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An Assessment of Potential Mining Impacts on Salmon Ecosystems of Bristol Bay, Alaska

Volume 1 – Main Report

U.S. Environmental Protection Agency, Seattle, WA
www.epa.gov/bristolbay

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NOTICE

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U.S. Environmental Protection Agency

Seattle, WA

DISCLAIMER

ABSTRACT

This report evaluates the potential impacts of large-scale mining development on salmon and other fish populations, wildlife, and Alaska Native cultures in the Nushagak River and Kvichak River watersheds of Bristol Bay, Alaska. It is not an assessment of a specific mine proposal for development, nor does it outline decisions made or to be made by the U.S. Environmental Protection Agency (USEPA). The assessment was conducted as an ecological risk assessment and starts with a review and characterization of the fisheries, wildlife, and Alaska Native cultures of the Bristol Bay watershed and specifically the Nushagak River and Kvichak River watersheds. We developed a hypothetical but realistic mine scenario that includes an open pit mine producing between 2 and 6.5 billion metric tons of ore and a 139-km (86-mile) transportation corridor. Based on this mine scenario, we conclude that, at a minimum, mining at this scale would cause the loss of spawning and rearing habitat for multiple species of anadromous and resident fish. A mine footprint of this scale would likely result in the direct loss of 87.5 to 141.4 km of streams and 10.2 to 17.3 km^2 of wetlands. Additionally, water withdrawals for mine operations would significantly diminish habitat quality in an additional 2 to 10 km of streams. Assuming no significant accidents or failures, the development and routine operation of one large-scale mine would result in significant impacts on fish populations in streams surrounding the mine site. Accidents, process failures, and infrastructure failures could increase the spatial scale and severity of mining impacts on fish populations. Potential accidents include (1) the release of acid, metal, and other contaminants from the mine site, waste rock piles, and tailings storage facilities (TSFs); (2) the failure of roads, culverts, and pipelines in the transportation corridor, including spills of copper concentrate; and (3) the catastrophic failure of a tailings dam. Although precise estimates of the probabilities of failure occurrence cannot be made, evidence from the long-term operation of similar large mines suggests that, over the life span of a large mine, at least one or more accidents or failures could occur, potentially resulting in immediate, severe impacts on salmon and detrimental, long-term impacts on salmon habitat and production. The Nushagak River and Kvichak River watersheds contain multiple sites under consideration for large-scale mining. Potential risks of mining development on salmon and other fish populations are likely to increase as a result of the cumulative impacts of multiple mines.

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Acronyms and Abbreviations

AAC	Alaska Administrative Code
ADEC	Alaska Department of Environmental Conservation
ADFG	Alaska Department of Fish and Game
ADNR	Alaska Departments of Natural Resources
AEIC	Alaska Earthquake Information Center
AFFI	Alaska Freshwater Fish Inventory
AP	acid-generation potential
AVS	acid volatile sulfides
AWC	Anadromous Waters Catalog
BLM	biotic ligand model
CCC	criteria continuous concentration
CMC	criterion maximum concentration
EBD	Environmental Baseline Document
EPA	U.S. Environmental Protection Agency
E-R	Exposure-Response relationship
FERC	Federal Energy Regulatory Commission
GIS	geographic information system
GMU	Game Management Unit
HEC-HMS	Hydrologic Engineering Center's Hydrologic Modeling System
HEC-RAS	Hydrologic Engineering Center's River Analysis System
HUC	hydrologic unit code
IGTT	Intergovernmental Technical Team
ISO	International Organization for Standardization
kg CaCO3/metric ton	kilograms of calcium carbonate per metric ton of waste material
MCE	maximum credible earthquake
MDE	maximum design earthquake
MDN	marine-derived nutrients
NAG	no acid generation potential
NDM	Northern Dynasty Minerals
NHD	National Hydrography Dataset
NNP	net neutralization potential
NP	neutralization potential
NPR	neutralizing potential ratio
NWI	National Wetlands Inventory
OBE	operating basis earthquake
PAG	potentially acid generating
PEC	probable effect concentration
PEL	probable effect level
PLP	Pebble Limited Partnership
PMF	probable maximum flood
PMP	probable maximum precipitation
Reclamation	Bureau of Reclamation
SEM	simultaneously extracted metals
TEC	threshold effect concentration
TEL	threshold effect level
TSF	Tailings Storage Facility
USACE	U.S. Army Corps of Engineers
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey

Units of Measure

μg	microgram
^{0}C	degrees Celsius
cm	centimeter
g	gram
ha	hectare
kg	kilogram
km	kilometer
km^2	square kilometers
L	liter
m	meter
m^2	square meter
m^3	cubic meter
Mcf	million cubic feet
mg	milligram
mm	millimeter
Mt	metric ton
s	second

Unit of Measure Conversion Chart

Metric	**Standard**
1 μg (microgram)	3.527396e-08 ounces
1 mg (milligram)	3.527396e-05 ounces
1 g (gram)	0.035 ounces
1 kg (kilogram)	2.202 pounds
1 Mt (metric ton)	1.103 short tons
1 mm (millimeter)	0.039 inches
1 cm (centimeter)	0.39 inch
1 m (meter)	3.28 feet
1 m^2 (square meter)	10.764 square feet
1 m^3 (cubic meter)	35.314 cubic feet
1 km (kilometer)	0.621 miles
1 km^2 (square kilometer) or 100 ha (hectares)	0.286 square miles
1 ha (hectare)	2.47 acres
1 L (liter)	0.264 gallons
1 ^{0}C (degrees Celsius)	1.8C + 32 Fahrenheit

PREFACE

This assessment was produced by the U.S. Environmental Protection Agency (USEPA) to assess the potential impacts of large-scale mining on salmon and other fish populations, wildlife, and Alaska Native cultures in the Bristol Bay watershed, Alaska. Clean Water Act Sections 104(a) and (b) provide the Agency with the authority to study the resources of the Bristol Bay watershed, evaluate the potential effects of pollution from large scale mining development on those resources, and make such an assessment available to the public. This assessment focuses on potential environmental impacts resulting from large-scale mining in the watershed. It does not address impacts associated with other development activities (e.g., airfield construction), and use of its findings to support or oppose types of development other than large-scale mining would not be appropriate.

We recognize that mining development is a controversial subject. Our goals in conducting this assessment are to complete an objective assessment of the potential impacts of large-scale mining on aquatic resources in the Bristol Bay watershed, and to identify uncertainties. To that end, we have sought input from federal, state, and Tribal representatives, and have used established procedures for evaluating data and information. With the distribution of this report, we look forward to receiving public comments on all aspects of this assessment including additional information on mitigation practices that may lessen the risks outlined in this assessment.

The USEPA convened an Intergovernmental Technical Team (IGTT), which included federal and state agency personnel and Tribal representatives, to provide us with background information for the assessment. We specifically asked this group for input on our assessment approach and the conceptual model diagrams we used to frame the assessment. We realize that some members of this group have specific positions on mining development, and have relationships (including financial ties) with mining companies and environmental groups. To ensure that this process was transparent and objective, and that the USEPA could understand and address any issues that could potentially harm the integrity of the assessment process, we developed IGTT Guidelines (available on the USEPA Bristol Bay Watershed Assessment Website, www.epa.gov/bristolbay) that included expectations for IGTT members and requested that all members identify any affiliations with non-government entities having a stake in the assessment outcome.

The USEPA has reviewed and considered information and data from a variety of sources, including environmental groups that oppose mining development and mining companies that are mining proponents. Where possible, we have relied on peer-reviewed, published data and information. However, much of the information on Bristol Bay has not been published in the peer-reviewed literature. We have used established guidelines for the use of those data, which include evaluating collection and analytical methods and identifying data limitations. All sources of information used in the assessment are identified in this report.

The assessment was developed by USEPA staff in the Office of Research and Development, the Office of Water, and Region 10, with additional support provided by contractors. All contractors who contributed to this assessment are identified as either authors or contributors, with additional information provided as part of our acknowledgements. Contractors contributing to this report were required to certify that they had no organizational conflicts of interest. As defined by Federal Acquisition Regulations subpart 2.101, an organizational conflicts of interest may exist when, "because of other activities or relationships with other persons, a person is unable or potentially unable to render impartial assistance or advice to the Government, or the person's objectivity in performing the contract work is or might otherwise be impaired or a person has an unfair competitive advantage."

The USEPA contracted with NatureServe to provide background information for the assessment and these background characterization reports are included as appendices to this assessment. NatureServe subcontracted with several experts in the Bristol Bay watershed. Concerns have been raised about whether several of these experts are able to be impartial, based on their expressed personal opinions or affiliations with non-government organizations that may oppose mining development. The USEPA used a screening process to ensure that these subcontractors have significant professional accomplishments and are highly qualified to perform the tasks they were assigned. The assessment process also includes several measures that we feel minimizes the impact of any potential bias in the reports prepared by these subcontractors. These measures include multiple reviews of contractor work products by USEPA personnel, and insistence that all information and conclusions are well documented and supported. In addition, these background characterization reports along with the main assessment report, will undergo scientific peer review by an independent panel of experts. The public, including industry and environmental groups, have the opportunity to comment on this assessment and identify any potential concerns regarding bias or other issues.

This draft report is being released for public comment and peer review by an external (i.e., outside the USEPA) panel of experts. It has been through an internal review process; the reviewers who participated in this internal review are listed in the pages that follow. Following the public comment period, a summary of public comments will be made and provided to the peer review panel. The review panel will meet over three days to discuss the report. The USEPA will evaluate comments received from the public and the peer review panel before developing the final assessment report.

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EXECUTIVE SUMMARY

The Bristol Bay watershed in southwestern Alaska supports the largest sockeye salmon fishery in the world, is home to 25 Federally Recognized Tribal Governments, and contains large mineral resources. The potential for large-scale mining activities in the watershed has raised concerns about the impact of mining on the sustainability of Bristol Bay's world-class fisheries, and the future of Alaska Native tribes in the watershed who have maintained a salmon-based culture and subsistence-based lifestyle for at least 4,000 years. The U.S. Environmental Protection Agency (USEPA) launched this assessment to determine the significance of Bristol Bay's ecological resources and evaluate the potential impacts of large-scale mining on these resources. The USEPA will use the results of this assessment to inform the consideration of options consistent with its role under the Clean Water Act. The assessment is intended to provide a scientific and technical foundation for future decision making; the USEPA will not address use of its regulatory authority until the assessment becomes final and has made no judgment about whether to use that authority at this time.

In addition to informing future USEPA actions, this report is of potential use to other federal and state government entities with an interest in mining in the Bristol Bay region. It is also of interest to both proponents and opponents of mining. By providing an unbiased assessment of potential risks, this assessment informs an active debate concerning the risks of mining development to the sustainability of the Bristol Bay salmon fishery.

Scope of the Assessment

This assessment reviews, analyzes, and synthesizes available information on the potential impacts of large-scale mining development on Bristol Bay fisheries and subsequent effects on the wildlife and Alaska Native cultures of the region. The primary focus of the assessment is the quality, quantity, and genetic diversity of salmonid fish. Because wildlife and Alaska Native cultures in Bristol Bay are intimately connected and dependent upon fish, the quantity and diversity of wildlife and the culture and human welfare of indigenous peoples, as affected by changes in the fisheries are additional endpoints of the assessment.

The geographic scope of the assessment is the Nushagak River and Kvichak River watersheds (Figure ES-1). These are the largest of the Bristol Bay watershed's six major river basins and compose about 50% of the total watershed area. These two watersheds are also identified as mineral development areas by the State of Alaska. The Pebble deposit, the most likely site for near-term large-scale mining development in the region, is located at the intersection of the Nushagak River and Kvichak River watersheds. The headwaters of three biologically productive tributaries originate in this region: the North Fork Koktuli River, located to the northwest of the Pebble deposit, which flows into the Nushagak River via the Mulchatna River; the South Fork Koktuli River, which drains the Pebble deposit area and converges with the North Fork west of the Pebble deposit; and Upper Talarik Creek, which drains the eastern portion of the Pebble deposit and flows into the Kvichak River via Iliamna Lake, the largest undeveloped lake in the United States (Figure ES-2).

The assessment addresses two general time periods for mine activities. The first is the development and operation phase, during which mine infrastructure is built and the mine is operated. This phase may last from 25 to 100 years or more. The second is the post-mining, or post-closure, phase, during which the site would be monitored and, as necessary, water treatment and other waste management activities continued and failures remediated. Because mining wastes would be altered by geologic processes but would not degrade, this period would continue for centuries and potentially "in perpetuity."

The assessment was conducted as an ecological risk assessment. We started with a thorough review of what is known about the Bristol Bay watershed fishery and wildlife and the Alaska Native cultures. We also reviewed information about copper mining and available information outlining proposed mining operations for the Pebble deposit that has been the focus of much exploratory study and has received much attention from various groups in and outside of Alaska. Using that information, we developed a set of conceptual models to show potential associations between the endpoints of interest—the salmon fishery and salmon populations—and the various types of environmental stressors that might reasonably be expected as a result of large-scale mining. Those conceptual models were refined through interactions with regional stakeholders. The assessment was then developed based upon the background characterization studies and the conceptual models.

Figure ES-1. The Nushagak River and Kvichak River Watersheds of Bristol Bay

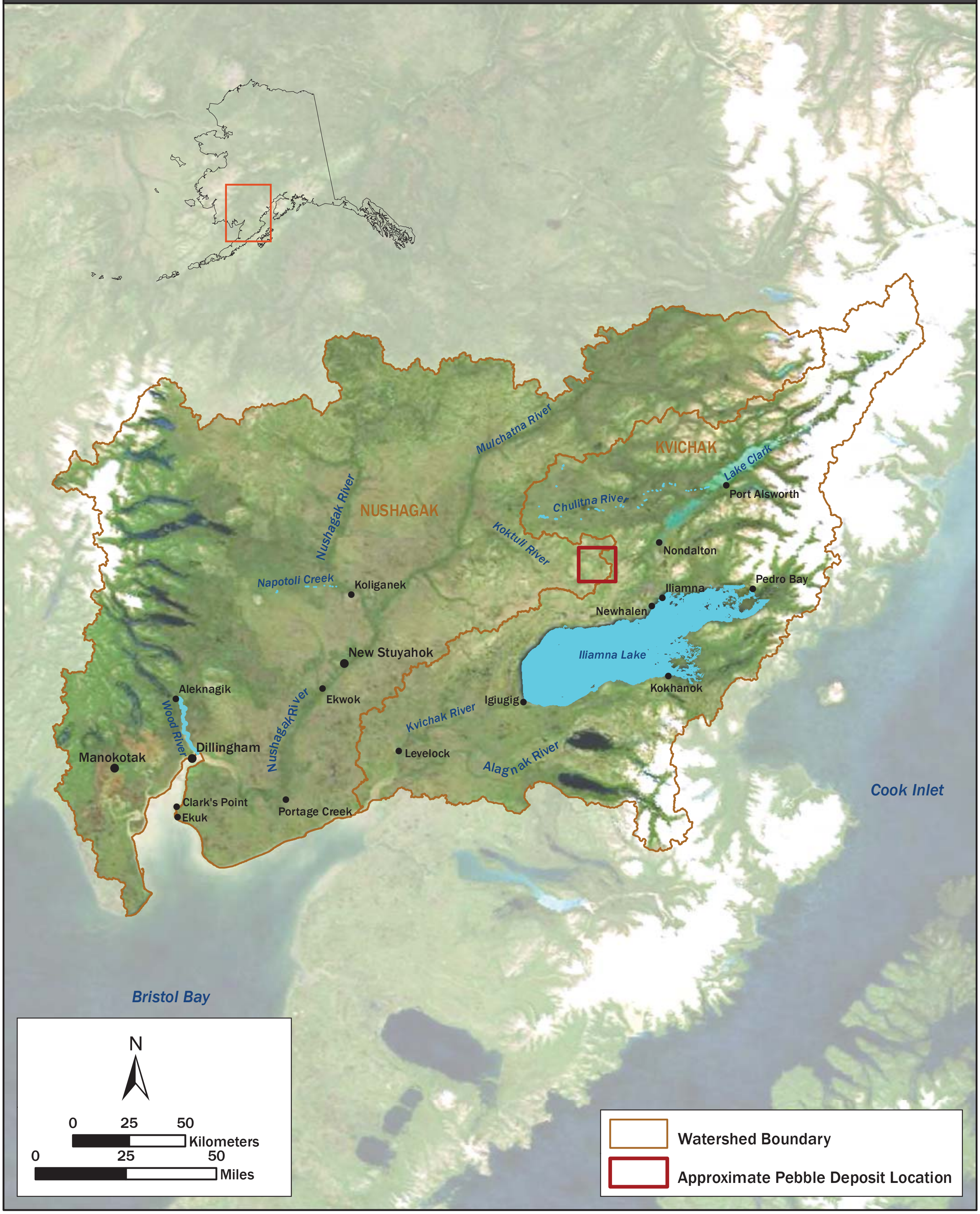
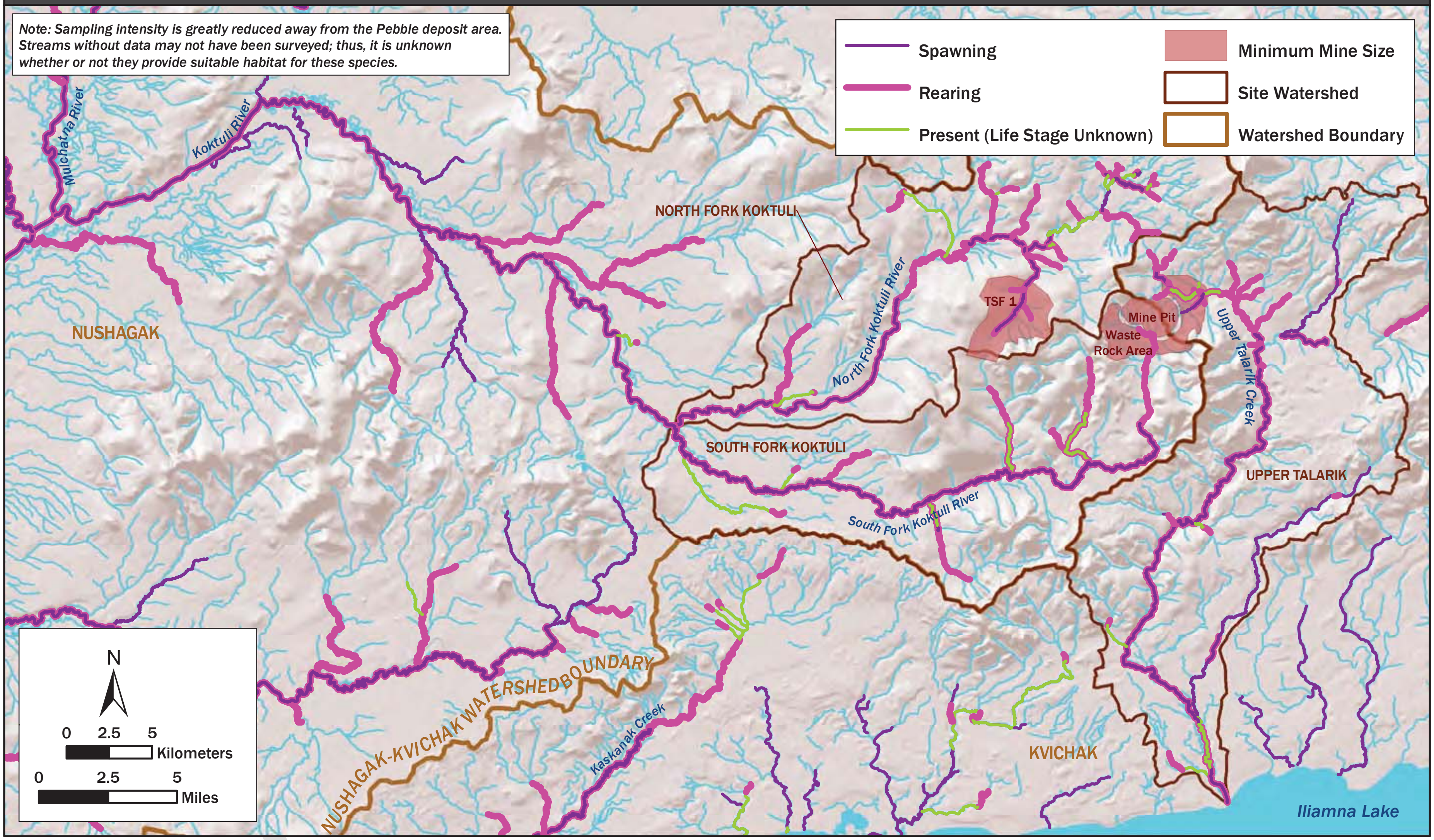


Figure ES-2. Reported Salmon (Sockeye, Chinook, Coho, Pink, and Chum Combined) Distribution in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek. Designation of species spawning, rearing, and presence is based on ADFG Draft 2012 Anadromous Waters Catalog (Johnson in press). Spawning spawning adults observed, rearing juveniles observed, present present, but life stage use not determined. Life-stage specific reach designations are likely underestimates, given the logistical constraints on the ability to accurately capture all streams that may support life-stage use at various times of the year.



This is not an in-depth assessment of a specific mine, but rather an examination of the impacts of mining activities at the scale and with the characteristics realistically foreseeable in the Bristol Bay region, given the nature of mineral deposits in the watershed and the requirements for successful mining development. Known information about the Pebble deposit is very relevant, because it is likely representative of any potential near-future mine development in the area. Thus, the assessment largely analyzes a mine scenario that reflects the expected characteristics of mining operations at the Pebble deposit. However, the analysis is intended to provide a baseline for understanding the potential impacts of mining development throughout the Nushagak River and Kvichak River watersheds. The potential mining of other existing copper deposits in the region would likely reflect the same type of mining activities and facilities analyzed for the Pebble deposit scenario (open pit mining, waste rock piles, tailing storage facilities) and, therefore, would present potential risks similar to those outlined in this assessment.

Ecological Resources

The Bristol Bay watershed provides habitat for numerous animal species, including 35 fishes, more than 190 birds, and more than 40 terrestrial mammals. Many of these species are essential to the structure and function of the region's ecosystems and economies. Chief among these resources is a world-class commercial and sport fishery for Pacific salmon and other important resident fishes. The watershed supports production of all five species of Pacific salmon found in North America: sockeye (*Oncorhynchus nerka*), coho (*O. kisutch*), Chinook or king (*O. tshawytscha*), chum (*O. keta*), and pink (*O. gorbuscha*). Because no hatchery fish are raised or released in the watershed, Bristol Bay's salmon populations are entirely wild. These fish are anadromous—hatching and rearing in freshwater systems, migrating to the sea to grow to adult size, and returning to freshwater systems to spawn and die (Figure ES-3).

The most abundant salmon species in the watershed is sockeye salmon. The Bristol Bay watershed supports the largest sockeye salmon fishery in the world, with approximately 46% of the average global abundance of wild sockeye salmon (Figure ES-4). Between 1990 and 2010, the annual average inshore run of sockeye salmon in Bristol Bay was approximately 37.5 million fish. Annual commercial harvest of sockeye over this same period averaged 27.5 million. Approximately half of the Bristol Bay sockeye salmon production is from the Nushagak River and Kvichak River watersheds—the area of focus for this assessment (Figure ES-4).

In addition to sockeye salmon, Chinook salmon are also abundant. For example, Chinook returns to the Nushagak River are consistently greater than 100,000 fish per year and have exceeded 200,000 fish in 11 years between 1966 and 2010, frequently placing Nushagak River Chinook runs at or near the world's largest. This is noteworthy given the Nushagak River's small watershed area compared to other Chinook-producing rivers such as the Yukon River, which spans Alaska, and the Kuskokwim River in southwest Alaska, just north of Bristol Bay.

Figure ES-3. Salmon Producing Watersheds in the Nushagak River and Kvichak River Watersheds. A total of 568 subwatersheds (total area of 61,317 km^2) were assessed in the Nushagak River and Kvichak River watersheds. The percentage of this area in each category is shown in parentheses in the legend. Note that the southwestern portion of the Nushagak River watershed (i.e., the Nushagak Bay watershed) was not included in this analysis. Data from Demory et al. (1964), Nelson (1967), Salomone et al. (2009), Johnson and Blanche (2011), and ADFG (2012).

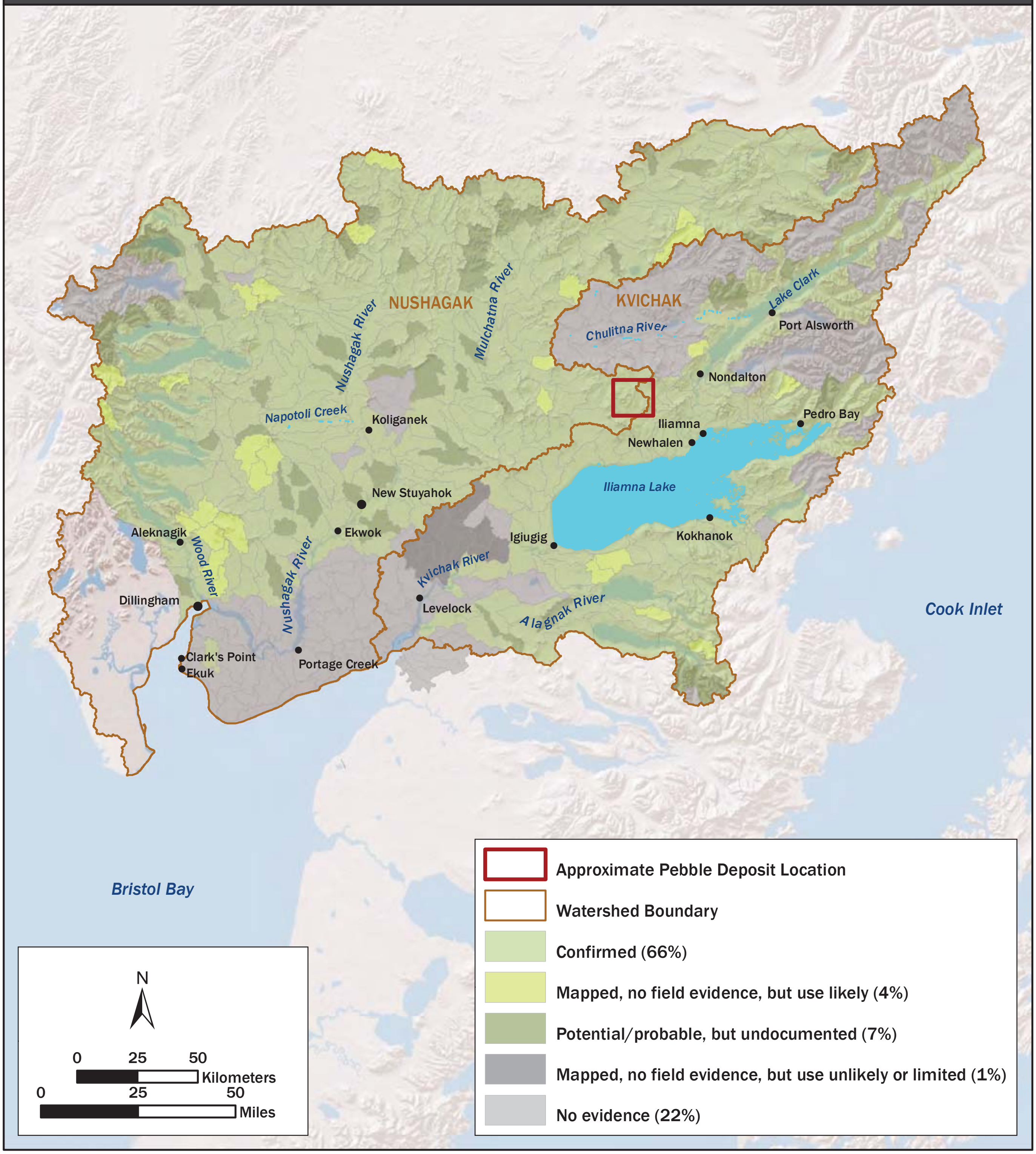
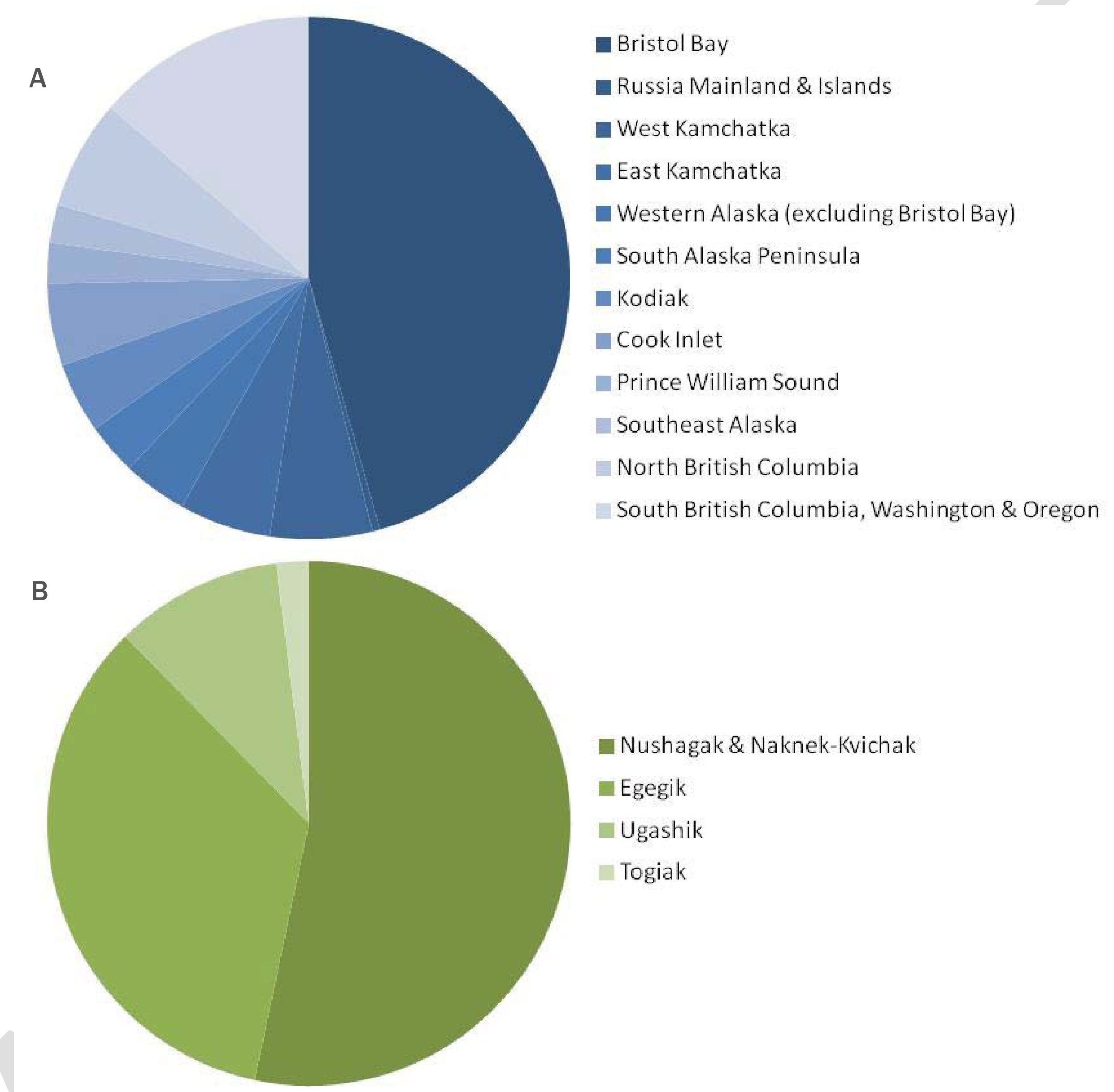


Figure ES-4. Average Annual Relative Abundance and Commercial Harvest of Wild Sockeye Salmon. A. Average annual relative abundance of wild sockeye salmon stocks in the North Pacific, 1956 to 2005; with the exception of Bristol Bay, stocks are ordered from west to east across the North Pacific, from Russia (Russia Mainland and Islands, West Kamchatka, East Kamchatka) to western North America (all other sites). B. Average annual relative commercial sockeye harvest in Bristol Bay watersheds, 1990 to 2009. Data from Ruggerone et al. (2010) and Salomone (pers. comm.).



The Bristol Bay watershed also supports populations of resident fishes that typically remain within the watershed's freshwater habitats throughout their life cycles. The region contains highly productive waters for such sport and subsistence fish species as rainbow trout (*Oncorhynchus mykiss*), Dolly Varden (*Salvelinus malma*), Arctic char (*Salvelinus alpinus*), Arctic grayling (*Thymallus arcticus*), and lake trout (*Salvelinus namaycush*). These fish species occupy a variety of habitats within the watershed, from headwater streams to wetlands to large rivers and lakes. The Bristol Bay region is especially renowned for the abundance and size of its rainbow trout: between 2003 and 2007 an estimated 196,825 rainbow trout were caught in the Bristol Bay Sport Fish Management Area.

The exceptional quality of the Bristol Bay watershed's fish populations can be attributed to several factors, the most important of which is perhaps the watershed's high-quality, diverse aquatic habitats, which are untouched by human-engineered structures and flow management controls. Surface and subsurface waters are highly connected, enabling hydrologic and biochemical connectivity between wetlands, ponds, streams, and rivers, thus increasing the diversity and stability of habitats able to support fish. The high diversity of habitats, high quality of surface and subsurface waters, and relatively low development pressures all contribute to making Bristol Bay a highly productive system. This high diversity of habitats also has enabled the development of high genetic diversity of fish populations. This genetic diversity acts to reduce year-to-year variability in total production and increases the stability of the fishery.

The return of salmon from the Pacific Ocean brings nutrients into the watershed and fuels terrestrial and aquatic food webs. The condition of terrestrial ecosystems in Bristol Bay, therefore, is intimately linked to the condition of salmon populations. Unlike most terrestrial ecosystems, the Bristol Bay watershed has undergone little development and remains largely intact. Consequently, the watershed continues to support its historic complement of species, including large carnivores such as brown bears (*Ursus arctos*), bald eagles (*Haliaeetus leucocephalus*), and gray wolves (*Canis lupus*); ungulates such as moose (*Alces alces gigas*) and caribou (*Rangifer tarandus granti*); and numerous waterfowl species.

Wildlife populations tend to be relatively large in the region, due to the increased biological productivity associated with Pacific salmon runs. Brown bears are abundant in the Nushagak River and Kvichak River watersheds. Moose and caribou also are abundant, with populations especially high in the Nushagak River watershed where felt-leaf willow, a preferred plant species, is abundant. The Nushagak River and Kvichak River watersheds are used by caribou, primarily the Mulchatna caribou herd. This herd ranges widely through these watersheds, but also spends considerable time in other watersheds.

Indigenous Cultures

The Alaska Native cultures present in the Nushagak River and Kvichak River watersheds—the Yup'ik and Dena'ina—are two of the last intact, sustainable salmon-based cultures in the world. In contrast, other Pacific Northwest salmon-based cultures are severely threatened due to development, degraded natural resources, and declining salmon resources. Pacific salmon are no longer found in 40% of their historical breeding ranges in the western United States, and where populations remain, they tend to be significantly reduced or dominated by hatchery fish. Salmon are integral to the entire way of life in these cultures as subsistence food and as the foundation for their language, spirituality, and social structure. The cultures have a strong connection to the landscape and its resources. In the Bristol Bay watershed, this connection has been maintained for at least the past 4,000 years and is in part due to and responsible for the continued pristine condition of the region's landscape and biological resources. The respect and importance given salmon and other wildlife, along with the traditional knowledge of the environment, have produced a sustainable subsistence-based economy. This subsistence-based way of life is a key element of indigenous identity and it serves a wide range of economic, social, and cultural functions in Yup'ik and Dena'ina societies.

Fourteen of Bristol Bay's 25 Alaska Native villages and communities are within the Nushagak River and Kvichak River watersheds, with a total population of 4,337 in 2010. Thirteen of the 14 communities are Federally Recognized Tribal Governments. In the Bristol Bay region, salmon constitute approximately 52% of the subsistence harvest. Subsistence from all sources (fish, moose, and other wildlife) accounts for an average of 80% of protein consumed by area residents. The subsistence way of life in many Alaska Native villages is augmented with activities supporting cash economy transactions. Alaska Native villages, in partnership with Alaska Native corporations and other business interests, are considering a variety of economic development opportunities—mining included. Some Alaska Native villages have decided for themselves that large-scale hard rock mining is not the direction they would like to go, while a few others are seriously considering this opportunity. All are concerned with the long-term sustainability of their communities.

Economics of Ecological Resources

The Bristol Bay watershed supports several economic sectors that are wilderness-compatible and sustainable: commercial, sport and subsistence fishing, sport and subsistence hunting, and non-consumptive recreation. Considering all these sectors, the ecological resources of the Bristol Bay watershed generated nearly $480 million (M) in direct economic expenditures and sales, in 2009, and provided employment for over 14,000 full- and part-time workers.

The Bristol Bay commercial salmon fishery generates the largest component of economic activity and was valued at approximately $300 M in 2009 (first wholesale value) and provided employment for over 11,500 full- and part-time workers at the peak of the season. These estimates do not include retail expenditures from national and international sales.

Based on 2009 data, the Bristol Bay sport-fishing industry supports approximately 29,000 sport-fishing trips, generates approximately $60 M per year, and directly employs over 850 full- and part-time workers. The vast majority of this revenue is spent in the Bristol Bay region. Sport hunting—mostly of caribou, moose, and brown bear—generates more than $8 M per year and employs over 130 full- and part-time workers. The scenic value of the watershed, measured in terms of wildlife viewing and tourism, is estimated to generate an additional $100 M per year and supports nearly 1,700 full- and part-time workers. The subsistence harvest of fish also contributes to the region's economy when Alaskan households spend money on subsistence-related supplies. These contributions are estimated to be slightly over $6 M per year.

Geological Resources

In addition to significant and valuable ecological resources, the Nushagak River and Kvichak River watersheds contain considerable mineral resources. The potential for large-scale mining development within the region is greatest for copper deposits and, to a lesser extent, for intrusion-related gold deposits. Because these deposits are low-grade—meaning that they contain relatively small amounts of metals relative to the amount of ore—mining will be economic only if conducted over a large area, and a large amount of waste material will be produced as a result of mining and processing.

The largest known deposit and the deposit most explored to assess future mining potential is the Pebble deposit. If fully mined, the Pebble deposit could produce more than 11 billion metric tons (1 metric ton = 1,000 kg, approximately 2,200 pounds) of ore, which would make it the largest mine of its type in North America. In comparison, the largest existing copper mine in the United States is the Safford Mine in Arizona with 7.3 billion metric tons of ore. Although the Pebble deposit represents the most imminent and likely site of mine development, other mineral deposits with potentially significant resources exist within the Nushagak River and Kvichak River watersheds. Several specific claims have been filed, many near the Pebble deposit. Findings of this assessment concerning the potential impacts of large-scale mining are generally applicable to these other sites.

Mine Scenario

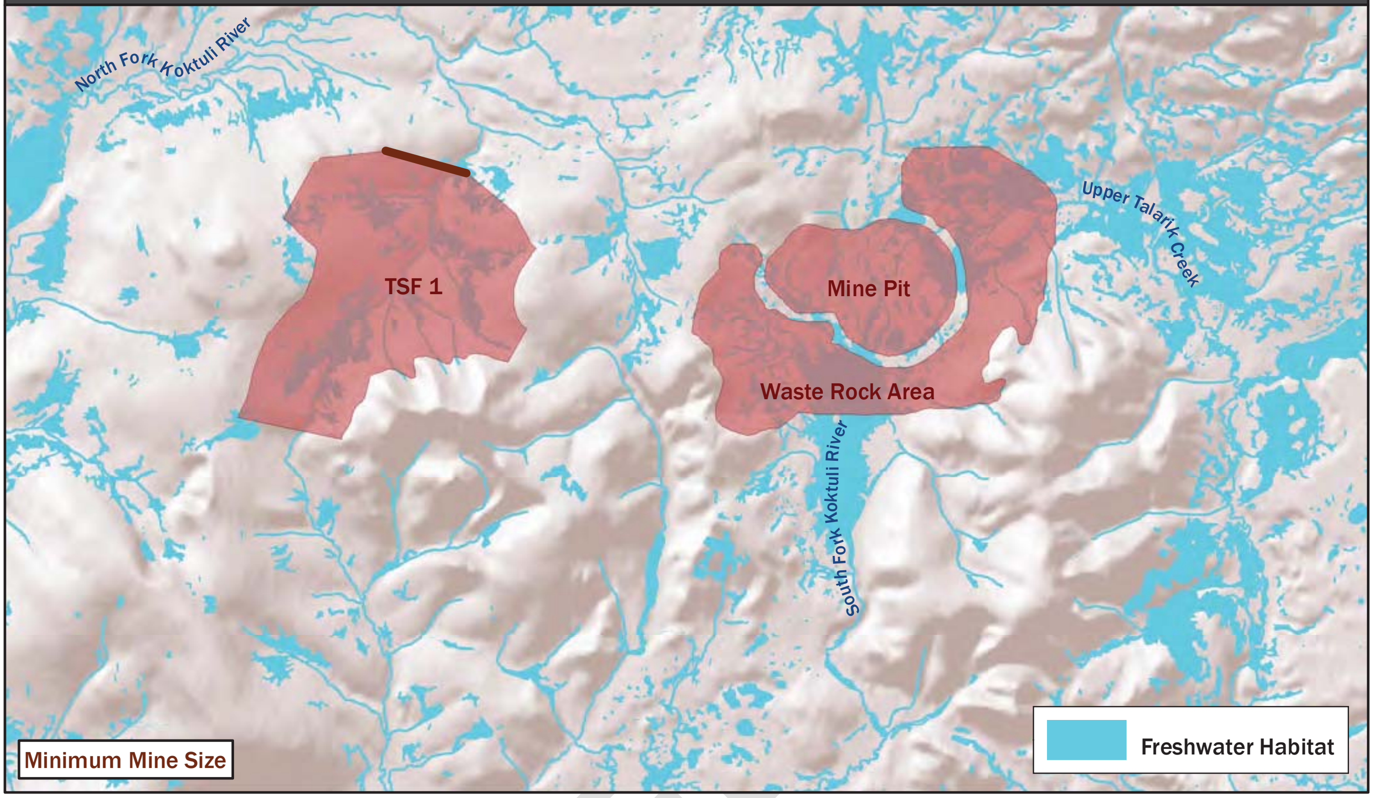
A detailed and final mine plan has not been made available for any of the copper deposits identified in the Bristol Bay watershed, nor is one strictly needed to conduct this assessment. To examine the mining-related stressors that could affect ecological resources in the watershed, we developed a hypothetical mine scenario, designed to be as realistic as possible. The mine scenario is based on mining of the Pebble deposit, because it is the best-characterized mineral resource and the most likely to be developed in the near term. Thus, the mine scenario draws on plans published by the Pebble Limited Partnership (PLP) and baseline data developed by PLP to characterize the likely mine site and surrounding environment. Details of a mining plan for the Pebble deposit or for other deposits in the watershed may differ from our mine scenario; however, our scenario reflects the general characteristics of mineral deposits in the watershed, contemporary mining technologies and best practices, the scale of mining activity required for economic development of the resource, and necessary development of infrastructure to support large-scale mining. Therefore, the USEPA concludes that the mine scenario represents the sort of development plan that can be anticipated for a copper deposit in the Bristol Bay watershed. Uncertainties associated with the mine scenario are discussed later in this executive summary.

The mine scenario includes minimum and maximum mine sizes, based on the amount of ore processed (2 billion metric tons vs. 6.5 billion metric tons), and approximate corresponding mine life spans of 25 to 78 years, respectively. Components of the minimum mine would include a 5.5 km^2 (1,358 acre) mine pit, a 14.9-km^2 (3,686-acre) tailings impoundment behind a 208 m-high (685-foot-high) earthen dam; a 13.3-km^2 (3,286-acre) waste rock pile; a 139-km (86-mile) road with four pipelines for product concentrate, return water, diesel, and natural gas; and facilities for ore processing and support services. The maximum size mine would include a much larger pit and waste rock pile, with a combined area of 38.4 km^2 (9,486 acres), potentially an underground mine, and three tailings impoundments, with a combined area of 43.7 km^2 (10,807 acres) (Figures ES-5 and ES-6).

The first part of the assessment considers routine operation, which assumes that the mine would be designed using practices to minimize environmental impacts and that no significant human or engineering failures occur during or for centuries after operation. The second part of the assessment considers various failures that have occurred during the operation of other mines and have the potential to occur here.

The assessment does not consider all mining-related development. Although the mine scenario assumes development of a deep-water port on Cook Inlet to ship concentrated product elsewhere for smelting and refining, impacts of the development and operation of a deep-water port are not assessed. Additionally, the assessment does not evaluate the potential environmental impacts of one or more electricity-generating power plants that would need to be constructed to provide power at the mine site and the deep-water port facility. This assessment also does not consider potential impacts resulting from secondary development that is likely to accompany a large-scale mine development. Secondary development includes, but is not limited to, additional support services for mine employees and their families, increased recreational development due to increased access, development of vacation homes, and increased transportation infrastructure (i.e., airports, docks, and roads).

Figure ES 5. Minimum and Maximum Footprints in the Assessment Scenario Individual mine components are the mine pit, waste rock piles, and one or more tailings storage facilities (TSFs). The dark bar at the north end of TSF 1 indicates the dam for which tailings dam failure is modeled.



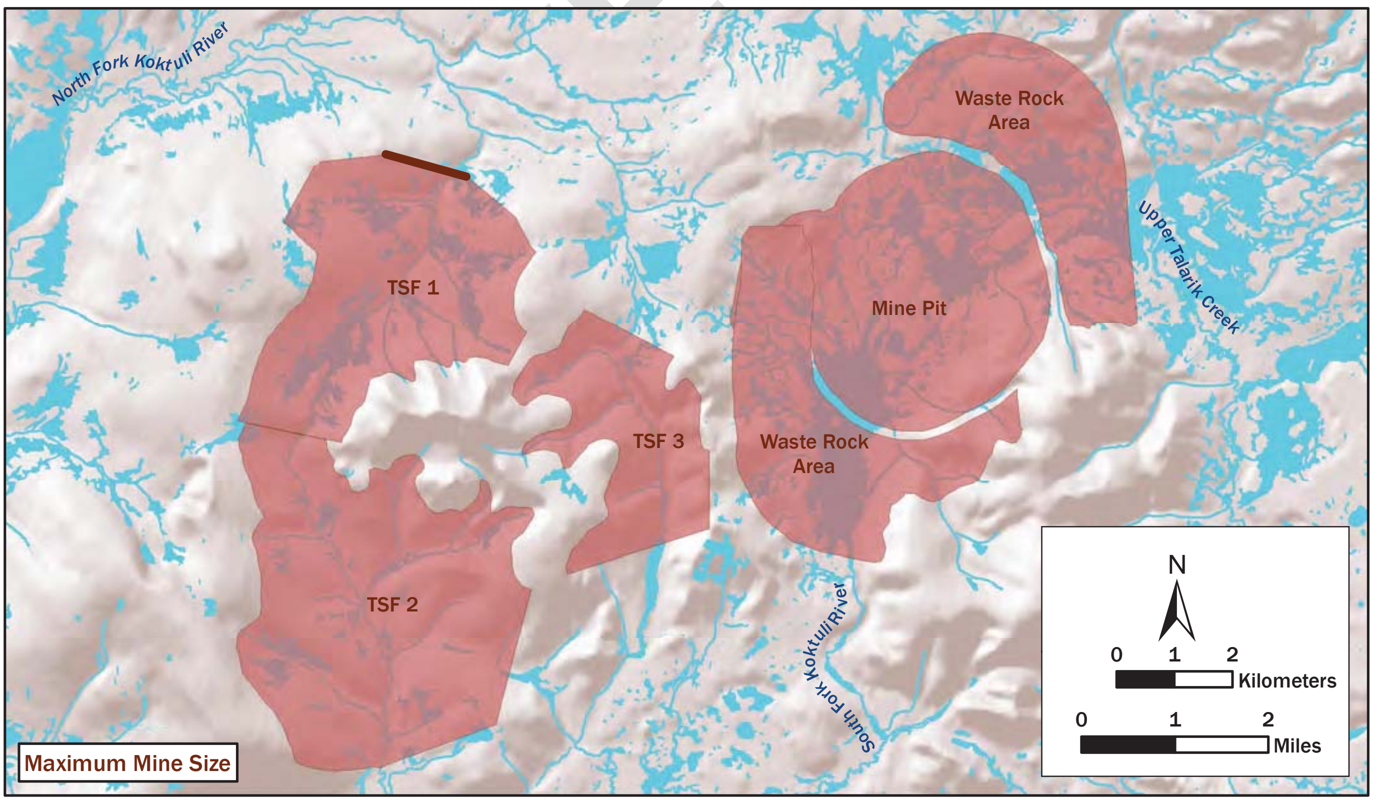
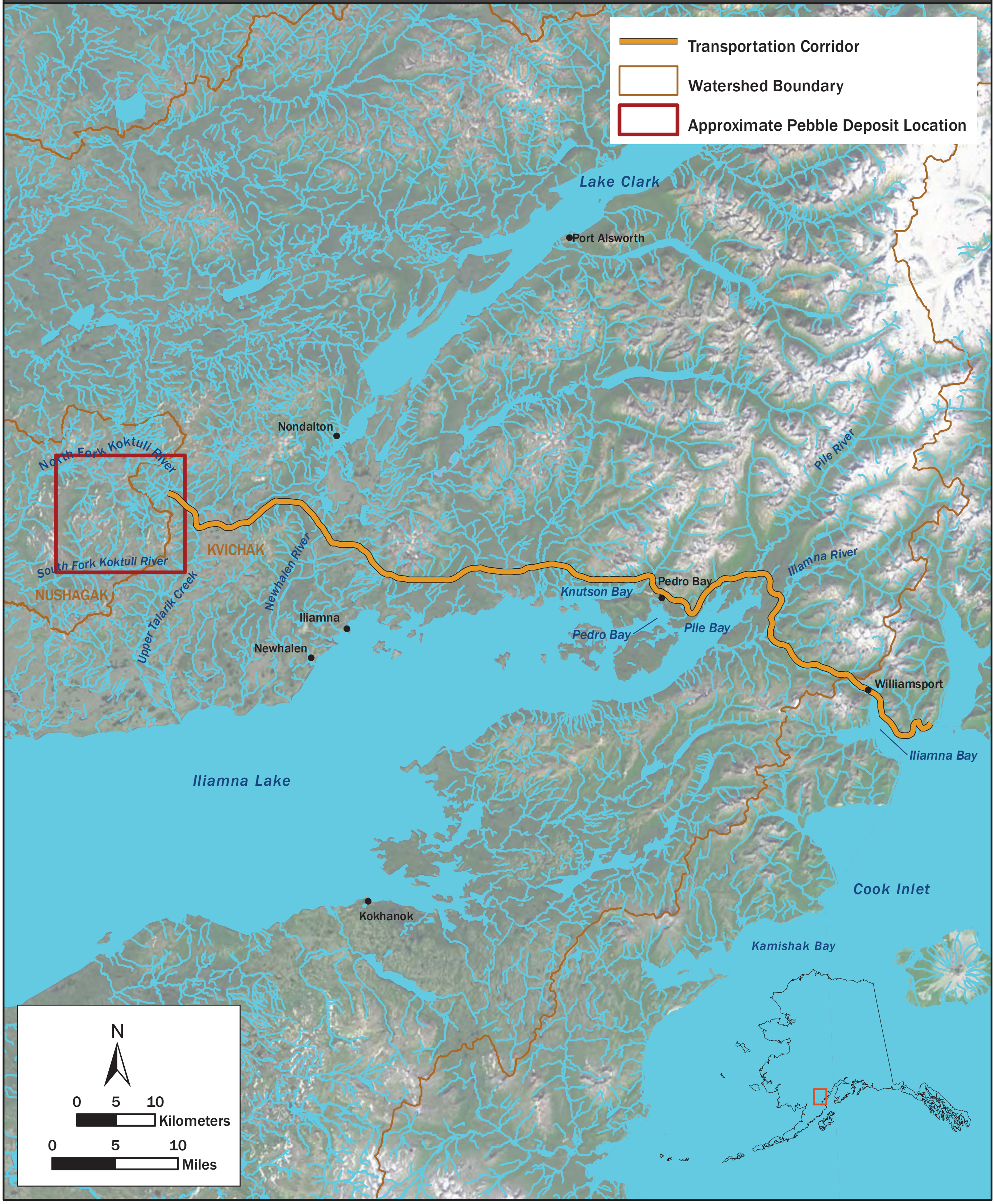


Figure ES-6. Potential 139-km (86-mile) Transportation Corridor Connecting the Pebble Deposit Area to Cook Inlet



Overall Risks to Salmon and Other Fish

Based on the mine scenario, the assessment defines potential mining-related stressors that could affect the Bristol Bay watershed's fish and would consequently have impacts on wildlife and human welfare.

No Failure

No failure, or routine operation, is a mode of operation defined as using the highest design standards and day-to-day practices, with all equipment and management systems operated in accordance with applicable specifications and requirements. In the no failure mode of operation, we assume that best practical engineering and mitigation practices are in place and in optimal operating condition. We do not specify all of those mitigation practices, but rather, we assume that they would be in place and properly functioning. Analyzing routine operations is not meant to imply that a failure-free mining operation is likely; rather, it is meant to isolate the inevitable and foreseeable effects of mining from those that are unintended and thus more difficult to predict. With no failures, adverse effects outside the mine footprint are minimized by complete containment of waste rock and mine tailings, reliable collection of all water from the site, and effective treatment of effluents. Nonetheless, impacts on fish resulting from habitat loss and modification within and beyond the area of mining activity would result from six key direct and indirect mechanisms.

1. **Eliminated or blocked streams** under the minimum and maximum mine footprints (i.e., the mine pit, waste rock piles, and tailings storage facilities) would result in the loss of 87.5 to 141.4 km (55 to 87 miles), respectively, of possible spawning or rearing habitats for coho salmon, Chinook salmon, sockeye salmon, rainbow trout, and Dolly Varden (Figure ES-7).
2. **Reduced flow** resulting from water retention for use in mine operations, ore processing, transport, and other processes would reduce the amount and quality of fish habitat. Reductions in streamflow exceeding 20% would adversely affect habitat in an additional 2 to 10 km (1.2 to 6.2 miles) of streams, reducing production of coho salmon, sockeye salmon, Chinook salmon, rainbow trout, and Dolly Varden. An unquantifiable area of riparian floodplain wetland habitat would either be lost or suffer substantial changes in hydrologic connectivity with streams due to reduced flow from the mine footprint.
3. **Removal of 10.2 to 17.3 km^2 (2,512 to 4,286 acres) of wetlands** in the footprint of the mine would eliminate off-channel habitat for salmon and other fishes. Wetland loss would reduce availability and access to hydraulically and thermally diverse habitats that can provide enhanced foraging opportunities and important rearing habitats for juvenile salmon.
4. **Indirect effects of stream and wetland removal** would include reductions in the quality of downstream habitat for the same species listed above in the three headwater streams draining the mine site. Sources of these indirect effects would include the following.

- Reduced food resources would result from the loss of organic material and drifting invertebrates from the 87.5 to 141.4 km (55 to 87 miles) of streams and streamside wetlands lost to the mine footprint.
- The balance of surface water and groundwater inputs to downstream reaches would shift, potentially reducing winter fish habitat and making the streams less suitable for spawning and rearing.
- Water treatment and reduced passage through groundwater flowpaths could increase summer water temperatures and decrease winter water temperatures, making streams less suitable for salmon, trout, and char.

These indirect effects cannot be quantified but likely would diminish fish production downstream of the mine site.

5. **Diminished habitat quality in streams below road crossings** would result primarily from altered flow, runoff of road salts, and siltation of spawning habitat and reduced invertebrate prey. The road is adjacent to Iliamna Lake and crosses multiple tributary streams. These habitats are important spawning areas for sockeye salmon, putting sockeye particularly at risk to impacts from the road.
6. **Inhibition of salmonid movement at road crossings** could result from culverts that may, over time, block or diminish use of the full stream length.

Failure

The assessment evaluates four failures that have occurred at other large-scale mining and related infrastructure projects and that could occur during mine operations or after mine closure: tailings dam failure, product concentrate or return water pipeline failure, water collection and treatment failures, and failures of roads and culverts. Risks associated with each of these failures are summarized in Table ES-1.

Tailings Dam Failure

Tailings are the waste materials produced during ore processing, which in our scenario would be stored in tailings storage facilities (TSFs) consisting of tailings dams and impoundments. The annual probability of failure for each tailings dam would be in the range of one-in-ten-thousand to one-in-a-million. The probability of one of several tailings dams failing increases with the number of dams. The minimum mine size outlined in the mine scenario includes one TSF with three dams; the maximum mine size includes three TSFs, with a total of eight dams. The TSFs and their component dams are likely to be in place for hundreds to thousands of years, long beyond the life of the mine. Although details for the actual design of mining operations at the Pebble deposit are unknown, available reports from the PLP suggest tailings dams as high as 208 m (685 feet) at TSF 1 (Figure ES-5). At this height, the tailings dam would be higher than the St. Louis Gateway Arch and the Washington Monument (Figure ES-8). We evaluated two dam failures in this assessment: one when the TSF was partially full (partial-volume failure) and one when it was completely full (full-volume failure). In both cases we assumed a release of

20% of the tailings, a conservative estimate that is well within the range of historical tailings dam failures.

Table ES-1. Summary of Probability and Consequences of Potential Failures under the Mine Scenario

Failure Type	Probability[a]	Consequences
Tailings dam	10^{-4} to 10^{-6} per dam-year = recurrence frequency of 10,000 to 1 million years[b]	More than 30 km of salmonid stream would be destroyed and more streams and rivers would have greatly degraded habitat for decades.
Product concentrate pipeline	10^{-3} per km-year = 98% chance per pipeline in 25 years	Most failures would occur between stream or wetland crossings and might have little effect on fish.
Concentrate spill into a stream	2 x 10^{-2} per year = 1.5 stream-contaminating spills in 78 years	Fish and invertebrates would experience acute exposure to toxic water and chronic exposure to toxic sediment in a stream and potentially extending to Iliamna Lake.
Concentrate spill into a wetland	3 x 10^{-2} per year = 2 wetland-contaminating spills in 78 years	Invertebrates and potentially fish would experience acute exposure to toxic water and chronic exposure to toxic sediment in a pond or other wetland.
Return water pipeline	Same as product concentrate pipeline	Fish and invertebrates would experience acute exposure to toxic water.
Culvert, operation	Low	Frequent inspections and regular maintenance would result in few impassable culverts.
Culvert, post-operation	3 x 10^{-1} to 6 x 10^{-1} per culvert-instantaneous = 4 to 10 culverts	In surveys of road culverts, roughly one-third to two-thirds are impassable to fish at any one time. This would result in 4 to 10 salmonid streams blocked.
Water collection and treatment, operation	High	Collection and treatment failures are highly likely to result in release of untreated leachates for hours to months.
Water collection and treatment, planned post-closure	High	Collection and treatment failures are highly likely to result in release of untreated leachates for days to months.
Water collection and treatment, premature post-closure or perpetuity	Certain	When water is no longer managed, untreated leachates would flow to the streams.

a Because of differences in derivation, the probabilities are not directly comparable.

b Based on expected state safety requirements. Observed failure rates for earthen dams are higher (about 5 x 10-4 per year or a recurrence frequency of 2,000 years).

Figure ES 7. Streams and Wetlands Lost (Eliminated and Blocked) Under the Minimum and Maximum Mine Footprints in the Assessment Scenario

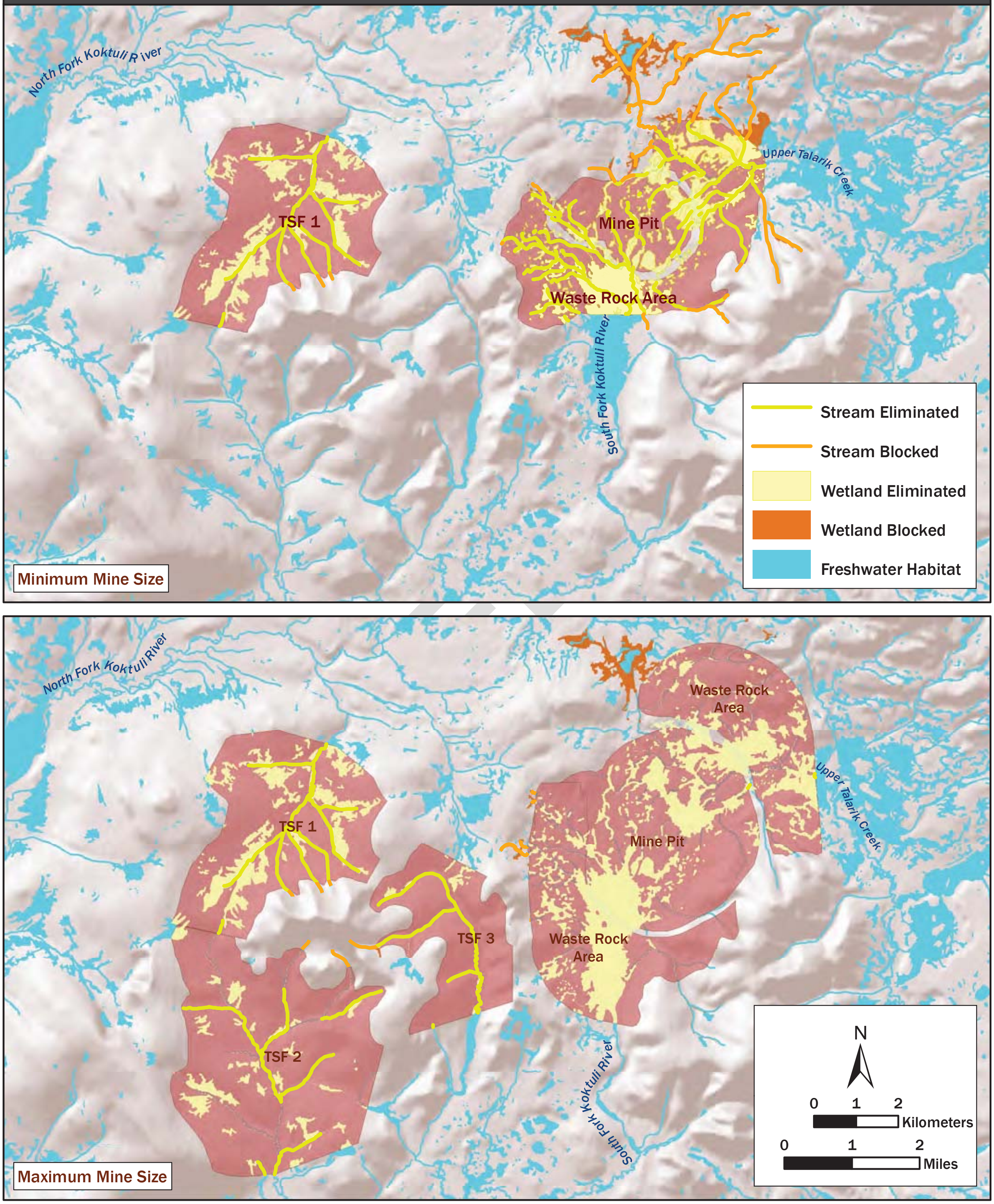
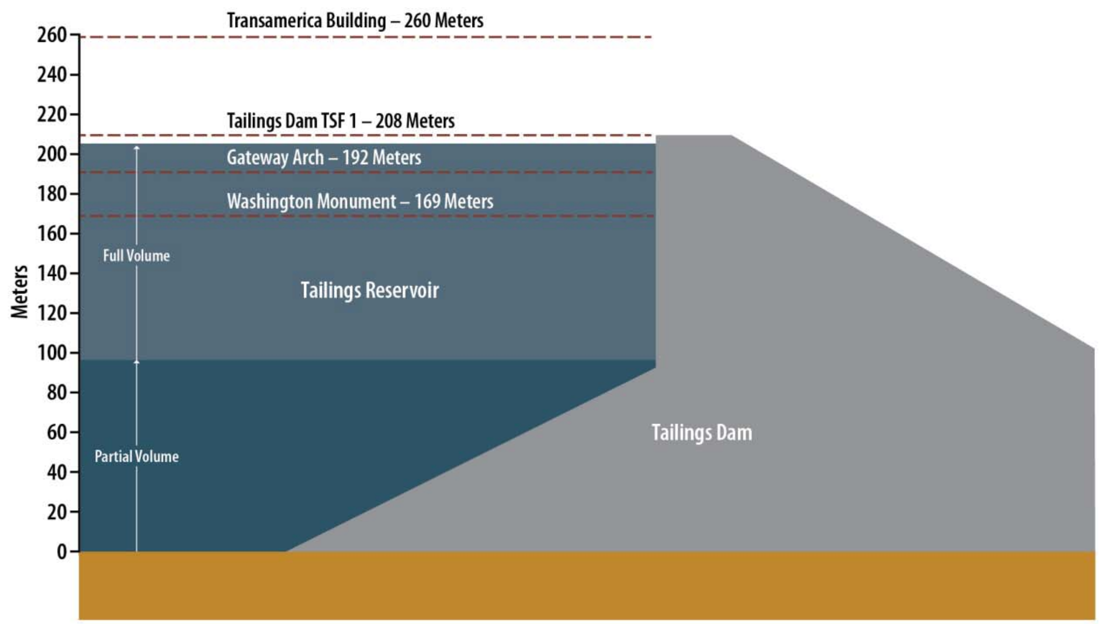


Figure ES-8. Height of the Partial-Volume and Full-Volume Dam at TSF 1, Relative to Common Landmarks



The range of estimated probabilities of dam failure is wide, reflecting the great uncertainty concerning such failures. The most straightforward method of estimating the annual probability of failure of a tailings dam is to use the historical failure rate of similar dams. Three reviews of tailings dam failures produced an average rate of approximately 1 failure per 2,000 dam years, or 5×10^{-4} failures per dam year. The argument against this approach is that it does not fully reflect current engineering practice. Some studies suggest that improved design, construction, and monitoring practices can reduce the failure rate by an order of magnitude or more, resulting in an estimated failure probability within our assumed range. The State of Alaska's guidelines suggest that an applicant follow accepted industry design practices such as those provided by the U.S. Army Corps of Engineers (USACE), Federal Energy Regulatory Commission (FERC), and other agencies. Both USACE and FERC require a minimum factor of safety of 1.5 against slope instability, for the loading condition corresponding to steady seepage from the maximum storage facility. An assessment of the correlation of dam failure probabilities with safety factors against slope instability suggests an annual probability of failure of 1 in 1,000,000 for Category I Facilities (those designed, built, and operated with state-of-the-practice engineering) and 1 in 10,000 for Category II Facilities (those designed, built, and operated using standard engineering practice). This spans the failure frequency used in our failure assessment. The advantage of this approach is that it addresses current regulatory guidelines and engineering practices. The disadvantage is that we do not know whether standard practice or state-of-the practice dams will perform as expected, particularly given the large size of potential dams. In addition, slope instability is only one type of failure; other failure modes, such as overtopping during a flood, would increase overall failure rates.

Failure of the dam at TSF 1 would result in the release of a flood of tailings slurry into the North Fork Koktuli River, scouring the valley and depositing tailings several meters (yards) in depth over the entire floodplain of the river. The complete loss of suitable salmon habitat in the North Fork Koktuli River along at least 30 km (18.6 miles) of stream habitat—the spatial limit of the modeling conducted for this assessment—in the short term (fewer than 10 years) and the high likelihood of very low-quality spawning and rearing habitat in the long term (decades) would result in near-complete loss of mainstem North Fork Koktuli River fish populations. The North Fork Koktuli River currently supports spawning and rearing populations of sockeye, Chinook, and coho salmon; spawning populations of chum salmon; and rearing populations of Dolly Varden and rainbow trout. The slurry flood would continue down the Koktuli River with similar effects, the extent of which cannot be estimated at this time due to model and data limitations.

The tailings dam failures evaluated here are predicted to have the following severe direct and indirect effects on aquatic resources, particularly salmonid fish.

1. **It is likely that the North Fork Koktuli River below the TSF 1 dam and very likely that much of the Koktuli River would not support salmonid fish in the short term (fewer than 10 years).**
   - Deposited tailings would degrade habitat quality for both fish and the invertebrates they eat. Based largely on their copper content, deposited tailings would be toxic to benthic macroinvertebrates, although existing data concerning toxicity to fish is less clear.
   - Deposited tailings would continue to erode from the North Fork Koktuli and Koktuli River valleys.
   - Suspension and redeposition of tailings would likely cause serious habitat degradation in the Koktuli River and downstream into the Mulchatna River.
2. **Those waters would provide very low-quality spawning and rearing habitat for a period of decades.**
   - Recovery of suitable substrates via mobilization and transport of tailings would take years to decades, and would affect much of the watershed downstream of a failed dam.
   - Ultimately, spring floods and stormflows would carry some proportion of the tailings into the Nushagak River.
   - For some years, periods of high flow would be expected to suspend sufficient concentrations of tailings to cause avoidance, reduced growth and fecundity, and even death of fish.
3. **Near-complete loss of North Fork Koktuli River fish populations would likely result from the habitat losses.**

- The Koktuli River watershed is an important producer of Chinook salmon. The Nushagak River watershed, of which the Koktuli River watershed is a part, is the largest producer of Chinook salmon in the Bristol Bay region, with annual runs averaging over 160,000 fish.
- The tailings spill would be expected to eliminate 28% of the Chinook salmon run in the Nushagak River due to loss of the Koktuli River watershed population; an additional 10 to 20% could be lost due to tailings deposited in the Mulchatna River and its tributaries.
- Sockeye are the most abundant salmon returning to the Nushagak River watershed, with annual runs averaging more than 1.3 million fish. The proportion of sockeye and other salmon species of Koktuli-Mulchatna origin is unknown.
- Similarly, populations of rainbow trout and Dolly Varden would be lost for years to decades. Quantitative estimates of the impacts on population sizes are not possible.

Effects would be qualitatively the same for both the partial-volume and full-volume dam failures, although effects from the full-volume failure would extend further and last longer. Failure of dams at the two additional TSFs under the maximum mine size (TSF 2 and TSF 3) were not modeled, but would have similar effects in the South Fork Koktuli River and downstream. However, because their volumes would be smaller, effects would be less extensive.

Pipeline Failures

Under the mine scenario, the primary product of the mine would be a concentrate of copper and other metals that would be pumped in a pipeline to a shipping facility on Cook Inlet. Water carrying the sand-like concentrate would be returned to the mine site in a second pipeline. Based on the record of pipelines in general, and the world's largest metal concentrate pipeline in particular, one to two near-stream failures of each of these pipelines would be expected to occur over the life of the maximum mine (78 years). Failure of either the product or the return water pipelines would release water that is expected to be highly toxic, potentially killing fish and invertebrates in the affected stream over a relatively brief period. If concentrate spilled into a stream, it would settle and form bed sediment predicted to be highly toxic based on its high copper content and acidity. Unless the receiving stream was dredged, causing additional long-term damage, this sediment would persist for decades before ultimately being washed into Iliamna Lake. Potential concentrations in the lake could not be predicted, but near the pipeline route Iliamna Lake contains important beach spawning areas for sockeye salmon that could be exposed to a toxic spill. Sockeye also spawn in the lower reaches of streams which could be directly contaminated by a spill.

Water Collection and Treatment Failures

There is a long history of unplanned discharges of contaminated waters from mine sites into surface and ground waters. Water in contact with tailings or waste rock would leach copper and other metals. The failure of collection and treatment systems due to imperfect design or operation, or the failure to maintain and operate these systems in perpetuity, could result in contamination of one or more streams draining the site. Based on a review of historical and currently operating mines, some failure of the collection and treatment systems is likely during operation or post-closure periods. These failures could range from operational failures resulting in short-term releases of untreated leachates, to long-term failures to operate the collection and treatment system in perpetuity. Our evaluation looked at the realistic possibility of leachate escaping at the base of TSF 1. We also considered a failure to collect and treat leachate from waste rock piles around the mine pit.

Test leachates from the tailings and non-ore-bearing Tertiary waste rocks—those formed between approximately 65 million to 2.5 million years ago—are mildly toxic; they would require an approximately two-fold dilution to achieve water quality criteria for copper, but they are not expected to be toxic to salmonids. If Tertiary rock were to be used as planned for construction of mining infrastructure, leachate from these areas would need to be collected and treated to avoid toxic effects on benthic invertebrates. Our risk assessment did not evaluate this potential pathway in detail.

Pre-Tertiary waste rocks, which would be excavated to expose the ore body, are acid-forming with high copper concentrations in test leachates and would require 2,900 to 52,000-fold dilution to achieve water quality criteria. If leachate from a waste rock pile surrounding the mine pit was not collected, the 10.6 million m^3 (approximately 2.8 billion gallons) of leachate per year from the waste rock pile could constitute source water for Upper Talarik Creek, which flows to Iliamna Lake. The total flow of Upper Talarik Creek would provide only 18-fold dilution, so failure to prevent leachate releases could cause the entire creek and a potentially large mixing zone in the lake to become toxic to fish and the sensitive invertebrates upon which they feed. The significance of such an event to salmon is illustrated by the abundance of spawning salmon in Upper Talarik Creek. As many as 33,000 sockeye and 6,300 coho spawners have been counted in the creek on a single day; in 2008, 82,000 sockeye were counted in Upper Talarik Creek and one of its tributaries in a single day. The toxic event described could kill adult fish or the millions of eggs, larvae, and fry that they generate.

Road and Culvert Failures

Within the Kvichak River watershed, the transportation corridor would cross 34 streams and rivers supporting migrating and/or resident salmonids, including 17 streams designated as anadromous waters at the location of the crossing. The most likely serious failure associated with the transportation corridor would be blockage or failure of culverts. Culverts commonly become blocked by debris that may not stop water flow but would block fish passage. If these blockages occurred during adult salmon immigration or juvenile salmon outmigration and were not cleared for several days, production of a year-class (i.e., fish spawned in the same year) could be lost or diminished.

Culverts can also fail to convey water as a result of landslides or, more commonly, floods that wash out the culvert. In such failures, the stream could be temporarily impassible to fish until the culvert is repaired or until erosion reestablishes the channel. If the failure occurs during a critical period in salmon migration, the effects would be the same as with a debris blockage (i.e., a lost or diminished year-class).

Culvert failures also would result in the downstream transport and deposition of silt, which could cause returning salmon to avoid a stream if they arrived during or immediately following the failure. More likely, deposition of silt would smother salmon eggs and larvae, if they were present, and would degrade the downstream habitat for salmonid fish and the invertebrates that they eat.

Extended blockage of fish passage at road crossings is unlikely during operation assuming best-case scenario daily inspection and maintenance. However, after mine operations cease, the road may be maintained less carefully or be transferred to a governmental entity. In that case, the proportion of culverts that are impassable would be expected to revert to levels found in published surveys of public roads (30 to 66%). Of the approximately 50 culverts that would be required, 17 would be on streams that are believed to support salmonids. Hence, over the long term, 4 to 10 streams would be expected to lose passage of salmon, rainbow trout, or Dolly Varden, and some proportion of those streams would have degraded downstream habitat resulting from the sedimentation from washout of the road.

Common Mode Failures

Multiple, simultaneous failures could occur as a result of a common event, such as the occurrence of a severe storm with heavy precipitation (particularly one that fell on spring snow cover) or a major earthquake. Such an event could cause one to three tailings dam failures that would spill tailings slurry into streams and rivers, road culvert washouts that would send sediments downstream and potentially block fish passage, and pipeline failures that would release product slurry, return water, or diesel fuel. The effects of each of these accidents individually would be the same as discussed previously, but their co-occurrence would cause cumulative effects on salmonid populations and make any mitigative response more difficult.

Over the perpetual timeframe that tailings, mine pit, and waste rock would be in place, the likelihood of multiple extreme precipitation events, earthquakes, or combinations of these events becomes much greater. Multiple events further increase the chances of weakening and eventual failure of facilities that are still in place.

Overall Loss of Wetlands

Wetlands are a dominant feature of the landscape in the Pebble deposit area and throughout the Bristol Bay watershed, and are important habitats for salmon and other fish. Ponds and riparian wetlands provide spawning, rearing, and refuge habitat for both anadromous salmonids and resident fish species. Other wetlands moderate flows and water quality, and can influence downstream delivery of dissolved organic matter, particulate organic matter, and aquatic macroinvertebrates that supply food sources to fish. Under the mine scenario, wetlands would be filled or excavated in 10.2 km^2 (2,512 acres) and 17.3 km^2 (4,286 acres) of the minimum and maximum mine footprints, respectively. An additional 1.9 km^2 (481 acres) and 1.1 km^2 (267 acres) of riparian wetlands would be blocked by the minimum and maximum footprints, respectively, and would be lost or suffer substantial changes in hydrologic connectivity with streams as a result of reduced flow from the mine footprint. Another 0.18 km^2

(44 acres) of wetlands would be filled in the Kvichak River watershed by the roadbed of the transportation corridor. By interrupting flow and adding silt and salts, the roadbed would also affect approximately 2.4 to 4.9 km^2 (593 to 1,211 acres) of wetlands. Finally, a tailings or product concentrate spill could damage wetlands and eliminate or degrade their capacity to support fish.

Fish-Mediated Risk to Wildlife

Although the effects of reduced salmon, trout, and char production on wildlife—the fish-mediated risk to wildlife—cannot be quantified given available data, some reduction in wildlife would be expected under the mine scenario. Changes in the occurrence and abundance of salmon have the potential to change animal behavior and reduce wildlife population abundances. Assuming no failures, routine operations would be expected to have local effects on brown bears, wolves, bald eagles, and other wildlife that consume salmon as a result of reduced salmon abundance from the loss and degradation of habitat in or immediately downstream of the mine footprint. Any of the accidents or failures evaluated would increase effects on salmon, which would proportionately reduce the abundance of their predators.

The abundance and production of wildlife also is enhanced by the marine nutrients that salmon carry on their spawning migration. Those nutrients are released into streams when the salmon die, enhancing the production of other aquatic species that feed wildlife. Salmon predators deposit these nutrients on the landscape, thereby fertilizing the vegetation and increasing the abundance and production of moose, caribou, and other wildlife that depend on vegetation for food.

Fish-Mediated Risk to Indigenous Culture

Under routine operations with no major accidents or failures, the predicted loss and degradation of salmon, char, and trout habitat in North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek is expected to have some impact on Alaska Native cultures of the Bristol Bay watershed. Fishing and hunting practices are expected to change in direct response to the stream, wetland, and terrestrial habitats lost due to the footprints of the mine site and the transportation corridor. Additionally, it is also possible that subsistence use of salmon resources could decrease based on the perception of reduced fish or water quality resulting from mining.

The potential for significant effects on indigenous cultures is much greater from a mine failure than from routine operations. As described above, failures could reduce or eliminate fish populations in affected areas, including areas significant distances downstream from the mine.

Any loss of fish production from these potential failures would reduce the availability of those subsistence resources to local Alaska Native villages, and the reduction of food supply potentially would have negative consequences on human health if alternative food resources are not available. Salmon-based subsistence is integral to Alaska Native cultures. If salmon quality or quantity is adversely affected, the nutritional, social, and spiritual health of Alaska Natives and their culture will potentially decline.

Cumulative Risks

This assessment has focused on the potential effects of a single, hypothetical mine on salmon and other resources in the Nushagak and Kvichak River watersheds, including the cumulative effects of multiple stressors associated with that mine. However, the potential exists for development of multiple mines and associated infrastructure in these watersheds. Each potential mine poses risks similar to those identified for the mine scenario. Estimates of the loss of stream and wetland habitats would differ across different deposits based on the size and location of mining operations within the watersheds. Individually, each mine footprint would eliminate some amount of fish-supporting habitat and, should human or engineering failures occur, affect fish habitats beyond the mine footprint. Cumulatively, multiple mines have the potential to decrease the abundance and genetic diversity of fish populations and thereby increase their annual variability.

We considered development of mines at several sites in the Nushagak River watershed, including Big Chunk, Groundhog Mountain, and Humble claims. These sites were chosen, because all contain copper deposits that have generated exploratory interest. If all four mine sites were developed, the cumulative area covered by TSFs alone would be close to 73 km^2 (19,038 acres). Loss of stream habitats as a result of eliminated or blocked streams could reach 233 km (144 miles). The combined facilities would eliminate an estimated 34.6 km (21.5 miles) of documented salmon streams. The length of salmon stream affected is likely an underestimate, because most streams have not been sampled for the presence of salmon. Loss of these distinct streams would likely result in the loss of their associated salmon populations, reducing the genetic and life-history diversity generated through the existence of numerous distinct populations.

Summary of Uncertainties in Mine Design and Operation

This assessment of a hypothetical mine scenario is generally applicable to the copper deposits in the Bristol Bay watershed and is based on specific characteristics of the Pebble deposit. The mine scenario does not represent the plans of any mining company; if the resource is mined in the future, actual events will undoubtedly deviate from this scenario. This is not a source of uncertainty, but rather an inherent aspect of a predictive assessment. Even an environmental assessment of a proposed plan by a mining company would be an assessment of a scenario that undoubtedly would differ from the ultimate development.

Multiple uncertainties are inherent in planning, designing, constructing, operating, and closing a mine.

- Mines are complex systems requiring skilled engineered design and operation. The uncertainties facing mining and geotechnical engineers include unknown geologic defects, uncertain values in geological properties, limited knowledge of mechanisms and processes, and human error in design and construction. Vick (2002) notes that models used to predict the behavior of an engineered system are "idealizations of the processes they are taken to represent, and it is well recognized that the necessary simplifications and approximations can introduce error in the model." Engineers use professional judgment in addressing uncertainty (Vick 2002).

- Accidents are inherently unpredictable. Though systems can be put into place to protect against system failures, seemingly logical decisions about how to respond to a given situation can have unexpected consequences resulting from human error (e.g., the January 2012 overflow of the tailings dam at the Nixon Fork Mine near McGrath, Alaska). Further, unforeseen events or events that are more extreme than anticipated can negate the apparent wisdom of prior decisions (Caldwell and Charlebois 2010).
- The ore deposit would be mined for decades and the waste would require management for centuries or even in perpetuity. Engineered waste storage systems of mines have only been in existence for about 50 years. Their long-term behavior is not known. The response of our best technology in the construction of tailings dams is untested and unknown in the face of centuries of extreme events such as earthquakes and weather.
- Mine management or ownership may change over time. Over the long timespan (centuries) of mining and post-mining care, generations of mine operators must exercise due diligence. Priorities are likely to change in the face of financial circumstances, changing markets for metals, new information about the resource, political priorities, or any number of currently unforeseeable changes in circumstance.

Such uncertainties are inherent in any complex enterprise, particularly when they involve an incompletely characterized natural system. However, the large scales and long durations implied by the effort required to exploit this resource make these inherent uncertainties more prominent.

Summary of Uncertainties and Limitations in the Assessment

Significant uncertainties about and limitations of the estimated potential effects of the mine scenario, as judged by the assessment authors, include the following.

- Any mine plan submitted by a mining company may not exactly reflect the location and sizes of the mine pit, waste rock pile, and tailings storage facilities, and the location and length of the transportation corridor used in the scenario for this assessment. An actual mine plan may be smaller, larger, or laid out differently than the mine scenario considered here.
- The estimated annual probability of tailings dam failure is uncertain and based on both design goals and historical experience. Actual failure rates could be higher or lower than the estimated probability.

- The proportion of the tailings that would spill in the event of a dam failure could be larger than the largest value modeled (20%).
- The long-term fate of the spilled tailings in the event of a dam failure could not be quantified. Analogous to other cases, it is likely that tailings would erode from the areas of initial deposition and move downstream over a period of more than a decade. However, the data needed to model that process and the resources needed to develop that model were not available.
- Consequences of the loss and degradation of habitat on fish populations could not be quantified because of the lack of quantitative information concerning salmon, char, and trout populations. The occurrence of salmonid species in rivers and major streams is known, but information on abundances, productivities, and limiting factors within each of the watersheds is not available. Estimating changes in populations would require population modeling, which requires knowledge of life-stage-specific survival and production as well as knowledge of limiting factors and processes that are not available. Further, it requires knowledge of how temperature, habitat structure, prey availability, density dependence, and sublethal toxicity influence life-stage-specific survival and production, which is not available. Obtaining that information would require more detailed monitoring and experimentation. Further, salmon populations naturally vary in size because of a great many factors that vary among locations and years. Collecting sufficient data to establish reliable salmon population estimates takes many years. Estimated effects of mining on habitat become the available surrogate for estimated effects on fish populations.
- Standard leaching test data are available for test tailings and waste rocks from the Pebble deposit, but these results are uncertain predictors of the actual composition of leachates from tailings impoundments, tailings deposited in streams and on their floodplains, and waste rocks in piles.
- The effects of tailings and product concentrate deposited in spawning and rearing habitat are uncertain. It is clear that they would have harmful physical and toxicological effects on salmonid larvae or sheltering juveniles, but the concentration in spawning gravels required to reduce salmonid reproductive success is unknown.
- The actual response of Alaska Native cultures to any impacts of the mine scenario is uncertain. Interviews with village elders and culture bearers, and other evidence suggest that responses would involve more than the need to compensate for lost food and would likely include some degree of cultural disruption. It is not possible to predict specific changes in demographics, cultural practices, or physical and mental health.

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CHAPTER 1. INTRODUCTION

The Bristol Bay watershed in southwestern Alaska supports the largest sockeye salmon fishery in the world, is home to 25 Federally Recognized Tribes and contains abundant natural resources, including mineral reserves. Worldwide attention to this watershed has increased because of widespread mineral exploration activities and the discovery of a large ore deposit in the watershed's northeast central region. The potential for large-scale mining activities has raised concerns about the quality and sustainability of Bristol Bay's world-class fisheries, and the future of Alaska Natives who have maintained a salmon-based culture and a subsistence-based lifestyle for at least 4,000 years.

This assessment represents a review and synthesis of available information to identify potential impacts of large-scale mining development on the Bristol Bay watershed's fisheries and the wildlife and Alaska Native cultures of the region. There are three main drivers for the assessment. The first driver is concern for the ecological goods and services provided by the Bristol Bay watershed. The watershed supports production of all five species of Pacific salmon found in North America, including almost half of the world's commercial sockeye salmon harvest. In 2009, Bristol Bay's wild salmon ecosystem, including its commercial, recreational, and subsistence fisheries, generated $480 million in direct annual economic expenditures in the region and sales, and provided employment for over 14,000 full- and part-time workers.

The second driver is mining. There are 17 existing mine claims in the watershed. The largest of these claims belongs to the Pebble Limited Partnership (PLP). Although PLP has not yet submitted an application for a mine, publicly available information strongly suggests that a mine at the Pebble deposit has the potential to become one of the largest mining developments in the world. The Pebble deposit is a large, low-grade deposit containing copper, gold, and molybdenum-bearing minerals. Extraction is expected to include the creation of a large open pit (as wide as 1 to 2 miles across and thousands of feet deep), the production of large amounts (as much as 23 billion tons) of waste rock and mine tailings, the creation of an approximately 139-km (86-mile) transportation corridor connecting the deposit area to

Cook Inlet, and the development of a deep-water port. Revenues from the mine have been estimated at between $300 billion and $500 billion over the life of the mine.

The third driver for this assessment is multiple requests for the U.S. Environmental Protection Agency (USEPA) to become involved to protect aquatic resources and salmon in the watershed. Nine Bristol Bay Federally Recognized Tribes, the Bristol Bay Native Association, the Bristol Bay Native Corporation, other Tribal organizations, and many groups and individuals have asked USEPA to restrict certain large-scale mining activities in the Bristol Bay watershed using its authorities under the Clean Water Act. These groups are concerned that large-scale mining could adversely affect the region's valuable natural resources, particularly its fisheries. In contrast, four Bristol Bay Federally Recognized Tribes, other Tribal organizations, the governor of Alaska, and others groups and individuals, including PLP, have asked USEPA to wait to evaluate the watershed until formal mine permit applications have been submitted.

Recognizing the importance of balancing potential future development with the goals of sustaining ecological resources and traditional Alaska Native cultures, and recognizing the high level of interest concerning potential development in the Bristol Bay watershed, USEPA initiated this assessment. Its focus is to examine the potential impacts of large-scale mining development on the region's fisheries, and associated impacts on wildlife and Alaska Native cultures dependent upon those fisheries. We have limited the assessment to the Nushagak River and Kvichak River watersheds, as they account for more than half of the Bristol Bay watershed's area and are most likely to be affected by large-scale mining development. This assessment does not provide an economic cost/benefit analysis of mining in the region.

We used the following approach to develop our assessment, based on USEPA guidelines for completion of an ecological risk assessment (USEPA 1998). First, we completed a comprehensive review of existing literature to provide background information on Bristol Bay, particularly the Nushagak River and Kvichak River watersheds. We compiled information on Pacific salmon, their biology, and their habitat preferences. We assembled background information on mining and other mine sites, with a focus on porphyry copper mining to reflect the Pebble deposit type. We also looked at watersheds that currently support both surface mine operations and salmon fisheries, using the Fraser River in British Columbia as a case study. Because mine claims in Bristol Bay are remote and substantial transportation corridors would need to be developed to remove minerals from the area, we also assembled background information on the potential impacts of road and pipeline crossings on aquatic systems. Given concerns about potential impacts on Alaska Native cultures and on the Bristol Bay salmon fishery, we also included background information on Alaska Native cultures and fishery economics. Much of this background characterization is provided in the appendices to this assessment.

Using these background characterization studies, we developed a series of conceptual models to show potential links between sources, stressors, and endpoints of interest (fish, wildlife, and Alaska Natives) in the assessment. These conceptual models were revised based on input received from an Intergovernmental Technical Team (IGTT) representing federal, state, local, and Tribal representatives.

Because none of the parties holding mine claims in Bristol Bay have submitted a formal application and mine plan, we developed a hypothetical but realistic mine scenario. This mine scenario, coupled with the conceptual models, was used to inform what information was needed for our assessment.

Our assessment is organized into nine chapters. This introduction is followed by Chapter 2, Characterization of Current Condition, which presents the background on current resource conditions in the Bristol Bay watershed, particularly the Nushagak River and Kvichak River watersheds. Information in Chapter 2 was taken from the detailed background material presented in the characterization studies provided as appendices to this assessment. Characterization study results incorporated into Chapter 2 include information on anadromous fish (Appendix A), non-anadromous fish (Appendix B), wildlife (Appendix C), Alaska Native culture (Appendix D), fishery economics (Appendix E), and marine resources (Appendix F).

Chapter 3, Problem Formulation, defines the problem addressed by the assessment, via more detailed consideration of the scope and endpoints for the assessment. Problem formulation is a critical part of the ecological risk assessment process (USEPA 1998). Chapter 4, Mining Background and Scenario, provides background information on mining, particularly porphyry copper mining, and details the mine scenario on which the subsequent risk assessment is based. Appendix G provides information on roads and pipelines, and Appendices H and I provide more detailed information supporting the mine scenario, in terms of geochemistry and mitigation practices.

Chapter 5, Risk Assessment: No Failure, presents a risk assessment analysis for routine mine operations. Chapter 6, Risk Assessment: Failure, presents a similar risk assessment analysis for potential accidents and infrastructure failures. Chapter 7, Cumulative and Watershed-Scale Effects considers potential effects of multiple mines. Chapter 8, Risk Characterization, provides the integrated risk characterization. Chapter 9 provides references cited in the assessment.

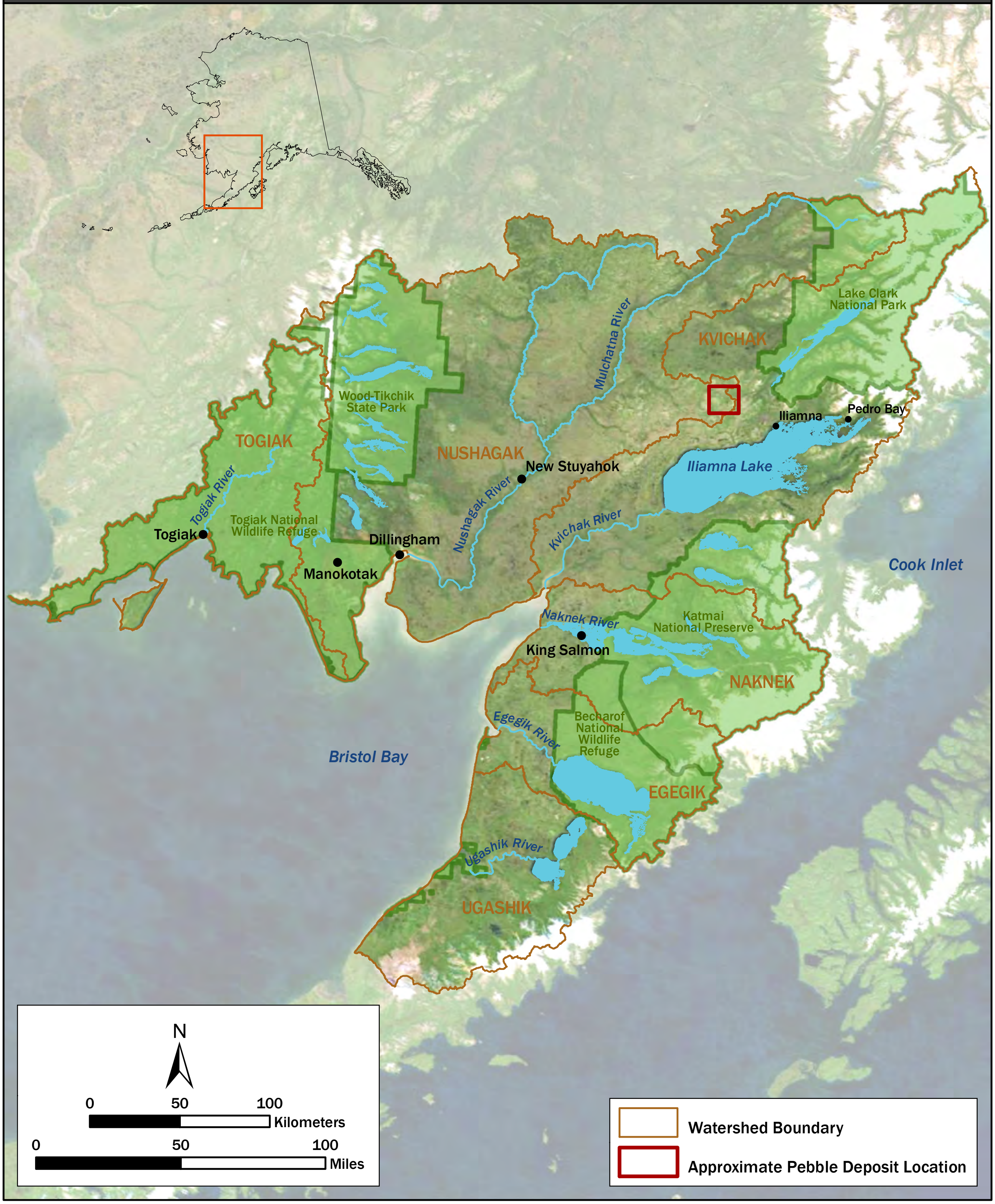
CHAPTER 2. CHARACTERIZATION OF CURRENT CONDITION

To assess potential impacts of mining development on the Bristol Bay watershed, one must first consider the current condition of the region's resources. In this section, we summarize the current status and condition of the Bristol Bay watershed's biological and cultural resources, the watershed characteristics that contribute to the quality and quantity of these resources, and the significance of these resources relative to those in other regions, particularly in terms of Pacific salmon stocks. More detailed characterizations of the Bristol Bay region's natural and cultural resources can be found in Appendices A through D.

2.1 Introduction to Bristol Bay Region

Bristol Bay is a large gulf of the Bering Sea located in southwestern Alaska. The land area draining to Bristol Bay consists of six major watersheds—from west to east, the Togiak, Nushagak, Kvichak, Naknek, Egegik, and Ugashik Rivers (Figure 2-1)—and seven small watersheds in the northern portion of the Alaska Peninsula. Vegetation across the region includes tundra, upland and lowland spruce hardwood forests, and shrub habitats. Freshwater habitats are abundant and diverse, and include headwater springs and streams, rivers, alpine and glacial lakes, spring-fed ponds, and tundra and floodplain wetlands.

Figure 2-1. The Bristol Bay Watershed, with the Togiak, Nushagak, Kvichak, Naknek, Egegik, and Ugashik Rivers and Their Watersheds



There are 25 Alaska Native villages and communities in the Bristol Bay watershed, with a total population of 7,475 in 2010 (Appendix E). The larger Bristol Bay area is home to 31 Federally Recognized Tribal Governments. The Bristol Bay economy is a mixed subsistence and cash economy. Most households use and share subsistence resources, and the great majority obtain most of their food resources from subsistence fishing, hunting, and gathering. Salmon account for a majority of the subsistence diet (Appendices D and E). Commercial fishing, with its limited season and close relationship to seasonal subsistence activities, is the primary cash economy (both the commercial and subsistence salmon economies are discussed in detail in Appendix E). Other cash economic sectors are related to recreational sport fishing and hunting, mineral exploration, and government.

