EPA 910-R-14-001A | January 2014

EPA
United States
Environmental Protection
Agency

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

Volume 1 – Main Report

Region 10, Seattle, WA
www.epa.gov/bristolbay

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Preferred citation: USEPA (U.S. Environmental Protection Agency). 2014. An Assessment of Potential Mining Impacts on Salmon Ecosystems of Bristol Bay, Alaska. Region 10, Seattle, WA. EPA 910-R-14-001.

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Acronyms and Abbreviations

AAC	Alaska Administrative Code
ADEC	Alaska Department of Environmental Conservation
ADF&G	Alaska Department of Fish and Game
ADNR	Alaska Department of Natural Resources
ADOT	Alaska Department of Transportation and Public Facilities
AFFI	Alaska Freshwater Fish Inventory
ANCSA	Alaska Native Claims Settlement Act
AP	acid-generation potential
APDES	Alaska Pollutant Discharge Elimination System
API	American Petroleum Institute
ASME	American Society of Mechanical Engineers
AUC	area under curve
AVS	acid volatile sulfides
AW	ambient waters
AWC	Anadromous Waters Catalog
BBAP	Bristol Bay Area Plan for State Lands
BLM	biotic ligand model
BMP	best management practice
CCC	criterion continuous concentration
CFR	Code of Federal Regulations
CH	Channel
CIBB	Cook Inlet-to-Bristol Bay
CMC	criterion maximum concentration
CRU	Climate Research Unit
CWA	Clean Water Act
DBB	Dillingham/Bristol Bay
DEM	digital elevation model
DOC	dissolved organic carbon
EC_{20}	20% effective concentration
EC_{50}	median effective concentration
EL_{50}	median effective level
E-R	exposure-response relationship
ERA	ecological risk assessment
FA	fish avoidance
FERC	Federal Energy Regulatory Commission
FK	fish kill
FP	high floodplain potential
FR	fish reproduction
FS	fish sensory
GCM	global climate model
GIS	geographic information system
GMU	Game Management Unit
HEC-RAS	Hydrologic Engineering Center's River Analysis System
HUC	hydrologic unit code
IA	invertebrate acute
IC	invertebrate chronic
IC_{20}	20% inhibitory concentration
IC_{50}	median inhibitory concentration
IFIM	Instream Flow Incremental Methodology

IGTT	Intergovernmental Technical Team
LC_{50}	median lethal concentration
LFP	left floodplain
MAF	mean annual streamflow
MCE	maximum credible earthquake
MDE	maximum design earthquake
MDN	marine-derived nutrients
Mi	Minerals
MOA	memorandum of agreement
NA	not applicable
NAG	non-acid-generating
NANA	NANA Regional Corporation, Inc.
NDM	Northern Dynasty Minerals
NED	National Elevation Dataset
NFP	no or low floodplain potential
NHD	National Hydrography Dataset
NNP	net neutralizing potential
NP	neutralizing potential
NPR	neutralizing potential ratio
NWI	National Wetlands Inventory
OBE	operating basis earthquake
OHW	ordinary high water
PAG	potentially acid-generating
PEC	probable effect concentration
PEL	probable effect level
PET	potential evapotranspiration
PHABSIM	Physical Habitat Simulation
PLP	Pebble Limited Partnership
PRISM	Parameter-elevation Regressions on Independent Slopes Model
Reclamation	Bureau of Reclamation
RFP	right floodplain
SCADA	supervisory control and data acquisition
SEM	simultaneously extracted metals
SNAP	Scenarios Network for Alaska and Arctic Planning
SWATP	Southwest Alaska Transportation Plan
SWPPP	stormwater pollution prevention plan
TDS	total dissolved solids
TEC	threshold effect concentration
TEL	threshold effect level
TLm	equivalent to LC_{50}
TSF	tailings storage facility
USACE	U.S. Army Corps of Engineers
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
WWTP	wastewater treatment plant

Units of Measure

µg	microgram
µS	micro-Siemens
°C	degrees Celsius
cm	centimeter
g	gram
ha	hectare
kg	kilogram
km	kilometer
km^2	square kilometers
km-yr	kilometer-year
L	liter
m	meter
m^2	square meter
m^3	cubic meter
mg	milligram
mm	millimeter
s	second
t	ton
yr	year

Unit of Measure Conversion Chart

Metric	Standard
1 μg (microgram)	3.527396×10^{-08} ounces
1 mg (milligram)	3.527396×10^{-05} ounces
1 g (gram)	0.035 ounce
1 kg (kilogram)	2.202 pounds
1 metric ton	1.103 tons
1 mm (millimeter)	0.039 inch
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 mile
1 km^2 (square kilometer) or 100 ha (hectares)	0.386 square mile
1 ha (hectare)	2.47 acres
1 L (liter)	0.264 gallon
1^0C (degrees Celsius)	1.8^0C + 32^0 Fahrenheit

Elements and Chemical Symbols

Ag	silver
Al	aluminum
As	arsenic
B	boron
Ba	barium
Be	beryllium
Bi	bismuth
Ca	calcium
$CaCO_3$	calcium carbonate
Cd	cadmium
Cl	chlorine
CN	cyanide
Co	cobalt
Cr	chromium
Cu	copper
F	fluorine
Fe	iron
Ga	gallium
Hg	mercury
In	indium
K	potassium
Mg	magnesium
Mn	manganese
Mo	molybdenum
Na	sodium
Ni	nickel
O	oxygen
Pb	lead
S	sulfur
Sb	antimony
Se	selenium
Se^{+4}	selenate
Se^{+6}	selenite
Si	silicon
SiO_2	silicon dioxide
Sn	tin
SO_4	sulfate
Sr	strontium
Te	tellurium
Th	thorium
Tl	thallium
U	uranium
V	vanadium
Zn	zinc

PREFACE

This assessment represents a collaboration among the U.S. Environmental Protection Agency's (USEPA's) Region 10, Office of Water, and Office of Research and Development. It was conducted as an ecological risk assessment to evaluate the potential impacts of large-scale porphyry copper mine development on salmon and other salmonid fishes and their habitats and consequent effects on wildlife and Alaska Native cultures in the Nushagak and Kvichak River watersheds of Bristol Bay, Alaska. It is not an assessment of a specific mine proposal for development, but the mine scenarios considered in the assessment are based on a published plan to mine the Pebble deposit. The assessment does not outline or evaluate decisions made or to be made by USEPA.

The first external review draft of this assessment (EPA 910-R-12-004) was released in May 2012 for a 60-day public comment period and external peer review by 12 independent expert reviewers. The revised, second external review draft was released in April 2013 (EPA 910-R-12-004B) for another 60-day public comment period and follow-on review by the same 12 peer reviewers. All public and peer review comments on the two drafts were considered in the development of this final assessment.

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PHOTO CREDITS

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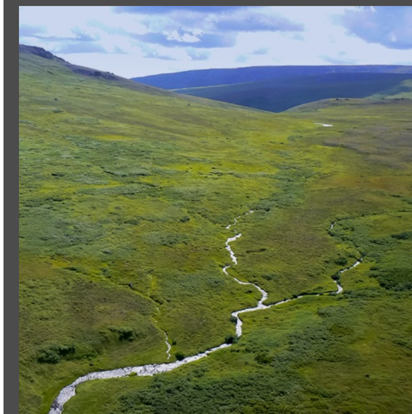
ACKNOWLEDGEMENTS

Assistance for this assessment was provided by ICF International under USEPA contract numbers EP-C-09-009 and EP-C-14-001, and by NatureServe under USEPA contract number EP-W-07-080. Additional assistance was provided by Colleen Matt (USEPA Contract EP-12-H-000099), Northern Ecologic LLC (USEPA Contract EPA-12-H-001001), Charles Schwartz (USEPA Contract EP-12-H-000105) and Halcyon Research (USEPA Contract EP-12-H-000107). The initial external peer review of the assessment was coordinated by Versar, Inc., under USEPA contract number EP-C-07-025. The external peer review of specific supplemental materials provided during the public comment period was coordinated by Versar, Inc., under USEPA contract number EP-C-12-045. Assistance with the management of public comments was provided by Horsley-Witten under USEPA contract EP-C-08-018. 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 conflict 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."

We would like to acknowledge the following people for their efforts in developing, completing, and releasing this assessment: David Allnut, Liz Blackburn, Andrew Bostrom, David Bottimore, Betzy Colon, Taukecha Cunningham, Kacee Deener, Brittany Ekstrom, Dave Evans, Ross Geredien, Rick Griffin, Rusty Griffin, Marianne Holsman, Carol Hubbard, Cheryl Itkin, Mark Jen, Nick Jones, Hanady Kader, Denise Keehner, Bill Kirchner, Matt Klasen, Terri Konoza, Rockey Louis, Don Maddox, Colleen Matt, Elizabeth McKenna, Julie Michaelson, Chris Moller, Heidi Nalven, Tim O'Neil, Jim Pendergast, Melissa Revely-Wilson, Tom Rothe, Stephanie Sarraino, Charles Schwartz, Vicki Soto, Gautam Srinivasan, Cara Steiner-Riley, Lowell Suring, Michael Szerlog, Jerry Tande, Mike Weimer, Bill Wilen, and Amina Wilkins.



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 significant 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 commercial, recreational, and subsistence fisheries and the future of Alaska Native tribes in the watershed, who have maintained a salmon-based culture and subsistence-based way of life 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. It uses the well-established methodology of an ecological risk assessment, which is a type of scientific investigation that provides technical information and analyses to foster public understanding and inform future decision making. As a scientific assessment, it does not discuss or recommend policy, legal, or regulatory decisions, nor does it outline or analyze options for future decisions.

This assessment characterizes the biological and mineral resources of the Bristol Bay watershed. It is intended to increase understanding of potential impacts of large-scale mining on the region's fish resources and serve as a technical resource for the public and for federal, state, and tribal governments as they consider how best to address the challenges posed by mining and ecological protection in the Bristol Bay watershed. It will inform ongoing discussions of the risks of mine development to the sustainability of the Bristol Bay salmon fisheries and thus will be of value to the many stakeholders in this debate.

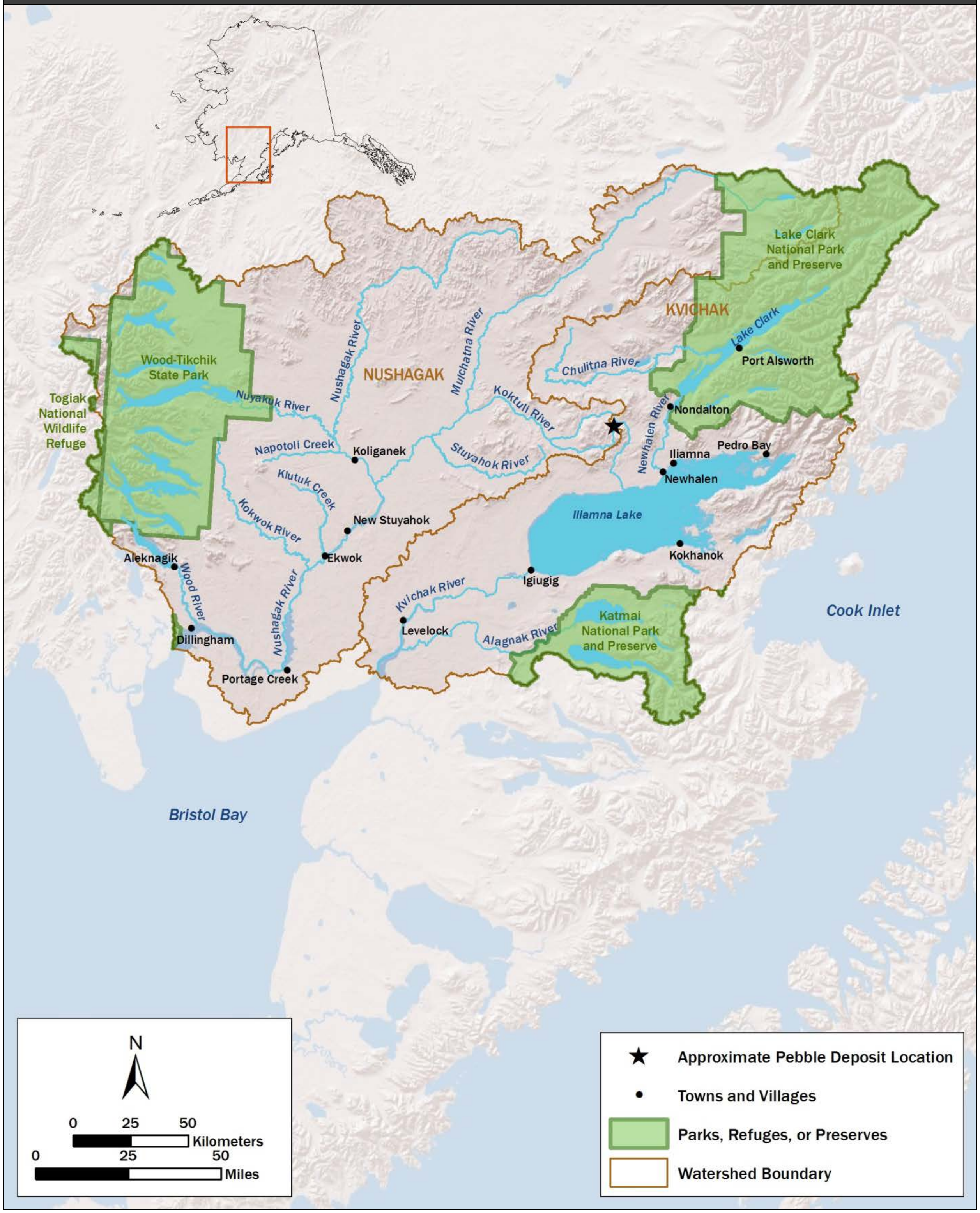
The assessment also will inform the consideration of options for future government action, including, possibly, by USEPA, which has been petitioned by multiple groups to address mining activity in the Bristol Bay watershed using its authority under the Clean Water Act (CWA). Should specific mine projects reach the permitting stage, the assessment will enable state and federal permitting authorities to make informed decisions to grant, deny, or condition permits and/or conduct additional research or assessment as a basis for such decisions. USEPA conducted this assessment consistent with its authority under the CWA Section 104(a) and (b).

Scope of the Assessment

This assessment reviews, analyzes, and synthesizes information relevant to potential impacts of large-scale mine development on Bristol Bay fisheries and consequent effects on wildlife and Alaska Native cultures in the region. Given the economic, ecological, and cultural importance of the region's key salmonids (sockeye, Chinook, coho, chum, and pink salmon, as well as rainbow trout and Dolly Varden) and stakeholder and public concern that a mine could affect those species, the primary focus of the assessment is the abundance, productivity, and diversity of these fishes. Because wildlife in Bristol Bay are intimately connected to and dependent on these and other fishes, changes in these fisheries are expected to affect the abundance and health of wildlife populations. Alaska Native cultures have strong nutritional, cultural, social, and spiritual dependence on salmon, so changes in salmon fisheries are expected to affect the health and welfare of Alaska Native populations. Therefore, wildlife and Alaska Native cultures are also considered as assessment endpoints, but only as they are affected by changes in salmonid fisheries.

The assessment considers multiple geographic scales. The largest scale is the Bristol Bay watershed, which is a largely undisturbed region with outstanding natural, cultural, and mineral resources. Within the larger Bristol Bay watershed, the assessment focuses on the Nushagak and Kvichak River watersheds (Figure ES-1). These are the largest of the Bristol Bay watershed's six major river basins, containing about 50% of the total watershed area, and are identified as mineral development areas by the State of Alaska. Given its size and extent of characterization, the Pebble deposit is the most likely site for near-term, large-scale mine development in the region. Because the Pebble deposit is located in the headwaters of tributaries to both the Nushagak and Kvichak Rivers, both of these watersheds are subject to potential risks from mining. The third geographic scale is the watersheds of the three tributaries that originate within the potential footprint of a mine on the Pebble deposit: the South Fork Koktuli River, which drains the Pebble deposit area and converges with the North Fork west of the Pebble deposit; the North Fork Koktuli River, located to the northwest of the Pebble deposit, which flows into the Nushagak River via the Koktuli and Mulchatna Rivers; 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-1). The mine footprints in the three realistic mine scenarios evaluated in the assessment make up the fourth geographic scale. These scenarios—Pebble 0.25, Pebble 2.0, and Pebble 6.5—define three potential mine sizes, representing different stages in the potential mining of the Pebble deposit. The final geographic scale is the combined area of the subwatersheds between the mine footprints and the Kvichak River watershed's eastern boundary that would be crossed by a transportation corridor linking the mine site to Cook Inlet.

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



The assessment also addresses two periods for mine activities. The first is the development and operation phase, during which mine infrastructure would be built and the mine would be operated. This phase may last from 20 to 100 years or more. The second is the post-mining phase, during which the site would be monitored and maintained. Water treatment and other waste management activities would continue as necessary and any failures would be remediated. Because mine wastes would be persistent, this period could continue for centuries and potentially in perpetuity.

We began the assessment with a thorough review of what is known about the Bristol Bay watershed, its fisheries and wildlife populations, and its Alaska Native cultures. We also reviewed information about copper mining and publicly available information outlining proposed mine operations for the Pebble deposit. The Pebble deposit has been the focus of much exploratory study and has received significant attention from groups in and outside of Alaska. With the help of regional stakeholders, we developed a set of conceptual models to show potential associations between salmon populations and the environmental stressors that might reasonably result from large-scale mining. Then, following the USEPA's ecological risk assessment framework, we analyzed the sources and exposures that would occur and potential responses to those exposures. Finally, we characterized the risks to fish habitats, salmon, and other fish populations, as well as the implications of those risks for the wildlife and Alaska Native cultures that use them.

This is not an in-depth assessment of a specific mine, but rather an examination of potential impacts of reasonably foreseeable mining activities in the Bristol Bay region, given the nature of the watershed's mineral deposits and the requirements for successful mine development. The assessment analyzes mine scenarios that reflect the expected characteristics of mine operation at the Pebble deposit. It is intended to provide a baseline for understanding potential impacts of mine development, not just at the Pebble deposit but throughout the Nushagak and Kvichak River watersheds. The mining of other existing porphyry copper deposits in the region would be expected to include the same types of activities and facilities evaluated in this assessment for the Pebble deposit (open pit mining and the creation of waste rock piles and tailings storage facilities [TSFs]), and therefore would present potential risks similar to those outlined in this assessment. However, because the region's other ore bodies are believed to be much smaller than the Pebble deposit, those mines would likely be most similar to the smallest mine scenario analyzed in this assessment (Pebble 0.25).

This assessment considers many but not all potential impacts associated with future large-scale mining in the Bristol Bay watershed. Although the mine scenarios assume development of a deep-water port on Cook Inlet to ship product concentrate elsewhere for smelting and refining, impacts of port development and operation are not assessed. The assessment does not evaluate impacts of the one or more large-capacity electricity-generating power plants that would be required to power the mine and the port. We recognize that large-scale mine development would induce the development of additional support services for mine employees and their families, vacation homes and other recreational facilities, and transportation infrastructure beyond the main corridor (i.e., airports, docks, and roads). The assessment describes but does not evaluate the effects of induced development resulting from large-scale mining in the region. Direct effects of mining on Alaska Natives and wildlife are not assessed. The assessment also does not include a cost-benefit analysis and does not compare mining to other ongoing activities such as commercial fishing.

Ecological Resources

The Bristol Bay watershed provides habitat for numerous animal species, including at least 29 fish species, more than 40 terrestrial mammal species, and more than 190 bird species. Many of these species are essential to the structure and function of the region's ecosystems and current economies. The Bristol Bay watershed supports several wilderness compatible and sustainable economic sectors, such as commercial, sport, and subsistence fishing; sport and subsistence hunting; and non-consumptive recreation. Considering all these sectors, the Bristol Bay watershed's ecological resources generated nearly $480 million in direct economic expenditures and sales in 2009 and provided employment for over 14,000 full- and part-time workers.

Chief among these ecological resources are world-class commercial and sport fisheries for Pacific salmon and other salmonids. The region's commercial salmon fishery generates the largest component of economic activity. The watershed supports production of all five species of Pacific salmon found in North America: sockeye (*Oncorhynchus nerka*), coho (*O. kisutch*), Chinook (*O. tshawytscha*), chum (*O. keta*), and pink (*O. gorbuscha*) (Figure ES-2). These fishes are anadromous, meaning that they hatch and rear in freshwater systems, migrate to sea to grow to adult size, and return to freshwater systems to spawn and die. Because no hatchery fish are raised or released in the watershed, Bristol Bay's salmon populations are entirely wild.

The most abundant salmon species in the Bristol Bay watershed is sockeye salmon. The watershed supports the largest sockeye salmon fishery in the world, with approximately 46% of the average global abundance of wild sockeye salmon (Figure ES-3). Between 1990 and 2009, 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 25.7 million fish. Approximately half of Bristol Bay's sockeye salmon production is from the Nushagak and Kvichak River watersheds, the main area of focus for this assessment (Figure ES-3).

Chinook salmon are also abundant in the region. 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 much of northwestern Canada, and the Kuskokwim River in southwestern Alaska, just north of Bristol Bay.

Figure ES-2. Reported salmon (sockeye, Chinook, coho, pink, and chum combined) distribution in the South and North Fork Koktuli River and Upper Talarik Creek watersheds. Designation of species spawning, rearing, and presence is based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are believed to be underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year.

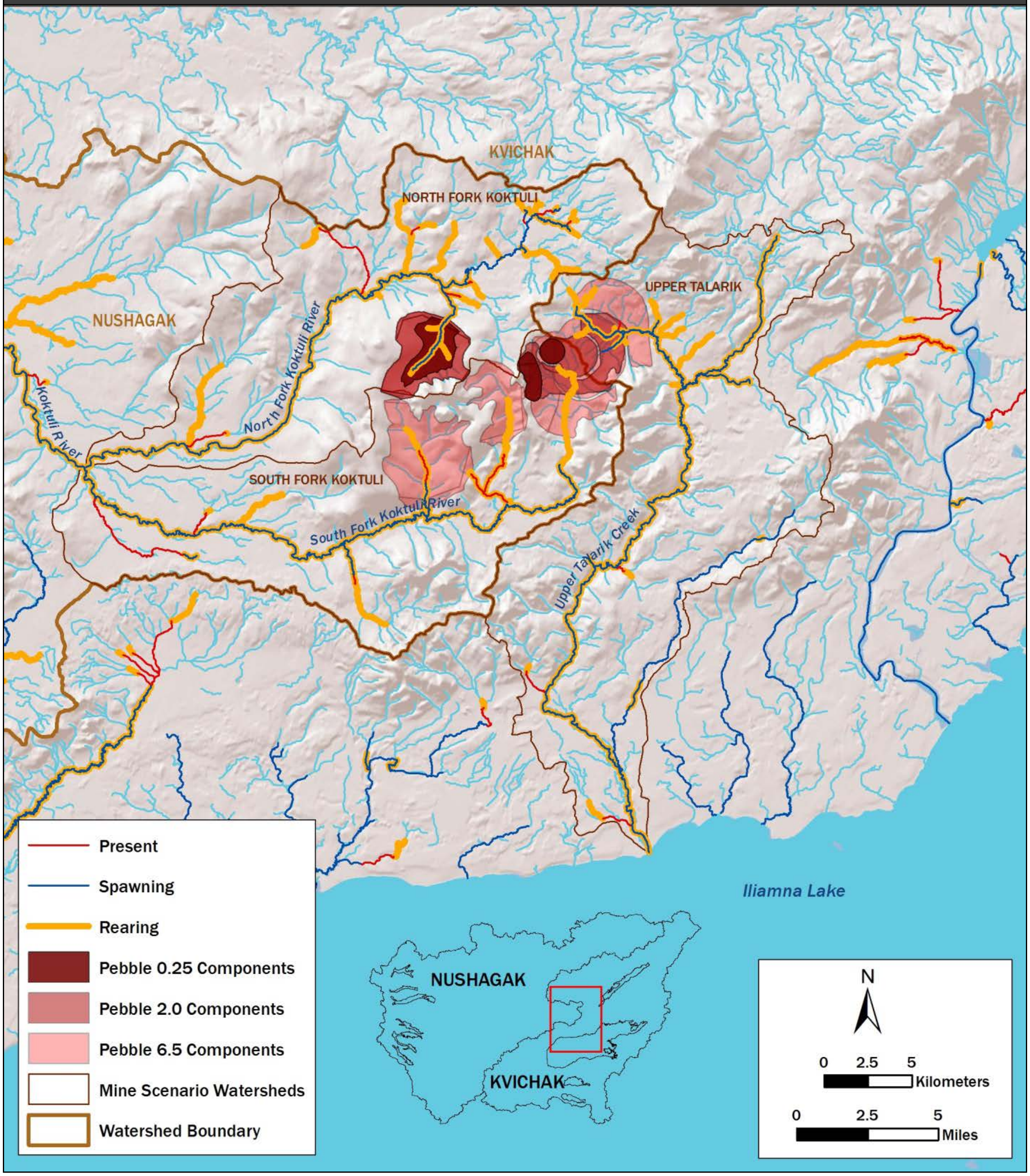
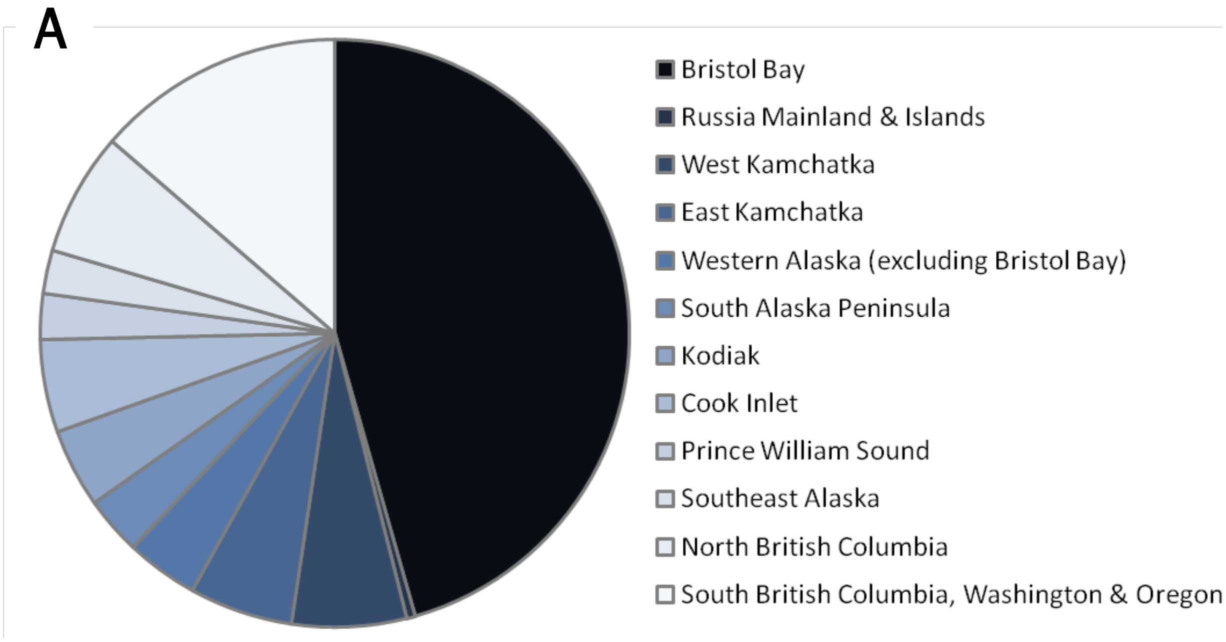
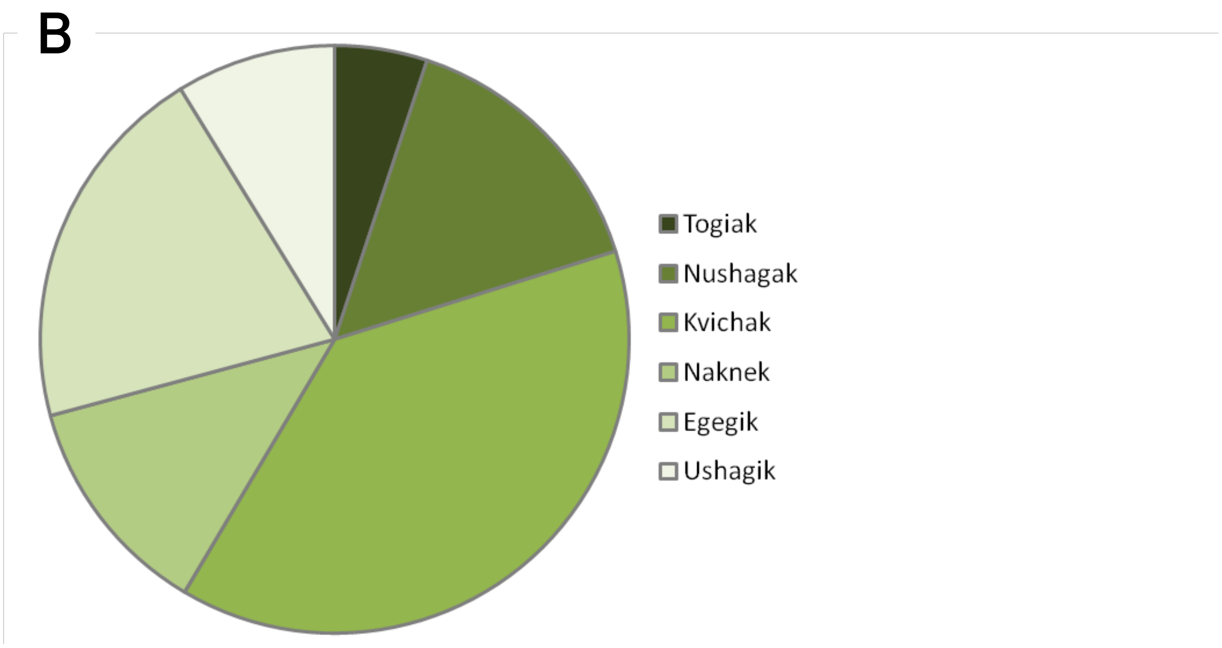


Figure ES-3. Proportion of total sockeye salmon run sizes by (A) region and (B) watershed in the Bristol Bay region. Values are averages from (A) 1956 to 2005 from Ruggerone et al. 2010 and (B) 1956 to 2010 from Baker pers. comm.





The Bristol Bay watershed also supports populations of non-salmon fishes that typically (but not always) remain in the watershed's freshwater habitats throughout their life cycles. The region contains highly productive waters for sport and subsistence fish species, including rainbow trout (*O. mykiss*), Dolly Varden (*Salvelinus malma*), Arctic char (*S. alpinus*), lake trout (*S. namaycush*), Arctic grayling (*Thymallus arcticus*), northern pike (*Esox lucius*), and humpback whitefish (*Coregonus pidschian*). These fishes occupy a variety of habitats in the watershed, from headwater streams to wetlands to large rivers and lakes. The Bristol Bay region is especially renowned for the size and abundance of its rainbow trout: between 2003 and 2007, an estimated 183,000 rainbow trout were caught in the Bristol Bay 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 the watershed's high-quality, diverse aquatic habitats unaltered 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 and thereby increasing the diversity and stability of habitats able to support fish. These factors all contribute to making the Bristol Bay watershed a highly productive system. High aquatic habitat diversity also supports the high genetic diversity of fish populations. This diversity in genetics, life history, and habitat acts to reduce year-to-year variability in total production and increase overall stability of the fishery.

The return of spawning salmon from the Pacific Ocean brings marine-derived nutrients into the watershed and fuels both aquatic and terrestrial foodwebs. Thus, the condition of Bristol Bay's terrestrial ecosystems is intimately linked to the condition of salmon populations, as well as to almost totally undisturbed terrestrial habitats. The watershed continues to support 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 and small mammal species. Brown bears are abundant in the Nushagak and Kvichak River watersheds. Moose also are abundant, particularly in the Nushagak River watershed where felt-leaf willow, a preferred forage species, is plentiful. The Nushagak 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.

Alaska Native Cultures

The predominant Alaska Native cultures present in the Nushagak 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 by development, degraded natural resources, and declining salmon resources. Salmon are integral to these cultures' entire way of life via the provision of subsistence food and subsistence-based livelihoods, and are an important 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 4,000 years and is in part both due to and responsible for the continued undisturbed condition of the region's landscape and biological resources. The respect and importance given salmon and other wildlife, along with traditional knowledge of the environment, have produced a sustainable subsistence-based economy. This subsistence-based way of life is a key element of Alaska Native identity and serves a wide range of economic, social, and cultural functions in Yup'ik and Dena'ina societies.

There are 31 Alaska Native villages in the wider Bristol Bay region, 25 of which are located in the Bristol Bay watershed. Fourteen of these communities are within the Nushagak and Kvichak River watersheds, with a total population of 4,337 in 2010. Thirteen of these 14 communities have federally recognized tribal governments and a majority Alaska Native population. Many of the non-Alaska Native residents in the watersheds have developed cultural ties to the region and they also practice subsistence. Virtually every household in the watersheds uses subsistence resources. In the Bristol Bay region, salmon constitute approximately 52% of the subsistence harvest; for some communities this proportion is substantially higher.

The subsistence-based way of life in many Alaska Native villages is augmented with activities that support cash economy transactions, including commercial fishing. Alaska Native villages, in partnership with Alaska Native corporations and other business interests, are considering a variety of economic development opportunities. Some Alaska Native villages have decided that large-scale mining is not the course they would like to pursue, whereas a few others are seriously considering this opportunity. All are concerned with the long-term sustainability of their communities.

Geological Resources

In addition to significant and valuable ecological resources, the Nushagak and Kvichak River watersheds contain considerable mineral resources. The potential for large-scale mine development in 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 large areas and will necessarily produce large amounts of waste material.

The largest known and most explored deposit is the Pebble deposit. If fully mined, the claim holder estimates that the Pebble deposit would produce more than 11 billion tons of ore, which would make it the largest mine of its type in North America. A mine at the Pebble deposit could ultimately generate revenues between $300 billion to $500 billion over the life of the mine, as well as provide more than 2,000 jobs during mine construction and more than 1,000 jobs during mine operation.

Although the Pebble deposit represents the most imminent site of mine development, other mineral deposits with potentially significant resources exist in the Nushagak and Kvichak River watersheds. Ten specific claims with more than minimal recent exploration (in addition to the Pebble deposit claim) have been filed for copper deposits. Most of these claims are near the Pebble deposit. The potential impacts of large-scale mining considered in this assessment are generally applicable to these other sites.

Mine Scenarios

Like all risk assessments, this assessment is based on scenarios that define a set of possible future activities and outcomes. To assess mining-related stressors that would affect ecological resources in the watershed, we developed realistic mine scenarios that include a range of mine sizes and operating conditions. These mine scenarios are based on the Pebble deposit because it is the best-characterized mineral resource and the most likely to be developed in the near term. The mine scenarios draw on preliminary plans developed for Northern Dynasty Minerals, consultation with experts, and baseline data collected by the Pebble Limited Partnership to characterize the mine site, mine activities, and the surrounding environment. The exact details of any future mine plan for the Pebble deposit or for other deposits in the watershed will differ from our mine scenarios. However, our scenarios reflect the general characteristics of mineral deposits in the watershed, modern conventional mining technologies and practices, the scale of mining activity required for economic development of the resource, and the infrastructure needed to support large-scale mining. Therefore, the mine scenarios evaluated in this assessment realistically represent the type of development plan that would be anticipated for a porphyry copper deposit in the Bristol Bay watershed. Uncertainties associated with the mine scenarios are discussed later in this executive summary.

The three mine scenarios evaluated in the assessment represent different stages of mining at the Pebble deposit, based on the amount of ore processed: Pebble 0.25 (approximately 0.25 billion tons [0.23 billion metric tons] of ore over 20 years), Pebble 2.0 (approximately 2.0 billion tons [1.8 billion metric tons] of ore over 25 years), and Pebble 6.5 (approximately 6.5 billion tons [5.9 billion metric tons] of ore over 78 years). The major parameters of the three mine scenarios are presented in Table ES-1, and their layouts are presented in Figure ES-4. The major components of each mine would be an open mine pit, waste rock piles, and one or more TSFs. Other significant features include plant and ancillary facilities (e.g., a water collection and treatment system, an ore-processing facility, and other facilities associated with mine operations) and the groundwater drawdown zone (the area over which the water table is lowered due to dewatering of the mine pit). An underground extension of the mine, which could increase the size of the mine to 11 billion tons of ore, is not included in this assessment.

Each of these mine scenarios includes a 138-km (86-mile) transportation corridor; 113 km (70 miles) of the corridor would fall within the Kvichak River watershed (Figure ES-5). This corridor would include a gravel-surfaced road and four pipelines (one each for product concentrate, return water, diesel fuel, and natural gas).

The assessment considers risks from routine operation of a mine designed using modern conventional design, practices, and mitigation technologies, assuming no significant human or engineering failures. The assessment also considers various types of failures that have occurred during the operation of other mines and that could occur in this case, including failures of a wastewater treatment plant, a tailings dam, pipelines, and culverts.

Table ES-1. Mine scenario parameters.

Parameter	Mine Scenario		
	Pebble 0.25	Pebble 2.0	Pebble 6.5
Amount of ore mined (billion metric tons)	0.23	1.8	5.9
Approximate duration of mining (years)	20	25	78
Ore processing rate (metric tons/day)	31,100	198,000	208,000
Mine Pit			
Surface area (km^2)	1.5	5.5	17.8
Depth (km)	0.30	0.76	1.24
Waste Rock Pile			
Surface area (km^2)	2.3	13.0	22.6
PAG waste rock (million metric tons)	86	580	4,700
NAG waste rock (million metric tons)	320	2,200	11,000
TSF 1[a]			
Capacity, dry weight (billion metric tons)	0.25	1.97	1.97
Surface area, exterior (km^2)	6.8	16.1	16.1
Maximum dam height (m)	92	209	209
TSF 2[a]			
Capacity, dry weight (billion metric tons)	NA	NA	3.69
Surface area, exterior (km^2)	NA	NA	22.7
TSF 3[a]			
Capacity, dry weight (billion metric tons)	NA	NA	0.96
Surface area, exterior (km^2)	NA	NA	9.82
Total TSF surface area, exterior (km^2)	6.8	16.1	48.6

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

Figure ES-4. Major mine components for the three scenarios evaluated in the assessment. Pebble 0.25 represents 0.25 billion tons of ore; Pebble 2.0 represents 2.0 billion tons of ore; Pebble 6.5 represents 6.5 billion tons of ore. Each mine footprint includes the mine components shown here, as well as the drawdown zone and the area covered by plant and ancillary facilities. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012).

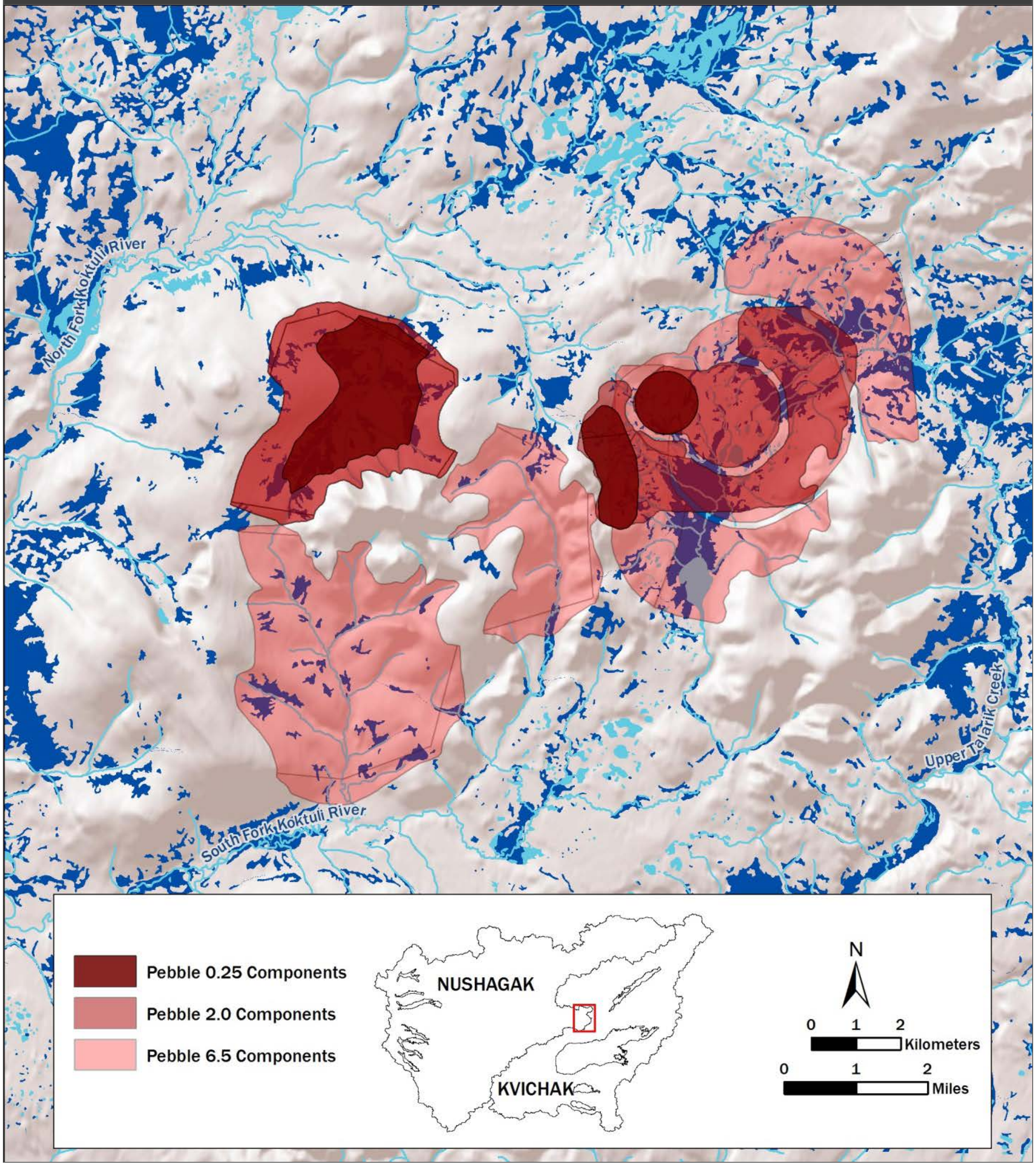
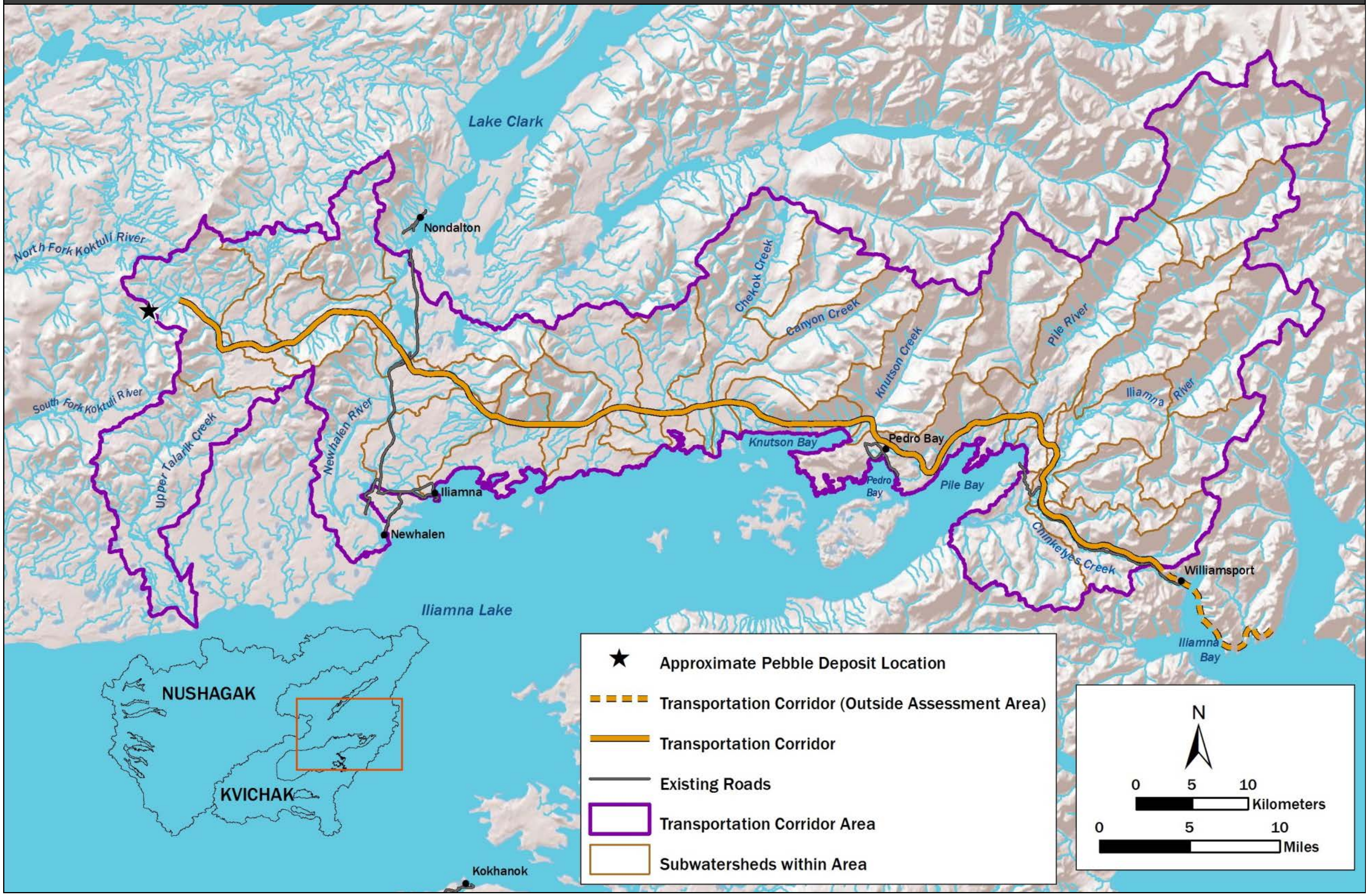


Figure ES-5. The transportation corridor area, comprising 32 subwatersheds in the Kvichak River watershed that drain to Iliamna Lake. Subwatersheds are defined by 12-digit hydrologic unit codes according to the National Hydrography Dataset (USGS 2012).



Risks to Salmon and Other Fishes

Based on the mine scenarios, the assessment defines mining-related stressors that would affect the Bristol Bay watershed's fish and consequently affect wildlife and human welfare. The scenarios include both routine operations (Tables ES-2 and ES-3) and several potential failure scenarios (Table ES-4).

Mine Footprint

Effects on fish resulting from habitat loss and modification would occur directly in the area of mine activity and indirectly downstream because of habitat destruction. These habitat loss estimates are believed to be low due to incomplete delineation of streams, wetlands, and salmon distribution across the region. However, it is possible that careful siting of mine facilities could reduce habitat losses to some degree.

- Due to the mine footprint (the area covered by the mine pit, waste rock piles, TSFs, groundwater drawdown zone, and plant and ancillary facilities), **38, 89, and 151 km (24, 55, and 94 miles) of streams would be lost**—that is, eliminated, blocked, or dewatered—in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively (Table ES-2). This translates to losses of 8, 22, and 36 km (5, 14, and 22 miles) of streams known to provide spawning or rearing habitats for coho salmon, sockeye salmon, Chinook salmon, and Dolly Varden (Table ES-2, Figure ES-6).

- Altered streamflow due to retention and discharge of water used in mine operations, ore processing and transport, and other mine activities would reduce the amount and quality of fish habitat. **Streamflow alterations exceeding 20% would adversely affect habitat in an additional 15, 27, and 53 km (9.3, 17, and 33 miles) of streams** in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively (Table ES-2), reducing production of sockeye salmon, coho salmon, Chinook salmon, rainbow trout, and Dolly Varden. Reduced streamflows would also result in the loss or alteration of an unquantifiable area of riparian floodplain wetland habitat due to loss of hydrologic connectivity with streams.

- Off-channel habitats for salmon and other fishes would be reduced due to **losses of 4.5, 12, and 18 km^2 (1,200, 3,000 and 4,900 acres) of wetlands and 0.41, 0.93, and 1.8 km^2 (100, 230, and 450 acres) of ponds and lakes** to the mine footprints in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively (Figure ES-6). These losses would reduce availability of and access to hydraulically and thermally diverse habitats that provide enhanced foraging opportunities and important rearing habitats for juvenile salmon.

- **Indirect effects of stream and wetland losses** would include reductions in the quality of downstream habitat for coho salmon, sockeye salmon, Chinook salmon, rainbow trout, and Dolly Varden. Although these indirect effects cannot be quantified, such effects would be expected to diminish fish production downstream of the mine site because fish depend on these habitats. Indirect effects would be caused by the following alterations.

- Reduced food resources would result from the loss of organic material and drifting invertebrates from 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 streams less suitable for spawning and rearing.
- Seasonal temperatures could be altered by water treatment and reduced groundwater flowpaths, making streams less suitable for salmonids.

Water Quality

Leakage during Routine Operations

Water from the mine site would enter streams through wastewater treatment plant discharges and in uncollected runoff and leakage of leachates from the waste rock piles and TSFs. Wastewater treatment is assumed to meet all state standards and national criteria, or equivalent benchmarks for chemicals that have no criteria. However, water quality would be diminished by uncollected leakage of tailings and waste rock leachates from the containment system, which would occur during routine operations. Test leachates from the tailings and non-acid-generating waste rocks are mildly toxic. They would require an approximately two-fold dilution to achieve water quality criteria for copper, but are not estimated to be toxic to salmonids. Waste rocks associated with 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. Several metals could be sufficiently elevated to contribute to toxicity, but copper is the dominant toxicant.

Uncollected leachates from waste rock piles and TSFs would elevate instream copper levels and cause direct effects on salmonids ranging from aversion and avoidance of the contaminated habitat to rapidly induced death of many or all fish (Table ES-2). Avoidance of streams by salmonids would occur in 24 and 34 to 57 km (15 and 21 to 35 miles) of streams in the Pebble 2.0 and Pebble 6.5 scenarios, respectively. Rapidly induced death of many or all fish would occur in 12 km (7.4 miles) of streams in the Pebble 6.5 scenario. Copper would cause death or reduced reproduction of aquatic invertebrates in 21, 40 to 62, and 60 to 82 km (13, 25 to 38, and 37 to 51 miles) of streams in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. These invertebrates are the primary food source for juvenile salmon and all life stages of other salmonids, so reduced invertebrate productivity would be expected to reduce fish productivity. These results are sensitive to the assumed efficiency of the leachate capture system, and a more efficient system could be devised. However, greater than 99% capture efficiency would be required to prevent exceedance of the copper criteria for the South Fork Koktuli River in the Pebble 6.5 scenario, which would require technologies beyond those specified in our scenarios or identified in the most recent preliminary mine plan.

Wastewater Treatment Plant Failure

Based on a review of historical and currently operating mines, some failure of water collection and treatment systems would be expected to occur during operation or post-closure periods. A variety of water collection and treatment failures are possible, ranging from operational failures that result in short-term releases of untreated or partially treated leachates to long-term failures to operate water collection and treatment systems in perpetuity. A reasonable but severe failure scenario would involve a complete loss of water treatment and release of average untreated wastewater flows into average dilution flows. In that failure scenario, copper concentrations would be sufficient to cause direct effects on salmonids in 27, 64 to 87, and 74 to 97 km (17, 40 to 54, and 46 to 60 miles) of streams in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. Aquatic invertebrates would be killed or their reproduction reduced in 78 to 100 km (48 to 62 miles) of streams in all three scenarios. In the Pebble 2.0 and 6.5 scenarios, a fish kill would occur rapidly in 3.8 and 31 km (2.4 and 19 miles) of streams, respectively, following treatment failure.

Spillway Release

In the event of TSF overfilling, supernatant water would be released via a spillway. If the water was equivalent to the test tailings supernatant, 2.6 km (1.6 miles) of streams would be avoided by fish and 3.4 to 23 km (2.1 to 14 miles) of streams would be toxic to invertebrates, independent of other sources.

Transportation Corridor

Construction and Routine Operation

In the Kvichak River watershed, the transportation corridor would cross approximately 64 streams and rivers. Of those, 55 are known or likely to support migrating and resident salmonids, including 20 streams designated as anadromous waters at the location of the crossing (Figure ES-7). The corridor would run near Iliamna Lake and cross multiple tributary streams near their confluences with the lake. These habitats are important spawning areas for sockeye salmon, putting sockeye particularly at risk from the road. Diminished habitat quality in streams and wetlands below road crossings would result primarily from altered streamflow, runoff of road salts, and siltation of habitat for salmon spawning and rearing and invertebrate prey production (Tables ES-2 and ES-3).

Culvert Failure

Culverts commonly fail to allow free passage of fish. They can become blocked by debris or ice that may not stop water flow but that create a barrier to fish movement. Fish passage also may be blocked or inhibited by erosion below a culvert that “perches” the culvert and creates a waterfall, by shallow water caused by a wide culvert and periodic low streamflows, or by excessively high gradients. If blockages occurred during adult salmon immigration or juvenile salmon emigration and were not cleared for several days, production of a year-class (i.e., fish spawned in the same year) would be lost from or diminished in the stream above the culvert.

Culverts can also fail to convey water due to landslides or, more commonly, floods that wash out undersized or improperly installed culverts. In such failures, the stream would be temporarily impassible to fish until the culvert is repaired or until erosion re-establishes the channel. If the failure occurs during a critical period in salmon migration, 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. Deposition of silt would smother salmon eggs and alevins if they were present, and would degrade downstream habitat for salmonids and the invertebrates that they eat.

Blockages of culverts could persist for as long as the intervals between culvert inspections. We assume that the transportation corridor would be inspected daily and maintained during mine operation. The level of surveillance along the corridor can be expected to affect the frequency of culvert failure detection. Driving inspections would likely identify a single erosional failure of a culvert that damaged the road, or a debris blockage sufficient to cause water to pool above the road. However, long-term fixes may not be possible until conditions are suitable for culvert replacement, and these fixes may not fully address fish passage, which may be reduced or blocked for longer periods. Extended blockage of migration would be less likely if daily road inspections included stops to inspect each end of each culvert.

After mine operations cease, the road would likely be maintained less carefully by the operator or may be transferred to a government entity that would be expected to employ a more conventional inspection and maintenance schedule. In either case, the proportion of impassable culverts at any one time would be expected to revert to levels found in published surveys of public roads (mean of 48% [range of 30 to 61%] of culverts that had failed and not been repaired when surveyed). Of the approximately 45 culverts that would be required, 36 would be on streams that are believed to support salmonids. Hence, 11 to 22 streams would be expected to have impeded passage of salmon, rainbow trout, or Dolly Varden for an indefinite period of time, and some proportion of those streams would have degraded downstream habitat resulting from sedimentation following washout of the road.

Truck Accidents

Trucks would carry ore processing chemicals to the mine site and molybdenum product concentrate to the port. Truck accident records indicate that truck accidents near streams are likely over the long period of mine operation. These accidents could release sodium ethyl xanthate, cyanide, other process chemicals, or molybdenum product concentrate to streams or wetlands, resulting in toxic effects on invertebrates and fish. However, the risk of spills could be mitigated by using impact-resistant containers.

Tailings Dam Failure

Tailings are the waste materials produced during ore processing. In our scenarios, these wastes would be stored in TSFs consisting of tailings dams and impoundments. The probability of a tailings dam failure increases with the number of dams. The Pebble 0.25 scenario would include one TSF with a single dam, the Pebble 2.0 scenario would include one TSF with three dams, and the Pebble 6.5 scenario would include three TSFs with a total of eight dams. Because their removal is not feasible, the TSFs and their component dams would be in place for hundreds to thousands of years, long beyond the life of the mine. Available reports from the Pebble Limited Partnership suggest a tailings dam as high as 209 m (685 feet) at TSF 1 (Figure ES-8). We evaluated two potential dam failures at TSF 1 in this assessment: one at a volume approximating the complete Pebble 0.25 scenario (92-m dam height) and one at a volume approximating the complete Pebble 2.0 scenario (209-m dam height). In both cases we assumed 20% of the tailings would be released, a conservative estimate that is well within the range of historical tailings dam failures. Failures of the TSF 2 and TSF 3 tailings dams were not analyzed but would be expected to be similar in terms of types of effects.

Table ES-2. Summary of estimated stream lengths potentially affected in the three mine size scenarios, assuming routine operations.

Effect	Stream Length Affected (km)		
	Pebble 0.25	Pebble 2.0	Pebble 6.5
Eliminated, blocked, or dewatered	38	89	151
Eliminated, blocked, or dewatered—anadromous	8	22	36
>20% streamflow alteration[a]	15	27	53
Direct toxicity to fish[a]	0	24	34–57
Direct toxicity to invertebrates[a]	21	40–62	60–82
Downstream of transportation corridor	272		

Notes:

[a] Stream reaches with streamflow alterations partially overlap those with toxicity.

Table ES-3. Summary of estimated wetland, pond, and lake area potentially affected in the three mine size scenarios, assuming routine operations.

Effect	Wetland, Pond, and Lake Area Affected (km^2)		
	Pebble 0.25	Pebble 2.0	Pebble 6.5
Lost to the mine footprint	4.9	13	20
Lost to reduced streamflow below mine footprint	Unquantified		
Filled by roadbed	0.11		
Influenced by the road (within 200 m)	4.7		

Table ES-4. Probabilities and consequences of potential failures in the mine scenarios.		
Failure Type	**Probability[a]**	**Consequences**
Tailings dam	4×10^{-4} to 4×10^{-6} per dam-year = recurrence frequency of 2,500 to 250,000 years[b]	More than 29 km of salmonid stream would be destroyed or degraded for decades.
Product concentrate pipeline	10^{-3} per km-year = 95% 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	1.5×10^{-2} per year = 1 stream-contaminating spill 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	2.6×10^{-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 spill	Same as product concentrate pipeline	Fish and invertebrates would experience acute exposure to toxic water if return water spilled to a stream or wetland.
Diesel pipeline spill	Same as product concentrate pipeline	Acute toxicity would reduce the abundance and diversity of invertebrates and possibly cause a fish kill if diesel spilled to a stream or wetland.
Culvert, operation	Low	Frequent inspections and regular maintenance would result in few impassable culverts, but for those few, blockage of migration could persist for a migration period, particularly for juvenile fish.
Culvert, post-operation	3×10^{-1} to $\sim 6 \times 10^{-1}$ per culvert; instantaneous = 11 to 22 culverts	In surveys of road culverts, 30 to 61% are impassable to fish at any one time. This would result in 11 to 22 salmonid streams blocked at any one time. In 10 to 19 of the 32 culverted streams with restricted upstream habitat, salmon spawning may fail or be reduced and the streams would likely not be able to support long-term populations of resident species.
Truck accidents	1.9×10^{-7} spills per mile of travel = 4 accidents in 25 years and 2 near-stream spills in 78 years	Accidents that spill processing chemicals into a stream or wetland could cause a fish kill. A spill of molybdenum concentrate may also be toxic.
Water collection and treatment, operation	0.93 = proportion of recent U.S. porphyry copper mines with reportable water collection and treatment failures	Water collection and treatment failures could result in exceedance of standards potentially including death of fish and invertebrates. However, these failures would not necessarily be as severe or extensive as estimated in the failure scenario, which would result in toxic effects from copper in more than 60 km of stream habitat.
Tailings storage facility spillway release	No data, but spills are known to occur and are sufficiently frequent to justify routine spillway construction	Spilled supernatant from the tailings storage facility could result in toxicity to invertebrates and fish avoidance for the duration of the event.
Water collection and treatment, managed post-closure	Somewhat higher than operation	Post-closure collection and treatment failures are very likely to result in release of untreated or incompletely treated leachates for days to months, but the water would be less toxic due to elimination of potentially acid-generating waste rock.
Water collection and treatment, after site abandonment	Certain, by definition	When water is no longer managed, untreated leachates would flow to the streams. However, the water may be less toxic.

