as much as three years in the environment prior to migration to sea. Juvenile winter and summer rearing habitat is a major factor in survival and production of Chinook salmon. The HSI model used to support the impact evaluation includes freshwater habitat requirements for all life stages, but is primarily concerned with embryo and juvenile habitat requirements (Raleigh *et al*. 1986).

A sustainable Chinook salmon population has habitat requirements that are similar to other salmonids. They require an abundant and diverse food supply of aquatic invertebrates; canopy cover to provide shade and allochthonous material; adequate stream flow to meet the needs of developing embryos and fry; appropriate substrate size to support spawning, embryo development and juvenile protection; and moderate water quality parameters to enhance productivity and development; in addition to other requirements.

Chum salmon (*Oncorhynchus keta*)

[Excerpts from ADF&G − http://www.adfg.state.ak.us/pubs/notebook/fish/chum.php]

Chum salmon have the widest distribution of any of the Pacific salmon. They range south to the Sacramento River in California and the island of Kyushu in the Sea of Japan.

Chum salmon are the most abundant commercially harvested salmon species in arctic, northwestern, and interior Alaska, but are of relatively less importance in other areas of the state. Chum salmon often spawn in small side channels and other areas of large rivers where upwelling springs provide excellent conditions for egg survival. Chum do not have a period of freshwater residence after emergence of the fry, as do Chinook, coho, and pink salmon. For those that migrate soon after emergence, their growth as fry is negligible. But growth may be significant for those that remain in freshwater for several weeks after emergence. Chums are similar to pink salmon in this respect, except that chum fry do not move out into the ocean in the spring as quickly as pink fry.

Escapement back to freshwater in Alaska typically occurs in the summer, but can extend to the fall. Eggs incubate in gravel redds for 50 to 130 days before hatching. Mortality during incubation is high; survival from egg deposition to fry emergence typically averages <10% (Hunter 1959). Fry emerge from the gravel in March to May. Some have delayed migrational patterns, allowing fish to feed in freshwater before entering the ocean. As with all salmonids, environmental factors that control downstream migration include temperature, photoperiod, light intensity, fish size, and level of river discharge (Brannon and Salo 1982).

Pink salmon (*Oncorhynchus gorbuscha*)

[Excerpts from ADF&G − http://www.adfg.state.ak.us/pubs/notebook/fish/pink.php]

Pink salmon occur throughout the Pacific northwest and into Canada and Alaska. Adult pink salmon enter Alaska spawning streams between late June and mid-October. Different races or runs with differing spawning times frequently occur in adjacent streams or even within the same stream. Most pink salmon spawn within a few miles of the coast and spawning within the intertidal zone or the mouth of streams is very common. Shallow riffles, where flowing water breaks over coarse gravel or cobble-size rock, and the downstream ends of pools are favored spawning areas. Sometime during early to mid-winter, eggs hatch. In late winter or spring, the fry swim up out of the gravel and migrate downstream into salt water.

Pink salmon adults and seaward migrating fry spend very little time in freshwater, and the entire juvenile stage is in seawater; thus, habitat requirements associated with potential impacts from mine creation concentrate on the requirements of the developing embryos and yolk sac fry.

Sockeye salmon (*Oncorhynchus nerka*)

[Excerpts from ADF&G − http://www.adfg.state.ak.us/pubs/notebook/fish/sockeye.php]

The sockeye salmon (*Oncorhynchus nerka*), often referred to as "red" or "blueback" salmon, occurs in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Klamath River in California and northern Hokkaido in Japan, to as far north as far as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia. The sockeye salmon is found in stream and river drainages from Southeast Alaska to Point Hope. Spawning rivers are those typically having lakes in their systems (Hart 1973). The largest sockeye populations in Alaska are in the Kvichak, Naknek, Ugashik, Egegik, and Nushagak Rivers that flow into Alaska's Bristol Bay. Adult sockeyes return to spawn between July and October. The female most often selects a redd site in an area of the stream with fine gravels. She deposits between 2,500 to 4,300 eggs in 3 to 5 redds that are fertilized by the male. Spawning can take place over three to five days.

Hatching occurs from mid-winter to early spring, and sac-fry, or alevins, remain in the gravel, living off the material stored in their yolk sacs, with emergence from the gravel between April and June. After emerging from the stream gravel, the fry swim upstream or downstream to a lake. They live there for one to two (or rarely three or four years) before migrating to the sea. Initially, the fry stay in the shallow water near the lake shore, but gradually move into deeper water. While in the lakes, they feed on aquatic insects and plankton. Peak migration from lakes to the ocean occurs in June in Bristol Bay. Once in the sea, sockeye salmon smolts stay close to shore initially, but gradually move into deeper water.

Egg hatching under experimental conditions has occurred across a wide range of temperatures (Scott and Crossman 1973). In Washington, Brett (1952) estimated an upper lethal temperature to juveniles of 24.4ºC, with preferred temperatures ranging from 12 to 14 ºC. Smolt out-migration takes place at surface temperatures approaching 4 to 7 ºC (Hart 1973). Adult spawning occurred at temperatures from 3 to 10 ºC (Scott and Crossman 1973). Water temperatures of 20 ºC have been shown to be lethal to upstream-migrating spawners (Foerster 1968).

Optimum pH values are typical to most taxa ranging from 6.7 to 8.3 (Bell 1973). Effects data was unavailable for optimum DO concentrations, but general water quality standards indicate viable populations require concentrations no less than 7 mg/L. As with other salmonids, high turbidities result in increased sedimentation and lethality to eggs and alevins by reducing water interchange in the redd. Bell (1973) noted effects to eggs (i.e., 85% mortality) as embeddedness reached 15 to 20%. Turbid water will absorb more solar radiation than clear water and may thus indirectly raise thermal barriers to migration (Reiser and Bjornn 1979). Sufficient water velocity and depth are needed to allow proper intragravel water movement. Reiser and Bjornn (1979) suggest optimal velocity at spawning sites ranging from 0.21 to 1.01 m/s [0.7 – 3.3 fps] with depth usually ≤0.15m.

3.0 RISK ANALYSIS

3.1 Physical Stressors

The following sections (3.1.1 – 3.1.5) provide detailed summaries of the extent and magnitude for physical stressors that are expected to impact salmonid resources. Water extraction/reduction from the proposed mine are predicted to be the root cause for stressors addressed in sections 3.1.1, 3.1.2, and 3.1.3. Because of this, methods for evaluating impacts from these three stressors are treated together and presented in sections 3.1.3.2 and 3.1.3.3. Methods for evaluating fish passage obstruction and turbidity impacts from road construction are presented in Section 3.1.4. Finally, the physical impacts to vegetation and their indirect effects on salmon from fugitive dust emissions generated at the mine are evaluated in Section 3.1.5.

3.1.1 Dewatering and Loss of Instream Flow

3.1.1.1 Stressor Description

All of the ground and surface water within the proposed mine (i.e., 'mine' refers to all non-peripheral elements such as roads, pipelines, etc.) area (see Figure 4), which encompasses the headwater of the NFK, SFK, and UTC, would be appropriated and subject for mine use over the 40- to 70-year life of the mine (NDM 2006a and 2006c). Because the number of fish produced is determined by the quality and amount of habitat available in a stream (NRC 1996, Heggenes *et al*. 1998), the loss of this flow over the lifetime of the mine (and possibly beyond) could reduce the number of resident and anadromous fish produced within the NFK, SFK and UTC.

The surface water appropriation for the mine and the tailings facilities would eliminate all flow and fish habitat in the upper portions of the main stem of the SFK and its headwater's tributaries, in tributary 1.190 to the NFK, and the tributaries to UTC, upstream of the point where the water is removed (see Figure 4). The fish stocks which may be genetically unique to these streams would be extirpated. The portions of these streams that would be excavated or buried under tailings would no longer produce fish even after the mine is closed.

Below the proposed mine, stream flow would be lost or reduced and fish habitat would be diminished for some distance downstream. Flow would gradually be restored downstream as unaffected tributaries (and groundwater) empty into the main channel for each of these three streams. NDM (2006c) has estimated that the net reduction in stream flow would be:

- 8% on the NFK, 18 miles downstream;
- 16% on the SFK, 12 miles downstream; and
- 9% on the UTC, 18 miles downstream.

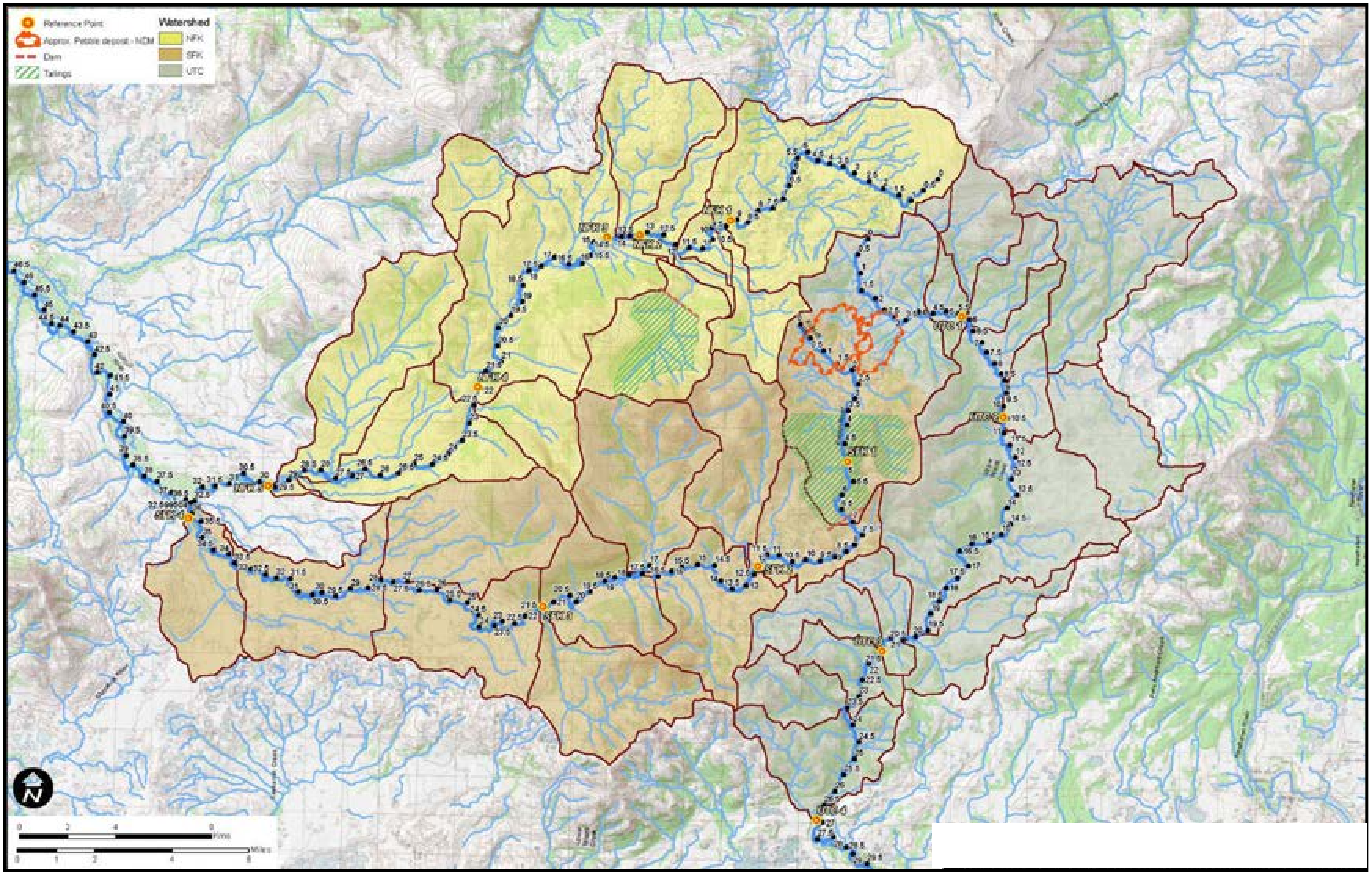


Figure 4. Proposed mine project components.

The reduction of habitat (stream width, depth and riparian zone) from these appropriations would substantially reduce available spawning and rearing habitat, particularly during the summer low flow period when Chinook, sockeye, and chum salmon are spawning, and would reduce the amount of available overwintering habitat for eggs and juvenile salmon during critical winter low flow periods.

A hydrogeology study also found that the middle section of the SFK goes dry during low flow periods during the summer (NDM 2005e). It is thought this is an indication that the SFK may be contributing cross drainage groundwater flow to Kaskanak Creek. The proposed surface water appropriations from the SFK would increase the frequency and length of the periods when upstream fish migration in the middle and upper SFK would be blocked [by a dry stream bed] and spawning and rearing habitat would not be available.

Predicting the impact of instream flow reductions from the proposed surface and groundwater appropriations for the proposed mine on fish production is complex and imprecise without long-term data. However, the loss of ground and surface water below the mine and tailings ponds may reduce salmon spawning and rearing habitat and overall production. It is likely that the proposed groundwater and surface water appropriations for the proposed mine would reduce downstream production of adult salmon and other fish species to some degree. Fish production would be incrementally reduced over the 40- to 70-year life of the project.

3.1.2 Groundwater Discharge

3.1.2.1 Stressor Description

Like streams and rivers, groundwater moves down gradient from high areas to low areas at right angles to subterranean contour lines. Because these contours are covered with layers of permeable soil, it is often difficult to model the direction and rate of flow with any degree of accuracy. Groundwater moves through the subsurface like water on the surface, except it travels more slowly. In sand and gravel it may move up to 5 feet per day; in other types of material it may move a few inches per day (EPA 2007b). NDM's 2004 baseline studies report indicates that the movement of groundwater in the mine area is relatively rapid (NDM 2005e).

Streams receiving ground and surface water from a mine site can be classified as “gaining” or “losing.” Gaining streams receive much of their water from groundwater, and the water level in the stream is generally at the same elevation as the water table in the adjacent aquifer. Water quality in the stream will then be affected by the quality of groundwater entering the stream. Because the water table elevation is approximately the same as the gaining stream surface elevation, both elevations may be used to predict the general direction of groundwater flow. Losing streams lose water to the adjacent aquifer because the water table has dropped below the stream level. If there is no major source of upstream flow, the stream may dry up between rainfall events (EPA 2007b).

NDM installed stream gauges to monitor surface flow and monitoring wells to provide information about groundwater quality and movement. Based on stream flow information provided in NDM's 2004 Environmental Studies and their 2006 water rights application, it appears that groundwater from the proposed mine area is an important contributor to stream flow in NFK, SFK and UTC (NDM 2005e and NDM 2006a); that is, the upper reaches of the streams may be classified as "gaining" streams. This appears to be particularly true during critical summer and winter low flow periods when there is little precipitation and surface run off.

Groundwater flow dynamics in the Pebble prospect area have important implications for fish habitat and water quality. In the upper reaches, groundwater is the most important water source to stream gravel during low flow periods in July and August when sockeye, Chinook, and chum salmon are spawning, and from January through March when incubating eggs and over wintering juvenile salmon require a consistent inflow of groundwater to survive (Groot and Margolis 1991). The removal of groundwater from these drainages would reduce both the amount of available spawning area for salmon and resident fish species and critical overwintering habitat. This, in turn, would reduce salmon and resident fish production from these three streams. In the lower reaches, adequate groundwater flowing from the upper reaches may be necessary to prevent these losing reaches from going dry.

3.1.3 Loss or Alteration of Habitat

3.1.3.1 Stressor Description

The proposed mine, including the Pebble West pit, the Pebble East pit or block cave, and the tailings ponds as currently proposed, would physically occupy approximately 33 square miles. This equates to over 68 miles of streams removed within the NFK, SFK, and UTC drainages (see Figure 4). All of the resident and anadromous fish habitat in the area would be destroyed, as would upgradient ephemeral streams that supply water to downstream environments. Based on information provided by NDM (2006c), approximately 3.5 miles of Stream 1.190, a tributary of the NFK which supports grayling, large numbers of Dolly Varden, spawning adult coho, and rearing juvenile coho and Chinook salmon, would be buried under Tailings Pond G. The headwaters of UTC and many of its tributaries are underlain by the Pebble East and West ore bodies and would be removed by mining, and the water appropriated for mine use.

A portion of the headwaters of UTC, which would be excavated for the proposed Pebble West pit or underlain by the proposed Pebble East Pit, is listed in the Alaska Department of Natural Resources Anadromous Waters Catalog (AWC). UTC has been identified as important for sockeye and coho salmon spawning, and high value rearing habitat for coho, Chinook, Dolly Varden, and rainbow trout. Although under our scenario it remains unclear how the deeper Pebble East Ore body would be mined (i.e., open pit or block caving), it seems almost certain that in either case the section of the main stem of UTC flowing over or adjacent to the proposed East pit, would flow into the mine and would have to be rerouted around the mine or appropriated for mine use. If this is the case, then fisheries production from a one- to two-mile segment of the main stem of UTC would be lost.

3.1.4 Impact Methodology for Loss of Instream Flow and Alteration of Habitat

The approach for determining effects to salmon from degradation of stream habitat has been to assess the present and future function and supportability of the streams that are predicted to be affected by mine development. As previously discussed, upper portions of the NFK, SFK and UTC would either be totally eliminated or re-directed for mine use. Figure 5 shows the proposed water extraction boundaries from the 2006 water rights application (NDM 2006c), which requests that all surface and groundwater within the designated water extraction boundaries be appropriated for mine usage.

Although NDM's applications provided data on pre- and post-development flows expected immediately outside of the designated extraction areas and within the mainstems at the most down-gradient monitoring site (e.g., most distal point in each watershed), they did not provide information on incremental changes within other portions of the watershed. This is important from an impact perspective because unaffected portions of the watershed may provide inputs to subsequent reaches of a mainstem and/or provide habitat support independent of mine-affected areas. Conversely, down-gradient portions of the watershed may be significantly affected by mine-related water removal actions, resulting in reduced viability for salmon (and resident taxa).

Surface runoff comprises all the water flowing on the earth's surface in response to precipitation. Although it is understood that groundwater will contribute to flow within watersheds (and their subbasins), the lack of site-specific groundwater discharge information prevented determination of this factor's relevance for stream flow calculation. For this evaluation, because quantitative groundwater contribution information was not available, it was presumed that all groundwater discharges in extraction and non-extraction areas were uniform and, thus, relative comparison of flow impacts along the river continuum was appropriate.

To determine the incremental changes in mainstem water flow a geographic information system (GIS)-based approach was used that considered and evaluated watershed subbasins' contributions to flow for both pre- and post-operational periods. To do this, drainages were identified within each of the three watersheds using a USGS-based Digital Elevation Model (DEM). First, Arc-GIS hydrologic modeling was used to determine flow direction, flow accumulation, stream order, and, finally, the watershed delineation. Inputs into the hydrologic model included a depressionless, 30-meter, 15-minute USGS DEM. The user parameter included an "expression," or a flow value, that determined when a cell is considered to be a stream.

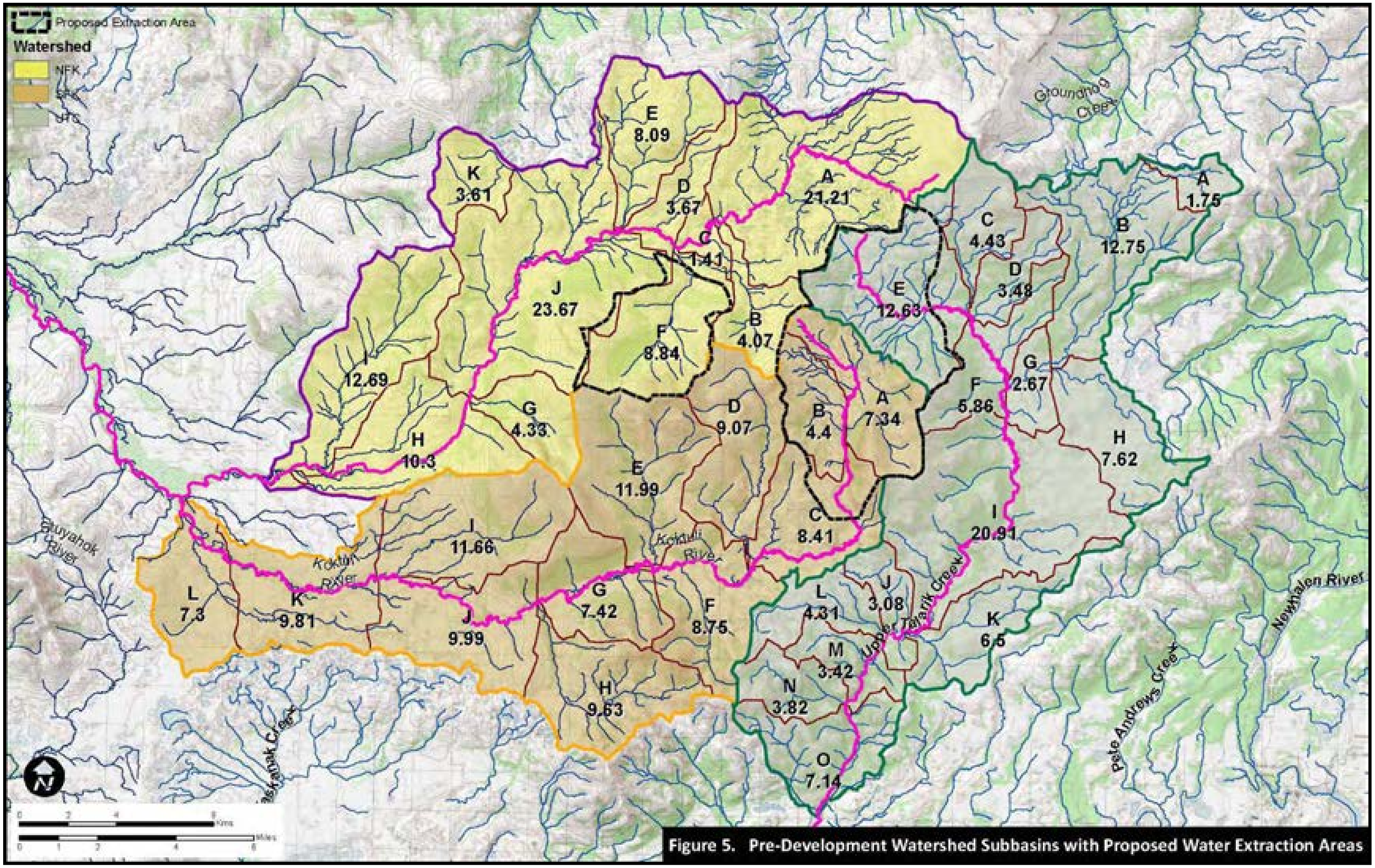
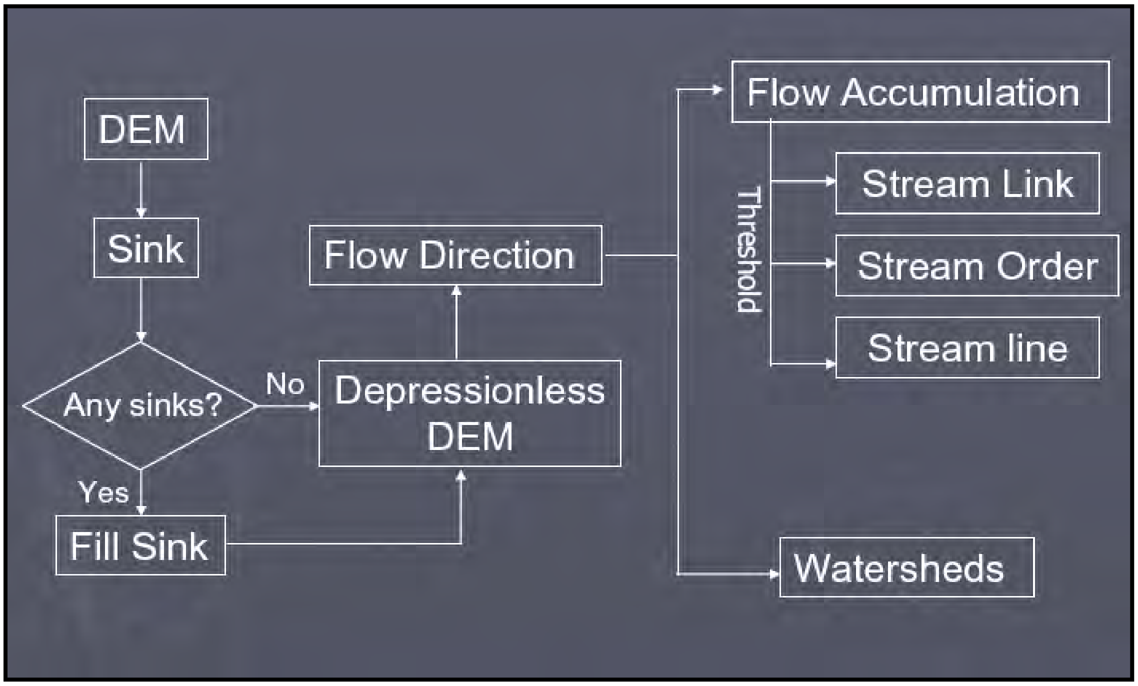


Figure 5. Pre-Development Watershed Subbasins with Proposed Water Extraction Areas

For the DEM, a cell size of 2000 was used in the analysis, although several tests were conducted using cell sizes of 500, 1000, 2000 and 2500, with 2000 resulting in the best resolution and scale for the project area. This means that a cell was considered to be a stream when 2000 other cells flowed into it. Flow was determined by the model using the topographic elevation model. Below, a basic flow chart diagrams the steps used in the hydrologic model. Watersheds were then delineated by the model based on stream order.



Subbasin (e.g., watershed sub-drainage areas) areas (square miles [mi^2]) that were determined using the GIS approach described above were then multiplied by average *unit runoff factors* (Table 3) that were provided in the 2006 water rights application (NDM 2006c). A *unit runoff factor* is based on overland flow per unit of land mass (mi^2) and expressed as a flow rate (cubic feet per second [cfs]). Unit runoff factors were provided on a monthly basis; thus, allowing for calculation of monthly flow for each subbasin (see Appendix A, Tables A-1, A-2, and A-3).

Table 3. Monthly Mean Watershed Unit Runoff Factors[1] (cfs)

Month	Watershed		
	N. Fork Koktuli	S. Fork Koktuli	Upper Talarik Ck.
January	0.7	1.3	1.2
February	0.6	0.8	0.7
March	0.6	0.6	0.5
April	2.2	1.0	3.1
May	9.5	8.4	5.0
June	3.3	2.8	2.4
July	1.7	1.4	1.8

Table 3. Monthly Mean Watershed Unit Runoff Factors[1] (cfs)

Month	Watershed		
	N. Fork Koktuli	S. Fork Koktuli	Upper Talarik Ck.
August	2	1.6	1.7
September	4.6	6.4	4.6
October	3.5	4.2	3.8
November	3.5	3.2	3.5
December	2.0	2.0	1.9

1 = From NDM 2006c.

Next, stations were selected along each stream for determination of incremental pre- and post-development flow rates (see Figures 5 and 6). Based on cumulative upstream subbasins' contributions, discharge (flow) rates at each station were derived (see Tables 4, 5, and 6). The results of this exercise revealed flow changes (e.g., percent change) along mainstem channels after project development. The flow results noted at each of the most downstream stations were fairly similar to stream percent reductions provided in NDM's water right application (NDM 2006c) and discussed previously in Section 3.1.1.

Table 4. Pre- and Post-Development Flow Rates at Five Selected Stations along the North Fork Koktuli River

Month	Flow (cfs)									
	Station 1[1]		Station 2[2]		Station 3[3]		Station 4[4]		Station 5[5]	
	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-
Jan	14.8	14.8	27.4	21.7	33.1	27.4	52.7	47.0	71.8	66.1
Feb	12.7	12.7	23.5	18.6	28.4	23.5	45.2	40.3	61.5	56.6
Mar	12.7	12.7	23.5	18.6	28.4	23.5	45.2	40.3	61.5	56.6
Apr	46.5	46.5	86.1	68.2	104.0	86.1	165.6	147.7	225.7	207.8
May	200.9	200.9	371.8	294.5	449.2	371.8	715.2	637.8	974.4	897.0
Jun	69.8	69.8	129.2	102.3	156.0	129.2	248.4	221.6	338.5	311.6
Jul	35.9	35.9	66.6	52.7	80.4	66.6	128.0	105.7	161.5	148.7
Aug	42.3	42.3	78.3	62.0	94.6	78.3	150.6	134.3	205.1	188.8
Sep	97.3	97.3	183.9	142.6	217.5	183.9	346.3	308.8	471.8	434.3
Oct	74.0	74.0	137.0	108.5	165.5	137.0	263.5	235.0	359.0	330.5
Nov	74.0	74.0	137.0	108.5	165.5	137.0	263.5	235.0	359.0	330.5
Dec	42.3	42.3	78.3	62.0	94.6	78.3	150.6	134.3	205.1	188.8
Annual Percent Change	0%		-21%		-17%		-11%		-8%	

1 = Includes drainage from subbasin A.
2 = Includes drainage from subbasins A, B, C, D and F.
3 = Includes drainage from subbasins A, B, C, D, F and E.
4 = Includes drainage from subbasins A, B, C, D, F, E, K and J.
5 = Includes drainage from subbasins A, B, C, D, F, E, K, J, G, H and I.

Table 5. Pre- and Post-Development Flow Rates at Four Selected Stations along the South Fork Koktuli River

Month	Flow (cfs)							
	Station 1[1]		Station 2[2]		Station 3[3]		Station 4[4]	
	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-
Jan	15.2	0	26.1	8.4	74.5	56.8	137.5	119.8
Feb	13.0	0	22.4	7.2	63.8	48.7	117.8	102.7
Mar	13.0	0	22.4	7.2	63.8	48.7	117.8	102.7
Apr	47.8	0	82.0	26.4	234.2	178.5	432.2	376.5
May	206.3	0	354.2	114.0	1011.0	770.8	1866.0	1625.8
Jun	71.7	0	123.0	39.6	351.2	267.8	648.2	564.7
Jul	36.9	0	63.4	20.4	181.0	138.0	334.0	291.0
Aug	43.4	0	74.6	24.0	212.8	162.3	392.8	342.3
Sep	99.9	0	171.5	55.2	489.5	373.2	903.5	787.2
Oct	76.0	0	130.5	42.0	372.5	284.0	687.5	599.0
Nov	76.0	0	130.5	42.0	372.5	284.0	687.5	599.0
Dec	43.4	0	74.6	24.0	212.8	162.3	392.8	342.3
Annual Percent Change	-100%		-68%		-24%		-13%	

1 = Includes drainage from subbasins A and B.
2 = Includes drainage from subbasins A, B and C.
3 = Includes drainage from subbasins A, B, C, D, E, F and G.
4 = Includes drainage from subbasins A, B, C, D, E, F, G, H, I, J, K and L.

Table 6. Pre- and Post-Development Flow Rates at Four Selected Stations along the Upper Talarik Creek

Month	Flow (cfs)							
	Station 1[1]		Station 2[2]		Station 3[3]		Station 4[4]	
	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-
Jan	15.2	3.0	61.4	49.2	103.2	91.0	120.5	108.3
Feb	13.0	2.6	52.6	42.3	88.4	78.0	103.3	92.8
Mar	13.0	2.6	52.6	42.3	88.4	78.0	103.3	92.8
Apr	47.8	9.4	193.0	154.6	324.4	286.0	378.7	340.4
May	206.3	40.7	833.3	667.7	1400.5	1235.0	1635.3	1469.7
Jun	71.7	14.1	289.4	231.9	486.5	429.0	568.0	510.5
Jul	36.9	7.3	149.1	119.5	250.7	221.0	292.7	263.1
Aug	43.4	8.6	175.4	140.6	294.8	260.0	344.3	309.4
Sep	99.9	19.7	403.5	323.3	678.1	598.0	790.5	711.6
Oct	76.0	15.0	307.0	246.0	516.0	455.0	602.5	541.5
Nov	76.0	15.0	307.0	246.0	516.0	455.0	602.5	541.5
Dec	43.4	8.6	175.4	140.6	294.8	260.0	344.3	309.4
Annual Percent Change	-80%		-20%		-12%		-10%	

1 = Includes drainage from subbasin E.
2 = Includes drainage from subbasins E, A, B, C, D, F, G and H.
3 = Includes drainage from subbasins E, A, B, C, D, F, G, H, I, K, J and L.
4 = Includes drainage from subbasins E, A, B, C, D, F, G, H, I, K, J, L, M, N and O.

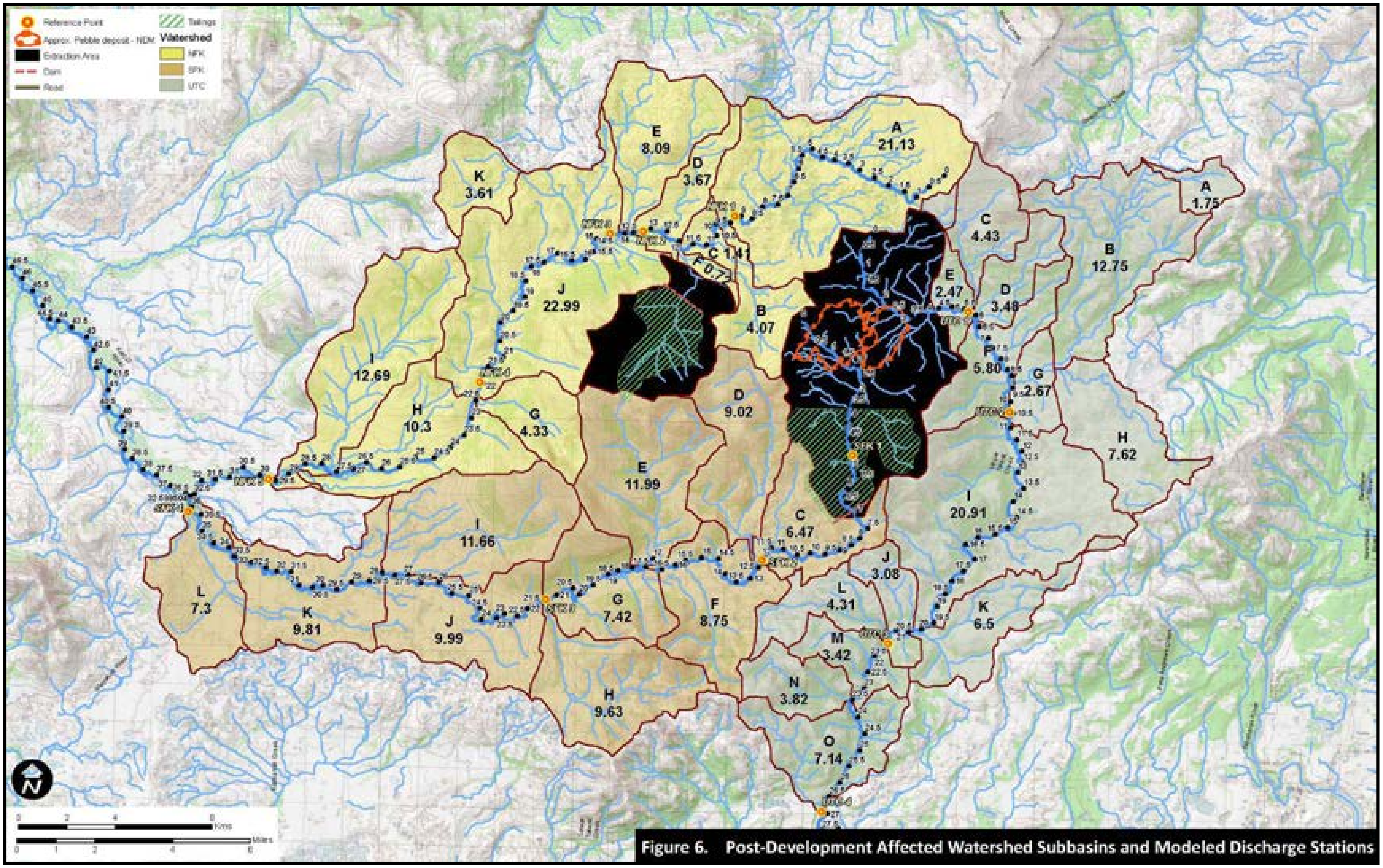


Figure 6. Post-Development Affected Watershed Subbasins and Modeled Discharge Stations

Next, to assess the potential effects to salmon from post-mining stream discharge and velocity, measurements collected by Woody (2009b) in headwater streams and by USGS (2009a) at mainstem gage stations were evaluated (Table 7). Again, a stream's velocity regime is an important habitat attribute that can potentially jeopardize salmon sustainability. Generally, during their August-September sampling period, Woody (2009b) measured velocities from 0.18 to 0.94 feet per second (fps), with an average of 0.5 fps. USGS collected point-in-time velocities during May, July, and September 2008 in downstream portions of each stream. Review of USGS gage data indicates that 2008 flow was similar to the previous three- to four-year period.

Table 7. Velocity and Discharge Information for Project Streams

Location	Date	Discharge (cfs)	Velocity (fps)	Source
SFK - Station 15302200	05-13-08	279	2.53	USGS[1]
	07-23-08	211	2.41	
	09-04-08	94.7	1.23	
NFK – Station 15302250	05-13-08	929	3.81	
	07-23-08	291	3.80	
	09-04-08	231	1.78	
UTC – Station 15300250	05-13-08	954	5.19	
	07-23-08	255	4.03	
	09-04-08	209	2.88	
NFK – 1b	09-02-08	0.649	0.21	Woody 2009b
NFK – 2	09-02-08	1.41	0.9	
NFK – 4	08-31-08	1.85	0.3	
UTC – 12/13	08-31-08	1.37	0.39	
UTC – 20	09-01-08	2.77	0.23	
SFK – 28	09-02-08	0.636	0.51	
UTC – 38L	08-29-08	0.89	0.18	
NFK – 40	08-30-08	3.81	1.01	
NFK – 41	08-30-08	2.08	0.58	
UTC – 49	09-01-08	3.04	0.94	

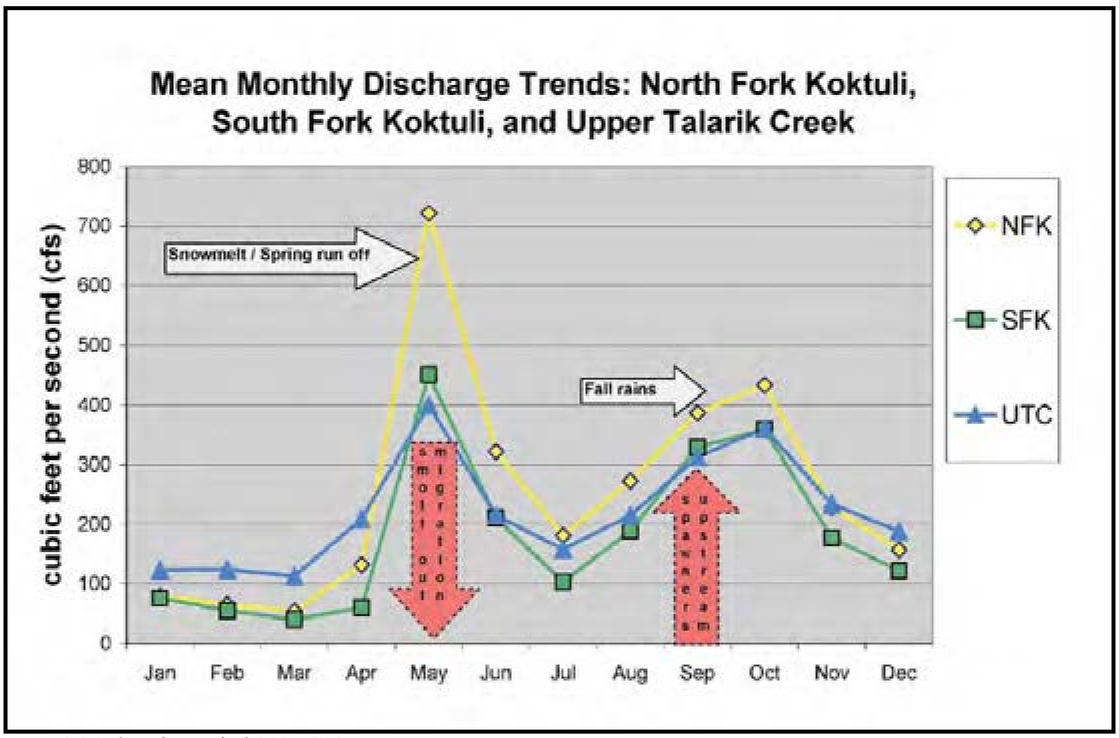
1 = C. Smith (USGS) personal communication with D. Trimm (E & E), date April 10, 2009.

The results of this analysis were then used to evaluate the impacts expected to target salmon species within each of the three watersheds based on long-term flow reduction (40 to 70-year life of the mine). Although the flow and velocity changes are, at present, theoretical for these watersheds, the impacts represented by reductions are based on well established habitat and environmental requirements for the species under consideration. For several species with similar life histories, effects may be similar and are, thus, treated simultaneously within discussions.

Watershed Flow Characteristics

Water flow levels in streams affect all aquatic life, and there is a definite relationship between the annual flow regime and the quality of salmonid riverine habitat (Raleigh *et al*. 1986). The complex life history of anadromous salmon have evolved with natural flow fluctuations of coastal rivers and are tuned to their home rivers for such things as spawn timing and smolt out-migration. Low flow conditions are recognized as potentially limiting to remnant wild salmon populations and have been addressed in other areas of the United States and Canada. The National Marine Fisheries Service (NMFS) and USFWS (2004) Draft Recovery Plan for the Gulf of Maine Distinct Population Segment (DPS) of Atlantic Salmon lists "*excessive or unregulated water withdrawal*" as one of the reasons for the need to protect the species under the Endangered Species Act. They noted that "*the potential impacts of water withdrawals from DPS rivers and streams include limiting summer habitat for parr, low winter flow effects on redds and egg incubation as well as adult immigration and smolt emigration*." Similarly, the Pacific Fishery Management Council (PFMC 1999) has noted that low flow can result in negative effects to salmon through "*crowding and increased competition for foraging sites, reduced primary and secondary productivity, increased vulnerability to predation and increased fine sediment deposition*."

The annual general flow cycles for rivers associated with the proposed project are presented in Figure 7.



Source: USGS data for period 2004-2007

Figure 7. Mean Monthly Discharge Trends in North Fork Koktuli, South Fork Koktuli, and Upper Talarik Creek Watersheds

For NFK, SFK, and UTC, there would be an incremental reduction within down-gradient portions of each of the three major watersheds based on water extraction and operational needs of the proposed mine (see Tables 4, 5, and 6, respectively). The nature of reduced flows would vary between each watershed based on the water extraction limits and nature of the watershed in question. For instance, for the NFK (see Table 4), flow in upper tributaries (i.e., subbasin A) above Station 1 (see Figure 6) should remain after development, but flow at Station 2 would reflect inputs/reductions from subsequent downstream watershed subbasins (e.g., [A] + B, C, D, and F). The nearly total [~100%] elimination of runoff from subbasin F (a fairly large subbasin) after development would result in an approximate 21% reduction in flow within the mainstem of the NFK at Station 2 (see Table 4). Proceeding downstream in the watershed, the impact of water extraction on flow would be reduced, but not eliminated.

The SFK watershed shows a different scenario as ‘all’ (subbasins A and B) or ‘some’ (subbasin C) flow from originating headwaters would be affected. As a result, downstream flow loss/reduction is predicted to be significant at Station 1 (100%), Station 2 (68%) and even as far downstream as Station 3 (24%), along the SFK mainstem (see Table 5).

In the UTC, most of the flow (80% at Station 1) from subbasin E would be lost. Upgradient subbasins A, B, C, and D would still provide ‘some’ flow as a result of unaffected headwater areas (see Table 6). As a result, downstream flow loss/reduction is predicted to be 20% at Station 2, 12% at Station 3, and at Station 4, near the end of the watershed, 10% (see Table 6).

This situation of variable flow reductions within the three watersheds would affect salmonid populations to varying degrees from one river to another. As potential impacts are described and predicted in the following paragraphs, statements regarding each impact’s relevance or severity will be discussed relative to each of the three rivers.

Habitat Evaluation Approach

Evaluation of the habitat presently available for coho (and other salmonids) and the prediction for change in that habitat after mine development is the method proposed for assessing effects to salmon viability in affected water bodies. In order to do this, an evaluation of the necessary habitat requirements and life requisites has been conducted using data developed for USFWS’s Habitat Evaluation Procedure (HEP) (USFWS 1980).

HEP (USFWS 1980) methods are based on habitat suitability index (HSI) models that provide habitat information for evaluating impacts on fish (and wildlife) habitat resulting from water or land use changes. Detailed descriptions of HEP methods are provided by Terrell *et al*. (1982) and Armour *et al*. (1984). HSI reports synthesize habitat information into explicit habitat models useful in quantitative assessments. Models in this series reference numerous literature sources in an effort to consolidate scientific information on species-habitat relationships. HSI models are usually presented in three basic formats: (1) graphic; (2) word; and (3) mathematical. Their value is to serve as a basis for improved decision-making and increased understanding of habitat relationships because they specify hypotheses of habitat relationships that can be tested and improved (USGS 2009b). HSI model are available for Chinook, chum, coho, and pink salmon (Raleigh *et al.* 1986, Hale *et al.* 1985, McMahon 1983, and Raleigh and Nelson 1985, respectively). For this ERA, it is important to note that HSI models were used only for correlating species-habitat relationships, and applied as a tool to evaluate change in habitat value that may result from changes to the environment expected within the watersheds under investigation.

A HSI can either be empirical regressions, mechanistic models, or descriptions (a judgment call based on opinion, literature, or other data). Mechanistic approaches are most commonly used and require the use of suitability index (SI) curves. A mechanistic model is structured as a tree diagram in which the variable at the end of every branch is thought or known to relate to the suitability of a given habitat for the given fish species and life stage (see Figure 8; from McMahon 1983). For example, percent cover may be represented by graphic V_{10} and percent pools represented by graphic V_{11} (Figure 9; from McMahon 1983).

V_{10} and V_{11} can contribute to the life requisites of cover and reproduction for a species. For any one habitat variable there may be more than one symbol, each representing a different life stage (i.e., a separate symbol representing a separate SI curve for adult, juvenile, and fry for any one variable name). The SI curves are then used to determine the value to assign to each variable symbol. The SIs are then aggregated to determine the HSI. This can be done in one of three ways:

1. The average value method (AVM) simply calculates the geometric mean: AVM = $(V_1 \text{ x } V_2 \text{ x } V_{i\ldots n})^{1/n}$.
2. The interactive limiting factor (ILF) method weights low SIs heavily which means all of the SI variables are considered equally important and if any receive a low SI value that will pull overall suitability down. It is calculated as: ILF = $(V_1 \text{ x } V_2 \text{ x } V_{i\ldots n})$.
3. Third is the Lowest SI (LSI) method in which HSI is assigned the lowest SI score. This approach assumes that the variable having the lowest SI will limit overall habitat suitability.

The third method is the approach used for all four of the anadromous salmonid HSI models used in the subsequent assessment. Thus, the lowest habitat variable SI score is considered to be the limiting factor for each life stage under consideration.

For the purposes of this evaluation, considering the self-sustaining and high-quality nature of the Bristol Bay salmon fishery, SI scores for each variable under consideration have been rated the highest possible (per Woody 2009a). For example, for the two SI variables described above, V_{10} (*percent pools during summer low flow period*) and V_{11} (*proportion of pools during summer low flow period that are 10 to 80 m^3* <u>or</u> *50 to 250 m^2 in size and have sufficient riparian canopy to provide shade*), both have been rated 1.0 (e.g., V_{10} – between 45% and 60%; V_{11} – greater than 75% - see Figure 9). Habitat variables for coho, Chinook, chum, and pink salmon species and descriptions of 'optimal levels' expected for each variable for optimal support of salmonids are provided in Appendix B, Table B-1.