The Nushagak River and Kvichak River watersheds, which account for more than half the area of the Bristol Bay watershed, represent complex mixtures of physiography, climate, geology, and hydrology, which interact to control the amount, distribution, and movement of water through these systems. Five distinct physiographic regions are represented by these watersheds (Wahrhaftig 1965): the Ahklun Mountains, the Southern Alaska Range, the Aleutian Range, the Nushagak-Big River Hills, and the Nushagak-Bristol Bay Lowland (Figure 2-2 and Table 2-1). Precipitation is greatest in the Southern Alaska Range, the Aleutian Range, and the Ahklun Mountains (Figures 2-2 and 2-3), and these regions serve as major water source areas for lower portions of the Nushagak and Kvichak River watersheds. Annual water balance in the mountains and hills is dominated by snowpack accumulation and subsequent melt, although late summer and fall rains are also important contributors to the hydrologic cycle, particularly in the Nushagak-Bristol Bay Lowland region.

Based on annual water surplus calculations (precipitation minus potential evapotranspiration), four climate classes (Feddema 2005) occur across these five physiographic regions (Table 2-2, Figures 2-2 and 2-3): very wet, wet, and moist classes experience an annual water surplus, whereas the dry class experiences an annual water deficit. Semi-arid and arid classes, which also experience an annual water deficit, are not found in this area. These combinations of physiographic region and climate class yield 17 different hydrologic landscapes within the Nushagak River and Kvichak River watersheds, representing the range of hydrologic characteristics across the area (Figure 2-4, Section 2.3.1).

2.2 Status and Condition of the Bristol Bay Region's Biological Resources and Alaska Native Cultures

The Bristol Bay watershed provides habitat for numerous animal species, including 35 fishes (Box 2-1), more than 190 birds, and more than 40 terrestrial mammals (Appendices A, B, and C). Many of these species are essential to the structure and function of the region's ecosystems and economies. The area supports world-class commercial and sport fisheries for Pacific salmon and resident fishes, in addition to other scenery and wildlife-based tourism. In this section, we examine the status and condition of key fish and wildlife populations across the Bristol Bay region, the economic value of those biological resources, and the Alaska Natives who depend on them.

Figure 2-2. Hydrologic Landscapes, as Defined by Physiographic Region and Climate Class within the Nushagak River and Kvichak River Watersheds. Physiographic regions (Wahrhaftig 1965) are classified as Ahklun Mountains, Southern Alaska Range, Aleutian Range, Nushagak-Big River Hills, and Nushagak-Bristol Bay Lowland; climate classes are defined as very wet, wet, moist, and dry. Climate classes (Feddema 2005) were calculated using Scenarios Network for Alaska and Arctic Planning (SNAP) data accessible at www.snap.uaf.edu.

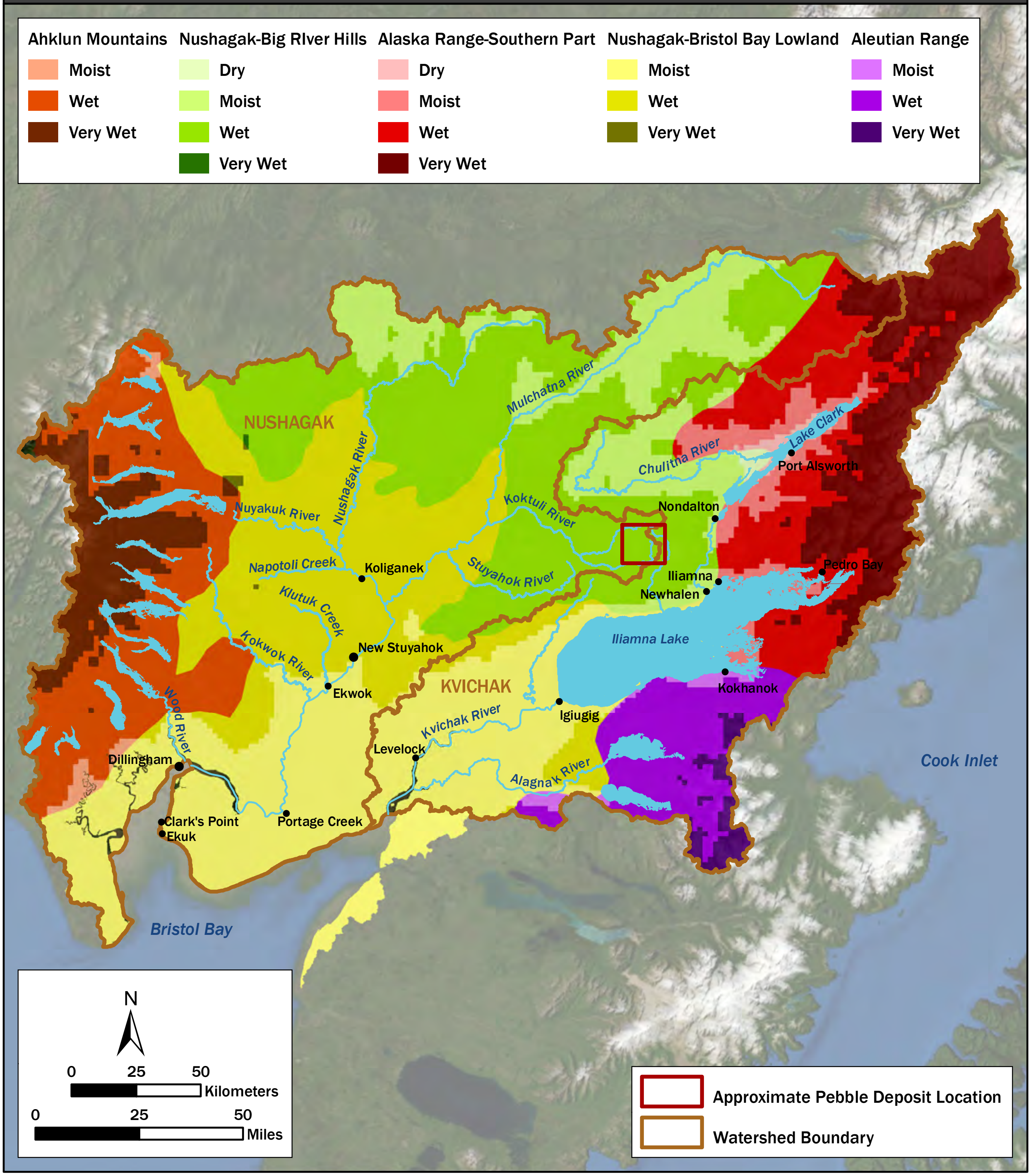


Figure 2-3. Distribution of Mean Annual Precipitation (mm) across the Nushagak River and Kvichak River Watersheds, 1971 to 2000. Values were calculated using Scenarios Network for Alaska and Arctic Planning (SNAP) data, accessible at www.snap.uaf.edu.

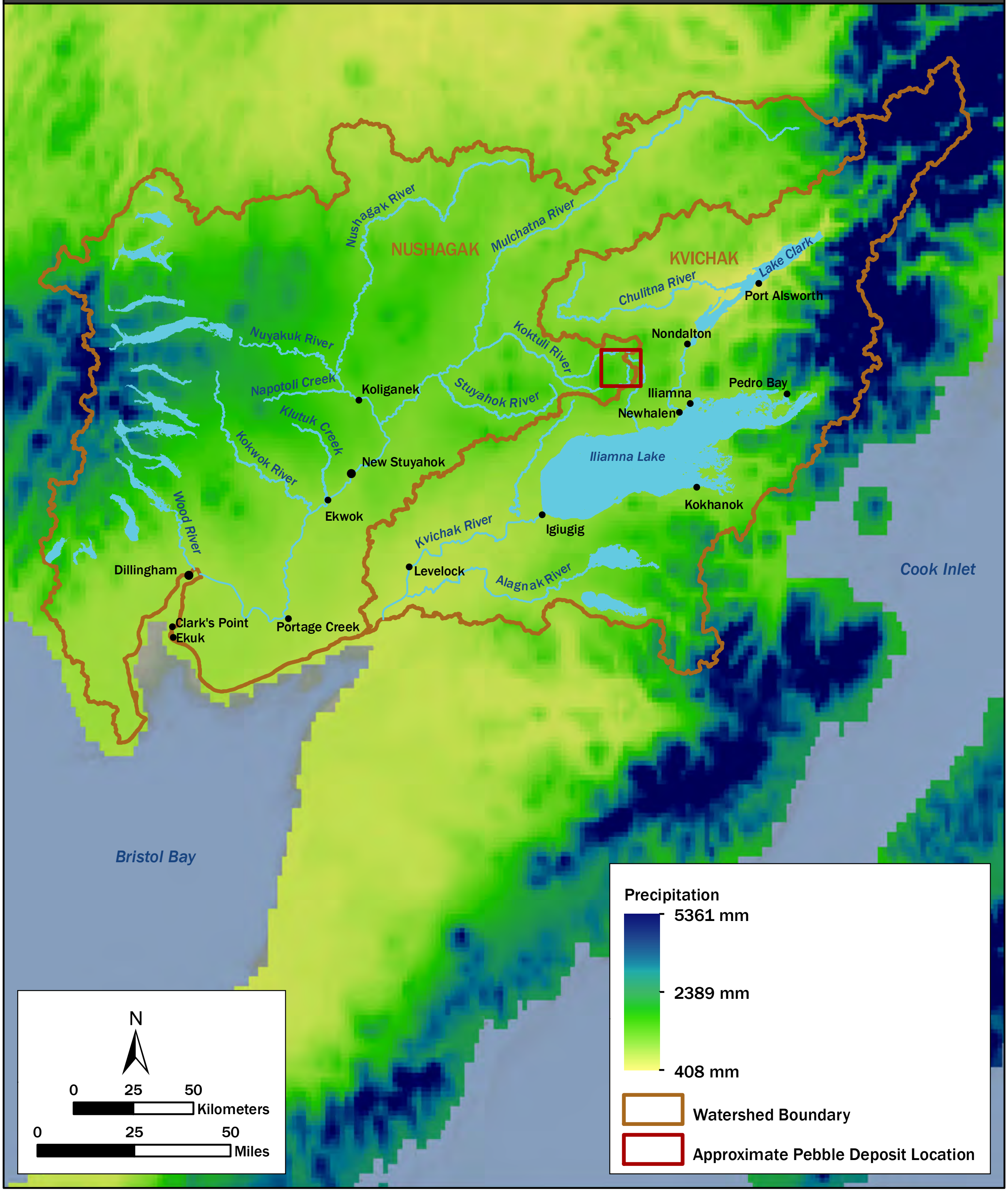


Table 2-1. Physiographic Regions (Wahrhaftig 1965) of the Nushagak River and Kvichak River Watersheds

Physiographic Region	Description	Elevation (meters)	Permafrost Extent	Freshwater Habitats
Ahklun Mountains	Sharp, steep glaciated mountains, separated by broad lowlands, with a few small glaciers in high mountain cirques	500–1,500	Sporadic	Incised streams in bedrock gorges; large glacial lakes in U-shaped valleys
Southern Alaska Range	Steep, glaciated mountains with land surfaces covered by rocky slopes, icefields, and glaciers	2,100–3,600	Unknown	Swift, braided streams and rivers with glacial headwaters; lakes in glaciated valleys
Aleutian Range	Rounded sedimentary ridges with common glacial features and active glaciers occurring on volcanoes	200–1,200 (intermittent volcanic peaks at 1,350–2,550)	Unknown	Streams that become braided upon reaching Nushagak-Bristol Bay Lowland; large lakes associated with ice-carved valleys and terminal moraines in northern part of region
Nushagak-Big River Hills	Rounded, flat-topped ridges with broad, gentle slopes and broad, flat or gently sloping valleys	450–750	Common	Glacial moraines and ponds in eastern part of region; braided and muddy rivers
Nushagak-Bristol Bay Lowland	Rolling landscape with low local topography and deep morainal and outwash deposits, but no glaciers	15–150	Sporadic or absent	Moraine and thaw lakes; large glacial lakes on southeast edge

Table 2-2. Distribution of Hydrologic Landscapes in the Nushagak River and Kvichak River Watersheds, as a Percentage of Entire Watershed Area

Physiographic Region	Ahklun Mountains			Southern Alaska Range				Aleutian Range			Nushagak-Big River Hills				Nushagak-Bristol Bay Lowland		
Climate class	V	W	M	V	W	M	D	V	W	M	V	W	M	D	V	W	M
Nushagak River watershed																	
Nushagak River (whole watershed)	7	16	1	1	2							25	9			24	15
Nushagak River at Ekwork [a]	4	9		2	3							40	14			27	1
Nuyakuk River	19	43	2									3			1	32	
Mulchatna River				4	7							53	22			14	
Nushagak River at Mulchatna River	8	18	1									30	9			35	
Koktuli River												99				1	
South Fork Koktuli River [b]												100					
North Fork Koktuli River [c]												100					
Kvichak River watershed																	
Kvichak River (whole watershed)				16	13	8	1	2	11	2		7	7			3	28
Kvichak River at Igiugig [d]				25	20	12	2			6		10	11	1			11
Kaskanak Creek near Igiugig [e]												21				28	50
Iliamna River near Pedro Bay [f]				94	6												
Upper Talarik Creek [g]												100					

Notes:

Climate classes are defined as very wet (V), wet (W), moist (M), and dry (D) according to Feddema (2005); no semi-arid or arid climates are found in the region.

[a] USGS gage 15302500.

[b] USGS gage 15302200.

[c] USGS gage 15302250.

[d] USGS gage 15300500.

[e] USGS gage 15302520.

[f] USGS gage 15300300.

[g] USGS gage 15300250.

Figure 2-4. Physiographic Regions of the Nushagak and Kvichak River Watersheds of Bristol Bay. The Nushagak and Kvichak River watersheds contain a wide range of aquatic habitats within five distinct physiographic regions (Wahrhaftig 1965) (see Figure 2-2). All photos taken between August 2003 and August 2010, courtesy of Michael Wiedmer.

Coastal plain south of the lower Nushagak River, Nushagak-Bristol Bay Lowland region

Nishlik Lake in the upper Nushagak River watershed, Ahklun Mountains region

Klutuk Creek in the lower Nushagak River watershed, western Nushagak-Bristol Bay Lowland region

Confluence of the Upper Nushagak River and the Nuyakuk River, Nushgak-Bristol Bay Lowland region

Source of the Mulchatna River, Southern Alaska Range region

Kvichak River immediately downstream of Iliamna Lake outlet, Nushagak-Bristol Bay Lowland region

Lake Clark, Southern Alaska Range region of the upper Kvichak River watershed

North Fork Swan River, Nushagak-Big River Hills region

BOX 2-1. SALMONID FISHES IN THE BRISTOL BAY WATERSHED

The Bristol Bay watershed's freshwater habitats support a diverse and robust assemblage of fishes, dominated by the family Salmonidae. This family comprises three subfamilies—salmon, trout, and char (Salmoninae), grayling (Thymallinae), and whitefish (Coregoninae)—all of which are represented in the region. In this assessment, we focus on fishes in the subfamily Salmoninae, particularly the five North American Pacific salmon species, rainbow trout, and Dolly Varden (a species of char). Collectively, we refer to these seven species as salmonids throughout this report.

All Salmonidae fishes spawn in freshwater, but they can differ in their life histories. Some populations (e.g., Bristol Bay's Pacific salmon) are anadromous, meaning that individual fish migrate to marine waters to feed and grow before returning to fresh water to reproduce. Other Bristol Bay populations (e.g., lake trout, Arctic grayling) are non-anadromous (resident), meaning that essentially all individuals remain in fresh waters to feed. Other populations (e.g., rainbow trout, Dolly Varden) can exhibit either anadromous or non-anadromous life histories. In this assessment, we consider non-anadromous or resident populations of rainbow trout and Dolly Varden.

Table 2-3. Life History, Habitat Characteristics, and Total Surveyed Occupied Stream Length for Bristol Bay's Five Pacific Salmon Species within the Nushagak River and Kvichak River Watersheds

Species	Freshwater Rearing Period (years)	Freshwater Rearing Habitat	Ocean Feeding Period (years)	Spawning Habitat	Surveyed Stream Length Occupied (kilometers)
Sockeye	0–3	Lakes, rivers	2–3	Beaches of lakes, streams connected to lakes, larger braided rivers	4,624
Coho	1–3	Headwater streams to moderate sized rivers, headwater springs, beaver ponds, side channels, sloughs	1+	Headwater streams to moderate sized rivers	5,860
Chinook	1+	Headwater streams to large-sized mainstem rivers	2–4	Headwater streams to large-sized mainstem rivers	4,788
Chum	0	None	2–4	Moderate-sized streams and rivers	3,435
Pink	0	None	1+	Moderate-sized streams and rivers, shallow rocky streams	2,155

Notes:
Data from ADFG 2011, Appendix A.

2.2.1 Pacific Salmon Populations

Five species of Pacific salmon spawn and rear in the Bristol Bay watershed's freshwater habitats: sockeye (*Oncorhynchus nerka*), coho (*O. kisutch*), Chinook or king (*O. tshawytscha*), chum (*O. keta*), and pink (*O. gorbuscha*). Confirmed salmon-producing watersheds—that is, watersheds where field reports have documented spawning or rearing salmon within their boundaries—make up more than 65% of the total area surveyed in the Nushagak River and Kvichak River watersheds (Figure 2-5). Because no hatchery fish are raised or released in the watershed, Bristol Bay's salmon populations are entirely wild. All of the species are anadromous, meaning that they at some point migrate to the ocean after hatching in freshwater, and then return to freshwater habitats to spawn. Adults return to their natal freshwater habitats to spawn (i.e., they exhibit homing behavior), and then die after spawning (i.e., they are semelparous). Sockeye, coho, and Chinook salmon spend a year or more rearing in freshwater before their ocean migration, and thus are more dependent on the quantity and quality of freshwater habitats than species such as pink and chum salmon, which migrate soon after hatching (Table 2-3). Freshwater habitats used for spawning and rearing vary across and within species, and include headwater streams, larger mainstem rivers, wetlands, and lakes (Table 2-3).

Sockeye is by far the most abundant salmon species in the Bristol Bay watershed (Table 2-4). The watershed supports the largest sockeye salmon fishery in the world, with approximately 46% of the average global abundance of wild sockeye salmon between 1956 and 2005 (Figure 2-6A) (Ruggerone et al. 2010). Bristol Bay was responsible for 63% of the nearly $8 billion landed value of the US sockeye salmon fishery from 1950 to 2008 (Schindler et al. 2010). Between 1990 and 2010, the annual average inshore run of sockeye salmon in Bristol Bay was approximately 37.5 million fish (ranging from a low of 16.8 million in 2002 to a high of 60.7 million in 1995) (Salomone et al. 2011). Annual commercial harvest of sockeye over this same period averaged 27.5 million (Table 2-4), translating to an average annual commercial value of $114.7 million for the Bristol Bay watershed's sockeye fishery (Section 2.2.4) (Salomone et al. 2011). The Bristol Bay region's salmon populations also support significant subsistence and recreational sport fisheries. For example, from 1990 to 2010, annual subsistence harvest averaged 140,767 salmon across all species, 78% of which were sockeye (Dye and Schwanke 2009, Salomone et al. 2011).

The Nushagak River also supports a large Chinook salmon fishery, and its commercial and sport fishing harvests are greater than those of all other Bristol Bay river systems combined (Table 2-4). Chinook returns to the Nushagak River are consistently greater than 100,000 fish per year, and have exceeded 200,000 fish per year in 11 years between 1966 and 2010 (Appendix A). This frequently places the Nushagak at or near the size of the world's largest Chinook runs, which is especially remarkable given its small watershed area compared to other Chinook-producing rivers such as the Yukon and Kuskokwim Rivers (Appendix A).

Figure 2-5. Salmon-Producing Watersheds in the Nushagak River and Kvichak River Watersheds.
A total of 568 subwatersheds (total area of 61,317 km^2) were assessed in the Nushagak River and Kvichak River watersheds. The percentage of this area in each category is shown in parentheses in the legend. Note that the southwestern portion of the Nushagak River watershed (i.e., the Nushagak Bay watershed) was not included in this analysis. Data from Demory et al. (1964), Nelson (1967), Salomone et al. (2009), Johnson and Blanche (2011), and ADFG (2012).

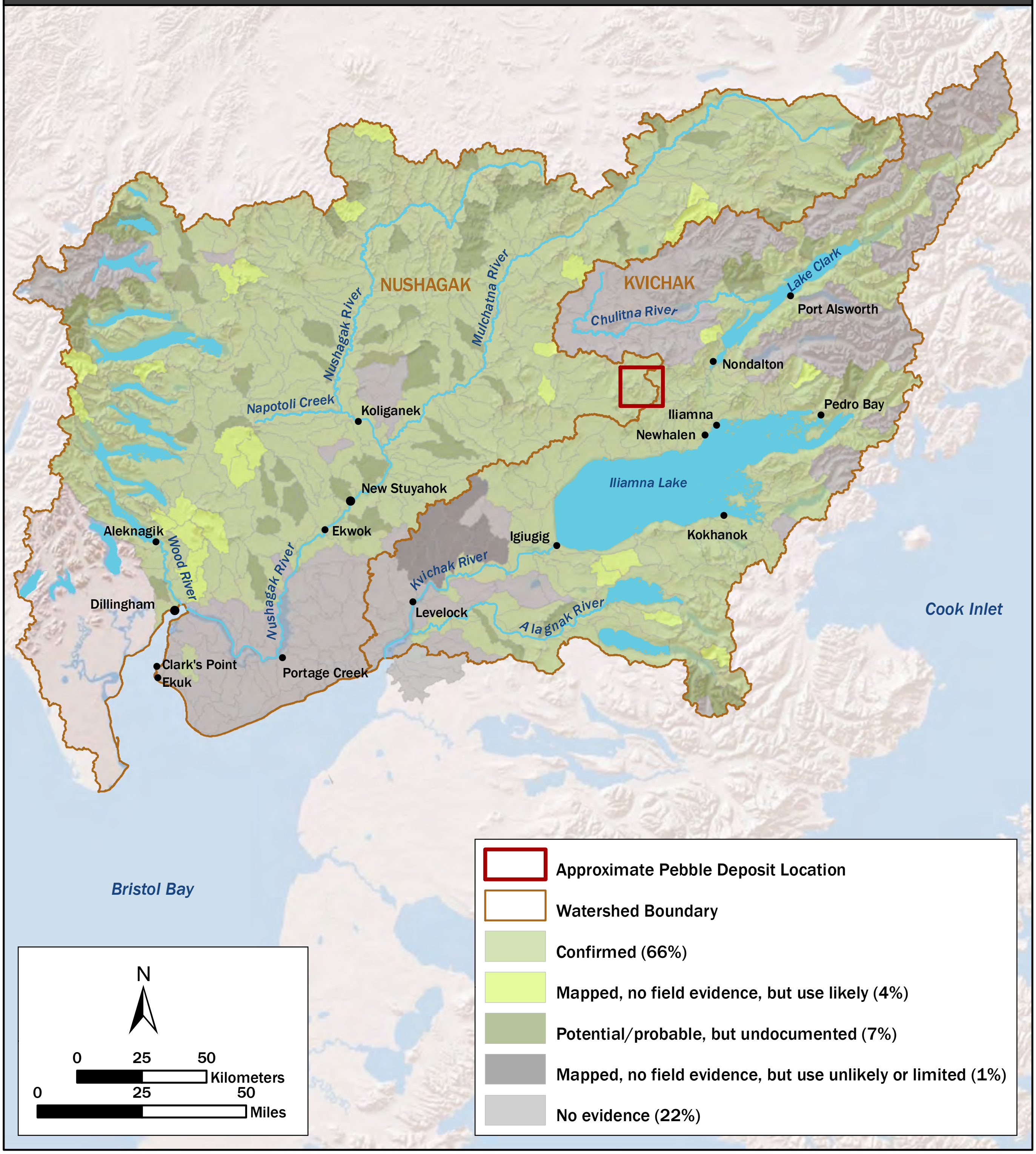


Figure 2-6. Average Annual Relative Abundance and Commercial Harvest of Wild Sockeye Salmon. A. Average annual relative abundance of wild sockeye salmon stocks in the North Pacific, 1956 to 2005; with the exception of Bristol Bay, stocks are ordered from west to east across the North Pacific, from Russia (Russia Mainland and Islands, West Kamchatka, East Kamchatka) to western North America (all other sites). B. Average annual relative commercial sockeye harvest in Bristol Bay watersheds, 1990 to 2009. Data from Ruggerone et al. (2010) and Appendix A.

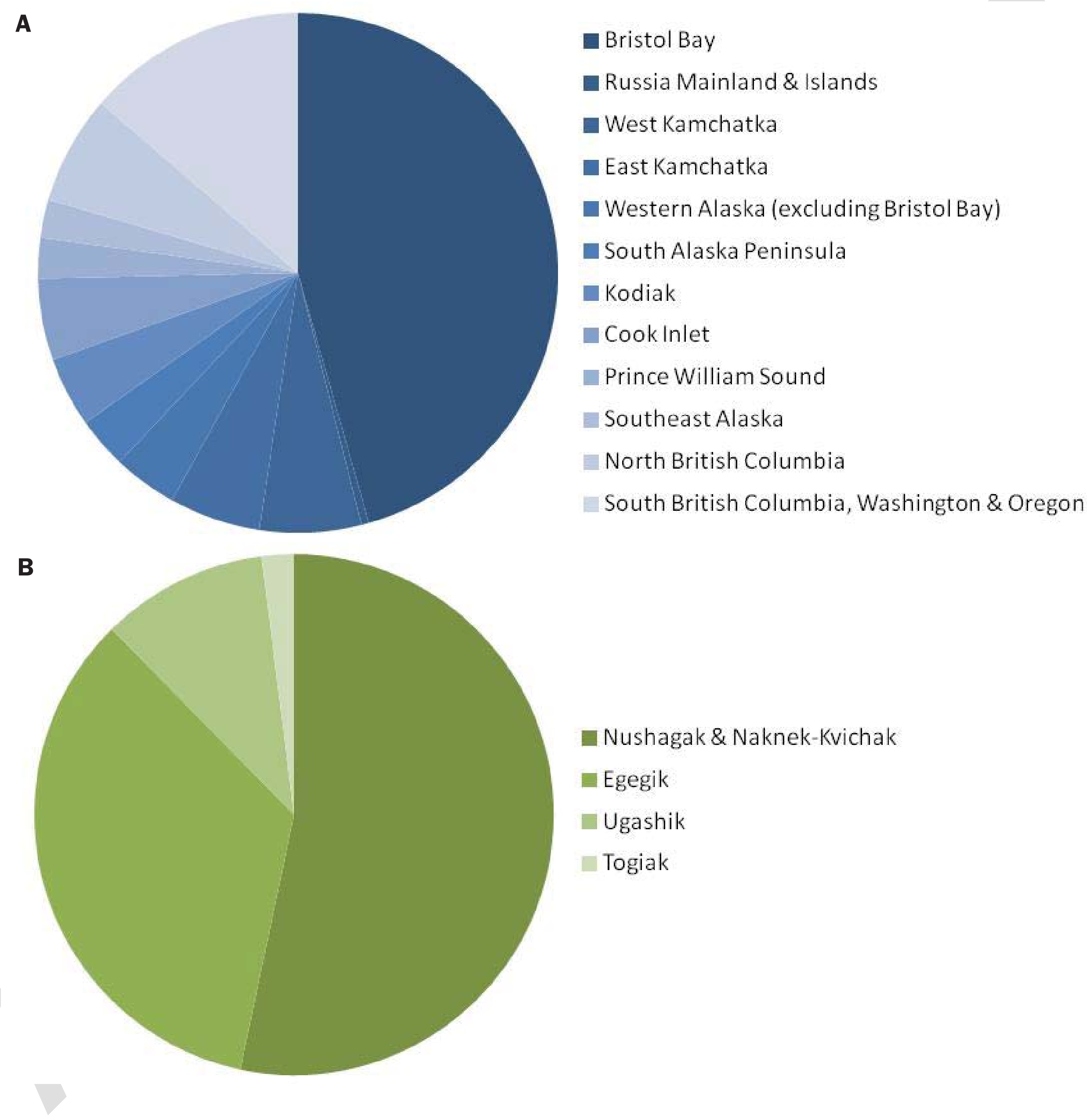


Table 2-4. Mean Annual Commercial Harvest (in Number of Fish) by Pacific Salmon Species and Bristol Bay Fishing District, 1990 to 2009

Salmon Species	Bristol Bay Fishing District					
	Naknek-Kvichak[a]	Egegik	Ugashik	Nushagak[a]	Togiak	Total
Sockeye	8,238,895	8,835,094	2,664,738	5,478,820	514,970	25,732,517
Chinook	2,816	849	1,402	52,624	8,803	66,494
Coho	4,436	27,433	10,425	27,754	14,234	84,282
Chum	184,399	78,183	70,240	493,574	158,879	985,275
Pink[b]	73,661	1,489	138	50,448	43,446	169,182

Notes:
[a] Naknek-Kvichak district includes the Alagnak River; Nushagak district includes the Wood and Igushik Rivers.
[b] Pink salmon data are from even-numbered years; harvest is negligible during odd-year runs.
Data from Appendix A.

2.2.2 Resident Fish Populations

In addition to the five Pacific salmon species discussed in Section 2.2.1, the Bristol Bay watershed supports populations of resident fishes, those that typically (but not always) remain within the watershed's freshwater habitats throughout their lifecycles. The region contains highly productive waters for such sport fish species as rainbow trout (*Oncorhynchus mykiss*), Dolly Varden (*Salvelinus malma*), Arctic char (*Salvelinus alpinus*), Arctic grayling (*Thymallus arcticus*), and lake trout (*Salvelinus namaycush*) (Dye and Schwanke 2009). These fish species occupy a variety of habitats in the watershed (Table 2-5), from headwater streams to large rivers and lakes. The Bristol Bay region is especially renowned for the abundance and size of its rainbow trout. Between 2003 and 2007, an estimated 196,825 rainbow trout were caught in the Bristol Bay Sport Fish Management Area (Table 2-5).

Table 2-5. Typical Habitats Occupied and the Number Caught and Harvested Listed by Common Fish Species of the Bristol Bay Watershed. Harvest represents a subset of catch, with harvested fish being removed from the system as opposed to caught and released back into the system.

Species	Habitat	Catch	Harvest
Rainbow trout *(Oncorhynchus mykiss)*	Medium-large streams and rivers, lakes	196,825[a]	1,762[a]
Arctic grayling *(Thymallus arcticus)*	Lakes, slow-flowing streams (not steep headwaters)	> 80,000[b]	1,711[a]
Dolly Varden char *(Salvelinus malma)*	Fast-flowing headwater and low order streams, upland lakes	NA	3,435[a,c]
Arctic char (*Salvelinus alpinus*)	Lakes, inlet streams		
Lake trout (*Salvelinus namaycush*)	Lakes, inlet/outlet streams	17,000[b]	NA

Notes:
[a] Estimated average annual harvest (2003–2007) in the Bristol Bay Sport Fish Management Area
[b] 2004 catch in Bristol Bay (Jennings et al. 2007)
[c] Dolly Varden and Arctic char harvest combined (data not separated by species)
NA = data not available

2.2.3 Wildlife Populations

Unlike most terrestrial ecosystems, the Bristol Bay watershed has undergone little development and remains largely intact. Thus, it still supports its historical complement of species, including large carnivores such as brown bears (*Ursus arctos*), bald eagles (*Haliaeetus leucocephalus*), and gray wolves (*Canis lupus*); ungulates such as moose (*Alces alces gigas*) and caribou (*Rangifer tarandus granti*); and numerous waterfowl species. Wildlife populations tend to be relatively large in the region, due to the increased productivity associated with Pacific salmon runs (Section 2.3.4). In many cases, little abundance data specific to the Bristol Bay watershed are available, but it is reasonable to assume that species distribution and abundance patterns in this region mirror those observed in similar habitats across southwestern Alaska.

Brown bear density estimates across portions of the Nushagak River and Kvichak River watersheds range from roughly 40 bears per 1,000 km^2 in the northern Bristol Bay region (Togiak National Wildlife Refuge and the Bureau of Land Management's Goodnews Block) (Walsh et al. 2010) to 150 bears per

1,000 km^2 along the shore of Lake Clark (Appendix C). From July 2006 to July 2007, 621 brown bears were reported harvested from the Alaska Department of Fish and Game's (ADFG's) Game Management Unit (GMU) 9, which includes the Kvichak River watershed and the Alaska Peninsula. Brown bears are not as abundant in the Nushagak River watershed as the Kvichak River watershed, and densities in both watersheds are lower than on the Alaska Peninsula's Pacific coast, which is home to the highest documented brown bear density in North America (551 bears per 1,000 km^2) (Miller et al. 1997).

Although no comprehensive survey of bald eagles or bald eagle nests has been conducted in the Bristol Bay watershed, limited count data are available for parts of the region. For example, 50 bald eagle nests were recorded along portions of the Nushagak, Mulchatna, and Kvichak Rivers in 2006; approximately half of those nests were categorized as active (Appendix C). The U.S. Fish and Wildlife Service's Bald Eagle Nest Database contains approximately 230 nest records for the Nushagak River and Kvichak River watersheds, with 169 of those records collected between 2003 and 2006 (Appendix C).

Gray wolf populations have not been well-studied in the Bristol Bay region, and it is difficult to assess population numbers. Wolves are currently thought to be abundant in the Nushagak River watershed; between 2003 and 2008, reported annual wolf harvest ranged from 60 to 141 in GMU 17, which includes the Nushagak and Togiak River watersheds. In the Kvichak River watershed, numbers are believed to be lower, although populations have increased since the 1990s (Butler 2009).

Moose and caribou are abundant in the Bristol Bay watershed. Moose abundance in the Nushagak River and Kvichak River watersheds was estimated at 8,100 to 9,500 in 2004 (Butler 2004, Woolington 2004). Populations are especially high in the Nushagak River watershed (ADFG 2011), where felt-leaf willow, a preferred plant species, is abundant (Bartz and Naiman 2005). The Nushagak River and Kvichak River watersheds are used primarily by the Mulchatna caribou herd (one of 31 caribou herds found in Alaska), which numbered roughly 200,000 in 1997 but had decreased to roughly 30,000 by 2008 (Valkenburg et al. 2003, Woolington 2009). The Mulchatna herd ranges widely through the Nushagak River and Kvichak River watersheds, but also spends considerable time in other watersheds.

Moose and caribou are significant subsistence food sources: a survey of Bristol Bay residents found that 86% and 88% of respondents has consumed moose and caribou meat, respectively, in the past year (Ballew et al. 2004). Between 1983 and 2006, moose harvest in GMU 17 increased from 127 to 380 per year; the upper Nushagak River watershed alone (GMU 17B) has a mean annual harvest of 149 moose (Appendix C). Caribou harvest ranged from 1,573 to 4,770 per year between 1991 and 1999, but this estimate is for the entire Mulchatna herd, including those taken outside of the Nushagak River and Kvichak River watersheds (Valkenburg et al. 2003).

More than 30 waterfowl species regularly occur in the Bristol Bay watershed, including ducks (e.g., northern pintail, scaup, mallard, and green-winged teal), geese (e.g., white-fronted, Canada), swans, and sandhill cranes. The region serves as an important staging area for many species, including emperor geese and Pacific brants, during spring and fall migrations and ducks are abundant. The Alaska Yukon Waterfowl Breeding Population Survey found average late May abundance indices of 497,000 ducks, 7,700 geese, 15,400 swans, and 5,300 sandhill cranes in the Bristol Bay Lowlands between 2002 and 2011 (Appendix C).

Although this assessment focuses on freshwater habitats of the Bristol Bay watershed, it should be noted that once the region's Pacific salmon populations migrate to the ocean, they also provide food for marine predators (Appendix F). Marine mammals such as northern fur seals, harbor seals, and stellar sea lions are known to feed on Pacific salmon. These interactions also can be important in freshwater habitats, as one of two freshwater harbor seal populations in North America is found in Iliamna Lake (Smith et al. 1996).

2.2.4 The Economics of Bristol Bay's Biological Resources

The Bristol Bay watershed supports several sustainable, wilderness-compatible economic sectors, including commercial fishing, subsistence use, sport fishing, recreational hunting, and wildlife viewing and other non-consumptive recreation. Each of these sectors generates expenditures or sales that drive the region's economy, generating more than $479 million (in 2009 dollars) in total direct annual economic benefit (Table 2-6).

Table 2-6. Summary of Regional Economic Expenditures Based on Salmon Ecosystem Services. Values are regional expenditures in different economic sectors, expressed in 2009 dollars. Note that estimates of certain year-specific total harvest and sales values vary slightly throughout this report, due to differences in how data were aggregated and reported. See Appendix E for additional information on these values.

Economic Sector	Estimated Direct Expenditure (sales per year, in $ millions)
Commercial fisheries, wholesale value	300.2
Sport fisheries	60.5
Sport hunting	8.2
Wildlife viewing / tourism	104.4
Subsistence harvest	6.3
TOTAL	479.6

The Bristol Bay commercial salmon fishery currently provides the region's greatest source of economic activity. From 2000 through 2010, the annual commercial salmon catch averaged 23 million fish (170 million pounds). The average annual commercial value of all Bristol Bay salmon fisheries from 1990 to 2010 totaled $116.7 million, $114.7 million of which resulted from the sockeye harvest (Salomone et al. 2011). Thus, sockeye salmon represent the principal species of economic value throughout the Bristol Bay region.

In 2009, fishermen received $144 million for their catch, and fish processors received approximately $300 million, which is referred to as the first wholesale value of the fish (Table 2-6, Appendix E). The commercial salmon fishery, which is largely centered in the region's salt waters rather than its freshwater streams and rivers, is closely managed for sustainability using a permit system. Approximately 26% of permit holders are Bristol Bay residents. The commercial fishery also provides significant employment opportunities, directly employing over 11,000 full- and part-time workers at the season's peak.

The uncrowded wilderness setting of the Bristol Bay watershed attracts recreational fishermen. Sport fishing in Bristol Bay accounts for approximately $60.5 million dollars in annual spending (Table 2-6), $58 million of which is spent in the Bristol Bay region. In 2009, approximately 29,000 sport fishing trips were taken to the Bristol Bay region (12,000 trips by people living outside of Alaska, 4,000 trips by Alaskans living outside the Bristol Bay area, and 13,000 trips by Bristol Bay residents). These sport fishing activities directly employ over 800 full and part-time workers; in 2010, 72 businesses and 319 guides were operating in the Nushagak River and Kvichak River watersheds alone (Appendix A).

Sport hunting for caribou, moose, brown bear, and other species also plays a role in the local economy of the Bristol Bay region. In recent years approximately 1,323 non-residents and 1,319 non-local residents of Alaska traveled to the region to hunt. Miller and McCollum (1994) estimate that non-residents and non-local residents spend approximately $5,170 and $1,319 per trip (values updated to 2009 dollars), respectively. These hunting activities result in an estimated $8.2 million per year in direct hunting-related expenditures (Table 2-6) and directly employ over 100 full- and part-time workers.

Many households participate in the subsistence harvest of fish, which generates regional economic benefits when Alaskan households spend money on subsistence-related supplies. In total, individuals in Bristol Bay communities harvest about 2.6 million pounds of subsistence harvest per year. In 2010, the U.S. Census Bureau reported an estimated 1,873 Alaska Native and 666 non-native households in the Bristol Bay Region. Goldsmith et al. (1998) estimated that Alaska Native households spend an average of $3,054 on subsistence harvest supplies; non-native households spend an estimated $796 on supplies (values updated to 2009 price levels). Based on these estimates, subsistence harvest activities resulted in expenditures of approximately $6.3 million (Table 2-6).

It is important to note that these estimates of expenditures reflect only the annual economic activity generated by these activities. It may be useful to consider calculations such as net economic value, or the value of the resource or activity over and above regular expenditures associated with it. These types of calculations, as well as the regional economic significance of Bristol Bay's salmon fishery, are discussed in Appendix E.

2.2.5 Alaska Native Cultures

Fourteen of Bristol Bay's 25 Alaska Native villages and communities are within the Nushagak River and Kvichak River watersheds, with a total population of 4,337 in 2010 (U.S. Census Bureau 2010). Population in the region grew substantially from 1980 to 2000, and remained relatively stable from 2000 to 2010 (Appendix D). Dillingham (population 2,329) is the largest community; other communities range in size from 2 residents (Portage Creek) to 510 residents (New Stuyahok). Because population in some communities is seasonal, these numbers increase during the subsistence fishing season. In all but one of these 14 villages, Alaska Natives were the population majority in 2010. There are 13 Federally Recognized Tribal Governments in the 14 villages.

The Alaska Native cultures present in the Kvichak River and Nushagak River watersheds—the Yup'ik and Dena'ina—are part of the last intact, sustainable salmon-based cultures in the United States. This is especially significant as other Pacific Northwest salmon-based cultures struggle with degraded resources. Cultures associated with salmon fishing appeared in these watersheds as early as 2000 BC and intensified around AD 1000 (Appendix D). Currently, the percentage of Alaska Native population in the region's villages ranges from 21.4% (Port Alsworth) to 95.7% (Koliganek) (Appendix D), and the Yup'ik and Dena'ina cultures still provides framework and values for everyday life. Among the Yup'ik, over 40% of the population continues to maintain their native language, one of the highest percentages among Alaska Native cultures in the United States (Appendix D).

Salmon are integral to the entire way of life in Yup'ik and Dena'ina cultures. Traditional and more modern spiritual practices place salmon in a position of respect and importance, as exemplified by the First Salmon Ceremony and the Great Blessing of the Waters (Appendix D). The salmon harvest provides a basis for many important cultural and social practices and values, including the sharing of resources among the people, fish camp, gender and age roles and the perception of wealth. While a small minority of Tribal Elders and culture bearers interviewed expressed a desire to bring in more market economy opportunities, most equated wealth with stored and shared subsistence foods (Appendix D).

Salmon as subsistence food and as the basis for Alaska Native cultures are inseparable, and the characteristics of these subsistence-based salmon cultures have been widely documented (Appendix D). The cultures have a strong connection to the landscape and its resources; in the Bristol Bay watershed, this connection has been maintained for centuries by the uniquely pristine condition of the region's landscape and resources. In turn, the respect and importance given salmon and other wildlife, along with the traditional knowledge of the environment, has produced a sustainable subsistence-based economy (Appendix D). This subsistence-based way of life is a key element of Alaska Native identity and it serves a wide range of economic, social, and cultural functions in Yup'ik and Dena'ina societies (Appendix D).

Alaska Native populations have managed to maintain continual access to a range of subsistence foods, and subsistence uses on these watershed's state lands are given priority by state law and regulations (i.e., the 1978 State of Alaska Subsistence Act). According to ADFG statistics, subsistence accounts for an average of 80% of protein consumed by area residents; in 2004 and 2005, annual subsistence consumption rates were over 300 pounds per person in many of the villages, and reached as high as 900 pounds per person (Appendix D). Percentage of salmon harvest in relation to all subsistence resources ranges from 29 to 82% in the villages (Appendix D). There is also a strong link between subsistence and the market economy (largely commercial fishing and recreation) in the area. Goods and services (e.g., boats, rifles, nets, snow mobiles, and fuel) are purchased by households and used for subsistence activities (Appendix E), and the market economy provides seasonal employment for residents, allowing them to participate year-round in subsistence activities. Continued access to high-quality subsistence resources is necessary for survival of the Alaska Natives and other local residents, because no alternative food sources are economically viable. Both federal and state legislation recognize the importance of salmon and other wild food resources and have designated subsistence as a priority for Alaska Natives (Appendix D).

Boraas and Knott (Appendix D) state:

> "… salmon and clean water are foundational to the Yup'ik and Dena'ina cultures in the Nushagak and Kvichak watersheds. The people in this region not only rely on salmon for a large proportion of their highly nutritional food resources; salmon is also integral to the language, spirituality, and social relationships of the culture. Because of this interconnection, the cultural viability, as well as the health and welfare of the local population, are extremely vulnerable to a loss of either quality or quantity of salmon resources."

2.3 Factors Contributing to Status and Condition of Resources

The exceptional quality of Bristol Bay's fish populations and their importance to the region's wildlife and Alaska Natives results from five key, interrelated characteristics of the Bristol Bay watershed: (1) the quantity, quality, and diversity of aquatic habitats found in the watershed; (2) the importance of groundwater inputs and flow stability in shaping these habitats; (3) the high level of biological complexity that these diverse habitats support; (4) the increased ecosystem productivity associated with anadromous salmon runs; and (5) the environmental integrity of the watershed's ecosystems.

2.3.1 Quantity, Quality, and Diversity of Aquatic Habitats

Differences in hydrology, geology, and climate across the Bristol Bay watershed interact to create the region's diverse hydrologic landscapes (Figure 2-2 and Table 2-2), ultimately shaping the quantity, quality, diversity, and distribution of aquatic habitats throughout the watershed (Figure 2-4) and determining their suitability for Pacific salmon. In general, conditions within the Bristol Bay watershed are highly favorable for Pacific salmon. Aquatic habitats are abundant and diverse, ranging from headwater streams to braided rivers, large lakes to wetlands, side channels to off-channel alcoves. The Bristol Bay watershed includes more than 90,000 km of streams and hundreds of km^2 of wetlands. The Nushagak River and Kvichak River watersheds contain over 58,000 km of streams; 13% of this total stream length has been documented as anadromous fish habitat, although this is likely a significant underestimate (Appendix A). The region's aquatic habitats provide a diverse assemblage of salmon spawning and rearing habitats, thereby supporting a diverse salmonid assemblage (Section 2.3.3). Gravel substrates—common throughout the region (Section 2.3.2)—are essential for Pacific salmon spawning, egg incubation, and early development (Appendix A).

Lakes are key spawning and rearing areas for sockeye salmon, and they cover relatively high percentages of watershed area in the Bristol Bay region: 7.9% for the entire Bristol Bay watershed area and 13.7% for the Kvichak River watershed (Luck et al. 2010). In other North Pacific river systems supporting sockeye salmon populations, from northern Russia to western North America, these values tend to be much lower (e.g., 0.2 to 2.9%) (Luck et al. 2010). Relatively low watershed elevations (especially in the extensive Nushagak-Bristol Bay Lowland region) and the absence of artificial barriers to migration (e.g., dams and roads; Section 2.3.5) mean that not only are streams, lakes and other aquatic habitats abundant in the Bristol Bay region, they tend to be accessible. With exception of Chikuminuk Lake, all major lakes within the watershed are accessible to anadromous salmon (Appendix A). Lakes and ponds also play a key role in groundwater dynamics and flow stability (Section 2.3.2).

Overall physical habitat complexity in the Bristol Bay watershed is higher than in many other systems supporting sockeye salmon populations. Of 1,509 North Pacific Rim watersheds, the Kvichak, Wood, and Nushagak (exclusive of Wood) Rivers ranked third, fourth, and forty-fourth, respectively, in physical habitat complexity, based on an index that included variables such as lake coverage, stream junction density, floodplain elevation and density, and human footprint (Luck et al. 2010).

2.3.2 Groundwater Exchange and Flow Stability

A key aspect of the Bristol Bay region's aquatic habitats is the importance of groundwater exchange. Because salmon rely on clean, cold water flowing over and through porous gravels for spawning, egg incubation, and rearing (Bjornn and Reiser 1991), areas of groundwater upwelling create high-quality salmon habitat (Appendix A). For example, densities of beach spawning sockeye salmon in the Wood River watershed were highest at sites with strong groundwater upwelling, and zero at sites with no upwelling (Burgner 1991). Densities of salmon-supporting streams tend to be lower in regions with lower permeability and less extensive exchange between groundwater and surface water (Johnson and Blanche 2011, ADFG 2012).

Portions of the Nushagak-Bristol Bay Lowland and Nushagak-Big River Hills physiographic regions, including the Pebble deposit area, contain coarse-textured glacial drift with abundant, high permeability gravels and extensive connectivity between surface waters and groundwater. Abundant wetlands and small ponds also contribute disproportionately to groundwater recharge (Rains 2011). This tight connection between groundwater and surface waters helps to moderate water temperatures and streamflows. For example, groundwater contributions that maintain water temperatures above 0°C are critical for maintaining winter refugia in streams that might otherwise freeze (Power et al. 1999).

These groundwater contributions to streamflow also support flows in the region's streams and rivers that are more stable than those typically observed in many other salmon streams (e.g., in the Pacific Northwest or southeastern Alaska). The lower mainstem Nushagak and Kvichak Rivers illustrate this tendency toward moderated, consistent streamflows (Figure 2-7). Coarse-textured glacial drift in the Kaskanek and Upper Talarik Creek drainages promotes high groundwater contributions to these streams, resulting in stable flows through much of the year (Figure 2-7A). High baseflows in the Nushagak River also are consistent with increased interactions between surface water and groundwater, as water flows from the Southern Alaska Range, Ahklun Mountains, and Nushagak-Big River Hills into the coarse-textured glacial drift of Nushagak-Bristol Bay Lowlands (Figure 2-7B).

Streamflow storage in upstream lakes plays a role in flow stabilization, as well. In the Kvichak watershed, Iliamna Lake dampens high flows from the Iliamna and Newhalen Rivers before they reach the mainstem. The effect of upstream lakes on flow storage is also evident in the Newhalen River, located downstream of Lake Clark (Figure 2-5). In the Nushagak watershed, large lakes occur in the Ahklun Mountain headwaters, and their moderating influence can be seen in the Nuyakuk River (Figure 2-5).

2.3.3 Biological Complexity

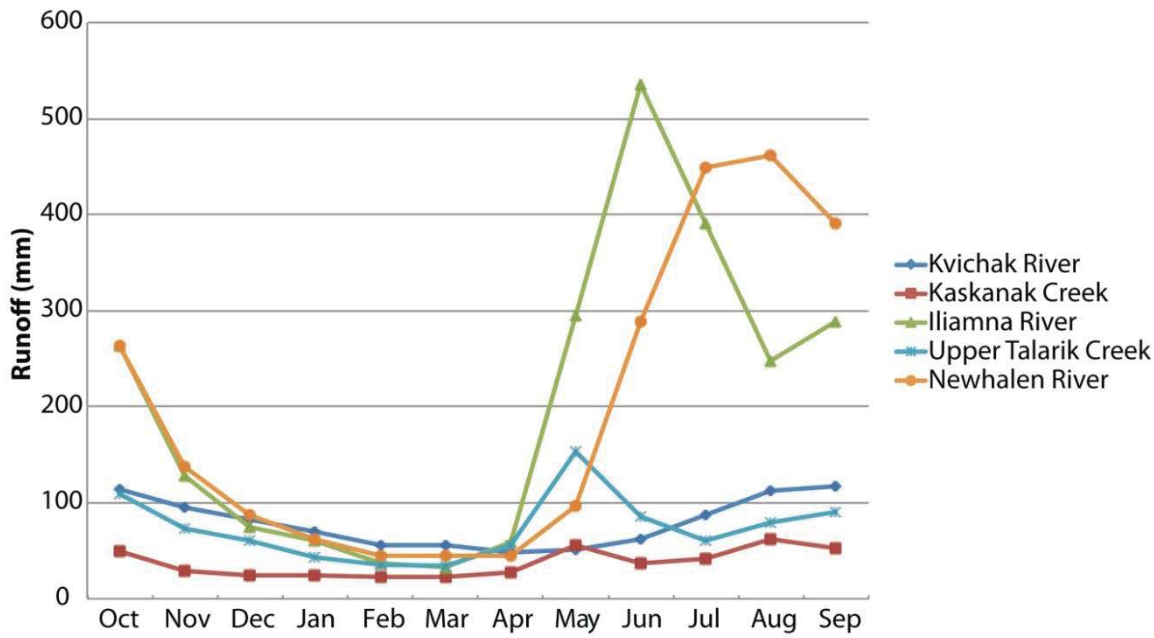
Closely tied to the Bristol Bay region's physical habitat complexity is its biological complexity, which—operating at multiple scales and across multiple species—greatly increases the region's ecological productivity and stability. This biological complexity is especially evident in the watershed's Pacific salmon populations, although other species (e.g., rainbow trout) also show considerable biological variability. As discussed in Section 2.2.1, the five Pacific salmon species found in Bristol Bay vary in many life history characteristics (Table 2-3), allowing them to fully exploit the range of habitats available. Even within a single species, life histories can vary significantly. For example, sockeye salmon may spend anywhere from 0 to 3 years rearing in freshwater habitats, then return to the Bristol Bay watershed anytime within a 4-month window (Table 2-7).

This life history variability, together with the Pacific salmon's homing behavior, results in distinct populations adapted to their own specific spawning and rearing habitats (Hilborn et al. 2003). Variations in temperature and streamflow associated with seasonality and groundwater-surface water interactions create a habitat mosaic supporting a range of spawning times across the watersheds. Spawning adults return at different times, to different locations, creating and maintaining a degree of reproductive isolation and allowing development of genetically distinct stocks (Hilborn et al. 2003, McGlauflin et al. 2011). The Bristol Bay watershed's sockeye salmon "population" is actually a sockeye salmon stock complex, or a combination of hundreds of genetically distinct populations, each adapted to specific, localized environmental conditions (Hilborn et al. 2003, Schindler et al. 2010). This stock complex structure acts to stabilize salmon productivity across the watershed as a whole, as the relative contribution of sockeye with different life history characteristics, from different regions of the Bristol Bay watershed, changes over time in response to changes in environmental conditions (Hilborn et al. 2003). For example, salmon stocks that spawn in small streams may be negatively affected by low-flow conditions, whereas stocks that spawn in lakes may not be affected (Hilborn et al. 2003). Thus, any population containing stocks that vary in spawning habitat is better able to persist as environmental conditions change.

Without this high level of system-wide biocomplexity, annual variability in the size of Bristol Bay's sockeye salmon runs would more than double and fishery closures would be more frequent (Schindler et al. 2010). In other watersheds with previously robust salmon fisheries, such as the Sacramento River's Chinook fishery, losses of biocomplexity have contributed to salmon population declines (Lindley et al. 2009). These findings suggest that even the loss of a small stock within an entire watershed's salmon population may have more significant effects than expected, due to associated decreases in biocomplexity of the population's stock complex.

Figure 2-7. Mean Monthly Runoff for Selected Streams and Rivers in the Kvichak River and Nushagak River Watersheds. USGS gages and dates used to generate each line: A. Kvichak River Watershed: Kvichak River (15300500, Aug 1967-Sep 1987); Kaskanak Creek (15300520, Jun 2008-Sep 2011); Iliamna River (15300300, Jun 1996-Sep 2010); Upper Talarik Creek (15300250, Sep 2004-Sep 2010); Newhalen River (15300000, Jul 1951-Sep 1986); B. Nushagak River Watershed: Nushagak River (15302500, Oct 1977-Sep 1993); Nuyakuk River (15302000, Jun 1953-Sep 2010); North Fork Koktuli River (15302250, Sep 2004-Sep 2010); South Fork Koktuli River (15302200, Sep 2004-Sep 2010).

A



B

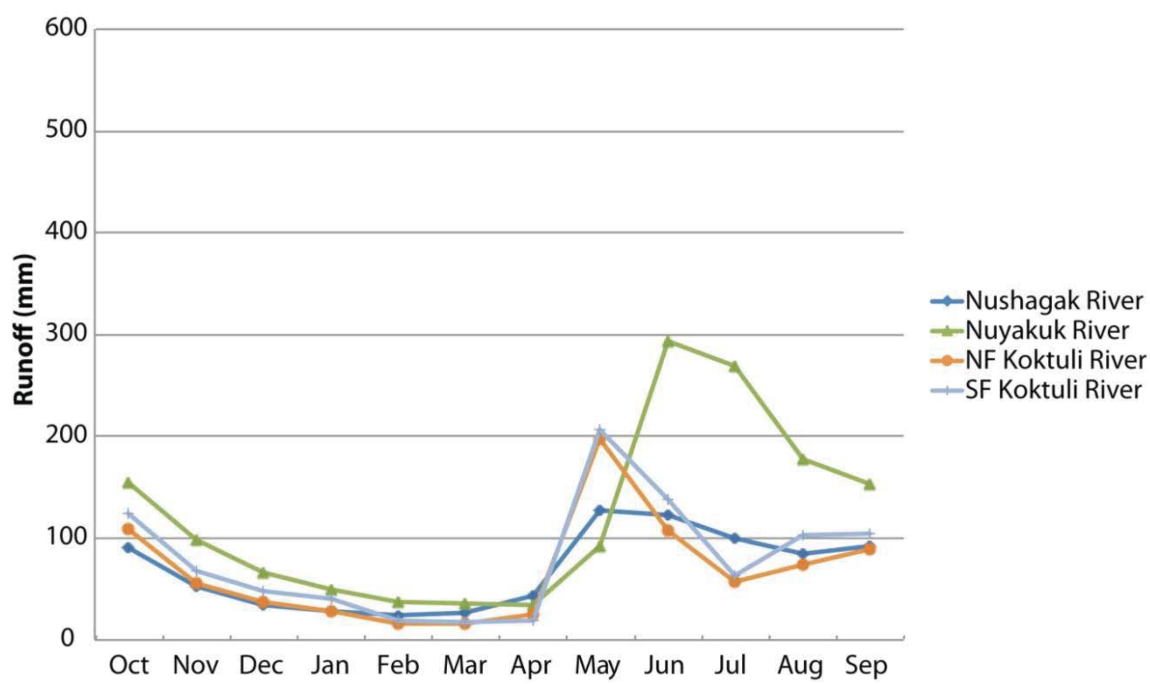


Table 2-7. Life History Variation within the Bristol Bay Sockeye Salmon Populations

Element of Biocomplexity	Range of Traits or Options
Location within the Bristol Bay watershed	7 major sub-watersheds, ranging from maritime-influenced systems on the Alaska Peninsula to more continental systems
Time of adult return to freshwater	June–September
Time of spawning	July–November
Spawning habitat	Major rivers, small streams, spring-fed ponds, mainland beaches, island beaches
Body size of adults	130 to 190-mm body depth at 450-mm male length
Body shape of adults	Sleek, fusiform to very deep-bodied, with exaggerated humps and jaws
Egg size	88–116 mg at 450 mm female length
Time between entry into spawning habitat and death	Days–weeks
Time spent rearing in freshwater	0–3 years
Time spent at sea	1–4 years
Source: Hilborn et al. 2003.	

2.3.4 Salmon-Derived Productivity

Most of the nitrogen, phosphorus and other elements in adult salmon bodies are derived from the marine environment (Larkin and Slaney 1997, Schindler et al. 2005). Adult salmon returning to their natal freshwater habitats import nutrients that they obtained during their ocean feeding period—that is, marine-derived nutrients (MDN)—back into those habitats. MDN from salmon accounts for a significant portion of nutrient budgets in the Bristol Bay watershed. For example, sockeye salmon are estimated to import approximately 12,700 kg of phosphorus and 101,000 kg of nitrogen into the Wood River system annually, and 50,200 kg of phosphorus and 397,000 kg of nitrogen into the Kvichak River system annually (Moore and Schindler 2004). Across the Kvichak River and Nushagak River, returns of 30 million to 40 million salmon each year import up to 20 million kg of nutrients into these watersheds (Appendix C). Returning salmon also redistribute nutrients within these systems by disturbing bottom substrates during spawning and increasing nutrient export downstream (Moore et al. 2007).

Productivity of the Bristol Bay region's fish and wildlife species is highly dependent on this influx of MDN into the region's freshwater habitats. When available, salmon-derived resources—in the form of live adult salmon, eggs, carcasses, and invertebrates that feed upon carcasses—are key dietary components for numerous animal species, including fishes (e.g., rainbow trout, Dolly Varden, Pacific salmon, Arctic grayling), mammals (brown bears, wolves, foxes, minks), and birds (bald eagles, waterfowl) (Appendices A and C). Availability and consumption of salmon-derived resources can have significant benefits for these species, including increased growth rates, energy storage, litter size, nesting success, and population density (Appendices A and C). The abundance of trophy-sized rainbow trout in the Bristol Bay system results from MDN from salmon. Terrestrial systems of the Bristol Bay watershed also benefit from these MDN. Bears, wolves, and other wildlife transport carcasses and excrete wastes throughout their ranges (Darimont et al. 2003, Helfield and Naiman 2006), which provide food and nutrients for other terrestrial species.

Finally, by dying in the streams where they spawn, adult salmon subsidize the next generation by adding their nutrients to the ecosystem that will feed their young. This positive feedback is missing from freshwater systems with depleted salmon runs, which probably inhibits attempts to renew those runs (Gresh et al. 2000).

2.3.5 Ecosystem Integrity

Unlike most other areas supporting Pacific salmon populations, the Bristol Bay watershed is a nearly pristine ecosystem, undisturbed by significant human development. Large-scale, human-caused modification of the landscape—a factor contributing to extinction risk for many native salmonid populations (Nehlsen et al. 1991)—is absent, and development in the watershed consists of only a small number of towns, villages, and roads. Iliamna Lake is the largest undeveloped lake in the United States.

The primary human manipulation of the Bristol Bay ecosystem is the marine harvest of approximately 70% of salmon returning to spawn. However, commercial salmon harvests are the ADFG's second priority for fish management; its first priority is to ensure that sufficient fish migrate into rivers to maintain a sustainable fishery, and thus a sustainable landscape. No hatchery fish are reared or released in the Bristol Bay watershed, whereas approximately 5 billion hatchery-reared juvenile salmon are released annually across the North Pacific (Irvine et al. 2009).

2.4 Bristol Bay and Pacific Salmon Stocks at a Global Scale

As the preceding sections illustrate, the Bristol Bay region is a unique environment supporting world-class Pacific salmon populations. However, the region takes on even greater significance when one considers the status and condition of Pacific salmon populations throughout their native geographic distributions (Figure 2-6A).

Although it is difficult to quantify the true number of extinct Pacific salmon populations around the North Pacific, estimates for the western United States (California, Oregon, Washington, and Idaho) range from 106 to 406 populations (Nehlsen et al. 1991, Augerot 2005, Gustafson et al. 2007). Pacific salmon are no longer found in 40% of their historical breeding ranges in the western United States, and where populations remain, they tend to be significantly reduced or dominated by hatchery fish (NRC 1996). For example, 214 salmon and steelhead stocks were identified as facing risk of extinction in the western United States; 76 of those stocks were from the Columbia River basin alone (Nehlsen et al. 1991). In general, these losses have resulted from cumulative effects of habitat loss, water quality degradation, climate change, overfishing, dams, and other factors (NRC 1996, Schindler et al. 2010). Species with extended freshwater rearing periods—that is, species like sockeye and Chinook, which dominate salmon production in the Bristol Bay watershed—are more likely to be extinct, endangered, or threatened than species, which spend less time in freshwater habitats (NRC 1996). No Pacific salmon populations from Alaska are known to have gone extinct, although many show signs of population declines (Appendix A).

The status of Pacific salmon throughout the United States highlights the value of the Bristol Bay watershed as a salmon sanctuary or refuge (Rahr et al. 1998, Pinsky et al. 2009). The Bristol Bay watershed contains intact, connected habitats that extend from headwaters to ocean with minimal influence of human development. These characteristics, combined with the region's high Pacific salmon abundance and life history diversity, make the Bristol Bay watershed a significant resource of global conservation value (Pinsky et al. 2009).

CHAPTER 3. PROBLEM FORMULATION

Before an assessment can be conducted, its scope must be identified. In ecological risk assessment, this process is called problem formulation. During problem formulation, key components that frame the assessment—such as the focal activity, region and endpoints—are defined. In this section, we describe these components for the assessment; Table 3-1 provides an overview.

Table 3-1. Summary of the Problem Formulation Components for the Bristol Bay Assessment

Component	Description
Type of development	Activities directly associated with large-scale porphyry copper mine development, operation and maintenance
Region	Pebble deposit, in the headwaters of the Nushagak River and Kvichak River watersheds
Endpoints	Quality, quantity, and genetic diversity of salmon populations Quality, quantity, and genetic diversity of non-anadromous fish populations Quantity and diversity of wildlife (as affected by fisheries) Alaska Native cultures (human welfare as affected by fisheries)
Timeframe	Operation: during mine operation Post-closure: After mine closure, when post-closure activities are on-going and oversight at mine is relatively high Perpetuity: after post-closure activities are completed and oversight at mine is minimal
Types of evidence and inference	Mine scenario Analogy to existing mines

3.1 Type of Development

The assessment addresses potential mining development in the watersheds of the Nushagak and Kvichak Rivers. It is limited to the mining of porphyry copper ores, which appear to be the major mineral resource type in the area. The assessment focuses on the Pebble deposit area, as this deposit is most likely to be developed in the near term and provides the most complete description of potential mining available to the public. However, there are a number of other claims in the region as well, and we consider cumulative effects of multiple potential mines in Chapter 7. The types of development considered in the assessment would be common to all porphyry copper mining in the area, and are limited to mineral extraction, beneficiation, waste disposal, and product and fuel transport. These activities directly associated with mining are described in the mine scenario (Chapter 4).

Certain activities associated with mining, but not directly related to mine operations, are not considered in this assessment. These include support activities such as housing workers and disposing of their wastes, power generation and transmission, construction and operation of a deepwater port at Cook Inlet, and secondary development (i.e., development that is not part of the mine project, but for which the mine project provides the impetus or opportunity, such as rural recreation or residential and commercial growth resulting from improved access). Exclusion of an activity from this assessment does not imply that it would be benign or have no effect on the environment, and many of these activities could have significant repercussions for the Bristol Bay ecosystem. The assessment focuses on activities directly associated with mine development, operation, and maintenance, which are most likely to have significant effects on the region's fish populations (Section 3.3).

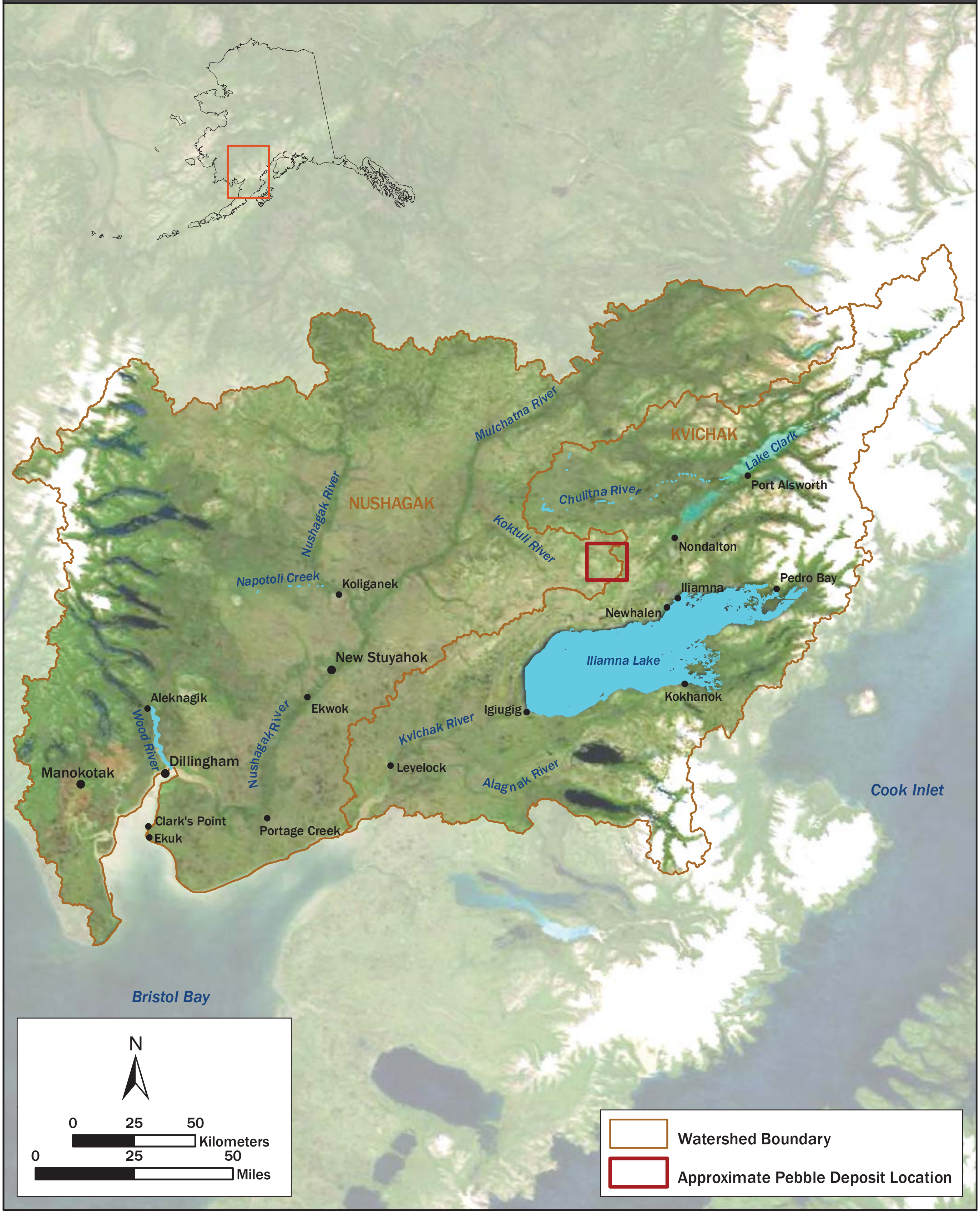
3.2 Region

The Pebble deposit represents the most likely site for near-term, large-scale mining development in the Bristol Bay watershed. This site is located in the headwaters of the Nushagak and Kvichak Rivers (Figure 3-1). Because the Nushagak River and Kvichak River watersheds account for more than half the land area of the Bristol Bay watershed, and are the watersheds most likely to be affected by large-scale mining development, this assessment focuses primarily on these two watersheds. Although the assessment applies to most sites in the Nushagak River and Kvichak River watersheds, three tributaries of these rivers are of particular note (Figure 3-1, inset): the North Fork Koktuli River, located to the northwest of the Pebble deposit, which flows into the Nushagak River via the Mulchatna; the South Fork Koktuli River, which drains the Pebble deposit area and converges with the North Fork west of the Pebble deposit; and Upper Talarik Creek, which drains the eastern portion of the Pebble deposit area and flows into the Kvichak River via Iliamna Lake.

3.3 Endpoints

The assessment focuses on four endpoints in the Nushagak River and Kvichak River watersheds: (1) quality, quantity, and genetic diversity of salmon populations; (2) quality, quantity, and genetic diversity of non-anadromous fish populations; (3) quantity and diversity of wildlife (as affected by fisheries); and (4) Alaska Native cultures (human welfare as affected by fisheries).

Figure 3 1. The Nushagak River and Kvichak River Watersheds of Bristol Bay



The primary endpoint of interest for this assessment is the quality, quantity, and genetic diversity of Pacific salmon in the Nushagak River and Kvichak River watersheds. As discussed in Chapter 2, the Bristol Bay region supports world-class fisheries among its five salmon species—sockeye, coho, Chinook, chum, and pink—with the Nushagak River and Kvichak River watersheds producing more than half of the region's salmon harvest (Appendix A). These fisheries generate significant economic benefit for commercial fishermen, provide subsistence for Alaska Natives, and support a significant recreational sector. Because sockeye, coho, and Chinook salmon spend a year or more rearing in the Bristol Bay watershed's streams, rivers, and lakes before their ocean migration—compared to chum and pink salmon, which migrate soon after emergence—these species are more dependent on upstream freshwater resources potentially affected by mining development. Accordingly, this assessment focuses on sockeye, coho, and Chinook salmon.

The region also supports subsistence fishing and world-class recreational sport fishing, for non-salmon fish species. The quality, quantity, and genetic diversity of two of these non-salmon fishes—rainbow trout and Dolly Varden are also included as assessment endpoints. Both are valuable sport and subsistence fish found throughout the watersheds. Dolly Varden may be especially vulnerable, because they are found in low-order, headwater streams likely to be affected by mining development. Other fish such as whitefish and grayling are also important, but are not as well-known and are believed to be less sensitive than the chosen representative species.

Because these fisheries benefit numerous other aquatic and terrestrial species, and are used extensively by Alaska Natives of the Bristol Bay region, the assessment also considers fish-mediated effects on wildlife and Alaska Native cultures—that is, it examines how changes in the region's fisheries, in turn, may affect wildlife and Alaska Native cultures. The assessment focuses on wildlife species that depend on salmon for food (e.g., brown bear, bald eagles, gray wolves) or that are important subsistence foods for Alaska Natives (e.g., moose, caribou). Direct effects of large-scale mine development on wildlife and Alaska Natives (e.g., direct alteration of wildlife habitat or direct effects of increased development on Alaska Native cultures) and secondary effects on the commercial and recreational economic sectors are beyond the scope of this assessment.

3.4 Timeframe

The assessment addresses three time periods: operation, when the mine active; post-closure, when mine operation has ceased but post-closure activities are ongoing and oversight is relatively high; and perpetuity, when post-closure activities have ceased and oversight is minimal. During operation, mine infrastructure would be built and ore would be extracted. The assessment evaluates this phase for a minimum and a maximum mine size, which assume different amounts of resource mined (Chapter 4). When mining is completed, either as planned or prematurely, the post-closure phase would begin. During this period, if the mine is closed as planned, the site would be monitored and water treatment and other waste management activities would continue, as necessary. Facilities needed to support on-going monitoring and maintenance activities—such as stormwater management ditches, monitoring wells, engineered covers on waste materials (if required), water treatment plants, and roads—would need to be maintained and replaced or remediated if they become compromised. At some point, given the limited lifetime of human institutions, the post-closure time period would lead into the perpetuity period. Active management of the mine site (e.g., monitoring and water treatment) would likely stop within decades to centuries of the end of mine operations, whereas mine wastes (e.g., tailings and waste rock) will remain in place in perpetuity.

3.5 Types of Evidence and Inference

The assessment is based on weighing two types of evidence: (1) analysis of a mine scenario in Bristol Bay and (2) analogy to existing mines. Under the first type of evidence, we develop a mine scenario that defines the potential direct impacts of mine development (e.g., length of streams filled), the effluents resulting from mine development, potential mitigation measures, and plausible accidents and failures (Chapter 4). We estimate the consequences of this mine scenario—using general scientific knowledge, mathematical and statistical models, data from the site, and data from laboratory studies—to evaluate exposure and exposure-response relationships. First, we estimate the magnitude of exposure to various consequences of the mine scenario (e.g., aqueous copper concentrations, kilometers of stream filled, kilometers of stream upstream of road crossings). Then, we consider the effects of these exposures—the exposure-response relationships—on our endpoints of interest (e.g., the relationship between water withdrawal and loss of salmon habitat, concentration-response relationships for copper and fish). We describe and quantify the exposure-response relationships to the estimated exposures and describe uncertainties. After these analyses, risk is characterized for each line of evidence by (1) combining exposures and exposure-response relationships to estimate effects and (2) considering uncertainties. For example, state standards, federal criteria, and effects models and toxicity tests for individual species are all lines of evidence for copper toxicity.

The second type of evidence involves analyzing monitoring results at existing mines. Prior mining activities in other, comparable watersheds provide examples of what can happen to the environment when metals are mined. This inference by analogy eliminates the uncertainties that come with modeling and prediction, but introduces other uncertainties related to site-specific differences in environmental conditions and mining practices. In this assessment, analogies are chosen to fit the individual issues being assessed, because no prior mine is similar in all aspects to potential mines in the Bristol Bay region. For example, we use the Fraser River watershed as an analogous system because it has similar mines and a similar salmon resource; however, we also recognize there are important differences between these systems, such as extensive urban development and forestry in the Fraser River watershed. We take care to use analogies that are defensible, despite their differences from our mine scenario. For example, metal mines in the Rocky Mountain metal belt (e.g., sites at Coeur d'Alene River, Idaho, and Clark Fork, Montana) were developed using mining practices that would not be allowed under current mining laws. However, failure of tailings dams or discharge of tailings onto floodplains at these sites, which also supported trout and salmon populations, offer some parallels to potential tailings dam failures in the Bristol Bay region—even if the underlying causes of failures differ.

Each risk is characterized by weighing these different lines and types of evidence, based on evidence strength and quality. The resulting qualitative or quantitative estimate of risk and uncertainty is based on either the best line of evidence or a combined estimate from multiple lines of evidence and inferences. Bounding analyses are used to express uncertainties concerning future mine activities and their effects. In particular, multiple sizes of mines and durations of mining are included in the mine scenario (Chapter 4). Bounding is also used to express stochasticity. For example, the occurrence and magnitude of tailings dam failures are random variables that cannot be reasonably defined. Hence, a range of tailings dam failure probabilities and a range of tailings release magnitudes are evaluated (Section 4.4.2).

3.6 Conceptual Models

To frame the assessment, we developed a series of conceptual model diagrams illustrating potential pathways by which activities and sources associated with large-scale mine development can lead to proximate stressors—that is, physical or chemical factors that can directly induce adverse effects—and, ultimately, impairment of salmon and resident fish resources in the focal watersheds (Box 3-1). These diagrams were initially developed by evaluating potential activities, sources, stressors, and ecological effects associated with large-scale mining development. These entities were organized into hypothesized cause-effect relationships leading from mine-related activities and sources to endpoints of interest, and revised based on feedback from the assessment team and other stakeholders (e.g., members of the Intergovernmental Technical Team).

The first four diagrams (Figures 3-2A through 3-2D) are organized according to stage of the mine life cycle (construction and operation vs. post-closure), type of mine operation (routine operations vs. accidents and failures), and the types of effects considered (habitat vs. water quality). The fifth diagram (Figure 3-2E) illustrates potential fish-mediated effects on Alaska Native cultures.

BOX 3-1. CONCEPTUAL MODELS

The conceptual model diagrams graphically represent the hypothesized pathways by which large-scale mine development may adversely affect Bristol Bay's salmon and resident fish resources and Alaska Native cultures. Inclusion of a pathway in these diagrams does not mean that pathway will occur with mine development, but rather that it is plausible that the pathway could occur.

When viewing these diagrams, it helps to keep the following principles in mind:

- Arrows leading from one shape to another indicate a hypothesized cause-effect relationship, whereby the first (or originating) shape can plausibly cause or result in the second shape.
- Arrows leading from a shape to another arrow indicate that the originating shape (always categorized as a *modifying factor*) plausibly influences the cause-effect relationship illustrated by the second arrow (e.g., by increasing or decreasing its probability or intensity of occurrence).
- Shapes within brackets are specific examples of the more general shape under which they appear.

Figure 3-2A. Conceptual Model Illustrating Potential Habitat Effects Associated with Mine Construction and Operation

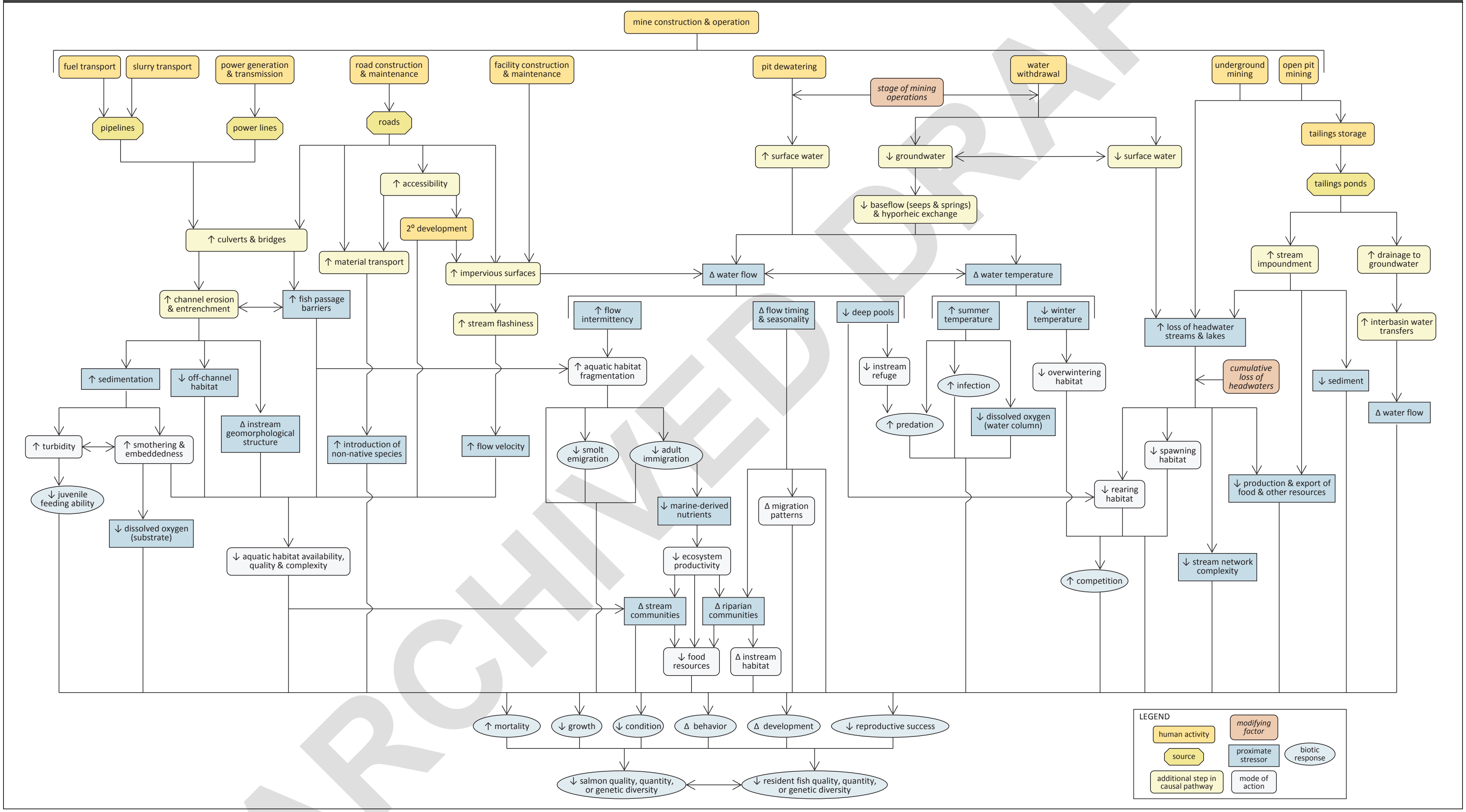


Figure 3-2B. Conceptual Model Illustrating Potential Water Quality Effects Associated with Mine Construction and Operation

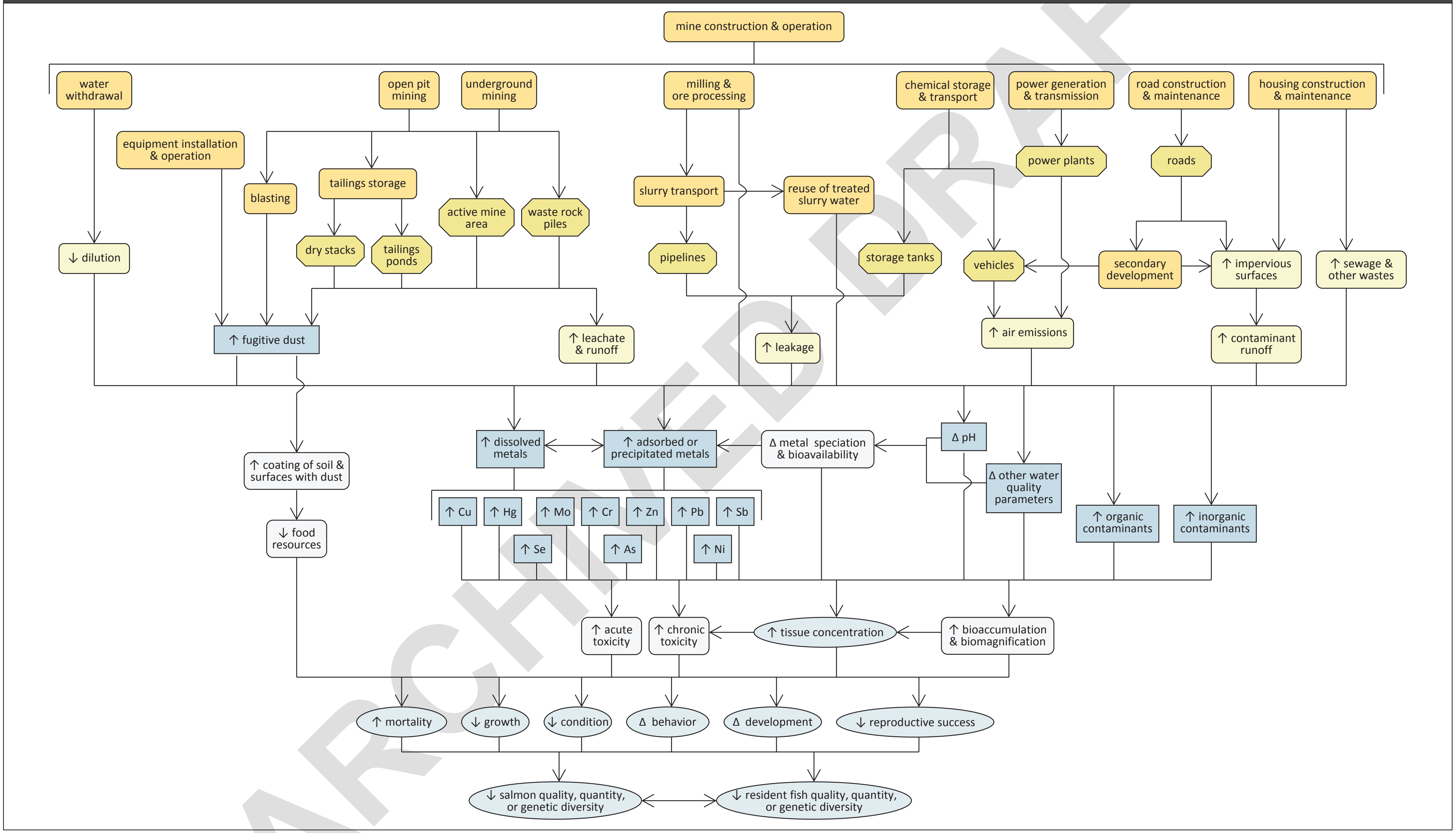


Figure 3-2C. Conceptual Model Illustrating Potential Habitat and Water Quality Effects Associated with Post Closure Mine Management

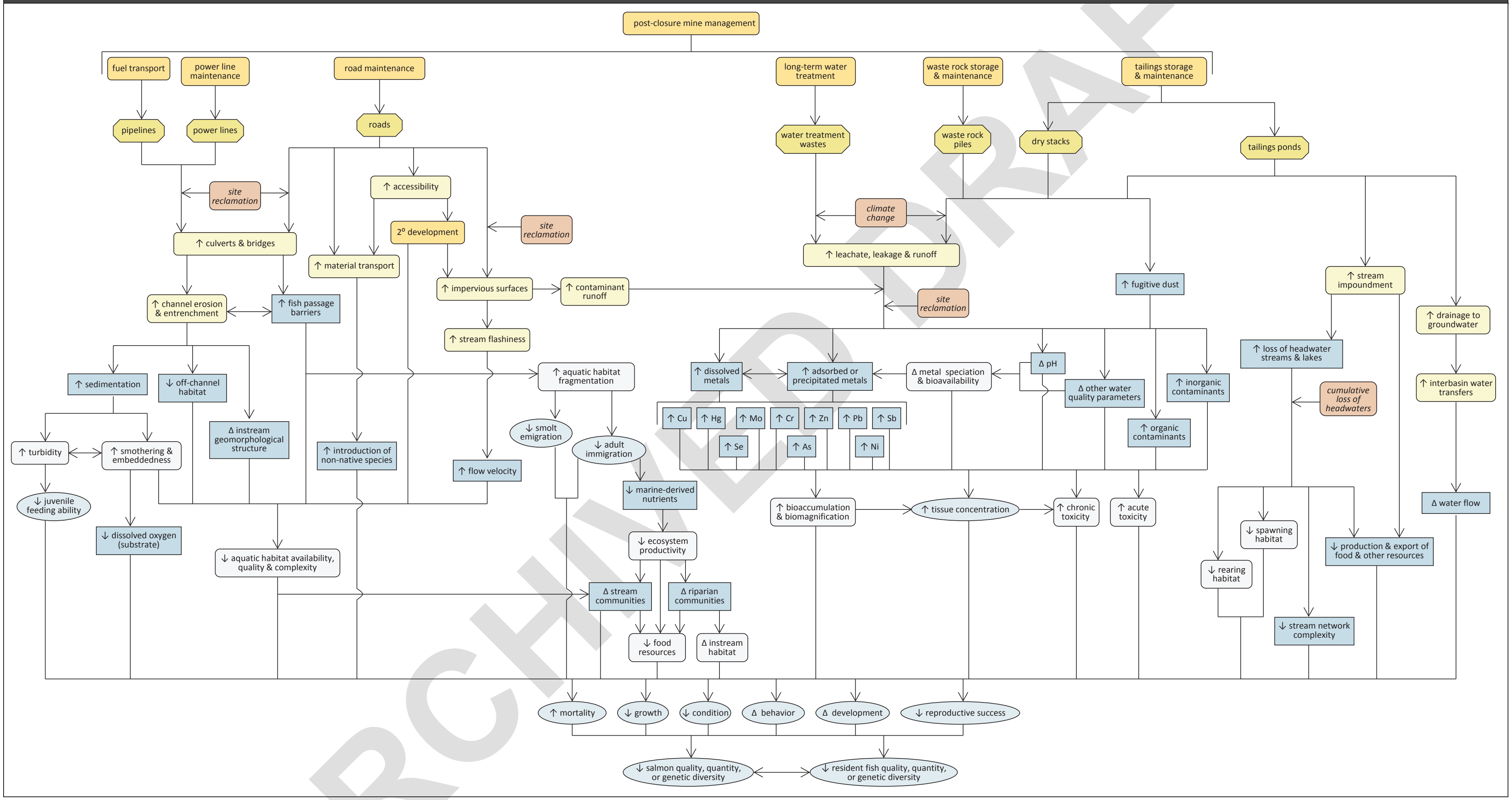


Figure 3-2D. Conceptual Model Illustrating Potential Habitat and Water Quality Effects Associated with Mine Accidents and Catastrophic Failures

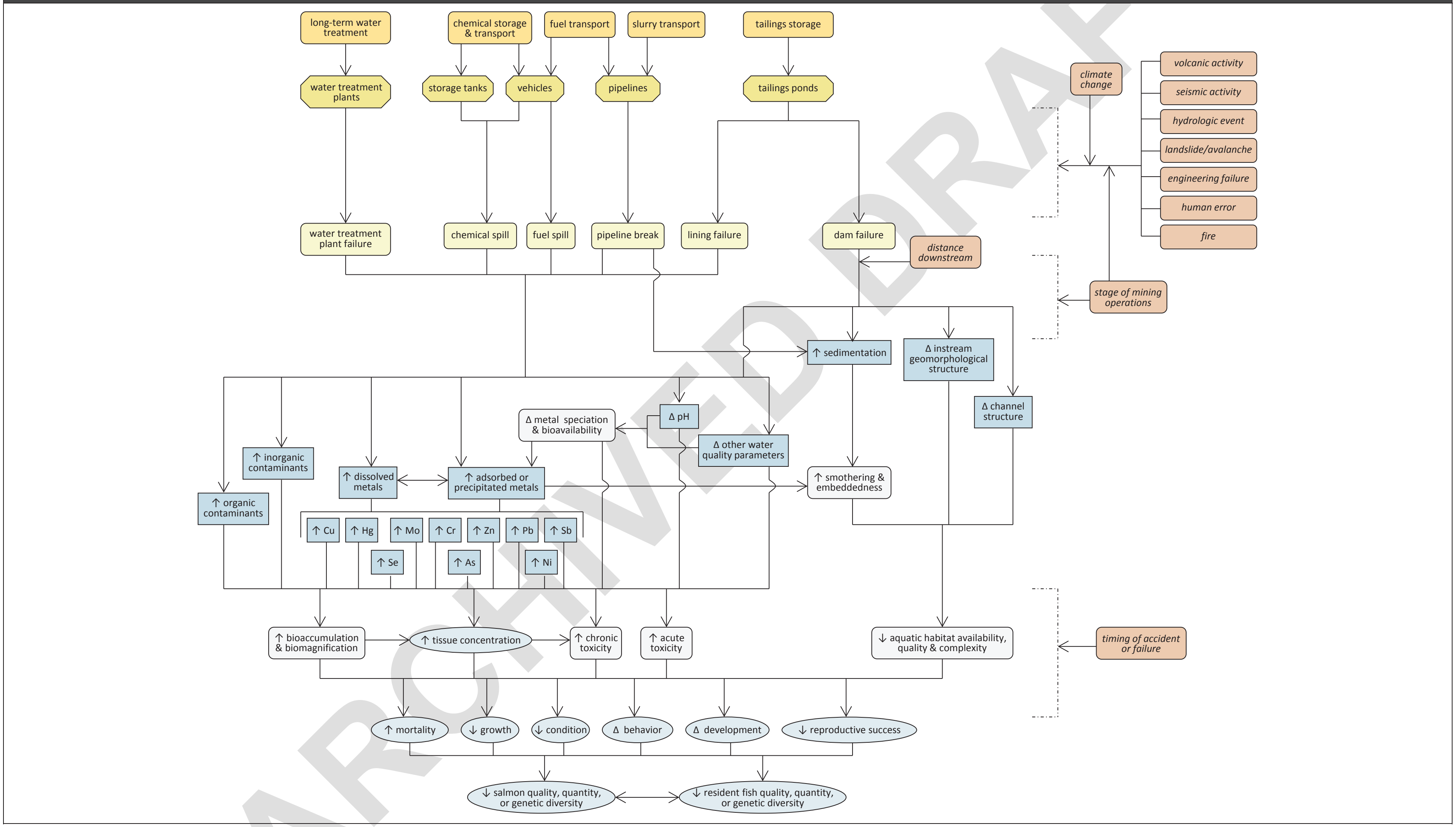
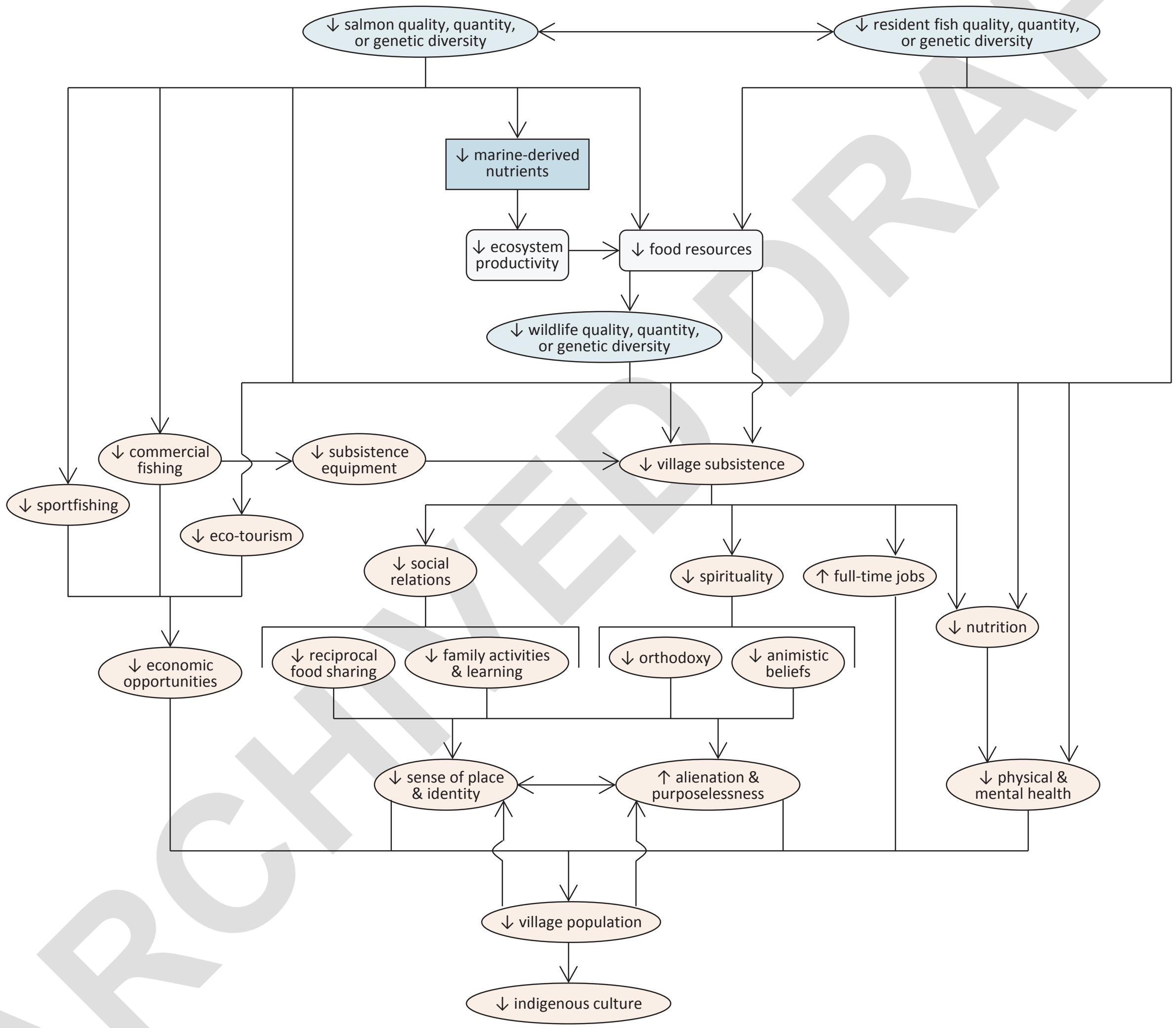


Figure 3-2E. Conceptual Model Illustrating Potential Fish-Mediated Effects on Alaska Native Cultures



CHAPTER 4. MINING BACKGROUND AND SCENARIO

In this section, we first provide background information on known mineral deposits in the Nushagak River and Kvichak River watersheds, with particular focus on porphyry copper deposits (Section 4.1). We then present a general overview of the processes and components associated with porphyry copper mining (Section 4.2). Specific processes and components from this overview are then incorporated into our hypothetical but realistic mine scenario (Section 4.3), which is used as the basis for subsequent analyses of potential mine failures (Section 4.4). We have included sources describing exploration and potential mining in the Bristol Bay watershed, as well as sources from the worldwide body of literature related to mining of porphyry copper deposits. Described mining practices and our mine scenario reflect the current practice for porphyry copper mining around the world, and represent current good, but not necessarily best, mining practices.