[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 5×10^{-4} per year or a recurrence frequency of 2,000 years).

Figure ES-6. Streams and wetlands lost (eliminated, blocked, or dewatered) in the Pebble 6.5 scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012).

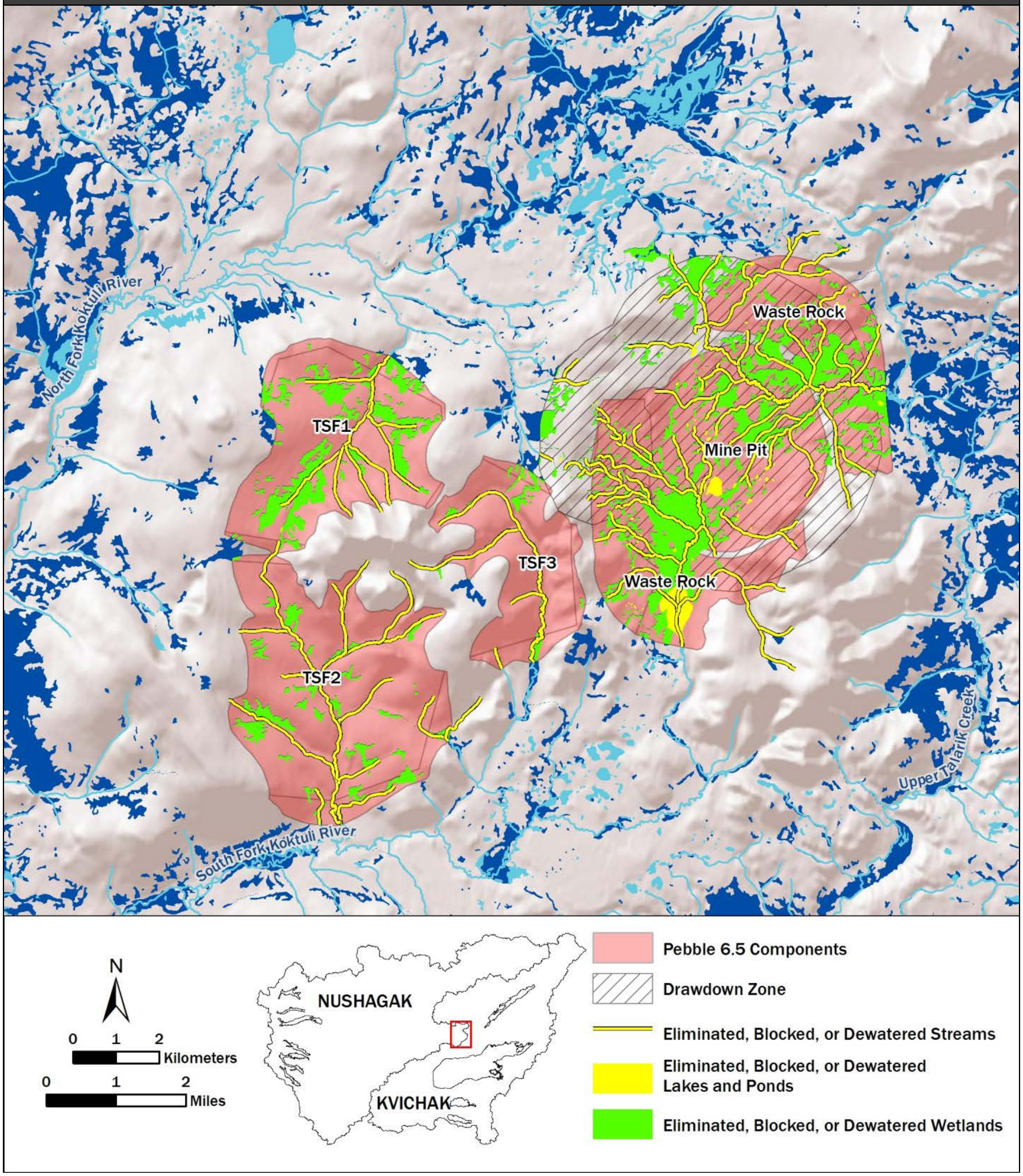


Figure ES-7. Reported salmon, Dolly Varden, and rainbow trout distribution along the transportation corridor. Salmon presence data are from the Anadromous Waters Catalog (Johnson and Blanche 2012); Dolly Varden and rainbow trout presence data are from the Alaska Freshwater Fish Inventory (ADF&G 2012). Note that rainbow trout have also been documented in the Iliamna River and Chinkelyes Creek, although these points are not indicated on this map.

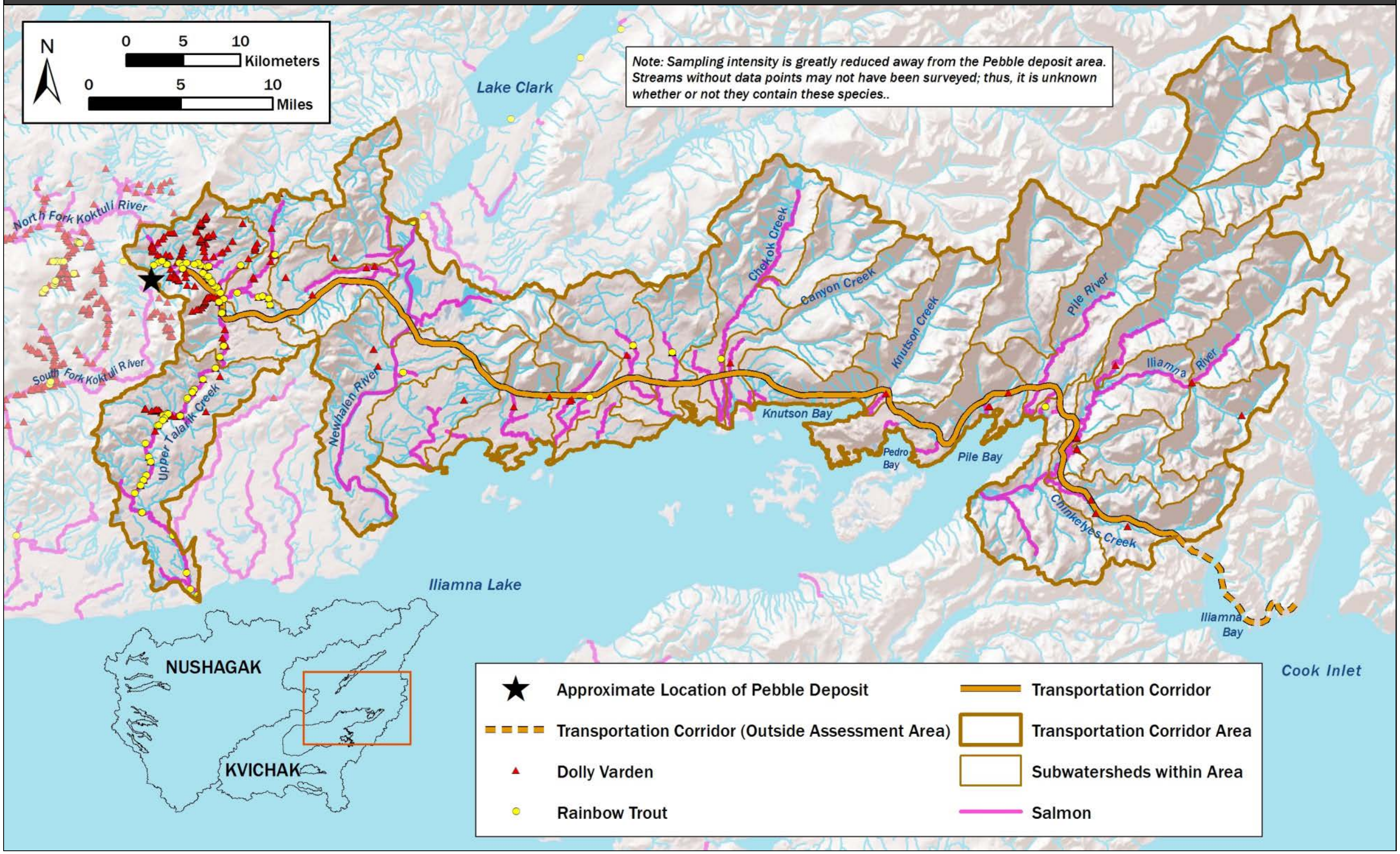
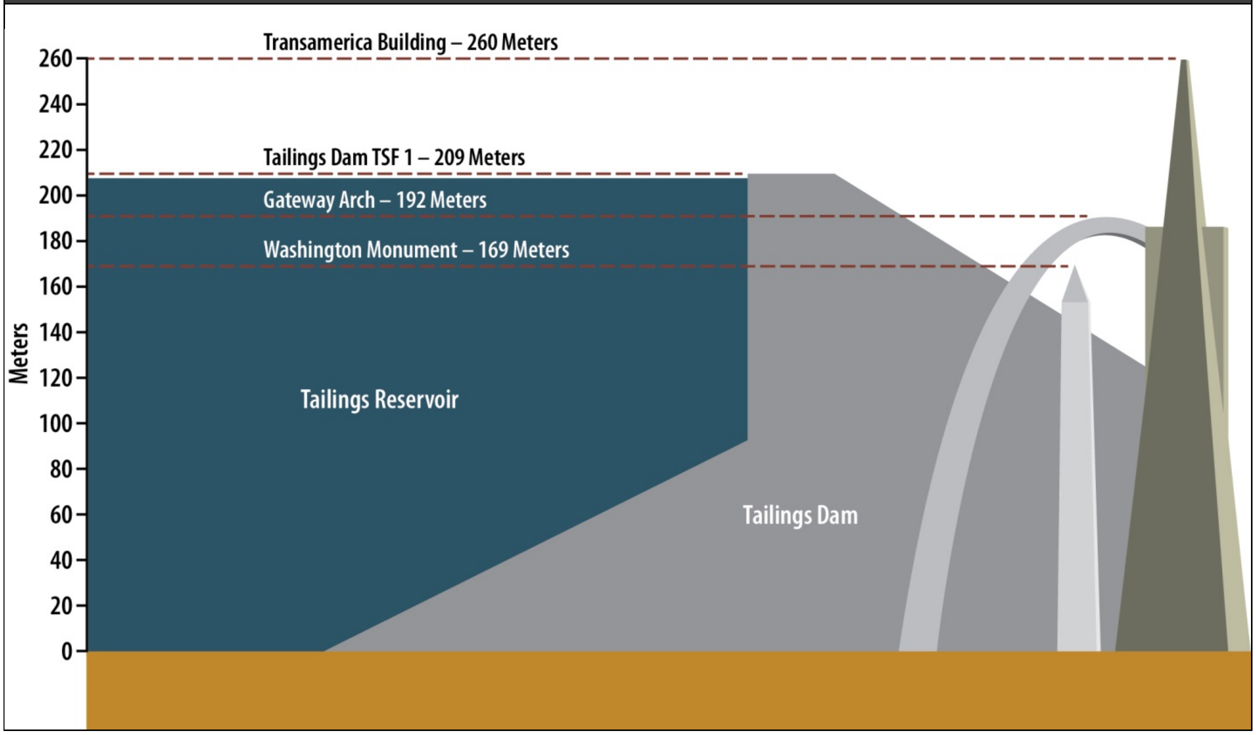


Figure ES-8. Height of the dam at tailings storage facility (TSF) 1 in the Pebble 2.0 and Pebble 6.5 scenarios, relative to U.S. landmarks.



The range of estimated dam failure probabilities is wide, reflecting the great uncertainty concerning such failures. The most straightforward method of estimating the annual probability of a tailings dam failure 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. Strictly speaking, these frequencies are properties that apply to a group of dams. However, by extension, if there is one dam and it is typical of the population, it would be expected to fail, on average, within a 2,000-year period. This does not mean it is expected to fail 2,000 years after it is built. Rather, it indicates that, after 2,000 years have passed, it is more likely than not that the dam would have failed and that expected failure could occur any year in that 2,000-year window with an average annual probability of 0.0005.

The argument against this method is that the record of past failures 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 the range assumed here (Table ES-4). 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), the Federal Energy Regulatory Commission (FERC), and other agencies. Based on safety factors in USACE and FERC guidance, we estimate that the probability of failure for all causes requires a minimum factor of safety of 1.5 against slope instability for the loading condition corresponding to steady seepage from the filled storage facility. An assessment of the correlation of dam failure probabilities with slope instability safety factors suggests an annual probability of failure of 1 in 250,000 per year for facilities designed, built, and operated with state-of-the-practice engineering (Category I facilities) and 1 in 2,500 per year for facilities designed, built, and operated using standard engineering practice (Category II facilities). 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 potential dam heights and subarctic conditions in these scenarios.

Failure of the dam at TSF 1 (the TSF included in all three mine scenarios) would result in the release of a flood of tailings slurry into the North Fork Koktuli River. This flood would scour the valley and deposit many meters of tailings fines in a sediment wedge across the entire valley near the TSF dam, with lesser quantities of fines deposited as far as the North Fork's confluence with the South Fork Koktuli River. The North Fork Koktuli River currently supports spawning and rearing populations of sockeye, coho, and Chinook salmon; spawning populations of chum salmon; and rearing populations of Dolly Varden and rainbow trout. The tailings slurry flood would continue down the mainstem 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 in the assessment would be expected to have the following severe direct and indirect effects on aquatic resources, particularly salmonids.

- **It is expected that the North Fork Koktuli River below the TSF 1 dam and much of the mainstem Koktuli River would not support salmonids in the short term (less than 10 years).**
    - In the tailings dam failure scenarios, spilled tailings would bury salmon habitat under meters of fines along nearly the entire length of the North Fork Koktuli River valley downstream of the dam (over 29 km or 18 miles in the Pebble 0.25 dam failure scenario), and beyond (in the Pebble 2.0 dam failure scenario).
    - 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, but existing data concerning toxicity to fish are less clear.
    - Deposited tailings would continue to erode from the North Fork Koktuli River and mainstem Koktuli River valleys.
    - Suspension and redeposition of tailings would be expected to cause serious habitat degradation in the mainstem Koktuli River and downstream into the Mulchatna River; however, the extent of these effects cannot be estimated at this time due to model and data limitations.
- **The affected streams would provide 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 the failed dam.

- Ultimately, spring floods and stormflows would carry some 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.

- **Near-complete loss of North Fork Koktuli River fish populations downstream of the TSF and additional fish population losses in the mainstem Koktuli, Nushagak, and Mulchatna Rivers would be expected to result from these 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 190,000 fish.
  - A tailings spill could eliminate 29% or more 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 1.9 million fish. The proportion of sockeye and other salmon species of Koktuli-Mulchatna origin is unknown.
  - Similarly, the North Fork Koktuli River populations of rainbow trout and Dolly Varden would be lost for years to decades if they could not successfully be maintained entirely in headwater networks upstream of the affected zone. Quantitative estimates of these losses are not possible given available information.

Effects would be qualitatively similar for both the Pebble 0.25 and Pebble 2.0 dam failures, although effects from the Pebble 2.0 dam failure would extend farther and last longer. Failure of dams at the two additional TSFs in the Pebble 6.5 scenario (TSF 2 and TSF 3) were not modeled, but would have similar types of effects in the South Fork Koktuli River and downstream rivers.

Pipeline Failure

In the mine scenarios, the primary mine product would be a sand-like copper concentrate with traces of other metals, which would be pumped via pipeline to a port on Cook Inlet. Water that carried the concentrate would be returned to the mine site in a second pipeline. Based on the general record of pipelines and further supported by the record of metal concentrate pipelines at existing mines, one near-stream failure and two near-wetland failures of each of these pipelines would be expected to occur over the life of the Pebble 6.5 scenario (approximately 78 years).

Failure of either the product or the return water pipeline would release water that is expected to be highly toxic due to dissolved copper and possibly processing chemicals. Invertebrates and potentially early fish life stages would be killed in the affected stream over a relatively brief period. If concentrate spilled into a stream, it would settle and form highly toxic bed sediment based on its high copper content and acid generation. The mean velocities of many streams crossed by the pipelines are sufficient to carry the concentrate downstream to Iliamna Lake, but some would collect in low-velocity areas of the receiving stream. If the spill occurred during low streamflows, dredging could recover some concentrate but would cause physical damage to the stream. Concentrations in Iliamna Lake could not be predicted, but near the pipeline route Iliamna Lake contains important sockeye salmon beach spawning areas that would be exposed to a spill. Sockeye also spawn in the lower reaches of streams that could be directly contaminated by a spill.

Based on petroleum pipeline failure rates, the diesel fuel pipeline also would be expected to spill near a stream over the life of the Pebble 6.5 mine. Evidence from modeling the dissolved and dispersed oil concentrations in streams, laboratory tests of diesel toxicity, and studies of actual spills in streams indicates that a diesel spill at a stream crossing would be expected to immediately kill invertebrates and likely fish as well. Remediation would be difficult but recovery would be expected to occur within 3 years. Failure of the natural gas pipeline would also be expected, but significant effects on fish would not be expected.

Spills into wetlands that support fish would be expected to have greater toxic effects because contaminants would be washed out slowly, if at all. However, retention of contaminants within the wetland would make remediation by removal more practical.

Common Mode Failures

Multiple, simultaneous failures could occur due to a common event, such as a severe storm with heavy precipitation (particularly precipitation that fell on spring snow cover) or a major earthquake. Over the long period that tailings impoundments, a mine pit, and waste rock piles 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 that facilities remaining in place will weaken and eventually fail.

Such an event could cause multiple tailings dam failures that would spill tailings slurry into both the South and North Fork Koktuli 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 remedial responses more difficult.

Fish-Mediated Risks to Wildlife

Although the effects of salmonid reductions on wildlife—that is, fish-mediated risks to wildlife—cannot be quantified given available data, some reduction in wildlife would be expected in the mine scenarios. Changes in the occurrence and abundance of salmon have the potential to change animal behavior and reduce wildlife population abundances. The mine footprints would be expected to have local effects on brown bears, wolves, bald eagles, and other wildlife that consume salmon, due to reduced salmon abundance from habitat loss and degradation in or immediately downstream of the mine footprint. Any of the accidents or failures evaluated would increase effects on salmon, which would further reduce the abundance of their predators.

The abundance and production of wildlife also is enhanced by the marine-derived nutrients that salmon carry upstream on their spawning migration. These 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 terrestrial vegetation that, in turn, provides food for moose, caribou, and other wildlife. The loss of these nutrients due to a reduction in salmon would be expected to reduce the production of riparian and upland species.

Fish-Mediated Risks to Alaska Native Cultures

Under routine operations with no major accidents or failures, the predicted loss and degradation of salmonid habitat in the South and North Fork Koktuli Rivers and Upper Talarik Creek would be expected to have some impact on Alaska Native cultures of the Nushagak and Kvichak River watersheds. Fishing and hunting practices would be expected to change in direct response to the stream, wetland, and terrestrial habitats lost to the mine footprints and the transportation corridor. It is also possible that subsistence use of salmon resources would decline based on perceptions of reduced fish or water quality resulting from mining.

The potential for significant effects on Alaska Native cultures is much greater from mine failures that reduced or eliminated fish populations in affected areas, including areas significant distances downstream from the mine. In the case of the tailings dam failures described in the assessment, the significant loss of Chinook salmon populations would have severe consequences, especially for villages in the Nushagak River watershed.

Any loss of fish production from these failures would reduce the availability of these subsistence resources to local Alaska Native villages, and the reduction of this highly nutritious food supply could have negative consequences on human health. Because salmon-based subsistence is integral to Alaska Native cultures, the effects of salmon losses go beyond the loss of food resources. If salmon quality or quantity was (or was perceived to be) adversely affected, the nutritional, social, and spiritual health of Alaska Natives would decline.

Cumulative Risks of Multiple Mines

This assessment has focused on the effects that a single large mine at the Pebble deposit would have on salmon and other resources in the Nushagak and Kvichak River watersheds, including the cumulative effects of multiple stressors associated with that mine. However, multiple mines and their associated infrastructure may be developed in these watersheds. Each mine would pose risks similar to those identified in the mine scenarios. Estimates of the stream and wetland habitats lost would differ across different deposits, based on the size and location of mine operations within the watersheds. Individually, each mine footprint would eliminate some amount of fish-supporting habitat and, should operator or engineering failures occur, affect fish habitats well beyond the mine footprint.

We considered development of mines at the Pebble South/PEB, Big Chunk South, Big Chunk North, Groundhog, AUDN/Iliamna, and Humble claims in the Nushagak and Kvichak River watersheds. These sites were chosen because all contain copper deposits that have generated exploratory interest. If all six mine sites were developed, the cumulative area covered by these six mine footprints could be 37 to 57 km^2 (9,100 to 14,000 acres). Stream habitats eliminated or blocked could be 43 to 70 km (27 to 43 miles). Cumulative wetland losses could be 7.9 to 27 km^2 (2,000 to 6,700 acres).

These are conservative estimates of habitat loss, because we did not estimate the hydrologic drawdown zones around each mine pit as was done for the Pebble scenarios. Inclusion of the drawdown area in the Pebble 0.25 scenario increased the area of stream and wetlands losses by roughly 50%. A similar increase might be expected at the other mine sites, depending on local geology. These mines also would be expected to modify streamflows and diminish water quality to approximately the same extent as the Pebble 0.25 scenario. Waters on these claim blocks include the Chulitna River and Rock, Jensen, Yellow, Napotoli, Klutuk, and Kenakuchuk Creeks, as well as over 250 unnamed tributaries and over 50 unnamed lakes and ponds. Although not all support salmon, many do. Loss of substantial habitat across the watersheds could contribute to diminishing the genetic diversity of salmon stocks and thereby increasing annual variability in salmon returns.

Mitigation and Remediation

The mine scenarios assessed here include modern conventional mitigation practices as reflected in Northern Dynasty Mineral's published plan for the Pebble deposit, plus practices suggested in the mining literature and consultations with experts. These practices include, but are not limited to, processing all potentially acid-generating waste rock before closure, managing effluent water temperatures, inspecting and maintaining roads daily, and providing automatic monitoring and remote shut-off for the pipelines. However, we recognize that risks could be further reduced by unconventional or even novel mitigation measures, such as dry stack tailings disposal or the use of armored containers on the trucks carrying process chemicals to the site. These practices may be unconventional because they are expensive, unproven, or impractical. However, these obstacles to implementation might be overcome and justified by the large mineral resource and the highly valued natural and cultural resources of the Bristol Bay watershed.

Although remediation would be considered if spills contaminated streams, features of the Pebble deposit area would make remediation difficult. Spilled tailings from a dam failure would flow into streams, rivers, and floodplains that are in roadless areas and that are not large enough to float a barge-mounted dredge. Recovery, transport, and disposal of hundreds of millions of metric tons of tailings under those conditions would be extremely difficult and would result in additional environmental damage. Compensatory mitigation measures could offset some of the stream and wetland losses, although there are substantial challenges regarding the efficacy of these measures to offset adverse impacts. Pipeline crossings of streams would be near Iliamna Lake, so the time available to block or collect spilled material before it reached the lake would be short. Spilled return water and the aqueous phase of the product concentrate slurry would be unrecoverable. The product concentrate itself would resemble fine sand, and mean velocities in many receiving streams would be sufficient to suspend and transport it. Hence, concentrate spilled or washed into streams could be recovered only where it collected in low-velocity locations. Diesel spills would dissolve, vaporize, and flow as a slick to Iliamna Lake. Booms and absorbents are not very effective in moderate- to high-velocity streams.

Summary of Uncertainties in Mine Design and Operation

This assessment considers realistic mine scenarios that are based on specific characteristics of the Pebble deposit and preliminary plans proposed by Northern Dynasty Minerals. These scenarios are generally applicable to copper deposits in the Bristol Bay watershed. If the Pebble deposit is mined, actual events will undoubtedly deviate from these scenarios. This is not a source of uncertainty, but rather an inherent aspect of a predictive assessment. Even an environmental assessment of a specific plan proposed for permitting by a mining company would be an assessment of a scenario that undoubtedly would differ from actual mine development.

Multiple uncertainties are inherent in planning, designing, constructing, operating, and closing a mine. These uncertainties, summarized below, are inherent in any complex enterprise, particularly when that enterprise involves an incompletely characterized natural system. However, the large spatial scales and long durations required to mine the Pebble deposit make these inherent uncertainties more prominent.

- Mines are complex systems requiring skilled engineering, design, and operation. The uncertainties facing mining and geotechnical engineers include unknown geological features, uncertain values of geological properties, limited knowledge of mechanisms and processes, and human error in design and construction. Models used to predict the behavior of engineered systems represent idealized processes and by necessity contain simplifications and approximations that potentially introduce errors.
- Accidents are unplanned and inherently unpredictable. Although 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 due to 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 apparently reasonable operation and mitigation plans. Climate change will likely exacerbate this uncertainty. In the Bristol Bay region, climate change is expected to lead to changes in snowpack and the timing of snowmelt, an increased chance of rain-on-snow precipitation, and increased flooding. All of these changes are likely to affect multiple aspects of any large-scale mining in the area, including mine infrastructure, the transportation corridor, water treatment and discharge, and post-closure management, in unknown and potentially unpredictable ways.

- The ore deposit would be mined for decades and wastes would require management for centuries or even in perpetuity. Engineered mine waste storage systems have been in existence for only about 50 years, and their long-term behavior is not known. The response of current technology in tailings dam construction is untested and unknown in the face of centuries of unpredictable events such as extreme weather and earthquakes.
- Over the long time span (centuries) of mining and post-mining care, generations of mine operators must exercise due diligence. Priorities could 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.

Summary of Uncertainties and Limitations in the Assessment

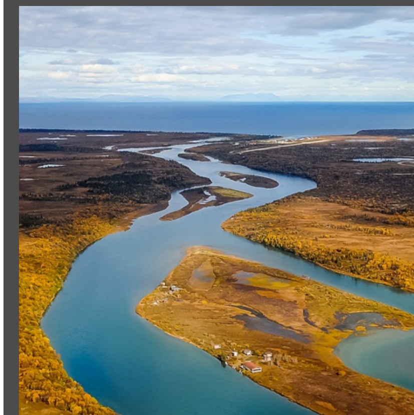
The most important uncertainties concerning estimated effects of the mine scenarios, as judged by the assessment authors, are identified below.

- Consequences of habitat loss and degradation for fish populations could not be quantified because of the lack of quantitative information concerning salmonid populations in freshwater habitats. The occurrence of salmonid species in rivers and major streams is known, but detailed and comprehensive information on abundances, productivities, and limiting factors in each of the watersheds is not available. Estimating fish population changes would require population modeling, which requires knowledge of life-stage-specific survival and production and limiting factors and processes. Further, it requires knowledge of how temperature, habitat structure, prey availability, density dependence, and sublethal toxicity influence life-stage-specific survival and production. Obtaining this information would require more detailed monitoring and experimentation. Salmon populations naturally vary in size due to many factors that vary among locations and years. At present, data are insufficient to establish reliable salmon population estimates, and obtaining such data would take many years. Estimated effects of mining on fish habitat thus become the best 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 waste rock piles, tailings impoundments, or tailings deposited in streams and on their floodplains.
- Leachate capture efficiencies are uncertain. We assume 50% capture for waste rock leachates outside of the mine pit drawdown zone. In the Pebble 2.0 scenario, for example, this would result in capture of 84% of the leachate by the pit drawdown zone and the wells combined. To avoid exceeding water quality criteria for copper, more than 99% capture would be required.
- The quantitative 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 estimated annual probability of tailings dam failure is uncertain because it is based on design goals. Historical experience is presumed to provide an upper bound of failure probability. Features that should reduce failure frequencies have not been tested for the thousands of years that they must function properly. Hence, actual failure rates could be higher or lower than the estimated probability.
- The proportion of 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 spilled tailings in the event of a dam failure could not be quantified. It is expected that tailings would erode from areas of initial deposition and move downstream over more than a decade. However, the data needed to model that process and the resources needed to develop that model are not available.
- The actual response of Alaska Native cultures to any impacts of the mine scenarios 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 be expected to include some degree of cultural disruption. It is not possible to predict specific changes in demographics, cultural practices, or physical and mental health.
- Because we mention but do not evaluate potential direct effects of mining on wildlife or on Alaska Natives, this assessment represents a conservative estimate of how these endpoints would be affected by mine development and operation.

Uses of the Assessment

This assessment is a scientific investigation. It does not reflect any conclusions or judgments about the need for or scope of potential government action, nor does it offer or analyze options for future decisions. Rather, it is intended to provide a characterization of the biological and mineral resources of the Bristol Bay watershed, increase understanding of the risks from large-scale mining to the region's fish resources, and inform future government decisions. The assessment will also better inform dialogues among interested stakeholders concerning the resources in the Bristol Bay watershed and the potential impacts of large-scale mining on those resources.



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 significant 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 way of life for at least 4,000 years.

Public interest in the Bristol Bay watershed has centered on the ecological goods and services provided by the watershed and on potential mining activity. The watershed is most noted for its abundant fish resources. The Bristol Bay watershed supports production of all five species of Pacific salmon found in North America (sockeye, Chinook, chum, coho, and pink), including almost half of the world's commercial sockeye salmon harvest. In 2009, Bristol Bay's ecosystems, which support the watershed's commercial, recreational, and subsistence fisheries, generated $480 million in direct economic expenditures in the region, and provided employment for over 14,000 full- and part-time workers (Appendix E). This consistently large fish production results from the watershed's high hydrologic diversity and pristine quality, both of which contribute to highly diverse fish populations.

In addition to these biological resources, 16 mine claim blocks have recently been explored in the Bristol Bay's Nushagak and Kvichak River watersheds. Eleven of these claims are associated with porphyry copper deposits, the largest belonging to the Pebble Limited Partnership (PLP). This partnership was created in 2007 by co-owners Northern Dynasty Minerals, Ltd. and Anglo American to design, permit, construct, and operate a long-life mine at the Pebble deposit (PLP 2013) (Anglo American withdrew from the partnership in late 2013). Although PLP has not yet submitted a permit application for a mine, preliminary mine plans have been developed and 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 require the creation of a large open pit, the production of large amounts of waste rock and mine tailings, the creation of a transportation corridor connecting the deposit area to Cook Inlet, and the development of a deep-water port. Revenues from such a mine have been estimated at between $300 billion and $500 billion over the mine's life (Chambers et al. 2012), and more than 2,000 and 1,000 jobs could be created during mine construction and operation, respectively (Ghaffari et al. 2011).

In light of these factors, nine Bristol Bay federally recognized tribes, the Bristol Bay Native Association, the Bristol Bay Native Corporation, other tribal organizations, and many groups and individuals petitioned the U.S. Environmental Protection Agency (USEPA) to use its authorities under Clean Water Act (CWA) Section 404(c) to restrict or prohibit the disposal of dredged or fill material associated with large-scale mining activities in the Bristol Bay watershed. These groups are concerned that large-scale mining could adversely affect the region's valuable natural resources, particularly its fisheries. Four Bristol Bay federally recognized tribes, other tribal organizations, the governor of Alaska, and other groups and individuals, including PLP, have asked USEPA to wait until formal mine permit applications have been submitted and an environmental impact statement has been developed.

USEPA initiated this assessment in response to these competing requests. The assessment's purpose is to characterize the biological and mineral resources of the Bristol Bay watershed, increase understanding of the potential impacts of large-scale mining, in terms of both day-to-day operations and potential accidents and failures, on the region's fish resources, and inform future decisions, by government agencies and others, related to protecting and maintaining the chemical, physical, and biological integrity of the watershed. The assessment represents a review and synthesis of available information to identify and evaluate potential risks of future large-scale mining development on the Bristol Bay watershed's fish habitats and populations and consequent indirect effects on the region's wildlife and Alaska Native cultures.

1.1 Assessment Approach

This assessment was conducted as an ecological risk assessment (ERA). ERA is a scientific process used to determine whether exposure to one or more stressors may result in adverse ecological effects, the findings of which are used to inform environmental decision making. USEPA routinely uses ERA methods to evaluate the potential impacts of current and future actions when considering management decisions (USEPA 2002a, 2002b, 2002c, 2008, 2011). USEPA is conducting this assessment consistent with its authority under CWA Section 104. CWA Sections 104(a) and (b) provide USEPA with the authority to study the resources of the Bristol Bay watershed, evaluate the effect of pollution from large-scale mining development on those resources, and make such an assessment available to the public. This assessment is not an environmental impact assessment, an economic or social cost-benefit analysis, or an assessment of any one specific mine proposal.

Risk assessors, decision makers, and community stakeholders determine the topical, spatial, and temporal scope needed to effectively address the decisions the ERA is informing. Within this scope, risk assessments consider the potential effects of an activity and use one or more scenarios, or sets of assumptions, to identify how resources of interest (in this case, fish habitat and populations) could be exposed to stressors generated by some activity (in this case, porphyry copper mining).

Assessment endpoints are explicit expressions of the environmental resources of interest to the risk assessors, decision makers, and stakeholders. We selected fish, and specifically salmon and other salmonids, as our primary assessment endpoint because of their critical importance to stakeholders and future decision making in the watershed. The sustainability of the Bristol Bay fisheries is a concern shared by all Bristol Bay stakeholders—including those who support mining—and the ecological, economic, and cultural importance of the region's commercial, sport and subsistence fisheries has been emphasized consistently by all stakeholders throughout the process. Our preliminary technical consultations with federal, state, and tribal representatives indicated that evaluating the potential risks of large-scale mining on the region's fishery resources was a top priority. During our public engagement efforts, stakeholders consistently emphasized that fish are the crucial resource of concern.

We also considered two key secondary endpoints: wildlife and Alaska Native cultures. Fish-mediated effects on wildlife were considered because fish, particularly salmon, are an important food resource for wildlife, via both direct consumption and as a source of marine-derived nutrients that contribute to the watershed's overall productivity. Fish-mediated effects on Alaska Natives were considered because sustainability of the region's fish populations is critical to the future of Alaska Natives in the Bristol Bay region, and because concern about the region's fishery resources prompted the original requests from Alaska Natives that USEPA examine potential mine development in the Bristol Bay watershed.

Multiple geographic scales are considered in the assessment. Background and characterization information is presented for the entire Bristol Bay watershed. The evaluation of potential large-scale mining impacts focuses on the Nushagak and Kvichak River watersheds. These two watersheds are the most likely to be affected by large-scale mining development, given the location of current mine claims and current federal and state restrictions on development in other portions of the Bristol Bay watershed. These two watersheds are responsible for approximately half of the Bristol Bay sockeye salmon production, and are also home to approximately half of the region's Alaska Native communities and federally recognized tribes. There are 31 federally recognized tribes in the larger Bristol Bay region, and 25 of these tribal communities are within the Bristol Bay watershed boundary defined in this assessment. Fourteen of these communities (13 of which have federally recognized tribes) are within the Nushagak and Kvichak River watersheds.

The assessment also considers smaller geographic scales for risk analysis and characterization. Because the Pebble deposit is the largest known and most explored deposit in the region, we use it as a case study for potential risks. Because none of the parties holding mine claims in the Bristol Bay watershed has submitted a formal application and mine plan, we developed a set of realistic mine scenarios for the assessment. The foundations for these scenarios are industry documents outlining approaches for mining porphyry copper deposits, as well as specific documents from the PLP outlining a basic, preliminary mine plan for the Pebble deposit. The mine scenarios were used to complete the risk analyses and characterizations in the assessment. Although these mine scenarios were developed for the Pebble deposit, the potential risks evaluated are expected to be qualitatively similar to potential risks associated with any mine of the same resource type (porphyry copper) anywhere in the Nushagak and Kvichak River watersheds.

Risk assessments are inherently uncertain, because they must predict the occurrence and consequences of future actions. In this assessment, expressions of uncertainty are treated differently for accidents and failures and for routine operations. Risks of accidents and failures are based on empirical frequencies (summarized in Table 14-1), but we acknowledge the possibility of lower risks due to advances in technology or practices. For example, data concerning risks of culvert failure provide frequencies of 0.3 to 0.6 per culvert. However, risks during operation are simply described as low, because our scenario specifies daily inspections and there are no data quantifying failure rates under such intensive maintenance programs. Risks that effects would occur due to routine operations are not described probabilistically, because they are unintended results of planned actions. However, these risks are uncertain due to lack of knowledge about the receiving environment and its response to mining activities. Those uncertainties are described based on the professional judgment of the authors using ordinary language such as "likely" and, when the evidence allows, in terms of possible deviations from expectations (e.g., thresholds for effects could be at least a factor of 2 lower). The term "likely" is used commonly as an abbreviation for "more likely than not" (>0.5 probability). The risk of a tailings storage facility (TSF) spillway release is different in that it is a hybrid between a failure (TSFs should not overflow during mine operation) and routine operations (spillways are installed to spill excess water, because overflows are a reasonable expectation). No statistics are available on overflow frequencies, but they are judged to be likely over the life of a mine and inevitable afterwards, if water treatment is not continued in perpetuity.

Throughout the assessment, we have reached out to interested parties to ensure transparency of the assessment process (Box 1-1). Through public comment opportunities and by engaging an Intergovernmental Technical Team (IGTT) of federal, state, and tribal representatives, we were able to identify additional information helpful for characterizing the biological and mineral resources of the watershed. These interactions with community members were also helpful in narrowing the scope of the assessment to issues that were most important to stakeholders.

BOX 1-1. STAKEHOLDER INVOLVEMENT IN THE ASSESSMENT

Meaningful engagement with stakeholders was essential to ensure that the U.S. Environmental Protection Agency (USEPA) heard and understood the full range of perspectives on the assessment and the potential effects of mining in the region. USEPA involved and informed stakeholders throughout the assessment process. Community involvement efforts included a project webpage and listserv to ensure that assessment-related information is shared with the public. Additional ways in which stakeholders and tribal governments were involved in the assessment process are summarized below.

- **Public and stakeholder meetings.** Throughout development of the assessment, USEPA visited many Bristol Bay communities, including Ekwok, Dillingham, Kokhanok, New Stuyahok, Koliganek, Iliamna, Newhalen, Nondalton, Naknek, King Salmon, Igiugig, and Levelock. USEPA also met with representatives from Bristol Bay tribal governments and corporations, as well as organizations representing the mine industry, commercial fishers, seafood processors, hunters and anglers, chefs and restaurant owners, jewelry companies, conservation interests; members of the faith community; and elected officials from Alaska and other states. USEPA heard from hundreds of people at these meetings and from thousands more via phone and email. USEPA was also invited to numerous conferences and meetings to discuss the assessment.
- **Intergovernmental Technical Team (IGTT).** In August 2011, USEPA met with the IGTT, which was established to provide USEPA with input on the structure of the assessment and to identify potential data sources. IGTT participants included tribal representatives from Ekwok, Newhalen, Iliamna, South Naknek, New Koliganek, Curyung, Nondalton, and Levelock and agency representatives from the Alaska Department of Public Health, the Alaska Department of Fish and Game, the National Park Service, the U.S. Fish and Wildlife Service, the National Oceanic and Atmospheric Administration, and the Bureau of Land Management. Feedback from this workshop was used to inform the early stages of problem formulation. USEPA also updated the IGTT on assessment progress in January 2012 via webinar.
- **Tribal consultation.** USEPA's policy is to consult on a government-to-government basis with federally recognized tribal governments when USEPA actions and decisions may affect tribal interests. Consultation is a process of meaningful communication and coordination between USEPA and tribal officials. In February 2011, USEPA invited all 31 federally recognized tribal governments (tribes) of the Bristol Bay region to enter formal consultation on the assessment, to ensure their involvement and to include their concerns and relevant information in the assessment. Throughout development of the assessment there have been numerous opportunities for tribes to participate in the tribal consultation process. Not all tribes elected to participate in consultation. USEPA met with representatives from 20 of the 31 tribes (including all 13 tribes with federally recognized tribal governments in the Nushagak and Kvichak River watersheds), either in person or on the phone, during the consultation process.
- **Alaska Native Claims Settlement Act (ANCSA) engagement.** USEPA provided multiple engagement opportunities for ANCSA Village and Regional Corporations throughout development of the assessment, consistent with Public Law 108-199, Division H, Section 161, and Public Law 108-447, Division H, Title V, Section 518. USEPA representatives traveled to King Salmon, Iliamna, and Anchorage for meetings at the request of multiple ANCSA Corporations, to share information about and receive input on the assessment. Additionally, ANCSA Corporation representatives were invited to participate in a webinar following the release of April 2013 draft of the assessment. Throughout assessment development, ANCSA Corporations have traveled numerous times to meet with USEPA officials in Anchorage, Seattle, and Washington, D.C. Seventeen of the 26 ANCSA Corporations within the Bristol Bay region were engaged through these mechanisms.
- **Public comments.** USEPA released two drafts of the assessment for public comment. Approximately 233,000 and 890,000 comments were submitted to the USEPA docket during the 60-day public comment period for the May 2012 and April 2013 drafts of the assessment, respectively. USEPA also held eight public comment meetings in June 2012, in Dillingham, Naknek, New Stuyahok, Nondalton, Levelock, Igiugig, Anchorage, and Seattle. Approximately 2,000 people attended these meetings. An overview of these meetings was shared via two webinars in July 2012.
- **Public involvement in peer review.** USEPA provided multiple opportunities for stakeholder involvement in the peer review process. In February 2012, the public was invited to nominate qualified scientists as potential peer reviewers; these nominations were submitted to the peer review contractor for consideration. In March 2012, USEPA requested public comments on the questions to be given to peer reviewers, and these questions were revised in response to comments received. In August 2012, the public was invited to participate in the first 2 days of the peer review meeting in Anchorage, to provide oral comments to and observe discussions among the peer reviewers.

Detailed background characterizations of the watershed's resources are included in the assessment's appendices. We used these characterization studies and input from the IGTT to develop a series of conceptual models illustrating potential linkages between sources and stressors associated with large-scale mining and the assessment endpoints. These models were then used to develop a plan for analyzing and characterizing risks. In the risk analysis, available data were used to assess potential exposures to stressors and potential effects on assessment endpoints stemming from those exposures. In the final phase, results of these analyses were integrated to provide a comprehensive picture of the risks to assessment endpoints within the defined scope of the assessment. The uncertainties and limitations associated with these analyses were also identified.

This assessment has undergone extensive review throughout its development. Two earlier drafts of the assessment, released in May 2012 and April 2013, were subjected to review by 12 independently selected, expert peer reviewers (Box 1-2). Both of these drafts also had 60-day public comment periods, during which interested parties could submit their comments on the assessment to USEPA (Box 1-1).

1.2 Uses of the Assessment

This assessment is a scientific investigation. It does not reflect any conclusions or judgments about the need for or scope of possible government action, nor does it offer or analyze options for future decisions. Rather, it is a scientific product intended to provide a characterization of the biological and mineral resources of the Bristol Bay watershed, increase understanding of the potential risks to fish resources from large-scale mining, and inform future government decisions.

USEPA and other stakeholders may use this assessment in several ways. The assessment will inform the public and interested government entities about the resources of the Bristol Bay watershed. Much of the information about these resources was previously found in a variety of sources. In this assessment, we have synthesized and integrated available literature and provided a useful summary characterizing the Bristol Bay watershed's resources.

The assessment also will inform the public and interested government entities about the potential impacts of large-scale mining. USEPA recognizes the high level of interest concerning the impacts of potential mine development on the watershed's ecological resources. That interest originates from Alaska Native communities within the watershed, other Alaska residents, and interested parties throughout the United States. It is expressed both by those interested in protecting the Bristol Bay fishery and by those interested in developing the watershed's extensive mineral resources. This assessment is a scientific and technical resource that is useful to members of the public as they weigh the challenges of both mining and protecting the ecological resources in the Bristol Bay watershed in the years ahead.

BOX 1-2. OVERVIEW OF THE ASSESSMENT'S PEER REVIEW PROCESS

The peer review process is designed to provide a documented, independent, and critical review of a draft assessment. Its purpose is to identify any problems, errors, or necessary improvements to a document prior to it being published or otherwise released as a final document. To this end, the U.S. Environmental Protection Agency (USEPA) tasked Versar, an independent contractor, with coordinating an external peer review of the May 2012 draft assessment. Versar assembled 12 independent experts to serve as peer reviewers. These reviewers were selected from a pool of candidates that included those suggested during a public nomination process. In assembling the peer reviewers, Versar evaluated the qualifications of each peer review candidate and conducted a thorough conflict of interest screening process.

The peer reviewers were asked to evaluate and provide a written review of the May 2012 draft of the assessment (the main report and its appendices) by responding to 14 questions developed by USEPA with input from public commenters. Peer reviewers were charged only with evaluating the quality of the science included in the draft assessment and were not charged with making any regulatory recommendations, commenting on any policy implications of USEPA's role or mine development in the region, or reaching consensus in either their deliberations (during the peer review meeting, see below) or their written comments. Peer reviewers were provided with a summary of public comments submitted during the 60-day public comment period for the May 2012 draft and were given access to the public comments themselves.

A 3-day peer review meeting, coordinated by Versar, was held in Anchorage, Alaska, on August 7 through 9, 2012. On the first day of the meeting, peer reviewers heard testimony from approximately 100 members of the public. Peer reviewers deliberated among themselves on the second and third days of the meeting; these deliberations were open to the public on the second but not the third day.

Following the public peer review meeting, peer reviewers were given additional time to complete their individual written reviews. Versar provided these final written comments to USEPA in their *Final Peer Review Meeting Summary Report* for the May 2012 draft, which USEPA released to the public in November 2012. USEPA considered these peer review comments, as well as comments received during the 60-day public comment period, as they revised the May 2012 draft of the assessment.

In April 2013, USEPA released a revised draft of the assessment. The same 12 peer reviewers were asked to conduct a follow-on peer review to evaluate whether the April 2013 draft of the assessment was responsive to their original comments. USEPA provided reviewers with a draft response to comments document, in which USEPA responses to peer review comments on the May 2012 draft assessment were added to the *Final Peer Review Meeting Summary Report* submitted by Versar.

In the follow-on review, peer reviewers were asked to go through their comments on the May 2012 draft, review USEPA's draft responses to their original comments, and evaluate whether their original review comments had been addressed sufficiently and whether appropriate changes had been incorporated into the April 2013 draft. USEPA received these follow-on peer review comments directly from the 12 peer reviewers in August to September 2013. Again, USEPA considered these peer review comments, as well as comments received during the 60-day public comment period, as they revised the April 2013 draft of the assessment.

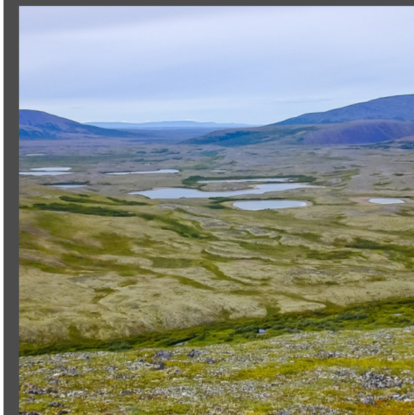
All drafts of the assessment (May 2012, April 2013, and final), as well as the peer review comments on the May 2012 and April 2013 drafts and USEPA's responses to those comments, are available online.

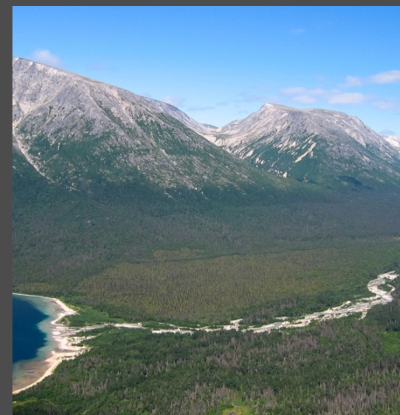
Our findings concerning the potential impacts of large-scale mining will help to inform future government decisions regarding mine development in the Bristol Bay watershed and potential actions to protect and maintain the integrity of the watershed's aquatic resources. One of the initiators for the assessment was the multiple petitions to USEPA to use its authority under CWA Section 404(c). It is expected that the assessment will provide an important base of information for any agency decision about whether or not to use Section 404(c), either now or in the future, and will facilitate a thoughtful decision regarding whether application of this authority is or is not warranted.

The assessment may also assist federal and state scientists and resource managers involved in the evaluation of future mine permit applications submitted for the deposits in the Bristol Bay watershed. It is likely that future mines in the watershed would require the filling of streams and wetlands and thus would require a Section 404 permit from the U.S. Army Corps of Engineers. USEPA reviews and comments on proposed Section 404 permit applications, and this assessment will be a valuable resource in the development and review of such permit applications.

If a Section 404 permit or other major federal action is required for a future mine in the watershed, it would trigger review of the proposed mine under the National Environmental Policy Act. This assessment, particularly in terms of its identification and analysis of potential direct, indirect, and cumulative effects of large-scale mining, will be a valuable resource in the development and review of any future environmental assessment related to mining in the Bristol Bay watershed.

Perhaps the most important use of this assessment is to better inform dialogues among interested stakeholders concerning the resources in the Bristol Bay watershed and the potential impacts of large-scale mining on those resources.





CHAPTER 2. OVERVIEW OF ASSESSMENT

2.1 Structure

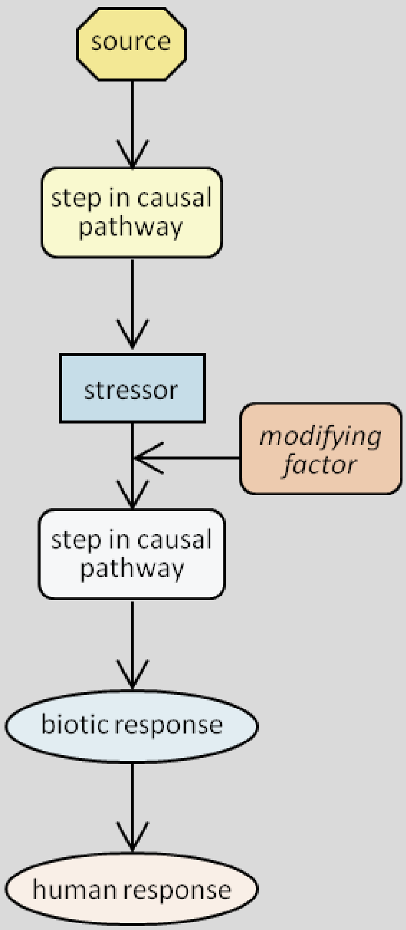
We based this assessment on U.S. Environmental Protection Agency (USEPA) guidelines for ecological risk assessment (ERA) (USEPA 1998). We began by reviewing existing literature to synthesize background information on the Bristol Bay region, particularly the Nushagak and Kvichak River watersheds. This information focused on several topics, including the ecology of Pacific salmon and other fishes; the ecology of relevant wildlife species; mining and mitigation, particularly in terms of porphyry copper mining; potential risks to aquatic systems due to road and pipeline crossings; fishery economics; and Alaska Native culture. These detailed background characterizations are included as appendices to this assessment.

In accordance with the different phases of an ERA, the assessment document itself is organized into two main sections: Problem Formulation (Chapters 2 through 6) and Risk Analysis and Characterization (Chapters 7 through 14). Problem formulation is the first phase of an ERA, during which the purpose and scope of the assessment are defined (USEPA 1998). Risk assessors, decision makers, and stakeholders determine the topical, spatial, and temporal scope needed to effectively address whatever decision process the assessment is meant to inform. Assessment endpoints, or explicit expressions of the environmental entities of interest (USEPA 1998), are identified. Conceptual models illustrating potential linkages among sources, stressors, and endpoints considered in the assessment (Box 2-1), as well as a plan for analyzing and characterizing risks, are developed.

The risk analysis and characterization phases follow problem formulation (USEPA 1998). During the risk analysis phase, available data are used to assess potential exposures to stressors and exposure-response relationships for those exposures and endpoint effects. In the risk characterization phase, information on exposures and effects is integrated, and the uncertainties and limitations associated with the assessment's analyses are identified.

BOX 2-1. CONCEPTUAL MODELS

Throughout this assessment we use conceptual model diagrams to illustrate potential ways in which large-scale mine development may adversely affect the Bristol Bay watershed's biota and Alaska Native cultures. These conceptual model diagrams show hypothesized pathways linking common sources associated with mining to potential stressors, and those stressors to responses of interest. Inclusion of a pathway indicates that the pathway *can* occur, not that it *will definitely* occur. Thus, these diagrams are not meant to illustrate worst-case scenarios in which all pathways occur simultaneously. Rather, they are meant to provide overviews of potential linkages among sources, stressors, and responses, one or more of which may plausibly result from mine development.



The conceptual model diagrams contain the following elements (note that not all elements are found in each diagram).

Sources are entities associated with mining that may directly or indirectly result in one or more stressors.

Steps in causal pathways are processes or states that may link sources to stressors or stressors to responses.

Stressors are physical or chemical entities that may directly induce a response of concern.

Modifying factors are processes, states, or other factors that may influence the delivery, expression, or effect of stressors (e.g., temperature, time or duration of exposure, mitigation).

Biotic responses are potential effects on salmon, other fishes, and wildlife.

Human responses are potential effects on Alaska Native people and culture.

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 could plausibly cause or result in the second shape.
- Arrows leading from a shape to another arrow (or a general section of the diagram) indicate that the originating shape (always categorized as a *modifying factor*) could plausibly influence the cause-effect relationships indicated (e.g., by increasing or decreasing its probability or intensity of occurrence).
- Shapes bracketed under another shape are specific components of the more general shape under which they appear.
- Within a shape, ↑ indicates an increase in the parameter, ↓ indicates a decrease in the parameter, and Δ indicates a change in the parameter.

2.1.1 Data Used in the Assessment

An ERA requires data of sufficient quantity and quality, from a variety of sources. Throughout the problem formulation, risk analysis, and risk characterization phases, relevant data are identified and acquired. These data may result from different kinds of studies, including field studies at the site of interest, field studies at other sites somehow relevant to the site or issue of interest, laboratory tests, and modeling applications.

In this assessment, we prioritized peer-reviewed, publicly accessible sources of information to ensure that the information and data we incorporated were of sufficient quality. In many cases, however, peer-reviewed data—particularly those directly relevant to potential mining in the Bristol Bay region—were not available. Thus, we incorporated credible, non-peer-reviewed data from multiple sources, including state government agencies (e.g., the Alaska Department of Fish and Game [ADF&G], the Alaska Department of Natural Resources [ADNR]), federal government agencies (e.g., the U.S. Geological Survey [USGS], the U.S. Fish and Wildlife Service [USFWS]), and academic organizations (e.g., Scenarios Network for Alaska and Arctic Planning [SNAP] data).

We also incorporated non-peer-reviewed data collected under the auspices of the Pebble Limited Partnership (PLP) (e.g., as presented in Ghaffari et al. 2011, PLP 2011), as these sources contain data directly relevant to the Pebble deposit and the surrounding region. Both Ghaffari et al. (2011) and the PLP's environmental baseline document (PLP 2011) are cited numerous times throughout the assessment. PLP is currently conducting its own peer review of the data presented in its baseline document, but that review had not been completed when this assessment was released.

Other non-governmental organizations have collected data relevant to the assessment. USEPA subjected some of these documents to external peer review and, where defensible, we have incorporated this information into the assessment (e.g., Chambers and Higman 2011, Woody and Higman 2011, Earthworks 2012).

In addition, some minor sources of information (e.g., permits and reports filed by mining companies) were used without peer review. In all cases, sources of information and data included in the assessment are appropriately cited (Chapter 15).

Throughout the assessment, we present numbers from the scientific literature or from PLP (2011) using the number of significant figures in the original source. Numbers derived for this assessment are presented with the appropriate number of significant figures given the precision of the input data and uncertainties due to modeling and extrapolation.

2.1.2 Types of Evidence and Inference

As in other ERAs, the risk analysis and characterization phase of this assessment is based on weighing multiple types of evidence. Available and relevant pieces of evidence from a variety of sources are used to follow different lines of inference and reach the best-supported conclusions.

In this risk analysis, we use general scientific knowledge, mathematical and statistical models, and data from the Bristol Bay region, other sites (e.g., mines in other regions), and laboratory studies to evaluate potential consequences of three mine size scenarios—that is, realistic potential mines of different sizes, the characteristics of which are based largely on a mining company report (Ghaffari et al. 2011)—in terms of sources, exposure to different stressors, and exposure-response relationships. First, we estimate the magnitude of exposures potentially resulting from both routine operation and accidents and failures in the mine scenarios, such as elevated aqueous copper concentrations, kilometers of streams eliminated, and kilometers of streams upstream of road crossings. Then, we consider the effects of these exposures—that is, 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, where possible, exposure-response relationships for the endpoints and estimated exposures. For some issues, multiple lines of evidence are available (e.g., state standards, federal criteria, effects models, field studies, and toxicity tests as lines of evidence for copper toxicity); for other issues, lines of evidence are more limited.

Evidence from existing mines and other analogous facilities is used where relevant. Prior mining activities in comparable watersheds provide examples of what can happen to the environment when metals are mined. Some components of our mine scenarios have analogues in other industries (e.g., oil and gas pipelines). These inferences by analogy reduce the uncertainties that come with modeling and prediction, but introduce other uncertainties related to industry-specific or site-specific differences in environmental conditions and potential changes in practices. Because no analogue is similar in all aspects to potential mines and their components in the Bristol Bay region, we choose analogues to fit the specific issues being assessed and take care to use analogues that are defensible despite their differences from our mine scenarios. For example, the Fraser River watershed could be considered an analogous system to the Bristol Bay watershed because it has similar mines and a similar salmon resource, but we recognize that there are important differences between these systems (e.g., extensive urban development, forestry, and agriculture in the Fraser River watershed). Metal mines in the Rocky Mountain metal belt (e.g., sites near the Coeur d'Alene River, Idaho, and the Clark Fork River, Montana) were developed using mining practices that would not be allowed under current mining laws. However, the fate and effects of tailings in streams and floodplains at these sites, which also supported trout and salmon populations, offer some parallels to the fate and effects of tailings following potential tailings dam failures in the Bristol Bay region, should they occur—even if the underlying causes of failure differ.

The use of data from the historical, operational records of mines, pipelines, and roads is necessary but controversial. It is essential and conventional for risk assessments to use the history of a technology to estimate failure rates. However, developers argue, with some justification, that the record of older technology is not relevant because of technological advances. Despite advances, no technology is perfect, and rates of past failures may be a better guide to future outcomes than the expectation that developers can design a system that will not fail. A classic example is the National Aeronautics and Space Administration (NASA) space shuttle program, which denied the relevance of the failure rate of solid rocket boosters and declared that the shuttle's rate of failure on launch would be one in a million. The Challenger failure showed that the prior failure rate was still relevant, despite updated technology.

For most potential failures, historical failure rates are the only available evidence. New technologies typically have not been in use long enough or widely enough to provide failure rates, and measures to correct past failure modes may unwittingly introduce new ones. Thus, in this assessment we choose failure rates that are most relevant and interpret them cautiously, using them to provide an upper bound estimate of future failure rates.

After these analyses and lines of evidence are presented, we characterize risk for each line of evidence by combining exposures and exposure-response relationships to estimate effects and by considering uncertainties. We weigh different lines and types of evidence based on evidence strength and quality. The resulting qualitative or quantitative estimates of risk and uncertainty are based on either the best line of evidence or a combined estimate from multiple lines of evidence and inferences. Bounding analyses, which set upper and lower limits for key parameters, are used to express uncertainties concerning future mine activities and their effects. In particular, multiple mine sizes and durations are included in the mine scenarios (Chapter 6). 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 (Chapter 9).