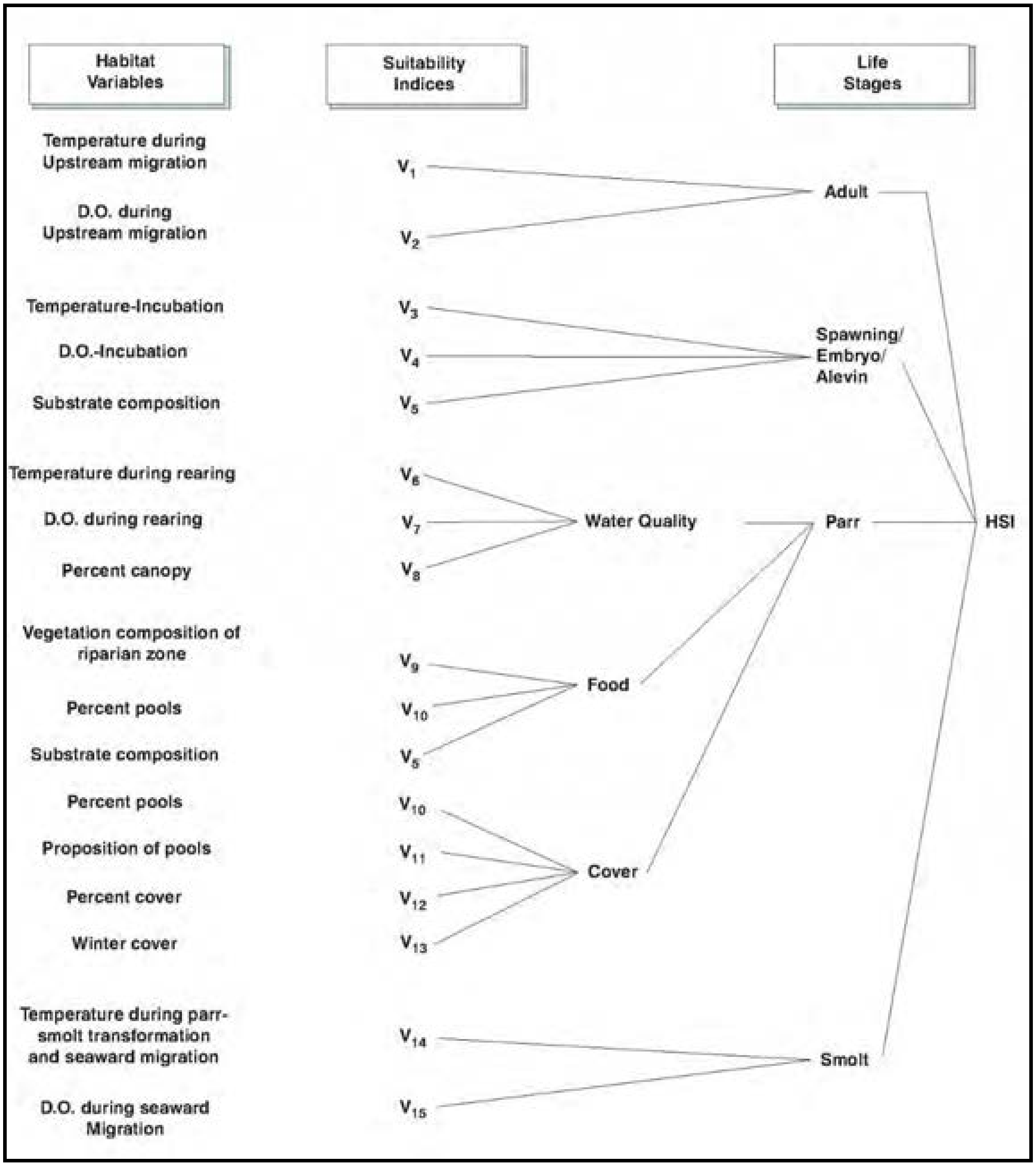


Figure 8. Diagram showing habitat variables included in the HSI model for coho salmon and the aggregation of the corresponding suitability indices (SI's) into an HSI; HSI = the lowest of the fifteen suitability index ratings (from McMahon 1983)

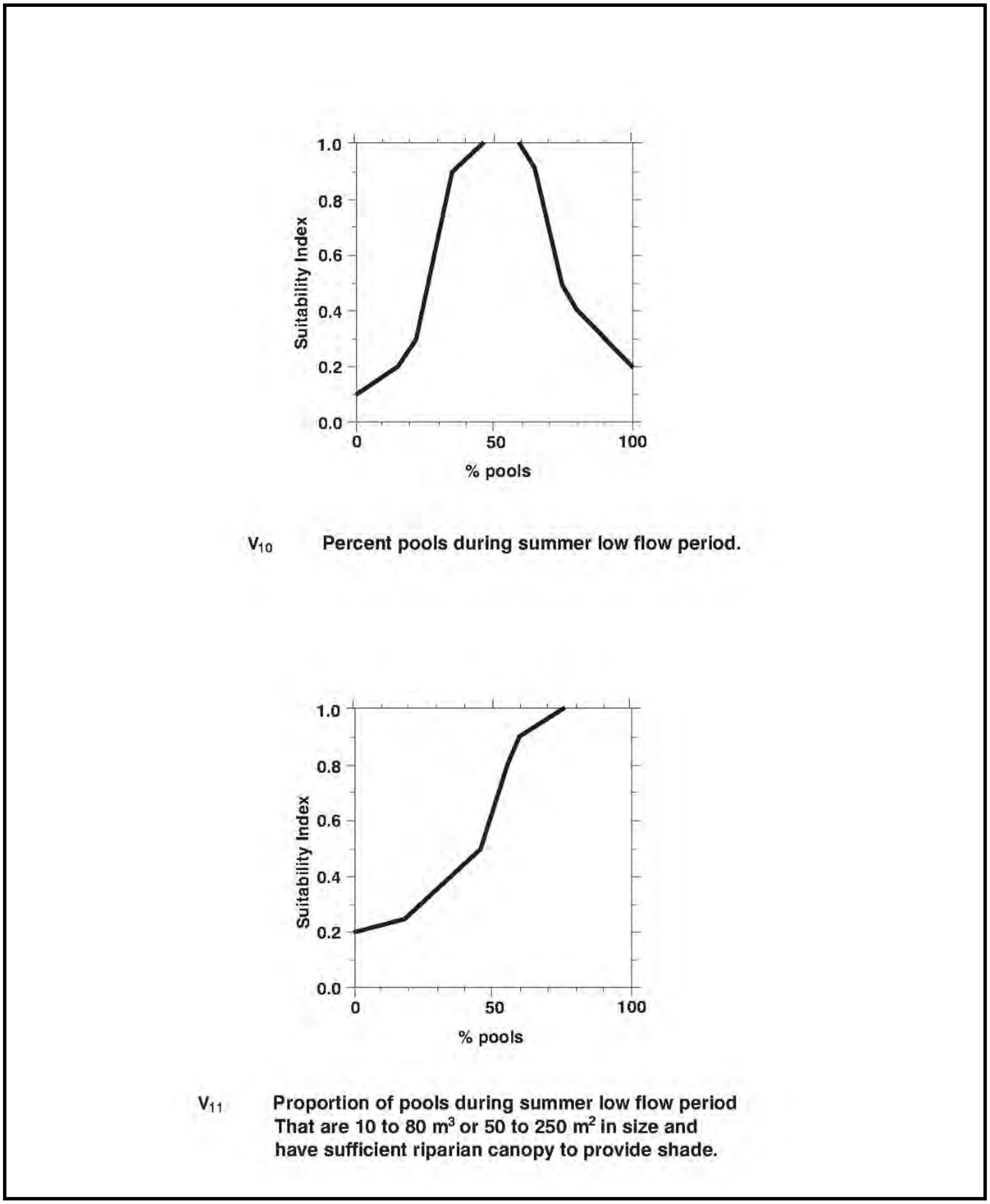


Figure 9. Example Habitat Suitability Indices for Coho Salmon (from McMahon 1983)

3.1.4.1 Impact Determination

The River Continuum Concept describes a downstream gradient of organisms that follow a downstream gradient of physical characteristics (Vannote *et al*. 1980). Although tributaries may not share the same physical characteristics as the mainstem, their contribution of organics, nutrients, and essential minerals, and their mitigation of effects for many physicochemical characteristics, results in a network that provides a somewhat smooth longitudinal gradient, and thus the essential life requisites for supporting the biological community. Kiffney *et al*. (2006) found that wood abundance and volume, variability in median substrate size (i.e., substrate heterogeneity), concentrations of nitrogen and phosphorus in water, algal biomass, and abundance of consumers and predators peaked with a higher frequency at or downstream of tributary junctions. As such, changes to a natural riverine complex will inevitably result in effects that could negatively impact fish (and other biota) viability. Loss of stream habitat is widely acknowledged as the single biggest cause of declines of anadromous salmonids in general [in the Pacific Northwest], and of coho salmon in particular (Nehlson *et al*. 1991; Reeves and Sedell 1992). For this ERA, the River Continuum Concept is of relevance as portions of the drainage basins that supply inputs into the three mainstem channels would be eliminated for, or due, to proposed mine operations.

Flow

Each new generation of salmon develops within a freshwater stream system and the success of the generation is dependent upon appropriate stream flow. Flow rates affect all life stages, including the upstream migration of adults, survival of eggs, the emergence and viability of fry, and timing of smolt out-migration. To reach spawning grounds, adults require access to the stream system and sufficient water flow to successfully navigate passage impediments while migrating upstream. The following sections discuss the impacts predicted from flow reduction within each of the three mainstems (NFK, SFK and UTC) potentially affected by the proposed mine.

Based on GIS evaluation of NDM information, approximately 33 square miles of drainage area within the NFK, SFK, and UTC watersheds would be eliminated for mine purposes (e.g., water extraction, tailings ponds, pits). This includes approximately 68 miles of stream channel. Based on this analysis, drainage characteristics within the most upper subbasin of the NFK watershed (Subbasin A; Figure 6) would not be affected. Down to NFK 1, this should result in normal runoff characteristics and flow within the NFK, similar to pre-project levels. However, the potential for continued use of first and second order streams above NFK 1 as rearing areas for coho (Woody 2009a) would be threatened by post-operation flow levels (a result of water depletion from Subbasin F that provides inputs to the NFK mainstem above NFK 2). Flow reduction of approximately 21% within the mainstem at NFK 2 could feasibly result in migrational restrictions to coho fry moving to upstream rearing habitat during summer low flow periods. Above NFK 2, approximately 3.5 miles of Alaska Department of Fish and Game- (ADFG) designated anadromous streams would be removed for water extraction or mine facilities (Table 8). The more aggravated summer low flow conditions down-gradient of NFK River Mile (RM) 12-13 (see Figure 6) would result in reduced pools and backwaters for supporting juveniles and, thus, more competition for resources including food, shelter, and cover. Sandercock (1991) found that low flows likely decreased wetted areas, increased stranding in isolated pools, and increased predation vulnerability, thereby reducing overall salmonid productivity. Nadeau and Lyons (1987) found that extremely low flow during the incubation or inter-gravel phase of salmonid life is one of the most limiting flow-related factors to salmon production in southeast Alaska. Host and Neal (2004) reported Jordan Creek (Auke Bay) smolt migration counts in the spring suggested that juvenile coho salmon smolt productivity may have been linked with streamflow. Based on communication with ADFG, they reported that following a relatively wet and warm winter and spring, in which flows did not fall below 2 fps, Spring 2001 counts were 26,600. This compared to a cold and dry late-winter and 10 days of zero stream flow in early-Spring 2002, when only 7,860 migrating smolts were counted.

Table 8. Designated Anadromous Waters Removed by the Project

Designation	NFK		SFK		UTC	
	Current	To be Removed	Current	To be Removed	Current	To be Removed
ADFG Anadromous Waters (miles)[1]	53.5	3.5	55.5	5.5	44.9	5.1

1 = based on GIS interpretation

Flow reductions in the SFK would be more dramatic with a 68% reduction in flow at SFK 2 (~5 miles below the tailings ponds dam) and a 24% reduction near RM 21.5 (SFK 3), which is approximately 14 miles below the dam (see Figure 6 and Table 5). Impacts in UTC would be most pronounced at UTC 1 and 2, where 80% and 20% flow reductions, respectively, would be expected (see Table 6). More than 5 miles of ADFG-designated anadromous waters would be eliminated in each of the SFK and UTC watersheds (see Table 8). General impact discussions below are considered similar for all three streams under investigation. Specific issues for individual streams are discussed, as appropriate.

Low flows can limit adult salmon entry into streams or movement up river to stage for spawning. Chum have a relatively restricted seasonal period (i.e., approximately one month in Alaska) and must arrive in good health for successful spawning (Hale *et al*. 1985). Along with effects related to increased temperatures (see below), low flow conditions can also be a barrier. In portions of Alaska, where the streams without a snow pack generally have low reservoir capacity and flow depends heavily on rainfall, migrating chum often have difficulty moving upstream during dry years (Hale *et al*. 1985). It has been observed that chum (Hale *et al*. 1985) and Chinook (personal observation, Woody 2009a) travel upstream in shallow riffles with the upper part of their bodies above water. Low flow conditions, along with other associated reductions in water quality conditions (e.g., reduced DO, higher water temperatures), would result in stress to individuals and potential mortality. This phenomenon was documented in southeastern Alaska when high temperatures and low DO during low flow conditions resulted in mortalities of up to 30,000 pre-spawn pink and chum salmon (Murphy 1985). Several suitability indices (SI) for coho (V_2, V_4), Chinook (V_9), chum (V_1, V_2), and pink (V_5) are directed at impacts associated with low flow conditions during spawning (Raleigh *et al*. 1986, Hale *et al.* 1985, McMahon 1983, Raleigh and Nelson, 1985 – see Appendix B).

Flow reduction of approximately 21% at NFK 2, and to some extent at NFK 3 (17%), would result in stressful conditions for coho and other salmon moving to upstream spawning habitat, especially during years of reduced rainfall. This constitutes a total stream reach of approximately 5 miles (as measured from NFK 1). The 5-mile segment above SFK 2, where there would be a 68% reduction in flow, would dramatically affect upstream migration during years with low precipitation. Similar impacts would extend for another 9 miles where flows would be reduced by up to 24%. It is expected that, during years of low rainfall, flow above UTC 1 (~2.5 miles) would be non-existent, and limited at NFK 2 (~ a 4.5-mile segment), where 20% flow reductions would be expected.

Permanent reduced flows result in smaller stream channel widths and, thus, less cover during re-establishment of riparian vegetation along stream channels. All of these factors would affect benthic community productivity, resulting in fewer food resources. Pools that remain within affected river reaches would experience increased temperatures, resulting in stress to seasonally remaining fry/juveniles. It is expected that changes to temperatures from flow reduction, especially in NFK (reaches above NFK 2 and 3), SFK (reaches above SFK 2 and 3), and UTC (reach above UTC 2), would negatively affect all salmonid life stages in those reaches.

The NMFS and USFWS (2004) have noted the following concerning salmon parr and stream flows:

> *"Parr growth and survival during the summer are positively correlated with various flow rates, demonstrating that the low flows limit parr populations. Population reductions during low flows probably occur because of reduction in habitat quantity and quality and possibly reduced foraging opportunities* (Frenette *et al*. 1984)*. This reduction in habitat quantity and quality can cause salmon parr to shift to sub-optimal habitat, reducing foraging opportunities and, thereby, impairing growth and survival."*

It was noted that standing crops of juveniles might vary with flow but that the variable juvenile and adult life histories of salmon tended to smooth out population swings caused by periodic low flow years. They noted, however, that

> *"If annual summer flows are <u>constantly</u>* {emphasis added} *low..., the population size will be constrained by the available habitat at those flows and will not vary as greatly as when flows were unregulated. The carrying capacity of the river to produce juveniles will be reduced for the long-term, not just for an occasional year."*

Frenette *et al.* (1984) also describe relationships of flow and survival in the alevin and fry stages:

> *"The timing of hatching and emergence, relative to spring runoff, affects egg-to-fry mortality and survival. Low flows in the 30 days prior to spring runoff may cause high mortality among pre-emergent alevins."*

Studies indicate that the amount of suitable winter habitat can be a major factor limiting coho production (Chapman 1966, Mason 1976, Chapman and Knudsen 1980). Winter cover is critical because as water temperatures drop swimming abilities of coho decreases. The extremely low winter flows that would be expected after mine development would reduce off-channel, back-water areas most significantly above SFK 2, where flow would be reduced by 68%. During winter periods, use of non-mainstem areas would not be available as rearing habitat.

Flow velocity is an important variable for spawning and embryo incubation: first, it maintains that substrate materials move downstream during redd construction; second, it carries oxygen to developing embryos; and last, it facilitates the removal of metabolic wastes from the redd. Redd construction occurs in swift, shallow areas at the head of riffles (Burner 1951, Briggs 1953). For coho, Smith (1973) noted preferred sites in riffle areas with velocities of 0.69 to 2.3 fps and minimum depths of ≥ 15 cm. Optimum stream velocities are provided for Chinook (V_9; 30-90 centimeters per second [cm/s] [1-3 fps] in Raleigh and Miller 1986) and pink (V_9; 40 cm/s [1.5 fps] in Raleigh and Nelson 1985) salmon (see Appendix B, Table B-1).

Woody (2009b) measured headwater stream velocities during their August–September sampling ranging from 0.18 to 0.94 fps (see Table 7), with an average of 0.5 fps. The only available data on mainstem velocities were from three USGS stations at the distal extent of each of the watersheds. It is expected that post-operation velocities would decrease in relation to flow reduction, but channel morphometry within the most impacted portions of the streams would dictate this relationship. As stated above, velocity would be affected to the greatest extent in those areas where flow was significantly reduced (e.g., NFK down to NFK 3; SFK reach above SFK 3; UTC reach above UTC 2), but the affected area would extend further downstream during years with low fall precipitation.

To estimate the effects on velocity from flow reductions, a relationship was established between observed discharge and velocity within the mainstems. Regressions were developed for USGS discharge/velocity data for three periods available (May, July, and September). Knighton (1998) showed that, for a particular gaging station, the relationship between mean velocity and discharge is linear in a log-log plot. That is, $V=kQ^m$, where V is velocity, Q is discharge, and *k* and *m* are fitting constants. Knighton (1998) presents a set of studies that show that *m* is typically in the range 0.3 to 0.5. In Figures 10, 11, and 12, *m* is the slope of the line for NFK, SFK, and UTC (e.g., 0.3707, 0.7056, and 0.3169, respectively). Table 9 provides measured and modeled velocity values based on regression analyses for the three mainstem stations. Velocities for various discharge magnitudes using regression predictions for each stream are provided in Table 10.

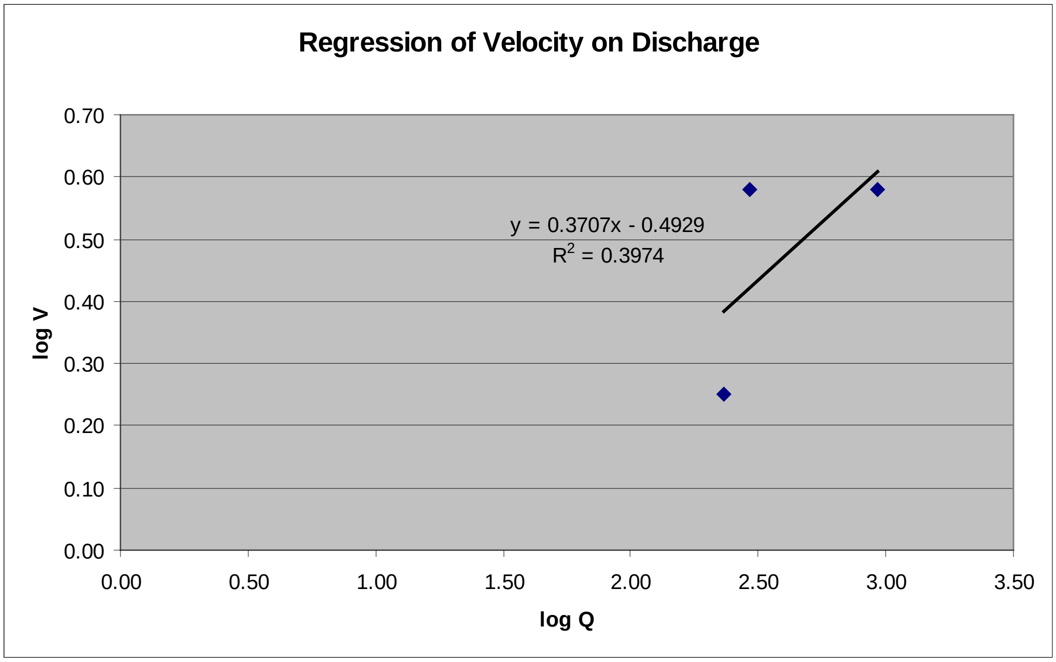


Figure 10. Results of Regression Analysis for Velocity to Discharge at the USGS Station 15302200 on the North Fork Koktuli

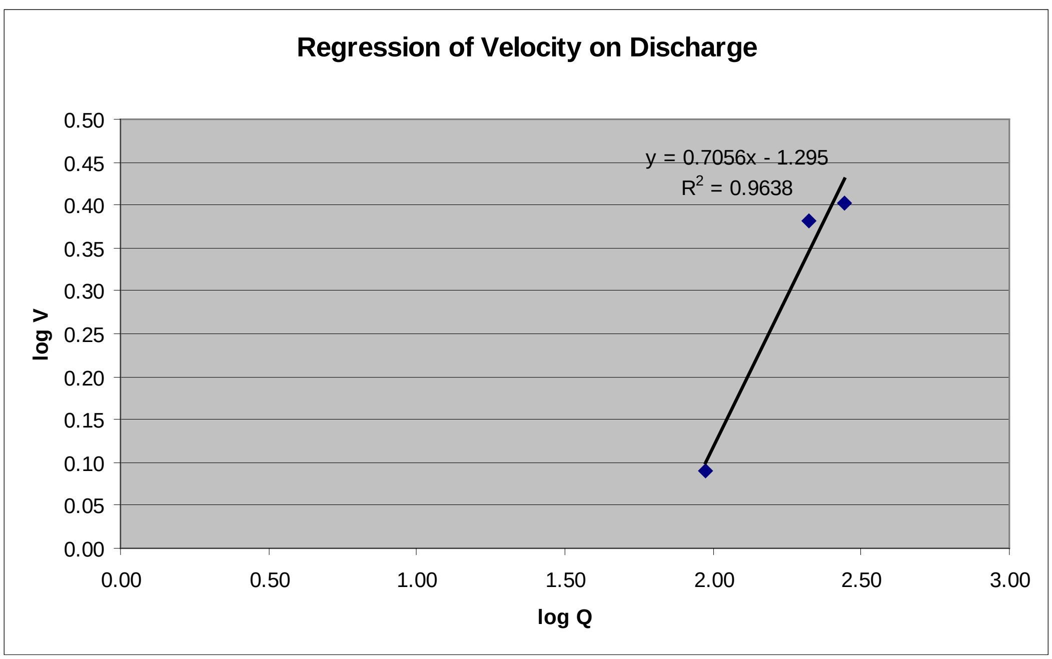


Figure 11. Results of Regression Analysis for Velocity to Discharge at the USGS Station 15302250 on the South Fork Koktuli

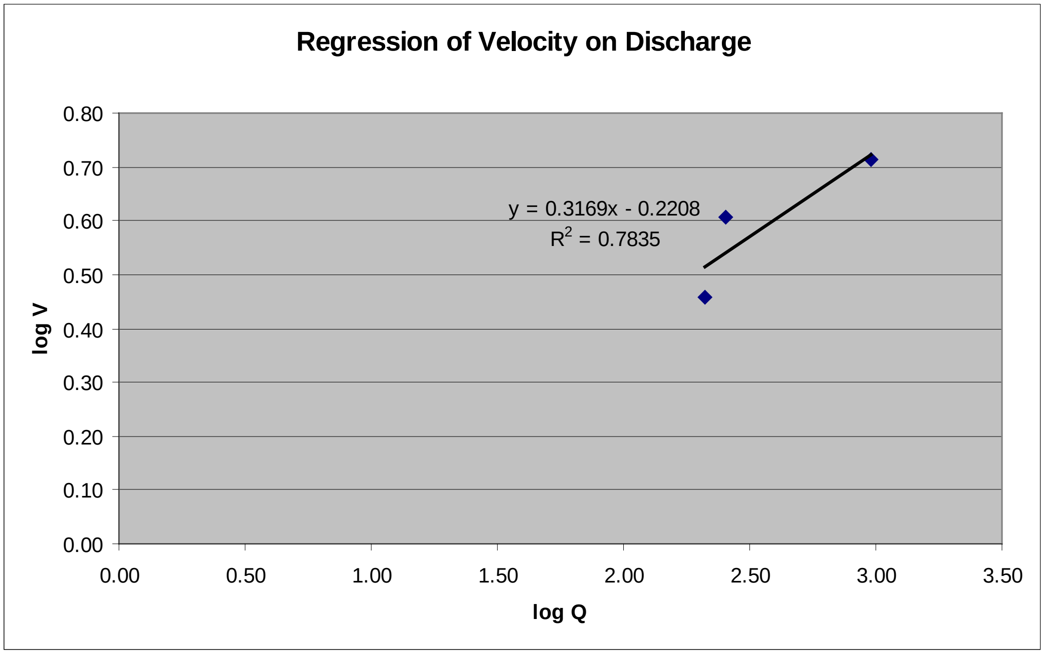


Figure 12. Results of Regression Analysis for Velocity to Discharge at the USGS Station 15300250 on the Upper Talarik Creek

Table 9. Results of Regression Model for Predicting Velocity in Impacted Streams

Stream	Discharge (cfs)	Measured Velocity (fps)	Modeled Velocity (fps)	Log Q	Log V
NFK	231	1.78	2.80	2.36	0.25
	291	3.8	3.06	2.46	0.58
	929	3.81	4.86	2.97	0.58
SFK	94.7	1.23	1.26	1.98	0.09
	211	2.41	2.21	2.32	0.38
	279	2.53	2.70	2.45	0.40
UTC	209	2.88	3.27	2.32	0.46
	255	4.03	3.48	2.41	0.61
	954	5.19	5.29	2.98	0.72

Table 10. Predicted Velocity in Impacted Streams during Low Flow Conditions

Discharge (cfs)	Model Predicted Velocities (fps)		
	NFK	SFK	UTC
10	0.80	**0.26**	1.25
20	**1.06**	0.42	1.55
50	1.52	0.80	**2.08**
100	2.00	1.31	2.59
200	2.64	2.13	3.22
500	3.80	4.07	4.31

Bold values represent regression-predicted post-development velocities in mainstems during low flow (Jan-Mar) periods.

Although channel configurations (and possibly their effects on velocity) may differ from downstream portions of the streams, it is predicted that after development, velocities during the critical spawning/embryo development period (January–March) at NFK 2 (~20 cfs), SFK 2 (~10 cfs), and UTC 2 (~45 cfs) would be less than optimum when compared to information provided in Smith (1973), Raleigh and Miller (1986), and Raleigh and Nelson (1985). Velocity reductions would be most prominent during the period directly after mine development due to the channel morphometry exhibiting pre-development characteristics. In time, channelization and deposition would most likely result in a more stable streambed and slight increase in velocities. Based on information presented previously, and as noted in Appendix B, Table B-1, impacts from reduced velocities would be greatest within the SFK (see Table 10), as predicted velocities during low-flow conditions (i.e., winter period; Jan-Mar) could be <0.3 fps. Again, for coho, Smith (1973) noted preferred sites in riffle areas had velocities of 0.69 to 2.3 fps, with optimum velocities for Chinook near 1.5 fps. The low flow conditions in portions of NFK (e.g., predicted velocity ~1.06 fps [see Table 10]) would also result in stressful conditions to both of these species especially in years with limited rainfall. Specific information on spawning locations within the upper reaches of the three rivers is unknown at this time.

Substrate/Dissolved Oxygen

Flow reduction would also affect substrate composition in riffle/run areas within the affected mainstem segments of the streams. Optimum spawning habitat noted by USFWS for coho (McMahon 1983) is composed of greater than 50% gravel and rubble or less than 5% fines (particle size<6 mm; V_5); Chinook (Raleigh *et al*. 1986) prefer ≤5% fines (silts and sand >30 mm; V_{10}); chum (Hale *et al*. 1985) <10% fines (particle size< 6mm; V_5); and pink (Raleigh and Nelson 1985) prefer a substrate particle size range of 1 to 5 cm (V_3).

Woody (2009b) found, generally, that the majority of sediments in headwater streams was composed of particles ranging from 2 to 64 mm. This indicates that fined-grained particles are present in headwater streams and most likely transported continually to downstream portions of the watershed. It is presumed that, because this is a natural system, fine-grained particles travel downstream during high flow events and current low flow conditions are still high enough to limit deposition in swift, shallow, riffle areas. However, low flow conditions expected in perpetuity after mine development may reduce overall particle transport mechanisms, potentially resulting in increased embedded conditions within riffle spawning habitat with each successive season, ultimately reducing the quality and quantity of spawning habitat. Generally with salmon, and as the models for coho and chum show (Figure 13; A [McMahon 1983] and B [Hale *et al.* 1985], respectively), SI values drop sharply as fine particle percentages in substrates increase above 5%.

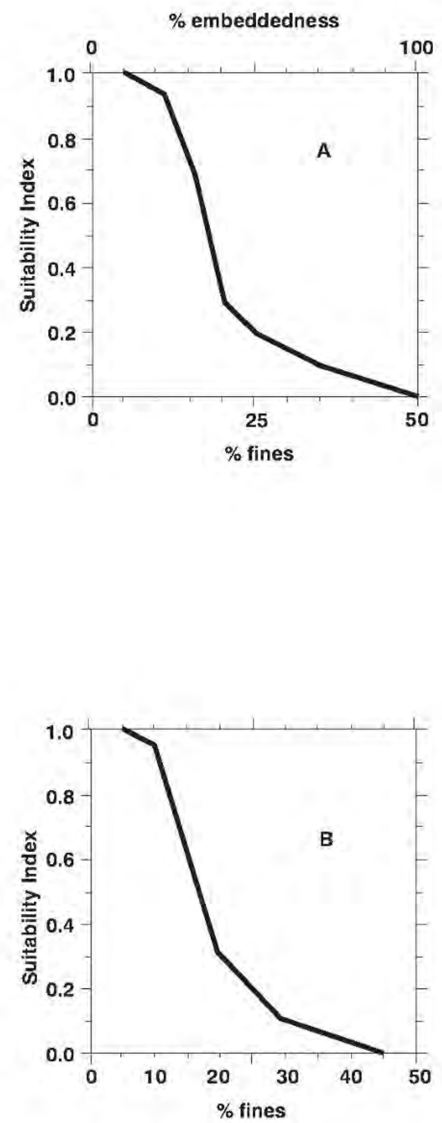


Figure 13. Substrate Composition Suitability Indices for Coho (A; V_5) and Chum (B; V_5) Salmon (from McMahon 1983; Hale et al. 1985)

As embedded conditions develop, the quality of redds would be reduced and embryonic development and fry emergence would be negatively affected. As discussed previously, survival and emergence of embryos and alevins is greatly influenced by DO supply within the redd. DO availability in redds relates to gravel permeability, water velocity, and instream DO concentrations. When any of these factors, alone or in combination, reduces intragravel DO supply below saturation, hypoxial stress occurs. Hypoxial stress results in delayed hatching and emergence, smaller size of emerging fry, and increased incidence of developmental abnormalities (Alderice *et al*. 1958, Coble 1961, Silver *et al.* 1963, Shumway *et al.* 1964, Mason 1976).

Low flow conditions expected in upper reaches of all three streams could result in increased down stream sediment deposition and reduced survival and emergence of fry. The area between NFK 1 and NFK 3 would have the highest probability for effects, but with an 11% overall flow reduction as far down stream as NFK 4 (~13 miles), impacts could be expected in this area, also. Again, flow reductions in the SFK would be more dramatic, with all headwater streams being eliminated resulting in a 68% reduction in flow at SFK 2 (~5 miles below the tailings ponds spillway), a 24% reduction near RM 21.5 (~14 miles below the dam) (Figure 6 and Table 5), and a 13% reduction all the way to RM 36, which is near the end of the watershed.

Parr require an abundance of food to sustain fast growth rates and have successful freshwater rearing during development into smolts. Reduced food availability can result in larger and fewer territories per unit area and increased emigration of resident fry, ultimately resulting in slower growth rates for remaining fish. Gravel-rubble substrate composition corresponds to a high production of aquatic invertebrates and, therefore, is excellent in providing food. Other substrates produce decreasing amounts of invertebrates in this order: rubble > bedrock > gravel > sand (Pennak and Van Gerpen 1947). This indicates that higher percent embeddedness or higher percentages of fines would ultimately result in lower invertebrate production and negative indirect effects to salmon and resident fish. Suitability indices noted above in Figure 13 can be applied for determining optimum (or less) food conditions for parr viability – e.g., a greater percentage of fines results in a lower suitability index. Again, areas noted in the previous paragraph for the NFK, SFK and UTC would also show effects from reduced flow and increased embeddedness.

Temperature

It is widely accepted that tributary inflows to riverine systems contribute colder water and help regulate riverine temperatures as a result of groundwater influx. Poole *et al*. (2001) found that stream temperatures are influenced by several external factors, one of which was tributary temperature. Malcolm *et al*. (2008) found that riparian woodland stream inputs into the mainstem of the Girnoc Burn had the most obvious effects during the spring, with a maximum in summer, before decreasing once again in autumn. This suggests that flow reductions would be the most critical during the summer when flow is already reduced and temperatures are highest.

Stream temperature is one of the primary controls on fish survival, growth, and reproduction. Temperature regulates salmonid metabolic function, determines rates of development, and motivates behavioral adjustments (Sullivan *et al*. 2000). Shrimpton and Blouw (2000) found that fish in streams with limited riparian habitat, and thus higher temperatures, had higher stress factors (e.g., lower concentrations of gill cortisol receptor) than fish in well-buffered streams. Higher tributary temperatures also resulted in greater diurnal temperature fluctuations in the mainstem river, which results in higher stress to fish. Temperature also affects the amount of DO in streams, a key limiting factor for fish survival, and can affect the amount of disease outbreaks. In addition to growth and survival, changes in stream temperature (and flow) have also been shown to have statistically significant effects on the timing of emigration of smolts, where increases in average spring water temperatures resulted in early emigration of juvenile Chinook salmon (Roper and Scarnecchia 1999).

McMahon (1983; see V_6 in Table B-1) suggests a maximum temperature during coho parr rearing of 9-13° C. Headwater stream temperatures in late August to early September near the proposed mine site ranged from 3.3 to 11.5° C, with a mean of 7.7° C (Woody 2009b); five of the 24 measured temperatures fell within this maximum SI range. Instream temperatures for mainstems (and downstream floodplain areas) were not available, but it is assumed that low flow July–August temperatures could be at or above those observed by Woody (2009b), and would increase after flow reduction. As a result, temperatures above NFK 3, SFK 3, and UTC 2 would likely increase, potentially falling above the optimum SI value during the low flow, warmest periods. Even though mortality may not occur, activation of latent infections would increase, ultimately impacting population health. Avoidance of areas with highest temperatures would most likely be the predominant effect, with areas of historically active rearing becoming depauperate.

In the face of climate change and warming summer stream temperatures, the biological benefits provided by the colder tributaries become even more crucial to protect as the mainstem is more likely to experience changes to annual summer maximum temperatures outside of the thermal tolerance of many aquatic species. The annual average temperature in Alaska has increased 3.5°F from 1949 to 2005. Temperatures have changed more in Alaska over the past 30 years than they have anywhere else on Earth: winters have warmed by a startling 5-6°F, compared with a global average of 1°F. According to Eaton and Scheller (1996), studies on climate warming effects on thermal habitat of fish species (including salmon) in the U.S. suggest that "habitat for cold and cool water fish would be reduced by ~50% and that this effect would be distributed through the existing range of these species". Bryant (2009) predicts that decreased summer stream flows and higher water temperatures, affecting juvenile coho growth and survival, are likely occur due to global warming. Also, higher temperatures during spawning and incubation may result in pink and chum early entry into the ocean when food resources are low (Bryant 2009).

Flow reductions can also have deleterious effects to egg/fry survival from reduced temperatures in winter. NMFS and USFWS (2004) note that "sources of egg mortality include de-watering, freezing, mechanical destruction (i.e., sedimentation) and predation." Baum (1997) noted that fewer than 10% of Atlantic salmon eggs survive to emerge as fry in Maine rivers.

Long-term exposure to temperatures below 4.4° C reduces survival of chum salmon embryos (Schroder 1973, Koski 1975, ADFG 1983). SI models for Chinook (V_7), chum (V_3), and pink (V_7) have been developed to evaluate this effect. Similar to SI models presented previously for sediment, there is a sharp decline in habitat quality as temperatures fall below the optimum range. The low flow conditions from January through March would increase the probability for reduced water temperatures (compared to present conditions) in upper reaches of the NFK, SFK, and UTC. It is predicted that this effect would be most pronounced in the 5-mile reach above SFK 2 where flow conditions during winter months would be less than 10 cfs. With flow this low, it is most likely that waters would freeze completely and stream flow would cease, resulting in reduced water interchange and high mortality to eggs that have been deposited. Bustard (1983) listed stranding and freezing as one of three major factors contributing to overwinter losses of juvenile Chinook and coho caused by too-low, late fall-winter flows.

3.1.5 Road Construction – Obstruction of Fish Passage and Turbidity

3.1.5.1 Stressor Description

Movement is an essential mechanism by which mobile animals acquire the resources necessary for successful completion of their life-cycles (Greenwood and Swingland 1983, Dingle 1996). Salmonids have coexisted with the presences of naturally occurring barriers to upstream movement in headwater streams for a very long time (Hoffman and Dunham 2007). However, human-placed movement barriers restrict or eliminate fish movement to upstream habitat and isolate or modify populations. To successfully negotiate a culvert, a fish must be able to enter the culvert, traverse the length of the barrel, exit the culvert, and proceed to an upstream resting area. Based on a review of current scientific literature, little is known about the capability of juvenile salmonids to access upstream habitat by overcoming barriers. Effects to salmon populations expected from road (or other) barriers include:

- Reduced ability to support declining upstream populations (Jackson 2003);
- Decreased ability to reach important headwater spawning and rearing sites (Wigington *et al.* 2006); and
- Upstream species richness is attenuated (Winston *et al*. 1991);

Culvert installations can significantly decrease the probability of fish movement between habitat patches (Schaefer *et al.* 2003). In the undisturbed case, fish are free to use the entire stream system as habitat. Road interruptions can result in stream discontinuity, and fragmented populations are forced to survive independently. Over a short time, smaller populations would be more sensitive to extirpation from chance events (Farhig and Merriam 1985), but over the long-term, lack of genetic diversity and natural disturbances also may cause extirpation, even in larger populations (Jackson 2003).

A culvert can become a barrier to fish passage if it creates conditions exceeding fishes' biological ability (Hotchkiss and Frei 2007). Obstructions to fish passage include excessive water velocities, drops at culvert inlets or outlets, physical barriers such as weirs, baffles, or debris caught in the culvert barrel, excessive turbulence caused by inlet contraction, and low flows that provide too little depth for fish to swim. Hotchkiss and Frei (2007) provide details on hydraulic mechanisms associated with each obstruction noted above, along with a general discussion on the physiological effects to fish during attempted passage. Also, conditions at or within a culvert may impede fish from entering or attempting passage, even when passage is possible. These conditions are termed 'behavioral barriers' and include long culverts, darkness, confined culverts, and shallow depths. Bates *et al.* (2003) and Robison *et al.* (1999) provide information on energy expenditures related to long culverts. Behavioral differences in light versus dark passage suggest that darkness may dissuade certain fish from entering a structure (Welton *et al*. 2002, Kemp *et al.* 2006, Stuart 1962).

According to the ADFG Sport Fish Division, "Poorly designed or inadequately maintained culverts can block or impede fish access to upstream spawning and rearing habitat." The connectivity of a diverse suite of fish habitats is integral to supporting the abundance of fish species and their life stages found in Alaska's fresh water habitats. Tributary streams, lakes, off-channel habitats, backwater areas, small ponds, and sloughs all provide critical fish habitat. Ensuring that these habitat components remain connected allowing for the free migration of spawning adults and rearing juvenile fish is critical for maintaining healthy fish populations. However, a variety of natural and man-made barriers (particularly culverts) may limit connectivity of habitats and can measurably reduce fish production in some watersheds" (ADFG 2007d).

Secondary physical effects at culverts include increased turbidity and sedimentation within downstream reaches from unconsolidated road material runoff. Water quality attributes such as total suspended solids (TSS) and turbidity are recognized as physical impacts that can be of concern in salmon rivers (Dill *et al*. 2002). The State of Alaska has established a water quality standard for turbidity for protection of 'water supply (aquaculture) and growth and propagation of fish, shellfish, and other aquatic life and wildlife.' This standard is 25 nephlometric turbidity units (NTU) above natural condition level (18 ACC 70, 2003). Bash *et al.* (2001) suggest that standards should also be based on evaluations of total suspended solids (TSS) levels that consider physiology, behavioral, and habitat effects. Therefore an evaluation of TSS should accompany NTU measurements.

3.1.5.2 Impact Methodology

Assessment of potential impacts from development of the proposed access road ranged from semi-quantitative, based on GIS evaluation, to somewhat qualitative in nature, as prediction of effects were derived from studies associated with other similar sites and conditions. To evaluate the potential short- and long-term impacts to salmonids from culvert placement and potential downstream elevated turbidities, GIS data were used to evaluate the number of crossings expected along the proposed corridor, along with those identified as ADFG-designated anadromous waters. Next, stream lengths for ADFG anadromous waters identified upstream of proposed crossing were enumerated. Habitat was presumed to be excellent for the streams under consideration based on their status as designated anadromous streams. Table 11 provides information on the upstream segment length for the anadromous streams crossed by the proposed road. Estimation of risk to salmon populations and habitat was evaluated based on; 1) potential short and long term effects from culvert installation; and 2) from road construction.

Table 11. Upstream Designated Anadromous Waters Affected by Road Crossings

Stream Name	Length (mi)
Tributary to UTC	0.54
Tributary to UTC	2.15
Newhalen River	14.6
Unnamed Creek	1.34
Eagle Bay Creek	2.52
Unnamed Creek	0.39
Unnamed Creek	2.27
Unnamed Creek	1.71
Unnamed Creek	0.44
Chokok Creek	14.85
Canyon Creek	1.61
Knutson Creek	0.35
Pile River	9.71
Iliamna River	22.66
Total	**75.14**

This analysis was based on historical data on habitat effects resulting from road crossings in general and does not reflect the proposed mine's specific construction techniques or mitigation efforts. With consideration of the possible effects to fisheries from road and culvert crossings, mine developers may reduce or eliminate potential impacts discussed in this assessment to reasonable levels.

3.1.5.3 Impact Determination

Culverts

The proposed access road and pipelines would cross at least 89 streams (Figure 14), with 14 of these officially designated as ADFG anadromous waters (see Table 11). The remaining streams may also be anadromous, but they have either not yet been surveyed or the results of surveys are not yet in the official record. These anadromous streams provide spawning, rearing and migratory habitat for salmon species of concern. As discussed, roads impact streams when inadequately designed, poorly installed, or inadequately maintained stream crossing structures (usually culverts) block fish passage to upstream fish habitat.

Figure 14. Streams designated in the Anadromous Waters Catalog crossed by the proposed mine access road*

***Additional anadromous streams are likely but have either not yet been surveyed for the AWC or results of surveys are not yet in the official record.**

Studies of culverts by the United States Forest Service (USFS) and others found that improperly installed and maintained stream crossing structures have blocked access to thousands of miles of formerly productive salmon and high value resident fish habitat in the Pacific Northwest and Alaska (Kemset *et al*. 1999). Secondly, construction activities at stream channels can result in short-term and long-term increases in turbidity and sedimentation, with both direct and indirect impacts on salmon and resident biota.

Construction and installation of culverts at the 14 streams identified as salmon waters in the AWC have the potential to affect long-term viability of populations. Approximately 75 miles of anadromous waters upstream of the proposed road would be at risk, but could directly affect a larger percentage of the streams' salmonid populations through over-stressed resources in below-culvert segments if fish are unable to move upstream to preferred spawning habitat.

Although fish passage guidelines exist for the installation of culverts, many of the culverts installed in the proposed access road may eventually end up as barriers to adult and juvenile fish migration. Unanticipated floods can erode stream channels, perch culverts, and block upstream migration. Incorrectly installed or poorly maintained culverts eventually become fish passage blockages. It is estimated that up to 50% of the culverts on public road systems may impede fish passage over time (Albert and Weiss, In review). Based on this estimate, it is possible that, over time, culverts on the proposed road may block access to many miles of rearing and/or spawning habitat and could reduce fish production annually.

Hauser (2008) provided examples of fish passage problems associated with inadequate design, installation, or maintenance of stream crossings, particularly culverts (Table 12). Moore *et al.* (1999) provide a bibliography of 96 annotated citations on culvert design for fish passage, risk analysis, and fish swimming ability.

Table 12. Studies Documenting Effects to Salmonid Populations from Culverts

Study Area	Passage Problems	Source
Labrador, Canada	53% with poor design or installation	Gibson *et al*., 2005
Tongass National Forest	66% of culverts across salmon streams and 85% of culverts across trout streams were considered inadequate for fish passage	Flanders and Cariello, 2000
Mat-Su Valley, AK	More than 44% of 130 culverts were deemed inadequate for fish passage; 10% were deemed adequate	Albert and Weiss, In review
Kenai Peninsula, AK	Results indicated that 78% of 97 culverts were deemed inadequate for fish passage; 9%were deemed adequate	Rich, In review
Near Tyonek, AK	Results indicated that 83% of 29 culverts were deemed inadequate for fish passage; 3% were deemed adequate	Rich, In review

Table 12. Studies Documenting Effects to Salmonid Populations from Culverts

Study Area	Passage Problems	Source
Western Montana	76 to 85% of culverts were velocity barriers depending on streamflow and fish life stage	Gresh *et al*., 2000
California, Washington, Oregon	Current salmon biomass in streams is 3 to 4% of historic biomass; much habitat loss is due, in part, to obstructed fish	Hilborn *et al*. 2003

Source: Hauser 2008

It is expected that the most serious and long-term impacts to local salmon populations would result at medium- to small-sized tributaries, such as most of the unnamed creeks, Chokok Creek, and Pile River, where culverts are expected to be installed. It is assumed that bridges or more sophisticated culverts would be required over the larger rivers such as the Newhalen and Iliamna. Over 37 miles (see Table 11) of upstream anadromous habitat could be totally eliminated or significantly affected for use by salmon as both spawning and rearing habitat in these small streams (e.g., Chokok Creek, Pile River, Canyon Creek, Eagle Bay Creek and most unnamed creeks). This would virtually eliminate, or substantially reduce, upper portions of these medium and smaller streams as viable habitat. As an example of the impacts that could be expected, Endou *et al*. (2006) reported that artificial barriers, including culverts and bridge bases, resulted in habitat fragmentation for salmonids (char and masu salmon) in the Fujigawa Basin, Japan. They surveyed 29 streams containing 356 artificial barriers and found that some isolated populations had been locally extirpated, even though both species had occurred throughout the headwaters during the 1970s. This is important for most of the smaller streams crossed by the proposed access road, because Endou *et al*. (2006) found that increased disappearance correlated with decreasing watershed areas (e.g., habitat size). Their model predicted that a minimum watershed area (0.39 mi^2 to 0.85 mi^2) was necessary for maintaining a population, suggesting that the probability of extirpation is highest if artificial barriers are constructed in upstream (or in already size-limited) portions of these smaller-sized watersheds. Although watershed sizes for streams crossed by the proposed access road have not yet been determined, the 'future' for fish populations in several of the unnamed streams from culvert installation could be at risk.

Based on evidence from Alaska and elsewhere (Table 12), risks to salmon populations from culvert placement during road construction for the mine are reasonably likely over the long term. As an example of the long-term implications from placement of culverts, the status of the Copper River, AK was reviewed (Copper River Knowledge System [CRKS] 2009). Unpublished ADFG data indicates that 244 culverts occur within the Copper River watershed. Each site may have more than one culvert (e.g., if two culverts are sitting side by side). According to ADFG's inventory, 64% of the culverts blocked the passage of fish, 32% of the culverts may not pass fish, and only 4% of the culverts provided adequate passage for fish (Table 13). Similar conditions were documented in other watersheds, with most exhibiting less than 'passable' conditions. The highest percentage of passable culverts (from studies evaluated by this ERA) occurred in the Tazlina-Nelchina watershed where a relatively poor rate of only 18% of the 11 culverts met this criterion. Of the watersheds investigated, the combined *Not Passable* to *May Not be Passable* constituted 75% to 100%.

Table 13. Watersheds Exhibiting Limited Passability as a Result of Culverts

Area	Total # of Culverts	Not Passable		May Not Be Passable		Passable		Uncategorized	
		#	%	#	%	#	%	#	%
Central Copper	24	19	79	3	13	0	0	2	8
Copper River Delta	115	59	51	39	34	4	3	13	11
Gulkana	24	12	50	7	29	1	4	4	17
Metnasta Chistochina	20	7	35	8	40	0	0	5	25
Mount Sanford	3	3	100	0	0	0	0	0	0
Tazlina Nelchina	11	6	55	3	27	2	18	0	0
Tonsina	41	33	80	4	10	2	5	2	5
Wrangell	6	0	0	5	83	0	0	1	17
Totals	244	139	57	69	28	9	4	27	11

Source: http://www.inforain.org/copperriver/content/pages/background/assessment_2.htm

Finally, Warren (1998) provided detailed analysis of the effects of road crossing methods to fish movement in small streams. He used mark-recapture techniques to examine the effects of four types of road crossings (culvert, slab, open-box, and ford crossing) on fish movement during spring base flows and summer low flows in small natural streams of the Ouachita Mountains, west-central Arkansas. For 21 fish species in seven families, he detected no seasonal or directional bias in fish movement through any crossing type or the natural reaches. Overall fish movement was an order of magnitude lower through culverts than through other crossings or natural reaches, except no movement was detected through the slab crossing. In contrast, open-box and ford crossings showed little difference from natural reaches in overall movement of fishes. Numbers of species that traversed crossings and movement also were reduced at culverts relative to ford and open-box crossings and natural reaches. Water velocity at crossings was inversely related to fish movement; culvert crossings consistently had the highest velocities and open-box crossings had the lowest.