The largest of the existing claim blocks in the Bristol Bay watershed, and the claim closest to submission of a formal application for mining, is that belonging to the Pebble Limited Partnership (PLP). Although the Pebble deposit is used as an example of potential mining in the region, the assessment does not predict what the PLP may eventually propose. The mine scenario described here is meant to reflect activities typically associated with large-scale porphyry copper mining in a general sense, rather than the specific characteristics of an individual mine.

4.1 Mineral Deposits in the Nushagak River and Kvichak River Watersheds

The geologic setting of the Nushagak River and Kvichak River watersheds has characteristics indicating the presence of several different mineral-deposit types (Schmidt et al. 2007). Of deposit types likely to occur in the region, porphyry copper, intrusion-related gold, and copper and iron skarn may be economically viable and prompt large-scale development. The potential for large-scale mining development within the watershed is greatest for porphyry copper deposits and, to a lesser extent, for intrusion-related gold deposits. Significant exploration activity associated with porphyry copper deposits is underway at the Pebble deposit and other sites. Accordingly, the remainder of this report will focus exclusively on porphyry copper deposits, although much of the discussion of mining methods applies to all types of disseminated ore deposits (i.e., ores with low concentrations of metal spread throughout the body of rock).

4.1.1 Genesis of Porphyry Copper Deposits

Porphyry copper deposits are found around the world, most commonly in areas with active or ancient volcanism (Figure 4-1). They are formed when hydrothermal systems are induced by the intrusion of magma into shallow rock in the Earth's crust. Water carries dissolved sulfur-metallic minerals (sulfides) into crustal rock where they precipitate (John et al. 2010). Minerals containing sulfur and metals are disseminated and precipitate throughout the affected rock zone in concentrations typically less than 1% (Table 4-1) (Singer et al. 2008).

Table 4-1. Global Grade and Tonnage Summary Statistics for Porphyry Copper Deposits[a]

Parameter	10th Percentile	50th Percentile	90th Percentile	Pebble Deposit
Tonnage (Mt)	30	250	1,400	10,777
Cu grade (%)	0.26	0.44	0.73	0.34
Mo grade (%)	0.0	0.004	0.023	0.023
Ag grade (g/t)	0.0	0.0	3.0	unknown
Au grade (g/t)	0.0	0.0	0.20	0.31

Notes:

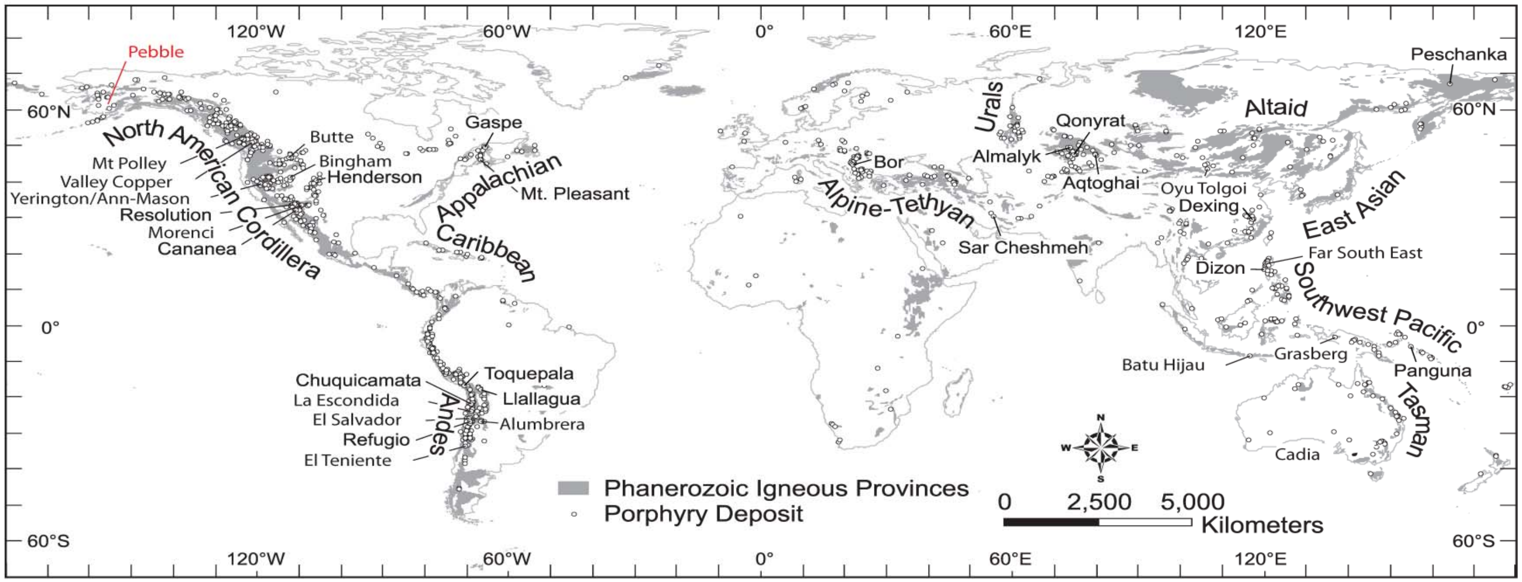
[a] Pebble deposit information is based on 0.3% copper cut-off grade, and includes measured, indicated, and inferred resources from PLP and other deposits (n = 256; Model 17).

Cu = copper; Mo = molybdenum; Ag = silver; Au = gold

Sources: PLP 2009, Singer et al. 2008

Porphyry copper deposits often occur in clusters (Lipman and Sawyer 1985, Singer et al. 2001, Anderson et al. 2009) and range in size from tens of millions to billions of metric tons. Singer et al. (2008) list the grade and tonnage of porphyry copper mines around the world. Mines in the 50th percentile have deposits of 200 to 250 million metric tons (Table 4-1). The well-delineated Pebble deposit is clearly at the upper end of the total size range; any additional deposits found in the Nushagak River and Kvichak River watersheds would be expected to be one or two orders of magnitude smaller.

Figure 4-1. Location of Phanerozoic Igneous Provinces and Representative Porphyry Deposits across the World. The location of the Pebble deposit indicated in red; the map is modified from Seedorff et al. (2005) and John et al. (2010).



4.1.2 Environmental Chemistry of Porphyry Copper Deposits

When mined, porphyry copper deposits can pose risks to aquatic and terrestrial ecosystems and to human health. These risks can range from insignificant to extremely harmful depending on a variety of factors, including site geology (both local and regional), hydrologic setting, climate, and mining and ore processing methods. There are a variety of geochemical models and approaches to understand and predict releases to the environment; however, there are limitations in our ability to make predictions with a high level of certainty because of the inherent complexity of natural materials and their environment.

Sources of risk from porphyry copper mines can be grouped into four broad categories: acid-generating potential, trace element associations and their mobility, mining and ore processing methods, and waste disposal practices. The relative importance of these categories will vary from deposit to deposit, but some generalization can be made for porphyry copper deposits as a whole. In this section we consider those categories related to environmental chemistry (i.e., acid-generating potential and trace elements); categories related to mining processes are described in Section 4.2.

Mining processes expose rocks and their associated minerals to atmospheric conditions that cause weathering. Grinding methods used in these processes create materials that have a high specific surface area, which accelerates the rate of weathering. Porphyry copper deposits are characterized by the presence of sulfide minerals, and oxidation of sulfide minerals creates acidity that can further accelerate weathering rates. Because most metals and other elements become more soluble as pH decreases, the acid-generating or acid-neutralizing potentials of waste rock, tailings, and mine walls are of prime importance in determining potential environmental risks associated with metals and certain elements in the aquatic environment.

One way to predict if acid generation will occur is to perform acid-base accounting tests. Acid-base accounting tests are rapid methods to determine the acid-generation potential (AP) and neutralization potential (NP) of a rock or mining waste material, independent of reaction rates. These potentials are then compared to one another by either their differences or their ratios, with the net neutralization potential (NNP) being NP-AP and the neutralizing potential ratio (NPR) being NP/AP, AP, NP, NNP, and NPR typically are expressed in units of kilograms of calcium carbonate per metric ton of waste material (kg CaCO3/metric ton). NNP values greater than zero are net alkaline, those equal to zero are net neutral, and those less than zero are net acidic.

Although methods used for acid-base accounting have known limitations, it is common industry practice to consider materials that have an NPR of 1 or less as potentially acid generating (PAG) and materials with an NPR greater than 4 (Brodie et al. 1991, Price and Errington 1998) as having no acid generation potential (NAG). Materials having a ratio between 1 and 4 require further testing via kinetic tests (e.g., ASTM D5744-07e1) and geochemical assessment for classification (Brodie et al. 1991, Price and Errington 1998). This further testing and assessment are necessary because if neutralizing minerals react before acid generating minerals, the neutralizing effect may not be realized and acid might be generated at a later time. Additionally, some toxic elements (e.g., selenium and arsenic) may be released from mining materials under neutral or higher pH conditions, which would be observed during kinetic leaching tests conducted at variable pH values.

In general, the rocks associated with porphyry copper deposits tend to straddle the boundary between being net acidic and net alkaline, as illustrated by Borden (2003) for the Bingham Canyon, Utah porphyry copper deposit (Figures 4-2 and 4-3). AP values for porphyry copper deposits typically correlate with the distribution of pyrite. The pyrite-poor, low-grade core corresponds to the central part of the Bingham Canyon deposit, where NNP values are greater than zero. Moving outward from the core to the ore shell and pyrite shell, pyrite abundance increases and NNP values become progressively more negative (Figure 4-3).

4.2 Porphyry Copper Mining Processes

Developing a mine requires establishing surface or underground mine workings that allow access to the ore body. The scope and complexity of development-related activities vary depending on the characteristics of each project, but typically include the following components.

- **Site preparation (clearing, stripping, and grading).** Topsoil and overburden are usually stockpiled for later use in mine reclamation.
- **Construction of mine infrastructure.** Specific requirements depend on the size and type of mine operation, location, and proposed mining and beneficiation methods. Typical infrastructure includes facilities for ore crushing, grinding, and other beneficiation processes; ore stockpiling and waste rock disposal facilities; tailings dams; water supply, treatment, and distribution facilities; roads; pipelines; conveyers; and other infrastructure (e.g., offices, shops, housing).
- **Establishment of mine workings.** Open pits and underground mine workings are usually excavated by drilling and blasting. Mine construction may include some ore production for use in testing the ore handling and processing facilities (Environment Canada 2009).

A significant part of mine development in the Nushagak River and Kvichak River watersheds would be infrastructure development. These watersheds encompass 6.1 million hectares (23,539 square miles), slightly smaller than the state of West Virginia (Figure 3-1). Existing infrastructure is limited to paved and lighted airstrips at Iliamna, Dillingham and King Salmon, and four segments of single- or double-lane roads: Williams Port to Pile Bay, Dillingham to Aleknagik, Naknek to King Salmon, and Iliamna to the upper Newhalen River near Nondalton (Figure 3-1). Any mine in these two watersheds would require new roads to coastal areas on Bristol Bay or Cook Inlet, as well as a port facility.

Figure 4-2. Plot of Neutralizing Potential (NP) vs. Acid-Generating Potential (AP) for Mineralized Rock Types at the Bingham Canyon Porphyry Copper Deposit, Utah. Bingham Canyon shares many geologic features with the Pebble deposit. Modified from Borden (2003).

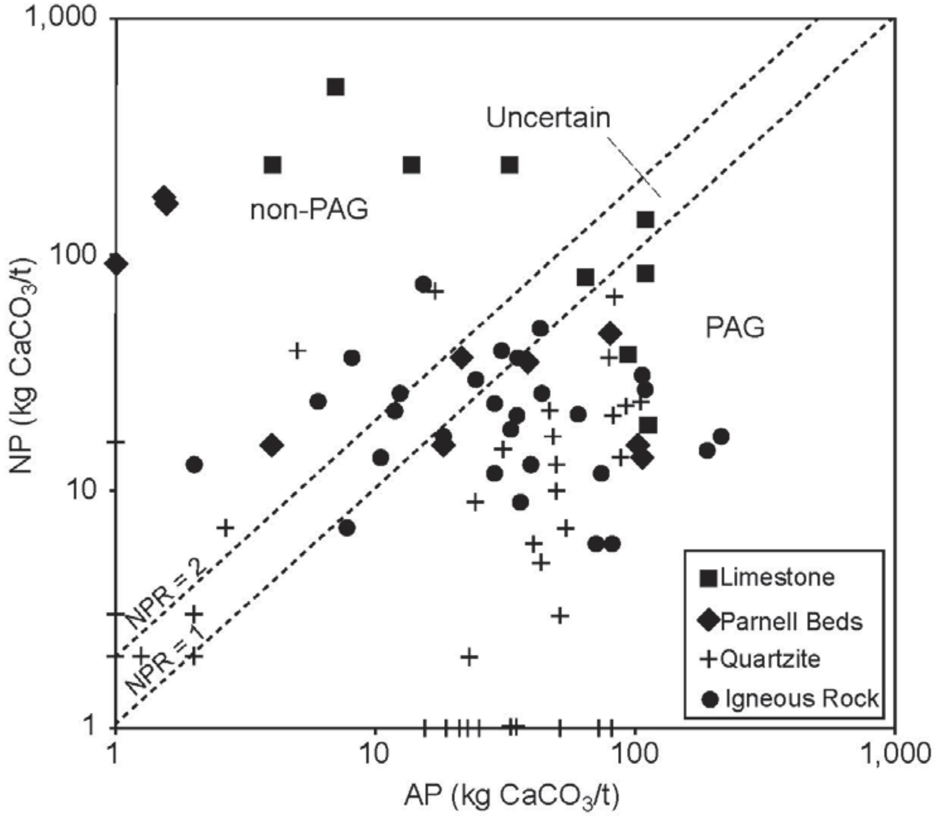
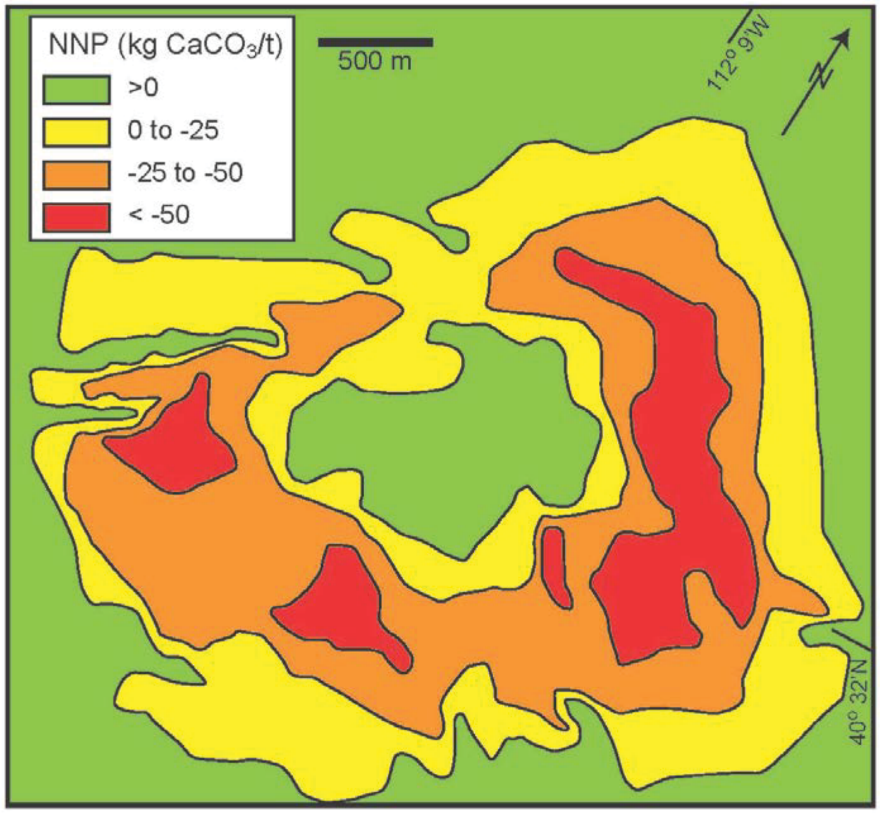


Figure 4-3. Plan View of the Distribution of Net Neutralizing Potential (NNP) Values at the Bingham Canyon Porphyry Copper Deposit, Utah. Bingham Canyon shares many geologic features with the Pebble deposit. NNP values greater than 0 are net alkaline; NNP values less than 0 are net acid. Modified from Borden (2003).



4.2.1 Extraction Methods

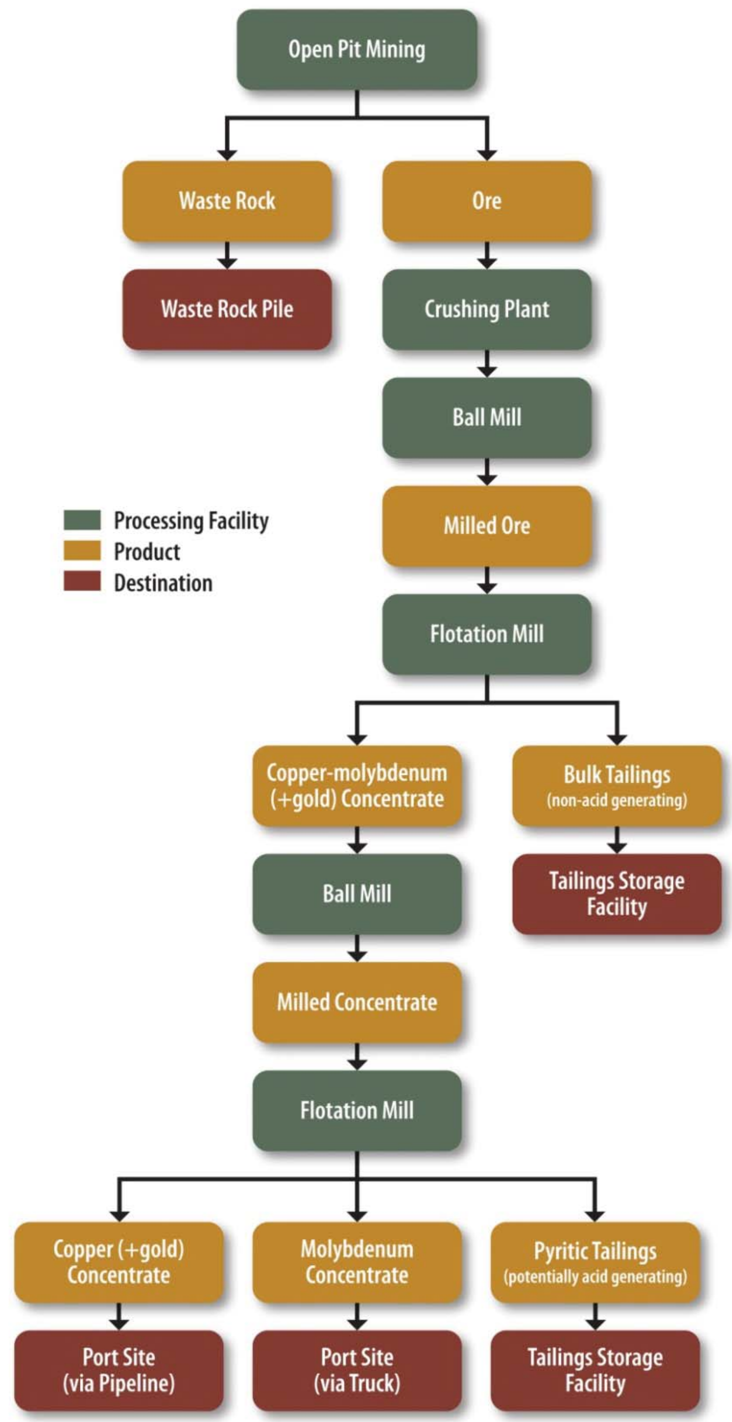
The low concentrations of disseminated metals in porphyry copper deposits require large amounts of ore to maximize the return on investment. Bulk or large-scale mining methods have been developed for this purpose, and specific mining methods depend on ore quality and depth. Open pit mining is typically used to extract ore where the top of a deposit is near the surface (less than 100 m deep). Excavation of a pit begins at the surface and the pit is successively enlarged until a break-even economic analysis establishes the pit limits. Block caving is an underground method used for large deposits with rock mass properties amenable to sustainable caving action (Lusty and Hannis 2009, Singer et al. 2008, Blight 2010). It requires tunneling to the bottom of the ore and undercutting the ore body, so that the deposit caves under its own unsupported weight; as ore is removed from below, the material above fragments and is removed at the bottom of an enlarging void. Eventually, surface material collapses into the void, leaving a depression on the landscape similar to, but not as stark as, the open pit. In contrast to open pit mining, block caving does not require the removal of overlying waste rock, eliminating the cost of handling some of the rock that is unprofitable to process.

4.2.2 Ore Processing

Generally, two streams of materials come from a mine: ore and waste rock (Figure 4-4). Ore is rock with sufficient amounts of metals to be economically processed. Waste rock is all other material that has little or no economic value at the time of disturbance, although it may have recoverable value at a future time (i.e., under different technology or economic conditions).

Ore blasted from a porphyry copper mine typically is hauled to a crushing plant near or in the mine pit (Figure 4-4). The crushing plant reduces ore to particle sizes manageable in the processing mill (e.g., less than 15 cm) (Ghaffari et al. 2011). Crushed ore is carried by truck or conveyer to a ball mill, where particle size is further reduced (e.g., less than 200 μm) (Ghaffari et al. 2011) to maximize the recovery of metals. The milled ore is subjected to a flotation process with an aqueous mixture of chemical reagents (e.g., pH controllers, collectors, and frothers) to recover valuable copper, molybdenum, and gold minerals into a copper-molybdenum concentrate (which also contains gold). Bulk tailings are the materials left after the first flotation circuit, and are directed to a tailings storage facility (TSF) (Section 4.2.3, Figure 4-4). The copper-molybdenum (+gold) concentrate may be fed through a second ball mill to grind the particles again (e.g., to less than 25 μm) (Ghaffari et al. 2011). Once sufficiently sized, the concentrate is directed into a second flotation process and then to a copper- molybdenum separation process. Final products are a copper concentrate that includes gold, a molybdenum concentrate, and pyritic tailings (Figure 4-4).

Figure 4-4. Simplified Schematic of Mined Material Processing



The most profound influence that ore processing can have on long-term management of a mine site centers on the fate of pyrite (Fuerstenau et al. 2007). At many porphyry copper mines, pyrite is discharged with the tailings, thereby contributing to the acid-generating potential of the TSF (Figure 4-4). A separate pyrite concentrate can be produced to decrease the acid-generating potential of the tailings; however, these concentrates are highly reactive and generate separate storage or transportation concerns.

The gold in porphyry copper deposits can be partitioned among the copper-sulfide minerals (chalcopyrite, bornite, chalcocite, digenite, and covellite), pyrite, and free gold (Kesler et al. 2002). Gold associated with the copper minerals will stay with the copper (+gold) concentrate and be recovered at an off-site smelter. Gold associated with pyrite will end up in the TSF unless a separate pyrite concentrate is produced, and gold is recovered from this concentrate by a vat leaching cyanidation process (Logsdon et al. 1999, Marsden and House 2006). The solution that remains after this cyanidation process is either treated in a water treatment plant or stored in the TSF, where cyanide concentrations may decrease through natural attenuation (e.g., volatilization, photodegradation, biological oxidation, and precipitation) (Logsdon et al. 1999). Tailings from this process, which have high concentrations of acid-generating sulfides, typically are directed to the TSF, where they are encapsulated in non-acid-generating tailings and kept saturated to minimize oxidation.

Porphyry copper deposits (and other metal deposits) often have marketable quantities of metals other than the primary target metals. These metals are carried through the flotation process and might be removed at some later point. As an example, the Pebble deposit is reported to have marketable quantities of silver, tellurium, rhenium, and palladium (Ghaffari et al. 2011), which are not sufficiently concentrated in the ore to warrant separation and production of an additional metal concentrate.

The process for removing metals from ore is not 100% efficient. At some point the cost of recovering more metals exceeds their value, so the amount of metals left in the tailings represents a tradeoff between revenues from more complete ore recovery and extraction costs. The process described by Ghaffari et al. (2011) recovers 86.1% of the copper, 83.6 % of the molybdenum and 71.2% of the gold from the ore. The residual metals remaining with the tailings are discharged to a TSF with the residue of blasting agents, flotation reagents, and inert portions of the ore.

4.2.3 Tailings Storage

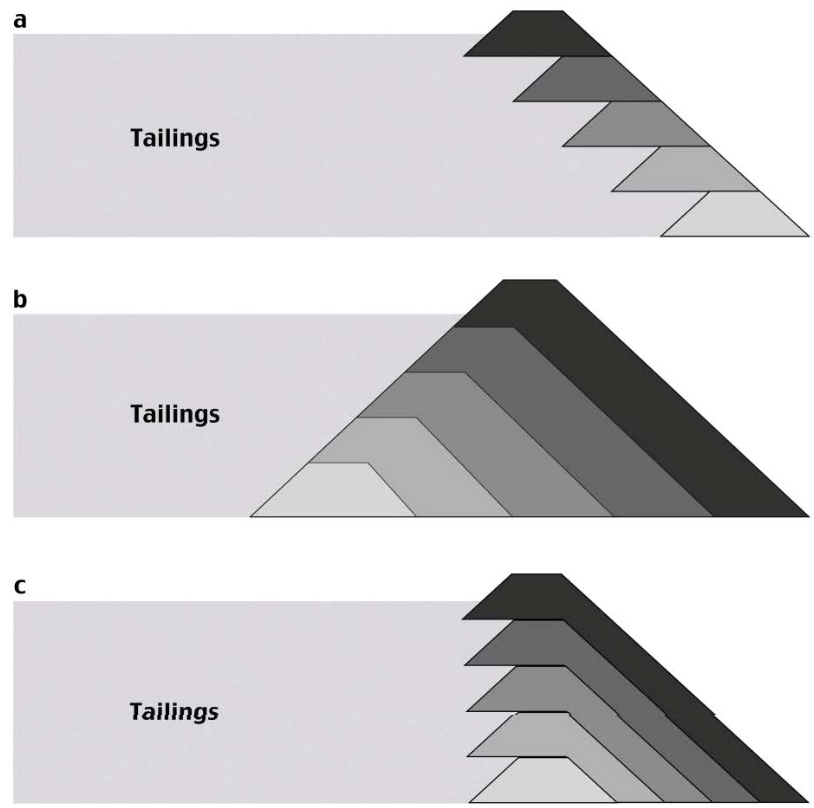
Tailings are a mixture of fine-grained particles, water, and residue of reagents remaining from the milling process. They are transported from the mill to a TSF as a slurry, of which solids—silt to fine sand (0.001- to 0.6-mm) particles with concentrations of metals too low to interact with flotation reagents—typically make up 30 to 50% by weight. Tailings may be thickened prior to disposal (i.e., via removal of water) to reduce evaporation and seepage losses and allow recycling of more process water back to the processing plant, thereby reducing operational water demand. Thickening also minimizes the amount of water stored in the TSF.

The most common method of tailings disposal is placement into tailings impoundments, which are water-holding structures typically built by creating a dam in a valley. Tailings dams are generally earthen or rockfill dams constructed from waste rock or the coarse fraction of the tailings themselves. The vast majority of tailings dams are less than 30 m (100 feet) in height, but the largest exceed 150 m (500 feet). The engineering principles governing the design and stability of tailings dams are similar to the geotechnical principles for earthen and rockfill dams used for water retention. They are typically built in sections over the lifetime of the mine, using upstream, downstream, or centerline methods (Figure 4-5), such that dam height increases ahead of the reservoir level. Tailings dams built by the upstream method are less stable against seismic events than dams built by either the downstream or the centerline method (ICOLD 2001), because it is not possible to compact the tailings that support the dam. Although upstream construction is considered unsuitable for impoundments intended to be very high or to contain large volumes of water or solids (State of Idaho 1992), this method is still routinely employed (Chambers and Higman 2011, Davies 2002). A dam designed as a hybrid upstream/centerline was recently constructed at the Fort Knox Mine tailings impoundment near Fairbanks, Alaska. The downstream method is considered more stable, but it is also the most expensive option. Centerline construction is a hybrid of upstream and downstream methods and has risks and costs lying between them (Martin et al. 2002).

As they fill with tailings, TSFs must store immense quantities of water (Davies 2011). Water level is controlled by removing excess water for use in the mining process or for treatment, and discharge to local surface waters. Tailings are deposited against the embankment through spigots or cyclones. Relatively coarser-grained sands are directed at the embankment to create a beach, causing water and fines to drain away from the dam to form a tailings pond. Care must be taken to prevent the formation of low-permeability lenses or layers on tailings beaches, as these layers may perch water in the TSF such that saturation of or flow through the dam may occur, leading to erosion or failure.

Although most of the tailings dam mass consists of fairly coarse and permeable material, the dams often have a low permeability core to limit seepage, as well as internal drainage structures to collect seepage water and to control pore pressures. Mitigation measures for seepage through or beneath a tailings dam may include any combination of liners, seepage cutoff walls, under-drains, or decant systems. Liners can include a high-density polyethylene, bituminous, or other type of geosynthetic material and/or a clay cover over an area of higher hydraulic conductivity. A clay liner may have a saturated hydraulic conductivity of 10^{-8} m/s, whereas a geomembrane may have a hydraulic conductivity of approximately 10^{-10} m/s (Commonwealth of Australia 2007). However, geomembrane technology has not been available long enough to know their service life, and geomembranes are generally estimated by manufacturers to last 20 to 30 years when covered by tailings (North pers. comm.). Overly steep slopes also may put stresses on geomembranes and cause them to fail.

Figure 4-5. Cross-Sections Illustrating a) Upstream, b) Downstream, and c) Centerline Tailings Dam Construction. In each case, the initial dike is illustrated in light gray, with subsequent dike raises shown in darker shades (modified from Vick 1983).Tailings dams in our mine scenario are assumed to use the downstream construction method.



Liners may cover the entire impoundment area, or only the pervious bedrock or porous soils. Full liners beneath TSFs are not always used and may not be practicable for large impoundments; however, the use of liners to minimize risks of groundwater contamination is increasingly required (Commonwealth of Australia 2007). If seepage is expected or observed, mitigation or remedial measures such as interception trenches or seepage recovery wells can be installed around the perimeter and downstream of the TSF to capture water and redirect it to a treatment facility. Precipitation runoff from catchment areas up-gradient of the TSF is typically diverted away from the impoundment to reduce the volume of stored liquid.

Dry stack tailings management, in which tailings thickened to a paste and filtered are “stacked” for long-term storage, is a newer, less commonly used tailings disposal method. Dry stacked tailings require a smaller footprint, are easier to reclaim, and have lower potential for structural failure and environmental impacts (Martin et al. 2002). However, the high energy cost of dry stack technology remains a barrier for mining low-grade ores such as porphyry copper, and this type of storage is less applicable to larger operations where tailings impoundments may store water as well as tailings. It is most applicable in arid regions, although dry stacks are also used in wet climates, or in cold regions where water handling is difficult (Martin et al. 2002). Currently, the only mines in Alaska that use dry stack disposal of tailings are underground mines with high-grade ore and relatively low quantities of tailings (e.g., Greens Creek, a lead, silver, zinc mine in southeast Alaska; and Pogo, a gold mine in eastern interior Alaska).

4.2.4 Waste Rock

Waste rock is rock overlying or removed with the ore body that contains uneconomic quantities of metals. A waste-to-ore ratio of 2:1— that is, the removal of 2 metric tons of waste rock for each metric ton of ore—is not uncommon for porphyry copper deposits (Porter and Bleiwas 2003). Waste rock is stored separately from tailings (Blight 2010). Some waste rock that contains marketable minerals may be stored such that it can be milled if commodity prices increase sufficiently or if higher than usual metal concentrations in ore require dilution to optimize mill operation. However, the potential for environmental impacts must be managed if the waste rock is PAG, via selective handling, drains, diversion systems, or other means. PAG waste rock might be placed in the open pit at closure to minimize oxidation of sulfide minerals and generation of acid drainage. Other waste rock, which is neither potentially acid-generating nor contains sufficient metals, likely will be placed in a rock dump somewhere near or at the back of the mine pit (Blight 2010).

4.3 Mine Scenario: No Failure

For this assessment, we used general information on porphyry copper deposits and mining practices to develop a mine scenario (Tables 4-2 and 4-3). In this scenario we make assumptions concerning the placement of our hypothetical mine; the size of the mine and the time period over which mining will occur; the size, placement, and chemistry of waste rock; the size, placement, and chemistry of TSFs;

on-site processing of the ore; and the removal of processed ore concentrate from the site. For comparison purposes, Table 4-4 provides similar information on other past, existing, and potential large mines in Alaska.

Table 4-2. Overview of the Mine Scenario

Characteristic	Value	Description
Size of Mine	Minimum	2.0 billion metric tons of ore extracted from mine
	Maximum	6.5 billion metric tons of ore extracted from mine
Mode of Operation	No failure	Mitigation measures work properly, with no operational failures during or after mine operations
	Failure	Mitigation measures do not work properly, with one or more operational failures during or after mine operations
Tailings dam failure	Partial	Failure of tailings dam when TSF is partially full (dam height = 98 m, tailings volume = 227 million m^3)
	Full	Failure of tailings dam when TSF is completely full (dam height = 208 m, tailings volume = 1,492 million m^3)
Phase	Operation	During mine operation
	Post-closure	After mine closure, when post-closure activities are ongoing and oversight at mine is relatively high
	Perpetuity	After post-closure activities are completed and oversight at mine is minimal
Type of closure	Premature	Closure of mine before planned mine lifespan is reached and without planned site management
	Planned	Closure of mine once planned mine lifespan is reached and with ongoing site management

Notes:
TSF = tailings storage facility

Table 4-3. Mine Scenario Components

Parameter	Mine Size	
	Minimum	Maximum
Amount of ore mined (billion metric tons)	2.0	6.5
Approximate duration of mining	25 years	78 years
Ore processing rate (metric tons/day)	200,000	200,000
Mine Pit		
Surface area (km^2)	5.5	17.8
Depth (km)	0.8	1.2
Waste Rock Pile		
Surface area (km^2)	13.3	22.6
PAG waste rock (million metric tons)	638	5,172
PAG waste rock bulk density (metric tons/m^3)	2.1	2.1
NAG waste rock (million metric tons)	2,379	12,013
NAG waste rock bulk density (metric tons/m^3)	2.1	2.1
TSF 1[a]		
Capacity (billion metric tons)	2	2
Surface area (km^2)	14.9	14.9
Maximum dam height (m)	208	208
Volume (million m^3)	1,492	1,492
Tailings dry density (metric tons/m^3)	1.46	1.46
NAG density, embankment (metric tons/m^3)	2.3	2.3
TSF 2[a]		
Capacity (billion metric tons)	NA	3.9
Surface area (km^2)	NA	21.2
Maximum dam height (m)	NA	267
Volume (million m^3)	NA	2,746
TSF 3[a]		
Capacity (billion metric tons)	NA	1.0
Surface area (km^2)	NA	7.6
Maximum dam height (m)	NA	226
Volume (million m^3)	NA	674
Total TSF surface area (km^2)	14.9	43.7
Dam Failure at TSF 1		
Partial-volume failure	dam height = 98 m volume = 227 million m^3	
Full-volume failure	dam height = 208 m volume = 1,492 million m^3	
Transportation Corridor		
Total length (km)	139	139
Length in assessment watersheds (km)	118	118

Notes:
[a] Final value, when TSF is full.
NA = not applicable; TSF = tailings storage facility; PAG = potentially acid generating; NAG = non-acid-generating

Table 4-4. Characteristics of Past, Existing, or Potential Large Mines in Alaska

Mine	Kennecott	Donlin	Fort Knox	Greens Creek	Kensington	Pogo	Red Dog
Location	Copper River basin, in Wrangell-St. Elias National Park	13 miles N of the village of Crooked Creek and the Kuskokwim River	26 miles NE of Fairbanks	18 mi SW of Juneau, in Admiralty Island National Monument	45 miles NW of Juneau, between Berners Bay and Lynn Canal	85 miles ESE of Fairbanks	western Brooks Range, 82 miles N of Kotzebue and 46 miles from the Chukchi Sea
Target metals	copper, silver	gold	gold	zinc, lead, silver, gold	gold	gold	zinc, lead
Ore type	massive sulfide	gold-bearing quartz	oxide ore body	massive sulfide	gold-bearing quartz	gold-bearing quartz	massive sulfide
Ore grade quality	very high	moderate	low	high	moderate	moderate	high
Mine life (years)	27 (1911–1938)	22	20	35–50	10	11	42 (1989–2031)
Extraction type	underground stope mining	open pits (2)	open pit	underground stope mining	underground stope mining	underground stope mining	open pits (2)
Total resource (million tons)	~ 5	634	442	32	27	10	190
Ore processing rate (metric tons/day)	~ 100	53,500	36,000–50,000	1,680	1,250	2,500	8300–9100
Total waste rock (million metric tons)	< 1	2100	372.5 million	~ 2	1.6	1.9	157
Tailings disposal	on Kennicott Glacier	dam/ponds (2)	dam/pond	dry tailings	lake disposal	dry tailings	dam/pond
Tailings amount (million metric tons)	< 1	471	200	~ 15	4.5	5.4	100
Tailings footprint (km^2)	NA	5.4	4.5	0.25	0.24	0.12	3
Dam height (m)	NA	143 (largest of multiple dams)	111	NA	NA	NA	63
Acid rock drainage potential	no	yes	no	yes	no	no	yes

Notes:
NA = not applicable
Source: Levit and Chambers 2012

Although we borrow details from Ghaffari et al. (2011), our mine scenario is not based on a specific mine permit application—rather, it reflects the general activities and processes typically associated with the kind of large-scale porphyry copper mining development likely to be proposed once a specific mine application is developed. Our mine scenario represents current good, but not necessarily best, mining practices.

Our mine scenario is defined by a suite of characteristics, each of which has multiple values (Table 4-2). The assessment is broadly organized in two parts, corresponding to different modes of operation (Table 4-2). First, we consider no failure (or routine operation) mode, which assumes that all appropriate practices and controls are used to prevent chemical contamination of stream habitats downstream of the mine site and associated TSFs, and no operational failures occur during or after mine operation. Second, we consider failure mode, which assumes that one or more appropriate practices and controls either are not used or do not work properly, and one or more system failures occur during or after mine operation.

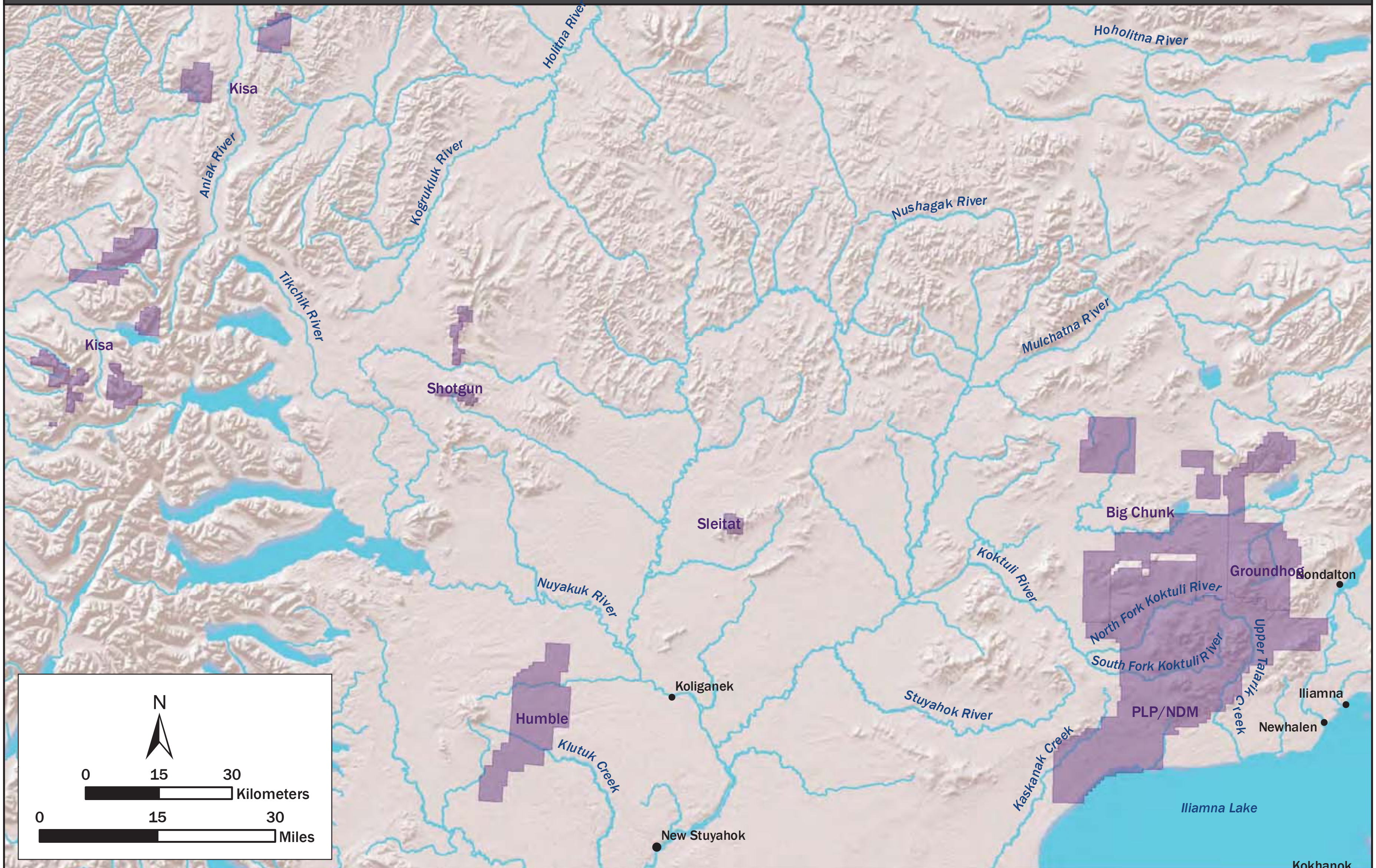
4.3.1 Mine Location

As discussed in Section 3.2, we have sited our hypothetical mine at the Pebble deposit in the headwaters of the Nushagak River and Kvichak River watersheds, where Upper Talarik Creek, the North Fork Koktuli, and the South Fork Koktuli come together (Figure 3-1). This area represents the most likely site for near-term, large-scale mining development in the Bristol Bay watershed, but it also is similar to other sites in the area where mineral exploration is proceeding (Figure 4-6). This similarity means that much of our analysis is transferable to other portions of the region. Potential mine sites, salmon, wildlife, and culturally important resources exist throughout the watershed—thus, a mine operation at any one of these sites could have qualitatively similar impacts to a mine operation at the site of the Pebble deposit. However, we recognize that specific placement of mine facilities is the result of a complex evaluation process that considers many site conditions, and that future mines may locate mine components differently.

4.3.2 Mine Size

Any mine development would need to be sufficiently large to offset the significant development costs associated with the infrastructure needed for hard-rock mining in this roadless region, as roads, power supply, pipelines and export facilities would need to be built at substantial cost. If fully mined, the Pebble deposit may exceed 11 billion metric tons of ore (Ghaffari et al. 2011), which would make it the largest mine of its type in North America. In comparison, the largest porphyry copper mine in the United States (based on 2008 data) is the Safford Mine in Arizona, at 7.3 billion metric tons of ore; the largest in the world (based on 2008 data) is Chuquicamata Mine in Chile, at 21.3 billion metric tons of ore.

Figure 4-6. Mine Claims and Approximate Locations of Significant Mineral Deposits in the Nushagak River and Kvichak River Watersheds (ADNR 2012)



In our mine scenario, we have defined a minimum and a maximum mine size of 2 billion metric tons and 6.5 billion metric tons of ore, respectively (Tables 4-2 and 4-3) (Ghaffari et al 2011). The minimum mine size represents an interim stage of mine development (i.e., approximately 25 years after the start of mine operations), before all currently economically viable ore has been extracted. The maximum mine size represents the most likely mine to be developed in the watersheds at this time (Ghaffari et al. 2011). Other deposits in the Nushagak River and Kvichak River watersheds are unlikely to exceed 250 million metric tons individually (Singer et al. 2008), but, if mined, these sites cumulatively may exceed our hypothetical maximum mine size (Chapter 7).

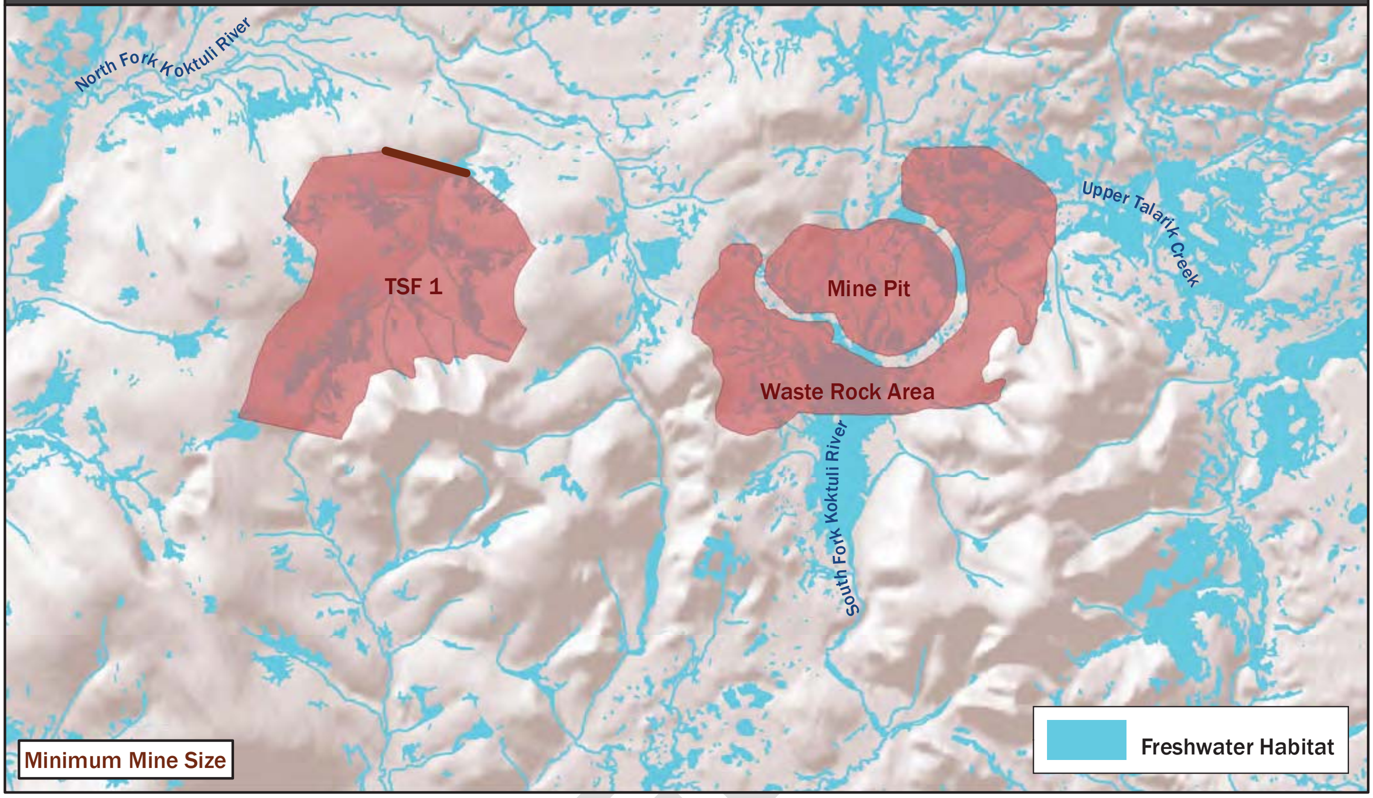
4.3.3 Mine Operations

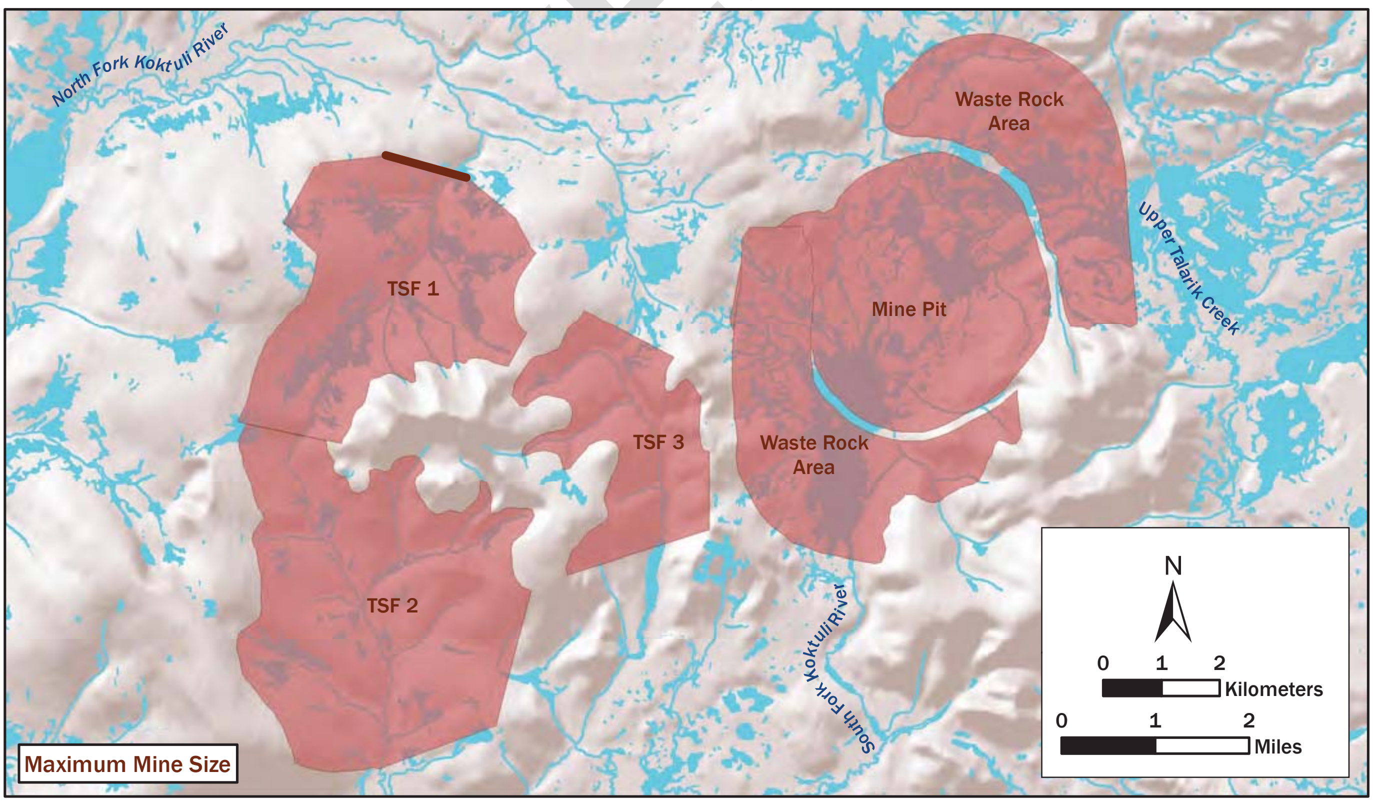
In this assessment, we assume the Pebble deposit is a porphyry copper ore body as described in Ghaffari et al. (2011). Based on standard mining practices, we assume that drill and blast methods would be used to excavate the rock, at a processing rate of approximately 200,000 metric tons/day for both the minimum and maximum mine sizes (Table 4-3). For the minimum mine size, we assume that an open pit method of excavation would be employed. Dimensions of the open pit are dictated by the size and shape of the ore deposit, and we estimate a pit with a surface area of 5.5 km^2 and depth of 800 m (Table 4-3, Figure 4-7). This hypothetical surface area and depth provide an approximate size of the open pit for an ore body of this size, although the dimensions of any specific mine could vary substantially from these numbers. For the maximum mine size, we assume mining operations would start with the open pit mine at the western portion of the deposit, following mining operations outlined in Ghaffari et al. (2011). The surface area of an open pit for the maximum mine size would be approximately 17.8 km^2, with a pit depth of 1.2 km (Table 4-3, Figure 4-7). If the operator develops an underground mine on the east side of the deposit, using block caving methods, the mine would initially occupy a smaller surface area, but subsidence would eventually increase the footprint to a larger size determined by the natural stable slope of the rock.

4.3.4 Ore Processing

In the mine scenario, an in-pit crusher would reduce the ore to a constant maximum size and a conveyor would bring the crushed ore to processing facilities (Figure 4-4). We assume ore would be processed in a flotation circuit as described in Section 4.2.2. Gold would be recovered from the pyrite fraction of the tailings in a secondary circuit (Figure 4-4). Pyritic tailings from this second circuit would be buried in the center of the TSF. The pyrite-rich tailings would be encapsulated in non-acid-generating tailings, with a water cap maintained in perpetuity to retard oxidation of sulfide minerals (Section 4.3.7).

Figure 4 7. Minimum and Maximum Footprints in the Assessment Scenario (Tables 4-2 and 4-3). Individual mine components are the mine pit, waste rock piles, and one or more tailings storage facilities (TSFs). The dark bar at the north end of TSF 1 indicates the dam for which tailings dam failure is modeled (Section 4.4.2).





4.3.5 Tailings Storage Facilities

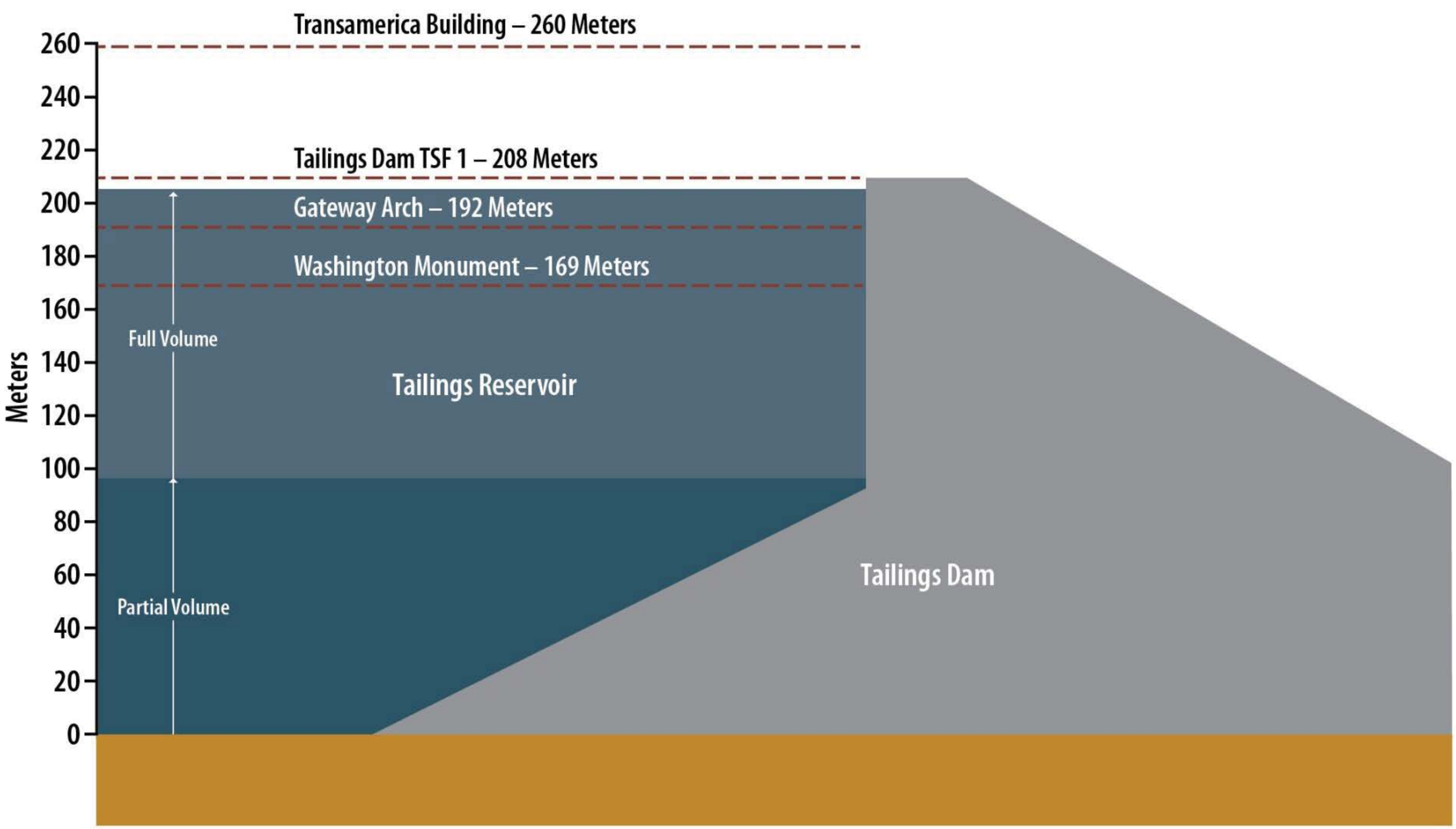
In our mine scenario, we assume that construction of dams at TSFs would proceed as described in Ghaffari et al. (2011), creating TSFs 1, 2, and 3 (Figure 4-7). The water rights application submitted by Northern Dynasty Minerals to the State of Alaska in 2006 described several potential locations for TSFs. We assume that the higher mountain valleys similar to the site of TSF 1, on the flanks of Kaskanak Mountain, are the most plausible sites given geotechnical, hydrologic, and environmental considerations. However, we do not imply a final determination that these sites are the least environmentally damaging practicable alternatives for purposes of Clean Water Act permitting. Permit-specific study, beyond the scope of this assessment, would go into determining if these or other sites met these criteria. Our purpose here is to analyze the risks of TSFs at sites we believe to be typical.

At each TSF, a rockfill starter dam would be constructed, with a liner on the upstream dam face and seepage capture and toe drain systems installed at the upstream toe, and with perpendicular drains installed to direct seepage toward collection ponds. The TSF would be unlined other than on the upstream dam face, and there would be no impermeable barrier constructed between tailings and underlying groundwater. As tailings accrued near the top of the starter dam, dam height would be raised using the downstream construction method (Figure 4-5) (Ghaffari et al. 2011). At some point, dam construction would shift to the centerline method (Figure 4-5), and a new stage would be constructed as the capacity of each previous stage is approached.

Given the low grade of ore expected in the region, our hypothetical mine would produce large amounts of tailings: approximately 99% of the mass of ore processed would be tailings, 85% as bulk tailings and 14% as pyritic tailings. Both types of tailings would be directed to TSFs (Figure 4-4). The discharge of bulk tailings would be managed such that the coarsest materials (fine sand) would be deposited at intervals along the inside perimeter of the TSF to form beaches, while finer materials (silt) would be carried with discharged water toward the center of the impoundment or tailings pond. Pyritic tailings would be discharged below the water surface of the tailings pond and encapsulated in NAG tailings to retard the rate of pyrite oxidation.

Our minimum mine size of 2 billion metric tons of ore is estimated to produce roughly 2 billion metric tons of tailings, requiring a dam at TSF 1 approximately 208 m high—much higher than most existing tailings dams (Section 4.2.3, Figure 4-8). The surface area covered by TSF 1 at full volume is estimated to be 14.9 km^2 (Table 4-3, Figure 4-7). Our maximum mine size would require the construction of TSFs 1, 2, and 3, with a combined tailings capacity exceeding 6.5 billion metric tons. We estimate that these three TSFs would have a combined surface area of 43.7 km^2 (Table 4-3).

Figure 4-8. Height of the Partial- and Full-Volume Dams at TSF 1, Relative to Common Landmarks



In a TSF, the low solubility of oxygen in water (less than 15 mg/L) limits the access of oxygen to unreacted sulfide minerals in the tailings, reducing dissolution reaction rates and thus the concentration of solutes. Furthermore, under anoxic conditions commonly encountered in sulfidic tailings, trace amount of carbonate or silicate minerals will partially neutralize acid, further limiting the solubility of metals and other trace elements (Blowes et al. 2003). At active mines, it is common practice to decant water from tailings ponds, treat it, and reuse the water in the mill. At the end of mining, it is expected that the composition of tailing pond water will be between the composition of local surface water and the water quality estimates produced by pre-mining humidity-cell test results resulting from the discontinuation of the introduction of process water (Box 4-1). The same is expected for the composition of any seepage from the base of the tailings impoundment, either during operation or after closure (Box 4-1, Appendix H). However, because the humidity cell tests used to predict pore water chemistry are a small sample of the ore body, water quality in the tailings impoundment may differ significantly from what is estimated (Appendix H). For example, the likely anoxic character and the opportunity to have its pH buffered by reactions with carbonate or silicate minerals would likely lead to lower concentrations of metals in the impoundment water than are seen in the humidity-cell tests.

A well field spanning the valley floor would be installed at the downstream base of the tailings dam to monitor groundwater flowing down the valley, including potential seepage from the TSF that was not captured by the seepage collection system. If seepage control requires the installation of collection wells to intercept groundwater, water from the well field would be either treated and released to the stream channel or pumped back into the TSF.

4.3.6 Waste Rock

In terms of surface area, waste rock piles would occupy approximately 13.3 km^2 under the minimum mine size and approximately 22.6 km^2 under our maximum mine size (Table 4-3). We assume that waste rock would be stored around the mine pit mostly within the cone of depression from mine pit dewatering (Figure 4-9), and that these piles would be constructed with a geometry designed to reduce the amount of runoff requiring treatment. Monitoring and recovery wells and seepage cutoff walls would be placed downstream of the piles to manage seepage, with seepage directed either into the mine pit or to collection ponds. Appendix H contains data on the potential composition of waste rock seepage. Stormwater would be pumped to runoff collection ponds and embankments would be constructed above seepage cutoff walls to contain any excess stormwater runoff.

PAG waste rock would be stored separately from NAG waste rock. As noted above, waste rock could be processed if commodity prices rose to the point where it was economical to process it, or if balancing the chemistry of the flotation process made this advantageous. Alternatively, PAG waste rock might be milled at the end of mining to both exploit the mineral content of the rock and to direct acid-generating pyrite to the TSF or the pit, where it might be more easily managed. Waste rock also might be placed back in the pit (e.g., waste rock from the eastern part of the ore body might be placed in the western portion of the pit once it is fully mined).

BOX 4-1. ESTIMATING THE GEOCHEMISTRY OF TAILINGS AND WASTE ROCK

The geochemistry and mineralogy of tailings from a porphyry copper deposit can be assessed through metallurgical testing. Ore is processed to remove a bulk sulfide concentrate that includes chalcopyrite, molybdenite, and pyrite—the three main sulfide minerals found in the Pebble deposit. The resulting tailings are characterized in terms of the acid-base accounting, mineralogy, bulk geochemistry, and metals leachability through humidity-cell tests. PLP tests of the Pebble deposit indicate that the tailings represent typical porphyry copper tailings.

The geochemistry of the tailings pond water and water that would seep from the base of the embankment is difficult to estimate but available data suggest a range and limits to constituents (PLP 2011). Likely sources of water for the tailings storage facilities (TSFs) would be process water from the flotation circuit (part of the slurry pumped to the TSF), local surface water, and precipitation. Likely sources of solutes would be process water from the mill, local surface water and groundwater, and geochemical interactions with the tailings solids.

The geochemistry of process water can be approximated by the average values reported for supernatant compositions from metallurgical testing using crushed drill-core samples (PLP 2011). Local surface water can be approximated by the mean composition of the North Fork of the Koktuli River (gage NK100A) (PLP 2011). The amount of solutes released by interactions with tailings can be approximated from humidity-cell tests on reject material (tailings) from previous metallurgical testing (PLP 2011). Here, we use solute concentrations based on average release rates for individual samples of tailings (PLP 2011). The chemical composition of the leachate can be calculated using weekly release rates as follows:

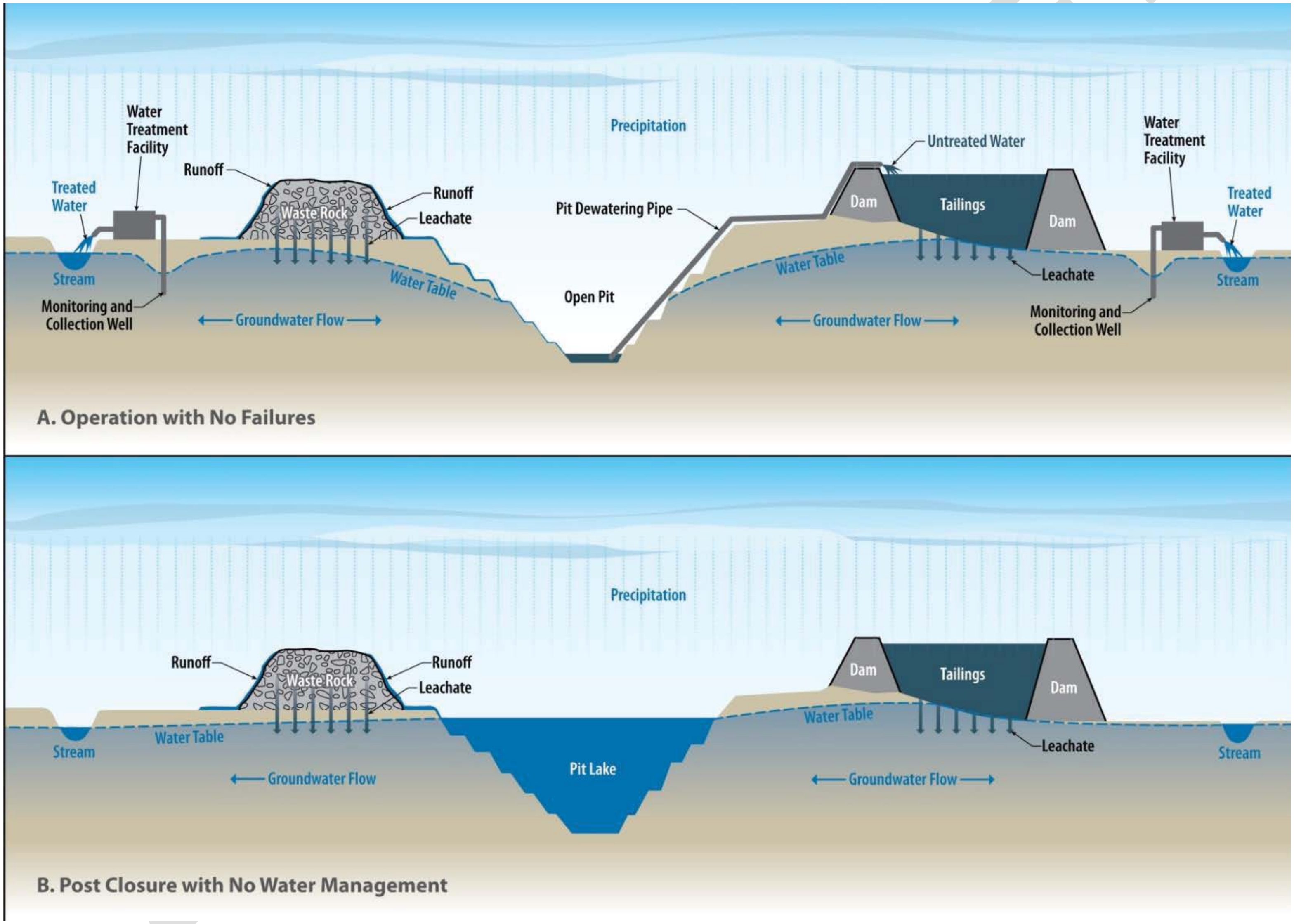
Concentration (mg/L) = [Release (mg/kg/week) x Mass of Sample (kg)]/Leachate Recovered (L/week).

These leachate compositions represent a worst-case scenario because the tests are conducted in an aerobic environment with unlimited access to atmospheric oxygen. The tailings used in the metallurgical testing are only approximations of actual tailings from an operating mine. Metallurgical testing uses composite samples of drill core, which may have been exposed to weathering in a core shack for extended periods of time, which may affect the surface properties of mineral grains. Tailings from an operating mine are the result of optimization of processes to maximize recovery of sulfide concentrates, and will likely have lower concentrations of copper and molybdenum.

The geochemistry of the seepage associated with both the potentially acid-generating (PAG) and non-acid-generating (NAG) waste rock piles is also a challenge to estimate, but available data suggest limits to its composition (PLP 2011). Primary sources of water would be local surface water and precipitation; primary sources of potential solutes would be local surface water and geochemical reactions with the waste rock. Humidity-cell tests conducted on the waste rock samples (PLP 2011) are more representative of site conditions than humidity-cell tests conducted on the tailings because the waste rock would be disposed where atmospheric oxygen would have access to the waste material. In addition, larger-scale barrel tests exposed samples to local climate conditions (i.e., temperature and precipitation variations). These tests should provide a better assessment of how the waste material would behave.

Subaqueous column tests were also performed. These tests would be most useful for assessing the efficacy of subaqueous disposal of waste rock, such as into a pit lake after mining has ceased. For the humidity-cell, subaqueous column, and barrel tests, the grain size of the test material is significantly smaller than that for waste rock at an operating mine. This difference provides more surface area per unit mass for reaction of the test materials, which should translate into higher concentrations of solutes in the test samples. In field settings at active mines, water flows through macropores and other preferential flowpaths through waste rock piles such that the entire surface of the waste material will not contact water, unlike typical conditions in humidity cells or barrels.

Figure 4-9. Simplified Schematic Illustrating Water Management and Movement at the Mine. Water movement and management is shown for two periods: A. routine operation, assuming no failures and B. post-closure, assuming no water management.



The geochemistry of plutonic rocks, sedimentary rocks, and volcanic rocks has been investigated by PLP. Data include acid-base accounting, bulk geochemistry, and leachability as generated by both standardized humidity-cell tests and larger volume on-site barrel tests under local climate conditions (Box 4-1, Appendix H). The ore-bearing Pre-Tertiary rocks investigated appear to represent typical hydrothermally altered rocks commonly found around porphyry copper deposits (Borden 2003). The non-ore-bearing Tertiary volcanic rocks were deposited after the hydrothermal activity in the Bristol Bay watershed and lack the acid-generating potential associated with hydrothermal sulfides. Because the Tertiary volcanic rocks were classified as NAG (PLP 2011), they may be useful for construction purposes such as building dams for tailing impoundments.

4.3.7 Water Management

In this section, we consider the major components of water movement at our hypothetical mine site (Figure 4-9). Development and operation of the mine would alter the natural flow of water within and from the mine site via several mechanisms.

- **Elimination of natural runoff and changes in infiltration resulting from the construction of mine components.** Uncontrolled runoff would be eliminated in any areas of the mine site in which the runoff could encounter areas disturbed by mining operations (e.g., the mine pit, waste rock piles, TSFs, ore processing facilities, or other mine infrastructure) or with materials that might degrade the water quality. In or immediately downstream of these areas, the mine operator would capture and collect surface runoff and either direct it to a storage location (e.g., a TSF or process water pond) or reuse or release it after testing and any necessary treatment.
- **Diversion of blocked streams upstream of the mine site.** If streams blocked by the mine pit or waste rock piles, or streams expected to dry up due to mine pit dewatering, have upstream reaches beyond the affected areas, water from these upstream reaches would be diverted around and downstream of the mine where practicable.
- **Capture of precipitation falling on the mine components.** Precipitation on the mine pit, waste rock piles, and TSFs would be collected and stored to use as process water, eliminating it as a source of stream recharge.
- **Extraction of groundwater from the mine pit and from leachate recovery wells.** Dewatering of the mine pit and groundwater extraction for leachate control (e.g., down-gradient of the waste rock piles and the tailings dams) would lower groundwater levels. Because many of the area's streams are fed from groundwater recharge, reductions in the groundwater level would reduce or eliminate the flow in streams draining the site.
- **Withdrawal of water for use in mine operations.** Many aspects of mine operations require water (e.g., power plant cooling, immersing tailings in TSFs, transporting product concentrates through pipelines). Similar to groundwater extraction, withdrawal of water for use in these processes would reduce or eliminate streamflows.

- **Testing and treatment of captured water prior to release.** Testing and treatment of captured water could change its chemical and thermal characteristics, potentially affecting streams even if all of that captured water was released. The physical movement of captured water—from collection points, through the water treatment plant, and to the points of effluent discharge—would likely alter flow quantities, rates, and timing, and redistribute water across the site both spatially and temporally.

The relative importance of each of these mechanisms, in terms of affecting the direction and magnitude of flow alterations, would vary with the stage of mine development. We consider three water management stages over the life of the mine: start-up, or the initial few years of mine operation as the mine becomes established; full operation for the minimum and maximum mine size; and post-closure (Section 4.3.8.4, Table 4-5). Developing a water balance for these stages is important to the assessment, because it determines the amount of water available at the site that could still contribute to downstream flows (Box 4-2). However, water balance development is challenging and requires a number of assumptions. It depends upon the amount of water needed to support mining operations, the amount of water delivered to the site via precipitation, the amount of water lost due to evapotranspiration, and the net balance of water to and from groundwater sources. Information exists to estimate precipitation and evapotranspiration, and estimates of water needed for mining operations are available based on typical mining practices (Ghaffari et al. 2011). More challenging, and potentially the largest source of uncertainty, is determining the net balance of water from groundwater sources.

Because the mining operation would always consume some water, there would always be less water available in the streams during active mining than there was before the mine was present. Major reductions in streamflow during mine operation would result from capture of precipitation falling on the mine pit, waste rock piles, and TSFs (Table 4-5). The mine pit would capture precipitation directly, but pit dewatering would also draw down the water table beyond the rim of the pit, creating a cone of depression that would extend underneath the waste rock piles (Figure 4-9). Leachate recovery wells downstream of the waste rock piles would extend the cone of depression (Figure 4-9). Because the mine pit would be located on a water divide, we estimate that there would be little net contribution from groundwater flow into the area defined by the cone of depression, and that the cone of depression would expand until water flow into the mine pit was balanced by recharge from precipitation over the cone of depression. The cone of depression would lower the groundwater table, drying up streams, ponds, and wetlands that depend on groundwater discharge and turning areas of groundwater discharge into areas of groundwater recharge. Water collected in the mine pit or from recovery wells would be pumped to a process water pond or to one of the TSFs. Water falling within the perimeter of a TSF would be captured directly in the TSF, but runoff from catchment areas up-gradient of the TSF would be diverted downstream. Some additional water would be collected as runoff at the port site and pumped to the mine site in the return water pipeline.

BOX 4-2. WATER BALANCE CALCULATIONS

To understand the impacts of water use in our mine scenario, we developed a water balance to account for major flows into and out of the mine area (Table 4-5). Our water balance does not account for flows within the mine site, but instead concentrates on changes in flows entering or leaving the mine site, relative to pre-mining conditions. These changes are divided into flows that would be withdrawn or captured from the natural system and flows that would be released to the natural system.

Captured flows include water captured at the mine site and at the TSFs (Table 4-5). The total amount of water captured at the mine site includes net precipitation (precipitation minus evapotranspiration) over the areas of the mine pit, the waste rock piles, and the cone of depression (without double-counting any areas of overlap). We estimated a net precipitation value of 803 mm/year and 804 mm/year at the mine site and TSFs, respectively. Areas of the mine pit, waste rock piles, and TSFs for the minimum and maximum sizes of the mine scenario are shown in Table 4-3. For water balance calculations, we also included a start-up phase of operations, during which the mine would be established. For start-up we assume a 5.8-km^2 waste rock pile and no contribution from the cone of depression—that is, the entire waste rock area under the minimum mine size has not yet been disturbed and the drawdown zone has not yet been created.

The flow of groundwater seeping into the mine pit was calculated using the Dupuit-Forcheimer discharge formula for steady-state radial flow into a fully penetrating well with a diameter equal to the average mine pit diameter. The hydraulic conductivity data gathered in the area of the mine during geologic investigations show significant scatter. We based our analysis on the hydraulic conductivity (k) varying with depth, with log k varying linearly from the surface to a depth of 200 m; specifically, with $k = 1 \times 10^{-4}$ m/s at the surface and $k = 1 \times 10^{-8}$ m/s at depths greater than or equal to 200 m. Given these values, negligible flow occurs below a depth of 200 m, so our model included a no-flow boundary at that depth. To apply the Dupuit-Forcheimer formula, we needed to transform the cross-section into an equivalent isotropic section by transforming the vertical dimension, so that the thickness at any depth was proportional to the hydraulic conductivity at that depth. The initial water table was at the ground surface, which was assumed to be horizontal in our simplified model. Our analysis assumed that the drawdown at the mine pit was 100 m, but we also verified that the results were not very sensitive to this assumption. The radius of influence was determined by balancing the net precipitation falling within the cone of depression with the calculated flow into the mine pit. Inflows were calculated to be 0.52 m^3/s (8,210 gpm) and 1.06 m^3/s (16,828 gpm) for the minimum and maximum mine sizes. The minimum mine inflow agrees closely with the estimate provided in Ghaffari et al. (2011). The cone of depression was determined to extend 1,222 m and 1,260 m from the edge of the idealized circular mine pit under the minimum and maximum mine sizes, respectively. In a geographic information system (GIS), we established the boundary of the cone of depression at those distances from the actual perimeter of the minimum and maximum sizes of mine pits.

All of the captured flows would be available for use by the mine operator. The summary of captured flows does not attempt to account for every possible or minor flow. For example, it does not include water from the portions of blocked streams that lie beyond the limits of the mine pit, waste rock piles, or drawdown cone of depression because our mine scenario calls for the diversion of this water around the mine site and back into the streams, where practicable. We also have not calculated flows from precipitation falling on the mill, other smaller facilities, or roads. To estimate the amount of water available for release, we subtracted consumptive losses associated with mining activities from the captured flows (Table 4-5). Consumptive losses would include water pumped to the port in the copper (+gold) concentrate pipeline, cooling tower evaporation and drift losses, interstitial water trapped in the pores of stored tailings, and water stored in the mine pit after closure. About 95% of the consumptive loss during mine operations would be the tailings pore water. When the tailings settle, about 46% of the volume would consist of voids between the solid particles. The water trapped in these pore spaces would no longer be available for use at the mine or release to streams.

Information on the flows in the concentrate and return water pipelines and on the cooling tower losses appears in Ghaffari et al. (2011). We also increased the amount of water available by flows brought onto the mine site, including water returned from the port (e.g., from dewatering the copper (+gold) concentrate and from stormwater runoff collected at the port site). We estimated the area of the port facilities over which runoff was likely to be collected (137,160 m^2) and multiplied the area by precipitation rate at the port (1,830 mm/year) to determine the contribution from port site runoff.

When the amount of withdrawn water exceeds the consumptive losses, water would be available, after testing and treatment, for release into area streams. This reintroduced water may differ from the baseline water in chemistry and temperature, and may be reintroduced at locations, flow rates, or times of year that differ from baseline conditions.

During start-up (the first few years of operation, as the mine is becoming established), we expect that the mine would require approximately 20% more water than would be captured by precipitation and groundwater (Table 4-5), with the difference being withdrawn from water stored in the TSF before active mining began. During mine operation, groundwater and precipitation would be pumped from the open pit to prevent flooding of the mine workings (Figure 4-9). Water would be needed for the flotation mill, to operate the TSF, and to maintain a concentrated slurry in the product pipeline (Section 4.3.8).

This captured water also would be available for mine operations. In hard rock metal mining, most water use occurs during milling and separation operations. However, much of this water is recycled and reused. For example, much of the water used to pump the tailings slurry from the mill to a TSF becomes available when the tailings solids settle, and excess overlying water is pumped back to the mill. Water losses occur when there is a consumptive use, and water is no longer available for reuse. In our mine scenario, consumptive uses would include cooling tower losses through drift and evaporation, water in the concentrate sent to the port that would exceed the amount recovered and returned in the return water line, and water that would fill the pore spaces in the TSF (Table 4-5). The TSF pore water accounts for about 95% of the mine operations water demand (Table 4-5). Consumptive losses would be made up by withdrawing water stored in a TSF or by pumping directly from the mine pit.

As the area of water capture expands (e.g., via the drawdown area, additional waste rock piles and TSFs, and pit expansion as we increase from the minimum to the maximum mine size), some of this captured water (16 to 63%, depending on the stage of water management) would not be needed at the mine site (Figure 4-9, Table 4-5). Assuming no water collection and treatment failures, this excess captured water would be treated to meet existing water quality standards and discharged to nearby streams, partially mitigating flow lost from eliminated or blocked upstream reaches.

Table 4-5. Water Balance Estimates for the Mine Scenario

Water Balance Component	Water Management Stage (10^6 m^3/year)			
	Start-Up	Operations: Minimum Mine (25 years)	Operations: Maximum Mine (78 years)	Post-Closure
Captured at mine site	10.5	20.2	41.2	41.2
Captured at TSF 1	12.0	12.0	12.0	12.0
Captured at TSF 2	0.0	0.0	17.0	17.0
Captured at TSF 3	0.0	0.0	6.1	6.1
Total captured	**22.4**	**32.2**	**76.3**	**76.3**
Cooling tower losses	1.3	1.3	1.3	0.0
Contained in concentrate to port	1.4	1.4	1.4	0.0
Contained in concentrate return	- 0.9	- 0.9	- 0.9	0.0
Runoff collected from port	- 0.3	- 0.3	- 0.3	0.0
Stored in TSFs as pore water	25.5	25.5	26.5	0.0
Stored in mine pit	0.0	0.0	0.0	41.2
Total consumed	**27.0**	**27.0**	**28.0**	**41.2**
Total reintroduced	**- 4.5**	**5.2**	**48.2**	**35.1**
% Reintroduced	**- 20**	**16**	**63**	**46**

Notes:
TSF = tailings storage facility

4.3.8 Post-Closure Site Management

Our assessment includes consideration of potential impacts from mine operations and potential impacts after mining activities have ceased, either as planned or prematurely (Table 4-2, Section 4.3.8). We assume that the mine would be closed after all currently identified economically profitable ore is removed from the site, leaving behind the mine pit, waste rock piles and TSFs. Water leaving the site via surface runoff or through groundwater would require capture and treatment for as long as it does not meet water quality standards. Weathering of the waste rock and pit walls would release contaminant concentrations of potential concern such as sulfates and metals. Weathering to the point where these contaminants are present in only trace amounts (at levels approaching their pre-mining background concentrations) would likely take hundreds to thousands of years, resulting in a need for management of materials and leachate over that time. We assume that, as part of post-closure operations, the existing seepage collection and treatment system would be maintained to capture and treat potentially toxic runoff and groundwater originating from the remaining facilities.

Such a seepage collection and treatment system might need to be maintained for hundreds to thousands of years. There are no examples of such successful, long-term collection and treatment systems for mines, because these time periods exceed the lifespan of most past large-scale mining activities, as well as most human institutions. Throughout this section, we refer to the need for treatment for extended periods of time. The uncertainty that human institutions have the stability to apply treatment for these timeframes applies to all treatment options.

4.3.8.1 Mine Pit

Because the pit would be the lowest point in the landscape by hundreds of meters, and water would need to be pumped from the mine while the mine is in operation, a cone of depression would be created in the landscape surrounding the pit that would persist for some time (Section 4.3.7, Figure 4-9). We assume that at closure the dewatering pumps in the pit would be turned off. Groundwater would continue to flow toward the pit in response to the local gradient. We estimate that the mine pit would take approximately 100 to 300 years to fill. Areas within the cone of depression that were groundwater discharge areas prior to the construction of the pit would continue to be groundwater recharge areas as the pit fills with water. Streams, ponds, and wetlands that depend on groundwater discharge would continue to be deprived of this source of water while the pit is filling. Surface flows upslope of the pit would also drain to the pit. Eventually water in the pit would reach equilibrium with surrounding groundwater, and pit water would flow into the groundwater system where the hydraulic gradient allows. Much of this groundwater would eventually discharge to down-gradient streams, ponds, and wetlands.

At least portions of the pit wall would consist of mineralized rock that was not economical to mine. These areas containing sulfide minerals would likely be acid-generating for as long as they remained above the water surface in the pit (if they were not sealed against oxidation), resulting in low-pH water running down the sides of the pit into the water body at the bottom. Oxidation of rocks exposed to air on the pit bottom or on side benches would also produce acidic metal-sulfate salts where sulfide minerals were present, which would create acid upon exposure to water, and carry dissolved metals. As the water level in the pit rises, the pit walls would become submerged and exposure to oxygen would be reduced. Eventually, acid generation would be expected to cease from rocks below the water's oxic zone. Exposed rock above the water surface or within the oxic zone would continue to produce acidic metal-sulfate salts that would run into the pit lake with precipitation and snowmelt. . However, predicting pit water quality has a high degree of uncertainty (Section 6.3.3).

4.3.8.2 Tailings Storage Facilities

We assume that water in the TSFs would be drawn down to prevent flooding, but that a small pond would be left to keep the core of the tailings hydrated and isolated from oxidation. Sulfide-rich materials that would generate acid if exposed to oxygen would have been placed in the core of the tailings impoundment. As long as a stagnant cover of water is maintained, oxygen movement into the tailings would be retarded, minimizing acid generation. Drawing down the level of water in the TSF would also provide capacity for unusual precipitation events, reducing the likelihood that a storm would provide enough precipitation to overwhelm capacity and cause tailings dam failure or overtopping. We assume that some NAG waste rock and a layer of soil would cover the tailings beaches and that they would be revegetated with native vegetation.

TSFs would require active management for hundreds to thousands of years. The tailings dam is an engineered structure that would require monitoring to ensure structural and operational integrity. An assumption in the mining industry is that tailings continue to compact, expelling interstitial water and becoming more stable over time. However, a recent analysis of data from oil sands tailings suggests that densification of tailings may stop after a period of time (Wells 2011). Thus, the system may require continued monitoring to ensure hydraulic and physical integrity. Interstitial water within the tailings would continue to seep into naturally fractured bedrock below the TSF. If, during operation, a well field was required for groundwater collection and treatment below the TSF, it would require continued operation in perpetuity or until the groundwater met regulatory requirements.

Retaining water in the tailings maintains a higher risk of tailings dam failure than if the tailings were drained. On the other hand, draining the tailings to stabilize them could allow sufficient oxygen-rich water to percolate through the tailings and allow oxidation of sulfides. An alternate approach to closure would separate pyritic tailings from bulk tailings. This would likely mean that pyritic tailings would be placed in the mine pit or shipped off site. Bulk tailings, which are not expected to be acid-producing, would then be drained and sloped at closure so that tailings could not flow down the valley in the event of a tailings dam failure.

4.3.8.3 Waste Rock

We assume that NAG waste rock would be sloped to a stable angle (less than 15%) (Blight and Fourie 2003), covered with soil/plant-growth media, and revegetated. At least some of the NAG waste rock would be placed on sand beaches around the TSF to retard wave-induced erosion during unusual precipitation events, when impoundment water levels rise to the level of the beach. However, at least some of the waste rock also could be placed in the mine pit. PAG waste rock would be processed through the flotation mill prior to mine closure, with tailings placed into the TSF or the mine pit. No PAG waste rock would remain on the surface. NAG materials would remain within the mine pit's cone of groundwater depression, so runoff from the waste pile that recharged groundwater would move into the pit for the time it took to fill the pit (approximately 100 to 300 years). Water that ran off as surface water would move downslope to the nearest surface water body.

4.3.8.4 Water Management

In the post-closure phase, water losses to stream systems would increase because the water filling the mine pit now acts as an additional consumptive loss (Table 4-5). Although operating consumptive losses from the cooling tower, concentrate transport, and TSF pore water would cease, annual flow into the pit would be about 50% greater than the annual consumptive losses during operations (Table 4-5). Water from precipitation falling on the TSFs, runoff from any of the former plant facility areas, and any water captured by leachate collection systems would be treated (until treatment was no longer necessary) and released. If this water was diverted to the mine pit instead, it would decrease the time necessary for the pit to reach equilibrium but would further reduce the amount of captured water released to streams.

4.3.8.5 Premature Closure

Many mines close before their ore reserves are exhausted. In one study of international mine closures between 1981 and 2009, 75% of the mines considered were closed before the mine plan was fully implemented (Laurence 2011). The Illinois Creek and Nixon Fork mines are examples of mines that closed prematurely in Alaska (although Nixon Fork has since reopened).

Closure before originally planned—or premature closure—may occur for many reasons, including technical issues, project funding, deteriorating markets, operational issues, and strategic financial issues of the owner. Premature closures can range from cessation of mining with continued monitoring of the site to complete abandonment of the site. As a result, environmental conditions at a prematurely closed mine may be equivalent to those under a planned closure, may require designation as a Superfund site, or may fall anywhere between these extremes. Environmental impacts associated with premature closure may be more significant than those associated with planned closure, as mine facilities may not be at the end condition anticipated in the closure plan and there may be uncertainty about future reopening of the mine. For example, PAG waste rock in our mine scenario would likely still be on the surface in the event of a premature mine closure. If the mine closed because of a drop in commodity price, there would be little incentive to incur the cost of moving or processing hundreds of millions of metric tons of PAG waste rock. Because premature closure is an unanticipated event, water treatment systems would likely be insufficient to treat the excessive and persistent volume of low pH water containing high metal concentrations.

When a mine reopens after premature closure, the owners may change the mining plan, may not implement the same mitigation practices, or may negotiate new effluent permits. For example, the

Gibraltar copper mine in British Columbia was permitted as a zero-discharge operation. When it closed, then reopened under new ownership, it was permitted to allow effluent discharge to the Fraser River, and this permit included a 92-m dilution zone for copper and other metals.

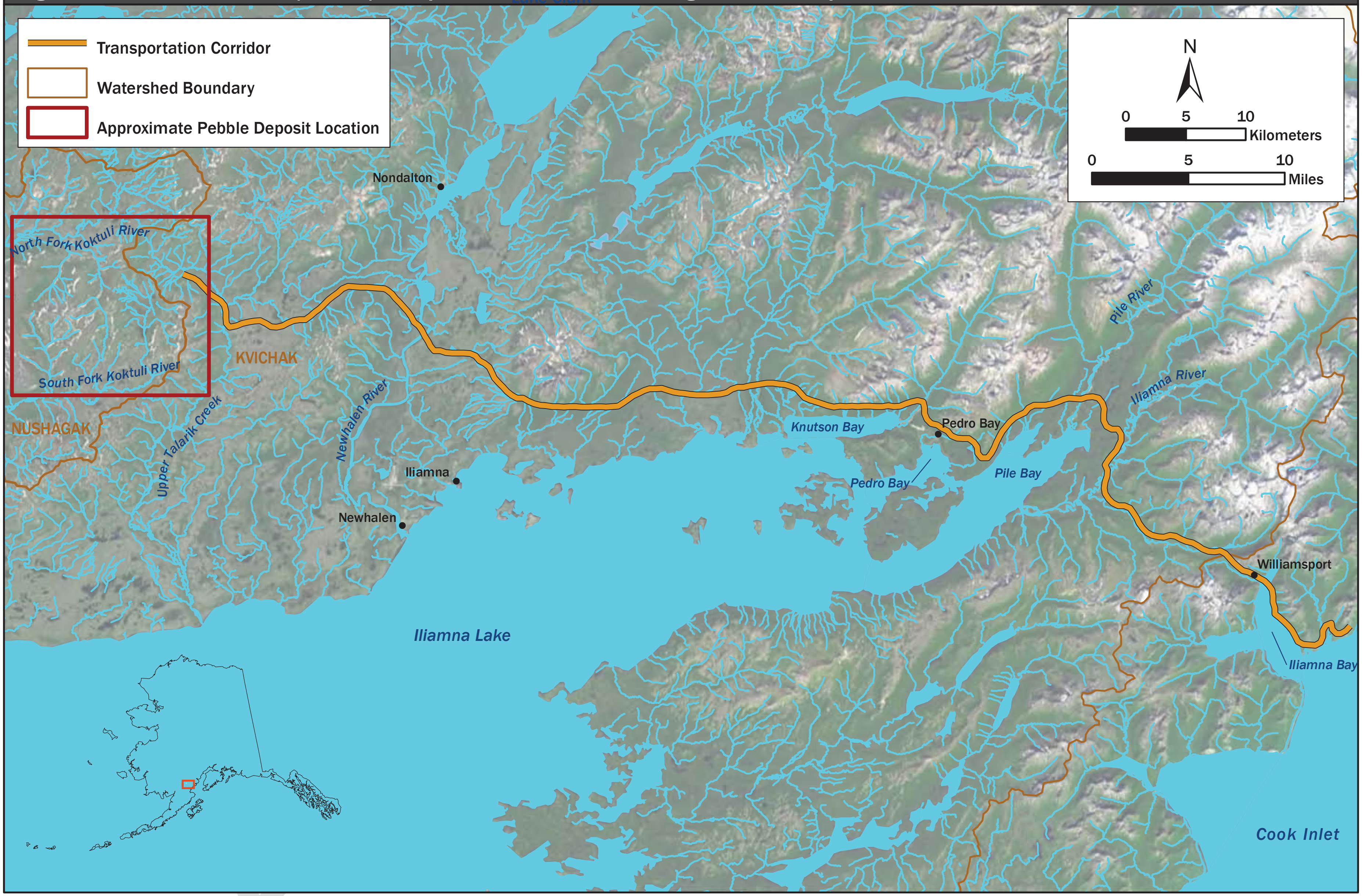
4.3.9 Transportation Corridor

4.3.9.1 Roads

Development of any mine in the Bristol Bay watershed would require substantial expansion and improvement of the region's transportation infrastructure. The Bristol Bay watershed is located in one of the last remaining, virtually roadless regions in the United States. There are no improved federal or state highways, and no railroads, pipelines, or other major industrial transportation infrastructure (Appendix G). Currently, the transportation system in the Bristol Bay watershed is limited to airstrips at each village and four short road segments, primarily between adjacent villages. Most people travel by air or boat during the ice-free season, and air or snow machine in winter.

In our mine scenario, a 139-km (86-mile) two-lane (30-foot-wide), gravel surface, all-weather permanent access road (Figure 4-10) would connect the mine site to a new deepwater port on Cook Inlet, from which concentrate would be shipped elsewhere for processing (Ghaffari et al. 2011). An estimated 118 km of this corridor would fall within the Kvichak River watershed. The primary purpose of this road would be to transport freight by conventional highway tractors and trailers, although critical design elements would be dictated by specific oversize and overweight loads associated with project construction. Material sources for road embankment fill, road topping, and riprap would be available at regular intervals along the road route, and we assume standard practices for design, construction, and operation of the road infrastructure, including design of bridges and culverts for fish passage. Costs for the road would include daily maintenance crew and equipment; crushed road topping every 5 years; culvert, embankment, riprap, guardrail and river training structures; regular bridge and other inspections; dust suppression; snow removal; and avalanche control and removal (Ghaffari et al 2011). Permanent structures would be designed for a service life of 50 years. Because the access road would be kept open permanently for ongoing care, maintenance, and environmental monitoring at the site after mine closure, maintenance and periodic replacement in perpetuity would be required.

Figure 4-10. Potential 139-km (86-mile) Transportation Corridor Connecting the Pebble Deposit Area to Cook Inlet



The proposed transportation corridor would cross many streams, rivers, wetlands, and extensive areas with shallow groundwater, including numerous mapped (and likely many more unmapped) tributary streams to Iliamna Lake (Figure 4-10). Approximately 20 bridges would be constructed over larger, anadromous streams and rivers along the length of the corridor, with spans ranging from approximately 12 to 183 m. At the remaining road-surface water crossings (i.e., at the majority of these crossings), culverts would be installed. In addition, there would be a 573-m (1,880-foot) causeway across the upper end of Iliamna Bay, and approximately 8 km of embankment construction along coastal sections in Iliamna Bay and Iniskin Bay (Ghaffari et al. 2011).

Topographically, moving from west to east, the transportation corridor would cross the Newhalen River and parallel the north shore of Iliamna Lake. It would cross rolling, glaciated terrain for approximately 97 road km until reaching steeper hillsides west of the village of Pedro Bay. After crossing gentler terrain around the northeast end of Iliamna Lake, the corridor would cross the Aleutian Range (the highest source of runoff in the Bristol Bay watershed) along the route of the existing Pile Bay Road to tidewater at Williamsport. From there it would cross Iliamna Bay and follow the coastline to the port site on Iniskin Bay, off Cook Inlet. Highly variable terrain and variable subsurface soil conditions, including extensive areas of rock excavation in steep mountainous terrain, are expected over this proposed route.

Avalanche hazards exist in isolated locations along the alignment, but routing would attempt to avoid any avalanche chutes and runout areas. Because of the steep mountain slopes and lack of significant vegetation at high elevations, storm runoff can rapidly accumulate and result in intense local runoff conditions. Road areas near the south slope of Knutson Mountain and the southeast slope of the mountain above Lonesome Bay and Pile Bay may be especially susceptible to these runoff events. In 2004, runoff from a storm washed out several culverts on the state-maintained Pile Bay Road.

4.3.9.2 Pipelines

The transportation corridor would include four pipelines, which would carry copper (+gold) concentrate, return water, natural gas (to fuel a natural gas-fired generating plant), and diesel fuel between the mine site and the Cook Inlet port (Table 4-6). Except at stream and river crossings, pipelines would be buried together in a trench adjacent to the road alignment, in the right-of-way. At short stream and river crossings, pipeline channels would be bored under channels to minimize waterway impacts. At longer crossings, pipelines would be supported aboveground on road bridges. Any aboveground pipeline sections would be constructed of double-walled pipe. Freeze protection would be provided by insulation (aboveground pipes) or burial (5 feet below ground surface). External corrosion would be prevented by a cathodic protection system. A leak detection system would be built into the pipelines, which would also assist in the detection and prevention of slack flows. A supervisory control and data acquisition (SCADA) system would monitor and control pumping facilities via a fiber optic line buried alongside the pipelines. Instruments such as pressure and temperature transducers located along the pipeline route would be tied into the fiber optic link.