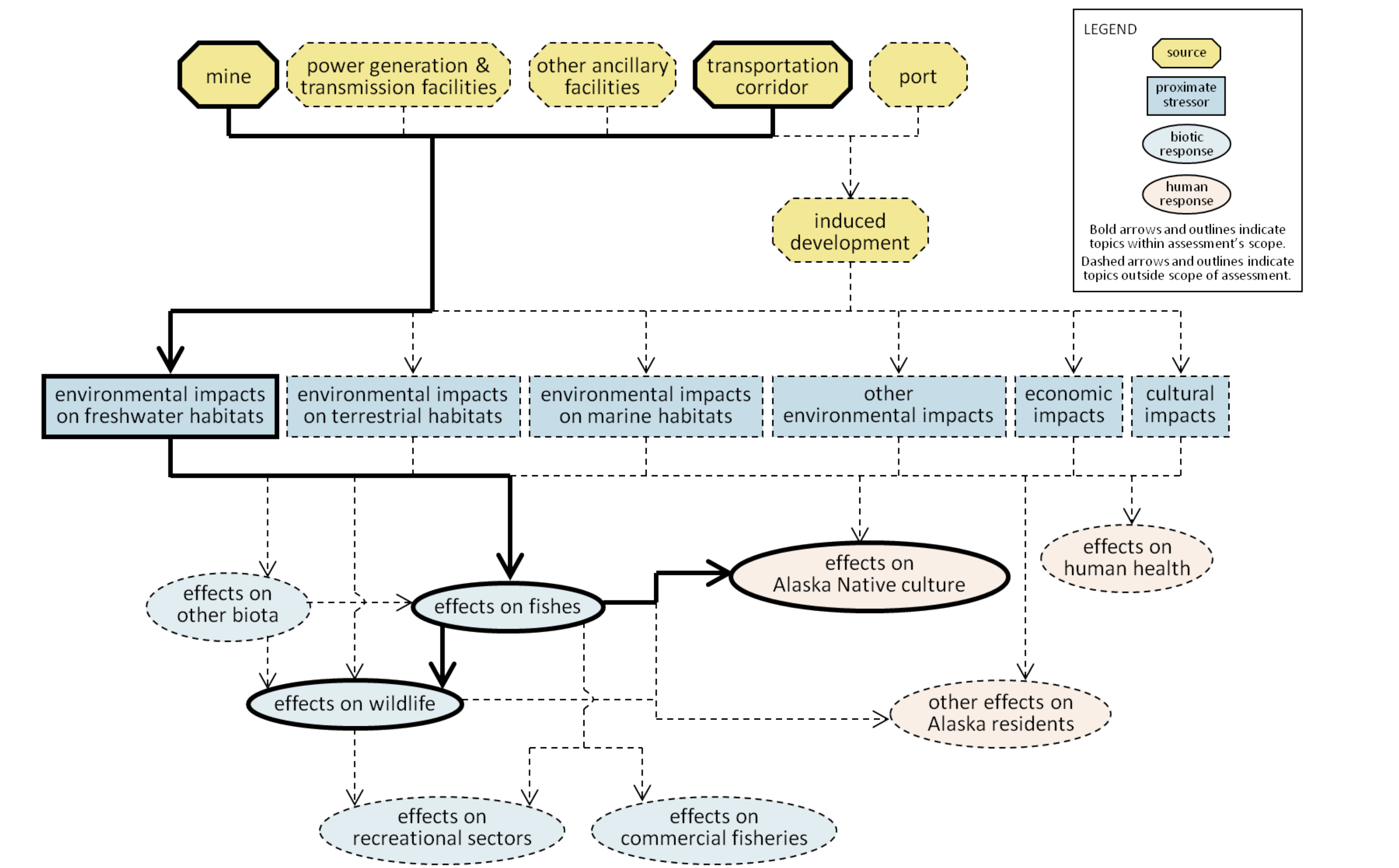
2.2 Scope

2.2.1 Topical Scope

Construction and operation of a large-scale mining operation require the development of extensive infrastructure and involve numerous processes and components, each of which may have repercussions for receiving environments. In this assessment, we do not consider all potential sources of risk associated with the development of large-scale mining in the Bristol Bay watershed, all the stressors that may result from these sources, and all the endpoints that may be affected. Rather, we focus on a more limited set of sources, stressors, and endpoints based on stakeholder concerns and potential decision-maker needs (Chapter 1). These focal components are described in broad terms below. In Chapters 3 through 6, we consider these components in greater detail, and more specifically define the focus of the assessment—in terms of geographic region, type of mining development, and ecological endpoints—for risk analysis and characterization purposes.

In terms of sources, we consider the mine infrastructure and transportation corridor components of a large-scale surface mining operation (Figure 2-1). Exploratory mining activities are ongoing in the region (Box 2-2), but these activities are considered outside the scope of the assessment. Certain sources associated with mining but not directly related to mine operations are not evaluated here, including power generation and transmission facilities and activities, ancillary facilities such as housing for mine workers and wastewater treatment plants to serve an increased human population, and construction and operation of a deep-water port at Cook Inlet (Figure 2-1). A thorough evaluation of induced development—development that is not part of the mine project, but for which the mine project provides the impetus or opportunity, such as residential and commercial growth resulting from increased accessibility—is also outside the scope of this assessment, although its importance is considered qualitatively in Chapter 13.

Figure 2-1. Conceptual model illustrating sources, stressors, and responses potentially associated with large-scale mine development in the Bristol Bay watershed. Pathways explicitly evaluated in this assessment are in bold; dashed pathways may be considered qualitatively in parts of the assessment, but are generally considered outside its scope. See Box 2-1 for a general discussion of how conceptual models are used and structured in the assessment.



BOX 2-2. EXPLORATORY MINING ACTIVITIES

Exploratory activities associated with the Pebble deposit—including geophysical, geochemical, and environmental surveys, geological mapping, and drilling—have been underway for several decades (Ghaffari et al. 2011). For example, 1,158 holes were drilled on the Pebble property through 2010, totaling 948,638 feet (289,145 m) (Ghaffari et al. 2011). These holes are concentrated in the Pebble deposit area, but occur throughout the Pebble claim block. According to the Pebble Limited Partnership's annual reclamation reports (submitted to the State of Alaska by the Pebble Limited Partnership in accordance with their land use permits), the total amount of land disturbed between 2009 and 2012 was approximately 3 acres.

Because these exploratory activities require water, power, personnel support, and the use of chemicals, heavy machinery, helicopters, and other equipment in relatively undeveloped areas, they likely have had some environmental impact on the region. Full evaluation of these effects is beyond the scope of this assessment, and it is likely that any effects of exploratory activities would be small relative to the effects of full mine development.

In terms of stressors, we focus on potential environmental effects on freshwater habitats (Figure 2-1). We focus on freshwater habitats because the Bristol Bay watershed supports exceptional fish populations, and these populations are intimately linked to the watershed's freshwater habitats. Although we recognize that large-scale mining could also have significant direct impacts on terrestrial and marine systems, as well as direct economic and cultural repercussions, we do not evaluate these impacts here (Figure 2-1).

Given the ecological and cultural significance of fishery resources in the Bristol Bay watershed, and the fact that the health and sustainability of the watershed's fish populations are primary concerns shared by all stakeholders interested in the Bristol Bay area (including those who support mining), we focus on effects on key salmonids (Box 2-3) and resulting effects on wildlife and Alaska Native cultures as assessment endpoints (Chapter 5). Direct effects of mining on wildlife and Alaska Native cultures, although potentially significant, are not evaluated in this assessment. For example, construction and operation of a transportation corridor would likely directly affect wildlife populations (Forman and Alexander 1998); however, because the assessment focuses on freshwater habitats, these direct wildlife effects are not considered here. The only effects on wildlife and Alaska Native cultures evaluated in the assessment are those resulting from impacts on fish populations (Chapter 12). We also recognize that many other endpoints may be directly affected by large-scale mining operations, including other biota (e.g., vegetation, small mammals), other recreational and commercial fisheries, and human health (Figure 2-1), but these topics are also outside the scope of the assessment.

It is important to keep in mind that exclusion of a source, stressor, or endpoint from this assessment does not imply that it would be insignificant or unaffected. We recognize that many of the pathways we identify as outside of the assessment's scope could have significant repercussions for the region's biota and people.

BOX 2-3. KEY SALMONIDS 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—Salmoninae (salmon, trout, and char), Thymallinae (grayling), and Coregoninae (whitefish)—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 (sockeye, Chinook, coho, chum, and pink), rainbow trout, and Dolly Varden (a species of char). Collectively, we refer to these seven species as salmonids throughout this report.

All Salmonidae 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 waters 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.

2.2.2 Geographic Scales

Throughout this assessment, we consider data across five geographic scales (Table 2-1, Figure 2-2).

- The **Bristol Bay watershed** (Scale 1, Figure 2-3) includes all the basins and waterways that flow into Bristol Bay.
- The **Nushagak and Kvichak River watersheds** (Scale 2, Figure 2-4) include those drainage areas that contain stream segments flowing either directly or via downstream segments into the mainstem Nushagak River or Kvichak River.
- The **mine scenario watersheds** (Scale 3, Figure 2-5) include the cumulative drainage areas of the South and North Fork Koktuli Rivers to their junction and Upper Talarik Creek to its junction with Iliamna Lake.
- The **mine scenario footprints** (Scale 4, Figure 2-6) include the footprints of the major mine components (i.e., the mine pit, waste rock piles, and tailings storage facilities), the groundwater drawdown zone, and plant and ancillary facilities for each mine size scenario (Chapter 6).
- The **transportation corridor area** (Scale 5, Figure 2-7) includes 32 subwatersheds in the Kvichak River watershed that drain to Iliamna Lake and would be crossed by the transportation corridor (Chapter 6); the transportation corridor does not cross into the Nushagak River watershed.

These geographic scales are defined using the USGS National Hydrography Dataset (USGS 2012) (Box 2-4, Table 2-1). In problem formulation, we use broader geographic scales to describe the physical, chemical, and biological environment in the Bristol Bay region (Table 2-1); we also use broader scales to consider the effects of multiple mines across the landscape. In risk analysis and characterization, we use finer geographic scales to evaluate the potential effects of mining operations.

BOX 2-4. THE NATIONAL HYDROGRAPHY DATASET

The National Hydrography Dataset (NHD) is a publicly available database of surface water information for the United States (USGS 2012). Within the NHD, the entire landscape of the United States is organized into a six-tiered system of nested hydrologic units, each with their own identifiable codes (hydrologic unit codes, or HUCs). These tiers are defined as regions (represented by 2-digit codes), subregions (4-digit codes), basins (6-digit codes), subbasins (8-digit codes), watersheds (10-digit codes), and subwatersheds (12-digit codes). In total, the entire United States is divided into roughly 160,000 subwatersheds (12-digit HUCs) within roughly 21 regions (2-digit HUCs). Due to the hierarchical nature of the system, all subwatersheds (12-digit HUCs) within the same watershed start with the same first 10 digits, all watersheds (10-digit HUCs) within the same subbasin start with the same first 8 digits, and so on.

It is important to note that the NHD hydrologic units do not always delineate true hydrologic watersheds (i.e., their boundaries do not always accurately indicate where water drains to a particular point). Nevertheless, these boundaries are useful in both water resource and land management and are used as a foundational geographic layer in this assessment.

Table 2-1. Geographic scales considered in the assessment.

Scale	Description	Hydrologic Unit Codes (HUCs)[a]	Area (% of scale above)	Representative Chapters
1	Bristol Bay watershed	19030202–19030206, 19030301–19030306, 1903010101–1903010113, 1903010201–1903010203, 1903020101–1903020110	116,000 km^2 (NA)	2, 3, 4, 5, 13
2	Nushagak and Kvichak River watersheds	19030301–19030304, 19030205, 19030206[b]	59,900 km^2 (52%)	2, 3, 4, 5, 13
3	Mine scenario watersheds	190303021103, 190303021104, 190303021101–190303021102 1903020607,	925 km^2 (2%)	6, 7, 8, 9, 12
4	Mine scenario footprints			
	Pebble 6.5	NA	103 km^2 (11%)	6, 7, 8, 9, 12
	Pebble 2.0	NA	45.3 km^2 (5%)	6, 7, 8, 9, 12
	Pebble 0.25	NA	18.9 km^2 (2%)	6, 7, 8, 9, 12
5	Transportation corridor area[c]	190302051403–190302051406, 190302060101–190302060104, 190302060201–190302060206, 190302060301–190302060302, 190302060701–190302060702, 190302060704, 190302060901–190302060905, 190302060907, 190302060914[d]	2,340 km^2 (4%[e])	6, 10, 11

Notes:

[a] From the National Hydrography Dataset (NHD) (USGS 2012). Scale 1 is defined by 8-digit and 10-digit HUCs; Scale 2 by 8-digit and 12-digit HUCs; Scale 3 by 10-digit and 12-digit HUCs; Scale 5 by 12-digit HUCs. See Box 2-4 for further discussion of the NHD.

[b] Except for 190302062301–190302062311.

[c] The transportation corridor would include a 113-km road in the Kvichak River watershed; the area presented here represents the area of the 12-digit HUCs incorporating this road.

[d] The 190302060914 area was clipped to remove the area of Iliamna Lake and any land area draining directly to Iliamna Lake.

[e] Represents % of Scale 2 encompassed by the transportation corridor area HUCs.

NA = not applicable

Figure 2-2. The five geographic scales considered in this assessment. Only selected towns and villages are shown on this map. See Figures 2-3 through 2-7 for detailed views of each scale.

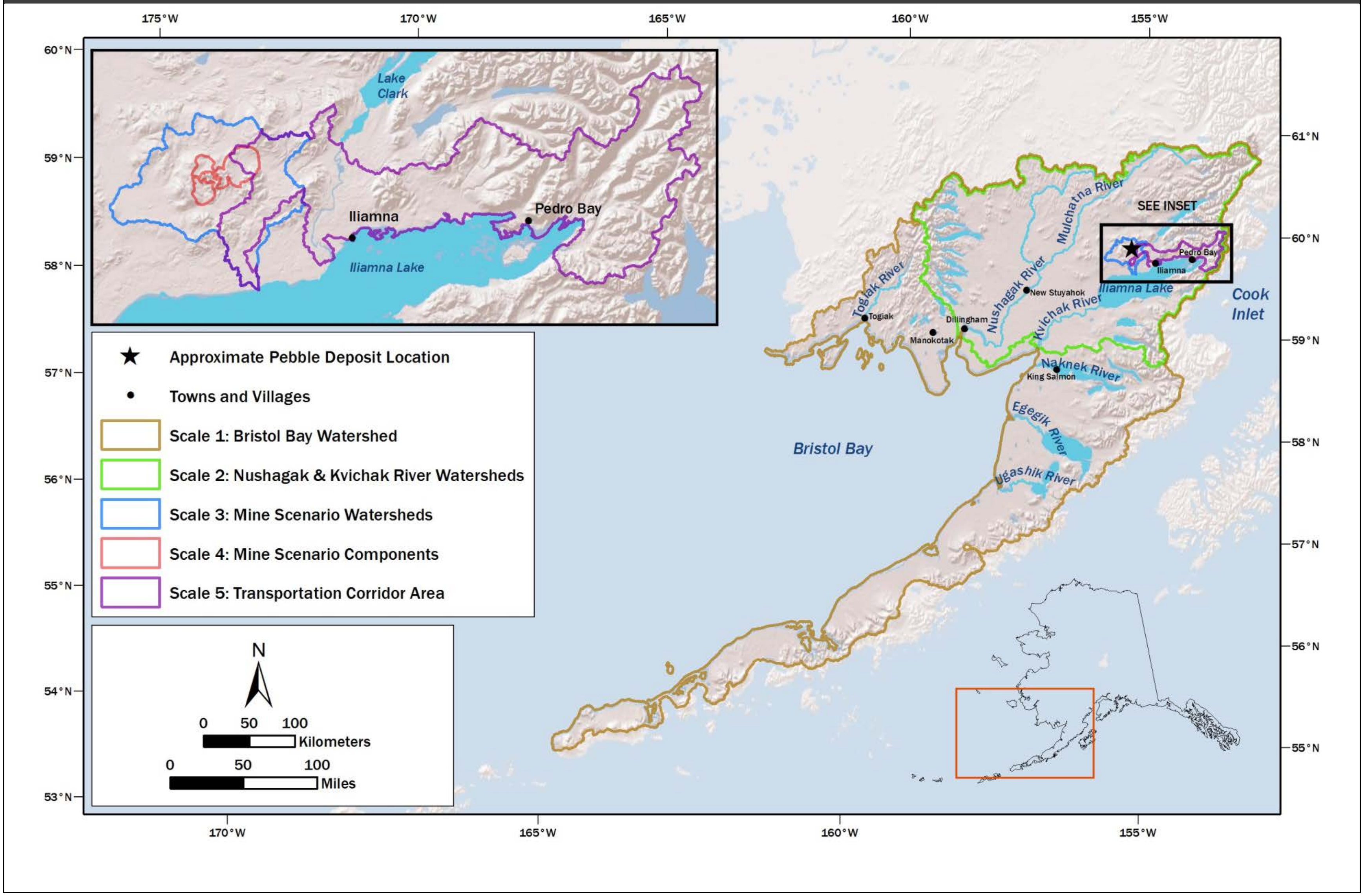


Figure 2-3. The Bristol Bay watershed (Scale 1), comprising the Togiak, Nushagak, Kvichak, Naknek, Egegik, and Ugashik River watersheds and the North Alaska Peninsula. Only selected towns and villages are shown on this map.

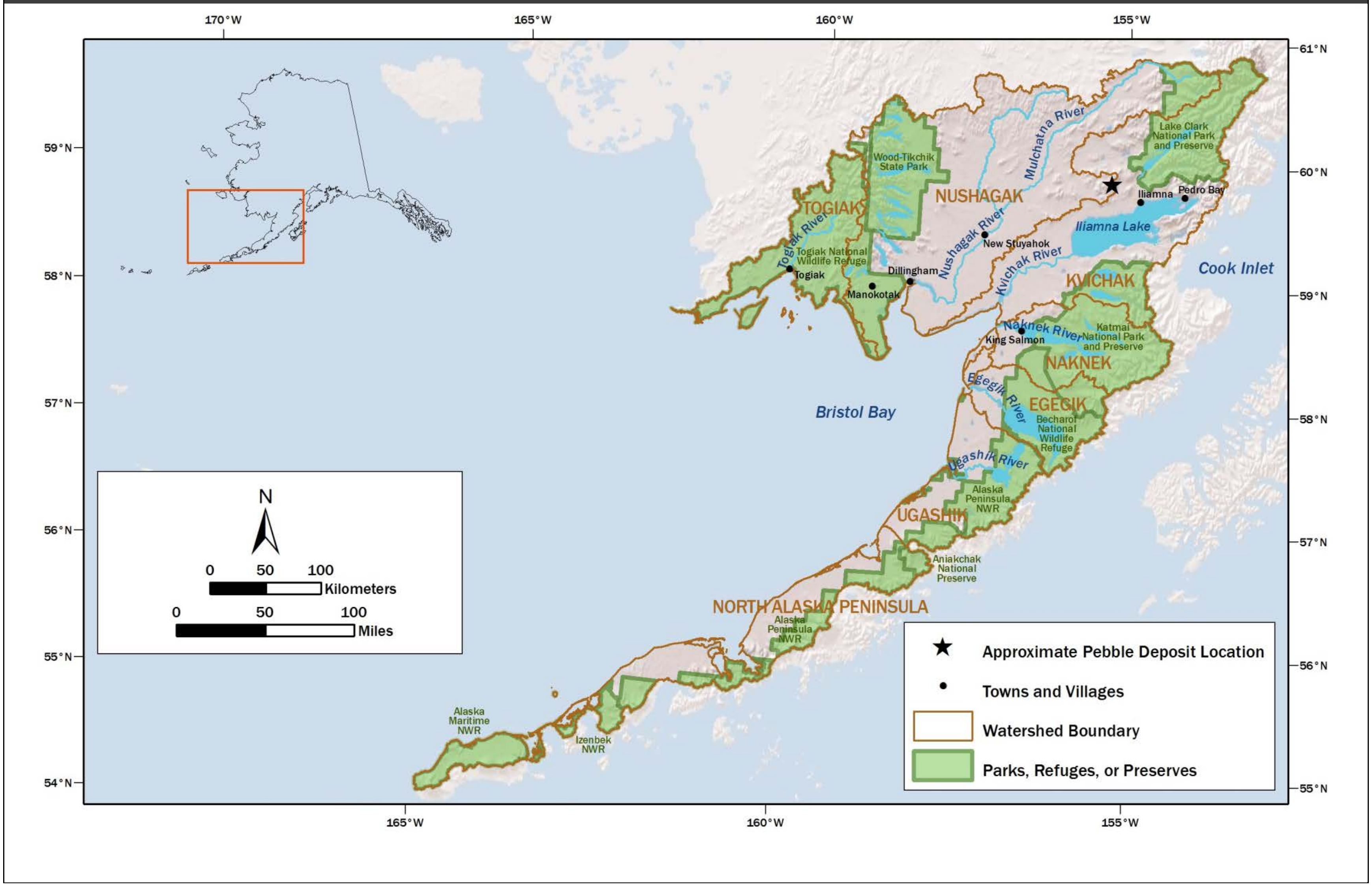


Figure 2-4. The Nushagak and Kvichak River watersheds (Scale 2).

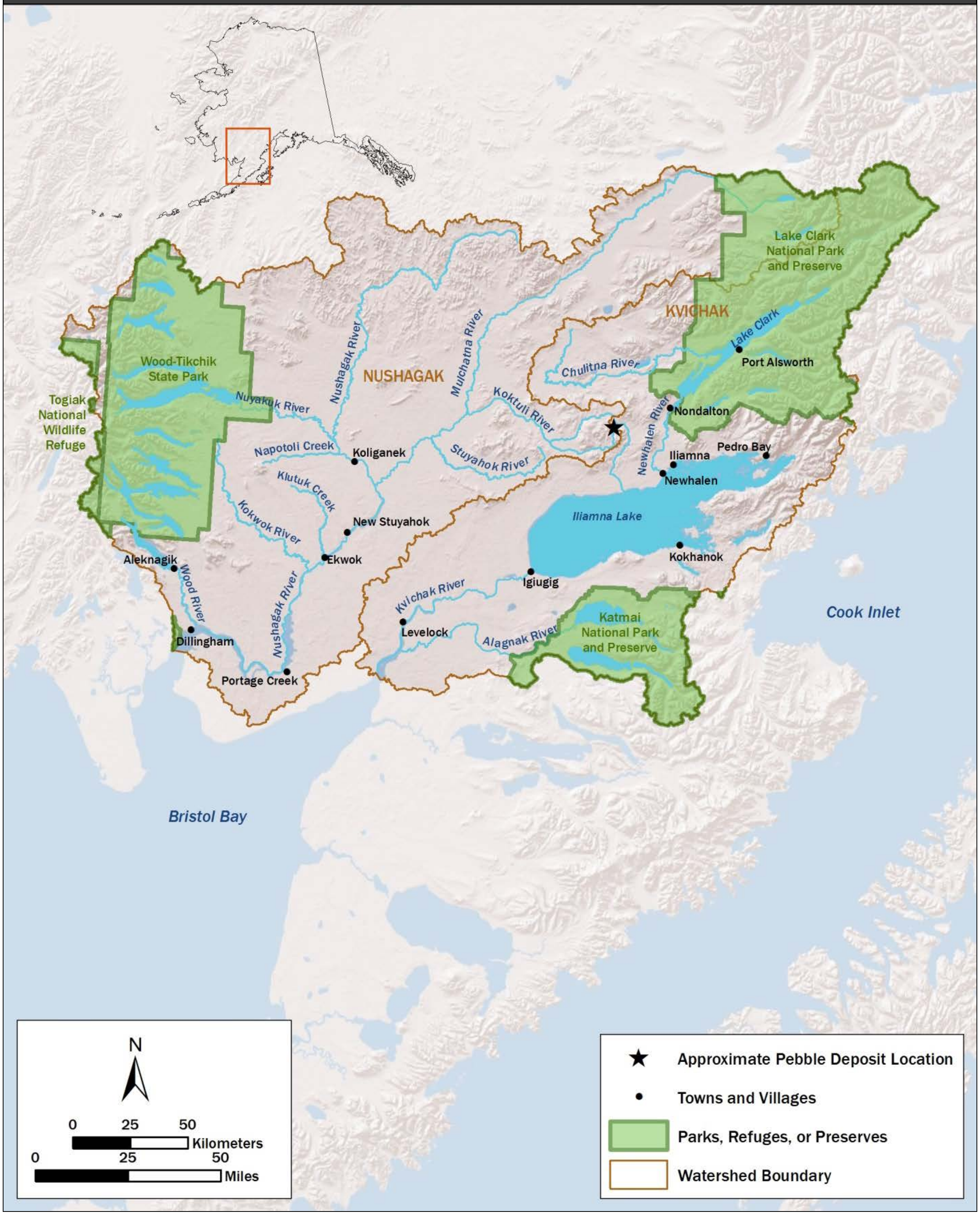


Figure 2-5. The mine scenario watersheds—South Fork Koktuli River, North Fork Koktuli River, and Upper Talarik Creek—within the Nushagak and Kvichak River watersheds (Scale 3).

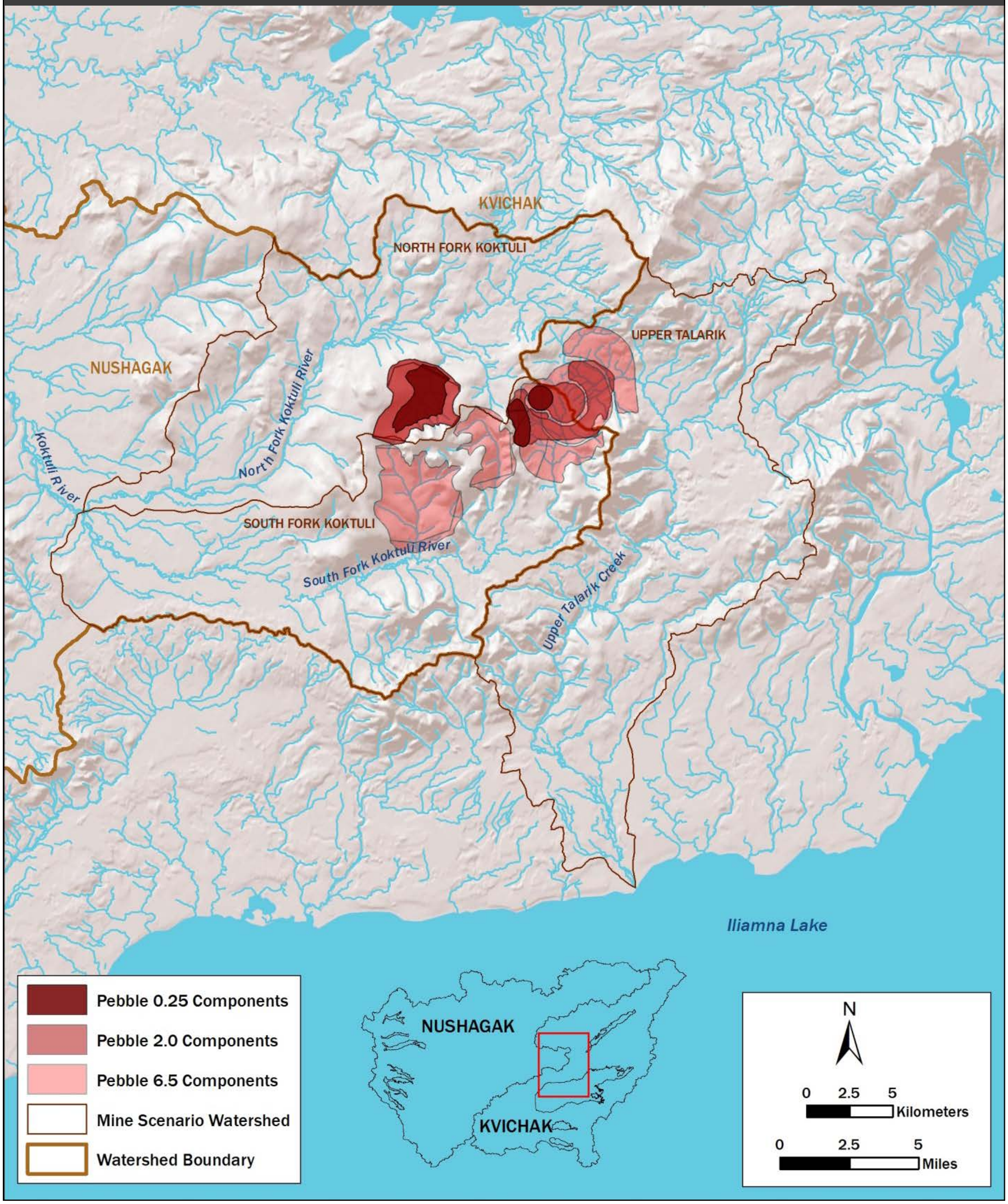


Figure 2-6. Footprints of the major mine components for the three scenarios evaluated in the assessment (Scale 4). Pebble 0.25 represents 0.25 billion ton of ore; Pebble 2.0 represents 2.0 billion tons of ore; Pebble 6.5 represents 6.5 billion tons of ore. Each mine footprint includes the footprints of the major mine components shown here, as well as the groundwater drawdown zone and the area covered by plant and ancillary facilities. See Figures 6-1, 6-2, and 6-3 for more detailed maps of the major mine components for each scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012).

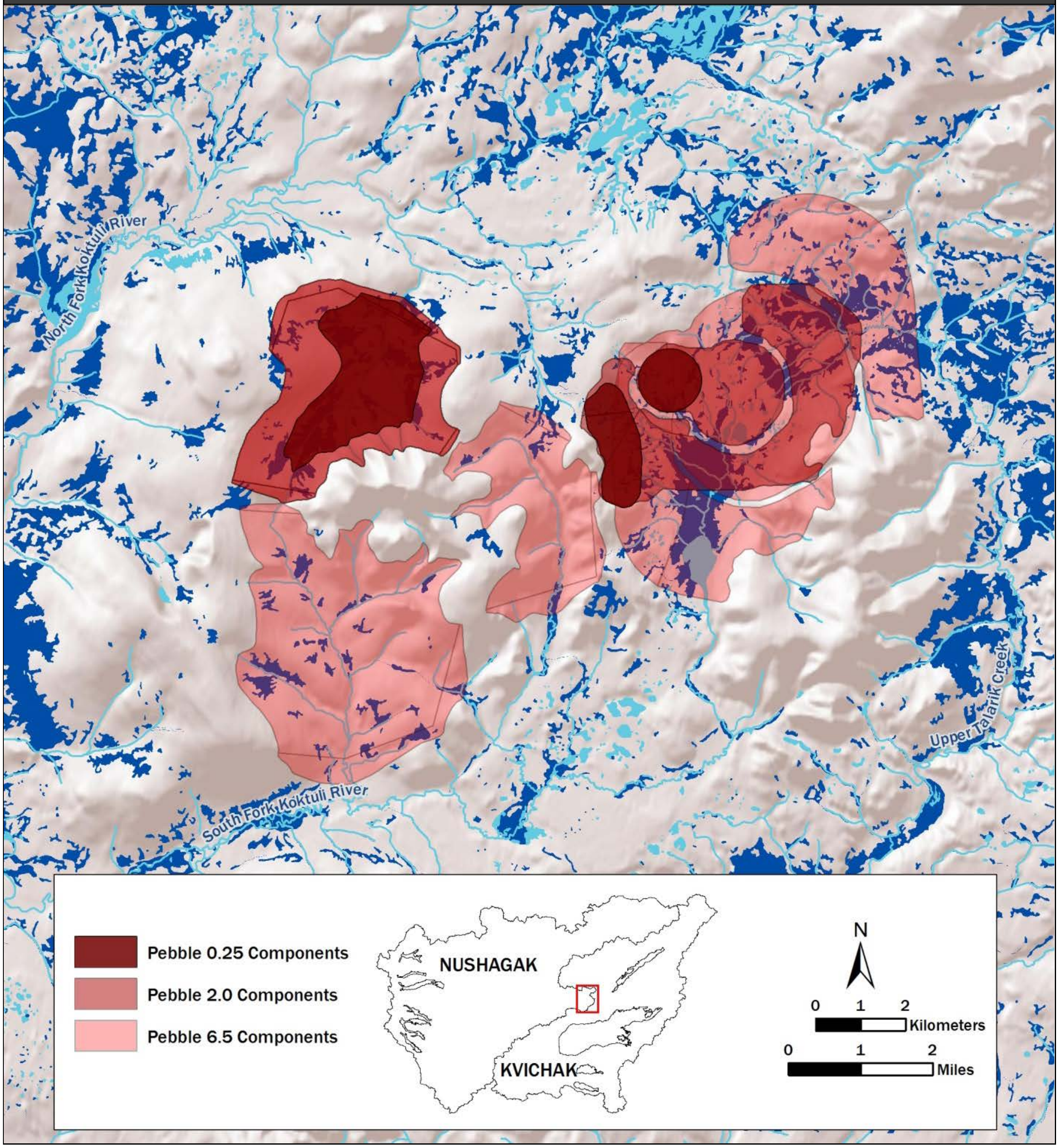
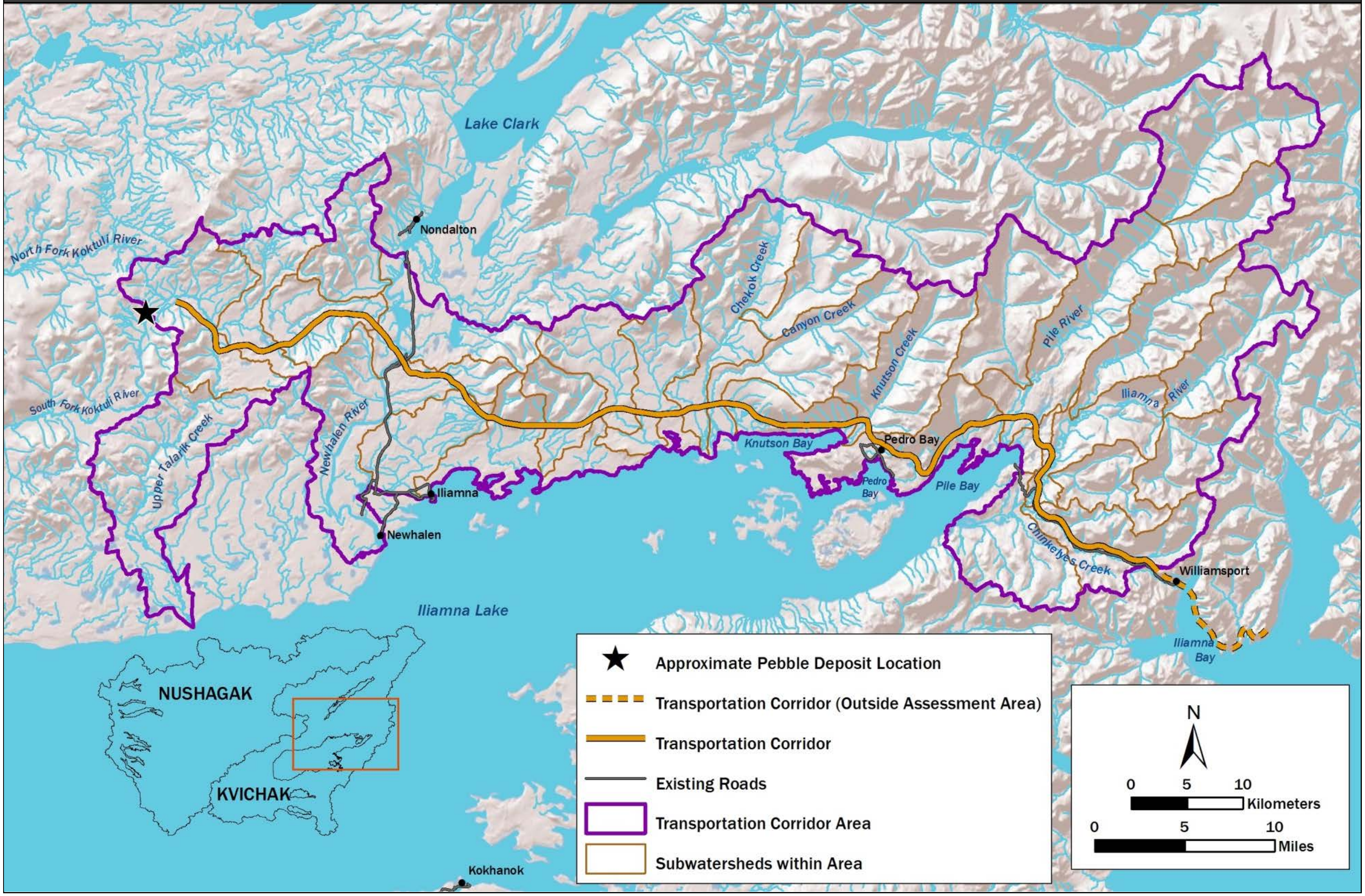
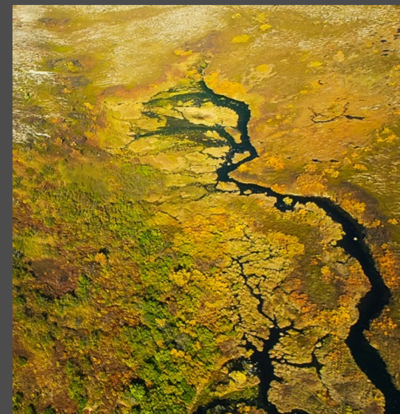


Figure 2-7. The transportation corridor area (Scale 5), comprising 32 subwatersheds in the Kvichak River watershed that drain to Iliamna Lake. Subwatersheds are defined by 12-digit hydrologic unit codes according to the National Hydrography Dataset (USGS 2012) (Box 2-4).







CHAPTER 3. 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 River watersheds—and a series of smaller watersheds draining the North Alaska Peninsula (Figure 2-3). The Bristol Bay region encompasses complex combinations of physiography, climate, geology, and hydrology, which interact to control the amount, distribution, and movement of water through a landscape shaped by processes such as tectonic uplift, glaciation, and fluvial erosion and deposition. The region's freshwater habitats are varied and abundant, and support a diverse and robust assemblage of fish (Chapter 5).

The Nushagak and Kvichak River watersheds account for more than half the land area in the Bristol Bay watershed (Table 2-1). The Pebble deposit, the largest known porphyry copper deposit in the region, is located in the headwaters of both watersheds (Figure 2-4) and represents the most likely site for near-term, large-scale mine development in the Bristol Bay watershed. In this chapter, we consider key aspects of the Bristol Bay watershed's physical environment, with particular emphasis on the Nushagak and Kvichak River watersheds (Figure 2-4).

3.1 Physiographic Divisions

The Nushagak and Kvichak River watersheds comprise five distinct physiographic divisions (Wahrhaftig 1965): the Ahklun Mountains, the Southern Alaska Range, the Aleutian Range, the Nushagak–Big River Hills, and the Nushagak–Bristol Bay Lowland (Table 3-1, Figure 3-1). Precipitation is greatest in the Southern Alaska Range, the Aleutian Range, and the Ahklun Mountains (Figures 3-1 and 3-2), and these physiographic divisions serve as major water source areas for lower portions of the watersheds. Annual water balance, especially 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 division (Selkregg 1974). Additional key attributes of each physiographic division are discussed below.

The Ahklun Mountain physiographic division, in the western portion of the Nushagak River watershed, is dominated by rolling hills to sharp, steep, glaciated mountains that receive high snowfall (Table 3-1, Figure 3-1) (Wahrhaftig 1965, Selkregg 1974, Gallant et al. 1995). Parent bedrock is deformed sedimentary rocks, intruded in several locations by igneous batholiths and stocks (Figure 3-3). A few small glaciers occur in high mountain cirques, and isolated masses of permafrost occur sporadically (Figure 3-4). Glacially carved lowland valleys are now filled with large, deep lakes, and adjacent streams are often incised in bedrock gorges. The surrounding area is mantled with colluvium, alluvium, and glacial drift and moraines (Figure 3-3). Soils are generally well drained and have medium erosion potential (Figures 3-5 and 3-6). Dwarf scrub is the dominant vegetation in the mountains and tall scrub and herbaceous plants are common in the valleys and lower mountain slopes (Figure 3-7).

The Southern Alaska Range physiographic division comprises a series of high, steep, glaciated mountains with land surfaces covered by rocky slopes, glacial drift and moraines, and glaciers (Table 3-1, Figure 3-1) (Wahrhaftig 1965, Selkregg 1974). Bedrock is a complex of granitic batholiths intruded into metamorphosed sedimentary and volcanic rock (Figure 3-3). Soils are shallow or not present (Figure 3-5) and permafrost occurs as isolated masses (Figure 3-4). Alpine tundra is the predominant vegetation (Figure 3-7). Streams are frequently swift and braided with several headwaters originating in glaciers (Figure 3-8). Several large, deep lakes occur in the glaciated valleys within the division (Figure 3-8). Braided, turbid streams flow into lakes, allowing sediment to settle, before flowing into the Nushagak and Kvichak River systems.

Within the Bristol Bay watershed, the Aleutian Range physiographic division consists of rolling hills to steep, glaciated mountains built of sedimentary, volcanic, and intrusive bedrock (Table 3-1, Figure 3-1) (Wahrhaftig 1965, Selkregg 1974). Cirque glaciers remain atop mountains in the extreme southeast corner of the Kvichak River watershed (Figure 3-3). This division is generally free of permafrost (Figure 3-4). Soils have formed in volcanic ash over glacial deposits at lower elevations, whereas rocky lands dominate at higher elevations (Figure 3-5). Erosion potential is high for some soils in the Aleutian Range division (Figure 3-6). Large, deep, moraine- and sill-impounded lakes are found in the ice-carved valleys. The Alagnak River, which drains most of the Aleutian Range physiographic division within the Bristol Bay watershed, is highly braided as it flows across the Nushagak–Bristol Bay Lowland division to the Kvichak River. Dwarf scrub vegetation is common (Figure 3-7) (Selkregg 1974, Gallant et al. 1995).

Table 3-1. Physiographic divisions (Wahrhaftig 1965) of the Nushagak and Kvichak River watersheds.

Physiographic Division	Description	Elevation (meters)	Permafrost Extent	Freshwater Habitats
Ahklun Mountains	Rolling hills to sharp, steep, glaciated mountains separated by broad lowlands, with a few small glaciers in high mountain cirques	10–1,600	Sporadic	Mix of unconstrained and constrained streams; Wood and Tikchik Lakes in U-shaped valleys
Southern Alaska Range	Rolling hills to steep, glaciated mountains covered by glacial drifts and moraines, rocky slopes, and glaciers	14–2,800	Unknown	Swift, braided streams and rivers, some with glacial headwaters; Lake Clark and other large lakes in glaciated valleys
Aleutian Range	Rolling hills to sharp, steep glaciated mountains, separated by broad lowlands, with a few small glaciers in high mountain cirques	14–1,600	Unknown	Large lakes associated with ice-carved valleys and terminal moraines; glacially fed lake tributaries
Nushagak–Big River Hills	Rounded ridges with broad, gentle slopes and broad, flat or gently sloping valleys	14–1,300	Sporadic	Glacial moraines and ponds in eastern part of region; upper reaches of the Nushagak and Mulchatna Rivers
Nushagak–Bristol Bay Lowland	Flat to rolling landscape with low local relief and deep morainal, drift, and outwash deposits, but no glaciers	0–800	Sporadic or absent	Morainal and thaw lakes; western half of Iliamna Lake; Kvichak, Alagnak, Nushagak, Nuyakuk, and Mulchatna River mainstems

Figure 3-1. Hydrologic landscapes within the Nushagak and Kvichak River watersheds, as defined by physiographic division and climate class. Physiographic divisions (Wahrhaftig 1965) are classified as Ahklun Mountains, Nushagak–Bristol Bay Lowland, Aleutian Range, Nushagak–Big River Hills, and Southern Alaska Range. Climate classes (Feddema 2005) were defined as very wet, wet, moist, dry, and semiarid, and calculated using 30-year (1971–2000) mean annual precipitation averages from the Scenarios Network for Alaska and Arctic Planning data (SNAP 2012). Points labeled A through H indicate approximate locations where photos in Figure 3-8 were taken.

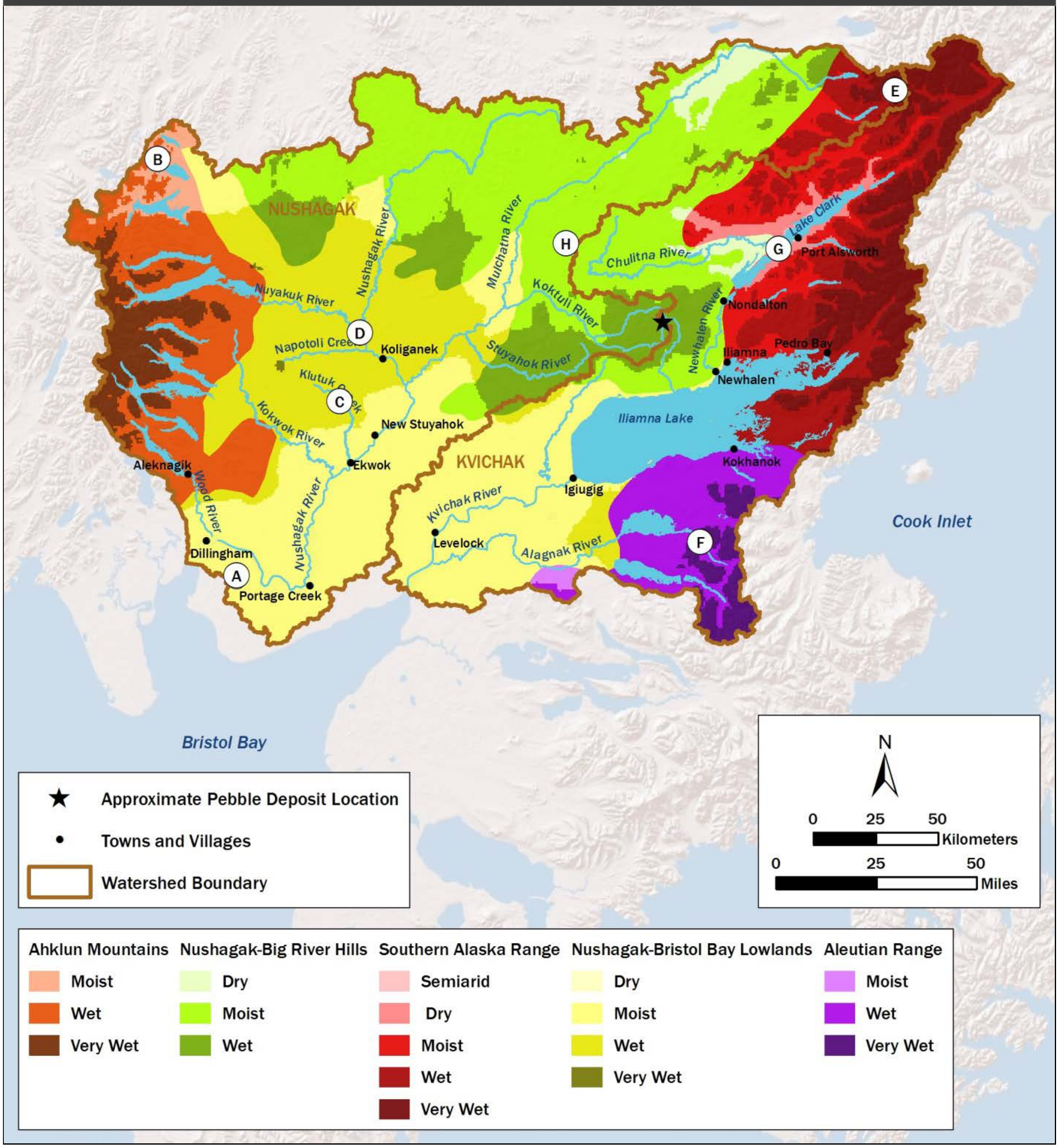


Figure 3-2. Distribution of mean annual precipitation (mm) across the Nushagak and Kvichak River watersheds, 1971 to 2000 (SNAP 2012).

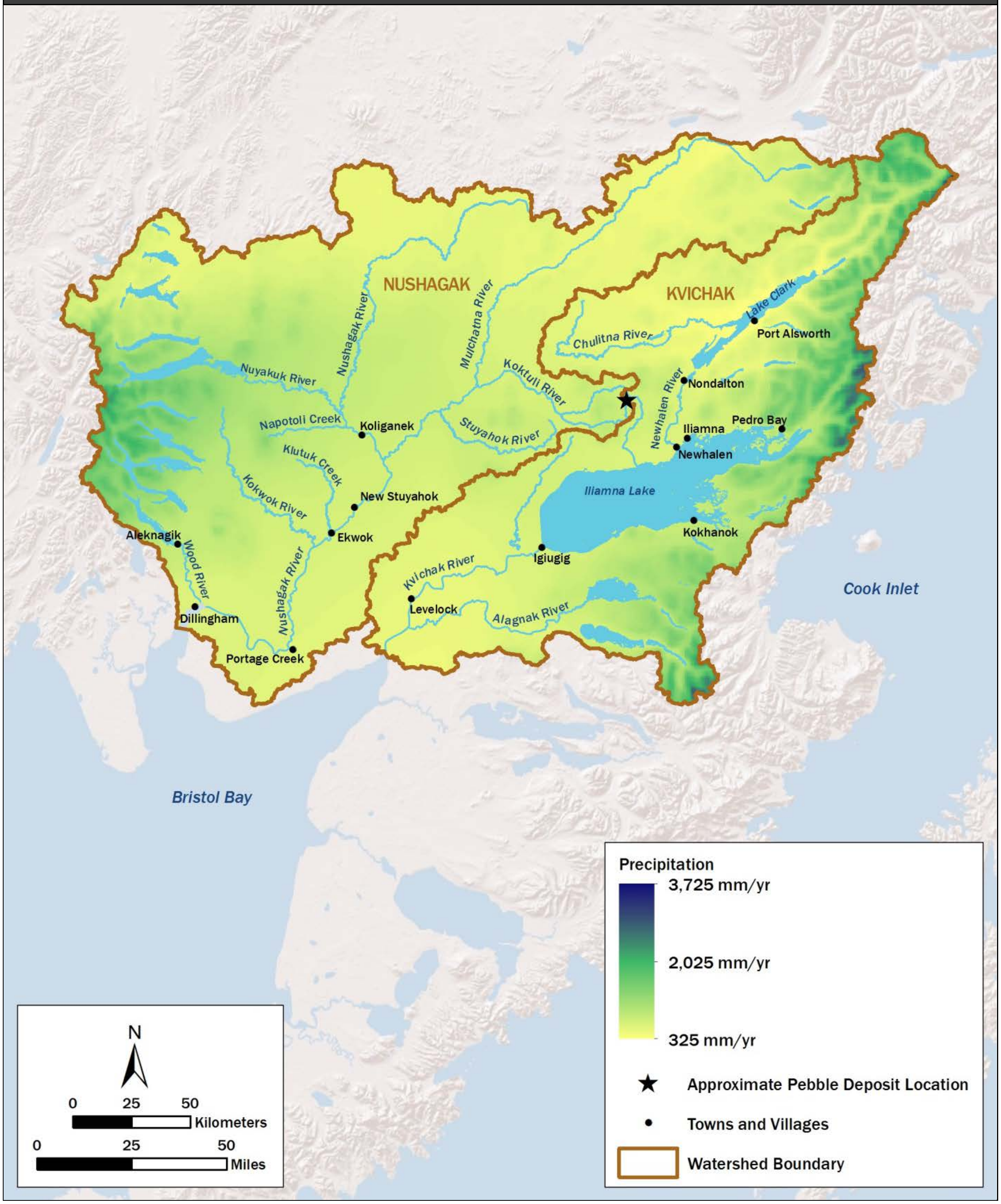


Figure 3-3. Generalized geology of the Bristol Bay watershed (adapted from Selkregg 1974).

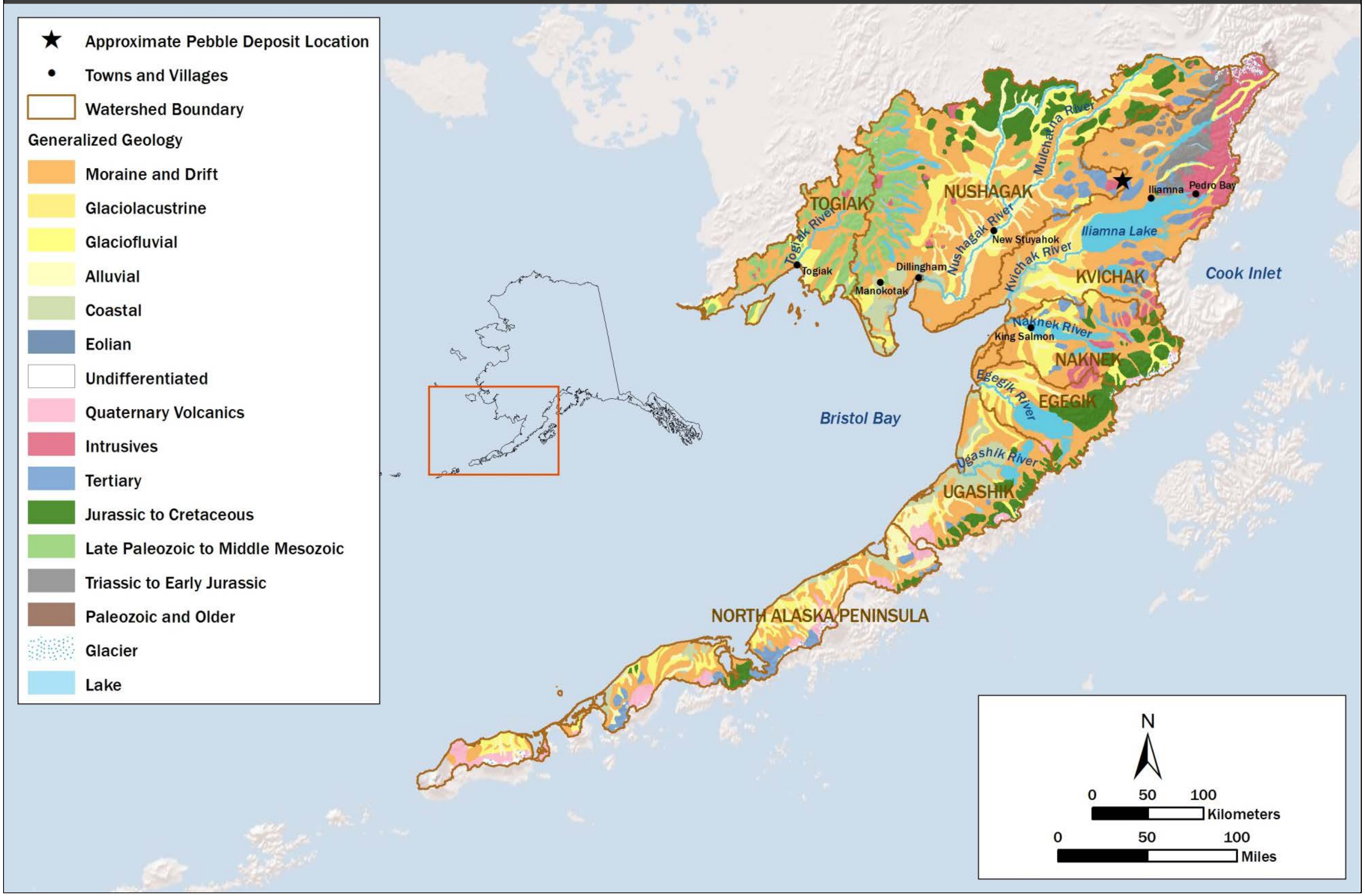


Figure 3-4. Occurrence of permafrost in the Bristol Bay watershed (adapted from Selkregg 1974).

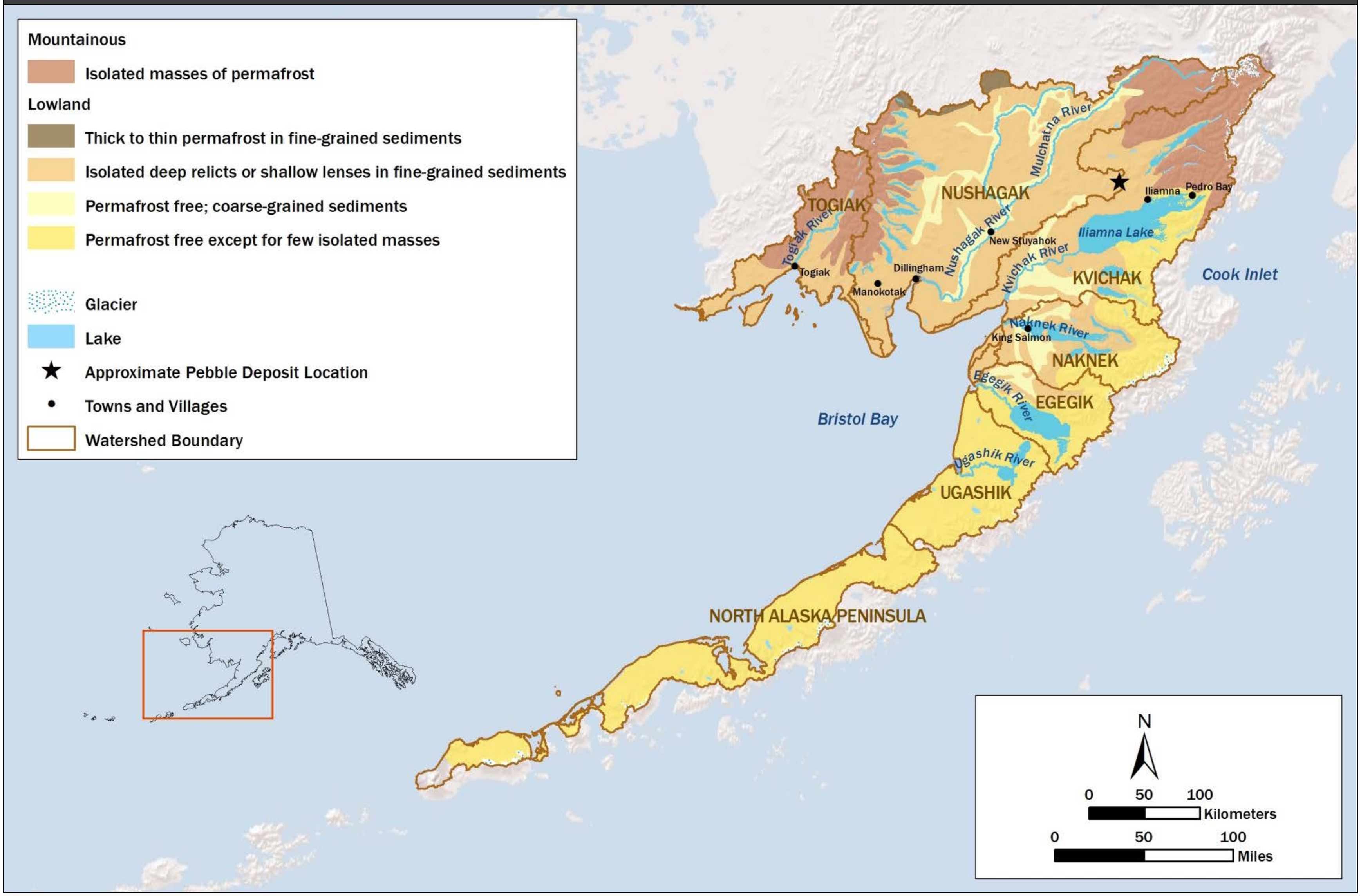


Figure 3-5. Dominant soils in the Bristol Bay watershed (adapted from Selkregg 1974).