Road Construction

Berman (1998) identified road construction as playing a significant role in altering instream physical and biological processes. The habitat complexity of a stream may be greatly compromised if there is a high sediment supply, where negative effects extend to spawning, egg and alevin survival, rearing habitat and adult holding habitat (Frissell 1992). Excess sediment can profoundly affect the productivity of a salmon stream (Cordone and Kelly 1961, McNeil and Ahnell 1964, McHenry *et al*. 1994). High turbidity impacts the feeding ability of juvenile salmon, although it may also provide them some cover from predation if it occurs during periods of smolt migration (Danie *et al*. 1984). Dill *et al.* (2002) cited Newcombe and Jensen (1996) when noting "that more than 6 days of exposure to TSS greater than 10 mg/L results in moderate stress for juvenile and adult salmonids. A single day of exposure to TSS in excess of 50 mg/L is also a moderate stress." Sigler *et al*. (1984) found that turbidities of 25 NTU or greater caused a reduction in juvenile salmonid growth.

The longer the duration of high turbidity the more damage is likely to fish and other aquatic organisms (Newcombe and MacDonald 1991). As noted by Arter (2004), "even moderate turbidity may affect a fish's ability to find food." Bash *et al*. (2001) provide a comprehensive review of the physiological and behavioral effects of elevated turbidities to salmon, along with expected impacts to habitat from this source.

Certain impacts of roads on habitats used by anadromous salmonids are widely recognized and well-understood: road-related landslides increase sediment loads and modify channel morphology, and culverts restrict access to parts of the channel network (Reid 1998). Other influences are less obvious, but may be even more pervasive. For example, road-related erosion significantly increases chronic turbidity levels in streams. Flow and turbidity data from Caspar Creek, California were used to model the potential influence of the presence and use of roads on cumulative duration curves for stream turbidity (USFS 2009). Results suggest that a proportional increase in fine-sediment production equivalent to that measured in coastal Washington (i.e., a 5.8-fold increase due to road-related erosion) could increase the average annual duration of turbidities greater than 100 NTU by a factor of 73 (i.e., from 0.5 day to 36.5 days). Published data suggest that feeding efficiency of juvenile coho salmon drops by 45% at a turbidity of 100 NTU.

Salmonid strategies for coping with high turbidity are likely to include use of off-channel, clean-water refugia and temporary holding at clean-water tributary mouths (USFS 2009). These coping strategies are partially defeated by the spatial distribution of roads: road runoff discharges into low-order channels that once would have provided clean inflows, and riparian roads restrict access to flood-plain and off-channel refugia. The temporal distribution of the high-turbidity inflows also decrease the effectiveness of coping strategies: turbidities are high even during low-magnitude events when flows may not be sufficient to allow access to refugia. The combined influences of increased turbidity and restricted opportunities for escape from the impact constitute a cumulative impact. Further, traffic-related turbidity is highest during the day, when salmonids feed, and traffic produces high turbidity even during small and moderate storm flows of autumn and spring, when water is warmer than during winter floods. Because salmonid metabolic rates are temperature-dependent, salmonids may be particularly sensitive to these unseasonal bouts of high turbidity. The type of road proposed is critical to downstream impacts.

The type of material forming the road bed, for example, is important. Lane and Sheridan (2002) conducted experiments at newly constructed, unsealed road stream crossing to determine the quantity and sources of sediment entering the stream. They continuously measured turbidity and estimates of TSS concentration upstream and downstream of a stream culvert over a five-month period. They found a statistically significant difference between up- and downstream measurements during baseflow conditions, with water quality good during non-rain periods. Rainfall events led to water quality decreases downstream of the crossing; overall, water quality was degraded during approximately 10% of the observations. Over the study period, sediment loads were ~3.5 times higher downstream of the culverts (compared to upstream loading), and it was estimated that approximately 2 to 3 tons of bedload material was added during crossing construction and from subsequent erosion. They predicted that this material would deposit on the cobble stream bed and most likely degrade aquatic ecosystem values.

As previously discussed, embedded conditions reduce the quality of redds and embryonic development and fry emergence, as survival and emergence of embryos and alevins is greatly influenced by the DO supply within the redd. Increased sedimentation will result in reduced intragravel DO supply, hypoxial stress, and ultimately delayed hatching and emergence, smaller size of emerging fry, and increased incidence of developmental abnormalities (Alderice *et al*. 1958, Coble 1961, Silver *et al*. 1963, Shumway *et al*. 1964, Mason 1976).

Turbidity impacts to aquatic life in streams are well documented. Alaska Department of Environmental Conservation (ADEC) noted that, in 2008, the majority of the streams that were designated under Category 4a (impaired water with a final/approved total maximum daily load [TMDL]), Category 4b (impaired water with other pollution controls) or Category 5 (impaired water, Section 303(d) listed and require TMDL) for water quality impairment from turbidity were associated with either mining or timber industries (http://dec.alaska.gov/water/wqsar/waterbody/2008ImpairedWaters.pdf) (see mining highlighted in Appendix C; Table C-1). This indicates that both direct mining activities and/or associated roads are critically impacting stream quality and thus reducing the viability of fish that use those habitats. The cumulative effect of the proposed access road construction, culvert placement, and maintenance for the 14 anadromous streams crossed could result in long-term reduction of habitat and subsequent reduction of viable salmonid populations presently found in these waterways. Importantly, the other 75 streams that have not [yet] been designated as anadromous streams would also be affected by road construction. Impacts similar to those predicted for the designated anadromous streams would also be likely occur to salmon and resident fish in these stream systems.

3.1.6 Fugitive Dust

3.1.6.1 Stressor Description

Fugitive dust would likely be dispersed within and outside of the mining area, depending on the wind speed and direction, soil moisture and other factors, at any given time. The tonnage of dust that escapes the mine will not be known until monitoring data are developed. Periodic high winds could mobilize and disperse dust for some distance. For example, Exponent (2007) cited Clark (2005) as noting detected dust dispersion at the Red Dog Mine in northern Alaska "*in sampling areas approximately 2 kilometers outside of the mine area, with visible dust extending well beyond the sampling sites* [and noted] *that dispersal was highest during dry periods, high winds, and associated with inversion phenomena*."

Sources of dust at the proposed mine are expected to be similar to other mines that have been studied and include:

- **Dust generated by open pit mining activities—**Dust can be generated from drilling, blasting, material handling, and truck haulage activities in the open pit;

- **Dust emissions from materials handling**—Dust can be generated from materials handling activities outside of the open pit, including truck haulage activities, placement of waste rock on waste rock stockpiles, and the stockpiling of ore;
- **Dust emissions from mill and concentrate storage facilities—**Dust can be generated from the ore crushers, the coarse ore stockpile building, and from concentrate storage and loading operations; and
- **Mechanical or wind-generated dust from surfaces—**Windblown dust can be generated from surfaces around the mine, including access roads and yards and tailings beaches, in addition to other mineralized surfaces.

There is a high potential for vegetation to be covered by dust emitted from the mining operations and thus habitat destroyed or reduced (Hoskin *et al*. 2000). This particulate layer can act to hinder plant functions by reducing light penetration or the exchange of gases by the leaves (International Council on Mining and Metals [ICMM] 2006). Fugitive dust can affect local vegetative and insect resources through coating important respiratory surfaces. The deposited particulate matter may block the plant leaf stomata, hence inhibiting gas exchange, or smother the plant leaf surfaces reducing photosynthesis levels (Moore and Mills 1977, Environment Australia 1998). Impacts can result in devegetation of large portions of land associated with instream salmon habitat. Without vegetative cover to restrict and mitigate surface runoff, stream turbidities and sedimentation can increase, with effects to salmon and salmon habitat similar to those previously discussed in Section 3.1.4, Road Construction.

3.1.6.2 Impact Determination

Distributions of fugitive dust emissions generally require either: (1) a particulate deposition collection study, or (2) an extensive air-transport modelling effort that considers particulate size, frequency of release and ambient weather conditions at the site under investigation. Neither of these methods was available for this risk assessment, but similar studies have been conducted for the Red Dog Mine, Alaska (Teck Cominco 2005). As such, information developed during the course of investigations directed at risk from fugitive dust emissions at Red Dog Mine have been used as a basis for predicting dust emission distributions, concentrations, and potential impacts at the proposed mine. Although it is well understood that mine specifics and locale vary between the proposed mine and Red Dog [as at all other mines evaluated], because of the detailed evaluation on dust deposition and surface soil concentrations conducted at Red Dog, it was selected as the most appropriate site for predicting dust impacts at the proposed mine.

Fugitive dust sources and volumes will vary between mines based on the processes used and geologic material being excavated. For this evaluation, it was assumed that processes and materials would be similar between Red Dog and the proposed mines, with meteorological conditions and ore/waste concentrations at both sites being the primary variables of interest. Meteorological conditions, specifically wind speed and direction, are relative for predicting dispersion distances and concentrations.

Data for Red Dog Mine (and vicinity) revealed that over an annual period, winds blow primarily from the north-northeast (Teck Cominco 2005) at an average speed of approximately 13 mph (http://www.city-data.com/city/Red-Dog-Mine-Alaska.html). Comparatively, over an annual cycle, winds near the proposed mine blow predominantly from two directions: north-northwest 38.3% of the time; and south-southeast 35% of the time (Hoefler Consulting Group 2006). The average annual wind speed is near 8 mph. As a result of this information, it can generally be expected that dust emissions would not travel as far at the proposed mine compared to Red Dog Mine. Dustfall jar deposition results for Red Dog show that greatest deposition occurs in the areas of the pit, ore stockpiles, mill and tailings beach (e.g., 'triangle area' in Teck Cominco 2005), with concentrations extending downwind to the south and southwest, but also distributed peripherally around the mine. Generally, based on data presented in Figure 5 of the Teck Cominco report, lead (Pb) concentrations exhibited a spatial pattern such that highest concentrations were found in the mine proper area (~100m from the source, referred to as zone 1 by this ERA), moderate concentrations occurred within a perimeter distance of approximately 100-1,000 m (zone 2), and lowest concentrations were in a perimeter zone from 1,000 to 2,000 m (zone 3) from zone 2.

Based on the results of the Red Dog Mine studies (Teck Cominco 2005), a conservatively-predicted scenario was developed for the proposed mine. Based on wind information from Hoefler Consulting Group (2006), Figure 15 was developed that provides northwesterly- and southeasterly-oriented isopleths which generally encompass distances of 100 [Zone A], 1,000 [Zone B] and 2,000 [Zone C] m. Because these directions account for winds during nearly 75% of the year, this scenario was considered appropriate.

Within Zone A (spatial area = 2.01 mi^2), plant community and drainage impacts are expected to be most observable and critical. Without important site management and mitigation measures, shifts and reductions of endemic plant community structure would likely result in patchy barren ground. Lichens and mosses, which are sensitive to dust impacts, would be affected to the greatest degree. At Red Dog Mine, for example, in the area closest to the mine, Teck Cominco Alaska, Inc., has been conducting vegetation impact studies to determine treatment options for these type of areas (ABR 2009).

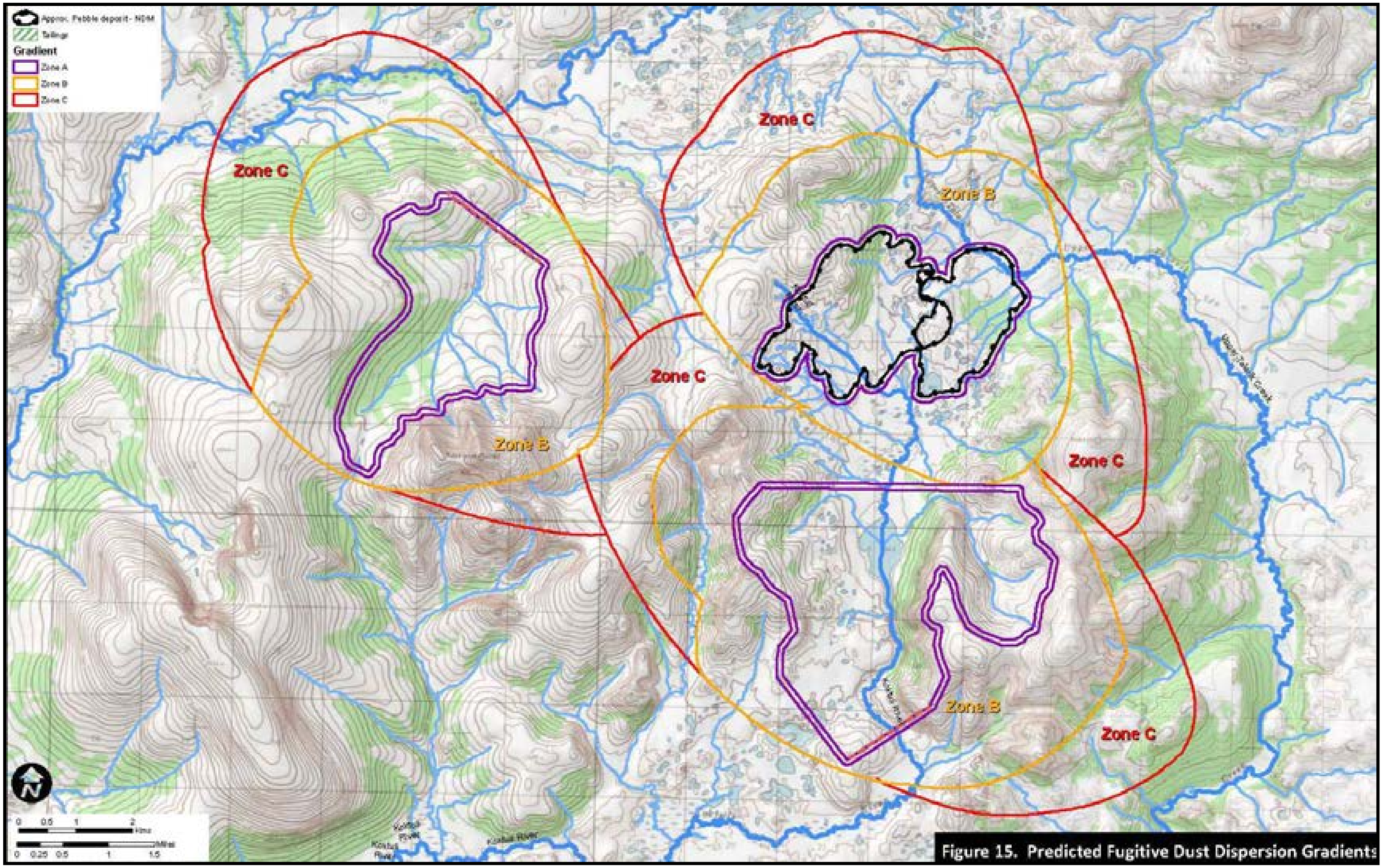


Figure 15. Predicted Fugitive Dust Dispersion Gradients

In Zone B (spatial area = 23.7 mi^2), effects may not be as obvious, but would be important. For a similar exposure at Red Dog Mine (Exponent 2007) a difference was observed between reference and 'Zone B' site communities; specifically there was a decrease in lichen cover which appeared to be a result of dust deposition – non-vascular plants are apparently more sensitive to dust (and metals in dust) than vascular plants.

Zone C (spatial area = 19.6 mi^2) is expected to show similar impacts, albeit at a slower pace. At Red Dog Mine it was reported "lichen cover values at stations 1,000 m and 2,000 m from the dust source were significantly lower than reference cover values (Exponent 2007).

Fugitive dust dispersion would affect a conservatively-predicted area of 33.5 mi^2 around the proposed mine, although areas outside of Zone C would most likely be affected to some degree. Within this 'predicted' area (but excluding water extraction areas proposed) are approximately 33 miles of ephemeral, intermittent, and perennial streams, which presently includes almost 10 miles of ADFG-designated anadromous waters that support rearing and juvenile salmonids. The measure of fugitive dust's impact on water quality is difficult to predict, but it would be expected that long-term (40-70 years) mining would result in denuded riparian habitat and increasingly degraded and embedded stream channels. Soils devoid of vegetation are especially prone to water and wind erosion and can result in the movement of soil particles off of the surface (USEPA 2007e). Based on review of similar hardrock mines, it is expected that highest dust levels would occur during earliest stages of mine development during pit creation, and that ore removal activities would generate somewhat less dust outside of the pit as it deepens during latter stages of the mine's life. Also, with proposed use of a pipeline for transporting the ore to the loading facility, dust created from haul roads would be eliminated.

Information regarding the overall ecological impact to other supportive invertebrate communities from dust deposition was not found. It is predicted that impacts to these resources, which are food for salmonids and other fishes, would be crucial and most likely long term. Over the life of the project, without specialized dust management practices, it would be expected that the immediate area (and beyond) for Zones A, B, and C would be significantly degraded. Down-gradient portions of the streams affected would show incremental negative changes over time as the ecological viability of headwaters that support salmonids, resident species, and other aquatic life diminishes.

3.2 Chemical Stressors

The following sections provide information on the extent and magnitude of chemical stressors expected to impact salmonid resources from development of the proposed mine. Many of these stressors of concern would result from metals' exposure in aqueous environments via reduced pH. Evaluation methods regarding temporal or spatial characteristics related to each source (as appropriate) are included in each Impact Determination section (as appropriate).

Both deposits (Pebble West and East) are referred to as sulfide ores because copper occurs in a compound containing iron and sulfur. Ore composition is important because sulfide ores are expected to form sulfuric acid when exposed to oxygen and water (United States Office of Surface Mining and Reclamation, 2007 and Acid Drainage Technology Initiative 2007). It is important to note that the mineral deposits associated with the proposed mine have little buffering capacity which increases risk of acid formation (NDM 2005c).

Sub-surface mining often progresses below the water table, so water must be constantly pumped out of the mine in order to prevent flooding. When a mine is abandoned or pumping ceases, or when precipitation or groundwater enters an operating open pit or underground mine, acid mine drainage (AMD), also known as acid rock drainage, can be triggered. Tailings ponds and waste rock piles can also be a source of AMD. When exposed to air and water, oxidation of metal sulfides (e.g., pyrite, which is iron-sulfide) within the surrounding rock and overburden generates acidity.

Colonies of bacteria and archaea (single-celled organisms) greatly accelerate the decomposition of metal ions, although the reactions also occur in abiotic environments. These microbes, called *extremophiles* (for their ability to survive in harsh conditions), occur naturally in the rock, but limited water and oxygen supplies usually keep their numbers low. Special *extremophiles* known as *acidophiles* especially favor the low pH levels in abandoned mines (Baker-Austin and Dopson 2007). In particular, *Acidithiobacillus ferrooxidans* is a key contributor to pyrite oxidation.

Metal mines may generate highly acidic discharges where the ore is a sulfide or is associated with pyrites. In these cases the predominant metal ion may not be iron but rather zinc, copper, or nickel. The Pebble deposit consists of the most commonly-mined ore of copper—chalcopyrite, itself a copper-iron-sulfide that occurs with other sulfides. Mining of similar copper sulfide ores in the U.S and worldwide has caused AMD (David 2003, Gilchrist *et al*. 2008, Gilchrist 2006, USFS 2009, Ashley *et al*. 2003).

Durkin and Herrmann (1994) reviewed data on mining waste generated from active and inactive mining sites in the western U.S. Their review revealed that in nine states over 2,500 miles of surface waterways were impacted by AMD. Of this total area, approximately 85 percent was attributed to copper, iron ore, uranium, and phosphate mining activities. Approximately one-half of the waste generated was mining rock waste and one-third was tailings, with the balance consisting of dump/heap leaching wastes and mine water. Scientific literature is plentiful with studies that quantify the adverse environmental effects of AMD on aquatic resources.

Most recent investigations focus on multiple bioassessments of large watersheds. These assessments include water and sediment chemistry, benthic macroinvertebrate sampling for taxa richness and abundance, laboratory acute water column evaluations, laboratory chronic sediment testing, caged fish within impacted streams, and development of models to explain and predict impacts of acid mine drainage on various aquatic species (Soucek *et al*. 2000, Woodward *et al*. 1997, Maret and MacCoy 2002, Hansen *et al*. 2002, Kaeser and Sharpe 2001, Baldigo and Lawrence 2000, Johnson *et al*. 1987, Griffith *et al*. 2004, Schmidt *et al*. 2002, Martin and Goldblatt 2007, Beltman *et al*. 1999, Hansen *et al*. 1999a, Boudou *et al*. 2005).

Farag *et al*. (2003) described streams in the Boulder River watershed in Montana impacted by nearly 300 abandoned metal mines as devoid of all fish near mine sources. Also, Barry *et al*. (2000) compared fish abundance, distribution and survival at contaminated and non-contaminated streams within Britannia Creek, BC. They noted that chum salmon (*O. keta*) fry abundance was significantly lower near the impacted waters (pH < 6 and dissolved copper > 1 mg/L) than the reference area. Reported laboratory bioassays confirmed AMD from the Britannia Mine was toxic to juvenile Chinook (*O. tshawytscha)* and chum salmon. Chinook salmon smolt transplanted to surface cages near Britannia Creek experienced 100% mortality within 2 days (Barry *et al*. 2000).

The U.S. EPA described 66 incidents in which environmental injuries from mining activities are detailed (EPA 1995). Nordstrom and Alpers (1999) reported that millions, perhaps billions, of fish have been killed from mining activities in the U.S. during the past century.

Chambers (2006) reviewed the geochemical characterization of rocks from 399 samples from the Pebble prospect area (NDM 2005c). Chambers noted these samples were analyzed for sulfur content which indicates acid generation potential and acid neutralizing potential (i.e., generally related to calcium carbonate content). Most regulatory agencies consider rock with a ratio of three times as much neutralizing material to acid generating material to be non-acid generating. Rock with equal amounts of neutralizing and acid-generating material, or with more acid-generating material than neutralizing material, are considered acid generating. EPA's Acid Mine Drainage Prediction Technical Document (1994) provides good information for evaluating the potential for AMD formation. As Chambers (2006) provides, if the analyses fall within the range of 3:1 to 1:1, then the rock is considered to be potentially acid-generating.

Figure 16 (source: NDM 2005c) shows a plot of Neutralization Potential (NP) versus Acid Potential (AP) for the 399 samples. The solid lines on the graph represent constant ratios of NP:AP for the ratios of NP:AP = 2:1 and NP:AP = 1:1. Some industry scientists consider rock with an NP:AP ratio of 2:1 to be non-acid generating, which is why this graph does not present the more conservative NP:AP = 3:1 line, which regulatory agencies would use (Chambers 2006). Over 95% of the 399 samples lie below the NP:AP = 1:1 line – that is, they are in the acid-generating category.

Several other conclusions were drawn from this data analysis in the NDM report, and Chambers provided comments to those [*as noted in brackets below*], they include:

- "… sulfur occurs primarily as sulfide minerals." [*rather than as sulfate minerals, which would not pose risks for acid generation*]

- "Sulfur concentrations in the pre-Tertiary rock types (i.e., much of the ore and non-overburden waste) are typically between 1 and 5 percent sulfur up to maximum concentrations near 9 percent." [*1% – 5% sulfur-as-sulfides is typically in the range for concern for acid mine drainage*]

- "Evidence that oxidation (of core samples) has occurred in storage is illustrated by the general increase in sulfate sulfur relative to sulfur as the age of the core increased." [*this says that some acid rock drainage has occurred in the older core samples taken from the site*]
- "… preliminary calculations indicate that it would take about 40 years for nearly all pre-Tertiary rock to become acidic under site conditions."

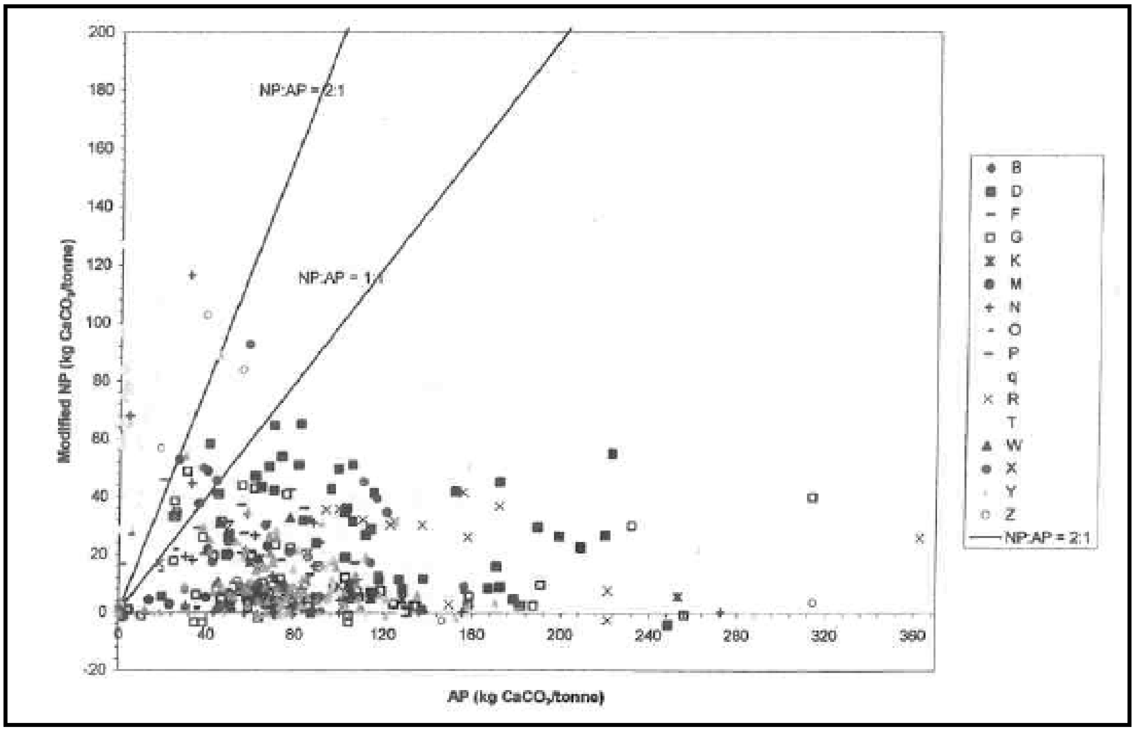


Figure 16. Plot of Neutralization Potential (NP) versus Acid Potential (AP): *showing that the majority of samples fall below the NP:AP ratio line of 1:1 and are, therefore, acid-generating* (Source: NDM 2005c)

Chambers pointed out that although the information presented in his report was not conclusive, it is clear from NDM's report that much of the rock from the proposed mine area is potentially acid-generating, and that great care will have to be taken in designing a mine to mitigate this potential.

Mine tailings and waste rock contain process chemicals and elements from natural rock that can be harmful to wildlife. Generally, metal concentration increases above background can negatively affect aquatic receptors, specifically salmon and resources they depend on that occur in local streams. Natural rock elements that occur in the Pebble prospect ore include aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, zinc, sulfides, and natural radioactive constituents (uranium, thorium, potassium-40) and others.

Determining the specific chemical fate and effects to biological receptors from release of ore constituents at the proposed mine into the environment is challenging. Predictive ecological risk assessments require information on the forms, transformations and geochemical environment of the metals under consideration. Smith (2007) provides a diagram of some processes and geochemical conditions that can redistribute cationic dissolved metals (such as expected at the proposed mine) in oxidizing, circumneutral-pH water systems near mining sites (Figure 17). Presently, no studies are available that predict the geochemical fate of released ore constituents in the Pebble prospect, or the potential risks to biological receptors as a result.

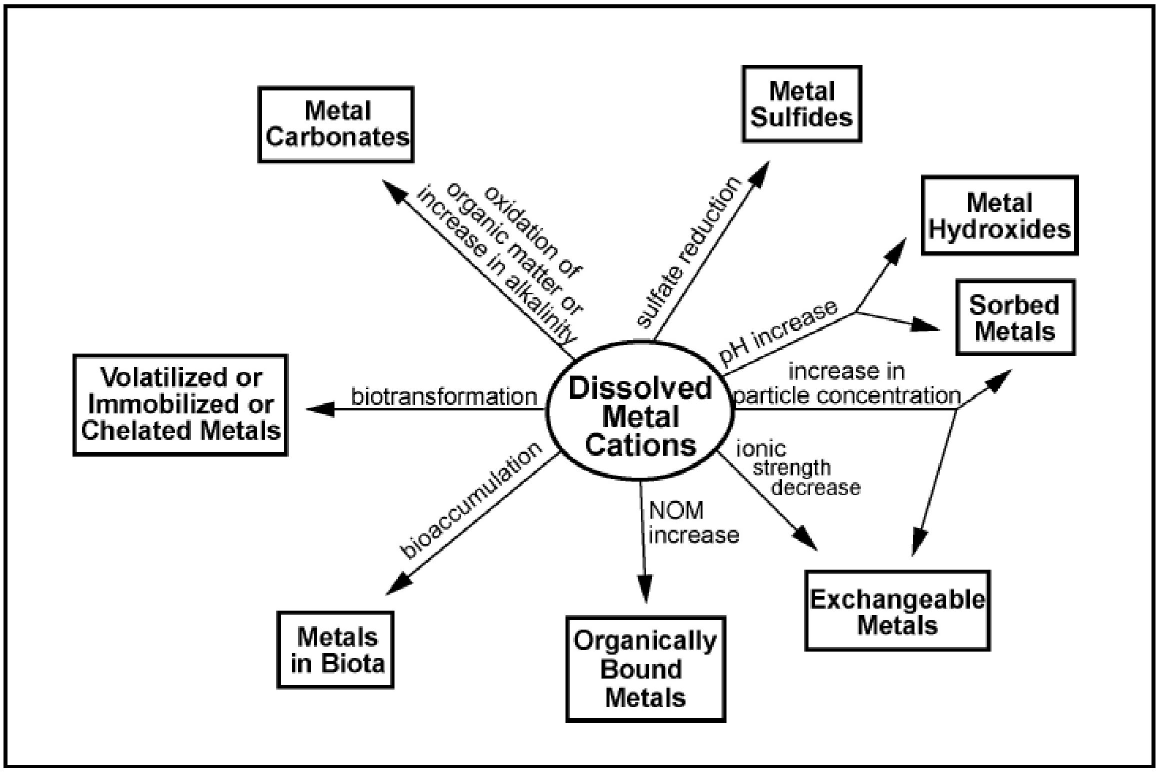


Figure 17. Processes and Geochemical Conditions Affecting Metals in Water
(Reprinted from Smith and Huyck 1999, with permission)

Ptacek and Blowes (2002) discussed transport mechanisms for metals to surface waters from mines in Canada. They provide that releases can take place over several decades to many centuries. Timing and duration of peak discharges vary from site to site, and depend on many factors including rate and extent of sulfide oxidation, acid neutralization potential, metal attenuation and release reactions. Groundwater velocity and length of flow path are critical factors to know in understanding release potentials. They reported that at sites where tailings had oxidized for 10 years, the pore waters contained elevated metal concentrations in the upper 5 m of the tailings. Metal concentrations were found in groundwater more than 100 meters from the tailings impoundment at a site where oxidizing had been occurring for more than 35 years, and for a 70-year old tailings pond at the Sherridon Mine, Manitoba, they reported very high metal concentrations in vadose zone [i.e., the portion of Earth between the land surface and the zone of saturation, extending from the top of the ground surface to the water table] pore water, and both groundwater and surface water were severely degraded (Ptacek and Blowes 2002). Similar to other older mines, ground water intrusion into surface water via lake sediments was found, suggesting that metals were available for diffusion or transport into the overlying water column. Even with the prevalence and use of existing predictive models, modeling for AMD has not yet found extensive applications in predicting oxidation rates and effluent quality at operating or proposed mines (Ferguson and Erickson 1988). One of the most significant issues is the inaccuracy of water quality predictions at hardrock mines. Maest *et al*. (2005) noted that large uncertainties are inherent in forward modeling predictions. Factors such as mine modification, lag times and duration of contamination have led modelers to emphasize ranges rather than precise values for water-quality predictions (Maest *et al*. 2005). Finally, Kuipers *et al*. (2006) found that an important cause of water quality impacts was errors in geochemical and hydrologic characterization of the mined materials and the mine site area. For the mines in their study that developed acid drainage, almost all either underestimated or ignored the potential for acid drainage in their EISs. In terms of predicted (post-mitigation) surface water quality impacts, 73% of the mines in their study having surface water quality impacts predicted low water quality impacts in their initial EISs, two predicted moderate impacts, and two had no information on post-mitigation impacts to surface water resources (Kuipers *et al*. 2006). Therefore, the predictions made about surface water quality impacts before the effects of mitigation were considered were more accurate than those made taking the effects of mitigation into account. Stated in another way, the ameliorating effect of mitigation on surface water quality was overestimated in the majority of the case study mines (Kuipers *et al*. 2006).

The most significant chemical stressor expected from hardrock mining operations in the Pebble prospect area is copper, along with other heavy metals. Previously reviewed studies on hard rock mining sites indicate that metals will occur within the watershed as a result of mining operations, but the level of concern expected within the Nushagak-Mulchatna and Kvichak river drainages is presently unknown. Metals' contamination from hard rock mines causes loading within various environmental media compartments including soils, sediment and water. Subsequent transfer or release into biological receptor groups, including vegetation and benthic organisms, can result in chronic exposure to fish via aqueous uptake and trophic exposure routes. Additionally, direct exposure to water-borne copper contamination can cause acute effects in fish, while impacts to their food resources (fish and benthic organisms) will likely result in an indirect impact on fish communities.

Of the three primary metals to be extracted from the proposed mine's ore, gold is benign but copper is known to be toxic to fish. Effects from fish exposure to molybdenum are not clear, but Reid (2004) noted that the toxicity of molybdenum in exercised fish is the result of adverse alterations in the oxygen and carbon dioxide exchange which was likely due to gill epithelium swelling and increased mucus production; a mechanism in common with effects from aluminum and nickel. A water quality criterion for molybdenum has not been developed. In addition, Morgan *et al.* (1986) reported that molybdenum accounted for only a very small proportion of tailings toxicity to freshwater mussels.

The analysis of water samples from the Pebble prospect area indicates that many other elements on the EPA's list of priority pollutants including antimony, arsenic, chromium, lead, nickel, selenium, and zinc are present in ground and surface waters and therefore are likely to be found in the ore body (Fey et al., 2008). These other metals are also toxic to salmon and other fish at low concentrations (Eisler 2000). To understand how mining related pollutants affect fish and aquatic life, copper is examined below in detail. However, the risks associated with the introduction of copper would, generally, be similar to other potentially harmful heavy metals likely present in the ore body and already present in ground and surface water within the region.

Copper Toxicity

Copper and copper compounds are acutely toxic to fish and other aquatic life at low parts per billion levels (ppb) (Eisler 1991; Eisler 2000; EPA 2007a; Hamilton and Buhl 1990). Copper (Cu) is essential to the growth and metabolism of fish and other aquatic life, but can cause irreversible harm at levels slightly higher than those required for growth and reproduction (Eisler 2000). When dissolved in water, elemental copper (Cu) and many copper compounds are toxic to fish and other aquatic life in the low parts per billion to parts per trillion ranges. As outlined below, copper ions have acute toxic, chronic toxic and behavioral effects on fish and aquatic life upon which they feed.

Exposure to sub-lethal levels of copper can have a detrimental effect on the behavior of salmonids. Salmonids are known to avoid waters with sub-lethal concentrations of copper and such concentrations alter other behavior as well. To put the potential for behavioral effects in context, background median dissolved copper levels reported from Zamzow's (2010) water chemistry data report ranged between 0.18 and 3.57 ppb., with a next highest value of 0.59 ppb. Five of the total 21 detections (N=22) in SFK were greater than 0.5 ppb. These data suggest that at present, median background levels of copper in the proposed project area may be below levels that would affect salmonid behavior.

Copper toxicity to freshwater fish and other aquatic life is affected by several factors including hardness, alkalinity, pH, water temperature, organic and inorganic complexation, synergistic effects with other metals such as zinc and age, size and species of fish (EPA 2007d, Chakoumas *et al*. 1979 and Eisler 2000). Water hardness, alkalinity and pH are interrelated and appear to be particularly important. Hardness is a measure of dissolved calcium and magnesium in water. Water with 0-60 mg/L (ppm) as calcium carbonate is considered soft, 61 to 120 ppm is moderately hard, and 121 and above as hard to very hard (USGS 2007). Alkalinity is a measure of the capacity of water to neutralize acid. Potential of Hydrogen or pH is a measure of the acidity or alkalinity of water. The ADEC acute and chronic copper Aquatic Life Criteria for Freshwater is calculated from a formula based on the hardness of the receiving waters (ADEC 2003).

Because of variability in individual species tolerance and the effect of water hardness on toxicity, some researchers have recommended that copper criteria should be developed on a site specific basis (Finlayson and Verrue 1982). For example, in tests to determine the relative sensitivity of bull trout (*Salvelinus confluentus*) and rainbow trout to acute copper toxicity, bull trout were found to be as sensitive to copper mortality as rainbow trout at water hardness levels of 100 ppm of $CaCO_3$, but 2.5 to 4 times less sensitive at 220 ppm $CaCO_3$ (Hansen *et al*. 2001). As such, researchers have predicted that the copper hardness-normalized criterion may be under-protective at low pHs. Presently, EPA's 2007 aquatic life freshwater quality criteria for copper is based on the Biotic Ligand Model (BLM) (EPA 2007c). The BLM is a metal bioavailability model that uses receiving water body characteristics and monitoring data to develop site-specific water quality criteria. Input data for the BLM include: temperature, pH, dissolved organic carbon (DOC), major cations (Ca, Mg, Na, and K), major anions (SO_4 and Cl), alkalinity, and sulfide.

Hardness

Copper and certain other metals such as cadmium are more toxic to fish in soft waters than in hard waters (Chakoumas *et al*. 1979, Sayer *et al*. 1989, Lauren and McDonald 1986, Lauren and McDonald 1984, Waiwood and Beamish 1978, Howarth and Sprague 1978). Hardness concentrations reported by NDM (2005a) are in the "soft to moderately soft" range (0-60 ppm): from 4.3 to 24.9 ppm in the North Fork Koktuli River, from 8.6 to 29.1 ppm in the South Fork Koktuli River, and from 10 to 45.2 ppm in Upper Talarik Creek (NDM 2005a). As such, because surface waters in the Pebble prospect area are relatively soft, and assuming water chemistry remains unchanged, it is likely that the introduction of only very small amounts of copper into the system would be needed before toxicity to salmon (and other species of fish) is observed.

Alkalinity

The acute and chronic toxicity of copper is also inversely correlated with alkalinity (Chakoumakos *et al*. 1979 and Lauren and McDonald 1986). Alkalinity is a measure of the capacity of substances (usually bicarbonate and carbonate) dissolved in water to neutralize acids, higher alkalinity equals higher capacity to neutralize acids essentially the capacity of water to resist changes in pH. Copper is more toxic at low alkalinity levels and increasing alkalinity levels reduce copper toxicity in rainbow trout and Chinook salmon (Lauren and McDonald 1986, Welsh *et al.* 2000). For protection of aquatic life, alkalinity should be at least 20 ppm calcium carbonate equivalent (ADEC 2003). Alkalinity concentrations in the proposed mine study area (reported as equivalent concentrations of $CaCO_3$) ranged from 11 to 32 ppm for the North Fork of the Koktuli River, from 7.0 to 35 ppm for the South Fork of the Koktuli River, and from 16 to 56 ppm for Upper Talarik Creek (NDM 2005a). Although, site specific sampling data were not available, NDM reports that most of the main stem sampling sites exceeded the minimum chronic aquatic-life criteria of 20 ppm during the May through October sampling period (NDM 2005a). Some of the sampling locations were below the minimum 20 ppm criteria for protection, and no data were provided during the winter low flow periods when alkalinity levels might be lower or higher (Sutcliffe and Carrick 1988).

pH

Copper is more toxic in acidic waters (Welch *et al*. 1993, Lauren and McDonald 1986). The ADEC Water Quality Standard For Designated Uses states that pH for Growth and Propagation of Fish, Shellfish, and Other Aquatic Life “May not be less than 6.5 or greater than 8.5 and may not vary more than 0.5 pH from natural conditions.” NDM (2005a) reported that pH levels ranged from 7.0 to 8.1 in the in the North Fork Koktuli River, from

6.6 to 8.4 in the South Fork Koktuli River, and from 6.8 to 7.7 in Upper Talarik Creek. Although, pH levels are currently within the ADEC standard range, based on the geochemical data released by mining proponents to date (NDM 2005c), it is important to note that some of the low alkalinity levels reported for these streams suggest a limited buffering capacity should acidity increase.

Acute Toxicity

Copper's acute toxicity to aquatic species has been well studied (Sorenson 1991; Eisler 2000). Exposure to copper causes ionoregulatory and respiratory problems in freshwater fish. Researchers at EPA’s Corvallis Environmental Research Laboratory found that dissolved copper (Cu) is acutely toxic to juvenile Chinook salmon and steelhead trout at levels of 17 to 38 ppb of copper. Steelhead trout *(Onchorynchus mykiss)* are more sensitive than Chinook salmon *(Onchorynchus tshawytscha)*, and salmon fry and smolt are more sensitive than newly hatched alevins (Chapman 1978a). They also found that copper is acutely toxic to adult male coho salmon and adult male steelhead at 46 and 57 ppb respectively (Chapman and Stevens 1978). California Department of Fish and Game toxicologists found that median lethal concentrations for juvenile Chinook salmon in 96 hour flow through tests were 26 to 34 ppb of copper. Exposure to 49 ppb of dissolved copper for 24 hours caused a rapid decline in blood sodium, chloride, and oxygen tension and increased heart beat in rainbow trout. At the same time arterial blood pressure doubled. Heart failure caused death. Because gill tissue controls oxygen and electrolyte levels in fish, these changes may be caused by gill tissue damage observed in fish which were exposed to copper (Wilson and Taylor 1993).

When exposed to copper, the incipient lethal level was between 37 and 78 ppb for sockeye salmon but between 25 and 55 ppb for pink salmon during the egg to fry stage (Eisler 1998). Growth and hatching were no better than mortality as indicators of toxic effects of copper. Copper inhibited egg capsule softening, but associated mortalities during hatching occurred only at concentrations also lethal to eggs and alevins. Copper was concentrated by eggs, alevins and fry in proportion to exposure concentrations. Copper concentrations of 105 and 6.8 ppm in pink salmon eyed eggs and fry, respectively, coincided with mortalities (Servizi and Martens 1978). Several studies found that salmon fry, smolt and adults acute copper toxicities were lower than developing eggs.

Canadian researchers conducted tests in an artificial stream to determine the attraction and avoidance responses of rainbow trout to lethal copper concentrations (0.5 to 4.0 ppm) over 96 hours (Pedder and Maly 1985). At all concentrations, there was an initial attraction period for copper and then subsequent avoidance of the more highly contaminated waters. Attraction was greatest in tests employing higher concentrations (3.0 and 4.0 ppm) of copper; but this attraction led to high mortality. These results indicate that observed trout behavior subsequent to copper discharges contributed to high mortality. The results also suggest that behavioral response of organisms to toxicants must be incorporated into work attempting to set reasonable water-quality standards in natural water bodies (Pedder and Maly 1985).

Chronic Toxicity

Exposure to elevated, but sub-lethal, levels of copper reduces the viability and increases the mortality rate of salmon and other fishes over time. For example, Coho salmon, which were exposed to sub-lethal levels of aqueous copper (1/4 and 1/2 of the dose which killed one half of the population in 4 days (LC50), lost their appetite and ceased growing or showed decreased rates of growth (Buckley *et al*. 1982).

Copper is broadly toxic to the salmon olfactory nervous system (Baldwin *et al*. 2003). Exposure to 1.0 to 20.0 ppb copper impaired the neurophysical responses of juvenile coho salmon olfactory receptor neurons to natural odorants within 10 minutes of exposure. The inhibitory effects of copper were dose dependent but were not influenced by water hardness. Toxic thresholds for the different receptor pathways were found to be 2.3 to 3.0 ppb over background. Short term influxes of copper to surface waters appear to interfere with olfactory senses that are critical for spawning, feeding, predation avoidance, and migration of wild salmonids (Baldwin *et al.* 2003). In laboratory tests, exposure to 25 to 300 ppb of copper significantly reduced the number of olfactory receptors in Chinook salmon and rainbow trout due to cellular necrosis (death of cells). These levels caused histological damage and neurological impairment to the olfactory system that these fish require for survival. Chinook salmon olfactory receptors were found to be harmed by lower doses of copper (50 ppb) than rainbow trout (200 ppb). Chinook salmon were more susceptible to olfactory damage at lower levels of copper than rainbow trout in copper contaminated waters (Hansen *et al*. 1999b).

Exposure to low levels of dissolved copper reduces the resistance of rainbow trout to disease. The effect of exposure to sub-lethal concentrations of copper (6.4, 16.0, and 29 ppb) on the immune systems of rainbow trout was measured after 3, 7, 14, and 21 days of exposure by researchers at the University of California Davis (Dethloff and Bailey 1989). They found that the immune system was altered at all concentrations with the greatest effects at higher concentrations. Consistent alterations in immunological parameters suggest that these parameters could serve as indicators of chronic metal toxicity in natural systems (Dethloff and Bailey 1989).

Exposure of Chinook salmon and rainbow trout to sub-lethal levels of copper increased their susceptibility to *Vibrio anguillarum* infections. *Vibrio* is a serious disease of fish. Exposure levels were 9% (parts per trillion) of the copper LC50 for 96 hours. Vibriosis mortality was also greater in exposed fish than unexposed fish. Rainbow trout stressed by copper required 50% less pathogens to induce a fatal infection than non-exposed fish (Baker *et al*. 1983). Exposure of rainbow trout to sub-lethal levels of copper in water increased their susceptibility to infectious hematopoietic necrosis (IHN) virus. In most instances, the percent mortality was twice as great in the stressed groups compared with those groups which were not stressed but received the same virus dose. Although the level of copper in the water influenced the mortality rates, the length of exposure did not prove to be critical, as similar results were obtained after 1, 3, 5, 7, or 9 days of exposure. When different virus challenges were employed, the percent mortalities were again greater in the stressed fish at all virus doses tested, and at one dose, mortalities were noted in the stressed group but not in the untreated group (Hetrick *et al.* 1979).

Behavioral Effects

Exposure to sub-lethal levels of copper can have a detrimental effect on the behavior of salmonids. Salmonids are known to avoid waters with sub-lethal concentrations of copper and such concentrations alter other behavior as well. Tests of the responses of Chinook salmon and rainbow trout to sub lethal levels of copper, cobalt, and a mixture of copper and cobalt found that behavioral avoidance of copper varied greatly between Chinook salmon and rainbow trout in soft water (less than 40 ppm hardness). Chinook salmon avoided at least 0.7 ppb of copper, whereas rainbow trout avoided at least 1.6 ppb copper. Furthermore, following acclimation to 2 ppb of copper, rainbow trout avoided 4 ppb of copper and preferred clean water, but Chinook salmon failed to avoid any copper concentrations and did not prefer clean water. The failure to avoid high concentrations of metals by both species suggests that the sensory mechanism responsible for avoidance responses was impaired. Exposure to copper concentrations that were not avoided could result in lethality from prolonged copper exposure or in impairment of sensory-dependent behaviors that are essential for survival and reproduction (Hansen *et al.* 1999b).

Rainbow trout exposed to copper and nickel solutions in a linear Plexiglas chamber with countercurrent flow avoided copper concentrations of 6.4 ppb total copper (Giattina *et al*. 1982). When copper concentrations were gradually increased, rainbow trout initially avoided low copper concentrations, but were attracted to higher concentrations (330-390 ppb) that are considered lethal. The 24-hour average concentration of these two metals presently considered adequate for the protection of freshwater aquatic life fell within the 95% confidence limits for threshold avoidance concentrations. This may indicate that environmental impacts predicted on the basis of toxicity tests alone do not reflect potentially important behavioral changes caused by sub-chronic concentrations of copper and nickel. Avoidance tests, therefore, may prove to be a valuable tool for screening toxic chemicals, providing additional information and a broader perspective for evaluating the impact of aquatic contaminants on fishery resources (Giattina *et al*. 1982).