Table 4-6. Characteristics of Pipelines in the Mine Scenario

Pipeline (# of pipes)	Route	Pipe Material	Diameter (cm)
Along Transportation Corridor			
Copper-(+gold) concentrate (1)	Mine to port	HDPE lined steel	20.3
Reclaimed water (1)	Port to mine	HDPE lined steel	17.8
Natural gas (1)	Port to mine	Steel	5. 0
Diesel fuel (1)	Port to mine	Steel	14.1 (OD)
At Mine Site			
Bulk tailings (2)	Process plant to TSF	Steel with liner	86
Pyritic tailings (2)	Process plant to TSF	Steel with liner	46
Reclaim water (1)	TSF 3arge to TSF head tank	HDPE	107
Reclaim water (1)	TSF head tank to process pond	Steel	107
Mine pit dewatering (1)	Pit to process pond or TSF	Steel	TBD

Notes:
HDPE = high density polyethylene; OD = outside diameter; TSF = tailings storage facility; TBD = to be determined
Source: Ghaffari et al. 2011

On the mine site itself, pipelines would carry tailings slurry from the process plant to the TSF, and reclaimed water from the TSF to the process plant (Table 4-6). In addition to these major on-site pipelines, there would be smaller pipelines for water supply, firefighting, and process flows within the plant. In this assessment, we assume that any leakage from pipelines in the process plant area would be captured and controlled by the plant's drainage system and either be treated prior to discharge or pumped to the process water pond or the TSF. Failures of these on-site pipelines could result in uncontrolled releases within the mine site, but these failures are not evaluated in this assessment.

At mine closure, concentrate and return water pipelines would be removed. Diesel and natural gas pipelines would be retained as long as fuel was needed at the site for monitoring, treatment, and site maintenance. It is also possible that local communities would select to retain the pipelines for continued use.

4.4 Mine Scenario: Failure

Our mine scenario assumes that engineering controls would be designed to capture and treat all surface and groundwater runoff from the site, and that no discharges would exceed existing water quality standards. However, human-engineered systems are imperfect: based on the experience of most large engineering projects, accidents and failures are likely to occur over the decades that a mine is in operation, and over the centuries that a TSF remains in the post-closure period and requires maintenance and monitoring. The potential for accidents and failures resulting from earthquakes may be of particular concern in our mine scenario, given that southwestern Alaska is a seismically active region (Box 4-3).

BOX 4-3. THE SEISMIC ENVIRONMENT OF BRISTOL BAY

The Alaska Earthquake Information Center (AEIC) and U.S. Geological Survey (USGS) collect data on earthquakes occurring in Alaska at seismological monitoring stations throughout the state. Earthquakes in Alaska range from minor events that are only detected by sensitive instruments, to the largest earthquake ever recorded in North America (the 1964 Anchorage earthquake, magnitude 9.2) (Table 4-7, Figure 4-11). The size of an earthquake is directly related to the length of the fault on which it occurs, with longer faults producing larger earthquakes. The damage caused by an earthquake is related to size of and distance from the earthquake. The effects of an earthquake diminish with distance, so more damage occurs at the epicenter than at a point several kilometers away. Southwest Alaska experiences a large number of earthquakes related to the numerous faults in the region. These faults are, from north to south, the Tintina-Kaltag Fault, the Iditarod-Nixon Fork Faults, the Denali-Farewell Fault, the Lake Clark-Castle Mountain Fault system, the Bruin Bay Fault, and the Border Ranges Fault. The Lake Clark-Castle Mountain Fault system, with an estimated length of 225 km, is located nearest to the Pebble deposit, and would likely have the most significant effect on the seismicity in the mine area.

The northeast-southwest trending Lake Clark Fault is the western extension of the Castle Mountain Fault (Koehler and Reger 2011). The western terminus of the Lake Clark Fault was originally interpreted to be near the western edge of Lake Clark, but more recent studies by USGS reinterpreted the position of the Lake Clark Fault further to the northwest, potentially bringing it as close as 16 km to the Pebble deposit (Haeussler and Saltus 2004). Haeussler and Saltus (2004) acknowledge that the fault could extend closer than 16 km, but data are not available to support this interpretation. USGS has concluded that there is no evidence for fault activity or seismic hazard associated with the Lake Clark Fault in the past 1.8 million years, and no evidence of movement along the fault northeast of the Pebble deposit since the last glaciations 11,000 to 12,000 years ago (Haeussler and Waythomas 2011). Recently, the Alaska Division of Geological and Geophysical Surveys and USGS investigated reports of a surface geological feature (the Braid Scarp) near the Pebble deposit that was reported to be a fault scarp, indicating recent movement of a fault (Koehler and Reger 2011, Haeussler and Waythomas 2011). Both agencies independently determined that the feature was a relic of glacial activity and did not represent evidence of recent faulting.

The 1980 USGS map of the structural geology of the Iliamna Lake quadrangle shows several mapped faults in the Tertiary-age volcanic rocks that host the area’s mineral deposits. Geologic mapping conducted by consulting firms for the Pebble Limited Partnership (PLP) identified numerous faults in the area of the Pebble deposit. The mapped faults shown in both these sources are all considerably shorter than the Lake Clark Fault, and therefore by themselves have a very limited capability to produce damaging earthquakes. The largest mapped fault in the Pebble deposit area is an unnamed northwest-trending fault approximately 13 km southwest of the deposit, approximately 16 km in length. There are several short (less than 4 km) faults mapped within and near the mine site (the Z-series faults), about half of which have northeast-southwest orientations. The faults show vertical displacement ranging from tens of meters to over 900 m, and are interpreted to have formed coincident with mineralization (Ghaffari et al. 2011). Although there is no current evidence that the Lake Clark Fault extends closer than 16 km from the Pebble deposit, and there is no evidence of a continuous link between the Lake Clark Fault and the northeast trending faults at the mine site, mapping the extent of subsurface faults over long, remote distances is difficult and has a high level of uncertainty.

Not all earthquakes occur along the mapped sections of faults. In some instances, stresses build up and cause earthquakes in rock outside of pre-existing faults, or along deeper faults that are not exposed at the surface or that are associated with faults identified by geophysical methods. While these “floating earthquakes” are generally smaller and less frequent than those associated with faults, they may occur at locations closer to critical structures than the nearest mapped capable fault. Small earthquakes can be induced when reservoirs or impoundments are constructed (Kisslinger 1976), altering the soil and rock stresses and increasing pore pressure along pre-existing zones of weakness. Induced earthquakes are generally small, but can occur frequently and cause landslides and structural damage to earthen structures.

Interpreting the seismicity in the Bristol Bay area is difficult because of the remoteness of the area for study, lack of historical records on seismicity, and complex bedrock geology that is overlain by multiple episodes of glacial activity. Thus, there is a high degree of uncertainty in determining the location and extent of faults, their capability to produce earthquakes, whether these or other geologic features have been the source of past earthquakes, and whether they have a realistic potential for producing future earthquakes. Large earthquakes have return periods of hundreds to thousands of years, so there may be no recorded or anecdotal evidence of the largest earthquakes on which to base future predictions. While geologic analyses and field studies of existing faults can provide evidence of surface rupture and bounding estimates of the age of movement, these data are not unique and are subject to many uncertainties.

In this section we consider four types of accidents or failures: (1) water collection and treatment failure, (2) tailings dam failure (3) pipeline failure, and (4) road and culvert failures. Each of these accidents is described in greater detail below.

4.4.1 Water Collection and Treatment Failure

Failure to properly collect and treat leachate from waste rock piles, TSFs, or other areas of the mine site may allow potentially toxic chemicals, soils, and particulate matter to enter streams. Here, we consider the failure of on-site collection and storage practices at TSF 1 as an example case. Based on the available data, estimation of potential flow through the substrate located under and around proposed TSFs requires several assumptions. The depth and hydraulic conductivity of the substrate material located near possible dam sites varies greatly. In addition, the presence of fractured bedrock allows for localized discontinuities in the rate of groundwater movement that can greatly influence overall groundwater conveyance. We assume that a reasonable hydraulic conductivity for the area at TSF 1 would be 1.45×10^{-6} m/s, and that the average depth of the permeable substrate layer would be 30 m (approximately 100 feet) (Ghaffari et al. 2011).

To estimate potential water flow under the tailings dam, we completed a simple flow net calculation by summing a "net" of different flows at different depths under the dam. The liner on the upstream face of the dam and the bedrock below 30 m were both considered impervious. This allowed the development of a flow net composed of equally proportioned grids from the unlined impoundment area behind the dam to the open valley floor located below the dam. When TSF 1 is partially full, we assume dam height would be 98 m (Section 4.4.2.4). The dam would be 575 m in cross-section (i.e., along the flowpath) and would create a flow net 100 m wide at the downstream face of the dam itself. When TSF 1 is completely full (after approximately 25 years), we assume dam height to be 208 m (Section 4.4.2.4). The dam would be 799 m in cross-section and create the same 100-m-wide flow net outlet area at the downstream face of the dam. It is possible that a larger flow net width could exist along the valley walls at the intersection of the dam construction, which would increase the estimate below. However, in this assessment it is assumed that the valley walls are impervious and seepage flows would conform to the basic valley topography and be expressed in a concentrated area along the existing surface flowpath. With a dam height of 98 m, estimated flow rate at the downstream face of the tailings dam would be 8.14×10^{-4} m^3/s; with a dam height of 208 m, estimated flow rate was 1.15×10^{-3} m^3/s.

These estimates are based on a simple and conservative assessment of seepage from the TSF. Actual hydraulic conductivity would likely span several orders of magnitude. Even a small number of flowpaths, with higher than expected hydraulic conductivity, could significantly affect the direction and quantity of flow. This pertains primarily to estimates of flow beneath the TSF 1 tailings dam, but would also apply to tailings leachate escaping the TSF in any direction.

4.4.2 Tailings Dam Failures

A tailings dam failure occurs when a tailings dam loses its structural integrity and releases tailings material from the tailings impoundment. The released tailings flow under the force of gravity as a fast-moving flood containing a dense mixture of solids and liquids, often with catastrophic results. This flood can contain several million cubic meters of material that can travel at speeds in excess of 60 km/hour (37 miles/hour). At dam heights ranging from 5 to 50 m—substantially less than the 98 m and 208 m tailings dam failures considered here (Section 4.3.5)—the flood wave can travel many kilometers over land and more than 100 km along waterways (Rico et al. 2008). There are many international examples of such failures (Box 4-4).

4.4.2.1 Causes of Tailings Dam Failures

Causes of tailings dam failure are similar to those for earthfill and rockfill water retention dams, and include the following circumstances.

- **Overtopping.** Overtopping occurs when insufficient freeboard is maintained and the water level behind the dam rises as a result of heavy rainfall, rapid snowmelt, flooding, or operator error.
- **Slope instability.** These failures occur when shear stresses in the dam exceed the shear resistance of the dam material, most frequently resulting in a rotational or sliding failure of a portion of the downstream slope, leading to overtopping or breaching of the dam.
- **Earthquake.** Shaking resulting from earthquakes (Table 4-7, Figure 4-11, Box 4-5) causes additional shear forces on the dam that can lead to a slope instability failure.
- **Foundation failure.** Weak soil or rock layers and high pore pressures below the base of the dam can lead to shear failures in the foundation, causing entire dams to slide forward or rotate out of position.

BOX 4-4. EXAMPLES OF HISTORICAL TAILINGS DAM FAILURES

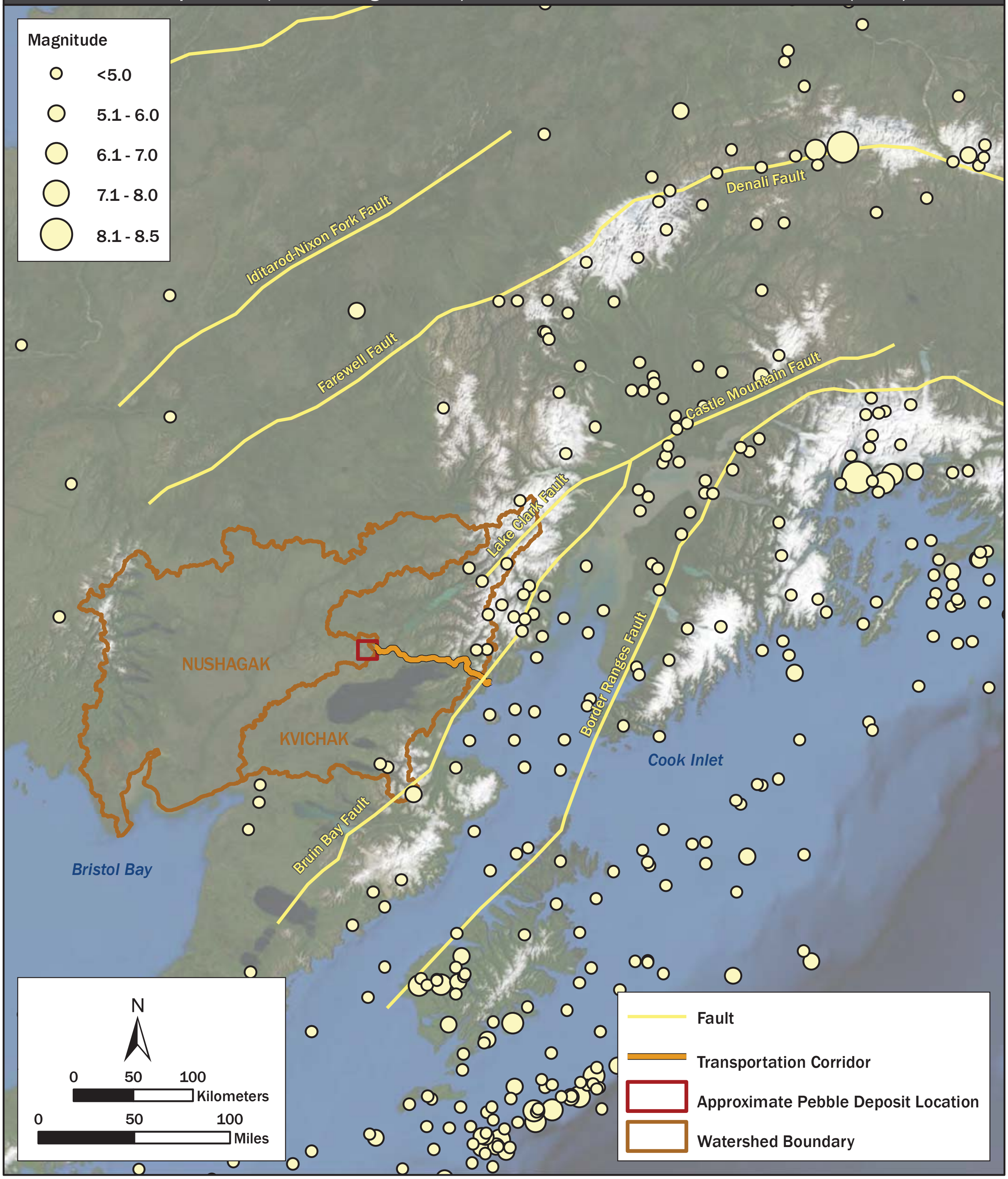
Aznalcóllar Tailings Dam, Los Frailes Mine, Seville, Spain, 1998. A foundation failure resulted in a 45-m-long breach in the 27-m-high, 600-m-long tailings dam, releasing up to 6.8 million m^3 of acidic tailings that traveled 40 km and covered 2.6 million ha of farmland (ICOLD 2001).

Stava, Italy, 1985. Two tailings impoundments were built, one upslope from the other, in the mountains of northern Italy. The upslope dam had a height of 29 m; the downslope dam had a height of 26 m. A stability failure of the upper dam released tailings, which then caused the lower dam to fail. The 190,000 m^3 of tailings, traveling at up to 60 km/hour, reached the village of Tesero 4 km downslope from the point of release, in 5 or 6 minutes. The failure killed 269 people (ICOLD 2001).

Aurul S.A. Mine, Baia Mare, Romania, 2000. A 5-km-long, 7-m-high embankment on flat land enclosed a tailings impoundment containing a slurry with high concentrations of cyanide and heavy metals. Heavy rains and a sudden thaw caused overtopping of the embankment, cut a 20- to 25-m breach, and released 100,000 m^3 of contaminated water into the Somes and Tisza Rivers. Flow continued into the Danube River and eventually reached the Black Sea. The contamination caused an extensive fishkill and the destruction of aquatic species over 1,900 km of the river system (ICOLD 2001).

Tennessee Valley Authority Kingston Fossil Plant, Roane County, Tennessee, 2008. After receiving nearly 20 cm of rain in less than 4 weeks, an engineered 18-m-high earthen embankment of a 34-ha storage impoundment failed, producing a 14-m-high surge wave and releasing 4.1 million m^3 of coal fly ash slurry. The release covered over 121 ha with slurry containing arsenic, cobalt, iron, and thallium. Over 2.7 million m^3 of coal ash and sediment were dredged from the Emory River to prevent further downstream contamination (AECOM 2009).

Figure 4 11. Seismic Activity in Southwestern Alaska. Location and magnitude of significant, historic earthquakes (USGS 2010) that caused deaths, property damage, and geological effects, or were otherwise experienced are shown, based on Seismicity of the United States (1568 to 1989) and the Preliminary Determination of Epicenters (1990 to August 2009). Fault lines based on Haeussler and Saltus (2004).



BOX 4-5. EARTHQUAKE EFFECTS

The effect of earthquakes on critical structures is a function of the strength of the seismic event, distance (depth and lateral) from the seismic event to the critical structure, and the nature of the geologic materials that carry the seismic waves. Earthquake damage can be caused by the following effects.

- Soil liquefaction, which causes the soil to turn into a semi-liquid material, reduces soil strength, and causes earthen structures to fail.
- Ground spreading and cracking of the earth surface, which causes structures above the rupture to separate and break.
- Shaking effects, including landslides and slope failures, and the creation of waves (seiches), which can cause overtopping of impoundments.

Unconsolidated sediments that are partially or fully saturated with water are susceptible to liquefaction. Smaller particles such as sands, silts, and clays are generally more susceptible to liquefaction than large-grained material such as gravel or boulders. Watersheds in the Bristol Bay area contain a wide range of soil conditions, but most slopes and areas outside stream deposits contain very coarse material. Streambeds and floodplains can contain sand and silt deposits up to tens of meters thick (PLP 2011), but because these deposits are typically in low gradient reaches they are less susceptible to liquefaction damage. If critical mine facilities are built on fine-grained sediments and not designed to withstand the effects of liquefaction, they could be susceptible to significant damage in the event of a large earthquake. Tailings storage facilities (TSFs) in our mine scenario would be located in an area of sand and silt deposits in the South Fork Koktuli River streambed, and could be susceptible to small-scale liquefaction.

Large and damaging earthquakes can rupture the surface of the earth and cause displacement from a few millimeters to several meters. The largest earthquake in Alaska (Table 4-7), the Anchorage earthquake of 1964 (magnitude 9.2), resulted in vertical displacements of up to 15 m and opened large crevices in streets. More recently, the Denali earthquake in 2002 (magnitude 7.9) caused vertical displacements of up to 4 m and lateral displacement along the fault of over 8 m. Such displacement is not likely to occur in the Bristol Bay watershed because of the absence of large faults, but there is a potential for a small amount of ground spreading and cracking from larger earthquakes.

As seismic waves travel through the ground, the earth surface rises and falls, much like the waves created in the ocean. Damage occurs as these waves move underneath buildings and support structures, and flex the rigid materials past their breaking points. Large tanks, pipelines and concrete structures must be designed to withstand such flexing. When seismic waves travel under large impoundments, they can create waves within the impoundments (seiches) that cause water to slosh in the impoundment and potentially over the edge of the dam.

- **Seepage.** Seepage through an earthfill embankment increases interstitial pore pressures and reduces the intergranular effective stresses and shear resistance, potentially leading to a slope instability failure. Seepage can also cause internal erosion and piping within a dam leading to a hydraulic failure.
- **Structural failure.** Tailings dams often contain structural components such as drainage systems or spillways that, if they fail to function properly, can cause overtopping or slope instability failure.
- **Erosion.** Erosion, especially along the toe of a dam, can reduce slope stability to the point of failure. Erosion near the crest can reduce freeboard and increase the risk of overtopping.
- **Mine subsidence.** If a tailings dam is near underground mining works, mine subsidence can cause displacement or cracking of the dam. Cracking can lead to a direct hydraulic breach or to slope instability. Settlement can reduce freeboard and increase the risk of overtopping.

Table 4-7. Examples of Earthquakes in Alaska

Date	Magnitude[a]	Distance and Direction from the Pebble Deposit
March 28, 1964	9.2	469 km east-northeast
November 3, 2002	7.2	593 km northeast
September 25, 1985	4.9	61 km southeast
July 13, 2007	4.3	30 km west-southwest
March 25, 2012	3.0	122 km east

Notes:
[a] Local magnitude as reported by the Alaska Earthquake Information Center. Note that earthquakes in the range of magnitudes 2.5 to 3.6 occur regularly in the Lake Clark area (data not shown).

A number of studies have attempted to analyze the historical record to determine the proximate causes and probability of tailings dam failures (ICOLD 2001, Davies 2002, Davies 2000 et al., Rico et al. 2008, Chambers and Higman 2011). These efforts have been hindered by the lack of a worldwide inventory of tailings dams, incomplete reporting of tailings dam failures, and incomplete data for known failures. Given these limitations, the U.S. National Inventory of Dams (NID 2005) lists 1,448 tailings dams in the United States, and the worldwide total is estimated at over 3,500 (Davies et al. 2000). The International Commission on Large Dams compiled a database of 221 tailings dam accidents and failures that occurred from 1917 through 2000 (ICOLD 2001). Causes of accidents and failures were reported for 220 of these; Table 4-8 summarizes information for 135 of the reported failures (ICOLD 2001).

Perhaps most noteworthy is the relatively high number of accidents or failures for active tailings dams relative to inactive tailings dams, primarily resulting from slope instability failure (Table 4-8). This suggests that the stability of tailings dams and impoundments may increase with time, as dewatering and consolidation of the tailings occurs and with the cessation of the application of additional loads (however, see Section 4.3.8.2). It could also be that any structural fault is more likely to cause a failure in the operating period, when loading conditions are still increasing. The primary cause of failure of inactive tailings dams is overtopping, accounting for 80% of the recorded failures for which the cause is known (Table 4-8).

Table 4-8. Number and Causes of Tailings Dam Failures at Active and Inactive Tailings Dams

Failure	Number of Tailings Dam Failures[a]		
Failure cause	Active Dams	Inactive Dams	Total
Overtopping	20	8	28
Slope instability	30	1	31
Earthquake	18	0	18
Foundation	11	1	12
Seepage	10	0	10
Structural	12	0	12
Erosion	3	0	3
Mine subsidence	3	0	3
Unknown	15	3	18
TOTALS	122	13	135

Notes:
[a] Data are presented for 135 tailings dam accidents and failures for which causes were reported, from 1917 to 2000.
Source: ICOLD 2001

4.4.2.2 Probability of Tailings Dam Failures

Several studies have estimated the probability of tailings dam failures, resulting in the failure probabilities listed below.

- An estimated 0.00050 failures per dam year, based on 88 failures from 1960 to 2010 (Chambers and Higman 2011). This translates to 1tailings dam failure every 2,000 mine years.
- An estimated 0.00049 failures per dam year, based on 3,500 appreciable tailings dams that experienced an average 1.7 failures per year from 1987 to 2007 (Peck 2007). This translates to 1 tailings dam failure every 2,041 mine years.
- An estimated 0.00057 to 0.0014 failures per dam year, based on a database including many unpublished failures that showed 2 to 5 major tailings dam failures annually from 1970 to 2001 (Davies 2002, Davies et al. 2000). This translates to 1 tailings dam failure every 1,754 to 714 mine years.

Available data do not permit estimation of failure rates based on causes of failure or tailings dam status. Most failures have occurred while the tailings dams were actively receiving tailings (Table 4-8), but the dam inventories do not indicate whether the thousands of dams in the inventory are active or inactive and do not include the years of operation. This prevents estimation of the proportion in each category and makes it impossible to calculate the number of active dam-years.

Low failure frequencies and incomplete datasets also make any meaningful correlations between the probability of failure and dam height or other characteristics questionable. Very few existing rockfill dams approach the size of the structures in our mine scenario, and none of these large dams have failed. For example, although the 1,448 tailings dams listed in the U.S. National Inventory of Dams create a statistically large and fairly complete database that includes dam heights, the International Commission on Large Dams failure database includes only 49 U.S. tailings dam failures—too small a dataset to develop a meaningful correlation between dam height and failure probability.

Silva et al. (2008) reported on over 75 earth dams, tailings dams, natural and cut slopes, and some earth-retaining structures to illustrate the relationship between the annual probability of slope failure in earth structures and factors of safety. They grouped projects into four categories based on the level of engineering applied to the design, site investigation, materials testing, analysis, construction control, operation, and monitoring of each project.

- **Category I:** Facilities designed, built, and operated with state-of-the-practice engineering. Generally these facilities are constructed to higher standards because they have high failure consequences.
- **Category II:** Facilities designed, built, and operated using standard engineering practice. Many ordinary facilities fall into this category.
- **Category III:** Facilities without site-specific design and substandard construction or operation. Temporary facilities and those with low failure consequences often fall into this category.
- **Category IV:** Facilities with little or no engineering.

The State of Alaska regulates its dams, including tailings dams, under Alaska Administrative Code (AAC) Title 11, Chapter 93, Article 3, Dam Safety (11 AAC 93). Each dam is assigned to a class based on the potential hazards of a tailings dam failure (Table 4-9). The tailings dams in our mine scenario would be classified as either Category I or Category II, both of which require a detailed computer stability analysis with verification by other methods, and may require more sophisticated finite element analyses in special circumstances. This analysis considers the effects of earthquakes based on a site-specific evaluation of seismicity in the area. Box 4-6 describes the selection of earthquake characteristics for design criteria.

Table 4-9. Summary of the State of Alaska's Classification of Potential Hazards of Dam Failure

Hazard Class	Effect on Human Life	Effect on Property
I (High)	Probable loss of one or more lives	Irrelevant for classification, but may include the same losses indicated in Class II or III
II (Significant)	No loss of life expected, although a significant danger to public health may exist	Probable loss of or significant damage to homes, occupied structures, commercial or high-value property, major highways, primary roads, railroads, or public utilities, or other significant property losses or damage not limited to the owner of the barrier Probable loss of or significant damage to waters identified under 11 AAC 195.010(a) as important for spawning, rearing, or migration of anadromous fish
III (Low)	Insignificant danger to public health	Limited impact on rural or undeveloped land, rural or secondary roads, and structures Loss or damage of property limited to the owner of the barrier

Notes:
Mine scenario would be classified as Hazard Class I or II
AAC = Alaska Administrative Code
Source: ADNR 2005

The Guidelines for Cooperation with the Alaska Dam Safety Program (ADNR 2005) do not specify a minimum safety factor for dams, but rather allow the applicant to propose one. Guidelines suggest that the applicant follow accepted industry design practices such as those provided by U.S. Army Corps of Engineers (USACE), the Bureau of Reclamation (Reclamation), Federal Energy Regulatory Commission (FERC), and other agencies. Both USACE and FERC require a minimum factor of safety of 1.5 for the loading condition corresponding to steady seepage with the maximum storage pool (FERC 1991, USACE 2003).

Combining the required factor of safety with the correlations between slope failure probability and factor of safety (Figure 4-12) derived from Silva et al. (2008) yields an expected annual probability of slope failure between 0.000001 and 0.0001. This translates to one tailings dam failure every 10,000 to 1 million mine years. The upper bound of this range is lower than the historic average of 0.00050 (1 failure every 2,000 mine years) for tailings dams, in part because slope failure is only one of several possible failure mechanisms, but also suggesting that past tailings dams may have been designed for lower safety factors or designed, constructed, operated, or monitored to lower engineering standards. Because 90% of tailings dam failures have occurred in active dams (Table 4-8), the probability of a tailings dam failure after TSF closure would be expected to be lower than the historical average for all tailings dams. However, Morgenstern (2011), in reviewing data from Davies and Martin (2009), did not observe a substantial downward trend in failure rates over time.

BOX 4-6. SELECTING EARTHQUAKE CHARACTERISTICS FOR DESIGN CRITERIA

Design criteria for dams specify that an evaluation be conducted to determine the effect of seismicity on stability and performance of the dam. The seismic evaluation must establish the operating basis earthquake (OBE) and maximum design earthquake (MDE). One important characteristic of determining earthquake sizes is the return period (recurrence period) over which the event is likely to occur. If long return periods are used in the analysis of earthquake size, the likelihood for a larger earthquake increases and the resulting design basis earthquake will have a greater margin of safety.

The OBE represents the characteristic earthquake with a reasonable probability of occurring during the functional lifetime of a project. Critical structures should be designed to withstand the effects of the OBE and remain functional, with little, easily repairable damage. The OBE can be defined using a probabilistic approach based on the likelihood that an earthquake of a certain magnitude and ground motion will be exceeded during a particular period of time. For a Class II dam, the return period that must be considered for the OBE is 70 to 200 years—that is, the OBE represents the largest earthquake likely to occur in 70 to 200 years.

The MDE represents the most severe earthquake considered at the site for which acceptable consequences of damage will result. All critical structures such as tailings dams must be designed to resist the effects of the MDE. The MDE can be determined based on historical earthquake patterns or through a rigorous probabilistic analysis. For a Class II dam, the return period considered appropriate for the MDE is 1,000 to 2,500 years. Underestimating the MDE could result in catastrophic tailings dam failure.

A third category of earthquake design level is the maximum credible earthquake (MCE). The term is not defined in the Alaska dam safety regulations, but supporting guidance defines it as the greatest earthquake that reasonably could be generated by a specific seismic source, based on seismological and geologic evidence and interpretations. Design engineers sometimes use the MCE to represent a floating earthquake located directly under a critical structure.

The return periods stated in Alaska dam safety guidance are inconsistent with the expected conditions for a large porphyry copper mine developed in the Bristol Bay watersheds, and represent a minimal margin of safety. The mine scenario in this assessment includes approximately 25 to 78 years of mineral extraction, with likelihood that additional long-term operations would be required for closeout and maintenance of the mine. This time period is barely within the OBE return period for Class II dams. The MDE analysis presents a potentially greater risk of underestimating the size of a characteristic earthquake. Tailings storage facilities (TSFs) will operate during the active mining period and could have a life expectancy of 10,000 years after operations cease. Because the return period for the MDE is 1,000 to 2,500 years, this could lead to significantly underestimating the largest earthquake that is likely to occur.

The Northern Dynasty Minerals Preliminary Assessment (NDM 2006) identified the following design criteria for the tailings storage facility.

- OBE return period of 200 years, magnitude 7.5.
- MDE return period of 2,500 years, magnitude 7.8, with maximum ground acceleration of 0.3g, based on Castle Mountain Fault data

NDM used a deterministic evaluation to select the MDE and MCE, which were deemed equivalent for the preliminary safety design. Northern Dynasty Minerals (NDM) also reports that the preliminary design incorporates additional safety factors, including design of storage facility embankments to withstand the effects of the MDE and a magnitude 9.2 event. In 2011, the NDM Preliminary Assessment Report states that an MCE of magnitude 7.5 with 0.44g to 0.48g maximum ground acceleration was used in the stability calculations for the tailings dam design.

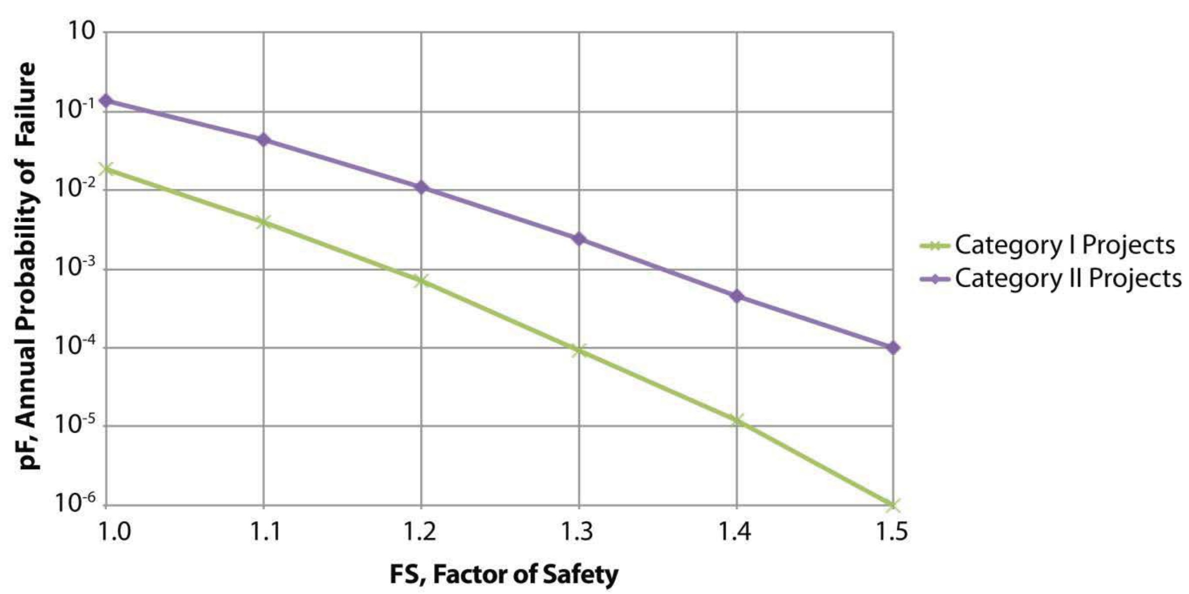
The variability in published probabilities of tailings dam failure reflects the uncertainty inherent in these estimates. Much of this uncertainty is due to incomplete data. Uncertainty also increases as time progresses, and TSFs may remain in place for long periods. Most dams are created as water holding dams that have a limited expected lifespan (generally 50 years). After mine closure, TSFs can be drained, eliminating the consequences of tailings dam failure. If TSFs remain in place after mine closure, the solid and liquid materials behind their tailings dams are expected to remain in place in perpetuity. This requires that dams be maintained in perpetuity, in the face of unpredictable seismic and weather events that may occur over thousands of years and may have cumulative effects.

4.4.2.3 Material Properties

Tailings Dam Rockfill

In our mine scenario, TSFs would be enclosed by rockfill dams constructed primarily of well-graded, non-acid-generating waste rock obtained from the mine pit during operations; the starter dike would contain material excavated from the upstream toe trench and local quarry. Waste rock from the mine pit would be used as it became available. The size of the rock used to construct the dam would depend on the rock's fracture characteristics, the methods used to blast and remove it from the mine pit, and the lift thickness specified for adequate compaction. Particle sizes typically range from sand to large boulders (Blight 2010). For a large rockfill dam with a high or significant hazard potential, the lift thickness would be expected to be limited to 1.5 m to guarantee adequate compaction, limiting the maximum particle size to about 1 m (Breitenbach 2007).

Figure 4-12. Annual Probability of Dam Failure vs. Factor of Safety (after Silva et al. 2008)



Well-graded rock would have a coefficient of uniformity, D_{60}/D_{10}, greater than 4 and would have a coefficient of curvature, $D_{30}/(D_{60}*D_{10})$, between 1 and 3. Combining these coefficients with Dawson and Morin's (1996) report of a D_{50} particle size greater than 200 mm for waste rock, one can generate a representative particle size distribution curve for the bulk of the tailings dam material (Figure 4-13).

Tailings Solids and Liquids

The tailings solids would include both bulk and pyritic tailings (Figure 4-4). The bulk tailings would be uniformly graded, consist largely of sand and silt-sized particles (D_{80} = 200 μm), and have a density of 1.36 metric tons/m^3. The pyritic tailings would consist of predominantly silt-sized particles, have a P_{80} of 30 μm, and would have a density of 1.76 metric tons/m^3. The mass of the bulk tailings and the pyritic tailings would equal 85% and 14% of the mass of the ore, respectively (Ghaffari et al. 2011).

Representative particle size distribution curves for the bulk, pyritic, and combined tailings are shown in Figure 4-13.

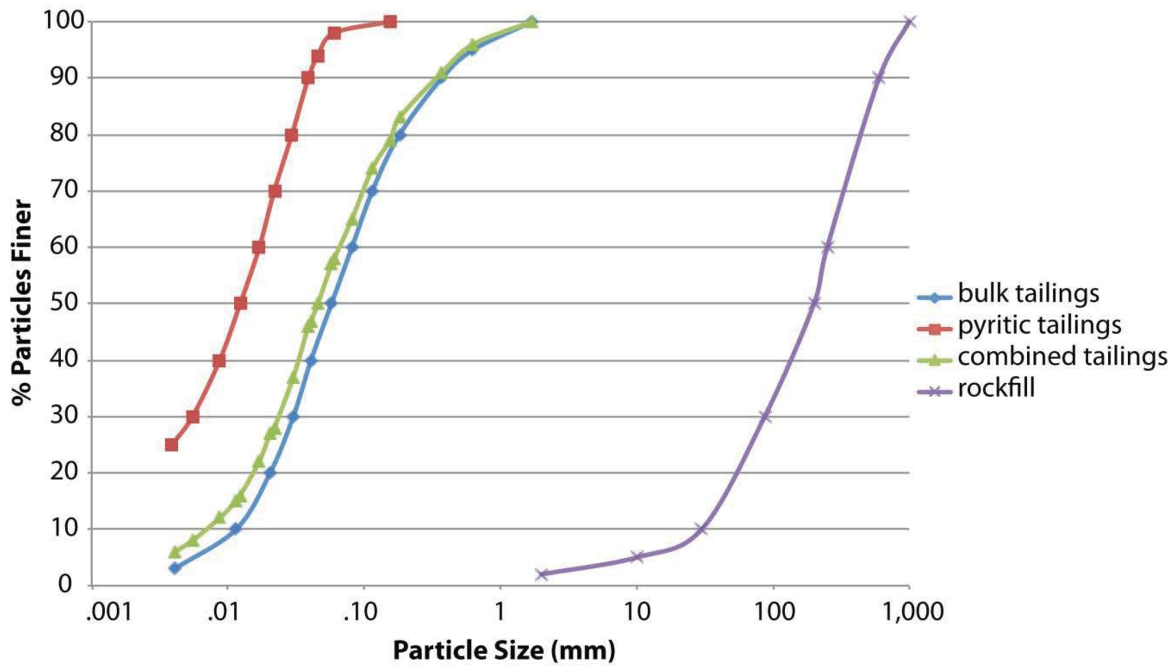
Given the bulk tailings dry density of 1.36 metric tons/m^3 and using the specific gravity reported for the ore of 2.61 for the solids, the bulk tailings would be 52% solids and 48% liquid by volume. The pyritic tailings, with a dry density of 1.76 metric tons/$m^3$and the same specific gravity, would be 68% solids and 32% water. Based on the proportions of bulk and pyritic tailings, the combined material in the TSF would be 55% solids and 45% water by volume, exclusive of any ponded water above the settled tailings. As the tailings consolidate, the bulk density of the deeper tailings would be expected to increase, although this consolidation may be limited (Section 4.3.8.2).

4.4.2.4 Tailings Dam Failure via Flooding and Overtopping

In this assessment, we consider the effects of two potential dam failures at TSF 1: a partial-volume failure, occurring during mine operations when TSF 1 would be only partially full (dam height = 98 m, tailings volume = 227 million m^3) and a full-volume failure, occurring during or after mine operations when TSF 1 would be filled to capacity (dam height = 208 m, tailings volume = 1,492 million m^3) (Tables 4-2 and 4-3). In both cases, we assumed 20% of the impounded tailings would be mobilized. We used a hydrologic model to simulate a maximum flood hydrograph (Box 4-7), and then modeled resulting hydrologic conditions in the stream channel and floodplains under partial and full-volume failure conditions, for a 30-km reach (Box 4-8).

Model results for hydrologic characteristics of the partial and full volume dam failures are shown in Tables 4-10 and 4-11. In both cases, estimated peak flows would be very large and atypical for flows experienced in this watershed, as the probable maximum flood (PMF) and impounded tailings would create a flood wave that could not result from a precipitation event alone. For comparison, a U.S. Geological Survey (USGS) gage located near the village of Ekwok, Alaska, experienced a record peak flood of 3,313 m^3/s in a 2,551-km^2 watershed. Under the partial volume dam failure, the peak flood is estimated at 1,862 m^3/s immediately downstream of the TSF 1 dam, where the contributing watershed area is only 1.4 km^2.

Figure 4-13. Representative Particle Size Distribution for Tailings Solids (Bulk and Cleaner or Pyritic Tailings) and Tailings Dam Rockfill



BOX 4-7. MODELING THE PROBABLE MAXIMUM FLOOD HYDROGRAPH AT TSF 1

We used the U.S. Army Corps of Engineers (USACE) Hydrologic Engineering Center's Hydrologic Modeling System (HEC-HMS) to generate a reasonable runoff hydrograph based on a 24-hour probable maximum precipitation (PMP) event of 356 mm (14 inches) (Miller 1963). Application of the PMP to calculate the probable maximum flood (PMF) is the accepted methodology for design and study of dams (ADNR 2005). The PMF is used to determine appropriate spillway/bypass facilities, or to predict the greatest flood that can cause failure. This conservative approach allows the full assessment of potential damage and impacts on the facilities and downstream reaches. However, this PMP value is extrapolated from limited precipitation gage data and has not been updated since 1963. It could be refined and may ultimately reduce the predicted flood peak, but no update is currently available. The HEC-HMS performs one-dimensional steady- and unsteady-state hydraulic calculations for river systems. Inputs of combined watershed parameters are used to model stormwater runoff characteristics for discrete watersheds. Basin characteristics for the TSF 1 site and the PMP were applied to the SCS Type 1A hydrograph methodology to model data for the probable PMF hydrograph (Box 4-7 Table).

Modeled Precipitation and Flow Data for the Probable Maximum Flood (PMF) at TSF 1

Time (hour)	Precipitation (mm)	Total Flow (m^3/s)
0:00	0.0	0.1
1:00	7.1	15.1
2:00	10.7	31.5
3:00	11.4	39.8
4:00	12.2	44.1
5:00	14.2	50.2
6:00	17.8	60.7
7:00	22.1	75.0
8:00	55.9	152.6
9:00	33.8	150.9
10:00	20.3	106.3
11:00	16.8	77.7
12:00	14.2	62.3
13:00	13.2	54.2
14:00	12.4	49.7
15:00	11.7	46.8
16:00	11.4	44.5
17:00	10.7	42.4
18:00	10.2	40.2
19:00	9.7	38.2
20:00	9.1	36.1
21:00	8.6	34.1
22:00	8.1	32.1
23:00	7.4	29.9
0:00	6.9	27.9
1:00	0.0	12.4

Notes:
Data are shown for a 24-hour period.

BOX 4-8. MODELING HYDROLOGIC CHARACTERISTICS OF TAILINGS DAM FAILURES

We used the U.S. Army Corps of Engineers (USACE) Hydrologic Engineering Center's River Analysis System (HEC-RAS) to model hydraulic characteristics of partial and full volume tailings dam failures caused by flooding and subsequent dam overtopping at tailings storage facility (TSF) 1. HEC-RAS inputs included geometry of an inline structure to simulate the dam cross-section and stream channel geometry data, both derived from a 30-m digital elevation model, as well as hydrograph data to simulate the probable maximum flood (PMF) (Box 4-7). Under both partial and full TSF volume conditions, results were modeled for 30 km (18.6 miles) downstream—from the face of the hypothetical dam to the confluence of the North Fork Koktuli and South Fork Koktuli Rivers (Figure 4-14)—because extension of the simulation beyond this point would have introduced significant error and uncertainty associated with the contribution of the South Fork Koktuli flows. The entire modeled flood wave hydrograph includes the PMF inflow, excess water on top of the tailings, and 20% of the total tailings volume. Channel roughness (i.e., Manning's *n* coefficient) was increased over typical values used in "clean water" models to better reflect the influence of sediment-rich water during tailings dam failure.

The headwater location of TSF 1 (and of other likely TSF locations in the Nushagak River and Kvichak River watersheds) would help to reduce the total volume of expected stormwater runoff into the TSF. If sufficient freeboard is maintained, it would be possible to capture and retain the expected volume of the PMF in the TSF. However, to examine potential downstream effects in the event of a tailings dam failure, we assume that sufficient freeboard would not exist and overtopping would occur. This may be less likely when the TSF would be actively monitored and maintained, but may be more representative of post-closure conditions.

Tailings dam failure via overtopping is expected to have similar effects as failures resulting from other causes (e.g., slope failure, earthquakes). We did not include a "dry weather" failure in our assessment but it is assumed that this kind of a failure (one that does not depend on a large precipitation event) would result in similar liquefaction of stored tailings; however, transport of tailings downstream may be reduced in a dry weather failure, as there is no precipitation generating additional flow. Available dry weather failure data indicate that sediment distribution varied greatly from site to site. Our results are well within reasonable limits.

Thus, on a unit area basis, the tailings dam area in the partial-volume failure analysis would result in a more than 1,000-fold increase in discharge compared to that observed in a record flood; for the full-volume failure analysis, there would be a more than 6,500-fold increase.

Maximum flood discharge would decrease with increasing distance downstream from the dam, as the downstream topography becomes flatter and the flood wave spreads out into the floodplain. When the flood wave recedes, water velocities would be expected to decrease similarly under the partial- and full-volume failures (as reflected in the same minimum flow velocities in Tables 4-10 and 4-11) and the potential for tailings deposition would be expected to increase.

Table 4-10. HEC-RAS Model Results for the Partial Volume TSF Dam Failure Analysis[a]. Values were modeled for more than 80 river stations along a 30-km length of stream; representative river stations along that length are shown here, listed by the distance upstream from the confluence of the North Fork Koktuli and South Fork Koktuli Rivers (River Station 30.0 km = foot of the dam for TSF 1 and River Station 0.6 km = downstream near confluence of North and South Fork Koktuli Rivers). Minimum flow values are based on an example flow expected to occur as the flood wave recedes (14.2 m^3/s).

	Maximum Flow Values					Minimum Flow Values		
			Velocities (m/s)			Velocities (m/s)		
River Station (km)	Discharge (m^3/s)	Depth (m)	LFP	CH	RFP	LFP	CH	RFP
30.0	1,862	10.52	3.37	5.40	3.45	0.28	0.66	0.34
26.8	1,751	5.96	1.78	4.09	1.76	0.12	0.34	0.15
24.7	1,723	6.27	2.13	4.04	1.37	0.23	0.56	0.00
17.2	1,024	5.01	0.00	1.93	0.00	0.00	0.30	0.00
12.7	386	2.90	0.21	0.69	0.17	0.00	0.27	0.00
9.4	301	3.71	0.12	1.18	0.30	0.00	0.23	0.00
5.4	276	2.41	0.28	0.74	0.00	0.06	0.30	0.00
0.6	243	3.37	0.27	0.57	0.28	0.08	0.23	0.06

Notes:

[a] Dam height = 98 m, tailings volume = 227 million m^3

LFP = left floodplain; CH = channel; RFP = right floodplain

Table 4-11. HEC-RAS Model Results for the Full Volume TSF Dam Failure Analysis[a]. Values were modeled for more than 80 river stations along a 30-km length of stream; representative river stations along that length are shown here, listed by the distance upstream from the confluence of the North Fork Koktuli and South Fork Koktuli Rivers (River Station 30.0 km = foot of the dam for TSF 1 and River Station 0.6 km = downstream near confluence of North Fork Koktuli and South Fork Koktuli Rivers). Minimum flow values are based on an example flow expected to occur as the flood wave recedes (14.2 m^3/s).

River Station (km)	Maximum Values					Minimum Values		
	Discharge (m^3/s)	Depth (m)	Velocities (m/s)			Velocities (m/s)		
			LFP	CH	RFP	LFP	CH	RFP
30.0	11,915	23.35	6.02	9.91	6.13	0.28	0.66	0.34
26.8	11,431	12.85	3.91	8.50	3.25	0.12	0.34	0.15
24.7	11,240	15.56	4.26	8.63	3.10	0.23	0.56	0.00
17.2	9,371	11.41	1.48	3.86	1.87	0.00	0.30	0.00
12.7	8,036	8.73	1.23	3.02	1.08	0.00	0.27	0.00
9.4	6,548	8.80	2.48	6.39	2.15	0.00	0.23	0.00
5.4	3,843	8.11	0.61	1.34	0.33	0.06	0.30	0.00
0.6	3,265	13.99	0.70	1.38	0.75	0.08	0.23	0.06

Notes:
[a] Dam height = 208 m, tailings volume = 1,492 million m^3.
LFP = left floodplain; CH = channel; RFP = right floodplain.

Dam failure flood waves and post-failure low flows under both partial- and full-volume failure conditions (Tables 4-10 and 4-11) suggest that transport and deposition of tailings material would occur throughout (and beyond) the modeled reach. After the initial deposition event, concentrated channel flows and floodplain conveyance areas would continue to transport sediment, as channel and valley morphology re-established within the newly deposited substrate.

Based on hydrologic model outputs, we estimated tailings deposition resulting from partial and full volume dam failures at TSF 1 along the 30-km stream length (Box 4-9), assuming mobilization of 20% of impounded tailings for both failures. Estimated amounts of tailings deposition at representative river stations are presented in Table 4-12. The depth of potential deposition varies across stations, based on the existing channel thalweg and floodplain terrace topography; however, this variability is small relative to uncertainty resulting from the low spatial resolution of the 30-m digital elevation model (Box 4-9).

The flood wave and tailings deposition that would result from a tailings dam failure under both partial and full volume conditions would have the potential to significantly alter the downstream channel and floodplain, even with only 20% of impounded tailings mobilized. The flood itself would have the capacity to scour the channel and floodplain, and the quantity of mobilized sediments that could be released from the TSF would bury the existing channel and floodplain under meters of fine-grained sediment. The sediment regime of the affected stream and downstream waters would be greatly altered. Nearly 30 km downstream of the TSF failed dam, estimated maximum depths of sediment deposition would be 3.4 m after a partial volume dam failure and 14.0 m after a full volume dam failure (Table 4-13). In both failure calculations, over 70% of the released tailings are modeled to remain in suspension at the 30-km model endpoint, indicating that effects would actually extend far beyond the 30-km reach. Based on historical tailings dam failure data, potential runout distances, or distance downstream where sediment from the failure is no longer evident, can range from hundreds to thousands of kilometers (Box 4-4).

BOX 4-9. USING HYDROLOGIC MODELS TO ESTIMATE TAILINGS DEPOSITION AFTER A TAILINGS DAM FAILURE

We used outputs from the one-dimensional Hydrologic Engineering Center's River Analysis System (HEC-RAS) hydraulic model (Box 4-8) to estimate tailings deposition along the stream network (Figure 4-14), based on calculated water depths and the assumption that tailings would settle at these depths as the velocity of sediment-rich water decreased across the floodplain. HEC-RAS most often used to simulate clear water flows. The flow calculation is completed between two adjacent cross-sections in the model, balancing the hydraulic energy to determine the water surface elevations and flow velocity, and then moving to the next cross- section in the sequence and repeating the process. When applied to tailings dam failure events, it is appropriate to increase channel roughness coefficients to better emulate flow characteristics of concentrated sediment flows. We assumed that sediment deposition could occur in the channel and the floodplain of each section at the maximum predicted channel depth during the peak of the flood wave. This creates a very conservative estimate of sediment deposition. Deposition at each cross-section at this maximum depth was used to calculate the volume between modeled river sections, and this volume was subtracted from the volume released from the tailings dam failure. It was assumed that the remaining sediment in the tailings dam failure flow was available to deposit at the next downstream section. This logic was carried downstream until the end of the modeled river length was reached.

We did not extend the analysis beyond the 30-km reach of the North Fork Koktuli River near its confluence with the South Fork Koktuli River. At some point downstream of the tailings dam failure, the gross deposition of sediment would cease and the flow dynamics of a typical sediment transport analysis would govern. We assumed that the confluence is where a more traditional sediment transport analysis would be appropriate. Given the scope of the current analysis, a traditional sediment transport analysis was not feasible. This discussion is limited to the estimation of probable sediment distribution after the immediate tailings dam failure and the total volume of sediment available to accommodate these assumptions.

We assume a particle size distribution of 0.1- to 1.0-m diameter for the dam construction material, and less than 0.01- to just over 1.0-mm diameter for the impounded tailings material (Figure 4-13). Based on the Hjulström curve—which estimates when a stream or river will erode, transport, or deposit sediment based on flow speed and sediment grain size—all of the mobilized tailings would remain in suspension at water velocities greater than 0.05 m/s (0.16 feet/s). This indicates that the channel would transport tailings under typical stormflow conditions and deposited tailings from floodplain terraces could be suspended and transported.

Based on historical tailings dam failure data, it is reasonable to assume that all construction material from the dam breach and from 30 to 66% of the impounded tailings material could contribute to debris flow following a tailings dam failure (Browne 2011). However, the volume of material mobilized, the distance it travels downstream, and the amount of deposition can vary greatly based on factors such as dam height, material size distribution, and material water content at the time of failure (Rico et al. 2008). Thus, we used conservative estimates for the percentages of impounded tailings material mobilized (5 to 20%, Table 4-13). Using a value less than measured historical release volumes allowed us to ensure we were not overestimating available sediment in the tailings dam failure calculations, and that volumes up to 20% would be considered reasonable at this level of investigation detail. We focus on transport and deposition of the fine-grained (less than 1.0 mm) tailings material, given the assumption that larger dam construction material would deposit within the first few kilometers downstream of the failure.

When the parameters for the partial- and full-volume dam failures are applied to runout distance equations from Rico et al. (2008), the expected runout distance under partial volume dam failure conditions is 35 km, reaching the mainstem Koktuli River; under full volume dam failure conditions, this distance increases to 307 km (190 miles), reaching the marine waters of Bristol Bay. Although the actual momentum of a failure flow would distribute some material upstream, we limit our analysis to downstream effects.

Table 4-12. Tailings Mobilized and Deposited During Partial and Full Volume Dam Failures at TSF 1. The volume of mobilized tailings and tailings remaining in transport were modeled for 30 km downstream of the tailings dam. The volume of mobilized tailings includes material within the dam cross section that has failed, plus a percentage (5 to 20%) of the stored tailings material.

Failure Volume	Volume of Stored Tailings (million m³)	% Mobilized	Volume of Mobilized Tailings (million m³)	Volume Remaining in Transport at Downstream Extent of Model (million m³)
Partial	227	20	55.4	40.6
		15	44.1	29.3
		10	32.7	18.0
		5	21.4	6.6
Full	1,489	20	317.5	239.3
		15	243.0	164.9
		10	168.5	90.4
		5	94.1	15.9

Table 4-13. Estimates of the Depth and Volume of Tailings Deposited Downstream of a Failed Dam at TSF 1. Values are presented for partial and full volume tailings dam failures at TSF1, assuming mobilization of 20% of impounded tailings (see Table 4 13). Values were modeled for more than 80 river stations along a 30 km length of stream; representative river stations along that length are shown here, listed by the distance upstream from the confluence of the North Fork Koktuli and South Fork Koktuli Rivers (River Station 30.0 km = foot of the dam for TSF 1 and River Station 0.6 km = downstream near confluence of North Fork Koktuli and South Fork Koktuli Rivers).

Failure Volume	River Station (km)	Cross-Sectional Area of Deposition (m^2)	Maximum Depth of Deposition (m)	Maximum Volume of Deposition (thousand m^3)
Partial	30.0	451	10.5	151
	26.8	777	6.0	129
	24.7	621	6.3	75
	17.2	532	5.0	158
	12.7	650	2.9	285
	9.4	285	3.7	95
	5.4	507	2.4	464
	0.6	644	3.4	361
Full	30.0	1,730	23.4	578
	26.8	2,659	12.9	442
	24.7	2,149	15.6	260
	17.2	2,801	11.4	832
	12.7	3,767	8.7	1,652
	9.4	1,655	8.8	550
	5.4	4,857	8.1	—
	0.6	3,635	14.0	2,035

These newly deposited tailings would create a completely different valley geomorphology. The existing channel and floodplain would be eliminated, and a new channel would develop in resulting topography. Given their fine size, these new deposits would be highly mobile under typical flows, and the channel would remain unstable. Newly deposited material on floodplains, and the remaining tailings in the breached TSF, would serve as concentrated sources of easily transportable, potentially toxic material (Section 6.1.3).

Use of a traditional sediment transport model would likely improve estimates of sediment movement and deposition, especially as the model is extended further downstream. As more sediment is deposited, flow would be expected to become less saturated. In addition, tributary streams would input clean water at each confluence. Because of the site-specific data required to implement a sediment transport model, we limited our model to the 30 km above the confluence of the North Fork Koktuli and South Fork Koktuli Rivers.

4.4.3 Pipeline Failures

4.4.3.1 Causes and Probabilities of Pipeline Failures

Over 4 million km of pipeline form an important component of the United States transportation system. Of these, over 3.8 million km are gas transmission or natural gas distribution mains and over 280,000 km (175,000 miles) carry hazardous liquids, primarily petroleum products (PHMSA 2012). The principal causes of pipeline failure are external corrosion and mechanical damage caused by third-party impacts. Internal corrosion and material breakdown also may cause pipeline failures, but are less common. The failure rate from third-party impacts, such as damage caused by excavating equipment, tends to be steady over time, whereas corrosion failures tend to increase with age of the pipe.

The most extensive analyses of pipeline failure statistics are derived from oil and gas industry data (Table 4-14). Although annual failure rates span a range of nearly two orders of magnitude (0.000046 to 0.0052), the range for pipelines most similar to the assessment pipelines along the transportation corridor is much narrower. For example, failure rate per kilometer-year for pipelines less than 20 cm in diameter equals 0.0010 and for pipelines in a climate similar to Alaska (Alberta, Canada) equals 0.0016, and for pipelines run by small operators (i.e., those with pipeline total lengths less than 670 km) equals 0.00062. The geometric mean of these three values yields an annual probability of pipeline failure per kilometer of pipeline equal to 0.0010.

This overall estimate of annual failure probability, coupled with the 139-km length of each pipeline as it runs along the transportation corridor, result in a 14% probability of a failure in each of the four pipelines each year. Thus, the probability of a pipeline failure occurring over the duration of the minimum mine scenario (i.e., approximately 25 years) would be 98% for each pipeline. Even if the mine operator achieves the average oil pipeline failure rate of 0.00028 failures per kilometer-year, the probability of a failure over 25 years would still be 63% for each pipeline, and 98% that at least one pipeline would fail.

Table 4-14. Studies that Examined Historical Pipeline Failure Rates

Study	Km-Years Analyzed	Pipeline or Failure Parameter Assessed	Annual Failure Rate (per km/year)
OGP 2010 (oil pipelines)	667,000	Diameter <20 cm	0.0010
		Diameter 20–36 cm	0.00080
		Wall thickness ≤5 mm	0.00040
		Wall thickness 5–10 mm	0.00017
OGP 2010 (gas pipelines)	2,770,000	1970 to 2004	0.00041
		2000 to 2004	0.00017
Caleyo 2007	34,595	Mexican gas pipelines	0.0030
	28,270	Mexican oil pipelines	0.0052
URS 2000 (56 US oil pipeline operators)	1,268,370	Highest failure rate	0.0011
		Average failure rate	0.00028
		Minimum failure rate	0.000046
		10 smallest operators (< 418 km)	0.00062
		10 largest operators (> 6900 km)	0.00020
Alberta Metal 2011	285,000	2000 failures, Alberta	0.0033
	394,000	2009 failures, Alberta	0.0016

4.4.3.2 Concentrate Pipeline Failure

The effects of a pipeline failure would depend upon many factors, including which pipelines are affected (copper [+gold] concentrate, reclaimed water, natural gas, or diesel), location of the pipeline failure along the transportation corridor, and the time of year at which the pipeline failure occurs. The volume of material released from a pipeline leak would depend on factors such as the type of failure, rate of loss from the pipe, pumping rate, duration of the leak, the diameter of the pipe, and distance to the nearest shutoff valves, and the time when those valves are closed. For the purposes of this assessment, we evaluate a break in the copper (+gold) concentrate pipeline that occurs at a stream crossing, thereby releasing slurry into that stream. We assume the following pipeline failure conditions.

- **Full pipeline break or a defect of equivalent size.** This could occur as a result of mechanical failure of the pipe from ground movement, vehicle impact, or material failure.
- **Pumping rates** (Ghaffari et al. 2011) of:
    - Copper (+gold) concentrate: 254.8 metric tons/hour
    - Reclaimed water: 106.7 metric tons/hour
    - Diesel fuel: 120,000 gallons/day
    - Natural gas: 50 million cubic feet /day
- **Pipe diameters** of:
    - Copper (+gold) concentrate: 20.3 cm
    - Reclaimed water: 17.8 cm

  - Diesel fuel: 14.1 cm
  - Natural gas: not specified
- **Remotely activated shutoff values**
- **Time to pipeline shutdown of 2 minutes**
- **Distance to nearest shutoff valve of 14 km.** This value assumes there would be isolation valves capable of being remotely activated on either side of nine major river crossings along the transportation corridor. This is similar to the plan laid out in Ghaffari et al. (2011), although they call for manual rather than automatic isolation valves.

Thus, the estimated volume of material released from a pipeline failure would equal the flow rate times 2 minutes plus the volume in the pipe between isolation valves (Table 4-15). Materials released from the pipelines would have different densities, affecting their persistence in the environment. The copper (+gold) slurry would have a density of 1.65 metric tons/m^3 and would sink rapidly if released into a water body. The reclaim water would have a density near 1.0 metric tons/m^3 and would more readily mix with surface waters. The diesel fuel would have a density less than 1.0 metric tons/m^3 and would float on water. The natural gas is lighter than air and upon release would rise and dissipate. If the gas cloud ignited, most of the heat would travel upward, but the initial blast and subsequent radiation heating could affect the road and the nearby environment.

Table 4-15. Estimated Releases from Pipeline Failures. Estimates are provided for the four pipelines that would connect the mine to the Cook Inlet port.

Product	Volume over 2 Minutes of Flow (m^3)	Volume Between Isolation Valves (m^3)	Total Release Volume (m^3)
Copper (+gold) concentrate	5.1	470	475
Reclaim water	3.6	362	366
Diesel fuel	0.6	184	185
Natural gas	2,000	1,250	3,200

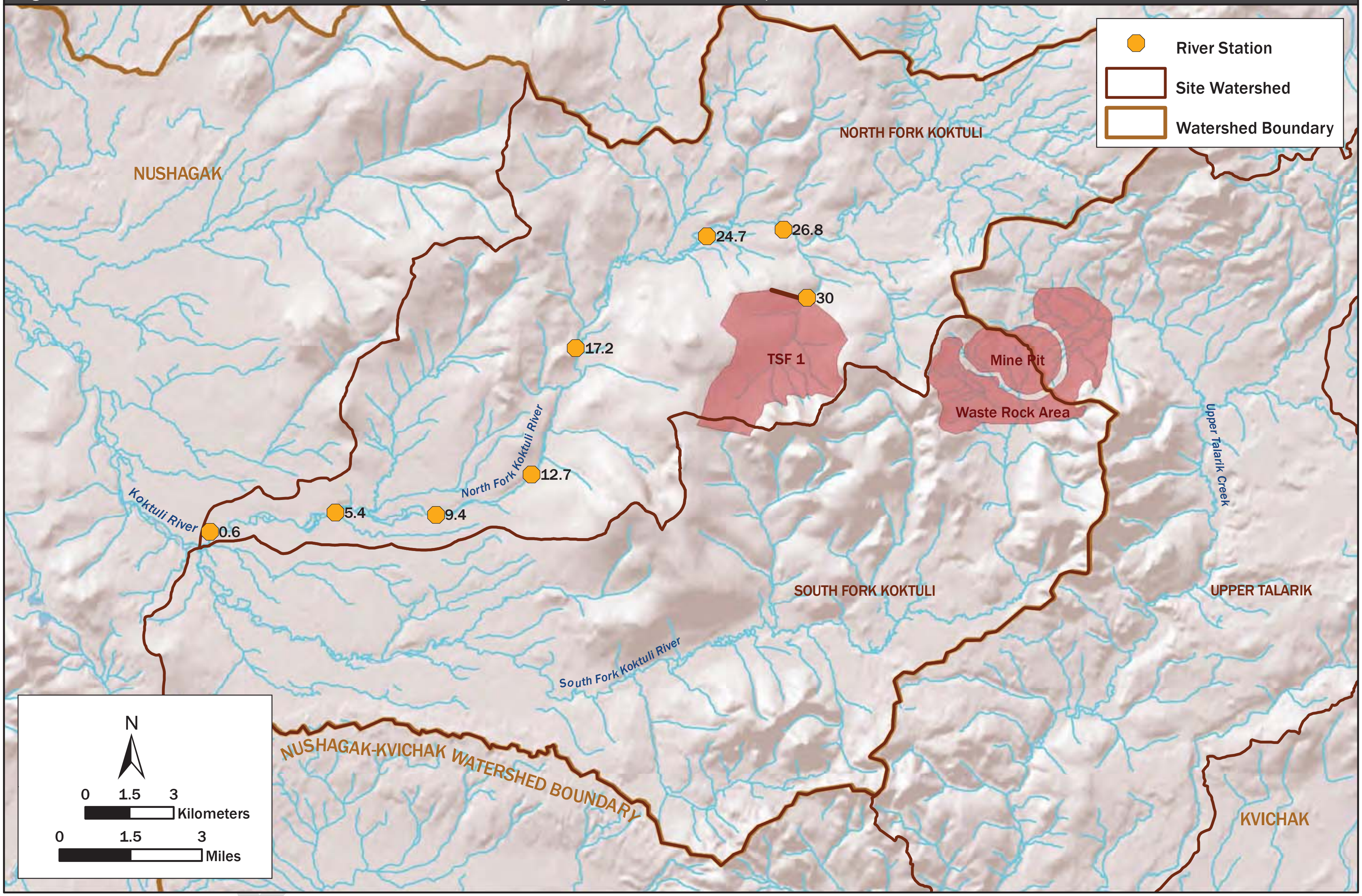
4.4.4 Road and Culvert Failures

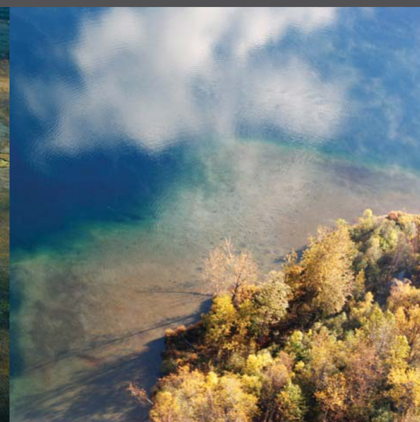
Construction of roads can increase the frequency of slope failures by orders of magnitude and result in episodic sediment delivery to streams and rivers, depending on such variables as soil type, slope steepness, bedrock type and structure, and presence of subsurface water. Mass soil movements triggered by roads can continue for decades after the roads are built (Furniss et al. 1991). Spills of transported chemicals and material also are likely events on the road (Angermeier et al. 2004), but they are not considered in this assessment (see Appendix G for additional information on roads).

Culverts are deemed to have failed if the passage of fish is blocked or if streamflow exceeds culvert capacity, thus resulting in washout of the road (Warren and Pardew 1998, Wellman et al. 2000). When culverts are plugged by debris or overtopped by high flows, road damage, channel realignment, and severe sedimentation also often occur (Furniss et al 1991). Reported culvert failure rates vary throughout the literature but are generally high: for example, 53% (Gibson et al. 2005), 30% (Price et al. 2010), 66% (85% for non-anadromous fish streams) (Flanders and Cariello 2000). The risk of road and culvert failure is substantial for most crossings, so *how* they fail is of critical importance. Road crossings may inhibit fish passage because of outfall barriers, excessive water velocity, insufficient water depth in culverts, disorienting turbulent flow patterns, lack of resting pools below culverts, or a combination of these conditions (Furniss et al. 1991). The mine access road would traverse varied terrain and subsurface soil conditions, including extensive areas of rock excavation in steep mountainous terrain (Ghaffari et al. 2011). Thus, although the road design, including placement and sizing of culverts, would take into account seasonal drainage and spring runoff requirements, road and culvert failures would be expected.

Failure of stream crossings can be a major source of increased sediment loading of streams. When stream crossings fail, they often do so catastrophically, causing extensive local scour and deposition and additional erosion downstream. Road and culvert failures that divert streamflow outside of stream channels are particularly damaging and persistent (Weaver et al. 1987). Changes in sediment load due to road and culvert failures change stream hydraulics and geomorphic pressures. Generally, habitat value in the stream is diminished as the channel becomes wider and shallower.

Figure 4-14. Selected River Stations Used in the Tailings Dam Failure Analysis (Boxes 4-8 and 4-9).



CHAPTER 5. RISK ASSESSMENT: NO FAILURE

This chapter addresses the risks associated with the routine operations of the mine scenario (Section 4.4). That is, it addresses the environmental effects of the mine operations in the absence of failures of any kind. This is not considered to be a realistic case, because accidents and failures always happen in complex and long-lasting operations. However, it serves to separate the inevitable effects of the mine scenario from those that are merely possible (Chapter 6).

Because these potential effects and the scenario on which they are based are more certain, they are analyzed in more detail than the potential effects of failures discussed in Chapter 6. These effects include elimination and modification of habitat (Section 5.2), release of effluents (Section 5.3), construction and operation of a transportation corridor (Section 5.4), indirect effects on wildlife (Section 5.5), and indirect effects on Alaskan Native cultures (Section 5.6).

5.1 Abundance and Distribution of Fish in Watersheds Draining the Mine Site

The potential effects of routine mine operations (this chapter) and failures (Chapter 6) depend on the abundance and distribution of the salmonid fish species that occur in the potentially exposed streams and rivers.

5.1.1 Fish Distribution

The watersheds draining the mine site—the North Fork Koktuli River, South Fork Koktuli River and Upper Talarik Creek watersheds (hereafter referred to as the site watersheds)—have been sampled extensively for summer fish distribution over several years. These data are captured in the Alaska Department of Fish and Game (ADFG) Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region (Anadromous Waters Catalog [AWC]) (Johnson and Blanche in press) and the Alaska Freshwater Fish Inventory (AFFI) (ADFG 2012). The AWC provides the

State of Alaska's official record of anadromous fish distribution and life-history information (spawning, rearing, or present but life stage unspecified) documented by individual stream reaches. The AFFI includes all fish species, including resident fishes, found at specific sampling points. The distribution of salmon-bearing subwatersheds in the Nushagak River and Kvichak River watersheds is shown in Figure ES-3. The documented distribution of the five species of Pacific salmon, Dolly Varden, and resident rainbow trout in site watersheds is shown in Figures 5-1 through 5-7. In addition, Arctic grayling, slimy sculpin, northern pike, ninespine stickleback, threespine stickleback, Alaskan or Arctic brook lamprey, burbot, round whitefish, humpback whitefish, least cisco, and longnose sucker occur in these watersheds (Johnson and Blanche in press, ADFG 2012). AWC and AFFI designations should be interpreted with care because not all streams could be sampled, and there are potential errors associated with fish identification or mapping. Caveats and uncertainties concerning interpretation of AWC and AFFI data are discussed in Section 5.2.4.

The distributions of pink and chum salmon are generally restricted to mainstem reaches where spawning and migration occur. Pink salmon have only been documented at very low numbers in the lowest section of Upper Talarik Creek and in the Koktuli River below the confluence of the north and south forks (Figure 5-1). Chum salmon have been found in all three site watersheds, and in the stream under the footprint of tailings storage facility (TSF) 3 (Figure 5-2). Sockeye salmon also use the mainstem reaches of all three site watersheds for spawning and rearing, including a portion of Upper Talarik Creek that is within the waste rock footprints of both the minimum and maximum mine sizes (Figure 5-3). Chinook salmon spawning has been documented throughout the mainstem reaches of the site watersheds (Figure 5-4). Chinook salmon are known to use small streams for rearing habitat, and juveniles have been observed in streams that are in the TSF 1 (North Fork Koktuli River), TSF 3 (South Fork Koktuli River), and waste rock pile (Upper Talarik Creek) footprints (Table 5-1, Figure 5-4). Coho salmon have the most widespread distribution of the five salmon species in the site watersheds, making extensive use of mainstem and tributary habitats (Figure 5-5). Coho salmon rear in the majority of the headwater streams that would be eliminated or blocked under both mine sizes (Figure 5-5). Dolly Varden are found even further upstream than coho salmon, and fish surveys indicate that they are commonly found in the smallest streams (i.e., first-order tributaries) throughout all three site watersheds (Figure 5-6). Their occurrence is limited above Frying Pan Lake, although they have been found in high-gradient streams draining the west side of Koktuli Mountain. Resident rainbow trout have been collected at many mainstem locations, especially in Upper Talarik Creek, and their reported distribution extends upstream throughout the TSF 1 footprint and in the portions of Upper Talarik Creek within the waste rock footprint (Figure 5-7).

Figure 5-1. Reported Pink Salmon Distribution in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek. Designation of species spawning and presence is based on 2012 ADFG Draft Anadromous Waters Catalog (Johnson and Blanche in press). Spawning=spawning adults observed and present=present, but life stage use not determined. Life stage-specific reach designations are likely underestimates, given the logistical constraints on the ability to accurately capture all streams that may support life stage use at various times of the year. See Section 5.2.4 for additional notes on interpretation of distribution data.

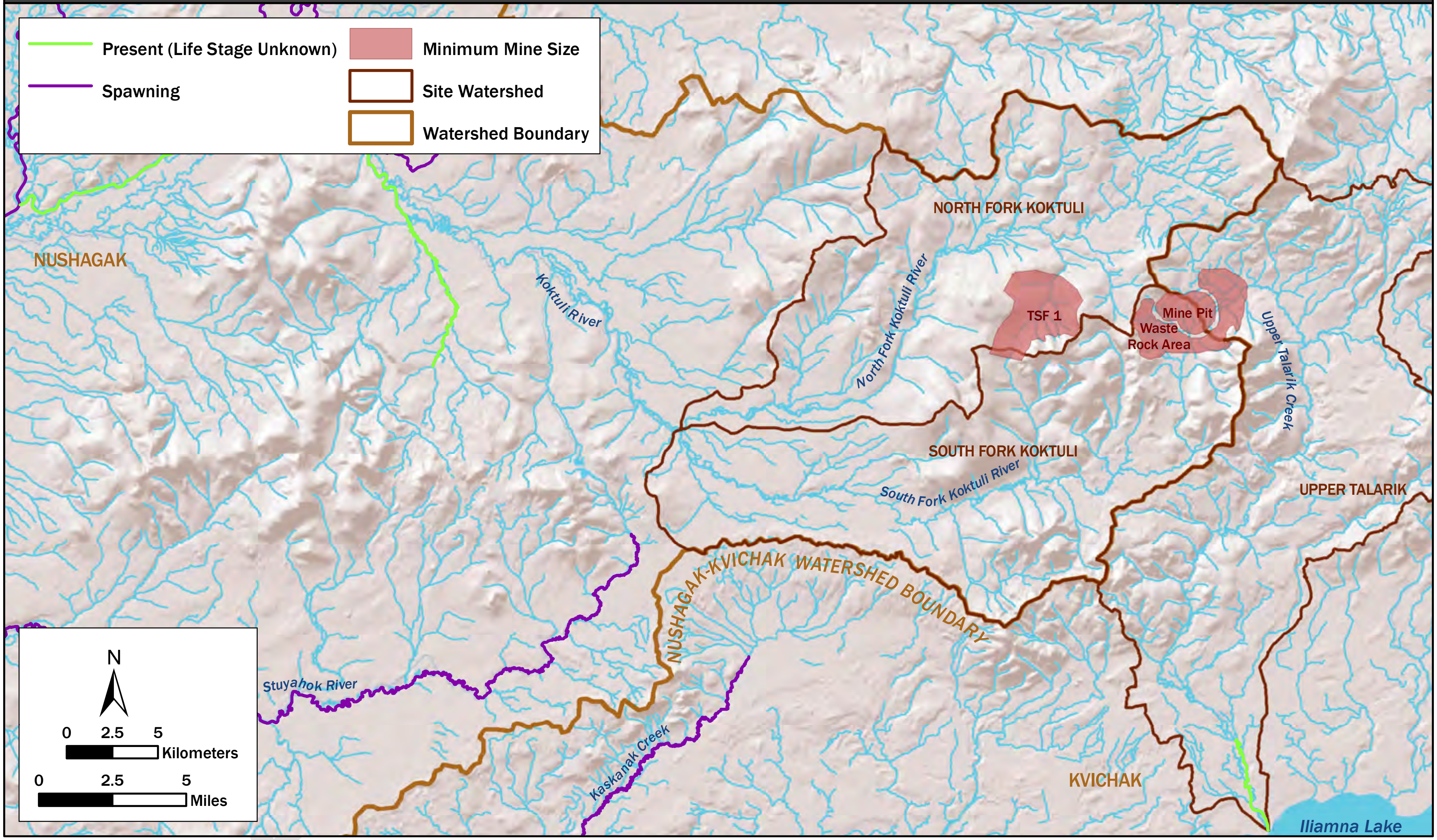


Figure 5-2. Reported Chum Salmon Distribution in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek. Designation of species spawning, rearing, and presence is based on 2012 ADFG Draft Anadromous Waters Catalog (Johnson and Blanche in press). Spawning = spawning adults observed, rearing = juveniles observed, present = present, but life stage use not determined. Life stage-specific reach designations are likely underestimates, given the logistical constraints on the ability to accurately capture all streams that may support life stage use at various times of the year. See Section 5.2.4 for additional notes on interpretation of distribution data.

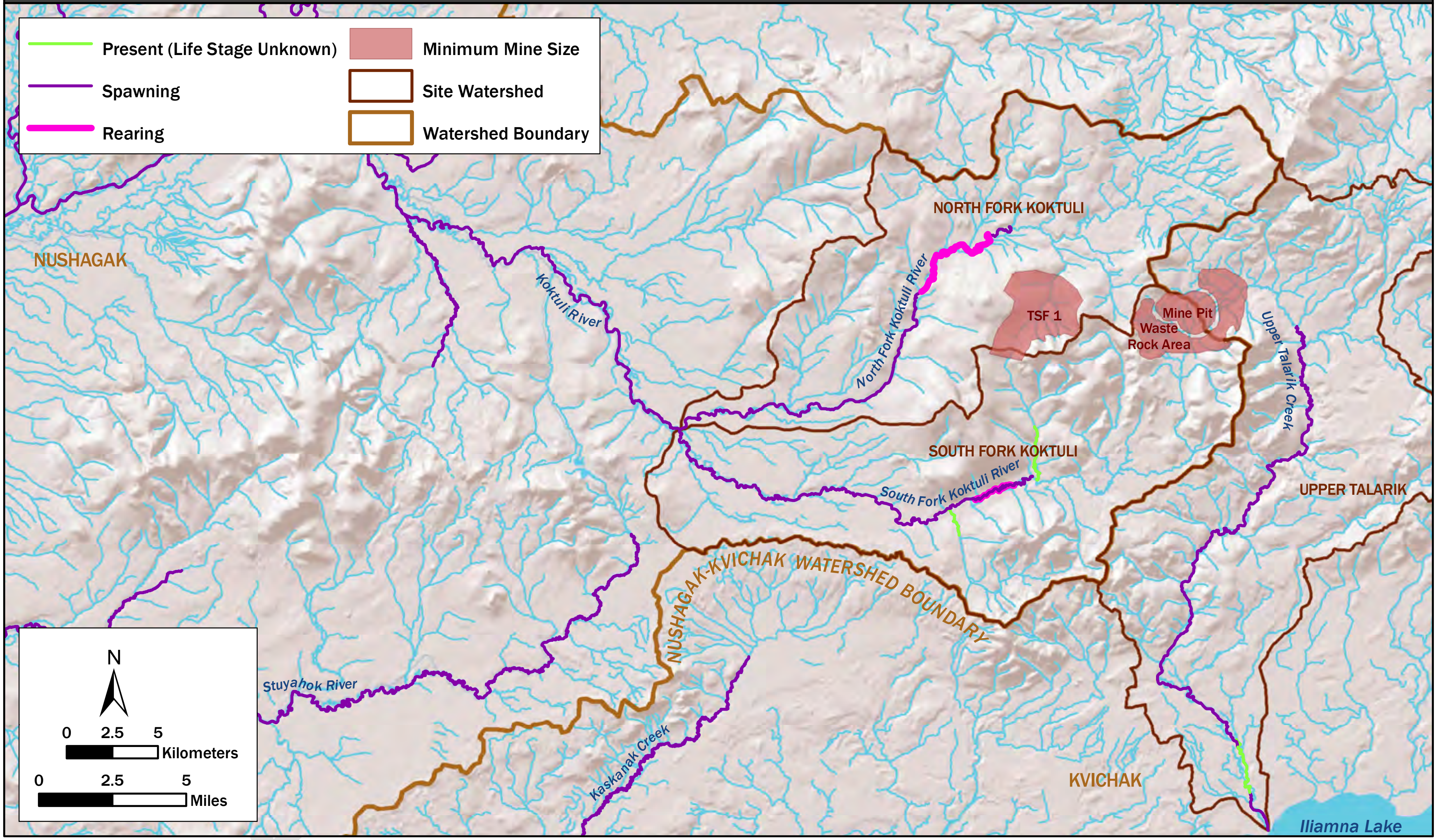


Figure 5-3. Reported Sockeye Salmon Distribution in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek. Designation of species spawning, rearing, and presence is based on 2012 ADFG Draft Anadromous Waters Catalog (Johnson and Blanche in press). Spawning = spawning adults observed, rearing = juveniles observed, present = present, but life stage use not determined. Life stage-specific reach designations are likely underestimates, given the logistical constraints on the ability to accurately capture all streams that may support life stage use at various times of the year. See Section 5.2.4 for additional notes on interpretation of distribution data.

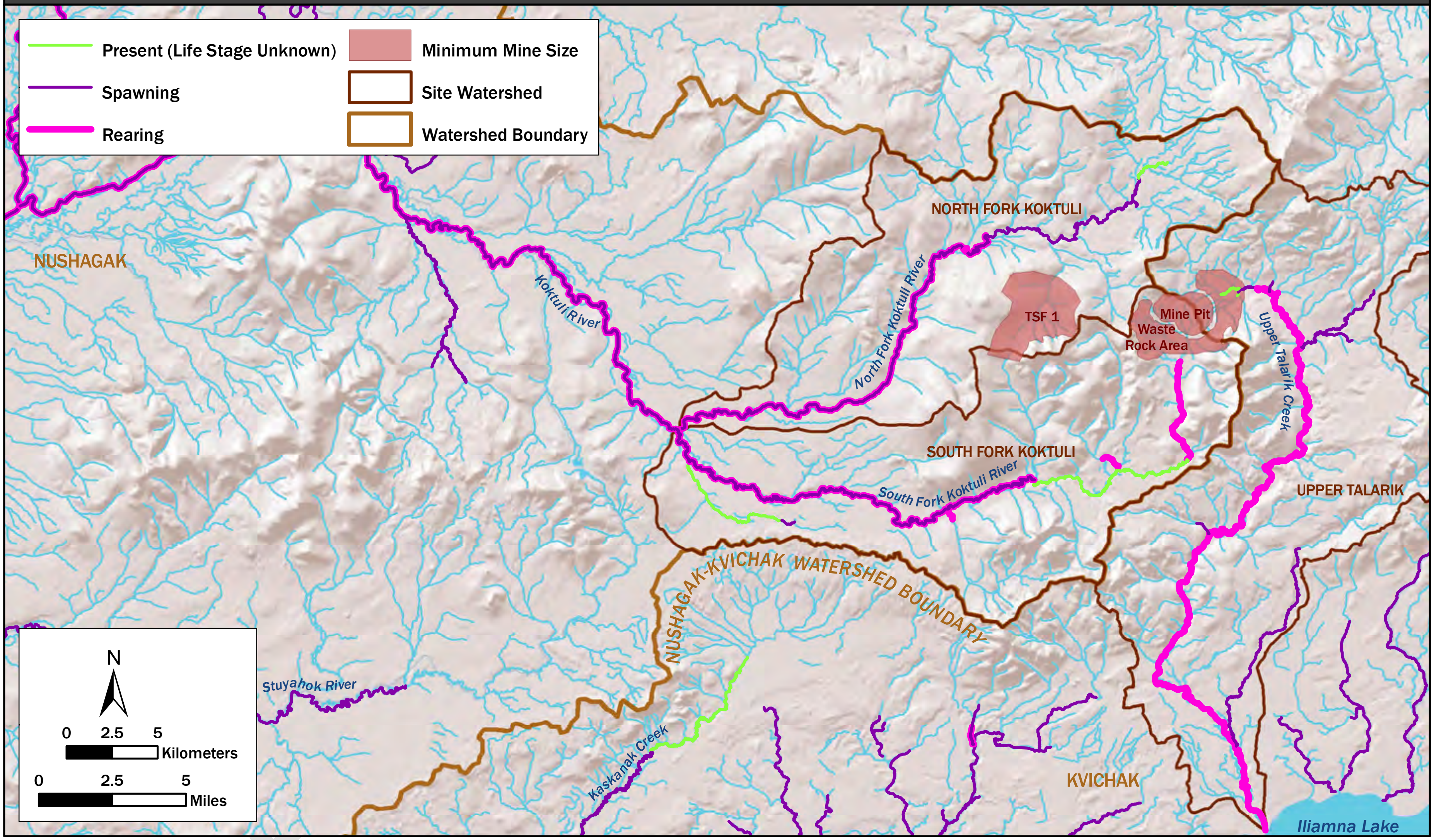


Figure 5-4. Reported Chinook Salmon Distribution in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek. Designation of species spawning, rearing, and presence is based on 2012 ADFG Draft Anadromous Waters Catalog (Johnson and Blanche in press). Spawning = spawning adults observed, rearing = juveniles observed, present = present, but life stage use not determined. Life stage-specific reach designations are likely underestimates, given the logistical constraints on the ability to accurately capture all streams that may support life stage use at various times of the year. See Section 5.2.4 for additional notes on interpretation of distribution data.

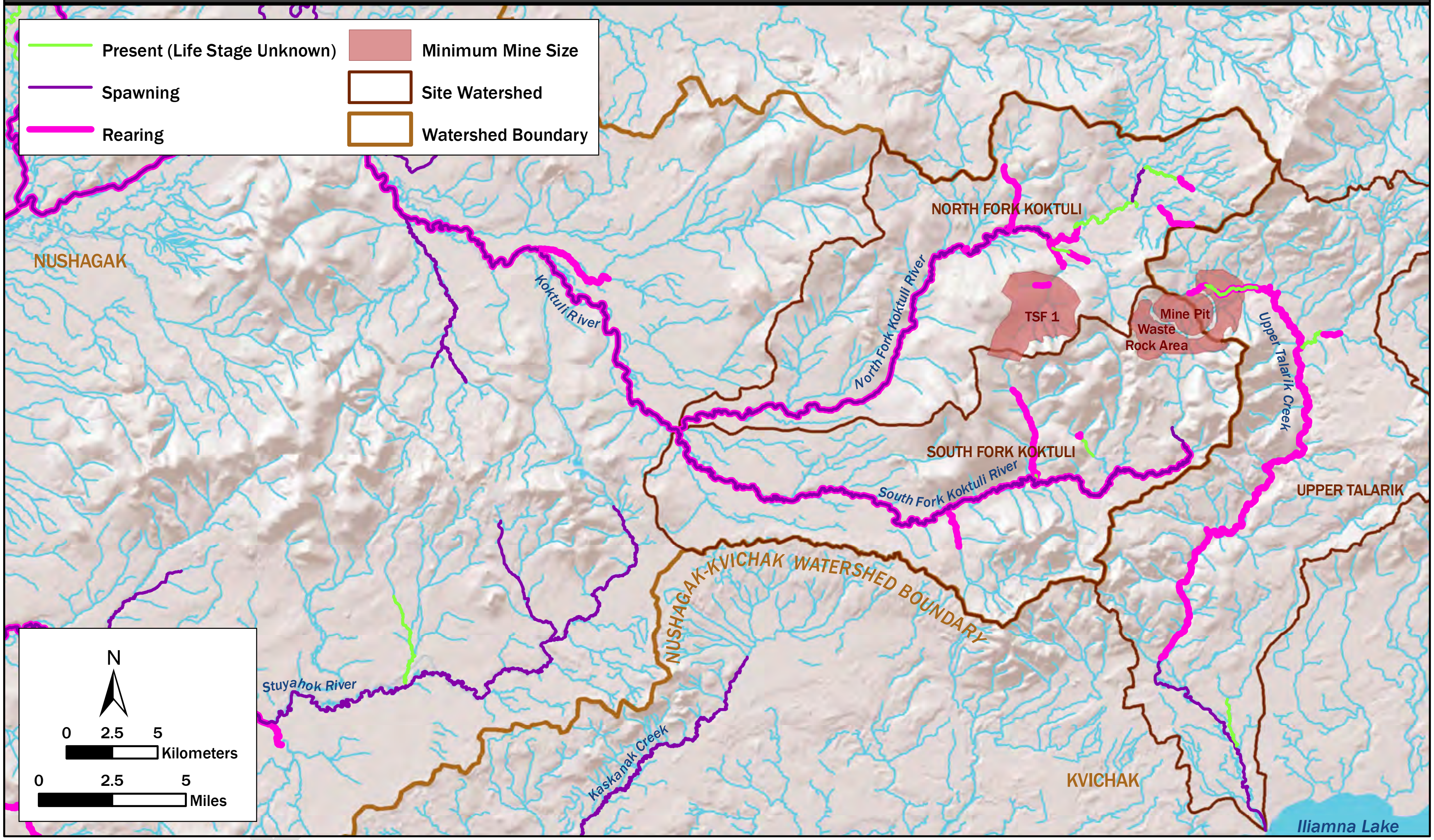


Figure 5-5. Reported Coho Salmon Distribution in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek. Designation of species spawning, rearing, and presence is based on 2012 ADFG Draft Anadromous Waters Catalog (Johnson and Blanche in press). Spawning = spawning adults observed, rearing = juveniles observed, present = present, but life stage use not determined. Life stage-specific reach designations are likely underestimates, given the logistical constraints on the ability to accurately capture all streams that may support life stage use at various times of the year. See Section 5.2.4 for additional notes on interpretation of distribution data.

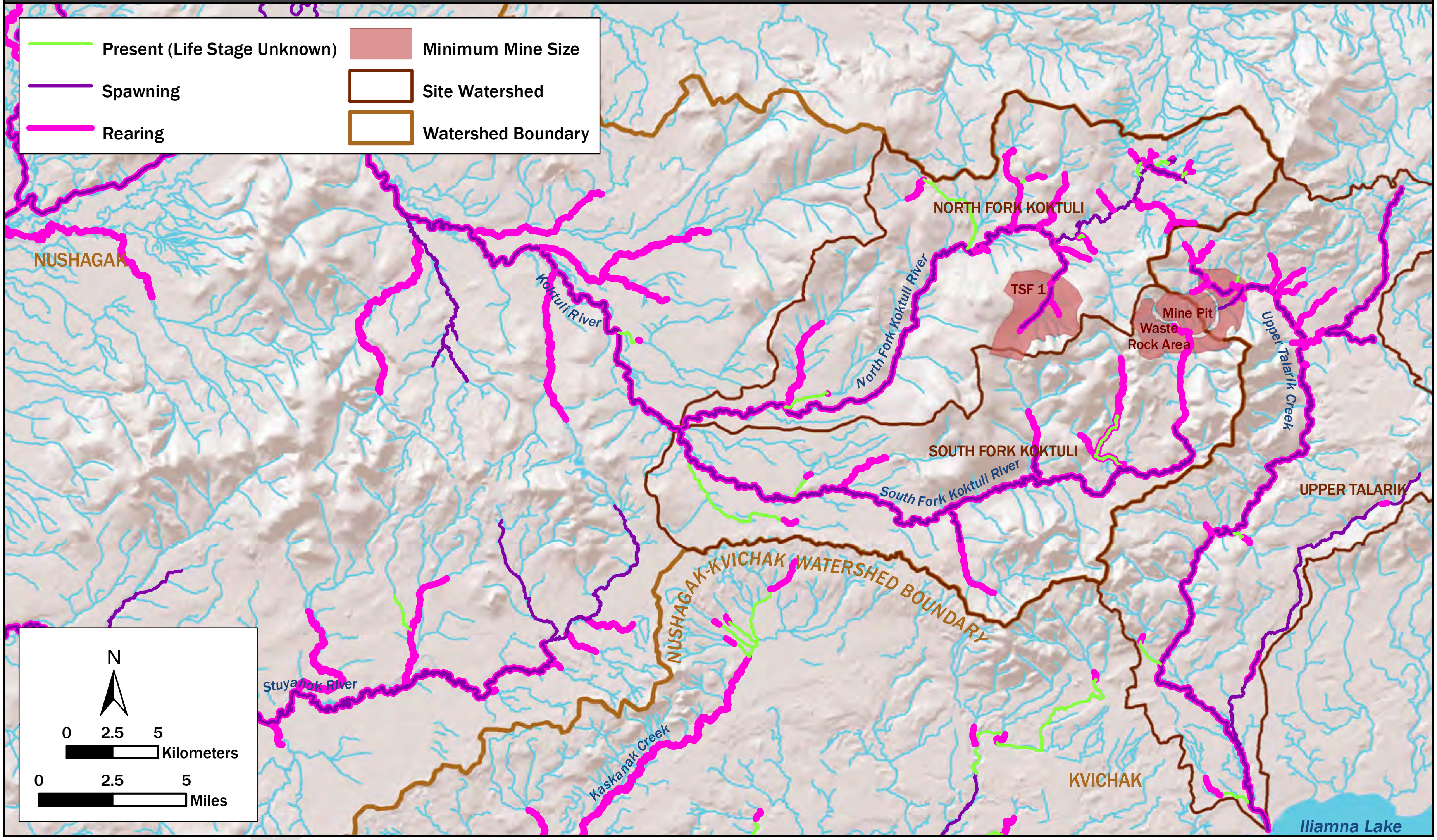


Figure 5-6. Reported Dolly Varden Occurrence in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek.
Designation of species presence is based on Alaska Freshwater Fish Inventory (ADFG 2012). Absence cannot be inferred from this map. See Section 5.2.4 for additional notes on interpretation of distribution data.

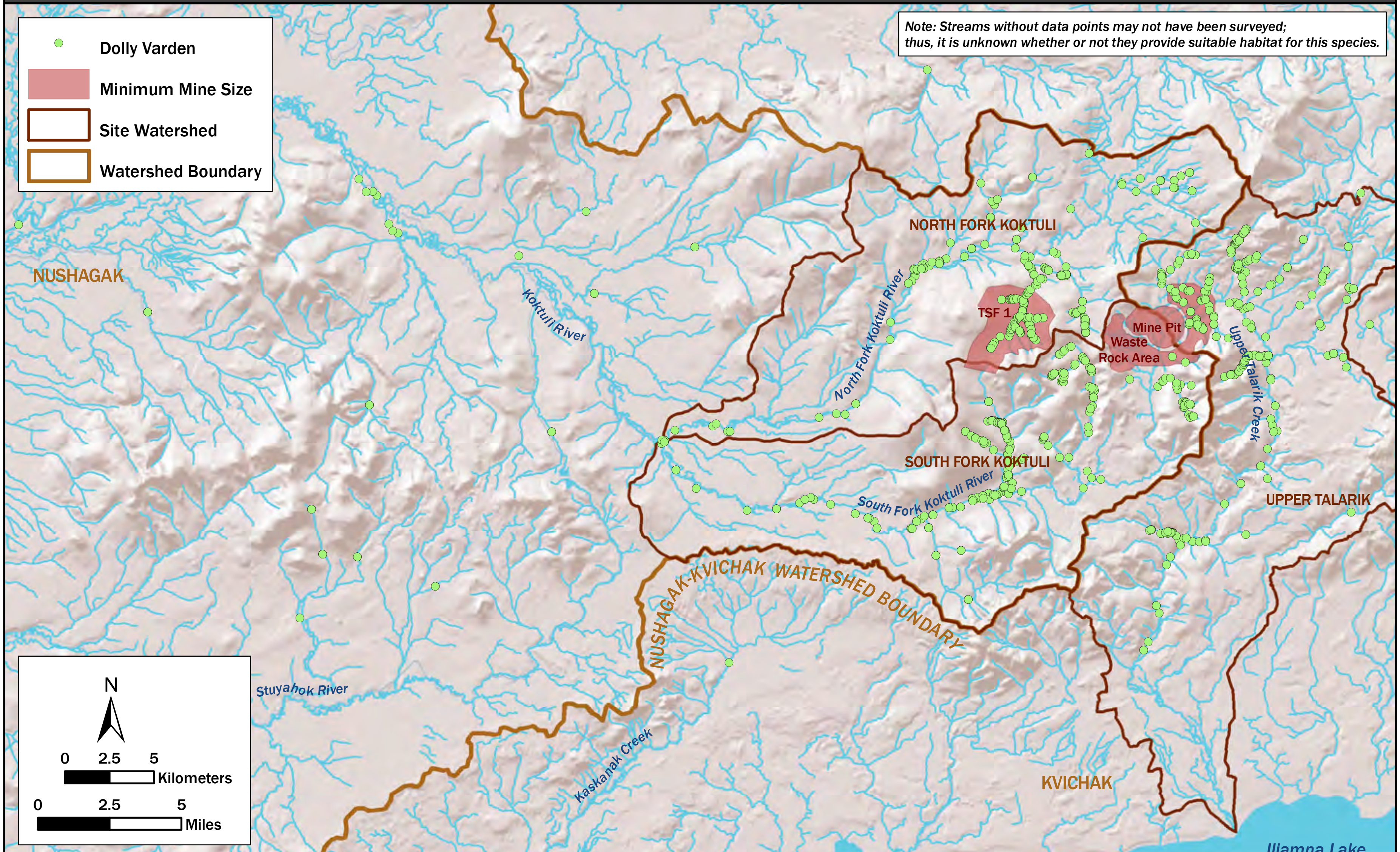


Figure 5-7. Reported Rainbow Trout Occurrence in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek. Designation of species presence is based on Alaska Freshwater Fish Inventory (ADFG 2012). Absence cannot be inferred from this map. See Section 5.2.4 for additional notes on interpretation of distribution data.