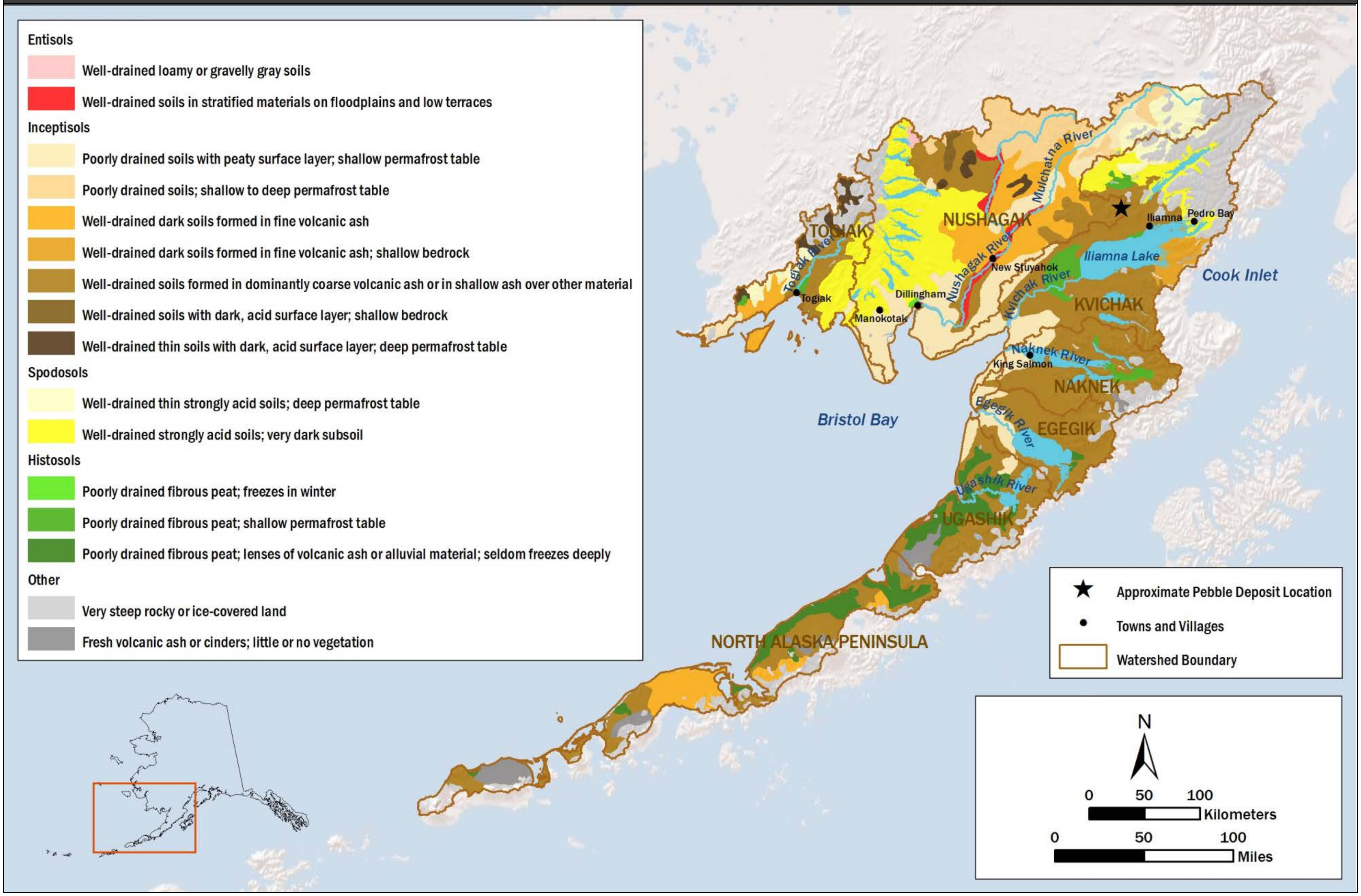


Figure 3-6. Erosion potential in the Bristol Bay watershed (adapted from Selkregg 1974).

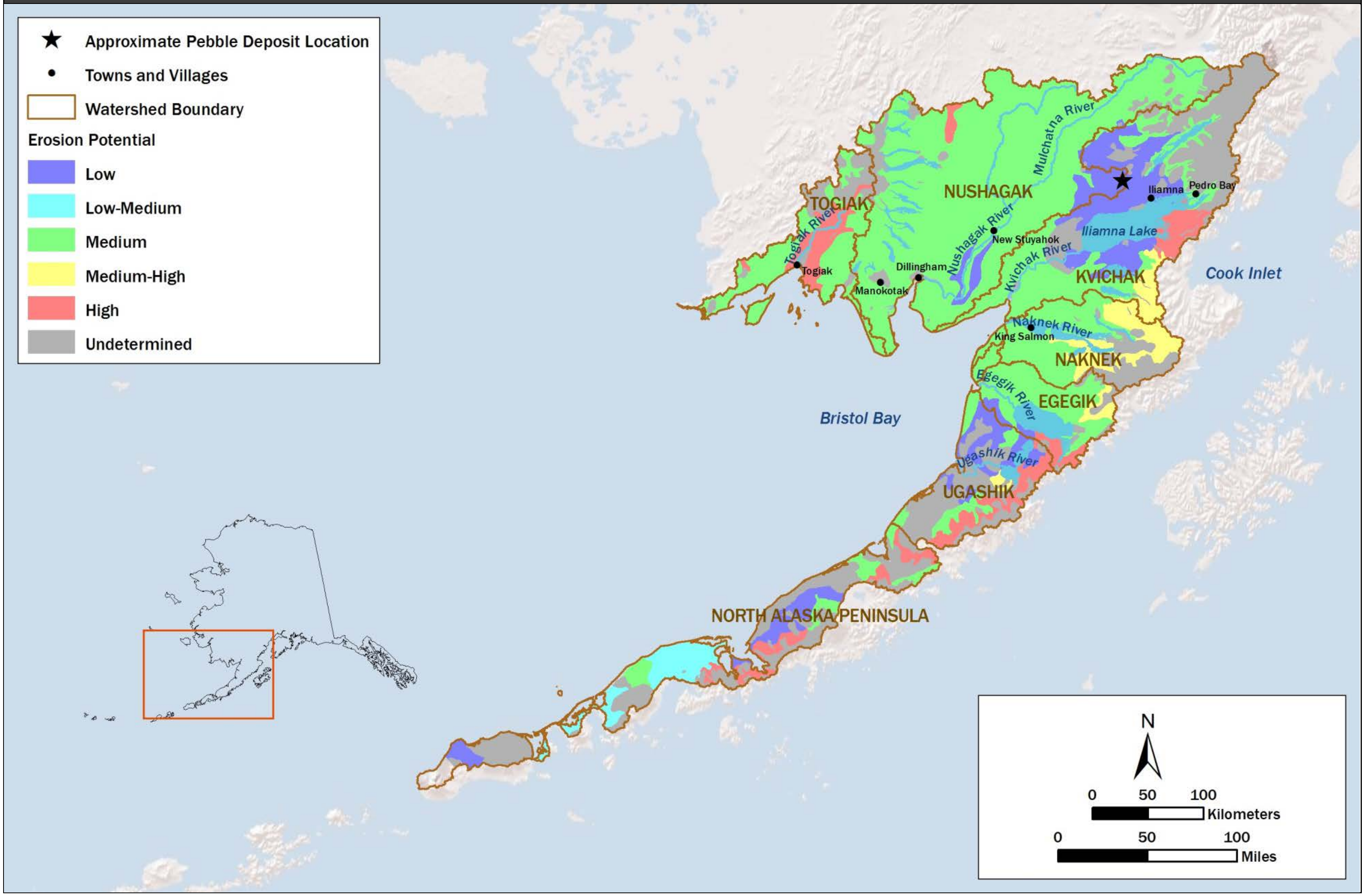


Figure 3-7. Dominant vegetation in the Bristol Bay watershed (adapted from Selkregg 1974).

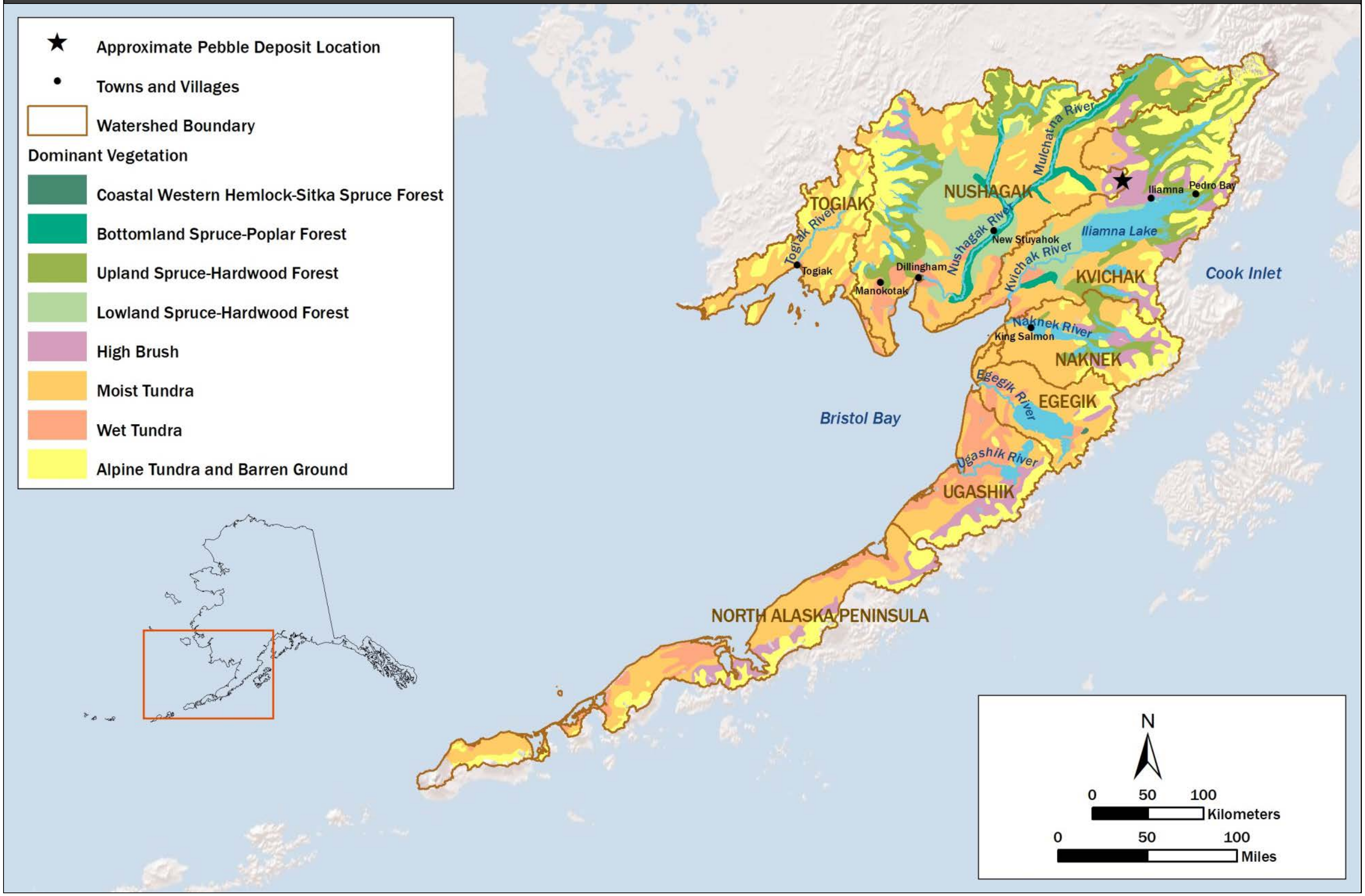
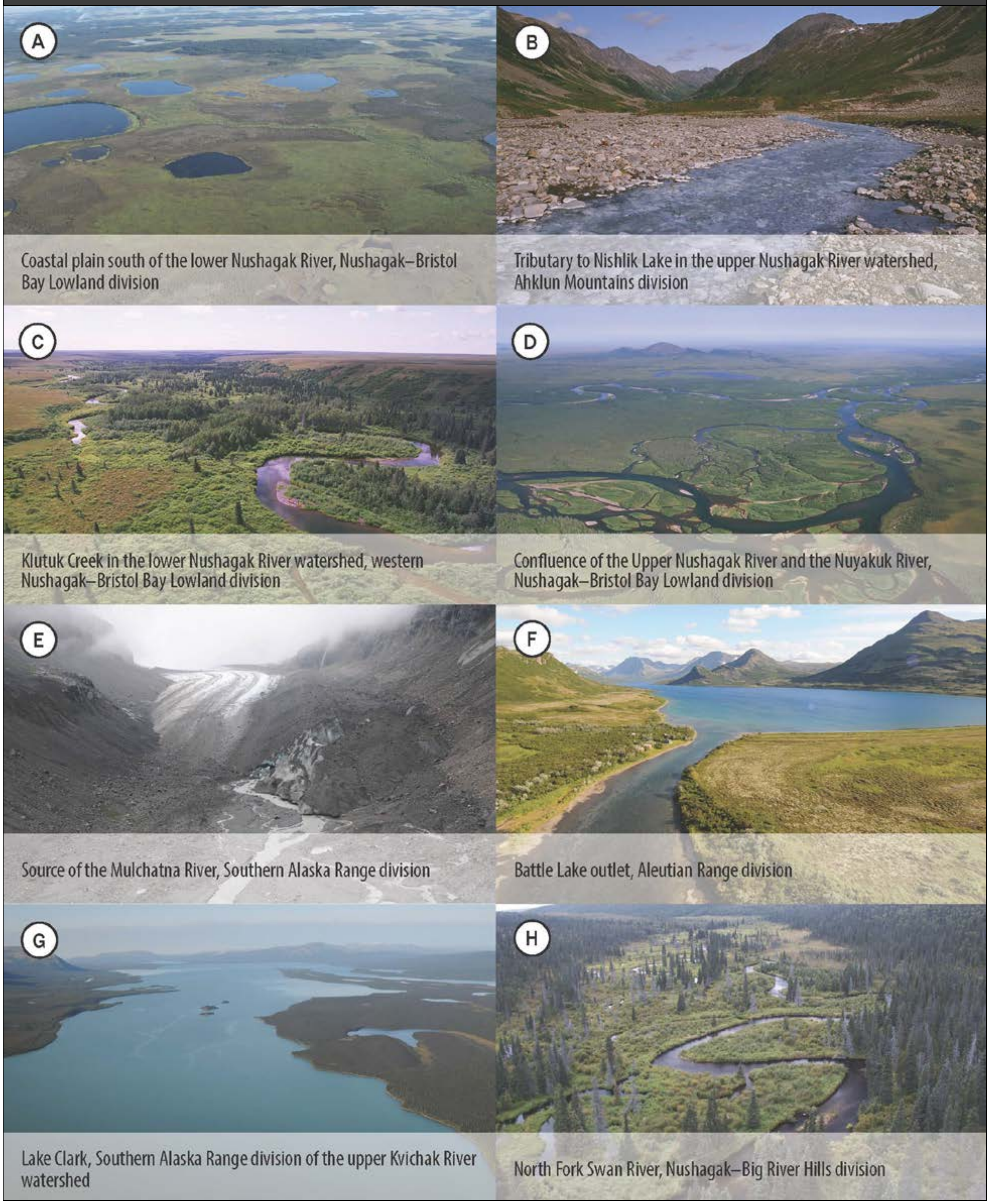


Figure 3-8. Physiographic divisions of the Nushagak and Kvichak River watersheds of Bristol Bay. The Nushagak and Kvichak River watersheds contain a wide range of aquatic habitats within five distinct physiographic divisions; see Figure 3-1 for a map of these divisions and the general location where each photo was taken. All photos taken between August 2003 and August 2013, courtesy of Michael Wiedmer.



Coastal plain south of the lower Nushagak River, Nushagak–Bristol Bay Lowland division

Tributary to Nishlik Lake in the upper Nushagak River watershed, Ahklun Mountains division

Klutuk Creek in the lower Nushagak River watershed, western Nushagak–Bristol Bay Lowland division

Confluence of the Upper Nushagak River and the Nuyakuk River, Nushagak–Bristol Bay Lowland division

Source of the Mulchatna River, Southern Alaska Range division

Battle Lake outlet, Aleutian Range division

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

North Fork Swan River, Nushagak–Big River Hills division

The Nushagak–Big River Hills physiographic division consists largely of rounded ridges that have moderate elevations and broad, gentle slopes and broad, flat or gently sloping valleys (Table 3-1, Figure 3-1) (Wahrhaftig 1965, Selkregg 1974). Major geologic formations include graywacke, argillite, conglomerate, and greenstone flows (Figure 3-3). No modern glaciers are present, but glacial drift and moraines are common throughout lower elevations and colluvium and alluvium mantle higher elevations. The Nushagak River headwaters are the only part of the Nushagak and Kvichak River watersheds that have not been glaciated. In most of this division falling within the Nushagak and Kvichak River watersheds, permafrost is found only in isolated masses or lenses (Figure 3-4). Soils throughout the division are typically shallow, occur in well-drained to poorly drained conditions, and have medium erosion potential (Figures 3-5 and 3-6). Rivers in the Mulchatna and Newhalen River systems originate from glaciers in the Southern Alaska Range. Sediment from these glaciers is trapped in large lakes, providing clearer water for downstream reaches.

The Pebble deposit is located in the eastern portion of the Nushagak–Big River Hills and is heavily influenced by past glaciation (PLP 2011: Chapter 3). At various times, Pleistocene glaciers blocked the South Fork Koktuli River, the North Fork Koktuli River, and Upper Talarik Creek, the three tributaries draining the Pebble deposit area (Figure 2-5). Unconsolidated glacial deposits, ranging from a few to several tens of meters in thickness, cover most of the area's lower elevations (Detterman and Reed 1973). All three of the stream valleys in the Pebble deposit area have extensive glacial sand and gravel deposits (PLP 2011: Chapter 8). Based on studies in the Pebble area, the Pebble Limited Partnership (PLP) (2011) concluded that the presence of permeable shallow aquifers, upward hydraulic gradients, and strong local relief indicate that local and intermediate groundwater flow systems dominate regional groundwater flow systems. Further, PLP (2011) noted the presence of many local, cross-cutting faults with high hydraulic conductivities in the Pebble deposit area.

The Nushagak–Bristol Bay Lowland physiographic division (Table 3-1, Figure 3-1) is mantled with glacial drift and moraine deposits up to hundreds of meters deep, forming a rolling landscape with low local relief (15 to 75 m) and maximum elevations of 90 to 150 m near the transitions from the lowland to adjacent mountains or hills (Wahrhaftig 1965, Detterman 1986, Lea et al. 1991, Stilwell and Kaufman 1996). Arc-shaped bands of morainal deposits ranging from 1.6 to 8 km wide enclose Iliamna Lake and are frequent in the lowlands between the Nushagak River and the Ahklun Mountains division (Figure 3-3). Steep outliers of the Wood River Mountains in the Ahklun Mountains physiographic division arise from the western part of the lowland. A small area with sand dunes occurs east of the Nushagak River (Lea and Waythomas 1990). Glacial drift is coarser near the mountains because of high amounts of outwash and grades to fine sand along the coast (Wahrhaftig 1965). The remainder of the lowland is dominated by low-relief (less than 20 m), rolling expanses of tundra underlain by Holocene peat and wind-born deposits (Lea et al. 1991). Glaciers do not occur today in the Nushagak–Bristol Bay Lowland division, and permafrost is sporadic or absent (Figure 3-4) (Wahrhaftig 1965). Morainal and thaw lakes are common, and mainstem rivers draining this area exhibit high channel complexity (Figure 3-8). Poorly drained soils dominate in the southern portions, whereas well-drained soils dominate across the remainder of the physiographic division (Figure 3-5). Soil erosion potential is moderate throughout the area (Figure 3-6). Extensive dwarf scrub communities occur on relatively well-drained soils, and moist and wet tundra communities cover large areas as well (Figure 3-7) (Selkregg 1974, Gallant et al. 1995).

3.2 Hydrologic Landscapes

To better evaluate the influence of inherent river basin attributes on streamflows and thus fish populations, we used the physiographic divisions discussed above to define different hydrologic landscapes across the Nushagak and Kvichak River watersheds. These landscapes can be considered hydrologic building blocks, in that they provide a broad-scale approach to spatially characterizing climate and watershed factors controlling the amount, timing, and flowpaths of water within the watersheds (Winter 2001).

We defined hydrologic landscapes by calculating water surplus (precipitation minus potential evapotranspiration) across the basins in each of the five physiographic divisions, using Scenarios Network for Alaska and Arctic Planning (SNAP) data (SNAP 2012) and procedures outlined by Feddema (2005). Feddema (2005) defined six annual climate classes ranging from very wet to arid conditions. The very wet, wet, and moist classes have an annual water surplus, whereas the dry, semi-arid, and arid classes have an annual water deficit. Combining these climate classes with the physiographic divisions (Section 3.1), we identified 18 different hydrologic landscapes across the Nushagak and Kvichak River watersheds (Table 3-2, Figure 3-1), which represent the range of hydrologic characteristics across the region.

3.3 Groundwater Exchange and Flow Stability

A key aspect of the Bristol Bay watershed's aquatic habitats is the importance of groundwater exchange. Because salmon rely on clean, cold water flowing over and upwelling and downwelling through porous gravels for spawning, egg incubation, and rearing (Bjornn and Reiser 1991), areas of groundwater exchange 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). Portions of the Nushagak–Bristol Bay Lowland and Nushagak–Big River Hills physiographic divisions, including the Pebble deposit area, contain coarse-textured glacial drift with abundant, high-permeability gravels and extensive connectivity between surface waters and groundwater (Figures 3-3, 3-4, and 3-9). Abundant wetlands and small ponds also contribute disproportionately to groundwater recharge (Rains 2011). This strong connection between groundwater and surface waters helps to moderate water temperatures and streamflows. For example, groundwater contributions that maintain water temperatures above 0^{0}C are critical for maintaining winter refugia in streams that might otherwise freeze (Power et al. 1999).

Table 3-2. Distribution of hydrologic landscapes in the Nushagak and Kvichak River watersheds. Values represent percentage of total area in the two watersheds.

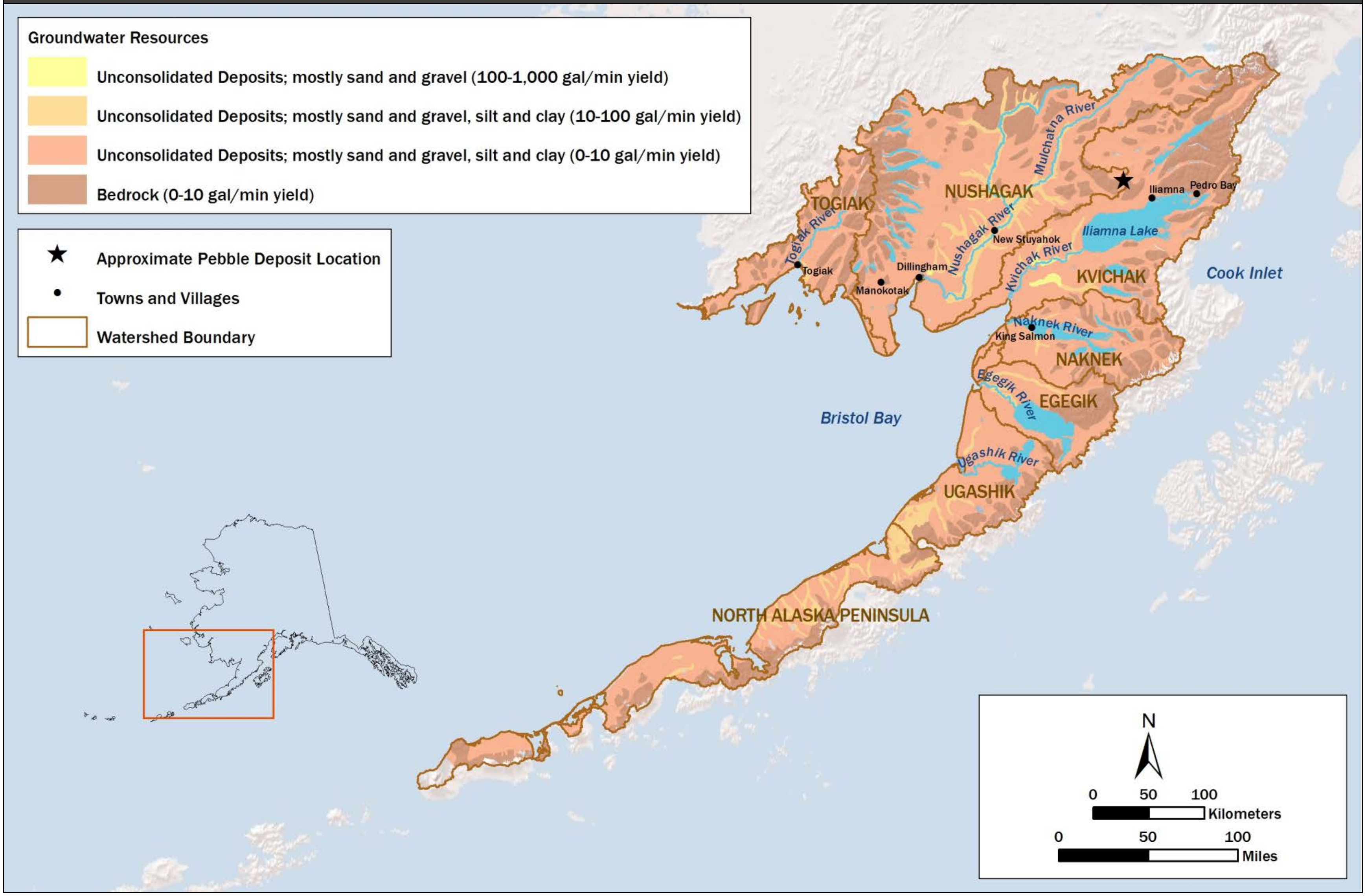
Physiographic Division	Ahklun Mountains			Southern Alaska Range				Aleutian Range			Nushagak–Big River Hills				Nushagak–Bristol Bay Lowland		
Climate Class	V	W	M	V	W	M	D	V	W	M	V	W	M	D	V	W	M
Nushagak River Watershed																	
Nushagak River (whole watershed)	7	16	1	1	2	-	-	-	-	-	-	25	9	-	-	24	15
Nushagak River at Ekwok[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:

Dashes (-) indicate hydrologic landscapes that are not found in that portion of the Nushagak or Kvichak River watersheds. 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 3-9. Groundwater resources in the Bristol Bay watershed (adapted from Selkregg 1974). Yields are presented in gallons per minute.



These groundwater contributions to streamflow, along with the influence of large and small lakes, 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). Greater groundwater contributions to streams result in more moderated streamflow regimes with lower peak flows and higher base flows, creating a less temporally variable hydraulic environment. The lower mainstem Nushagak and Kvichak Rivers illustrate this tendency toward moderated, consistent streamflows (Figure 3-10). Coarse-textured glacial drift in the Kaskanak and Upper Talarik Creek drainages promotes high groundwater contributions to these streams, resulting in stable flows through much of the year (Figure 3-10). High baseflow in the Nushagak River also is 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 the Nushagak–Bristol Bay Lowland (Figure 3-10).

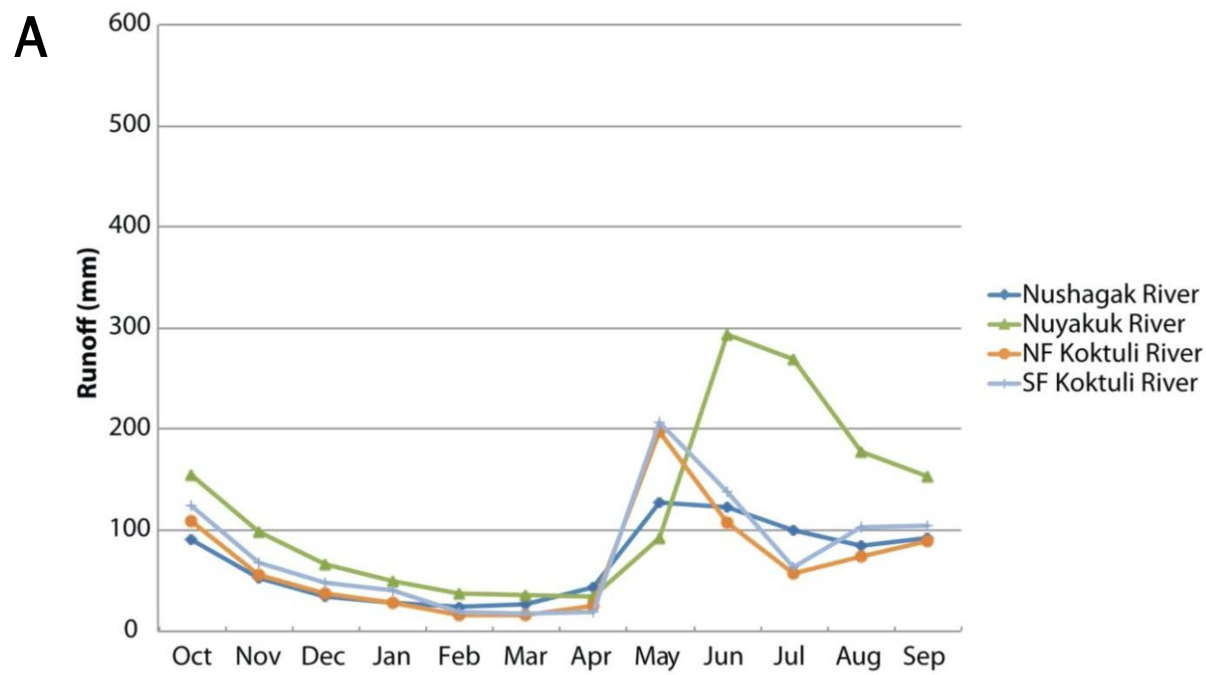
Water storage in upstream lakes plays a role in flow stabilization, as well. For example, in the Kvichak River watershed, Iliamna Lake dampens high flows from the Iliamna and Newhalen Rivers before they reach the mainstem. The attenuating effect of upstream lakes on streamflow is also evident in the Newhalen River, located downstream of Lake Clark (Figure 3-10).

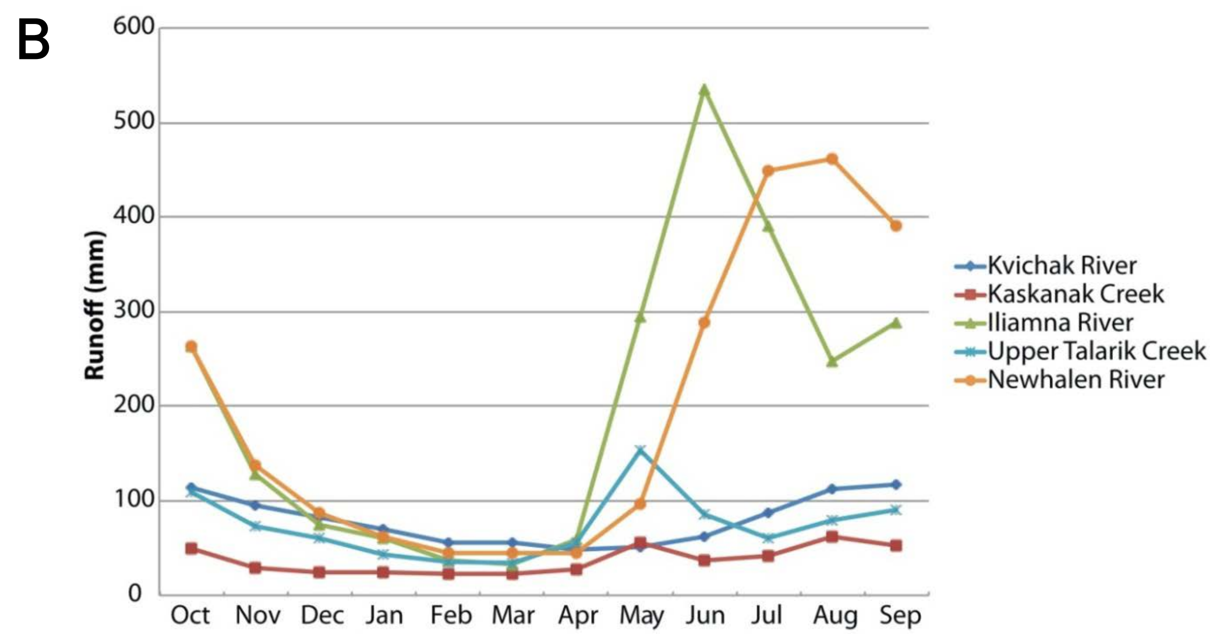
3.4 Quantity and Diversity of Aquatic Habitats

Differences in hydrology, geology, and climate across the Bristol Bay watershed interact to create the region's diverse hydrologic landscapes (Table 3-2, Figure 3-1) and ultimately shape the quantity, quality, diversity, and distribution of aquatic habitats throughout the watershed. These diverse habitats, in conjunction with the enhanced ecosystem productivity associated with anadromous salmon runs, support a high level of biological complexity that contributes to the environmental integrity and resilience of the watershed's ecosystems (Schindler et al. 2010, Ruff et al. 2011, Lisi et al. 2013).

In general, conditions in the Bristol Bay watershed are highly favorable for Pacific salmon. The Nushagak and Kvichak River watersheds encompass an abundant and diverse array of aquatic habitats and support a diverse salmonid assemblage (Section 5.2). Freshwater habitats range from headwater streams to braided rivers, small ponds to large lakes, side channels to off-channel alcoves. These watersheds contain over 54,000 km of streams, 14% of which have been documented as anadromous fish streams (Johnson and Blanche 2012). This percentage is likely a significant underestimate of the actual extent of anadromous waters across the watersheds (Box 7-1, Appendix A).

Figure 3-10. Mean monthly runoff for selected streams and rivers in the Nushagak and Kvichak River watersheds. USGS gages and dates used to generate each line: A. Nushagak River watershed: Nushagak River (15302500, Oct 1977–Sep 1993); Nuyakuk River (15302000, Jun 1953–Sep 2010); North Fork (NF) Koktuli River (15302250, Sep 2004–Sep 2010); South Fork (SF) Koktuli River (15302200, Sep 2004–Sep 2010). B. 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).





Lakes and associated tributary and outlet streams are key spawning and rearing areas for sockeye salmon. Lakes 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 (RAP 2011). 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%) (RAP 2011). Relatively low watershed elevations (especially in the extensive Nushagak–Bristol Bay Lowland physiographic division) and the absence of artificial barriers to migration (e.g., dams and roads) mean that not only are streams, lakes, and other aquatic habitats abundant in the Bristol Bay region, but they also tend to be accessible to anadromous salmonids. With very few exceptions, all major lakes in the watershed are accessible to anadromous salmon (Appendix A). Lakes and ponds also play a key role in groundwater dynamics and flow stability (Section 3.3).

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, RAP 2011).

3.4.1 Stream Reach Characterization: Attributes

To characterize the stream and river habitats in the Nushagak and Kvichak River watersheds, we described stream and river valley attributes for each of the 52,277 stream and river reaches (54,427 km) in the Nushagak and Kvichak River watersheds documented in the National Hydrography Dataset (NHD) (USGS 2012). We excluded another 27,186 reaches (7,936 km) for which we could not identify reach-specific drainage areas from the analysis. For each reach, we estimated the mean annual streamflow (m^3/s), mean channel gradient (%), and percent of flatland in the contributing watershed lowland (% flat); each attribute is described in detail in the following sections. These attributes were selected because they represent fundamental aspects of the physical and geomorphic settings in streams, providing context for stream and river habitat development and subsequent fish habitat suitability (Burnett et al. 2007). It also was feasible to obtain these attributes for the entire area given available data. These attributes have been used to model habitat suitability for salmon at large scales, for example via intrinsic potential modeling (Burnett et al. 2007, Shallin Busch et al. 2011). We did not develop intrinsic potential models for salmon species in this assessment, as that effort would require multiple years of field data collection for model validation and testing and those data are not currently available. However, our characterization results do provide insights into the distribution of broad-scale habitat conditions within the watersheds, and could provide the basis for future intrinsic potential model development.

3.4.1.1 Channel Gradient

Channel gradient broadly characterizes channel steepness and geomorphic form. Channel gradient and associated aspects of channel morphology influence channel capacity to transport sediment, affecting channel response to disturbance (Montgomery and Buffington 1997). Channel morphology can strongly influence suitability for salmon rearing and spawning. Specific substrate and hydraulic requirements vary slightly by species (Appendix A), but stream-spawning salmon generally require relatively clean gravel-sized substrates with interstitial flow, and sufficient bed stability to allow eggs to incubate in place for months prior to fry emergence (Quinn 2005).

Montgomery and Buffington (1997) proposed a process-based classification of mountain streams. Field data from their study indicated that gradients estimated by digital elevation models (DEMs) provide a useful predictor of channel morphology. We estimated the channel gradient of each stream reach in the Nushagak and Kvichak River watersheds by assessing the gradient of correlated flowpaths across a 30-m-cell National Elevation Dataset DEM (Gesch et al. 2002, Gesch 2007, USGS 2013) (Box 3-1). We adapted the classification scheme put forth by Montgomery and Buffington (1997) to define four gradient classes and predicted channel morphologies for stream reaches at different watershed scales.

- Less than 1%, dune-ripple or pool-riffle morphology.
- At least 1% and less than 3%, plane-bed morphology.
- At least 3% and less than 8%, step-pool morphology.
- At least 8%, cascade morphology.

The substrate and hydraulic conditions required by stream-spawning salmon are most frequently met in stream channels with gradients less than 3% (Montgomery et al. 1999). At the lowest gradients, the channel's capacity to transport fine sediments will be low and substrates may be dominated by sands and other fines, providing suboptimal salmon spawning habitat. A notable exception to this generality occurs in low-gradient, off-channel habitats and ponds that may be dominated by fine sediments but that contain areas of upwelling. These areas are used by riverine-spawning (Eiler et al. 1992) and pond-spawning (Quinn et al. 2012) sockeye salmon. At gradients above 3%, channels develop step-pool or cascade morphologies and the size, stability, and frequency of pockets of suitable spawning substrates decrease substantially (Montgomery and Buffington 1997). In the Bristol Bay region, gradients of productive stream reaches for salmon are typically less than 3%, with gradients less than 1% characterizing the most productive reaches; these habitats include lake outlets and lower tributary reaches, and most of the major spawning reaches and tributaries of the Nushagak and Kvichak River watersheds (Figures 3-11 and 3-12) (Demory et al. 1964). We note, however, that low-gradient watersheds in the coastal plain region of the Nushagak–Bristol Bay Lowland that lack upland headwaters are generally not productive salmon habitats. These streams tend to have lower dissolved oxygen levels, be characterized by fine-textured substrates with high proportions of organic material, and may lack substrates coarser than sand, presumably due to lack of higher-gradient source areas for gravel recruitment (ADF&G 2012, Wiedmer pers. comm.).

Environmental conditions determining suitability for juvenile salmon and adult resident salmonids (e.g., resident Dolly Varden; Box 2-3) are also influenced by gradient. Fish movement can be restricted by the high water velocities and frequent drops found in streams with gradients exceeding 12%, although Dolly Varden have been found at gradients exceeding 15% in southeast Alaska streams

(Wissmar et al. 2010). Gradient and channel roughness also influence the distribution of water velocities and hydraulic conditions in streams, influencing food delivery rates and availability and subsequent energetic demands of drift feeding fish (Hughes and Dill 1990).

BOX 3-1. METHODS FOR CHARACTERIZING CHANNEL GRADIENT

The valley gradient of each stream reach in the Nushagak and Kvichak River watersheds was estimated by assessing the gradient of correlated flowpaths along across a 30-m cell National Elevation Dataset digital elevation model (DEM) (Gesch et al. 2002, Gesch 2007, USGS 2013). We found the measured gradient of the National Hydrography Dataset (NHD) flowlines (based on the elevation of the underlying DEM) was not an accurate representation of channel gradient because of inconsistencies between the mapped streams and rivers in the NHD and the topography described by the DEM. Channel traces in the NHD did not reliably follow the valley floor, and upslope traces and misalignment with the DEM resulted in inaccurate measures of stream gradients and sampled elevations.

We determined that the gradient of streams in a drainage network described by a flow analysis across the DEM would more accurately represent channel morphology given the data available. The drainage network of the DEM paralleled the network of NHD flowlines, but included or excluded some small tributaries and lacked the sinuosity mapped in the NHD.

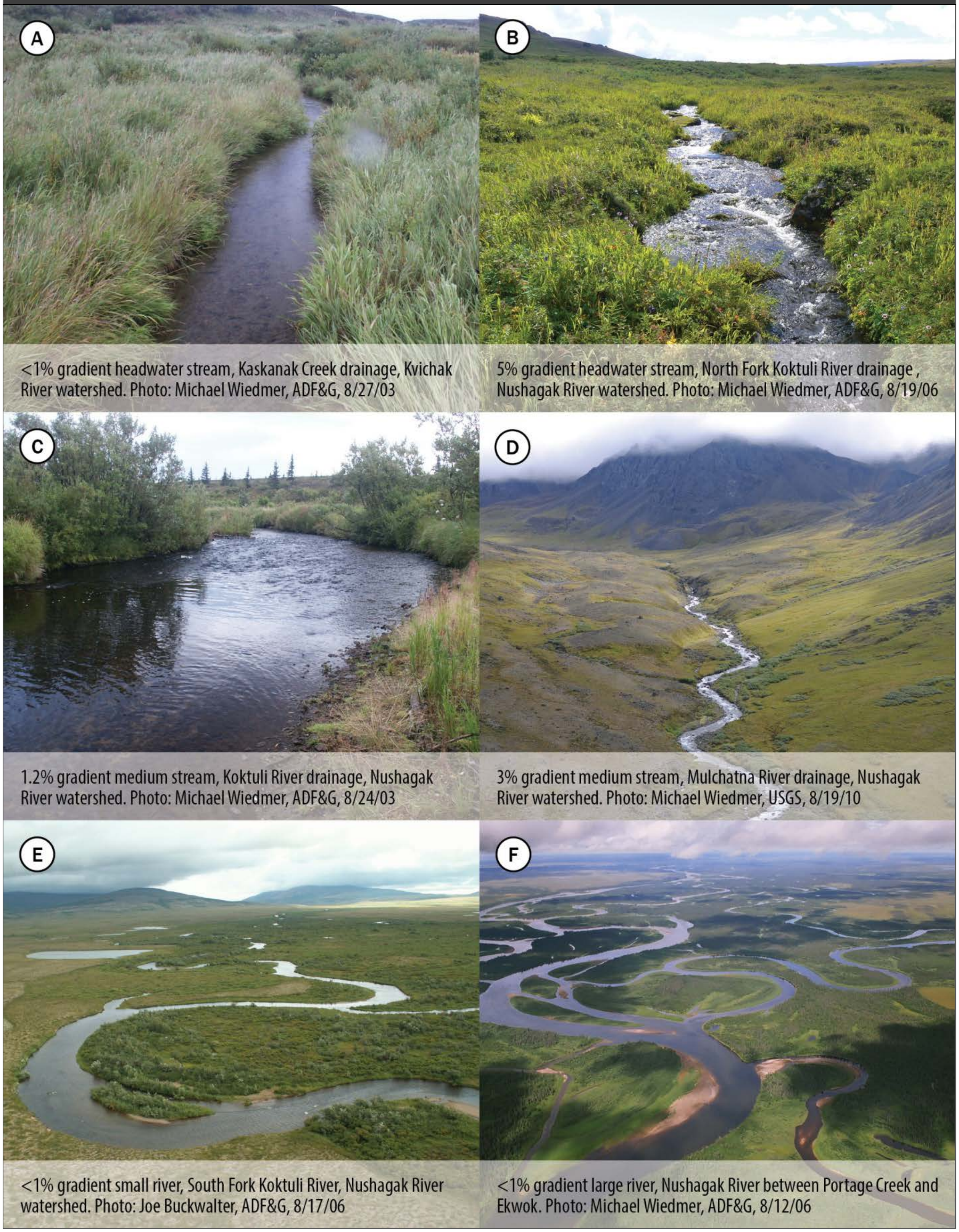
Gradients of flowlines across the DEM were determined using the hydrology tools of the Spatial Analyst extension of ArcGIS. First, the hydraulic network was generated based on the topography of the NHD DEM. Generation of the hydraulic network involved the following tools:

- **Fill.** Sinks in the DEM were filled so that continuous flowpaths could be described.
- **Flow direction.** The steepest path or flow direction was determined from each cell in the DEM.
- **Flow accumulation.** Based on the direction of flow, the total number of cells, or receiving area for each cell in the DEM, was determined.
- **Reclassify.** A threshold value of 0.25 km^2 was applied to the total receiving area output from the previous step to distinguish streams from non-streams.
- **Stream link.** The resulting network was processed to assign unique identifiers to each link in the drainage network.

To determine the gradient of each stream link in the drainage network, and to generate geometry that could assign these values to the reaches of the NHD flowlines, the following tools were used:

- **Extract by mask.** Elevation values underlying the drainage network were isolated from the DEM so that cross-valley slopes would not be measured when determining gradient.
- **Slope.** Gradient along the drainage network was measured between each cell of the isolated drainage network DEM. The drainage DEM confined the slope measures to the flowpath of the drainage network, providing an estimate of stream gradient at each 30-m cell.
- **Watershed.** The output of the Stream Link tool (see above) and the results of the flow direction analysis were used to delineate the drainage basin for each stream link. This geometry was then used to transfer gradient values to the NHD stream reaches.
- **Zonal statistics.** In the drainage basin for each stream segment, the average gradient was determined for all cells with values (i.e., a mean gradient of the stream segment). Mean gradient values were then assigned to the drainage basin geometry.
- **Zonal statistics as table.** The mean gradient for each drainage basin was used to calculate the channel gradient for each NHD flowline. This tool measured the length-weighted mean of the gradients for each reach (as defined by the NHD Reach Code attribute) from the means calculated for each drainage basin. Typically, the NHD flowlines occupied no more than two drainage basins. The resulting gradient estimates were appended to the table of NHD flowlines.

Figure 3-11. Examples of different stream size and gradient classes in the Nushagak and Kvichak River watersheds.



<1% gradient headwater stream, Kaskanak Creek drainage, Kvichak River watershed. Photo: Michael Wiedmer, ADF&G, 8/27/03

5% gradient headwater stream, North Fork Koktuli River drainage , Nushagak River watershed. Photo: Michael Wiedmer, ADF&G, 8/19/06

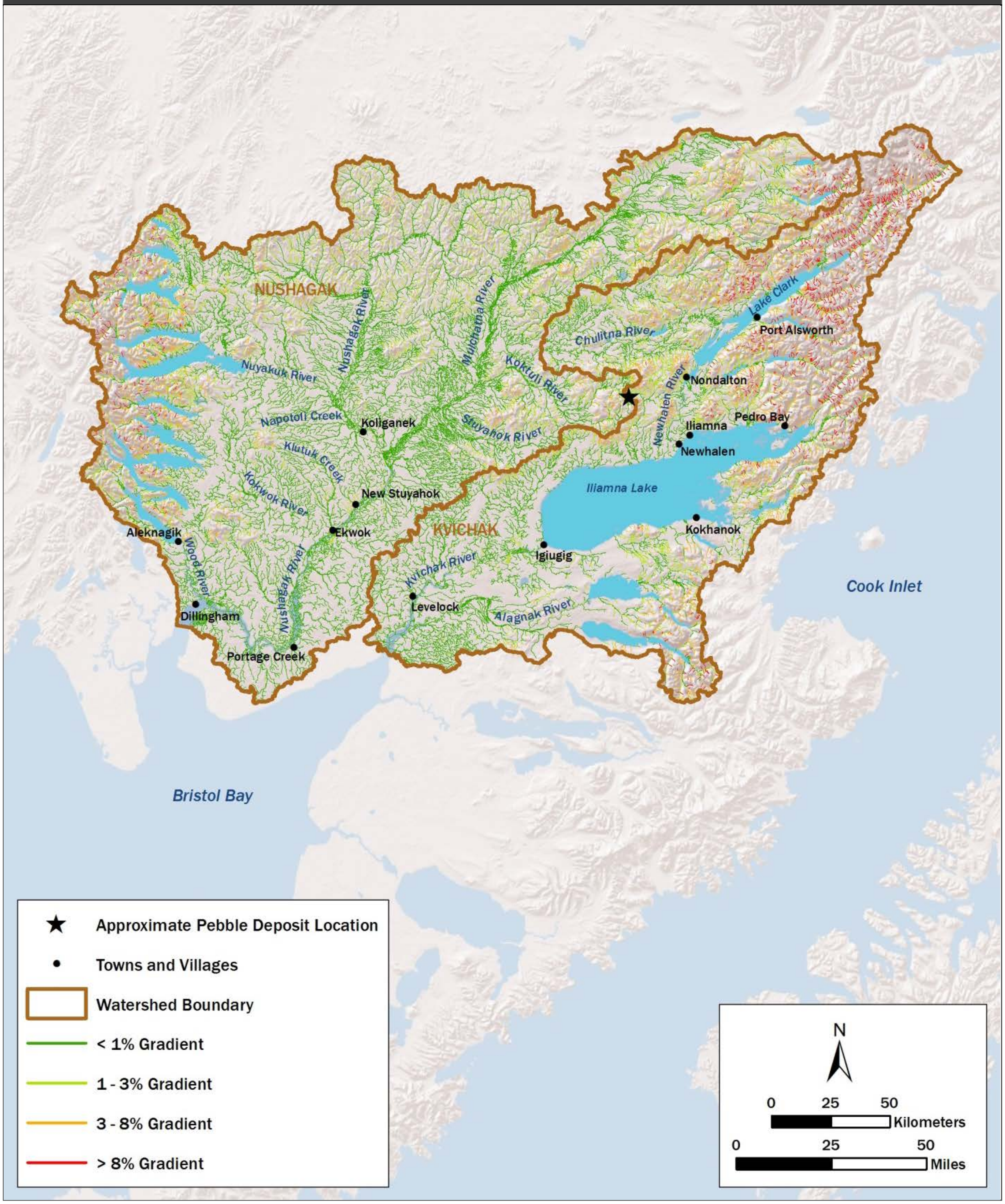
1.2% gradient medium stream, Koktuli River drainage, Nushagak River watershed. Photo: Michael Wiedmer, ADF&G, 8/24/03

3% gradient medium stream, Mulchatna River drainage, Nushagak River watershed. Photo: Michael Wiedmer, USGS, 8/19/10

<1% gradient small river, South Fork Koktuli River, Nushagak River watershed. Photo: Joe Buckwalter, ADF&G, 8/17/06

<1% gradient large river, Nushagak River between Portage Creek and Ekwok. Photo: Michael Wiedmer, ADF&G, 8/12/06

Figure 3-12. Channel gradient classes in the Nushagak and Kvichak River watersheds. Valley gradient was assessed by measuring drainage channel slope across the watersheds' landscapes (Box 3-1).



3.4.1.2 Mean Annual Streamflow

Mean annual streamflow is a metric of stream size, an important determinant of available habitat space (capacity) for stream fishes. The relationship between mean annual streamflow and habitat capacity for rearing juvenile salmon can vary with streamflow regime and other limiting factors, but is generally positive when other factors are not constraining.

Mean annual streamflow for each stream reach within the Nushagak and Kvichak River watersheds was estimated using regression equations for the prediction of mean annual streamflow, based on drainage area and historical mean annual precipitation in southwestern Alaska (Parks and Madison 1985) (Box 3-2). We defined four classes of stream size based on these mean annual streamflow calculations.

- Small headwater streams (less than 0.15 m^3/s), including many of the tributaries of the South and North Fork Koktuli Rivers and Upper Talarik Creek.
- Medium streams (0.15 to 2.8 m^3/s), including the upper reaches and larger tributaries of the South and North Fork Koktuli Rivers and Upper Talarik Creek.
- Small rivers (2.8 to 28 m^3/s), including the middle to lower portions of South and North Fork Koktuli Rivers, and Upper Talarik Creek, and the mainstem Koktuli River.
- Large rivers (greater than 28 m^3/s), including the Mulchatna River below the confluence with the Koktuli River, the Newhalen River, and other larger rivers.

All five species of Pacific salmon present in the Bristol Bay region use portions of large and small rivers and medium streams for migration, spawning, and/or rearing habitat. Research in the Wood River system suggests that larger stream sizes allow multiple salmon species to coexist, perhaps due to habitat partitioning made possible by increased space and habitat diversity (Pess et al. 2013). Salmon also use small streams in the Bristol Bay region for spawning and rearing, but use of these habitats may be constrained by shallow depths, insufficient streamflow to allow passage, the unavailability of open water in winter, or other limitations related to stream size.

Salmonid species differ in their propensities for small streams. Dolly Varden have been documented using all stream sizes, including some of the smallest channels. Of the Pacific salmon species, coho salmon are most likely to use small streams for spawning and rearing, and have been observed in many of the smaller streams near the Pebble and other deposits. Larger-bodied Chinook salmon adults are less likely to access smaller streams for spawning (Quinn 2005). However, juvenile Chinook salmon are observed in small tributaries where spawning has not been documented.

BOX 3-2. METHODS FOR CHARACTERIZING MEAN ANNUAL STREAMFLOW

Mean annual streamflow for each stream reach in the Nushagak and Kvichak River watersheds was estimated using regression equations, based on drainage area and historical mean annual precipitation data in southwestern Alaska (Parks and Madison 1985). Total drainage area was determined for reaches along the National Hydrography Dataset (NHD) flowlines by developing a drainage-corrected digital elevation model (DEM) based on the National Elevation Dataset (NED). Although the underlying topography and catchments described by the NED remained the same, the elevations underlying the NHD flowlines and in their immediate vicinity were lowered and smoothed such that runoff conformed to the geometry of the NHD flowlines.

Using the drainage-corrected DEM, we estimated total catchment area above any location in the drainage network. The NED DEM was corrected to better conform to the NHD flowlines and drainage areas were calculated using the following tools of the ArcHydro and Spatial Analyst tools of the ArcGIS suite:

- **DEM reconditioning.** The elevations of the DEM were altered along the NHD flowlines and in their immediate vicinity. Parameters used for this tool were a 10-m reduction of elevations along the flowline, a 5-cell (150-m)-wide transition zone on either side of the flowline, and a post-process 1-km reduction in elevations along the flowlines. The initial elevation reduction and transition width were found to adequately capture flows and maintain those flows within the channel geometry. The post-processing adjustment is a more arbitrary value intended to confine flows to the channels once captured.
- **Fill.** Sinks in the reconditioned DEM were filled so that continuous flowpaths could be described.
- **Flow direction.** The steepest path or flow direction was determined from each cell in the DEM.
- **Flow accumulation (drainage area).** Based on the direction of flow, the total number of cells, or receiving area for each cell in the DEM, was determined. These values were multiplied by 0.0009 to convert the area of each cell (900 m^2) to square kilometers.
- **Flow accumulation (accumulated precipitation).** Due to variation in precipitation patterns across the study area, the average accumulated precipitation was calculated by using the flow accumulation tool with a weight assigned to each cell based on the average annual precipitation data for 1971 to 2001 (SNAP 2012). The result was divided by the total number of cells accumulated at each location on the grid to determine the average accumulated annual precipitation.

The output drainage area raster and raster coverage of average annual precipitation were used as inputs for the mean annual streamflow regression equation developed by Parks and Madison (1985) for southwestern Alaska:

$$Q = (10^{-1.38})*(DA^{0.98})*(P^{1.13})$$

where Q is mean annual flow in cubic feet per second, DA is drainage basin area in square miles, and P is mean annual precipitation in inches per year. We used the median mean annual streamflow value from the cells within the drainage network that corresponded to each NHD flowline as the estimate of mean annual streamflow for the stream segment.

3.4.1.3 Proportion of Flatland in Lowland

Stream channels in mountainous and foothill terrain are laterally constrained by their valley walls to varying degrees. Degree of channel constraint influences channel form, including the development of off-channel habitats, variability in local channel gradients, and hydraulic conditions during over-bank flows. Unconstrained channels generally have higher complexity of channel habitat types and hydraulic conditions and higher frequencies of off-channel habitats such as side channels, sloughs, and beaver ponds. Such habitat complexity can be beneficial to salmon by providing a diversity of spawning and rearing habitats throughout the year (Stanford et al. 2005).

To provide an index of the degree of channel constraint expected within each stream reach, we estimated the percent of flatland (less than 1% slope) within lowland (area below median elevation) for each stream reach's adjacent drainage basin (Box 3-3). Visual inspection of portions of the study area where high-resolution aerial photographs were available showed that channels were typically unconstrained when the proportion of flatland in lowland exceeded 5%. This threshold was used to identify two classes:

- Less than 5% flatland in lowland, indicating reaches are constrained and have limited floodplain area. These reaches are classified as having low or no floodplain potential.
- Greater than or equal to 5% flatland in lowland, indicating reaches are unconstrained and have high likelihood for floodplain development. These reaches are classified as having floodplain potential.

In the Bristol Bay region, streams that are unconstrained and able to develop complex off-channel habitats are more likely to provide a diversity of channel habitat types and hydraulic conditions, creating favorable conditions, particularly for salmonid rearing. For Chinook and coho salmon, as well as river-rearing sockeye salmon that may overwinter in streams, such habitats may be particularly valuable. The percent flatland in lowland metric is not a perfect index of channel constraint, however. Channels in flat lowlands such as the coastal Nushagak–Bristol Bay Lowlands physiographic division (Figure 3-1) may actually be incised into fine-grained sediments with very little off-channel habitat complexity. In the glacially worked landscapes of the Bristol Bay region, streams may be constrained by relatively flat valley terraces and moraine deposits that are not distinguishable on the coarse-scale DEM available for the region. Terraces are a common feature in portions of the region, but the degree to which terrace constraint influences these results could not be determined from the existing DEM. In steep, mountainous terrain, narrow valleys may occasionally allow for unconstrained stream channel development across low-gradient floodplains, but these features are likely not always detected with the DEM resolution currently employed for this effort.

3.4.2 Stream Reach Characterization: Results

We estimated the three stream-reach attributes discussed above in four geographically defined areas that vary in scale and location (as described in Section 2.2.2).

- The Nushagak and Kvichak River watersheds (Scale 2).
- The mine scenario watersheds—that is, the South Fork Koktuli River, the North Fork Koktuli River, and the Upper Talarik Creek watersheds (Scale 3).
- The streams lost to the Pebble 6.5 scenario footprint (Scale 4).
- The subwatersheds of the transportation corridor area (Scale 5).

In this section, we summarize results for the Nushagak and Kvichak River watersheds to broadly characterize the region. Results for the other three geographic scales are reported later in the assessment (Sections 7.2.1 and 10.2), where we evaluate potential impacts of large-scale mining.

BOX 3-3. METHODS FOR CHARACTERIZING PERCENT FLATLAND IN LOWLAND

The relative degree of channel constraint in the Nushagak and Kvichak River watersheds was estimated by calculating the percent of flatland (<1% slope) within lowland (area below median elevation) in each stream reach's adjacent drainage basin. These calculations included the delineation of drainage basins of the drainage-corrected drainage network (developed for the mean annual streamflow analysis; see Box 3-2) as well as elevation and slope analyses of the unaltered digital elevation model (DEM).

To establish the drainage basin geometry of the drainage-corrected flow analysis, the following Spatial Analyst tools were applied within an ArcGIS workspace.

- **Reclassify.** A threshold value of 0.25 km^2 was applied to the total receiving area output from the drainage-corrected flow analysis to distinguish streams from non-streams.
- **Stream link.** The resulting network was processed to assign unique identifiers to each link in the drainage network.
- **Watershed.** The output of the Stream Link tool (see above) and the results of the flow direction analysis were used to delineate the drainage basin for each stream link. This geometry was used as the geographic extent of analysis for each stream segment.