In field tests in the South Fork of the Coeur d'Alene River in Idaho, the spawning migration of adult male Chinook salmon was monitored by radio telemetry to determine their response to the presence of copper, lead, zinc and cadmium contamination. The majority of the fish avoided the contaminated South Fork and moved up the non-contaminated North Fork to spawn. Metals levels in the South Fork waters were 6.90 ppb cadmium, 2.0 ppb copper 23.0 ppb lead and 2,220 ppb zinc at hardness of 108 ppm. The results agree with laboratory findings that wild fish avoid spawning streams with high levels of metals contamination (Goldstein *et al*. 1999). The results were also consistent with a study of wild Atlantic salmon (*Salmo salar*). Similarly, in a study of wild Atlantic salmon in a stream contaminated by a base metal mine (Saunders and Sprague 1967), spawning salmon avoided sub-lethal concentrations of copper and zinc by returning downstream prematurely.

3.2.1 Chemical Spills

3.2.1.1 Stressor Description

Mining requires the use of many types of hazardous chemicals. The process of flotation is the most widely used method of mineral separation of sulfides, oxides and native metals from silicates and of the separation of specific minerals. Flotation is applied to finely ground ores - the upper size is determined by what an air bubble will lift. Several specific hazardous chemicals are used during this process including:

Frothing Agents – The frother provides strength to the bubbles formed in the flotation cells. This prevents the bubbles from bursting when reaching the surface and allows the froth to be mechanically removed to recover the sulfide minerals. Frothers are organic surfactants that are absorbed at the air/water interfaces (bubbles), and create a sudsy situation that allows the minerals that have bonded with xanthates to attach themselves to air bubbles in the froth. The two main functions of frothers are to ensure the dispersion of fine bubbles in the ore-pulp and to maintain an adequate stability of the froth on top of the pulp. (e.g., Methyl Isobutyl Carbinol (MIBC), and also Pine Oil).

Collecting Agents – The collector makes the bubbles attract the sulfide minerals. Collectors induce specific minerals to adhere to froth bubbles; and modifying agents induce or depress adhesion of specific minerals to the bubbles. The collectors are organic molecules or ions that are absorbed selectively on certain surfaces to make them hydrophobic. These are thus the most important and the most critical flotation agents. Typically these are ethyl, butyl, propyl and amyl xanthates (e.g., potassium amyl xanthate).

Depressors - Depressors are inorganic compounds which selectively cover the mineral surfaces to make them hydrophilic and thus decreasing their affinity for collectors. The use of depressors increases the selectivity of flotation by preventing the flotation of undesirable particles. (e.g., cyanide – While cyanide is primarily used to dissolve gold from ore/concentrate, it is sometimes used in small amounts in base metal flotation operations to keep pyrite from being collected in the flotation cells.)

Activators – Activators essentially re-sulfides those ore particles that may be partially oxidized or contain a mixture of sulfides and gangue to make them more amenable to the flotation process. This is done by adsorbing onto the mineral surface where the sulfur atom provides a site to attract the collector. This ensures those particles that are difficult to float (i.e., contain minor amounts of sulfide) go to the concentrate rather than the tailings. Activators are generally soluble salts that ionize (dissolve) in water. The ions in solution react with the mineral surfaces to favor the absorption of a collector. Activators are used when collectors and frothers cannot adequately float the concentrate. (e.g., copper sulfate)

Flocculant – Flocculants are used to collect suspended particles to help separate water and solids. Flocculants are polymers, essentially water-in-oil emulsions. Flocculants are found in tailings, but generally adheres to particles and is not particularly mobile in the soil.

Surfactant – Surfactants are products that carry out sensibly the same role as detergents (for example washing detergents). They are designed to reduce the hydrophobic characteristic of organic contaminants to such a level that they are removed from the solid particles.

Lime – used primarily to raise the pH of the processing solution to the desired level.

Acid – might be added at the end of the water treatment process to bring a discharge of treated mill water, which may be elevated due to the addition of lime, down into a pH range mandated by water quality standards.

Transportation and storage of hazardous chemicals near water bodies can result in inadvertent spills which may result in contamination producing fish kills or other acute impacts to fishery populations. Generally, it is expected that quantities of hazardous material will be limited or contained in areas near aquatic systems. Except for road building and other construction activities, impacts to salmonids from these sources should be limited. In the event of a pipeline break (see Section 3.2.3) or episodic and large scale pollution event [tailings dam failure] (see Section 3.2.4) clean-up activities could result in large pieces of heavy equipment and maintenance materials being required instream at the site. In these instances, moderate-sized spills could potentially occur, but generally spill response materials are required on these types of equipment. Spills could result in release of hydrocarbons such as diesel, oil, gasoline or other similar products. Impacts would be critical if spills occurred in spawning or rearing habitat. Limited information was available on impacts to salmonids from these stressors. Because of the variable nature of chemical spills over the life of the proposed mine, no direct evaluation of impacts from this stressor has been conducted, but potential effects are considered in the Cumulative Risk Analysis (see Section 4.3).

3.2.2 Fugitive Dust

3.2.2.1 Stressor Description

Section 3.1.5 covered potential physical effects expected from fugitive dust dispersion associated with mining activities. Based on the scenario presented in that section (see Figure 15), fugitive dust generated at the proposed mine would be dispersed outside of the mine area, generally, as predicted, within a three-tiered gradient and consistent with seasonal wind patterns. The deposition analysis in Section 3.1.5, shows the highest metal concentrations are expected closest to the mine proper (i.e., Zone A: <100 m), with secondary (Zone B: 100-1000 m) and tertiary (Zone C: 1000-2000 m) areas exhibiting relative reductions based on dispersal mechanisms.

Impacts to salmonids from metal-laden dust particles transported by runoff into streams or leached from soils, could occur. Aslibekian and Moles (2003) showed that a long history of mining near Tipperary, Ireland, resulted in elevated soil metals' concentrations in depositional areas associated with water-related pathways. Studies associated with the Clark Fork River Complex provide documentation on the effects of river transport of metals from primary sources of contamination, including particulates which ultimately contaminated sediments. Near the proposed mine, dust-adsorbed metals could ultimately be deposited in sediment, with subsequent release to surface waters, biologically incorporated into benthic macroinvertebrates which serve as food resources for salmon and other resident fish species, or bound within sediment matrices. If surface waters become contaminated by AMD (e.g., lowered pH), metals could be leached into the water column from metal-laden sediments. Studies in the western U.S. (USDOI 2009) and internationally (Herr 1998) show mine dust can produce extensive problems that can persist for decades because such sites have low soil pH and lack normal soil stabilization processes. As a result, these sites do not develop normal soil structure or support the establishment of vegetation cover.

3.2.2.2 Impact Determination

The concentration of various metals expected within dust escaping from the proposed mine is unknown at this time, but estimates for the proposed mine were made based on ambient winds near the mine and from historical information for metals in dust deposited at Red Dog Mine, Alaska (Teck Cominco 2005). First, we looked at surface soil results for the 0-2 cm horizon (see Figure 2 in Teck Cominico's 2005 report) that provide background surface soil concentrations taken prior to mine development in 1977-1979. Generally, this figure showed that exploration mineral Pb soil samples were near 1000 ppm over the mine-proper area, and around 100 ppm (or less) to the southwest. Samples were focused in the SW area based on winds that predominate from the northeast. In 2003/2004, the tundra soil sampling program (directed at fugitive dust evaluation) revealed that surface soil Pb concentrations were approximately 22,000 to 23,000 ppm (see Teck Cominco 2005 Figure 4) over the mine proper area (denoted as the 'triangle' area); 5,000-10,000 ppm within a ~1,000m perimeter from the 'triangle' area (i.e., zone 2); and 3,000-5,000 ppm in an area ~1,000m from 'zone' 2 perimeter [see Figure 5, Tech Cominco 2005]. Because the mine began construction in 1987, we assumed a 16-year period (i.e., 1987-2003) for fugitive dust to increase surface soil concentrations to that observed in 2003-2004. Importantly, the Teck Cominco (2005) report states that *"The data expressed in Figure 10 represents the lead deposition rate recorded by the dustfall jars and shows a similar pattern to the tundra soil results."* This statement indicates that the authors considered that the increased soil concentrations were likely a result of the dust deposition. So, based on this information, we assumed an incremental increase in surface soil concentrations over the period from mine initiation (1987) until the 2003-2004 sampling program (16 years). Because no annualized increase in soil Pb concentrations was provided, we assumed a linear increase over the 16-year period for each of the three areas (zones) studied (e.g., zone 1: 1000 to 22,500 mg/kg; zone 2: 100 to 7,500 mg/kg; zone 3: 100 to 4,000 mg/kg). To determine if the soil concentrations found at Red Dog were reasonable, we compared them to other studies. Kribek *et al*. (2010) found copper concentrations in dust ranging from the 100's to 100,000's mg/kg near mines in the Zambian Copperbelt Mining District. Relative to distribution, other studies have shown that metals have been scattered by prevailing winds from 5.4 (Grzebisz *et al*. 2001) to 9-12 km (Kalandadze 2003) around mines.

Based on the information presented above, we estimated dust deposition for the proposed mine within each of the three zones (Figure 15) based on observed rates of deposition for the 16-year period within each respective area at Red Dog (Figure 18). Based on this approach, we estimated the maximum increase from baseline at the proposed mine to concentrations in the future, while accounting for the differences in ore concentrations between Pb at Red Dog and Cu at the Pebble prospect. Although Pb at Red Dog is ~4.8%

and Cu at Pebble is ~0.6%, we assumed that total dust dispersion from mining operations would be similar even if metal loads varied. To estimate future concentrations for the proposed mine, we used initial soil concentration information from Pebble prospect area baseline studies (Fey *et al*., 2008) that showed the upper soil horizon had Cu concentrations of ~26 ppm, and estimated future concentrations if soil increases (i.e., dust accumulations) were similar to those at Red Dog. Because it was known how much the Pb concentration had increased at Red Dog in the three 'zones' over the 16-year time period, and the Pb content of the ore was known, the ratio of 8:1 (e.g., 4.8% Pb to 0.6% Cu) was used to translate the effects observed at Red Dog to those predicted at the proposed mine. Therefore, using the factor for this ratio (0.125) proposed mine yearly concentrations determined during Step 1 were recalculated by multiplying each year's increase by 0.125, and then adding it back to the original concentration for that year to get a predicted annual soil concentration.

The evaluation meant that the 16-year increase had 1), considered an increase in soil metal from a similar dust accumulation scenario (i.e., Red Dog), and 2), accounted for the relative difference in ore content between the two mines (e.g., lead-to-copper). It is understood that there are variables that cannot be addressed by this method, and there are assumptions to be made relative to rainfall, snowmelt, etc., but for this analysis it was felt that this method would at a minimum provide an order of magnitude for metal-laden dust accumulation concentrations based on a real-world study. Because the Red Dog study only looked at dust accumulation concentrations up to 2003-2004, it is unclear how the concentrations would increase over subsequent years at the proposed mine. Although, we do not believe that soil concentrations will increase along the same slope over the 70-year life of the mine. As a result, for subsequent sections of this analysis, predicted soil concentrations only reflect those values determined for the first 16 years of operation (Table 14). In addition, the analysis does not consider possible dust control best management practices (BMP) such as those that have recently occurred at Red Dog Mine.

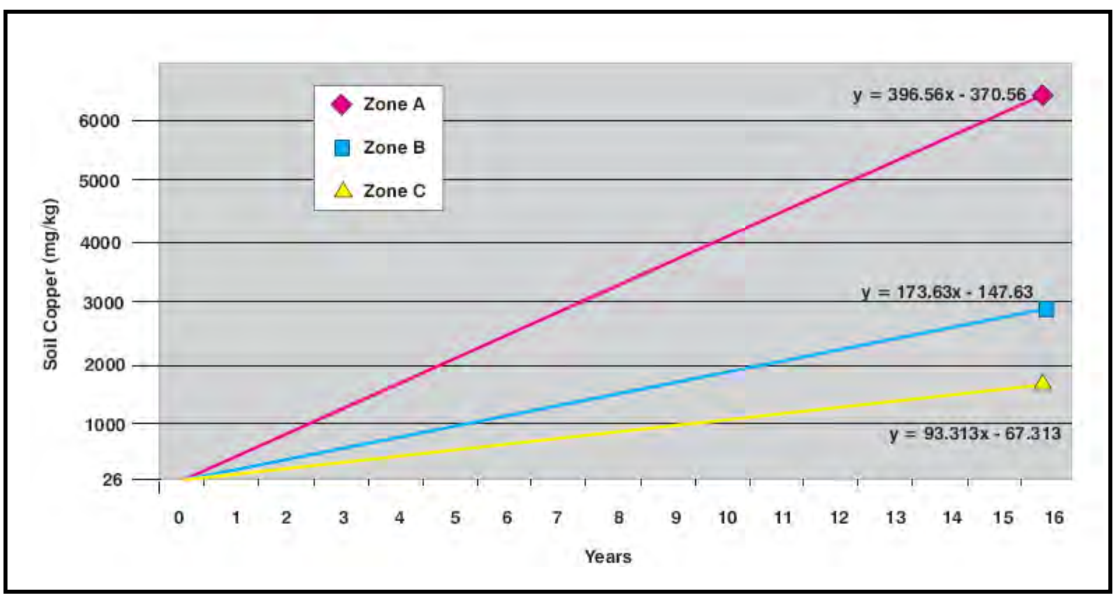


Figure 18. Estimated Soil Copper Concentrations in Zones A, B, and C, based on observed rates of deposition from Red Dog Data

To understand the potential for transport of metals in dust within the three zones to proximal surface waters at the proposed mine, two transport scenarios were considered:

- The bulk metal concentration expected from physical transport of ore dust particles to surface water; and
- The leachable metal concentration from dust in soil that is expected to reach surface water.

Transport of metals in soil matrices depends on both chemical-specific factors in addition to site-specific media considerations (Appendix D, Table D-1). A metal's chemical characteristics, the site's ambient conditions, and soil type and chemistry will regulate potential migration both vertically and horizontally. The degree to which each of these factors affects the fate and final disposition of a metal will depend on both chemical and physical factors. For example, soil clay content has a strong influence on the transport behavior of copper. A small difference in clay content can result in a significant difference in copper migration behavior. Importantly, the concentration of a trace metal in mine waste does not necessarily reflect its potential for release (Lapakko 2002). The phase in which that trace metals exists determines how readily available it is for release to the environment.

Table 14. Estimates of Annual Copper Contributions to Surface Soil Horizons in Three Zones around Proposed Mine

Metal	Average U.S. Soil Conc[1] (mg/kg)	Proposed Mine Average Soil Concentration (mg/kg)	Predicted Annual Post-Operation Deposition Rates in Zones[3] (mg/kg/Yr)			Maximum Predicted Concentrations in Mine Zones[2] (mg/kg)								
						10 yrs			16 yrs			>16 yrs		
			A	B	C	A	B	C	A	B	C	A	B	C
Copper	30	26	396	136	72	3,960	1,360	720	6,371	2,201	1,180	6,371	2,201	1,180

1 = Lindsay, W.L., 1979. *Chemical Equilibria in Soils*. New York: John Wiley & Sons. 449 pp.
2 = Assumes that soil concentrations past 16 years will remain constant.

Generally, heavy metals are transported: 1) horizontally via runoff to nearby surface water bodies then to sediment; 2) vertically to surficial groundwater then to surface waters; 3) to deep groundwater aquifers through infiltration; and 4) to air via evapotranspiration. For this analysis, transport factors 1 and 2 were evaluated. Hellweg *et al*. (2005) found that surface run-off accounted for 35% of the transport, with leaching accounting for approximately 25%. Thus, it is suspected that these two factors will account for a significant source of metal transport to surface water from soil.

Erosional Transport

Erosional transport is difficult to predict without site-specific information. This includes precipitation frequency and duration, infiltration rate, topography, vegetative cover, soil type and chemistry, surface water flow information and other variables. Water flowing off the soil surface provides the mechanism for transporting particles loosened by rainfall.

Although described as sheet flow, this type of flow seldom occurs in an uninterrupted sheet. Usually the water detours around clods, spills out of small depressions, and in general moves with sluggish irregularity. Even so, the water is able to carry soil particles. This transport ability is influenced by the energy level of the flow, which in turn is dependent on the depth of flow and slope of the land. Flat areas have little or no runoff; consequently, no transport occurs. Runoff from steeper areas flows at greater velocities and may have considerable transport capability. Each type of soil has its own inherent susceptibility to the forces of erosion, in large part because of chemical composition and organic matter content. Although large-grained materials are easily detached by raindrop splash or flowing water, they are not easily transported. On the other hand, fine soils such as clays and silts that bond together tightly are not easily detached, but once free they are transported with little difficulty (Pidwirny and Draggan 2008). For this reason, fine materials can be carried considerable distances, whereas larger particles are deposited somewhere along the flow path.

Site specific data were unavailable for this analysis, therefore transport was predicted based on a study conducted by Striffler (1969) that measured erosion rates in alpine tundra. The study concluded that erosion rates were very low (<1 m per 2-year period), even though the study took place on slopes with grades of 5% to 60%. Soil particle movement was most highly correlated with snow deposition – melt water running over saturated soil carries particles a short distance downslope.

At Red Dog Mine, particle size analysis of soil near the DeLong Mountain Regional Transportation System (DMTS) indicated that 98% of soil particles were larger than 1 micron in diameter and about 80% of soil particles were larger than 10 microns (Exponent 2007). Lamprecht and Graber (1996) found that the most common size fraction of soil dust particles collected over 24 hours ranged in diameter from 10–20 microns along the Dalton Highway in Alaska. The particle size of zinc and lead concentrates at Red Dog were determined to be <40 microns, with 80 percent <20 microns (Teck Cominco 2003b and f <u>in</u> Exponent 2007). Particle size is important because studies show that the smallest size particles contained the highest percentage of weak acid leachable copper (Hansen *et al*. 2005) and are typically the most mobile.

From this information a prediction was made concerning potential migration of metal-laden dust particles into streams within each of the three designated zones. The analysis assumed that land slope in the area was near 1-2%. This was based on data from Woody (2009) where the mean stream gradient was 1.4% for 22 surveyed streams. Per Striffler (1969), a conservative movement estimate of 10 cm per year was made for dust particles in areas near streams remaining after operations begin and water extraction commences. This will likely result in a slow progression of metal-laden dust particles entering streams at annual soil copper concentrations of up to 396 mg/kg in Zone A, 136 mg/kg in Zone B and 72 mg/kg in Zone C. After being dislodged during heavy rainfall events and transported to streams, metals associated with fine particulate matter (dust) would most likely accumulate at slow-moving, depositional areas within streams. Although mixing with other sediment particles will likely occur, cumulative annual contributions can result in increased sediment concentrations over time. Because the thin surficial soil layer will have the highest metal concentrations (e.g., with no mixing considered), the contaminant concentration in the delivered sediment may be several times greater than the concentration in the 'bulk' soil from where it came. Erosion studies in agriculture plots have shown mean metals concentrations in sediments up to 4 times higher than the parent soil (Quinton and Catt 2007). Their study revealed that in some erosion events the sediment had 13.5 times the concentration of metals found in the soil. Similar impacts were observed near mining sites in Tennessee's Copper Basin. Mayfield *et al*. (2009) reported sediment enrichment from impacted soils for arsenic, cadmium, lead, uranium and other radioactive metals. Their study also showed that metals in sediment were both attenuated (arsenic and lead) and enriched (lead, thorium and uranium) as streams progressed away from the mining source.

As a first step for predicting impacts to salmon from metals' enrichment from the erosional transport of dust particles, baseline sediment metal concentrations were reviewed for streams near the proposed mine (Fey *et al*., 2008, Zamzow 2010). Although it was expected that sediment concentrations would be similar for the three stream systems under investigation, this was not necessarily the case. Data indicates that sediment copper concentrations in the SFK watershed were higher (mean = 46 mg/kg) compared to the other two watersheds (NFK mean = 29.8 mg/kg; UTC mean = 4.4 mg/kg). It was difficult to determine the reason for this anomaly. It could be due to sample location or greater fine-grained sediments (and thus metal accumulation), but this is unknown. At any rate, future-projected concentrations from annual dust contribution were compared to established sediment screening benchmarks.

Per Hellweg *et al*. (2005), predicted concentrations assumed that of the averaged combined maximum annual dust concentrations expected for the three zones (e.g., A+B+C/3, see Table 14), 35% would be transported via erosion from soil and deposited as sediment. The remaining soil concentrations would either be leached into surface groundwater or bound within soil matrices. Some of the dust particles that reach a stream would be discharged downstream within the water column and deposited away from the source. This deposition assumption was based on information provided in Thomas *et al.* (2001) where studies on longitudinal loss rates for various sized particles (e.g., very fine, 15-52 microns; fine, 53-106 microns; medium, 107-250 microns) showed that local hydrological and benthic conditions establish a minimum rate of particle deposition and that departures from this rate due to gravitational forces begin to occur at particle diameters similar to the larger size classes used in their study (50-100 microns). Again, it was predicted that dust particle size at the proposed mine would be similar to that at Red Dog Mine; <40 microns, with 80 percent <20 microns (Teck Cominco 2003b and F in Exponent 2007).

With the understanding that the smallest soil particles would be transported most readily it was conservatively predicted that 50% of these smallest particulates would be deposited within the first kilometer, with the other 50% portion of the concentration transported to the successive 1-km stream segment. Also, to account for real-world events such as snow-melt and rain storms that could move these small particles out of upstream stream segments, two uncertainty levels were considered. These uncertainty levels (or factors in the following equations) were subjectively applied with the assumption that 25% (high prediction) or 10% (low prediction) of the metals making it to the stream would be deposited and retained in sediments within the stream segment being evaluated. Based on the factors and processes presented above, the following soil-to-sediment transport model was developed:

$$\textbf{\textit{Annual copper concentration in km 1}} = [_{4}(\max A+B+C)/3^{1} * (0.35)^{2}] * [0.5]^{3} * [0.25]$$

then,

$$\textbf{\textit{Annual copper concentration in km 2}} = [1^{st}\ \text{km concentration}] * [0.5] * [0.25]$$

then,

$$\textbf{\textit{Annual copper concentration in km 3}} = [2nd\ \text{km concentration}] * [0.5] * [0.25]$$

et cetera, until concentration is nill.

[*This process was also used for an assumed 10% retention*].

Notes:

1 = Cumulative annual mean soil concentration for three zones
2 = Percent of copper concentration transported from soils via erosion
3 = Predicted percent sediment deposition
4 = Estimated percent retention

Based on the above, predicted concentrations for each kilometer downstream segment were added to mean baseline concentrations derived from Fey *et al*. (2008) and Zamzow (2010) for each watershed in order to relatively evaluate increased contributions. Also, for long term assessment, concentrations were developed for each 10-year increment over the expected life (i.e., 70 years) of the mine (Table 15). Predicted concentrations were then compared to relevant sediment screening benchmarks to assess potential impacts to aquatic organisms.

Consensus-based Probable Effect Concentration (PEC) sediment quality guidelines (SQGs) from MacDonald *et al.* (2000) and NOAA Effects Range Medium (ERM; Long *et al.* 1995, Long and Morgan 1991) were selected for comparison to predicted temporal concentrations. The 149 mg/kg copper PEC represents the geometric mean of published SQGs from a variety of sources. Sources for PECs include probable effect levels, effect range median values, severe effect levels, and toxic effect thresholds (see MacDonald *et al.* 2000 for references). PECs are intended to identify contaminant concentrations above which harmful effects on sediment-dwelling organisms are expected to occur (MacDonald *et al.* 2000). NOAA's National Status and Trends Program, Sediment Quality Guidelines copper ERM of 270 mg/kg was obtained from:
http://response.restoration.noaa.gov/book_shelf/121_sedi_qual_guide.pdf.

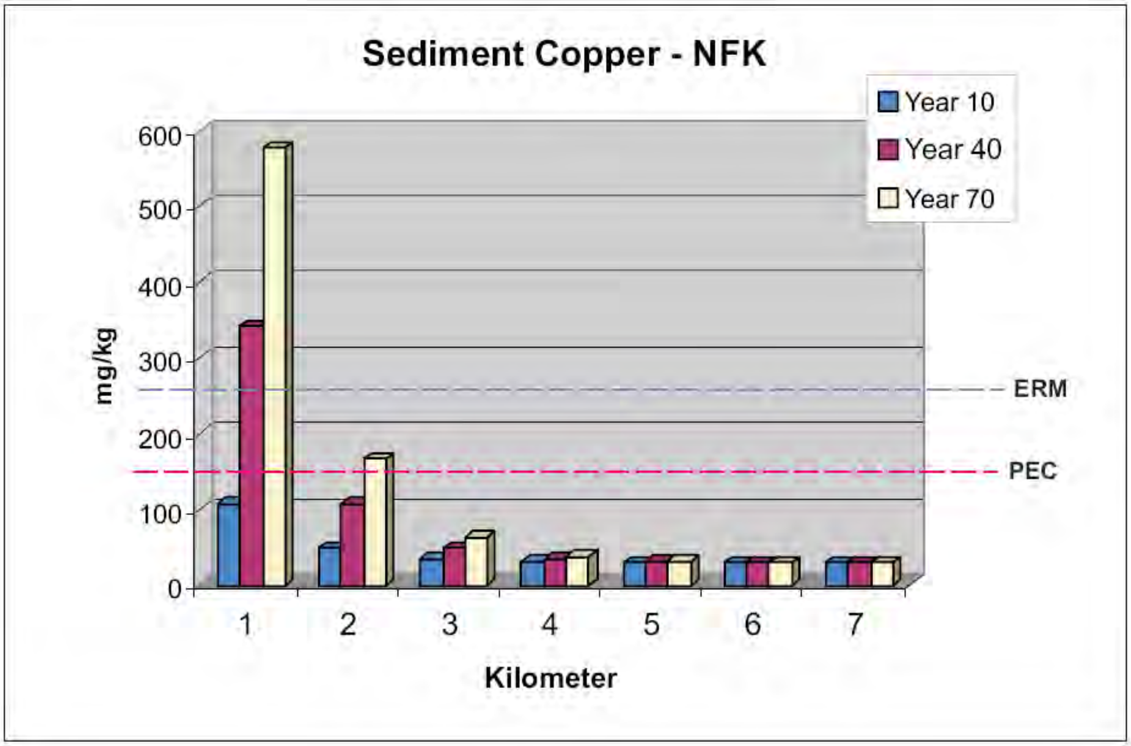
The ERM was developed as an interpretive tool for the NS&T Program (Long *et al.* 1995; Long and Morgan 1991) and represents a concentration above which effects frequently occur. The guidelines were initially intended for use by NOAA scientists in ranking areas that warranted further detailed study on the actual occurrence of adverse effects such as toxicity. The guidelines are not criteria or standards. Their sole intent is for use as informal (non-regulatory) guidelines for interpreting chemical data from analyses of sediments. This data set was developed from toxicity tests on marine benthic organisms and is used only for comparison purposes to the PEC for freshwater ecosystems. Figures 19a and b [assumed deposition/retention: ‘a’ = 25% ‘b’ = 10%], 20a and b and 21a and b show that over the life of the proposed mine dust contributions to sediment copper concentrations should be most pronounced and critical in upstream portions of streams associated with the mine [although this model does not evaluate resuspension and thus downstream sediment copper movement and deposition resulting from annual high flow events]. Again, predicted concentrations were compared to two sediment quality guidelines’ benchmarks related to possible/probable effects to sediment dwelling organisms from copper concentrations.

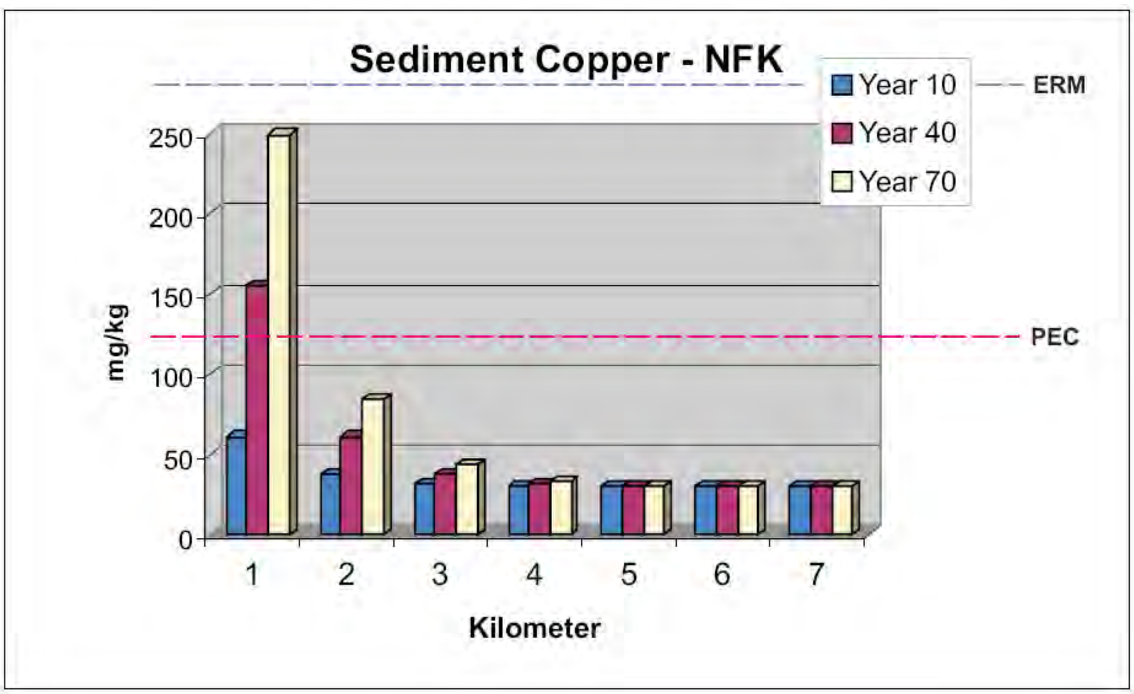
Under both the 25% and 10% dust migration/deposition/retention scenarios (see Table 15), predicted early [e.g., 10 years] mine-stage increases to sediment copper concentrations throughout all watersheds do not appear critical. As the mine ages (years 30-50) and fugitive dust emission impacts are more sustained (e.g., if dust management practices are not implemented), stream concentrations may reach levels where chronic toxicological effects to benthic macroinvertebrates, resident fish and salmon are imminent and acute effects possible. By mid-life stages of the mine (years 30-50), discrete effects to sensitive benthic macroinvertebrates (e.g., mayfly, caddisfly, stonefly) could occur in the most upstream segments where concentrations feasibly could exceed baseline mean concentrations by factors ranging from 11X [NFK] to 72X [UTC].

Table 15. Modeled Predicted Sediment Copper Concentrations (mg/kg) from Dust Deposition within the North Fork Koktuli, South Fork Koktuli, and Upper Talarik Creek (Considering Retention Levels of 25% and 10%)

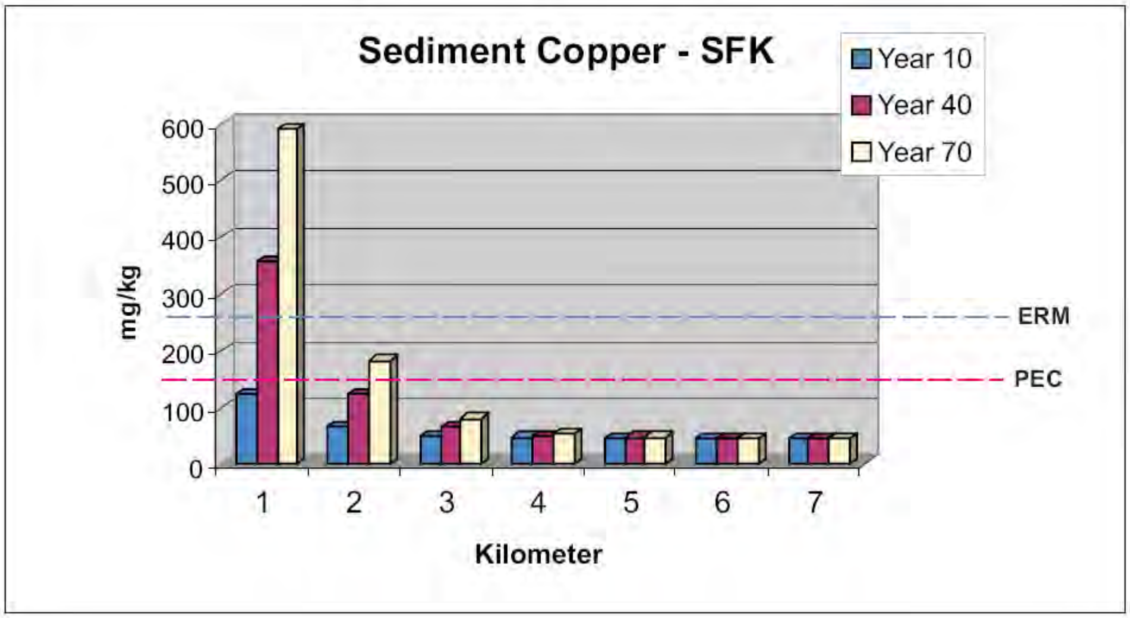
Stream Segment	Baseline Concentration[1]	Year 10		Year 20		Year 30		Year 40		Year 50		Year 60		Year 70	
		25%	10%	25%	10%	25%	10%	25%	10%	25%	10%	25%	10%	25%	10%
North Fork Koktuli															
Km – 1	29.8	107.8	61.1	185.8	92.4	263.8	123.7	341.8	155.0	419.8	186.3	497.8	217.6	575.8	248.9
Km – 2		49.3	37.7	68.8	45.5	88.3	53.3	107.8	61.1	127.3	69.0	146.8	76.8	166.3	84.6
Km – 3		34.7	31.8	39.6	33.7	44.5	35.7	49.4	37.7	54.3	39.6	59.2	41.6	64.0	43.5
Km – 4		31.0	30.3	32.3	30.8	33.5	31.3	34.7	31.8	35.9	32.3	37.1	32.8	38.4	33.3
Km – 5		30.1	29.9	30.4	30.1	30.8	30.2	31.1	30.3	31.4	30.4	31.7	30.5	32.0	30.7
Km – 6		29.9	29.9	30.0	29.9	30.1	29.9	30.1	29.9	30.2	30.0	30.3	30.0	30.4	30.0
Km – 7		29.8	29.8	29.9	29.8	29.9	29.9	29.9	29.9	29.9	29.9	29.9	29.9	30.0	29.9
South Fork Koktuli															
Km – 1	46.1	124.1	77.4	202.1	108.7	280.1	140.0	358.1	171.3	436.1	202.6	514.1	233.9	592.1	265.2
Km – 2		65.6	54.0	85.1	61.8	104.6	69.6	124.1	77.4	143.6	85.3	163.1	93.1	182.6	100.9
Km – 3		51.0	48.1	55.9	50.0	60.8	52.0	65.7	54.0	70.6	55.9	75.5	57.9	80.4	59.8
Km – 4		47.3	46.6	48.6	47.1	49.8	47.6	51.0	48.1	52.2	48.6	53.4	49.1	54.7	49.6
Km – 5		46.4	46.2	46.7	46.4	47.1	46.5	47.4	46.6	47.7	46.7	48.0	46.8	48.3	47.0
Km – 6		46.2	46.2	46.3	46.2	46.4	46.2	46.4	46.2	46.5	46.3	46.6	46.3	46.7	46.3
Km – 7		46.1	46.1	46.2	46.1	46.2	46.1	46.2	46.2	46.2	46.2	46.2	46.2	46.3	46.2
Upper Talarik Creek															
Km – 1	4.4	82.4	35.7	160.4	67.0	238.4	98.3	316.4	129.6	394.4	160.9	472.4	192.2	550.4	223.5
Km – 2		23.9	12.2	43.4	20.1	62.9	27.9	82.4	35.7	101.9	43.6	121.4	51.4	140.6	59.2
Km – 3		9.3	604	14.2	8.3	19.1	10.3	24.0	12.2	28.9	14.2	33.7	16.2	38.6	18.1
Km – 4		5.6	4.9	6.8	5.4	8.1	5.9	9.3	6.4	10.5	6.9	11.7	7.4	12.9	7.9
Km – 5		4.7	4.5	5.0	4.6	5.3	4.8	5.6	4.9	5.9	5.0	6.3	5.1	6.6	5.2
Km – 6		4.5	4.4	4.6	4.5	4.6	4.5	4.7	4.5	4.8	4.6	4.9	4.6	5.0	4.6
Km – 7		4.4	4.4	4.4	4.4	4.5	4.4	4.5	4.4	4.5	4.5	4.5	4.5	4.5	4.5

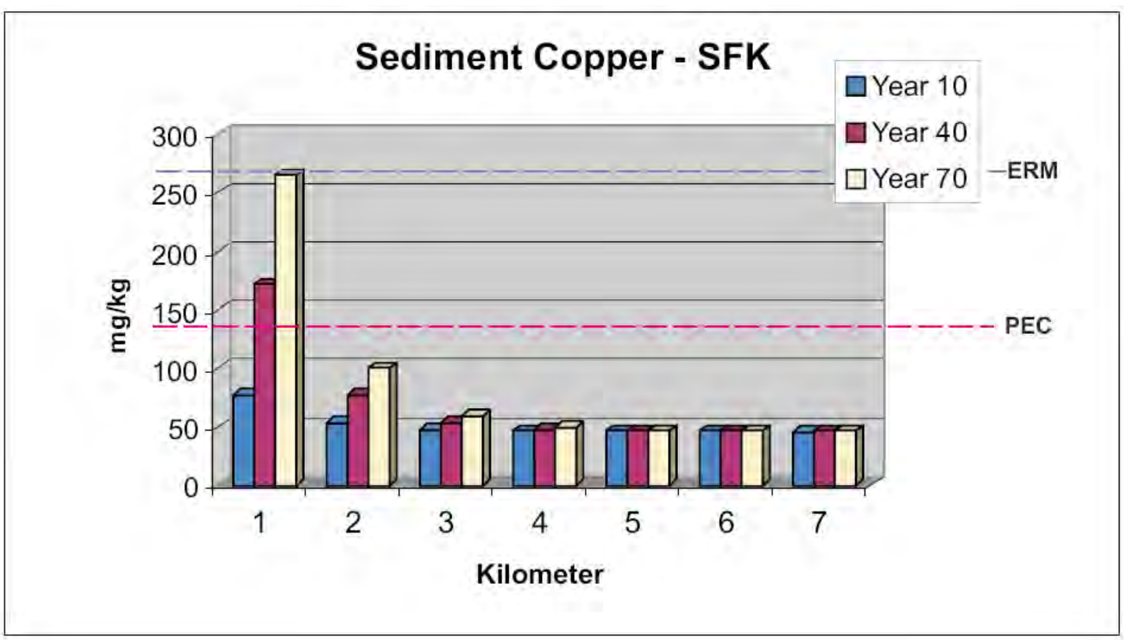
1 = Derived from data in Fey et al., (2008) and from Zamzow (2010).



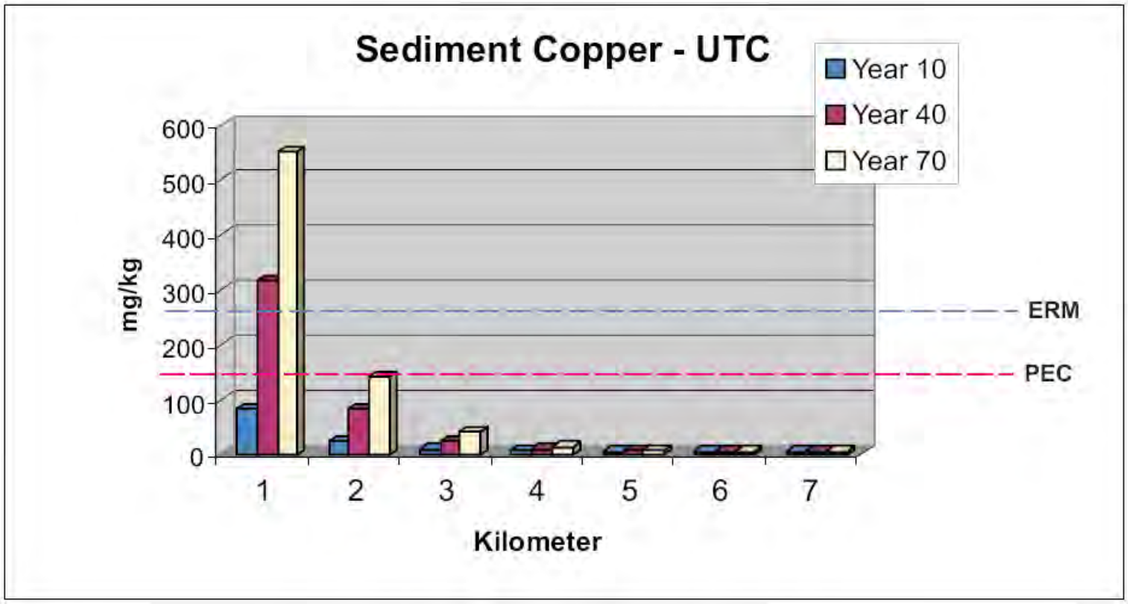


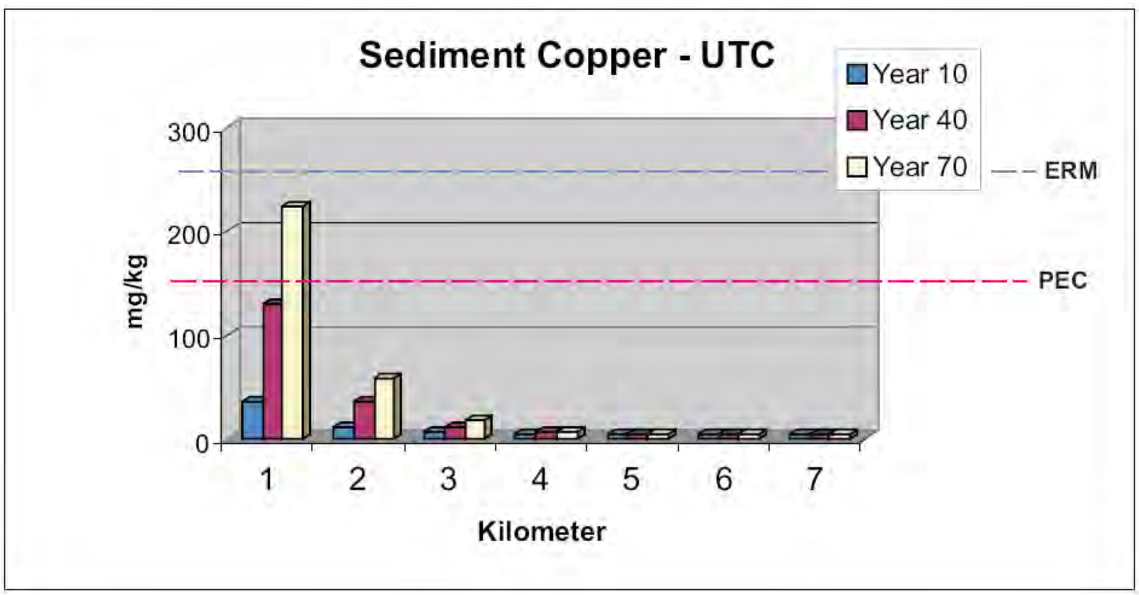
Figures 19a (top - 25%) and 19b (bottom - 10%). Comparison of Estimated Sediment Concentrations from Dust Deposition to Sediment Quality Guidelines in North Fork Koktuli





Figures 20a (top - 25%) and 20b (bottom - 10%). Comparison of Estimated Sediment Concentrations from Dust Deposition to Sediment Quality Guidelines in South Fork Koktuli





Figures 21a (top - 25%) and 21b (bottom - 10%). Comparison of Estimated Sediment Concentrations from Dust Deposition to Sediment Quality Guidelines in Upper Talarik Creek

Malmqvist and Hoffsten (1999) found that copper [and zinc] exposure in sediment resulted in reduced taxa richness, especially for sensitive species of mayfly, stonefly and caddisfly. Bakkala (1970) described benthic organisms, primarily aquatic insects, as the primary food source for chum salmon in freshwater. Based on the approach used in this study, sediment copper concentrations would not be critical in areas away from the mine, even in latter stages of the mine's life. But, this evaluation only considers static dust contributions to sediment loads and does not consider other potential sources of copper such as waste piles, spills, leaks or transport in acid mine drainage, or re-entrainment, transport and deposition in quiescent (slow-moving) portions of a downstream segment. Finally, the model only predicts absolute sediment concentrations based on very conservative assumptions. It is likely that spatial distribution, mixing and periodic flushing of stream channels would reduce real-time concentrations in upstream segments. But, these factors could also result in lengthening of the contaminant pathway and thus extrapolate effects to larger portions of the watershed. Real-world contributions of metal-laden dust in streams would require a long term monitoring program with control sites.

The specific fate of copper deposited into stream sediment is unclear at this time. Copper (and other metals) would reach equilibrium, with sediment copper being continually released into interstitial (pore) water / surface waters and with suspended particulate matter in the water column adsorbing free copper ions to be re-deposited back into the substrate. Studies on a variety of benthic invertebrates indicate that interstitial water concentrations of metals correspond very well with the bioavailability of metals in test sediments (Ankley *et al.* 1994). The bioavailability will depend on many physicochemical factors of both the sediment and overlying waters, including pH. For instance, up to 29 different species of copper can be present in an aqueous solution in the pH range from 6 to 9 (Eisler 1998). Aqueous copper speciation and toxicity will depend on the ionic strength of the water (EPA 2004). The hydroxide species and free copper ions are mostly responsible for toxicity, while copper complexes consisting of carbonates, phosphates, nitrates, ammonia, and sulfates are weakly toxic or nontoxic. Copper in the aquatic environment can partition to dissolved and particulate organic carbon. The bioavailability of copper also can be influenced to some extent by total water hardness (EPA 2004). Water quality changes (i.e., reduced pH) from AMD discharge into watersheds are expected to result in increased potential for bioavailability of copper from sediments, with higher proportions of ionic copper within the water column.

In anaerobic sediments a key phase controlling partitioning of cationic metals such as copper into pore water is acid volatile sulfide (AVS) (Berry *et al*. 1996, ICMM 2002). It is understood that bioavailability (i.e., uptake by organisms and subsequent toxicity) is controlled primarily by the dissolved metal concentration in the sediment porewater. Proponents of this theory contend that using simultaneously extracted metal (SEM) to AVS molar ratios to estimate sediment porewater concentrations for cadmium, copper, mercury, nickel, lead, and zinc (generally present as divalent species) provides a better indicator of sediment toxicity than total metals concentrations on a dry weight basis (DeWitt *et al*. 1996; Hansen *et al*. 1996). AVS is usually the dominant-binding phase for divalent metals in sediment. Metal-sulfide precipitates are typically very insoluble and this limits the amount of dissolved metal available in the sediment porewater. For an individual metal, when the amount of AVS exceeds the amount of the SEM metal (i.e., the SEM/AVS molar ratio is below 1), the metal concentration in the sediment porewater will be low because of the limited solubility of the metal sulfide. Although SEM and AVS have been measured in stream sediments associated with the proposed mine, it is unclear what conclusions can be drawn from preliminary data. No discussion of the SEM/AVS ratios and the potential bioavailability of copper in sediments is currently available. An evaluation of the bioavailability of copper presently in sediments would be important for assessing the potential risks should sediment concentrations increase in these watersheds over the life of a mine.

Leaching

For comparison of leachability and transport potentials at the proposed mine, lead data available for Red Dog Mine (Applied Research & Technology [ART] 2007) was used. Their assays indicate that the lead concentrations in the proximal regions close to the mine and mill were 2.9% for surface samples. Lead and zinc extractabilities determined for samples near the Red Dog Mine were assessed using a diagnostic leaching procedure. Results indicate that approximately 0.3% of the lead is leached, zinc leaching was approximately 4%. Kinetic testing indicated that only one sample had the potential for acidic leaching over a long period, but low-metals leaching was predicted based on kinetic testing. Approximately 30% of the zinc could be extracted under aggressive oxidative conditions of a humidity cell test.

Leachability is dependent on many variables, including the testing method used. Ward *et al*. (2003) compared leachability test methods and element mobility for selected fly ash samples. Generally, copper showed some mobility in acid-generating ashes, but was virtually immobile in alkali-generating samples. Copper mobility generally ranged from 2-6% for all but column leaching methods (Ward *et al*. 2003). The expectation at the proposed mine for soil impacted by ore dust would be similar – soils at the site are considered as having low alkalinity based on data from USGS's 2007 study at the mine site that showed surface soils had a mean pH of 5.4 (range 3.6-7.0) (Fey *et al.* 2008).

In natural media, metal contaminants undergo reactions with solid materials with which the water is in contact. Reactions in which the metal is bound to the solid matrix are referred to as sorption reactions and metal that is bound to the solid is said to be sorbed. The metal *partition coefficient* (K_d) is the ratio of sorbed metal concentration (expressed in mg metal per kg sorbing material) to the dissolved metal concentration (expressed in mg metal per L of solution) at equilibrium. During transport of metals in soils, metal sorption to the solid matrix results in a reduction in dissolved metal concentrations and this affects the rate of overland metal transport to water bodies. During transport of metals in soils-to-surface water systems, metal sorption to the solid matrix results in a reduction in the dissolved concentration of metal and this affects the overall rate of metal transport.