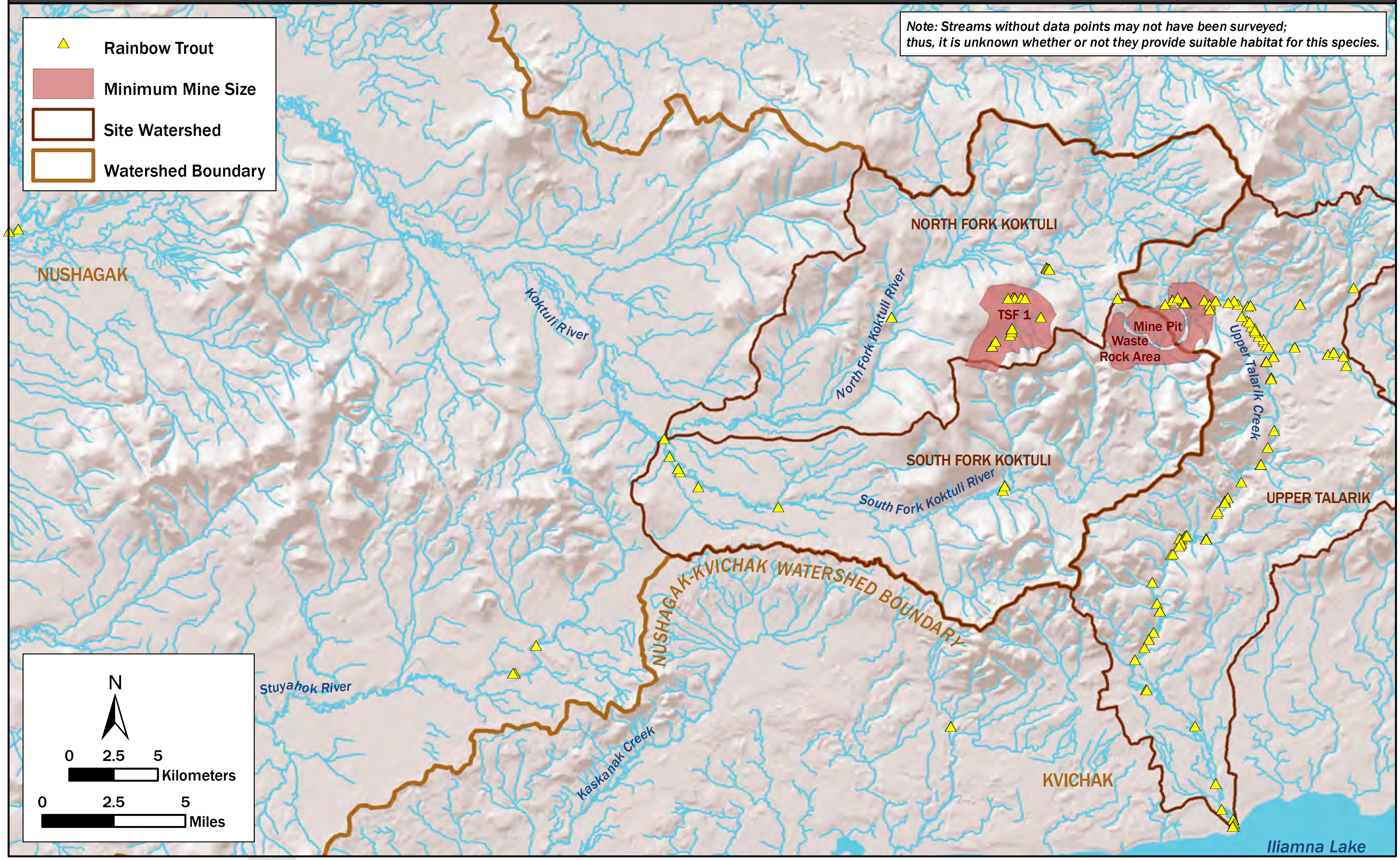


Table 5-1. Highest Reported Index Spawner Count for Each Year

River or Creek	Salmon Species	Highest Index Spawner Count Per Year (Number Of Counts)[a]				
		2004	2005	2006	2007	2008
Upper Talarik	Chinook	275 (2)	100 (3)	80 (3)	150 (9)	100 (8)
	chum	(0)	3 (1)	13 (2)	8 (8)	18 (5)
	coho	3,000 (4)	(0)	6,300 (3)	4,400 (9)	6,300 (14)[b]
	sockeye	33,000 (2)	15,000 (4)	10,000 (6)	10,000 (14)	82,000 (14)[b]
North Fork Koktuli River	Chinook	2,800 (3)	2,900 (4)	750 (4)	600 (8)	500 (8)
	chum	400 (1)	350 (4)	750 (4)	800 (9)	1,400 (7)
	coho	300 (3)	350 (1)	1,050 (4)	125 (8)	1,700 (15)
	sockeye	550 (2)	1,100 (5)	1,400 (7)	2,200 (10)	2,000 (12)
South Fork Koktuli River	Chinook	2,750 (3)	1,500 (4)	250 (5)	300 (8)	500 (9)
	chum	(0)	350 (4)	850 (7)	200 (11)	950 (7)
	coho	250 (2)	550 (4)	1,375 (3)	250 (10)	1,875 (20)
	sockeye	1,400 (2)	2,000 (5)	2,700 (8)	4,000 (11)	6,000 (13)

Notes:
[a] Values likely underestimate true spawner abundance by a substantial amount.
[b] Tributary 1.60, a major tributary to Upper Talarik Creek, was included in this count.
Source: PLP 2011

5.1.2 Spawning Salmon Abundance

No quantitative estimates of spawner abundance are available for any fish species in the site watersheds. Some aerial index counts of spawning salmon are available. These are primarily used as a crude index to track variation in run size over time, and we report values here recognizing that they tend to underestimate true abundance by a large and unknown factor (Jones et al. 2007).

ADFG conducts aerial index counts of sockeye salmon on Upper Talarik Creek and Chinook salmon on the Koktuli River that target peak spawning periods. Sockeye salmon counts have been conducted most years since 1955 (Morstad 2003), and Chinook salmon counts most years since 1967 (Bue et al. 1988, Dye and Schwanke 2009). Between 1955 and 2011, sockeye salmon counts in Upper Talarik Creek have ranged from 0 to 70,600, with an average of 7,021 over 49 count periods (Morstad pers. comm.). Between 1967 and 2009, Chinook salmon counts in the Koktuli River ranged from 240 to 10,620, with an average of 3,828 over 29 count periods (Dye and Schwanke 2009). It must be stressed, however, that surveys coinciding with the peak of spawning activity underestimate true abundance because (1) an observer in an aircraft is not able to count all of the fish in dense aggregations and (2) only a fraction of the fish that spawn at a given site are present at any one time (Bue et al. 1988, Jones et al. 2007). Additionally, surveys intended to capture peak abundance may not always do so. Thus, we present the ADFG data recognizing that the true spawner abundance is probably substantially higher than the values presented here.

The Pebble Limited Partnership's (PLP's) Environmental Baseline Document (EBD) provides aerial index counts for Chinook, chum, coho, and sockeye salmon in the site watersheds from 2004 to 2008 (PLP 2011). Multiple counts were usually made for each stream and species in a given year (Table 5-1).

Because of difficulties in establishing reliable estimates of observer efficiency, the EBD reports the average of each year's index counts as an abundance index for each population. Instead, we report the highest of each year's index counts for each population (Table 5-1), because this number is closer to the true abundance, and the averaged estimates reported in the EBD are often pulled downward by counts outside of the spawning period when no fish were counted (PLP 2011: Figure 15.1-93). The highest index counts for coho and sockeye salmon were in Upper Talarik Creek, and the highest counts for Chinook and chum salmon were in the Koktuli River (Table 5-1). The overall highest count was for sockeye salmon in Upper Talarik Creek in 2008, when approximately 82,000 fish were tallied. For the reasons discussed in the previous paragraph, the reported index values probably underestimate true spawner abundance by a substantial amount.

5.1.3 Juvenile Salmon and Resident Fish Abundance

Quantitative density estimates for juvenile salmon and resident fishes are available for 12 headwater stream sites in the mine area (O'Neal and Woody 2012). Electrofishing was used to conduct mark-recapture studies in tributaries of the North Fork Koktuli River (three tributaries), South Fork Koktuli River (three tributaries), Upper Talarik Creek (three tributaries), Kaskanak Creek (one tributary), the Chulitna River (one tributary), and the Stuyahok River (one tributary). Density estimates (number per 100 m^2 ± standard deviation) averaged across the 12 sites were 46±70 for coho salmon, 42±123 for Dolly Varden, 0.5±1 for Arctic grayling, and 1±5 for rainbow trout. Standard deviations, which were larger than the means for each of these estimates, indicate that abundance of each of these species varied widely across the tributaries sampled.

The EBD reports index counts for juvenile salmon and resident fishes in the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek based on extensive sampling efforts from 2004 through 2008 (PLP 2011). Snorkel surveys were the primary data collection method, but electrofishing, minnow traps, beach seines, gill nets, angling, and dip netting were used in certain situations. It is not clear which survey methods generated which counts. Raw field counts were expressed as densities (count per 100-m reach was the only unit reported for all three streams). These counts should not be viewed as quantitative abundance estimates, because they are very likely underestimates as a result of the extreme difficulty of observing or capturing all fish in complex habitats (Hillman et al. 1992). Other methods generate density estimates with confidence bounds (e.g., mark-recapture or depletion estimates) but are much more time-consuming or labor-intensive.

Fish densities reported in the EBD (averaged over the 4 years) vary widely by stream, sample reach, and habitat type (PLP 2011: Figures 15.1-23, 15.1-52, and 15.1-82). Species that attain densities of several hundred per 100-m reach in one setting were often absent or sparse in other habitat types or reaches within the same stream, which is typical for fish in heterogeneous environments like streams. Table 5-2 presents maximum fish densities, approximated from figures in the EBD, for the focal species that rear for extended periods in the surveyed streams: Chinook and coho salmon, Arctic grayling, and Dolly Varden. We report maximum density to give a sense of the magnitude attained in the surveyed streams, but it should be stressed that abundance varied widely by stream reach and habitat type within a given stream (PLP 2011: Figures 15.1-23, 15.1-52, and 15.1-82). The highest reported densities were approximately 25,000 Arctic grayling and 16,000 coho salmon per km from adjacent reaches on Upper Talarik Creek and 1,400 coho salmon per km from a reach on the North Fork Koktuli River.

Table 5-2. Highest Index Counts of Selected Stream-Rearing Fish Species

	Highest Reported Density (count per 100 m)[a]				
Stream	Chinook Salmon	Coho Salmon	Arctic Grayling	Dolly Varden	Source
North Fork Koktuli River	500	1400	40	40	EBD Table 15.1-23
South Fork Koktuli River	450	600	275	55	EBD Table 15.1-52
Upper Talarik Creek	400	1600	2500	10	EBD Table 15.1-82

Notes:
[a] Values were approximated from tables listed in the source column.
Source: PLP 2011

5.2 Habitat Modification

Routine mine operations would modify habitat for salmonid fish (salmon, trout, and char) by eliminating headwater streams within and up-gradient of the mine footprint (Section 5.2.1) and by using or redirecting water that would otherwise flow into streams draining the site (Section 5.2.2). Downstream flow changes have complex effects, including reducing the amount of aquatic habitat (Section 5.2.2.1), changing water temperatures (Section 5.2.2.2), and affecting fish populations (Section 5.2.2.3). These effects are described for start-up conditions and both mine sizes. The combined risks from habitat modifications are characterized (Section 5.2.3), and uncertainties and assumptions are described (Section 5.2.4).

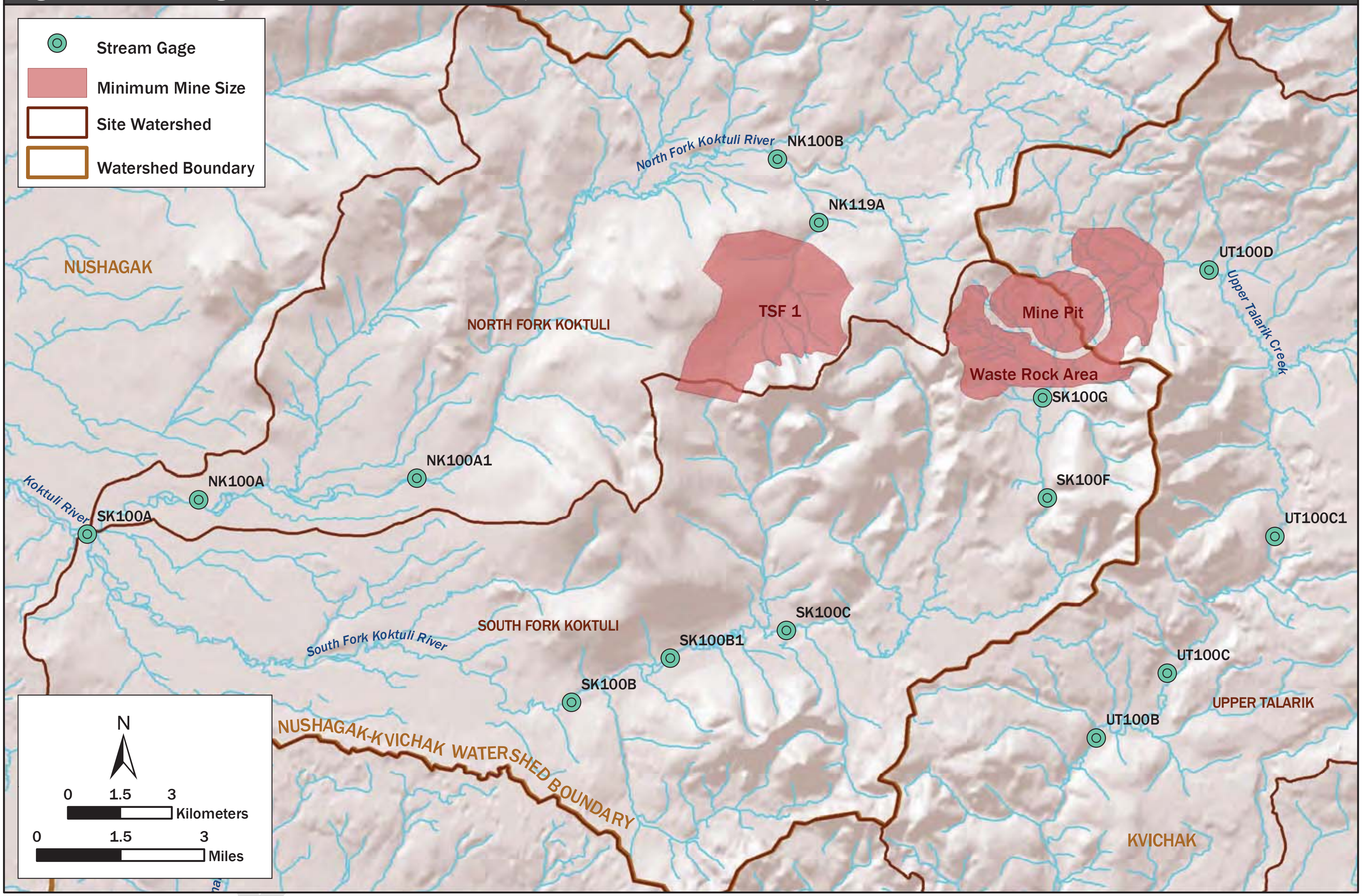
5.2.1 Habitat Lost or Blocked in the Mine Footprint

The total mine footprint consists of the area devoted to mining, including the mine pit, waste rock piles, TSFs, ore processing facilities, and other mine-related constructs. Streams and wetlands habitats would be lost within and upstream of the footprint (Figure 5-8), and downstream habitat would be degraded by the loss of the headwater streams and wetlands.

5.2.1.1 Stream and Wetland Loss in the Mine Footprint

The mine scenario described in Chapter 4 dictates our estimates of direct fish habitat losses expected from mining activity. We assume that streams under or upstream of the mine footprint would be effectively lost to access by fish from downstream reaches as a result of (1) removal (e.g., loss of stream channels in pit area), (2) elimination under a TSF or waste rock pile, (3) capture into the water treatment footprint of the mine, or (4) diversion of the stream channel in a manner that prevents fish passage (e.g., via pipes or conveyances too steep for fish passage).

Figure 5 8. Stream Gages in the North Fork Koktuli River, South Fork Koktuli River, and Upper Talarik Creek Watersheds



Under the minimum mine size, 87.5 km of first- through third-order streams located in the site watersheds would be eliminated or blocked by the mine footprint (Table 5-3 and Figure 5-9). Under the maximum mine size, an additional 19 km of streams in the pit and waste rock pile area, and an additional 34.9 km of first- through third-order streams in the South Fork Koktuli River watershed (TSF 2 and TSF 3) would be eliminated or blocked, for a total of 141.4 km of streams eliminated or blocked in the mine area (Table 5-3). In addition to streams, 10.2 km^2 of wetland habitat would be eliminated by the minimum footprint, and 17.3 km^2 of wetland habitat would be eliminated by the maximum mine size footprint (Table 5-3). The methods used to estimate these losses are described in Box 5-1.

BOX 5-1. CALCULATION OF STREAMS AND WETLANDS AFFECTED BY MINE SITE AND ROAD NETWORK DEVELOPMENT

For calculation of stream kilometers eliminated, blocked, or altered in flow as a result of mine site development we used the Alaska National Hydrography Dataset (NHD) (USGS 2012). The scale of this dataset is 1:63,360. For the purposes of this assessment, a stream segment is classified as eliminated if it falls within the boundaries of the mine pit, the waste rock pile, or the tailings storage facility (TSF). A stream segment is classified as blocked if it or a downstream segment it connects to directly intersects the mine pit, waste rock pile, or TSF. For calculation of stream kilometers either eliminated or blocked that are inhabited by anadromous and resident fish species we used the Alaska Department of Fish and Game (ADFG) Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region (AWC) (Johnson and Blanche in press) and the Alaska Freshwater Fish Inventory (AFFI) (ADFG 2012). We followed the same methodology for classification of these stream segments as eliminated and blocked as outlined for those in the NHD. Stream lengths either blocked or eliminated were summed across each classification for both NHD and fish distribution stream segments (Table 5-4).

Estimates of wetland area either eliminated or blocked due to mine site development were derived from the NWI available at http://www.fws.gov/wetlands/index.html). For the State of Alaska, the scale of this dataset is 1:63,360. A wetland is classified as eliminated if it falls within the boundaries of the mine pit, waste rock pile, or tailings storage facility. Blocked wetlands were those wetlands that directly intersected a previously categorized blocked NHD stream (Figure 5-9). Wetland area either blocked or eliminated was summed within each classification (Table 5-4).

The NHD, AWC, and AFFI were similarly used to calculate effects of the road corridor on hydrologic features and fish populations. A 30-m NHD digital elevation model (USGS 2012) was used to characterize the slope along NHD stream segments for the calculation of stream length likely to support fish (Table 5-22). For the analysis of road length intersecting and within 200 m of either a stream or wetland, each stream (NHD) or wetland (NWI) was buffered to a distance of 100 m and 200 m and the length within this range was summed across the length of road in the two site watersheds. Similarly, for the area of wetlands within 200 m of the road corridor, the road corridor was buffered and the area of wetlands within that buffered area summed across the length of road. For the area of wetlands directly filled by the road corridor, a road width of 9.1 m was used.

It is important to note that the characterization of both stream length and wetland area affected represents a conservative estimate of the potential effect. The NHD does not capture all stream courses and may underestimate channel sinuosity resulting in underestimates of affected stream length. Additionally, the AWC and the AFFI do not necessarily characterize all potential fish-bearing streams because it is not possible to sample all streams, and there may be errors in identification and mapping. The characterization of wetland area is limited by the resolution of the available NWI data product. Further, in this analysis the mine site components and road network often bisected wetland features and the wetland area falling outside the boundary was assumed to maintain its functionality. We were also unable to determine the effect that mine site and road network development may have on wetlands that had no direct surface connection to a blocked NHD stream segment, but may be connected via groundwater pathways. Together, these limitations likely make our calculations an underestimate of the effect that mine site development would have on hydrologic features in this region. These estimates could be enhanced with improved, higher-resolution mapping, increased sampling of possible fish-bearing waters, and ground-truthing.

Figure 5 9. Streams and Wetlands Lost (Eliminated and Blocked) Under the Minimum and Maximum Mine Footprints

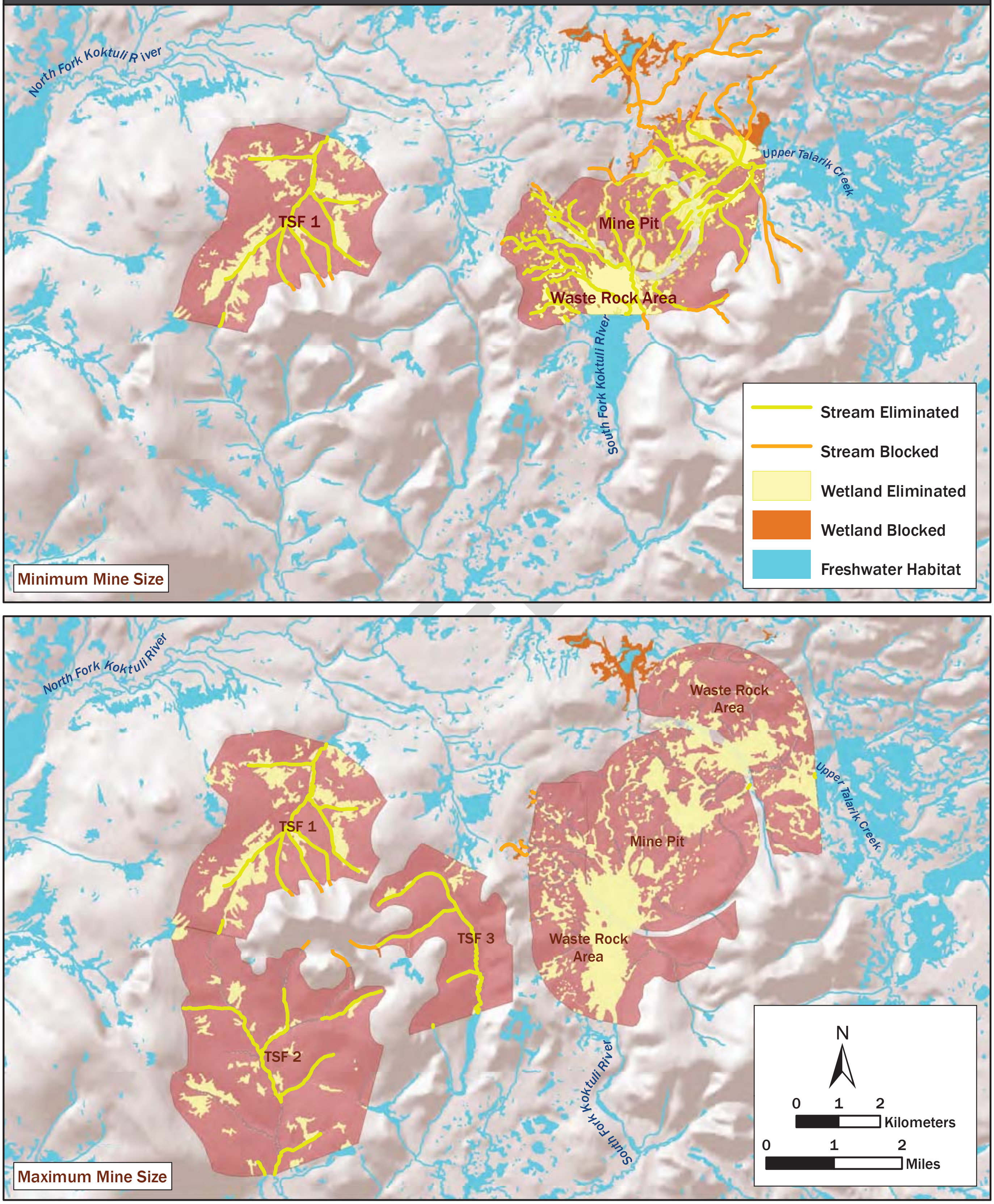


Table 5-4 provides a summary of the total documented anadromous stream length in the site watersheds included in the 2011 AWC. Approximately 7% and 10% of the total anadromous stream kilometers in these watersheds would be either eliminated or blocked by the minimum and maximum mine size footprints, respectively (Table 5-3). Although the amount of both total and documented anadromous headwater streams lost represents a relatively small portion of each watershed, loss of these headwater habitats would also have indirect impacts on fishes and their habitats in downstream mainstem reaches of each watershed (Section 5.2.1.2).

5.2.1.2 Implications of Headwater Stream and Wetland Loss for Fish

Fish Occurrence in Streams and Wetlands Lost to the Mine Footprint

Table 5-3 provides an estimate of salmon habitat directly affected by the mine footprint under the two mine sizes. A total of 21.7 km and 33.8 km of documented anadromous streams would be eliminated or blocked by the minimum and maximum mine sizes, respectively. The distribution of anadromous Dolly Varden in the Kvichak River and Nushagak River watersheds is not known, making an estimate of the total anadromous fish habitat affected by the mine scenario impossible. Of the total wetlands area eliminated or blocked by the footprint, the proportion used by anadromous salmonids or resident fish species is unknown. Fish access to and use of wetlands are likely to be extremely variable in the mine area. This would be expected because of differences in the duration and timing of surface water connectivity with stream habitats, distance from the main channel, or physical and chemical conditions (e.g., dissolved oxygen concentrations (King et al. 2012). Wetlands can provide refuge habitats (Brown and Hartman 1988) and important rearing habitats for juvenile salmonids by providing hydraulically and thermally diverse conditions. Wetlands can also provide enhanced foraging opportunities (Sommer et al. 2001). Given our insufficient knowledge of how fish use wetlands in the mine area, it is not possible to calculate the effects of lost wetland connectivity and abundance on stream fish populations.

Spawning habitat for coho salmon would be lost in the North Fork Koktuli River and South Fork Koktuli River watersheds as a result of TSF 1 and TSF 3, respectively; coho and sockeye salmon spawning habitat would be lost in the Upper Talarik Creek watershed as a result of the waste rock pile footprint (Figures 5-3 and 5-5) (Johnson and Blanche in press). No information on spawning populations of resident fish was found, but in other areas use by anadromous and resident forms of Dolly Varden has been observed in the most upstream and high-gradient habitats available for spawning, indicating that headwaters may be important source areas for downstream populations (Bryant et al. 2004).

In addition to spawning, headwater streams provide rearing habitat for fishes of the site watersheds. Species known to rear in habitats within and upstream of the mine footprint are chum salmon (Figure 5-2), sockeye salmon (Figure 5-3), Chinook salmon (Figure 5-4), coho salmon (Figure 5-5), Dolly Varden (Figure 5-6), rainbow trout (Figure 5-7), Arctic grayling, slimy sculpin, northern pike, and ninespine stickleback (Johnson and Blanche in press, ADFG 2012).

Table 5-3. Stream Kilometers and Wetland Areas (km²) Blocked or Eliminated under the Minimum and Maximum Mine Size Footprints

Mining Impact	Streams Eliminated by Footprint[a] (km)	Streams Blocked by Footprint[a,b] (km)	Wetlands Eliminated by Footprint[a] (km²)	Wetlands Blocked by Footprint[a,b] (km²)	Streams in AWC Eliminated by Footprint[c] (km)	Streams in AWC Blocked by Footprint[b,c] (km)	Anadromous Fish Species Present
Minimum Mine Size							
Mine pit and waste rock	46.6	25.5	6.7	1.9	11.3	4.2	Chinook, sockeye, coho salmon
TSF 1	14.8	0.6	3.5	0.0	6.1	0.0	Chinook, coho salmon
Total	61.4	26.1	10.2	1.9	17.4	4.2	
Maximum Mine Size							
Mine pit and waste rock	77.0	14.1	11.8	1.1	19.2	1.2	Chinook, sockeye, coho salmon
TSF 1	14.8	0.6	3.5	0.0	6.1	0.0	Chinook, coho salmon
TSF 2	24.5	0.9	1.7	0.0	4.9	0.0	Chinook, coho, chum salmon
TSF 3	8.8	0.7	0.3	0.0	2.4	0.0	Coho salmon
Total	125.1	16.3	17.3	1.1	32.6	1.2	

Notes:
[a] From National Hydrography Dataset (USGS 2012)
[b] Includes all streams or lakes and ponds in the watershed at a higher elevation than the footprint
[c] AWC= Anadromous Waters Catalog (Johnson and Blanche in press)

Table 5-4. Total Documented Anadromous Stream Length and Stream Length Documented to Contain Different Fish Species in Site Watersheds

	North Fork Koktuli River (km)	South Fork Koktuli River (km)	Upper Talarik Creek (km)	Total (km)
Total Mapped Streams[a]	343	315	427	1,085
Total Anadromous Streams[b]	104	95	123	322
By species				
Chinook salmon	61	59	63	183
Chum salmon	31	37	45	113
Coho salmon	103	93	122	318
Pink salmon	0	0	7	7
Sockeye salmon	47	64	80	191
Dolly Varden[c]	0	48	26	75

Notes:
[a] From the National Hydrography Dataset (USGS 2012)
[b] From Anadromous Waters Catalog (Johnson and Blanche in press)
[c] Listed as Arctic char in some cases, but assumed to be Dolly Varden (Appendix B)

Importance of Headwater Stream and Wetland Habitats

As a result of their narrow width, headwater streams receive proportionally larger inputs of organic material from the surrounding terrestrial vegetation than do larger stream channels (Vannote et al. 1980). This material is either used within the headwater environment (Tank et al. 2010) or transported downstream as a subsidy to higher-order streams in the network (Wipfli et al. 2007). Consumers in headwater stream food webs, such as invertebrates and juvenile salmon, have been shown to rely heavily on the terrestrial inputs that enter the stream (Doucett et al. 1996, Dekar et al. 2012). Because of their shallow depths and propensity to freeze, headwater streams may be largely uninhabitable in the winter (but see discussion of overwintering below), and fish distribution in headwater systems in southwestern Alaska is likely greatest in summer (Wiedmer pers. comm.). This coincides with the period of maximum growth rates for rearing juvenile salmon—early spring and summer—when both stream temperatures and food availability increase (Quinn 2005:195–196).

Data on riparian vegetation communities specific to the mine footprints were not available, but the EBD vegetation study describes vegetation in the mine area (PLP 2011). Shrub vegetation communities account for 81% of the total area, with four dominant vegetation types: dwarf ericaceous shrub tundra, dwarf ericaceous shrub lichen tundra, open willow low shrub, and closed alder tall shrub (PLP 2011: Chapter 13:10). Riparian areas were dominated by willow and alder shrub communities (PLP 2011: Chapter 13:11). Deciduous shrub species such as alder and willow provide abundant and nutrient-rich leaf litter inputs, which are used more rapidly in stream food webs than coniferous plants or grasses (Webster and Benfield 1986). In addition, alder is a nitrogen-fixing shrub known to increase headwater stream nitrogen concentrations (Compton et al. 2003, Shaftel et al. 2012), which can result in more rapid litter processing rates (Ferreira et al. 2006, Shaftel et al. 2011). The presence of both willow and alder in headwater stream riparian zones implies high-quality basal food resources for stream fishes in the mine area.

In addition to increasing the amount of summer rearing habitat, headwater streams and wetlands may also provide important habitat for stream fishes during other seasons. Loss of wetlands is a common symptom of land development (Pess et al. 2005), and in more developed regions has been associated with reductions in habitat quality and salmon abundance, particularly for coho salmon (Beechie et al. 1994, Pess et al. 2002). Off-channel wetlands can provide thermally diverse habitats that provide rearing and foraging conditions that may be unavailable in the main stream channel (Sommer et al. 2001, Henning et al. 2006), increasing capacity for juvenile salmon rearing (Brown and Hartman 1988). Winter habitat availability for juvenile rearing has been shown to limit salmonid productivity in streams of the Pacific Northwest (Nickelson et al. 1992, Solazzi et al. 2000, Pollock et al. 2004) and may be limiting for fishes in the site watersheds because of the relatively cold temperatures and long winters in southwest Alaska. Overwintering habitats for stream fishes must provide suitable instream cover, dissolved oxygen, and protection from freezing (Cunjak 1996). Beaver ponds and groundwater sources in headwater streams and wetlands in the mine footprints likely meet these requirements.

In winter, beaver ponds typically retain liquid water below the frozen surface, which makes them important winter refugia for coho salmon (Nickelson et al. 1992, Cunjak 1996). Beavers preferentially colonize headwater streams because of their shallow depths and narrow widths, and several studies have indicated that dam densities are reduced significantly at stream gradients above 6 to 9% (Collen and Gibson 2001, Pollock et al. 2003). Beaver ponds provide excellent habitat for rearing salmon because they have high macrophyte cover, low flow velocity, and increased temperatures; and they trap organic materials and nutrients (Nickelson et al. 1992, Collen and Gibson 2001, Lang et al. 2006). Studies in Oregon have shown that salmon abundance is positively related to pool size, especially during low-flow conditions (Reeves et al. 2011), and beaver ponds provide particularly large pools.

An aerial survey of active beaver dams in the mine area, conducted in October 2005 (PLP 2011: Chapter 16:16.2-8), mapped a total of 113 active beaver colonies. The area surveyed did not include the streams draining the TSF 1 footprint (PLP 2011: Figure 16.2-20). Several active beaver colonies were mapped in streams that would be eliminated or blocked by the mine pit and waste rock piles. These are lower-gradient habitats than the headwater streams draining the TSF 1, 2, and 3 footprints. The loss of beaver pond habitats in the headwaters of the South Fork Koktuli River and Upper Talarik Creek watersheds would reduce both summer and winter rearing opportunities for anadromous and resident fish species.

For juvenile salmon, areas with groundwater inputs may be critical for maintaining sufficient free-water areas suitable for overwintering (Cunjak 1996, Huusko et al. 2007, Brown et al. 2011). The best available information on groundwater inputs to headwater streams draining the mine footprint is from two aerial surveys of the site watersheds (PLP 2011, Woody and Higman 2011). Results from the PLP seep inventory indicate that no groundwater sources in these headwater streams would be affected by the TSF 1 and 2 footprints, although numerous seeps are shown in the streams draining the TSF 3, mine pit, and waste rock pile footprints (PLP 2011: Figure 9.1-5). Results from a March 2011 aerial survey indicate partially open water throughout the TSF 2 footprint, in the lower half of the TSF 1 footprint, and in the uppermost extent of the TSF 3 footprint (Woody and Higman 2011). No open waters were documented in the mine pit footprint, but partially open water and open water were documented in the section of Upper Talarik Creek in the waste rock pile footprint. These surveys provide preliminary evidence that the mine scenario would have direct impacts on groundwater sources in the mine area and could result in lost overwintering habitats for stream fishes.

Other Effects of Headwater Stream and Wetland Loss

In addition to providing habitat for stream fishes, headwater streams and wetlands serve an important role in the stream network by contributing nutrients, water, organic material, and macroinvertebrates downstream to higher order streams in the watershed. In the northeastern United States, headwaters contribute approximately 70% of the water volume and 65% of the nitrogen flux to second-order streams and 55% of the volume and 40% of the nitrogen flux to fourth- and higher-order rivers (Alexander et al. 2007). The contributions of headwaters to downstream systems results from their high density in the dendritic stream network. Headwater streams also have high rates of instream nutrient processing and storage due to extensive hyporheic zone interactions resulting from a large bed surface area compared to the volume of the overlying water (Alexander et al. 2007). In addition to nutrients, both invertebrates and detritus are exported from headwaters to downstream reaches and provide an important energy subsidy for juvenile salmonids (Wipfli and Gregovich 2002). This effect can be mediated by the surrounding vegetation; riparian alder (a nitrogen-fixing shrub) was positively related to aquatic invertebrate densities and the export rates of invertebrates and detritus (Piccolo and Wipfli 2002, Wipfli and Musslewhite 2004). Headwater wetlands and associated wetland vegetation can also be important sources of dissolved organic matter, particulate organic matter, and macroinvertebrate diversity (King et al. 2012), contributing to the chemical, physical, and biological condition of downstream waters (Shaftel et al. 2011, Dekar et al. 2012, Walker et al. 2012). The losses of headwater streams and wetlands from the mine footprint would greatly reduce inputs of organic material, nutrients, water, and macroinvertebrates to reaches downstream of the mine footprints, but the effect on fish cannot be quantified.

The inputs of groundwater-influenced streamflow from headwater tributaries likely benefit fish by moderating mainstem temperatures, resulting in reduced freezing in winter and reduced heating in summer (Power et al. 1999, Armstrong et al. 2010). PLP collected temperature data from stream sampling sites using in-situ field meters according to the procedures outlined in their Quality Assurance Project Plan (PLP 2011: Figure 9.1-8). Maximum summer (June through August) water temperatures recorded at gage NK119A, which drains the TSF 1 footprint, were approximately 5°C colder than the mainstem reach that it flows into (PLP 2011: Tables 15.1 through 15.4). This difference was not as pronounced for the maximum summer water temperatures recorded at gage SK119A, which drains the TSF 2 footprint and was approximately 2°C colder than the mainstem reach that it flows into (PLP 2011: Tables 15.1 through 15.21). Longitudinal temperature profiles for the North Fork Koktuli River and South Fork Koktuli River watersheds from August and October indicate that the mainstem reaches (NFK-C and SFK-B) just downstream of the tributaries draining TSF 1 and TSF 2 experience significant cooling in the summer and warming in the winter compared to the adjacent upstream reaches (PLP 2011: Figures 15.1-11 and 15.1-41). Headwater streams in the North Fork Koktuli River and South Fork Koktuli River watersheds may provide a temperature-moderating effect, providing temperatures beneficial to fishes in summer and possibly winter as well.

5.2.2 Effects of Downstream Flow Changes

5.2.2.1 Streamflow

In this section, we describe projected changes in the hydrology of the site watersheds and associated effects on downstream flows resulting from mine development and operation. The mine scenario described in Chapter 4 dictates our estimates of direct fish habitat losses expected from mining activities. We assume that streams under or upstream of the mine footprint would be effectively lost to access by fish from downstream reaches as a result of (1) removal (e.g., loss of stream channels in pit area), (2) elimination under a TSF or waste rock pile, (3) capture into the water treatment footprint of the mine, or (4) diversion of the stream channel in a manner that prevents fish passage (e.g., via pipes or conveyances too steep for fish passage).

The alteration in streamflows resulting from mine operations was estimated by reducing the flows recorded at existing stream gages (Figure 5-8, Table 5-5) for the site watersheds by the percentage of the expected surface area lost to the mine footprint and the area of any drawdown caused by groundwater flow back to the mine pit or locations of dewatering operations (Table 5-6, Box 4-9).

The periods of record varied for the gages in the three site watersheds, but they generally covered the period from 2004 through 2010 and were distributed from the upper reaches to the lower reaches of the watershed along the mainstem drainage course. The tributary area to each stream gage was reported by others (PLP 2011). Using geographic information system (GIS) data, the footprint of each major mine component (e.g., pit, TSF, waste rock piles) was determined (Figure 4-7) and divided as appropriate across the boundaries of the three watersheds (Table 5-6). Assuming that no natural flow or uncontrolled runoff would be generated from the mine footprint, the gage record was reduced by the percentage of area lost to mining.

Expected changes to surface water flows were assessed for three water management stages: start-up, minimum mine operations, and maximum mine operations (Table 4-5, Section 4.3.7). We also considered water balance issues for the post-closure period, but flow estimates were not assessed. The start-up footprint consists of the mine pit, one waste rock pile, and TSF 1. Minimum mine operations would add a second or expanded waste rock pile and the effects of drawdown from groundwater flow to the pit (Section 4.3.7). Maximum mine operations would add effects associated with the fully expanded mine footprint (including TSFs 2 and 3) to accommodate expanded mine operations. The post-closure analysis assumes that active dewatering of the pit has ceased, but that water leaving the site via surface runoff or through groundwater would require capture and treatment for as long as it does not meet water quality standards.

For minimum and maximum mine operations, it was assumed that some flows would be recovered from the mine footprint. These recovered flows could be treated and returned as surface flow to downstream areas. From the minimum and maximum mine sizes (Section 4.3.2), we estimated that the recovery rate for minimum mine operations would be 16% and the recovery rate for maximum mine operations would be 63% (Table 4-5, Table 5-6) of the total water captured. For each of the watersheds, the percentage of recovered flow was applied to the area previously considered as no longer contributing to the natural flow within the watershed (i.e., the mine footprint) and added back to the estimated streamflow for the gaging station downstream of this same area. The spatial extent of these projected changes in streamflow and implications for fish and aquatic habitat are discussed in Section 5.2.2.3.

Table 5-5. Stream Gages and Related Characteristics for Upper Talarik Creek, South Fork Koktuli River, and North Fork Koktuli River

River and Gage Name	Drainage Area (km^2)	Measured Mean Annual Flow (m^3/s)[a]	Mean Annual Unit Runoff ($m^3/s/km^2$)
Upper Talarik Creek			
UT100D	31.0	0.84	0.000030
UT100C1	156.4	3.49	0.000026
UT100C	179.9	4.60	0.000030
UT100B[b]	223.4	6.56	0.000033
South Fork Koktuli			
SK100G	14.2	0.42	0.000031
SK100F	30.9	0.80	0.000032
SK100C	97.1	1.48	0.000016
SK100B1	140.9	3.20	0.000031
SK100B[c]	180.0	5.41	0.000034
North Fork Koktuli			
NK119A	20.1	0.70	0.000041
NK100B	96.6	2.45	0.000030
NK100A1	221.0	5.77	0.000031
NK100A[d]	274.2	7.36	0.000031

Notes:
[a] Reported stream gage data, pre-mine conditions (PLP 2011)
[b] USGS 15300250
[c] USGS 15302200
[d] USGS 15302250

Table 5-6. Pre-Mining Watershed Areas and Mine Footprint Areas for Start-Up, Minimum, and Maximum Mine Sizes for the Site Watersheds

	Pre-Mining	Start-Up			Minimum Mine Size (16% recapture efficiency)				Maximum Mine Size (63% recapture efficiency)			
Stream Gage	Water-shed Area (km²)	Mine Footprint Drainage Area (km²)	% of Original Drainage Area	Flow Returned From Footprint (%)	Mine Footprint Drainage Area (km²)	% of Original Drainage Area	Flow Returned From Footprint (%)	Net Flow Reduction (%)	Mine Footprint Drainage Area (km²)	% of Original Drainage Area	Flow Returned From Footprint (%)	Net Flow Reduction (%)
Upper Talarik Creek Watershed												
UT100D	31.0	2.6	8	0	12.1	39	6	33	27.0	87	55	32
UTC100C2	125.0	2.6	2	0	12.1	10	2	8	27.0	22	14	8
UT100C1	156.4	2.6	2	0	12.1	8	1	7	27.0	17	11	6
UT100C	179.9	2.6	1	0	12.1	7	1	6	27.0	15	9	6
UT100B (USGS 15300250)	223.4	2.6	1	0	12.1	5	1	5	27.0	12	8	4
South Fork Koktuli River												
SK100G	14.2	11.1	78	0	13.13	94	15	79	23.9	100	n/a	100
SK100F	30.9	11.1	36	0	13.13	43	7	36	23.9	78	49	29
SK100C	97.1	11.1	11	0	13.13	14	2	12	32.2	33	21	12
SK100B1	140.9	11.1	8	0	13.13	9	2	8	54.4	39	24	14
SK100B (USGS 15302200)	180.0	11.1	6	0	13.13	7	1	6	54.4	30	19	11
North Fork Koktuli River												
NK119A	20.1	14.6	73	0	15.1	75	12	63	16.9	84	53	31
NK100B	96.6	14.6	15	0	15.1	16	2	13	16.9	17	11	6
NK100A1	221.0	14.6	7	0	15.1	7	1	6	16.9	8	5	3
NK100A (USGS 15302250)	274.2	14.6	5	0	15.1	5	1	5	16.9	6	4	2

Notes:
Minimum and maximum mine sizes assume 16% and 63% water recapture efficiency, which is then returned to streams to yield net flow reduction (%)
See Box 4-9 and text in Section 5.2 for details

Start-Up

For mine start-up, it was assumed that all precipitation falling on the mine footprint would be excluded from approximately 2.6, 11.1, and 14.6 km^2 in the Upper Talarik Creek, South Fork Koktuli River, and North Fork Koktuli River watersheds, respectively (Table 5-6). This is based on the assumption that mine start-up would require capture of surface water and shallow groundwater equivalent to the precipitation-minus-evapotranspiration falling on the mine footprint. Capture of water would be necessary for on-site consumption and for storage of water for use in early construction and start-up activities. The water balance conditions associated with mine start-up would gradually, over a period of years, transition to those described for the minimum mine operations (Section 5.2.2.1).

Based on these defined conditions for the start-up period, we estimate that in each watershed the upper-most gages below the mine site would experience the most significant reductions in streamflow during the start-up period, because they have the highest proportion of contributing area lost to the mine footprint and no water would be returned to streams (Table 5-6). A 8% reduction in streamflow is projected at gage UT100D in the Upper Talarik Creek watershed, a 78% reduction at gage SK100G in the South Fork Koktuli River watershed, and a 73% reduction at gage NK119A in the North Fork Koktuli River watershed (Table 5-7). Projected flow reductions decline in a downstream direction as tributaries and groundwater inputs contribute additional flows. At the lower-most gages in each watershed, projected reductions in streamflow are 1% (Upper Talarik), 6% (South Fork Koktuli River), and 5% (North Fork Koktuli River) (Table 5-6).

Operations: Minimum Mine Size

Under the minimum mine size, the area lost to the mine footprint would increase from the start-up footprint with the addition of a second or expanded waste rock pile and a groundwater cone of depression that would develop around an excavated mine pit and further reduce water flowing to surrounding streams (Figure 4-9, Section 4.3.7). From 1 to 15% of the water captured would be returned to the respective stream as treated water (Table 5-7). After accounting for this returned water, reductions in flow would be most severe for gages UT100D (33% reduction), SK100G, and SK100F (79 and 36% reductions, respectively), and gage NK119A (63% reduction in flow) (Table 5-6). Factoring in this flow return to streams from mining operations results in less severe reductions than if considering only the percentage of surface area lost to the mine footprint (Table 5-6).

Table 5 7. Measured Mean Monthly Pre Mining Flow Rates (m^3/s) (in bold), and Estimated Mean Monthly Flow Rates Under Start up Conditions and the Minimum and Maximum Mine Sizes, at Five Stations Along the South Fork Koktuli River

	SK100G				SK100F				SK100C				SK100B1				SK100B			
Month	Pre	Start-up	Min	Max	Pre-	Start-up	Min	Max	Pre	Start-up	Min	Max	Pre	Start-up	Min	Max	Pre	Start up	Min	Max
January	0.23	0.05	0.05	NA	0.44	0.28	0.28	0.31	0.37	0.33	0.33	0.33	1.54	1.42	1.42	1.33	2.47	2.33	2.33	2.20
February	0.14	0.03	0.03	NA	0.25	0.16	0.16	0.18	0.03	0.03	0.03	0.03	0.79	0.73	0.73	0.68	1.40	1.32	1.32	1.25
March	0.11	0.02	0.02	NA	0.19	0.12	0.12	0.14	0.00	0.00	0.00	0.00	0.57	0.53	0.53	0.49	1.09	1.02	1.02	0.97
April	0.18	0.04	0.04	NA	0.24	0.16	0.16	0.17	0.13	0.11	0.11	0.11	0.80	0.74	0.74	0.69	1.41	1.32	1.32	1.25
May	0.72	0.16	0.15	NA	1.95	1.25	1.25	1.38	4.30	3.83	3.79	3.79	10.75	9.89	9.89	9.25	12.70	11.93	11.93	11.30
June	0.50	0.11	0.10	NA	1.38	0.89	0.89	0.98	2.77	2.46	2.43	2.43	6.67	6.13	6.13	5.73	8.56	8.05	8.05	7.62
July	0.29	0.06	0.06	NA	0.59	0.38	0.38	0.42	0.73	0.65	0.65	0.65	2.56	2.36	2.36	2.21	3.85	3.62	3.62	3.43
August	0.42	0.09	0.09	NA	0.83	0.53	0.53	0.59	1.17	1.04	1.03	1.03	4.05	3.73	3.73	3.48	5.92	5.56	5.56	5.26
September	0.55	0.12	0.12	NA	1.20	0.77	0.77	0.86	2.05	1.82	1.80	1.80	5.18	4.76	4.76	4.45	7.75	7.28	7.28	6.89
October	0.64	0.14	0.13	NA	1.47	0.94	0.94	1.04	2.80	2.49	2.46	2.46	6.12	5.63	5.63	5.26	9.08	8.54	8.54	8.08
November	0.35	0.08	0.07	NA	0.75	0.48	0.48	0.53	1.04	0.92	0.91	0.91	2.84	2.62	2.62	2.44	4.44	4.17	4.17	3.95
December	0.28	0.06	0.06	NA	0.53	0.34	0.34	0.38	0.54	0.48	0.48	0.48	1.92	1.76	1.76	1.65	3.02	2.84	2.84	2.69

Notes:
NA – not applicable, as gage SK100G would be eliminated by TSF 2 under the maximum mine size

Operations: Maximum Mine Size

Under the maximum mine size, the area lost to the mine footprint would increase with inclusion of a larger pit and its associated cone of depression, a substantially larger waste rock pile, and the development of two additional TSFs in the South Fork Koktuli River watershed (Figure 4-7). The drainage area for the maximum mine size would increase downstream with the addition of TSF 2 (on a tributary of South Fork Koktuli River upstream of gage SK100C) and TSF 3 (on a tributary upstream of gage SK100B1) (Figure 5-9, Table 5-6). Gage SK100G would be eliminated under the maximum mine size waste rock piles. Efficiency of water recapture is estimated to be 63%, which would allow higher proportions of water captured in the footprint to be returned to streams. The net effects of lost effective watershed area and recapture and release of water would result in reductions in streamflow that would be most severe for gages UT100D (32% reduction), SK100F (29% reduction), and NFK119A (31% reduction). The physical extent and connectivity of wetlands to one another and to the stream network in the cone of depression would also be reduced, with a concomitant reduction in their associated contributions to salmon rearing habitat as well as detrital inputs and macroinvertebrate support. Furthermore, where the associated streams experience reduced flow, loss of connectivity to wetlands with their associated refugia and contributions to food supply would further impact fish populations which could already experience impacts as a result of the impacts of reduced flow.

Uncertainty

Our assessment of changes in streamflow distributes the losses according to the percentage of the area lost to the mine footprint in a given watershed, and uses flow per unit area of measured data. We assume that the reduced flows follow the same spatial patterns of gaining or losing groundwater reaches as initial (pre-mine) conditions. We acknowledge, however, that mine operations could alter the relative importance of groundwater flowpaths and, therefore, result in a different spatial distribution of streamflow changes than we have reported.

Post-Closure

After the mine closes, pit dewatering would cease, leading to pit filling. As the pit fills, water from the pit that had been returned to streams via pumping to the water treatment facility would no longer be available for streamflow. This period is projected to last at least 100 to 300 years, after which the pit would reach equilibrium with surrounding groundwater, and pit water would flow into the groundwater system where the piezometric gradient allows. Much of this groundwater would eventually discharge to down-gradient streams and ponds (Section 4.3.8.1).

Post-closure streamflows would be a function of the pit cone of depression, and, as necessary, the capture, treatment, and release of water through the water treatment facility. Temporary augmentation of streamflows via TSF drawdown (Section 4.3.8.2) could be possible during this period. Given uncertainties in the post-closure water balance, we have not attempted to estimate streamflows during that period.

5.2.2.2 Stream Temperature

Stream temperatures in the site watersheds could be substantially altered as a result of changes in streamflow, changes in sources of streamflow (e.g., groundwater contributions, inputs of water from a water treatment facility), or other changes to the heat balance of waters eventually entering surface waters. We expect treated water returned to streams to have different thermal characteristics than water derived from groundwater sources (the dominant source prior to mining). The extent and duration of temperature effects depends not only on source water temperatures, but also on the quantity and timing of water contributed from various additional sources, such as tributaries, natural groundwater inputs, or process water released from the water treatment facility. Simple mixing models can be used to estimate stream temperatures below the confluence of multiple sources with known temperature and discharge. However, we cannot use such models here, because we cannot account for all contributions, particularly groundwater (Leach and Moore 2011). In the absence of models, we have relied on available literature to identify the most likely risks to fish associated with projected changes in the mine area.

Changes in water temperature associated with mine development activities are a concern given the importance of suitable water temperatures to Pacific salmon. Water temperature controls the metabolism and behavior of salmon; if temperatures are stressful, fish can be more vulnerable to disease, competition, predation, or death (McCullough et al. 2001). Recognizing the importance of water temperature to healthy salmon populations, the State of Alaska requires that maximum water temperatures not exceed 13^{o}C in spawning areas and 15^{o}C in migration routes and rearing areas (ADEC 2011). This standard is designed to protect against increases in summer temperature, a serious concern for salmon populations particularly in light of projected climate change effects on streamflow and temperatures (Bryant 2009).

Summer is not the only period during which salmon are sensitive to temperatures. Salmon and other native fishes in the mine area rely on suitable temperature regimes to successfully complete their life cycles (Quinn 2005). For locally adapted populations, timing of key life-history events (i.e., spawning, incubation, and out-migration) can be closely tied to the timing of other ecosystem functions that provide critical resources for salmon (Brannon 1987, Quinn and Adams 1996). Thus, changes to thermal and hydrologic regimes that disrupt life-history timing cues can result in mismatches between fish and their environments or food resources, adversely affecting survival (Jensen and Johnsen 1999, Angilletta et al. 2008).

Migration, spawning, and incubation timing are closely tied to fall, winter, and spring water temperatures, allowing a diversity of spawning migration timing to persist (Hodgson and Quinn 2002). For the Bristol Bay region, this asynchrony in spawning timing helps buffer Bristol Bay sockeye salmon populations from climatic events or other environmental changes that may adversely affect a particular run timing (Schindler et al. 2010). An additional benefit of staggered spawner return timing is the extended availability of spawning sockeye salmon to mobile consumers like brown bear (Schindler et al.

2010). Deviations from the thermal regime to which local populations of salmon may be adapted can have serious population-level consequences (Angilletta et al. 2008).

Thermal Regimes in the Mine Area

Extensive glacially reworked deposits with high hydraulic conductivity allow for extensive connectivity between groundwater and surface waters in the region (Power et al. 1999). This groundwater-surface water connectivity has a strong influence on the hydrologic and thermal regimes of streams in the Nushagak River and Kvichak River watersheds, providing a moderating influence against both summer heat and winter cold extremes in stream reaches where this influence is sufficiently strong.

Water temperature data collected by PLP and published in the EBD (PLP 2011: Appendix 15.1E-Attachment 1) indicate significant spatial variability in thermal regimes. The range of spatial variability in temperatures provided in the EBD is consistent with streams influenced by upstream lakes and groundwater contributions (Mellina et al. 2002, Armstrong et al. 2010).

Winter water temperatures are also spatially variable, as indicated by instream temperature monitoring data provided in the EBD and aerial surveys of ice cover (PLP 2011, Woody and Higman 2011). Winter water temperatures can be critical for fish that remain in streams, as freezing conditions can severely limit the availability of suitable habitat (Reynolds 1997), particularly in smaller streams where portions of the channel may freeze solid. Under these conditions, areas of groundwater upwelling can be critical for overwintering fish survival by providing habitat refugia free of anchor ice or surface ice (Brown et al. 2011). Open water can also allow oxygen exchange with the atmosphere to alleviate low oxygen conditions that can otherwise exist in ice-covered streams (Reynolds 1997).

Projecting specific mining-associated changes to groundwater and surface water interactions in the mine area is not feasible at this time. Disruptions or changes to groundwater flowpaths could have significant adverse effects on winter habitat suitability for fish, particularly if groundwater-dominated stream reaches are converted to surface water-dominated systems. Irons et al. (1989 as cited in Reynolds 1997) reported that groundwater-mediated unfrozen refugia were dependent on fall rains maintaining groundwater, but that during a dry year, groundwater levels declined and allowed full freezing of stream surface waters and the streambed. This suggests that the threshold between completely frozen and partially frozen streams can be a narrow one, particularly for small streams with low winter discharge. Maintaining winter groundwater connectivity may be critical for fish in such streams (Cunjak 1996, Huusko et al. 2007, Brown et al. 2011).

5.2.2.3 Fish Populations

Water from streams originating upstream of the footprint (i.e., blocked streams) could be captured by the footprint of the mine, for use or storage on site or eventual treatment and return to the stream via the water treatment facility. We assume that water from blocked streams would be returned to downstream stream segments via diversion channels or pipes. Habitat upstream of the footprint (in blocked streams) is assumed to no longer be accessible to fish downstream because of the inability of fish to move upstream through diversion channels or pipes.

Altered Streamflow Regimes: Start-Up

Altered streamflows can have various effects on aquatic life. Short-term effects include reduced habitat availability resulting from water withdrawal (effects on winter habitat reviewed by West et al. 1992, Cunjak 1996) and reduced habitat quality resulting from extreme and rapid fluctuations in flow if withdrawals are intermittent (Curry et al. 1994, Cunjak 1996). Temporal variability in flows is a natural feature of stream ecosystems (Poff et al. 1997), although the degree of variability differs depending on hydrologic controls, including climate, geology, landform, human land use, and relative groundwater contributions (Poff et al. 2006). Fish populations may be adapted to periodic disturbances such as droughts, and may quickly recover under improved hydrologic conditions but this is contingent upon many factors (Matthews and Marsh-Matthews 2003). Longer-term effects of prolonged changes in streamflow regime can have lasting impacts on fish populations (Lytle and Poff 2004).

The natural flow paradigm is widely supported and is based on the premise that natural flow variability, including the magnitude, frequency, timing, duration, rate of change and predictability of flow events, and the sequence of conditions, is crucial to maintaining healthy aquatic ecosystems (Postel and Richter 2003, Arthington et al. 2006, Poff et al. 2009). However, numerous human demands can directly alter the natural flow of the system, potentially affecting ecosystem function and structure. Guidelines for minimizing impacts of altered hydrologic regimes have been offered by several researchers (Poff et al. 1997, Poff et al. 2009, Richter 2010). Determining the natural flow regime is a data-intensive process, but it is crucial to understanding how to manage flow within a system (Arthington et al. 2006).

Given the high likelihood of complex groundwater-surface water connectivity in the mine area, predicting and regulating flows to maintain key ecosystem functions associated with groundwater-surface water exchange is particularly challenging. PLP has invested in a relatively intensive network of stream gages, water temperature monitoring sites, fish assemblage sampling sites, groundwater monitoring wells, and geomorphic cross-section locations. The integration of information gathered by this process will help identify relationships among surface water flow, groundwater and surface water temperatures, and instream habitat for fish (Bartholow 2010). However, until linkages between biology, groundwater, surface water, and proposed activities can be better predicted and understood, a protective approach would identify and maintain surface and groundwater flows in the mine area within natural flow regimes.

The sustainability boundary approach is one way to balance the maintenance of aquatic ecosystems with human demands on the system (Richter et al. 2011). With this approach, percentage-based deviations from natural conditions are used to set the limits of flow alteration daily. These percentages are based on natural flow and do not focus on the more simplistic approach of setting a percentage based on a high-flow or low-flow event. Numerous case studies have tested this type of approach, and the percentage bounds of flow alteration around natural daily flow that caused measurable ecological harm were determined to be similar regardless of the geographic location (Richter et al. 2011). Based on these studies, Richter et al. (2011) proposed that flow alteration be managed based on the following daily percentage flow alteration thresholds.

- A flow alteration below 10% would cause minor impacts on the system with a relatively high level of ecosystem protection.
- A flow alteration of 11 to 20% would cause measurable changes in structure and minor impacts on ecosystem functions.
- A flow alteration greater than 20% would result in moderate to major changes in ecosystem structures and functions. Increasing alteration beyond 20% would cause significant losses of ecosystem structures and functions. Losses could include reduced habitat availability for salmon and other stream fish particularly during low-flow periods (West et al. 1992, Cunjak 1996), reductions in macroinvertebrate production (Chadwick and Huryn 2007), and increased fragmentation of stream habitats through increased frequency and duration of stream drying. These losses could significantly decrease salmon habitat quantity and quality in these watersheds.

We used this sustainability boundary approach to determine natural daily flows and evaluate the risks associated with potential alterations to flow throughout the site watersheds. Daily flow data were obtained using the EBD data from four gages in Upper Talarik Creek, five gages in the South Fork Koktuli River , and four gages in the North Fork Koktuli River (Table 5-5). We determined mean monthly and minimum monthly flows for each gage, for start-up and the minimum and maximum mine sizes (Tables 5-8 through 5-12). We then compared the predicted flows (Section 5.2.2.1) with the boundary limits of 10 and 20% flow alterations around mean daily flow. Figures 5-9 through 5-12 show natural flows, 10 and 20% alteration flows, and predicted flows for each gage under the minimum size mine operating conditions. Values are plotted as mean monthly flows for clarity.

Table 5 8. Measured Mean Monthly Pre Mining Flow Rates (m^3/s) (in bold), and Estimated Mean Monthly Flow Rates Under Start Up Conditions and the Minimum and Maximum Mine Sizes, at Four Stations Along Upper Talarik Creek

	UT100D				UT100C1				UT100C				UT100B			
Month	Pre-	Start-up	Min	Max	Pre-	Start-up	Min	Max	Pre-	Start-up	Min	Max	Pre-	Start-up	Min	Max
January	**0.32**	0.30	0.22	0.22	**1.74**	1.70	1.62	1.63	**2.45**	2.43	2.31	2.31	**3.62**	3.59	3.44	3.48
February	**0.28**	0.26	0.19	0.19	**1.55**	1.52	1.44	1.45	**2.25**	2.23	2.12	2.12	**3.31**	3.28	3.14	3.18
March	**0.22**	0.20	0.15	0.15	**1.28**	1.26	1.19	1.20	**1.98**	1.96	1.86	1.86	**2.88**	2.85	2.74	2.76
April	**0.55**	0.51	0.37	0.37	**2.51**	2.46	2.34	2.36	**3.44**	3.40	3.23	3.23	**4.79**	4.74	4.55	4.60
May	**1.95**	1.79	1.31	1.33	**7.43**	7.28	6.91	6.98	**9.11**	9.02	8.57	8.57	**12.80**	12.68	12.16	12.29
June	**1.02**	0.94	0.68	0.69	**4.29**	4.21	3.99	4.04	**5.63**	5.58	5.29	5.29	**7.40**	7.33	7.03	7.11
July	**0.62**	0.57	0.41	0.42	**2.76**	2.71	2.57	2.60	**3.77**	3.74	3.55	3.55	**5.13**	5.08	4.87	4.92
August	**0.78**	0.72	0.52	0.53	**3.30**	3.24	3.07	3.11	**4.38**	4.34	4.12	4.12	**6.48**	6.42	6.16	6.22
September	**1.03**	0.95	0.69	0.70	**4.67**	4.58	4.34	4.39	**6.09**	6.03	5.72	5.72	**7.82**	7.74	7.43	7.51
October	**1.18**	1.08	0.79	0.80	**5.25**	5.16	4.89	4.95	**6.66**	6.61	6.27	6.27	**9.08**	8.99	8.63	8.72
November	**0.74**	0.68	0.50	0.51	**3.68**	3.60	3.42	3.45	**4.60**	4.55	4.32	4.32	**6.34**	6.28	6.02	6.09
December	**0.52**	0.48	0.35	0.35	**2.61**	2.56	2.43	2.46	**3.37**	3.33	3.16	3.16	**5.00**	4.95	4.75	4.80

Table 5 9. Measured Mean Monthly Pre Mining Flow Rates (m^3/s) (in bold), and Estimated Mean Monthly Flow Rates Under Start up Conditions and the Minimum and Maximum Mine Sizes, at Four Stations Along the North Fork Koktuli River

	NK119A				NK100B				NK100A1				NK100A			
Month	Pre	Start-up	Min	Max	Pre	Start-up	Min	Max	Pre	Start-up	Min	Max	Pre	Start-up	Min	Max
January	**0.15**	0.04	0.05	0.10	**1.04**	0.89	0.91	0.98	**2.08**	1.93	1.96	2.02	**2.85**	2.71	2.71	2.79
February	**0.10**	0.03	0.04	0.07	**0.67**	0.57	0.58	0.63	**1.44**	1.34	1.36	1.40	**1.88**	1.79	1.79	1.84
March	**0.08**	0.02	0.03	0.06	**0.54**	0.46	0.47	0.51	**1.23**	1.14	1.15	1.19	**1.55**	1.48	1.48	1.52
April	**0.21**	0.06	0.08	0.14	**0.88**	0.75	0.76	0.83	**2.17**	2.02	2.04	2.11	**2.66**	2.53	2.53	2.61
May	**2.28**	0.62	0.84	1.58	**7.03**	5.97	6.12	6.61	**16.57**	15.41	15.58	16.07	**20.10**	19.10	19.10	19.70
June	**1.15**	0.31	0.42	0.79	**3.64**	3.09	3.16	3.42	**9.48**	8.81	8.91	9.19	**11.39**	10.82	10.82	11.16
July	**0.55**	0.15	0.20	0.38	**2.04**	1.73	1.78	1.92	**5.13**	4.77	4.83	4.98	**5.88**	5.59	5.59	5.77
August	**0.71**	0.19	0.26	0.49	**2.44**	2.08	2.13	2.30	**6.21**	5.77	5.83	6.02	**7.40**	7.03	7.03	7.25
September	**1.10**	0.30	0.41	0.76	**3.31**	2.81	2.88	3.11	**7.98**	7.42	7.50	7.74	**9.35**	8.88	8.88	9.16
October	**1.10**	0.30	0.41	0.76	**4.01**	3.41	3.49	3.77	**9.40**	8.74	8.84	9.12	**11.14**	10.58	10.58	10.91
November	**0.52**	0.14	0.19	0.36	**2.12**	1.81	1.85	2.00	**4.79**	4.45	4.50	4.64	**5.95**	5.65	5.65	5.83
December	**0.24**	0.07	0.09	0.17	**1.35**	1.15	1.18	1.27	**2.89**	2.69	2.72	2.80	**3.84**	3.65	3.65	3.76

Table 5 10. Measured Minimum Monthly Pre Mining Flow Rates (m^3/s) (in bold), and Estimated Minimum Monthly Flow Rates Under Start up Conditions and the Minimum and Maximum Mine Sizes, at Four Stations Along Upper Talarik Creek

	UT100D				UT100C1				UT100C				UT100B			
Month	Pre	Start-up	Min	Max	Pre	Start-up	Min	Max	Pre	Start-up	Min	Max	Pre	Start-up	Min	Max
January	**0.12**	0.11	0.08	0.08	**0.80**	0.78	0.74	0.75	**1.55**	1.54	1.46	1.46	**2.09**	2.07	1.99	2.01
February	**0.10**	0.10	0.07	0.07	**0.73**	0.71	0.68	0.68	**1.48**	1.47	1.39	1.39	**1.98**	1.96	1.88	1.90
March	**0.12**	0.11	0.08	0.08	**0.80**	0.78	0.74	0.75	**1.37**	1.36	1.29	1.29	**2.09**	2.07	1.99	2.01
April	**0.11**	0.10	0.08	0.08	**0.76**	0.75	0.71	0.72	**1.42**	1.41	1.34	1.34	**2.04**	2.02	1.94	1.96
May	**0.22**	0.21	0.15	0.15	**1.25**	1.23	1.16	1.18	**2.02**	2.00	1.90	1.90	**2.83**	2.80	2.69	2.72
June	**0.23**	0.21	0.16	0.16	**1.57**	1.54	1.46	1.48	**2.85**	2.82	2.68	2.68	**2.58**	2.55	2.45	2.47
July	**0.21**	0.19	0.14	0.14	**1.37**	1.35	1.28	1.29	**2.50**	2.47	2.35	2.35	**2.55**	2.52	2.42	2.45
August	**0.22**	0.20	0.15	0.15	**1.58**	1.55	1.47	1.48	**2.40**	2.37	2.25	2.25	**2.97**	2.94	2.82	2.85
September	**0.20**	0.18	0.13	0.13	**1.52**	1.49	1.41	1.43	**2.37**	2.34	2.22	2.22	**2.83**	2.80	2.69	2.72
October	**0.33**	0.30	0.22	0.23	**2.24**	2.19	2.08	2.10	**3.03**	3.00	2.85	2.85	**3.82**	3.78	3.63	3.67
November	**0.31**	0.28	0.20	0.21	**2.04**	1.99	1.89	1.91	**2.36**	2.33	2.21	2.21	**3.68**	3.64	3.50	3.53
December	**0.22**	0.21	0.15	0.15	**1.25**	1.23	1.16	1.18	**1.83**	1.81	1.72	1.72	**2.83**	2.80	2.69	2.72

Table 5 11. Measured Minimum Monthly Pre Mining Flow Rates (m^3/s) (in bold), and Estimated Minimum Monthly Flow Rates Under Start up Conditions and the Minimum and Maximum Mine Sizes, at Five Stations Along the South Fork Koktuli River

	SK100G				SK100F				SK100C				SK100B1				SK100B			
Month	Pre	Start up	Min	Max	Pre	Start up	Min	Max	Pre	Start up	Min	Max	Pre	Start up	Min	max	Pre	Startu p	Min	Max
January	**0.11**	0.02	0.02	NA	**0.20**	0.13	0.13	0.14	**0.00**	0.00	0.00	0.00	**0.60**	0.55	0.55	0.52	**1.13**	1.06	1.06	1.01
February	**0.08**	0.02	0.02	NA	**0.15**	0.09	0.09	0.10	**0.00**	0.00	0.00	0.00	**0.40**	0.37	0.37	0.35	**0.85**	0.80	0.80	0.76
March	**0.07**	0.01	0.01	NA	**0.11**	0.07	0.07	0.08	**0.00**	0.00	0.00	0.00	**0.27**	0.24	0.24	0.23	**0.65**	0.61	0.61	0.58
April	**0.04**	0.01	0.01	NA	**0.11**	0.07	0.07	0.08	**0.00**	0.00	0.00	0.00	**0.27**	0.24	0.24	0.23	**0.65**	0.61	0.61	0.58
May	**0.08**	0.02	0.02	NA	**0.14**	0.09	0.09	0.10	**0.00**	0.00	0.00	0.00	**0.38**	0.35	0.35	0.32	**0.79**	0.75	0.75	0.71
June	**0.20**	0.04	0.04	NA	**0.46**	0.30	0.30	0.33	**0.12**	0.11	0.10	0.10	**1.51**	1.39	1.39	1.30	**2.49**	2.34	2.34	2.22
July	**0.08**	0.02	0.02	NA	**0.22**	0.14	0.14	0.15	**0.00**	0.00	0.00	0.00	**1.12**	1.03	1.03	0.96	**1.64**	1.54	1.54	1.46
August	**0.08**	0.02	0.02	NA	**0.16**	0.10	0.10	0.11	**0.00**	0.00	0.00	0.00	**0.67**	0.62	0.62	0.58	**1.25**	1.17	1.17	1.11
September	**0.06**	0.01	0.01	NA	**0.08**	0.05	0.05	0.06	**0.00**	0.00	0.00	0.00	**0.51**	0.47	0.47	0.44	**1.02**	0.96	0.96	0.91
October	**0.22**	0.05	0.05	NA	**0.63**	0.40	0.40	0.45	**0.71**	0.63	0.62	0.62	**2.10**	1.93	1.93	1.80	**3.54**	3.33	3.33	3.15
November	**0.18**	0.04	0.04	NA	**0.34**	0.22	0.22	0.24	**0.12**	0.10	0.10	0.10	**1.16**	1.07	1.07	1.00	**1.93**	1.81	1.81	1.71
December	**0.12**	0.03	0.02	NA	**0.21**	0.14	0.14	0.15	**0.00**	0.00	0.00	0.00	**0.66**	0.61	0.61	0.57	**1.22**	1.14	1.14	1.08

Notes:
NA – not applicable, as gage SK100G would be eliminated by TSF 2 under the maximum mine size

Table 5 12. Measured Minimum Monthly Pre Mining Flow Rates (m^3/s) (in bold), and Estimated Minimum Monthly Flow Rates Under Start up Conditions and the Minimum and Maximum Mine Sizes, at Four Stations Along the North Fork Koktuli River

	NK119A				NK100B				NK100A1				NK100A			
Month	Pre-	Start up	Min	Max	Pre-	Start up	Min	Max	Pre-	Start up	Min	Max	Pre-	Start up	Min	Max
January	**0.08**	0.02	0.03	0.05	**0.43**	0.37	0.38	0.41	**0.93**	0.86	0.87	0.90	**1.10**	1.05	1.05	1.08
February	**0.07**	0.02	0.03	0.05	**0.44**	0.38	0.38	0.42	**0.95**	0.88	0.89	0.92	**1.13**	1.08	1.08	1.11
March	**0.06**	0.02	0.02	0.04	**0.33**	0.28	0.29	0.31	**0.80**	0.74	0.75	0.77	**0.91**	0.86	0.86	0.89
April	**0.04**	0.01	0.02	0.03	**0.18**	0.16	0.16	0.17	**0.84**	0.78	0.79	0.81	**0.96**	0.91	0.91	0.94
May	**0.01**	0.00	0.00	0.01	**0.54**	0.46	0.47	0.51	**1.16**	1.07	1.09	1.12	**1.44**	1.37	1.37	1.41
June	**0.30**	0.08	0.11	0.21	**1.31**	1.11	1.14	1.23	**3.75**	3.48	3.52	3.63	**4.27**	4.06	4.06	4.19
July	**0.21**	0.06	0.08	0.14	**1.04**	0.88	0.90	0.98	**2.57**	2.39	2.42	2.49	**2.35**	2.23	2.23	2.30
August	**0.14**	0.04	0.05	0.10	**0.96**	0.81	0.83	0.90	**2.02**	1.88	1.90	1.96	**1.93**	1.83	1.83	1.89
September	**0.12**	0.03	0.05	0.08	**0.91**	0.77	0.79	0.85	**1.89**	1.76	1.78	1.83	**1.76**	1.67	1.67	1.72
October	**0.20**	0.05	0.07	0.14	**1.53**	1.30	1.33	1.44	**3.19**	2.97	3.00	3.09	**4.39**	4.17	4.17	4.30
November	**0.12**	0.03	0.05	0.08	**0.71**	0.61	0.62	0.67	**1.51**	1.40	1.42	1.46	**1.98**	1.88	1.88	1.94
December	**0.10**	0.03	0.04	0.07	**0.57**	0.48	0.50	0.53	**1.21**	1.12	1.14	1.17	**1.53**	1.45	1.45	1.50

Figure 5-10. Sustainability Boundary for Upper Talarik Creek Based on Mean Monthly Flow for the Minimum Mine Size, through Four Gages (Upstream to Downstream: UT100D, UT100C1, UT100C, UT100B).

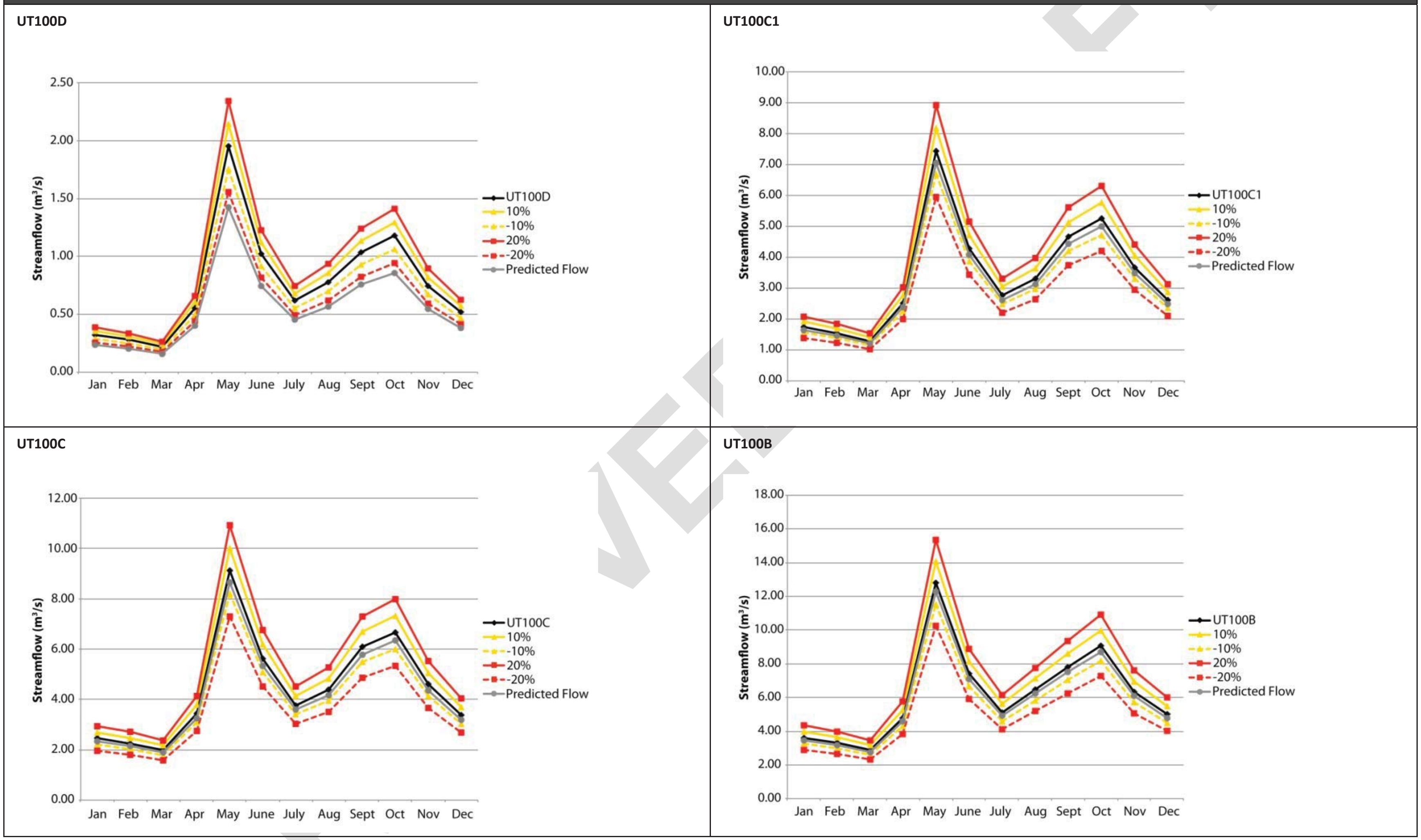


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Figure 5-11. Sustainability Boundary for South Fork Koktuli River Based on Monthly Mean Flow for the Minimum Mine Size, through Four Gages (Upstream to Downstream: UT100D, UT100C1, UT100C, UT100B).

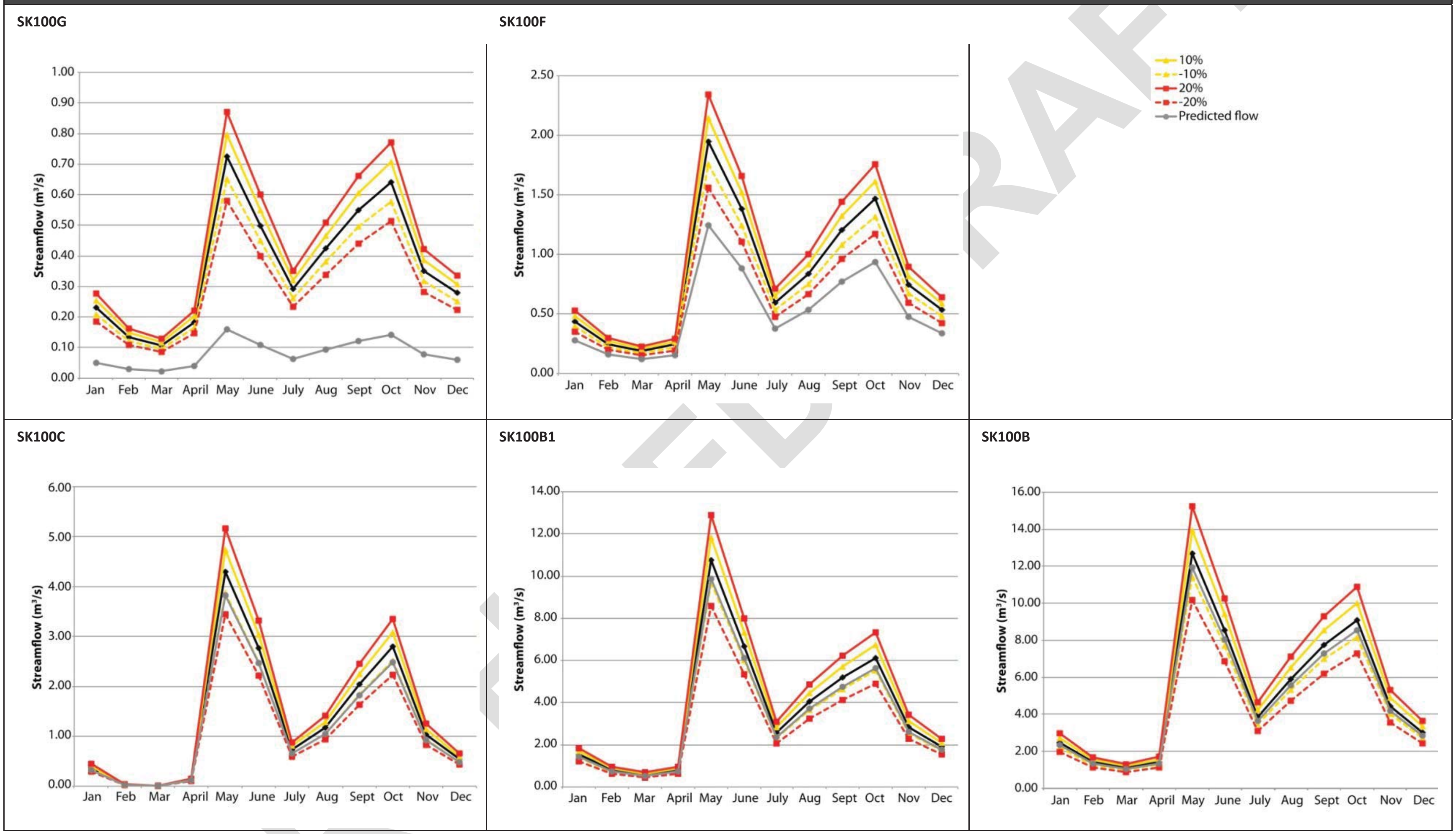
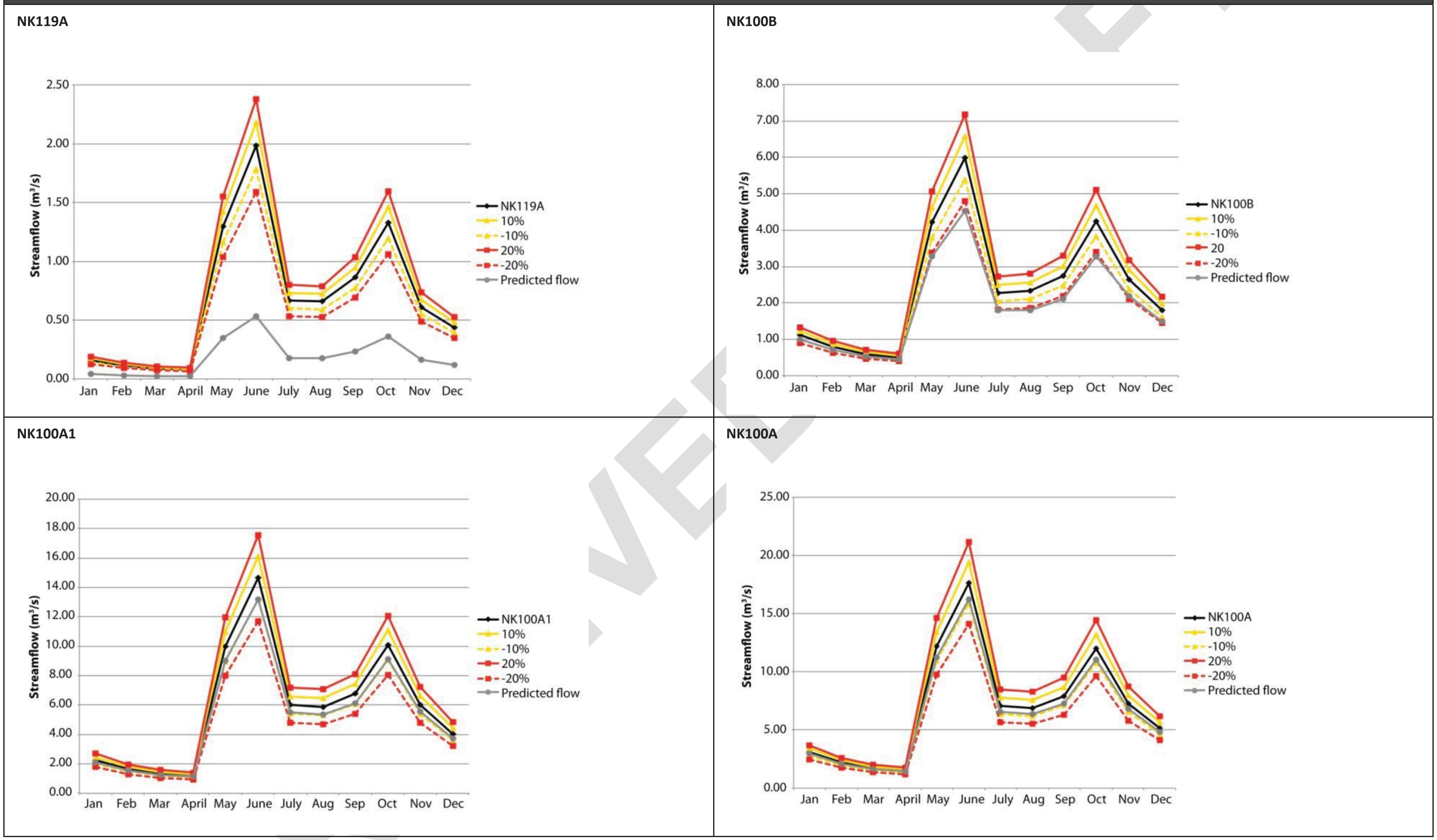


Figure 5-12. Sustainability Boundary for North Fork Koktuli River Based on Monthly Mean Flow for the Minimum Mine Size, through Four Gages (Upstream to Downstream: NK119A, NK100B, NK100A1, and NK100A).



To estimate the spatial extent of deleterious reductions in streamflow in the site watersheds, we calculated the length of stream network upstream of the upper-most stream gage in each site watershed. This estimate was made for the minimum mine size, to illustrate the spatial extent of streamflow modification that would be expected for a mine of that size. The projected streamflow estimates are based on a percentage of the watershed area above a stream gage that would be removed by the mine footprint and would longer contribute to streamflow (Section 5.2.2). Thus, the estimates for reduction in flow are for downstream endpoints at the stream gage. These point estimates may be inferred to reflect an overall reduction in flow that the subwatershed might experience, but not all sections of stream in these upper portions of the watersheds would experience similar reductions in flow. Some stream sections directly under the mine footprint would be totally lost (Section 5.2.1.1), whereas stream sections closer to the mine footprint would experience greater reductions in flow than those at the downstream gages. Additionally, other stream sections that drain outside of the mine footprint might maintain pre-mine streamflows. Water from streams originating upstream of the footprint (i.e., blocked streams) could be captured by the footprint of the mine, for use or storage on site, or eventual treatment and return to the stream via the water treatment facility. We assume that water from blocked streams would be returned to stream segments downstream, via diversion channels or pipes. Habitat upstream of the footprint (in blocked streams) would no longer be accessible to fish downstream because fish could not move upstream through diversion channels or pipes. Stream sections throughout the stream network could be affected indirectly, via reductions in flow downstream that could preclude use of downstream habitats by fish that move seasonally between headwater and mainstem habitats. Similarly, these stream sections could be isolated by downstream flow reductions that reduce or eliminate the potential for movement of fish into those areas from downstream.

During the mine start-up period, the Upper Talarik Creek watershed would be affected by preparation and development of the mine pit footprint. Resulting streamflows at gage UT100D are expected to be reduced by 8% (Table 5-6). The mainstem reaches downstream of gage UT100D in Upper Talarik Creek would experience flow reductions ranging from 1 to 2%.

In the South Fork Koktuli River watershed, the South Fork Koktuli River mainstem and tributaries upstream of gage SK100G would either be eliminated by the mine footprint or would suffer severe flow reductions (78% at SK100G Table 5-13). The South Fork Koktuli River below Frying Pan Lake appears to be a losing reach, and under pre-mine conditions experiences periods of zero discharge at gage SK100C (Table 5-11). With projected reductions in streamflow, the frequency and duration of periods of zero flow would be expected to increase, resulting in increased habitat fragmentation for fish including salmon. Alteration to the natural flow regime of this magnitude would have a very significant adverse effect on salmonid populations and overall ecosystem functioning in these portions of the watershed.

Table 5-13. Estimated Decreases in Streamflow Under the Minimum and Maximum Mine Size, and Subsequent Stream Lengths Affected

River and Gage Name	Minimum Mine Size		Maximum Mine Size	
	Estimated Decrease in Streamflow (%)	Stream Length Affected (km)	Estimated Decrease in Streamflow (%)	Stream Length Affected (km)
Upper Talarik Creek				
UT100D	**33**	4.9	**32**	0.15
UT100C1	7	14.0	6	14.0
UT100C	6	8.3	6	8.3
UT100B[a]	5	4.8	4	4.8
South Fork Koktuli				
SK100G	**79**	0.5	NA	NA
SK100F	**36**	3.3	**29**	0.8
SK100C	*12*	18.9	*12*	18.9
SK100B1	8	6.6	*14*	6.6
SK100B[b]	6	4.4	*11*	4.4
North Fork Koktuli				
NK119A	**63**	0.8	**31**	0.8
NK100B	*13*	0.8	6	0.8
NK100A1	6	22.1	3	22.1
NK100A[c]	5	8.4	2	8.4

Notes:

When % streamflow decrease exceeds 20% (bold), major effects on salmon populations would be expected; when % streamflow decrease falls between 11% and 20% (italics), moderate effects on salmon populations would be expected.

For UT100D, SK100G, and NK119A, stream length affected includes mainstem length upstream to edge of mine footprint only, and does not include upstream lengths, including tributaries, that are blocked or eliminated by the mine footprint.

[a] USGS 15300250

[b] USGS 15302200

[c] USGS 15302250

In the upper reaches of the North Fork Koktuli River (upstream of NK119A), the mainstem and tributaries would experience direct loss of habitat to the mine footprint or substantial loss in flow (73% reduction at gage NK119A). Downstream of gage NK119A, flow reductions of 15%, 7%, and 5% at gages NK100B, NK100A1, and NK100A respectively, would be expected (Table 5-6).

In summary, reductions in flow across all three site watersheds are predicted to occur as a result of water demand associated with mine start-up conditions. The Upper Talarik Creek watershed is projected to experience an 8% reduction in flow at gage UT100D (Table 5-6). The South Fork Koktuli River watershed is projected to experience a 78% reduction in flow at gage SK100G. and a 36% reduction in flow at gage SK 100F (Table 5-6). The North Fork Koktuli River watershed is projected to experience a 73% reduction in flow at gage NK119A (Table 5-6). The flow reductions predicted in the upper South Fork Koktuli and North Fork Koktuli River watersheds are well beyond the 20% limit set by the sustainability boundary approach.

Altered Streamflow Regimes: Minimum and Maximum Mine Sizes

Under the minimum and maximum mine size operations, the area considered lost to the mine footprint would increase from the start-up footprint because of a second or expanded waste rock pile and because of a groundwater cone of depression that would develop around an excavated mine pit and further reduce water flowing to surrounding streams. The proportional reductions in streamflow from the mine footprint under the minimum and maximum mine sizes would be partially offset by water recapture and return to the streams at the mine site (Table 5-6). As a result, increasing proportions of the streamflow under the minimum and maximum mine sizes would be made up of recaptured water that was returned to the stream as a point source and likely passed through a water treatment facility (Section 4.3.7). The implications of this for water temperature and chemistry are discussed in Sections 5.2.2.2 and 5.3.1, respectively.

Minimum Mine Size

For the minimum mine size, the mine footprint captures 39% of the Upper Talarik watershed above gage UT100D (Table 5-6). As a result, most of the total stream length in the upstream reaches of Upper Talarik Creek watershed, including the mainstem and all tributaries above gage UT100D, would experience either total loss of habitat from the mine footprint, or indirect effects of fragmentation (Section 5.2.1, Figure 5-9). Of this stream length, 4.9 km of mainstem would experience a significant loss of habitat and decline in habitat quality from the predicted 33% reduction in streamflow. Downstream of gage UT100D in Upper Talarik Creek, flow reductions would range from 5 to 7% (Table 5-13). Impacts on salmon habitat from flow reduction would be moderated by inputs of tributary flow and groundwater that may help ameliorate flow losses originating upstream, assuming that groundwater sources and flowpaths are not also altered by the mine footprint. This assumption is questionable (Section 5.2.4).

In the South Fork Koktuli River and North Fork Koktuli River watersheds, reductions in streamflow would be slightly less severe under the minimum mine operations than under start-up conditions as a result of increased rates of water return to streams (Table 5-6). However, anticipated reductions in streamflow would still exceed the 20% sustainability threshold for stream gage stations in the upper South Fork Koktuli River and North Fork Koktuli River watersheds (gages SK100G, SK100F, and NK119A). In the South Fork Koktuli River mainstem and tributaries upstream of gage SK100G, the majority of the stream length would be eliminated by the mine footprint (Figure 5-9), resulting in severe flow reductions (78%) at gage SK100G (Table 5-6). The impact of reduced flow in the South Fork Koktuli River would continue downstream for an additional 22 km of mainstem habitat, with flow reductions of 36% (3 km) and 12% (19 km) between the uppermost gage (SK100G) and the next two gages downstream (SK100F and SK100C). Downstream of gage SK100C, flow reductions of 8% and 6% at SK100B and SK100A, respectively, would be expected for an additional 11 km of mainstem stream (Table 5-13).

In the North Fork Koktuli River, the majority of stream length above gage NK119A would be eliminated by construction of TSF 1 (Figure 5-9), resulting in substantial loss in flow (73% reduction at gage

NK119A) for approximately 1 km of stream between TSF 1 and gage NK119A. Downstream of NK119A, flow reductions of 13%, 6%, and 5% at gages NK100B, NK100A1, and NK100A, respectively, would be expected for an additional 31 km of the North Fork Koktuli River mainstem (Table 5-13).

Maximum Mine Size

The maximum mine size would capture an even larger portion of the Upper Talarik Creek, South Fork Koktuli River, and North Fork Koktuli River watersheds, but increased rates of water recapture and return to the stream would largely compensate for reduced streamflows. As a result, predicted streamflow reductions for the maximum size mine are slightly less severe than for the minimum footprint for gages in the Upper Talarik Creek and North Fork Koktuli River watersheds (Tables 5-6, 5-13). In the South Fork Koktuli River, flow reductions are more severe in the lower mainstem at gages SK100B1 and SK 100B because of the additional water demands of TSF 2 and TSF 3 under the maximum mine size (Table 5-13, Figure 5-9). Additional losses of streamflow are anticipated in the tributaries to the South Fork Koktuli River in response to the construction of TSF 2 and TSF 3 under the maximum mine size. These reductions influence flow calculations in the South Fork Koktuli River mainstem, but are not assessed for the tributaries as only mainstem gages were used for this assessment.

Post-closure streamflows would be a function of several factors, including but not limited to the pit cone of depression, pit refilling, and the capture, treatment, and release of water that fails to meet water quality standard through the water treatment facility. Temporary augmentation of streamflows via TSF drawdown (Section 4.3.8.2) could be possible during this period. Given uncertainties in the post-closure water balance, we have not attempted to estimate streamflows during the post-closure period.

Reductions in flow and losses of stream habitat of the magnitudes estimated for the start-up, minimum, and maximum operation periods represent substantial risks to spawning and rearing habitat for populations of coho, sockeye, and Chinook salmon; Dolly Varden; and rainbow trout in the upper portions of these watersheds. Habitat quantity and quality would be significantly diminished by the loss of flow from the mine site resulting from multiple mechanisms, including a direct reduction in the area and volume of habitat, the loss of channel to off-channel habitat connectivity, increased periods of zero flow, and reduced food production. Although the loss of salmonid production cannot be estimated, flow reductions greater than 20% would be expected to have substantial effects based on those mechanisms and on the substantial effects on stream structure and function (Richter et al. 2011).