Areas of flatland and lowland were then identified for each drainage basin. The unaltered National Elevation Dataset DEM was processed with the following Spatial Analyst tools from ArcGIS.

- **Slope.** The original (not drainage-corrected) DEM was analyzed to determine slope (%) across the extent of the Nushagak and Kvichak River watersheds.
- **Reclassify.** A threshold value of 1% was applied to the slope analysis, and attributes were assigned across the study area as meeting or not meeting the flatland criteria.
- **Zonal statistics.** In the drainage basin for each stream segment, the minimum and maximum elevations were determined using the Zonal Statistics tool. These values were used to identify the median elevation for each watershed.
- **Reclassify.** The DEM was classified as meeting or not meeting the lowland criteria based on results of the previous step.

Finally, the percent flatland in lowland for each stream reach's drainage basin was calculated using the following steps.

- **Times.** Areas of flatland outside of lowland areas were eliminated by multiplying the flatland and lowland rasters. The flatland and lowland rasters used 1 and 0 values for true and false, respectively, so both conditions were required to return a positive result for flatland in lowland.
- **Zonal statistics.** The total areas of lowland and flatland within lowland were calculated for each drainage basin.
- **Divide.** The percent flatland in lowland was determined for each drainage basin by dividing the area of flatland in lowland by the area of lowland in each drainage basin.
- **Zonal statistics as table.** The average value of percent flatland in lowland for each stream reach was calculated and added to a table, which was then appended to the National Hydrography Dataset (NHD) flowline data table. Although the mean statistic was used to ascertain these values for the NHD flowlines, the flowlines typically had a one-to-one correlation with drainage basins, as the basins were based on the drainage-corrected flow analysis.

We characterized 54,427 km of streams and 52,277 stream and river reaches in the Nushagak and Kvichak River watersheds. Reach attributes reflected the hydrologic landscapes in which the reaches occurred and upstream within each reach's drainage (Section 3.2). Relatively low-gradient stream channels extend far up into the headwaters of the upper Mulchatna and Nushagak River watersheds (Figure 3-12), allowing salmon to access headwater streams. High-gradient conditions are primarily found in the headwaters of Lake Clark and Iliamna Lake tributaries and the headwaters of the Alagnak, Wood, Kokwok, and Nuyakuk Rivers (Figure 3-12). Valley flatland is heavily concentrated in the Nushagak–Bristol Bay Lowlands physiographic division and along the larger rivers, but includes significant wider-valley reaches in the Nushagak–Big River Hills, Southern Alaska Range, and Aleutian Range divisions (Figure 3-13).

The majority of stream channel length (75%) in the Nushagak and Kvichak River watersheds is composed of low-gradient (less than 3%), medium and small (less than 2.8 m^3/s mean annual streamflow) streams (Table 3-3, Figures 3-12 and 3-14). The extent of flatland in valley lowlands is strongly associated with gradient. For streams with less than 1% gradient, 55% have high floodplain potential (i.e., greater than or equal to 5% flatland in lowland). In contrast, less than 5% of streams with gradients greater than 1% have high floodplain potential. Stream reaches with greater than 3% gradient were only found in landscapes where floodplain potential was low (i.e., less than or equal to 5% flatland in lowland). Overall, these results reveal the high proportion of stream channels in these watersheds that possess the broad geomorphic and hydrologic characteristics enabling the development of stream and river habitats highly suitable for fishes such as Pacific salmon, Dolly Varden, and rainbow trout.

3.5 Water Quality

3.5.1 Water Chemistry

Water quality of streams near the Pebble deposit has been characterized extensively (PLP 2011, Zamzow 2011). The streams draining the watersheds in the Pebble deposit area (Figure 2-5) are neutral to slightly acidic, with low conductivity, hardness, dissolved solids, suspended solids, and dissolved organic carbon (see Section 8.2.1.1 for more detailed discussion of water chemistry in streams draining the mine scenario watersheds). In those respects, they are characteristic of undisturbed streams. However, as would be expected for a metalliferous site, levels of sulfate and some metals (copper, molybdenum, nickel, and zinc) are elevated, particularly in the South Fork Koktuli River. PLP (2011) 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 or close to the deposit and the number and magnitude of exceedances decreased with distance downstream (PLP 2011: Figure 9.1-35, 60, 61, 65, and 66).

Figure 3-13. Likelihood of floodplain potential, as measured by the percent flatland in lowland areas, for the Nushagak and Kvichak River watersheds. Flatland refers to land with less than 1% slope; lowland areas are defined as areas below the midpoint elevation within the drainage basin of each stream reach (Box 3-3).

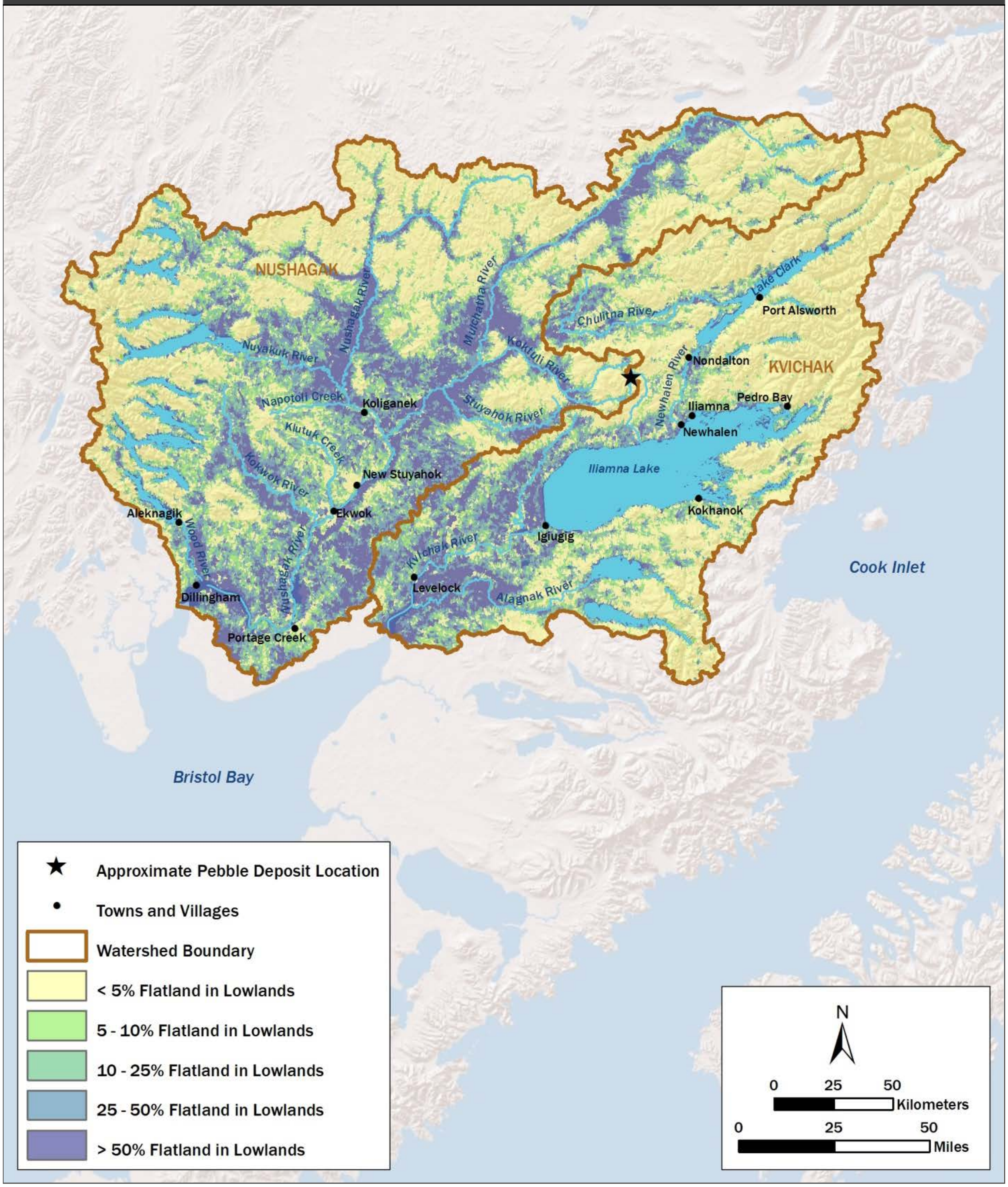


Figure 3-14. Stream size classes in the Nushagak and Kvichak River watersheds as determined by mean annual streamflow. Mean annual streamflow for streams and rivers was estimated using drainage area and mean annual precipitation (Box 3-2).

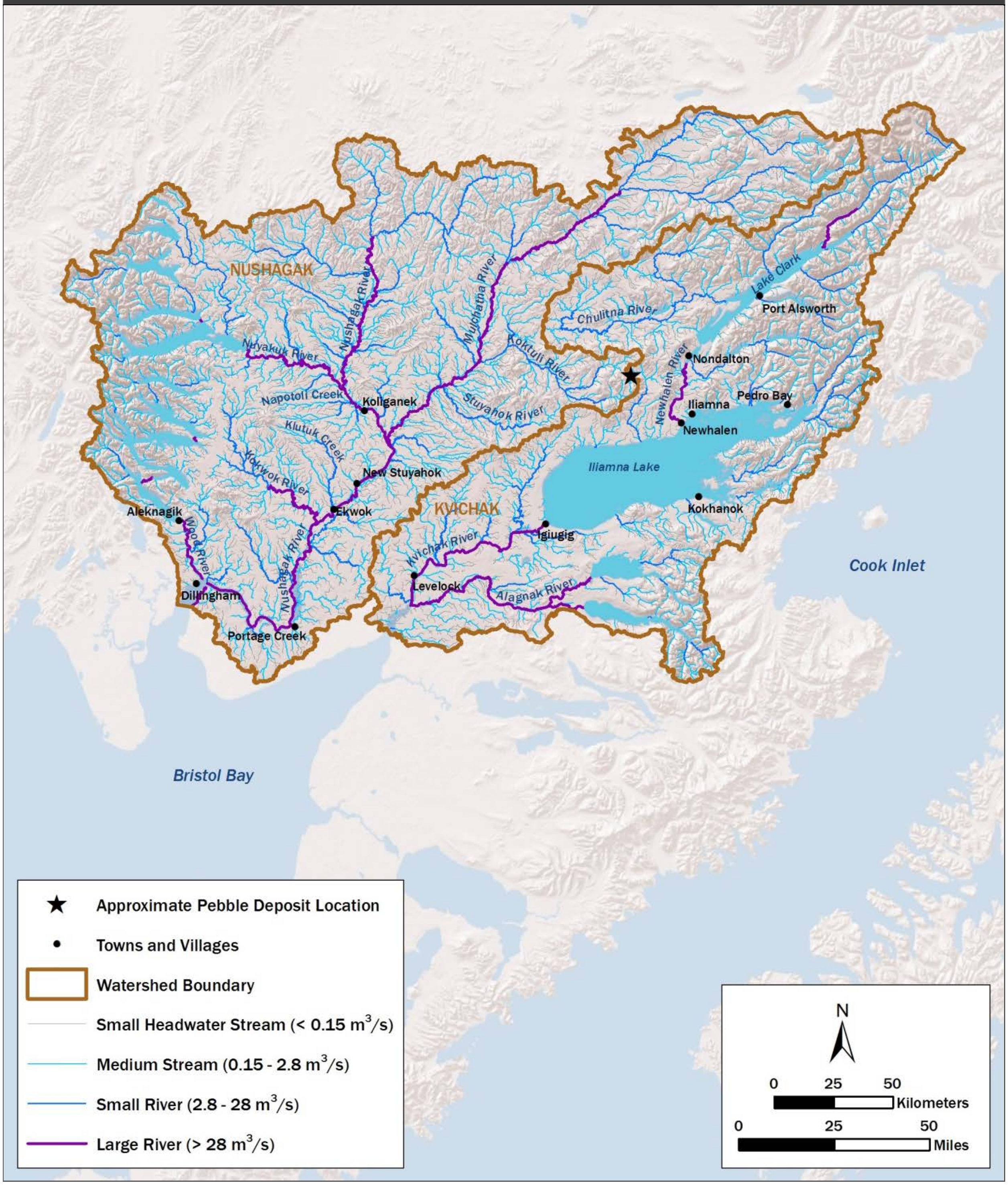


Table 3-3. Proportion of stream channel length within the Nushagak and Kvichak River watersheds classified according to stream size (based on mean annual streamflow in m³/s), channel gradient (%), and floodplain potential (based on % flatland in lowland). Gray shading indicates proportions greater than 5%; bold indicates proportions greater than 10%.

	Gradient							
	<1%		≥1% and <3%		≥3% and <8%		≥8%	
Stream Size	FP	NFP	FP	NFP	FP	NFP	FP	NFP
Small headwater streams[a]	**27%**	5%	3%	**13%**	0%	8%	0%	3%
Medium streams[b]	**20%**	3%	1%	3%	0%	2%	0%	1%
Small rivers[c]	6%	1%	0%	0%	0%	0%	0%	0%
Large rivers[d]	2%	0%	0%	0%	0%	0%	0%	0%

Notes:
[a] 0–0.15 m³/s; most tributaries in the mine footprints.
[b] 0.15–2.8 m³/s; upper reaches and larger tributaries of the South Fork Koktuli, North Fork Koktuli, and Upper Talarik Creek.
[c] 2.8–28 m³/s; mid to lower portions of the South Fork Koktuli, North Fork Koktuli, and Upper Talarik Creek, including the mainstem Koktuli River.
[d] >28 m³/s; the Mulchatna River below the Koktuli confluence, the Newhalen River, and other large rivers.
FP = high floodplain potential (≥5% flatland in lowland); NFP = no or low floodplain potential (<5% flatland in lowland).

3.5.2 Water Temperature

Water temperature data (PLP 2011: Appendix 15.1E, Attachment 1) indicate significant spatial variability in thermal regimes. Average monthly stream water temperatures in the Pebble deposit area in July or August can range from 6°C to 16°C. Longitudinal profiles of temperature indicate that stream temperatures in the Pebble deposit area do not uniformly increase with decreasing elevation (PLP 2011). This is often due to substantial inputs of cooler water from tributaries or groundwater (PLP 2011). 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 and Kvichak River watersheds, and provides a moderating influence against both summer heat and winter cold extremes in stream reaches where this influence is sufficiently strong. The range of spatial variability in temperatures in the Pebble deposit area (PLP 2011) is consistent with streams influenced by a variety of thermal modifiers, including upstream lakes, groundwater, or tributary contributions (Mellina et al. 2002, Armstrong et al. 2010).

3.6 Seismicity

The Alaska Earthquake Information Center 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 detected only by sensitive instruments, to the largest earthquake ever recorded in North America (the 1964 Good Friday earthquake near Anchorage, magnitude 9.2) (Table 3-4, Figure 3-15).

Table 3-4. Examples of earthquakes in Alaska.

Date	Depth (km)	Magnitude[a]	Distance and Direction from the Pebble Deposit
March 28, 1964	25	9.2	469 km east-northeast
November 3, 2002	4.2	7.2	593 km northeast
September 25, 1985	184	4.9	61 km southeast
July 13, 2007	6.2	4.3	30 km west-southwest
March 25, 2012	12	3.0	122 km east

Notes:

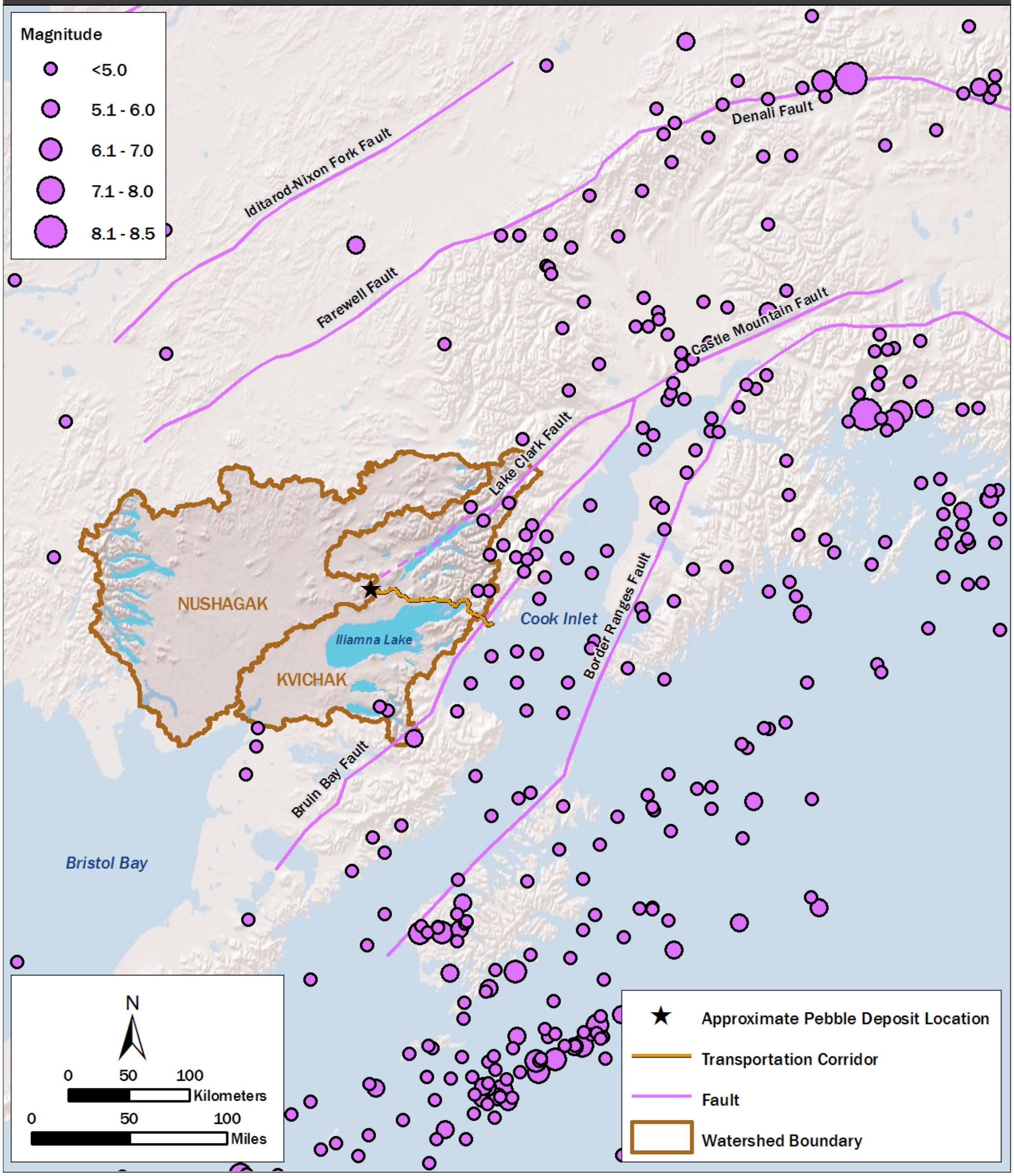
[a] Local magnitude as reported by the Alaska Earthquake Information Center. Note that earthquakes in the range of magnitudes 1.5 to 3.6 occur regularly in the Lake Clark area (data not shown). These earthquakes are centered at a depth of 100 km or greater.

Southwestern Alaska experiences a large number of earthquakes related to the presence of four active moving blocks of crust associated with large fault systems. These faults are, from north to south, the Tintina-Kaltag Fault, the Iditarod-Nixon Fork Fault, the Denali-Farewell Fault, the Lake Clark–Castle Mountain Fault system, the Bruin Bay Fault, and the Border Ranges Fault (Figure 3-15). Some sections along these faults are seismically active and have generated earthquakes in the past. The size of an earthquake is directly related to the area of the fault that ruptures; thus, longer faults are capable of producing larger earthquakes. The damage caused by an earthquake is related to the 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.

The Lake Clark–Castle Mountain Fault system, with a mapped length of 225 km, is the fault located nearest to the Pebble deposit. 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 has not been identified, but was originally interpreted to be near the western edge of Lake Clark. 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.

There are few residents and no long-term seismic monitoring station records in the area of the Pebble deposit, which make it difficult to assess accurately the recent seismic history of the area. As a result, the paleoseismic history of the western part of the Lake Clark Fault is unknown (Koehler and Reger 2011). 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).

Figure 3-15. 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. Fault lines are based on Haeussler and Saltus (2004), including the preferred drawing of the Lake Clark Fault (dashed purple line).



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 PLP identified numerous faults in the Pebble deposit area. 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 scenario watersheds (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 evidence that the Lake Clark Fault extends closer than 16 km to the Pebble depositor that there is 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 known pre-existing faults. Earthquakes can occur on previously unidentified, minor, or otherwise inactive faults, or along deeper faults that are not exposed at the surface. Although 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 seismicity in the Bristol Bay area is difficult because of the remoteness of the area, its complex bedrock geology overlain by multiple episodes of glacial activity, and the lack of historical records on seismicity. 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.

3.7 Existing Development

Unlike most other areas supporting Pacific salmon populations, the Bristol Bay watershed is undisturbed by significant human development. It is located in one of the last remaining virtually roadless areas in the United States (Section 6.1.3.1). 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. The Bristol Bay watershed also encompasses Iliamna Lake, 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 Alaska Department of Fish and Game's (ADF&G'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 sustainable salmon-based ecosystems. 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). Given the potential for hatchery fish to have negative effects on wild fish (e.g., Araki et al. 2009, Rand et al. 2012), this lack of hatchery fish is notable.

3.8 Climate Change

Thus far, this chapter has focused on the current physical environment in the Bristol Bay watershed. In the future, over time scales at which large-scale mining will potentially affect these watersheds, this physical environment is likely to change substantially—particularly in terms of climate and, by extension, hydrology. Over the past 60 years, much of Alaska has been warming at twice the average rate of the United States and many parts of the world (ACIA 2004). Throughout Alaska, changes such as warmer temperatures, melting glaciers, declining sea ice, and declining permafrost have already occurred (Serreze et al. 2000, Stafford et al. 2000, ACIA 2004, Hinzman et al. 2005, Liston and Hiemstra 2011, Markon et al. 2012). However, there is limited evidence over the last decade that suggests air temperature in much of Alaska has cooled, due to changes in the Pacific Decadal Oscillation and weakening of the Aleutian low (Wendler et al. 2012). Climate models suggest that warming throughout Alaska is projected to continue, and it is likely to lead to changes in the type and timing of precipitation, decreased snowpack and earlier spring snowmelt, and subsequent changes in hydrology similar to projections in Arctic regions (Hinzman et al. 2005).

Using methods detailed in Box 3-4, we used the multi-model average A2 emissions scenario developed by SNAP (2012) to generate 30-year means for future temperature and precipitation patterns in the Bristol Bay region. We focused on characterizing possible climate change impacts using the A2 emissions scenario 30-year mean for the end of this century (2071–2100) as an upper bound estimate of climate change effects expected for this region with current modeling. Similar trends in temperature and precipitation, but with smaller magnitudes, are shown for effects earlier in the century or with more benign emission scenarios.

BOX 3-4. METHODS FOR CLIMATE CHANGE PROJECTIONS

To project temperature and precipitation changes over the next century, we used data from the Scenarios Network for Alaska and Arctic Planning (SNAP). A full description of the SNAP data and methodology used is available on the SNAP website (SNAP 2012).

From the SNAP dataset, we used downscaled values of monthly mean temperature and precipitation. The historical dataset is derived from the Climate Research Unit (CRU) at the University of East Anglia for 1901 to 2009 (CRU 2012). The CRU data are downscaled using the Parameter-elevation Regressions on Independent Slopes Model (PRISM) 1971 to 2000 monthly climatologies for Alaska (PRISM Climate Group 2012), which take into account elevation, slope, and aspect. SNAP then developed downscaled monthly projections of temperature and climate for Alaska under three emissions scenarios developed by the Intergovernmental Panel on Climate Change for the Coupled Model Intercomparison Project. SNAP uses five global climate models (GCMs) [cccma_cgcm31, mpi-echam5, gfdl_cm21, ukmo_hadcm3, and miroc3_2_medres] that best characterize the Arctic region up to the year 2100 (Walsh et al. 2008). These emissions scenarios are:

- the B1 scenario, which represents a best-case emissions scenario;
- the A1B scenario, which represents a middle-of-the-road emissions scenario; and
- the A2 scenario, which represents a worst-case emissions scenario.

For this assessment, we use the SNAP 5-model average for the A2 scenario of the best-performing GCMs to consider a worst-case climate change scenario for the Bristol Bay region. Although uncertainty is inherent in climate modeling due to many factors, the SNAP 5-model average tends to perform better than any single model under the A2 scenario. Using the SNAP model, we calculated 30-year normal values, or average values over a 30-year period, for temperature and precipitation over 1971 to 2000 (historical) and over 2011 to 2040, 2041 to 2070, and 2071 to 2100 under the three emissions scenarios. We focused on the A2 scenario for the years 2071 to 2099 (the year 2100 is not included because one of the GCMs used in the average did not include that year). Using the SNAP data, we calculated changes in temperature and precipitation at three scales: the Bristol Bay watershed (Figure 2-3), the Nushagak and Kvichak River watersheds (Figure 2-4), and the mine scenario watersheds (Figure 2-5). We also calculated annual potential evapotranspiration (PET) (Hamon 1961) and annual water surplus (annual precipitation minus PET) for the Bristol Bay watershed and the Nushagak and Kvichak River watersheds.

Data for the appropriate watersheds were extracted from the SNAP dataset, which covers the entire state of Alaska. The resolution of the SNAP dataset is a 771-m grid. Any grid pixel intersecting a watershed boundary was included, even if the intersection was minimal, to account for the full range of possible temperature and precipitation values across the watersheds. In all cases, the values reported in the assessment represent the geographic spatial average across the entire watershed over an average of 30 years. Precipitation and temperature differences between the two periods were calculated as the geographic spatial average across the entire watershed of the raster representing the A2 scenario (2071 to 2099), minus the present period. Precipitation percent differences were calculated as the geographic spatial average across the entire watershed of the raster representing the difference between the A2 scenario (2071 to 2099) and the present period, divided by the present period and multiplied by 100.

Water surpluses under historical and future periods were calculated for each calendar month and summed to arrive at annual values. Differences between periods were calculated by subtracting the present value from the A2 scenario (2071 to 2099) value. It is important to remember that surplus measurements were calculated at the annual level and do not represent monthly or seasonal differences across a single scenario or between multiple scenarios.

Uncertainty is an inherent issue when dealing with projected temperature, precipitation, and water surplus values because of local variability and uncertainty in GCMs. Using average values for the five best-performing GCMs for the Arctic and calculating mean values over 30-year periods helps to reduce uncertainty; however, this averaging also decreases precision in predicting extreme events.

By the end of the century, based on SNAP (2012) data for the A2 emissions scenario, the multi-model average annual air temperature in the Bristol Bay region is projected to increase by approximately 4°C, with an approximately 6°C increase occurring in the winter months. Increases in air temperature are likely to affect the accumulation and melt of snowpack, the extent of lake ice, and the timing of spring ice break up, and result in increased water temperatures. Research from adjacent regions provides some basis for estimating water temperature changes that may result from climate change. Kyle and Brabets (2001) estimated that air temperature increases of 7.2°C to 8.5°C projected for Cook Inlet watersheds by 2100 would be associated with water temperature increases of 1.2°C to 7.1°C. It is important to note that although air temperature can be a useful metric for modeling water temperature, other factors (e.g., quantity, type, and seasonality of precipitation, snow and glacier cover) can also be critical water temperature drivers (Webb and Nobilis 1997, Mohseni and Stefan 1999).

Although we are unable to predict a change in extreme events, changes in precipitation patterns are likely to occur (Salathé 2006, Christensen et al. 2007, Peacock 2012, Markon et al. 2012), with rain-on-snow events becoming more common. The effect of increased rain-on-snow events on the frequency or volume of floods is unclear. Storm patterns also may change, although the increased likelihood of extreme events occurring and potential impacts on flooding are unknown. Changes in the seasonality of precipitation, snowpack, and the timing of snowmelt will likely affect streamflow regimes and may result in water availability changes, particularly in terms of decreased water availability in summer. Based on temperature, precipitation, and evapotranspiration projections, the landscape will likely be warmer and wetter annually; however, due to method limitations we are not able to determine how evapotranspiration will affect water availability on the landscape seasonally (Box 3-4).

3.8.1 Climate Change Projections for the Bristol Bay Region

Across the entire Bristol Bay watershed, average temperature is projected to increase by approximately 4°C by the end of the century (Table 3-5, Figure 3-16), and winter temperature is projected to increase the most (Table 3-5). Similar patterns are projected in the Nushagak and Kvichak River watersheds (Table 3-5).

By the end of the century, precipitation is projected to increase roughly 30% across the Bristol Bay watershed, for a total increase of approximately 250 mm annually (Table 3-6, Figure 3-17). In the Nushagak and Kvichak River watersheds, precipitation is projected to increase roughly 30% as well, for a total increase of approximately 270 mm of precipitation annually (Table 3-6). At both spatial scales, increases in precipitation are expected to occur in all four seasons (Table 3-6). Based on evapotranspiration calculations, annual water surpluses of 144 mm and 165 mm are projected for the Bristol Bay watershed and the Nushagak and Kvichak River watersheds, respectively (Table 3-7, Figure 3-18). Our simulated temperature and precipitation changes based on SNAP (2012) data for the Bristol Bay region are within the range of changes projected by other studies concentrating on Alaska and the Arctic (Christensen et al. 2007, Peacock 2012, Markon et al. 2012).

Table 3-5. Average annual and seasonal air temperature for historical and projected periods across the Bristol Bay watershed and the Nushagak and Kvichak River watersheds. Values were calculated using the SNAP (2012) dataset (Box 3-4). Temperature was calculated as average values over each 30-year period. Number in parentheses equals one standard deviation.

Scale	Season	Historical Temperature (1971–2000) (°C)	Projected Temperature (2017–2099) (°C)	Difference (°C)
Bristol Bay Watershed (Scale 1)	**Annual**	**1** (1)	**5** (1)	**4** (0.2)
	Winter	-8 (2)	-2 (2)	6 (1)
	Spring	0 (1)	4 (1)	4 (0.2)
	Summer	11 (2)	14 (2)	3 (0.07)
	Fall	1 (2)	5 (2)	4 (0.3)
Nushagak and Kvichak River Watersheds (Scale 2)	**Annual**	**1** (1)	**5** (1)	**4** (0.2)
	Winter	-9 (1)	-3 (1)	6 (0.4)
	Spring	0 (1)	4 (1)	3 (0.2)
	Summer	11 (2)	14 (2)	3 (0.05)
	Fall	0 (2)	5 (2)	4 (0.07)

Table 3-6. Average annual and seasonal precipitation for historical and projected periods across the Bristol Bay watershed and the Nushagak and Kvichak River watersheds. Values were calculated using the SNAP (2012) dataset (Box 3-4). Precipitation was calculated as average values over each 30-year time period. Number in parentheses equals one standard deviation.

Scale	Season	Historical Precipitation (1971–2000) (mm)	Projected Precipitation (2017–2099) (mm)	Difference (mm)
Bristol Bay Watershed (Scale 1)	**Annual**	**847** (421)	**1,095** (512)	**248** (104)
	Winter	177 (121)	229 (143)	52 (27)
	Spring	150 (91)	196 (112)	45 (25)
	Summer	234 (97)	303 (117)	69 (25)
	Fall	286 (141)	367 (170)	81 (34)
Nushagak and Kvichak River Watersheds (Scale 2)	**Annual**	**795** (336)	**1,062** (430)	**267** (95)
	Winter	160 (79)	215 (97)	55 (21)
	Spring	138 (67)	189 (90)	51 (23)
	Summer	226 (84)	300 (107)	75 (24)
	Fall	271 (123)	357 (152)	86 (32)

Table 3-7. Average annual water surplus for historical and projected periods across the Bristol Bay watershed and the Nushagak and Kvichak River watersheds. Values were calculated using the SNAP (2012) dataset (Box 3-4). Number in parentheses equals one standard deviation.

Scale	Historical Surplus (1971–2000) (mm)	Projected Surplus (2017–2099) (mm)	Difference (mm)
Bristol Bay Watershed (Scale 1)	400 (441)	544 (534)	144 (106)
Nushagak and Kvichak River Watershed (Scale 2)	341 (359)	506 (456)	165 (99)

Figure 3-16. Mean annual temperature across the Bristol Bay watershed under (A) historical conditions (1971 to 2000) and (B) the A2 emissions scenario (2071 to 2099), and (C) the temperature change between these two climate scenarios (SNAP 2012). See Box 3-4 for additional details.

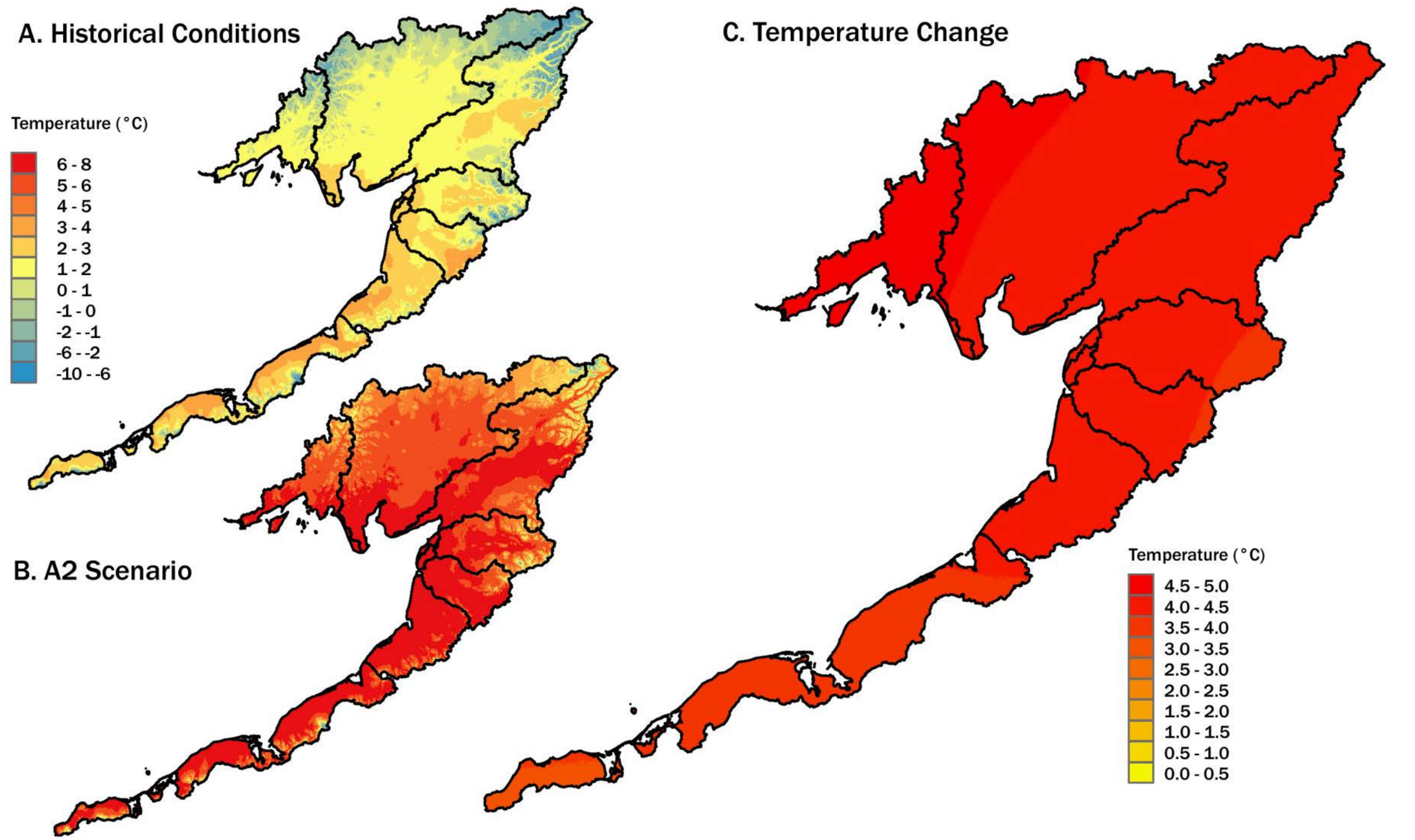


Figure 3-17. Mean annual precipitation across the Bristol Bay watershed under (A) historical conditions (1971 to 2000) and (B) the A2 emissions scenario (2071 to 2099), and (C) the precipitation change between these two climate scenarios (SNAP 2012). See Box 3-4 for additional details.

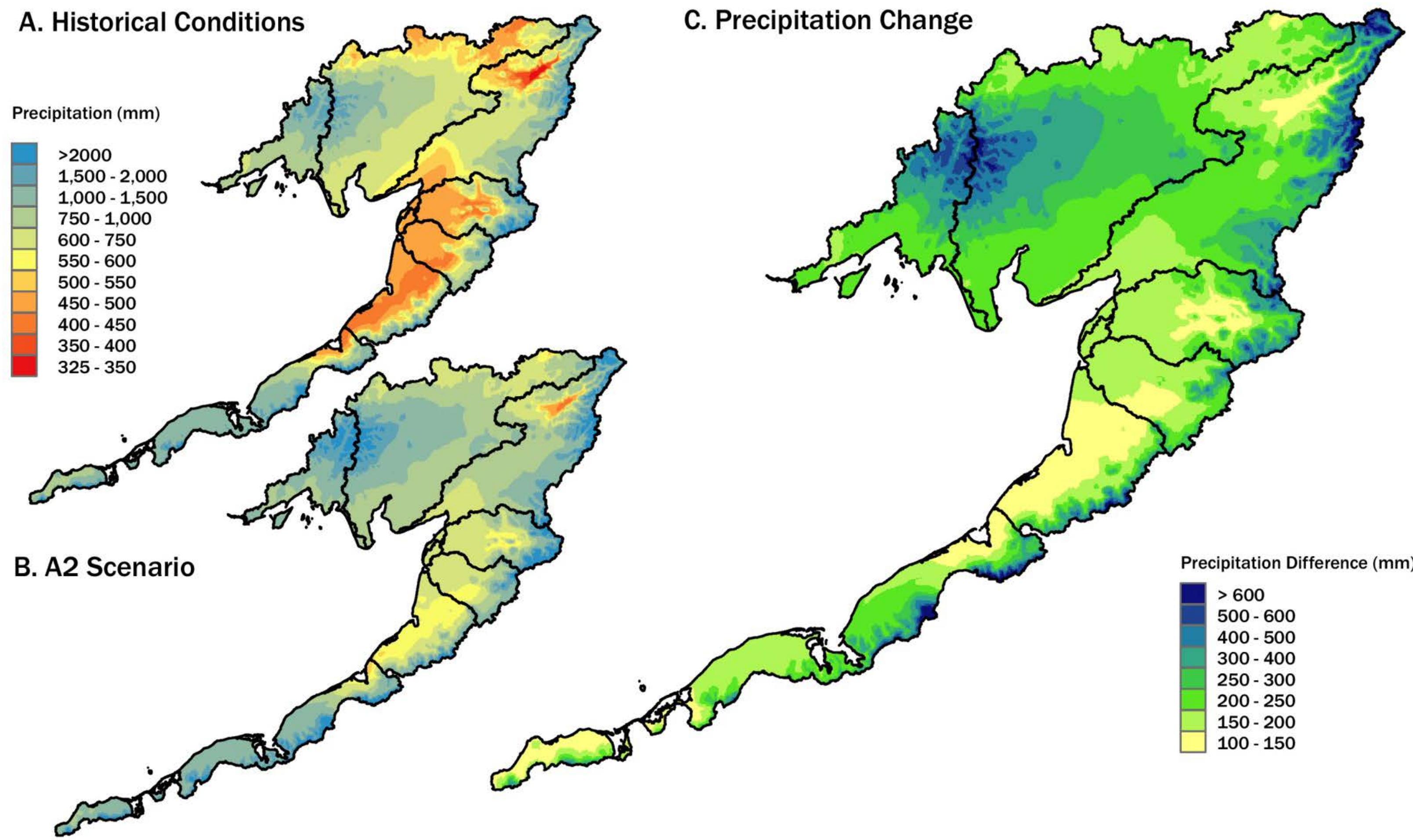
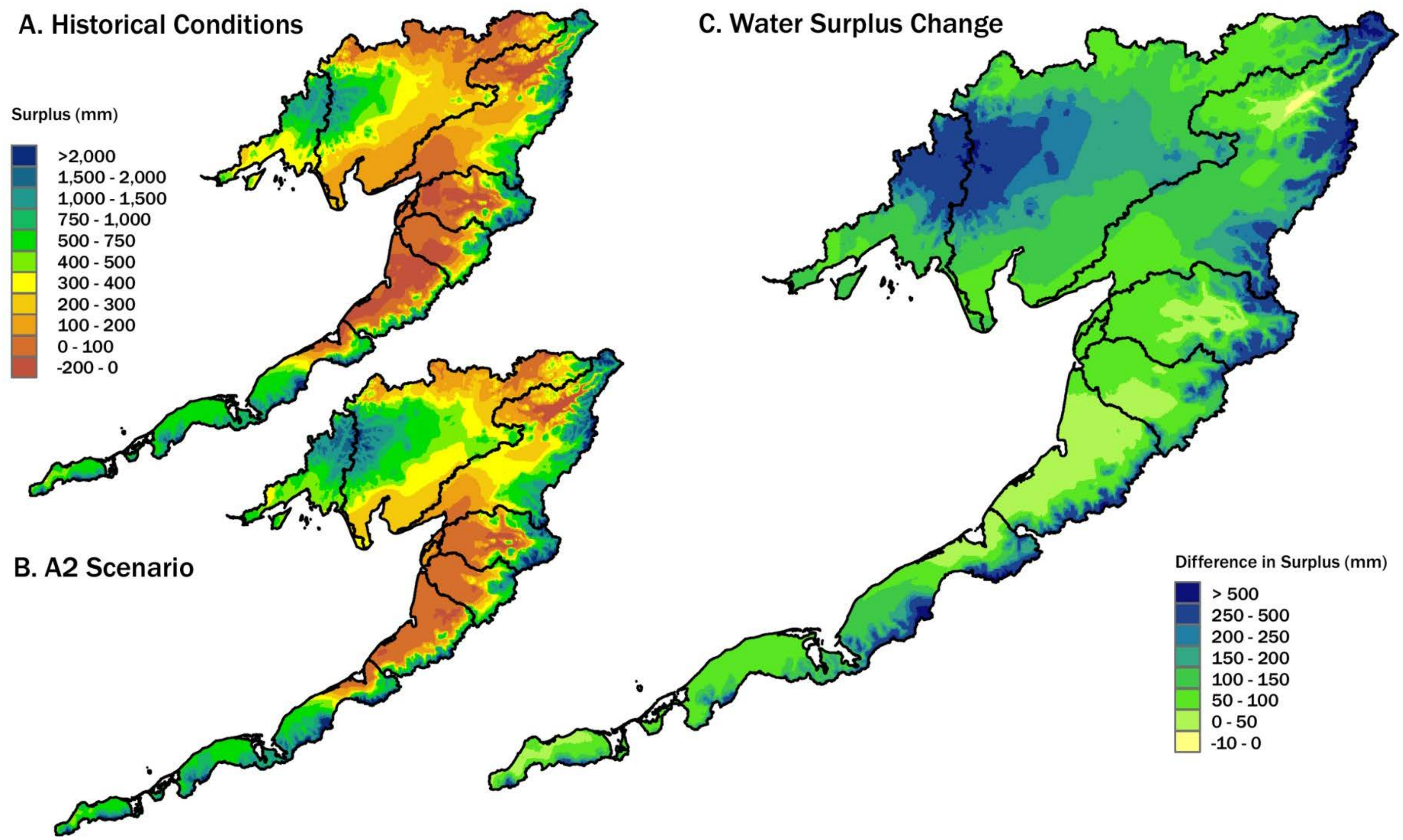


Figure 3-18. Mean annual water surplus (precipitation minus evapotranspiration) across the Bristol Bay watershed under (A) historical conditions (1971 to 2000) and (B) the A2 emissions scenario (2071 to 2099), and (C) the water surplus change between these two climate scenarios (SNAP 2012). See Box 3-4 for description of surplus calculations.



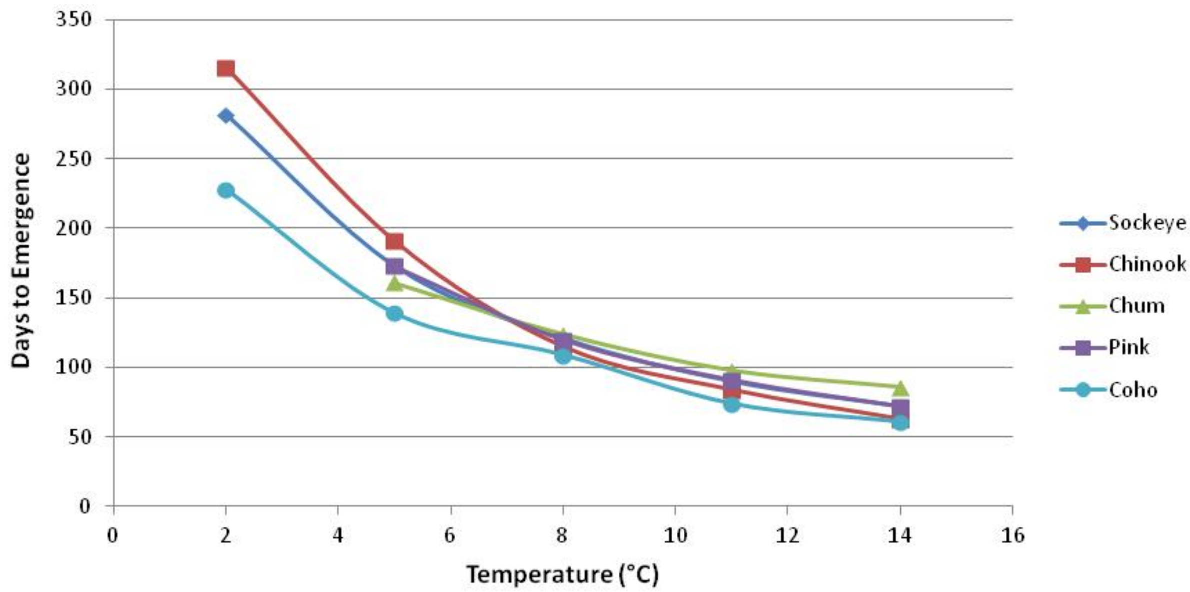
3.8.2 Potential Climate Change Effects

There are likely to be hydrological impacts associated with projected changes in temperature, precipitation, and evapotranspiration in the Bristol Bay watershed, including changes in the magnitude and timing of streamflow that are likely to affect salmon habitat and populations. When temperature increases in freshwater environments, community structure, habitat, and salmon populations can be affected (Eaton and Scheller 1996, Hauer et al. 1997). With warmer temperatures and changes in the type, timing, and amount of precipitation, there likely will be changes in snowpack, a shift in the timing of spring snowmelt, and changes in the type of precipitation falling (Barnett et al. 2005). With these changes, there will be alterations to both the magnitude and timing of the natural streamflow regime and a likely decline in seasonal water availability, mirroring already observed changes in other systems such as the Pacific Northwest (Mote et al. 2003).

These hydrologic flow regime changes may affect salmon populations during spawning and smolt migrations, and can scour streambeds leading to the loss of salmon eggs (Lisle 1989, Montgomery et al. 1996, Steen and Quinn 1999, Mote et al. 2003, Lawson et al. 2004, Stewart et al. 2004). Changes in hydrology are likely to affect existing habitat via changes in water volume and velocity along with channel forms, which may lead to declines in habitat availability for spawning and rearing salmon populations. Changes to baseflow, depending on groundwater and surface water interactions, are likely to affect the amount of wetlands in the Bristol Bay watershed, in that wetlands are likely to decrease under drier baseflow conditions. Although we are unable to predict whether baseflow will increase or decrease, any changes in baseflow will likely affect water temperature (in addition to the direct effects of increased air temperature on water temperature).

Both the hydrology and water temperature of freshwater systems affect critical life stages of salmonid species. Furthermore, these hydrological changes are likely to have different effects on salmon populations depending on the amount of time they spend rearing in freshwater habitats, their life stage, and their ability to adapt to changes in environmental conditions. Pink and chum salmon are likely to be affected by temperature increases early in egg incubation, which can affect timing of emergence, migration to the ocean, and potential mismatch in the timing of peak food abundance in the marine environment (Bryant 2009). For example, the average migration time for one population of pink salmon in southeast Alaska now occurs nearly 2 weeks earlier than it did 40 years ago (Kovach et al. 2012). For sockeye salmon that typically rear in fresh water for 1 to 2 years, temperature increases may affect life-stage timing, including spawning and fry emergence, as well as the growth and survival of lake-rearing fry (Healey 2011, Martins et al. 2012). Across all five Pacific salmon species, time to fry emergence decreases as water temperature increases (Figure 3-19); thus, warmer winters may result in earlier fry emergence.

Figure 3-19. Relationship between time from fertilization to emergence and temperature for the five Pacific salmon species. Data are from Quinn 2005.



Changes in precipitation and hydrology also may affect access to lakes and spawning locations, and high-intensity rainfall may increase sedimentation in spawning streams and rearing lakes for sockeye salmon (Bryant 2009). Rich et al. (2009) hypothesized that warmer temperature was a factor in poor sockeye salmon recruitment in the Kvichak River watershed. For Chinook salmon, increases in temperature are likely to affect incubation and fry emergence (Beer and Anderson 2001), which may affect growth, survival, and timing of migration to the ocean (Heming et al. 1982, Taylor 1990, Berggren and Filardo 1993). Coho salmon incubation and timing of emergence are also affected by increases in temperature (Tang et al. 1987).

Populations of Pacific salmon species are likely to respond and adapt to changes in temperature, precipitation, and hydrology in different ways, and the geographic location of populations is likely to affect their ability to adapt to these changes. Studies have predicted that the reproductive success of salmon populations in Washington is likely to decline over the next century (Battin et al. 2007, Mantua et al. 2010), and freshwater temperature increases in the Fraser River will negatively affect growth and survival of sockeye salmon at all life stages (Healey 2011). The genetic and life history diversity within and among the Bristol Bay Pacific salmon populations (Section 5.2.4) will likely be crucial for maintaining the resiliency of the region's salmon stocks under a future environment characterized by climate change and increased anthropogenic stressors (Hilborn et al. 2003, Schindler et al. 2010, Rogers and Schindler 2011).



CHAPTER 4. TYPE OF DEVELOPMENT

4.1 Mineral Deposits and Mining in the Bristol Bay Watershed

Significant mineral resources are located in Alaska, and the state has a long mining history. Russian explorers began searching for placer gold in the early 1800s, and substantial placer deposits have been found in many areas of the state. More recently, hard rock exploration has increased throughout the region. Alaska mines range in size from small, recreational suction dredging operations to large-scale commercial operations, for a variety of deposit types (Table 4-1).

Several known mineral deposits with potentially economically significant resources are located in the Nushagak and Kvichak River watersheds, and active exploration of deposits is occurring in a number of claim blocks (deposits other than Pebble are considered in greater detail in Chapter 13; see Table 13-1 and Figure 13-1 for the names and locations of these deposits). Of deposit types occurring or likely to occur in the region, porphyry copper, intrusion-related gold, and copper and iron skarn may indicate economically viable mining, thereby prompting large-scale development. Thus, the development of a number of mines, of varying sizes, is plausible in this region—and once the infrastructure for one mine is available, it would likely facilitate the development of additional mines (Chapter 13).

The potential for large-scale mining development within the Nushagak and Kvichak River watersheds is greatest for porphyry copper deposits, most notably the Pebble deposit. Significant exploration activity has been ongoing at this deposit for many years, and the information available provides the most complete description of potential mining in the region. Because the Pebble deposit is the most likely deposit to be developed in the near term, this assessment focuses exclusively on porphyry copper deposits. However, much of the discussion of mining methods (Section 4.2.3) applies to all types of disseminated ore deposits (i.e., ores with low concentrations of metal spread throughout the body of rock).

Table 4-1. Characteristics of past, existing, or potential large mines in Alaska.

Mine	Kennecott	Donlin	Fort Knox	Greens Creek	Kensington	Pogo	Red Dog	Pebble (78-yr)[a]
Location	Copper River basin, in Wrangell–St. Elias National Park	13 miles N of village of Crooked Creek and Kuskokwim River	26 miles NE of Fairbanks	18 miles 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	Headwaters of three streams running into the Nushagak and Kvichak Rivers
Target metals	Copper, silver	Gold	Gold	Zinc, lead, silver, gold	Gold	Gold	Zinc, lead	Copper, gold, molybdenum
Ore type	Massive sulfide	Gold-bearing quartz	Oxide ore body	Massive sulfide	Gold-bearing quartz	Gold-bearing quartz	Massive sulfide	Porphyry copper
Ore grade quality	Very high	Moderate	Low	High	Moderate	Moderate	High	Low
Operational life (years)	27 (1911–1938)	22	20	35–50	10	11	42 (1989–2031)	78
Extraction type	Underground stope mining	Open pits (2)	Open pit	Underground stope mining	Underground stope mining	Underground stope mining	Open pits (2)	Open pit
Total resource (million metric tons)	~ 4.5	491[b]	401	29	24	9.1	171	5,920
Ore processing rate (metric tons/day)	~ 91	48,524	33,000–45,000	1,524	1,134	2,267	7,500–8,300	208,000
Total waste rock (million metric tons)	<0.9	1900	338	~ 1.8	1.5	1.7	142	14,600
Tailings disposal	On Kennicott Glacier	Dams/ponds (2)	Dam/pond	Dry tailings	Lake disposal	Dry tailings	Dam/pond	Dams/ponds (multiple)
Tailings amount (million metric tons)	<0.9	426	181	~ 13.6	4.1	4.9	91	5,860
Tailings footprint (km^2)	NA	5.4	4.5	0.25	0.24	0.12	3	46
Dam height (m)	NA	143 (largest of multiple dams)	111	NA	27[b]	NA	63	209 (largest of multiple dams)
Acid mine drainage potential	No	Yes	No	Yes	No	No	Yes	Yes

Notes:
[a] Ghaffari et al. 2011.
[b] Novagold 2012.
NA = not applicable.
Source: Levit and Chambers 2012, except as noted.

4.2 Porphyry Copper Deposits and Mining Processes

4.2.1 Genesis of Porphyry Copper Deposits

Porphyry copper deposits are found around the world, often occurring in clusters (Lipman and Sawyer 1985, Singer et al. 2001, Anderson et al. 2009) 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 4-2) (Singer et al. 2008). Porphyry copper deposits range in size from millions to billions of tons (Table 4-2). The well-delineated Pebble deposit is at the upper end of the total size range; thus, any additional deposits found in the Nushagak and Kvichak River watersheds are likely to be much smaller than the Pebble deposit.

Table 4-2. Global grade and tonnage summary statistics for porphyry copper deposits.

Parameter	10th Percentile	50th Percentile	90th Percentile	Pebble Deposit[a]
Tonnage (Mt)	30	250	1,400	10,777
Copper grade (%)	0.26	0.44	0.73	0.34
Molybdenum grade (%)	0.0	0.004	0.023	0.023
Silver grade (g/t)	0.0	0.0	3.0	unknown
Gold 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 Pebble Limited Partnership.

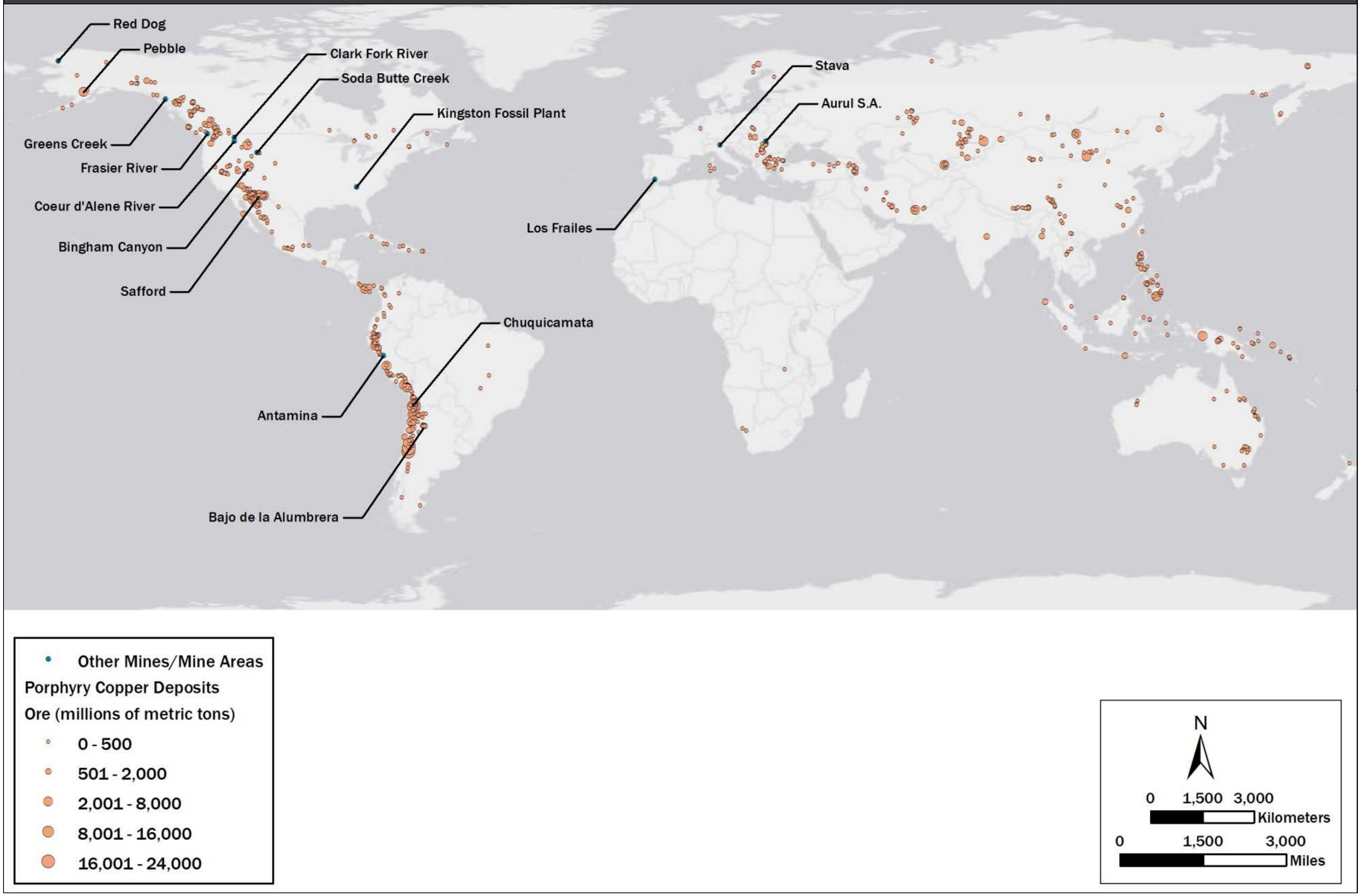
Mt = million tons; g/t = grams per ton.

Sources: Singer et al. 2008; Appendix H.

4.2.2 Chemistry and Associated Risks of Porphyry Copper Deposits

Exposure to hazards associated with mining 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 that control the hazards, 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 the water quality of releases to the environment; however, our ability to make predictions is limited because of data insufficiency and the inherent complexity of natural materials and their environment.

Figure 4-1. Porphyry copper deposits around the world. Values are from the database compiled by and described in Singer et al. 2008. Other mines and mining regions mentioned in the text also are shown on the map.