For a particular metal, K_d values in soil are dependent upon various geochemical characteristics of the soil and its porewater. The derived soil partitioning coefficient for copper in Allison and Allison's (2005) study was 2.7; zinc was 3.1. Based on this information, it was assumed that leaching characteristics for copper at the proposed mine would be similar to those found for zinc at Red Dog Mine (e.g., ~4%). Although, soil pH and organic matter can both be critical to determining the partitioning of these and other metals to soil solutions (Leybourne and Cameron 2007). General information in the *Handbook of Chemistry and Physics* (Lide and Frederikse 1998) shows that 3% of copper is extractable from soil at a pH of 4.5. Although the mean pH observed at the proposed site by USGS was 5.4, for this analysis it was predicted that soil pH would decrease as the sulfide ore dust accumulates. Again, ore [dust] composition is important because sulfide ores are expected to form sulfuric acid when exposed to oxygen and water (United States Office of Surface Mining and Reclamation, 2007 and Acid Drainage Technology Initiative 2007).

To determine potential copper concentrations in surface waters as a result of leaching from soil, the following approach was used. First, predicted minimum and maximum annual copper concentrations for the three dust zones, as presented in Table 14, were added (e.g., minimum A+B+C). Next, summed annual values were multiplied by the predicted leachable percentage (e.g., 4% for copper) to determine concentrations that would be expected to be discharged into proximal surface waters. Concentration ranges were evaluated over the life of the mine in order to assess potential impacts from the contribution of metals in stream systems. Finally, due to soil characteristics such as hydraulic restrictions, or other factors including ‘leachable’ distance from a stream, it was assumed that only a small fraction (10%) of the leached copper would reach local surface water bodies. So, estimated concentrations were developed as such:

Annual metals contributions to surface waters = [(soil conc. A+B+C) * (0.04) * # Years] * 0.01

The results of this exercise indicate that dust deposition could contribute approximately 0.242 mg/kg of dissolved copper each year into streams near the proposed mine. Extrapolation of these aggregate concentrations at various stages in the mine’s life (e.g., 10, 40 and 70 years) are presented in Table 16.

Table 16. Future Estimates of Dissolved Copper Contributions to Surface Water near Proposed Mine as a Result of Leaching from Soil

Metal	Annual Contribution (mg/kg)	Year 10	Year 40	Year 70
Copper	0.242	2.42	9.66	16.94

The fate of copper has been discussed previously in Section 3.2. Generally, for copper and other heavy metals in freshwater, the many physicochemical factors of a receiving water body will dictate a metal’s speciation. For instance, in freshwater, the solubility of copper salts is decreased under reducing conditions and is further modified by water pH, temperature, and hardness; size and density of suspended materials; rates of coagulation and sedimentation of particulates; and concentration of dissolved organics (Eisler 1998). The cupric ion is the dominant toxic copper species at pH levels less than 6; the aqueous copper carbonate complex is dominant from pH 6.0 to 9.3 (USEPA 1980). This equilibrium is altered in the presence of humic acids, fulvic acids, amino acids, cyanide, certain polypeptides, and detergents (USEPA 1980). Such chemical speciation affects bioavailability because relative uptake rates can differ among chemical species and the relative concentrations of chemical species can differ among exposure conditions. At equilibrium in oxygenated waters, "free" copper exists as the cupric ion [Cu(II) weakly associated with water molecules], but this species is usually a small percentage of the total copper. Substantial amounts of copper can also be adsorbed to or incorporated into suspended particles. Water quality standards have been developed based on this understanding.

For comparison purposes, surface water samples collected by Zamzow (2010) for ambient water quality were evaluated in order to predict relative increases of leached copper from dust deposition. Table 17 provides a comparison of mean site copper concentrations to ambient water quality criteria derived for the watersheds under investigation. Presently, watershed dissolved copper concentrations constitute from approximately 60 to 80 percent of the total copper found within the water column (e.g., SFK = 63.5%, NFK = 73.8%, UTC = 82.4%). Based on a mean hardness value of 25 mg/L $CaCO_3$ used for determining the copper water quality criteria (WQC; 2.85 µg/L), only one sample for the selected stations in the three watersheds exceeded the copper WQC. The 3.57 µg/L concentration was in the SFK where, generally, concentrations were higher than the other two watersheds. The potential effect of continuous contributions of dissolved copper into stream systems is expected to result in long term degradation of water quality, especially considering that the exposure and oxidation of sulfides in both dust and other mine sources will likely result in acid being generated and thus pH being reduced within proximal water bodies, especially during latter stages of the mine's life, and beyond. Yim *et al.* (2006) showed that reduced surface water pH from AMD will likely result in lower hardness concentrations. In their study, approximately four to five times higher toxicity was observed in 'soft' rather than 'hard' water test solutions. Although the expected impacts may not be as readily distinguishable in site water bodies due to the already soft water conditions, continued reduction of the buffering capacity along with increased metals concentrations could result in various long term systemic and behavioral impacts to salmon.

This would be most pronounced in upstream portions of the watersheds where dilution from non-affected surface runoff and groundwater is unavailable [due to proposed water extraction]. Section 3.1.3, Tables 4, 5 and 6, show that flow reduction would be most pronounced down to Stations 3 in the NFK and SFK watersheds, with greatest reductions in flow volume apparent at Station 2 in UTC watershed (see Figure 6). Downstream portions of all watersheds would most likely show reduced/limited impairment from this source as a result of dilution from inflowing tributaries during the mine's life. Several studies (Abraham *et al*. <u>no date</u>, Bidhendi *et al*. 2007, Malinovsky *et al*. 2002, Beltman *et al*. 1999) have noted that 'general' mining impacts on surface water quality are ameliorated in downstream reaches as a result of dilution, as well as increased complexation capacity. Although, it is important to understand that reduced surface water concentrations may not in itself indicate that downstream segments will be free from mining influences. For a mine in the Philippines, David (2003) found that although water quality conditions showed improvement at downstream stations (e.g., pH and dissolved copper concentrations were comparable to reference streams), elevated copper concentrations in caddisflies suggested that other potential pathways for metals, such as through contamination of a food source (algae), may be as important. Similarly, Brumbaugh *et al*. (1994) noted that although metals' concentrations in surface sediments decreased gradually downstream in the Clark Fork River, they increased similar to the uppermost river stations at stations located within the Milltown Reservoir. Review of their data indicates that this characteristic was most likely a result of extensive downstream depositional areas resulting in higher metal concentrations. In addition, they also observed spikes in metal concentrations during floods and multiple fish kills have been observed due to the extensive contamination deposited downstream of Milltown Reservoir (Brumbaugh *et al*. 1994).

NOAA provided benchmark concentrations (BMCs) for sensory effects to juvenile salmonids exposed to dissolved copper (Hecht *et al*. 2007). They suggest BMCs ranging from 0.18 to 2.1 µg/L above background (e.g., background was defined by NOAA as surface waters with less than 3 µg/L dissolved copper) which correspond to the ability of juvenile salmonids to avoid predators in freshwater. These concentrations for juvenile salmonid sensory and behavioral responses fall within the range for other sub-lethal endpoints (i.e., behavior, growth, primary production) affected by dissolved copper concentrations of 0.75 to 2.5 µg/L. Relative to these BMCs for potential behavioral effects to juvenile salmonids, watershed dissolved copper concentrations (see Table 17) ranged from 0.05 µg/L to 3.57 µg/L. Thus, based on comparison, even a small increase in dissolved copper above observed background concentrations could result in sub-lethal effects to rearing juveniles throughout the watersheds.

Table 17. Comparison of Copper Concentrations[1] in Surface Waters near the Proposed Mine to Ambient Water Quality Criterion

Sample Location	Copper		Sample Location	Copper		Sample Location	Copper	
	T	D		T	D		T	D
UTC-01-01	0.22	0.16	SFK-01-01	0.27	0.18	NFK-01-01	0.43	0.27
UTC-01-02	0.22	0.17	SFK-01-02	0.27	0.19	NFK-01-02	0.41	0.28
UTC -01-01	0.15	0.12	SFK-01-03	0.25	0.2	NFK-01-03	0.38	0.26
UTC -02-01	0.48	0.23	SFK-01-01	0.83	0.56	NFK-01-01	0.21	0.21
UTC-02-02	0.48	0.22	SFK-01-02	0.78	0.59	NFK-01-02	0.3	0.3
UTC -02-03	0.5	0.23	SFK-01-03	0.79	0.57	NFK-01-03	0.2	0.2
UTC -02-01	0.27	0.19	SFK-01-04	0.8	0.57	NFK-02-01	0.2	0.2
UTC -02-01	0.2	0.2	SFK-11-01	2.7	0.82	NFK-11-01	0.12	0.08
UTC -03-01	0.2	0.2	SFK-11-02	2.42	0.78	NFK-11-02	0.13	0.07
UTC -03-02	0.2	0.2	SFK-12-01	5.6	1.89	NFK-11-03	0.11	0.07
UTC -03-03	0.2	0.2	SFK-12-01	2.2	1.4	NFK-11-01	0.41	0.15
UTC -03-04	0.3	0.2	SFK-12-02	2.5	1.4	NFK-11-01	0.5	0.1
UTC -11-01	0.2	0.17	SFK-12-03	2.1	1.7	NFK-21-01	0.35	0.27
UTC -11-02	0.21	0.17	SFK-12-04	2.2	1.4	NFK-21-02	0.31	0.27
UTC -11-03	0.2	0.16	SFK-21-01	0.15	0.09	NFK-21-01	0.29	0.29
UTC -11-01	0.15	0.1	SFK-21-02	0.16	0.09	NFK-21-01	0.2	0.19
UTC -11-01	0.1	0.1	SFK-21-03	0.15	0.09			
UTC -31-01	0.05	0.05	SFK-31-01	5.31	***3.57***			
UTC -41-01	0.1	0.1	SFK-31-01	4.4	2.5			
UTC -41-02	0.1	0.1	SFK-41-01	0.2	0.2			
UTC -41-03	0.1	0.1	SFK-51-01	1.3	0.9			
UTC -41-04	0.1	0.1						
UT-01-01	0.22	0.16						
UT-01-02	0.22	0.17						
ADEC Water Quality Criterion[2]		**2.85**			**2.85**			**2.85**

T = Total concentration / D = Dissolved concentration
1 = All concentrations in µg/L (ppb)
2 = Calculated at hardness of 25 mg/L

Predicted Effects to Salmon

The models predict that, without treatment measures, dust generated at the mine would result in metal-laden soils, with transport mechanisms resulting in continuous, long-term contamination of local surface waters that support multiple salmon life stages. Although the preceding discussions may present an overly simplistic approach to evaluating impacts from dust generated by the proposed mine, a certainty exists that, even with mitigation measures employed at the mine, copper and other metals will likely be mobilized in runoff or leached into surface and/or groundwater over the 40-70 year life of the mine. The actual amount may be higher or lower than predicted, but current ambient metals' concentrations in surface waters within the watershed indicate that any increase in dissolved metals' fractions could result in negative effects to the most sensitive salmon life stages. A study of recently permitted large sulfide-based copper and gold mines found that mining <u>often</u> increases the concentrations of copper and other pollutants in ground and surface waters to levels that are toxic to fish and other aquatic life (Kuipers *et al*. 2006). It is fairly certain then that <u>some</u> of the copper sulfide ore dispersed as fugitive dust will degrade to copper sulfate, with some percentage of this copper sulfate conveyed in runoff to surface water or seeped into the soil to become groundwater. Most importantly, the chronic metals' contributions to surface waters from dust generated at the mine would act to compound other physical (habitat loss, flow reduction) and chemical (AMD) impacts expected from the mine's creation and operation.

At the elevated concentrations predicted, it is presumed that salmon would be exposed to copper through three primary routes; 1) directly, through olfactory bulbs (Hansen *et al*., 1999b); 2) gill uptake of waterborne free cupric ions (Taylor *et al*. 2002); and 3) biotransfer from food resources (Dallinger *et al*. 1987). It is expected that the first and second route would be the primary mechanisms for copper exposure. But, Clearwater *et al.* (2002) suggests that contrary to popular belief, the relative efficiency of copper uptake from water and diet is very similar when daily doses are compared, rather than comparison of copper concentrations in each media. EPA (1999) recommended a water-to-fish bioconcentration factor (BCF) of 710 for copper, but little evidence exists to support the concept of biomagnification of copper in aquatic environments.

It is uncertain how salmon will be affected from anthropogenic inputs of copper in light of the naturally high copper concentrations within the Nushagak and Kvichak river drainages. Marr *et al.* (1995) showed that brown trout acclimated to elevated mixtures of metals (including copper) suffered fewer mortalities than unacclimated populations. But, their study also indicated that the potential for increased tolerance to metals was related to the mediating effect of dissolved organic carbon and hardness on chronic and acute copper toxicity. Also, Marr *et al.* (1995) documented the association between reduced growth and increases in metallothionein during acclimation. Other studies have shown similar correlations between increased metallothionein and reduced growth (Dixon and Sprague 1981; Roch and McCarter 1984).

3.2.3 Slurry Pipeline Breaks and Spills

3.2.3.1 Stressor Description

Slurry pipeline breaks and spills occur frequently in spite of governmental regulations and oversight. Aquatic life would be adversely affected by a spill of copper concentrate slurry into any of the 89 streams crossed by the slurry pipeline. For example, large numbers of Newhalen River and Lake Clark sockeye stocks could be impacted by a pipeline break which spilled copper concentrate and contaminated water into the Newhalen River during adult spawning migration, or smolt out migration. Rearing salmon in the Newhalen River and Iliamna Lake would be directly and indirectly impacted by dissolved copper which is toxic to both fish and the planktonic food organisms that juvenile sockeye salmon feed on.

Adult salmon attempting to enter the Newhalen River might be injured or killed by copper levels in the river or abort their spawning run up river. Depending on the size, time and location of a pipeline spill, a slurry pipeline break could impact thousands to hundreds of thousands of adult salmon and high value resident fish, and hundreds of thousands to millions of juvenile fish.

3.2.3.2 Impact Determination

Slurry pipeline spills appear to be common in the mining industry. Information on pipeline spills at the Phelps Dodge Chino Mine indicated that 45 spills had been reported by the State of New Mexico between 1990 and 2001. Three large distinctive pipeline spills included an approximate 480,000-gallon spill, an approximate 18,000-gallon spill in 2000 and a spill of approximately 20,000 gallons in January 2001 (NMED 2003). For similar spill information, Environment Canada (http://www.ec.gc.ca/) provides web-based access to National Spill Statistics for major spills from various industries. Data from 1984-1995 indicates that the mining industry is classified as a business sector for which large spills are regularly reported.

As an example of the potential impacts from pipeline spills, in 1966, a year after open pit mining began at Chevron's Molycorp mine, a baseline water quality survey of the Red River determined that for the river segment adjacent to Molycorp mine, quality was high. In November 1971, USEPA conducted a study of the Red River and concluded that the chemical quality and biological conditions of the Red River remained very good, but that occasional breaks in the Molycorp tailings pipelines resulted in some degradation of river quality. Also in the early 1970s, during routine fish population studies in the middle reach of the Red River, the New Mexico Game and Fish Department discovered that once-thriving populations were conspicuously absent. Beginning in the early 1980s, EPA and the Bureau of Land Management (BLM) began documenting major impacts to the Red River due to mining and mining-related activities. In 1992, the New Mexico Water Quality Control Commission submitted a report to Congress documenting elevated levels of numerous metals within the vicinity of the Molycorp mine, including cadmium, copper, lead, silver, and zinc. [source: http://www.epa.gov/superfund/sites/npl/nar1599.htm] Although the Molycorp mine pipeline was only 10-miles long, it has been documented as having over 100 spills, many within sensitive riverine systems.

From the information presented above, proposed mine pipeline spills feasibly could result in from 20,000 to over 500,000 gallons of metal-laden slurry being deposited into sensitive anadromous streams. Generally, impacts from small spills would be similar in perennial streams such as the Newhalen River and Iliamna River, with fine-grained slurry particles being quickly entrained in flowing waters and transported downstream. The degree to which spilled slurry would be dispersed and transported into downstream environments will depend on the mobility of the slurry particles within the flowing system. Although there is a body of knowledge concerning mobility of coarse-grained, non-cohesive sediments such as sands, understanding of the potential for mobility of fine-grained cohesive sediment particles is less certain. Early investigations by sedimentologists (Shields 1936) revealed that sediments of varying particle size begin to move at different critical velocities (Varoni 1964). A model developed by Middleton and Southard (1977) relates *Reynolds* number (*Re*) to current velocity (*U*), diameter of the particle (*D*), the fluid density (*P*) and the dynamic viscosity (*u*) as such;

$$Re = UDP/u$$

Several modifications to this original model (Blatt *et al*. 1980; Newbury 1984) suggested that particles from 1 to 100 mm diameter deviated significantly from Shield's (1936) prediction for critical velocities. The commonly used Shields diagram plots Shields Number (dimension-less critical shear stress) versus stream *Reynolds* number. The velocities needed to mobilize sand (grain size 0.05 to 0.5 mm) are typically lower than for other sediment particle sizes. As particle size increases from sand to boulders (0.5 to 1,000 mm), the critical velocity increases due to the increase in mass (Figure 22). As particle size decreases from sand to silt and clay (0.05 to 0.001 mm), the critical velocity also increases if the material is consolidated because of the adhesive properties of the fine particles. For unconsolidated silts and clays, the critical velocity remains similar to that for fine sands. Table 18 lists the velocities allowable by United States Army Corps of Engineers (USACE) Guidance (USACE 1994) for stable stream systems with different channel materials.

Only major floods will generate the velocities required to disturb the consolidated silts or boulders. Although greater velocities and shear stress values are required to transport these smaller particles, fine sediments are also cohesive and will normally be eroded as floccules rather than individual particles, further discouraging their detachment (Richards 1982). The two primary types of fine sediment transport can be identified as (1) along the surface of the substrate as bedload rolling or sliding, and (2) as turbulence increases, the weight of the particle may be upheld as suspended load by a succession of eddy currents (Petts and Foster 1985). Deposition of silts and clays occurs when trajective forces are less than the settling velocity (gravitational forces) exerted upon the grain, as expressed by Stokes's Law (Richards 1982). For particles larger than 0.1 mm, the relationship between grain diameter and fall velocity is nonlinear due to the influence of inertial forces (Wood and Armitage 1997).

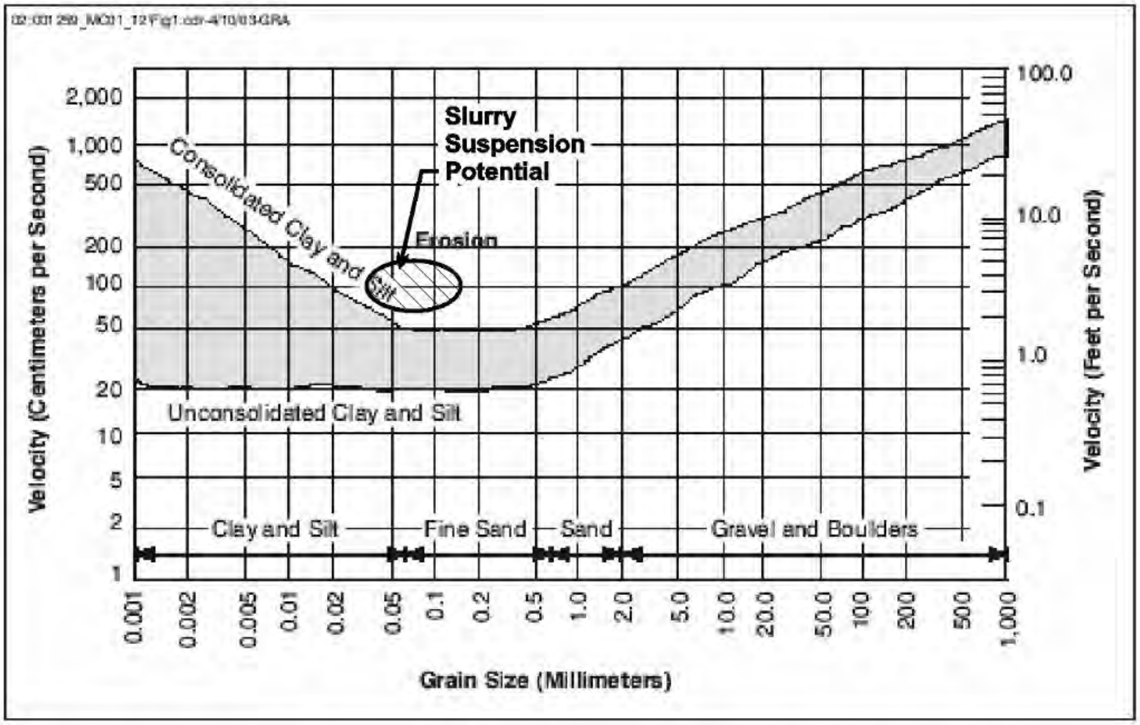


Figure 22. Critical Velocities for Movement of Sediment Particles
Source: *Environmental Management*, Vol. 14, No. 5, Reice *et al*. 1990

Table 18. USACE Allowable Channel Velocities for Various Sediment Grain Sizes

Channel Material	Mean Channel Velocity (fps)
Fine Sand	2
Coarse Sand	4
Fine Gravel	6
Sandy Silt	2
Silt-Clay	3.5
Clay	6

Again, a characteristic that will affect the mobility of the slurry released into stream systems is the behavior of the slurry material itself. The physical characteristics (size) of slurry particles, which are primarily crushed ore, are predicted to be those that will pass through sieve sizes of 100 to 400, or 37 to 149 microns (Table 19). Also, based on conversations with mining engineers at various commercial companies, slurry is predicted to be approximately 50-60% solids.

Table 19. Particle Size Estimates of Ore in Slurry

U.S. Sieve Size	Tyler Equivalent	Opening	
		micron	inch
No.100	100 Mesh	149	0.0059
No. 120	115 Mesh	125	0.0049
No. 140	150 Mesh	105	0.0041
No. 170	170 Mesh	88	0.0035
No. 200	200 Mesh	74	0.0029
No. 230	250 Mesh	63	0.0025
No. 270	270 Mesh	53	0.0021
No. 325	325 Mesh	44	0.0017
No. 400	400 Mesh	37	0.0015

In a first step for predicting the physical fate of ore particles in a stream should the slurry pipeline break or rupture at a crossing, stream velocity data was obtained from ADNR for the Newhalen River near Iliamna, AK. This information indicates that velocity was approximately 3.15 fps during October of 1984 (ADNR 1984). Based on seasonal flow information for the streams under investigation (see Figure 7), and assuming monthly flow variances would be similar in the Newhalen River, it was predicted that the October velocity data represented nominal rates, with expected moderately higher and/or lower rates during other months throughout the year. Finally, the slurry particle size (range) was plotted against velocity on Figure 22 (see notation: *Slurry Suspension Potential*). This evaluation predicts that slurry particle sizes would be held in suspension as a result of the velocity of the Newhalen River. Although it is understood that the author's (Reice *et al*. 1990) intent for the graph was to predict particle mobility from sediment, it was felt that the theory behind particle size to velocity was relevant for predicting entrainment after a spill, especially considering that due to its less viscous (i.e., 40-50% water) nature the released slurry would be mobilized more readily than sediment material.

Based on this understanding of the physical fate of slurry particles, a prediction was made regarding the spatial extent expected from a spill. For comparison, studies directed at responses to coal slurry spills near waterways were reviewed. Refuse-coal slurry has similar size properties as ore slurry (e.g., <149 microns; Parekh 2007). As a result, information on travel distances for coal spills can be directly correlated with those expected for ore slurry. In 2000, 250 million gallons of coal slurry was discharged into two first-order streams in Martin County, Kentucky. Subsequent discharge downstream resulted in slurry contamination for over 70-80 miles (Armstead 2009). The predominant evidence/impact of the coal spill occurred in smaller channels where coal slurry was several feet thick. In larger downstream channels slurry deposits were less obvious and generally oriented in depositional areas.

For a nominal spill into the Newhalen River of 100,000 to 200,000 gallons, it is expected that initial impacts would be from the large volume of slurry being deposited directly into the stream channel. ADNR (1984) noted that although the Newhalen River is

300-350 feet wide, during their survey it was only about 3 feet deep. At 50-60% solids in the transported slurry, this would equate to approximately 60,000 to 120,000 gallons of ore solids being deposited in the stream channel. This would mean that from bank to bank, 13 to 26 linear feet of the Newhalen River could be completely clogged with contaminated slurry. Based on the seasonal flow structure, slurry would most likely end up as depositions or 'sand bars' in the vicinity of the spill with water flowing over and around it, systematically entraining ore particles to be swept downstream and re-deposited as velocities fall below entrainment thresholds. Primary physical impacts then would be embeddedness in riffle habitats proximal to the spill site, with increased turbidities potentially resulting in fish gill abrasion. In addition, habitat quality would be diminished from increased turbidities, lost riparian habitat and equipment leaks and spills during clean-up activities which could last for weeks to months. Except for spills associated with cyanide, fish kills from acute metals' exposure have not been reported for hard rock mines.

Metals' concentrations in proposed mine slurry solids have been estimated at 30% copper, 27% iron, and 33% sulfur, with smaller quantities of gold, molybdenum and other heavy metals (Phelps Dodge 2007). Based on the scenario presented above, salmonids and other biota downstream of the spill site would be exposed to 30,000 to 36,000 pounds of copper and large amounts of other heavy metals. Based on review of spills in other flowing systems, recovery of most of the spilled material would not be possible. The bulk of the spilled slurry would remain in the stream to be transported and chemically incorporated into the sediment. Oxidation of the large volume of copper (and other metals) is also expected, with decreases in pH and increases in relative bioavailable copper within the water column. Subsequent short and long term effects would be expected. As with the Molycorp example presented previously, degradation of the ecological complex would be expected. Direct and indirect (i.e., sub-lethal) impacts on salmonids, especially eggs and fry development in redds, would most likely continue for an extensive period after the spill. Increased copper and other metals' concentrations would result in increased total and dissolved metal fractions in both surface water and sediment interstitial (pore) water. Biouptake and transfer within food chains would result from exposure of forage fish species and benthic macroinvertebrates to both water and sediment metals' concentrations. It is expected that the relatively moderate flow regime and flood events in the Newhalen River would cause slurry to be transported all the way to Iliamna Lake which is approximately 20 miles downstream from where the pipeline is proposed to cross the river. Additional USGS (Station 15300000) velocity data (1981-1982) for the river indicates that reduced flow (3040 cfs @ velocity of 0.940 fps) can occur in the spring, suggesting that a spill during this period would result in less material being transported downstream, with impacts being more localized at the spill site.

Only one anadromous stream crossed by the proposed pipeline has been addressed in detail by the preceding analysis. It is feasible that due to the length of the pipeline, the rugged terrain crossed, and the fact that pipelines are known to rupture and spill their contents, over the life of the project spills would likely occur and some could impact salmon-bearing streams. Our analysis suggests that impacts would most likely be exacerbated in smaller streams. Impacts to salmon viability in these streams would be critical because of the lack of tributaries which may dilute or reduce exposure concentrations downstream. Also, several of these noted in Table 11 which are in the midsection of the pipeline route are located only short distances upstream from Iliamna Lake where millions of sockeye fry rear one to two years before migrating to sea (see Figure 14). As such, long term risks to sockeye salmon would be present if spills occurred at these stream crossings.

USGS 2008 velocity data for the Iliamna River (Station 15300300; 0.78 fps [Oct] to 2.84 fps [July]) near Pedro Bay, AK, was reasonably similar to that found for the Newhalen River, although flow (i.e., discharge) was much less. Based on the physical characteristics of the Iliamna River (i.e., width ~135 ft.), it is expected that water depths during nominal flow periods would be similar to the Newhalen with slurry transport characteristics also similar. The potential for slurry transport to Iliamna Lake would be more critical in the Iliamna River because the distance to the lake from the pipeline crossing is significantly less (~8-10 mi; see Figure 14).

3.2.4 Episodic and Large Scale Pollution Event(s)

3.2.4.1 Stressor Description

A failure of one of the tailings dams planned for the proposed mine would have both short and long term impacts on receiving waters, with severity dependent on dam release volume, timing, and location. A tailings dam failure due to a structural failure, flood or earthquake would release both potentially toxic water downstream along with tons of silt-like mine wastes. This silt would clog gravel in stream riffle areas and make the water in these clear water streams opaque. Generally, during an initial tailings release event there is a highly toxic plume associated with dissolved contaminants, but the long-term threat is the ‘as yet’ unoxidized metal sulfide that is deposited as a solid. This sulfide material would continue to oxidize over time, especially material that is in proximity of the seasonal water table, and can accumulate metal salts which are released all at once when flushed by the next water event. Streams such as Silver Bow Creek, MT, and Clark Fork River, MT, still exhibit fish kills and impacts to other aquatic life below stretches that have been historically contaminated by mine tailings (USEPA 2010).

The failure of one of the proposed mine tailings dams, containing billions of tons of potentially reactive mine waste and hundreds of billions of gallons of contaminated water, could theoretically impact aquatic life as far downstream of Iliamna Lake or out into Bristol Bay. Mine tailings would be washed downstream, and when exposed to oxygen, would release acid and heavy metals. A major failure of a tailings storage facility could kill adult salmon and high value resident fish depending on when and where the spill occurred. Fish production might be permanently eliminated or impaired in the stream impacted by the spill. Planktonic food organisms that are the food source for juvenile sockeye salmon in Iliamna Lake would decline if copper levels in the Lake increase as result of a spill (Roch *et al*. 1985). Depending on concentrations of heavy metals in the water column, returning salmon that encounter heavy metals in a spill plume might incur olfactory damage or be deterred from their upstream migration (Goldstein *et al.* 1999).

3.2.4.2 Impact Determination

Understanding and predicting the magnitude and effects of ore releases from episodic and large scale pollution events are difficult. Ore releases from dust deposition and pipeline spills have been addressed, but the most critical release would be if a tailings dam were breached or failed. World-wide, thousands of spills directly related to mine tailings dams' failures have occurred. In the past two decades, major spills have been reported in Central and Southeast Asia, Australia, Africa, and North and South America. Recent spills in Europe have occurred in Sweden, Spain, Italy, Portugal, Bulgaria, Estonia and the U.K. The website, http://www.wise-uranium.org/mdaf.html, provides tabulated data on over 100 tailings dam failures for uranium and non-uranium mines world-wide since the 1950s (Appendix E: Table E-1). Non-uranium mines, including hard rock copper mines, have resulted in spills into the millions of tons and more. Impacted streams ranged from those immediately downstream of the mine to large rivers over one hundred kilometers away from the mine site. In the U.S., since the 1970s, spills resulting from impoundment failure, embankment failure, slope instability or earth movement have ranged from cumulative annual volumes of 10 to 179 million gallons.

Rico *et al.* (2008) provided a detailed search and re-evaluation of the known historical cases of tailings dam failures world-wide. Their review of the dam failure databases (i.e., International Commission on Large Dams [ICOLD], U.S. Commission on Large Dams [USCOLD], USEPA and UN Environmental Program) revealed that 147 cases of world-wide tailings dam disasters have occurred. In Europe, the most prevalent of the 15 different failure causes was associated with unusually high rain events. They also noted that failures attributed to weather events (including rainfall, hurricanes, rapid snowmelt and ice accumulation in tailings dam) may also be associated with overflow/overtopping, seepage, foundation failure or bad impoundment management. Outside of Europe, seismic liquefaction ranked as the second cause of tailings dam failure. They noted that over 90% of the incidents occurred in active mines.

Relative to impacts that may occur in stream systems near mines, Hudson-Edwards *et al.* (2003) reported that, for a 1998 tailings dam breach in Spain, although clean-up efforts attempted to remove much of the 5.5 million m^3 of acidic water and more than 1.3 million m^3 of contaminated tailings, sediment contamination still remains above pre-spill concentrations. Much of the highly contaminated sediment remaining in the floodplain and channel still contains a large proportion of tailings-related sulfide minerals which are potentially reactive and may continue to release contaminants to the river system. Similarly, a tailings dam failure in 1950 in the New World Mining District, Montana resulted in metal-rich sediment being deposited within high flood-plain levels along Soda Butte Creek (Marcus *et al*. 2001). The earthen impoundment dam failed releasing ~41,000 m^3 of water and an unknown mass of tailings. During the 1990's, mean copper and lead concentrations in floodplains deposits were about one order of magnitude greater than pre-mining concentrations. Their evaluation noted that there was no significant downstream trend in particle size, sorting or deposit thickness, which they deemed consistent with rapid deposition during a brief sediment-charged flood. Metal concentrations were found to decrease exponentially downstream, most likely a result of dilution by uncontaminated sediment entrained in the flood. But, they noted that copper concentrations in some depositional layers exceeded 1000 mg/kg as far as 16 km below the impoundment (Carolan 1997). These concentrations had remained unchanged even after major floods in 1995, 1996 and 1997, as a result of snowmelt runoff (Marcus *et al*. 2001).

It appears that catastrophic releases from tailings ponds can happen at any stage of a mines' life, and the preponderance of information indicates that dam failures occur at older impoundments. But, Davies (2002) noted in his paper on tailings impoundment failures that even relatively young dams (5-20 years) built in the 'modern age' of engineering have also failed. So, the prospect that one of the proposed mine's tailings pond dams will break early in its life is just as relevant as for those at older-engineered mines. The proposed mine would employ two tailings ponds with total storage capacity of 2.5 billion tons. This equates to approximately 3.29 billion cubic yards, or 2.5 billion cubic meters of tailings. Based on the proposed storage capacity expected for the two tailings ponds, total volumes were determined for each pond (e.g., 'Pond' A = 82% or 2.064 billion cubic meters; 'Pond' G = 18% or 0.450 billion cubic meters). Pond A has been proposed to have 3 dams, each approximately 700 ft (213 m) high; Pond G has been proposed to have one dam approximately 450 ft (137 m) high, with a second smaller dam (i.e., 150 ft high).

Rico *et al.*'s (2008) analyses of tailings dam failure relative to dam height (Figure 23) showed that ~56% occurred in dams over 49 ft (15 m), with ~22% of incidents in dams higher than 98 ft (30 m). Again, the tailings dams at the proposed mine are over 700, and near 450, feet high. In contrast, dams at Fort Knox and Red Dog (both in Alaska) are 330 feet and 177 feet high, respectively, and are not in seismically active areas. For purposes of this assessment, it was necessary to devise a reasonable method to predict the size of a potential release from either of the proposed mine's tailings ponds. Rico, Benito and Díez-Herrero (2008) compiled information on historic tailings dam failures and examined correlations between tailings ponds geometric parameters (e.g., dam height, tailings volume) and hydraulic characteristics of floods from dam failures. They showed a strong correlation ($r^2 = 0.86$) between tailings' volume at the time of a failure and tailings outflow volume (see Figure 24 below; and Eq. 7 – $V_F = 0.354 \text{ x } V_T^{1.01}$ *from* Rico, Benito and Díez-Herrero 2008). The volume of spilled tailings was also correlated with its run-out distance ($r^2 = 0.57$) (see Figure 25 below; and Eq. 5 – $D_{max} = 1.61 \text{ x } (HV_F)^{0.66}$ *from* Rico, Benito and Díez-Herrero 2008), and they state that "*in average, one-third of the tailings and water at the decant pond is released during dam failures*". An envelope curve drawn to encompass the majority of data points, estimated potential maximum downstream distance affected by a tailings' spill. Although they indicated that predictive application of the described regression should be treated with both caution and support of on-site dam measurements, they suggest that their method may provide a universal baseline approximation on tailings outflow characteristics (even if detailed dam information is unavailable), which is of great importance for risk analysis purposes.

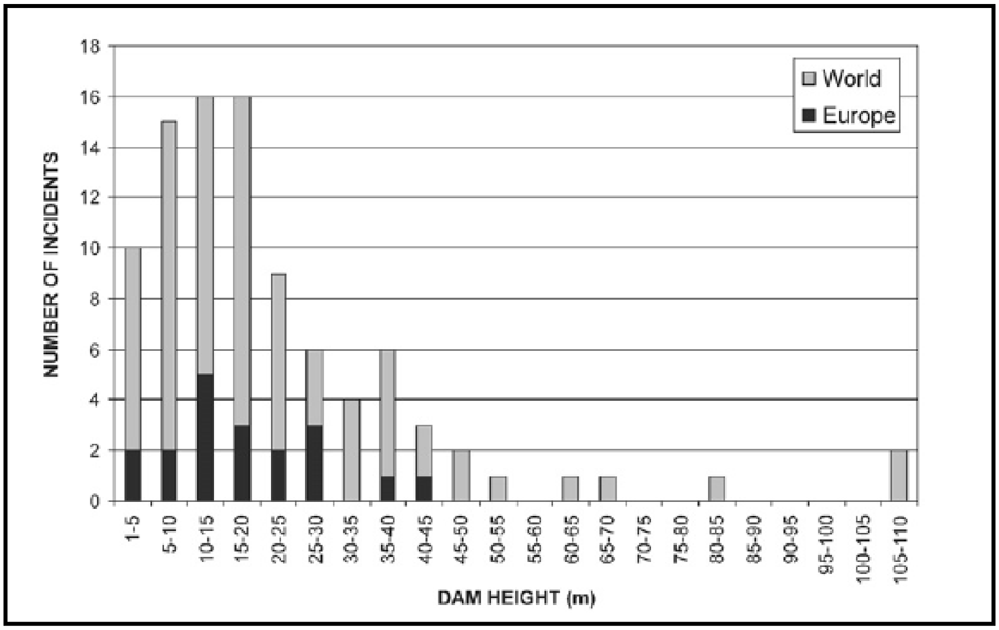


Figure 23. Distribution of Number of Incidents Related to Dam Height (from Rico *et al.* 2008)

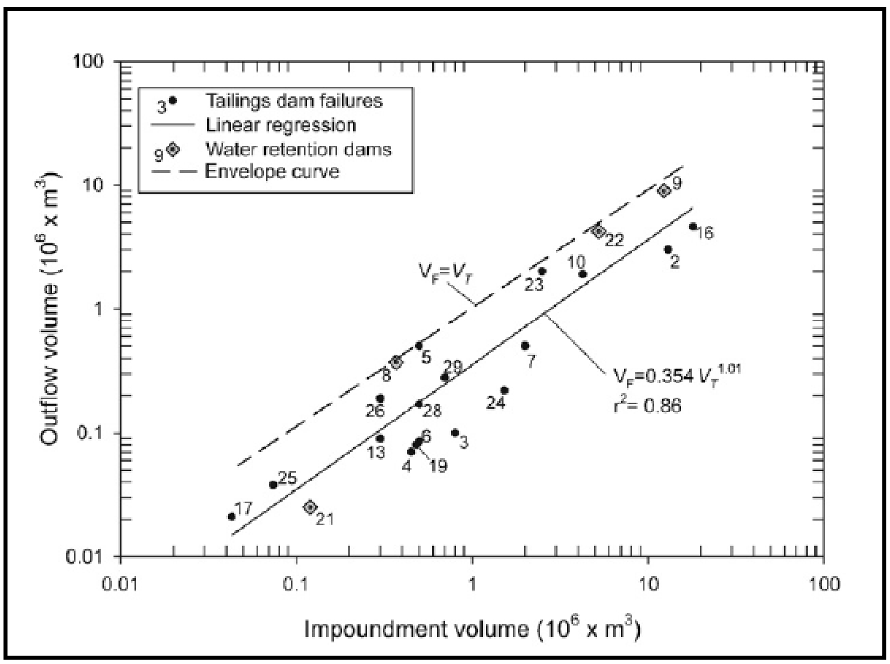


Figure 24. Graph Showing the Tailings Outflow Volume from the Tailings Dam vs. the Volume of Tailings Stored at the Dam at the Time of the Incident (Fig. 4 in Rico, Benito and Díez-Herrero, 2008)

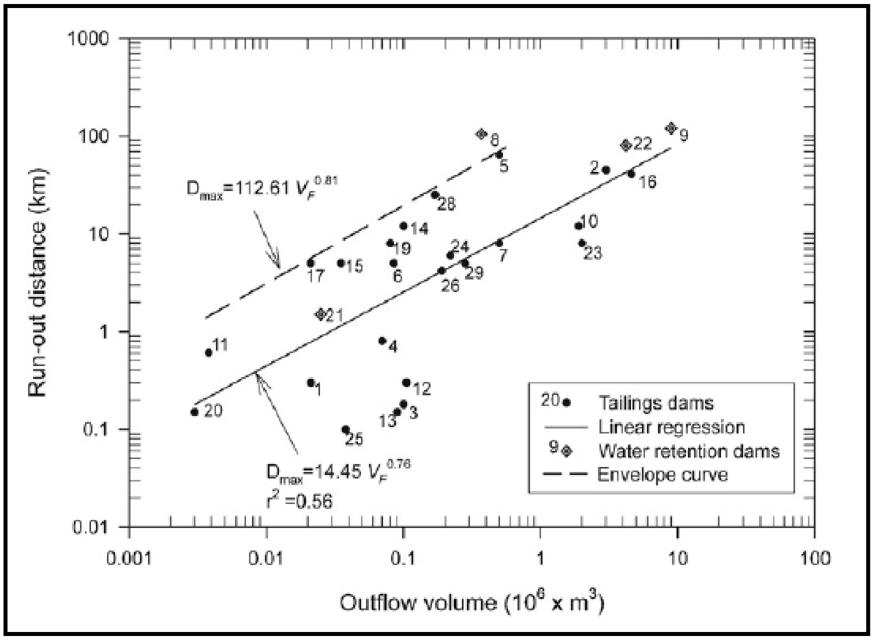


Figure 25. Graph Showing the Tailings Outflow Volume due to Tailings Dam Incidents vs. the Run-Out Distance of Tailings from Historic Failure Cases (Fig. 2 in Rico, Benito and Díez-Herrero 2008)

For the proposed mine, the initial approach for determining runout distance was to first consider pond(s) volume (see X axis on Figure 24), then determine the corresponding outflow volume per Eq. 7 provided above and in Appendix F. The final step would then be to determine the outflow volume from this first step onto the X axis of Figure 25 using Eq. 5 provided above [see Appendix F), and thus determine the corresponding run-out distance (km). The equations were linked within an Excel® spreadsheet so that the various pond volumes could be applied, with the corresponding *outflow runout distance* determined (see Appendix F). Although this appeared to be a reasonable approach, the sheer size of the proposed ponds [again, 'Pond' A = 2.064 billion m^3; 'Pond' G = 0.450 billion m^3] were much greater than those addressed by Rico, Benito and Díez-Herrero in their analysis. Considering this situation, it was nevertheless deemed relevant to use the proposed mine's pond volumes to predict runout distances. The justification for this approach was based on the need for understanding the 'potential' impacts that could be associated with dam failures. It was understood that the approach contained much uncertainty. But, in fact, Rico, Benito and Díez-Herrero (2008) stated that even with an empirical approach, errors may be large on extreme cases – "*The diversity of the tailings dam characteristics [dam type, dam situation, dam foundation, storage volume, etc.] make any universal prediction assessing failure impacts very speculative. In addition, detailed risk assessments involve timely and costly geotechnical, hydrological and hydraulic studies which can only be completed with either or both the complicity of mining companies and political authorities*". Thus, for our analysis, this approach seemed reasonable, considering that there are no other methods presently available for pre-predicting dam failure impacts.

Based on the approach described above, four release volumes (e.g., 25%, 50%, 75% and 100%) for Ponds A and G were determined per Eq. 7 (in Rico, Benito and Díez-Herrero 2008). Next, based on the results of step one, predicted outflow volumes were determined using Eq. 5 from Rico, Benito and Díez-Herrero (2008) to predict a run-out distance (km) for each of the four released tailings volumes (Table 20).

Although it is understood that many site-specific variables, including water content, viscosity of tailings, topography, and barriers, can affect run-out distance estimates, this approach seemed reasonable for predictive risk assessment purposes. Results of the analysis presented above show that run-out distances for Pond A, based on the four outflow volumes predicted from the impoundment, all could extend to Bristol Bay (and beyond), or to 270 km (Table 20) in SFK. Pond G run-out distances for the four volumes was similar, extending all the way to Bristol Bay in NFK, no matter the volume considered.

Table 20. Predicted Run-Out Distances[1] on South Fork Koktuli and North Fork Koktuli from Tailings Ponds Spills

	25%	50%	75%	100%
Pond A Capacity = 2.064 billion m^3				
Run-out Distance (km)[2]	270	270+	270+	270+
Pond G Capacity = 0.450 billion m^3				
Run-out Distance (km)[2]	270	270+	270+	270+

1 = Based on regression models provided in Rico, Benito and Díez-Herrero, 2008 as shown in Appendix F.
2 = All run-out distances were much greater than the entire stream length from tailings dam to Bristol Bay (270 km).

Although these distances seem extraordinary, and the results from the methodology used may appear to be over-stated, it is important to understand the perspective for this determination. In Rico, Benito and Díez-Herrero's evaluation, the largest tailings volume assessed was for the Los Frailes (Aznalcollar) dam failure in Spain (~20 million m^3) – the dam was 88 feet high and the run-out distance extended 41 km. By comparison, the proposed mine tailing ponds' dams would be from 5 to over 8 times higher and volumes, as per NDM's (2006c) present permit application, could be from 22 to 103 times greater. Since maximum predicted runout distances for the proposed mine's ponds ranged from 12 to 61 times greater than Los Frailes, it appears that the proposed mine prediction is not overstated.

Although Rico, Benito and Díez-Herrero found that the release volume for Los Frailes was approximately 25% of the impoundment volume, they state that for most information they reviewed *"one-third of the tailings and water at the decant pond is released during dam failures"*. Under either scenario, a release from the proposed mine's tailings ponds would be from one to two orders of magnitude greater than Los Frailes. A release of tailings from either impoundment would result in significant impacts to salmonid populations and habitat. Based on the model's prediction, even a moderate release (e.g., 25% of the requested capacity; per NDM 2006c) from either Pond A or G would physically inundate stream channels that would remain (after water extraction) throughout both the SFK and

NFK watershed, and beyond. Lethal effects to all biota living in the affected streams would be instantaneous as the slurry travels quickly (up to 60 km/hr) down the stream valley. Examples of the devastation caused by these types of events are provided in Appendix E, Table E-1. Although the bulk of the tailings should remain and not travel to a great extent outside of areas within the focus waterways, the overlying, acidic waters (with dissolved copper and other metals in solution) would contaminate surface water and adjacent terrestrial areas (riparian zones that would be affected) well away from the impact zone, potentially to Bristol Bay. For example, in April 1998, the tailings dam failure of the Los Frailes lead-zinc mine at Aznalcóllar, Spain, released 5.5 million m^3 of acidic water and 1.3×10^6 m^3 of heavy metal-bearing tailings into nearby Río Agrio, a tributary to Río Guadiamar (Hudson-Edwards *et al*. 2003). The slurry wave covered several thousand hectares of farmland, and threatened the Doñana National Park, a UN World Heritage Area. Turner *et al*. (2002) reported that more than 60% of the solid wastes were deposited in the first 13 km downstream of the breach, reaching depths of 4 m in areas close to the impoundment and a few millimeters at the spill margins. Studies on heavy metals in the watershed four years after the spill noted that an accumulation zone had formed up to 30 cm in the underlying soil. It was predicted that in the future there could be further penetration of heavy metals to greater depths (Kraus and Wiegand 2006).

As noted above for clean-up impacts for pipeline spills, habitat quality would be severely diminished from increased long-term turbidities, lost riparian habitat and equipment leaks and spills during clean-up activities which could last for years to decades. For example, for the Los Frailes spill, due to the vast (~4.7×10^6 m^3; Hudson-Edwards *et al*. 2003) quantities of soil and sediment removed during clean-up, the impacts on the river channel and valley floor morphology were considerable (in river channels and flood plain areas) (Turner *et al*. 2002). Channelization, involving dredging, re-sectioning and realignment of the river, was undertaken from the dam downstream to reach 6 (~35 km), and all in-channel and most riparian vegetation (in upper reaches) was removed. Channel banks and valley floor soils and sediments were therefore prone to erosion, increasing both the risk of channel instability and mobilization of residual contaminants (Turner *et al*. 2002). For the October 2000 Martin County Coal Impoundment Spill in Kentucky, approximately 9-11 kilometers of streams were impacted by clean-up activities (Armstead 2009). Restoration and recovery took more than seven years to complete.

Spill response activities would result in long term critical stress to salmonid populations within the two proximal watersheds (NFK and SFK), and further downstream, that could be affected. It is expected that post-spill effects would cause direct spawning and rearing habitat losses both within and outside (downstream) of the watershed affected. Also, contamination could well result in impacts to groundwater.