Connectivity and Timing/Duration of Off-Channel Habitats

Loss of streamflow resulting from the mine footprint and potential water withdrawals (Section 5.2.2.1) would affect connectivity between the main channel and off-channel habitats important to juvenile salmonids. Loss of flood peaks could alter groundwater recharge rates, influencing characteristics of floodplain percolation channels, seeps, or other expressions of the hyporheic zone (Hancock 2002). Rapid reductions in streamflow that exceed recession rates typically experienced by fish in these systems could result in stranding or isolation in off-channel habitats (Bradford et al. 1995). Off-channel habitats, particularly those with groundwater connectivity, are critical rearing habitats for several species of juvenile salmonids and can be important spawning habitats for sockeye salmon (Quinn 2005). Maintaining connectivity and the physical and chemical attributes of these habitats in conditions similar to baseline conditions will be important for minimizing risks to salmon and other native fishes.

Wetlands that are hydrologically connected to affected streams would also respond to alterations in streamflow and groundwater. Fish access to and use of wetlands are likely to be extremely variable in the mine area because of differences in the duration and timing of surface water connectivity with stream habitats, distance from the main channel, or physical and chemical conditions (e.g., dissolved oxygen concentrations) (King et al. 2012). Projecting the effects of lost wetland connectivity and abundance on stream fish populations is beyond the scope of this assessment, but could be a significant unknown.

Once the mine is no longer a net consumer of water, we assume that flow regulation through the water treatment facility could be designed to somewhat approximate natural hydrologic regimes, which could provide appropriate timing and duration of connectivity with off-channel habitats. Channel cross-section data and gage data gathered as part of the EBD (PLP 2011) would provide useful insights into flow-connectivity relationships and could help guide a flow management plan.

Changes in Groundwater Inputs and Importance to Fish

There is limited information describing potential surface water-groundwater interaction in the site watersheds, but groundwater is likely the dominant source of streamflow in these streams (Rains 2011). High baseflow levels in the monthly hydrographs of the site watersheds illustrate groundwater's important influence on these streams (Figure 2-6).

Aerial winter open-water surveys (PLP 2011: Figure 7.2-5, Woody and Higman 2011) consistently suggest the presence of upwelling groundwater maintaining ice-free conditions in portions of area streams and rivers. Highly permeable glacial outwash deposits create a complex mosaic within less permeable, silty Pleistocene lake deposits and bedrock outcrops, which can control surface water-groundwater interactions in landscapes like this one (Power et al. 1999). Mine operations that reduce surface water contributions in the natural drainage course, or that lower groundwater tables, may influence groundwater paths and connections within and among streams in the mine area in ways that are unpredictable, but that could have significant impacts on fish. In our analyses of the water management regimes for the mine scenario, we projected increasing proportions of streamflow derived from water released from water treatment and collection facilities as the mine develops (Sections 4.3.7 and 5.2.2.1). The increased releases would result from increased interception of groundwater associated with the mine pit cone of depression, rainwater, and surface runoff collection. Water treated and discharged would be replacing a portion of the groundwater that would otherwise be feeding stream systems, and could have substantially different chemical characteristics (Section 5.3). Additionally, interception of groundwater that is collected then released as a point-source through a water treatment facility would alter the ways in which groundwater feeds stream channels through dispersed and complex pathways. Groundwater-surface water interaction in streams can create thermal heterogeneity, enhancing the diversity of habitats available to fish (Power et al. 1999). Interruption of this process could fundamentally alter the physical environment in headwater streams influenced by the mine (Hancock 2002).

Fish in the region are highly attuned to groundwater signals in the hydrologic and thermal regimes (Power et al. 1999). Spatial heterogeneity in flow and temperature, largely mediated by groundwater-surface water exchange, provides a template for diverse sockeye salmon life histories and migration timing (Hodgson and Quinn 2002, Rogers and Schindler 2008, Ruff et al. 2011). For example, groundwater moderates winter temperatures, which strongly control egg development and hatch and emergence timing (Brannon 1987, Hendry et al. 1998). Spatial thermal heterogeneity allows diverse foraging strategies for consumers of sockeye salmon and their eggs such as brown bear and rainbow trout, thereby benefitting not only sockeye salmon populations, but also the larger food web (Armstrong et al. 2010, Ruff et al. 2011).

Interruption of groundwater flowpaths and connectivity to surface waters in the mine area could have profound effects on the thermal regimes and cued life histories of aquatic biota. Curry et al.(1994) examined the influence of altered hydrologic regimes on groundwater-surface water interchange at spawning locations for brook trout in an Ontario stream. Responses of groundwater-surface water exchange to changes in river discharge varied among sites, precluding predictable responses. The complexity that can be inherent in groundwater-surface water interactions can make regulating or controlling such interactions during large-scale landscape development very difficult (Hancock 2002). Adequately protecting the critical services that groundwater provides to fish is complicated by the fact that flowpaths vary at multiple scales, and connections between distant recharge areas and local groundwater discharge areas are difficult to predict (Power et al. 1999).

5.2.3 Risk Characterization

The volume of water that would require treatment by the mine wastewater treatment plant is unknown at this point, but could be very high. To avoid or minimize risks associated with altered streamflows in downstream effluent-receiving areas (Section 5.2.2.1), capacity for water storage and release would be required in order to maintain natural flow regimes or any minimum flows required by ADFG. Maintenance of mine discharges in terms of water quality, quantity, and timing, to avoid adverse impacts would require long-term commitments for monitoring and facility maintenance. As with other long-term maintenance and monitoring programs, the financial and technological requirements could be very large, and the cumulative risks (and likely instantaneous consequences) of facility accidents, failures, and human error would increase with time. We know of no precedent for the long-term management of water quality and quantity on this scale at an inactive mine.

5.2.4 Uncertainties and Assumptions

The losses of anadromous fish-bearing streams (Table 5-3) in the site watersheds are likely underestimated because of the difficulty of accurately capturing data on all streams that may support fish use throughout year. We rely on the AWC and AFFI for documentation of species distributions, but these records are necessarily incomplete (not all stream reaches have been surveyed) and may be subject to errors in fish identification. Additionally, depictions of species and life-history distributions in the AWC reflect a wide range of mapping policies, and it is difficult to interpret under which policies a particular water body was mapped. That said, the fish sampling documented in the EBD (PLP 2011) is one of the highest-density efforts conducted to date in this portion of Alaska, such that estimates of anadromous fish distributions are likely better represented here than elsewhere in Alaska.

Losses of headwater streams and anadromous fish-bearing streams (Table 5-4) in the site watersheds may also be underestimated because of challenges with stream network mapping. Estimates of headwater stream extent were derived from the Alaska National Hydrography Dataset (USGS 2012), which does not capture all stream courses and may underestimate channel sinuosity, resulting in underestimates of stream length. A LiDAR-derived stream network map would likely yield substantially different results than those presented here. Similarly, actual wetland loss or blockage as a result of the mine footprint (Table 5-3) would likely be higher than estimated here, as the NWI is based on satellite imagery and generally underestimates wetland area. See Box 5-1 for additional discussion of uncertainties associated with stream and wetland mapping.

Alternatively, estimates of headwater streams blocked by the mine footprint may be overestimates if stream diversion channels can be engineered to successfully connect headwater sources above the mine footprint with stream sections downstream of the footprint. Success of diversions would depend on flow and habitat conditions that were suitable for fish passage in both upstream and downstream directions. Diversions would need to avoid potential exposure to sources of contamination along the diversion route, and be maintained and engineered in a manner that safeguards against diversion canal failure.

Lacking specific information on effective contributing area to streamflow in these areas, we relied on simplifying assumptions when estimating changes in streamflow resulting from mine operation. Estimates of changes in streamflow are based on the proportional area of each watershed that would be lost to the mine footprint. Based on this area, streamflow reductions are calculated as a proportional loss that is uniform across the watershed and remains constant throughout the year. Additionally, the effects of TSF 2 and TSF 3 on streamflow are captured for stream gaging stations on the mainstem South Fork Koktuli River, but not for the tributaries themselves, which would experience much more extreme (but unquantified in this assessment) effects of water loss. Seasonal differences in the relative contribution of different parts of the watershed and the confounding influence of potentially complex groundwater flowpaths in the mine area contribute an unknown degree of uncertainty to the streamflow estimates.

It is assumed that more water would be required for mine start-up than is available from runoff from the start-up footprint. In this case, additional water could be withdrawn from area streams, groundwater, or from some other source, further reducing streamflows during mine start-up (Section 5.2.2.1). We do not attempt to quantify that magnitude or the sources that would meet these additional water requirements. Thus, streamflow for mine start-up may be overestimated.

The temperature of waters discharged from the mine, whether directly from the water treatment facility or indirectly via groundwater or surface water runoff, would be influenced by a number of factors controlling heat exchange that cannot be known with certainty at this point. Likewise, the influence of these discharges on temperatures of streams downstream of the mine site is unknown. Because exchange with groundwater is so important to surface water properties in the mine area, simple models that assume primarily surface water heat exchange would be incomplete and inaccurate.

Projecting changes to groundwater-surface water interaction in the mine area with any specificity is not feasible at this time. Local geology and stream hydrographs are indicative of systems that are largely driven by groundwater. Disruptions or changes to groundwater flowpaths in the mine area could have significant adverse effects on winter habitat suitability for fish, particularly if groundwater-dominated stream reaches are converted to stream reaches dominated by effluent from a water treatment system. Given the high likelihood of complex groundwater-surface water connectivity in the mine area, predicting and regulating flows to maintain key ecosystem functions associated with groundwater-surface water exchange will be particularly challenging.

Our approach for assessing potential risks of flow alteration rests on simplifying assumptions regarding changes to the natural streamflow regime under the mine scenario (Section 5.2.2.1). The natural flow regime consists of multiple components, including flow magnitude, frequency, duration, timing, and rate of change, all of which can have important implications for fish and other aquatic life (Poff et al. 1997). We were unable to anticipate changes to the streamflow regime beyond simplistic reductions in flow magnitude, yet it is very likely that other aspects of the flow regime would be modified as well, depending on how flows respond to water management at the mine site. Our analysis does not account for these possibilities.

Additionally, we assume that larger deviations from the natural flow regime pose greater risks of ecological change. This assumption is supported by the literature as a general trend (Poff et al. 2009, Poff and Zimmerman 2010, Richter et al. 2011); however, as pointed out by Poff and Zimmerman (2010), specific responses to changes in streamflow vary. While all stream studies reviewed by Poff and Zimmerman (2010) showed declines in fish abundance, diversity, and demographic rates with any level of flow modification, other ecological responses (e.g., macroinvertebrate abundance, riparian vegetation metrics) sometimes increased. The responses of fish populations and other ecological metrics to flow modification would be dependent on a suite of interacting factors, including but not limited to stream structural complexity, trophic interactions, and the ability of fish to move seasonally (Anderson et al. 2006).

5.3 Pollutants

Under routine operations, our mine scenario presumes that all runoff water, leachate, and wastewater would be collected and properly treated to meet state and federal criteria before release (Section 4.3.7). This section begins with a description of the potential exposures to contaminated water from routine operations (Section 5.3.1). It then describes the exposure-response relationships that are used to screen the constituents of leachates and the more detailed toxicology of the major contaminant of concern, copper (Section 5.3.2). This information is also applied to the discussion of toxic risks from accidents (Chapter 6). The section ends with a characterization of the potential risks from routine effluents (Section 5.3.3) and a discussion of uncertainties (Section 5.3.4).

5.3.1 Exposure

Under the mine scenario (Section 4.3), water that has been in contact with tailings, waste rock, ore, product concentrate, or mine walls would leach minerals from those materials (Appendix H). In addition, chemicals would be added to the water used in ore processing. Most of the water used to transport tailings or products, or used in ore processing would be reused. Leachates from TSFs or waste rock piles would be collected and stored in the TSF or treated for use or discharge (Figure 4-9). Waste rock used in the construction of dams, berms, and other mine structures would be leached by rain and snowmelt and the leachate would be collected and treated as well. Water pumped from the mine pit is assumed to have similar composition to waste rock leachates, and would also be used or treated for disposal. Surplus water on the site would be treated to meet applicable standards and discharged under permit. Based on Alaskan Water Quality Standards (18 Alaska Administrative Code [AAC] 70), no mixing zones would be authorized for anadromous streams or spawning habitat for most game or subsistence fish species, so it is expected that effluents would be required to meet criteria (i.e., no exemptions would be granted).

During the start-up phase, all water from the site would be collected and used in operations. However, during the minimum and maximum mine operations, 5 million to 48 million cubic meters of water available on the site per annum would exceed operational needs, and treated water would be discharged (Section 4.3.7). Our mine scenario does not specify where this effluent would be discharged or what its composition or discharge rates would be, but a complex discharge plan would be required, as far as possible given the water loss, to maintain streamflow, groundwater recharge, temperature, and seasonal variation in support of fish production in the site watershed (Section 5.2). The effluent could contain domestic wastewater, possibly tailings leachate captured below the impoundments, and any excess transport or process waters. However, the primary concern during routine operation would be waste rock leachate. That leachate would become more voluminous as the waste rock piles and uses of waste rock for construction increased during operation. After mine closure, it would be a major source of routinely generated wastewater along with water pumped from the TSF and pit. Leachate composition from tests of the three waste rock types (Tertiary, East Pre-Tertiary, West Pre-Tertiary) is presented in Tables 5-14 through 5-16.

Table 5-14. Composition of Test Leachate from Tertiary Waste Rock in the Pebble Deposit and Quotients Relative to Acute (CMC) and Chronic (CCC) Water Quality Criteria

Parameter	Average Value	CMC	CCC	CMC Quotients	CCC Quotients
pH	7.2		6.5–9		
Alkalinity (mg/L $CaCO_3$)	65.9				
Hardness (mg/L $CaCO_3$)	74.0				
Cl	530				
F	62				
SO_4	27,970				
Ag	0.01121	1.9		0.0059	
Al	79.95	750	87	0.11	0.92
As	2.741	340	150	0.0081	0.018
B	17.70				
Ba	57.23				
Be	0.3072				
Bi	0.5392				
Ca	21,282				
Cd	0.2189	1.5	0.20	0.15	1.1
Co	3.919				
Cr	0.5464	445	58	0.0012	0.0094
Cu	3.200	10[a]	6.9[a]	0.32[a]	0.46[a]
Cu	3.200	2.5[b]	1.6[b]	1.3[b]	2.0[b]
Fe	139.8				
Hg	0.01025	1.4	0.77	0.0073	0.013
K	1,854				
Mg	5,064				
Mn	101				
Mo	6.289				
Na	7,216				
Ni	4.369	360	40	0.012	0.11
Pb	0.1151	46	1.8	0.0025	0.06
Sb	2.118				
Se	1.914		5.0		0.38
Sn	1.253				
Tl	0.068				
V	1.77				
Zn	15.89	91	91	0.17	0.17
Sum of metals				0.78[a] : 1.8[b]	3.3[a] : 4.6[b]

Notes:
Values are presented in micrograms per liter (μg/L) unless indicated otherwise. Average leachate values are from Appendix H.
[a] From Alaska's hardness-based standard
[b] From the biotic ligand model (BLM)-based national water quality criteria
CMC = criterion maximum concentration; CCC = criterion continuous concentration

Table 5-15. Composition of Test Leachate from Pebble East Pre Tertiary Waste Rock and Quotients Relative to Acute (CMC) and Chronic (CCC) Water Quality Criteria

Parameter	Average Value	CMC	CCC	CMC Quotients	CCC Quotients
pH	4.8		6.5–9		
Alkalinity (mg/L $CaCO_3$)	9.9				
Hardness (mg/L $CaCO_3$)	21.9				
Cl	907.9				
F	109.8				
SO_4	51,901				
Ag	0.01928	0.24		0.082	
Al	380.3	750	87	0.51	4.4
As	8.000	340	150	0.023	0.053
B	12.53				
Ba	4.522				
Be	0.5493				
Bi	0.6250				
Ca	6302				
Cd	3.220	0.46	0.085	7.0	38
Co	9.683				
Cr	1.571	160	21	0.0096	0.073
Cu	1,416	3.20[a]	2.4[a]	440[a]	580[a]
Cu	1,416	0.043[b]	0.027[b]	33,000[b]	52,000[b]
Fe	10,195				
Hg	0.01012	1.40	0.77	0.0072	0.013
K	961.8				
Mg	1,498				
Mn	338.6				
Mo	4.270				
Na	2,065				
Ni	10.48	130	14	0.081	0.73
Pb	0.3515	12	0.47	0.029	0.75
Sb	0.7824				
Se	3.243		5.0		0.65
Sn	1.870				
Tl	0.08767				
V	2.436				
Zn	478.5	32	32	15	15
Sum of metals				460[a] : 33,000[b]	640[a] : 52,000[b]

Notes:
Values are presented in micrograms per liter (µg/L) unless indicated otherwise. Average leachate values are from Appendix H.
[a] From Alaska's hardness-based standard
[b] From the biotic ligand model (BLM)-based national water quality criteria
CMC = criterion maximum concentration; CCC = criterion continuous concentration

Table 5-16. Composition of Test Leachate from Pebble West Pre Tertiary Waste Rock and Quotients Relative to Acute (CMC) and Chronic (CCC) Water Quality Criteria

Parameter	Average Value	CMC	CCC	CMC Quotients	CCC Quotients
pH	6.6		6.5–9		
Alkalinity (mg/L $CaCO_3$)	18.5				
Hardness (mg/L $CaCO_3$)	59.2				
Cl	520.0				
F	120.0				
SO_4	60,800				
Ag	0.02698	1.3		0.021	
Al	318.2	750	87	0.42	3.7
As	1.493	340	150	0.0044	0.0100
B	15.88				
Ba	13.62				
Be	0.3273				
Bi	0.6936				
Ca	12,720				
Cd	0.3970	1.2	0.17	0.33	2.3
Co	7.027				
Cr	0.6948	370	48	0.0019	0.014
Cu	1,599	8.2[a]	5.7[a]	190[a]	280[a]
Cu	1,599	0.88[b]	0.55[b]	1,800[b]	2,900[b]
Fe	1,671				
Hg	0.01068	1.4	0.77	0.0076	0.014
K	1,410				
Mg	6,692				
Mn	728.8				
Mo	1.781				
Na	2,053				
Ni	6.805	300	33	0.023	0.20
Pb	0.1724	36	1.4	0.0047	0.12
Sb	3.071				
Se	3.799		5.0		0.76
Sn	0.1403				
Tl	0.4139				
V	0.6825				
Zn	55.58	75	75	0.74	0.74
Sum of metals				200[a] : 1,800[b]	290[a] : 2,900[b]

Notes:
Values are presented in micrograms per liter (μg/L) unless indicated otherwise. Average leachate values are from Appendix H.
[a] From Alaska's hardness-based standard
[b] From the biotic ligand model (BLM)-based national water quality criteria

Because the streams draining a mine site are the receptors for wastewaters, their water quality constitutes the dilution water quality. The water quality of streams in the mine area has been extensively characterized (PLP 2011, Zamzow 2011). The site watersheds are neutral to slightly acidic with low conductivity, hardness, dissolved solids, suspended solids, and dissolved organic carbon (Table 5-17). In those respects, they are characteristic of undisturbed streams. However, as would be expected for a metalliferous site, the levels of sulfate and some metals (copper, molybdenum, nickel, and zinc) are elevated, particularly in the South Fork Koktuli River. PLP found that copper levels in some samples from the South Fork Koktuli River exceeded Alaska's chronic water quality standard. However, most of the exceedances were "in sampling locations within, or in proximity to, the general deposit location" and the number and magnitude of exceedances decreased with distance downstream (PLP 2011: Figure 9.1-35).

Table 5-17. Mean Background Surface Water Characteristics of the Site Watersheds, 2004–2008

Analyte	North Fork Koktuli River	South Fork Koktuli River	Upper Talarik Creek
TDS (mg/L)	37	44	51.2
pH (field)	6.74	7.0	6.99
DO (mg/L)	10.2	10.2	10.5
Temperature (°C)	4.39	4.77	4.04
Specific Conductivity (µS/cm)	46.0	55.5	73.4
TSS (mg/L)	1.39	2.21	2.52
Ca (mg/L)	5.09	6.34	8.77
Mg (mg/L)	1.32	1.41	2.12
Na (mg/L)	2.38	2.35	2.82
K (mg/L)	0.41	0.38	0.44
Alkalinity (mg/L)	20.5	17.4	31.8
SO_4 (mg/L)	2.26	8.78	5.48
Cl (mg/L)	0.66	0.69	0.29
F (mg/L)	0.03	0.05	0.39
Hardness (mg/L)	14.4	21.6	26.5
Al (µg/L)	13	11	13
As (µg/L)	0.2	0.31	0.67
Ba (µg/L)	3.1	4.1	5.5
Cd (µg/L)	0.012	0.013	0.012
Cu (µg/L)	0.39	1.3	0.42
Fe (µg/L)	110	120	110
Mn (µg/L)	10	20	21
Mo (µg/L)	0.19	0.66	0.2
Ni (µg/L)	3.0	0.41	0.63
Pb (µg/L)	0.39	0.087	0.067
Zn (µg/L)	1.8	2.7	2.0
CN (µg/L)	1.9	2.8	1.5
DOC (mg/L)	1.5	1.36	1.57

Notes:
Filtered concentrations are used for hardness and trace elements.
Source: PLP 2011

5.3.2 Exposure-Response

5.3.2.1 Leachates

Tests performed for the EBD (PLP 2011) provide empirical evidence of the potential composition of waste rock leachates from the mine (Appendix H). We screen those leachate constituents against criteria and benchmarks to identify the potentially most toxic constituents, indicate the degree of treatment that would be required, and indicate what sorts of exposures might occur in the event of accidents or failure (Chapter 6). Screening was performed against mean concentrations across samples, because it is assumed that effluents would be mixtures of leachates from tailings and the three types of waste rock. The results of screening waste rock tests are presented in Tables 5-12 through 5-14.

5.3.2.2 Copper

Although the ore and waste rock from porphyry copper mines contain a mixture of metals, copper is the major resource metal and is particularly toxic to aquatic organisms. Hence, it is the most likely to cause toxic effects, and actions taken to prevent copper effects are likely to mitigate effects from co-occurring metals. For these reasons, copper criteria, standards, and toxicity are considered in detail.

Copper Standards and Criteria

The State of Alaska's copper standard is a function of hardness and is based on a prior national criterion (USEPA 1985a). The formulas for the Alaska's acute value (the criterion maximum concentration [CMC]) and chronic value (criterion continuous concentration [CCC]), in micrograms per liter and based on hardness in milligrams per liter, are:

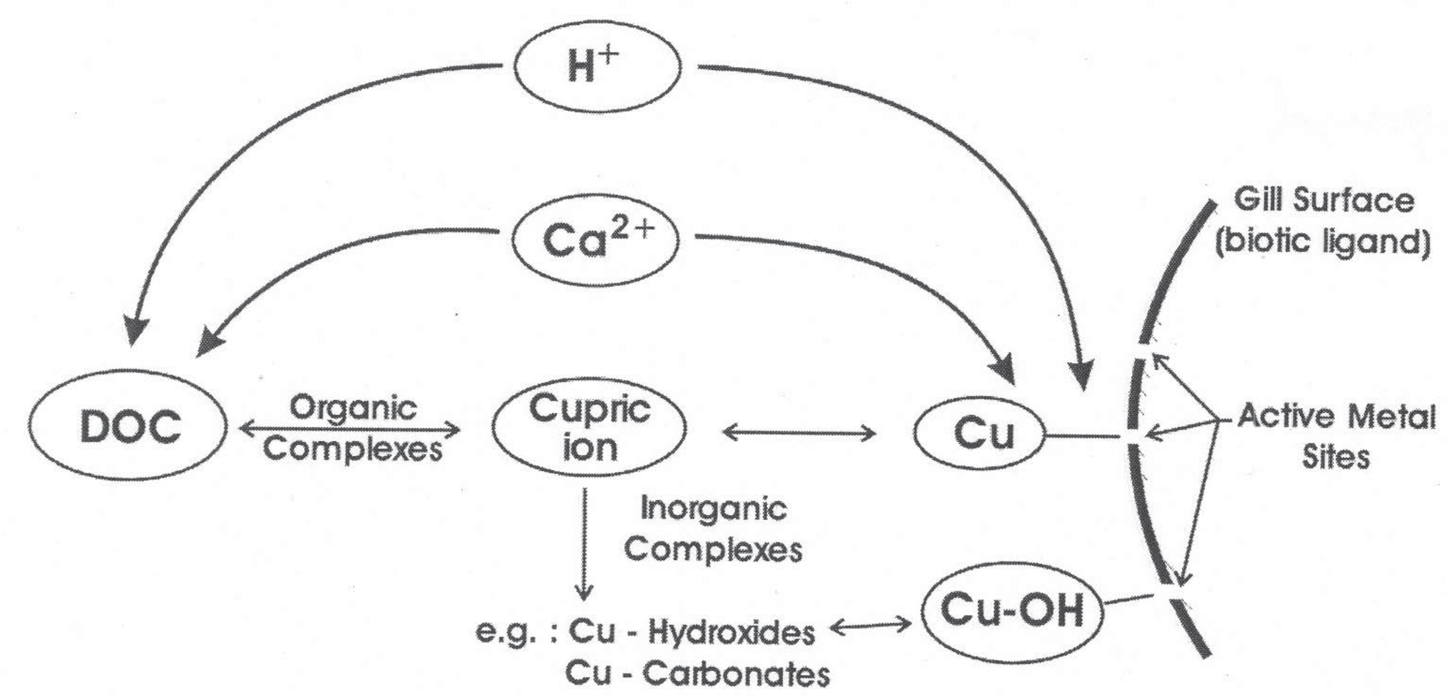
$$\text{Cu acute criterion} = e^{0.9422 \times \ln \text{hardness} - 1.700} \times 0.96;$$

$$\text{Cu chronic criterion} = e^{0.8545 \times \ln \text{hardness} - 1.702} \times 0.96.$$

Note that the formulae are similar and yield similar values—that is, when copper causes toxic effects, they occur relatively quickly. At 20 mg/L hardness (soft water typical of the Bristol Bay region), the acute and chronic values for copper are 2.95 and 2.26 μg/L, respectively.

The federal government has developed new National Ambient Water Quality Criteria for Protection of Aquatic Life (criteria) for copper (USEPA 2007). They are calculated using the biotic ligand model (BLM), which derives the effects of copper as a function of the amount of metal bound to biotic ligands on gills or other receptor sites on an aquatic organism. The ligands bind free copper ions and, to a lesser degree, copper hydroxide ions (Figure 5-13). Copper competes for ligands with calcium and other cations. The competitive binding model for the biotic ligand requires a metal speciation model and estimates of basic water chemistry parameters. The BLM is an advance over hardness normalization, because it more fully accounts for the mechanisms controlling variance in toxicity. In practice, its most important consequence is to estimate the often large reduction in toxicity resulting from binding of copper by dissolved organic matter. The BLM is freely available from USEPA (http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/pollutants/copper/2007_index.cfm) and from the model's developer Hydroqual Inc. (http://www.hydroqual.com/blm).

Figure 5-13. Processes Involved in Copper Uptake as Defined in the Biotic Ligand Model (USEPA 2007)



The results of applying the BLM to mean water chemistries of the North Fork Koktuli and South Fork Koktuli Rivers and Upper Talarik Creek (Table 5-17) are presented in Table 5-18. These values are lower than Alaska's hardness-based values and the variance among streams is potentially significant.

Table 5-18. Results of Applying the Biotic Ligand Model to Mean Water Chemistries in the Site Watersheds to Derive Receiving Water-Specific Copper Criteria

Stream	Acute Cu Criterion (CMC in µg/L)	Chronic Cu Criterion (CCC in µg/L)
North Fork Koktuli River	1.73	1.07
South Fork Koktuli River	2.37	1.47
Upper Talarik Creek	2.70	1.68

Notes:
CMC = criterion maximum concentration; CCC = criterion continuous concentration
Biotic ligand model (BLM) source: USEPA 2007

The results of applying the BLM to mean chemistries of the waste rock leachates are presented in Table 5-19. The model runs used mean water chemistries from the PLP tests (Appendix H). These effluent-specific values are higher than those for background surface water because of the higher content of mineral ions.

Table 5-19. Results of Applying the Biotic Ligand Model to Mean Water Chemistries in Waste Rock Leachates to Derive Effluent-Specific Copper Criteria

Leachates	Acute Cu Criterion (CMC in µg/L)	Chronic Cu Criterion (CCC in µg/L)
Pebble Tertiary	2.5	1.6
Pebble East Pre-Tertiary	0.88	0.55
Pebble West Pre-Tertiary	0.43	0.027

Notes:
CMC = criterion maximum concentration; CCC = criterion continuous concentration
Biotic ligand model (BLM) source: USEPA 2007

For both the background waters and the leachates, temperature was set to the mean from streams on the site (4.5°C). For the leachates, dissolved organic carbon was set to 1 mg/L (the lowest level accepted by the model), and humic acid was set to the default value of 10% of dissolved organic carbon.

Both the state standards and the national criteria are derived from the 5th percentile of the sensitivity distribution for copper of aquatic genera. The most sensitive 33% of genera in acute tests and 42% of genera in chronic tests are all invertebrates (USEPA 2007). Hence, the regulatory benchmarks are determined by invertebrate sensitivities. However, the most sensitive vertebrates in both types of tests are fish of the genus *Oncorhynchus,* which includes rainbow trout and the five Pacific salmon species. Rainbow trout is a standard test species that is at least as sensitive to copper as Chinook and coho salmon, brook trout, and brown trout in acute tests (CH2M Hill and LLC 2004). Acute and chronic values for rainbow trout can be derived for background water quality using the BLM method (Table 5-20).

Table 5-20. Rainbow Trout Site-Specific Acute and Chronic Toxicity Derived by Applying the Biotic Ligand Model to Mean Water Chemistries in the Site Watersheds

Stream	Acute Cu Toxicity[a] (LC_{50} in µg/L)	Chronic Cu Toxicity (CV in µg/L)
North Fork Koktuli River	59.4	20.6
South Fork Koktuli River	62.78	21.8
Upper Talarik Creek	75.4	26.2

Notes:
[a] Acute toxicity: median lethal concentration (LC50)
CV = chronic value, calculated using the species-specific acute to chronic ratio of 2.88.
Biotic ligand model (BLM) source: USEPA 2007

Alternative Endpoints

The standards and criteria are based on conventional test endpoints: survival, growth, and reproduction. However, research has shown that the olfactory sensitivity of salmon is diminished at lower copper concentrations than those that reduce conventional endpoints in salmon (Hecht et al. 2007). Salmon use olfaction to find their spawning stream, detect and avoid predators, find food, detect reproductive and alarm pheromones, and perform other life processes. Although effects on fish olfaction have not been shown to affect the viability of field populations, it is reasonable to expect that interference with these essential processes would have population-level consequences (DeForest et al. 2011b).

Meyer and Adams (2010) applied the hardness-corrected criteria and the BLM to data from multiple laboratory tests for olfactory effects and found that the BLM accounted well for variance among tests, and that BLM-based criteria were consistently protective of those effects in the test systems. However, hardness-corrected criteria were not consistently protective. DeForest et al. (2011a) extended those results by applying the same models to 133 ambient waters in the western United States, including Alaska, which exhibited a wide range of water chemistries. Using the 20% inhibitory concentration (IC_{20}) for coho salmon olfaction from McIntyre et al. (2008a, 2008b) as the endpoint, they found that the hardness-corrected criteria were not consistently protective, but the BLM-based chronic criteria were protective of this chronic effect in 100% of the waters. Even the acute BLM-based criteria were protective of this chronic effect in 98% of waters. That is because, as noted previously, the criteria are determined by sensitive invertebrates that experience diminished survival, growth, or reproduction at even lower levels than those that inhibit fish olfactory receptors.

Dietary Exposure-Response

Dietary exposure to metals, particularly at mine sites, has become a topic of investigation in recent years (Meyer et al. 2005). Studies of the tailings-contaminated Clark Fork River in Montana and the Coeur d'Alene River in Idaho have shown that macroinvertebrates can accumulate metals at levels that result in toxicity to fish that consume them (Farag et al. 1994, Woodward et al. 1994, Woodward et al. 1995, Farag et al. 1999). Participants in a recent Pellston Workshop (convened by the Society for Environmental Toxicology and Chemistry to examine toxicology issues in aquatic environments) reviewed the literature and developed an estimate of the degree to which aqueous toxicity thresholds should be adjusted to account for dietary exposures in rainbow trout (Borgmann et al. 2005). The estimate is based on an average bioconcentration factor of 2,000 L/kg and an average dietary chronic value for rainbow trout of 646 μg/g. Because the resulting factor is 0.95, the adjustment is not large. If the factor is applied to the lowest chronic value for rainbow trout (11.3 μg/L) (USEPA 2007), the result (10.7 μg/L) is still much higher than the national water quality criteria and state standards, because of the relative insensitivity of fish. This result applies to aqueous-only exposures. Dietary exposure of fish to copper in sediments is considered in Section 6.1.4.

Exposure-Response Data from Analogous Sites

Evidence concerning exposure-response relationships for copper and other metals in streams at metal mines also comes from field studies. Because the mine scenario presumes that water quality criteria would be met during routine operations, the critical question is whether effects are observed at those levels. The most relevant high-quality studies are those performed in the Colorado metal belt, particularly near the Animas and Arkansas Rivers. These sites are contaminated predominately by mine drainage and mine waste leachates, and field and laboratory experiments have confirmed that aqueous metals, not tailings or other particles, cause the observed effects (Courtney and Clements 2002). These studies have identified effects on aquatic insect populations and invertebrate communities at concentrations below water quality criteria for the dominant metals (cadmium, copper, and zinc) (Buchwalter et al. 2008, Schmidt et al. 2010). Application of the BLM and an additive combined effects model reduced the discrepancy but did not eliminate it, suggesting that chronic criteria for metals are not protective against effects on invertebrates (Schmidt et al. 2010). In particular, while the combined criteria approximated thresholds for taxa richness, abundances of sensitive taxa were reduced at exposures below the combined criteria (Griffith et al. 2004, Schmidt et al. 2010). Potential reasons for the discrepancy are the absence of sensitive species or life stages from the criteria, less-than-life-cycle exposures, and the absence of dietary exposures.

Unexpected field effects might be caused by an unknown factor that is correlated with both the concentration of metals and the biological effects (i.e., a confounding variable). However, no such factor is known, and the hypothesized mechanisms for the greater sensitivity of field communities are supported by evidence from laboratory and field experiments.

It also must be noted that the occurrence of biological effects below criterion concentrations does not necessarily indicate that criteria are not adequately protective. By design, the criteria allow acute or chronic effects on as much as 5% of species (USEPA 1985b).

Uncertainties

The copper criterion is based on a large body of data and a mechanistic model of exposure and effects. Hence, it is one of the best-supported criteria. However, it is always possible that it would not be protective in particular cases due to unstudied conditions or responses. Because the most sensitive taxa are aquatic invertebrates, unknown aspects of invertebrates are most likely to be influential. In particular, field studies, including studies of streams draining metal mine sites, show that Ephemeroptera (mayflies) are often the most sensitive species and the smaller instars are particularly sensitive (Kiffney and Clements 1996, Clements et al. 2000). However, the copper criteria do not include any ephemeroptera in the sensitivity distribution (USEPA 2007). If the ephemeropteran, plecopteran, trichopteran, or other invertebrate species in the site watershed streams are more sensitive than the cladocerans (the most sensitive tested species), then they may not be protected by the criteria.

In addition, the chronic copper criterion is derived by applying an acute-chronic ratio to the BLM-derived final acute value (USEPA 2007). Because of the complex dynamics of chronic uptake, distribution, and sequestration of metals in aquatic insects, the BLM, which focuses on binding to a surface ligand, may not adequately adjust chronic toxicity (Luoma and Rainbow 2005, Buchwalter et al. 2008). Brix et al. (2011) reviewed the toxicity testing literature and found that aquatic insects are highly sensitive to copper in chronic exposures, relative to acute exposures, and may not be protected by current criteria. Hence, the protectiveness of the chronic criterion is more uncertain than the acute criterion.

5.3.3 Risk Characterization

If the leachates and excess process waters are collected and treated before discharge to achieve state standards and national criteria, unacceptable toxic effects should not occur. The toxicity of copper is expected to be the greatest concern. Therefore, discharges should meet the BLM-based national criteria as well as the hardness-based state standard. Although those regulatory benchmarks are based on invertebrate sensitivities, they are highly relevant to protecting salmon and other valued fish. Immature salmon rely on invertebrates as food and all post-larval life stages of resident rainbow trout and Dolly Varden feed on invertebrates. In streams, these invertebrates are primarily aquatic insects, but immature sockeye salmon in lakes are dependent on zooplankton. Hence, protection of fish requires protection of sensitive invertebrates.

5.3.4 Uncertainties

Although effects of permitted effluents are not expected to be significant, the following uncertainties remain.

- Chemical criteria and standards do not address the interactions or combined effects of the individual constituents or any unusual sensitivities of the biotic community. The waste rock leachates all exceed criteria for more than one metal (Tables 5-12 through 5-14). Therefore, meeting all criteria could still result in toxicity resulting from combined effects.
- Studies of streams receiving mine effluents and laboratory studies suggest that the abundance of important insect taxa could be reduced even if criteria are met.
- Criteria for chemicals other than copper do not address site water chemistry, or they address it in a simple way. Hence, they may be inaccurate estimates of threshold concentrations for toxic effects in these highly pure waters.

- Some leachate and process water constituents have no water quality criteria (e.g., sulfate), or the criteria and standards are based on old literature.
- The identities of the ore processing chemicals are unknown, so their potential toxicity is not considered.
- If the tested rock and tailings samples are not representative, other wastewater constituents may be of concern. That is, some waste rocks or tailings may have high levels of elements other than those identified in the screening analysis.
- Dissolved salts (expressed as conductivity or total dissolved solids) are a potential risk to stream biota from the leaching of waste rocks, and routine water treatment does not handle them well (USEPA 2011). However, there are no applicable criteria and the actual salinity and the mixture of ions in the effluent are highly uncertain. For these reasons, any discharge permits for mines in the Bristol Bay watershed should include relevant whole-effluent toxicity testing and monitoring of biotic communities in receiving streams.

5.4 Roads and Stream Crossings

Only rarely can roads be built that have no negative effects on streams (Darnell et al. 1976). Roads modify natural drainage networks and accelerate erosion processes, which, in turn, can lead to changes in streamflow regimes, sediment transport and storage, channel bank and bed configurations, substrate composition, and the stability of slopes adjacent to streams. These changes can have important biological consequences for anadromous and resident fishes, for example by negatively affecting food, shelter, spawning habitat, water quality, and access for upstream and downstream migration (Furniss et al. 1991).

The physical effects of roads on streams and rivers often propagate long distances from the site of a direct road incursion, as a result of the energy associated with moving water (Richardson et al. 1975). Alteration of hydrodynamics and sediment deposition can result in changes in channels or shorelines many kilometers away, both down- and up-gradient of a road crossing.

Background discussion of important issues with respect to roads and stream crossings are introduced in Sections 5.4.1 to 5.4.3. Risks are assessed for road crossings as barriers to fish movement (Section 5.4.4), dust and sediment deposition (Section. 5.4.5), chemicals in runoff (Section 5.4.6), and filling and alteration of wetlands (Section 5.4.7). The extent of habitat alteration and the fish populations potentially affected along the road corridor are described in Sections 5.4.8 and 5.4.9. Finally, risks from all aspects of the road corridor are characterized in Section 5.4.10.

5.4.1 Culverts

Culverts are the most common migration barriers associated with road networks. Hydraulic characteristics and culvert configuration can impede or prevent fish passage. Where flow restrictions such as culverts are placed in stream channels, the power of streamflow is increased. This can lead to increased channel scouring and down-cutting, streambank erosion, and undermining of the stream crossing structure and fill. Although the well-planned installation of culverts allows natural flow upstream and downstream of crossings, failure rates are generally high (Sections 4.4.3.3 and 6.4).

5.4.2 Stormwater Runoff

During runoff events, traffic residues produce a contaminant "soup" of metals (especially lead, zinc, copper, chromium, and cadmium), oil, and grease, which can run off road surfaces, enter streams, and accumulate in sediments (Van Hassel et al. 1980) or disperse into groundwater (Van Bohemen and Van de Laak 2003). Fish mortality in streams has been related to high concentrations of aluminum, manganese, copper, iron, or zinc, with effects on populations recorded as far as 8 km downstream (Forman and Alexander 1998). Although this is an important issue for streams near highways, it is unlikely that a mine access road would have sufficient traffic to significantly contaminate runoff with metals or oil. However, because the salts or other materials used for winter treatment of roads could present a significant issue, these are addressed below (Section 5.4.4).

Increased runoff associated with roads may also increase the rates and extent of erosion, reduce percolation and aquifer recharge rates, alter channel morphology, and increase stream discharge rates (Forman and Alexander 1998). These effects on flow are not assessed, however, because they are highly location-specific and are not likely to have significant effects on salmonids in our mine scenario.

5.4.3 Near-Surface Groundwater and Hyporheic Flows

The high incidence of seeps and springs noted on glaciolacustrine, alluvial, and slope till deposits in the mine area (Hamilton 2007, Woody and O'Neal 2010) and the abundance of wetlands testify to the pervasiveness of shallow subsurface flow processes and high connectivity between groundwater and surface water systems in the areas traversed by the transportation corridor (Appendix G). The construction and operation of roadways and pipelines can fundamentally alter connections between shallow aquifers and surface channels and ponds by intercepting shallow groundwater flowpaths, leading to further impacts on surface water hydrology, water quality, and fish habitat (Darnell et al. 1976, Stanford and Ward 1993, Forman and Alexander 1998, Hancock 2002).

5.4.4 Road Crossings as Barriers to Fish Movement

5.4.4.1 Exposure

Within the Kvichak River watershed, the transportation corridor would cross 34 streams and rivers supporting migrating and/or resident salmonids, including 17 streams designated as anadromous waters at the location of the crossing. Of these crossings, 20 would be bridges and 14 would be culverts.

5.4.4.2 Exposure-Response

Free access to spawning and early rearing habitat in headwater streams is critical for a number of fish species. Culverts pose the most common migration barriers associated with road networks. Persistent barriers to fish movement are assessed in Section 6.4, because they are considered to constitute maintenance failures. Culverts designed to meet the State of Alaska's requirements and regularly maintained should not block fish passage; however, hydraulic characteristics such as low water depth or high water velocities and culvert configurations can impede or prevent fish passage.

Salmonids and other riverine fishes also actively move into seasonal floodplain wetlands and small valley floor tributaries to escape the stresses of main-channel flood flows (Copp 1989). Culverts can reduce flow to these habitats by directing flow from the entire floodplain through the culvert into the main channel. High water velocities in a stream channel may result from storm flows being forced to pass through a culvert rather than spread across the floodplain. Higher velocities cause scour and down-cutting of the channel downstream of the culvert, hydrologically isolating the floodplain from the channel and consequently blocking fish access to floodplain habitat. Entrenchment of the channel also prevents fish from reaching slow-water refugia in a storm event and eliminates nutrient and sediment cycling processes on the floodplain.

5.4.4.3 Risk Characterization

The mine scenario assumes that culverts would be installed along the transportation corridor with adequate size for the streams crossed, and that the roadway would be monitored daily to ensure that failures could be rapidly identified and repaired. Even with these assumptions, inhibition of fish passage and reductions in habitat still could occur. The behavioral responses to culverts of the up-migrating and down-migrating life stages of the salmonid species that use the potentially crossed streams are uncertain. Standards for culvert installation on fish-bearing streams in Alaska target road safety and fish passage, but not the physical structure of the stream or habitat quality (ADFG and ADOT&PF 2001). Culverts' capacities are allowed to be less than channel capacity. Culverts must be 0.9 times the ordinary high-water channel width in most cases. Where the channel slope is less 0.5%, the culvert is allowed to be 0.75 times the ordinary high-water channel width. During flood flows this reduced effective channel width results in slower than normal velocities upstream of the culvert and higher water velocities exiting the culvert. Downstream channel beds may be scoured, channel dynamics changed, and channels and the floodplains may become disassociated. This process would reduce the capacity of the downstream reaches to support salmonid fish. The high flows in and immediately downstream of the culvert and the structure of the culvert may inhibit fish passage even if movement is not blocked. Downstream erosion would result in perched culverts, if they are not inspected and maintained, which would inhibit and ultimately block passage (Section 6.4). Floodplain habitat and floodplain/channel ecosystem processes would be disrupted by entrenchment of the channel resulting from culvert-induced erosion. These potential reductions in downstream habitat quality and inhibited fish passage could occur in the 14 culverted streams that support salmonids.

5.4.5 Dust and Fine Sediment

5.4.5.1 Exposure

During rain and snowmelt, soil eroded from road cuts, borrow areas, road surfaces, shoulders, cut-and-fill surfaces, and drainage ditches, along with road dust deposited on vegetation, would be washed into streams and other water bodies. The sediment contribution per unit area from roads is often much greater than that from all other land management activities combined (Gibbons and Salo 1973). The chief variables in surface erosion are the inherent erodibility of the soil, slope steepness, surface runoff, slope length, and ground cover. Erosion and siltation are likely to be greatest during road construction.

5.4.5.2 Exposure-Response

Sediment loading from roads can severely affect streams below the right-of-way (Furniss et al. 1991) and references therein). As described in Section 6.1.3, salmonids are adapted to episodic exposure to suspended sediment, but as concentration or duration of exposure increase, effects on survival and growth can occur. As described in Section 6.1.2, increased deposition of fine sediment decreases the abundance and production of fish and benthic invertebrates. Increased loading of road-derived fine sediments, in particular, has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, increased predation on fishes, and reduced benthic organism populations and algal production (Newcombe and MacDonald 1991, Newcombe and Jensen 1996, Gucinski et al. 2001, Angermeier et al. 2004, Suttle et al. 2004). In low-velocity stream reaches, an excess of fine sediment can completely cover suitable spawning gravel, rendering it useless for spawning. Excessive sediment loading of streams can also result in channel braiding, increased width-depth ratios, increased incidence and severity of bank erosion, reduced pool volume and frequency, and increased subsurface flow. These changes can result in a reduction in quality and quantity of available spawning habitat (Furniss et al. 1991). During high-discharge events, accumulated sediment tends to be flushed out and re-deposited in larger water bodies (Forman and Alexander 1998). Because the streams crossed by the road connect directly or indirectly to Iliamna Lake, accelerated sedimentation could have an impact on the concentrated spawning populations of sockeye salmon in the lake's shallow waters (Woody 2007).

5.4.5.3 Risk Characterization

Suspended and deposited sediment washed from roads, shoulders, ditches, cuts, and fills would diminish habitat quality in the streams below road crossings. The magnitude of effects cannot be estimated in this assessment; however, published studies of the influence of silt on salmonid streams indicate that the magnitude could be locally significant (Section 6.1).

5.4.6 Salts and Dissolved Solids in Runoff

5.4.6.1 Exposure

Roads are treated with salts and other materials to reduce dust and improve winter traction. In Alaska, calcium chloride is commonly used for dust control and is mixed with sand for winter application.

During periods of rain and snowmelt, these materials are washed off roads and into streams, rivers, and wetlands, where fish and their invertebrate prey can be directly exposed. We found no relevant data for chloride levels in streams treated in this way.

5.4.6.2 Exposure-Response

Compounds used to control ice and dust (Hoover 1981) have been shown to cause toxic effects when they run off and enter surface waters. Dissolved calcium, like sodium, has little influence on the toxicity of dissolved chloride salts (Mount et al. 1997). Hence, the toxicity of the calcium chloride used commonly in Alaska would be expected to be similar to that of the more studied sodium chloride, based on chlorine concentrations. Salmonids are sensitive to salinity, particularly at fertilization (Weber-Scannell and Duffy 2007). According to the USDA Forest Service (1999), application of chloride salts should be avoided within at least 8 m of surface waters or anywhere groundwater is near the surface. Adverse biological effects are likely to be particularly discernible in naturally low-conductivity waters, such as those of the Bristol Bay watershed, but research is needed to substantiate this (Appendix G).

5.4.6.3 Risk Characterization

The risks to salmonids from de-icing salts would depend on the amount and frequency of application; however, the risks are potentially locally significant. The transportation corridor would intersect more than 30 streams and rivers supporting spawning anadromous and/or resident salmonids, including 270.3 km of stream between road crossings and Iliamna Lake (Table 5-21). Additionally, 19.4 km of roadway would intersect wetlands within and beyond those mapped by the National Wetlands Inventory (NWI). Runoff from these segments of roadway could have a significant impact on these wetlands.

Table 5-21. Stream Lengths Downstream of Road Crossings, Measured from Road-Stream Intersections to Iliamna Lake

HUC-12 Name or Description	Downstream Length (km)
Headwaters, Upper Talarik Creek	17.6
Upper tributary stream to Upper Talarik Creek[a]	9.1
Outlet, Upper Talarik Creek	34.3
Tributary to Newhalen River portion upstream of corridor[b]	18.0
Headwaters, Newhalen River	9.7
Outlet, Newhalen River	34.3
Roadhouse Creek	22.0
Iliamna Lake	16.4
Eagle Bay Creek	10.8
Youngs Creek Mainstem (Roadhouse Mountain HUC)	4.2
Youngs Creek East Branch[c]	7.6
Chekok Creek	8.7
Canyon Creek	5.4
Iliamna Lake – Knutson Bay	20.0
Knutson Creek	3.6
Iliamna Lake – Pedro Bay	8.7
Iliamna Lake – Pile Bay	11.4
Outlet, Pile River	5.7
Lower Iliamna River	4.1
Middle Iliamna River	2.6
Chinkelyes Creek	16.1
TOTAL	270.3

Notes:
[a] 190302060701
[b] 190302051404
[c] 190302060904
Values are summed by 12-digit Hydrologic Unit Code (HUC-12), arranged from west (top) to east (bottom) along the potential transportation corridor.

5.4.7 Wetland Filling and Alteration

5.4.7.1 Exposure

Construction of the transportation corridor, as described in the mine scenario (Section 4.3.9.1), would result in the direct filling of wetlands. In addition, by damming and diverting surface flow and inhibiting subsurface flow, road construction could alter wetland hydrology and limit access by fish.

5.4.7.2 Exposure-Response

The loss of wetlands can result in the loss of resting habitat for adult salmonids and of spawning and rearing habitat in ponds and riparian side channels. Within wetlands, hydrologic disruptions from roads, by altering hydrology, mobilizing minerals and stored organic carbon, and exposing soils to new wetting and drying and leaching regimes, can lead to changes in vegetation, nutrient and salt concentrations, and reduced water quality (Ehrenfeld and Schneider 1991). These changes in wetland dynamics and structure can affect the utility of wetlands for fish and water quality in streams receiving wetland drainage.

5.4.7.3 Risk Characterization

The filling of wetlands would directly eliminate habitat for salmonids and would indirectly alter wetlands in ways that could reduce the quality, quantity, and accessibility of habitat for fish. The area that would be filled by the roadbed is estimated to be 0.18 km^2 and the area that would be altered is estimated to be 4.9 km^2 (Section 5.4.6.3). Effects on fish production cannot be estimated; however, the loss of long riparian side channels to streams and rivers that are crossed with culverts or bridges that do not span the entire floodplain could be locally significant.

5.4.8 Potential Extent of Habitat Altered by the Transportation Corridor

The streams and wetlands along the transportation corridor would be affected by their combined exposure to sediment, salts, culverts, and the filling of connected wetlands. The areas and resources potentially affected are described in this section.

5.4.8.1 High-Impact Areas along the Transportation Corridor

The transportation corridor would affect fish and aquatic resources throughout its approximately 139-km length. The largest impact on sockeye salmon would likely occur where the road would run parallel the Iliamna River and Chinkelyes Creek, where many sockeye salmon spawn (Figure 5-15, Iliamna River inset). Other high-impact areas include where the road would run parallel to Knutson Bay, intersecting many small streams (Figure 5-15, Knutson Bay inset), and where the road crosses wetlands north of Iliamna Lake (Figure 5-15, Newhalen River inset).

5.4.8.2 Stream Length Upstream and Downstream of Crossings

The transportation corridor has the potential to affect 270.3 km of stream between the road crossings and Iliamna Lake. This is based on the length of streams below crossings, in each hydrologic unit code (HUC), as shown in Table 5-21. In some cases there would be multiple stream crossings in sinuous streams, but no streams have been double-counted. The Knutson Bay and Pedro Bay HUCs contain ten and six outflows, respectively, to Iliamna Lake.

The length of stream upstream of the transportation corridor likely to support fish, based on a stream gradient higher than 10%, is 240 km. The upstream length summed by stream length in each HUC is shown in Table 5-22.

Figure 5-14. Reported Salmon, Dolly Varden, and Rainbow Trout Distribution along the Potential Transportation Corridor

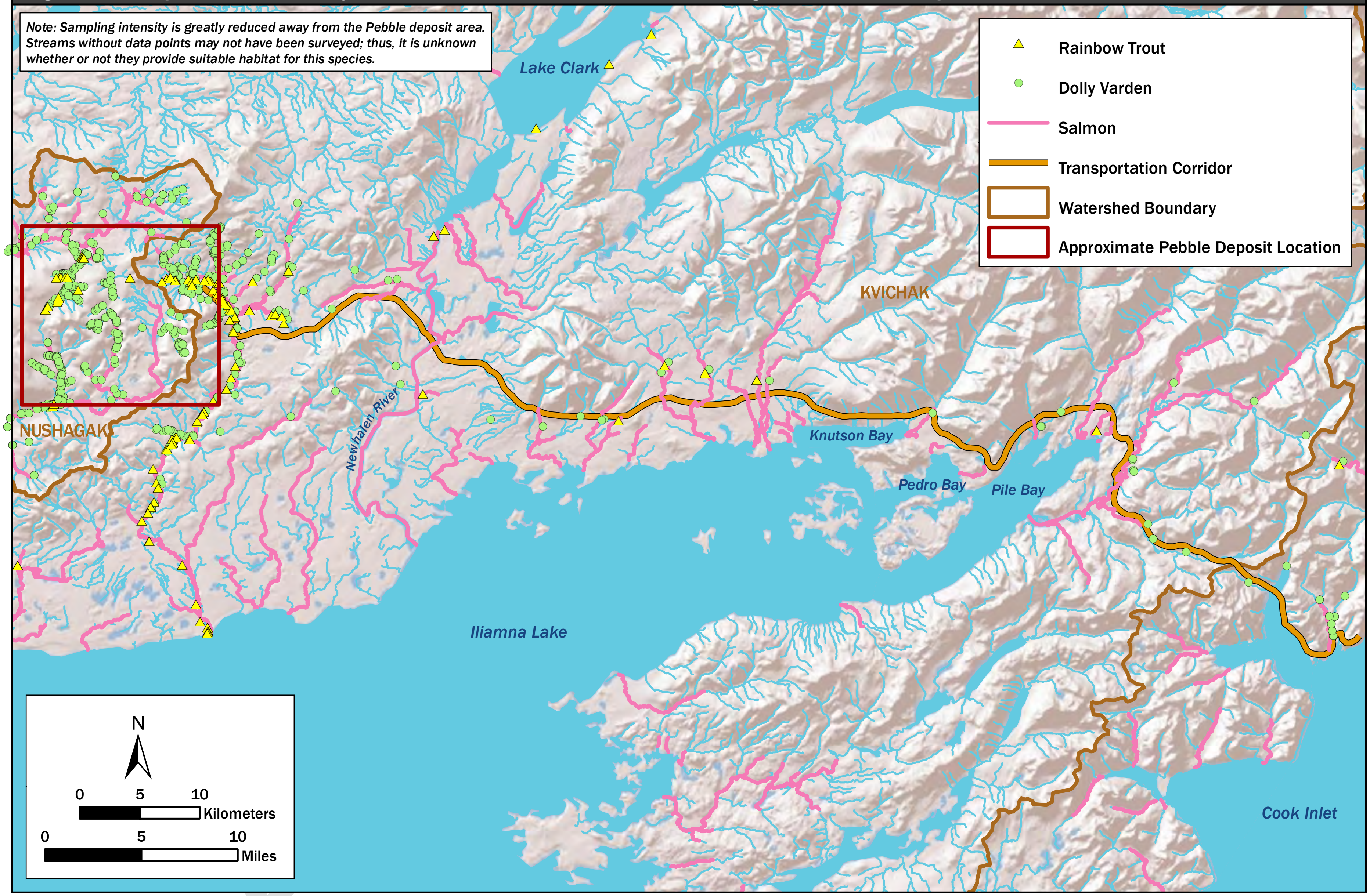


Figure 5-15. High-Impact Areas in the Potential Transportation Corridor (Insets)

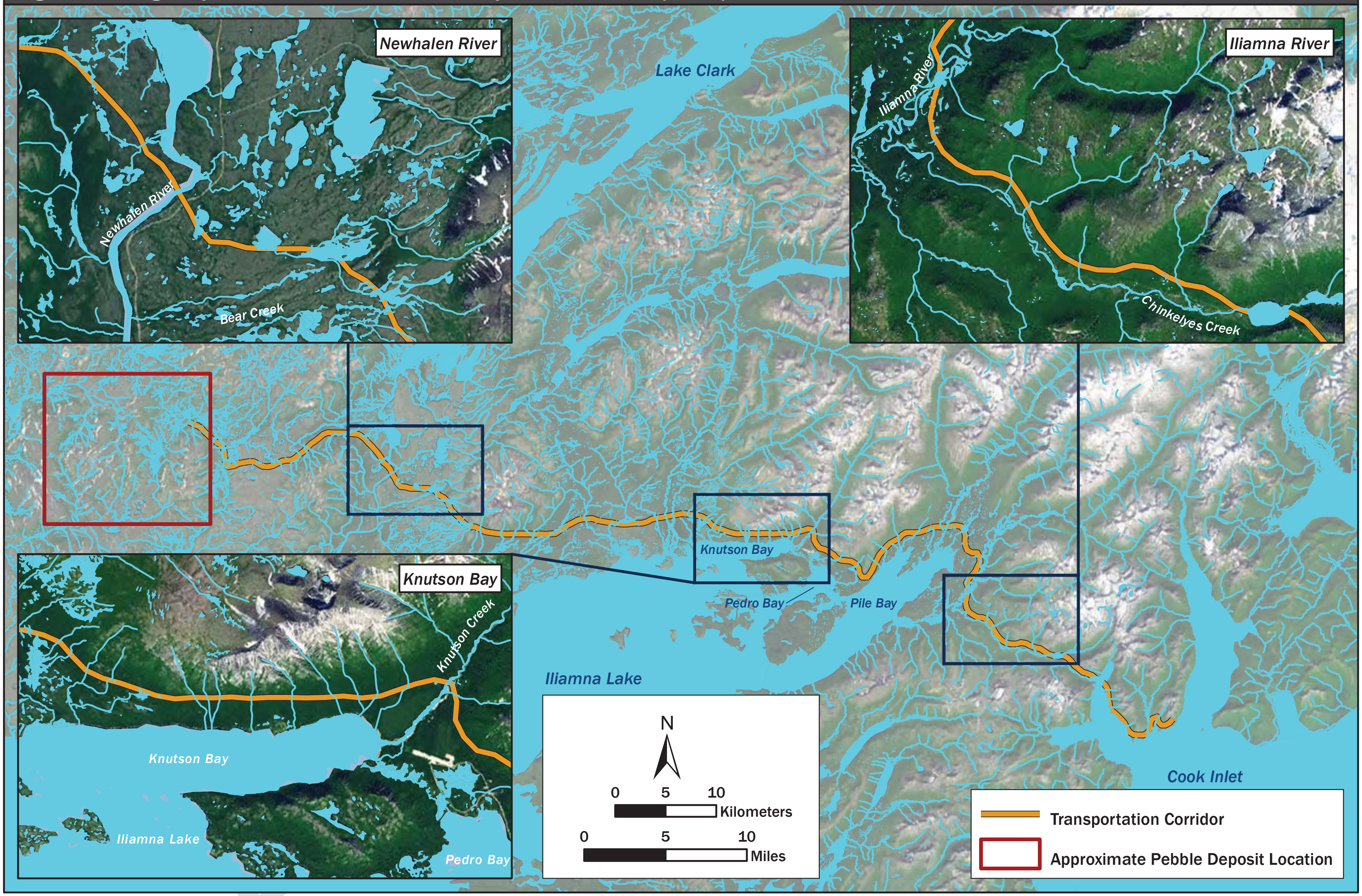


Table 5-22. Stream Lengths Upstream of Road Crossings that are Likely to Support Salmonid Fish (Gradient <10%)

HUC-12 Name or Description	Upstream Length (km)
Headwaters, Upper Talarik Creek	53.2
Upper tributary stream to Upper Talarik Creek[a]	19.9
Tributary to Newhalen River portion upstream of corridor[b]	5.9
Tributary headwaters, Newhalen River	3.2
Headwaters, Newhalen River	13.1[d]
Iliamna Lake	0.5
Eagle Bay Creek	10.5
Youngs Creek Mainstem (Roadhouse Mountain HUC)	21.9
Youngs Creek East Branch[c]	16.8
Chekok Creek	40.9
Canyon Creek	4.0
Iliamna Lake – Knutson Bay	0.8
Knutson Creek	0.5
Iliamna Lake – Pile Bay	0.5
Outlet, Pile River	12.9
Middle Iliamna River	22.2
Chinkelyes Creek	13.2
TOTAL	240.0

Notes:
[a] 190302060701
[b] 190302051404
[c] 190302060904
[d] Includes upstream Newhalen River length only to Sixmile Lake and Lake Clark
Values are summed by 12-digit Hydrologic Unit Code (HUC-12), arranged from west (top) to east (bottom) along the potential transportation corridor.

5.4.8.3 Road Lengths Crossing or Near Water

The lengths of the transportation corridor located in different proximities to National Hydrology Dataset (NHD) streams and NWI wetlands are shown in Table 5-23 and Table 5-24, respectively. These lengths do not encompass the section of corridor outside of the Kvichak watershed (i.e., in watersheds flowing into Cook Inlet). Approximately 16.7 % of the transportation corridor (19.7 km) would be located within 100 m of an NHD stream, and 33.6 % (39.6 km) of the corridor would be located within 200 m of an NHD stream (Table 5-23).

Table 5-23. Lengths of the Potential Transportation Corridor Located in Different Proximities to NHD Streams

HUC-12 Name or Description	Proximity to Streams			
	Not nearby	< 100 m	100-200m	Total
Headwaters, Upper Talarik Creek	5.0	1.5	1.0	7.5
Upper tributary stream to Upper Talarik Creek[a]	4.2	0.2	0.2	4.6
Tributary to Newhalen River portion upstream of corridor[b]	7.4	1.4	2.3	11.1
Headwaters, Newhalen River	2.5	0.4	0.5	3.4
Outlet, Newhalen River	3.8	1.0	1.8	6.6
Roadhouse Creek	0.6	1.6	1.2	3.4
Iliamna Lake	27.4	6.2	6.6	40.2
Eagle Bay Creek	2.8	1.0	0.7	4.5
Youngs Creek Mainstem (Roadhouse Mountain HUC)	3.0	0.3	0.2	3.5
Youngs Creek East Branch[c]	1.1	0.8	1.2	3.1
Chekok Creek	1.6	0.4	0.5	2.5
Canyon Creek	0.9	0.2	0.2	1.3
Knutson Creek	1.1	0.5	0.4	2.0
Outlet, Pile River	2.0	0.9	0.7	3.6
Middle Iliamna River	4.9	0.8	1.2	6.9
Chinkelyes Creek	9.8	2.6	1.2	13.6
TOTAL	78.1	19.7	19.9	117.8

Notes:
[a] 190302060701
[b] 190302051404
[c] 190302060904
Values are summed by 12-digit Hydrologic Unit Code (HUC-12), arranged from west (top) to east (bottom) along the potential transportation corridor.
NHD = National Hydrography Dataset

Approximately 16.5% (19.4 km) of the transportation corridor would intersect wetlands, an additional 23.4% (27.6 km) would be located within 100 m of wetlands, and an additional 16.4% (19.3 km) would be located within 100 to 200 m of wetlands (Table 5-24). Thus, 56.3 % (66.3 km) of the corridor would fill or otherwise alter wetlands. Wetlands constitute nearly 11% of the total area within 200 m of the transportation corridor. The areas of wetlands within 100 m and 200 m of the corridor would be 2.4 km^2 and 4.9 km^2, respectively. The area of wetlands filled by the roadbed would be 0.18 km^2.

Table 5-24. Lengths of the Potential Transportation Corridor Located in Different Proximities to NWI Wetlands

HUC-12 Name or Description	Proximity to NWI Wetlands				
	Avoids	Intersects	< 100 m	100-200 m	Total
Headwaters, Upper Talarik Creek	0.2	2.0	4.1	1.3	7.6
Upper tributary stream to Upper Talarik Creek[a]	0.3	1.7	1.4	1.2	4.6
Tributary to Newhalen River portion upstream of corridor[b]	3.5	0.9	4.0	2.6	11.0
Headwaters, Newhalen River	2.4	0.1	0.4	0.5	3.4
Outlet, Newhalen River	1.1	2.4	1.7	1.4	6.6
Roadhouse Creek	0.7	0.3	1.9	0.5	3.4
Iliamna Lake	30.4	1.8	4.1	3.9	40.2
Eagle Bay Creek	1.3	0.7	1.7	0.8	4.5
Youngs Creek Mainstem (Roadhouse Mountain HUC)	0.9	0.2	1.1	1.3	3.5
Youngs Creek East Branch[c]	0.3	0.5	0.8	1.5	3.1
Chekok Creek	1.8	0.2	0.3	0.2	2.5
Canyon Creek	0.8	0.0	0.2	0.3	1.3
Knutson Creek	1.0	0.1	0.6	0.3	2.0
Outlet, Pile River	0.3	1.2	1.6	0.5	3.6
Middle Iliamna River	3.1	0.6	1.7	1.5	6.9
Chinkelyes Creek	3.4	6.7	2.0	1.5	13.6
TOTAL	51.5	19.4	27.6	19.3	117.8

Notes:
[a] 190302060701
[b] 190302051404
[c] 190302060904
Values are summed by 12-digit Hydrologic Unit Code (HUC-12) within NWI digitized area, arranged from west (top) to east (bottom) along the potential transportation corridor.
NWI = National Wetland Inventory

In sum, the length of road within 200 m of NHD streams or NWI wetlands would be 80.2 km (Table 5-25). This takes into account the fact that the NWI dataset includes riverine wetlands that are also included in the NHD dataset. The methods used to estimate these values are described in Box 5-1.

5.4.9 Fish Populations along the Transportation Corridor

The Kvichak River watershed includes over 100 separate sockeye salmon spawning locations (Demory et al. 1964, Morstad 2003), including small tributary streams, rivers, mainland beaches, island beaches, and spring-fed ponds. The spatial separation and unique spawning habitat features within the watershed have influenced genetic divergence among spawning populations of sockeye salmon at multiple spatial scales (Gomez-Uchida et al. 2011). These distinct populations can occur at very fine spatial scales, with sockeye salmon that use spring-fed ponds and streams approximately 1 km apart exhibiting traits such as spawn timing, spawn site fidelity, and productivity consistent with a group of discrete populations (Quinn et al. 2012).

Table 5-25. Lengths of the potential transportation corridor located near water (within 200 m of NHD streams or NWI wetlands)

HUC-12 Name or Description	Proximity to Streams or Wetlands		
	Not Nearby	Within 200 m	Total
Headwaters, Upper Talarik Creek	0.0	7.5	7.5
Upper tributary stream to Upper Talarik Creek[a]	0.1	4.5	4.6
Tributary to Newhalen River portion upstream of corridor[b]	3.3	7.8	11.1
Headwaters, Newhalen River	2.3	1.1	3.4
Outlet, Newhalen River	1.1	5.5	6.6
Roadhouse Creek	0.0	3.4	3.4
Iliamna Lake	20.7	19.5	40.2
Eagle Bay Creek	0.9	3.6	4.5
Youngs Creek Mainstem (Roadhouse Mountain HUC)	0.7	2.8	3.5
Youngs Creek East Branch[c]	0.3	2.8	3.1
Chekok Creek	1.4	1.1	2.5
Canyon Creek	0.8	0.5	1.3
Knutson Creek	0.7	1.3	2.0
Outlet, Pile River	0.3	3.3	3.6
Middle Iliamna River	2.4	4.5	6.9
Chinkelyes Creek	2.6	11.0	13.6
TOTAL	37.6	80.2	117.8

Notes:
[a] 190302060701
[b] 190302051404
[c] 190302060904
Values are summed by 12-digit Hydrologic Unit Code (HUC-12) within NWI digitized area, arranged from west (top) to east (bottom) along the potential transportation corridor.
NHD = National Hydrography Dataset, NWI = National Wetland Inventory

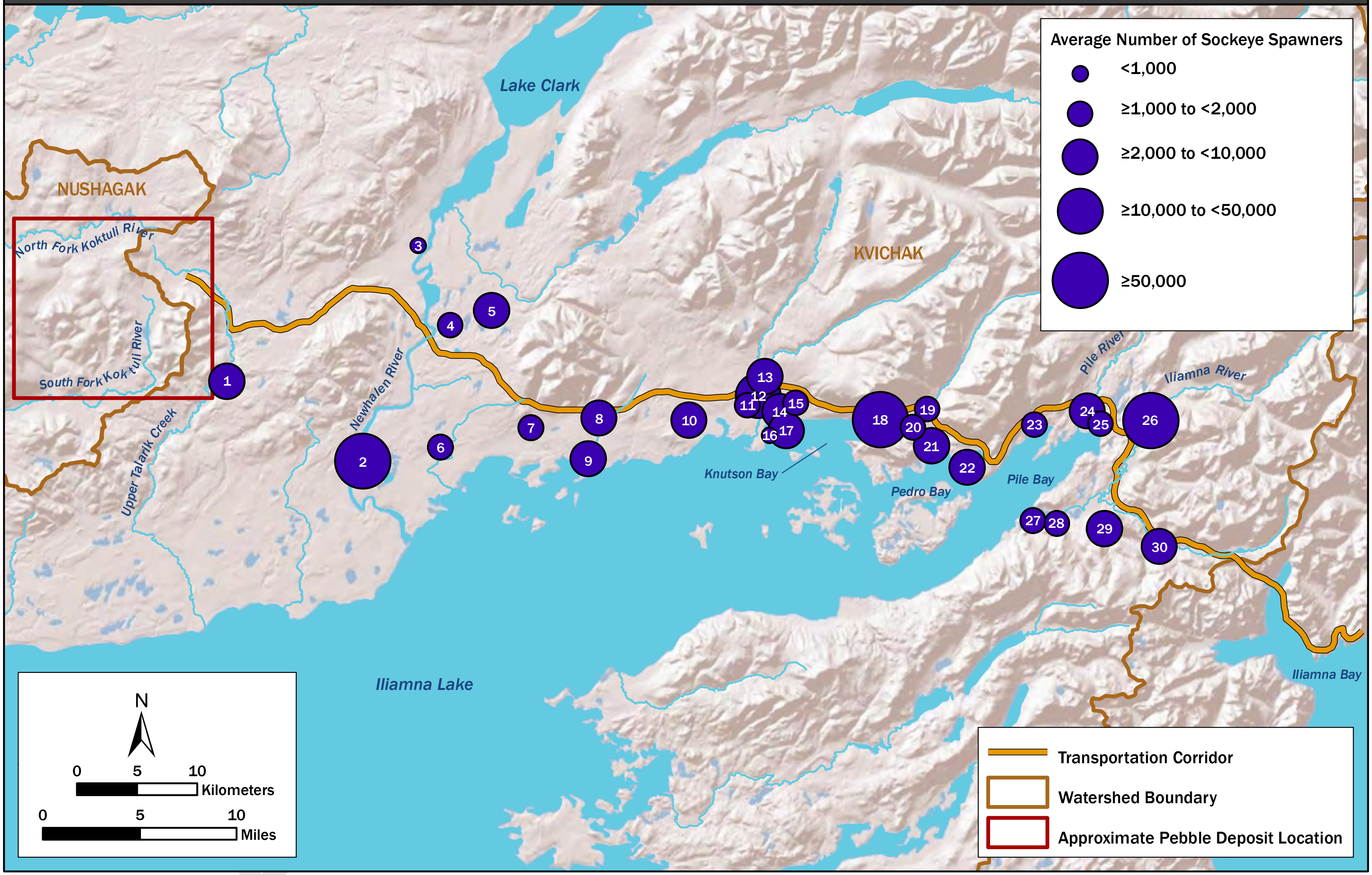
The transportation corridor would intersect multiple streams and rivers along the northern end of Iliamna Lake. Nearly a third of the spawning locations in Iliamna Lake identified by Demory et al. (1964) and Morstad (2003) are located in this portion of the lake. These locations include tributary streams, rivers, and spring-fed ponds draining into the lake (Figure 5-15, Knutson Bay inset). The transportation corridor would also run parallel to and 400 to 600 m from the Knutson Bay mainland beach spawning population. Sockeye salmon spawn along the north and south beaches of the bay, with the highest concentration in the northeast portion at depths between approximately 1 and 33 m (Demory et al. 1964). Sockeye salmon spawn at 29 locations along the transportation corridor. Indices of sockeye salmon spawning abundance at each of these locations vary considerably (Table 5-26, Figure 5-16).

Table 5-26. Average Number of Spawning Adult Sockeye Salmon at Locations near the Transportation Corridor

Map Point	Area	Area Name	Type	Average Number of Sockeye Salmon Spawners (1955–2011)	Number of Years Spawners were Counted (Max = 57)	Range
1	Upper Talarik	Upper Talarik Creek	Stream	7,021	49	0 –70,600
2	Newhalen River System	Newhalen River	River	84,933	34	97–730,900
3	Newhalen River System	Little Bear Creek/Ponds	Ponds	527	20	0–1,860
4	Newhalen River System	Alexi Creek	Stream	1,176	27	0–13,200
5	Newhalen River System	Alexi Lakes	Lake	7,121	33	11–38,000
6	North East	Roadhouse Creek	Stream	1,052	28	0–4,950
7	North East	N.W. Eagle Bay Creek	Stream	1,649	32	0–17,562
8	North East	N.E. Eagle Bay Creek/Ponds	Stream	3,416	38	0–18,175
9	North East	NE Eagle Bay Cr. Ponds	Ponds	4,766	5	200–11,700
10	North East	Youngs Creek	Stream	3,532	38	0–26,500
11	North East	Chekok Creek/Ponds	Stream	1,840	32	0–8,700
12	North East	Tomkok Creek	Stream	10,882	38	300–56,600
13	North East	Canyon Creek	Stream	8,015	38	200–48,000
14	North East	Wolf Creek Ponds	Ponds	4,469	26	0–28,000
15	North East	Mink Creek	Stream	1,144	35	0–6,000
16	North East	Canyon Springs	Ponds	884	20	0–5,000
17	North East	Prince Creek Ponds	Ponds	3,797	34	5–34,800
18	North East	Knutson Bay	Lake	72,845	47	1,000–1,000,000
19	North East	Knutson Creek	Stream	1,548	41	1–6,600
20	North East	Knutson Ponds	Ponds	1,200	39	0–6,350
21	North East	Pedro Creek & Ponds	Ponds	4,259	48	0–38,150
22	North East	Russian Creek	Stream	2,263	17	0–20,000
23	North East	Lonesome Bay Creek	Stream	1,026	6	32–2,675
24	North East	Pile River	River	6,431	38	0–39,200
25	North East	Swamp Creek	Stream	1,091	18	25–7,700
26	Iliamna River System	Iliamna River	River	101,306	53	3,000–399,300
27	Iliamna River System	Bear Creek & Ponds	Ponds	1,748	30	40–10,300
28	Iliamna River System	False Creek	Stream	1,317	21	0–13,300
29	Iliamna River System	Old Williams Creek	Stream	3,726	27	0–38,000
30	Iliamna River System	Chinkelyes Creek	Stream	9,128	46	50–44,905

Notes:
Locations are organized from west to east along the corridor. Adult counts from aerial surveys conducted by Alaska Department of Fish and Game and University of Washington
Sources: Morstad 2003; Morstad pers. comm.

Figure 5-16. Potential Transportation Corridor near the Shore of Iliamna Lake, Showing the Locations of Sockeye Salmon Surveys and Number of Spawners



Sockeye salmon are most abundant in Knutson Bay, Iliamna River, and the Newhalen River, averaging over 100,000, 80,000, and 70,000 spawners, respectively. These populations can have very large runs in good years. For example, the 1960 survey for Knutson Bay reported 1 million adults. Sockeye salmon spawn along the north side of Knutson Bay, adjacent to the transportation corridor. Spawning is associated with upwelling groundwater along the northern and eastern portions of the bay. Sockeye salmon use of spring-fed ponds is notable and occurs at eight locations along the corridor. These locations tend to have fewer spawners (approximately 2,700 on average), but fish using these locations may be more adapted to the unique abiotic features of ponds (Quinn et al. 2012).

Less is known about the occurrence or abundance of other salmon species in streams and rivers crossing or adjacent to the transportation corridor. Chinook, coho, and chum salmon are present in the Kvichak River watershed, but data for spatial occurrence are for isolated points in the system (ADFG 2012). Chinook and coho salmon are reported in the Newhalen River; Chinook, coho and chum salmon are reported in the Iliamna River; and coho salmon are reported in Tomkok and Youngs Creeks.

Rainbow trout and Dolly Varden are found in all of the sockeye salmon-bearing streams that would be crossed by or adjacent to the transportation corridor, such as Knutson Creek, Iliamna River, and Chinkelyes Creek (ADFG 2012). Rainbow trout may exhibit multiple life-history patterns (Meka et al. 2003), and seasonal movements between lakes and streams are likely in response to feeding opportunities and the need for winter thermal refuge. If fish passage were impaired due to poorly designed crossings, then those life histories that rely on moving between the lake and portions of streams above the road might be removed from the population.

5.4.10 Overall Risk of Transportation Corridor to Salmonid Populations

The risks to salmonids from siltation, hydrologic modification, filling of wetlands, and road salts are likely to diminish the production of anadromous and resident salmonids in more than 30 streams. Salmonid migrations and other movements may be impeded by culverts in 14 streams. The habitat potentially affected below the road crossings totals 270 km of stream, and an additional 240 km of stream upstream of the crossings would be affected if culverts impede fish movement. The magnitude of changes in fish populations cannot be estimated at this time.

5.5 Salmon-Mediated Effects on Wildlife

Routine operations under the mine scenario (Section 4.3) would cause the direct loss of wildlife habitat in the mine footprint and the transportation corridor. However, this assessment is limited to the effects on wildlife mediated by effects on salmon, so direct effects of habitat loss are not analyzed.

As described above, effects on salmon, trout, and char during routine operations would result from:

- the loss of 87.5 to 141.4 km of streams within or upstream of the mine footprint,
- reduced flow in each of the site watersheds,

- reduced habitat quality in the site watersheds, and
- reduced habitat quality in the streams crossed by the transportation corridor.

Each of these mechanisms would result in reduced salmon production, which would cause roughly proportionate reductions in wildlife that feed on salmon including brown bears, wolves, and bald eagles. The returning and spawning salmon are also important to wildlife in that they provide marine-derived nutrients (MDN) that fuel much of the productivity of the Bristol Bay watershed. Those MDN are deposited on the landscape by the salmon predators, where they increase the plant production that supports moose, caribou, song birds, and other terrestrial wildlife. Therefore, reduced salmon production would reduce the abundance and production of wildlife, but those effects cannot be quantified.

Concerns have been expressed that wildlife may be affected by consuming contaminated fish. The primary aquatic contaminant from a porphyry copper mine would be copper. Although copper is accumulated by both aqueous and dietary exposures, it does not biomagnify. In fact, in the Clark Fork River, copper concentrations were lower in fish than in invertebrates, and lower in invertebrates than in periphyton (ARCO 1998). Hence, contaminated fish do not pose a significant dietary risk to wildlife.

5.6 Salmon-Mediated Effects on Human Welfare and Alaska Native Cultures

In this section, we evaluate potential salmon-mediated effects of large-scale mining development on human welfare and Alaska Native cultures. Because the Alaska Native cultures (and to some extent the larger resident culture) is subsistence-based and particularly reliant on salmon, any negative impact on salmon quality or quantity could lead to a negative impact on health and welfare because of loss or change in food resources, and because of effects on an integral part of the culture. We do not attempt to quantify these impacts in this report, but discuss them qualitatively.

Because routine mine operations would destroy habitat within the mine footprint and preclude use in its vicinity, these areas would no longer be available for collection or production of subsistence resources, and current users would be displaced. According to subsistence data collected by ADFG and discussed in the EBD (Braund and Associates 2011 cited in PLP 2011), subsistence use of the mine area is high and centers on caribou, moose, and trapping. Because no subsistence salmon fisheries are documented in the mine footprint, the loss of non-salmon subsistence food resources likely would represent a greater direct effect than loss of salmon.

Section 5.2 discusses the relationship of these headwater areas to downstream salmon fisheries and estimates impacts related to habitat changes. Any negative impacts on downstream fisheries from headwater disturbance would affect subsistence salmon resources beyond the footprint. Likewise, any salmon-mediated effects on subsistence wildlife resources in the area would have corresponding impacts on subsistence users. For example, a reduction in plant material resulting from a decrease in MDN from salmon would result in a reduction in subsistence wildlife resources. PLP recently released significant subsistence use data for individual villages, which may help to quantify potential losses of subsistence resources in and around the mine site; however, it was not available in time for thorough review and analysis for this assessment.

A review of ADFG data indicates that some residents use the area along the transportation corridor for subsistence salmon harvest. Based on the analysis in Section 5.4, we anticipate that routine transportation operations would have some negative effects on salmon habitat in streams along and downstream from the transportation corridor, and likewise, some subsistence users in these areas could be displaced. The corridor also could result in long-term increased access opportunities, which could increase subsistence use of these streams but also create greater competition for this resource from new users of this corridor.

Human health and cultural effects related to potential decreases in salmon resources would vary with the magnitude of these reductions. A small reduction in salmon quality or quantity may not have significant impacts on subsistence food resources, human health, or cultural and social organization, but a significant reduction in salmon quality or quantity would certainly have significant negative impacts on these salmon-based cultures. It is not possible to develop a quantitative relationship between predicted effects on salmon and resulting effects on human health and culture; however, significant negative impacts on salmon or other subsistence resources would have negative impacts on elements of the Alaska Native cultures that are highly interrelated with and dependent on subsistence resources (Appendix D) (PLP 2011), such as:

- nutrition and physical health;
- mental and emotional health related to traditional culture;
- language and traditional ways to express relationships to the land, one another, and spiritual concepts;
- extended family relationships;
- strong social networks relating to the sharing of subsistence foods; and
- economic viability.

Even a negligible measurable reduction in salmon quantity or quality related to mining could decrease use of salmon resources, solely based on the perception of subtle changes in the salmon resource. Interviews with Tribal Elders and culture bearers indicate that perceptions of subtle changes to salmon quality are important to subsistence users, even if there are no measureable changes in the quality and quantity of salmon (Appendix D). This perception or fear of contamination could create a decrease in use of the salmon resource, or lead to cultural effects. Literature regarding the responses of Alaska Native communities to contamination of subsistence foods has not been fully evaluated for this assessment but could provide additional information about the role of perception in avoidance of subsistence foods.

Moreover, a reduction in downstream seasonal water levels as a result of mine-related withdrawals could pose obstacles for subsistence users who are dependent on water for transportation to fishing, hunting, or gathering areas.

It is not likely that any direct or indirect loss of subsistence use areas resulting from mine operations could be avoided. Under the mine scenario, the mine pit, waste rock piles, and TSFs would remain on the landscape in perpetuity and thus represent permanent habitat loss. Because the Alaska Native cultures in this area have significant ties to the specific land and water resources, which have evolved over thousands of years, it is not possible to replace elsewhere these subsistence use areas lost to mine operations.

Although this assessment is focused on salmon-mediated effects of mine operations on Alaska Native cultures, it should be noted that potential direct effects on other subsistence resources also could affect these cultures. Tribal Elders who were interviewed expressed concerns about ongoing mine exploration activities directly affecting wildlife resources, especially the caribou herd range (Appendix D). Development of a large-scale mine operation would have direct effects on wildlife subsistence resources within and around the mine footprint during operation, both from loss of habitat and disturbance from mining activities.

Mine construction and operation also would have direct economic and social effects on the Alaska Native culture. An influx of new residents in response to mine development could decrease the local population percentage of Alaska Natives and have a corresponding effect on local culture. Increased full-time employment in mining and secondary development could decrease subsistence activities and social relationships derived from these activities. While some residents have expressed a desire for jobs and development related to large-scale mining and a market economy, other residents have expressed concerns that this type of economic shift would be detrimental to their culture (Appendix D).

In summary, it is unlikely that there would be significant loss of salmon subsistence resources related to the mine footprint. Habitat modification in areas downstream of the mine site (Section 5.2) would have related impacts on downstream subsistence users. Some changes to salmon subsistence activities likely would result from development of the transportation corridor. In addition to the actual changes in subsistence resources, based on interviews with Tribal Elders, subsistence use could decrease downstream of the mine footprint, based solely on the perception that the salmon are being affected by the mine operation. Subsistence use could also decrease if fluctuations in downstream water levels reduce access for subsistence activities. Although this assessment is focused on salmon, the non-salmon-related impacts on Alaska Native cultures from routine mine operation are likely to be more significant, including cultural changes resulting from a shift to a market economy, increased access to the area, and direct effects on non-salmon subsistence resources.



CHAPTER 6. RISK ASSESSMENT: FAILURE

This analysis focuses on accidents and failures that are particular to metal mining and, if they occurred, would be most likely to cause significant effects on fish. Specifically, the analysis considers two magnitudes of a tailings dam failure, a break in the pipelines carrying product concentrate slurry and concentrate return water, and failure to collect or treat leachate waters from the mine site (Section 4.4). In addition, the assessment considers road and culvert failures that would block streams or degrade habitat. Other accidents or failures that could occur but are not considered include spills of process chemicals on site or during transportation, failure of a tailings slurry pipeline, diesel fuel spills, waste rock slides or erosion, fires, and explosions. These were judged to be less important, less well-specified or less germane to mining.

6.1 Tailings Dam Failure

As discussed in Section 4.4.2, we modeled two tailings dam failures resulting from flooding and overtopping at tailings dam facility (TSF) 1: a partial-volume failure that would occur when the TSF is partially full (dam height = 98 m), and a full-volume failure that would occur when the TSF is completely full (dam height = 208 m). For each failure, we assumed that 20% of the tailings stored in the TSF would be mobilized.

6.1.1 Overview of a Tailings Dam Failure

A breach of the TSF 1 dam would result in a flood wave and subsequent tailings deposition that would greatly alter the downstream channel and floodplain (Section 4.4.2). The initial flood wave for either a partially full or full TSF 1 breach would far exceed the typical flood event currently experienced in the study watersheds. The flood itself would have the capacity to scour the channel and floodplain and alter the landscape. In addition to the hydraulic flow, the quantity of tailings that could discharge from the TSF has the potential to bury the existing channel and floodplain system with meters of fine-grained material, and varying depths of sediment could create a completely different valley geomorphology. It is expected that the existing channel and floodplain would be eliminated and a new channel form would develop in the resulting topography. Given the fine grain size distribution (70% being finer than 0.1 mm) of these new deposits, channel form would remain unstable as the sediment would be highly mobile under typical flow events and could easily create scouring and transport flow velocities. The quantity of sediment on the floodplains and the remaining sediment in the breached dam would create a concentrated source of highly mobile material that does not currently exist in the study watersheds. The sediment regime of the affected stream and downstream waters would be greatly altered. This would transform the existing and well-defined gravel bed stream to an unstable, silt-dominated channel. A sediment transport study would be required to quantify the temporal and spatial distributions of effects, and the collection of data for such a study was beyond the scope of this assessment.

Remediation may occur following a tailings spill, but it is uncertain. A spill would flow into a roadless area and into streams and rivers that are too small to float a dredge, so the proper course of remediation is not obvious. The remediation process could be delayed by planning, litigation and negotiation, particularly concerning the proper disposal of the excavated tailings. If the operator was no longer present at the site or was no longer in existence, the response would, at best, be further delayed. Once started, the building of a road and support facilities and the excavation, hauling, and disposal of tailings could take years, particularly given the long winter season. Therefore, the extent to which tailings exposure downstream of the initial runout would be diminished by remediation cannot be estimated. Given this uncertainty, the assessment assumes that significant amounts of tailings would remain in the receiving watersheds for some time and remediation may not occur at all.

Similar effects would occur following tailings dam failures of TSF 2 or 3. However, the effect magnitudes would be smaller because a smaller quantity of tailings would be released.

6.1.2 Scour, Sediment Deposition, Turbidity

A tailings dam failure (described in Section 4.4.2) could have devastating effects on aquatic life and habitat. Both smaller (107 m) and larger (208 m) dam failures were modeled, providing estimates of instantaneous discharges and velocities associated with the dam break event, and the volume of sediment remaining in transport at the downstream end of the 30-km modeled reach (Section 4.4.2). We identified three processes associated with a tailings dam failure that would pose risks to aquatic habitat:

- Hydraulic scour and bed mobilization
- Deposition of tailings fines
- Mobilization and suspension of tailings fines creating turbidity

Additional risks associated with suspended sediments are discussed in Section 6.1.3, and those associated with toxicity of spilled water and deposited sediments are discussed in Section 6.1.4.

In the case of a tailings dam failure at TSF 1, the flood itself would mobilize existing sediments in the North Fork Koktuli River watershed. The volume of sediment mobilized would supplement the tailings released and could leave meters of material deposited in the floodplain. While a full sediment transport analysis is required to quantify actual deposition, it is very likely, based on this investigation, that a sediment volume and flood of this magnitude could greatly alter the valley morphology and introduce large volumes of fine-grained sediments that would continue to be transported downstream beyond the mouth of the North Fork Koktuli River. Failures at tailings dams in other headwater streams would be expected to cause qualitatively similar effects.

6.1.2.1 Exposure

Initial Deposition

The tailings dam failures described above would result in intense scour and deposition in the North Fork Koktuli valley, from the tailings dam downstream to at least the confluence with the South Fork Koktuli River, a distance of approximately 30 km. The volume of available fine tailings material that could be mobilized could result in meters of deposition of tailings fines across the entire valley, to at least the confluence with the South Fork Koktuli River (Tables 4-11 through 4-14), with continued erosion and transport of fines as the channel adjusts to the vastly increased fine sediment supply.

To translate this tailings dam failure into effects on aquatic habitat and biota, we assumed that the velocities calculated during the tailings dam failure flood event (Table 4-11) would result in a nearly complete reworking of the existing North Fork Koktuli channel and much of the valley. Given the volumes of material that would be exported from the TSF, we assumed that the new valley floor would be predominately tailings material with particle sizes ranging from less than 0.01 mm to just over 1.0 mm, of which 70% would be finer than 0.1 mm. Following the recession of the tailings dam failure flood event, we assume that the bed and bank would be primarily tailings material, with incorporated dam fill and valley fill material accounting for less than 20%.

Both magnitudes of tailings dam failure would completely eliminate suitable spawning and rearing habitat for salmon and other native fishes in the North Fork Koktuli River downstream of the tailings dam, immediately following the tailings dam failure event. Tributaries of the North Fork Koktuli River, including portions of the watershed upstream of North Fork Koktuli River Tributary 1.190, could also be adversely affected. Temporary flooding of tributary junctions during the tailings dam failure event, and subsequent deposition of sediments at confluence zones that caused local aggradation, steepening, or shallowing of tributary confluences, could make movement of resident and anadromous fish between tributaries and the mainstem more difficult. Recovery of channel dimensions and substrate size distributions suitable for salmonid (salmon, trout, and char) spawning and rearing habitat would be contingent upon rates of fine sediment export and recruitment of gravels and larger substrates from tributaries or pre-failure valley fill.

No tailings dam failure has been monitored sufficiently to provide information on recovery. However, stream recovery following the Mount St. Helens volcanic eruption in 1980 provides an analogy.

Recovery of stream channels was relatively rapid where the only disturbance was airfall deposition of up to 1 m of silt to gravel-sized sediments generated from the blast (Meyer and Martinson 1989). Post-eruption sediment yields diminished to background within 5 years (Major et al. 2000). However, for stream valleys subject to lahars and debris flows following the Mount St. Helens eruption, stream channels experienced periods of channel widening and aggradation interspersed with episodic channel incision, and stream channels remained unstable and contributors of sediment volumes up to 10 times background levels 20 years later (Major et al. 2000). These stream valleys provide a better analogy to our modeled tailings dam failure. Further, the relatively low gradients in the Koktuli River watershed would likely result in slower sediment erosion than at Mount St. Helens. We estimate that recovery of suitable structural habitat in the North Fork Koktuli River watershed would likely take decades given the volume of sediment that would potentially be delivered under the tailings dam failure described above.

Subsequent Sediment Transport and Re-Deposition

The TSF 1 tailings dam failure described above would have the potential to fill the North Fork Koktuli valley with extensive deposits of tailings fines less than 0.1 mm in size and still carry a substantial volume of fine sediments downstream into the mainstem Koktuli, Mulchatna, and Nushagak Rivers. The volume of material remaining in transport at the confluence of the North Fork and South Fork Koktuli Rivers and available for deposition in the mainstem Koktuli, Mulchatna, and Nushagak Rivers following the 107-m tailings dam failure would range from 6.6 to 40.6 million m^3, depending on the proportion of TSF fill material that is mobilized in the spill (5 to 20%; Table 4-13). The volume of sediment remaining in transport at the confluence following the 208-m (full) tailings dam failure would range from 15.9 to 239.3 million m^3 (Table 4-13). The depth and distribution of fines in the mainstem Koktuli, Mulchatna, and Nushagak Rivers cannot be estimated at this time, but it is reasonable to expect that continued pulses of fine sediments would be transported through and transiently stored in these mainstem river sections during spring snow melt and fall rain events for many years (Major et al. 2000). Transport of suspended material and deposition of tailings fines would have significant, adverse effects on spawning and rearing salmon in these lower river reaches, via habitat impacts described above as well as via reductions in primary production and abundance of macroinvertebrates (Lloyd et al. 1987).

6.1.2.2 Exposure-Response

Natural Sediment

Natural background conditions provide an indication of the sediment levels that could be achieved and that support the current productivity of salmonid populations. Two available sources provide data on substrate size distribution and fine sediment concentrations in the study area. Pebble Limited Partnership's (PLP's) Environmental Baseline Document (EBD) (PLP 2011: Appendix 15.1F, Fluvial Geomorphology Studies) reports concentrations of fine sediments from sieved bulk gravel samples collected at one known salmon spawning site in each of the study streams: North Fork Koktuli River, South Fork Koktuli River, and Upper Talarik Creek (PLP 2011: Figure 4 in Appendix 15.1F). Average concentration of fines (less than 0.84 mm) was less than 6% for all streams and dates, except for the August sample from the uppermost South Fork Koktuli site (SGSK3) (PLP 2011: Figure 4 in Appendix 15.1F), which had nearly 8% fines. The geometric mean diameter was greater than 15 mm at all sites for both sampling periods, except the uppermost Upper Talarik Creek site (SGUT3) (PLP 2011: Figure 4 in Appendix 15.1F), where the mean diameter for both seasons was between 10 and 15 mm. These data led the authors to conclude that gravel quality was generally high and that, based on published criteria (Shirazi et al. 1981, Chapman and McLeod 1987, Kondolf 2000), salmonid survival to emergence would be "high" (presumably above 80%) at all sites except the uppermost Upper Talarik Creek site, where criteria predicted survival between 50 and 80% (PLP 2011).

Areal coverage of substrate sizes is available for 77 wadeable stream sites around the Kvichak and Nushagak watersheds, including one site each on the North Fork Koktuli River, South Fork Koktuli River, and Upper Talarik Creek (Table 6-1). These pebble counts followed U.S. Environmental Protection Agency (USEPA) methodology (Peck et al. 2006), where five particles were systematically selected across each of 21 evenly spaced transects (from each wetted margin and from three locations in between). These data indicate that a mix of substrate sizes occurs in streambeds in the region, and that cobble and gravel are generally abundant. Pebble counts from riffles at many sites in the study watersheds (15 on the North Fork Koktuli, 16 on the South Fork Koktuli, 1 on the Main Fork Koktuli, and 17 on Upper Talarik Creek) also show a mix of substrate sizes with abundant gravel and generally small amounts of fine sediment, although the smallest size class reported is 2 mm (PLP 2011: Appendix 15.1F).

Fish

The State of Alaska standard for accumulation of fine sediment (0.1 to 4.0 mm) is "no more than 5% increase by weight above natural conditions (as shown by a grain size accumulation graph) with a maximum of 30% fines in waters used by fish for spawning" (ADEC 2011). Bryce et al. (2010) found that even slight increases (exceeding 5% fines or 13% sands and fines) in streambed fine sediments were associated with declines in sediment-sensitive aquatic vertebrates, including salmonids. The tailings dam failure described above would completely scour and transport or bury existing substrates in the North Fork Koktuli River valley under several meters of tailings fines, greatly exceeding all sediment criteria for salmonid spawning. Continued erosion and transport of fines deposited on bars, floodplains and terraces would provide a chronic source of additional fines during precipitation events, providing new inputs of fines during fall spawning and early egg incubation. Exceedance of fine sediment standards in the entire North Fork Koktuli River would be a likely outcome for years to decades.

Table 6-1. Sediment Size Distributions. Surveyed at Upper Talarik Creek, North Fork Koktuli River, South Fork Koktuli River, and wadeable stream sites in the Nushagak and Kvichak watersheds. Figures represent % areal coverage based on 105 systematically selected particles at each site, following USEPA methods. All data were collected during June.