Sources of hazards from porphyry copper mines can be grouped into four broad, interrelated categories: acid-generating potential, trace elements and their mobilities, 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, acid-generating potential, and trace elements (categories related to mining processes are described in Section 4.2.3).

Mining processes expose rocks and their associated minerals to atmospheric conditions that cause weathering, which releases minerals (e.g., copper minerals) from the rock matrix. Grinding methods used in these processes create materials that have high specific surface areas, which accelerates the rate of weathering. Porphyry copper deposits are characterized by the presence of sulfide minerals, and oxidation of sulfide minerals creates acidity, sulfate, and free metal ions (e.g., iron in the case of pyrite); in addition, the acid produced 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 exposure to metals and certain elements in the aquatic environment.

One way to predict if acid generation has the potential to occur is to perform acid-base accounting tests. Acid-base accounting tests are rapid methods to determine the acid-generating potential (AP) and neutralizing 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 neutralizing potential (NNP) being NP-AP and the neutralizing potential ratio (NPR) being NP/AP. AP, NP, and NNP typically are expressed in units of kilograms of calcium carbonate per metric ton of waste material (kg $CaCO_3$/metric ton). Positive NNP values are net alkaline and negative values 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 that have an NPR greater than 4 as being non-acid-generating (NAG) (Brodie et al. 1991, Price and Errington 1998). Materials that have a ratio between 1 and 4 require further testing via kinetic tests and geochemical assessment for classification (Brodie et al. 1991, Price 2009, 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—that is, pH of the system may decrease over time as neutralizing materials are used up, resulting in acid mine drainage. 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. Depending on the water chemistry of both a receiving water body and any mine drainage, released elements may either be transported downstream as dissolved ions or form precipitates that travel as suspended solids or settle to the streambed.

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 porphyry copper deposit in Utah (Figure 4-2A). 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 4-2B).

4.2.3 Overview of the Mining Process

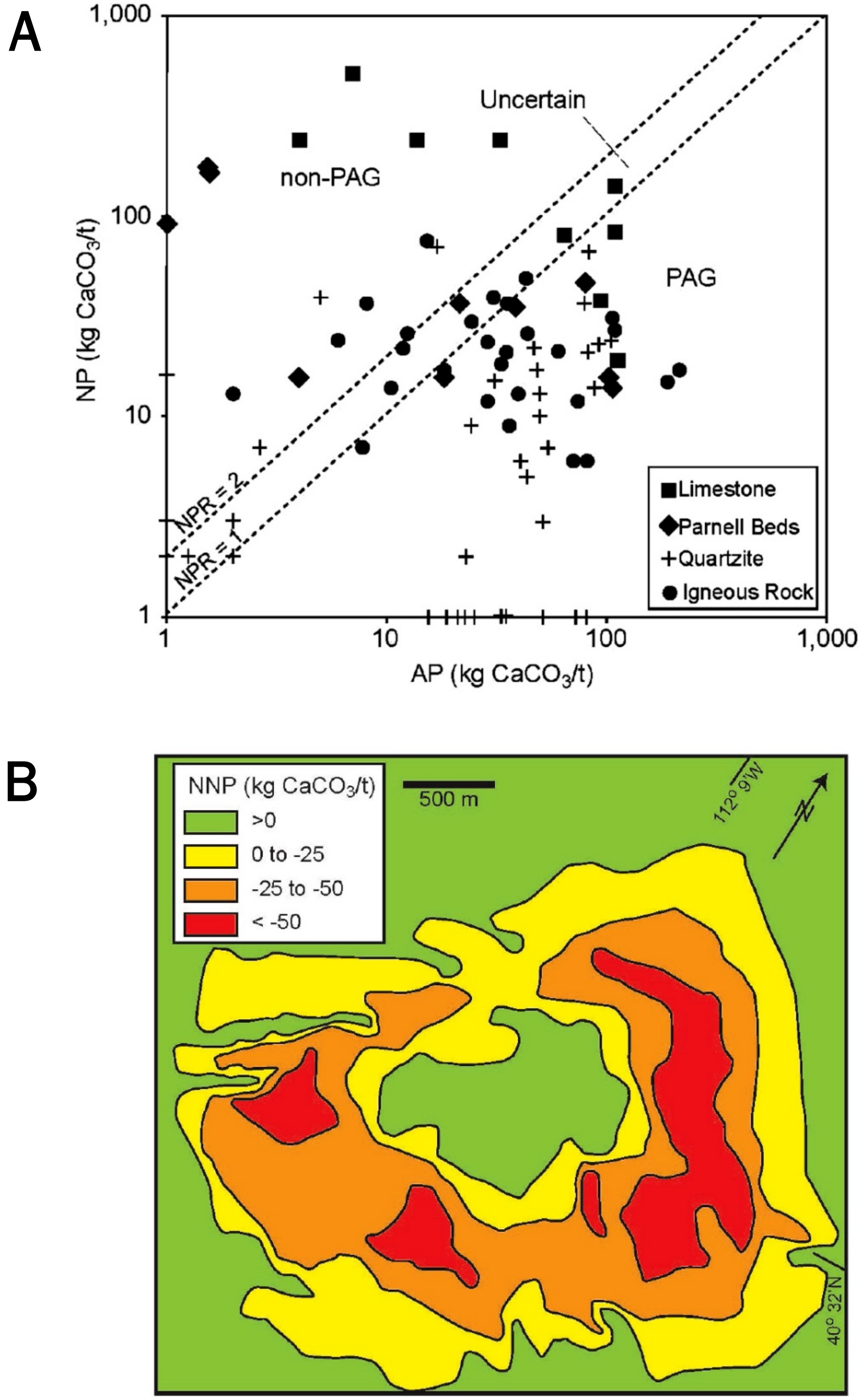
Developing a mine requires establishing surface or underground mine workings that allow access to the ore body. The scope and complexity of development-related activities vary depending on the characteristics of each project, but typically include the following components.

- **Site preparation (clearing, stripping, and grading).** Topsoil and overburden are removed and typically stockpiled for later use in mine reclamation.
- **Construction of mine site infrastructure.** Specific requirements depend on the size and type of mine operation, its location, and proposed mining, milling, and processing methods. Typical infrastructure includes facilities for ore crushing, grinding, and other mineral separation processes; ore stockpiling and waste rock disposal facilities; tailings storage facilities; water supply, treatment, and distribution facilities; transportation infrastructure such as roads or railways; pipelines; conveyers; and other infrastructure (e.g., offices, shops, housing).
- **Establishment of mine workings.** Once the site is prepared and infrastructure is constructed, mine workings are established: ore is extracted and processed, water at the site is managed and treated, and tailings and waste rock are stored and managed.

At each stage of mine development, potential impacts on the environment and human health can be reduced by ensuring effective implementation of proper design, construction, operation, and management techniques and protocols (Box 4-1).

Any mining company must comply with a number of federal, state, and local laws when developing and operating a mine. Compliance is facilitated through the regulatory permitting process and involves multiple state and federal agencies (see Box 4-2 for additional detail on these regulatory requirements). Regulations also serve to hold an operator accountable for potential future impacts, through establishment of financial assurance requirements and imposition of fines or compliance orders upon non-compliance with permit requirements (Box 4-3).

Figure 4-2. Neutralizing potential at the Bingham Canyon porphyry copper deposit in Utah. (A) Plot of neutralizing potential (NP) vs. acid-generating potential (AP) for mineralized rock types. PAG denotes potentially acid-generating. Note that the range of uncertainty is indicated as 1 to 2 in this figure; in the assessment, we use the more conservative range of 1 to 4. (B) Plan view of the distribution of net neutralizing potential (NNP) values. Plots modified from Borden (2003).



BOX 4-1. REDUCING MINING'S IMPACTS

Reducing mining's impacts on the environment and human health requires proper mine planning, design, construction, and operation; appropriate management and closure of waste and water containment and treatment facilities; and monitoring and maintenance over all mine-life phases, including post-closure. Some general methods for reducing adverse impacts of mining are provided here, along with information about how these concepts are incorporated into the assessment.

Best management practices refer to specific measures for managing non-point source runoff (40 CFR 130.2(m)). Measures for minimizing and controlling sources of pollution in other situations are often referred to as best practices, state of the practice, or simply mitigation measures. These are not the best possible or conceivable practices, but rather the current practices of the best operators. We assume that these types of measures would be applied throughout a mine as it is constructed, operated, closed, and post-closure. Although we describe some measures as they are relevant to a discussion, it is not necessary, for the purpose of this assessment, to describe them all.

Mitigation refers to all steps taken to avoid, minimize, treat, or compensate for potential adverse impacts on the environment from a given activity. One example of a mitigation measure for avoidance is to avoid mining a particularly reactive type of rock that might make future leachate management too difficult. Minimization of an impact is practiced when avoidance is not feasible, and includes measures taken to lessen the amount of contaminant released. An example of a mitigation measure to minimize an impact is to blend known acid-producing material with sufficient neutralizing material. Treatment is required when contaminants are released. An example is the diversion and collection of seepage from a waste rock pile for passage through a wastewater treatment plant to meet appropriate water quality criteria prior to release to the environment. Many elements of our mine scenarios include mitigation measures and all are assumed to meet minimum regulatory requirements. Appendix I contains further discussion of mitigation measures.

Compensatory mitigation refers to the restoration, establishment, enhancement, and/or preservation of wetlands, streams, or other aquatic resources to offset environmental losses resulting from unavoidable impacts on waters of the United States, as authorized by Clean Water Act Section 404 permits issued by the U.S. Army Corps of Engineers (40 CFR 230.93(a)(1)). This becomes an option only after all opportunities for aquatic resource impact avoidance and minimization have been exhausted. See Box 7-2 and Appendix J for a more complete discussion of compensatory mitigation.

Reclamation refers to restoration of a disturbed area to an acceptable form and planned use following closure of a mining operation. Our mine scenarios assume that the site would be reclaimed according to statutory requirements and present some options that are feasible and common, but it is outside the scope of this assessment to evaluate a specific post-closure plan.

Remediation refers to fixing a problem that has become evident, such as an accidental release or spill of product or waste material. For example, a tailings slurry spill would require remediation. The dam may have been designed and constructed to properly mitigate (i.e., avoid or minimize) the potential for a spill, but an accident or failure could cause contaminant release, thereby creating the need for remediation.

BOX 4-2. PERMITTING LARGE MINE PROJECTS IN ALASKA

Large mine projects in Alaska must comply with federal and state environmental laws, and many federal, state, and local government permits and approvals are required before construction and operation of a large hard rock mine can begin. The specific permits and approvals vary from project to project, depending on the unique challenges posed by each mine.

Federal laws and agencies. The involvement of federal agencies varies for each mine, but most projects at least require authorizations from the U.S. Army Corp of Engineers. Other agencies that may be involved include (but are not limited to) the U.S. Environmental Protection Agency, the U.S. Fish and Wildlife Service, the National Marine Fisheries Service, the U.S. Coast Guard, and the U.S. Department of Transportation.

Federal agency authorizations ensure that projects comply with the following applicable federal laws.

- Clean Water Act
- Clean Air Act
- National Environmental Policy Act
- National Historic Preservation Act
- Resource Conservation and Recovery Act
- Rivers and Harbors Act
- Endangered Species Act
- Bald Eagle Protection Act
- Migratory Bird Act
- Magnuson-Stevens Act
- Mine Safety and Health Act

Alaska Department of Natural Resources permits and approvals. The Alaska Department of Natural Resources (ADNR) Office of Project Management and Permitting coordinates the permitting of large mine projects via the establishment of a large mine project team for each project. This project team is an interagency group, coordinated by ADNR, that works cooperatively with large mine permit applicants and operators, federal resource agencies, and the Alaskan public to ensure that projects are designed, operated, and reclaimed in a manner consistent with the public interest.

ADNR may require the following permits and approvals.

- Plan of operations approval
- Reclamation plan and bond approval
- Right-of-way for access and utilities (roads, power lines, pipelines)
- Millsite lease
- Permit to appropriate water
- Dam safety certification (certificates of approval to construct and operate a dam)
- Upland or tideland leases
- Material sale
- Winter travel permits
- Cultural resource authorization
- Mining license

Alaska Department of Environmental Conservation permits and approvals. The Alaska Department of Environmental Conservation may require the following permits related to wastewater management and water and air quality.

- Waste management permit
- Alaska pollutant discharge elimination permit
- Domestic and non-domestic wastewater disposal permits
- Certificate of reasonable assurance for 404 permits
- Stormwater discharge pollution prevention plan
- Air quality permits
- Approval to construct and operate in a public water supply system
- Plan review for non-domestic wastewater treatment system
- Plan review and construction approval for domestic sewage system
- Oil discharge prevention and contingency plan

Other state permits and approvals. The state may require the following permits and approvals.

- Fish passage permit
- Fish habitat permit
- Utility permit on right of way
- Driveway permit
- Approval to transport hazardous materials
- Life and fire safety plan check
- State fire marshal plan review certificate
- Certificate of inspection for fired and unfired pressure vessel
- Employer identification number

BOX 4-3. FINANCIAL ASSURANCE

Many of the regulatory checks listed in Box 4-2 help to reduce potential impacts of mining on the environment, but they do not ensure that a permitted mine will have negligible effects on the environment. Even with the most stringent requirements, accidents and human error may cause mine systems to fail—and the most unpredictable accidents and errors often result in the most economically and environmentally costly failures. Thus, regulations also serve to hold an operator accountable during mine operations via both the imposition of fines for non-compliance with permit regulations and the establishment of financial assurance requirements for closure and reclamation of the mine. Financial assurance basically means that operators must ensure that sufficient funds are available for future remediation, closure, and reclamation of a mine.

Operators of Alaska's hard rock mining facilities, including copper and gold facilities, are required by the state to demonstrate financial assurance for reclamation, waste management, and dam safety costs.

- Prior to the start of hard rock mining operations on state-owned, federal, municipal, or private land, the Alaska Department of Natural Resources (ADNR) must approve a reclamation plan and financial assurance must be demonstrated in an amount necessary to ensure performance of the plan (Alaska Statute 27.19).
- The Alaska Department of Environmental Conservation may require hard rock mining operations that dispose of solid or liquid waste material or heated process or cooling water under a waste management and disposal permit to demonstrate financial assurance in an amount based on the estimated costs of required closure activities and post-closure monitoring for the waste management area (Alaska Statute 46.03.100(f)).
- Operators of hard rock mines on state-owned or privately owned land seeking ADNR approval to construct mine tailings dams must demonstrate financial assurance to cover the cost of reclamation and post-closure monitoring and maintenance of the dam (Alaska Statute 46.17).
- Operators of hard rock mining facilities on land managed by the Bureau of Land Management or U.S. Forest Service can be required by these agencies to demonstrate additional financial assurance for reclamation (43 CFR 3809 and 36 CFR 228 Subpart A, respectively).
- In addition to State of Alaska and Bureau of Land Management financial assurance requirements, facilities operating under leases, permits, or other agreements for the development of hard rock minerals on tribal lands can be required by the Bureau of Indian Affairs to demonstrate financial assurance to ensure compliance with the terms and conditions of the mineral agreement and applicable statutes and regulations (25 CFR 211.24 and 225.30).

Financial assurance calculations assume that a government entity would have to enter the site and commence reclamation activities without the benefit of any equipment or labor that may be at the site. The process determining the cost of every shovel, loader, gallon of fuel, and hour of labor is revisited and adjusted as necessary every 5 years. The State of Alaska allows several types of assurance (e.g., cash, gold bullion, surety bonds, reclamation trust funds, irrevocable letters of credit).

Example Financial Assurance Amounts for Alaska Mines

Mine	Amount
Fort Knox	$68,852,293
Kensington	$28,727,011
Pogo	$44,430,000
Red Dog	$305,150,000

It is important to note that effective financial assurance depends on accurate estimates of costs, which poses challenges when dealing with the potentially long-term, unpredictable, and costly events that a hard rock mining operation must consider. For example, current financial assurance requirements do not address chemical or tailings spills because of the greater degree of uncertainty related to these accidents; whereas the costs associated with reclamation and closure can be estimated, the cost of cleaning up a spill is unpredictable. However, financial assurance calculations increasingly include long-term water treatment.

4.2.3.1 Extraction Methods

The low concentrations of disseminated metals in porphyry copper deposits require large amounts of ore to enable a 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. A long-range mining plan is usually developed first to match the final mine design with the available ore reserves, weighing economics against engineering restrictions. This plan is re-evaluated throughout the life of the mine to reflect changes in the economy, increased knowledge of the ore body, and potential changes in mining technology.

Porphyry copper deposits are most commonly mined using open pit and, less commonly, underground mining methods (John et al. 2010). Open pit mining is typically used to extract ore where the top of a deposit is within 100 m of the surface (Blight 2010). Excavation of a pit begins at the surface, with drilling and blasting to strip overburden from the ore body surface. The equipment and materials used will fit the economies of scale for the project (e.g., mine life, daily production). The ore is drilled and blasted according to a blasting pattern. The size and spacing of the drill holes and the amount of explosives used determine the size of the material that is loaded and hauled to the crushing plant. The pit is successively enlarged until the pit limits are established by the extent of ore that can be profitably mined.

Pit design depends on the material characteristics of the ore and waste rock. The moisture content, strength, and load-bearing capacity of the ore and waste rock help determine the angle of the pit slopes, which generally are designed to be as steep as possible while still maintaining stability. A properly designed pit reduces the stripping ratio, or the volume of waste rock to ore, thereby increasing efficiency, potentially decreasing costs, and optimizing the amount of ore that can be mined economically.

Block caving is an underground mining method used for large deposits with rock mass properties amenable to sustainable caving action (Singer et al. 2008, Lusty and Hannis 2009, Blight 2010). Such deposits typically have mineralization throughout the rock (e.g., porphyry copper deposits) and are too deep to be mined economically by open pit methods. Block caving uses gravity to reduce the amount of drilling and blasting required to extract ore. It involves tunneling to the bottom of the ore and undercutting it, so that the deposit caves under its own unsupported weight. As ore is removed from below, fractures spread throughout the block, which breaks into fragments and is removed from the bottom of the enlarging void (Box 4-4).

Underground mining via block caving has a different set of costs than open pit mining, because of the extensive drilling of tunnels and shafts through non-ore-bearing rocks needed to gain access to the ore. Once begun, block caving generally requires less drilling and blasting, allows for less ore selectivity in the mining process, and may require less labor relative to open pit mining. As with other types of mining, the economics of block caving are determined by the prices of the metals being extracted, operational costs, and a number of other factors. If block caving allows the mining of additional ore that could not be mined using open pit mining methods, it creates the need for additional tailings storage capacity, increased capacity at the mill, increased consumption of utilities such as water and power, increased production of metal concentrates, and possible extension of the mine life.

BOX 4-4. BLOCK CAVING AND SUBSIDENCE

Subsidence at the ground surface is an inevitable result of the extraction of any underground resource (SME 2011). Block caving causes the surface above the worked-out mine to collapse into the void created by the removed ore. The area of subsidence on the ground's surface generally is larger than the area actually block-caved underground (Whittaker and Reddish 1989, USDA 1995). The extent and rate at which subsidence occurs depend on a number of factors, including the strength and thickness of the overburden, the extent of faulting and fracturing, and the depth of the mine workings (Whittaker and Reddish 1989).

In addition to altering surface topography, subsidence can affect both the quantity and quality of surface-water and groundwater systems, either directly or indirectly. For example, Slaughter et al. (1995) observed both increases and decreases in groundwater levels and changes in groundwater total dissolved solids concentrations due to subsidence at a coal mine in Utah. The authors attributed the rise in the water table to stream water seeping through fractures in the streambed, the subsequent decrease in the water table to connectivity between streambed fractures and the mine workings, and the total dissolved solids changes to exposure of the water to mine workings (Slaughter et al. 1995).

Backfilling a mining void is known to reduce subsidence. However, this requires a sufficient amount of suitable material, which may need to be imported in areas mined with methods that generate little waste material (SME 2011). Void-filling grout also may be used to mitigate subsidence, as well as to minimize oxidation of mined surfaces to reduce the potential for production of acid mine drainage.

4.2.3.2 Water Treatment and Management

Because mine workings must be kept dry for the duration of mining activities, dewatering is required for both open pit mines and block caving operations. Dewatering is accomplished by pumping water either directly from the pit or underground workings or from wells surrounding these areas. This pumping of water may create a cone of depression, which is a cone-shaped reduction in water level extending outward from the point of water withdrawal, where water levels are lowest. Water extracted during dewatering typically is pumped to lined process water ponds for use in the milling process. Excess water typically is tested and, if necessary, treated before discharge.

In hard rock metal mining, most water use occurs during milling and separation operations. This water is obtained from the mine site area and then held in storage facilities until its use. However, much of the water used in the mining process is recycled and reused. For example, the water used to pump tailings slurry from the mill to the tailings storage facility (TSF) becomes available when the tailings solids settle and excess overlying water is recycled back to the mill. Other water use needs include power plant cooling and transport of metal concentrate slurry (where transport occurs via pipeline).

In general, stormwater runoff is diverted around mine components (e.g., the open pit or waste rock piles) to keep it from becoming contaminated, and then collected in sedimentation ponds to settle out suspended solids prior to use or discharge to a stream. Stormwater runoff that contacts mine components may be contaminated with pollutants. Such water is directed to collection ponds and treated before being used in mine processes or released. Seepage and leachate are directed to storage ponds for containment, treated, and released to the environment. Tailings may be dewatered, and reclaimed water directed to process water holding ponds for reuse. Surface water and groundwater are monitored for contamination throughout mine operations, and are routed to a treatment facility if significant contamination is detected.

Water treatment options include physical or chemical methods—for example, reverse osmosis (physical) and formation of precipitated solids (chemical)—used together or independently. The choice of treatment methods and the chemicals used for treatment depends on the site's specific water chemistry and the water's end use.

Once mining ceases, an open pit is typically allowed to fill with water. Acid-generating waste rock and other potentially acid-generating (PAG) materials (e.g., pyrite-rich tailings) may be placed at the bottom of the pit to submerge these materials and reduce the potential for acid mine drainage once the pit fills. In block caving, ore is removed from the ground and the resulting void is filled by overlying materials (Box 4-4). After mining operations cease, groundwater fills in the remaining pore spaces in the void.

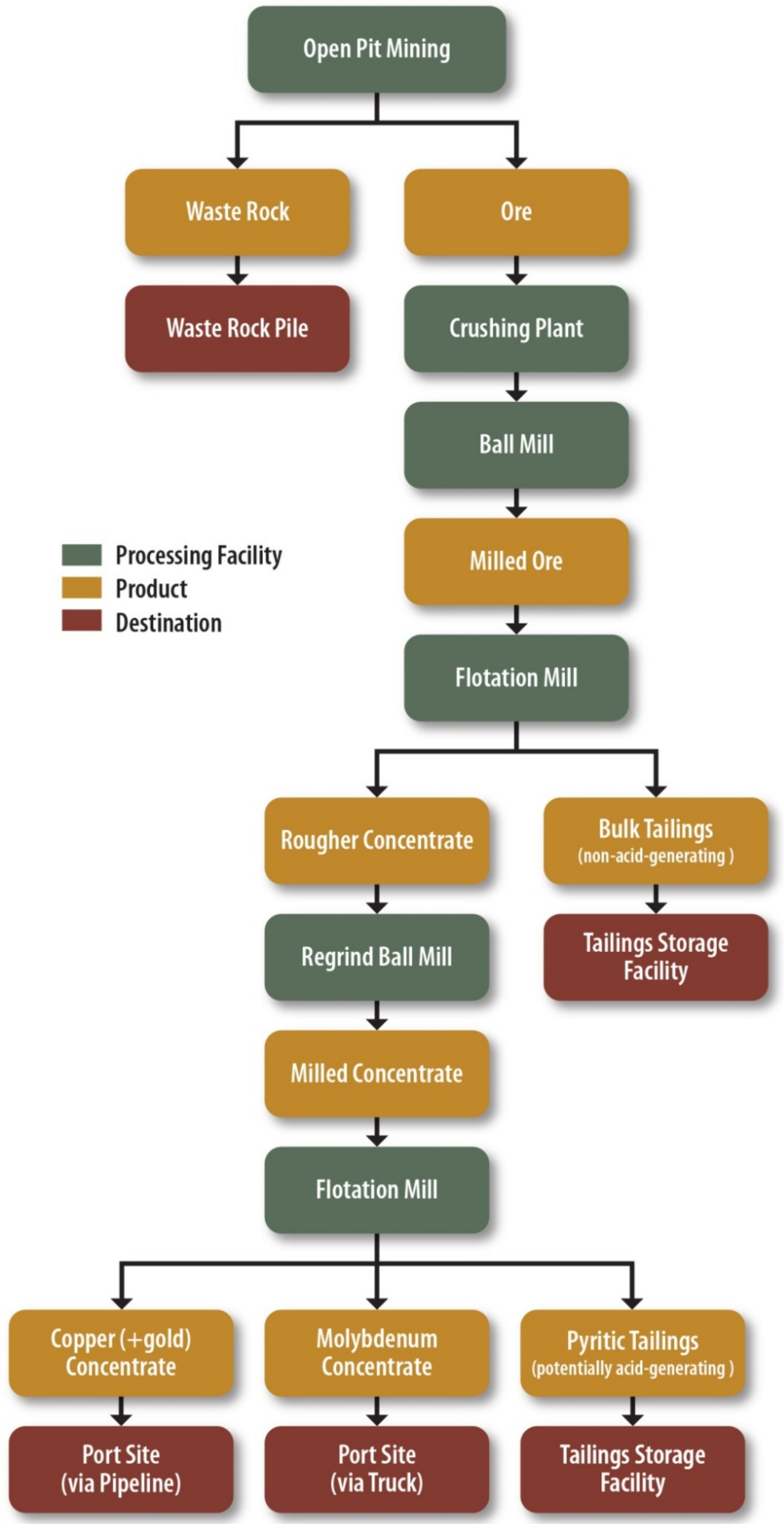
4.2.3.3 Ore Processing

Generally, two streams of materials come from a mine: ore and waste rock (Figure 4-3). Ore is rock with sufficient amounts of metals to be economically processed. Waste rock is 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 technological or economic conditions).

Ore blasted from a porphyry copper mine typically is hauled to a crushing plant near or in the mine pit (Figure 4-3). 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., 80% to 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 (Box 4-5) to collect valuable copper, molybdenum, and gold minerals in a copper-molybdenum concentrate, which also contains gold. Bulk tailings are the material remaining after the first flotation circuit, which are directed to a TSF (Figure 4-3). Figure 4-3 assumes NAG bulk tailings; however, if prior testing has indicated the potential for acid production, they can be treated further to minimize this potential prior to their disposal. The copper-molybdenum (+gold) concentrate may be fed through a second ball mill to regrind the particles (e.g., 80% to less than 25 μm; Ghaffari et al. 2011). Once sufficiently sized, the regrind 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 4-3).

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). Traditionally, PAG and NAG tailings were discharged together, thereby contributing to the acid-generating potential of the TSF. It is possible to use a technique called selective flotation to separate most of the pyrite into the cleaner circuit tailings (PAG) with the rougher tailings (bulk tailings in Figure 4-3) comprising predominantly NAG minerals. The PAG tailings would need to be stored separately and kept isolated from oxygen.

Figure 4-3. Simplified schematic of mined material processing.



BOX 4-5. CHEMICALS USED IN ORE PROCESSING AND HANDLING

After dry grinding and milling, water is added to the fine ore particles to create a slurry. This slurry undergoes further beneficiation using chemical reagents to separate minerals from gangue (rock barren of target minerals) and to separate one mineral from another. Reagents are added to the slurry at different points in the process to chemically or physically modify the surface of particles and facilitate separation. The amounts and types of reagents used are site-specific and depend on many factors such as particle size variation, particle density, ore grade, and host rock character. The volume of reagents used per metric ton of ore is closely monitored to optimize the mineral concentration process and minimize the unnecessary use of reagents. Although highly site-specific, most reagents are used at a rate of 0.01 to 0.3 kg of reagent per metric ton of ore (USEPA 1994a, Khoshdast and Sam 2011). To ensure the flotation system is optimized, the incoming ore composition is monitored and the reagent mix is modified as changes occur due to variations in the ore.

The reagents used in flotation generally fall into five categories.

- **Collectors** (e.g., xanthates, dithiophosphates) increase the ability of air bubbles to stick to a particle. Toxicity of collectors varies widely within the group, but some commonly used collectors, such as sodium ethyl xanthate, are toxic to freshwater organisms (Alto et al. 1977, Vigneault et al. 2009).
- **pH regulators** (e.g., lime, caustic soda, sulfuric acid) are added to maintain the proper pH level in the slurry. If released, these reagents could affect pH in natural waters.
- **Frothers** (e.g., aliphatic alcohol, methylisobutyl carbinol, propylene glycol) increase the stability of air bubbles so they do not burst before bringing a particle to the surface. These reagents are generally considered to have low toxicity (Fuerstenau 2003).
- **Flocculants and dispersants** (e.g., polyacrylamides, aluminum salts, polyphosphate) promote settling of fine materials and separation of fine gangue materials. They are generally considered to have low toxicity (Vigneault et al. 2009).
- **Modifiers** (e.g., cyanide salts, carboxymethylcellulose) make collectors more effective by either activating or depressing certain reactions. Toxicity of these reagents varies widely.

Although some of these reagents can be transported to a mine site as powder or pellets, most material arrives in liquid form.

The gold in porphyry copper deposits is 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 would stay with the copper (+gold) concentrate and be recovered at an off-site smelter. Gold associated with pyrite would end up in the TSF unless a separate pyrite concentrate were produced, and gold could be recovered from this concentrate by a vat leaching cyanidation process (Logsdon et al. 1999, Marsden and House 2006) (Box 4-6).

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 processing and extraction costs. The process proposed by Ghaffari et al. (2011) would recover 86.1% of the copper, 83.6 % of the molybdenum and 71.2% of the gold from the Pebble deposit ore. The residual metals remaining with the tailings would be discharged to a TSF along with the residue of blasting agents, flotation reagents, and inert portions of the ore.

BOX 4-6. USE OF CYANIDE IN GOLD RECOVERY

At mines producing both copper and gold, copper concentrate and gold doré (unrefined gold produced at the mine site) are extracted using standard processes such as gravity separation and froth flotation. If enhanced gold recovery is undertaken at the mine site, cyanide is universally used for such gold extraction (Marsden and House 2006).

The gold recovery process involves a cyanide leach step. The solution that remains after the cyanidation process is commonly passed through either a cyanide recovery unit or a cyanide destruction unit. Cyanide recovery allows the recycling of cyanide for reuse in the cyanidation process. Cyanide destruction converts the cyanide ion to less toxic cyanate, which is then treated in a wastewater treatment plant for discharge or transferred to a tailings storage facility (TSF). Because the tailings from this process have high concentrations of acid-generating sulfides, they are typically directed to the TSF, encapsulated in non-acid-generating tailings, and kept saturated to minimize oxidation. If water is recycled from the TSF into the copper process water system, cyanide can interfere with the flotation process; to prevent this interference, some mines isolate cyanidation tailings in a separate TSF (Scott Wilson Mining 2005).

Once in the TSF, cyanide concentrations may decrease through natural attenuation (e.g., volatilization, photodegradation, biological oxidation, precipitation) (Logsdon et al. 1999). Cyanide may escape the TSF through seepage or as dust from tailings beaches. Because cyanide dissolves other metals such as copper, fauna also may be exposed to high metal concentrations and toxic copper-cyanide complexes.

Reported rates of cyanide use at gold mines average about 0.15 to 0.50 kg of cyanide (as sodium cyanate) per metric ton of concentrate after cyanide recovery (Stange 1999).

4.2.3.4 Tailings Storage

Tailings are a mixture of fine-grained particles, water, and residues of reagents remaining from the milling process. The most common method of tailings storage is disposal in an impoundment (i.e., a TSF) (Porter and Bleiwas 2003). Tailings are transported from the mill to a TSF as a slurry, of which solids—silt to fine sand particles (0.001 to 0.6 mm) with concentrations of metals too low to interact with flotation reagents—typically make up 30 to 50% by weight. Tailings may be thickened (dewatered) prior to disposal. Thickening reduces evaporation and seepage losses and allows recycling of more process water back to the processing plant, thereby reducing operational water demand. It also minimizes the amount of water stored in the TSF.

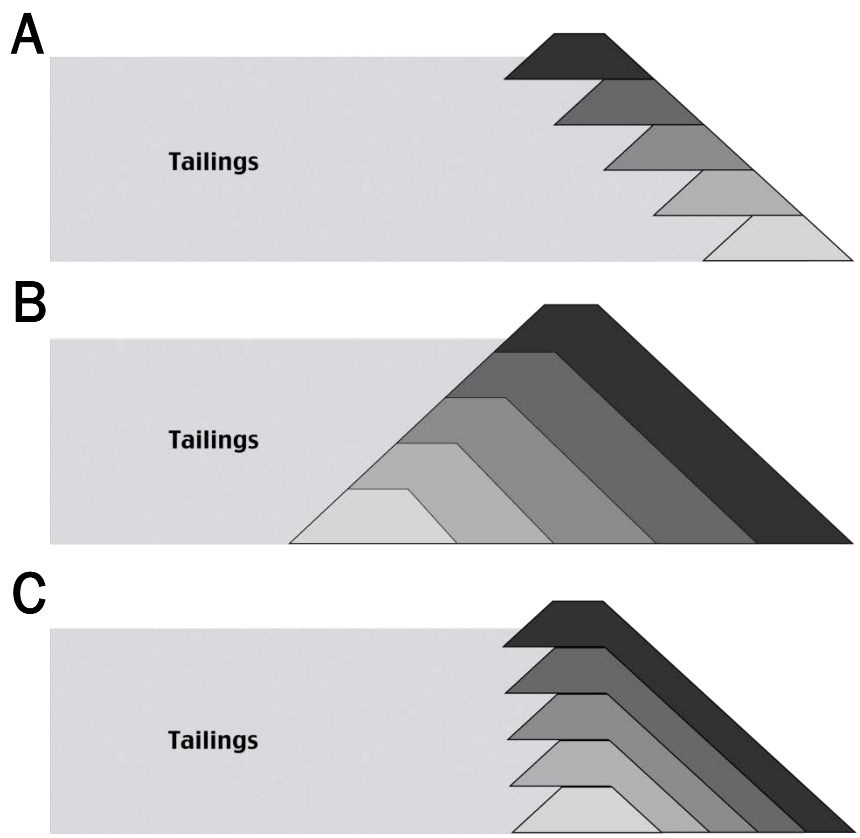
Tailings impoundments are water-holding structures typically built by creating a dam in a valley. Tailings dams are generally earthen or rockfill dams constructed from waste rock or the coarse fraction of the tailings themselves. The majority of existing tailings dams are less than 30 m in height, but the largest exceed 150 m (McLeod and Murray 2003, National Inventory of Dams 2005, Rico et al. 2008).

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, called lifts, over the lifetime of the mine, such that dam height increases ahead of reservoir level, using upstream, downstream, or centerline methods (Figure 4-4). 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). This is because part of the dam rests on the tailings, which have a lower density and a higher water saturation than the dam materials (USEPA 1994b). 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 employed (Davies 2002). For example, an upstream dam lift was recently designed and constructed on the Fort Knox Mine tailings impoundment dam near Fairbanks, Alaska (USACE 2011). The downstream method is considered more stable from a seismic standpoint, but it is more expensive to implement than the upstream method. Centerline construction has characteristics of both upstream and downstream types (USEPA 1994b, 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 either for use in the mining process or for treatment and subsequent discharge to local surface waters. Tailings are deposited against the embankment through spigots or cyclones. 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.

Figure 4-4. 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 scenarios are assumed to use the downstream construction method initially and at some point change to centerline construction.



Liners may cover the entire impoundment area (e.g., as proposed for the Donlin Creek Mine TSF in Alaska) or only the pervious bedrock or porous soils. Full liners beneath TSFs are not always used; however, at least in Australia, mining companies are required to justify why a liner would not be necessary (e.g., the foundation has a sufficiently low saturated hydraulic conductivity or the groundwater has no beneficial use) (Commonwealth of Australia 2007). Full liners may not be economically practicable, in which case partial liners may be used to cover areas of pervious bedrock or porous soils.

Liners can include a high-density polyethylene, bituminous, or other type of geosynthetic material (geomembrane) 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 the service life of these liners. Laboratory tests and data from landfills suggest that high-density polyethylene liner lifespans range from 69 to 600 years, depending on whether it is the primary (upper) or secondary (lower or backup) liner (Rowe

2005, Koerner et al. 2011). In general, longer lifespans are expected at lower temperatures and exposures to light (Rowe 2005, Koerner et al. 2011). Breakdown of the liner material and punctures by equipment or rocks may limit the effective life of liners (Rowe 2005). Overly steep slopes also may put stresses on geomembranes and cause them to fail. Service life data for other types of geomembranes are anecdotal and based on field performance, since no laboratory studies have been conducted (Koerner et al. 2011).

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 are filtered and "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) (Box 4-7). Dry stack technology has found greatest acceptance in arid regions where water is scarce or expensive, 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 tailings disposal 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; Pogo, a gold mine in eastern interior Alaska; and Nixon Fork, a gold mine in west-central Alaska).

BOX 4-7. DRY STACK TAILINGS MANAGEMENT

In a dry stacking operation, tailings are dried using filter presses or vacuum technologies such that water content typically falls below 20%. The dewatered tailings are either loaded into trucks or transported by conveyer to the tailings storage facility (TSF), where they are spread in lifts and compacted, similar to a traditional earth-moving operation.

The compacted tailings have a higher in-place bulk density than tailings placed using more conventional slurry methods. We estimate that dry stacking would reduce the required volume for tailings storage by approximately 15%. The lower water content of dry stack tailings means that less water is captured in the void spaces between solid tailings particles, reducing the amount of water "lost" to the TSF by approximately one-third. The additional water that is not captured in the TSF is available for treatment and release, potentially reducing streamflow losses in local streams. The higher density and lower water content of the tailings also increase their stability. In many cases, the need for a confining embankment and the risk of a tailings dam failure and tailings liquefaction can be eliminated with dry stack management.

The additional capital costs for dewatering equipment and the high energy cost of dewatering have often been barriers to adopting dry stack tailings management for low-grade ores such as porphyry copper. However, higher production costs may be at least partially offset by cost savings in other areas. For example, the increased stability of a dry stacked TSF may reduce closure costs, post-closure monitoring costs, and post-closure financial assurance requirements.

Dry stacked tailings are typically placed in unsaturated conditions, which can increase the exposure of tailings to oxygen. Thus, this type of storage may be less appropriate for potentially acid-generating tailings or may require additional engineering controls to limit, collect, or treat acid drainage. Where TSFs are typically used to store water as well as tailings, the use of dry stack tailings may not eliminate the need for construction and operation of a separate water impoundment facility.

4.2.3.5 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), typically in large, terraced stockpiles. 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 also may be blended with ore in the mill to maintain a steady and predictable composition of feed material for the flotation process over time. NAG waste rock may be placed in piles near the open pit, with ditches to divert stormwater around the piles and drains (or other systems) to capture leachate or direct it toward the open pit. At closure, a dry cover (e.g., encapsulation) can be placed over the waste rock pile to isolate it from water and oxygen, or the pile could be placed into the completed open pit and kept below the water line if it contains PAG material, depending on site-specific characteristics (O'Kane and Wels 2003). With small pits and in some settings, it is beneficial to fill the pit with waste rock and other waste material and then construct a dry cover over the filled pit area.

4.2.4 Timeframes

The mining process described above can be thought of in terms of three distinct periods.

- **Operation** refers to the period during which the mine is active—that is, the period when mine infrastructure is being built and ore is being extracted and processed.
- **Closure** refers to the period following completion of mining operations (either as planned or prematurely) when mining has ceased and activities related to reclamation and preparation of the site for future stability continue. During this period, waste areas are reclaimed and facilities needed to support ongoing monitoring and maintenance activities—such as stormwater management ditches, monitoring wells, engineered covers on waste materials (if required), wastewater treatment plants, and roads—are created, retained from the operational period, or replaced or remediated if they had become compromised.
- **Post-closure** refers to the extended period following closure activities when monitoring and maintenance activities continue. During this time, water leaving the site is monitored and treated for as long as contaminants are present at levels exceeding regulatory standards. The post-closure phase may last decades, centuries, or longer, until only minimal oversight is required. Such minimal oversight is necessary, perhaps in perpetuity, to ensure the remaining infrastructure's structural integrity and to minimize environmental impacts. Given the limited lifetime of human institutions, continued monitoring and maintenance of the site might become increasingly unlikely as the time from mine closure increases.

CHAPTER 5. ENDPOINTS

5.1 Overview of Assessment Endpoints

Selection of assessment endpoints is a key component of the problem formulation stage of an ecological risk assessment. Each endpoint is an explicit expression of the environmental values of concern in the assessment, in terms of both the entity valued (e.g., a species, community, or ecological process) and a potentially at-risk characteristic or attribute of that entity (USEPA 1998). Endpoints can be defined at any level of ecological organization, from within an organism to across ecosystems, depending on the needs of the assessment. In all cases, however, selected endpoints should be relevant to both ecology and decision-maker needs, as well as susceptible to potential stressors (USEPA 1998).

We consider three endpoints in this assessment: (1) the abundance, productivity, or diversity of the region's Pacific salmon and other fish populations; (2) the abundance, productivity, or diversity of the region's wildlife populations; and (3) the health and welfare of Alaska Native cultures. Endpoint 1 is evaluated in terms of direct effects of mining; endpoints 2 and 3 are evaluated indirectly, in terms of effects resulting from fish-related impacts (i.e., via fish-mediated effects). Each of these endpoints meets the criteria of ecological relevance, management relevance, and potential susceptibility to stressors associated with large-scale mining.

The assessment focuses most heavily on Endpoint 1, which is the only endpoint for which direct effects of mining are considered (Section 2.2.1). Most analyses center on Pacific salmon, rainbow trout, and Dolly Varden. This focus reflects the ecological, economic, and cultural significance of these fish species, as well as data availability. Other parts of the region's aquatic ecosystems, including algae, aquatic invertebrates, and smaller resident fishes such as sculpins, also may be affected by large-scale mining. However, these taxa are not as relevant to decision makers and data on their distributions, abundances, and susceptibilities are more limited.

We evaluate Endpoints 2 and 3 indirectly, in terms of the effects of large-scale mining on Endpoint 1 (i.e., via fish-mediated effects). This focus on indirect effects is not meant to suggest that mining would directly affect only fish populations, or that direct effects of mining on wildlife and Alaska Native populations would be inconsequential. Rather, it reflects the ecological and regulatory importance of the region's fisheries and their susceptibility to potential impacts. Under Endpoint 2, we focus on wildlife species that depend on salmon for food (e.g., brown bear, bald eagles, gray wolves, waterfowl) or that are important subsistence foods for Alaska Natives (e.g., moose, caribou). Although Alaska Natives are not the only people who would potentially be affected by mining in the region, Endpoint 3 focuses on Alaska Native populations because of the centrality of salmon and other salmon-dependent resources to their way of life and well-being, and because this assessment was initiated in response to requests from federally recognized tribal governments to restrict large-scale mining in the watersheds. We focus on the primary Alaska Native cultures of the Nushagak and Kvichak River watersheds, the Yup'ik and Dena'ina. Sugpiaq people, who traditionally lived along the Alaska Peninsula within the greater Bristol Bay watershed, still live in this region. However, because the Alaska Peninsula falls outside the Nushagak and Kvichak River watersheds, these cultures were not included in the assessment (Box 5-1). We also recognize that non-Native people have lived in the Bristol Bay region for hundreds of years, and also consider salmon integral to their way of life. Further discussion of the scope of the assessment and how this scope was defined can be found in Chapters 1 and 2.

In the following sections, we discuss each of the three assessment endpoints in greater detail. We present information on the fish and wildlife species considered, including what is known about their life histories, distributions, and abundances both across the Bristol Bay watershed (Scale 1) and within the Nushagak and Kvichak River watersheds (Scale 2). We discuss the Alaska Native populations in the region and examine why the region's salmon fisheries are an ecologically, economically, and culturally important resource.

BOX 5-1. CULTURAL GROUPS IN THE BRISTOL BAY WATERSHED

Within the Bristol Bay watershed there are three main cultural groups: the Yup'ik, the Dena'ina, and the Sugpiaq. Prior to western contact, these three groups tended to be seasonally dispersed, with large populations periodically gathering in a central location. Westernization efforts by both Russia and the United States promoted permanent communities with year-round occupation. Some communities grew around traditional Alaska Native sites (e.g., Nondalton); other communities were built where resources were more concentrated or accessible. Naknek is one of the older recorded communities in the Bristol Bay region, with archaeological surveys indicating that Alaska Natives have occupied the Naknek area for at least 6,000 years.

Although there are descendants of the Sugpiaq that currently live both along the Alaska Peninsula and within the Nushagak and Kvichak River watersheds, this assessment focuses on the primary cultural groups found within the Nushagak and Kvichak River watersheds, the Yup'ik and the Dena'ina.

5.2 Endpoint 1: Salmon and Other Fishes

The Bristol Bay watershed is home to at least 29 fish species, representing at least nine different families (Table 5-1). The region is renowned for its fish populations, and it supports world-class fisheries for multiple species of Pacific salmon and other game fishes (Dye and Schwanke 2009). These resources generate significant benefit for commercial fishers, support valued recreational fisheries (Figure 5-1), and provide sustenance for Alaska Native populations and other rural residents (Figure 5-2, Box 5-2).

In this section we summarize key fish species found in the Bristol Bay watershed, their distributions and abundances in the region, and some of the factors contributing to the significance of these resources. This background information is provided to underscore the uniqueness of the region's fisheries and support the assessment's focus on potential impacts of large-scale mining on these fishes. More detailed discussion of the region's fishes can be found in Appendices A and B.

Table 5-1. Fish species reported in the Nushagak and Kvichak River watersheds. (H) indicates species considered to be harvested—that is, they are well-distributed across these watersheds and are or have been targeted by sport, subsistence, or commercial fisheries. This list does not include primarily marine species that periodically venture into the lower reaches of coastal streams. See Appendix B, Table 1, for references and additional information on the abundance and life history of each species.

Family	Species	Relative Abundance
Salmonids (Salmonidae)	Bering cisco (*Coregonus laurettae*)	Very few specific reports
	Humpback whitefish (H) (*C. pidschian*)	Common in large upland lakes; locally and seasonally common in large rivers
	Least cisco (*C. sardinella*)	Locally common in some lakes (e.g., Lake Clark, morainal lakes near Iliamna Lake); less common in Iliamna Lake and large slow-moving rivers such as the Chulitna, Kvichak, and lower Alagnak
	Pygmy whitefish (*Prosopium coulterii*)	Locally common in a few upland lakes or adjacent streams
	Round whitefish (*P. cylindraceum*)	Abundant/widespread throughout larger streams in upland drainages; not found in headwaters or coastal plain areas
	Coho salmon (H) (*Oncorhynchus kisutch*)	Juveniles abundant/widespread in upland flowing waters of Nushagak River watershed and in some Kvichak River tributaries downstream of Iliamna Lake; present in some Iliamna Lake tributaries; not recorded in the Lake Clark watershed
	Chinook salmon (H) (*O. tshawytscha*)	Juveniles abundant and widespread in upland flowing waters of Nushagak River watershed and in Alagnak River; infrequent upstream of Iliamna Lake
	Sockeye salmon (H) (*O. nerka*)	Abundant
	Chum salmon (H) (*O. keta*)	Abundant in upland flowing waters of Nushagak River watershed and in some Kvichak River tributaries downstream of Iliamna Lake; rare upstream of Iliamna Lake
	Pink salmon (H) (*O. gorbuscha*)	Abundant (in even years), with restricted distribution, in the Nushagak River watershed and in some Kvichak River tributaries downstream of Iliamna Lake; rare upstream of Iliamna Lake
	Rainbow trout (H) (*O. mykiss*)	Frequent/common; in summer, closely associated with spawning salmon
	Arctic char (H) (*Salvelinus alpinus*)	Locally common in upland lakes
	Dolly Varden (H) (*S. malma*)	Abundant in upland headwaters and selected lakes
	Lake trout (H) (*S. namaycush*)	Common in larger upland lakes and seasonally present in lake outlets; absent from the Wood River lakes
	Arctic grayling (H) (*Thymallus arcticus*)	Abundant/widespread
Lampreys (Petromyzontidae)	Arctic lamprey (*Lethenteron camtschaticum*)	Juveniles common/widespread in sluggish flows where fine sediments accumulate[a]
	Alaskan brook lamprey (*L. alaskense*)	
	Pacific lamprey (*Entosphenus tridentatus*)	Rare
Suckers (Catostomidae)	Longnose sucker (*Catostomus catostomus*)	Common in slower flows of larger streams
Pikes (Esocidae)	Northern pike (H) (*Esox lucius*)	Common/widespread in still or sluggish waters

Table 5-1. Fish species reported in the Nushagak and Kvichak River watersheds. (H) indicates species considered to be harvested—that is, they are well-distributed across these watersheds and are or have been targeted by sport, subsistence, or commercial fisheries. This list does not include primarily marine species that periodically venture into the lower reaches of coastal streams. See Appendix B, Table 1, for references and additional information on the abundance and life history of each species.

Family	Species	Relative Abundance
Mudminnows (Umbridae)	Alaska blackfish (*Dallia pectoralis*)	Locally common/abundant in still or sluggish waters in flat terrain
Smelts (Osmeridae)	Rainbow smelt (*Osmerus mordax*)	Seasonally abundant in streams near the coast
	Pond smelt (*Hypomesus olidus*)	Locally common in coastal lakes and rivers, Iliamna Lake, inlet spawning streams, and the upper Kvichak River; abundance varies widely interannually
	Eulachon (*Thaleichthys pacificus*)	No or few specific reports; if present, distribution appears limited and abundance low
Cods (Gadidae)	Burbot (*Lota lota*)	Infrequent to common in deep, sluggish, or still waters
Sticklebacks (Gasterosteidae)	Threespine stickleback (*Gasterosteus aculeatus*)	Locally abundant in still or sluggish waters; abundant in Iliamna Lake
	Ninespine stickleback (*Pungitius pungitius*)	Abundant/widespread in still or sluggish waters
Sculpins (Cottidae)	Coastrange sculpin (*Cottus aleuticus*)	Abundant/widespread[b]
	Slimy sculpin (*C. cognatus*)	

Notes:

[a] These species are combined here, because juveniles, the most commonly encountered life stage for each, are indistinguishable.

[b] These species are combined here, because they are not reliably distinguished in field conditions, although slimy sculpin is thought to be more abundant and widely distributed.

Figure 5-1. Approximate extents of popular Chinook and sockeye salmon recreational fisheries in the vicinity of the Nushagak and Kvichak River watersheds. Areas were digitized from previously published maps (Dye et al. 2006). Recreational rainbow trout fisheries are also distributed throughout the watersheds.

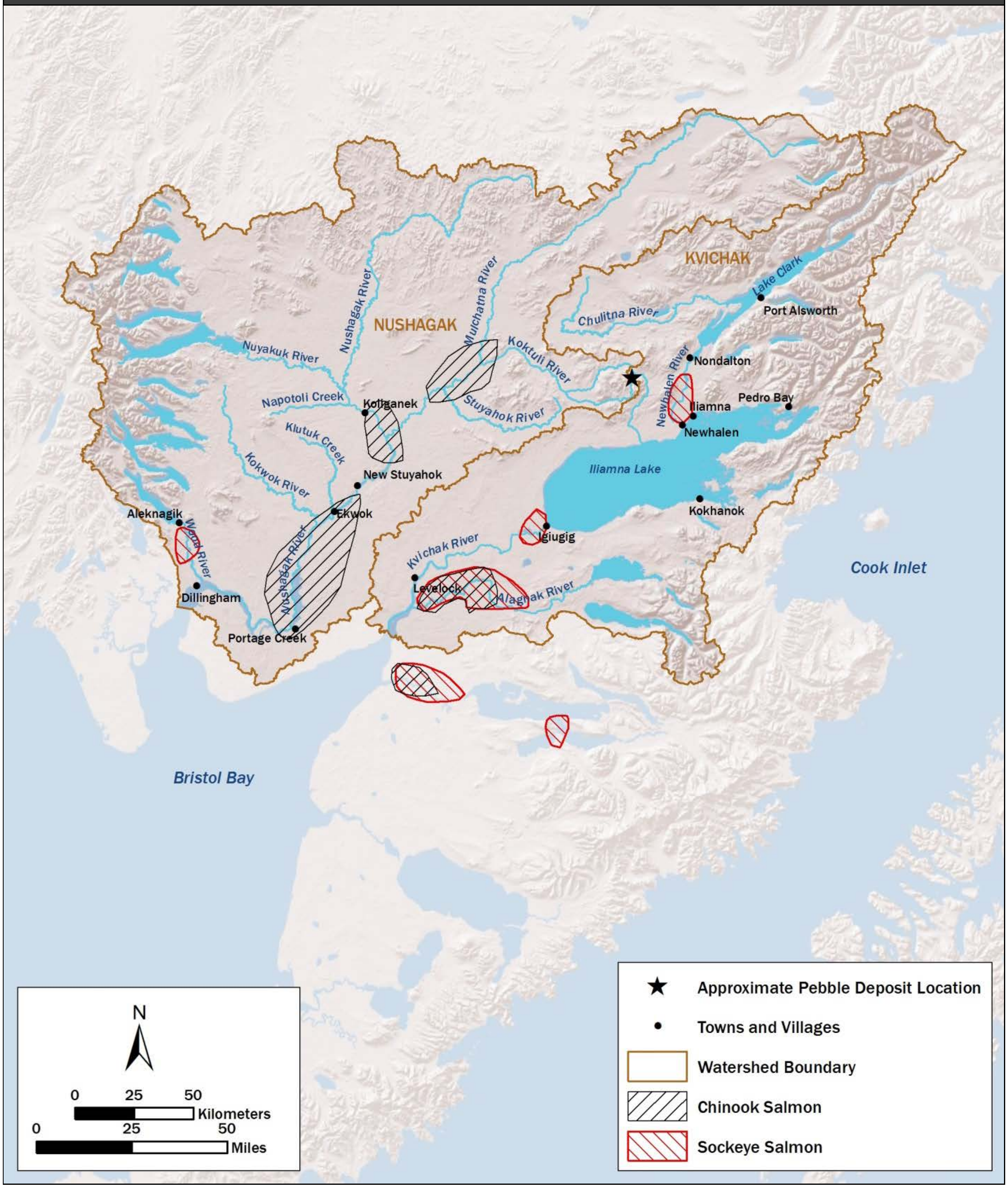
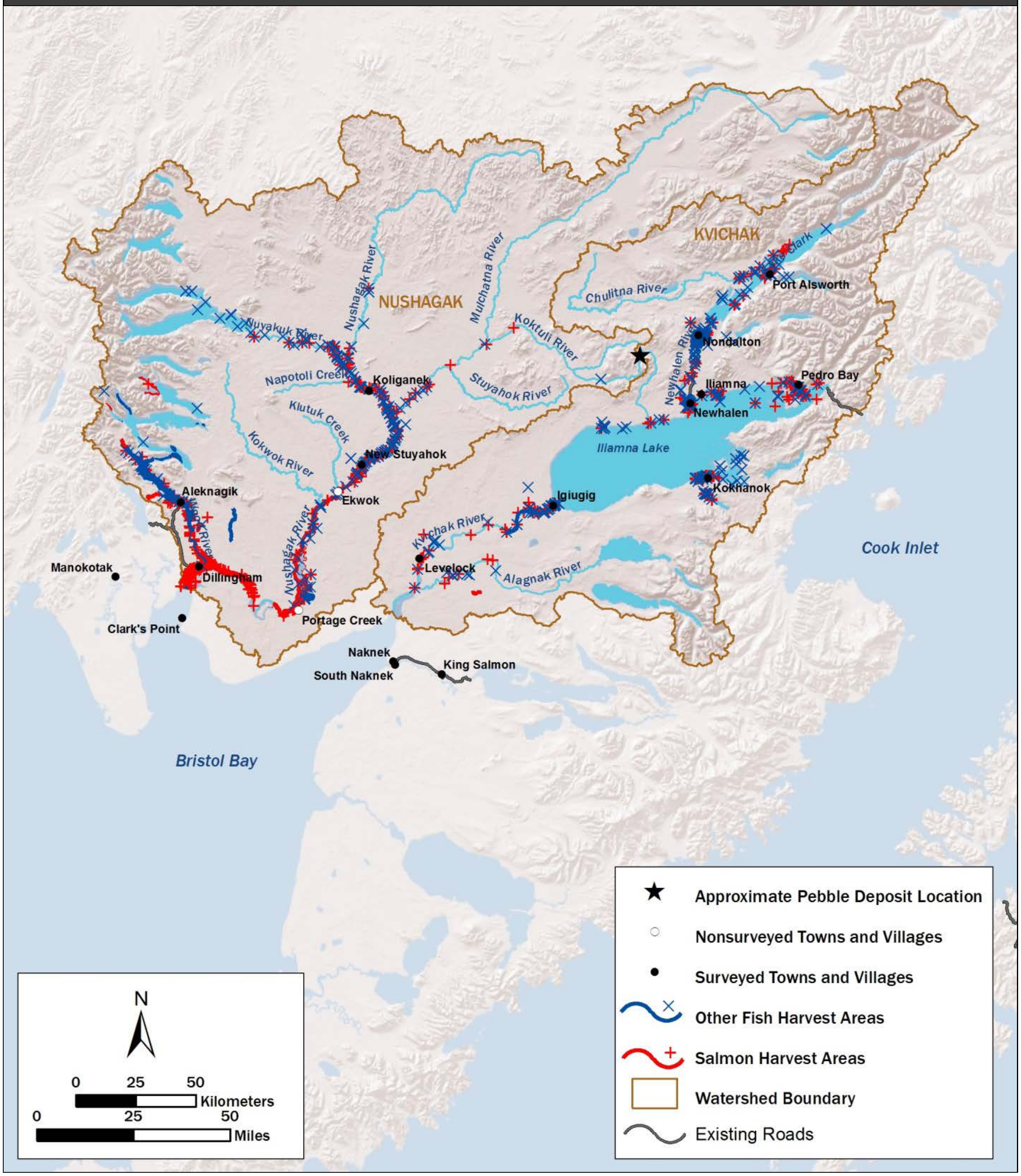


Figure 5-2. Subsistence harvest and harvest effort areas for salmon and other fishes within the Nushagak and Kvichak River watersheds. Other fishes are defined as those non-salmon and whitefish species discussed in the text. Each fish category is designated by a representative individual color and includes all harvest points, lines, or polygons meeting that classification. See Box 5-2 for more detailed discussion of methodology.



BOX 5-2. SUBSISTENCE USE METHODOLOGY

Subsistence use and harvest data were extracted from data collected by the Alaska Department of Fish and Game in collaboration with Stephen R. Braund and Associates (Fall et al. 2006, Krieg et al. 2009, Holen and Lemons 2010, Holen et al. 2011, Holen et al. 2012). These data are a compilation of a multi-year study to document and examine baseline subsistence use and harvest (via both directed or targeted efforts and incidental catches), along with demographic and economic data within the communities near the Pebble deposit. Eighteen communities were interviewed: Aleknagik, Clark's Point, Dillingham, Igiugig, Iliamna, King Salmon, Kokhanok, Koliganek, Levelock, Lime Village, Manokotak, Naknek, New Stuyahok, Newhalen, Nondalton, Pedro Bay, Port Alsworth, and South Naknek.