Recovery can take many years to decades. Few studies have been made on the natural capacity of aquatic systems to recover following impacts from a stressor (Cairns 1978). Most studies have focused on understanding how various stressors alter the chemical, physical, and biological function and structure of aquatic ecosystems (Niemi *et al*. 1990). A "disturbance scenario" is generally considered the event responsible for the modification of a stream system. Niemi *et al.* (1990) discuss this "disturbance scenario" in the context of the specific cause as a stressor (e.g., chemical, nutrient, siltation, or acid precipitation). A disturbance is reserved for the situation when the stressor or stressors result in a change in the state of the system that is different from normal behavior (Niemi *et al.* 1990). Two terms are used to define disturbances: "pulse" and "press." A pulse disturbance is defined as a disturbance of limited and easily definable duration. Pulse disturbances have little effect on the surrounding watershed (e.g., floods). Press disturbances are longer in duration and often involve changes in the watershed or stream channel (e.g., timber harvesting or channelization). Based on examples presented in Niemi *et al.* (1990), a proposed mine impoundment release could be categorized as both a pulse (e.g., initial physical release of tailings) and a press disturbance (e.g., chronic release of contaminants into water, soil and/or sediment).

For benthic assemblage recovery within lotic systems studied by Neimi *et al.* (1990), 85% of the benthic macroinvertebrate recovery endpoints met pre-disturbance densities within 18 months (see Figure 26). Niemi *et al.* (1990) suggests that the adaptations necessary for species' survival in 1^{st} to 3^{rd} order streams should lead to rapid recoveries following pulse disturbances. The resiliency of impacted populations stems from a variety of factors that contribute to the rate of recovery. These include: the persistence of impact including changes in system productivity, habitat integrity, and stressor persistence; the life history of the organism, including generation time, emergence time and propensity to disperse (e.g., drift); the time of year when the disturbance occurred; the presence of refugia; and the distance to source for recolonization (upstream and downstream). Recovery after pulse disturbances represent the abilities of the organisms to repopulate after catastrophic events. But, Marcus *et al*. (2001) noted that for Soda Butte Creek, the distance below mine tailings where taxa numbers recovered remained the same since 1967, implying that metals' contamination, and their effects to benthic communities, was long-term along the stream's length. Warner (1971) found that more species of insects and algae were found in unpolluted West Virginia streams (pH $\geq$ 4.5) compared to those streams polluted by acid (pH 2.8 to 3.8). Reductions of benthic fauna in a West Virginia stream severely affected by acid mine water were reported by Menendez (1978).

Recovery endpoints for fish were provided in Niemi *et al.* (1990) following a wide range of natural (e.g., flooding or drought), single anthropogenic (e.g., chemical, rotenone, and DDT), and watershed (e.g., mining and logging) disturbances. Although recovery times to all endpoints (i.e., density, time to first appearance, recovery of average age) ranged from 0.01 to > 52 years (Figure 27), recovery from anthropogenic pulse disturbances were generally less than 5 years. But again, recovery estimates are not available considering the scale of impacts that could potentially occur at the proposed mine. Studies by Binns (1967) and Olmstread & Cloutman (1974) showed that adjoining headwaters were a significant source for immigration of fish species. Niemi *et al.* (1990) suggested that minnows from headwater regions may be well adapted to colonize variable environments under stressful conditions. Niemi *et al*.'s (1990) analysis also showed that presence of undisturbed stretches up or downstream of a stressed system did appear to affect recovery time for fish densities; over one-third of those systems with recolonization sources up and downstream from a site had a median recovery time of less than or equal to one year. In the majority of pulse case studies, authors identified migration and recolonization rather than increase of resident populations as the main recovery mechanism.

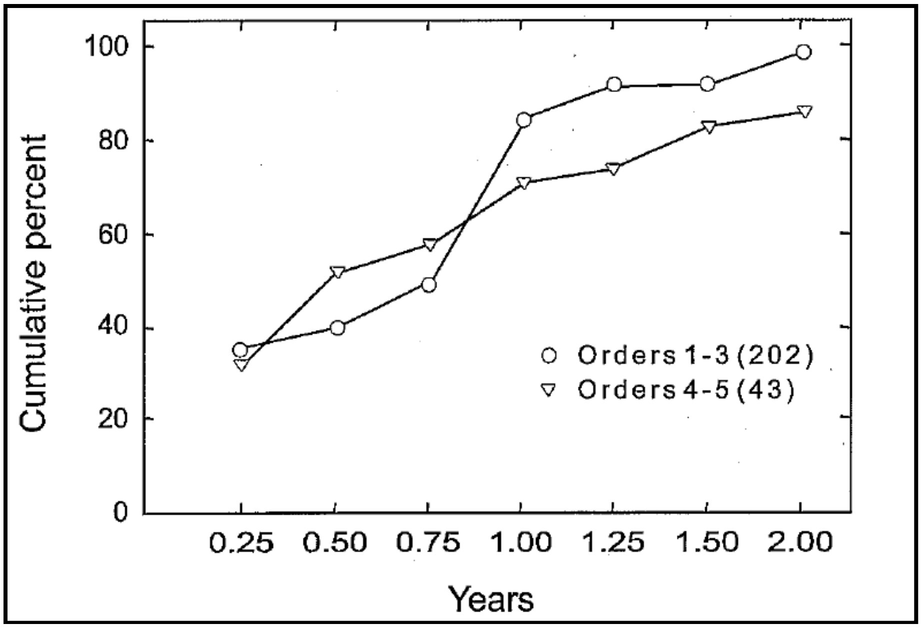


Figure 26. Recovery of Benthic Macroinvertebrate Populations following Pulse Disturbances (from Niemi *et al*. 1990)

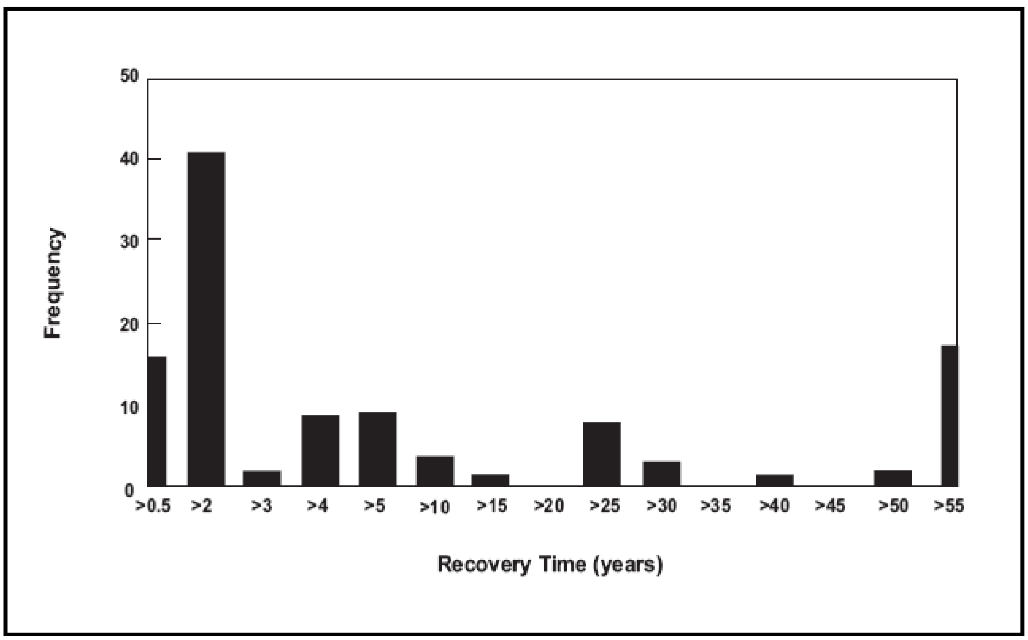


Figure 27. Recovery of Fish Populations over Time

Based on conclusions provided in recent studies directed at recovery of lotic systems, the recovery potential for NFK, SFK and downstream watersheds after disturbance would appear challenging. As previously discussed, the lack of significant headwater streams (due to water extraction proposed) would prove a hindrance to overall recovery. Several studies (Sander 1969, Slobodkin and Sanders 1969, and Holling 1973) theorize that lotic populations and communities from more variable and disturbed systems should respond more quickly to non-novel disturbances than those from less variable environments, because species in unstable environments are adaptable (Poff and Ward 1990). Since watersheds within the proposed project site would not be considered variable and disturbed, it is predicted that recovery would be much longer. Also, Turner *et al*. (2002) note that for the Los Frailes spill, flow from upper reaches of the river allowed for dilution from dissolved metals' concentrations which became elevated during low flow conditions. In the SFK specifically, no upper reach contributions would be available because all headwaters (and groundwater) would be appropriated for mine uses (see Table 5). Although the NFK below the dam area would receive continued upgradient inputs from subbasins A, B, C, D and, partially, F (see Figure 6 and Table 4), the reduced contributions from subbasin F (e.g., ~21% of the discharge) would limit the effect of dilution. Finally, although high flows may allow for dilution of released metals in the water column, it would also allow for greater downstream transfer of contaminated sediments. For salmonid streams, it would be expected that sediment transport and deposition into distal portions of the watershed would result in further negative impacts, including riffle (redd) embeddedness and chronic metals' exposure to the most sensitive life stages (i.e., eggs and fry). Similarly, EPA's Ecological Risk Assessment for the Clark Fork River (EPA 1999b) suggests that time-limited events such as periods of high water and thunderstorms probably do more harm than the steady, but less-lethal, impact of metals' pollution. They found that acute weather events led to pulses of metals' pollution – especially copper – that resulted in mortality to fish and aquatic insects.

For the proposed mine it is impossible to predict the probability that a tailings dam would fail and result in impacts similar to what has been presented above. Generally, in an ecological risk assessment, probabilities are provided in order to make an informed decision on the risk posed to receptors from an expected or assumed effect. Relative to a tailings dam failure and subsequent release at the proposed mine, considering that dam failures are highly correlated with extraordinary and unexpected rainfall/snowmelt flood events, no such probability can be made. In fact, throughout the life of the mine, a dam failure may not occur and salmonid resources may not be affected. But, for this predictive ERA, and based on historical information related to the overall impacts to downstream resources, should a dam fail, it is highly certain that impacts would be significant to catastrophic for affected watersheds, and that a long-term recovery period could be expected. Finally, it should be noted that the spill scenario presented above is based on information from NDM's water use permit application (NDM 2006c) for tailings ponds' storage capacity of approximately 2.5 billion tons. If this storage capacity were to be expanded over time, the release would likely expand as well; subsequent impacts to salmon resources and habitat would be expected to increase similarly.

3.2.5 Acid Mine Drainage

3.2.5.1 Stressor Description

USEPA (1994) has developed information on issues and information necessary for predicting AMD at non-coal mining sites. Their report examines acid generation prediction methods and summarizes current methods used to predict acid formation including sampling, testing and modeling. As noted in Section 3.2, Chambers (2006) provided a memorandum concerning the geochemical characterization of rocks from the proposed mine with a conclusion that the majority of sampled rock is potentially acid-generating (see Figure 16; per NDM 2005c). Thus, for the proposed mine, the understanding that the mine would be developed in an area with moderate precipitation (>36 inches of precipitation per year), a high water table, numerous small streams, and over geological formations that are susceptible to ground water movement, makes AMD formation and movement highly likely and a high risk proposition (Kuipers *et al*. 2006). Again, more detailed discussion of the formation and fate for AMD is discussed in Section 3.2, above.

For the purposes of this ERA it is assumed that AMD will be formed at the proposed mine. What is not clear, and never really is for most mines, is when and how much will form and be discharged over the life of the mine (Kuipers *et al*. 2006). There are several ways that metallic and acid pollution from sulfide ore body mines can enter surface and groundwater. These include chronic leaks of acid and heavy metal contaminated water from tunnels and mine pit walls before and after mine closure; chronic leaks from waste rock piles and tailings storage areas; discharges from treatment facilities; partial or complete failures of tailings dams and storage areas; slurry pipeline breaks; fugitive dust; and, failure of pollution control measures after closure or abandonment (Lottermoser 2003). But, even with the prevalence and use of existing predictive models, modeling for AMD has not yet found extensive applications in predicting oxidation rates and effluent quality at operating or proposed mines (Ferguson and Erickson 1988, Kuipers *et al*. 2006).

Under our scenario, tailings dams would be constructed and operated that would have the capacity to contain over 2.5 billion tons of mine waste, with a complex system of drains, collection systems, and pumps to collect and return leakage back into the impoundments. If these facilities are not maintained in perpetuity, the waste will eventually leak into the ground water and adjacent streams. The mine could develop an AMD problem over its 40 to 70 year life or, more likely, after it is closed or abandoned and turned back over to the State of Alaska. Information in Ptacek and Blowes (2002) suggests that releases can take place over several decades to many centuries, with timing and duration of peak discharges varying between sites. They provided a general timeline that indicates reduction of pH and subsequent metals' contamination does not progress quickly in either tailings or associated groundwater. Of course, they note that groundwater velocity and length of flow path are critical to understanding release potentials. One of the mines they reviewed, Nickel Rim Mine, had been inactive since 1958. They concluded that sulfide oxidation had been occurring for over 40 years. Bain *et al*. (2000) reported that residual dissolved constituents at Nickel Rim were being transported to a down-gradient surface water body.

From review of information available, AMD formation and/or discharge at older mines (i.e., 40-50+ years) usually occurs in the latter stages of the mine's life or after abandonment or closure. Impacts to biological communities in downstream receiving waters have been documented at many such mines via National Priority List (i.e., Superfund) reports and other informational sources.

Tailings vary considerably in their physical, chemical, and mineralogical characteristics. Particular characteristics can affect tailings behavior in a storage area and ultimately affect drainage water from the latter, seeping into either surface water or shallow groundwater systems. Tailings can also have poor settling and consolidation characteristics. Generally, when initially deposited, the tailings are alkaline (pH > 10), but pH drops to between 2 and 4 as the pyrite becomes oxidized. Potential for seepage from the tailings into groundwater is a major issue with many tailings disposal operations. Generally, synthetically-lined dams always leak even if the dams do not fail. The effect of 'vertical' seepage from a tailings dam to the local groundwater flow would depend on the relative quantity of water flow from the tailings to the total flow in the underlying groundwater system. The low permeability of many tailings deposits and the flooding that occurs in both operating and abandoned tailings impoundments limit the rate of AMD generation and release. Thus, the full potential effect of the very large deposits of more recent acid-generating tailings may not have had sufficient time to develop. Drainage of waters placed with the tailings will likely occur for some period after the tailings are no longer being deposited. Also, seepage derived from precipitation on the surface would also continue indefinitely.

3.2.5.2 Impact Determination

For this evaluation, two primary AMD-release mechanisms at the proposed mine are addressed: 1) AMD discharge from tailings dams and waste piles; and 2) infiltration into groundwater with subsequent discharge to surface waters. The specific method for AMD release from tailings dams/waste piles is dealt with semi-quantitatively based on predicted pH values. Discharge of groundwater-derived AMD into sediments is addressed qualitatively due to the lack of hydrogeologic information for the proposed mine.

In general, Kuipers *et al.* (2006) found that very few EISs predicted that surface water and groundwater quality standards <u>would not</u> be met after mitigation was in place. A number of mines in their study mentioned the effect of time on predicted surface water quality impacts. For example, the EIS for the Greens Creek Mine in Alaska predicted a lag time for acid generation in tailings of 20 to 50 years. Similarly, although surface water quality impacts were predicted to be low at the Grouse Creek Mine in Idaho, the EIS mentioned that if AMD occurs, the effects could be long-term. The majority (72%) of the 25 case study mines reviewed by Kuipers *et al.* (2006) predicted [in one or more EIS] a low potential for acid drainage. Of the 25 mines studied, to date, 9 (36%) had developed acid drainage on site. Of these nine mines, eight (89%) had initially predicted low acid drainage or had no information on acid drainage potential. Therefore, per Kuipers *et al.* (2006), nearly all the mines that developed acid drainage either underestimated or ignored the potential for acid drainage in their EISs.

As at other mine sites, it is predicted that AMD discharges at the proposed mine would occur during latter stages of the mine's life or after closure. So, for this assessment, flow characteristics in down-gradient NFK and SFK stream systems would be reflective of post-development flow rates as provided in Tables 4 and 5, respectively. Information gathered for hard rock mines indicate that AMD results in chronically low pH, generally below 4. For example, Gilchrist *et al*. (2006) found acid water (pH 2.25-4.06; minimum of 1.78) below tailings piles in first-order streams near Copper Brook Mine. USFS (2009) noted that pH from AMD was near 3.3 at Ore Hill Mine, with at least one mile of stream affected. At the Mt. Perry Copper Mine in Queensland, Australia, the most proximal streams were contaminated with AMD, with pH ranging from 3.3 to 4.5. Mahiroglu *et al*. (2009) found relatively higher pH (~4.8) in streams near copper mines in Turkey. Finally, Bambic *et al.* (2006) found seasonal and spatial variations in metal concentrations and pH in a stream at a restored copper mine site located near a massive sulfide deposit in the Foothill copper-zinc belt of the Sierra Nevada, California. Spatial variation was assessed in a 400 m reach encompassing an acidic, metal-laden seep. At the seep, pH remained low (2-3) throughout the year, and copper concentrations were highest.

Based on this information, it was predicted that pH for AMD discharges into stream systems from mine workings and tailings ponds at the proposed mine would be near 4. Because the evaluation is primarily to show the relative spatial effects expected from this type of release, this pH concentration was deemed appropriate. Similar unique reaction kinetics have been observed at the Bingham Canyon Mine in Utah where fresh waste rock exhibits a paste pH of 7.0. Within 6 years, the pH of the waste rock dumps decline to 4.7 further decreasing to pH 3.7 after 50 years of weathering (Borden 2001).

Starting with a pH of 4 [NFK started with assumed pH of 5 due to upstream water contributions prior to the theoretical discharge point], our analyses then assumed linear water contributions (e.g., groundwater and surface water) along the stream in question to predict changes from dilution. One assumption was that all non-impacted waters in downstream reaches had initial pH values of 7. This was considered reasonable for this exercise based on data from Woody (2009b) who found pH values very close to 7 for the 20 stations collected in August 2008 during her study of salmon streams in and near the Pebble prospect area. [*Although, it must be pointed out that a distinct change in water chemistry typically occurs during breakup (late April/early May), and that pH may be below neutral during these time periods. For example, when precipitation falls through the air, it dissolves gases such as carbon dioxide and forms a weak acid. Natural, unpolluted rain and snow are slightly acidic - it has a pH between 5 and 6. When snow melts rapidly it may not percolate through the soil before reaching the stream; soil minerals can't buffer it. At these times the stream water may also be slightly acidic (e.g., <6).*]

Next, per-mile tributary contributions were determined based on information presented in Tables 4 and 5, and considering stream miles between Stations 1-4 (see Figure 6). Since flow was determined cumulatively for each station denoted on Tables 4 and 5 and on Figure 6, contributions per mile were calculated based on the total pre-station value and normalized by mile. For instance, between Stations 1 and 2 on SFK flow increases 32% (e.g., from 0 to 7.2 cfs, or to 114 cfs, depending on month). Therefore, the increase in flow was divided by the number of miles in that reach (~5 miles) to determine an average flow increase per mile (i.e., January = 1.7 cfs/mi). Since all months for each stream have the same ratio of increase, pH changes would be the same regardless of month. [*Although it is understood that initial dilution effects at the discharge point will vary based on discrete precipitation events, this could not be accounted for in our analysis. As high flow conditions modify (increase) the discharge/stream flow ratio, dilution will act to increase pH. But, impacts from metals in low pH surface water are generally exacerbated during low flow periods*.]

Next, the effect of dilution on pH was reviewed. pH is the negative log of the hydrogen ion concentration. As such, as the concentration of the ion changes, the pH of a solution should change also, based on dilution. So, if a strong acid is diluted, the solution H+ concentration decreases. For example, if a 1 M HCl solution pH is 1, when diluted to 0.01M then the pH changes to 2, but it can only attain a maximum value of 7 at infinite dilution with water. Therefore, a dilution factor of ten for a strong acid changes the $[H^+_{(aq)}]$ by a factor of 10 and thus the pH by one unit.

Based on this understanding, pH values were calculated starting for each mile segment, beginning with a discharge pH of 4 for upper portions of the SFK and a pH of 5 for NFK nearest to the mine tailings ponds. The results for SFK show that post-development surface water inputs into the system downstream from the mine would provide limited capacity for buffering [dilution] of the low pH discharge (Table 21 and Figure 28). Results for NFK account for up-gradient, additional stream volume prior to mine discharge that would act to dilute AMD discharge as it enters the stream.

Table 21. Predicted Change in AMD Discharge pH within North Fork Koktuli and South Fork Koktuli Watersheds

Stream	Discharge pH	1st Predicted Value	2nd Predicted Value	3rd Predicted Value
NFK	5.00*	6.03 @ RM 24	6.43 @ RM 28	ND
SFK	4.00	4.41 @ RM 17	4.52 @ RM 27	4.56 @ RM 37

ND = Not determined.
Note: * = based on assumed discharge pH of 4, with predicted in-stream pH of 5 based on dilution from up-gradient channels in NFK watershed (see Figure 28).

A similar situation is likely if discharge of groundwater-derived AMD into sediment occurs. Without site-specific hydrogeological information it is assumed that major AMD discharge via this route would occur in near-site portions of the watersheds addressed by the tailings discharge scenario, but also in UTC if the hydrogeologic connection is assumed.

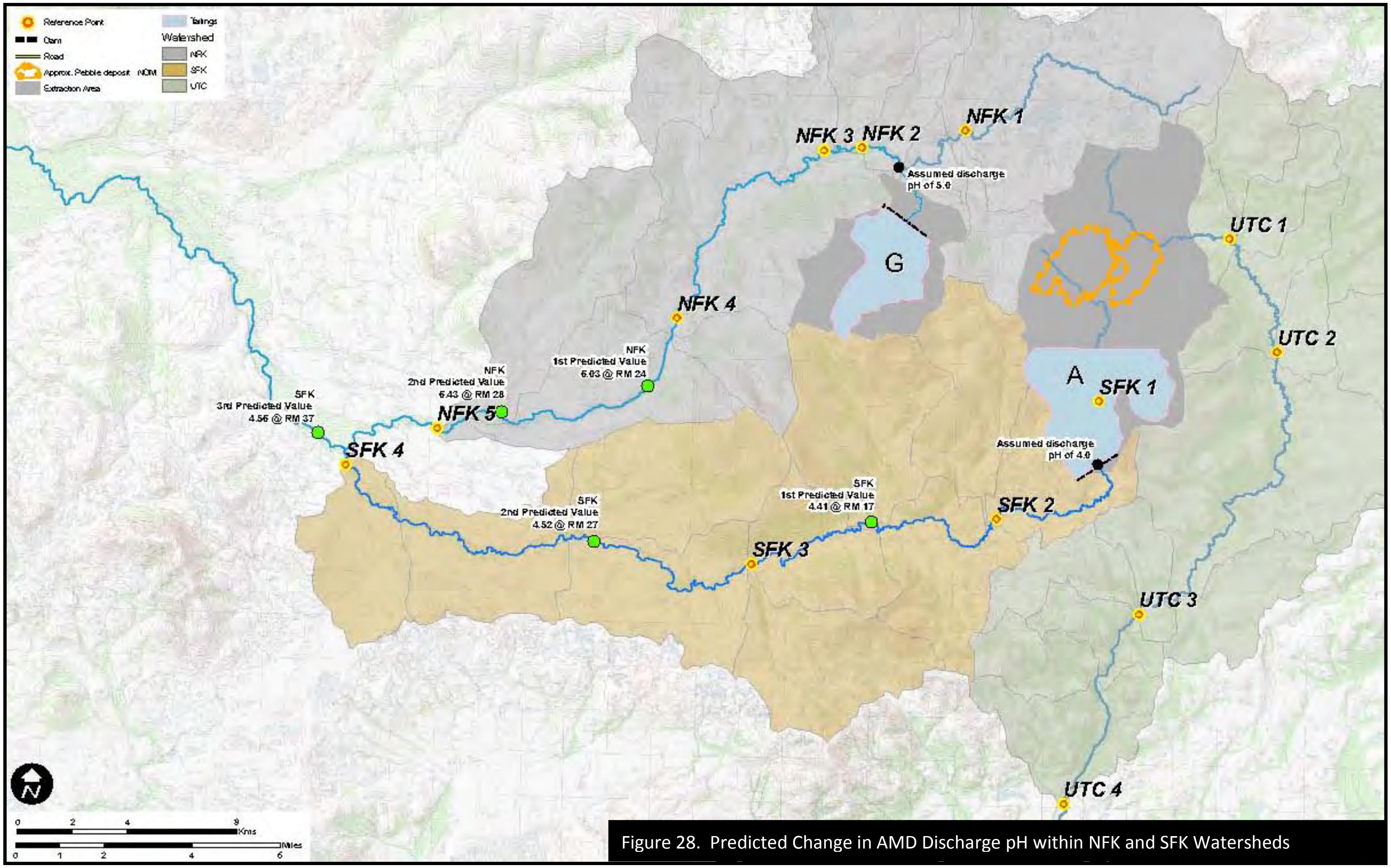


Figure 28. Predicted Change in AMD Discharge pH within NFK and SFK Watersheds

Groundwater transport within downstream portions of the streams would be a long-term phenomenon, but information from studies on historic contamination from other mines does indicate that this is possible. Again, Ptacek and Blowes (2002) found that metal concentrations were found in groundwater more than 100 meters from the tailings impoundment at a site where oxidizing had been occurring for more than 35 years. For a 70-year old tailings pond at the Sherridon Mine, Manitoba, they reported very high concentrations of metals present in the vadose zone (i.e., unsaturated surficial zone or portion of earth between the land surface and the zone of saturation) pore water, and both groundwater and surface water were severely degraded. Similar to other older mines, ground water intrusion into surface water via lake sediments was found, suggesting that metals were available for diffusion or transport into the overlying water column (Ptacek and Blowes 2002).

Water quality changes from AMD discharge (and metals in solution) into watersheds could result in increased bioavailability of copper already found in surface water and sediments, in addition to metals added to the system from other mine sources previously described (e.g., dust, ore releases, etc. which may oxidize and lower pH in addition to AMD), with higher proportions of ionic copper occurring within the water column. Aqueous copper speciation and toxicity depends on the ionic strength of the water. Again, the hydroxide species and free copper ions are mostly responsible for toxicity, while copper complexes consisting of carbonates, phosphates, nitrates, ammonia, and sulfates are weakly toxic or nontoxic. Copper in the aquatic environment can partition to dissolved and particulate organic carbon [although expected to be low in site waters]. The bioavailability of copper also can be influenced to some extent by total water hardness. Water quality changes from AMD discharge into watersheds would result in increased bioavailability of copper in sediments, again with higher proportions of ionic copper within the water column. Impacts to salmonids from free cupric ions would be expected.

Healthy, unpolluted streams generally support several species and moderate abundance of individuals; whereas impacted streams are dominated by fewer species and often low to moderate numbers of only a few organisms (Jennings *et al*. 2008). Streams affected by AMD are typically poor in taxa richness and abundance. Cooper and Wagner (1973) found that the distributions of fish in AMD-affected streams in Pennsylvania were severely impacted at pH 4.5 to 5.5. Ten species revealed some tolerance to the acid conditions of pH 5.5 and below; 38 species were found living in waters with pH values ranging from 5.6 to 6.4; while 68 species were found only at pH levels greater than 6.4. They noted a complete loss of fish in 90% of streams with waters of pH 4.5 and total acidity of 15 mg/L. Also, Raleigh (1985) provided that the optimum habitat suitability index for pink salmon fell between pH 6.5 and 8. The index value dropped to 0 (on a scale of 0 to 1.0) below pH 5.5.

A second water quality impairment is likely when the pH of AMD is raised past 3, either through contact with fresh water or neutralizing minerals, previously soluble Iron(III) ions can precipitate as Iron(III) hydroxide, a yellow-orange solid colloquially known as ‘yellow boy’ (Herbst 1995). ‘Yellow-boy’ has numerous effects on streams including oxygen removal, acidification, and depletion of the buffering capacity. It can also cause a decrease in light penetration, and subsequent negative effects on photosynthesis and the clarity of the water. The precipitate will smother-out bottom dwellers and decrease food resources for fish (Robb and Robinson 1995). The process also produces additional hydrogen ions, which can further decrease pH.

4.0 RISK SUMMARY AND CONCLUSIONS

The 'predictive' nature of this ERA includes consideration and treatment of unknowns. Historical studies, literature and documented effects to salmon and other aquatic life from current and past hard rock mines were used to reduce unknowns to the greatest extent possible and refine the overall prediction of risk.

4.1 Summary of Risks by Stressor

To characterize the risk posed to salmon resources within watersheds associated with the proposed mine, both quantitative and qualitative information developed through the risk process was used to determine an overall (predictive) *weight-of-evidence* conclusion. Generally, weighing of evidence begins by summarizing the evidence developed for each endpoint selected (e.g., salmon species and/or their supporting habitat), then evaluating whether there is strong *proof* for supporting conclusions of potential *effect, non-effect,* or a point somewhere along this gradient. A major goal of this ERA process was to use the most relevant historical and literature-based findings to reduce the overall uncertainty. This process provided that predicted risk from mine creation, operation and closure would be appropriate and relevant. It is important to understand that the very nature of this **'predictive'** analysis is based upon unknowns. But these unknowns have been considered to the extent possible as potential risks to salmon from mine creation, operation and closure were explored. The information developed by this ERA is not conclusive, nor does it attempt to state with confidence that specific impacts to salmon will or will not occur. It has been developed to provide a gage with which to evaluate the risks that will likely be present should a mine such as the proposed mine be permitted and developed.

The risk assessment focused on two general stressor categories that may affect the viability of salmon within the watersheds under consideration. First, as a result of mine development and operation, physical stressors would occur that directly affect the viability of salmon resources. These include: the loss of instream flow [via changes to surface and groundwater] and subsequent alteration of habitat; impacts from road construction, including culverts' placement; and, effects from fugitive dust during construction and mining activities. Secondly, impacts associated with chemical [primarily metals] stressors within surface waters and/or sediments were evaluated for sources including: fugitive dust; slurry pipeline spills; chemical spills; tailings releases from episodic and large scale pollution events; and acid mine drainage (AMD).

4.1.1 Physical Stressors

Physical stressors include permanent removal/reduction of waterways (*Dewatering and Loss of Instream Flow* [*including Groundwater Discharge*] and *Loss or Alteration of Supporting Habitat*) that either directly support fisheries resources or provide necessary flow for species and population viability in downstream reaches. Similarly, stream crossing impacts during *Road Construction* may limit upstream migration and reduce reproductive potential for affected salmon populations. Reduced down-gradient stream water quality and quantity, and subsequent secondary effects to fisheries, could be expected from *Fugitive Dust* emissions, as a result of mine activities.

Dewatering and Loss of Instream Flow* [*Including Groundwater Discharge*] *and Loss or Alteration of Supporting Habitat

- ***33 sq. miles of drainage area lost.***
- ***Approximately 68 stream miles lost.***
- ***14 miles designated salmon streams lost.***
- ***Reduced flow can result in higher temperatures; lower dissolved oxygen; restricted upstream migration.***
- ***Potential effects to spawning and embryonic development.***
- ***Up to 78 stream miles would exhibit some form of flow reduction in the three watersheds evaluated.***

The analysis predicts that physical stressors, including *Dewatering and Loss of Instream Flow* (including *Groundwater Discharge Loss*) would be critical and related to secondary effects such as *Loss or Alteration of Supporting Habitat* for salmon species (especially Chinook and coho) occurring within the watersheds under evaluation. First, approximately 33 square miles of drainage area within the three watersheds is proposed to be lost due to mining uses (e.g., water extraction, tailings ponds, excavation pits, mills, etc.). This 33 square mile area includes approximately 68 linear miles of stream channels, of which over 14 miles are ADFG-designated anadromous streams. As a result of lost up-gradient source water from the eliminated streams, summer low flow conditions in down-gradient mainstem segments of all three streams under evaluation would be exacerbated resulting in reduced pools and backwaters that support juveniles –approximately 78 stream miles would exhibit some form of flow reduction. This in turn would likely result in greater competition for resources such as food and cover. Pools that remain within affected stream reaches could experience increased temperatures.

Reduced low flow during the incubation or inter-gravel phase would also act to reduce salmon production within affected streams. Low flows would limit adult salmon entry into streams or affect their movement up river to stage for spawning. It is predicted that after mine development, velocities during the critical spawning/embryo development period (January–March) within all three streams would be less than optimum. Low flow conditions, along with other associated reductions in water quality conditions (i.e., lowered dissolved oxygen, higher water temperatures) would likely increase stress on individuals, potentially resulting in mortality. Flow reduction would also affect substrate composition in riffle areas within affected mainstem segments through embedded conditions and reduced sediment oxygen concentrations. This is turn would act to diminish the quality of redds, ultimately resulting in negative impacts during embryonic development and fry emergence.

Temperature changes can also occur as a result of stream flow reductions. The most critical period would be summer, when flow is already reduced and temperatures are highest. Summer water temperatures would likely increase due to diminished riparian areas providing less shade and reduced upstream tributary inflows. Increased temperatures can cause higher stress to salmon (and forage fish). Temperature increases also affect the amount of dissolved oxygen in a stream, a key limiting factor for fish survival, resulting in increased disease outbreaks. In addition to growth and survival, changes in stream temperatures would affect the timing of smolt emigration. Finally, flow reductions have been shown to result in long-term reduced temperatures in winter, ultimately causing deleterious effects to egg/fry survival.

Road Construction

- ***Installed culverts along the proposed road could affect 89 streams, 14 of which are officially designated as supporting salmon and 75 of which may support salmon but have not yet been surveyed or survey results are not yet public.***
- ***35 miles of upstream anadromous habitat may be significantly affected for salmon spawning and rearing in the 14 designated anadromous streams.***
- ***Barriers to upstream migration for spawning could result in population fragmentation.***

Culverts installed during *Road Construction* can restrict or eliminate fish movement to upstream habitat, and isolate or modify populations. Effects to populations from culvert placement can include reduced ability to support upstream populations; habitat fragmentation; decreased ability to reach important headwater spawning and rearing sites; and attenuation of upstream species richness. The proposed access road would cross at least 89 streams; 14 of which are designated as ADFG anadromous waters. At these 14 stream crossings, over 35 miles of upstream anadromous habitat could be eliminated or significantly affected for use by salmon as spawning and rearing habitat. In addition, rainfall events could lead to water quality reductions downstream of crossings. Studies have shown that sediment loads are up to 3.5 times higher downstream of road culverts, with material being deposited in cobble stream beds downstream. Again, embedded riffle conditions would reduce the quality of redds and embryonic development and fry emergence, as survival and emergence of embryos and alevins is greatly influenced by the dissolved oxygen supply within the redd. Similar impacts could be expected at anadromous streams which are not yet designated. The overall impact of the proposed road construction, culvert placement, and maintenance at the 14 anadromous streams (and others) crossed could result in long-term reduction of habitat and subsequent reduction of viable salmonid populations presently found in these waterways.

Fugitive Dust

- ***Fugitive dust dispersion could conservatively cover 33.5 square miles surrounding the proposed mine.***
- ***33 miles of streams, of which 10 miles are designated salmon habitat, would be affected within the 33.5 mile dispersion zone.***
- ***Over the life of the mine, water quality could be negatively impacted due to vegetation loss and subsequent increased runoff, resulting in elevated stream turbidities and embedded conditions in riffle areas used for spawning.***

Fugitive Dust is expected to be generated during open construction and pit mining activities, materials handling, mill and concentrate storage facilities, and from wind-generated dust at mineralized surfaces. Dust dispersion would conservatively affect an area of 33.5 mi^2 around the proposed mine, but most likely a larger area. Within this area are approximately 33 miles of ephemeral, intermittent, and perennial streams, of which approximately 10 miles are ADFG-designated anadromous waters. Fugitive dust's impact on water quality over the 40 to 70-year life of the mine would result from denuded riparian habitat and subsequent degraded, embedded stream channels. Plant community and drainage impacts would be most obvious, with shifts and reductions of endemic plant communities replaced by patchy barren ground in areas having highest dust accumulation. Lichens and mosses are sensitive to dust impacts and would be affected to the greatest degree. Down-gradient streams would show incremental negative changes over time as the ecological viability of headwaters that support salmonids, resident species and other aquatic life diminishes.

4.1.2 Chemical Stressors

Chemical stressors, including those from *Fugitive Dust*, *Pipeline Spills*, *Episodic and Large Scale Pollution Events*, *Chemical Spills* and *Acid Mine Drainage* will likely act both on short- and long-term time scales, with the magnitude of their effects based on factors such as locale, season, volume and/or stressor type. Evaluation of the risk(s) posed from most of these stressors centered primarily on the potential for exposure of salmonids [and their habitat, including food resources] to copper expected in dust, tailings, slurry and mining wastes. Effects predicted from AMD centered on potential degradation of supporting habitat [surface waters] from reduced pH, but also included evaluation of AMD for mobilizing metals in the water column and directly affecting salmon. Chemical spills focused primarily on potential for impacts to aquatic environments from hazardous materials that are typically used in the hard rock mining industry.

Chemical Spills

- ***Hazardous chemical spills could cause fish kills and habitat destruction.***
- ***Spills would be critical during clean up activities associated with pipeline breaks or tailings dam failures.***
- ***Impacts would be critical if spills occurred in spawning or rearing habitat.***

Transportation and storage of hazardous chemicals near water bodies could result in inadvertent *Chemical Spills* producing fish kills or other acute impacts to fishery populations. Clean-up activities associated with a pipeline break or tailings dam failure may pose the biggest risk to salmon due to the heavy equipment and maintenance materials being required at a site. Impacts would be critical if spills occurred in spawning or rearing habitat.

Fugitive Dust

- ***During early stages of the proposed mine (10 years) copper from dust dispersion could affect benthic communities and subsequently salmon.***
- ***As the mine ages (30-50 yrs), copper from dust accumulation & transport could result in acute and/or chronic effects to aquatic resources, including salmon.***
- ***Toxicity would increase with oxidation of dust particles and in association with and acid mine drainage.***

Fugitive Dust is expected to be generated during open pit mining activities, materials handling, mill and concentrate storage facilities, and from wind-generated dust at mineralized surfaces. Risk was evaluated for two potential transport mechanisms; erosion of metal-laden soil particles and metals' leaching. Based on the depositional rates and patterns presented, erosion of soil particles indicate that during the early stages of mining operations [10 years] sediment copper concentration increases within the three watersheds would not be critical, but could include effects to sensitive benthic macroinvertebrates (e.g., mayfly, caddisfly, stonefly) which would occur in the most upstream segments where concentrations feasibly could exceed baseline mean concentrations by factors ranging from 2 to 18. As the mine ages (30-50 years), and dust (metals) accumulation along with erosion impacts are more sustained, stream concentrations could reach levels where chronic aquatic toxicological effects are imminent and acute effects possible. Copper (and other metals) would reach equilibrium, with sediment copper being continually released into interstitial (pore) water / surface waters, and suspended particulate matter in the water column adsorbing free copper ions to be re-deposited back into the substrate. Water quality changes (i.e., reduced pH) from AMD into watersheds would increase the bioavailability of copper, with higher proportions of ionic copper within the water column. Factors such as mixing and floods could both ameliorate local effects or lengthen the contaminant pathway, extending effects to larger portions of the watershed. At the concentrations predicted, salmon would be exposed to copper directly, through olfactory bulbs; through gill uptake of waterborne free cupric ions; and biotransfer in food resources.

Leaching of metals from dust-laden soils suggests that a continuous contribution of dissolved copper into stream systems would be expected to result in long term degradation of water quality. The model predicts that dust generated at the mine would result in metal-laden soils, with transport mechanisms resulting in continuous, long-term contamination of local surface waters that support multiple salmon life stages. This is important, especially considering that the exposure and oxidation of sulfides in both dust [and other mine sources] would result in acid generation and thus pH reduced in local water bodies. This would be most pronounced in upstream portions of the watersheds because dilution, due to proposed water extraction, would not be available. Small increases in dissolved copper above present background concentrations could result in sub-lethal effects to rearing juveniles throughout the watersheds. Salmon genetic acclimation to 'historic' dissolved copper concentrations in the watershed may make impacts from any increase in these concentrations critical. Downstream portions of all watersheds would most likely show reduced impairment as a result of dilution from inflowing tributaries.

Slurry Pipeline Spill

- ***Pipeline releases could send thousands of gallons of slurry into sensitive salmon streams.***
- ***Embedded riffles and increased turbidities would result in down-gradient stream segments.***
- ***Long-term exposures and food chain transfer of copper in water and sediment would impact salmon and other aquatic life.***

A pipeline break or spill could result in thousands of gallons of metal-laden slurry being deposited into sensitive anadromous streams. Impacts from small spills would be similar in perennial streams such as the Newhalen River and Iliamna River, with fine-grained slurry particles being quickly entrained in flowing waters and transported downstream. For a nominal spill into the Newhalen River (100,000 to 200,000 gallons), slurry would be deposited directly into the stream channel. Primary physical impacts would be embeddedness in riffle/spawning proximal areas and increased turbidities resulting in potential gill abrasion and respiratory distress. Habitat quality would be diminished from increased turbidities, lost riparian habitat, and equipment leaks and spills during clean-up activities, for weeks to months. Long-term biouptake and transfer within food chains would likely result from exposure of forage fish species and benthic macroinvertebrates to both water and sediment metals' concentrations. The analysis suggests that impacts would most likely be exacerbated in smaller streams compared to larger streams.

Episodic and Large Scale Pollution Events

- ***A tailings dam release could extend to Bristol Bay;***
- ***Fish kills would occur and tailings in streams would cause long-term effects;***
- ***Spills would result in loss of spawning and rearing habitat;***
- ***Recovery could take decades or longer.***

A failure of one of the tailings dams planned for the proposed mine would have both short and long term impacts on receiving waters, with severity dependent on dam release volume, timing, and location. Analysis predicts that run-out distances could be extensive, ranging to Bristol Bay. Lethal effects to biota in an affected stream would be instantaneous as the slurry travels quickly (up to 60 km/hr) down a stream valley. The bulk of the tailings would likely remain near the spill site and not travel outside of impact area, but overlying, acidic waters (containing dissolved copper and other metals) would affect surface water and adjacent terrestrial areas (affected riparian zones) well downstream of the impact zone.

Response activities would result in long-term stress to salmonid populations that were affected. Post-spill effects could cause direct spawning and rearing habitat losses both within and outside (downstream) of the primarily watershed affected. A conservative estimate of lost stream functional viability within the NFK and SFK watersheds shows that all anadromous streams would be affected. It is expected that salmon further downstream would also be affected to some varying degree. Because affected watersheds are not considered variable or disturbed, it is predicted that recovery would be slow and on the order of decades or longer.

Acid Mine Discharge (AMD)

- ***AMD is expected during the proposed mine's life, and after.***
- ***Instream pH levels from AMD below 5 could occur up to 30 miles from the mine.***
- ***Low pH would result in fish kills and benthic community impacts.***
- ***AMD into streams would result in increased bioavailability of copper (and other metals) from various mine sources (dust, waste piles, accidental ore releases).***

Geochemical characterization of rocks from the proposed mine indicates that they would be acid-generating. Because the mine is proposed to be developed in an area with moderate precipitation, a high water table, numerous small streams, and over geological formations that are susceptible to ground water movement, AMD movement is predicted to be highly likely. Based on the literature reviewed, a pH of 4 (for SFK) and 5 (for NFK) for AMD discharges from tailings ponds/waste piles was used to show the relative spatial changes expected from AMD development and discharge. [AMD formation and discharge via groundwater was assumed but not specifically addressed by the analysis.] Results of the analysis showed that surface water pH values less than 5 would be possible up to 30 miles downstream of the mine.

Water quality changes from AMD into watersheds would result in increased bioavailability of copper (and other metals) already found in surface water and sediments, in addition to metals added to the system from other mine sources previously described (e.g., dust, ore releases, waste piles, etc., may also oxidize and reduce pH in concert with AMD), with higher proportions of ionic copper occurring within the water column. Impacts to salmonids from free cupric ions would be expected. Streams affected by AMD are typically poor in taxa richness and abundance. Based on literature findings, a complete loss of fish in 90% of streams having a pH less than 4.5 could be expected.

4.2 Multiple Stressors and Relative Risk

It is important to understand that the potential stressors of concern identified through this risk assessment process would work both independently and concurrently to impact salmon and their supporting ecosystem. For example, stream flow reduction from water extraction/use proposed for the mine has the potential to directly affect individuals and their habitat, with fugitive dust impacts and inadvertent spills and releases also occurring in the same locale. Both physical and chemical impacts from dust and mining activities would act to exacerbate an already stressed fish community in those stream segments where flow has been reduced and habitat has been altered. This example would be considered a chronic, long-term issue, with effects to populations and habitat increasing over decades or longer.

Conversely, episodic and large-scale pollution events alone are generally considered to be the most critical to salmon from a short-term perspective. Based on their size, these events likely would result in acute impacts, but impacts such as habitat destruction and chemical exposures could occur over much longer periods – beginning during initial response and clean-up, and extending into channel rehabilitation and beyond. Additionally, an episodic spill event in streams already stressed by flow reductions, dust or other on-going mining-related impacts, would limit a salmon population's recovery as compared to a stream system that has not experienced reductions in flow and is lacking impacts associated with mining dust dispersion and other similar mining-related impacts.

Based on information developed during the risk process and as described in the preceding summary, stressors of concern impacts were objectively evaluated for each salmon species at three ecologically relevant levels; *individual, population* and *habitat*. Impacts to individuals would be those that affect limited portions of a population, typically over short time frames, and are generally not critical for sustaining populations. Chemical and pipeline slurry spills that result in fish kills or temporary relocation are considered relevant stressors for impacts to individuals. Although individual fish would be killed, their loss would not, in most cases, result in changes to stream communities over the long term, if clean-up measures are adequate. Typically the most vulnerable segment of a fishery population are juveniles. So, although subsequent year-class strength may be temporarily diminished in the near term, the overall long-term reproductive potential of a population may not be significantly affected. It is understood that spills may result in significant short-term modification to habitat and local fishery resources during ensuing months following the event, with many factors ultimately influencing the intensity and duration of effects.

Impacts that would be critical to sustainability of salmon populations would include any that negatively influence survivability, reproductive success, limit movement and thus restrict continued populations' interaction or spawning potential, and/or result in long-term degradation of salmon habitat and associated ecological components/attributes. Water flow in a stream affects all aquatic life, and there is a definite relationship between annual flow regimes and the long-term quality of salmonid riverine habitat. Flow rates affect all salmon life stages, including the upstream migration of adults, survival of eggs, the emergence and viability of fry, and timing of smolt out-migration. A long-term reduction of flow within a system would increase the potential for systemic effects to resident salmon populations.

Impacts on populations from metals' contamination, as a result of hard rock mining, would result from loading within various environmental media (sediment and water). Transfer or release into biological receptor groups, including vegetation and benthic organisms, results in chronic exposure to fish via aqueous uptake and trophic exposure routes. Direct exposure to water-borne metals' contamination can cause both acute and chronic effects in fish, while impacts to their food resources (fish and benthic organisms) will likely result in indirect and long term impacts on fish populations. These effects can be associated with stressors of concern such as: fugitive dust dispersion; pipeline spills and episodic and large scale pollutions events when metal-laden tailings/slurry remains in a system; and AMD.

Impacts to habitat are associated with reduced flow, and with other stressors that result in elevated turbidities or embedded conditions, other changes to water quality parameters that are not conducive to fish sustainability, and physical changes to the environment during spill cleanups. AMD that results in long-term reduced water quality or reductions in food resources would also be considered as an impact to habitat.

As a result of the analysis for effects for each stressor, species were assessed relative to life history and life requisite information from both a temporal and spatial perspective. Using criteria of potential scope, severity and duration, impacts predicted for each stressor of concern were qualitatively ranked based on their potential to negatively alter salmon within the primary watersheds addressed by this ERA. For example, based on life history information on spawning habitat requirements, along with data from Woody (2009b) that identified coho and Chinook juveniles in streams associated with the mine site, *Dewatering and Loss of Instream Flow* impacts were deemed more critical to those species than other salmon species. Similarly, impacts expected from pipeline spills would be associated with site-specific downstream watersheds. As such, for a spill into the Newhalen River for example, habitat and species supported both in and downstream (i.e., Lake Iliamna) may potentially be affected. Based on this criterion, sockeye salmon fry would be at risk and thus a relatively higher impact factor was selected. Because of the spatial extent for each of the species under investigation and the large scope of the proposed mine, it was not possible to individually evaluate each stressor's effect for each stream. Also, because the location and extent for some of the chemical stressors is presently unknown, the impact factor was developed based on the preponderance of information for effects to ‘most’ salmonid species (or their habitat and/or supporting biological community) if an event occurred.