River or Stream(s)	Date	Latitude	Longitude	% Large Boulder (>1000mm)	% Small Boulder (250-1000mm)	% Cobble (64-250mm)	% Coarse Gravel (16-64mm)	% Fine Gravel (2-16mm)	% Sand (0.06-2mm)	% Fines (<2mm)
Upper Talarik	6/13/2011	59.91820	-155.27771		2	30	29	13	24	2
North Fork Koktuli	6/6/2009	59.84033	-155.71272			17	49	24	10	
South Fork Koktuli	6/8/2010	59.83047	-155.27719		3	3	51	15		22
77 Streams	2008 to 2011			3(±2)	4(±5)	15(±13	39(±15)	17(±11)	17(±11)	12(±10)

Sources: Rinella pers. comm., Peck et al. 2006

Interstitial spaces between channel substrate particles used by juvenile salmonids for overwintering and concealment habitats are a critical habitat resource, particularly in northern ice-bound rivers and streams (Bustard and Narver 1975, Cunjak 1996, Huusko et al. 2007, Brown et al. 2011). Interstitial habitat would likewise be initially eliminated by the tailings dam failure, and then subject to high levels of embeddedness by infiltrating tailings fines as new channels erode into the new valley fill composed of tailing fines. The new sediment regime in the North Fork Koktuli River watershed and associated transport and storage of massive quantities of fine sediments would essentially eliminate interstitial habitat from the watershed for years to decades, if not longer. The altered valley morphology and substrate composition would also very likely lead to changes in groundwater flowpaths and interactions with surface waters. Infiltration and burial of coarse valley fill by fine sediments could greatly reduce hydraulic conductivity and result in decreased rates of exchange between surface water and groundwater (Hancock 2002). As a result of these habitat changes, suitable spawning environments and overwintering habitats for salmon would be greatly diminished in this watershed. This would likely lead to severe declines in salmon spawning success and juvenile survival (Wood and Armitage 1997).

Invertebrates

Aquatic macroinvertebrates are an important food for Chinook salmon and coho salmon, rainbow trout, Dolly Varden, Arctic grayling, and other fishes that rear in the study area's streams (Nielsen 1992, Scheuerell et al. 2007). Two available data sources describe the existing macroinvertebrate communities for streams in the study area: the EBD (PLP 2011: Chapter 15.2) and Bogan et al. (2012). Both documents describe broadly similar communities that are consistent with those reported from other regions of Alaska (Oswood 1989). Communities are reasonably diverse: Bogan et al. (2012) reported 137 taxa from 38 families, with 9 to 40 taxa occurring at a given site (Chironomidae were lumped at the family level). Communities are dominated by Diptera (true flies), primarily Chironomidae (non-biting midges), with lesser numbers of Ephemeroptera (mayflies) and Plecoptera (stoneflies) and relatively few Trichoptera (caddisflies). Macroinvertebrate densities were characteristically variable, ranging two orders of magnitude (102 to11,371 organisms per m^2) (Bogan et al. 2012).

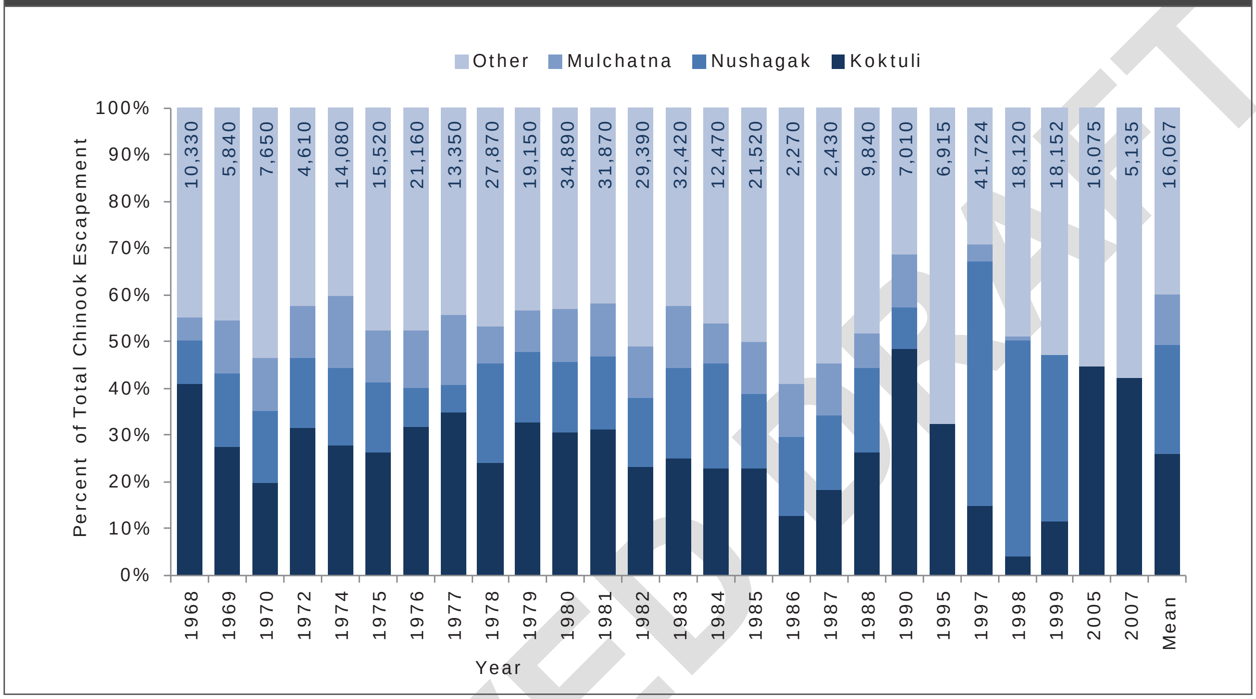
Catastrophic sedimentation associated with the tailings dam failure, in addition to the direct impacts described above, would likely affect fish populations through reductions in macroinvertebrate food resources (the toxicology of released tailings is addressed in Section 6.1.4, so this discussion addresses only changes in macroinvertebrate communities due to changes in habitat). Sedimentation can affect benthic macroinvertebrates through abrasion, burial, reductions in living space, oxygen supply, and food availability (Jones et al. 2011). The effects of sedimentation have been reviewed thoroughly, and are largely deleterious (Wood and Armitage 1997, Jones et al. 2011). Sedimentation typically leads to reductions in density and taxonomic diversity (Wagener and LaPerriere 1985, Culp et al. 1986, Quinn et al. 1992, Milner and Piorkowski 2004), even at sediment loads substantially lower than those modeled under the tailings dam failure (Wood and Armitage 1997, Jones et al. 2011). The conversion of a stable streambed dominated by gravel and cobble to a highly unstable one composed entirely of fine sediments, as described in the tailings dam failure, would certainly lead to reductions in the biomass and diversity of macroinvertebrate prey available to fish populations.

6.1.2.3 Risk Characterization

The complete loss of suitable salmon habitat in the North Fork Koktuli mainstem in the short term (less than10 years), along with the likelihood of very low-quality spawning and rearing habitat in the long term (decades), would result in near-complete loss of the mainstem North Fork Koktuli fish populations downstream of the tailings dam. These impacts would persist for multiple salmon life cycles, so salmon cohorts that are at sea during the tailings dam failure would eventually return to find degraded spawning and rearing habitat. The North Fork Koktuli River watershed currently supports spawning and rearing populations of sockeye salmon, Chinook salmon, and coho salmon, and spawning populations of chum salmon (Johnson and Blanche in press)). Dolly Varden and rainbow trout rearing is also supported in the North Fork Koktuli (ADFG 2012). The Koktuli River watershed has been recognized as an important producer of Chinook salmon for the greater Nushagak River Management Zone (Dye and Schwanke 2009, ADFG 2011), which, in turn, is the largest producer of Chinook salmon for the Bristol Bay region, with annual runs averaging over 160,000 fish (1966 through 2010) (Dye and Schwanke 2009, Buck pers. comm.) Of all the Chinook salmon tallied during annual aerial index counts in the Nushagak River watershed, on average 28% (range 4 to 55%) are counted in the Koktuli River (Figure 6-1) (Dye and Schwanke 2009). The Mulchatna River accounts for another 10% (range 1 to 15%) of the Nushagak Chinook salmon count, and the Stuyahok River (drains to the Mulchatna downstream of the Koktuli) represents another 17% (range 3 to 42%). Hence, Chinook salmon production could be significantly degraded by loss of habitat downstream of the tailings dam.

Sockeye salmon are the most abundant salmon returning to the Nushagak River watershed, with annual runs averaging more than 1.3 million fish (1956 through 2010) (Baker pers. comm.). Spatially extensive sockeye salmon spawner data are not available for the Nushagak River watershed, so it is impossible to estimate what proportion of the population spawns in the Koktuli River. Sockeye salmon are generally dependent on nursery lakes for 1 to 2 years of juvenile residence, suggesting that sockeye salmon distribution in the Nushagak River watershed should be associated with lakes outside of the Koktuli River watershed. However, in northern climates sockeye salmon may also migrate directly to sea after emergence (“sea-type”) or reside in rivers for 1 to 2 years (“river-type”) before going to sea. The river-type can be common and represent a substantial proportion of the total return (Wood et al. 1987) if lakes are not available and riverine conditions are favorable. Most sockeye salmon from the Nushagak and Mulchatna Rivers are sea-type (Yuen and Bill 1990), as is approximately 20% of the overall Nushagak River sockeye salmon population (1979 through 2003) (Sands pers. comm.). The tailings dam failure would likely affect sockeye salmon production throughout the Koktuli River, but the proportion of the total Nushagak River production that would potentially be affected is unknown. See Section 5.1 for more information on fish abundance.

Figure 6-1. Escapement Counts of Chinook Salmon in Select Streams of the Nushagak-Mulchatna River Watershed, as Assessed via Aerial Surveys (Dye et al. 2006). Values are total counts from all watersheds for that year. Totals include counts from the Iowithla, Kokwuk, Klutispak, King Salmon, and Stuyahok Rivers (combined as "Other" in plot), and the Koktuli, Nushagak and Mulchatna Rivers. Data from some years was not included because only a few watersheds or no watersheds were surveyed. Survey conditions in 1997 were noted as especially favorable for aerial surveys.



Populations of resident and anadromous fishes present in North Fork Koktuli headwaters and tributaries at the time of the tailings dam failure would not immediately suffer loss of habitat, but would suffer indirect effects resulting from alteration of the North Fork Koktuli River valley. Many species in the region's rivers, including resident non-anadromous species, undergo extensive seasonal migrations (West et al. 1992). Such movements are important for juveniles moving from natal areas to overwintering habitats, for adult spawning migrations, or, in the case of resident species, for migration between areas for spawning, foraging, and over-winter thermal refugia. Sediment deposition at tributary mouths in the North Fork Koktuli River valley could adversely affect passage of juvenile and adult fish into and out of these tributaries. For several years, mainstem river habitats upon which many tributary fish depend upon for portions of their life history could be gone or severely degraded.

Successful re-colonization of the North Fork Koktuli by resident fish would depend on whether unimpaired tributary habitats function as suitable refugia and source areas for re-colonization of the North Fork Koktuli following disturbance. Salmon would require sufficient tributary habitat to complete their entire life history, as it is likely that downstream habitat would be unusable for multiple generations. Re-colonization of fish from their tributary refugia or downstream areas would require suitable passage at tributary junctions, and suitable migratory corridors throughout the mainstem. Aquatic macroinvertebrate food resources would likely also be adversely affected in the main river channel (Section 6.1.2.2), limiting rearing potential for insectivorous fish like juvenile salmonids. Given estimates of the depth of fine-sediment deposition and the unstable, silt/sandbed channels that would likely form across the valley floor, successful migratory conditions seem unlikely for several years after a tailings dam failure.

The near-complete loss of the North Fork Koktuli fish populations and long-term transport of fine sediment to downstream locations would have significant adverse effects on the Koktuli and Nushagak salmon, Dolly Varden, and rainbow trout populations. Direct loss of habitat in the North Fork Koktuli, and downstream impairments through either direct impairment of spawning and rearing habitat from transported sediment settling out or suspended sediment affecting water quality and juvenile or adult migration or rearing, could adversely affect a substantial portion of Chinook salmon returning to the Nushagak depending on the extent of impairment. Assuming that Alaska Department of Fish and Game (ADFG) aerial survey counts reflect the proportional distribution of Chinook salmon in the Nushagak River watershed, habitat destruction of the North Fork Koktuli River valley, downstream transport of sediment to the Koktuli mainstem, and the subsequent loss of access to the South Fork Koktuli would affect, on average, 28% of the Nushagak River Chinook salmon run in a given year (Figure 6-1) (Dye and Schwanke 2009). If the deposited tailings material is deep enough to impede fish access to the Mulchatna and Stuyahok Rivers, then a tailings dam failure could affect more than half of the Nushagak River Chinook salmon population (Figure 6-1) (combined counts from the Stuyahok, Koktuli, and Mulchatna Rivers average 52% of total Nushagak count; range 8 to 72% of total).

6.1.2.4 Uncertainties

While it is certain that a tailings dam failure could have devastating effects on aquatic habitat and biota, the distribution and magnitude of effects is uncertain. Uncertainties associated with the initial events, including the dam failure likelihood, sediment transport, and sediment deposition, are discussed in Section 4.4.2. Uncertainties regarding the risks to habitat are related to the timeframe for geomorphic recovery, the longitudinal extent and magnitude of habitat impacts downstream of the end of our modeled reach of the North Fork Koktuli River, and the fish populations affected. These uncertainties are discussed here.

We estimate that recovery of suitable structural habitat in the mainstem North Fork Koktuli and off-channel areas would likely take decades, given the scouring action of the flood wave and the volume of fine-grain sediment that would potentially be delivered under the tailings dam failure. However, the time period for recovery could be substantially longer. Recovery of suitable gravel substrates and development of channel morphology suitable for salmon habitat could be delayed even further if the flood wave were to scour much of the North Fork Koktuli valley to bedrock, which would then be buried under massive deposits of tailings fines. Recruitment of gravels to the North Fork Koktuli valley could be delayed by low supplies and/or low rates of transport of gravels and coarser substrate particles from tributaries or unaffected upstream sources. Recovery may also be delayed if the riparian vegetation does not recover because the tailings are toxic to plants. However, that causal pathway is not assessed.

The tailings dam failure simulation (Section 4.4.2) was restricted to approximately 30 km of the North Fork Koktuli River, from the face of the TSF 1 dam downstream to the confluence of the North Fork

Koktuli River and South Fork Koktuli Rivers. Extension of the simulation beyond the confluence would introduce significant error and uncertainty associated with the contribution of South Fork Koktuli flows. This analysis would require a more sophisticated sediment transport model. As a result, we were unable to quantify sediment transport and deposition in the mainstem Koktuli, Mulchatna, and Nushagak Rivers. Given the high volume of tailings fines that would be transported beyond the confluence of the North Fork Koktuli and the South Fork Koktuli Rivers (Table 4-13), it is highly likely that impacts on fish habitat estimated for the North Fork Koktuli would extend for some significant distance down the mainstem Koktuli River and possibly further. We are unable to quantify those effects.

We estimate that the combined effects of direct losses of habitat in the North Fork Koktuli, downstream in the mainstem Koktuli and beyond, and impacts on macroinvertebrate prey for salmon could adversely affect 30 to 50% of Chinook salmon returning to spawn in the Nushagak River watershed. Uncertainty around this estimate is associated with the downstream extent of habitat impacts (described above) and the variable and imprecise estimate of the relative abundance of Chinook salmon in the Nushagak, Mulchatna and Koktuli Rivers. While aerial counts can substantially underestimate true abundance (Jones et al. 1998), we based our estimate on long-term (1967 to 2007) aerial counts of Chinook salmon collected and interpreted by ADFG (Dye and Schwanke 2009).

Because long-term abundance data are lacking for most other fish species and locations in the project area, losses caused by a tailings dam failure could not be quantified. Information documenting known occurrence of fish species in rivers and major streams is available (Johnson and Blanche in press, ADFG 2012), but not abundances, productivities, or limiting factors.

6.1.3 Suspended Tailings Particles

6.1.3.1 Exposure

During a tailings dam failure, aquatic biota would be exposed to a slurry of suspended tailings moving at up to 6.1 m/s (Table 4-12). Thirty km downstream, at the confluence of the South Fork Koktuli (the limit of the model), much of this material would still be flowing (Table 4-13).

For years after a tailings dam failure, settled tailings would be re-suspended and carried downstream. At first, this process would be frequent if not continuous (except during periods of freezing), as a channel and floodplain structure is established by erosional processes suspending the tailings (Section 4.4.2). Gradually, as the tailings flowed downstream, a substrate consisting of gravel embedded in tailings fines would become established, and the flow velocities necessary to suspend sediment would increase until they resembled those of an undisturbed stream.

Studies at other tailings-contaminated sites do not usefully address suspended tailings, as they have been carried out long after the spills occurred, are based on events that differ from the one large spill that would result from a tailings dam failure, and focus on toxic properties of the tailings (Section 6.1.4). However, based on studies of volcanic ash deposition at Mount St. Helens, reduction of suspended sediments to natural levels is expected to take decades (Section 6.1.1).

6.1.3.2 Exposure-Response

Suspended sediment has a variety of effects on fish that are equivalent to effects of toxic chemicals. Like chemical effects, the severity of effects increases with concentration and duration of exposure (Newcombe and Jensen 1996). At low levels, suspended sediment causes physiological and behavioral effects; at the highest levels it causes death. Salmonids avoid turbid waters when possible, which may result in loss or underutilization of traditional spawning habitats (Bisson and Bilby 1982, Newcombe and Jensen 1996). However, salmonids must withstand brief periods of high suspended sediment concentrations associated with spring floods (Rowe et al. 2003). Empirically derived effective exposures for lethal and sublethal effects (i.e., reduced abundance or growth or delayed hatching) on juvenile and adult salmonids may be summarized as 22,026 mg/L for 1 hour, 2,981 mg/L for 3 hours, 1,097 mg/L for 7 hours, 148 mg/L for 1 to 2 days, 55 mg/L for 6 days, 7 mg/L for 2 weeks, and 3 mg/L for 7 weeks to 11 months (derived from (Newcombe and Jensen 1996).

6.1.3.3 Risk Characterization

During and immediately after a tailings spill, exposure to suspended sediment would be far higher than any of the effects thresholds. Fish could be literally smothered and buried in the slurry. For years thereafter, erosion of tailings from the re-formation of the channel and floodplain is likely to exceed 1,000 mg/L of suspended sediment for days at a time, so fish are likely to avoid these streams or experience lethality, reduced growth, or reduced abundance. Avoidance could also block migrating salmon and other fish from their spawning areas in upstream tributaries at these times. The potential for tailings to be more aversive or toxicologically effective than natural suspended sediment is unknown. Exposure levels would gradually decline over time as tailings are carried downstream, channel stability increase, and the floodplain becomes revegetated. Rates of these processes are unknown, but, based on analogy to volcanic ash, it is reasonable to assume that decades would be required for suspended sediment loads to drop to levels that occur with normal high flows in stable channels of the Bristol Bay watershed.

6.1.3.4 Uncertainties

There can be little doubt that, during and in the years immediately following a tailings dam failure, suspended sediment concentrations would be sufficient to cause the loss of fish populations for many kilometers downstream of a failed tailings dam. A major uncertainty, however, is the number of years required to reduce suspended sediment concentrations to levels that are not adverse. Another major uncertainty is the downstream extent of the effects. The data and modeling effort required to determine how far the initial slurry deposition would extend, how far re-suspended sediments would travel, and how long erosional processes would continue were not feasible for this assessment.

6.1.4 Tailings Constituents

The most dramatic effect of a tailings dam failure would be exposure to the flowing tailings slurry and subsequent habitat destruction and modification; however, exposures to potentially toxic materials would also occur. While the effects of a tailings dam failure can be assessed using the composition of the tailings and of experimental tailings leachates, experience with tailings spills at other sites also provides important evidence. Descriptions of these cases are presented in Box 6-1.

6.1.4.1 Exposure

Aqueous Exposures to Waters from the Impoundment

During a tailings dam failure, aquatic biota would be exposed to water that had been in contact with tailings during processing and in the TSF. This water includes pore water associated with the deposited tailings and water overlying the tailings. If the spill was caused by flow through a fault in the dam or by a seismically induced tailings dam failure, pore water and supernatant water would be released. However, if the dam was eroded or overtopped by a flooding event, as in a tailings dam accident (Section 4.4.2), the pore and surface water could be diluted by fresh water.

A spill would have two phases. At first, tailings slurry would pour through the breach for approximately 3 hours based on the assumed rate of dam erosion and slurry flow. Then pore water would drain from the residual tailings. The latter process is slow and could continue until the dam was repaired. If a tailings dam failure occurred after the mine site was abandoned, equilibrium would be achieved in which rain, snow, and upstream flows were balanced by outflow of leachate through the breach.

Once in the stream, toxic constituents dissolved in the water, unlike the tailings, would not settle out. Because the potentially toxic constituents are not degradable or volatile, they would flow to Bristol Bay. However, the constituents would be diluted along the way. In a potential maximum failure of the tailings dam at TSF 1, the flow of spilled water at the bottom of the North Fork Koktuli River is estimated to be 3,266 m^3/s (Section 4.4.2). The Nushagak River at Ekwok would be the first gaging station downstream where most of the tailings would have settled out and dilution could be estimated. Using the annual average and highest monthly average flows (668 and 1,215 m^3/s, respectively), proportionate dilution of the spilled water by the Nushagak River would be only 0.83 and 0.73. The highest monthly average flow is a reasonable comparison. Because the hypothesized flood that would cause the dam to fail is an intense local storm, a flood at the scale of the Nushagak River watershed is not implied, but relatively high flows would occur. Minimum flow is not considered because we assume that overtopping failure would not occur in winter (although winter overtopping did occur at Nixon Fork Mine as a result of human error; Box 6-2).

BOX 6-1. BACKGROUND ON RELEVANT ANALOGOUS TAILINGS SPILL SITES

Past deliberate or accidental spills of metal mine tailings into salmonid streams and rivers provide evidence concerning the nature of exposures to aquatic biota. In the United States, some of these sites are relatively well-studied because the observed effects of such spills have led to their classification as Superfund sites. Other tailings spills caused extensive fish kills and other significant effects, but have not generated useful long-term monitoring data. These brief descriptions provide background information and support the use of evidence from these cases in analyzing risks from a hypothetical tailings dam failure in the Bristol Bay watershed.

Clark Fork River, Montana

The Clark Fork River Operable Unit of the Milltown Reservoir/Clark Fork River Superfund Site includes 120 river miles (193 km), extending from the river's headwaters to the Milltown Reservoir, just east of Missoula, Montana. Mining for gold, silver, copper, lead, and zinc began in the Clark Fork watershed in the late 1800s. Most of the wastes released were tailings from copper mines in Butte and Anaconda, but aqueous mine discharges and aerial smelter emissions also contributed. Two sedimentation ponds were constructed by 1918, with a third constructed by 1959. Mine water treatment was initiated between 1972 and 1975. By the mid-1970s, waste inputs to the Clark Fork were largely limited to movement of previously released solids. It became a Superfund site in 1983. Contaminants of concern are arsenic, cadmium, copper, lead, and zinc, but copper was the focus of assessment and planning because of its high toxicity.

The primary source of exposure is tailings deposited on the floodplains, resulting in aquatic pollution through erosion and leaching. Large areas with acidic tailings (both acidic and neutral tailings were deposited) are barren of plant life due to metal toxicity, which contributes to erosion and leaching. The river was fishless from the late 1800s to the 1950s, but has begun to recover. Trout and other fish continue to exhibit low growth and abundance, and intermittent fish kills have followed metal pulses from rain storms or rapid snow melt. However, sedimentation was also thought to contribute to effects on fish populations through habitat degradation.

More detailed information can be found in the responsible party's remedial investigation (ARCO 1998) and in USEPA documents (USEPA 2012a).

Coeur d'Alene River, Idaho

The Coeur d'Alene River basin in northern Idaho flows from the Bitterroot Mountains to Lake Coeur d'Alene. From the late 19th to late 20th century, the upper basin was mined for silver, lead, zinc, and other metals, and much of the ore was smelted locally. Tailings were dumped into gullies, streams, and the river until dams and tailings impoundments were built beginning in 1901. Plank tailings dams failed in the 1917 and 1933 floods; direct discharge of tailings did not end until 1968. According to the USEPA's remedial investigation, approximately 56 million metric tons (62 million tons) of tailings were discharged to the Coeur d'Alene River. In 1983, the area of the Bunker Hill smelter was added to the Superfund national priority list and in 1998, the contaminated river watershed, Lake Coeur d'Alene, and part of the Spokane River were explicitly included.

Metals concentrations above ambient water quality criteria, lethality in tests of ambient waters, and the absence of some fish species from reaches with high metal concentrations were all attributed to leachates from tailings and other mine wastes in floodplains and tributary watersheds. In addition, toxicity of bed sediments, which include tailings, was found in the Coeur d'Alene and Spokane Rivers and tributaries. Aquatic effects were attributed primarily to zinc, but cadmium, lead, and copper also reached toxic levels.

More detailed background information can be found in the Ecological Risk Assessment for the Coeur d'Alene Basin Remedial Investigation/Feasibility Study (USEPA 2001), other USEPA documents (USEPA 2012b) and the National Research Council's review of USEPA's assessment and management documents (NRC 2005).

Soda Butte Creek, Montana and Wyoming

The headwaters of Soda Butte Creek drain the New World mining district in Montana before entering Yellowstone National Park. From 1870 to 1953, porphyry deposits were mined for gold and copper with some arsenic, lead, silver, and zinc. In June 1950, the earthen tailings dam at the McLaren mine failed, releasing approximately 41 million m^3 of water and an unknown mass of tailings into Soda Butte Creek (Marcus et al. 2011). In 1969, the creek was rerouted around the tailings pile was covered and seeded. 1n 1989, a Superfund emergency response re-created and riprapped the creek channel to accommodate a 100-year flood. Despite these actions, metal levels remain high in the creek and floodplain sediments and the biota are impaired. The lack of any decrease in sediment copper despite floods in 1995, 1996, and 1997 and the lack of macroinvertebrate recovery following remediation of acid drainage in 1992 indicate that the tailings are persistent and are the primary cause of biological impairments. The primary sources of information on effects of the tailings spill are academic studies (Nimmo et al. 1998, Marcus et al. 2001).

BOX 6-2. AN ACCIDENTAL TAILINGS WATER RELEASE: NIXON FORK MINE, ALASKA, WINTER 2012

The Nixon Fork Mine is an underground gold mine that was intermittently mined between 1917 and 1950. The modern mine opened in 1995 then closed in 1999 (ADNR 2012) and reopened under new ownership again in 2007. The current operation is mining two ore bodies with a defined resource of 241,966 metric tons (266,755 tons) of ore containing an estimated 4.6 million grams (162,550 ounces) of gold (ADED 2012). An additional 856,156 grams (30,200 ounces) of gold is estimated to be recovered by reprocessing tailings on site. The mine is located on federal lands managed by the Bureau of Land Management. The mine operates under authorizations from the Bureau of Land Management, Alaska Departments of Natural Resources (ADNR) and Alaska Department of Environmental Conservation (ADEC). Below is the chronology of events described by the mine operator that lead to the overtopping of the tailings impoundment in January and February of 2012, based on a March 15, 2012, memo to Alaska State Mine Safety Engineer from Mystery Creek Resources, Inc.

- Prior to October 25, 2011, the mine staff monitored the freeboard in the tailings impoundment per requirements of agency authorizations.
- After October 25, 2011, the staff decided to waive gage observation until spring melt because the gage was frozen in ice.
- During a mid-January trip to the site, the Mystery Creek Resources, Inc., President noticed insufficient freeboard in the tailings pond. He notified Bureau of Land Management, ADNR, and ADEC.
- Corrective action was taken and the pond level began to drop.
- In late February 2012, mill operations that had been completed in batches were switched to continuous operation without recognizing the implications for water balance (more water would be flowing to the tailings impoundment).
- On March 9, 2012, mine personnel noticed evidence of overtopping of the dam. Bureau of Land Management, ADNR, and ADEC were notified and action was taken to draw down the pond and stop the overtopping.
- On March 10, 2012, agency inspections began. It was found that water from the tailings impoundment was not likely to have reached nearby streams. An estimated 32,400 gallons of tailings water were discharged from the impoundment.

On inspection of the dam it was found that the engineered spillway for the dam had been frozen over by a previously undiscovered tailings water release. The ice prevented the spillway from operating as designed so that the later spill overtopped the dam at another location not designed for overflow.

The composition of the aqueous phase is uncertain. None of the tests performed by PLP represents the leaching conditions in a tailings impoundment, and no model exists to mathematically simulate the leaching process. The aqueous phase may be represented by some mixture of tailings supernatant, which represents the source water for the impoundment (Table 6-2); humidity cell leachate, which represents aqueous leaching from tailings under oxidizing conditions (Table 6-3); and local water (Table 5-15).

Tailings impoundment surface waters would consist of water used to transport the tailings (supernatant) and any other waters stored in the impoundments prior to reuse or treatment and discharge. Hence, the surface water is expected to resemble the PLP's test supernatant (Table 6-2) with some dilution by precipitation. However, those results do not include process chemicals (other than unspecified thiosalts) that would be associated with the supernatant and are not included in this assessment. Supernatant water would be slightly diluted by rain and snow onto the surface of the impoundment, but peripheral berms should prevent dilution by runoff except during flood events.

The waters released from a tailings spill could consist of surface water, surficial pore water, and a much larger volume of deep pore water. The surficial tailings pore water would be generated by leaching tailings in the presence of some oxygen. The composition and concentrations of constituents in that water may be roughly similar to a mixture of those observed in the supernatant and humidity cell tests (Tables 6-2 and 6-3). Pore water from deeper anoxic tailings would have begun primarily as supernatant, but may have lower metal content due to chemical precipitation under anoxic conditions. Leachate flowing from an abandoned and failed impoundment would be more oxidized because the cover water and much of the pore water would have drained away.

Table 6-2. Aquatic Toxicological Screening of Tailings Supernatant against Acute Water Quality Criteria (CMC) and Chronic Water Quality Criteria (CCC). Values are µg/L unless otherwise indicated. Average leachate values are from Appendix H.

Analyte	Average Value	CMC	CCC	CMC Quotients	CCC Quotients
pH (S.U.)	7.9				
Alkalinity (mg/L $CaCO_3$)	74.8				
Hardness (mg/L $CaCO_3$)	322.8				
SO_4	318,708				
Ag	0.000018	24		0.0007	
Al	71.8	750	87	0.096	0.8249
As	17.2	340	150	0.051	0.1146
Ca	116004				
Cd	<0.1	6.3	0.55	<0.012	<0.1415
Co	<0.1				
Cr	<1.0	1500	190	<0.0007	<0.0051
Cu	7.8	40[a]	24[a]	0.19[a]	0.3179[a]
Cu	7.8	7.2[b]	4.4[b]	1.1[b]	1.8[b]
Fe	16.8				
Hg	0.0	1.4	0.77	<0.027	<0.0485
K	25951				
Mg	8001				
Mn	71.9				
Mo	69.7				
Na	43781				
Ni	<0.8	1300	140	<0.0006	<0.0056
Pb	0.2	220	8.8	0.0010	0.0261
Sb	6.0				
Se	7.6		5		1.5
Tl	0.0				
Zn	4.3	316	316	0.014	0.014
Sum of metals				0.31[a] : 1.7[b]	2.3[a] : 3.8[b]

[a] From Alaska's hardness-based standard.
[b] From the national water quality criteria based on the biotic ligand model (BLM)
CMC = criterion maximum concentration; CCC = criterion continuous concentration

Table 6-3. Aquatic Toxicological Screening of Tailings Humidity Cell Leachates against Acute Water Quality Criteria (CMC) and Chronic Water Quality Criteria (CCC).

Analyte	Average Value	CMC	CCC	CMC Quotients	CCC Quotients
pH (S.U.)	7.8		6.5-9		
Alkalinity (mg/L $CaCO_3$)	59.7				
Hardness (mg/L $CaCO_3$)	66.8				
Cl	515.5				
F	450.9				
SO_4	17448				
Ag	0.01	1.6		0.0062	
Al	23.64	750	87	0.031	0.27
As	5.46	340	150	0.016	0.036
B	10.67				
Ba	9.25				
Be	0.20				
Bi	0.49				
Ca	22551				
Cd	0.05	1.4	0.19	0.038	0.28
Co	0.19				
Cr	0.50	410	53	0.0012	0.0094
Cu	5.33	9.2[a]	6.4[a]	0.58[a]	0.84[a]
Cu	5.33	4.8[b]	3.0[b]	1.1[b]	1.8[b]
Fe	29.66				
Hg	0.01	1.4	0.77	0.0071	0.013
K	4015				
Mg	2547				
Mn	44.15				
Mo	33.46				
Na	2099				
Ni	0.54	330	37	0.0016	0.014
Pb	0.06	41	1.6	0.0015	0.039
Sb	1.80				
Se	1.48		5		0.30
Sn	2.93				
Tl	0.05				
V	0.78				
Zn	3.16	83	83	0.038	0.038
Sum of metals				0.72[a] : 1.2[b]	1.8[a] : 2.8[b]

Notes:
Values are presented in micrograms per liter (μg/L) unless indicated otherwise. Average leachate values are from Appendix H.
[a] From Alaska's hardness-based standard.
[b] From the national water quality criteria based on the biotic ligand model (BLM)
CMC = criterion maximum concentration; CCC = criterion continuous concentration

Aqueous Exposures from Deposited Tailings

After a tailings dam failure, aquatic biota would be exposed to potentially toxic tailings that covered the substrate of streams or rivers. Thus, benthic organisms would be the most exposed. These organisms would include aquatic insects and other invertebrates that burrow into the substrate or crawl upon its surface. In addition, eggs and larvae (fry) of any salmon, trout, or char that spawned in the contaminated substrate would be exposed. In either case, the bioavailable contaminants are those that are dissolved in the pore water of the deposited tailings. Hence, exposure is determined by the rate of leaching of the tailings and the rate of dilution of the leachate, which depend on hydrological conditions. Unlike the lakes and estuaries that are the usual sites of sediment pollution studies, streams have a high level of interaction between substrates and surface water. Shallow, turbulent water is typically near oxygen saturation. Bedload sediment bounces and slides downstream during high flows, and during higher flows sediment is suspended, exposing it to oxygen. In addition, water flows longitudinally and laterally through bed and floodplain sediments and vertically between groundwater and surface water.

Because the biologically active zone is oxidized, the tailings leachate to which biota would be exposed could resemble leachates from the supernatants and humidity cells. Ideally, a leaching test would be performed that simulated conditions in a streambed, but no such test results are available. In theory, the leachate composition could be estimated using a mechanistic model, but no such model is available. Dilution of the leachate would be minimal in low-flow areas such as pools and backwaters and during low-flow periods. Dilution would be greatest in high-flow and turbulent locations such as riffles, in groundwater up-wellings or down-wellings, and during high-flow periods such as spring runoff and floods. However, high flows would be expected to increase leaching rates.

Although we assume that spilled tailings would be mixed and would have average metal compositions (Table 6-4), stream processes would be expected to sort them. In Soda Butte Creek (Box 6-1), copper concentrations in riffles and glides gradually decreased downstream from the tailings spill site. However, fine sediments in pools had higher copper concentrations than the high energy segments, and some of the highest copper concentrations were found in fine pool sediments more than 10 km downstream (Nimmo et al. 1998).

Table 6-4. Comparison of Mean Metal Concentrations of Tailings (Appendix H) to Threshold Effect Concentration (TEC) and Probable Effect Concentration (PEC) Values for Fresh Water and Sums of the Quotients ($\sum$ TU)

Tailings Constituents	Mean	TEC[a]	TEC Quotient	PEC[a]	PEC Quotient
Ag	0.7				
As	25.2	9.8	2.6	33	0.76
Ba	30.0				
Be	0.3				
Bi	0.6				
Cd	0.1	0.99	0.10	5.0	0.020
Co	8.1				
Cr	149.9	43	3.5	110	1.3
Cu	682.9	32	21	150	4.5
Hg	0.1	0.18	0.56	1.1	0.091
Mn	359.9	630	0.57	1200	0.30
Mo	51.9				
Ni	67.7	23	2.9	49	1.4
Pb	15.0	36	0.41	130	0.12
Sb	1.0				
Se	1.8				
Tl	0.3				
U	0.4				
V	87.3				
Zn	87.4	120	0.72	459	0.19
Sum			32		8.7

Notes:

[a] TECs and PECs are consensus values from (MacDonald et al. 2000), except for Mn which are the TEL and PEL for *Hyalella azteca* 28 d tests from (Ingersoll et al. 1996).

TEC = threshold effect concentration; PEC = probable effect concentration; TEL = threshold effect level; PEL = probable effect level

All concentrations are mg/kg dry weight.

After the spill, aquatic biota would also be indirectly exposed to tailings deposited on land, primarily in the floodplains. Erosion of these tailings would result in deposition in streams, potentially replacing tailings lost through streambed erosion (Marcus et al. 2001). In addition, rain and snowmelt would run across and percolate through tailings deposited on floodplains, leaching metals and carrying them into the stream. Leachate would also form during lateral groundwater movement through tailings, particularly where tailings deposited in wetlands. Floodplain-deposited tailings are leached in the presence of oxygen with episodes of saturation and drainage (ARCO 1998). Hence, humidity cell leachates would be more relevant to this exposure route than to others, and leachate concentrations in Table 6-3 may roughly estimate leachate composition from floodplain-deposited tailings. This leachate could have three fates: it could move upward during dry periods and deposit on the surface as soluble salts (e.g., hydrated metal sulfates); it could move down into buried soils and deposit as weak acid-extractable compounds (e.g., metal sulfides); it could sorb to organic matter or move laterally to the surface channel as dissolved metal ions (Nimik and Moore 1991, ARCO 1998). Runoff from tailings-contaminated floodplains of the Clark Fork River had high copper levels (67.8-8,380 μg/L) (Nimik and Moore 1991, ARCO 1998). Concentrations in Bristol Bay would probably be lower than for the acidic Clark Fork tailings and salt accumulation on the surface would be less as a result of greater precipitation, but the same processes would occur. Dilution of the leachate that moves into the stream would be highly location- and condition-specific. Once in a stream, leached metals could remain dissolved or precipitate or be sorbed to clays or organic matter, depending on the conditions.

Remobilization of deposited tailings during high flows could result in acute exposures to suspended tailings and extend the downstream range of exposure to deposited tailings. In the Coeur d'Alene River, floods occurring in 1995, 1996, and 1997, more than 30 years after the last release of tailings, carried metal-enriched sediment from both the floodplain and streambed more than 210 km (130 miles) downstream (the furthest extent of the study) (USGS 2005).

Less dramatic increases in flow would cause bedload transport (movement of sediment without suspension in the water column), which could release sediment pore water (leachate) into the water column. First, copper could leach from the tailings and accumulate in sediment pore water during low-flow periods. Then when flows increase sufficiently to mobilize the sediment, pore water would mix with surface water, resulting in exposure of aquatic biota and downstream copper transport. Studies in the tailings-contaminated Clark Fork River found that copper concentrations in interstitial water were 3-36 μg/L in depositional areas and 3-22 μg/ L in riffles (ARCO 1998). Concentrations would differ with Bristol Bay tailings, but this result demonstrates that deposited tailings can have significant interstitial water concentrations, even in a hydrologically active stream where leaching has proceeded for decades. If sediment movement was sufficient to mobilize deep anoxic sediments, precipitated or complexed metals may be mobilized and, depending on local water chemistry, dissolved.

Solid Phase Exposure to Deposited Tailings

Although the most bioavailable metals in sediment are those dissolved in pore water, it is useful to consider the whole sediment as a source of exposure. This approach avoids uncertainties associated with using leaching tests to represent field processes. It is reasonable to consider the average tailings composition to represent stream sediment to which biota downstream of a spill would be exposed (Table 6-4). During and after the spill, there may be some sorting of the tailings by size or density that would result in locally higher metal compositions, but that cannot be predicted. While the material in the failed dam would dilute the tailings initially, particles in the dam would be larger than the tailings and would settle out in the first few kilometers downstream (Section 4.4.2). Some soil would be scoured from the receiving stream, but that would be associated with the first wave of the slurry. Hence, given the volume spilled, the tailings in most of the initial depositional area would be effectively undiluted. After the spill, the tailings sediment would be diluted by clean sediment from tributaries, but that process would be slow because the watershed is nearly undisturbed except for potential mine facilities, and the volume of tailings deposited in the watershed is so large. The background sediment load (1.4 to 2.5 mg/L total suspended solids; Table 5-15) is miniscule compared to the multiple meters of tailings that would be deposited (Table 4-14). The washing of tailings from floodplains into streams and rivers would be more important for many years, so the sediments in streams and rivers below a tailings spill would resemble average tailings.

Dietary Exposures

As discussed in Section 5.3.2.2, dietary exposures of fish to metals have been an issue of concern at mine sites. An adjustment factor for rainbow trout to account for a dietary component to aqueous exposures (0.95) is presented there. It may be applied to cases, such as flow into a stream through floodplain tailings or from upwelling through tailings, in which both direct aqueous and dietary exposures may occur. Dietary exposures with respect to sediment levels may also be estimated. In such cases, the direct aqueous exposures of fish may be negligible, but invertebrates, particularly metal-tolerant insects such as chironomids, may accumulate metals, carry them out of the sediment, and then serve as sources of dietary exposure. This phenomenon has been documented in both the Clark Fork and Coeur d'Alene River basins (Kemble et al. 1994, Farag et al. 1999).

A review of metal bioaccumulation by freshwater invertebrates (mostly Ephemeroptera and Diptera) derived models for two relevant feeding guilds:

$$\text{Collector/Gatherer Cu} = 0.294\,x$$

$$\text{Scraper/Grazer Cu} = 1.73\,x$$

where x is sediment concentration and Cu is tissue concentration, both in µg/g dry weight (Goodyear and McNeill 1999). Studies of the Soda Butte Creek tailings spill found that copper concentrations in mixed invertebrates were slightly lower than sediment concentrations (Marcus et al. 2001). Studies of the Clark Fork River give bioaccumulation factors for copper and river invertebrates ranging from 0.18 to 1.62, with factors generally rising as sediment concentrations declined (calculated from (Brumbaugh et al. 1994, Ingersoll et al. 1994). Equivalent studies in the Coeur d'Alene River give very similar factors (0.15 to 1.77) (calculated from (Farag et al. 1998). These results support the use of the average bioaccumulation factor of 1.0 (Goodyear and McNeill 1999). This implies copper concentrations in invertebrates equal to those in sediments, which in this case are tailings with an average copper concentration of 683 mg/kg (Table 6-4).

Another method used to estimate the bioaccumulation and toxicity of divalent metals in sediment is the acid volatile sulfides (AVS) / simultaneously extracted metals (SEM) approach (Ankley et al. 1996). However, it requires measurements of SEM and AVS with the sediment of concern. The source of copper in the tailings is sulfide ores, so one might assume that there is adequate sulfide for the copper, but experience with tailings spills refutes that assumption. The availability process of concern is oxidation of the sulfides, not binding of added copper by sediment sulfides. Studies in the Clark Fork River found that, contrary to expectations of that model, invertebrates accumulated metals at locations with AVS greater than SEM (Ingersoll et al. 1994). This discrepancy may be due to spatial variability, the high oxidizing conditions in riffles where most invertebrates are found, and the fact that much of the metals in these sediments are in a form (metal sulfide particles of the tailings) that is very different from the lake and estuary sediments for which the model was developed. Hence, for practical and empirical reasons, the AVS/SEM model is not appropriate to estimate bioaccumulation or toxicity in this system.

Persistence of Exposures

Evidence that tailings persist in streams as sources of metals exposures is provided by prior tailings releases. A review by (Miller 1997) found persistence of high metal content sediment in streams after 10 to 100 years. One well-documented case is provided by a tailings dam failure in Soda Butte Creek, Montana, in 1950 (Box 6-1) (Marcus et al. 2001). Sediment was still characterized by high copper concentrations after 48 years despite two 100-year floods, indicating that tailings are retained by streams and maintain high metal levels after decades of leaching. Similarly, the Coeur d'Alene River basin was contaminated by direct discharge of tailings to floodplains, tailings dam failures, and mine drainage, causing extensive damage to the watershed (Box 6-1) (NRC 2005). Treatment of the mine drainage improved biotic communities, but they were still impaired, apparently as a result of metals leaching from deposited tailings which entered the river until 1968 (Hoiland et al. 1994, NRC 2005). At least as late as 2000, metals (cadmium, lead, and zinc) concentrations were elevated in caddisflies and were more highly correlated with sediment concentrations than with surface water concentrations, suggesting that the deposited tailings were the primary source of exposure (Maret et al. 2003).

For the Clark Fork River (Box 6-1), a new study has modeled future decline in sediment metals concentrations (Moore and Langner 2012), assuming an exponential decay in concentrations over time due to loss and dilution. Although there was no significant change over time (1991 to 2009) in downstream concentration declines (which one would expect as tailings wash downstream), concentrations did decline over time at three individual sites. Based on regression for each of those sites, Moore and Langer (2012) estimated that average copper concentrations would decline below the probable effect concentration (PEC) in less than 85 years. At the most contaminated of the three sites, copper is predicted to reach the threshold effect concentration (TEC) in 163 years. In Bristol Bay, dilution with clean sediment would likely be slowed by denser vegetation and less land disturbance. The lower gradients in Bristol Bay relative to Montana would also tend to slow recovery, as recovery is primarily achieved by tailings transport downstream. It should also be noted that these time estimates are not from the date of a spill, but rather from a date decades later when channel structure stabilized and much of the tailings had been carried downstream.

6.1.4.2 Exposure-Response

Exposure-Response for Aqueous Chemicals

The toxic effects of exposure to a tailings spill can be estimated from aquatic toxicity data. Ambient water quality criteria are used to screen the metals in the two types of tailings leachates (Tables 6-2 and 6-3). Copper is the dominant contaminant in tailings leachates, and criteria rainbow trout median lethal concentration values based on the biotic ligand model (BLM) are used as benchmarks (Table 6-5). Acutely lethal levels for rainbow trout exposed to the humidity cell leachate and supernatant are estimated to be 93 and 188 µg/L respectively, based on the BLM.

Table 6-5. Results of Applying the Biotic Ligand Model to Mean Water Chemistries in Tailings Leachates and Supernatants to Derive Effluent-Specific Copper Criteria.

Stream	Acute Cu Criterion (CMC in μg/L)	Chronic Cu Criterion (CCC in μg/L)
Tailings humidity cell leachates	4.8	3.0
Tailings supernatants	7.16	4.45

CMC = criterion maximum concentration; CCC = criterion continuous concentration
Source: USEPA 2007

Note that these criteria are calculated for the water chemistry of the supernatant and leachate. This is clearly appropriate for the acute exposures immediately following a tailings dam failure, when the slurry volume would greatly exceed natural flows. It would also be appropriate for situations like sediment pore water, where dilution is minimal. However, for situations in which significant dilution occurs, these criteria would be too high because uncontaminated water has lower hardness and lower levels of chemicals that reduce copper binding to biotic ligands. Hence, dilution by a factor of two would not quite reduce toxicity of metal-contaminated water by a factor of two.

Exposure-Response for Sediment Chemicals

The toxicity of settled tailings may also be estimated from tailings metal concentrations. Various approaches have been employed to derive sediment quality guidelines, but the most common are the threshold effect level (TEL) and the probable effect level (PEL). TELs and PELs have been used in assessments of sites contaminated by mine wastes (USEPA 2001, USGS 2004, 2007). These levels are derived from distributions of sediment concentrations that do or do not exhibit apparent toxicity in laboratory or field studies. (MacDonald et al. 2000) performed a meta-analysis of published values, proposed consensus threshold effect concentrations (TECs) and probable effect concentrations (PECs), and then tested them using additional sediment studies. One of the sites in the test data set was the tailings-contaminated Clark Fork River. For copper, that validation study found toxic effects in 17.7% of sediments with concentrations less than the TEC, in 64% of sediments with concentrations between the TEC and PEC, and in 91.8% of sediments with concentrations above the PEC, out of 347 total sediments from 17 rivers and lakes (MacDonald et al. 2000). The consensus TECs and PECs are used to evaluate tailings as potential sediment, because they are the best supported values.

Exposure-Response for Dietary Chemicals

Effects may also be estimated from dietary exposures. If the primary source of exposure is dissolved copper in the water column (e.g., if significant upstream and floodplain leaching occurs), then the 0.95 adjustment factor (Section 5.3.2.2) is applicable. However, if sediment is the primary source of exposure, a dietary value is needed for consumption of benthic invertebrates. A dietary chronic value for rainbow trout derived from multiple studies is 646 μg/g (micrograms of copper per gram of dry diet) (Borgmann et al. 2005), at which survival and growth are observed to decline in multiple studies.

Exposure-Response for Analogous Sites

The effects of exposure to leachate from tailings can also be estimated from effects at analogous sites. In the Clark Fork River, Coeur d'Alene River, and Soda Butte Creek, both field effects on fish and invertebrates and toxicity have been associated with deposited tailings. However, the magnitude and nature of effects are so site-specific that quantitative empirical exposure-response models from these sites would not be reasonably applied to the tailings dam failure analyzed here. Nevertheless, the qualitative relationships are applicable.

6.1.4.3 Risk Characterization

Characterization of Acute Toxic Risks

At sites closest to the failed TSF, acutely toxic effects of a tailings spill would, in practice, be indistinguishable from the concurrent effects of being smothered by tailings particles. Aquatic life within the range of the tailings slurry would be devastated by its physical effects. Dissolved components of the spill would continue to flow to Bristol Bay, beyond the extent of significant particle deposition. Undiluted leachates of both types would be expected to exceed the acute national criterion for copper, which suggests that they would kill invertebrates (Tables 6-2 and 6-3). However, even the minimal dilution by a factor of 0.73 in the Nushagak River at Ekwok would dilute leachate from the maximum spill to below the national criterion. Even copper in undiluted tailings leachates (5.3 and 7.8 μg/L for the humidity cell and supernatant, respectively) would be well below levels required to kill post-larval salmonids in an acute exposure (93 and 188 μg/L for the humidity cell and supernatant, respectively). Hence, in the tailings dam failures, acute exposure to dissolved copper in the near-field would be sufficient to kill sensitive invertebrates but not salmonids, but those effects would be eclipsed by the physical effects. Far downstream, where physical effects would be minimal, toxic effects would not be expected due to dilution.

Characterization of Chronic Toxic Risks for Aqueous Exposure

Risks from chemicals leaching from tailings in streambed and riverbed sediments and associated floodplains are addressed by screening leachate concentrations against chronic water quality criteria and standards. Hazard quotients (Tables 6-2 and 6-3) can be interpreted as relative degrees of toxicity of leachate constituents or as an indication of the degree of dilution required to avoid significant toxic effects. The two estimates of tailings leachate composition give similar results (Tables 6-2 and 6-3). Undiluted leachate of both types would be expected to exceed the chronic national criterion for copper but not the Alaskan standard. If combined toxic effects of metals are considered (see the *Sum of Metals* line in Tables 6-2 and 6-3), chronic toxicity would be expected with both the hardness-based and BLM-based copper criteria, and acute lethality would be expected with the BLM-based copper criterion. However, direct aqueous exposures of fish to copper are unlikely to be toxic unless concentrations in the actual field leachates are much higher than the tailings test leachate concentrations.

The quotients with respect to chronic criteria (criteria continuous concentrations [CCCs]) imply that dilution by a factor of two to four would be sufficient to render leachate nontoxic. Low dilutions would be expected in the years immediately after a spill, when flows would pass through large volumes of tailings. After tailings have eroded and a more normal channel and floodplain are established, low dilution of tailings could occur in sediments during normal flows and in locations where water contaminated by floodplain tailings feeds a stream. In those situations, sensitive invertebrates could be reduced or eliminated.

Characterization of Chronic Toxic Risks from Sediment Chemicals

Sediment quality guidelines provide another line of evidence to assess risks from tailings after a tailings dam failure. Table 6-4 shows that tailings would be expected to cause severe toxic effects on the organisms that live in or on them. Notably, the copper concentration is 4.5 times the PEC; chromium and nickel concentrations would also exceed their PECs. The sum of TEC quotients of 32 implies that tailings would need to be diluted by 32 parts clean sediment to one part tailings before toxic effects would be unlikely (below the TEC). Because the Bristol Bay watershed is relatively undisturbed, background levels of total suspended solids are low (Table 5-15), so the time required to achieve that degree of dilution would be very long.

Characterization of Chronic Toxic Risks from Dietary Chemicals

The most relevant estimate of fish dietary exposure to tailings is provided by bioaccumulation factors with respect to sediment. The best estimate bioaccumulation factor of 1 implies copper concentrations in invertebrates of 683 mg/kg (Section 6.1.4.2). Dividing this concentration by a consensus dietary chronic value for rainbow trout of 646 µg/g (micrograms of copper per gram of dry diet) (Borgmann et al. 2005) results in a quotient of 1.1. This implies that the undiluted tailings would produce toxic prey for fish. As discussed above, dilution of the tailings with clean sediment is likely to be a slow process. Benthic invertebrates are a major component of the diet of salmon and Dolly Varden that rear in streams and rivers.

Characterization of Chronic Toxic Risks—Analogous Sites

Some well-documented cases indicate that adverse effects of chronic toxicity on aquatic communities in general, and salmonids in particular, can occur in streams and rivers that receive tailings spills. These cases have shown that effects continue indefinitely, but that the nature and magnitude of those effects vary among sites. In every case that we found in the literature in which the ecological consequences of a major spill of metal ore tailings to a stream or river was studied, extensive and long-lasting toxic effects were observed.

The most relevant case appears to be Soda Butte Creek in Montana, where a tailings spill from a porphyry gold and copper mine occurred in 1950 (Box 6.1). In the Soda Butte Creek case, the copper content of macroinvertebrates was positively correlated with sediment copper ($r^2 = 0.80$) and their taxa richness was inversely correlated ($r^2 = 0.48$) (Marcus et al. 2001). Although copper concentrations generally decreased downstream, sediments and sediment pore waters were toxic to the amphipod *Hyalella azteca* for the full 28-km length of the study area (Nimmo et al. 1998). Macroinvertebrate community effects persisted for at least 40 years after the spill. These effects were attributed to sediment toxicity (Nimmo et al. 1998), but habitat effects of deposited tailings also may have contributed. Although they were less well studied, it was clear that trout were also affected. Only two trout were found in the 300-m reach downstream of the spill site in 1993, although prior to mining, Soda Butte Creek was known for "fast fishing and large trout" (Nimmo et al. 1998).

In the Coeur d'Alene River and its tributaries, elevated metals concentrations and effects on both benthic invertebrates and fish persisted more than 30 years after tailings releases ended and after treatment of mine drainage. Some fish species were absent; others were reduced in abundance and experienced toxic effects from both aqueous and dietary exposures (Farag et al. 1999, Maret and Maccoy 2002, Maret et al. 2003). Returning Chinook salmon avoided the more contaminated South Fork in favor of the North Fork (Goldstein et al. 1999). Macroinvertebrate communities and taxa were also impaired (Hoiland et al. 1994, Maret et al. 2003).

In the Clark Fork River, a sediment quality triad approach demonstrated that tailings-containing sediments had high metal levels, were toxic to the amphipod *Hyalella azteca*, and shifted the macroinvertebrate community to generally metal-tolerant Oligochaeta (worms) and Chironomidae (midges) (Canfield et al. 1994). Rainbow and brown trout abundances were low in contaminated reaches of the Clark Fork, fish kills occurred apparently due to metals washing from floodplain tailings deposits, and metals in invertebrates were sufficient to cause toxic effects in laboratory tests of trout (Kemble et al. 1994, Pascoe et al. 1994, ARCO 1998).

6.1.4.4 Uncertainties

All of the lines of evidence concerning risks to aquatic communities from the toxic properties of spilled tailings have notable uncertainties.

Toxic Risks from Aqueous Exposures

The use of leachate and supernatant concentrations to estimate risks from a tailings spill is uncertain primarily because of uncertainty concerning test relevance to leaching in the field. Leaching of tailings in the impoundment, streambeds, and floodplains would occur under very different conditions than in humidity cell tests. In addition, it is possible that tailings could become more acidic over time as their acid neutralizing capacity is consumed or as acid neutralizing chemicals are dissolved, resulting in increased metal concentrations. Test leachates are available for the bulk tailings but not pyritic tailings. The assessment assumes that the content of the tailings impoundment is tailings, but acid-generating rock may also be deposited there. Finally, the degree of leachate dilution in the field would be highly variable and could be roughly estimated, at best.

The exposure-response relationships for this line of evidence are also uncertain. As noted above (Section 5.3.2.2), the water quality criteria and standards used in this assessment may not be protective of all macroinvertebrate taxa that are important prey for fish. However, direct aqueous exposures of fish to copper are unlikely to be toxic unless the field concentrations are much higher than test leachate and supernatant concentrations.

Toxic Risks from Sediment

Although the consensus TECs and PECs are the best available effects benchmarks for sediment, their applicability to tailings in streams and rivers of Bristol Bay is uncertain. The studies from which the values are derived include lakes, reservoirs, and other systems that differ ecologically from the rivers and streams in the Bristol Bay watershed. However, the Clark Fork River (a tailings-contaminated salmonid stream) was one of the confirmation sites for the TECs and PECs, which suggests that they are relevant to this type of situation.

Because the TECs and PECs are geometric means of prior sediment guidelines, the range of guidelines provides an estimate of uncertainty. Alternate threshold values range from 16 to 70 mg/kg and probable effect values range from 86 to 390 mg/kg (MacDonald et al. 2000). The average copper concentration of tailings (683 mg/kg) is well above all of these values, so this uncertainty is immaterial.

Some evidence suggests that these sediment guidelines may not be fully protective. When quotients of sediment concentrations/TELs (one of the sources of the TECs and a numerically similar value) were summed to address the combined toxicity of cadmium, copper, lead, and zinc, that value was not a threshold for effects on stream invertebrates in the Colorado mining belt (Griffith et al. 2004), and reductions in four different community metrics occurred below the sum of TEL values. However, this result may be confounded by mine drainage.

Dietary Risks

Dietary risks depend on the tailings composition, the copper bioaccumulation factor for aquatic invertebrates, and the chronic toxic threshold for dietary exposures of rainbow trout. Tailings composition may differ in practice, but that uncertainty is unknown. Ecological uncertainties are likely to be larger. Bioaccumulation factors for invertebrates and sediment range from 0.15 to 1.77, even in a single river (above), which translates to invertebrate body burdens of 102 to 1,210 μg/g. That range encompasses the seven available estimates for the copper toxic dietary threshold in rainbow trout, which ranges from 458 to 895 μg/g (Borgmann et al. 2005). This range of bioaccumulation factors is not surprising given the differences in feeding habits, morphology, and physiology among invertebrates.

Analogous Sites

The analogous sites for a potential tailings spill are all salmonid streams or rivers that received large deposits of tailings from metal mines and that were well studied over an extended period (Box 6-1). A large source of uncertainty when evaluating the effects at those sites relative to the current situation is the composition of the tailings. The Pebble test tailings are, in general, less acidic and contain less copper. On that basis, the nature and magnitude of effects are likely to be less. However, the setting is different in ways that might increase effects. For example, low hardness and low levels of dissolved materials in the Koktuli receiving waters would make biota of the receiving streams more susceptible to metals than in the analogous sites. However, these cases can be used with confidence to identify or confirm important modes of exposure and the processes leading to exposure. They also confidently demonstrate the persistence of tailings and the leaching of their metals for multiple decades.

6.1.5 Weighing Lines of Evidence

This risk characterization is based on weighing multiple lines of evidence, and evidence for the various routes of exposure is complex, as summarized in Table 6-6. For each route, sources of the exposure estimate and the exposure-response relationship are indicated. All evidence is qualitatively weighed based on three attributes: its logical implication, its strength, and its quality (Suter and Cormier 2011). In this case, the logical implication is the same for all lines of evidence: they all suggest that a spill from a tailings dam failure would have adverse effects. The strength of the evidence is based primarily on the magnitudes of the hazard quotients (exposure concentrations divided by effects concentrations): 0 signifies a low quotient, + a moderate quotient and ++ a high quotient. In this case there are no moderate quotients. Quality is a more complex concept. It includes conventional data quality issues, but in this case the primary determinate is the relevance of the evidence to the mine scenario. Because this is a predictive assessment, none of the evidence is based on observations of an actual spill. Hence, the evidence is based on assumptions about the spill, laboratory studies, or field studies at other sites where tailings have spilled into streams or rivers or where biota were exposed to other sediments with high copper levels. Separate quality scores are provided for the exposure estimate and for the exposure-response relationship. The scores indicated in Table 6-6 are not a substitute for the actual evidence, but rather are intended to remind the reader what evidence is available and show the pattern of strength and quality of the several lines of evidence that might not be apparent from reading the text.

Table 6-6. Summary of Evidence Concerning Risks to Fish from a Tailings Dam Failure. The risk characterization is based on weighing multiple lines of evidence for different routes of exposure. All evidence is qualitatively weighed (using +, 0, -) on three attributes: logical implication, strength, and quality. Here, all lines of evidence have the same logical implication—that is, all suggest a tailings dam failure would have adverse effects. Strength refers to the overall strength of the line of evidence, and quality refers to the quality of the evidence sources in terms of data quality and relevance of evidence to mine scenario. See Section 6.1.5 for more detailed discussion of weighing these lines of evidence.

Route of Exposure Source of Evidence (Exposure / E-R)	Logical Implication	Strength	Quality	
			Exposure	E-R
Suspended sediment Assumption/synthesis of laboratory and field studies	+	++	0	+
Acute aqueous exposure Leachate measurements/laboratory-based criteria	+	0	0	+
Chronic aqueous exposure Leachate measurements/laboratory-based criteria	+	0	0	+
Chronic sediment exposure Tailings measurements/sediment guidelines	+	++	+	+
Chronic dietary exposure Tailings measurements and BAFs/mean of laboratory-based effects levels	+	0	+	+
All routes in the field Exposure and effects at analogous sites	+	++	+	0

Notes:
E-R = Exposure-Response relationship
BAF = bioaccumulation factor

6.1.6 Risk Characterization Summary for a Tailings Spill

Following a tailings spill, fish in the receiving stream and the invertebrates on which they depend would be exposed to deposited tailings, suspended tailings, and tailings leachates. The fine texture of deposited tailings would make them unsuitable for salmonid spawning and development, and a poor substrate for the invertebrates that serve as food for developing salmon and resident trout and char. Suspended tailings would have lethal and sublethal physical effects on fish and invertebrates immediately following the spill, which are likely to continue with gradually diminishing intensity for years thereafter. The most toxic constituent of the leachate and tailings would be copper, and exposures would be both direct and through diet. Copper in leachate and in food is mildly toxic for fish, but copper and other constituents in the tailings themselves would be moderately toxic to benthic invertebrates and potentially toxic to fish eggs and larvae spawned in tailings-contaminated streams.

The physical and chemical effects of tailings on fish and invertebrates would be extensive in both space and time. Elevated levels of suspended tailings would last for years. Deposited tailings and their leachate would persist at toxic levels for decades. The acute effects of a tailings spill would extend far beyond the modeled distance, which resulted in modeled tailings deposition of 3 to 14 m approximately 30 km downstream (Section 4.4.2). Based on data from other sites, tailings deposition from a spill would extend for more than 100 km downstream, resulting in chronic exposures and effects (Section 4.4.2). From the confluence of the North Fork Koktuli and South Fork Koktuli Rivers, the mouth of the Koktuli River is 63.6 km; from there, the mouth of the Mulchatna River is another 66.5 km, and the mouth of the Nushagak River at Dillingham is another 170.5 km.

6.1.7 Risks from Remediation of a Tailings Spill

Although streams typically recover from aqueous effluents in less than a decade, the effects of tailings deposition in streams and floodplains persist for as long as they have been monitored at analogous sites. For that reason, tailings-contaminated streams, rivers and lakes in the United States have been or will be dredged, riprapped, or redirected under the federal Superfund or state cleanup programs. Although such remedial actions have net benefits, they create long-term impacts on aquatic habitats. For example, riprapping reduces downstream exposure to tailings and associated metals by reducing erosion of floodplain tailings, but it also reduces habitat complexity and quality for fish by channelizing the stream or river (Schmetterling et al. 2001).

Remediation in this case would be particularly difficult and damaging because the streams and their floodplains are pristine and because a road would need to be built into a roadless area to bring in equipment and to haul out the tailings. At the upper end of the affected area, the process of removing the tailings would do little additional damage because the structure of the watershed would have been destroyed. If the removal of tailings extended to streams that were not scoured in the initial release, the removal would destroy those streams and associated wetlands. If removal was not undertaken, the substrate of the streams would still consist of tailings until flood flows scoured them out.

6.2 Pipeline Failure

In this section, we assess accidents involving the pipelines for the product concentrate slurry and return water (Section 4.4.3.2). We do not assess failures of the natural gas or diesel pipelines here because such pipelines are common, their risks are well known, and they are not particularly associated with mining. Iliamna Lake is described as the receptor for spills, because the portion of the pipelines that is within the scope of this assessment is within the watershed of the lake.

6.2.1 Product Slurry Spill

No analyses of product concentrate slurry or its leachate are available from the Pebble deposit or any other ore body in the region of concern. Therefore, to estimate the concentration of metals and other constituents in the transport water we use analyses from the Atik (Sweden) porphyry copper mine (Table 6-7) as described in Appendix H.

The fine particles of product concentrate would, like spilled tailings (Section 6.1.2), degrade habitat quality for fish and benthic invertebrates. However, these effects would be much less than for a tailings dam failure because of the much lower volume, and would be minor compared to the potential toxic effects. Therefore, this assessment focuses on toxic effects rather than habitat effects.

6.2.1.1 Exposure

Pipelines carrying product concentrate slurry would be associated with the road to Cook Inlet (Section 4.3.9.2). The potential pipelines would have approximately 70 crossings of streams and rivers; 35 of these water bodies are believed to support salmonids and all could convey contaminants to Iliamna Lake. For 16% of their length (20.7 km), the pipelines would be within 100 m of a stream or river (Table 5-21), creating the potential for spilled slurry to flow into surface waters either directly or by overland flow. Downstream of those crossings lie 269 km of streams (Table 5-19) and Iliamna Lake. (Note that the number of crossings is much larger than the number of hydrologic units in Tables 5-19 through 22 because the hydrologic units may contain multiple watersheds and each watershed may have multiple crossings related to tributaries.)

For 23.4% of their length (27.6 km), the pipelines would be within 100 m of a designated wetland (Table 5-22), creating the potential for spilled slurry to flow into wetlands either directly or by overland flow. Some of these wetlands include ponds that support salmonids, but the number and extent of salmonids are unknown. Further, spilled slurry water and leachate from spilled concentrate in wetlands could flow to streams and Iliamna Lake.

A pipeline failure and spill would be expected to release 475 m^3 of product concentrate (Table 4-16). All or part of that mass could enter a stream, where it would form a sand-like sediment. Over time, it would be spread downstream by erosion, eventually entering Iliamna Lake where it could mix into sand and gravel beaches used by spawning sockeye salmon. This process cannot be quantified with existing data and modeling resources, but it would occur.

Table 6-7. Aquatic Toxicological Screening of Leachates From Atik (Sweden) Mine Copper Concentrate (Appendix H) based on Acute and Chronic Criteria (CMC/CCC) and Quotients of Concentrations Divided By CMC and CCC Values

Analyte	Concentrations	Criteria CCM/CCC	Quotients
pH (S.U.)	5.36	6.5-9	NA
Spec. conductivity (µS/cm)	264	-	-
Alkalinity (mg/L)	0	-	-
Sulfate (mg/L)	121	-	-
SiO_2 (mg/L)	58.8	-	-
Ag	<1	0.90/-[a]	<1/-
Al	844	750/87	1.1/9.7
As	<1	340/150	<0.0029/<0.0067
Ba	38.4	-	-
Ca	26,900	-	-
Cl	800	19/11	42/73
Cd	3.53	1.73/0.22[a]	2.0/16
Co	136	-	-
Cr	<1	500/65[a]	<0.002/<0.0067
Cu	8400	11.61/7.9[a] 0.046/0.028[b]	720/1100 180,000/290,000
F	1,600	-	-
Fe	210	-	-
K	3,980	-	-
Mg	4,450	-	-
Mn	644	-	-
Mo	< 2	-	-
Na	889	-	-
Ni	484	410/46[a]	1.2/10
Pb	10.6	54/2.1[a]	0.20/5.0
Sb	12.8	-	-
Se	7.3	-/5.0	-/1.5
U	10.5	-	-
Zn	1300	100/100[a]	13/13
Sum of metals			740[a]/1,200[a] 180,000[b]/290,000[b]

Notes:
CMC = criterion maximum concentration; CCC = criterion continuous concentration
Unless otherwise designated, units are µg/L.
[a] Based on hardness of 85.5 estimated from 2.5 Ca + 4.1 mg in mg/L applied to hardness based criteria and standards.
[b] From the national ambient water quality criterion for copper based on the biotic ligand model (BLM)

The estimated pipeline failure rate of one per 1,000 km per annum (Section 4.4.3.1) results in an estimated failure rate of 0.118 per annum. If the probability of a pipeline failure is independent of location, and if it is assumed that spills within 100 m of a stream could flow to that stream, a spill would have a 16% probability of entering a stream within the Kvichak watershed (Section 5.4). This would result in an estimate of 0.019 stream-contaminating spills per annum, or 1.5 stream-contaminating spills over the duration of the maximum mine size (approximately 78 years). Similarly, a spill would have a 23.4% probability of entering a wetland (Section 5.4) resulting in an estimate of 0.028 wetland-contaminating spills per annum or 2 wetland-contaminating spills over the pipeline lifetime. A proportion of those wetlands are ponds or backwaters that support fish. Depending on the stream, a slurry spill could contaminate 2.6 to 34 km of stream with leachate and product concentrate before entering Iliamna Lake. The potential extent of contamination of wetlands cannot be readily estimated.

Exposure to Aqueous Phase Chemical Constituents

As with a tailings spill, toxicologically relevant exposures could occur by multiple routes in the event of a product pipeline spill (these routes are described in more detail in Section 6.1.4.1**)**. During and immediately following the spill, organisms would be acutely exposed to leachate (i.e., the slurry water that has leached ions from the product concentrate) and suspended particles. After a spill, product concentrate deposited on the stream or lake bed would result in chronic aqueous exposures to pore water and acute aqueous exposures during re-suspension events. In each case, aqueous exposure is estimated from the leachate concentrations in Table 6-7. Unlike the tailings spill, which would inevitably enter a stream and its floodplain, the slurry spill might directly enter a stream, pond, or wetland; it might flow overland to a nearby water body; or it might flow across a terrestrial habitat without reaching water. Terrestrial slurry deposits are likely to be collected by the operator, so rain and snowmelt are unlikely to leach those deposits and contaminate streams. However, the spilled leachate from the pipeline slurry could enter a stream, wetland, pond, or lake by groundwater flow.

The spill would result in flows of 2,567 L/s (255 metric tons/hour) of product concentrate and 1,767 L/s of leachate for 2 minutes (Section 4.4.2). The potential receiving streams vary considerably in their flows. Measurements in streams along the road corridor in 2004 and 2005 yielded a maximum observed flow of 58,000 L/s in the Iliamna River and a lowest observed flow of 2.8 L/s in an unnamed stream (PLP 2011). Hence, full mixing of spilled leachate could result in as much as a 33-fold dilution, but in the smaller streams there would be effectively no dilution. Of 12 monitored streams on the corridor, only two had observed August 2004 flows (an estimate of summer low flow) greater than the leachate flow (Table 7.3-10 in PLP 2011).

Exposure to Solid Phase Chemical Constituents

If spilled product concentrate entered a stream, pond, or wetland directly or by overland flow or erosion, it would settle and become the substrate for invertebrates and possibly salmon eggs and fry. Product concentrate in a stream would wash into Iliamna Lake, where it could serve as substrate for spawning sockeye salmon. Metal concentrations in copper product concentrate are presented in Table 6-8. While the concentrate spilled into a stream would settle rapidly, forming an area with essentially undiluted concentrate as sediment, concentrations downstream and in Iliamna Lake would be diluted to an extent that could not be estimated.

Table 6-8. Comparison of Mean Metal Concentrations in Copper Concentrate from the Atik (Sweden) Porphyry Copper Mine (Appendix H) to Threshold Effect Concentration (TEC) and Probable Effect Concentration (PEC) Values for Fresh Water. All concentrations are mg/kg dry weight.

Concentrate Constituents	Concentrations	TEC[a]	TEC Quotient	PEC[a]	PEC Quotient
Ag	>10				
As	12	9.8	1.2	33	0.36
Ba	59				
Bi	44.9				
Cd	2.4	0.99	2.4	5.0	0.48
Co	53.9				
Cu	>10000	32	>310	150	>67
Ga	0.88				
In	2.35				
Mn	345	630	0.55	1200	0.29
Mo	1100				
Ni	72.1	23	3.1	49	1.5
Pb	64.9	36	1.8	130	0.50
Sb	43.4				
Te	4.1				
Th	1.5				
Tl	0.2				
U	2.2				
V	23				
Zn	2190	120	18	459	4.8
Sum of metals			>340		>75

TEC = threshold effect concentration; PEC = probable effect concentration (PEC); TEL = threshold effect level; PEL = probable effect level

[a] TECs and PECs are consensus values from (MacDonald et al. 2000) except for Mn, which are the TEL and PEL for *Hyalella azteca* 28 d tests from Ingersoll et al. 1996.

Dietary exposure is not considered because, as a result of toxicity, few if any invertebrates would be expected to live in sediment formed of spilled concentrate, even with considerable dilution by clean sediment.

6.2.1.2 Exposure-Response

Acute water quality criteria (criterion maximum concentrations [CMCs]) and CCCs are used as thresholds for aqueous toxicity. Consensus sediment quality guidelines are used as thresholds for sediment solids toxicity. These benchmark values are discussed in Section 6.1.4.2 and presented in Tables 6-7 and 6-8. The BLM generates extremely low acute and chronic water quality criteria values because of the extreme water chemistry of the leachate. However, the parameters are all within the calibration range of the model (alkalinity and dissolved organic carbon were set to minimum values because they were absent from the leachate, which slightly raises the criteria values) (HydroQual 2007).

6.2.1.3 Risk Characterization

Risk Characterization Based on the Mine Scenario

A pipeline failure and spill would be expected to release 366,000 L of leachate (Table 4-16). The leachate exceeds CMCs for six metals, including exceeding the copper acute criterion by a factor of more than 700. None of the rivers or streams along the transportation corridor could provide enough dilution to avoid exceeding the acute criterion. This spill would last 2 minutes, which may be sufficient to cause acute injury or lethality to invertebrates or fish in receiving streams, given the high concentrations of toxic constituents. However, it would be more likely to cause acute effects in backwaters and ponds, which would retain spilled water. Those habitats are important rearing areas for salmon (Appendix A).

Exposure to pore water in sediments consisting of spilled product concentrate would be chronic. The screening assessment performed here suggests that a pipeline failure and product slurry spill would cause severe toxic effects (Table 6-7). The 8,400 μg/L of dissolved copper in leachate would be sufficient to kill benthic invertebrates (those that live in the gravel or sediment) and fish eggs and larvae in pore water, or in epibenthic water (water just above the bottom) of a receiving stream or pond.

The estimated 475 m^3 release of product concentrate would form a toxic substrate in a receiving stream (Table 6-8). The concentrate itself exceeds the sediment PEC for copper by more than a factor of 67. Hence, based on experience with other high-copper sediments, the concentrate would be certain to cause toxic effects on benthic organisms, including invertebrates and fish eggs and larvae. Because copper is aversive to salmonids (Goldstein et al. 1999, Meyer and Adams 2010), it is possible that the chronic leaching of copper from deposited product concentrate would prevent returning salmon from using a contaminated stream or river.

Risk Characterization Based on Analogy

The 316-km, 175-mm-diameter product slurry pipeline for the Bajo de la Alumbrera porphyry copper-gold mine in Argentina provides an analogue for pipeline considered here. It was reported that a 6.5-magnitude earthquake on September 17, 2004, caused a break in the pipeline, releasing an unknown quantity of concentrate that caused the Villa Vil River to overflow for approximately 2 km (Clap 2004, Mining Watch Canada 2005). The operators reported that the 2004 spill was controlled in less than 2 hours and water for drinking and irrigation was not contaminated (Minera Alumbrera 2004). They do not mention an earthquake, do not explain why the automatic shutoff did not function, and attribute the failure to “an existing outer mark on the pipe” (Minera Alumbrera 2004). They reported other pipeline failures with concentrate slurry spills in 2006 and 2007 but not in 2005 or from 2007 to 2010 (Minera Alumbrera 2004, 2005, 2006, 2007, 2008, 2009, 2010). They reported that those releases were small due to automatic shutoff, the concentrate from those spills did not reach water, and “no hazard is involved in concentrate handling since it is a harmless product consisting of ground rock” (Minera Alumbrera 2006). They reported that the composition of the harmless ground rock includes 28% copper and 32% sulfur (Minera Alumbrera 2006). They subsequently built collection pits at pumping stations, monitored streams at pipeline crossings, and brought in water to the community of Amanao in part to mitigate effects of "potential pipeline failure" (Minera Alumbrera 2008, 2010). They stated that pipeline crossings of streams have no adverse effects on biodiversity, but they do not address the effects of or recovery from the 2004 spill (Minera Alumbrera 2010). Although the interval during which Minera Alumbrera has provided sustainability reports is too short to reliably estimate an annual failure probability, it is remarkable that, despite International Organization for Standardization (ISO) 14001 certification of the pipeline, it failed and released concentrate in 3 of 7 years.

Although the Alumbrera case does not provide good evidence concerning the ecological effects of a concentrate spill, it does support the plausibility of pipeline failures leading to tailings slurry flowing into a stream. Our estimated pipeline failure rate of one per 1,000 km per annum (Section 4.4.3) implies a failure rate of 0.32 per annum for this 316-km pipeline, which is similar to the observed rate from 2004 to 2010 at Alumbrera of 0.43. Further, the 2004 spill provides a case of an accident that was more severe than assumed in our hypothetical accident, in that the spill lasted less than 2 hours rather than 2 minutes. Hence, it suggests that concentrate pipeline failures are common at a modern copper mine and they can result in spills that are potentially more severe than our assumptions indicate.

Risk Characterization Summary

The experience with pipelines in general and with the Alumbrera copper concentrate pipeline suggests that pipeline failures and product spills would be likely in the maximum size of mine scenario. A spill of product concentrate slurry into a stream may kill fish and invertebrates immediately, but would certainly cause long-term local loss of fish and invertebrates. The settled concentrate would become sediment, which would be toxic to fish and invertebrates for many years until it washed into Iliamna Lake, where it could be toxic to the eggs and larvae of sockeye salmon until it was sufficiently mixed with or buried by clean sediment.

If the spill were remediated, some fraction of the concentrate (but none of the leachate) could be recovered by excavation and the extent of the chronic (but not acute) toxic effects would be diminished. The proportion of concentrate recovered would depend on the location, time of year, and diligence of the operator.

6.2.1.4 Uncertainties

The composition of the product concentrate and its leachate are uncertain because they are based on a surrogate material and because leaching test conditions are inevitably somewhat artificial. However, given that the material is inevitably high in copper and sulfur, it is implausible that it would be nontoxic to aquatic biota. Although the copper concentration in the product concentrate leachate is very high, effects of copper exposures of less than an hour are unknown. A 2-minute spill duration depends on successful operation of an automatic shutoff. The potential for a larger spill if automatic shutoff failed (e.g., if an earthquake damaged the pipeline and the shutoff system) is unknown. The frequency and location of spills is also uncertain, but experience with pipelines in general and the Alumbrera case in particular suggest that pipeline failures are likely.

Return Water Spill

A spill from failure of a return water pipeline could result in an acute aqueous phase exposure as discussed above for a product slurry spill (Section 6.2.1). Flow and composition of return water are expected to be the same as the product concentrate failure, but without the solid phase. Hence, based on the acute criteria, the concentration would be sufficient to kill aquatic organisms until dilution reached a factor of more than 700 (Table 6-7). Because of the short duration of the spill, effects are most likely in low-flow habitats such as backwaters and ponds. We know of no analogous return water pipeline failure. However, experience with pipelines in general suggests that multiple failures and spills would occur over the life of the mine, and at least one would be expected to occur at or near a stream (Section 6.2.1).

6.3 Water Collection and Treatment Failure

During mine operation, collection or treatment of leachate from mine tailings, pit walls or waste rock piles could fail in various ways. This water collection and treatment failure could be continuous (e.g., failure to collect all leachate from the tailings storage facility) or episodic (e.g., failure due to a power loss). In such cases, leachate might enter groundwater and not be collected by the pit sumps or the tailings impoundment's collection system, or could discharge to surface waters directly or through a non-functioning water treatment system.

Following the termination of mine operations, collection and treatment may cease immediately (premature closure) or may continue for some period (planned closure), but eventually will cease (perpetuity). If the water is nontoxic, in compliance with all criteria and standards, and its composition is stable or improving, the collection and treatment system may be shut down under permit. Otherwise, treatment would continue until institutional failures ultimately resulted in abandonment of the system, at which time untreated leachate discharges would occur.

6.3.1 Exposure

The magnitudes of exposures to untreated leachates would depend on leachate composition, flow rates, temporal variability, and spatial distribution (Section 4.4.1). Leachate may come from the tailings impoundments, waste rock piles, the walls of the pit, and any material deposited in the pit. The compositions of tailings and waste rock leachates are presented in Tables 5-12, 5-13, 5-14, 6-2, and 6-3. Water collection and treatment failures may be acute or chronic. A recent example is the overfilling of the tailings impoundment at the Nixon Fork, Alaska, mine that resulted in overtopping of the dam (Box 6-2). Chronic exposures would occur during operation if a lengthy process were required to repair a failure during operation. After operation, a chronic water collection and treatment failure may be due to intermittent or imperfect monitoring, collection or treatment or to abandonment of the site. The mine scenario describes, but does not quantify, the potential failures of water collection and treatment because they are so potentially diverse, so exposures cannot be quantified except in terms of the amount of dilution required to avoid toxic effects.

Potential flows of leachate from the TSF to the North Fork Koktuli from TSF 1 are estimated to be relatively low (31,500 m^3/year; Section 4.4.1). That leachate would resemble a mixture of the tailings test leachates and supernatants (Section 6.1.4.1) but, because it would come from the bottom of the impoundment, it could have undergone reduction and metal precipitation, which would lower concentrations. However, if the acid-generating Pre-Tertiary waste rock or pyritic tailings were deposited in the TSF, they would contribute to the leachate and increase concentrations, particularly if not kept immersed. The composition of that mixed rock and tailings leachate is not predictable at this time. Dissolved materials in the leachate would be oxidized when the leachate flows to the stream.

After mine closure, the mine pit would no longer be dewatered and would fill until precipitation and groundwater flow equilibrated (Section 4.3.7). The water that interacted with the walls of the pit and with any waste rock deposited in the pit might resemble a mixture of the waste rock leachates and ambient water. Once the pit is filled, it will flow to one of the streams. The rate of flow would be the amount of precipitation falling on the pit (approximately 4 and 14 million m^3 per year for the minimum and maximum sizes of the mine scenario) plus whatever water flowed into the pit from up-gradient (potentially including waste rock leachate). The path of that flow cannot be determined at this time, but the most likely receptor would be Upper Talarik Creek.

Experimental leachates from the Tertiary waste rocks of the Pebble deposit are neutral on average (Table 5-12), and rocks would be used for construction of the tailings dam, to line the edges of the TSF, and for other uses that require fill. Those uses could result in uncollected leachates. Excess Tertiary rock would be segregated from Pre-Tertiary waste rock in the piles.

Leachates of the Pre-Tertiary waste rocks of the Pebble deposit are acidic (Tables 5-13 and 5-14), and would require segregation and storage in such a way that the leachate would be collected and treated. At mine closure, it is expected that acid-generating rock would be disposed of in the TSF or the mine pit. However, premature closure could leave waste rock piles in place.

Net precipitation (rain and snow minus evaporation) on the waste rock pile would generate approximately 10 to 18 million m^3 of leachate per year in the two sizes of the mine scenario. If waste rock leachate was not collected and treated, it could potentially form the source of Upper Talarik Creek and South Fork Koktuli River because the piles would be located in their current headwaters. Exposure of fish and invertebrates to untreated Pre-Tertiary waste rock leachate would occur primarily through direct exposure to dissolved constituents. Dietary exposures of fish to copper could also occur. If acidic and metal-bearing leachate entered streams, acid neutralization would occur downstream, resulting in the formation of metal hydroxide flocs and the classic orange streams that occur below acid mine or acid rock drainage. Neutralization would be a temporally and spatially lengthy process because of the low alkalinity of the potential receiving streams.

6.3.2 Exposure-Response

As with other sources, leachate constituents are screened against acute and chronic criteria. For copper dietary exposure of fish, application of the dietary factor of 0.95 to the lowest chronic value for rainbow trout of 11.3 μg/L (USEPA 2007) results in a dietary benchmark of 10.7 μg/L (Section 5.3.2.2).

6.3.3 Risk Characterization

Failure of the water collection and treatment system during the operation or planned post-closure periods would, like failures of any water treatment system, be a relatively common occurrence of limited duration. Loss of power, mechanical failures, pipeline breaks, operator errors, or other events could result in the release of untreated wastewater to a stream. The composition of that water could be a mixture of the tailings and waste rock leachates, discussed below, plus domestic wastewater or other waters from the operation. The toxic effects would depend on the wastewater composition, which could exceed acutely or chronically toxic levels for invertebrates or fish, and on the duration of exposure, which could range from hours to weeks. Alternatively, water collection and treatment failure could be a result of an inadequately designed water treatment system which could result in the release of inadequately treated water as at the Red Dog Mine, Alaska (Ott and Scannell 1994, USEPA 1998, 2008). In that case, the failure could continue for years until a new or upgraded treatment system is designed and constructed.

Failure to collect tailings leachates would result in exposure to waters resembling those described in Tables 6-2 and 6-3 with some ambient water dilution. As discussed above, with respect to a possible tailings dam failure (Section 6.1.4), these leachates would be toxic to metals-sensitive invertebrates, at least until dilution by a factor of three to four, which would bring them below the chronic criterion. Immediately below TSF 1, flow of the North Fork Koktuli could be 100% leachate with dilution occurring downstream. Below TSF 2 and TSF 3, flow of the South Fork Koktuli could consist of tailings leachate mixed with whatever flowed from the area of the pit and waste rock pile.

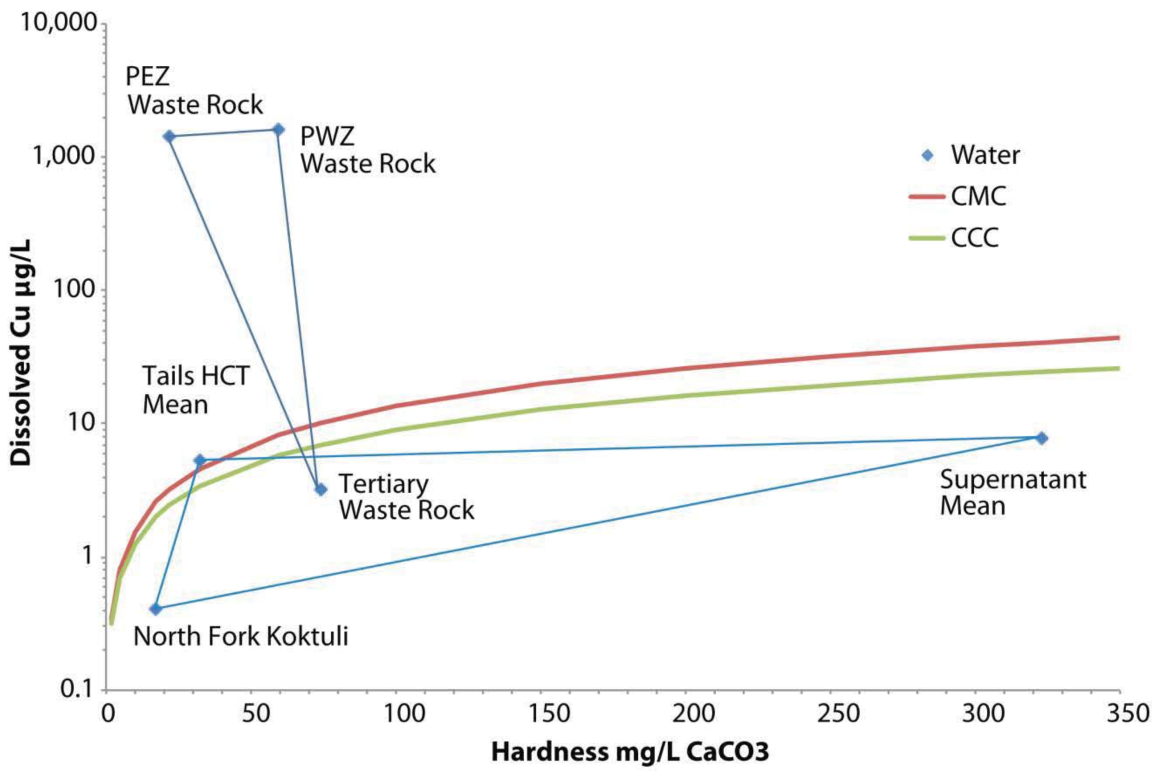
Tertiary waste rock leachate is neutral and is assumed to be the rock that would be used for construction of the tailings dam and berms, and potentially other structures requiring fill. Although the leachate from the tested Tertiary rock is much less toxic than from Pre-Tertiary rock, it still exceeds the acute (CMC) and chronic (CCC) national ambient water quality criteria for copper (Table 5-12), but not the diet-adjusted chronic value for rainbow trout. Hence, based on the available tests, leachate from mine structures would also require collection and treatment to avoid exceeding criteria and causing toxic effects on benthic invertebrates. Failure of collection and treatment of leachate from Tertiary waste rock could cause acute lethality in sensitive invertebrates and chronic toxicity to invertebrates at up to two times dilution.

Failure to collect Pre-Tertiary waste rock leachate could result in classic acid rock drainage. Leachates from both Pebble East and Pebble West Pre-Tertiary waste rocks would be highly toxic in both acute and chronic exposures (Tables 5-13 and 5-14). Figure 6-2 shows the much greater toxicity of waste rock versus tailings leachates. The 1,416 and 1,599 μg/L copper concentrations are far higher than the median lethal concentration values for rainbow trout estimated by the BLM for those waters (10 and 39 µg/L for Pebble East and West, respectively). Thus, even a short (less than 1 day) emission of untreated Pre-Tertiary waste rock leachate would be expected to result in a kill of fish and invertebrates. Even if the Pre-Tertiary leachate was equally mixed with leachate from Tertiary waste rock, not used primarily in construction, and half the leachate was from Tertiary waste rock, the leachate would be highly toxic. If it is not collected, part of the 10 to 18 million m^3/year of leachate from the waste rock pile could constitute the source of Upper Talarik Creek, which flows to Iliamna Lake. Assuming that half of the waste rock pile drained that way, the mean total flow of the creek (6.2 m^3/s) would provide only 36- to 20-fold dilution of the waste pile leachate, whereas the Pre-Tertiary waste rock leachates would require 2,900- to 52,000-fold dilution to meet the chronic criterion for copper. Hence, the entire creek and a potentially large mixing zone in the lake could be toxic to fish and invertebrates. This is a rough calculation, but it serves to indicate the large potential risk from improperly managed waste rock leachate.

An indication of the resources at risk is provided by aerial surveys of spawning salmon in Upper Talarik Creek that were conducted from 2004 through 2008 (PLP 2012, Table 5-1). The maximum index counts of adult salmon observed over this study period ranged as follows:

- Chinook salmon: 80 to 275
- Chum salmon: 0 to 18
- Coho salmon: 0 to 6,300
- Sockeye salmon: 10,000 to 82,000

Figure 6-2. Comparison of Copper Concentrations in Leachates and Background Water to Alaska's Hardness-Based Acute (CMC) and Chronic (CCC) Copper Standards. North Fork Koktuli is background water, Tails HCT is leachate from humidity tests of tailings, Supernatant is leachate from column tests of tailings, PWZ is Pebble West Pre-Tertiary, and PEZ is Pebble East Pre-Tertiary. Copper concentrations in tailings leachate in the field would be expected to lie in the lower blue triangle. Copper concentrations in waste rock leachate would be expected to lie in the upper blue triangle. Data are from Appendix H.



The mine pit would fill with water for hundreds of years after closure and eventually would be a source of leachate to streams if it was not collected and treated. Leachate would form from precipitation on the pit walls, from shallow groundwater entering the pit and from water collected in the pit dissolving metals and anions from the rock walls and any waste rock that was returned to the pit. The composition of the leachate would be approximated by some mixture of the waste rock test leachates (Tables 5-12, 5-13 and 5-14) with some dilution by ambient water. These tests are run in oxidizing conditions, so they maximize leaching rates. Oxygen levels are expected to be lower in the pit than in the tests, but oxygen would be provided in the pit by atmospheric diffusion from the surface, precipitation, shallow groundwater, and vertical mixing of water in the pit during turnover. If some or all of the waste rock leachate flowed to the pit after closure, it would contribute to the mix. The composition of the pit water cannot be predicted with any confidence, but some degree of leaching is inevitable. The experience with closed pit mines is quite variable, and some mines, such as the Berkeley Pit in Montana, are acidic and have high metal concentrations. After the pit is no longer pumped, it would fill and then drain to streams through groundwater or by overland flow. Leachate from waste rock piles would mix with pit water either directly, if the pit drained through the waste rock piles, or in the receiving stream. The pit water would be expected to flow to Upper Talarik Creek where, mixed with waste rock leachate, it would constitute the source of the stream. .

Acid mine or rock drainage has been a common phenomenon at metal and coal mines around the world, so analogies are numerous (Marchand 2002, Jennings et al. 2008). Such drainage has been shown to eliminate fish and invertebrates from streams and, after dilution, to reduce abundance, production, and diversity of stream and river ecosystems. A particularly relevant case is Britannia Creek, British Columbia, where acid drainage formed in an abandoned copper mine (Barry et al. 2000). Spring copper concentrations exceeded 1,000 µg/L and pH was below 6. The abundance of chum salmon fry was lower in the creek than in reference areas and 100% of Chinook salmon smolts died when placed in the creek in cages. In addition, sustained discharges have resulted in the loss of habitat through precipitation of metal hydroxides.

In sum, failure to collect and treat wastewaters could expose the biota of one or more of the streams draining the mine site to mildly or highly toxic water. Although it is clear from other mines that acid drainage can cause severe ecological effects, the probability of such drainage at the mine cannot be estimated. Unlike pipelines, there are no data on failure rates for wastewater management at mines. However, premature closures of mines are common and such closures are likely to leave acidic materials on the surface. Further, it is much too soon to know whether mines that are permitted for perpetual water collection and treatment (e.g., the Red Dog Mine in Alaska) can in fact carry out those functions in perpetuity.

6.3.4 Uncertainties

The risks from water collection and treatment failure are highly uncertain. The following factors contribute to these uncertainties.

- The range of failures is wide and the probability of occurrence of any of them cannot be estimated from available data.
- The waste rock leachate concentrations are from humidity cell tests. Because these tests involve repeated flushing of rock under oxic conditions, they may reasonably represent rock piles or pit walls leached intermittently by precipitation and snowmelt. However, laboratory tests of relatively small samples are imperfect models of large rock piles in the field.
- If the tested rock and tailings samples are not representative, other wastewater constituents may be of concern. For example, selenium concentrations are not high on average but are far above criteria in some individual leachate samples.
- The routes by which pit water and waste rock leachate would reach surface waters and the degree of dilution received are unclear. However, the acidity and high copper concentrations of the Pre-Tertiary waste rock leachates make it unlikely that they could be released without treatment and not cause severe toxic effects on invertebrates and fish.

6.4 Road and Culvert Failure

Roads and culverts can fail in various ways. The failure that is most likely to affect fish is the failure of a culvert.

6.4.1 Exposure

For purposes of this assessment we define culvert failure as the inability to provide passage for fishes or road failure due to culvert-caused redirection of stormwater and ensuing erosion. As noted in Section 4.4.3.4, road crossings often fail because of outfall barriers, excessive water velocity, insufficient water depth in culverts, disorienting turbulent flow patterns, lack of resting pools below culverts, or a combination of these conditions (Furniss et al. 1991). When culverts are plugged by debris or overtopped by high flows, road damage, channel realignment, and severe sedimentation also often occur. Observed frequencies of failed culverts vary in the literature but are generally high: 30% (Price et al. 2010), 53% (Gibson et al. 2005), 58% (Langill and Zamora 2002), and 66% (Flanders and Gariello 2000). As noted in Chapter 4, several culverts maintained by the State of Alaska failed in a flood at Pile Bay Road (Iliamna Lake to Cook Inlet) in 2004.

Blockages could persist for as long as the intervals between culvert inspections. Because of its importance to the mine, the access road would receive daily inspection and maintenance during the operation of the mine. Under such surveillance, a single erosional failure of a culvert that damaged the road would likely be identified soon after it occurs and repaired within a week. However, multiple failures such as might occur during an extreme precipitation event could require more than a month to repair. Inspections are likely to identify debris blockages sufficient to cause water to pool above the road, and such blockages would be cleared to prevent overflow of the road. Other failures that would reduce or block fish passage, such as downstream channel erosion that perches the culvert, might not be noticed by a driving inspection.