Members of participating households within each community were asked to document where they hunted, fished, and gathered subsistence resources during the previous year by adding points (used for harvest locations), polygons (used for harvest effort areas), and lines (used to depict trap lines or courses travelled during fish trolling) to various maps. Interviews were conducted from 2004 to 2011; not every community was interviewed in the same year, so the reported years differed between communities. Following completion of interviews, hand-drawn maps were digitized and data compiled for use within a geographic information system. In this assessment, only towns and villages documenting subsistence use and harvest within the Nushagak and Kvichak River watersheds were considered; data points or sections of polygons and lines falling outside the boundary of these watersheds were omitted.

Subsistence use and harvest data were extracted for four representative use categories: salmon, other fishes, wildlife, and waterfowl, based on tables found within each report (e.g., Holen et al. 2012: Table 1-16). Species or other general classifications within each category include:

- **Salmon:** chum salmon, Chinook (king) salmon, pink salmon, salmon, coho (silver) salmon, sockeye salmon, and spawning sockeye (red) salmon
- **Other fishes** *(i.e., non-salmon fish species and whitefishes)*: Arctic char, Dolly Varden, humpback whitefish, lake trout, least cisco, rainbow trout, round whitefish, steelhead trout, trout, and whitefish
- **Wildlife:** black bear, brown bear, caribou, and moose
- **Waterfowl:** black scoter, brant, Canada goose, eggs, geese, gull eggs, lesser snow goose, mallard, pintail, sandhill crane, teal, tern eggs, tundra swan, waterfowl, and white-fronted goose

Data were extracted for all points, lines, and polygons in each category, for each interviewed community. Data were then summed across all communities to produce a cumulative layer for the entire Nushagak and Kvichak River watersheds. Subsistence intensity across the landscape was derived by first generating a 1-km square grid across the Nushagak and Kvichak River watersheds. Each documented point, line, and polygon shapefile was spatially joined and summed across the 1-km grid to account for multiple or overlapping points, lines, and polygons within the same 1-km pixel. Therefore, each pixel represents the total number of points and sections of lines and polygons within its boundaries. Subsistence use was then summed across the four representative use categories to derive total cumulative subsistence use across the Nushagak and Kvichak River watersheds.

This subsistence use metric provides a coarse measure of areas that are used for subsistence uses more than others within the watersheds. However, it is important to note some of the limitations of the subsistence intensity metric. Points represent harvest locations, but the way these data are tabulated does not confer abundance of species harvested within the pixel. Therefore, a point may represent either a single capture or multiple captures of a given species. Although abundance information was collected by the researchers, it was not consistently reported in the geospatial data. Further, the line and polygon files represent general catch areas and not point of actual capture, allowing broad areas to have the same value as an actual point of capture. Finally, since this assessment is focused on fish as the main assessment endpoint, we focus on aquatic species and habitats. Many other plant and animal species included in the subsistence use databases were not used to arrive at this subsistence intensity metric.

5.2.1 Species and Life Histories

5.2.1.1 Salmon

Five species of Pacific salmon spawn and rear in the Bristol Bay watershed's freshwater habitats: sockeye or red (*Oncorhynchus nerka*), coho or silver (*O. kisutch*), Chinook or king (*O. tshawytscha*), chum or dog (*O. keta*), and pink or humpback (*O. gorbuscha*). Because no hatchery fish are raised or released in the watershed, Bristol Bay's salmon populations are entirely wild.

All five salmon species share a trio of life-history traits that contribute to the success and significance of these species in the Bristol Bay region. First, they are anadromous: they hatch in freshwater habitats, migrate to sea for a period of relatively rapid growth, and then return to freshwater habitats to spawn. Second, the vast majority of adults return to their natal freshwater habitats to spawn. This homing behavior fosters reproductive isolation, thereby enabling populations to adapt to the particular environmental conditions of their natal habitats (Blair et al. 1993, Dittman and Quinn 1996, Eliason et al. 2011). Homing is not absolute, however, and this small amount of straying increases the probability that suitable habitats will be colonized by salmon (e.g., Milner and Bailey 1989). Finally, each species is semelparous: adults die after spawning a single time. After completing their upstream migration, females excavate nests (redds) in the gravel and release eggs into them. These eggs are fertilized by one or more competing males as they are released, and the females bury them in the nests. The females and males then die, depositing the nutrients incorporated into their bodies in their spawning habitats (Section 5.2.5).

The seasonality of spawning and incubation is roughly the same for all five species, although the timing can vary somewhat by species, population, and region. In general, salmon spawn from summer through fall, and fry emerge from spawning gravels the following spring to summer. Freshwater habitats used for spawning and rearing vary across and within species, and include headwater streams, larger mainstem rivers, side- and off-channel wetlands, ponds, and lakes (Table 5-2). With some exceptions, preferred spawning habitat consists of gravel-bedded stream reaches of moderate water depth (30 to 60 cm) and current (30 to 100 cm/s) (Quinn 2005). Sockeye are unique among the species, in that most populations rely on lakes as the primary freshwater rearing habitat (Table 5-2).

Both chum and pink salmon migrate to the ocean soon after fry emergence (Heard 1991, Salo 1991). 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 (Table 5-2), these species are more dependent on upstream freshwater resources than chum and pink salmon. As a result, potential large-scale mining in this region likely poses greater risks to sockeye, coho, and Chinook salmon.

Table 5-2. Life history, habitat characteristics, and total documented stream length occupied for Bristol Bay's five Pacific salmon species in the Nushagak and Kvichak River watersheds.

Salmon Species	Freshwater Rearing Period (years)	Freshwater Rearing Habitat	Ocean Feeding Period (years)	Spawning Habitat	Documented Stream Length Occupied (kilometers)
Sockeye	0–3	Lakes, rivers	2–3	Beaches of lakes, streams connected to lakes, larger braided rivers	4,600
Coho	1–3	Headwater streams to moderate-sized rivers, headwater springs, beaver ponds, side channels, sloughs	1+	Headwater streams to moderate sized rivers	5,900
Chinook	1+	Headwater streams to large-sized mainstem rivers	2–4	Moderate-sized streams to large-sized mainstem rivers	4,800
Chum	0	Limited	2–4	Moderate-sized streams and rivers	3,400
Pink	0	Limited	1+	Moderate-sized streams and rivers	2,200

Notes:
Data compiled from Appendix A, pages 4–13.

5.2.1.2 Other Fishes

In addition to the five Pacific salmon species discussed above, the Bristol Bay region is home to at least 24 other fish species, most of which typically (but not always) 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 (*O. mykiss*), Dolly Varden (*Salvelinus malma*), Arctic char (*S. alpinus*), Arctic grayling (*Thymallus arcticus*), humpback whitefish (*Coregonus pidschian*), northern pike (*Esox lucius*), and lake trout (*S. namaycush*), as well as numerous other species that are not typically harvested (Table 5-1). These fish species occupy a variety of habitats throughout the watershed, from headwater streams to rivers and lakes.

In this assessment, we focus primarily on the five Pacific salmon species, rainbow trout, and Dolly Varden (Box 2-3). This focus is not meant to imply that other fish species found in the Bristol Bay watershed are not economically, culturally, or ecologically important, or that they are unlikely to be affected by potential mining-related activities. Rather, it reflects the value of Pacific salmon, rainbow trout, and Dolly Varden as both sport and subsistence fisheries throughout the region, the potential sensitivity of these species to mine development and operation, and the relatively greater amount of information available for these species, particularly in terms of their distributions and abundances.

The species *O. mykiss* includes both a non-anadromous or resident form (commonly referred to as rainbow trout) and an anadromous form (commonly referred to as steelhead). In the Bristol Bay watershed, steelhead generally are restricted to a few spawning streams near Port Moller, on the Alaska Peninsula; thus, most populations throughout the region of the assessment are the non-anadromous form.

The spawning habitat and behavior of rainbow trout are generally similar to that of the Pacific salmon species, with a few key exceptions. First, rainbow trout are iteroparous, meaning that they can spawn repeatedly. Second, spawning occurs in spring, versus summer and early fall for salmon. Juveniles emerge from spawning gravels in summer (Johnson et al. 1994, ADF&G 2012), and immature fish may remain in their natal streams for several years before migrating to other habitats (Russell 1977).

Rainbow trout in the Bristol Bay watershed exhibit complex migratory patterns, moving between spawning, rearing, feeding, and overwintering habitats. For example, many adults in the region spawn in inlet or outlet streams of large lakes, then migrate shortly after spawning to feeding areas within those lakes. Some mature fish may seasonally move distances of 200 km or more (Russell 1977, Burger and Gwartney 1986, Minard et al. 1992, Meka et al. 2003). Often, these migratory patterns ensure that rainbow trout are in close proximity to the eggs and carcasses of spawning salmon, which provide an abundant, high-quality food resource (Meka et al. 2003). The variety of habitat types utilized by rainbow trout is reflected by different life-history types identified in the region, including lake, lake-river, and river residents (Meka et al. 2003). See Appendix B (pages 11–16) for additional information on rainbow trout life history.

Dolly Varden is a highly plastic fish species, with multiple genetically, morphologically, and ecologically distinct forms that can co-exist in the same water bodies (Ostberg et al. 2009). Both anadromous and non-anadromous Dolly Varden are found in the Bristol Bay watershed, and both life-history forms can exhibit complex and extensive migratory behavior (Armstrong and Morrow 1980, Reynolds 2000, Scanlon 2000, Denton et al. 2009). Anadromous individuals usually undertake three to five ocean migrations before reaching sexual maturity (DeCicco 1992, Lisac and Nelle 2000, Crane et al. 2003). During these migrations, Dolly Varden frequently leave one drainage, travel through marine waters, and enter a different, distant drainage (DeCicco 1992, DeCicco 1997, Lisac 2009). Non-anadromous individuals also may move extensively between different habitats (Scanlon 2000).

Dolly Varden spawning occurs in fall, upstream of overwintering habitats (DeCicco 1992). Northern-form anadromous Dolly Varden (the geographic form of Dolly Varden found north of the Alaska Peninsula) overwinter primarily in lakes and in lower mainstem rivers where sufficient groundwater provides suitable volumes of free-flowing water (DeCicco 1997, Lisac 2009). Within the Nushagak and Kvichak River watersheds, juveniles typically rear in low-order, high-gradient stream channels (ADF&G 2012). Because Dolly Varden occur in upland lakes and high-gradient headwater streams (ADF&G 2012)—farther upstream than many other fish species and above migratory barriers to anadromous salmon populations—they may be especially vulnerable to mine development and operation in these headwater areas. See Appendix B (pages 20–25) for additional information on Dolly Varden life history.

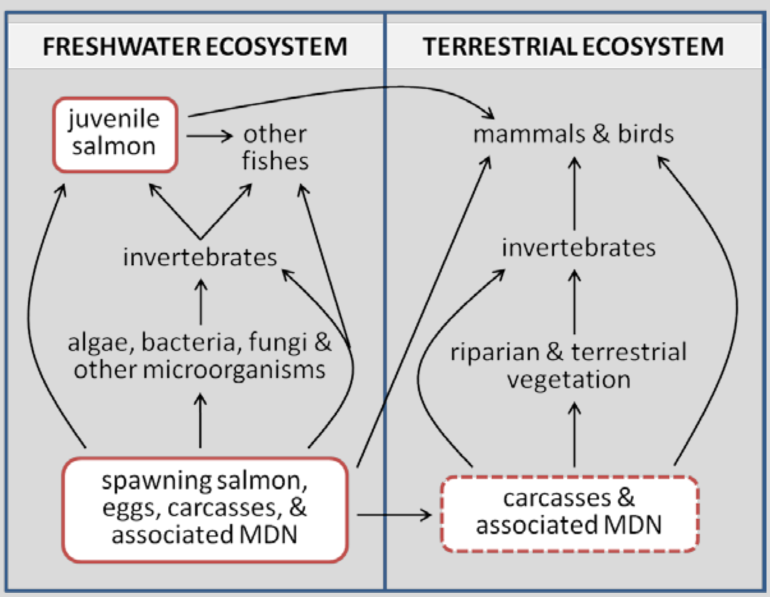
It is important to note that these endpoint species do not exist in isolation from other fish species. The biomass carried into the Bristol Bay watershed's aquatic habitats by spawning salmon is a fundamental driver of aquatic foodwebs (Box 5-3). Many of the species listed in Table 5-1 are prey for, predators of, or competitors with the endpoint species. For example, sculpins, Dolly Varden, and rainbow trout are well-known predators of salmon eggs and emergent fry, and northern pike can be effective predators of juvenile salmon and other fish species (Russell 1980, Sepulveda et al. 2013). Insectivorous and planktivorous fishes (e.g., Arctic grayling and pond smelt, respectively) may prey on similar species as juvenile salmonids (e.g., Hartman and Burgner 1972). Given these foodweb interactions, we recognize that shifts in the relative abundance of species are likely to have repercussions throughout the aquatic community; however, evaluation of the myriad foodweb interactions that could result from large-scale mining is beyond the scope of this assessment.

BOX 5-3. SALMON IN FRESHWATER AND TERRESTRIAL FOODWEBS

Salmon are a cornerstone species in the Bristol Bay region, in that they comprise a significant portion of the resource base upon which both aquatic and terrestrial ecosystems in the region depend (Willson et al. 1998). Adults returning to freshwater systems to spawn import marine-derived nutrients (MDN) back into these freshwater habitats. These nutrients provide the foundation for aquatic and terrestrial foodwebs via two main pathways: direct consumption of salmon in any of its forms (spawning adults, eggs, carcasses, and/or juveniles) and nutrient recycling (Gende et al. 2002).

Because salmon are a seasonally abundant, high-quality food resource in the Bristol Bay watershed, many aquatic and terrestrial species take advantage of this resource (e.g., see Sections 5.3 and 12.1). For example, Willson and Halupka (1995) found that more than 40 species of mammals and birds feed on salmon in southeastern Alaska. Salmon eggs and juveniles are eaten by many fishes, such as other salmon, rainbow trout, northern pike, and Dolly Varden (Appendix B).

The nutrients incorporated into spawning salmon biomass also can have a bottom-up effect on both freshwater and terrestrial ecosystems via nutrient recycling (Gende et al. 2002). Given that these systems tend to be nutrient-poor, MDN contributions play a significant role in the Bristol Bay region’s productivity. In lakes and streams, MDN help to fuel the production of algae, bacteria, fungi, and other microorganisms that make up aquatic biofilms. These biofilms in turn provide food for aquatic invertebrates, which are preyed on by juvenile salmon and other fishes. Terrestrial vegetation and invertebrates also receive a salmon-related nutrient subsidy, in the form of carcasses and excreta deposited on land by mammal and bird consumers.



Note that the simplified foodweb above (modified from Willson et al. 1998) focuses on how salmon serve as a resource base within and across freshwater and terrestrial ecosystems. Not all interactions, particularly those mediated by other species (e.g., invertebrates) and those that cross between freshwater and terrestrial ecosystems, are shown on this schematic. It also does not illustrate the role of salmon in estuarine and marine foodwebs, as these habitats are outside the scope of this assessment.

5.2.2 Distribution and Abundance

Fish populations throughout the Bristol Bay watershed have not been sampled comprehensively; thus, estimates of total distribution and abundance across the region are not available. However, available data (e.g., the Anadromous Waters Catalog, the Alaska Freshwater Fish Inventory, escapement and harvest data) provide at least minimum estimates of where key species are found and how many individuals of those species have been caught. More information on the distribution and abundance of key fish species can be found in Appendices A and B. See Section 7.2.5 for additional information on the interpretation of available fish distribution data.

5.2.2.1 Salmon

Most (63%) of the subwatersheds in the Nushagak and Kvichak River watersheds are documented to contain at least one species of spawning or rearing salmon within their boundaries, and 12% are documented to contain all five species (Figure 5-3). Reported distributions for each salmon species in the Nushagak and Kvichak River watersheds are shown in Figures 5-4 through 5-8.

Sockeye is by far the most abundant salmon species in the Bristol Bay watershed (Table 5-3) (Salomone et al. 2011). Bristol Bay is home to the largest sockeye salmon fishery in the world, with 46% of the average global abundance of wild sockeye salmon between 1956 and 2005 (Figure 5-9A) (Ruggerone et al. 2010). Between 1990 and 2009, the average annual 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 period averaged 25.7 million fish (Table 5-3), and 78% of the average annual subsistence salmon harvest (140,767 salmon) over this period were sockeye (Dye and Schwanke 2009, Salomone et al. 2011). Escapement goals—that is, the number of individuals allowed to escape the fishery and spawn, to ensure long-term sustainability of the stock—vary by species and stock. The current sockeye escapement goal for the Kvichak River ranged from 2 to 10 million fish (Box 5-4). Annual sport harvest of sockeye in recent years has ranged from approximately 8,000 to 23,000 fish (Dye and Schwanke 2009).

More than half of the Bristol Bay watershed's sockeye salmon harvest comes from the Nushagak and Kvichak River watersheds (Figure 5-9B). Sockeye returns to the Kvichak River averaged 10.5 million fish between 1963 and 2011, and this number climbs to 12.1 million fish when returns to the Alagnak River are included (Cunningham et al. 2012). Kvichak River sockeye runs have exceeded 30 million fish three times since 1956, with 48.6, 34.9, and 37.9 million fish in 1965, 1970, and 1980, respectively (Cunningham et al. 2012).

Tributaries to Iliamna Lake, Lake Clark, and the Wood-Tikchik Lakes (Figure 2-4) are major sockeye spawning areas, and juveniles rear in each of these lakes (Figure 5-4). Iliamna Lake provides the majority of sockeye rearing habitat in the Kvichak River watershed, and historically has produced more sockeye than any other lake in the Bristol Bay region (Fair et al. 2012). Riverine sockeye populations spawn and rear throughout the Nushagak River watershed (Figure 5-4).

Table 5-3. Mean annual commercial harvest (number of fish) by Pacific salmon species and Bristol Bay fishing district, 1990 to 2009[a]. Number in parentheses indicates percentage of total found in each district.

Salmon Species	Bristol Bay Fishing District					
	Naknek-Kvichak[a]	Egegik	Ugashik	Nushagak[a]	Togiak	Total
Sockeye	8,238,895 (32)	8,835,094 (34)	2,664,738 (11)	5,478,820 (21)	514,970 (2)	25,732,517
Chinook	2,816 (4)	849 (1)	1,402 (2)	52,624 (80)	8,803 (13)	66,494
Coho	4,436 (5)	27,433 (33)	10,425 (12)	27,754 (33)	14,234 (17)	84,282
Chum	184,399 (19)	78,183 (8)	70,240 (7)	493,574 (50)	158,879 (16)	985,275
Pink[b]	73,661 (43)	1,489 (1)	138 (<1)	50,448 (30)	43,446 (26)	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.
Source: Appendix A, Table 1.

Chinook salmon spawn and rear throughout the Nushagak River watershed and in several tributaries of the Kvichak River (Figure 5-5), and they are an important subsistence food for residents of both watersheds. Although Chinook is the least common salmon species across the Bristol Bay region, the Nushagak River watershed 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 5-3). 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. This frequently places the Nushagak at or near the size of the world's largest Chinook runs, which is notable given the Nushagak River's small watershed area compared to other Chinook-producing rivers such as the Yukon River, which spans Alaska and much of northwestern Canada, and the Kuskokwim River in southwestern Alaska, just north of Bristol Bay.

Coho salmon spawn and rear in many stream reaches throughout the Nushagak and lower Kvichak River watersheds (Figure 5-6). Juveniles distribute widely into headwater streams, where they are often the only salmon species present (Woody and O'Neal 2010, King et al. 2012). Production of juvenile coho is often limited by the extent and quality of available overwintering habitats (Nickelson et al. 1992, Solazzi et al. 2000).

Chum salmon is the second most abundant salmon species in the Nushagak and Kvichak River watersheds (Table 5-3). Both chum and pink salmon spawn throughout the Nushagak and lower Kvichak River watersheds (Figures 5-7 and 5-8), but do not have an extended freshwater rearing stage.

BOX 5-4. COMMERCIAL FISHERIES MANAGEMENT IN THE BRISTOL BAY WATERSHED

Commercial fisheries management in Alaska is largely focused on achieving escapement goals—management goals based on the optimum range of fish numbers allowed to escape the fishery and spawn—rather than harvest rates (Fair et al. 2012). Thus, management involves allowing an adequate number of spawners to reach each river system while maximizing harvest in the commercial fishery (Salomone et al. 2011). Bristol Bay's commercial salmon fisheries are considered a management success (Hilborn et al. 2003, Hilborn 2006). Several factors have contributed to this success, including a clear management objective of maximum sustainable yield, the escapement goal system, management responsibility falling to a single agency, a permit system that limits the number of fishers, and favorable freshwater habitats and ocean conditions (Hilborn et al. 2003, Hilborn 2006).

Escapement goals for sockeye salmon in the nine major rivers draining the Bristol Bay watershed are listed in the table below. The Alaska Department of Fish and Game (ADF&G) regularly reviews escapement goals for the major salmon stocks in Bristol Bay. These reviews include updates to escapement estimates, revisions to how catch is partitioned to stocks, and revisions to stock-recruit models used to recommend escapement goals. For example, data on sockeye genetic stock composition, age composition, and run timing were used to reconstruct brood tables for the major stocks in 2012 (Cunningham et al. 2012, Fair et al. 2012).

The Kvichak River frequently did not meet its sockeye escapement goal from 1991 through 1999, and in 2001 it was placed into special management status due to chronic low yields (Fair 2003). The cause of this low productivity in Kvichak River sockeye is not entirely known, but marine conditions likely led to this decline (see Appendix A, pages 31–33, for a more detailed discussion of this decline). However, the Kvichak River stock is considered to be rebuilding: escapement goals have been met for the last 5 years, and in 2012 ADF&G recommended that it be removed from special management status (Morstad and Brazil 2012).

Sockeye Salmon Escapement Goals in the Bristol Bay Watershed

River	Escapement Range (thousands of fish)
Kvichak	2,000–10,000
Alagnak	320 minimum
Naknek	800–1,400
Egegik	800–1,400
Ugashik	500–1,200
Wood	700–1,500
Igushik	150–300
Nushagak-Mulchatna	370–840
Togiak	120–270

Once escapement goals are set, the timing and duration of commercial fishery openings are adjusted throughout the fishing season to ensure that escapement goals are met and any additional fish are harvested. Fishery openings are based on information from a number of sources, including pre-season forecasts (expected returns of the dominant age classes in a given river system, based on the number of spawning adults that produced each age class); the test fishery at Port Moller on the Alaska Peninsula; early performance of the commercial fishery; and in-river escapement monitoring. At the beginning of the fishing season, the frequency and duration of openings are primarily based on pre-season forecasts and are managed conservatively. As the season progresses and additional information becomes available, fishing times and areas are continuously adjusted via emergency orders. If the escapement goal is exceeded at a given monitoring station, the fishery is opened longer and more frequently. If the escapement goal is not reached, the fishery is closed.

This type of in-season management is also used to meet a Chinook salmon escapement goal for the Nushagak River (55,000–120,000 fish). There is a chum salmon escapement goal for the Nushagak River (200,000 fish minimum) and there are Chinook salmon escapement goals for the Alagnak and Naknek Rivers; however, in-season management is not used to help attain these goals (Baker et al. 2009).

See Appendix A for a more detailed discussion of historical and current fisheries management in the Bristol Bay region.

Figure 5-3. Diversity of Pacific salmon species production in the Nushagak and Kvichak River watersheds. Counts of salmon species (sockeye, Chinook, coho, pink, and chum) spawning and rearing, based on the Anadromous Waters Catalog (Johnson and Blanche 2012), are summed by 12-digit hydrologic unit codes. See Section 7.2.5 for details on interpretation of distribution data.

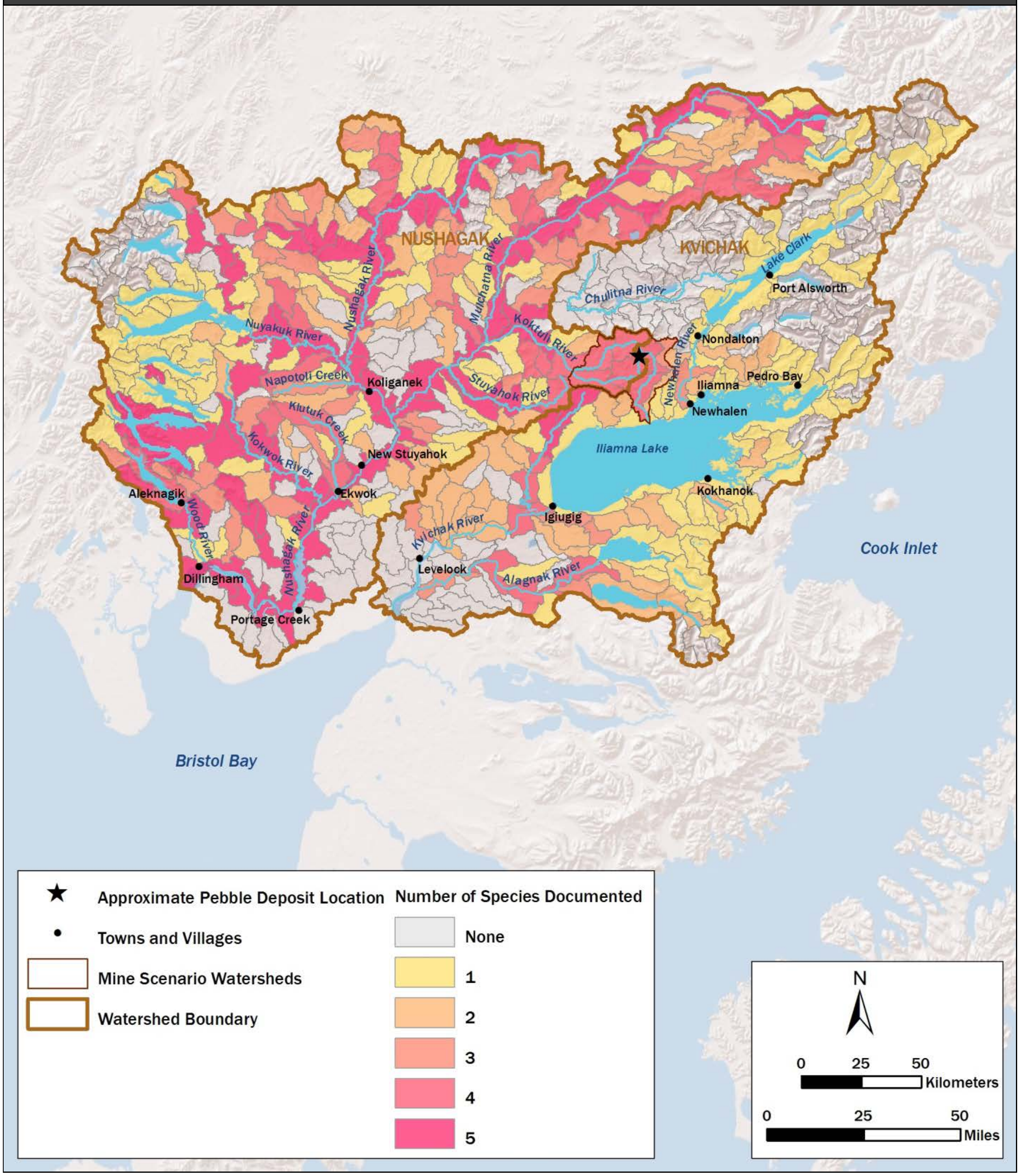


Figure 5-4. Reported sockeye salmon stream distribution in the Nushagak and Kvichak River watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year. See Section 7.2.5 for details on interpretation of fish distribution data.

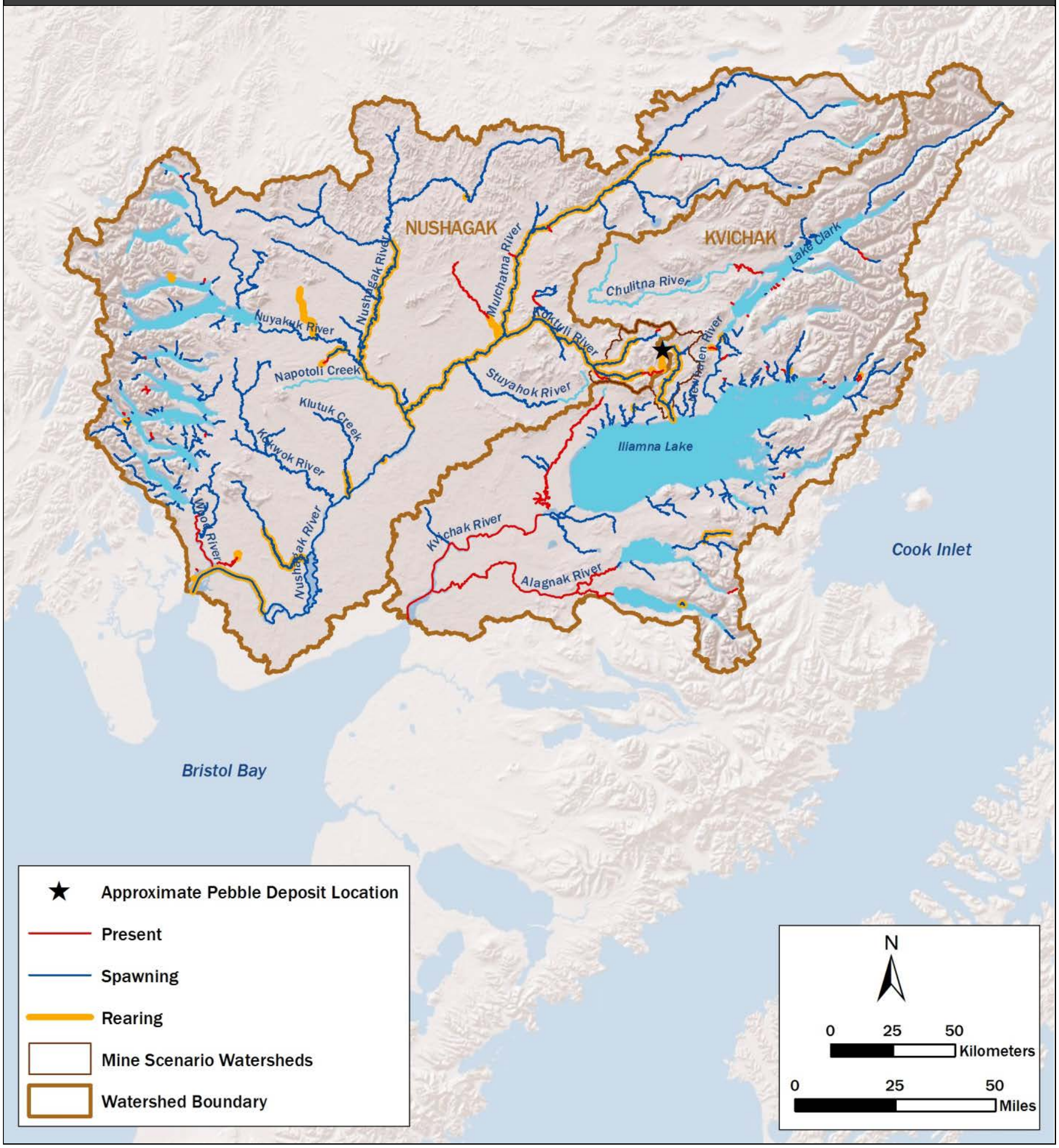


Figure 5-5. Reported Chinook salmon distribution in the Nushagak and Kvichak River watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year. See Section 7.2.5 for details on interpretation of fish distribution data.

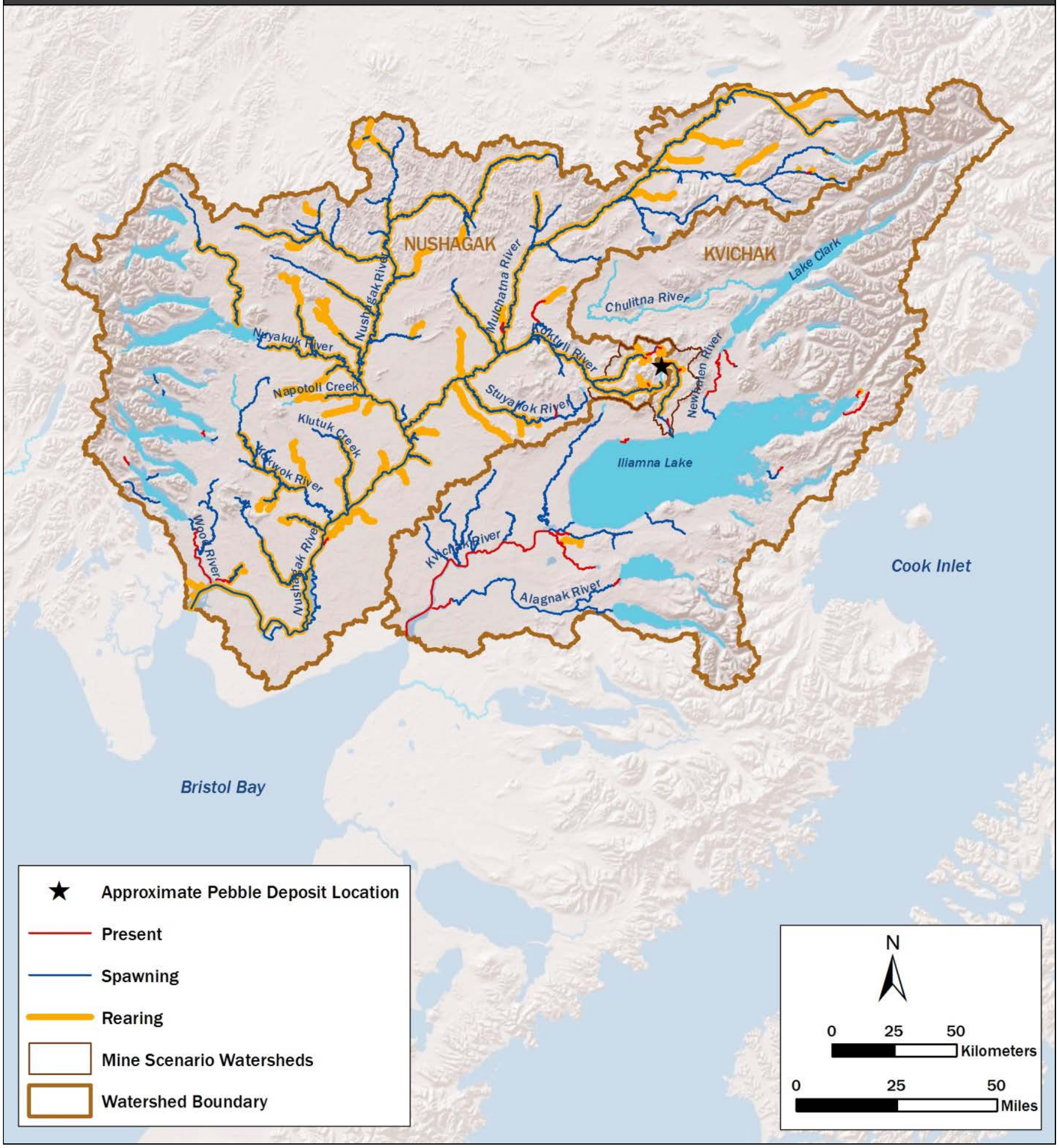


Figure 5-6. Reported coho salmon distribution in the Nushagak and Kvichak River watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year. See Section 7.2.5 for details on interpretation of fish distribution data.

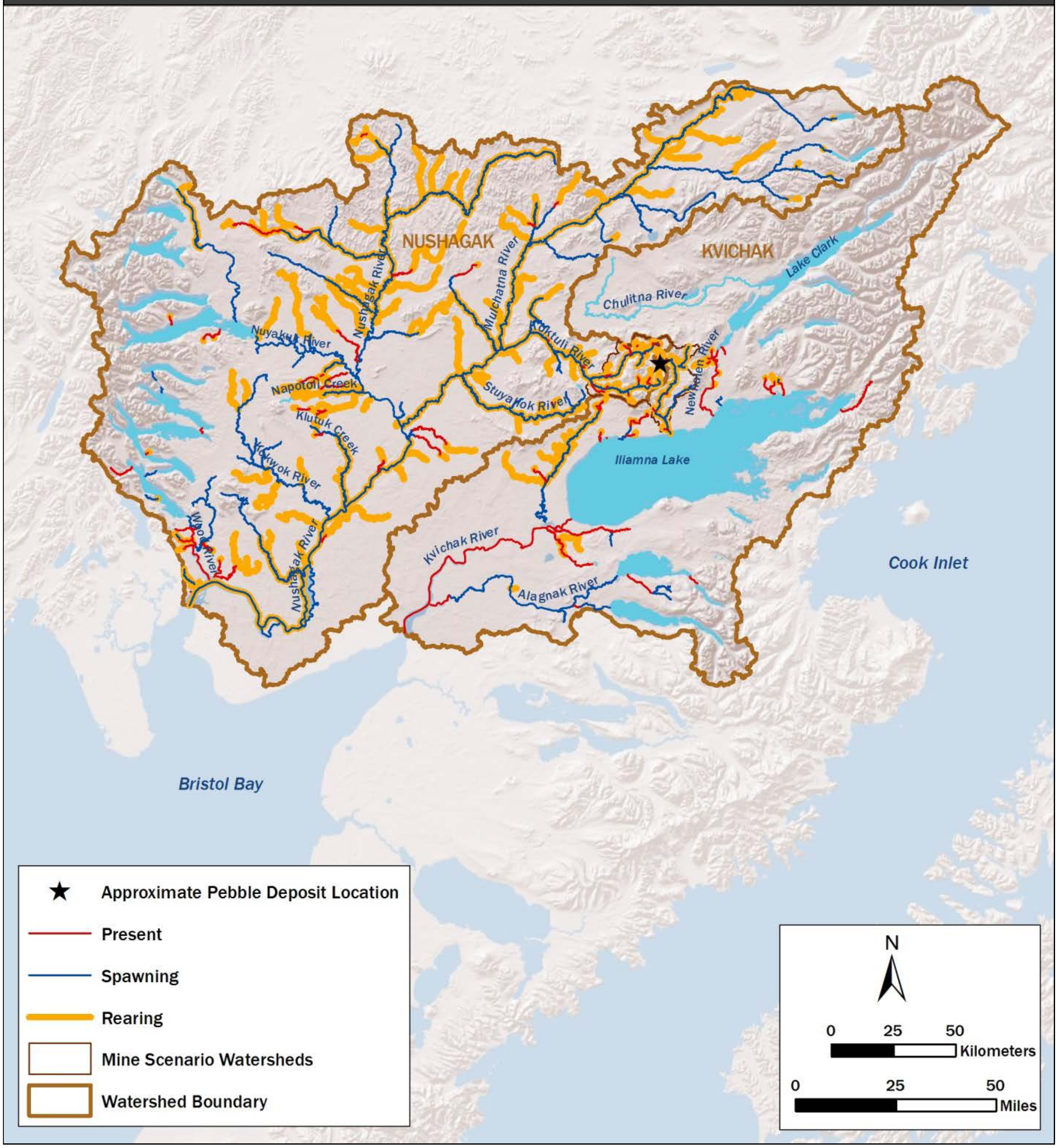


Figure 5-7. Reported chum salmon distribution in the Nushagak and Kvichak River watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year. See Section 7.2.5 for details on interpretation of fish distribution data.

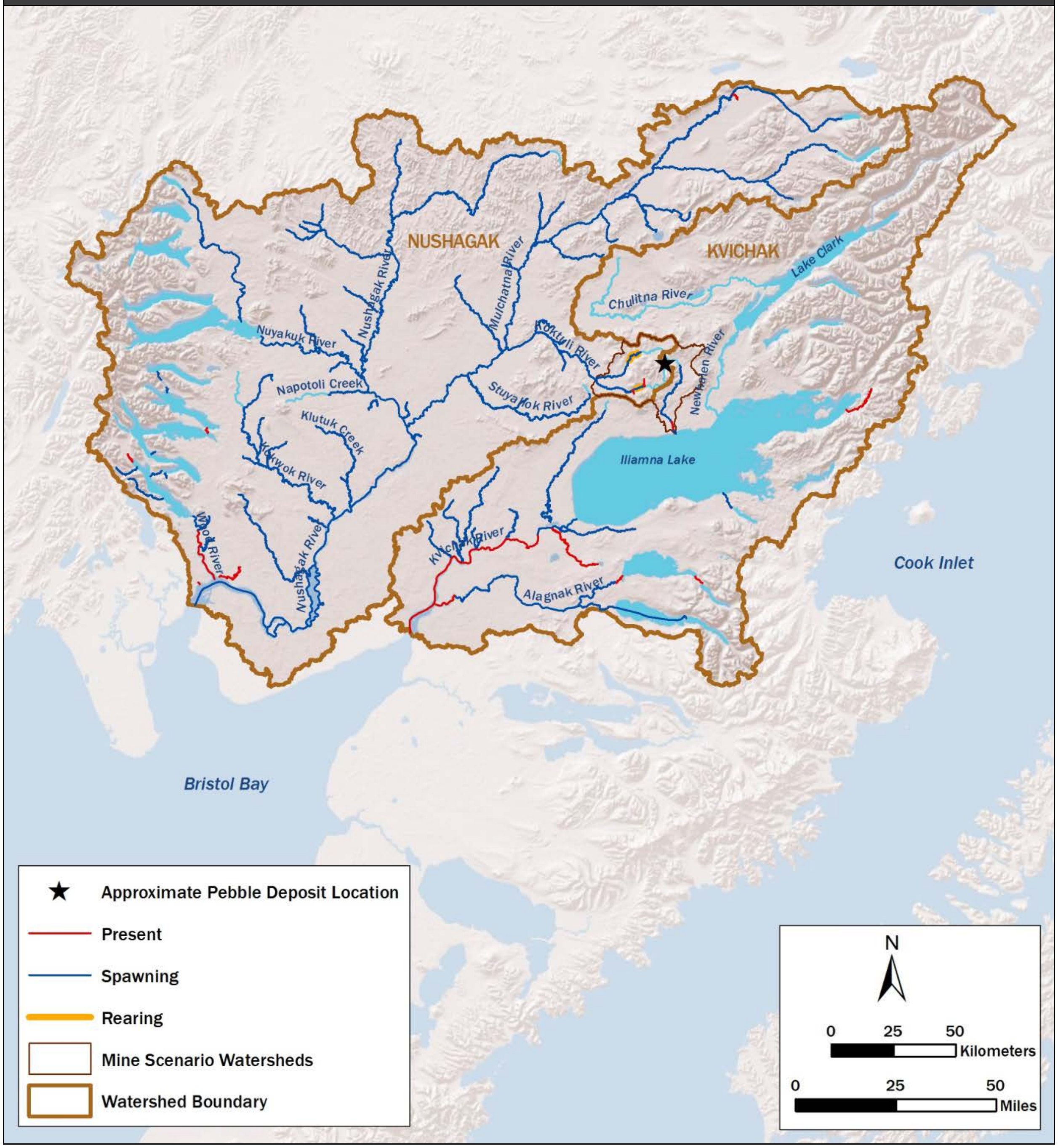


Figure 5-8. Reported pink salmon distribution in the Nushagak and Kvichak River watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed. Present and spawning designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year. See Section 7.2.5 for details on interpretation of distribution data.

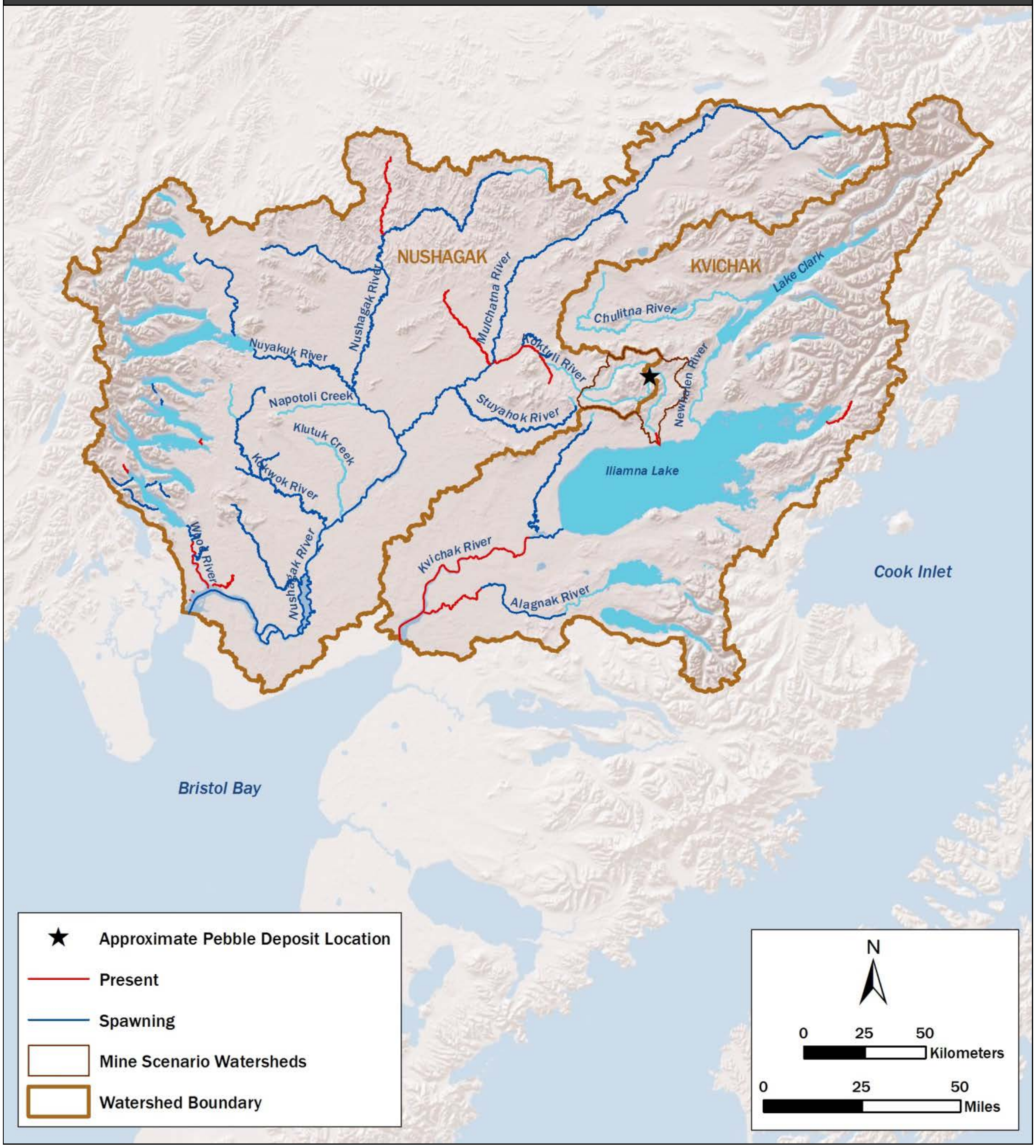
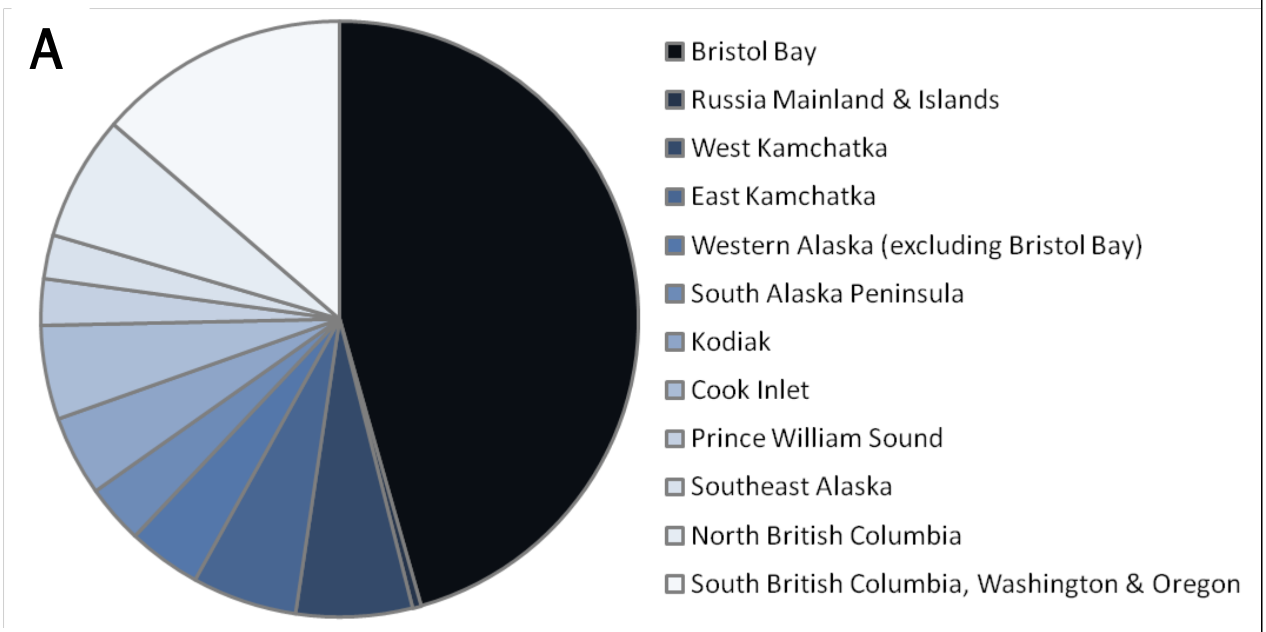
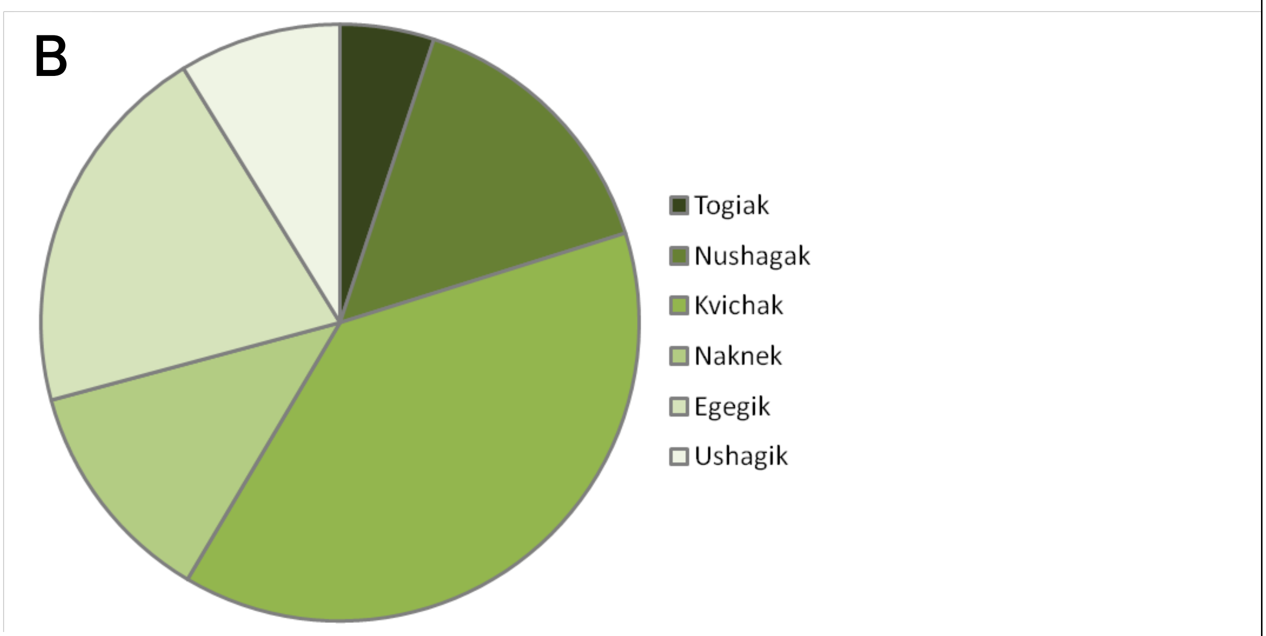


Figure 5-9. Proportion of total sockeye salmon run sizes by (A) region and (B) watershed within the Bristol Bay region. Values are averages from (A) 1956–2005 from Ruggerone et al. 2010 and (B) 1956–2010 from Baker pers. comm. (Appendix A: Tables A2 and A3).





5.2.2.2 Other Fishes

Extensive sampling for rainbow trout and Dolly Varden has not been conducted throughout the Bristol Bay region, so total distributions and abundances are unknown. Figures 5-10 and 5-11 show the reported occurrence of rainbow trout and Dolly Varden throughout the Nushagak and Kvichak River watersheds and provide minimum estimates of their extents.

Between 2003 and 2007, an estimated 183,000 rainbow trout were caught in the Bristol Bay Management Area (Dye and Schwanke 2009). Radio telemetry, tagging, and genetic studies indicate that multiple rainbow trout populations are found within Bristol Bay watersheds (Gwartney 1985, Burger and Gwartney 1986, Minard et al. 1992, Krueger et al. 1999, Meka et al. 2003). The most popular rainbow trout fisheries are found in the Kvichak River watershed, the Naknek River watershed, portions of the Nushagak and Mulchatna River watersheds, and streams of the Wood River lakes system (Dye and Schwanke 2009).

Dolly Varden populations are a significant subsistence resource. In the mid-2000s, subsistence harvests of Dolly Varden and Arctic char combined (Alaska's fisheries statistics do not distinguish between the two species) were estimated at 3,450 fish for 10 communities in the Nushagak and Kvichak River watersheds (Fall et al. 2006, Krieg et al. 2009). From the mid-1970s to the mid-2000s, these two species were estimated to represent between 16.2 and 26.9% of the total weight of the Kvichak River watershed's non-salmon freshwater fish subsistence harvest (Krieg et al. 2005). Dolly Varden also support a popular sport fishery.

Figure 5-10. Reported rainbow trout occurrence in the Nushagak and Kvichak River watersheds. Designation of species presence is based on the Alaska Freshwater Fish Inventory (AFFI point data, ADF&G 2012). Note that points shown on land actually occur in smaller streams not shown on this map. Absence cannot be inferred from this map. See Section 7.2.5 for details on interpretation of fish distribution data.

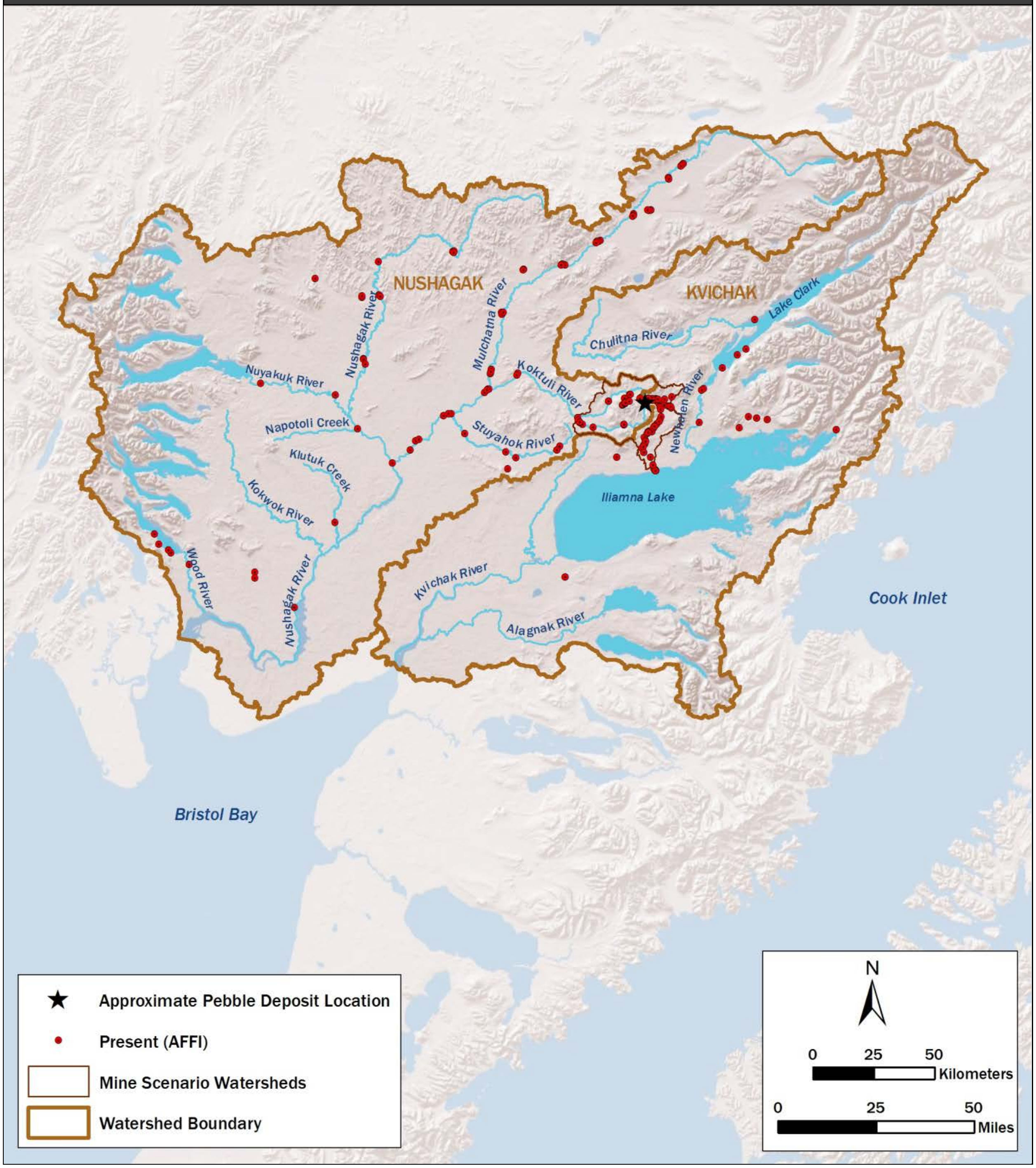
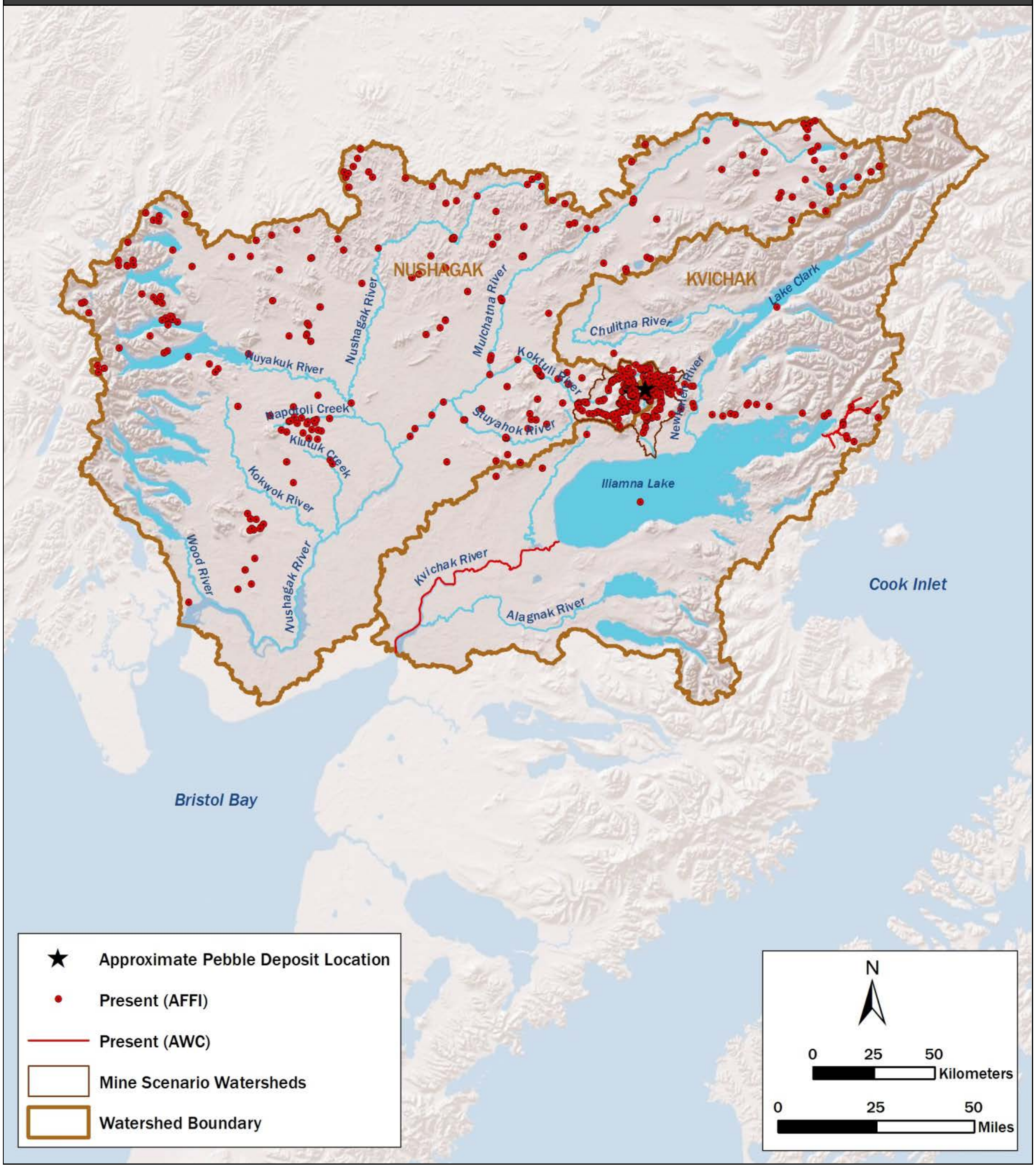


Figure 5-11. Reported Dolly Varden occurrence in the Nushagak and Kvichak River watersheds. Designation of species presence is based on the Alaska Freshwater Fish Inventory (AFFI point data, ADF&G 2012) and the Anadromous Waters Catalog (AWC line data, Johnson and Blanche 2012). Note that points shown on land actually occur in smaller streams not shown on this map. Absence cannot be inferred from this map. See Section 7.2.5 for details on interpretation of fish distribution data.