The risk analysis indicates that physical stressors would act to create secondary effects such as loss or reduction of supporting habitat for Chinook and coho salmon for the watersheds evaluated. This determination was based on data that indicates these two species were more prevalent in the local watersheds compared to sockeye, pink or chum. Overall, impacts expected from other individual physical stressors such as fugitive dust dispersion, pipeline spills and chemical spills, were deemed important primarily in portions of watersheds nearest to proposed extraction areas or near the spill location, and thus impacts should be relatively lower. But, it must be considered that these impacts will likely occur much earlier in the mine’s life and thus may act to magnify subsequent effects from ore spills and releases, or from long-term AMD.

Contribution of ‘clean’ surface water and groundwater along mainstem channels away from the mine should act to ameliorate negative impacts to habitat for salmon. But, the potential for AMD through both of these sources could result in habitat degradation further from the mine. Episodic and large scale pollution events and AMD, that could result in significant and long-term effects to populations and habitat (water quality) much further downstream, would result in a higher prediction of risk.

It must be reiterated that many of these stressors would occur simultaneously, creating synergistic effects which would tend to elevate a stressor’s risk potential. For instance, it is highly probable that even with mitigation and BMPs employed at the mine, copper and other metals will be mobilized in runoff or leached into surface and/or groundwater during the life of the mine. Long-term metals’ contributions to surface waters from dust generated at the mine would act to compound other physical (habitat loss, flow reduction) and chemical (spills, releases AMD) impacts expected from the mine’s creation and operation, resulting in cumulative impacts (see Cumulative Risk Analysis, Section 4.3) to salmon populations.

4.3 Cumulative Risk Analysis

- ***The magnitude and extent of the 'effect' of an action on a resource depends on whether cumulative impacts exceed the capacity of the resource to sustain itself and remain productive (USCEQ 1997).***
- ***Incremental increases in effects would slowly reduce salmon resistivity and result in magnification of each stress factor.***
- ***Over time, stressors would act synergistically to reduce habitat and food resources, increase effects to sensitive life stages, increase potential for fish kills, increase metals' bioavailability with short and long-term effects, and reduce genetic variability and disease resistance.***
- ***It is predicted that impacts to the surrounding ecosystem would expand over the course of the proposed mine's life. Risks to salmon and their supporting habitat would also increase over time and space as the mine expands. Development of additional mining interests in the area would increase risks.***

A cumulative impact has been defined as "*...the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future action...*" (U.S. Council on Environmental Quality [USCEQ] 1978). The National Environmental Policy Act [NEPA] directs that cumulative analyses are essential for effectively managing the consequences of human activities on the environment. The cumulative analysis necessarily involves assumptions and uncertainties, but provides a method for bringing useful information for making informed decisions (USCEQ 1997).

One of the fundamental issues related to determining cumulative impacts is defining the pre-development baseline condition (Dubé 2003). Baseline conditions provide a measure by which to assess changes to watersheds from a directed project and all other activities that may affect the watershed or resource in the future. Importantly, cumulative risk must consider both the spatial and temporal perspectives of the proposed action, all effects related to the action, and other actions that may have bearing on the resource or species of concern. Spatially, the scale of distribution for the identified species at risk may dictate the level of concern warranted. For instance, for wide-ranging species, society may be willing to accept a larger risk of error than for species that are specialized, endemic or in imminent danger of extinction (Ziemer 1994). Over time, care must be afforded to species that could be negatively affected by changes to supporting habitat through natural and anthropogenic factors in the near and distant future. Thus, the magnitude and extent of the 'effect' of an action on a resource depends on whether cumulative impacts exceed the capacity of the resource to sustain itself and remain productive (USCEQ 1997).

Analysis of cumulative risk on salmon viability within proximal watersheds associated with the proposed mine was based on a two-pronged approach. First, evaluation was made on the potential for individual stressors of concern to affect salmon and/or their supporting habitat, both from a spatial and temporal perspective. Second, the probability that stressors of concern could act synergistically to disrupt salmon populations' viability was considered. Again, both tools were used in the context of temporal and spatial prediction of effects (risk), as compared to current baseline salmon conditions.

From a temporal perspective, a stressor of concern's potential to affect or alter salmon populations considered factors such as distribution, longevity, target organism(s), form, persistence, toxicity and/or magnitude. As provided in the weight-of-evidence analysis (see above), impact potentials for 'populations' and 'habitat' generally indicate that some stressors would be relatively less important (Fugitive Dust, Chemical Spills, Pipeline Spills), with others more critical (Dewatering and Loss of Instream Flow, Loss or Alteration of Habitat, Episodic and Large Scale Pollution Events, AMD). First, an objective long-term prediction for independent effects to salmon population viability for each individual stressor of concern was considered over the proposed life of the mine, and beyond. For the purpose of evaluating a reasonably likely scenario, this analysis assumed that two (2) significant (i.e., ~100,000 - 200,000 gallon) pipeline spills would occur during the mine's operational life (see Section 3.2.4; Slurry Pipeline Breaks and Spills), and one significant episodic and large-scale pollution event (i.e., tailings pond release of ~25% of capacity) would occur (see Section 3.2.5; Episodic and Large Scale Pollution Events).

Generally, magnitudes and extent of all other stressors, excluding AMD, assumed continuous operations would result in increasing incremental stress (and thus risk) to salmonid populations within the watersheds under investigation. It was predicted that AMD generation would occur later in the mine's life (and beyond) and that impacts would increase dramatically near the mid-life stage of the mine. Finally, it was predicted that stressors of concern would act synergistically to exacerbate other physical and chemical effects and result in increased overall risk and lower viability for local salmon populations. For instance, when significant events occur in a watershed, such as an inadvertent dam release or other similar episodic spill event, salmon populations would most likely have little success recovering to pre-event levels because of the historical stress exerted on them from other mine-related stressors. AMD development in the older mine would exacerbate the negative effects on all life stages (and other biota), with risk increasing dramatically and population viability suffering for decades, or even centuries, into the future. The results of this exercise indicate that, based on risk predicted from the various stressors of concern, cumulative risk will likely follow an increasing upward trend over the life of the mine. The trend generally would be most relevant for risk to salmon over time within localized watersheds, but would also be important for other endemic species. The upward trend expected assumes that stressors would act in concert, but does not necessarily assume that all stressor effects would be additive. It also does not attempt to incorporate stressors (e.g., road construction, pipeline spills) that would affect salmon in other watersheds.

The evaluation of long-term impacts to salmon populations from man-made (anthropogenic) disturbances, as predicted for mines such as this proposed mine, is not new to fisheries scientists. The National Academy of Sciences (1996) provided discussions on salmon populations' responses to natural and anthropogenic disturbances. As provided in the NAS report, natural disturbances coupled with frequent, small anthropogenic disturbances results in long-term declines in salmon productivity (Figure 29a). They also note that a very large anthropogenic disturbance has typically been shown to have a significant short-term reduction in salmon productivity, with long-term consequences, where future productivity is much lower than prior to the large-scale event (Figure 29b).

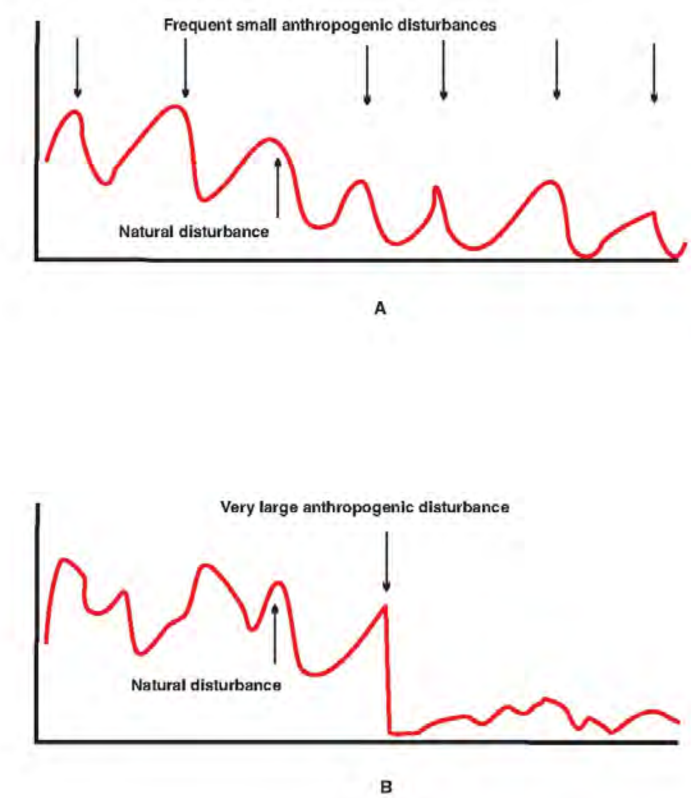


Figure 29. Hypothetical Response of Fish Populations to Natural and Anthropogenic Disturbances: *(A) Frequent small anthropogenic disturbances in concert with Natural Disturbances; (B) Single very Large Anthropogenic Impacts in concert with Natural Disturbance Regime* (Source: NAS 1996)

This evaluation predicts that mine construction and development would begin to affect local groundwater and surface water resources prior to mining commencement (see Figure 29a). Mine development includes land clearing, building of mine structures (mills, buildings, tailings storage structures and dams) and processing plants, and installation of all necessary equipment (Fourie and Hohm 1992). Next, access to the ore body encompasses removal of soil and barren rock to expose the ore bodies. This process is known as pre-production stripping. This process of stripping the surface away can take months to years. Throughout this process, dust from blasting, trenching, and excavation, in addition to truck and other vehicle traffic, would be created and dispersed across the mine site and beyond. Surface waters would be enveloped by the mine's footprint and groundwater would be used exclusively for construction and future production. It is predicted that construction of the proposed road and pipeline will likely result in impacts over many months. Although regulatory BMPs will be required, it is likely that impacts to streams would occur during this process. All of the pre-production activities, which could take several years, would initially act independently to alter proximal salmon habitat, although specific effects to populations may not yet be measureable during these initial phases.

After mining begins, ore exposure and removal would result in an incrementally larger mine footprint, with increasing amounts of tailings and waste rock generated on site. Through the extended mining period (40-70 years), effects exhibited on salmon habitat and populations (e.g., viability) from each of the stressors of concern would increase. This incremental increase in effects would slowly reduce salmon resistivity and result in magnification of each stress factor (i.e., reduced flow and water quality, reduced habitat quantity and quality, increased copper concentrations) produced. [This step in the risk analysis process did not include consideration of the stressors *Road Construction* or *Pipeline Spills* because it was understood that they would occur outside of the primary watersheds under consideration.] Although from a holistic perspective, it is expected that both of these stressors would act to reduce salmon viability in other watersheds over time. So, from a temporal perspective, cumulative risk to salmon populations associated with the proposed mine area is predicted to be moderate during early stages (years 0 – 25); with subsequent stages resulting in greater risk as each stressor, and their cumulative impact with other stressors, begin to exhibit greater and more pronounced effects on habitat, individual salmon health and population structure.

An *Episodic and Large Scale Pollution Event* during the mine's mid-life (at ~30 years) would most likely exacerbate pre-event natural and anthropogenic stress within local watersheds, with recovery of salmon populations to pre-event levels dubious [per information in Figure 29a and 29b]. The magnitude of the physical and chemical effects during latter stages of a mine's life (and beyond) could act to create environments where salmon, although possibly surviving, would have reduced distributions, limited available habitat, and be genetically susceptible to minor natural or anthropogenic disturbances. Long-term sustainability would most likely be jeopardized in the most critically affected portions of the watersheds. Also, it is predicted that AMD effects could occur during this period and well beyond.

The result of this exercise suggests that risk from the stressors of concern addressed by this ERA would act synergistically over time through: 1) reduction of habitat and food resources; 2) increased negative effects to sensitive salmon life stages as a result of reduced water budgets; 3) increased potential for fish kills; 4) increased bioavailability of metals in solution with subsequent short- and long-term systemic effects to individuals; 5) and reduced genetic variability and disease resistance.

Spatially, cumulative risks from stressors of concern will most likely develop in concert with temporal aspects as described above. *Dewatering and Loss of Instream Flow* would be expected in those portions of the watershed nearest to the mine proper during mine development and operation. Subsequently, reduction of groundwater discharge into down-gradient streams would be expected based on extraction for mine use and reduced upgradient recharge. *Loss or Alteration of Habitat* is expected as flows are reduced and channels re-established. Although most obvious in areas nearest the mine, lesser downstream reductions could affect tributaries and back-water areas that are important as salmon rearing habitat, and could lead to increased stranding, greater predation vulnerability and decreased productivity. As the mine ages (20-30 years), components such as refuse piles, waste rock and/or chemical storage areas would increase in size and become more difficult to manage properly. It is predicted that dust accumulation and transport, discharges, and/or spills would be likely to cause additive stress within the near-mine watersheds. Over time, it is expected that degradation of current high-value salmon habitat and its potential to sustain optimum populations would become more prevalent further away from the mine. Based upon the volume and distance of discharge, an *Episodic and Large Scale Pollution Event* could lead to both acute and chronic impacts within near and distal stream channels. The event in and of itself would most likely disrupt seasonal reproductive cycles and lead to reduced production outside of the zone of impact. Much of the discharged material would remain in the system with secondary effects such as embeddedness, turbidity and copper (and other metals) accumulation in sediment occurring in portions of the watershed much farther from the initial impact zone. These type effects would continue over time with fine-grained, copper-laden sediments (i.e., tailings) being continually transported further downstream with each major flood or snow-melt. As mine tailings ponds increase in size and duration, AMD is likely to occur. Effects within the near-mine watersheds would be expected first as groundwater becomes contaminated. As ponds and waste piles provide a continual AMD source, water quality reductions and downstream shifts in resident fish and invertebrate communities would be expected and result in reduction of salmon sustainability and production.

Although spatial cumulative risks are more difficult to predict, it is important to understand that the preceding risk characterization was based on the preliminary plan for the proposed mine as submitted to the Alaska Department of Natural Resources in 2006 as a part of Northern Dynasty's water rights application. That plan proposed mining 2.5 billion tons of ore (NDM 2006c). A recent news release by Pebble Limited Partnership (2010) indicates that the Pebble deposit has a mineral resource of 10.78 billion tons. History suggests that it is fairly standard in the industry to secure a permit for a smaller mine and then request expansion permits for more mining once the mine is in operation, has a workforce in place, and is paying taxes to local and state jurisdictions. For example, at the Zortman-Landusky mines in Montana, 21 amendments were approved by the regulatory agencies after the mines were initially permitted. This process of initial mine permitting, with subsequent expansions, was demonstrated in 2009 at several mines worldwide:

- Red Dog Mine, AK – expansion will double the life of the mine from 20 to 40 years;
- Keetac-Taconite Mine, MN – expansion will add 2000+ acres and increase output by approximately 33%;
- Smoky Canyon Mine, ID – expands mine by 1,100 acres and increases capacity by 38%;
- Cloudbreak Mine, Australia – major expansion project;
- Antamina Mine, Peru - extends the life of the mine until 2029 and increases ore processing by 38%;
- Metropolitan Colliery, Australia – extends life of mine by 20 years;
- Absaloka Mine, MT – increases mine size by 3,660 acres; and
- Kemess North Mine, BC – expands mine by using 269 hectare lake to store tailings and waste rock [was denied].

This information suggests that even if the initial design of a mine in the Pebble prospect area were to be smaller than that proposed by NDM in 2006, expansion in the future is possible and probable. Moreover, mining development on other claims in the region is also possible. This information suggests that impacts to the surrounding ecosystem would expand over the course of a mine's existence; with noted risks to salmon and their supporting watersheds also expected to increase over time and space as the mine grows. For example, Figure 30 provides a spatial rendition of future tailings ponds' locations that would be needed for storage based on the February 2010 news release. Although the locations and pond sizes as shown are speculative, they were developed considering local topography and information on currently described mine attributes.

In conclusion, this ERA has been developed based on both predicted and expected systematic perturbations and high-profile contamination events within the Nushagak-Mulchatna and Kvichak watersheds that presently support sustainable salmonid populations. Although it is uncertain if all the stressors described by this ERA will actually occur and result in degradation of habitat and reduced health and viability for salmon species (and their supporting ecosystems) that occur, based on historical information gathered for other similar mines and known effects of mining-related heavy metals to salmon and other biological populations, significant negative impacts to the aquatic ecosystem are to be expected over the life of the mine, and beyond.

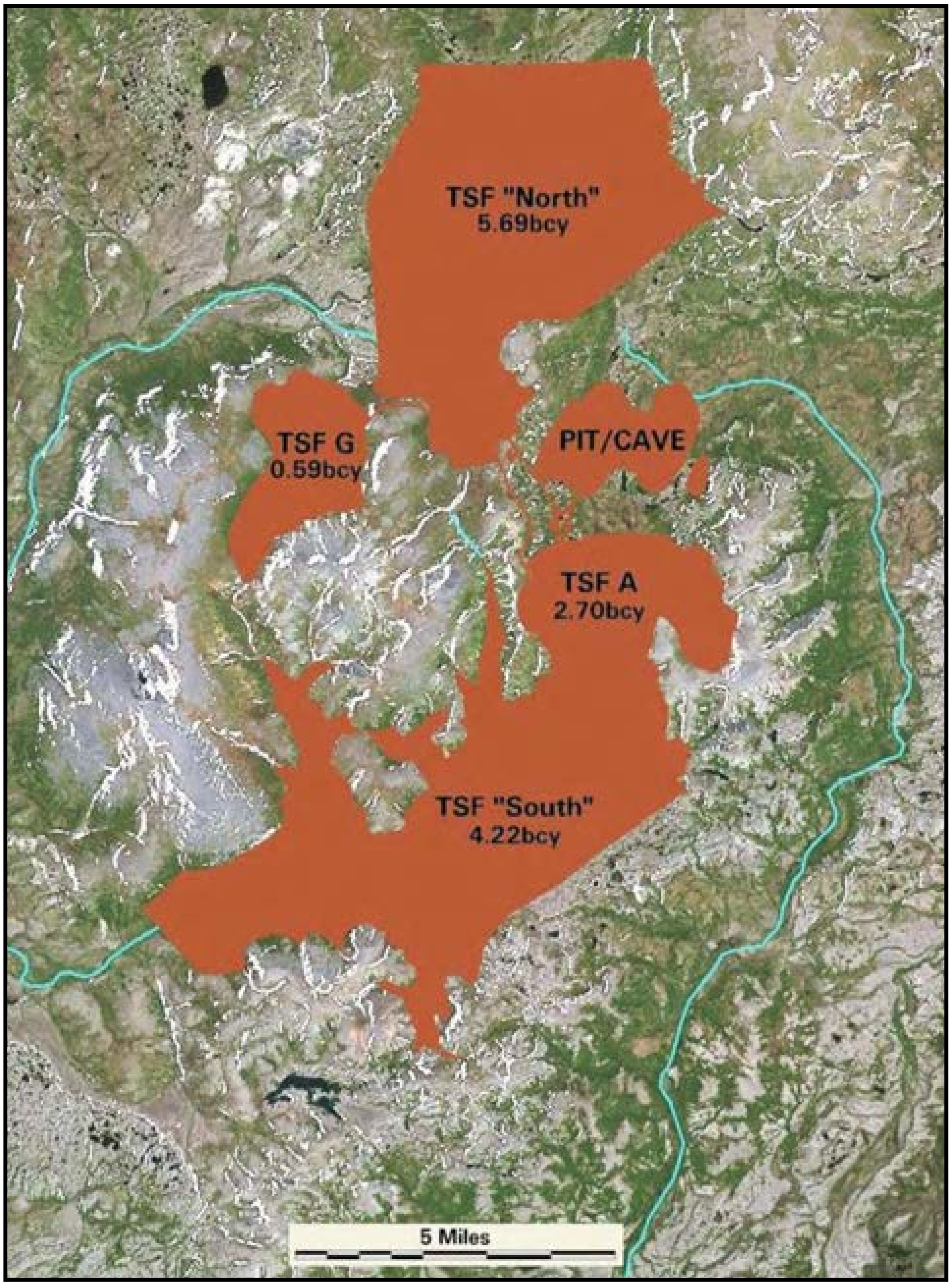


Figure 30. Hypothetical Rendition of Future Tailings Ponds Placement based on NDM February 2010 News Release of Pebble Deposit Mineral Ore Resource of 10.78 Billion Tons [image courtesy of The Center for Science in Public Participation and SkyTruth]

4.4 Loss of Salmon Production

- ***Determination of lost salmon production requires knowledge of stream habitat within affected watersheds.***
- ***ADFG predicted 2010 sockeye runs of 3.84M and 2.32M, respectively, for Kvichak and Nushagak-Mulchatna river systems***
- ***Stressors that act to affect proximal water bodies could reduce annual salmon production by 1% of 2010 levels.***
- ***Large-scale dam failures could potentially reduce salmon production in downstream portions of the Nushagak-Mulchatna watershed by 25-50% of 2010 levels.***

As discussed throughout this ERA, various impacts from proposed mine development, operation and/or closure could likely result in significant long-term changes to supporting, spawning and rearing habitat in portions of the Nushagak-Mulchatna and Kvichak watersheds. Although various impacts from mining have been shown to be highly probable based on historic information from similar hard rock mining methods, the specific relevance of these impacts to salmon production in affected streams has not yet been addressed.

Habitat alteration and loss can lead to salmon production loss (NAS 1996). Production declines when habitat alteration and loss impair the successful completion of life-history stages in the context of a watershed's landscape, its natural disturbance regime, and its anthropogenic changes (NAS 1996). Research has demonstrated that the quality of freshwater habitat (particularly over-winter habitat) has a direct influence on survival rate. Habitat quality determines the number of salmon smolts that a stream can produce as well as the efficiency with which those smolts are produced (i.e., survival rate).

Historically, models have been developed to estimate production potential and spawner escapement that account for differences in habitat quality (Nickelson 1998). The habitat limiting factors model (HLFM version 5.0; Nickelson *et al.* 1992) was first used in Oregon to estimate smolt potential based on population abundance for the spawning, spring rearing, summer rearing, and winter rearing life stages of coho salmon. The HLFM applied habitat-specific densities by the areas of individual habitat types that were derived from both summer and winter stream inventory data (Nickelson 1998). From this information, the model can then estimate potential smolts by applying survival rates from each of these life stages to the smolt stage. Typically, suitable winter-rearing habitat is in least supply compared with the other habitat types and thus can be the limiting factor to smolt production.

Similar approaches for determining production potential have been used in Oregon (USDOI) and throughout the Pacific Northwest:

- *Coho Salmon Production Potential in the Cle Elum River Basin, Storage Dam Fish Passage Study, Yakima Project, Washington, Technical Report Series No. PN-YDFP-007*, Bureau of Reclamation, Boise, Idaho, March 2007.
- *Assessment of Sockeye Salmon Production Potential in the Cle Elum River Basin,*
- *Storage Dam Fish Passage Study, Yakima Project, Washington, Technical Report Series No. PN-YDFP-008*, Bureau of Reclamation, Boise, Idaho, March 2007.
- *Coho Salmon Production Potential in the Bumping River Basin, Storage Dam Fish Passage Study, Yakima Project, Washington, Technical Report Series No. PN-YDFP-009*, Bureau of Reclamation, Boise, Idaho, March 2007.

These studies generally used two approaches to estimate coho salmon production potential; first, by estimating the number of spawning adults that the available spawning habitat would support, and second by estimating juvenile rearing/overwintering habitat that would be available in accessible river reaches. Suitable spawning habitat is primarily a function of substrate composition and suitable water velocity and depth – spawning site selection by fish is complex and likely based on a range of environmental or microhabitat conditions such as depth, flow, and substrate size (Bjornn and Rieser 1991). This can differ for the same species in different streams (McHugh and Budy 2004).

Other approaches that have been used for smolt density modeling include similar data requirements. Information such as habitat quantity and quality, and whether the habitat supports spawning and rearing, rearing only, or is only used for brief periods as transit corridors and is thus not considered to be spawning or rearing habitat, is typically required. In order to predict lost production from the various impacts discussed throughout the ERA, a comprehensive knowledge of salmon habitat parameters noted above in the affected portions of the watersheds is required. Critical to overall production estimation would be an understanding of the use of stream habitat during the winter period.

Alaska Department of Fish and Game (ADFG) develop annual forecasts for sockeye salmon run to Bristol Bay. Their most recent forecast (ADFG 2009) uses adult escapement and return data from brood years 1976-2006. The annual forecast is the sum of individual predictions for nine river systems (Kvichak, Alagnak, Naknek, Egegik, Ugashik, Wood, Igushik, Nushagak-Mulchatna and Todiak rivers) and for four age classes (age 1.2, 1.3, 2.2 and 2.3; plus ages 0.3 and 1.4 for Nushagak River). ADFG's current forecast predicts a total of 39.77 million sockeye are expected to return to Bristol Bay in 2010, which compares closely to their last 6 year forecasts where total runs were close to or exceeded 40 million sockeye salmon. For 2010, ADFG predicted total runs for the Kvichak and Nushagak-Mulchatna river systems of 3.84M and 2.32M sockeye, respectively (i.e., 15% of Bristol Bay forecast).

To predict the relative impact to [sockeye] salmon in the three streams evaluated by this ERA, it was necessary to assume that the number of sockeye returning to the watersheds was uniform across all streams that make up the Kvichak and Nushagak-Mulchatna river systems. Return data from ADFG (2010) was used as a surrogate for production based on the presumption that returning sockeye represented some [unknown] percentage of a stream system's smolt production. This is generally referred to as the smolt-to-adult return (SAR) rate and is specific to a watershed; generally varying from year to year.

SARs are typically determined by counting smolts as they migrate downstream and comparing this to the number of returning adults within a stream or river. Although it was well understood that the many habitat [and other environmental] features within these systems do not provide support to salmon uniformly, this approach was deemed reasonable and was determined to be a method for evaluating relative potential impacts from mining within the focus streams. For instance, both river systems contain large lakes with hundreds of tributaries. Sockeye salmon spawn not only in the rivers and streams in these systems but also along the beaches in these lakes, and some of these habitats may be utilized more than others and it is most likely that some are more productive than others (Baker 2010). Nevertheless, using this approach, an attempt was made to generally predict the impacts to sockeye from several of the stressors presented throughout the ERA.

No long term monitoring studies were found that compared pre- and post-mining salmon production rates, but it is predicted that effects within the localized smaller watersheds would act systemically to negatively affect production in the larger watersheds. In other words, decreased production would be exacerbated downstream and within connective tributaries as stressors' effects expand spatially. Again, as noted in Section 3.1.4.1, the *River Continuum Concept* for a stream translates to a smooth longitudinal gradient of conditions; thus the life requisites for supporting the biological community are interconnected and thus similarly at risk from upstream perturbations. As such, changes to a natural riverine complex will inevitably result in effects, to some degree, that can negatively impact fish (and other biota) viability along this gradient.

Using the National Hydrographic Dataset (NHD; USGS 2008), total stream lengths for both the Nushagak-Mulchatna and Kvichak watersheds were determined (see Figures 31 and 32). Next, for each of the various impact stressors considered, spatial and temporal loss of *production* was predicted based on the stressors' characteristics. Finally, the predicted relative percent loss of *production* [based on the 2010 ADFG forecast] was determined. For example, the Nushagak-Mulchatna watershed has a total stream length of 35,326 km. If all of the 326 km NFK watershed was rendered 'non-producing' as a result of some stressor, then this was determined to represent a loss of approximately 0.9% smolt *production* of the larger Nushagak-Mulchatna watershed.

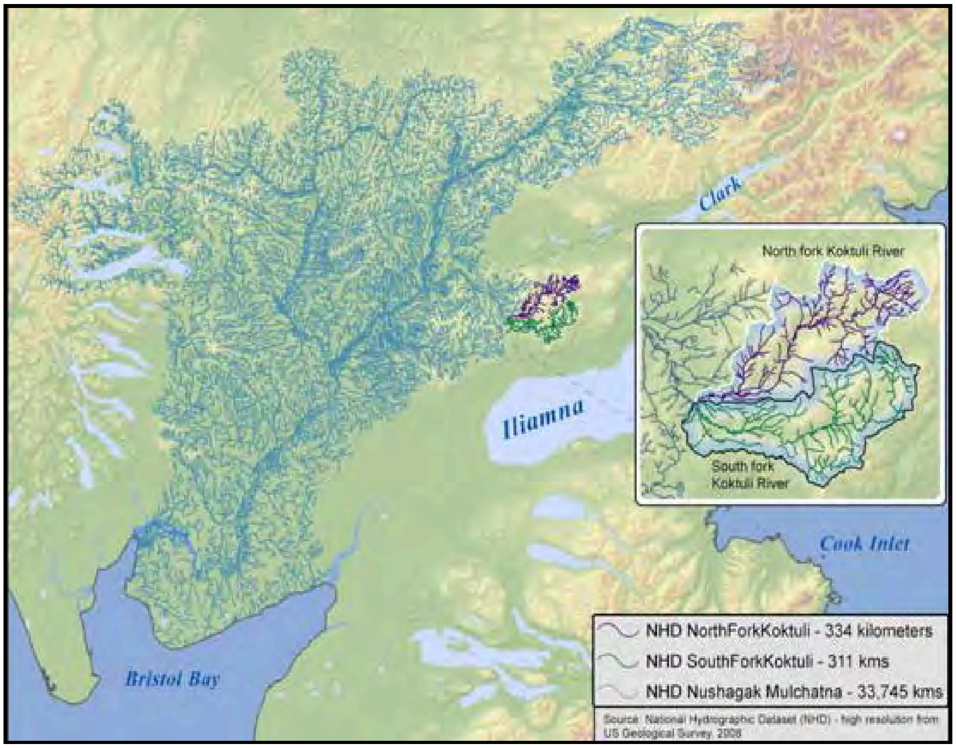


Figure 31. Watershed Size Comparison of North and South Fork Koktuli to Nushagak-Mulchatna River Systems

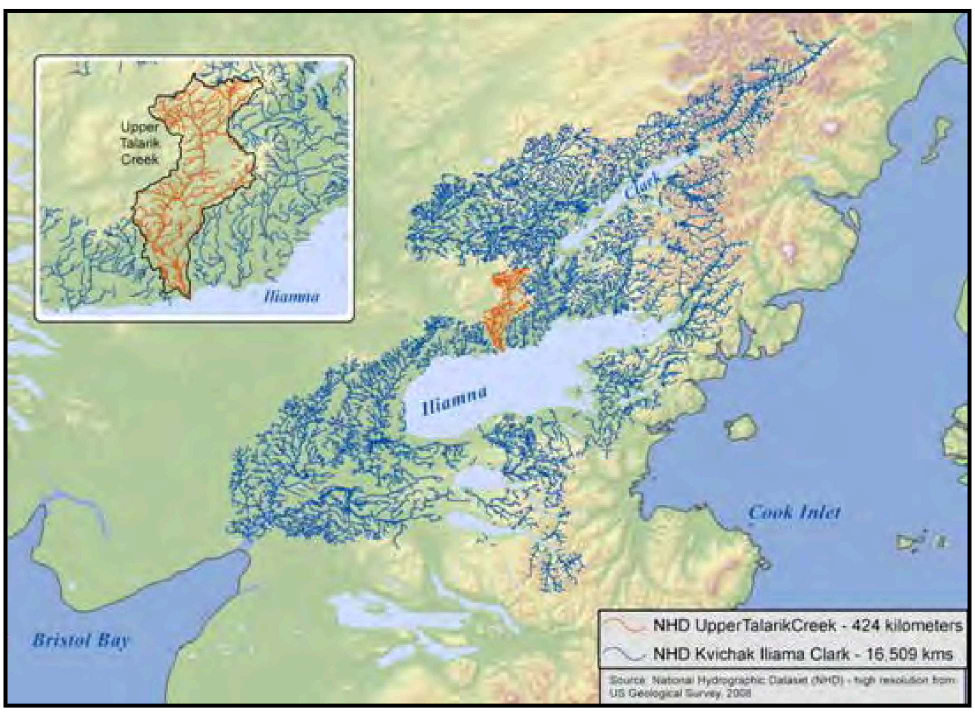


Figure 32. Watershed Size Comparison of Upper Talarik Creek to Kvichak-Iliamna-Clark River Systems

It was predicted that for those stressors which may impact a stream incrementally over time and space [i.e., copper and other metals' increased accumulation in water, sediments and biota from fugitive dust, AMD from waste piles runoff and/or increased discharge of contaminated groundwater further from the mine source], annual production losses within a watershed could result from the spatial expansion of these impacts and thus subsequent/additive production losses in the future.

Earlier analysis predicted that *Dewatering and Loss of Instream Flow* (including *Groundwater Discharge Loss*) and subsequent *Loss or Alteration of Supporting Habitat* would eliminate approximately 68 linear miles of stream channels, with 78 stream miles likely exhibiting some form of flow reduction. This in turn would result in greater competition for resources such as food and cover. For example, Washington's Department of Fish and Wildlife have developed empirically-based regression models that use stream flow indices as predictors for wild coho smolt production for several index stocks (Seiler *et al*. 2003). *Fugitive Dust* is expected to be generated during open construction and pit mining activities. Physical impacts from dust dispersion could affect a large area around the proposed mine, with metal-laden soil particles potentially contaminating sediment and water downstream from the impacted area. Other impacts such as inadvertent *Chemical Spills* could cause fish kills or other acute and/or chronic impacts to supporting food sources and habitat. All of these type impacts, if they occur, would likely result in reduced salmon production in the portions of the NFK, SFK and/or UTC watersheds proximal to the mine. The additive nature of these impacts could result in annual sockeye production loss near 1% of that derived by ADFG for the Kvichak and Nushagak-Mulchatna river systems. Temporal and spatial changes to sockeye production over time would depend on adherence to BMPs and/or any mitigation measures designed to reduce off-site contamination.

AMD could cause water quality changes in local and downstream watersheds resulting in increased bioavailability of copper (and other metals). If AMD develops, this would most likely result in impacts to relatively greater portions of the Kvichak and Nushagak-Mulchatna river systems compared to impacts discussed above, and could potentially make them unable to support salmon. The spatial extent for the effects from AMD would be dependent on whether discharges are continuous, as a result of groundwater contamination or surface water runoff, or a result of episodic spills or discharges from tailings ponds, or both. If chronic AMD occurs in mid to latter stages of the mine's life, it could impact most of the watersheds under investigation. Xiao *et al.* (2010) described environmental impacts from AMD from the Dexing Copper Mine (largest open pit mine in Asia). Their investigation found AMD up to 75 km from the mine in the Le An River (Xiao 2010). Similar effects within the Nushagak-Mulchatna and Kvichak watersheds could result in changes to water quality well outside of the focus watersheds and potentially cause production losses of greater than 1-2% of the ADFG forecast for both the Nushagak-Mulchatna and Kvichak river systems.

Spills or releases could result in extended impact areas, with salmon production losses associated with the volume released. If AMD develops and is released, sockeye salmon production losses to 1% (or greater) could occur within the Kvichak and Nushagak-Mulchatna river systems. Using the 2010 ADFG sockeye run forecast noted above as baseline, this would mean that if all annual NFK *production* was lost, conservatively, there would be a decrease of 23,200 smolts (e.g., 1%) throughout each of the Nushagak-Mulchatna and Kvichak watersheds. Again, return data from ADFG (2010) was used as a surrogate for production based on the presumption that returning sockeye represented some [unknown] percentage of a stream system's smolt production. Thus, smolt losses would theoretically be much greater than the predicted number of returning adult sockeye predicted to be lost.

An *Episodic and Large Scale Pollution Event* could affect large portions of the watershed down-gradient from the tailings dam. Lost salmon production could be significant considering the size of the dams planned and the tailings volumes proposed. Based on the analysis of the travel distance for a release within either the NFK or SFK, affected portions of the Nushagak-Mulchatna watershed could result in production losses throughout mainstem channels all the way to Bristol Bay. Post-event production losses very well could be up to 25-50% of the ADFG 2010 levels for many years following a release.

In summary, it is impossible to predict the specific loss of production for salmon found within the watersheds associated with proposed mine activities. As stated previously, mine management practices have not yet been provided, and extraordinary weather events that could trigger large-scale impacts are always unknown. But, based on historical findings from other similar large hard rock mines, it can be predicted with some certainty that salmon (and other indigenous species) will exhibit some effects both temporally and spatially, with subsequent production loss inevitable. Considering the potential for further mine development, as noted in Section 4.3 Cumulative Risk Analysis, continued emphasis is needed on assessing the possible impacts to salmon production.

4.5 Uncertainty Analysis

- ***The predictive ERA relied on assumptions, but attempted to reduce uncertainty through professional judgment, analogy with similar chemicals and conditions, and data/information from other mine sites.***
- ***Uncertainty is low for water loss and habitat alteration and creation of dust at Pebble Mine, but high for the potential failure of a tailings dam.***
- ***The uncertainty of impacts from dust runoff into streams and road construction is high, but it is highly certain that over the life of the mine pipeline breaks will occur with subsequent impacts to salmon highly likely.***
- ***Based on historical information from other hard rock mines, there is a high certainty that AMD will develop during the life of the mine and affect downstream waterbodies.***

In an ERA, the uncertainty analysis is an integral part of the characterization of exposure assessment and thus risk prediction. Since this is a predictive ERA, data were unavailable for directly characterizing exposure. As a result, the assessors had to rely on assumptions that inherently include varying degrees of uncertainty. In order to reduce the uncertainties associated with these assumptions, a combination of professional judgment, inferences based on analogy with similar chemicals and conditions, along with estimation techniques and developed data for other mine sites, were used. The following provides information on sources of these uncertainties, and describes the methods used to minimize them, with a final goal of providing relevant characterization of ecological risks associated with each stressor addressed within the ERA.

During the initial stages of the ERA, information developed to date by mining proponents was reviewed to understand the potential scope and breadth of the project and to characterize the large-scale mining proposal. Project specific information that was publically available was reviewed from The Pebble Partnership website (http://www.pebblepartnership.com/), ADNR's Division of Mining, Land and Water (http://dnr.alaska.gov/mlw/mining/largemine/pebble/env_baseline_studies.htm) or any other relevant and reliable sources that provided information on the proposed project location, such as the USGS site (http://pubs.usgs.gov/of/2008/1132/). This effort resulted in a general understanding of the project for facilitating more precise estimates and characterization of risk, but was not comprehensive enough to answer many of the questions regarding mine operations and management. As a result, uncertainty exists for issues such as:

- Tailings/Waste locations (large areas will be required, but specific locations are presently unknown);
- Seismic risk (information is needed on the exact location of the nearest faults and what the proposed mine would use as the design event for tailings dams construction);
- AMD potential (it is known that site rock is potentially AMD producing, but it is unclear how much will be formed);
- Mining methods, amounts, and sequence of mining (it is expected that the proposed mine will use open pit and underground block caving,);
- Road construction materials and method; and
- Dust (it is well known that dust could be a factor is spreading contamination, but quantitative determination has not yet been made).

Development of a comprehensive MMP would facilitate more precise estimates and characterization of risk related to mine development, operation and closure.

For many of the stressors of concern evaluated, ambient meteorological conditions in the proposed project area can play major roles for both timing and sequence, and in their influence on the level of effect. For instance, although flow reduction and subsequent habitat alteration are expected during and after mine creation, rainfall timing and magnitudes can both exacerbate or ameliorate effects predicted within watersheds. Precipitation amounts can influence groundwater levels and movement, surface water dilution capacities and increase or decrease the potential for large-scale pollution events and AMD formation and release. Rainfall, in addition to wind direction and speeds, can affect dust creation, dispersion and concentrations. Finally, seasonal temperature fluctuations and extremes can play a major role in the health and viability of salmon populations, by ultimately affecting a population's resistivity to possible impacts associated with mining. Based on the above, there is much uncertainty associated with predicted stressor impact levels and timing when ambient meteorological conditions and trends are considered.

The amount and degree of impacts to salmon individuals and populations, as predicted in the ERA, generally assumes that the five species spawn, rear and mature within the watersheds being evaluated. Also, without site-specific knowledge, the ERA assumes that habitat requisites are consistent throughout the watersheds. These assumptions were necessary in order to predict with consistency the effects from potential changes in the environment from mining-related activities. As a result, uncertainty exists regarding the level of effects expected to each salmon species under investigation. Although the assessors understood that this could affect predicted risk, it was felt that uncertainty was reduced through information on ADFG-designated anadromous streams, which indicated that salmon species were prevalent within the directed watersheds.

The following provides information on uncertainty for each of the stressors of concern identified and analyzed in the ERA.

Dewatering and Loss of Instream Flow [including Groundwater Discharge] and Loss or Alteration of Supporting Habitat

Based on the details of the 2006 water use permit applications, there is very little uncertainty associated with the analysis of loss and reduction of stream flow in watersheds near the proposed mine. Of course, as mentioned above, occurrence of specific salmon species in all portions of these watersheds has not yet been definitively established, but the various effects levels (per HSI models) used in the analysis are fairly consistent between species. The geographic applicability of the HSI models used for assessing potential effects will add some uncertainty to the evaluation, but many of these models have been developed from studies in Alaska. The extent for instream flow reduction in streams was based on surface runoff potentials only, because no groundwater flow data was available. Although the uncertainty associated with this analysis could not be quantified, it is important to understand that the mine proposes to use all groundwater within the project site for operational activities, so groundwater contributions to down-gradient flow may be compromised. An understanding of the impacts from mining activities on groundwater migration and flow would help reduce the uncertainty associated with impact prediction.

Road Construction

Uncertainties associated with construction of the proposed road include those related to timing, materials and long-term management practices. Since, to date, no information has been provided on these issues, there is much uncertainty regarding the specific impacts to salmon from this activity. Seasonal timing of road construction activities near stream channels could reduce or increase effects predicted from turbidity and sedimentation. Material used for road construction would be critical to understanding the risk potentials for both construction and operational phases of the project. Although information on culvert types expected for stream crossings is unknown, the uncertainty associated with effects to salmon movement from culverts placement is low. Information from Alaska and other states which showed that culverts have historically resulted in impacts to salmon was used to reduce the uncertainty of impacts predicted from this source.

Fugitive Dust (Physical and Chemical)

Uncertainties related to impacts predicted from fugitive dust are associated with lack of information on specific construction methods and timing, and management practices to be used for dust suppression. As mentioned above, unknown meteorological conditions during construction and operations can either increase or decrease predicted impact levels and distributions, but local wind conditions were included in the assessment to reduce these as much as possible. Based on data developed from another Alaska mine, there is little uncertainty regarding the potential for dust generation, but it is still unclear what management steps will be taken to minimize these levels. The chemical impacts predicted from copper runoff and leaching from dust-laden soils has a high degree of uncertainty, but factors related to soil pH, copper leachability, transport and retention were built into the models to reduce this to the greatest degree possible.

Chemical Spills

Impacts associated with chemical spills were not addressed beyond listing the types of chemicals that are present during mining activities and the potential effects that can occur from releases. The potential for chemical spills to occur during the multi-year construction and 40-70 year life of the mine is high. There is little uncertainty that some of these spills will affect salmon. The longevity of these effects is unknown and would not be known until a spill occurs. A MMP that addresses the chemicals of concern and response activities required would provide the approach for dealing with spills and limiting salmon exposures.

Pipeline Spill

Discharge scenarios and potential impacts associated with slurry pipeline breaks were based on well-documented historical information at similar mines across the U.S. and Canada. Based on the information that was developed, there is a high certainty that at some time during the proposed mine’s life a pipeline break will occur. Since many pipeline breaks occur near stream crossings, it is also expected that streams and biota would be at risk. Although the magnitude of a spill is always an unknown, for this ERA, the analysis used a nominal predicted volume that was well within upper and lower release volumes identified in historical records and reports. Transport mechanisms and stream flow values were based on literature- and agency-derived information and thus have a high degree of certainty. Potential fate and effects within a stream, should a spill occur, are generally related to flow volume and seasonality, so only inferences to salmon impacts could be derived from the analysis.

Episodic and Large Scale Pollution Events

As with other aspects of this ERA, there is little certainty that a dam failure and slurry release will occur at the proposed mine. Although, there is a very high probability that if a spill does occur, it would be catastrophic within and down-gradient of the primary watersheds addressed throughout this ERA. The regression models used for determining runout distances from other mine discharges had a moderate degree of success by the original authors in their evaluation. The proposed size of the dams at the proposed mine were off the scale and much larger than any of the events used in their evaluation. As such, one can only infer that larger dams and greater volumes would equate to worse effects. But, these factors in and of themselves do not increase the certainty that a dam failure will occur. Other information sources have noted the increased potential for dam failures due to seismic activity in the area near the mine. This, along with the size of the dams and volumes expected, makes even a low probability for dam failure critical to predicting effects to salmon populations and habitat.

Acid Mine Discharge (AMD)

Based on geochemical evaluation of rock samples from the Pebble formation, there is a high certainty that acid would be produced in waste piles, other mine refuse and within tailings ponds. Also, because the mine is to be developed in an area with moderate precipitation, numerous small streams, a high water table, and over geological formations that are susceptible to ground water movement, AMD formation and movement is highly likely. Based on these conditions, and historical information on AMD at other hard rock mine, there was a high certainty that waters with pH levels near (or below) 4 would be discharged near mid-life stages of the mine and/or beyond. As a result, some effects would be exhibited in downstream water bodies. Calculation of potential pH levels in the SFK and NFK watersheds only considered surface water contributions for dilution, so there is much uncertainty regarding groundwater's influence on these predictions.

4.5 Conclusion

This ecological risk assessment identified a wide range of specific, significant risks to the salmon ecosystems of the Nushagak-Mulchatna and Kvichak watersheds under a specific large-scale mining scenario. Overall, the risk to wild salmon populations of such large-scale mining in this region is very high. Of the wide range of risks analyzed in this assessment, the high likelihood of acid mine drainage both during and after mine operations, the potentially catastrophic though highly uncertain nature of a large-scale pollution event, and the potential cumulative effects of various ecosystem stressors over time, are reasonable cause for significant concern regarding the long-term abundance, diversity and sustainability of salmon species (and their supporting ecosystems) in this region. Although it is uncertain what will actually occur, based on historical information on physical and chemical stressors gathered for other large mines, and the known effects of mining-related heavy metals to salmon and other biological populations, significant negative impacts to the aquatic ecosystem would be expected over the life of large-scale mines in this region. Additionally, such impacts would be likely to persist and in some cases increase long after mine closure.

Finally, cumulative risk associated with construction, maintenance and closure of such mines may also be magnified by concurrent or subsequent development of additional mining interests in the region depending on their location and design.