After mine operations end, traffic would be reduced to that which is necessary to maintain any residual operations on the site, and inspections and maintenance would likely decrease. However, if the road was adopted by the state or local governmental entity, the frequency of inspections and quality of maintenance could decline to those provided for other roads. Either of these possibilities could result in a proportion of failed culverts similar to those described in the literature (30 to 66%, Section 4.4.4).

6.4.2 Exposure-Response

Blockage of a culvert by debris or downstream erosion would prevent the in-and-out migration of salmon and the movement of other fish among seasonal habitats. The effects of a blockage would depend on its timing and duration. A blockage would result in the loss of spawning and rearing habitat if it occurred during in-migration of salmon and persisted for several days. It could cause the loss of a year class of salmon from a stream if it occurred during outmigration and persisted for several days.

Erosional failure of a road resulting from failure of a culvert to convey streamflow would create suspended sediment that would be carried downstream and deposited in the stream or lake bed.

Relationships between the concentration and duration of elevated sediment concentrations and effects on fish are presented in Section 6.1.3.2. Relationships between the amount of deposited sediment and effects on fish and invertebrates are presented in Section 6.1.2. A failure of this sort could also temporarily block the movement of fish.

6.4.3 Risk Characterization

Both blockage and erosional failure of culverts are common occurrences and both types of failure would be likely to occur at some stream crossings during the mine operations and thereafter. Blockages of culverts during operation could lead to the loss of a year class if they occurred during migrations and were not promptly cleared or repaired. The likelihood that such consequences would occur under daily inspection and maintenance is low. However, the likelihood of such a loss would greatly increase after mine operation if inspection and maintenance frequencies declined to those of typical roads.

Erosional failure of the road at a culvert could also temporarily block movement of fish resulting in the loss of a year class from the affected streams. Because this failure is most likely to result from flooding due to an extreme precipitation event, it is likely that multiple culverts would fail at the same time, so that repairs could be delayed and the blockage would persist, unless the failures were sufficient to create new channels for fish passage.

As noted in Section 5.4.2.2., culverts and other road crossings that do not provide free passage between upstream and downstream reaches can fragment populations into small demographic isolates vulnerable to extinction (Hilderbrand and Kershner 2000, Young et al. 2004). Drawing inference from natural long-term isolates of coastal cutthroat trout and Dolly Varden in southeastern Alaska, (Hastings 2005) found that about 5.5 km of perennial headwater stream habitat, supporting a census population size of greater than 2,000 adults, is required for a high likelihood of long-term population persistence.

Table 6-9 shows that, of the 34 potential salmonid-supporting streams, 24 containing less than 5.5 km of upstream habitat (stream length) would be intersected by the proposed road crossing. These 24 stream crossings contain a total of 33 km of upstream habitat and 227.6 km of downstream habitat. Eight of these represent anadromous river crossings that would likely be bridged. Three bridges would be built over non-anadromous streams, most likely including a Chinkelyes Creek crossing with 10.6 km of upstream habitat. Thus, two of the remaining 16 streams with less than 5.5 km of upstream habitat might be bridged, leaving 14 salmonid streams with culverts. Assuming typical maintenance practices after mine operations, roughly 50% of these streams, or 7 streams, would be entirely or in part blocked. As a result, salmon spawning would fail or be reduced in the upper reaches of the streams and the streams would likely not be able to support long-term populations of resident species such as rainbow trout or Dolly Varden.

Table 6-9. Upstream Length (km) Likely to Support Fish (Based on a Gradient Less Than 10%) and Downstream Length to Iliamna Lake at Road-Stream Crossings along the Potential Transportation Corridor

HUC-12 Name or Description	Stream Crossing Code	Upstream Fish Habitat Length (km)[x]			Downstream Length (km)[x]
		Main Channel	Tributaries	Total	
Headwaters, Upper Talarik Creek	**19030206007015**	15.8	37.3	53.1	43.2
	19030206007159	0.1	0.0	0.1	41.7
Upper tributary stream to Upper Talarik Creek[a]	**19030206007015_2**	18.1	0.0	18.1	43.2
	19030206007175	1.8	0.0	1.8	52.4
Tributary to Newhalen River portion upstream of corridor[b]	19030205007587	1.9	0.1	2.0	13.8
	19030205007593	0.4	0.0	0.4	12.4
	19030205007598	1.6	1.1	2.7	12.4
	19030205007606	0.3	0.0	0.3	5.0
	19030205007602	0.4	0.0	0.4	4.8
Headwaters, Newhalen River	19030205007615	1.5	1.7	3.2	0.8
	19030205000002	13.1	0.0	13.1[d]	26.2
Iliamna Lake	**19030206006678**	0.2	0.0	0.2	9.7
	19030206006644	0.3	0.0	0.3	11.2
Eagle Bay Creek	19030206006671	1.2	0.0	1.2	6.3
	19030206006663	4.9	0.4	5.3	6.2
	19030206006654	4.0	0.0	4.0	4.1
Youngs Creek Mainstem (Roadhouse Mountain HUC)	**19030206006598**	10.8	11.1	21.9	9.2
Youngs Creek East Branch[c]	**19030206006553**	5.6	11.2	16.8	7.6
Chekok Creek	**19030206006533**	4.8	1.5	6.3	4.9
	19030206032854	24.9	9.7	34.6	2.0
Canyon Creek	**19030206006359**	4.0	0.0	4.0	5.6
Iliamna Lake – Knutson Bay	19030206006336	0.3	0.0	0.3	3.8
	19030206006337	0.3	0.0	0.3	3.7
	19030206006236	0.1	0.0	0.1	3.5
Knutson Creek	**19030206006255**	0.4	0.0	0.4	3.5
	19030206006280	0.1	0.0	0.1	3.4
Iliamna Lake – Pile Bay	**19030206006228**	0.2	0.0	0.2	1.6
	19030206006227	0.3	0.0	0.3	3.0
Outlet, Pile River	19030206006222	0.7	0.0	0.7	4.6
	19030206000474	10.2	0.0	10.2[e]	4.1
	19030206010632	2.0	0.0	2.0	3.8
Middle Iliamna River	**19030206000033**	17.5	4.7	22.2[f]	6.5
Chinkelyes Creek	19030206005761	2.7	0.0	2.7	10.3
	19030206005737	9.9	0.7	10.6	17.9

Notes:
Values are arranged by 12-digit Hydrologic Unit Code (HUC-12), from west (top) to east (bottom). Bold stream crossing codes indicate these sites are listed in the Alaska Department of Fish and Game Anadromous Waters Catalog.

[x] Because the lengths at each crossing represent contiguous lengths, a portion of stream may be included in more than one crossing

[a] 190302060701; [b] 190302051404; [c] 190302060904

[d] Includes upstream length only to Six-mile Lake and Lake Clark

[e] Based on the ADFG Anadromous Waters Catalog, the amount of stream with documented anadromous fish habitat upstream of road crossing = 13.2 km.

[f] Based on the ADFG Anadromous Waters Catalog, the amount of stream with documented anadromous fish habitat upstream of road crossing = 41.2 km.

Source: Anadromous Waters Catalog (Johnson and Blanche in press)

6.5 Effects on Wildlife

Brown bears, wolves, and bald eagles depend on salmon for a large fraction of their summer diets. A tailings dam failure and spill, pipeline failure and spill, or water collection and treatment failure, would reduce the resources available to salmon and result in a potential reduction of those species. In addition, all terrestrial wildlife in the Bristol Bay watershed depend on the enhanced aquatic and terrestrial production provided by the marine nutrients that are brought into the watershed by returning and spawning salmon. Those nutrients are deposited on the landscape by salmon predators, where they increase the production of plants that feed moose, caribou, and other important wildlife species. Aquatic insects, which are more sensitive than fish, also provide nutrients to terrestrial ecosystems when they emerge. These potential effects of mine accidents on wildlife abundance and production cannot be quantified at this time, but they would inevitably result from any reduction in salmon abundance.

6.6 Effects on Human Welfare and Alaska Native Cultures

Salmon-mediated effects from potential accidents and failures associated with large-scale mining may have an effect on human welfare and Alaska Native cultures. Because the cultures are subsistence-based and reliant on salmon in particular, any negative impact on salmon quality and/or quantity resulting from failures or accidents should be assumed to cause a negative impact on human health and welfare, both directly from loss or change in food resources, and indirectly from disruption to an integral part of the culture. We are not attempting to quantify these impacts, but provide a qualitative assessment of them.

The potential salmon-mediated effects on Alaska Native cultures differ across these watersheds. Villages near the transportation corridor would be affected by spills from a pipeline or road and culvert failure. Villages downstream of a mine would be more affected by a water collection and treatment failure of a waste containment system. Salmon-mediated effects on Alaska Native cultures would be much greater from a failure of waste containment systems than from routine mine operations for three reasons. First, because all aspects of these cultures are subsistence- based, cultural vulnerability to long-term environmental disruption is very high (Appendix D). Second, although these cultures have evolved with fluctuations in salmon runs, a major failure or accident that would result in long-term disruption of salmon habitat and ongoing toxicity to salmon or their food would be significant. Third, because these cultures are closely tied to the local landscape and resources, it is virtually impossible for the cultures to be relocated elsewhere in response to an accident or failure.

A significant reduction in salmon quality or quantity would certainly have significant negative impacts on the salmon-based cultures in these watersheds. As with potential effects from the mine operation itself, it is not possible to develop a quantitative relationship between predicted effects on salmon and predicted effects on human health and Alaska Native cultures that would result from a failure or accident causing long-term habitat loss or toxicity downstream from the mine. However, as discussed in Section 2.2.4., the integration of the Alaska Native cultures with salmon is well documented (Appendix D). Because these cultures are so intimately related to the local landscape and the resources it provides, any change to salmon or other subsistence resources would likely result in changes to the culture itself. The magnitude of the changes could be assumed to be dependent on the magnitude and duration of the loss of subsistence resources, as well as disruption to the landscape itself.

The initial effect of an accident or failure on Alaska Native cultures would be the loss or decrease of subsistence salmon resources downstream. It is not possible to quantify the magnitude of subsistence resources that would be lost, nor is it possible to evaluate to what extent these subsistence users could be absorbed elsewhere in the watersheds. However, if these events were to occur, there would be negative effects on the ability of subsistence users to harvest salmon in these areas.

Subsistence foods used in rural Alaska have demonstrated the following health benefits.

- Consumption of subsistence foods results in lower cumulative risk of nutritionally mediated health problems, including diabetes, obesity, high blood pressure, and heart disease (Murphy et al. 1995, Dewailly et al. 2001, Dewailly et al. 2002, Din et al. 2004, Alaska Department of Health and Social Services 2005, Chan et al. 2006, Ebbesson and Tejero 2007).
- Traditional foods provide a range of micronutrients essential to health (Bersamin et al. 2007); iron (Nobmann et al. 2005) and very high levels of omega-3 fatty acids, the anti-inflammatory substances found in oily cold-water fish such as salmon (Murphy et al. 1995, Ebbesson and Tejero 2007).

As previously discussed, subsistence foods make up a substantial proportion of the human diet in the Nushagak and Kvichak watersheds. Subsistence accounts for an average of 80% of protein consumed by area residents, and the percentage of salmon harvest in relation to all subsistence resources ranges from 29% to 82% in the villages (Appendix D). Dietary transition away from subsistence foods in rural Alaska carries a high risk of excess consumption of processed simple carbohydrates and saturated fats similar to urban communities that have low availability and high cost of fresh produce, fruits, and whole grains (Kuhnlein et al. 2001, Bersamin et al. 2006). Also, alternative food sources may not be economically viable.

The loss of subsistence resources, especially salmon, has implications beyond the loss of food resources with demonstrated health benefits. The inability to harvest salmon from portions of these watersheds would also result in some degree of cultural disruption. Potential cultural disruption from negative effects on the salmon population is fundamental and goes well beyond a loss of food supply. Boraas and Knott (Appendix D) state, "The people in this region not only rely on salmon for a large proportion of their highly nutritional food resources; salmon is also integral to the language, spirituality, and social relationships of the culture."

The potential vectors of cultural change that would be related to a long-term reduction of salmon, or other subsistence resources that are dependent on salmon, are numerous. It is not possible to predict the magnitude of these effects, nor is it possible to predict what level of subsistence resource loss would be necessary to overcome the adaptive capacity of these cultures. On a physical level, the loss of salmon as a highly nutritious wild food, and the substitution of purchased food supplies, would have a negative effect on individual and public health (Appendix D). Salmon is especially valued around the world for nutrition and disease prevention. Also, the physical benefits of engaging in a subsistence lifestyle would be reduced (Appendix D). On an economic level, the necessity of purchasing expensive foods from outside the region in conjunction with limited opportunities to obtain paid employment in the region, would make it extremely difficult for families to survive in this region. While a large-scale mining industry would inject some market-based economic benefits for some time, it would likely employ a small fraction of Alaska Natives. Even these jobs would not be permanent, because mines have a finite lifespan, as well as "boom and bust" cycles.

On a cultural level, a significant loss of salmon would result in negative stress on a culture that is highly integrated with this resource. Boraas and Knott (Appendix D) discuss and document several of the social values and activities that are integrated with subsistence such as sharing and generalized reciprocity, fish camp, steam baths, gender and age equity, and wealth. Likewise, they document how spirituality and psychological health of the cultures is integrated with the natural world, specifically with salmon.

There are some measures that could be put in place to prevent and respond to accidents and spills. For small spills and releases that are contained in a timely manner, there may not be effects on the salmon subsistence resource. However, for large-scale releases, even with active remediation, effects on the salmon subsistence resource will be long-term. Because the Alaska Native cultures in this area have significant ties to the specific land and water resources in these watersheds, which have evolved over thousands of years, it would not be possible to replace subsistence use and culturally important areas lost to large-scale environmental contamination.

In summary, should an accident or failure related to a large-scale mine reduce the availability and/or increase the toxicity of salmon resources, there would be a negative impact on the health and welfare of the Alaska Native cultures. The potential for significant effects is much greater from a large accident or long-term failure than from routine mine operations. It is not possible to quantify the magnitude of cultural disruption in the event of accident or failure, nor is it possible to evaluate at what point these effects would overcome the adaptive capacity of the culture. However, if these events were to occur, they likely would have considerable long-term negative consequences on the Alaska Native cultures in these watersheds.

CHAPTER 7. CUMULATIVE AND WATERSHED-SCALE EFFECTS OF MULTIPLE MINES

7.1 Introduction

Thus far, this assessment has focused on the potential effects of a single, hypothetical mine. Although the Pebble deposit represents the most imminent and likely site of mine development in the Nushagak River and Kvichak River watersheds, the development of a number of mines, of varying sizes, is plausible in this region. Several known mineral deposits with potentially significant resources are located in the two watersheds (Table 7-1), and active exploration of deposits is occurring in a number of claims blocks (Figure 4-6). Once the infrastructure for one mine is built, it would likely facilitate the development of additional mines (e.g., initial road construction in the largely roadless area could make otherwise marginal ore deposits profitable). Thus, the potential exists in these watersheds for the development of multiple mines and their associated infrastructure (Box 7-1). In this chapter, we briefly consider potential cumulative effects of multiple mines in the Nushagak River and Kvichak River watersheds on Pacific salmon and other fish, and through these fish resources, their effects on wildlife and Alaska Native culture.

Table 7-1. Deposit Types with Significant Resource Potential in the Nushagak River and Kvichak River Watersheds (see Appendix H).

Deposit Type	Commodities	Example Deposits	References
Porphyry copper	Copper, molybdenum, gold, silver	Pebble, Humble, Big Chunk, Kijik River	Schmidt et al. 2007, Bouley et al. 1995
Intrusion-related gold	Gold, silver	Shotgun/Winchester, Kisa, Bonanza Hills	Schmidt et al. 2007, Rombach and Newberry 2001
Copper(+gold) skarn	Copper, gold	Kasna Creek, Lake Clark	Schmidt et al. 2007, Newberry et al. 1997
Iron skarn	Iron	Iliamna, Lake Clark	Schmidt et al. 2007, Newberry et al. 1997

BOX 7-1. THE FRASER RIVER

The Fraser River watershed, which supports sockeye and other salmon and contains multiple copper mines, serves as an analog for proposed mining development in Bristol Bay . Mining proponents have argued that the Fraser River fishery demonstrates that mining and fishing can co-exist (Joling 2011). However, the Fraser River is much less productive per unit of habitat than the Bristol Bay rivers, the fishery has been closed in some recent years, and most of the salmon runs are listed as threatened or endangered (Cohen 2010, O'Neal and Woody 2011).

The Cohen Commission for Inquiry into the Decline of Sockeye Salmon in the Fraser River has commissioned scientific projects to investigate the potential causes of decline. The report on freshwater ecological factors considered mining as one issue (Nelitz et al. 2011). The authors concluded that metal mining was a minor issue for sockeye habitat relative to other developments in the watershed, because there are only five active mines and only one (Endako) was in proximity to sockeye rearing habitat. Other developments that potentially affect habitat include logging; pulp, paper, and other wood products manufacturing; coal, placer, and gravel mining; urbanization; hydroelectricity; oil and gas drilling; agriculture; and water withdrawal. Although the authors argued that acid and metal drainage from closed mines poses a risk to salmon, they did not analyze that exposure. They concluded that mining was a plausible contributor, but not the major contributor to the decline in sockeye salmon, based on sedimentation of stream habitats.

Another Cohen Commission report that addressed contaminants listed mine-related contaminants, but could not specifically quantify the effects of mines (MacDonald et al. 2011). However, the authors concluded that concentrations of six metals (including copper) and phenols were sufficient to reduce survival, growth, or reproduction of sockeye salmon in the Fraser River.

In light of this information, Cohen Commission reports on the Fraser River do not provide evidence that mining and salmon co-exist. The fishery has declined, but available evidence is insufficient to conclude whether metal mining is a significant contributor. Neither the Cohen Commission nor USEPA’s contractor, ICF International, was able to assess the effects of metals mines in the Fraser River watershed, because compliance documents are not readily available.

Some raw monitoring data show episodes of low pH and frequently elevated dissolved copper in waters at the Gibraltar and Mount Polly mines. Other effects have been associated with closed mines. In particular, a tailings impoundment failed at the Pinchi Lake Mine in 2004, during reclamation activities, releasing tailings and leachate to Pinchi Lake. This accident, along with prior releases, resulted in the imposition of a very restrictive fish consumption advisory related to mercury bioaccumulation.

In sum, other activities in the watershed obscure any effects of the mines at the watershed scale. This diverse and relatively intensive development makes the Fraser River watershed a poor analogue for the development of mines in the nearly pristine Bristol Bay watershed.

Outside of Bristol Bay and throughout the range of Pacific salmon, most ecosystems face the cumulative effects of multiple land and water uses, creating a variety of stressors that occur in combination. Anadromous, and to a lesser extent, resident fish stocks in these watersheds are subject to persistent disturbance-induced stresses, the effects of which accumulate through the river network. For example, sedimentation of spawning beds from accelerated erosion, loss of rearing habitat from filling of streamside wetlands, and reduced out-migration success from downstream channelization are separate effects that together have a cumulative impact on salmon in a river system. The effect of each stressor accumulates regardless of whether factors occur at the same time, or even in temporal proximity. Since Pacific salmon, Dolly Varden, and rainbow trout are migratory, at least within a given stream system, adverse impacts can even accumulate when fish are absent from a particular reach. The overall consequences are diminished and extinct salmonid populations.

The Nushagak River and Kvichak River watersheds have not yet experienced these cumulative stresses associated with human activity, and their ecosystems are relatively pristine. Bristol Bay salmon runs are resilient because the abundance, diversity, and quality of Bristol Bay habitats result in large and diverse salmon populations (Chapter 2). Ordinary fluctuations in habitat availability or quality across the watersheds related to natural processes (e.g., landslides causing sedimentation of a river reach, floods causing scouring, drought) typically result in temporary loss or reduction in a discrete portion of habitat, but are easily absorbed by Bristol Bay's diverse salmon populations. In contrast, the types of impacts attributed to mining in Chapters 5 and 6 of this assessment may be long-lasting and extensive, eliminating habitat for extended periods and potentially killing or otherwise eliminating cohorts of fish. These impacts may remove component populations permanently or for long periods of time, weakening the overall population's ability to absorb and rebound from disturbance.

7.2 Potential Mine Development in the Bristol Bay Watershed

We cannot predict what mining activities would occur in the future, in what order mines would be developed, or what their specific impacts would be. However, we can identify a plausible example of potential cumulative effects on fisheries in the Nushagak River and Kvichak River watersheds, based on current patterns of mineral exploration.

7.2.1 Potential Mine Locations

Ghaffari et al. (2011) describe several "high priority" exploration targets beyond the Pebble deposit, including the Sill prospect and 25, 37, and 38 Zones. These target areas could be future mine sites if exploration identifies marketable quantities of metals. Other mining companies are actively exploring potential porphyry copper deposits in the Big Chunk, Humble, and Groundhog claims blocks (Szumigala et al. 2011). There is also active interest in exploring for gold, silver, or tin at two other prospects in the Nushagak River watershed (Shotgun/Winchester, and Sleitat Mountain) and a third with claims that straddle the divide between the Nushagak River and Kuskokwim River watersheds (Kisa). Other mineral claim blocks exist, but at the time of this writing are not currently being explored (Szumigala et al. 2009).

To examine the potential scope of cumulative impacts from multiple additional mines, we consider development of mines at the Humble, Big Chunk, Groundhog, Sill, and 38 Zone prospects. The Humble prospect is located approximately 135 km (84 miles) southwest of the Pebble deposit, and is thought to be geologically and geochemically similar to that deposit (Szumigala et al. 2011). All of the other prospects are within 25 km (16 miles) of the Pebble deposit and may be of the same geologic origin. Construction of mining infrastructure at the Pebble deposit would substantially reduce development costs for surrounding prospects and could facilitate creation of a mining district that could include these sites.

7.2.2 Mine Size and Components

As described in Chapter 4, each potential mine site would presumably include a mine pit and an adjacent waste rock disposal area. Most, if not all, would also include a mill, related processing facilities, and at least one tailings storage facility (TSF). Based on the range of worldwide porphyry copper deposits

(Section 4.1.1, Table 4-1), we assume ore bodies in the area would be smaller than the Pebble deposit, with an average size of 200 to 250 million metric tons—well below the minimum size of 2 billion metric tons considered in the mine scenario (Table 4-3).

We assume that future mines at the Sill and 38 Zone prospects would use the mill and TSFs built for potential mining at the Pebble deposit. For Humble, Big Chunk and Groundhog prospects, we develop plausible TSFs based on topography near the exploration sites and the projected need to store roughly 200 million metric tons of tailings (Figure 7-1). Although we cannot predict exact location and size of these TSFs, were they to be developed, these hypothetical locations should be representative enough to allow consideration of potential cumulative impacts of multiple mines.

7.2.3 Transportation Corridors

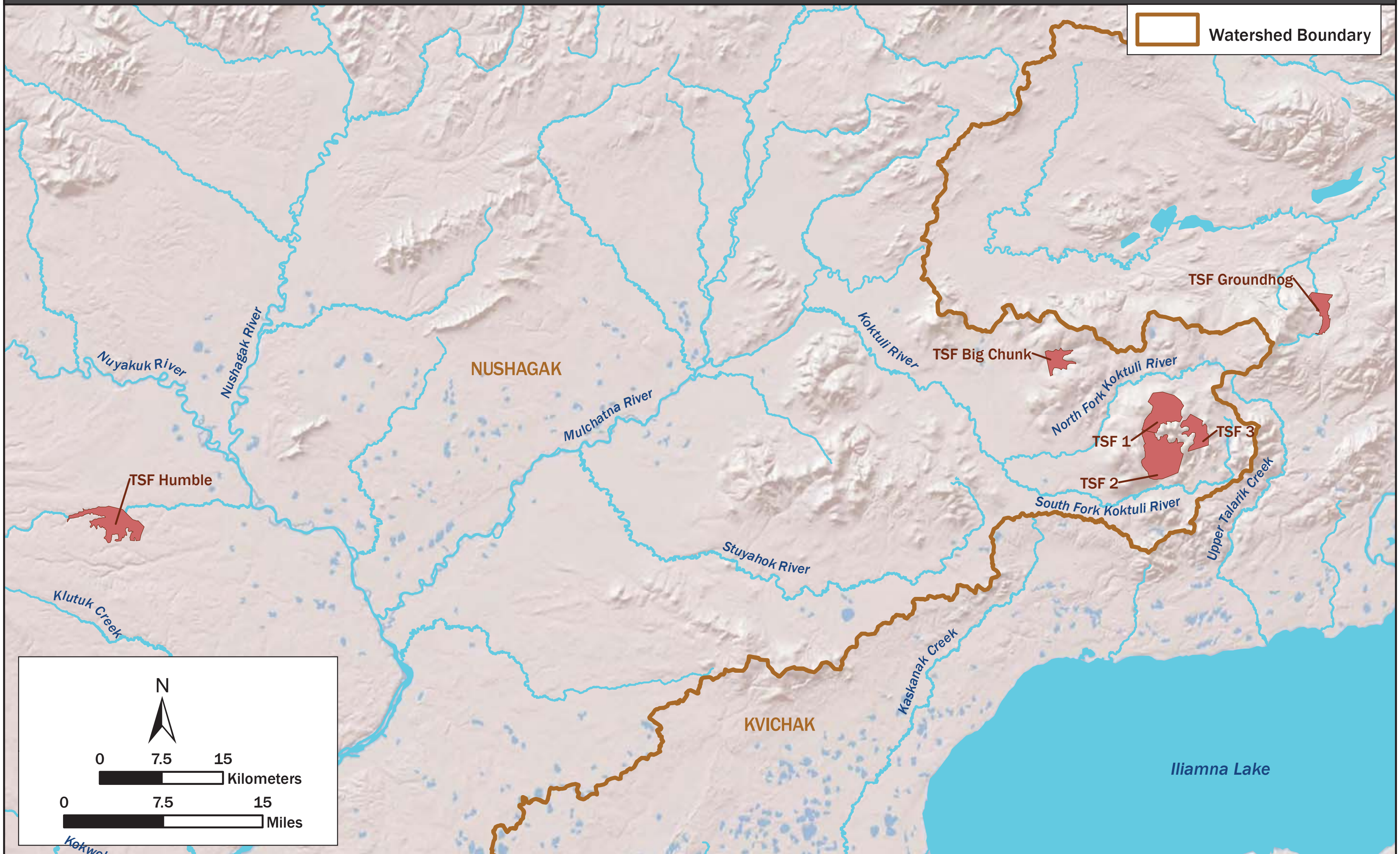
Any additional mines would also require construction of transportation infrastructure, including access roads and pipelines. Mines at the Sill, 38 Zone, Big Chunk, and Groundhog prospects presumably would connect to any roads and pipelines between the Pebble mine site and Cook Inlet (Section 4.3.8). For the Humble site, the Dillingham-Aleknagik Road (75 km to the southwest) is the closest link to existing road infrastructure; other possible routes would be to Dillingham (90 km to the southwest) or to a future roadway linking Aleknagik to the Alaska Peninsula (Appendix G).

7.3 Potential Mine Sites

7.3.1 Humble Prospect

Unlike the other potential mines, the Humble prospect occurs at low elevation (less than 150 m, Table 2-1) in the Nushagak-Bristol Bay Lowland physiographic region (Figure 2-2). The Humble claims drain into a number of the Nushagak River's tributaries (Figure 4-6).

Figure 7-1. Plausible Locations of Tailings Storage Facilities for Potential Mine Sites in the Nushagak River and Kvichak River Watersheds. TSFs 1, 2, and 3 are associated with long-term extraction from the Pebble deposit, while TSF Groundhog, TSF Humble, and TSF Big Chunk are hypothetical TSFs to support future mining in surrounding regions.



7.3.1.1 Hypothetical Tailings Storage Facility Location

The hypothetical Humble TSF would be about midway up Napotoli Creek, a tributary that empties into the Nushagak River approximately 30 km (19 miles) upstream of Koliganek (Figure 7-1). The Napotoli Creek stream network supports spawning and/or rearing Chinook, coho, and chum salmon, both within and upstream of the hypothetical Humble TSF. At least four of its headwater tributaries support rearing coho (Johnson and Blanche 2011, including nomination forms 11-371 and 11-383 through 11-386), and surveys have documented the presence of both adult and juvenile Dolly Varden within and above the TSF footprint. At the stream's outlet, the Nushagak River supports both adult and juvenile rainbow trout. Information on local population sizes for these species is not available. The Napotoli Creek stream network contains numerous beaver complexes, as well as frequent seeps and springs (Johnson and Blanche 2011, nomination forms 04-171, 06-753, 06-754, 11-369 through 11-372, and 11-384 through 11-386), which may provide important overwintering habitat for juvenile salmonids (Section 5.1.1.2). Villagers from Koliganek and New Stuyahok engage in subsistence fishing, hunting, and gathering in and along Napotoli Creek and the Nushagak River downstream of the Napotoli Creek confluence (Krieg et al. 2009). Subsistence targets include Chinook salmon, coho salmon, chum salmon, Dolly Varden, brown bear (in the headwaters area), moose, caribou, small mammals, waterfowl, upland birds, berries, and other plants.

7.3.1.2 Other Waters

Klutuk Creek and several other streams near the potential Humble TSF support Chinook, coho, and/or chum salmon. There is also documented sockeye salmon spawning and rearing habitat in Klutuk Creek, rearing in an unnamed stream in the northern part of the claims block, and both adult and juvenile Dolly Varden in an unnamed headwater tributary immediately downstream of the claims block's southwest corner. A large number of headwater streams originate within the claims block, and New Stuyahok and Koliganek villagers engage in subsistence activities in or downstream from some of these areas (Krieg et al. 2009).

7.3.2 Big Chunk Prospect

Like the Pebble deposit, the Big Chunk prospect occupies headwater areas in both the Nushagak River and Kvichak River watersheds. Portions of the Big Chunk prospect drains to the Koktuli River in the Nushagak River watershed, whereas others areas drain to the Chulitna River in the Kvichak River watershed. The Chulitna River flows through the northern edge of the claim and then to Lake Clark, in the upper part of the watershed; Lake Clark then drains into Iliamna Lake via the Newhalen River. Big Chunk is approximately 116 km (72 miles) upstream of New Stuyahok and approximately 114 km (71 miles) upstream of the village of Nondalton (with Lake Clark located in between).

7.3.2.1 Hypothetical Tailings Storage Facility Location

Big Chunk's hypothetical TSF would be located in the headwaters of an unnamed stream that drains to the mainstem Koktuli River (Figure 7-1). To date, surveys have documented rearing coho and Chinook salmon in this tributary downstream of the site, as well as both adult and juvenile Dolly Varden;

information on local population sizes for these species is not available. The stream system may be important for overwintering, given the presence of numerous beaver complexes (Johnson and Blanche 2011, nomination forms 06-885 and 06-887). The unnamed stream is within the subsistence brown bear, black bear, moose, caribou, small mammal, and waterfowl hunting grounds for villagers from Iliamna, Newhalen, Nondalton, Port Alsworth, and New Stuyahok (Fall et al. 2006, Krieg et al. 2009). Downstream of the tributary's confluence with the mainstem, the Koktuli River supports spawning coho, Chinook, and sockeye salmon, as well as adult Dolly Varden and rainbow trout.

7.3.2.2 Other Waters

Little of the fish habitat in most of the Big Chunk claims block has been surveyed, particularly in the portions that drain to the Chulitna River. The stream on which our hypothetical TSF is located has a tributary downstream that also supports rearing Chinook and coho, as well as both adult and juvenile Dolly Varden. In addition, in the southeast corner of the claims block, there are a number of tributaries to the North Fork Koktuli River (upstream of the outlet of our hypothetical TSF 1) that have documented spawning and/or rearing habitat for Chinook and coho salmon. There is documented sockeye salmon spawning in one of these tributaries, with sockeye presence extending into its headwater channels. Both adult and juvenile Dolly Varden are documented to occur in two of the headwater tributaries to the North Fork Koktuli River, as well as in one headwater stream that drains to the Chulitna River, at the northern part of the claims block. As at the other sites, information on local population sizes for these species is not available. The claims block includes a large number of headwater streams and numerous lakes and ponds, including at least four that support coho salmon and one—Big Wiggly Lake on the North Fork Koktuli River—with documented sockeye salmon spawning. Villagers from Port Alsworth, Nondalton, Newhalen, Iliamna, and Kokhanok use either these portions of the claims block or the downstream Chulitna River/Lake Clark/Newhalen River drainage for subsistence fishing (sockeye salmon, coho salmon, chum salmon, Dolly Varden, rainbow trout), hunting (brown and black bear, moose, caribou, small mammals, birds), and gathering (berries and other plants) (Fall et al. 2006, Krieg et al. 2009).

7.3.3 Groundhog Prospect

The majority of the Groundhog prospect also lies in the headwaters of the Chulitna River, and the southernmost portion occupies headwaters in both the Upper Talarik Creek and North Fork Koktuli River watersheds. The village of Nondalton lies approximately 88 km (55 miles) downstream on the Chulitna River. Igiugig lies approximately the same distance downstream on Upper Talarik Creek and across Iliamna Lake. New Stuyahok is approximately 152 km (94 miles) downstream in the Koktuli/Mulchatna/Nushagak River watershed.

7.3.3.1 Hypothetical Tailings Storage Facility Location

Groundhog's hypothetical TSF would be near the headwaters of Groundhog Creek, which rise in a series of lakes and ponds and drain to the Chulitna River via Rock Creek (Figure 7-1). The Chulitna River flows through the northernmost portion of the claims block, where there are also a number of lakes and ponds. Both the Chulitna River and Lake Clark support sockeye salmon, with spawning occurring at least in Lake Clark (Johnson and Blanche 2011). There are also Chinook and coho salmon, Dolly Varden, and rainbow trout in the Newhalen River stream network (Fall et al. 2006, Krieg et al. 2009, Johnson and Blanche 2011). The extent of salmonid habitat upstream in the Chulitna River system, including in Groundhog Creek, is unknown. In addition to subsistence activities along the Chulitna River, Lake Clark, and the Newhalen River, residents of Newhalen, Nondalton, and Port Alsworth hunt and gather along Groundhog Creek (Fall et al. 2006).

7.3.3.2 Other Waters

As described in Section 7.2.2, there is insufficient information with which to estimate location or size of other facilities associated with a potential mine at the Groundhog prospect. The southeast corner of the claim block includes a number of headwater tributaries to Upper Talarik Creek, at least one of which supports coho salmon as well as both adult and juvenile Dolly Varden. This tributary system originates in the same series of lakes and ponds as Groundhog Creek, and enters Upper Talarik Creek downstream of the hypothetical mine pit for the Pebble deposit. The southwest corner of the claims block drains to the North Fork Koktuli River and contains at least three headwaters streams that support both adult and juvenile Dolly Varden. The majority of the streams in the claims block are headwaters tributaries.

7.3.4 Sill and 38 Zone Prospects

We assume that hypothetical mines at the Sill and 38 Zone prospects would use the mill and TSFs built for mining at the Pebble deposit. Thus, we anticipate that the primary additional development at these sites would be limited to the mine pits, waste rock areas, and transportation corridors between the site and the other infrastructure.

7.3.4.1 Sill Prospect

The Sill prospect is on the east slope of the ridge between Upper Talarik Creek and Frying Pan Lake (Ghaffari et al. 2011), approximately 6 km (4 miles) southeast of the mine pit in the mine scenario (Section 4.3) and approximately 27 km (17 miles) upstream of Iliamna Lake on Upper Talarik Creek. The headwaters of three unnamed tributaries of Upper Talarik Creek drain the slope near this prospect, with the southerly two joining before entering Upper Talarik Creek. A single survey in the lower reach of this latter stream system found both adult and juvenile Dolly Varden, as well as juvenile coho salmon (ADFG 2012).

7.3.4.2 38 Zone Prospect

The 38 Zone prospect lies above the South Fork Koktuli River, on the north slope of Sharp Mountain (Ghaffari et al. 2011), opposite the outlet of the unnamed stream draining TSF 2 in the mine scenario (Figure 7-1). Six unnamed streams drain the mountain slope; five flow directly to the South Fork Koktuli River, and the sixth flows to the South Fork via an unnamed tributary that drains the lake and valley on the mountain's south side. Based on a single survey, at least one of the mountainside streams supports both adult and juvenile Dolly Varden, as does the south-side stream. In addition, three surveys in the south-side stream documented the presence of juvenile Chinook, coho, and sockeye salmon and Arctic grayling (ADFG 2012).

7.4 Potential Cumulative Effects on Assessment Endpoints

Chapters 5 and 6 describe the direct and indirect impacts resulting from routine operations (Chapter 5) and accidents and failures (Chapter 6) associated with a single, large-scale porphyry copper mine and its related infrastructure. Although the extent and nature of potential impacts would vary somewhat according to project specifics, the risks examined for that single mine apply, in a general sense, to any similar development in the Nushagak River and Kvichak River watersheds. Impacts on assessment endpoints resulting from multiple large-scale mines in the watersheds, their associated transportation corridors, and any related secondary development would accumulate over time and space, potentially affecting the region's populations of fish, wildlife, and human residents.

7.4.1 Routine Operations

7.4.1.1 Habitat Eliminated Under the Mine Footprint

Chapter 5 of this assessment describes the extent, nature, and effects of habitat modification and pollutant exposure resulting from our single mine scenario. For example, the maximum mine size at the Pebble deposit would eliminate or block 141.4 km of stream channel, including 33.8 km of documented anadromous fish streams (Table 5-4). The nature of habitat modification and pollutant exposure would be similar for these additional potential mines, although the extent and magnitude of their effects on assessment endpoints would vary by location.

We estimate that the Big Chunk, Humble, and Groundhog TSFs would eliminate or block an estimated 27.3, 97.0, and 43.2 km (16.9, 66.3 and 26.8 miles) of stream, respectively, in addition to any channels lost to mine pits and other features (including any at the Sill or 38 Zone prospects) (Table 7-2, Figure 7-2). At Humble, at least 22% of the directly affected stream habitat, 97 km (60 miles), currently supports Pacific salmon, and an additional 8%, 7.8 km (4.8 miles), supports Dolly Varden. Some of the lost streams support rainbow trout, and at least some of the stream length in the vicinity of the 38 Zone prospect also supports Dolly Varden. Because streams that may be affected have not been adequately surveyed for fish, these values for the length of habitat directly affected are likely underestimates.

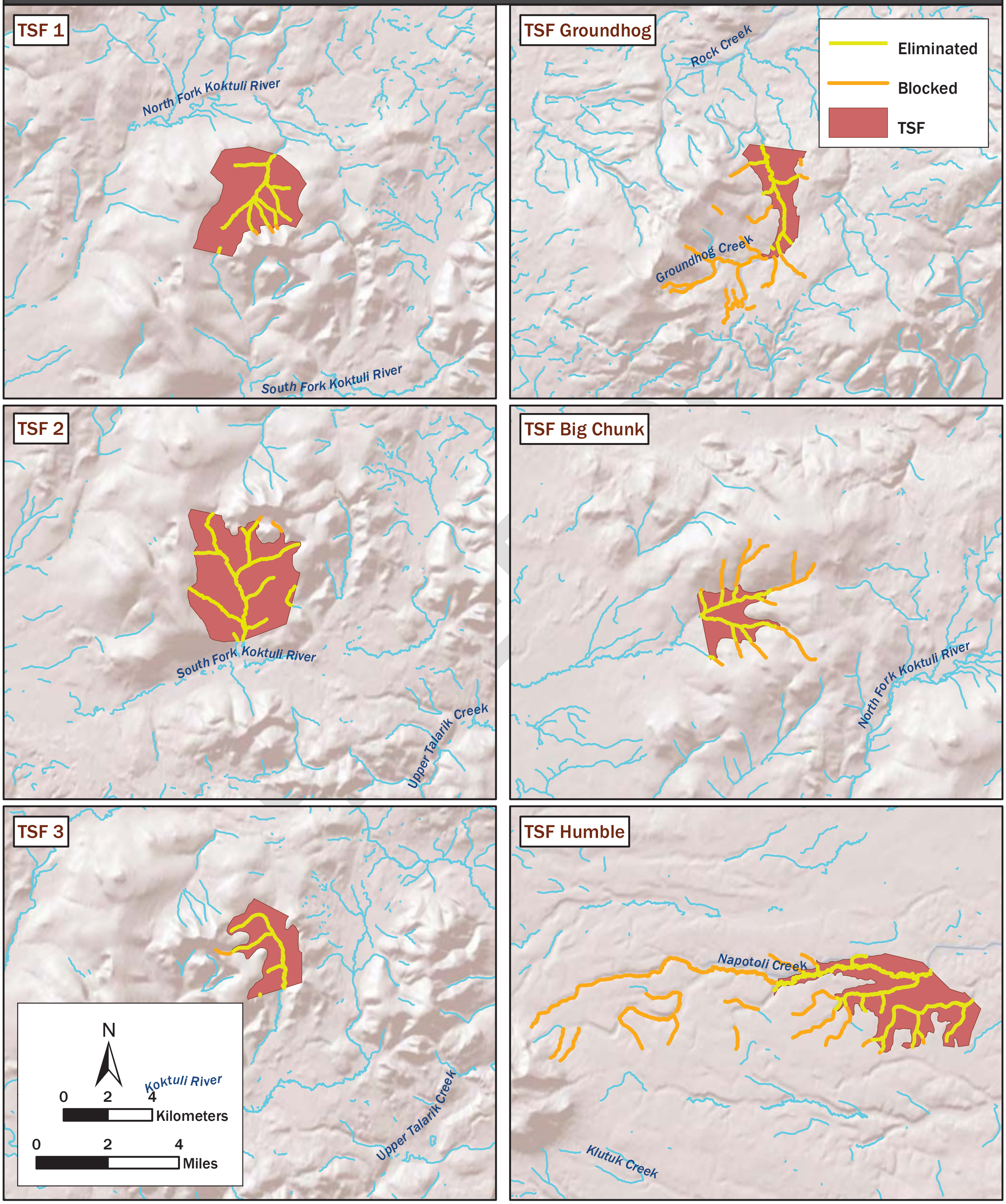
Table 7-2. Length of Stream Eliminated or Blocked by the Footprint of Each Mine Facility. Eliminated streams would be buried beneath the facility footprint; blocked streams would exist, but may be rendered inaccessible to fish by channel obstruction. These sites are hypothetical examples of facilities that would likely be constructed for a mine with 200 to 250 million metric tons of low-grade ore. See Box 5-1 for description of data sources.

Mine	Facility	Length of Stream (km)		Length of Documented Anadromous Stream (km)	
		Eliminated	Blocked	Eliminated	Blocked
Pebble deposit mine scenario (minimum size)	mine pit + waste rock pile	46.6	25.5	11.3	4.2
	TSF 1	14.8	0.6	6.2	0.0
Pebble deposit mine scenario (maximum size)	mine pit + waste rock pile	77.0	14.1	18.2	2.1
	TSF 1	14.8	0.6	6.2	0.0
	TSF 2	24.5	0.9	4.9	0.0
	TSF 3	8.8	0.7	2.4	0.0
Humble	TSF	32.6	64.4	6.2	8.1
Big Chunk	TSF	11.4	15.9	0.0	0.0
Groundhog	TSF	11.6	31.6	0.0	0.0

All three of the hypothesized additional TSFs are located in or near the headwaters of their stream watersheds. Because they occur in smaller watersheds, proportionately more of the headwaters would be lost than those associated with the mine scenario in Upper Talarik Creek or the North or South Fork Koktuli River. All three claims include a high density of headwater streams, indicating that other mine facilities, including any associated transportation corridors, would likely involve additional loss of headwaters, as would facilities at the Sill or 38 Zone prospects. Besides the documented and potential salmonid habitat that would be directly lost, routine operations at additional mines would also likely degrade or destroy downstream habitat quantity and quality as a result of reduced organic matter and inorganic nutrient transport, reduced groundwater inputs, and increased pollutant inputs (Sections 5.1.1.2 and 5.2). Such indirect impacts would likely extend to salmonid populations unaffected by the direct habitat losses associated with the mine footprint, including those in the lower Napotoli Creek/Nushagak River watershed and the Chulitna River/Lake Clark/Newhalen River watersheds. They could also contribute to cumulative degradation in the Koktuli River and Upper Talarik Creek watersheds.

Besides the impacts associated with the sheer quantity of lost or degraded habitat, additional large-scale mines could also cumulatively threaten the biological complexity of the Nushagak-Kvichak salmonid stock complex via effects on additional distinct populations of different species (Section 2.3.3, Figure 7-1, and Appendices A and B). It is reasonable to assume that some loss of genetic and life-history diversity would occur if multiple mines are developed, given the extent of stream loss and habitat degradation. Those losses would occur in geographically and hydrologically distinct parts of the Bristol Bay watershed (Section 7.3).

Figure 7 2. Streams Eliminated and Blocked by TSFs 1, 2, and 3 of the Mine Scenario and Hypothetical TSFs at Three Additional Claims (Groundhog, Big Chunk, and Humble) in the Nushagak River and Kvichak River Watersheds



Beyond the stream channels, the mine scenario would convert 33.7 km^2 (minimum size) or 84.1 km^2 (maximum size) of the Nushagak River and Kvichak River watersheds to mining facilities (Table 7-3). Conversion of these areas would result in losses of extensive floodplain, riparian habitat, and wetland areas. The Humble, Big Chunk, and Groundhog TSFs would convert additional portions of watershed area to mine footprint, resulting in increased habitat loss. With the addition of mine pits and waste rock disposal areas at those facilities and/or at the Sill and 38 Zone prospects, cumulative direct losses from all mine footprints in the watersheds would be substantially greater. The loss of these habitats would contribute to additional degradation of salmonid habitat—through the loss of nutrient, detrital, and baseflow inputs, temperature maintenance, and flow attenuation—beyond the areas lost as a direct result of mine development.

Table 7-3. Estimated Footprint Areas for the Mine Scenario (Section 4.3) and for Potential TSFs at the Humble and Big Chunk Prospects

Mine	Component	Area (km^2)
Pebble deposit mine scenario (minimum size)	mine pit + waste rock pile	18.8
	TSF 1	14.9
Pebble deposit mine scenario (maximum size)	mine pit + waste rock pile	40.4
	TSF 1	14.9
	TSF 2	21.2
	TSF 3	7.6
Humble	TSF	18.1
Big Chunk	TSF	5.9
Groundhog	TSF	6.7

7.4.1.2 Water Withdrawal

In addition to direct habitat loss to the mine footprint, habitat could be lost or degraded by water withdrawal and management of precipitation at the mine facilities, as described in Chapters 4 and 5. Mines require water to operate a mill and to transport tailings and concentrate. Reduced runoff from the collection of precipitation would effectively reduce the size of the watershed contributing to flow. Dewatering of a mine pit would further reduce the contributing watershed by creating a zone of depression as described in Section 4.3.7. Streams, wetlands, and ponds within this zone that receive their water through groundwater would dry up, discontinuing any water and nutrient contributions or biogeochemical modifications they made to surface waters. Groundwater flow down the valley would also be disrupted, potentially affecting spawning and wintering habitat downstream. Section 5.1.2 describes the downstream effects of changes in flow.

In some cases, operational water needs would be exceeded by precipitation and water withdrawal. In these cases, water would be treated and discharged to stream channels as surface water. Although surface flow may be partially restored, these point-source modifications could significantly alter natural flow regimes, and groundwater movement would continue to be modified for some distance downstream.

7.4.1.3 Roads and Stream Crossings

If additional mines are developed, additional transportation corridors would be needed. A transportation corridor could conceivably extend from the Sill prospect to an existing processing facility at the Pebble mine site without crossing any additional streams. A corridor from the 38 Zone prospect to processing facilities would require at least one crossing of and construction in proximity to the South Fork Koktuli River or Upper Talarik Creek. Our hypothetical Groundhog TSF would be located approximately 11 km (7 miles) north of our hypothetical transportation corridor (Section 4.3.8), and the Big Chunk TSF would be approximately 15 km (9 miles) to the west. Given the distribution and density of aquatic habitats in the landscape, connecting these TSFs to the assessment corridor would likely require multiple crossings of streams, lakes, ponds, and wetlands (e.g., for Big Chunk, there would have to be at least one crossing of the North Fork Koktuli River). A mine near the hypothetical Humble TSF would require a much longer transportation corridor, with a correspondingly greater number of crossings. Section 5.4 describes the impacts associated with routine operations of such crossings. The nature of impacts from additional transportation corridors would be similar to those discussed in Section 5.4 (e.g., stormwater runoff, siltation, salt runoff, and stream channel modifications). Although we cannot quantify the magnitude and extent of impacts with currently available information, adverse effects would increase as road length and number of stream and wetland crossings increased.

7.4.2 Accidents and Failures

Section 4.4 and Chapter 6 describe the probability of and consequences from a variety of accidents and failures, including leachate collection and treatment failures, pipeline breaks, road crossing failures, and TSF dam failures. Although the probability of such failures at individual facilities at any given time is low, the cumulative probability of failure increases with increasing number of facilities. For example, historical data described in Section 4.4.3.1 suggest a 98% cumulative probability of failure in one of the four 139-km pipelines over the life of the minimum (25-year) size of the mine scenario. Additional pipelines at additional mines would increase the overall probability of failure at some location in the watershed each year. Similarly, the chances of a road failure with significant consequences for downstream waters is substantial and becomes more so with increased road kilometers in the watershed (Section 4.4.3.3). The consequences could extend for many kilometers both upstream and downstream and would likely persist for many years. The cumulative effect would likely be a slow decline in productivity in these systems as the affected reaches grow and accumulate.

Another potential source of pollutant discharge into waters results from the failure to adequately understand the mining environment and the long-term needs for controlling pollutants at the site. Failures have a variety of sources: inadequate characterization of the geochemistry of an ore body or surrounding rock or of site hydrology, or even underestimating the potential mine longevity. At the Red Dog mine, near Kotzebue, in Northwest Alaska, treatment of waste rock wastewater for metals resulted in excessive total dissolved solids requiring that water be directed to the TSF rather than discharged. Compounding this problem, failure to implement planned surface water diversions early in mine development resulted in unpredicted rapid filling of the TSF. Unscheduled discharges were needed to prevent overfilling the TSF. At the Greens Creek Mine, near Juneau, Alaska, mine closure was planned for a specific timeframe. Reclamation of the dry stack tailings facility was designed to prevent acid drainage. However, the prolonged mine life prevented reclamation and resulted in acid drainage from the tailings. A new understanding of the geochemistry indicates that perpetual water treatment would be necessary even after reclamation. This was not part of the original design. In addition, the operators did not anticipate local wetlands chemistry, which resulted in a treatment system that re-dissolved metals before discharge. A new water treatment facility was needed to address this unanticipated source of pollution. These are unintended but essential failures in human judgment that may result in the discharge of wastewater from mine sites.

Mechanical failure and human error can also result in water bypassing a treatment system. Human error resulted in an uncontrolled discharge from a TSF in January 2012 at the Nixon Fork Mine, near McGrath, Alaska (see Box 6-2 for a description of events). Consequences of a bypass may be inconsequential or substantial. Waters from the January 2012 Nixon Fork Mine bypass were not thought to have reached nearby streams at the time of this writing and, therefore, were thought to have caused no environmental harm.

Although much less common, adverse impacts of TSF dam failures at these additional potential mines would likely be similar in nature to the partial failure described in Section 6.1, although magnitudes may vary with TSF size and the degree of failure. At the Humble prospect, such a failure could encompass Napotoli Creek and extend down the Nushagak River to Koliganek and beyond. At the Big Chunk prospect, slurry could flow into the mainstem Koktuli River, to within 15 km or less of the Mulchatna. At the Groundhog prospect, slurry could reach down the Chulitna River to within 10 km or less of Lake Clark. As Chapter 6 illustrates, although it would be a low-probability event, any single TSF dam failure could be catastrophically damaging to fisheries in the Nushagak River or Kvichak River watersheds. The presence of multiple large-scale mines and associated facilities would increase the probability of at least one failure occurring in the watershed over the lifetimes of the mines, and thus increase the chance of long-term adverse downstream effects.

7.4.3 Post-Closure Site Management

We assume that the post-closure management considerations described in Section 4.3.7 would generally apply to each additional mine, although the specifics would be based on design and operational assumptions of a mine and thus would differ from site to site. Closure would typically include hundreds to thousands of years of monitoring, maintenance, and treatment of any water that may flow off site. However, over these timeframes we would expect multiple and more frequent system failures. And, as mentioned above, given the relatively ephemeral nature of human institutions over these timeframes, we would expect that eventually monitoring, maintenance, and treatment would cease. The water quality of leachate at that time would control the effect of downstream waters.

7.4.4 Effects on Wildlife, Human Welfare, and Alaska Native Cultures

As the number of large-scale mines in the region increased, so would any mine-related and salmon-mediated effects on wildlife and humans, primarily via direct and indirect loss of food sources. A mine in the vicinity of our hypothetical Big Chunk TSF would increase the mine-related and salmon-mediated effects on wildlife and Alaska Native cultures by adding to impacts in the Koktuli River watershed, whereas operations at the Humble and Groundhog prospects would affect subsistence areas that would be relatively unaltered by operations of the Pebble mine claim and the Big Chunk prospect. Additional roads and pipelines would increase the number of sites across the landscape where failures affected habitat quality, incrementally affecting fisheries on which wildlife and humans depend.

7.4.5 Effects of Secondary Development

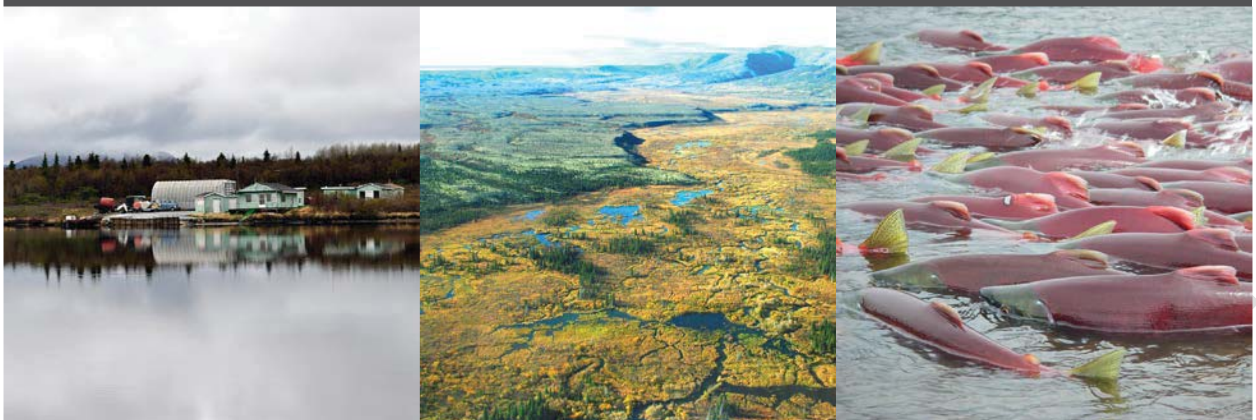
Although detailed analysis of secondary development effects is beyond the scope of this assessment, it is important to note that cumulative effects of secondary development associated with multiple mines would contribute to adverse effects on fish, wildlife, and Alaska Native culture. The construction of transportation corridors in this largely roadless area likely would facilitate non-mining development as a result of improved access and infrastructure. Induced development would take at least two forms. First, and less significant, would be facilities built to support mine operations (e.g., housing, service, office space for mine operators and employees). Second, improved accessibility to recreational opportunities would lead to the construction of additional cabins, lodges, and other residential and recreational facilities. For example, the road link to Cook Inlet, with a planned ferry connection between Williamsport and the Kenai Peninsula, would provide easier access to the area from Anchorage and the rest of Alaska (ADOT 2004). Improved accessibility would also increase fishing, hunting, and off-road vehicle impacts on nearby habitats, in turn potentially increasing competition and conflict between local and non-resident users. In addition, the introduction of workers and families from outside of the region would result in the development of facilities to meet their needs, including everything from entertainment facilities to schools.

7.4.6 Common Mode Failures

Multiple failures with a common cause are referred to as common mode failures. As discussed in Section 7.1.2.5, multiple failures such as failed dams, washed-out culverts, and broken pipelines could be caused by an earthquake or severe storm. This problem would be multiplied if there are multiple mines in the same area—that is, if multiple mines were developed in a mining district in the area of the Pebble deposit, they could all experience failures due to a single severe event.

7.4.7 Summary of Cumulative Impacts

The nature of impacts from mine footprints and from accidents and failures associated with mine components would be similar to impacts discussed in Chapters 4, 5, and 6. The footprints would eliminate substantial amounts of habitat, both directly and through dewatering effects. The consequences of leachate collection or treatment failure (Section 6.3) would depend on the chemical nature of the rock or tailings over which it flows. Since porphyry copper deposits tend to straddle the threshold between acid and non-acid generating (Section 4.1.2), there is a reasonable expectation that some of the waste rock and a portion of the tailings at any of these additional mines could be acid-generating. Each additional facility would increase the likelihood of collection and treatment failures, which would increase the frequency of discharge of untreated leachate or other wastewater in the Nushagak River and Kvichak River watersheds, with each event resulting in an increment of impact. Longer roads and pipelines associated with additional mines, coupled with a greater number of aquatic area crossings, would increase the probability of events such as culvert failures, pipeline breaks, and truck accidents that would damage aquatic systems, incrementally decreasing habitat value over an extensive area. In the long term, cessation of maintenance and treatment would likely result in the degradation of fisheries in waters downstream of each mine in the Nushagak River and Kvichak River watersheds.



CHAPTER 8. INTEGRATED RISK CHARACTERIZATION

This chapter summarizes the risk analysis results, organized by assessment endpoint. For each endpoint, it integrates the various sources of risk, including those from routine operations and accidents and failures, different physical and chemical exposures, and different pathways of exposure and mechanisms of effects. In addition, it combines multiple types of evidence, including evidence from analysis of the mine scenario and from knowledge of analogous mining operations. Limitations and uncertainties in the risk characterization are discussed. See Chapters 5 and 6 for the derivation of these conclusions. Finally, these results are extrapolated to the cumulative effects of multiple mines.

8.1 Overall Risk to Salmon and Other Fish

8.1.1 Routine Operations

Routine operations are defined as mine operations conducted according to conventional practices, including common mitigation measures, and that meet applicable criteria and standards. This mode is based on the assumption that there would be no accidents, failures, or other events that would cause any releases of mining products or wastes. Under these conditions, toxic effects would be minimized by reliable collection of all water from the site and effective treatment of effluents. Adverse effects on fish caused by habitat loss and modification would be directly and indirectly induced.

1. **Removal of 87.5 to 141.4 km of streams** in the footprint of the mine pit and waste storage areas, under the minimum and maximum mine sizes, respectively, would result in the loss of 21.7 to 33.8 km of streams that provide spawning or rearing habitats for coho salmon, sockeye salmon, Chinook salmon, and Dolly Varden.
2. **Reduced streamflow** resulting from water retention for mine operations, ore processing, transport, and other processes would reduce the amount and quality of fish habitat. Reductions in streamflow exceeding 20% would adversely affect habitat in an additional 1.8 to 9.5 km of streams, reducing

production of coho salmon, sockeye salmon, Chinook salmon, rainbow trout, and Dolly Varden. An unquantifiable area of riparian floodplain wetland habitat would be lost or would suffer substantial changes in hydrologic connectivity with streams due to reduced flow from the mine footprint.

3. **Removal of 10.2 to 17.3 km² wetlands** in the footprint of the mine would eliminate off-channel habitat for salmon and other fishes. Wetland loss would reduce availability and access to hydraulically and thermally diverse habitats that can provide enhanced foraging opportunities and important rearing habitats for juvenile salmon.

4. **Indirect effects of stream and wetland removal** would include reductions in downstream habitat quality in the three headwater streams draining the mine site, affecting the same species as the direct effects. Modes of action for these effects include the following.

    - A reduction in food resources would result from the loss of organic material and drifting invertebrates exported from the 87.5 to 141.4 km of streams lost to the mine footprint.
    - The balance of surface and groundwater inputs to downstream reaches would change. Shifting the source water flow from groundwater to surface water could reduce winter habitat and make the streams less suitable for spawning and rearing.
    - Water treatment and discharge, resulting in reduced passage through groundwater flowpaths, could increase summer water temperatures and decrease winter water temperatures, making streams less suitable for salmon and char.
    - These indirect effects cannot be quantified, but it is likely that one or more of these mechanisms would diminish fish production downstream of the mine in each watershed.

5. Diminished habitat quality in streams below road crossings would result primarily from runoff of road salts and of soil, leading to sedimentation of spawning habitat and reduced invertebrate prey. Because the road skirts Iliamna Lake, sockeye salmon are particularly at risk.

6. Inhibition of salmonid movements could result from culverts that may block or diminish use of the full stream length.

8.1.2 Failures

The assessment addressed four potential failures that could occur during mine operations or after mine closure in perpetuity: tailings dam failure, failure of a product concentrate or return water pipeline, failure to collect and treat contaminated water, and failures of roads and culverts. The probabilities and consequences of these failures are summarized in Table 8-1, and the derivation of these estimates is discussed in Box 8-1. Many other potential failures are not analyzed, including failures of the tailings, diesel, and natural gas pipelines; spills of ore processing chemicals on site or along the transportation corridor; failures of tailings dams on streams other than the North Fork Koktuli River; fires; waste rock slides; or failures at the port.

Table 8-1. Summary of Probability and Consequences of Potential Failures under the Mine Scenario

Failure Type	Probability[a]	Consequences
Tailings dam	10^{-4} to 10^{-6} per dam-year = recurrence frequency of 10,000 to 1 million years[b]	More than 30 km of salmonid stream would be destroyed and more streams and rivers would have greatly degraded habitat for decades.
Product concentrate pipeline	10^{-3} per km-year = 98% chance per pipeline in 25 years	Most failures would occur between stream or wetland crossing and might have little effect on fish.
Concentrate spill into a stream	2 x 10^{-2} per year = 1.5 stream-contaminating spills in 78 years	Fish and invertebrates would experience acute exposure to toxic water and chronic exposure to toxic sediment in a stream and potentially extending to Iliamna Lake.
Concentrate spill into a wetland	3 x 10^{-2} per year = 2 wetland-contaminating spills in 78 years	Invertebrates and potentially fish would experience acute exposure to toxic water and chronic exposure to toxic sediment in a pond or other wetland.
Return water pipeline	Same as product concentrate pipeline	Fish and invertebrates would experience acute exposure to toxic water.
Culvert, operation	Low	Frequent inspections and regular maintenance would result in few impassable culverts.
Culvert, post-operation	3 x 10^{-1} to 6 x 10^{-1} per culvert-instantaneous = 4 to10 culverts	In surveys of road culverts, roughly one-third to two-thirds are impassable to fish at any one time. This would result in 4 to 10 salmonid streams blocked.
Water collection and treatment, operation	High	Collection and treatment failures are highly likely to result in release of untreated leachates for hours to months.
Water collection and treatment, planned closure	High	Collection and treatment failures are highly likely to result in release of untreated leachates for days to months.
Water collection and treatment, premature closure or perpetuity	Certain	When water is no longer managed, untreated leachates would flow to the streams.

[a] Because of differences in derivation, the probabilities are not directly comparable.

[b] Based on expected state safety requirements. Observed failure rates for earthen dams are higher (about 5 x 10^{-4} per year or a recurrence frequency of 2,000 years).

BOX 8-1. FAILURE PROBABILITIES

Table 8-1 presents probability estimates and consequences of different kinds of failures. Here, we explain the derivation of these estimates. As much as possible, multiple methods are used within a failure type to determine how robust the estimates may be. The methods differ among failure types and the results are not strictly equivalent; however, they convey the likelihood of occurrence. More details can be found in Chapters 4 and 6.

Tailings Dam Failures

The most straightforward method of estimating the annual probability of failure of a tailings dam is to use the failure rates of existing dams. Three reviews of earthen dam failures produced an average rate of 1 failure per 2,000 dam years (i.e., a recurrence frequency of 2,000 years), or 5×10^{-4} per year. The argument against this approach is that it does not reflect current engineering practice.

The State of Alaska's guidelines suggest that an applicant follow accepted industry design practices such as those provided by U.S. Army Corps of Engineers (USACE) and the Federal Energy Regulatory Commission (FERC). Both USACE and FERC require a minimum factor of safety of 1.5 for the loading condition corresponding to steady seepage with the maximum storage facility. An assessment of the correlation of dam failure probabilities with safety factors against slope instability suggests an annual probability of failure of 1 in 1,000,000 years for Category I Facilities (those designed, built, and operated with state-of-the-practice engineering) and 1 in 10,000 years for Category II Facilities (those designed, built, and operated using standard engineering practice). This corresponds to risks of 10^{-4} to 10^{-6} per year. The advantage of this approach is that it addresses current regulatory expectations and engineering practices. The disadvantage is that we do not know whether standard practice or state-of-the-practice dams designed with safety factors will perform as expected. Another disadvantage is that this method was based on slope failures, and does not include other failure modes such as overtopping during a flood. The mine scenario includes three TSFs, each with multiple dams. However, we may assume that failure of one dam would relieve pressure on other dams on the same TSF. Hence, we may estimate that, after all three TSFs are operational, the risks would rise to 3×10^{-4} to 3×10^{-6} or a recurrence frequency of 3,000 to 300,000 years.

Pipeline Failures

A review of observed pipeline failure rates for oil and gas pipelines yields an average annual probability of failure per kilometer of pipeline of 10^{-3} or a frequency of 1 failure per 1,000 km per annum. This average risk comes very close to estimating the observed failure rate of the copper concentrate pipeline at the Minera Alumbrera mine, Argentina.

This annual failure probability, over the 118-km length of each pipeline within the Kvichak River watershed, results in a 0.12 (10^{-1}) probability of a failure in each of the four pipelines each year or a recurrence frequency of 8.5 years. If the probability of a failure is independent of location, and if it is assumed that spills within 100 m of a stream could flow to that stream, a spill would have a 0.16 probability (6-year recurrence frequency) of entering a stream within the Kvichak River watershed. This would result in an estimate of 0.019 stream-contaminating spills per annum, or 1.5 stream-contaminating spills over the duration of the maximum mine size (approximately 78 years). Similarly, a spill would have a 0.23 probability (4-year recurrence frequency) of entering a wetland, resulting in an estimate of 0.028 wetland-contaminating spills per annum or 2 wetland-contaminating spills over the duration of the maximum mine.

Water Collection and Treatment Failures

Although there are many anecdotal cases, we could find no data on the frequency of failures to fully collect and properly treat waters from mining operations. Hence, qualitative probabilities are used. During mine operation, collection or treatment of leachate from mine tailings, pit walls, or waste rock piles could fail in various ways. The probability that some failures would occur is judged to be high, but during operation the failures should be brief unless they involve a faulty system design. During a planned post-closure period, the probability of a collection or treatment failure would continue to be high, and would be less likely to be detected and stopped quickly because of the lower level of activity and oversight. Finally, if the mine is closed prematurely or post-closure water management ended, the discharge of untreated water would become inevitable.

Culvert Failures

Culvert failure is defined as a condition that blocks fish passage. Empirical data for culvert failures are not based on rates of failure of culverts but rather on instantaneous frequencies of culverts that were found to have failed in road surveys. The frequencies in recent surveys range from 0.30 to 0.66 (3×10^{-1} to 6×10^{-1}) per culvert. Fourteen streams in the Kvichak River watershed that are believed to support salmonid fish (salmon, trout, or char) would have culverts, so at any time 4 to 10 culverted streams would be expected to have blocked fish passage.

8.1.2.1 Tailings Dam Failure

Failure of a tailings dam would have a one in ten thousand to one in a million probability of occurrence per year for each tailings storage facility (TSF). Probability of a tailings dam failure increases with an increase in the number of dams. The minimum mine size includes one TSF and the maximum mine size includes three TSFs. Each TSF has multiple dams, but the probability of a spill from a TSF would not increase in proportion to the number of dams. The dam failures evaluated in this assessment simulated the release of 20% of the tailings (a conservative estimate) from a partial-volume (98-m) and a full-volume (208-m) dam at TSF 1.

Failure of the TSF 1 dam would result in the release of a flood of tailings slurry into the North Fork Koktuli River, scouring the valley and depositing tailings. The complete loss of suitable salmon habitat in the North Fork Koktuli River (30 km of habitat, which was the extent of the model) in the short term (less than 10 years) and the high likelihood of very low-quality spawning and rearing habitat in the long term (for decades) would result in near-complete loss of mainstem North Fork Koktuli River fish populations. Even salmon at sea during the failure would not find suitable spawning habitat on their return to the North Fork Koktuli River as adults. The river currently supports spawning and rearing populations of sockeye, Chinook, and coho salmon, spawning populations of chum salmon, and rearing populations of Dolly Varden and rainbow trout. Suspended mine tailings sediments would continue for an unknown (due to model and data limitations) distance further down the Koktuli River, and probably into the Mulchatna and Nushagak Rivers, with similar effects. Salmon anywhere in the flowpath below a tailings dam failure would be killed or forced downstream. Fish migrating into tributaries of affected rivers would be blocked from migration for some period of time, which our model could not predict.

Following the slurry flood, deposited tailings would continue to erode from the North Fork Koktuli and Koktuli River valleys. After many years, a new channel with gravel substrate and a natural floodplain structure would become established. However, that recovery would come at the expense of the downstream Mulchatna and Nushagak Rivers, as much of the spilled tailings initially deposited in the North Fork Koktuli and Koktuli Rivers would be re-suspended by erosion and transported down the drainage. This process could not be modeled with existing data and resources, but would be inevitable if a tailings spill occurred.

High concentrations of suspended tailings would occur following a tailings dam failure, but over time they would decline as erosion progressed. For some years, periods of high flow would be expected to suspend sufficient concentrations of tailings to cause avoidance, reduced growth and fecundity, and even death of fish. Migration to and from any affected tributaries would be blocked, if flow from the tributaries was not sufficient to adequately dilute suspended sediment concentrations, meaning that fish would not be able to reach spawning grounds, winter refugia, or seasonal feeding habitats.

Deposited tailings would degrade habitat quality for both fish and the invertebrates they eat. Salmon and trout spawn in gravels, and their eggs and larvae require sufficient space within the gravel for water to circulate; juvenile salmonids require even larger clear spaces for concealment from predators and for overwintering habitat. Tailings would fill those interstitial spaces. An increase in fines of more than 5% causes unacceptable effects on salmonid reproduction. Until considerable erosion occurred and a gravel-bedded channel was re-established, female salmonids would be unable to clean the gravel to spawn. Even where gravel is available, high deposition from upstream erosion of tailings could smother eggs and larvae. Recovery of suitable substrates via mobilization and transport of tailings fines would take decades, and would require and affect much of the watershed downstream of the failed dam.

In addition to degrading fish habitat, deposited tailings are potentially toxic. Based largely on their copper content, deposited tailings would be toxic to benthic macroinvertebrates, although existing data concerning toxicity to fish is less clear. Estimated pore water concentrations are less than published thresholds for chronic effects in fish, but directly relevant tests of salmonid early life stages have not been conducted. The combined effects of copper toxicity and poor habitat quality (particularly low dissolved oxygen concentrations) caused by fine sediment are unknown. Dietary exposures of salmonid fish via invertebrate prey exposed to tailings are estimated to be marginally toxic.

In sum, a TSF 1 dam failure would have severe direct and indirect effects on aquatic resources, and specifically on salmonid fish. In the short term (less than 10 years), certainly the North Fork Koktuli River below the TSF 1 dam failure location and very likely much of the Koktuli River would not support salmonid fish. For a period of decades, those waters would provide very low-quality spawning and rearing habitat, likely resulting in the nearly complete loss of North Fork Koktuli fish populations. Deposition, re-suspension, and re-deposition of tailings would likely cause serious habitat degradation in the Koktuli River and downstream into the Mulchatna River. Ultimately, spring floods and stormflows would carry some proportion of the tailings into the Nushagak River. Effects would be qualitatively the same for both the partial-volume and full-volume dam failures, although effects from the full-volume failure would extend up to 272 km further and last longer.

The Koktuli River watershed is an important producer of Chinook salmon for the larger Nushagak Management Zone. The Nushagak River watershed is the largest producer of Chinook salmon in the Bristol Bay region, with an annual escapement of nearly 160,000 fish (1966 to 2010) (Buck pers. comm.). Assuming ADFG aerial survey counts (Dye and Schwanke 2009) reflect the proportional distribution of Chinook salmon within the Nushagak River watershed, the tailings dam failure would eliminate 28% of that run due to loss of the Koktuli River salmon population, and an additional 10 to 20% could be lost because tailings deposited in the Mulchatna River would affect its tributaries. Sockeye salmon are the most abundant salmon returning to the Nushagak River watershed, with spawning escapement averaging more than 1.3 million fish. However, the proportion of sockeye and other salmon species that originates in the Koktuli River and Mulchatna River watersheds is unknown. Similarly, populations of rainbow trout and Dolly Varden of unknown size would be lost for decades.

The dam failure evaluated in the assessment used TSF 1 as a hypothetical but plausible location. Failure of the other hypothesized tailings dams at TSF 2 and TSF 3 were not modeled, but would have similar effects in the South Fork Koktuli River and downstream. However, because their volumes would be smaller, the effects would be less extensive. It would be expected that dam failures at TSFs located in other headwater areas would have similar impacts on different streams.

8.1.2.2 Pipeline Failures

The primary product of the mine would be a concentrate of copper and other metals that would be pumped in a pipeline to a shipping facility on Cook Inlet. Water that carried the sand-like concentrate would be returned to the mine site in a second pipeline. Based on the record of pipelines in general, and the world's largest metal concentrate pipeline in particular, one to two near-stream failures of each of these pipelines would be expected to occur over the duration of the life of the maximum mine size (approximately 78 years). In either case, water that is expected to be highly toxic would be released, potentially killing fish and invertebrates in the affected stream over a relatively brief period. If the concentrate pipeline spilled into a stream, it would settle and form bed sediment predicted to be highly toxic based on its high copper content and acidity. Unless the receiving stream was dredged, causing additional damage, this sediment would persist for decades before ultimately being washed into Iliamna Lake. Potential concentrations in the lake could not be predicted, but near the pipeline route Iliamna Lake contains important beach spawning areas for sockeye salmon that could be exposed to a toxic spill. Sockeye also spawn in the lower reaches of streams that could be directly contaminated by a spill.

8.1.2.3 Water Collection and Treatment Failures

Water in contact with tailings or waste rock would leach copper and other metals. The failure of collection and treatment systems due to imperfect design or operation, or the failure to maintain and operate these systems in perpetuity, could result in contamination of one or more of the streams draining the site. Based on a review of historical and operating mines, it is likely that there would be some failure of the collection and treatment systems, during the operation or post-closure periods. This could range from operations failures that result in short-term releases of untreated leachates, to long-term failures to operate the collection and treatment system in perpetuity. Our evaluation looked at one realistic possibility of leachate escaping at the base of the TSF 1 dam, for the minimum and maximum mine sizes. We also considered a failure to collect and treat leachate from waste rock piles around the mine pit.

Test leachates from the tailings and Tertiary waste rocks are mildly toxic (i.e., they would require an approximately two-fold dilution to achieve water quality criteria for copper but are not expected to be toxic to salmonids). If Tertiary rock was used for construction of mine infrastructure, leachate from these areas would need to be collected and treated to avoid toxic effects on benthic invertebrates. Our risk assessment did not evaluate this potential pathway in detail.

Pre-Tertiary waste rocks are acid-forming with high copper concentrations in test leachates (i.e., they would require 2,900- to 52,000-fold dilution). If leachate from a waste rock pile surrounding the mine pit was not collected, the 10.6 million m^3 of leachate per year from the waste rock pile could constitute source water for Upper Talarik Creek, which flows to Iliamna Lake. The total flow of Upper Talarik Creek would provide only 18-fold dilution, so the entire creek and a potentially large mixing zone in the lake could be toxic to fish and the sensitive invertebrates upon which they feed. The runs of sockeye and coho salmon in Upper Talarik Creek would be jeopardized by even a day-long event.

8.1.2.4 Road and Culvert Failures

The most likely serious failure associated with the access road would be blockage or failure of culverts. Culverts can commonly become blocked by debris that may not stop water flow, but that would block fish passage. Culverts can also fail to convey water because of landslides or, more commonly, flooding that washes out the culvert. In such failures, the stream may temporarily be impassable to fish until the culvert is repaired or until erosion re-establishes the channel. If either of these failures occurred during adult salmon immigration or juvenile salmon outmigration and the blockage was not cleared for several days, production of a year class could be lost or diminished.

Culvert failures would also result in the downstream transport and deposition of fine sediment. This could cause returning salmon to avoid the stream if they arrived during or immediately following the failure. More likely, the deposition of fine sediment from the washed-out culvert would smother salmon eggs and larvae, if they were present, and would degrade the downstream habitat for salmonid fish and the invertebrates that they eat. It would also change stream hydraulics and morphology, diminishing habitat value.

Extended blockage of fish passage at road crossings is unlikely during operation, because the mine scenario assumes daily inspection and maintenance. However, after mine operations end, the road may be maintained less carefully or maintenance may be transferred to a governmental entity. In that case, the proportion of culverts that are impassable would be expected to revert to the levels found in published surveys (30 to 66% at any time). Of the many culverts that would be required, 14 would be on streams that are believed to support salmonids. Hence, four to 10 streams would be expected to lose passage of salmon or resident trout or char and some proportion of those would have degraded downstream habitat resulting from sedimentation caused by road washout.

8.1.2.5 Common Mode Failures

Multiple failures could result from a common event, such as an earthquake or a severe storm with heavy precipitation. Failures resulting from such an event could include one to three tailings dam failures that spill tailings slurry to streams and rivers, road culvert washouts that send fine sediment downstream and potentially block fish passage, and product slurry and return water pipeline failures resulting from a culvert washout and scouring of the streambed or a slide of the roadbed. The effects of these accidents individually would be the same as discussed previously, but the co-occurrence of these failures would cause cumulative effects on salmonid populations and would make any mitigative response more difficult.

Over the perpetual timeframe that tailings, mine pit, and waste rock would be in place, the likelihood of multiple extreme precipitation events, earthquakes, or combinations of these events becomes much greater. Multiple events further increase the chances of weakening and eventual failure of facilities that are still in place.

8.2 Overall Loss of Wetlands

Wetlands are a dominant feature of the landscape in the mine area, and are important habitats for salmon and other fish. Ponds and riparian wetlands provide spawning, rearing, and refuge habitat for both anadromous salmonids and resident fish species. Other wetlands moderate flows and water quality, and can influence downstream delivery of dissolved organic matter, particulate organic matter, and aquatic macroinvertebrates that supply energy sources to fish. Wetlands would be filled or excavated in 10.2 km^2 and 17.3 km^2 of the mine footprint under the minimum and maximum mine sizes, respectively. In addition, an unquantifiable area of riparian floodplain would be lost or would suffer substantial changes in hydrologic connectivity with streams, due to reduced flow from the mine footprint. Another 0.18 km^2 of wetlands would be filled in the Kvichak River watershed by the roadbed of the transportation corridor. By interrupting flow and adding silt and salts, the roadbed would also affect approximately 2.4 to 4.9 km^2 of wetlands. Finally, a tailings or product concentrate spill could damage wetlands and eliminate or degrade their capacity to support fish.

8.3 Overall Fish-Mediated Risk to Wildlife

The effects of reduced salmon, trout, and char production on wildlife cannot be quantified at this time. However, some degree of reduction in wildlife would occur under the mine scenario. Routine operations would have local effects on brown bears, wolves, bald eagles, and other wildlife that consume salmon, resulting from reduced salmon abundance due to loss and degradation of habitat in or immediately downstream of the mine footprint. Any accidents or failures would have larger effects on salmon, which would reduce the abundance of their predators.

The abundance and production of wildlife is also enhanced by the marine nutrients that salmon carry on their spawning migration. Those nutrients are released into streams when the salmon die, enhancing the production of other aquatic species that feed wildlife. Salmon predators deposit these nutrients on the landscape, fertilizing the vegetation and increasing the abundance and production of moose, caribou, and other wildlife.

8.4 Overall Fish-Mediated Risk to Alaska Native Cultures

Under routine operations, the predicted loss and degradation of salmon, char, and trout habitat in the North and South Fork Koktuli River and Upper Talarik Creek would have some effect on Alaska Native cultures of the Bristol Bay watershed or in individual villages, because some subsistence resource areas would be lost. It is also possible that subsistence use of salmon resources would decrease, based on the perception of effects from mining. In addition, access to some subsistence use areas may be impeded by reductions in water levels resulting from water withdrawals.

The failures listed below could have sufficient effects on salmonids to influence subsistence resources and Alaska Native cultures.

- A spill of product concentrate or return water is likely and could severely affect fish populations in a stream or river, and potentially an area of Iliamna Lake.
- Flow of untreated waste rock leachate could destroy the fishery of Upper Talarik Creek and some portion of Iliamna Lake or of other streams below TSFs.
- A tailings dam failure would have more extensive effects. If the TSF 1 dam were to fail, fish populations would be lost for years to decades from the North Fork Koktuli River and likely from much of the Koktuli River. As tailings were carried downstream by erosion for decades after the spill, they would degrade spawning and rearing habitat in the Koktuli River and to a lesser but still potentially significant extent in the Mulchatna and Nushagak Rivers. Failures of other headwater TSFs could have similar effects.

The loss of fish production from these failures would reduce the availability of those subsistence resources to local villages, with negative consequences to human health and cultural identity. Salmon-based subsistence is integral to these indigenous cultures. If salmon quality or quantity is negatively affected, there would be negative consequences for the nutritional, social, and spiritual health of these Alaska Natives and their cultures. Because of the close cultural and nutritional connection with salmon that has developed over thousands of years, replacement of salmon with alternate food supplies or displacement of villages would not be effective in maintaining the health and welfare of Alaska Natives or their culture.

8.5 Summary of Uncertainties and Limitations in the Assessment

This is an assessment of a particular mine scenario, which makes various assumptions about mining, processing, and transporting of the porphyry copper resource in the Pebble deposit. The scenario does not represent specific plans of any mining company and, if the resource is mined in the future, actual events would not be identical to the mine scenario. This does not represent a source of uncertainty, but rather is an inherent aspect of any predictive assessment. Even an environmental assessment of a mining company's proposed plan would be an assessment of a scenario that undoubtedly would differ from actual events.

This assessment does have uncertainties and limitations in the extent to which the potential effects of the routine operation and potential accidents and failures can be estimated. These uncertainties are summarized below.

- The estimated annual probability of a tailings dam failure is uncertain and based on design goals rather than historical experience. Actual failure rates could be higher or lower than the estimated probability.
- The proportion of the tailings that would spill in the event of a dam failure could be larger than the largest value modeled (20%). However, even this conservative assumption results in an initial outflow beyond the capabilities of the model.
- The ultimate fate of spilled and deposited tailings in the event of a dam failure could not be quantified. From principles of geohydrology and analogy to other cases, we know that slurry would erode from areas of initial deposition and move downstream over a period of more than a decade. However, the data needed to model that process and the resources to develop the model were not available.
- It is uncertain whether a tailings spill would be remediated, how it would be remediated, how long it would take to remediate, and to what extent remediation could reduce effects downstream of the initial runout of the slurry.
- The effects of mining on fish populations could not be quantified because of the lack of quantitative information concerning salmon, char, and trout populations and their responses. The occurrence of salmonid species in rivers and major streams is generally known, but not their abundances, productivities, or limiting factors. Estimating changes in populations would require population modeling, which requires knowledge of life-stage-specific survival and production as well as knowledge of limiting factors and processes that were not available for this case. Further, it requires knowledge of how temperature, habitat structure, prey availability, density dependence, and sublethal toxicity influence life-stage-specific survival and production, which is not available. Obtaining that information would require more detailed monitoring and experimentation. Further, salmon populations naturally vary in size because of a great many factors that vary among locations and years. Collecting sufficient data to establish reliable salmon population estimates takes many years. Thus, we used estimated effects of mining on habitat as an available surrogate for estimated effects on fish populations.
- Standard leaching test data are available for test tailings and waste rocks from the Pebble deposit, but these results are uncertain predictors of the actual leachate composition from a tailings impoundment, tailings deposited in streams and their floodplains, and waste rocks in piles. Test conditions are artificial, and the materials tested may not be representative; in particular, the pyritic tailings were not tested. Additionally, data and resources were insufficient to allow geochemical modeling of water quality expected in the TSF or downstream of the mine site under varied chemical and hydrological conditions, or to model expected pit water chemistry at closure.
- The effects of tailings and product concentrate deposited in spawning and rearing habitat are uncertain. It is clear that they would be harmful to salmonid eggs, fry, or sheltering juveniles due to

both physical and toxicological effects, but the concentration in spawning gravels required to reduce reproductive success of salmonids is unknown.

- The actual response of Alaska Native cultures to any of these scenarios is uncertain. Interviews with Tribal Elders and culture bearers and other evidences suggests that responses would involve loss of food resources and cultural disruption, but it is not possible to predict specific changes in demographics, cultural practices, or physical and mental health.
- Although some tailings would eventually reach the estuarine portions of the Nushagak River and even Bristol Bay, exposures at that distance could not be estimated. Therefore, risks to salmonids resulting from marine and estuarine contamination could not be addressed.
- The assessment is limited by its focus on the effects of mining on salmonid fish and the indirect effects of diminished fish resources on wildlife and people. Direct effects on humans, wildlife, and terrestrial ecosystems are not included, and neither is secondary development associated with mine development.

8.6 Summary of Uncertainties in Mine Design and Operation

In addition to uncertainties in assessment, some uncertainties are inherent in planning, designing, constructing, operating, and closing a mine. Such uncertainties are inherent in any complex enterprise, particularly when they involve an incompletely characterized natural system. However, the large scales and long durations implied by the effort required to exploit this resource make these inherent uncertainties more prominent.

- Mines are complex systems requiring skilled engineered design and operation. The uncertainties facing mining and geotechnical engineers include unknown geologic defects, uncertain values in geological properties, limited knowledge of mechanisms and processes, and human error in design and construction. Vick (2002) notes that models used to predict the behavior of an engineered system are "idealizations of the processes they are taken to represent, and it is well recognized that the necessary simplifications and approximations can introduce error in the model." Engineers use professional judgment in addressing uncertainty (Vick 2002).
- Accidents are inherently unpredictable. Though systems can be put into place to protect against system failures, seemingly logical decisions about how to respond to a given situation can have unexpected consequences resulting from human error (as happened in January 2012, when the tailings dam at the Nixon Fork Mine near McGrath, Alaska, overflowed). Further, unforeseen events or events that are more extreme than anticipated can negate the apparent wisdom of prior decisions (Caldwell and Charlebois 2010).
- The ore deposit would be mined for decades, and the waste must be managed for centuries or even in perpetuity. Engineered waste storage systems of mines have only been in existence for about 50 years, so their long-term behavior is not known. The response of our best technology in the

construction of tailings dams is untested and unknown in the face of centuries of extreme events such as earthquakes and weather.

- Human institutions change. Over the long time span of mining and post-mining care, generations of mine operators must exercise due diligence. Priorities are likely to change in the face of financial crises, changing markets for metals, new information about the resource, political priorities, or any number of currently unforeseeable changes in circumstance. The promises of today's mine developers may not be carried through by future generations of operators whose sole obligation is to the shareholders of their time (Blight 2010).

8.7 Summary of Risks under the Mine Scenario

Even if the mining and mitigation practices described in the mine scenario were performed perfectly, an operation of this size would inevitably destroy or degrade habitat of salmonid fish. The mine footprint would eliminate or block 87.5 km of streams under the minimum mine size and 141.4 km under the maximum mine size, of which 21.7 and 33.8 km, respectively, support spawning and rearing habitat for coho, Chinook, and sockeye salmon and Dolly Varden. Wetlands would be filled or excavated in 10.2 km^2 and 17.3 km^2 of the mine footprint under the minimum and maximum mine sizes, respectively. Reduced flow from water use would degrade additional stream and wetland habitat. Leachates and other waste waters would be treated to meet standards, but the temperature and distribution of effluents could further degrade habitat.

The assessment considered failures of a tailings dam, product concentrate or return water pipeline, roads and culverts, and water collection and treatment system. Tailings dam failures are improbable, but become likely in the extremely long term. A tailings dam failure would destroy salmonid habitat in more the 30 km of the North Fork Koktuli River and associated wetlands for years to decades. A pipeline failure near a stream would be expected to occur during the life of a mine and would cause acute lethal effects on fish and create highly toxic sediment. Culvert failures are routine, and would block fish passage and could degrade downstream habitat. Failures to collect and treat leachates and other wastewaters could cause releases ranging from short-term and innocuous to long-term and toxic.

8.8 Summary of Cumulative and Watershed-Scale Effects of Multiple Mines

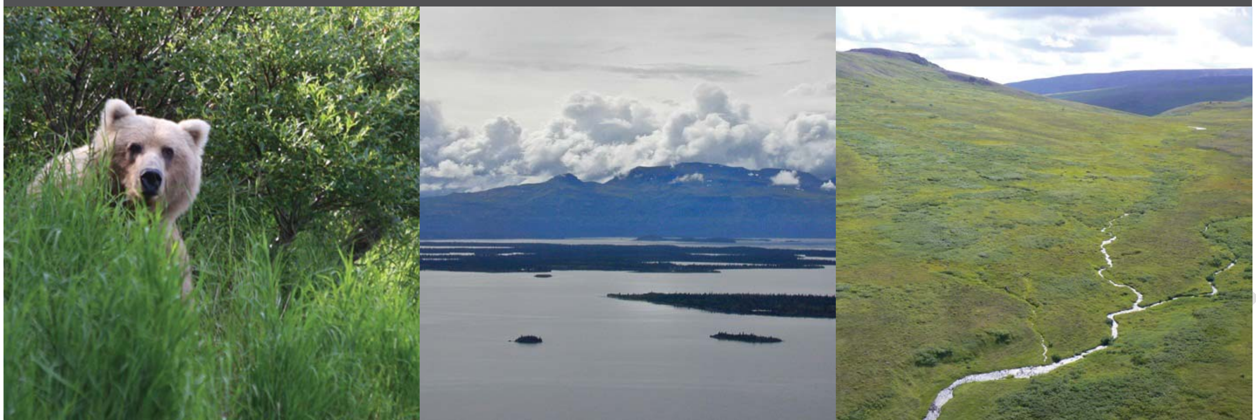
In order to provide reasonable realism and detail, this assessment largely addresses the potential effects of a single, hypothetical mine at the Pebble deposit. However, the development of multiple mines, of various sizes, in the Nushagak River and Kvichak River watersheds is plausible. Several known mineral deposits with potentially significant resources are located in the two watersheds and there is active exploration of a number of claims blocks. The construction of roads, pipelines, and other infrastructure for one mine would likely facilitate the development of additional mines. Thus, the development of multiple mines and their associated infrastructure may affect the environment of these watersheds.

Outside of Bristol Bay, most ecosystems that support Pacific salmon have been modified by the cumulative effects of multiple land and water uses. Anadromous fish are particularly susceptible to these effects because they require suitable habitat in spawning areas, rearing areas and along the migration corridors. Because Pacific salmon, Dolly Varden, and rainbow trout migrate among freshwater habitats seasonally or between life stages, loss or degradation of habitat in one location can diminish the ability of other locations to support salmonids. As a result of their particular susceptibility, anadromous salmonid fisheries have declined in most of their range due to the combined effects of habitat loss and degradation, pollution, and harvesting.

The Nushagak River and Kvichak River watersheds have not yet experienced these cumulative stresses associated with human activity, and their ecosystems are relatively pristine. Bristol Bay salmon runs are resilient because the abundance, diversity, and quality of Bristol Bay habitats result in large and diverse salmon populations. Fluctuations in habitat availability or quality across the watersheds caused by natural processes typically result in temporary loss or reduction in a discrete portion of habitat, but are easily absorbed by Bristol Bay's diverse salmon populations. In contrast, the effects of mining may be long-lasting and extensive, eliminating habitat for extended periods and potentially killing or otherwise eliminating cohorts of fish. Such effects may remove component populations permanently or for long periods of time, weakening the overall population's ability to absorb and rebound from disturbance.

To examine the potential cumulative effects from multiple mines, we considered development of mines at the Humble, Big Chunk, Groundhog, Sill, and 38 Zone prospects. The Humble prospect is located approximately 135 km (84 miles) southwest of the Pebble deposit, and is thought to be geologically and geochemically similar to that deposit. All of the other prospects are within 25 km (16 miles) of the Pebble deposit and may be of the same geologic origin. Construction of mining infrastructure at the Pebble deposit would substantially reduce development costs for surrounding prospects and could facilitate creation of a mining district that could include these sites.

The impacts from mine footprints and from accidents and failures associated with mine components would be similar to impacts projected for the Pebble deposit. The footprints would eliminate substantial amounts of stream and wetland habitat, both directly and through dewatering. We estimate that, at the Big Chunk, Humble, and Groundhog sites, the tailings impoundments alone would eliminate or block an estimated 27.3, 97.0, and 43.2 km of stream habitats. The consequences of leachate collection or treatment failure would depend on the chemical nature of the rock or tailings over which it flows. Because porphyry copper deposits tend to straddle the threshold between acid and non-acid generating, there is a reasonable expectation that some of the waste rock and a portion of the tailings at any of these additional mines could be acid-generating. Each additional facility would increase the likelihood of collection and treatment failures, which would increase the frequency of discharge of untreated leachate or other wastewater in the Nushagak River and Kvichak River watersheds, with each event resulting in an increment of impact. Longer roads and pipelines associated with additional mines, coupled with a greater number of stream crossings, would increase the frequency of events such as culvert failures, pipeline breaks, and truck accidents that would damage aquatic systems, incrementally decreasing habitat value over an extensive area. In the long term, cessation of maintenance and treatment would likely result in the degradation of fisheries in waters downstream of each mine in the Nushagak River and Kvichak River watersheds. Extreme natural events such as earthquakes and floods could cause failures of dams, roads, pipelines or water treatment systems at multiple mines.



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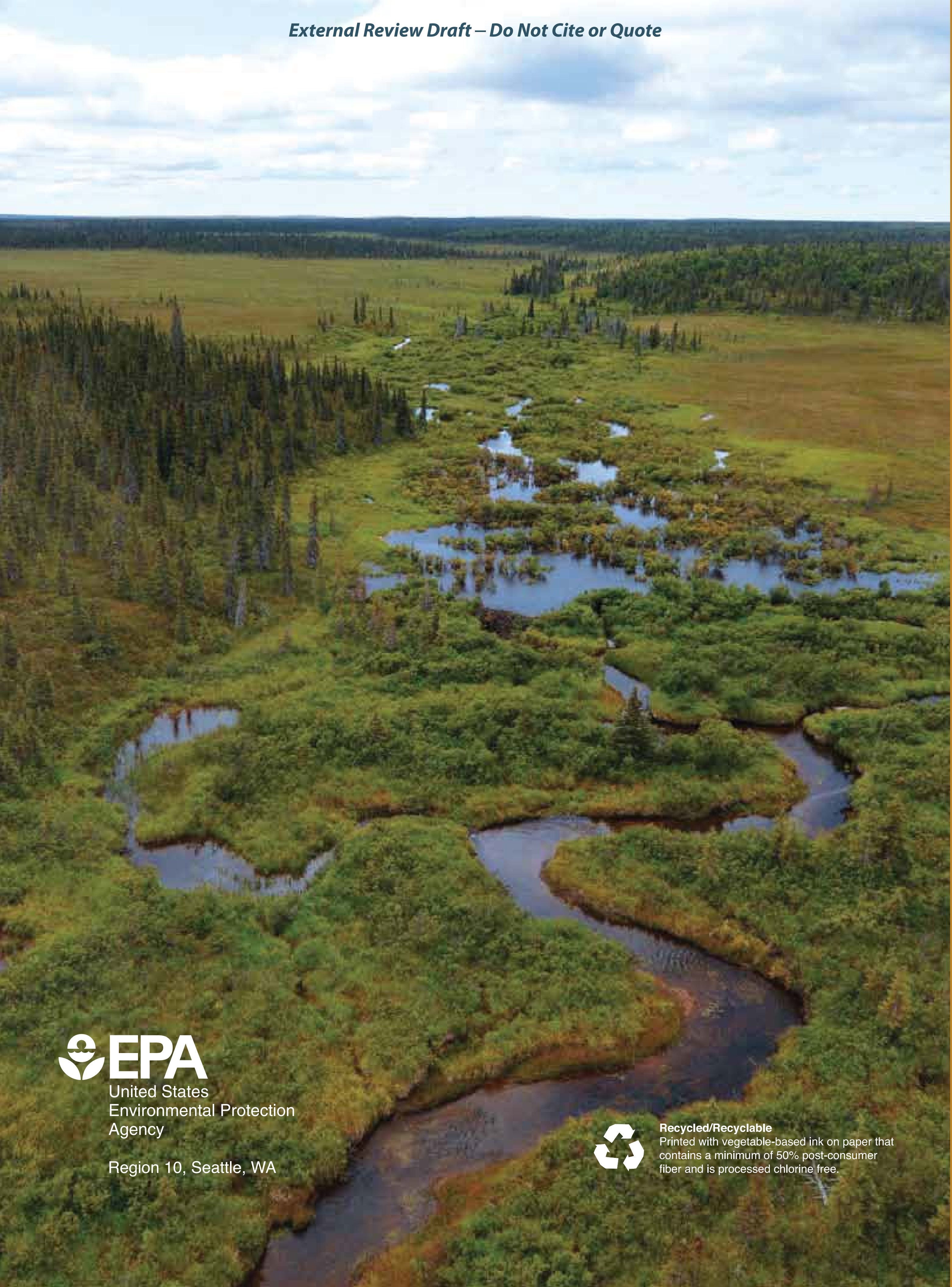

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