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APPENDIX A

Estimated Pre- and Post-Development Subbasin Monthly Discharges

Table A-1. Estimated Pre- and Post-Development Subbasin Monthly Discharges for North Fork Koktuli River

Pre-Development												
Subbasin	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**Total**
Jan	14.8	2.8	1.0	2.6	5.7	6.2	3.0	7.2	8.9	16.6	2.5	71.3
Feb	12.7	2.4	0.8	2.2	4.9	5.3	2.6	6.2	7.6	14.2	2.2	61.1
Mar	12.7	2.4	0.8	2.2	4.9	5.3	2.6	6.2	7.6	14.2	2.2	61.1
Apr	46.7	9.0	3.1	8.1	17.8	19.4	9.5	22.7	27.9	52.1	7.9	224.2
May	201.5	38.7	13.4	34.9	76.9	84.0	41.1	98.1	120.6	224.9	34.3	968.2
Jun	70.0	13.4	4.7	12.1	26.7	29.2	14.3	34.1	41.9	78.1	11.9	336.3
Jul	36.0	6.9	2.4	6.2	13.8	15.0	7.4	17.6	21.6	40.2	6.1	173.3
Aug	42.4	8.1	2.8	7.3	16.2	17.7	8.7	20.7	25.4	47.3	7.2	203.8
Sep	97.5	18.7	6.5	16.9	37.2	40.7	19.9	47.5	58.4	108.9	16.6	468.8
Oct	74.2	14.2	4.9	12.8	28.3	30.9	15.2	36.2	44.4	82.8	12.6	356.7
Nov	74.2	14.2	4.9	12.8	28.3	30.9	15.2	36.2	44.4	82.8	12.6	356.7
Dec	42.4	8.1	2.8	7.3	16.2	17.7	8.7	20.7	25.4	47.3	7.2	203.8
Post-Development												
Jan	14.8	2.8	1.0	2.6	5.7	0.5	3.0	7.2	8.9	16.1	2.5	65.1
Feb	12.7	2.4	0.8	2.2	4.9	0.4	2.6	6.2	7.6	13.8	2.2	55.8
Mar	12.7	2.4	0.8	2.2	4.9	0.4	2.6	6.2	7.6	13.8	2.2	55.8
Apr	46.5	9.0	3.1	8.1	17.8	1.6	9.5	22.7	27.9	50.6	7.9	204.6
May	200.7	38.7	13.4	34.9	76.9	6.8	41.1	97.9	120.6	218.4	34.3	883.6
Jun	69.7	13.4	4.7	12.1	26.7	2.4	14.3	34.0	41.9	75.9	11.9	306.9
Jul	35.9	6.9	2.4	6.2	13.8	1.2	7.4	17.5	21.6	39.1	6.1	158.1
Aug	42.3	8.1	2.8	7.3	16.2	1.4	8.7	20.6	25.4	46.0	7.2	186.0
Sep	97.2	18.7	6.5	16.9	37.2	3.3	19.9	47.4	58.4	105.8	16.6	427.8
Oct	73.9	14.2	4.9	12.8	28.3	2.5	15.2	36.1	44.4	80.5	12.6	325.5
Nov	73.9	14.2	4.9	12.8	28.3	2.5	15.2	36.1	44.4	80.5	12.6	325.5
Dec	42.3	8.1	2.8	7.3	16.2	1.4	8.7	20.6	25.4	46.0	7.2	186.0

Table A-2. Estimated Pre- and Post-Development Subbasin Monthly Discharges for South Fork Koktuli River

Subbasin	A	B	C	D	E	F	G	H	I	J	K	L	Total
Pre-Development													
Jan	9.5	5.7	10.9	11.8	15.6	11.4	9.6	12.5	15.2	13.0	12.8	9.5	137.5
Feb	6.1	3.7	7.0	7.5	10.0	7.3	6.2	8.0	9.7	8.3	8.1	6.1	87.8
Mar	4.6	2.8	5.3	5.7	7.6	5.5	4.7	6.1	7.3	6.3	6.2	4.6	66.6
Apr	7.3	4.4	8.4	9.1	12.0	8.8	7.4	9.6	11.7	10.0	9.8	7.3	105.8
May	61.7	37.0	70.6	76.2	100.7	73.5	62.3	80.9	97.9	83.9	82.4	61.3	888.5
Jun	20.6	12.3	23.5	25.4	33.6	24.5	20.8	27.0	32.6	28.0	27.5	20.4	296.2
Jul	10.3	6.2	11.8	12.7	16.8	12.3	10.4	13.5	16.3	14.0	13.7	10.2	148.1
Aug	11.5	6.9	13.1	14.1	18.7	13.7	11.6	15.0	18.2	15.6	15.3	11.4	165.0
Sep	47.0	28.2	53.8	58.0	76.7	56.0	47.5	61.6	74.6	63.9	62.8	46.7	676.9
Oct	30.8	18.5	35.3	38.1	50.4	36.8	31.2	40.4	49.0	42.0	41.2	30.7	444.2
Nov	23.5	14.1	26.9	29.0	38.4	28.0	23.7	30.8	37.3	32.0	31.4	23.4	338.5
Dec	14.7	8.8	16.8	18.1	24.0	17.5	14.8	19.3	23.3	20.0	19.6	14.6	211.5
Post-Development													
Jan	0.0	0.0	8.4	11.7	15.6	11.4	9.6	12.5	15.2	13.0	12.8	9.5	119.7
Feb	0.0	0.0	5.4	7.5	10.0	7.3	6.2	8.0	9.7	8.3	8.1	6.1	76.4
Mar	0.0	0.0	4.1	5.7	7.6	5.5	4.7	6.1	7.3	6.3	6.2	4.6	58.0
Apr	0.0	0.0	6.5	9.0	12.0	8.7	7.4	9.6	11.7	10.0	9.8	7.3	92.0
May	0.0	0.0	54.3	75.8	100.7	73.5	62.3	80.9	97.9	84.0	82.4	61.3	773.2
Jun	0.0	0.0	18.1	25.3	33.6	24.5	20.8	27.0	32.6	28.0	27.5	20.4	257.7
Jul	0.0	0.0	9.1	12.6	16.8	12.2	10.4	13.5	16.3	14.0	13.7	10.2	128.9
Aug	0.0	0.0	10.1	14.1	18.7	13.6	11.6	15.0	18.2	15.6	15.3	11.4	143.6
Sep	0.0	0.0	41.4	57.7	76.8	56.0	47.5	61.6	74.6	64.0	62.8	46.7	589.1
Oct	0.0	0.0	27.2	37.9	50.4	36.7	31.2	40.4	49.0	42.0	41.2	30.6	386.6
Nov	0.0	0.0	20.7	28.9	38.4	28.0	23.8	30.8	37.3	32.0	31.4	23.3	294.6
Dec	0.0	0.0	12.9	18.0	24.0	17.5	14.8	19.3	23.3	20.0	19.6	14.6	184.1

Table A-3. Estimated Pre- and Post-Development Subbasin Monthly Discharges for Upper Talarik Creek

Pre-Development																
Subbasin	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**	**O**	**Total**
Jan	2.1	15.3	5.3	4.2	15.2	7.0	3.2	9.1	25.1	3.7	7.8	5.2	4.1	4.6	8.6	120.4
Feb	1.2	8.9	3.1	2.4	8.8	4.1	1.9	5.3	14.6	2.2	4.6	3.0	2.4	2.7	5.0	70.3
Mar	0.9	6.4	2.2	1.7	6.3	2.9	1.3	3.8	10.5	1.5	3.3	2.2	1.7	1.9	3.6	50.2
Apr	5.4	39.5	13.7	10.8	39.2	18.2	8.3	23.6	64.8	9.5	20.2	13.4	10.6	11.8	22.1	311.2
May	8.8	63.7	22.2	17.4	63.2	29.3	13.4	38.1	104.6	15.4	32.5	21.6	17.1	19.1	35.7	501.9
Jun	4.2	30.6	10.6	8.4	30.3	14.1	6.4	18.3	50.2	7.4	15.6	10.3	8.2	9.2	17.1	240.9
Jul	3.2	22.9	8.0	6.3	22.7	10.5	4.8	13.7	37.6	5.5	11.7	7.8	6.2	6.9	12.9	180.7
Aug	3.0	21.7	7.5	5.9	21.5	10.0	4.5	13.0	35.5	5.2	11.1	7.3	5.8	6.5	12.1	170.6
Sep	8.1	58.6	20.4	16.0	58.1	27.0	12.3	35.1	96.2	14.2	29.9	19.8	15.7	17.6	32.8	461.7
Oct	6.7	48.4	16.8	13.2	48.0	22.3	10.1	29.0	79.5	11.7	24.7	16.4	13.0	14.5	27.1	381.4
Nov	6.1	44.6	15.5	12.2	44.2	20.5	9.3	26.7	73.2	10.8	22.8	15.1	12.0	13.4	25.0	351.3
Dec	3.3	24.2	8.4	6.6	24.0	11.1	5.1	14.5	39.7	5.9	12.4	8.2	6.5	7.3	13.6	190.7
Post-Development																
Jan	2.1	15.3	5.3	4.2	3.0	7.0	3.2	9.1	25.1	3.7	7.8	5.2	4.1	4.6	8.6	108.2
Feb	1.2	8.9	3.1	2.4	1.7	4.1	1.9	5.3	14.6	2.2	4.5	3.0	2.4	2.7	5.0	63.1
Mar	0.9	6.4	2.2	1.7	1.2	2.9	1.3	3.8	10.5	1.5	3.2	2.2	1.7	1.9	3.6	45.1
Apr	5.4	39.5	13.7	10.8	7.7	18.0	8.3	23.6	64.8	9.5	20.1	13.4	10.6	11.8	22.1	279.5
May	8.8	63.7	22.2	17.4	12.4	29.0	13.3	38.1	104.6	15.4	32.5	21.6	17.1	19.1	35.7	450.8
Jun	4.2	30.6	10.6	8.4	5.9	13.9	6.4	18.3	50.2	7.4	15.6	10.3	8.2	9.2	17.1	216.4
Jul	3.2	22.9	8.0	6.3	4.5	10.4	4.8	13.7	37.6	5.5	11.7	7.8	6.2	6.9	12.9	162.3
Aug	3.0	21.7	7.5	5.9	4.2	9.9	4.5	13.0	35.6	5.2	11.0	7.3	5.8	6.5	12.1	153.3
Sep	8.1	58.6	20.4	16.0	11.4	26.7	12.3	35.1	96.2	14.2	29.9	19.8	15.7	17.6	32.9	414.8
Oct	6.7	48.4	16.8	13.2	9.4	22.0	10.1	29.0	79.5	11.7	24.7	16.4	13.0	14.5	27.1	342.6
Nov	6.1	44.6	15.5	12.2	8.7	20.3	9.3	26.7	73.2	10.8	22.7	15.1	12.0	13.4	25.0	315.6
Dec	3.3	24.2	8.4	6.6	4.7	11.0	5.1	14.5	39.7	5.8	12.3	8.2	6.5	7.3	13.6	171.3

APPENDIX B

Habitat Suitability Index Variables, Description, and Associated Life Stage for Coho, Chinook, Chum, and Pink Salmon

Table B-1. Habitat Suitability Index Variables, Description, and Associated Life Stage for Coho, Chinook, Chum, and Pink Salmon[1]

Variable Number	Habitat Variable Description	Life Stage Affected	Maximum Suitability Index Description
Coho Salmon			
V_1	Maximum temperature during upstream migration	Adult	up to 11 degrees C
V_2	Minimum dissolved oxygen during upstream migration		> 6.5 mg/l
V_3	Maximum temperature from spawning to fry emergence	Spawning/embryo /alevin	Between 5 degrees C and 12 degrees C
V_4	Minimum dissolved oxygen saturation levels from spawning to fry emergence		80%
V_5	Substrate composition in riffle/run areas		>50% gravel and rubble <u>or</u> <5% fines (e.g., particles < 6mm)
V_6	Maximum temperature during rearing (parr)	Parr	9 – 13 degrees C
V_7	Minimum dissolved oxygen during rearing (parr)		up to 8 mg/l
V_8	Percent canopy over rearing stream		50% to 75%
V_9	Riparian vegetation index in summer		150 and above (based on formula where ≥ 75% deciduous shrubs and trees rates excellent)
V_{10}	Percent pools during summer low flow periods		Between 45% and 60%
V_{11}	Proportion of pools during summer low flow period that are 10-80 m^3 or 50-250 m^2, and have sufficient riparian canopy cover		Above 75%
V_{12}	Percent instream and bank cover during summer low flow period		Above 35%
V_{13}	Percent total area with quiet backwaters and deep (≥ 45 cm) pools with good in water habitat.		Above 30%
V_{14}	Maximum temperature during (A) winter in rearing streams and (B) spring-early summer in streams where seaward smolt migration occurs	Smolt	(A) – not greater than 8 degrees C (B) – not greater than 12 degrees C
V_{15}	Minimum dissolved oxygen during spring-early summer period in streams where seaward migration occurs		Not less than 8 mg/l

Table B-1. Habitat Suitability Index Variables, Description, and Associated Life Stage for Coho, Chinook, Chum, and Pink Salmon[1]

Variable Number	Habitat Variable Description	Life Stage Affected	Maximum Suitability Index Description
Chinook Salmon			
V_1	Annual maximum or minimum pH as measured in summer and fall (using lowest SI value).	Adult	6.5 to 8.0
V_2	Maximum temperature during warmest periods when adults or juveniles present	Adult, Juvenile	A = prespawning adults – 7 to 12 degrees C B = juveniles – 12 to 18 degrees C
V_3	Minimum dissolved oxygen levels during egg and pre-emergent yolk sac fry period; and during occupation by adults and juveniles	Embryo, Juvenile	8 mg/l at ≤ 5 degrees C 12 mg/l at >10 degrees C
V_4	Percent pools during late growing season / low water period	Adult, Juvenile	40% to 60%
V_5	Pool class rating during the late growing season / low flow period		Variable based on percentage of pools in habitat
V_6	Maximum or minimum temperature at beginning and end of first month of spawning of late summer or fall spawning stocks. (using lowest SI value) [minimum temperature must remain ≥ 4.5 degrees C for ≥ 3 ½ weeks after fertilization	Spawning/embryo	4.5 to 13 degrees C
V_7	Maximum or minimum temperature at beginning and end of embryo incubation period. Use the temperature that yields the lowest SI. [applicable to spring spawning stocks only]	Embryo	6.0 to 14 degrees C
V_8	Percentage of spawning gravel in two classes	Spawning, Embryo, Fry	Based on spatial assessment of gravel types
V_9	Average water column velocity (cm/s) over areas of spawning gravel used by Chinook salmon		Velocity of 30 cm/s to 90 cm/s
V_{10}	Average percentage of fines in spawning gravel – includes silts (≤0.8mm) and sand (0.8 to 30mm)		~ 5% or less

Table B-1. Habitat Suitability Index Variables, Description, and Associated Life Stage for Coho, Chinook, Chum, and Pink Salmon[1]

Variable Number	Habitat Variable Description	Life Stage Affected	Maximum Suitability Index Description
V_{11}	Average annual base flow during the late summer to later winter low-flow period as percentage of the average daily flow. For embryo and pre-emergent fry use the average and low flows that occur during intergravel occupation period.	Embryo, Juvenile	50%
V_{12}	Average annual peak flow as multiple of average annual daily flow	Embryo, Standing crop	Multiple of 2 to 3
V_{13}	Predominant (≥50%) substrate type in riffle-run areas for food production indicator – for juvenile rearing and upstream areas.	Juvenile, Standing crop	Rubble or small boulders dominate; limited amounts of gravel, large boulders or slab rock present; no fines.
V_{14}	Average percentage of fines (<3 mm) in riffle-run areas		10% or less
V_{15}	Nitrate-nitrogen (mg/l) in late summer after spawner die off		0.15 – 0.25 mg/l
V_{16}	Percentage of stream area providing escape cover – late summer-fall average to low flow period at depths ≥ 15 cm and with bottom velocities ≤ 40 cm/s.	Juvenile	20 – 50 %
V_{17}	Percentage of stream area with 10 to 40 cm average sized boulders. [only for juveniles that overwinter in freshwater]		15 – 25 %
Chum Salmon			
V_1	Maximum temperature during upstream migration	Spawning Adult	Between 8 degrees C and 12 degrees C
V_2	Minimum dissolved oxygen during upstream migration		> 6.5 mg/l
V_3	Extreme intragravel temperatures from spawning to fry emergence	Embryo, Fry	Maximum – 7.2 to 12.8 degrees C Minimum – 6 to 8 degrees C
V_4	Minimum dissolved oxygen concentration from spawning to fry emergence		6 mg/l
V_5	Substrate composition within riffle-run areas. A: percent gravel substrate 10-100mm diameter B: percent fines (< 6 mm)	Spawning Adult, Embryo, Fry	A: ≥ 60% B: <10% fines

Table B-1. Habitat Suitability Index Variables, Description, and Associated Life Stage for Coho, Chinook, Chum, and Pink Salmon[1]

Variable Number	Habitat Variable Description	Life Stage Affected	Maximum Suitability Index Description
V_6	Stream discharge pattern from egg deposition to downstream migration of fry	Embryo, Alevins	Best condition is stable streamflow, < 100-fold difference between extreme average daily stream discharges; stream channel stable, with little shifting.
V_7	Mean intragravel salinity for embryos and alevins	Embryo	< 4 ppt
V_8	Temperature extremes during rearing and downstream migration of fry. A: maximum B: minimum	Smolts	A: 12 degrees C B: 7 degrees C
V_9	Minimum dissolved oxygen during rearing and downstream migration of fry	Fry	8 mg/l
Pink Salmon			
V_1	Annual maximal or minimal pH (summer to fall period)	Adult, Juvenile	6.5 to 8.0
V_2	Maximal or minimal water temperature during the adult upstream migration and spawning period	Spawning Adult	8 degrees C to 13 degrees C
V_3	Average size range of substrate particle used for spawning	Spawning Adult, Embryo, Fry	1 to 5 cm
V_4	Percent fines (<0.3 cm) for survival of embryos and emergent fry	Embryo, Fry	6%
V_5	Average water velocity for spawning and embryo incubation	Spawning Adult, Embryo	40 cm/s
V_6	Minimal dissolved oxygen during egg incubation and pre-emergent yolk sac fry period	Embryo	8 mg/l
V_7	Maximal or minimal water temperature during early embryo development period	Embryo, Fry	7.5 degrees C to 12.5 degrees C
V_8	Maximal salinity during embryo development	Embryo	30 ppt
V_9	Average base flow during embryo incubation period (as percentage of average daily flow during spawning)		50%

Table B-1. Habitat Suitability Index Variables, Description, and Associated Life Stage for Coho, Chinook, Chum, and Pink Salmon[1]

Variable Number	Habitat Variable Description	Life Stage Affected	Maximum Suitability Index Description
V_{10}	Peak flow during incubation period (as multiple of average base flow)	Embryo	2 to 5
V_{11}	Maximum temperature during the period of seawater migration	Fry	2.5 degrees C to 17 degrees C

1 Habitat Variables from USFWS Habitat Suitability Index Models: Coho – McMahon, 1983; Chinook – Raleigh, Miller and Nelson, 1986; Chum – Hale, McMahon and Nelson, 1985; Pink – Raleigh and Nelson, 1985.

APPENDIX C

Alaska's Impaired Waters – 2008

Table C-1
ALASKA's IMPAIRED WATERS – 2008

<u>Impaired Water body Categories:</u>

Category 4a – Impaired water with a final/approved TMDL
Category 5 – Impaired water, Section 303(d) list, require TMDL

Within the tables waters are listed by region - -Interior, Southcentral, Southeast – and alphabetically.

Region	Category	Alaska ID #	Water body	Location	Area of Concern	Water Quality Standard	Pollutant Parameters	Pollutant Sources
Category 4a Waterbodies – Impaired but not needing a TMDL, TMDL has been completed								
IN	Category 4a	40402-001	Birch Creek Drainage:- Upper Birch Creek; Eagle Creek; Golddust Creek	North of Fairbanks	N/A	Turbidity	Turbidity	Placer Mining
SE	Category 4a	10203-005	Granite Creek	Sitka	N/A	Turbidity Sediment	Turbidity, Sediment	Gravel Mining
SE	Category 4a	10301-001	Lemon Creek	Juneau	N/A	Turbidity Sediment	Turbidity, Sediment	Urban Runoff, Gravel Mining
Category 5 Section 303(d) Listed Waterbodies – Impaired by pollutant(s) for one or more designated uses and requiring a TMDL ;Clean Water Act Section 303(d) Listed Waters								
IN	Category 5 Section 303(d) listed	20502-101	Caribou Creek	Denali National Park	16.1 miles	Turbidity	Turbidity	Mining
IN	Category 5 Section 303(d) listed	40402-010	Crooked Creek Bonanza Crooked Deadwood Ketchem Mammoth Mastodon Porcupine	North of Fairbanks	77 miles	Turbidity	Turbidity	Placer Mining
IN	Category 5 Section 303(d) listed	40402-010	Crooked Creek Bonanza Crooked Deadwood Ketchem Mammoth Mastodon Porcupine	North of Fairbanks	77 miles	Turbidity	Turbidity	Placer Mining

Region	Category	Alaska ID #	Water body	Location	Area of Concern	Water Quality Standard	Pollutant Parameters	Pollutant Sources
IN	Category 5 Section 303(d) listed	40509-001	Goldstream Creek	Fairbanks	70 miles	Turbidity	Turbidity	Placer Mining
IN	Category 5 Section 303(d) listed	40510-101	Slate Creek	Denali National Park	2.5 miles	Turbidity	Turbidity	Mining
SE	Category 5 Section 303(d) listed	10203-002	Katlian River	N. of Sitka, Baranof Island	4.5 miles	Sediment, Turbidity	Sediment, Turbidity	Timber Harvest
SE	Category 5 Section 303(d) listed	10203-602	Klag Bay	West Chichagof Island	1.25 acres	Toxic & Other Deleterious Organic and Inorganic Substances	Metals	Mining
SE	Category 5 Section 303(d) listed	10203-001	Nakwasina River	Baranof Island, Sitka	8 miles	Sediment, Turbidity	Sediment, Turbidity	Timber Harvest
SE	Category 5 Section 303(d) listed	10303-004	Pullen Creek (Lower Mile)	Skagway	Lower mile of Pullen Creek	Toxic & Other Deleterious Organic and Inorganic Substances	Metals	Industrial
SE	Category 5 Section 303(d) listed	10303-601	Skagway Harbor	Skagway	1.0 acre	Toxic & Other Deleterious Organic and Inorganic Substances	Metals	Industrial

APPENDIX D

Factors Affecting Contaminant Transfer to Environmental Groundwater, Surface Water, and Soil

Table D-1. Factors Affecting Contaminant Transfer to Environmental Groundwater, Surface Water and Soil

Transport Mechanism	Factors Affecting Transport	
	Chemical-Specific Considerations	Site-Specific Considerations
Groundwater		
Movement within and across aquifers and to surface water	• Density (more or less dense than water) • Water solubility • K_{OC} (organic carbon partition coefficient)	• Site hydrogeology • Precipitation • Infiltration rate • Porosity • Hydraulic conductivity • Groundwater flow direction • Depth to aquifer • Groundwater/surface water recharge and discharge zones • Presence of other compounds • Soil type • Geochemistry of site soils and aquifers • Presence and condition of wells (well location, depth, and use; casing material and construction; pumping rate) • Conduits, sewers
Volatilization (to soil gas, ambient air, and indoor air)	• Water solubility • Vapor pressure • Henry's Law Constant • Diffusivity	• Depth to water table • Soil type and cover • Climatologic conditions • Contaminant concentrations • Properties of buildings • Porosity and permeability of soils and shallow geologic materials
Adsorption to soil or precipitation out of solution	• Water solubility • K_{OW} (octanol/water partition coefficient) • K_{OC}	• Presence of natural carbon compounds • Soil type, temperature, and chemistry • Presence of other compounds
Biologic uptake	• K_{OW}	• Groundwater use for irrigation and livestock watering
Soil (Surface and Subsurface) and Sediment		
Runoff (soil erosion)	• Water solubility • K_{OC}	• Presence of plants • Soil type and chemistry • Precipitation rate • Configuration of land and surface condition

Transport Mechanism	Factors Affecting Transport	
	Chemical-Specific Considerations	**Site-Specific Considerations**
Leaching	• Water solubility • K_{OC}	• Soil type • Soil porosity and permeability • Soil chemistry (especially acid/base) • Cation exchange capacity • Organic carbon content
Volatilization	• Vapor pressure • Henry's Law Constant	• Physical properties of soil • Chemical properties of soil • Climatologic conditions
Biologic uptake	• Bioconcentration factor • Bioavailability	• Soil properties • Contaminant concentration
Surface Water		
Overland flow (via natural drainage or man-made channels)	• Water solubility • K_{OC}	• Precipitation (amount, frequency, duration) • Infiltration rate • Topography (especially gradients and sink holes) • Vegetative cover and land use • Soil/sediment type and chemistry • Use as water supply intake areas • Location, width, and depth of channel; velocity; dilution factors; direction of flow • Floodplains • Point and nonpoint source discharge areas
Volatilization	• Water solubility • Vapor pressure • Henry's law constant	• Climatic conditions • Surface area • Contaminant concentration
Hydrologic connection between surface water and groundwater	• Density	• Groundwater/surface water recharge and discharge • Stream bed permeability • Soil type and chemistry • Geology (especially Karst conditions)
Adsorption to soil particles and sedimentation (of suspended and precipitated particles)	• Water solubility • K_{OW} • K_{OC} • Density	• Particle size and density • Geochemistry of soils/sediments • Organic carbon content of soils/sediment

Transport Mechanism	Factors Affecting Transport	
	Chemical-Specific Considerations	Site-Specific Considerations
Biota		
Biologic uptake	• K_{OW} • Bioconcentration factor	• Chemical concentration • Presence of fish, plants, and other animals
Bioaccumulation	• K_{OW} • Persistence/half-life	• Presence of plants and animals • Consumption rate
Migration	• NA	• Commercial activities (farming, aquaculture, livestock, dairies) • Sport activities (hunting, fishing) • Migratory species
Vapor sorption	• NA	• Soil type • Plant species
Root uptake	• NA	• Contaminant depth • Soil moisture • Plant species

APPENDIX E

Historic Information on World-Wide Dam Failures

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
May 14, 2009	Huayuan County, Xiangxi Autonomous Prefecture, Hunan Province, China	?	manganese	tailings dam failure (capacity: 50,000 cubic metres)	?	The landslide set off by the tailings dam failure destroyed a home, killing three and injuring four people.
Dec. 22, 2008	Kingston fossil plant, Harriman, Tennessee, USA	Tennessee Valley Authority	coal ash	retention wall failure	Release of 5.4 million cubic yards [4.1 million cubic metres] of ashy slurry	The ash slide covered 400 acres [1.6 square kilometres] as deep as 6 feet [1.83 metres]. The wave of ash and mud toppled power lines, covered Swan Pond Road and ruptured a gas line. It damaged 12 homes, and one person had to be rescued, though no one was seriously hurt.
Sep. 8, 2008	Taoshi, Linfen City, Xiangfen county, Shanxi province, China	Tashan mining company	iron	Collapse of a waste-product reservoir at an illegal mine during rainfall	?	A mudslide several metres high buried a market, several homes and a three-storey building. At least 254 people are dead and 35 injured.
Nov. 6, 2006	Nchanga, Chingola, Zambia	Konkola Copper Mines Plc (KCM) (51% Vedanta Resources plc)	copper	failure of tailings slurry pipeline from Nchanga tailings leaching plant to Muntimpa tailings dumps	?	Release of highly acidic tailings into Kafue river; high concentrations of copper, manganese, cobalt in river water; drinking water supply of downstream communities shut down

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
April 30, 2006	near Miliang, Zhen'an County, Shangluo, Shaanxi Province, China	Zhen'an County Gold Mining Co. Ltd.	gold	tailings dam failure during sixth upraising of dam	?	The landslide buried about 40 rooms of nine households, leaving 17 residents missing. Five injured people were taken to hospital. More than 130 local residents have been evacuated. Toxic potassium cyanide was released into the Huashui river, contaminating it approx. 5 km downstream.
April 14, 2005	Bangs Lake, Jackson County, Mississippi, USA	Mississippi Phosphates Corp.	phosphate	phosphogypsum stack failure, because the company was trying to increase the capacity of the pond at a faster rate than normal, according to Officials with the Mississippi Department of Environmental Quality (the company has blamed the spill on unusually heavy rainfall, though)	approx. 17 million gallons of acidic liquid (64,350 m3)	liquid poured into adjacent marsh lands, causing vegetation to die
2004, Nov. 30	Pinchi Lake, British Columbia, Canada	Teck Cominco Ltd.	mercury	tailings dam (100-metres long and 12-metres high) collapses during reclamation work	6,000 to 8,000 m3 of rock, dirt and waste water	tailings spilled into 5,500 ha Pinchi Lake

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
2004, Sep. 5	Riverview, Florida, USA	Cargill Crop Nutrition	phosphate	a dike at the top of a 100-foot-high gypsum stack holding 150-million gallons of polluted water broke after waves driven by Hurricane Frances bashed the dike's southwest corner	60 million gallons (227,000 m3) of acidic liquid	liquid spilled into Archie Creek that leads to Hillsborough Bay
2004, May 22	Partizansk, Primorski Krai, Russia	Dalenergo	coal ash	A ring dike, enclosing an area of roughly 1 km^2 and holding roughly 20 million cubic meters of coal ash, broke. The break left a hole roughly 50 meter wide in the dam.	approximately 160,000 cubic meters of ash	The ash flowed through a drainage canal into a tributary to the Partizanskaya River which empties in to Nahodka Bay in Primorski Krai (east of Vladivostok). For details download Sept. 2004 report (PDF) by Paul Robinson, SRIC
2004, March 20	Malvési, Aude, France	Comurhex (Cogéma/Areva)	decantation and evaporation pond of uranium conversion plant	dam failure after heavy rain in preceding year (view **details**)	30,000 cubic metres of liquid and slurries	release led to elevated nitrate concentrations of up to 170 mg/L in the canal of Tauran for several weeks
2003, Oct. 3	Cerro Negro, Petorca prov., Quinta region, Chile	Cia Minera Cerro Negro	copper	tailings dam failure	50,000 tonnes of tailings	tailings flowed 20 kilometers downstream the río La Ligua

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]						
	Location	Parent company	Ore type	Type of Incident	Release	Impacts
2002, Aug. 27 / Sep. 11	San Marcelino, Zambales, Philippines	Dizon Copper Silver Mines, Inc.		overflow and spillway failure of two abandoned tailings dams after heavy rain (**view details**)	?	Aug. 27: some tailings spilled into Mapanuepe Lake and eventually into the Sto. Tomas River Sep. 11: low lying villages flooded with mine waste; 250 families evacuated; nobody reported hurt so far
2001, Jun. 22	Sebastião das Águas Claras, Nova Lima district, Minas Gerais, Brazil	Mineração Rio Verde Ltda	iron	mine waste dam failure (**view details**)	?	tailings wave traveled at least 6 km, killing at least two mine workers, three more workers are missing
2000, Oct. 18	Nandan county, Guangxi province, China	?	?	tailings dam failure	?	at least 15 people killed, 100 missing; more than 100 houses destroyed
2000, Oct. 11	Inez, Martin County, Kentucky, USA	Martin County Coal Corporation (100% A.T. Massey Coal Company, Inc. , Richmond, VA (100% Fluor Corp.))	coal	tailings dam failure from collapse of an underground mine beneath the slurry impoundment (**view details**)	250 million gallons (950,000 m3) of coal waste slurry released into local streams	About 75 miles (120 km) of rivers and streams turned an iridescent black, causing a fish kill along the Tug Fork of the Big Sandy River and some of its tributaries. Towns along the Tug were forced to turn off their drinking water intakes.

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
2000, Sep. 8	Aitik mine, Gällivare, Sweden	Boliden Ltd.	copper	tailings dam failure from insufficient perviousness of filter drain (**view details**)	release of 2.5 million m3 of liquid into an adjacent settling pond, subsequent release of 1.5 million m3 of water (carrying some residual slurry) from the settling pond into the environment	
2000, Mar. 10	Borsa, Romania	Remin S.A.		tailings dam failure after heavy rain	22,000 t of heavy-metal contaminated tailings	contamination of the Vaser stream, tributary of the Tisza River. View Romanian Govt. report · UNEP report (527k PDF)
2000, Jan. 30	Baia Mare, Romania	Aurul S.A. (Esmeralda Exploration, Australia (50%), Remin S.A. (44.8%))	gold recovery from old tailings	tailings dam crest failure after overflow caused from heavy rain and melting snow (**view details**)	100,000 m^3 of cyanide-contaminated liquid	contamination of the Somes/Szamos stream, tributary of the Tisza River, killing tonnes of fish and poisoning the drinking water of more than 2 million people in Hungary
1999, Apr. 26	Placer, Surigao del Norte, Philippines	Manila Mining Corp. (MMC)	gold	tailings spill from damaged concrete pipe	700,000 tonnes of cyanide tailings	17 homes buried, 51 hectares of riceland swamped
1998, Dec. 31	Huelva, Spain	Fertiberia, Foret	phosphate	dam failure during storm (view details)	50,000 m3 of acidic and toxic water	

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]						
	Location	**Parent company**	**Ore type**	**Type of Incident**	**Release**	**Impacts**
1998, Apr. 25	Los Frailes, Aznalcóllar, Spain	Boliden Ltd., Canada	zinc, lead, copper, silver	dam failure from foundation failure (**view details**)	4-5 million m3 of toxic water and slurry	thousands of hectares of farmland covered with slurry
1997, Dec. 7	Mulberry Phosphate, Polk County, Florida, USA	Mulberry Phosphates, Inc.	phosphate	phosphogypsum stack failure	200,000 m3 of phosphogypsum process water	biota in the Alafia River eliminated
1997, Oct. 22	Pinto Valley, Arizona, USA	BHP Copper	copper	tailings dam slope failure	230,000 m3 of tailings and mine rock	tailings flow covers 16 hectares
1996, Nov. 12	Amatista, Nazca, Peru	?	?	liquefaction failure of upstream-type tailings dam during earthquake	more than 300,000 m3 of tailings	flow runout of about 600 meters, spill into river, croplands contaminated
1996, Aug. 29	El Porco, Bolivia	Comsur (62%), Rio Tinto (33%)	zinc, lead, silver	dam failure	400,000 tonnes	300 km of Pilcomayo river contaminated
1996, Mar. 24	Marcopper, Marinduque Island, Philippines	Placer Dome Inc., Canada (40%)	copper	Loss of tailings from storage pit through old drainage tunnel	1.6 million m3	Evacuation of 1200 residents, 18 km of river channel filled with tailings, US$ 80 million damage
1995, Dec.	Golden Cross, New Zealand	Coeur d'Alène, Idaho, USA	gold	Dam movement of dam containing 3 million tonnes of tailings (continuing) (view details)	Nil (so far)	Nil (so far)

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1995, Sep. 2	Placer, Surigao del Norte, Philippines	Manila Mining Corp.	gold	Dam foundation failure	50,000 m3	12 people killed, coastal pollution
1995, Aug. 19	Omai, Guyana	Cambior Inc., Canada (65%), Golden Star Resources Inc., Colorado, USA (30%)	gold	tailings dam failure from internal dam erosion (preliminary report on technical causation)	4.2 million m3 of cyanide slurry	80 km of Essequibo River declared environmental disaster zone (view details)
1994, Nov. 19	Hopewell Mine, Hillsborough County, Florida, USA	IMC-Agrico	phosphate	dam failure	Nearly 1.9 million m3 of water from a clay settling pond	spill into nearby wetlands and the Alafia River, Keysville flooded
1994, Oct. 2	Payne Creek Mine, Polk County, Florida, USA	IMC-Agrico	phosphate	dam failure	6.8 million m3 of water from a clay settling pond	majority of spill contained on adjacent mining area; 500,000 m3 released into Hickey Branch, a tributary of Payne Creek
1994, Oct.	Fort Meade, Florida, USA	Cargill	phosphate	?	76,000 m3 of water	spill into Peace River near Fort Meade
1994, June	IMC-Agrico, Florida, USA	IMC-Agrico	phosphate	Sinkhole opens in phosphogypsum stake	?	Release of gympsum and water into groundwater
1994, Feb. 22	Harmony, Merriespruit, South Africa	Harmony Gold Mines	gold	Dam wall breach following heavy rain	600,000 m3	tailings traveled 4 km downstream, 17 people killed, extensive damage to residential township

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1994, Feb. 14	Olympic Dam, Roxby Downs, South Australia	WMC Ltd.	copper, uranium	leakage of tailings dam during 2 years or more	release of up to 5 million m3 of contaminated water into subsoil	?
1993, Oct.	Gibsonton, Florida, USA	Cargill	phosphate	?	?	Fish killed when acidic water spilled into Archie Creek
1993	Marsa, Peru	Marsa Mining Corp.	gold	dam failure from overtopping	?	6 people killed
1992, Mar. 1	Maritsa Istok 1, near Stara Zagora, Bulgaria	?	ash/cinder	dam failure from inundation of the beach	500,000 m3	?
1992, Jan.	No.2 tailings pond, Padcal, Luzon, Philippines	Philex Mining Corp.	copper	Collapse of dam wall (foundation failure)	80 million tonnes	?
1991, Aug. 23	Sullivan mine, Kimberley, British Columbia, Canada	Cominco Ltd	lead/zinc	dam failure (liquefaction in old tailings foundation during construction of incremental raise)	75,000 m3	the slided material was contained in an adjacent pond
1989, Aug. 25	Stancil, Perryville, Maryland, USA	?	sand and gravel	dam failure during capping of the tailings after heavy rain	38,000 m3	tailings flowside covered 5000 m2
1988, Apr. 30	Jinduicheng, Shaanxi province, China	?	molybdenum	breach of dam wall (spillway blockage caused pond level to rise too high)	700,000 m3	approx. 20 people killed

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1988, Jan. 19	Tennessee Consolidated No.1, Grays Creek, TN, USA	Tennessee Consolidated Coal Co.	coal	dam wall failure from internal erosion, caused from failure of an abandoned outlet pipe	250,000 m3	?
1988	Riverview, Florida, USA	Gardinier (now Cargill)	phosphate	?	acidic spill	Thousands of fish killed at mouth of Alafia River
1987, April 8	Montcoal No.7, Raleigh County, West Virginia, USA	Peabody Coal Co. (now Peabody Energy)	coal	dam failure after spillway pipe breach	87,000 cubic meters of water and slurry	tailings flow 80 km downstream
1986, May	Itabirito, Minas Gerais, Brazil	Itaminos Comercio de Minerios	?	dam wall burst	100,000 tonnes	tailings flow 12 km downstream
1986	Huangmeishan, China	?	iron	dam failure from seepage/slope instability	?	19 people killed
1985, July 19	Stava, Trento, Italy	Prealpi Mineraia	fluorite	dam failure, caused from insufficient safety margins and inadequate decant pipe construction (**view details**)	200,000 m3	tailings flow 4.2 km downstream at 90 km/h; 268 people killed, 62 buildings destroyed (**view details**)
1985, Mar. 3	Veta de Agua No.1, Chile	?	copper	dam wall failure, due to liquefaction during earthquake	280,000 m3	tailings flow 5 km downstream
1985, Mar. 3	Cerro Negro No.4, Chile	Cia Minera Cerro Negro	copper	dam wall failure, due to liquefaction during earthquake	500,000 m3	tailings flow 8 km downstream

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1985	Olinghouse, Wadsworth, Nevada, USA	Olinghouse Mining Co.	gold	embankment collapse from saturation	25,000 m3	tailings flow 1.5 km downstream
1982, Nov. 8	Sipalay, Negros Occidental, Philippines	Marinduque Mining and Industrial Corp.	copper	dam failure, due to slippage of foundations on clayey soils	28 million tonnes	Widespread inundation of agricultural land up to 1.5 m high
1981, Dec. 18	Ages, Harlan County, Kentucky, USA	Eastover Mining Co.	coal	dam failure after heavy rain	96,000 m3 coal refuse slurry	the slurry wave traveled the Left Fork of Ages Creek 1.3 km downstream, 1 person was killed, 3 homes destroyed, 30 homes damaged, fish kill in Clover Fork of the Cumberland River
1981, Jan. 20	Balka Chuficheva, Lebedinsky, Russia	?	iron	dam failure	3.5 million m3	tailings travel distance 1.3 km
1980, Oct. 13	Tyrone, New Mexico, USA	Phelps Dodge	copper	dam wall breach, due to rapid increase in dam wall height, causing high internal pore pressure	2 million m3	tailings flow 8 km downstream and inundate farmland
1979, July 16	Church Rock, New Mexico, USA	United Nuclear	uranium	dam wall breach, due to differential foundation settlement	370,000 m3 of radioactive water, 1,000 tonnes of contaminated sediment	Contamination of Rio Puerco sediments up to 110 km downstream

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1979 or earlier	(unidentified), British Columbia, Canada	?	?	piping in the sand beach of the tailings dam	40,000 m3 of ponded water	considerable property damage
1978, Jan. 31	Arcturus, Zimbabwe	Corsyn Consolidated Mines	gold	slurry overflow after continuous rain over several days	30,000 tonnes	1 person killed, extensive siltation to waterway and adjoining rough pasture
1978, Jan. 14	Mochikoshi No.1, Japan	?	gold	dam failure, due to liquefaction during earthquake	80,000 m3	1 person killed, tailings flow 7-8 km downstream
1977, Feb. 1	Homestake, Milan, New Mexico, USA	Homestake Mining Company	uranium	dam failure, due to rupture of plugged slurry pipeline	30,000 m3	no impacts outside the mine site
1976, Mar. 1	Zlevoto, Yugoslavia	?	lead, zinc	dam failure, due to high phreatic surface and seepage breakout on the embankment face	300,000 m3	tailings flow reached and polluted nearby river
1975, June	Silverton, Colorado, USA	?	(metal)	dam failure	116,000 tonnes	tailings flow slide polluted nearly 100 miles (160 km) of the Animas river and its tributaries; severe property damage; no injuries
1975, Apr.	Madjarevo, Bulgaria	?	lead, zinc, gold	rising of tailings above design level caused overloading of the decant tower and collectors	250,000 m3	?

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1975	Mike Horse, Montana, USA	?	lead, zinc	dam failure after heavy rain	150,000 m3	?
1974, Nov. 11	Bafokeng, South Africa	?	platinum	embankment failure by concentrated seepage and piping through cracks	3 million m3	12 people killed in a mine shaft inundated by the tailings; tailings flow 45 km downstream
1974, Jun. 1	Deneen Mica, North Carolina, USA	?	mica	dam failure after heavy rain	38,000 m3	tailings released to an adjacent river
1973	(unidentified), Southwestern USA	?	copper	dam failure from increased pore pressure during construction of incremental raise	170,000 m3	tailings traveled 25 km downstream
1972, Feb. 26	Buffalo Creek, West Virginia, USA	Pittston Coal	coal	collapse of tailings dam after heavy rain (view Citizens' Commission report)	500,000 m3	the tailings traveled 27 km downstream, 125 people lost their lives, 500 homes were destroyed. Property and highway damage exceeded $65 million. (see details)
1971, Dec. 3	Fort Meade, Florida, USA	Cities Service Co.	phosphate	Clay pond dam failure, cause unknown	9 million m3 of clay water	tailings traveled 120 km downstream with Peace River, large fish kill
1970	Mufulira, Zambia	?	copper	liquefaction of tailings, flowing into underground workings	some 1 million tons	89 miners killed

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1970	Maggie Pie, United Kingdom	?	china clay	dam failure after raising the embankment and after heavy rain	15,000 m3	tailings spilled 35 meters downstream
1969 or earlier	Bilbao, Spain	?	?	dam failure (liquefaction) after heavy rain	115,000 m3	major downstream damage and loss of life
1968	Hokkaido, Japan	?	?	dam failure (liquefaction) during earthquake	90,000 m3	tailings traveled 150 meters downstream
1967, Mar.	Fort Meade, Florida, USA	Mobil Chemical	phosphate	dam failure, no details available	250,000 m3 of phosphatic clay slimes, 1.8 million m3 of water	spill reaches Peace River, fish kill reported
1967	(unidentified), United Kingdom	?	coal	dam failure during regrading operations	?	tailings flow covered an area of 4 hectares
1966	(unidentified), East Texas, USA	?	gypsum	dam failure	76,000 - 130,000 m3 of gypsum	flow slide traveled 300 meters; no fatalities
1966	Derbyshire, United Kingdom	?	coal	dam failure from foundation failure	30,000 m3	tailings traveled 100 meters downstream
1966, Oct. 21	Aberfan, Wales, United Kingdom	Merthyr Vale Colliery	coal	dam failure (liquefaction) from heavy rain	162,000 m3	the tailings traveled 600 meters, 144 people were killed (**view details**, watch video)

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]

	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1966, May 1	Mir mine, Sgorigrad, Bulgaria	?	lead, zinc, copper, silver, (uranium?)	dam failure from rising pond level after heavy rains and/or failure of diversion channel	450,000 m3	the tailings wave traveled 8 km to the city of Vratza and destroyed half of Sgorigrad village 1 km downstream, killing 488 people. (View **details** · historic photographs)
1965, Mar. 28	Bellavista, Chile	?	copper	dam failure during earthquake	70,000 m3	tailings traveled 800 meters downstream
1965, Mar. 28	Cerro Negro No.3, Chile	?	copper	dam failure during earthquake	85,000 m3	tailings traveled 5 km downstream
1965, Mar. 28	El Cobre New Dam, Chile	?	copper	dam failure (liquefaction) during earthquake	350,000 m3	tailings traveled 12 km downstream, destroyed the town of El Cobre and killed more than 200 people
1965, Mar. 28	El Cobre Old Dam, Chile	?	copper	dam failure (liquefaction) during earthquake	1.9 million m3	
1965, Mar. 28	La Patagua New Dam, Chile	?	copper	dam failure (liquefaction) during earthquake	35,000 m3	tailings traveled 5 km downstream
1965, Mar. 28	Los Maquis, Chile	?	copper	dam failure (liquefaction) during earthquake	21,000 m3	tailings traveled 5 km downstream

Table E-1. Historic Information on World-Wide Dam Failures [Hard rock mines highlighted]						
	Location	Parent company	Ore type	Type of Incident	Release	Impacts
1965	Tymawr, United Kingdom	?	coal	dam failure from overtopping	?	tailings traveled 700 meters downstream, causing considerable damage
1962	(unidentified), Peru	?	?	dam failure (liquefaction) during earthquake and after heavy rainfall	?	?
1961	Tymawr, United Kingdom	?	coal	dam failure, no details available	?	tailings traveled 800 meters downstream

tonnes = metric tonnes

Sources:

- **Tailings Dam Incidents**, U.S. Committee on Large Dams - USCOLD, Denver, Colorado, ISBN 1-884575-03-X, 1994, 82 pages *[compilation and analysis of 185 tailings dam incidents]*
- **Environmental and Safety Incidents concerning Tailings Dams at Mines**: Results of a Survey for the years 1980-1996 by Mining Journal Research Services; a report prepared for United Nations Environment Programme, Industry and Environment. Paris, 1996, 129 pages *[compilation of 37 tailings dam incidents]*
- **Tailings Dams - Risk of Dangerous Occurrences, Lessons learnt from practical experiences**, Bulletin 121, Published by United Nations Environmental Programme (UNEP) Division of Technology, Industry and Economics (DTIE) and International Commission on Large Dams (ICOLD), Paris 2001, 144 p. *[compilation of 221 tailings dam incidents mainly from the above two publications, and examples of effective remedial measures]*

APPENDIX F

Tailings Dam Failure Runout and Volume Estimates

From: Rico, Benito and Diez-Herrero: Floods from tailings dam failures. J.Hazard.Mat 154(2008).

Potential Tailings Outflow Volume = V_f

V_T = Total Volume of the Tailings

Eq. 7: $V_f = 0.354 \times V_T^{1.01}$

V_T in millions (10^6) in cubic meters = 2,064 ← Enter number Here

V_f = Potential Tailings Outflow Volume

V_f = **788.6** V_f in millions (10^6) cubic meters

The above equation shows, that in average, one-third of the tailings and water at the decant pond is released during dam failures. The envelope curve (not included here) represents the maximum tailings volume that can be released in the most extreme situation in which pond volume was emptied following the dam break, as is the case of water-storage dam accidents or those of industrial (diluted) waste ponds.

D_{max} = Outflow Runout Distance

Eq. 5: $Dmax = 1.61 \times (HV_F)^{0.66}$

Dam Height in meters (H) = 225 ← Enter number Here

D_{max}	=	**4,690**	Outflow Distance in Kilometers
	=	**2,914**	Outflow Distance in Miles

conversion factors:

1 kilometer	=	0.621371	statute miles
1 foot	=	0.3048	meters
1 cubic yard	=	0.764555	cubic meters

Proposed Mine Tailings Dam Information

From: 2006 Water Rights Applications

TSF A

(Pebble Project Tailings Impoundment A, Initial Application Report (Ref. No. VA101-176/16-13), Knight Piesold Ltd, September 5, 2006)

Dam Height

"On-going staged expansion of the north embankment will result in a final height of 700 feet. The southeast and southwest embankments will be developed to heights of 710 feet and 740 feet, respectively." (Knight Piesold, p. 14 of 24)

Southwest Embankment (South Fork Koktuli) = 740 feet = 225 meters
Southeast Embankment (South Fork Koktuli) = 710 feet = 216 meters
North Embankment (South Fork Koktuli) = 700 feet = 213 meters

Waste Volume

"The design basis for the TSF at Site A will allow for secure storage of approximately 2 billion tons of tailings solids…" (Knight Piesold, p. 1 of 24)

TSF Total Storage Volume (tailings & waste rock) = 2.7 billion cubic yards (Knight Piesold, Figure 5.3)

Volume (yd^3)	Volume (m^3)
2.70E+09	2.06E+09

TSF G

(Pebble Project Tailings Impoundment G, Initial Application Report (Ref. No. VA101-176/16-13), Knight Piesold Ltd, September 5, 2006)

Dam Height

"On-going staged expansion of the north embankment will result in a final height of 700 feet. The southeast and southwest embankments will be developed to heights of 710 feet and 740 feet, respectively." (Knight Piesold, p. 14 of 24)

Main Embankment (Unnamed Tributary NK1.190 to the North Fork Koktuli River) = 450 feet = 137 meters

Saddle Dam (Unnamed Tributary NK1.190 to the North Fork Koktuli River) = 175 feet = 53 meters

Waste Volume

"The design basis for the TSF at Site G will allow for secure storage of approximately 500 million tons of tailings solids discharged into an engineered containment impoundment." (Knight Piesold, p. 1 of 24)

TSF Total Storage Volume (tailings & waste rock) = 580 million cubic yards
(Knight Piesold, Figure 5.3)

Volume (yd^3)	Volume (m^3)
5.80E+08	4.43E+08

Ultimate Mine Buildout

10.78 billion tonnes	=	13.5 bcy for the total waste storage requirement, based on the average waste density implied by TSF A and TSF G of 59 lbs per cubic foot. This "average density" is derived from comparing the ratio of the amount of tailings in TSF A & G (Knight Pieslod, 2006) to the waste volume (Knight Pielsod, 2006, Appendix A). We had to use this approach since Knight Piesold did not disclose the waste rock volume weight or volume to be added to the impoundments in the applications, but the volume figures taken from the TSF A & G Appendices A did include both tailings and waste rock.
	=	10.32 billion cubic meters
10.78 billion tons	=	8.78 bcy at an average density of 100 pounds per cubic foot (this does not consider waste rock)
	=	6.72 billion cubic meters