5.2.3 Economic Implications

The Bristol Bay watershed supports several sustainable, wilderness-compatible economic sectors, including commercial fishing, sport fishing, subsistence hunting and 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 roughly $480 million (in 2009 dollars) in total direct annual economic benefit (Table 5-4).

Table 5-4. 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**

Roughly 75% of this annual economic benefit results directly from the commercial, sport, and subsistence fishing supported by the Bristol Bay watershed. The 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, fishers 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 5-4, 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 (Box 5-4). 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, pristine wilderness setting of the Bristol Bay watershed attracts recreational fishers, and aesthetic qualities are rated as most important in selecting fishing locations by Bristol Bay anglers. Sport fishing in Bristol Bay accounts for approximately $60.5 million in annual spending (Table 5-4), $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 and Kvichak River watersheds alone, down from a peak of 92 businesses and 426 guides in 2008 (Appendix A, Table 4).

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 foods 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, whereas 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 5-4). It is important to note that these estimates of expenditures reflect only the annual economic activity generated by these activities and not the value of the subsistence resources harvested. 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.

5.2.4 Biological Complexity and the Portfolio Effect

As the previous sections illustrate, the Bristol Bay watershed supports world-class salmon fisheries. These fisheries result from numerous, interrelated factors. Closely tied to the Bristol Bay region's physical habitat complexity (Chapter 3) is its biological complexity, which greatly increases the region's ecological productivity and stability. This biological complexity operates at multiple scales and across multiple species, but it is especially evident in the watershed's Pacific salmon populations. As discussed in Section 5.2.1.1, the five Pacific salmon species found in the Bristol Bay watershed vary in many life-history characteristics (Table 5-5). This variability allows them to fully exploit the range of habitats available throughout the watershed. 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 1 to 4 years feeding at sea, before returning to the Bristol Bay watershed anytime within a 4-month window (Table 5-5).

Table 5-5. Life-history variation within Bristol Bay sockeye salmon populations.

Element of Biological Complexity	Range of Traits or Options
Location within the Bristol Bay watershed	7 major subwatersheds, ranging from maritime-influenced systems on the Alaska Peninsula to more continental systems
Time of adult return to freshwater habitats	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

Notes:
Data from Hilborn et al. 2003.

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). In the Bristol Bay region, hydrologically diverse riverine and wetland landscapes provide a variety of large river, small stream, floodplain, pond, and lake habitats for salmon spawning and rearing, and environmental conditions can differ among habitats in close proximity. Variations in temperature and streamflow associated with seasonality and groundwater–surface water interactions create a habitat mosaic that supports a range of spawning times across the watersheds. Spawning adults return at different times and 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). These distinct stocks can occur at fine spatial scales, with sockeye salmon that use spring-fed ponds and streams approximately 1 km apart exhibiting differences in spawn timing, spawn site fidelity, productivity, and other traits that are consistent with discrete populations (Quinn et al. 2012).

Thus, the Bristol Bay watershed's sockeye salmon "population" is actually a sockeye salmon stock complex—that is, 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 can be likened to a financial portfolio in which assets are divided among diverse investments to increase financial stability. Essentially, it creates a biological portfolio effect (Schindler et al. 2010), stabilizing 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-streamflow 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 biological complexity, annual variability in the size of Bristol Bay's sockeye salmon runs would be expected to more than double and fishery closures would be expected to become more frequent (Schindler et al. 2010). In other watersheds with previously robust salmon fisheries, such as the Sacramento River's Chinook fishery, losses of biological complexity 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 biological complexity of the population's stock complex.

5.2.5 Salmon and Marine-Derived Nutrients

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 (Cederholm et al. 1999, Gende et al. 2002). Because approximately 95 to 99% of the carbon, nitrogen, and phosphorus in an adult salmon's body are derived from the marine environment (Larkin and Slaney 1997, Schindler et al. 2005), MDN from salmon account for a significant portion of nutrient budgets in the Bristol Bay watershed (Kline et al. 1993). 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). The distribution and relative importance of the trophic subsidies provided by MDN within salmon-bearing watersheds are not expected to be spatially or temporally uniform (Janetski et al. 2009). The magnitude and density of spawning salmon and their by-products (i.e., excreta and gametes) will be highest in areas of high spawning density and where carcasses accumulate. In contrast, MDN influences on aquatic foodwebs may be negligible in headwater streams above the upstream limit of anadromous fish distributions. In these systems, other sources of energy, such as terrestrial inputs and benthic production, will be important (Wipfli and Baxter 2010).

Where salmon are abundant, 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 (Box 5-3). When and where available, salmon-derived resources—in the form of eggs, carcasses, and invertebrates that feed upon carcasses—are important dietary components for many fishes (e.g., rainbow trout, Dolly Varden, juvenile Pacific salmon, Arctic grayling). Eggs from spawning salmon are a major food source for Bristol Bay rainbow trout and are likely responsible for much of the growth attained by these fish and the abundance of trophy-sized rainbow trout in the Bristol Bay system. Upon arrival of spawning salmon in the Wood River basin, rainbow trout shifted from consuming aquatic insects to primarily salmon eggs, resulting in a five-fold increase in ration and energy intake (Scheuerell et al. 2007). With this rate of intake, a bioenergetics model predicts a 100-g trout to gain 83 g in 76 days; without the salmon-derived subsidy, the same fish was predicted to lose 5 g (Scheuerell et al. 2007). Rainbow trout in Lower Talarik Creek were significantly fatter (i.e., had a higher condition factor) in years with high salmon spawner abundance than in years with low abundance (Russell 1977). Research in Iliamna Lake suggests that between 29 and 71% of the nitrogen in juvenile sockeye salmon, and even higher proportions in other aquatic taxa, comes from marine-derived sources, and that the degree of MDN influence increases with escapement (Kline et al. 1993).

Terrestrial mammals (e.g., brown bears, wolves, foxes, minks), and birds (e.g., bald eagles, waterfowl) also benefit from these subsidies (Box 5-3) (Brna and Verbrugge 2013; this document was originally published as Appendix C of this assessment, but has since been released as a U.S. Fish and Wildlife Service [USFWS] report). 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 (Appendix A, Brna and Verbrugge 2013). Terrestrial systems of the Bristol Bay watershed also benefit from these MDN (Cederholm et al. 1999, Gende et al. 2002) (Box 5-3). Bears, wolves, and other wildlife transport carcasses and excrete wastes throughout their ranges (Darimont et al. 2003, Helfield and Naiman 2006), which then 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 may inhibit attempts to renew those runs if trophic resources are limiting those populations (Gresh et al. 2000). It is important to note that, although there is ample evidence for the significant benefits provided by trophic subsidies associated with spawning salmon in the Bristol Bay region, trophic limitations to fish population productivity should not be assumed. For example, Schindler et al. (2005) showed that MDN are indeed important for lake productivity in the Wood River system, but that interception of MDN inputs by the commercial fishery did not appear to be a driver of sockeye salmon population dynamics—likely because spawning habitat is a more limiting resource for this population.

5.2.6 Bristol Bay Fisheries in the Global Context

The Bristol Bay region is a unique environment supporting world-class fisheries, particularly in terms of Pacific salmon populations. The region takes on even greater significance when one considers the status and condition of Pacific salmon populations throughout their native geographic distributions. These declines are discussed briefly below; for additional information on threatened and endangered salmon stocks, see Appendix A (pages 37–41).

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 populations tend to be significantly reduced or dominated by hatchery fish where they do remain (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, which dominates 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.

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). Because the region's salmon resources have supported Alaska Native cultures in the region for at least 4,000 years and continue to support one of the last intact wild salmon-based cultures in the world (Appendix D), the watershed also has global cultural significance.

5.3 Endpoint 2: Wildlife

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 5.2.5). MDN provide a foundational element for the foodwebs in these watersheds and are important for many species of wildlife. Wildlife, in turn, distribute these nutrients from the aquatic to the terrestrial environment, cycling them through the entire ecosystem (Box 5-3). Thus, interactions between salmon and wildlife species are complex and reciprocal.

In this section we summarize key wildlife species in the Nushagak and Kvichak River watersheds, with particular focus on how these species are related to salmon resources. The species selected for characterization—brown bear, moose, barren-ground caribou, gray wolf, bald eagle, waterfowl (as a guild), shorebirds (as a guild), and land birds (as a guild)—are important to ecosystem function, have a direct link to salmon, and/or are important to Alaska Native and non-native residents. Within the Nushagak and Kvichak River watersheds, there are no known breeding or otherwise significant occurrences of any species listed as threatened or endangered under the Endangered Species Act, nor any designated critical habitat. For additional information on wildlife species, readers should consult Brna and Verbrugge (2013). 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.

Although this assessment focuses on inland aquatic and nearshore 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, stellar sea lions, orcas and beluga whales 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). Although this population is not evaluated in this assessment, the National Oceanic and Atmospheric Administration is currently conducting a status review on Iliamna Lake seals to determine if they represent a distinct population segment that may warrant protection under the Endangered Species Act (Appendix F).

5.3.1 Life Histories, Distributions, and Abundances of Species

5.3.1.1 Brown Bears

Brown bears are wide-ranging and feed on many different plant and animal species. They typically spend July through mid-September near streams supporting salmon runs, then move to higher elevations in the fall to feed on berries and other food items before denning in October to November. They emerge in spring and feed on vegetation and carrion, as well as moose and caribou calves. Because of their wide-ranging behavior, they distribute MDN via both deposition of salmon carcasses and excretion of wastes throughout their ranges.

Brown bear density estimates 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 (Olson and Putera 2007). From July 2006 to July 2007, 621 brown bears were reported harvested from the Alaska Department of Fish and Game's (ADF&G'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). Brown bears are reported as common in the area surrounding the Pebble deposit, with a 2009 estimated density of 18.4 to 22.5 per 1,000 km^2 (PLP 2011).

5.3.1.2 Moose

Moose habitat is determined by forage opportunities and includes both aquatic and upland areas. Alluvial habitats along the Nushagak and Mulchatna Rivers, where willows and other plants regenerate after scouring and subsequent deposit of river silt, support an abundant moose population. High-quality summer forage, especially near wetlands, is important for nursing cows and calves. It is likely that MDN contribute to increased plant productivity in these alluvial areas (Cederholm et al. 1999, Gende et al. 2002).

Moose abundance in the Nushagak 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 (ADF&G 2011), where felt-leaf willow, a preferred plant species, is abundant (Bartz and Naiman 2005). Moose were considered "low density" (0.04 moose/km^2) in the immediate area of the Pebble deposit and the transportation corridor, but there is a large variance around this estimate (PLP 2011).

5.3.1.3 Caribou

Caribou feed in open tundra, mountain, and sparsely forested areas and can travel for long distances. The Nushagak and Kvichak River watersheds are primarily used by caribou from the Mulchatna herd, one of 31 caribou herds found in Alaska. The Mulchatna herd ranges widely through the Nushagak and Kvichak River watersheds, but also spends considerable time in other watersheds. It numbered roughly 200,000 in 1997 but had decreased to roughly 30,000 by 2008 (Valkenburg et al. 2003, Woolington 2009). Recent surveys reported only a few caribou near the Pebble deposit area and potential transportation corridor (PLP 2011). However, caribou populations and ranges in the Bristol Bay region fluctuate significantly over time, and in previous years the herd was much larger and there was higher-density use of the Pebble deposit area (PLP 2011). Barren-ground caribou on the North Slope of Alaska have demonstrated avoidance of exploration activities (Fancy 1983), and some tribal Elders in the Nushagak and Kvichak River watersheds believe that mining exploration has contributed to avoidance of the Pebble deposit area (Brna and Verbrugge 2013).

5.3.1.4 Gray Wolf

Gray wolf abundance is influenced by prey abundance and availability, but populations are primarily limited by mortality caused by humans. Wolves have flexible diets and can shift to non-ungulate prey species when ungulate prey are scarce, or take advantage of seasonally abundant species such as salmon. Wolves often transport salmon away from streams for consumption or to feed pups through regurgitation.

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).

5.3.1.5 Bald Eagle

Bald eagles generally nest near riparian and beach areas and are primarily piscivorous, although they have a variable diet. Nesting bald eagles rely on salmon resources (Hansen 1987), and inland bald eagles nesting near spawning streams have higher nesting success than those with more distant nests (Gerrard et al. 1975). Birds and non-salmon fishes are also important prey for bald eagles. Salmon abundance in the Nushagak and Kvichak River watersheds affects bald eagle abundance, distribution, breeding, and behavior. Bald eagles, in turn, distribute MDN in their excretions.

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 (Brna and Verbrugge 2013); approximately half of those nests were categorized as active. The USFWS Bald Eagle Nest Database contains approximately 230 nest records for the Nushagak and Kvichak River watersheds, with 169 of those records collected between 2003 and 2006. Raptor studies in the Pebble deposit area indicate that bald eagles were the most abundant nesting raptor (30% of all raptor nests in 2005) (PLP 2011).

5.3.1.6 Waterfowl

More than 30 species of waterfowl, including ducks (e.g., northern pintail, scaup, mallard, and green-winged teal), geese (e.g., white-fronted, Canada), swans, and sandhill cranes, regularly use the Bristol Bay region (PLP 2011). Diversity of habitat and extent of wetlands and waters provide habitat for migrants and wintering waterfowl, and the region is an important staging area for many species, including emperor geese, Pacific brant, and ducks, during spring and fall migrations.

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 (Brna and Verbrugge 2013). Salmon are used by some waterfowl as direct sources of prey and carrion, and used indirectly through invertebrates and vegetation. Of the 24 duck species in the Bristol Bay region, at least 11 prey on salmon eggs, parr, or smolts, or scavenge on salmon carcasses (Brna and Verbrugge 2013).

5.3.1.7 Shorebirds

Thirty of 41 shorebird species or subspecies that regularly occur in Alaska can be found in the Bristol Bay watershed (see Brna and Verbrugge [2013] for a summary of different shorebird surveys). Shorebirds use the Bristol Bay watershed primarily during migration and breeding. Significant areas of intertidal habitat exist at Kvichak Bay (530 km^2) and Nushagak Bay (400 km^2). Important foods include abundant intertidal invertebrates and fruits and tubers in upland areas. Shorebirds likely play an important role in the distribution of MDN to terrestrial ecosystems. Adults, young, and eggs also provide a source of food for predatory birds and terrestrial mammals. Although there is not a strong direct link between salmon and shorebirds, it is reasonable to assume that MDN contribute to the abundance of invertebrates in the intertidal zone.

The Bristol Bay/Alaska Peninsula lagoon system, which includes the Nushagak and Kvichak River deltas, is one of the most important migratory shorebird stop-over areas in Alaska. Surveys of the Pebble deposit area in 2004 to 2005 identified 14 shorebird species in the Pebble deposit area (PLP 2011).

5.3.1.8 Land Birds

Approximately 80 species of land birds, both migratory and year-round residents, breed in and adjacent to the Nushagak and Kvichak River watersheds. Land birds eat vegetation (e.g., seeds, berries), invertebrates, and vertebrates. Studies indicate that the abundance of many songbird species is related to the presence of salmon carcasses (Willson et al. 1998, Gende and Willson 2001, Christie and Reimchen 2008). Salmon carcasses provide food for aquatic invertebrate larvae, and MDN contribute to increased plant productivity (Cederholm et al. 1999, Gende et al. 2002), both important food sources for land birds. Few abundance studies have focused on the Nushagak and Kvichak River watersheds, but 2004 to 2005 surveys identified 28 land bird species in the Pebble deposit area (PLP 2011).

5.3.2 Recreational and Subsistence Activities

Many of the species discussed in the preceding sections are important subsistence resources. For example, a 2002 survey of Bristol Bay residents found that 86% and 88% of respondents have consumed moose and caribou meat, respectively (Ballew et al. 2004). Between 1983 and 2006, moose harvest in GMU 17 increased from 127 to 380 moose per year; the upper Nushagak River watershed alone (GMU 17B) had a mean annual harvest of 149 moose (Brna and Verbrugge 2013). 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 and Kvichak River watersheds (Valkenburg et al. 2003).

Waterfowl support recreational and subsistence harvests, as well as wildlife viewing opportunities. There are no reliable estimates of recreational harvests specific to the Nushagak and Kvichak River watersheds. Subsistence harvest of waterfowl is very important in the watershed. The spring harvest provides fresh meat early in the season, after winter food supplies are depleted. Harvest data from 1995 through 2005 for the Dillingham, Nushagak River, and Iliamna subregions (Wentworth 2007, Wong and Wentworth 1999) indicate annual harvests of roughly 10,000 ducks, 2,500 to 2,900 geese, and up to 300 tundra swans, as well as fewer than 500 waterfowl eggs (Brna and Verbrugge 2013).

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 5-4) and directly employ over 100 full- and part-time workers.

5.4 Endpoint 3: Alaska Natives

Alaska Natives are the majority population in the Bristol Bay region, and salmon has been central to their health, welfare, and culture for thousands of years. In fact, Alaska Native cultures in the region represent one of the last intact salmon-based cultures in the world (Appendix D). Much of the region's population practices subsistence, with salmon making up a large proportion of subsistence diets—making Alaska Natives particularly vulnerable to potential changes in salmon resources.

The effect on Alaska Natives resulting from potential mining-related changes in salmon and other fishes was selected as an assessment endpoint because of the nutritional and cultural importance of salmon to Alaska Natives, and because of the U.S. Environmental Protection Agency's (USEPA's) responsibilities to work with federally recognized tribes on a government-to-government basis to protect, restore, and preserve the environment. These responsibilities are set forth in Executive Order 13175, Executive Order 12898, President Obama's 2009 Indian Policy, former USEPA Administrator Jackson's Reaffirmation of USEPA's Indian Policy 2009, USEPA's Policy on Tribal Consultation and Coordination, and USEPA's Region 10 Tribal Consultation and Coordination Procedures. Nine Bristol Bay federally recognized tribes and other tribal organizations petitioned the USEPA in 2010, requesting that the agency use its authority under the Clean Water Act Section 404(c) to restrict or prohibit the disposal of dredged or fill material associated with large-scale mining activities in the Bristol Bay watershed.

5.4.1 Alaska Native Populations

There are 31 Alaska Native villages in the wider Bristol Bay region, 25 of which are located in the Bristol Bay watershed. Fourteen of these communities are within the Nushagak and Kvichak River watersheds, with a total population of 4,337 in 2010 (U.S. Census Bureau 2010). Dillingham (population 2,329) is the largest community; other communities range in size from two (year-round) residents (Portage Creek) to 510 residents (New Stuyahok). Because population in some communities is seasonal, these numbers increase during the subsistence fishing season. Thirteen of these 14 villages—all but Port Alsworth—have federally recognized tribal governments and had an Alaska Native population majority in 2010.

Overall population in the region grew 55% from 1980 to 2000, and remained relatively stable from 2000 to 2010 (U.S. Census Bureau 2010). Population has fluctuated in individual villages since 1980 (Appendix D, Table 2). From 2000 to 2010, nine villages decreased and five villages increased in population. The extent to which these changes reflect natural population fluctuations or whether any gains or losses indicate a long-term trend is unknown. Four of the villages that decreased in population (Dillingham, Igiugig, Aleknagik, and Kokhanok) and one of villages that increased in population (Iliamna) changed less than 10%. Port Alsworth has experienced steady population growth since 1980. Its economy is more closely tied to Lake Clark National Park, and its population contains the smallest proportion of Alaska Natives among the 14 villages. Portage Creek is the smallest village in the region, and its year-round population has fluctuated significantly over the past 40 years (e.g., 48 in 1980, 5 in 1990, 36 in 2000, 2 in 2010), making it difficult to draw conclusions about trends.

5.4.2 Subsistence and Alaska Native Cultures

5.4.2.1 Importance of Salmon to Alaska Native Cultures

The primary Alaska Native cultures present in the Nushagak and Kvichak River watersheds—the Yup'ik and Dena'ina (Box 5-1)—are part of the last intact, sustainable salmon-based cultures in the United States (Appendix D). This is especially significant as other Pacific Northwest salmon-based cultures struggle with degraded resources (Colombi and Brooks 2012). Cultures associated with salmon fishing appeared in these watersheds as early as 4,000 before present (BP) and intensified around 1,000 BP (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, Table 2). The Yup'ik and Dena'ina cultures still provide the framework and values for everyday life in the region. Among the Yup'ik, over 40% of the population continues to maintain their native language, one of the highest percentages among native cultures in the United States (Appendix D).

In the Bristol Bay region, the subsistence way of life is irreplaceable. Subsistence resources provide high quality foods, foster a healthy lifestyle, and form the basis for social relations for both Alaska Natives and non-Alaska Natives in the villages. These resources, particularly salmon, are integral to the entire way of life in Yup'ik and Dena'ina cultures. The Alaska Federation of Natives (2010) describes subsistence as follows.

> The hunting, fishing, and gathering activities which traditionally constituted the economic base of life for Alaska's Native peoples and which continue to flourish in many areas of the state today…Subsistence is a way of life in rural Alaska that is vital to the preservation of communities, tribal cultures, and economies. Subsistence resources have great nutritional, economical, cultural, and spiritual importance in the lives of rural Alaskans…Subsistence, being integral to our worldview and among the strongest remaining ties to our ancient cultures, are as much spiritual and cultural as it is physical.

For Alaska Natives today, subsistence is more than the harvesting, processing, sharing, and trading of land and sea mammals, fish, and plants. Subsistence holistically subsumes the cultural, social, and spiritual values that are the essence of Alaska Native cultures. There is a strong tradition and practice of sharing and trading subsistence resources. Food is shared with tribal Elders, family living outside of the watershed, and others who may not be able to fully participate in subsistence (Appendix D). This practice was confirmed by tribal Elders interviewed for Appendix D and those who testified at public meetings on the May 2012 draft of the assessment (Box 5-5).

Cultural and personal identity largely revolve around traditional cultural practices such as hunting, fishing, and gathering of wild food resources—that is, subsistence. Tribal Elders and culture bearers continue to instruct young people, particularly at fish camps where cultural values as well as fishing and fish processing techniques are shared. The social system that forms the backbone of the culture, by nurturing the young, supporting the producers, and caring for the tribal Elders, is based on the virtue of sharing wild foods harvested from the land and waters. Sharing networks extend to family members living far from home. The first salmon catch of the year is recognized with a prayer of thanks and shared in a continuation of the ancient First Salmon Ceremony (Appendix D), when those who have caught the first Chinook (king) salmon in the spring share them with tribal Elders and all those in need, as well as friends and family.

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, fish camp, gender and age roles, and the perception of wealth. Although a small minority of tribal Elders and culture bearers interviewed expressed a desire to increase market economy opportunities (including large-scale mining), most equated wealth with stored and shared subsistence foods (Appendix D). In interviews conducted for Appendix D, the Yup'ik and Dena'ina communities of the Nushagak and Kvichak River watersheds consistently define a "wealthy person" as one with food in the freezer, a large extended family, and the freedom to pursue a subsistence way of life in the manner of their ancestors. Their ability to continue their reliance on subsistence and their concept of wealth have contributed to the maintenance of vital and viable cultures for at least 4,000 years.

BOX 5-5. TESTIMONY ON THE IMPORTANCE OF SUBSISTENCE USE

The USEPA held a series of public meetings to collect input on the May 2012 draft of the assessment. Many Alaska Natives, including tribal Elders and other tribal leaders, provided testimony on the importance of salmon and the subsistence way of life to Alaska Native cultures in the region. The following are selected quotes representative of this testimony; complete public meeting transcripts are available at www.epa.gov/BristolBay.

- "Our subsistence way of life plays a substantial role in our health both spiritually and physically."
- "From traditional knowledge we keep our culture going. My subsistence life is with my family, which consists of four boys and my wife. I also help my grandmother, grandfather, mother, father, and our other family members. I hold a Bristol Bay drift permit, my family fishes with me both commercially and subsistence. My family processes approximately 4,000 pounds of salmon, kings, reds, silvers, etc. We start when the fish first come into the river, all the way to the very end. My family and I smoke, dry, and freeze the salmon. I brought you some canned salmon to share that we keep year round."
- "The king salmon is a very important part of our fishery. If you cover that portion of the king [Chinook] salmon spawning beds, it is going to make it very hard for us to maintain our culture of people who eat king every year. Is the first fish of the year, it's a very important fish for us and we can't have that huge loss."
- "Fishing is our life and our livelihood. It's what we do for healthy communities, healthy lifestyles. Going out and catching the subsistence fish, smoking these. Passing the traditional knowledge on to younger generations. You hear about how they will make you free, the fish. We have been doing this for 6,000 years and we will want to do it for 6,000 more."
- "The generations that are coming who can be fed from this resource and this land and it's a beautiful interaction and it's one that we are losing around the world. When we realize that we have lost it we strive to get it back, but it is taking a long time for this beautiful balance between human, animal and subsistence lifestyle to come about and evolve."
- "The survival of our culture directly depends on the health of our land, the fish and the wildlife. No amount of money or jobs can replace our way of life and our culture."
- "I am a Dena'ina, and Athabascan Indian. This village is my home. We are very rich people in our culture, our resources, plants, animals and salmon. They all need clean water. That includes us, the Dena'ina people of the land. But only because we are so blessed to have clean water. Salmon have been a great part of our diet for generations and will be in the future."
- "Right now we are getting excited for the kings to come up our river. For everyone works together cutting fish. To dry, salt or vacuum pack for the winter. We do not waste anything, because we fish. Around here it is gold, gold to us which we treasure. When we fill our dry rack, we go walking and help one another."
- "I've lived here for 30 years and I moved here by choice. My experience of living in this area is that people choose to be here whether born or coming here. It's a choice. It is not a scientific fact, but three reasons people choose to be in Bristol Bay is because clean water, the fishery and the lifestyle."
- "This environment has sustained our culture for thousands of years. It sustained jobs and commercial fishing for hundreds of years, and recreation and sport fishing and everything."

The Alaska Native community is also dependent on the regional economy, which is primarily driven by commercial salmon fishing and tourism. The commercial fishing and recreation market economies provide seasonal employment for many residents, giving them both the income to purchase goods and services needed for subsistence and the time to participate year-round in subsistence activities. The fishing industry provides half of all jobs in the region, followed by government (32%), recreation (15%), and mineral exploration (3%) (Appendix E). It is estimated that local Bristol Bay residents held one-third of all 2009 jobs and earned almost $78 million (28%) of the total income traceable to the Bristol Bay salmon ecosystem (Appendix E).

5.4.2.2 Use of Subsistence Resources in the Bristol Bay Watershed

Alaska Native populations, as well as non-Alaska Native residents, have continual access to a range of subsistence foods. As described by Fall et al. (2009), these subsistence resources are the most consistent and reliable component of the local economies in the Bristol Bay watershed, even given the world-renowned commercial fisheries and other recreational opportunities the region supports (Table 5-4). Subsistence uses on state lands are given priority by state law and regulations (i.e., the 1978 State of Alaska Subsistence Act). All citizens of Alaska benefit from a subsistence priority in areas specifically designated as subsistence areas by the State of Alaska. State hunting and fishing regulations apply to lands of the Alaska Native Corporations. These lands were often selected because of their significant value for subsistence activities, and Alaska Native peoples have the exclusive right to occupy and use these lands for subsistence. These rights are not recognized in the State of Alaska Constitution; however the Alaska Federation of Natives has passed resolutions for several years asking for the constitution to be revised. In addition, the Alaska Federation of Natives recommended improvements to management of state and federal subsistence programs. Indigenous hunting and fishing rights are recognized by statute only and therefore can be diminished over time. Their lack of special status makes these rights vulnerable to constitutional challenges, especially challenges based on the right to equality (Duhaime and Bernard 2008).

Virtually every household in the Nushagak and Kvichak River watersheds uses subsistence resources (Appendix D: Table 12). No watershed data are available for the proportion of Bristol Bay watershed residents' diets made up of subsistence foods, as most studies focus on harvest data and are not dietary surveys. A study that included the nearby Yukon-Kuskokwim region found that 22.8 % of calories came from Native (subsistence) foods (Johnson et al. 2009). In 2004 and 2005, annual subsistence consumption rates in the Nushagak and Kvichak River watersheds were over 300 pounds per person in many villages, and reached as high as 900 pounds per person (Appendix D, Table 12; for comparison, an average American consumes 1,996 pounds of food per year). Villages with the highest per capita subsistence usage were Koliganek, Ekwok, Newhalen, Kokhanok, Igiugig, and Levelock.

Subsistence use varies throughout the Bristol Bay watershed, as villages differ in the per capita amount of subsistence harvest and the variety of subsistence resources used. Salmon and other fishes provide the largest portion of subsistence harvests of Bristol Bay communities. On average, about 50% of the subsistence harvest by local community residents (measured in pounds usable weight) is Pacific salmon, and about 10% is other fishes (Fall et al. 2009). The percentage of salmon harvest in relation to all subsistence resources ranges from 29 to 82% in the villages (Appendix D, Table 11). Salmon accounts for an especially high percentage compared to all subsistence resources for Iliamna, Kokhanok, and Pedro Bay. Igiugig, Levelock, and New Stuyahok show the lowest percentage of salmon usage relative to other subsistence resources. Villages in the Nushagak River watershed, especially New Stuyahok, Ekwok, and Dillingham, rely on Chinook salmon to a great extent, whereas villages in the Kvichak River watershed and Iliamna Lake area (e.g., Iliamna, Kokhanok, Iguigig, Newhalen, Nondalton, Pedro Bay, and Port Alsworth) rely more on sockeye salmon. All communities also rely on non-salmon fishes (Table 5-1), but to a lesser extent than salmon. These fishes are taken throughout the year by a variety of harvest methods and fill an important seasonal component of subsistence cycles (Fall et al. 2009). For example, whitefish and other freshwater species provide fresh fish during winter ice-fishing season (Appendix D).

The ADF&G overview of subsistence fisheries in the Bristol Bay watershed (Fall et al. 2009) provides the following information.

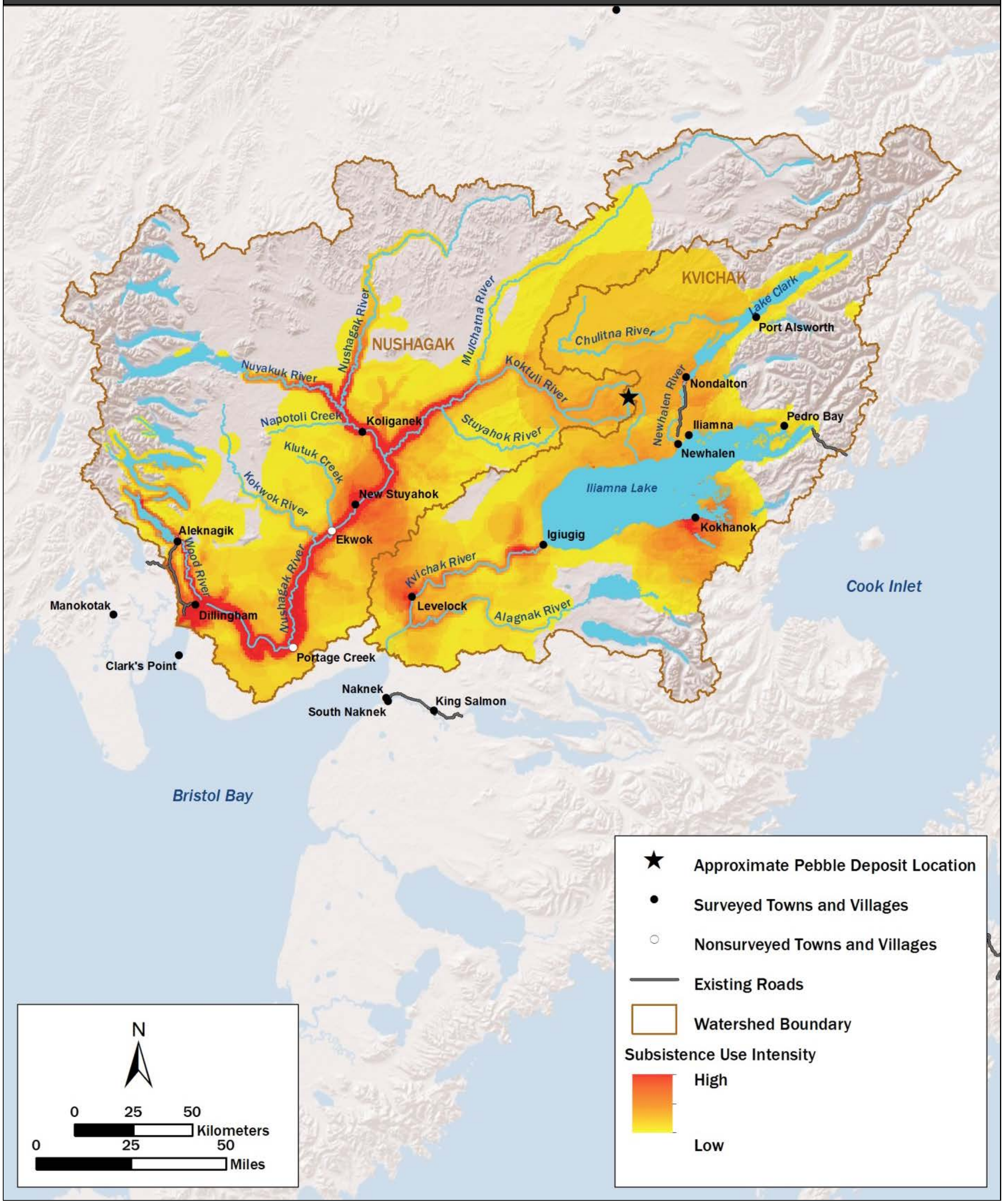
- The number of Bristol Bay subsistence salmon permits issued has been stable since 1990, and the recent 10-year average is 1,146 permits. Most permit holders (84%) are residents of Bristol Bay communities, and most permits are issued for the Nushagak and Naknek/Kvichak districts. Sockeye salmon make up the largest portion of the Bristol Bay subsistence salmon harvest (79% of the 1998–2007 average, based on subsistence salmon permits), followed by Chinook (19%), coho (5%), chum (5%), and pink (2%).
- Annual subsistence harvests for the Bristol Bay management area vary from year to year. Salmon harvest declined from the early 1990s to the early 2000s but has recovered slightly since 2002. Since 1975, the average annual harvest was about 152,371 salmon; the recent 5-year average (2003–2007) was 126,717 salmon.
- The largest decline over the last 15 years has occurred in the Kvichak River watershed subsistence sockeye salmon fishery, historically the largest component of the Bristol Bay subsistence salmon harvest. Declines are due to lower harvests per permit, rather than reduced fishing effort. Since 1996, harvest per day is down 26% in years of escapements under 2 million fish, compared to the previous 13-year average. The long-term average (45 years, for which permit data are available) for this fishery is 66,614 sockeye salmon.
- There has been an overall harvest decline in the Nushagak district from a high of 86,400 fish in 1986 to a low of 40,373 salmon in 2006. The 24-year average harvest (the time for which data are available) is 50,740 fish. However, the number of subsistence salmon permits issued in the Nushagak district has remained relatively stable since 1983.
- Subsistence salmon harvests in the Nushagak district are similar to those in the Kvichak district in terms of harvest levels. For example, in 2007the communities in the Nushagak district harvested 44,944 salmon, compared to 47,538 salmon in the Kvichak River/Iliamna Lake subdistrict, based on permit returns. However, there are differences in the two fisheries. Whereas salmon harvest in the Kvichak River watershed is almost all sockeye salmon (47,473 out of 47,538 in 2007), salmon harvest in the Nushagak district is more varied, with larger harvests of Chinook, coho, and chum salmon. There are also larger communities in the Nushagak district, including Dillingham, Manokotak, Aleknagik, New Stuyahok, and Koliganek.
- Chinook salmon returns are higher in the Nushagak River watershed than in the Kvichak River watershed. In the upper portion of the Nushagak River, residents attempt to harvest large numbers

> of Chinook salmon, their traditionally preferred salmon resource. Chinook salmon spawn early in the season, and it is important to put up these fish for subsistence before commercial fishing starts in earnest (Holen et al. 2012). Substitution of Chinook for sockeye salmon accounts for some, but not all, of the decline in the Nushagak district. Subsistence sockeye salmon harvests in the Kvichak River watershed, including Iliamna Lake and Lake Clark (historically the largest component of the Bristol Bay subsistence salmon fishery), declined by more than 50% during the 1990s and early 2000s. Local subsistence fishers attributed these lowered harvests to poor returns and scarcities of salmon in once reliable and abundant traditional harvest locations. Effort has increased in harvesting salmon in these areas since the low harvest levels seen in early 2000.

Figures 5-2 and 5-12 show areas of subsistence use identified by ADF&G in the Nushagak and Kvichak River watersheds. Clark's Point subsistence use areas overlap with Nushagak and Kvichak River watersheds for caribou, coho salmon, and moose. Clark's Point high per capita harvest rate (1,210 lbs per capita) resulted from a high harvest rate of salmon in 2008. This was three times higher than the harvest levels reported in 1973 and 1989 (Holen et al. 2012). Manokotak subsistence use areas overlap with the Nushagak communities for caribou and moose. Aleknagik moose search areas include part of Nushagak River area (Holen et al. 2012). South Naknek, Naknek, and King Salmon subsistence use areas for waterfowl, rainbow trout, unspecified trout, moose, and berry picking, as well as caribou search areas, overlap the Nushagak and particularly the Kvichak River watersheds (Holen et al. 2011). It should be noted that available subsistence data are coarse and incomplete (Box 5-2), and it is likely that subsistence activities occur outside of the areas identified on the figures. Data used to generate the figures were collected in different years, and at least one village with high recorded subsistence harvests (Ekwok) declined to be surveyed. Also note that these figures do not indicate abundance or harvest, only use.

Although subsistence is a non-market economic activity that is not officially measured, the effort put into subsistence activities is estimated to be the same or greater than full-time equivalent jobs in the cash sector (Appendix E). There is a strong and complex relationship between subsistence and the market economy (largely commercial fishing and recreation) in the area (Wolfe and Walker 1987, Krieg et al. 2007). Market economy income funds goods and services purchased by households and used for subsistence activities (e.g., boats, rifles, nets, snow mobiles, and fuel). In addition to the economic activity generated by the purchase of subsistence goods, subsistence harvests are valued at approximately $60 to $86 per pound, or 34 to 42% of the 2009 per capita income of regional residents (Appendix E).

Figure 5-12. Subsistence use intensity for salmon, other fishes, wildlife, and waterfowl within the Nushagak and Kvichak River watersheds. See Box 5-2 for more detailed discussion of methodology.



The salmon-dependent diet of the Yup'ik and Dena'ina benefits their physical and mental well-being in multiple ways, in addition to encouraging high levels of fitness based on subsistence activities. The interviews conducted for Appendix D confirm ADF&G harvest data that people of the Nushagak and Kvichak River watersheds primarily eat two species of Pacific wild salmon, sockeye and Chinook. These are consumed in different ways, including fresh, salted, pickled, canned, dried, and smoked. Salmon and other traditional wild foods comprise a large part of the people's daily diet throughout their lives, beginning as soon as they are old enough to eat solid food. (Appendix D). Subsistence foods consumed in rural Alaska have demonstrated multiple nutritional benefits, including lower cumulative risk of nutritionally mediated health problems such as 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) and provision of essential micronutrients and omega-3 fatty acids (Murphy et al. 1995, Nobmann et al. 2005, Bersamin et al. 2007, Ebbesson and Tejero 2007).

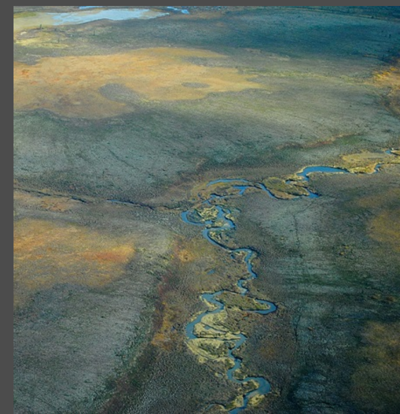
A disproportionately high amount of total diet protein and some nutrients comes from subsistence foods. For example, a 2009 study of two rural regions found that 46% of protein, 83% of vitamin D, 37% of iron, 35% of zinc, 34% of polyunsaturated fat, 90% of eicosapentaenoic acid, and 93% of docosahexaenoic acid came from subsistence foods consumed by Alaska Natives (Johnson et al. 2009).

In summary, the roles of salmon as a subsistence food source and as the basis for Alaska Native cultures are inseparable. 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, and 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 Alaska Natives' 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 serves a wide range of economic, social, and cultural functions in Yup'ik and Dena'ina societies (Appendix D). Appendix D states the following:

> … 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.

It should be noted that, even though the scope of the assessment is focused on villages in the Nushagak and Kvichak River watersheds, subsistence harvest areas do not necessarily correspond with watershed boundaries. As noted previously, villages outside of these watersheds use areas within the watersheds for subsistence activities.





CHAPTER 6. MINE SCENARIOS

6.1 Basic Elements of the Mine Scenarios

For this assessment, we used information on porphyry copper deposits and mining practices summarized in Chapter 4 to develop three mine size scenarios: Pebble 0.25 with 0.25 billion ton (0.23 billion metric tons) of ore, Pebble 2.0 with 2.0 billion tons (1.8 billion metric tons) of ore, and Pebble 6.5 with 6.5 billion tons (5.9 billion metric tons) of ore. The word Pebble in the names of the scenarios represents the fact that we place our scenarios at the Pebble deposit. These three mine size scenarios, as well as other scenario types considered in later chapters of the assessment, are summarized in Table 6-1.

The three mine size scenarios evaluated in the assessment represent realistic, plausible descriptions of potential mine development phases, consistent with current engineering practice and precedent. The scenarios are not mine plans: they are not based on a specific mine permit application and are not intended to be the detailed plans by which the components of a mine would be designed. However, the scenarios are based on preliminary mine details put forth in Northern Dynasty Minerals' *Preliminary Assessment of the Pebble Mine* (Ghaffari et al. 2011), as well as information from scientific and industry literature for mines around the world (see Chapter 4 and Appendix H for background information on mining and the geology of porphyry copper deposits). Thus, the mine scenarios reflect the general activities and processes typically associated with the kind of large-scale porphyry copper mine development likely to be proposed once a specific mine application is developed. We use these scenarios to benchmark potential risks resulting from this type of development, to provide decision makers with a better understanding of potential risks associated with any specific action proposed in the future.

Table 6-1. Summary of scenarios considered in the assessment.			
Scenario Type	**Scenario**	**Description**	**Assessment Chapter(s)**
Mine size	Pebble 0.25	Mine size of 0.25 billion ton (0.23 billion metric ton) of ore.	7, 8
	Pebble 2.0	Mine size of 2.0 billion tons (1.8 billion metric tons) of ore.	
	Pebble 6.5	Mine size of 6.5 billion tons (5.9 billion metric tons) of ore.	
Water collection, treatment, and discharge	Routine operations[a]	All water collection and treatment at site works properly, and wastewater is treated to meet state and national standards before release; however, some leachate from waste rock and TSFs is not captured.	8
	Wastewater treatment plant failure[a]	Wastewater treatment plant fails and releases untreated wastewater through its two outfalls.	
	TSF spillway release	Excess water stored in TSF 1 is released over the spillway.	
Tailings dam failure	Pebble 0.25	Failure of 92-m dam at TSF 1.	9
	Pebble 2.0	Failure of 209-m dam at TSF 1.	
Transportation corridor		113-km gravel road with four pipelines, within the Kvichak River watershed.	10
Pipeline failure	Product concentrate pipeline failure[b]	Complete break or equivalent failure of the product concentrate pipeline.	11
	Return water pipeline failure[b]	Complete break or equivalent failure of the return water pipeline.	
	Diesel pipeline failure[b]	Complete break or equivalent failure of the diesel pipeline.	

Notes:
[a] Scenario was considered for each mine size scenario.
[b] Each pipeline failure scenario was considered at two locations: Chinkelyes Creek and Knutson Creek.
TSF = tailings storage facility.

In the scenarios, we make decisions concerning mine placement; the size of the mine and the time over which mining would occur; the size, placement, and chemistry of waste rock; the size, placement, and chemistry of tailings storage facilities (TSFs); on-site processing of the ore; and the removal of processed ore concentrate from the site. For comparison purposes, Table 4-1 provides similar information for other past, existing, and potential large mines in Alaska. The mine components described in the scenarios are placed on the landscape based on information either from Ghaffari et al. (2011) or where, in our experience, modern mining practice suggests a component would be placed. For example, the pit is located on the deposit; TSFs are placed in locations described by Ghaffari et al. (2011) and where topography provides an efficient location to store a large volume of tailings; waste rock is placed around the pit to minimize the cost of hauling millions to billions of metric tons of material (Table 6-2); and the transportation system is located within the corridor described by Ghaffari et al. (2011).

We focus on the major mine components (mine pit, waste rock piles, and TSFs) that have the potential to adversely affect aquatic resources regulated under the federal Clean Water Act (33 USC 1251–1387) (Box 6-1). Smaller mine facilities such as crushing and screening areas, the mill, laydown areas, workshops, offices, and housing would be expected to be placed in uplands to avoid wetlands, ponds, and streams; thus, they are only addressed as they relate to stormwater runoff.

BOX 6-1. CUMULATIVE IMPACTS OF A LARGE-SCALE PORPHYRY COPPER MINE

In this assessment, we focus on the areas of the major mine components (mine pit, tailings storage facilities, and waste rock piles) and the transportation corridor. The actual infrastructure needed to operate any large-scale mine would be significantly more extensive than these four components and would result in larger cumulative impacts of a single mine. These additional infrastructure needs (based on Ghaffari et al. 2011) would include, but are not limited to, the following.

- **Mining and processing facilities**, including grinding mills, ore stockpiles, conveyers, a wastewater treatment plant, and process water ponds and distribution lines.
- **Drainage management structures**, such as seepage cutoff walls, stream diversion channels, drainage ditches, and sediment control ponds.
- **Other storage and disposal facilities**, such as overburden and topsoil stockpiles, explosives storage, a non-hazardous waste landfill, process water storage tanks, waste incinerators, a fuel storage compound, and hazardous waste storage.
- **Other operational infrastructure**, such as administrative buildings, dormitories, a sewage treatment plant, a power generation plant, power distribution lines, potable water treatment plant and distribution lines, and a truck shop.

These cumulative plant and ancillary areas are included in the total mine footprint for each scenario (Tables 6-5 through 6-7) but are not specifically placed on the landscape because of the greater uncertainty regarding their placement.

The cumulative impacts of a large-scale mine at the Pebble deposit likely would be much larger than the footprints evaluated in the mine scenarios.

- According to Ghaffari et al. (2011), the total area of direct impact for a 25-year mine at the Pebble deposit would cover approximately 125 km^2; in comparison, the mine footprint for the 25-year mine scenario (Pebble 2.0) considered in this assessment covers approximately 45 km^2 (Table 6-6).
- Net power generation for such a mine would be approximately 378 megawatts (Ghaffari et al. 2011). This is more than 100 times the maximum electrical load of the largest population center in the Bristol Bay watershed, the Dillingham/Aleknagik area (Marsik 2009), and slightly less than half of the combined capacity of the two electric utilities that serve more than 40% of Alaska's total population (CEA 2011, ML&P 2012).
- Dormitories for such a mine would house more than 2,000 people during construction and more than 1,000 people during mine operation (Ghaffari et al. 2011). Thus, the mine site would rival Dillingham as the largest population center in the Bristol Bay watershed during construction and would remain the second largest population center during operation.
- The mine site could contain more than 19 km of main roads, as well as numerous pit and access roads, and would depend on a fleet of 50 to 100 vehicles, in addition to 150 or more large ore-hauling trucks (Ghaffari et al. 2011). Potential risks associated with these roads would be similar in type to those described in Chapter 10.

We specify that all mine components would be developed using modern conventional design and technologies and operated under standard industry practices. Our purpose in this assessment is to evaluate the potential effects of mining porphyry copper deposits in the Nushagak and Kvichak River watersheds given design and operation to these standards. We have included basic descriptions of design features intended to mitigate potential adverse effects of mine operation.

In spite of these design and operation standards, however, any large-scale mine in the Bristol Bay region would have a footprint that would affect aquatic resources (Figures 6-1 through 6-3). These footprint-related impacts are addressed in Chapter 7. Additional impacts that may result from human error, mechanical failure, accidents, and other unplanned events are considered in Chapters 8 through 11. Compensatory mitigation for effects on aquatic resources that cannot be avoided or minimized by mine design and operation is discussed in Appendix J.

It is important to remember that this is an assessment of mine scenarios, and that like any predictive assessment it is hypothetical. Although major features of the scenarios will undoubtedly be correct (e.g., a pit at the location of the ore body and the generation of a large volume of tailings), some specifics would inevitably differ in an official mine plan submitted for permitting. All plans—even those submitted to and approved by state and federal regulators—are scenarios, and unforeseen changes in design and practice inevitably occur over the course of mine development and operation. The Fort Knox Mine near Fairbanks, Alaska, provides an example. On October 1, 2012, an Alaska Pollution Discharge Elimination System permit authorized the Fort Knox Mine to discharge wastewater to nearby Fish Creek, although the mine was originally designed and permitted in 1994 as a no-discharge facility.

It is also important to note that the largest scenario considered in this assessment, based upon 6.5 billion tons (5.9 billion metric tons) of ore, does not represent complete extraction of the Pebble deposit. Ghaffari et al. (2011) estimate the entire Pebble mineral resource at 11.9 billion tons (10.8 billion metric tons); were a mine to be developed that fully extracted this amount of ore, potential effects could be significantly greater than those estimated in the assessment.

This section describes the mine components common to the three mine size scenarios (and most other mines of this type, as described in Chapter 4). Section 6.2 describes specific characteristics of each mine size scenario relevant to our assessment, including water treatment and discharge. Section 6.3 describes closure of the mines, and Section 6.4 provides conceptual models of the relationships between mine components, potential stressors, and biotic responses.

Figure 6-1. Footprint of the major mine components (mine pit, waste rock piles, and tailings storage facility [TSF]) in the Pebble 0.25 scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012a) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012).

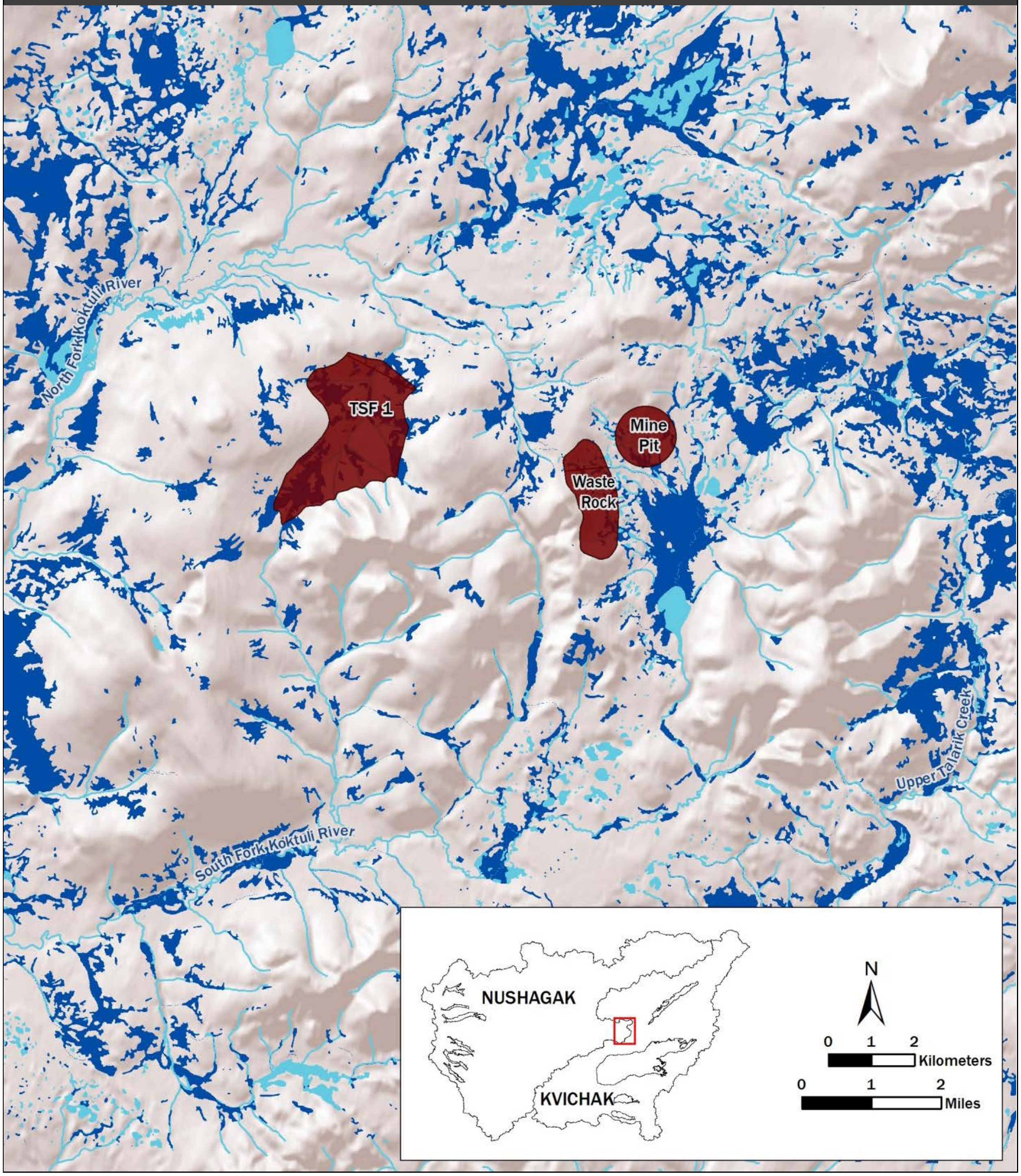


Figure 6-2. Footprint of the major mine components (mine pit, waste rock piles, and tailings storage facility [TSF]) in the Pebble 2.0 scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012a) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012).

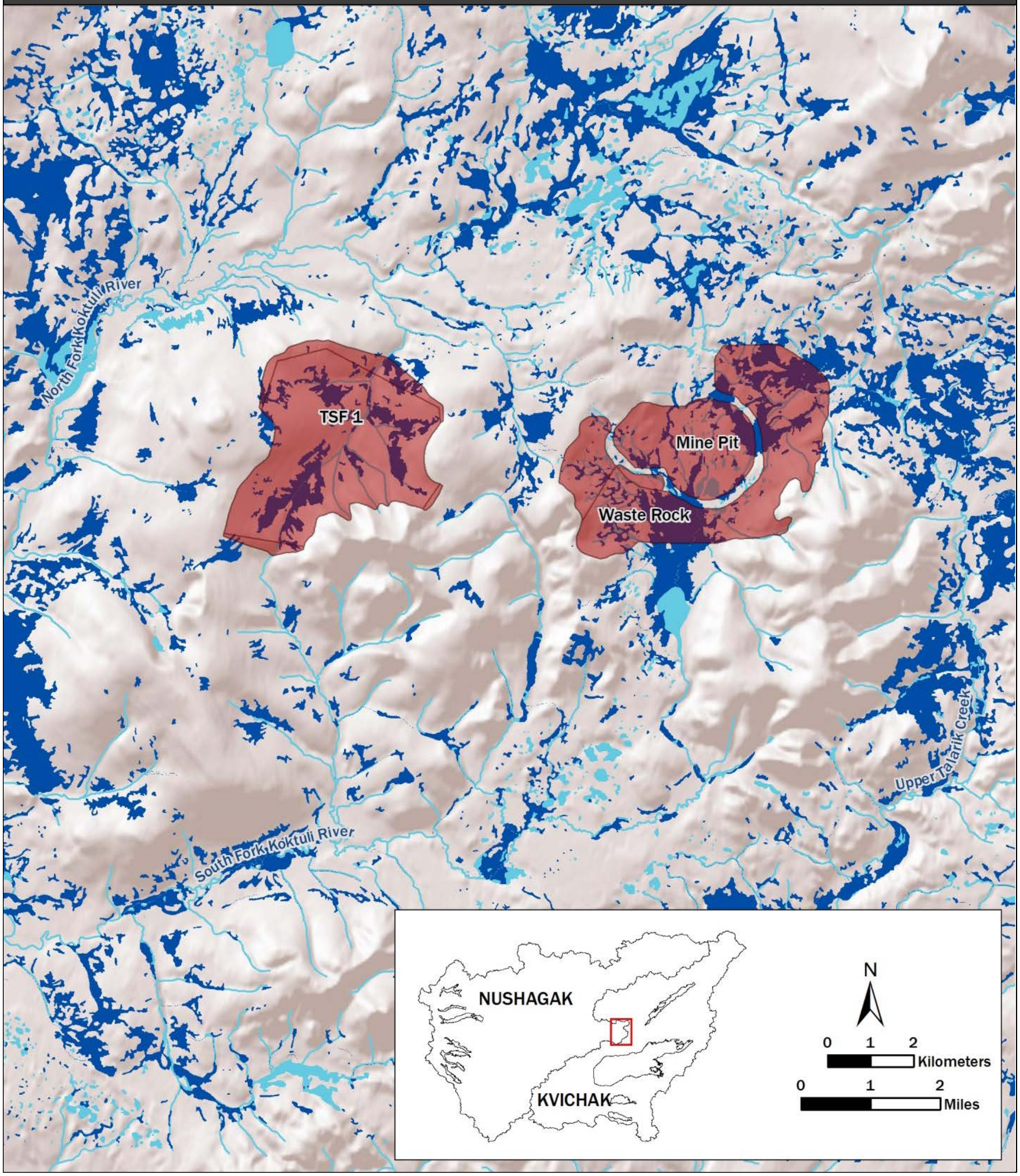
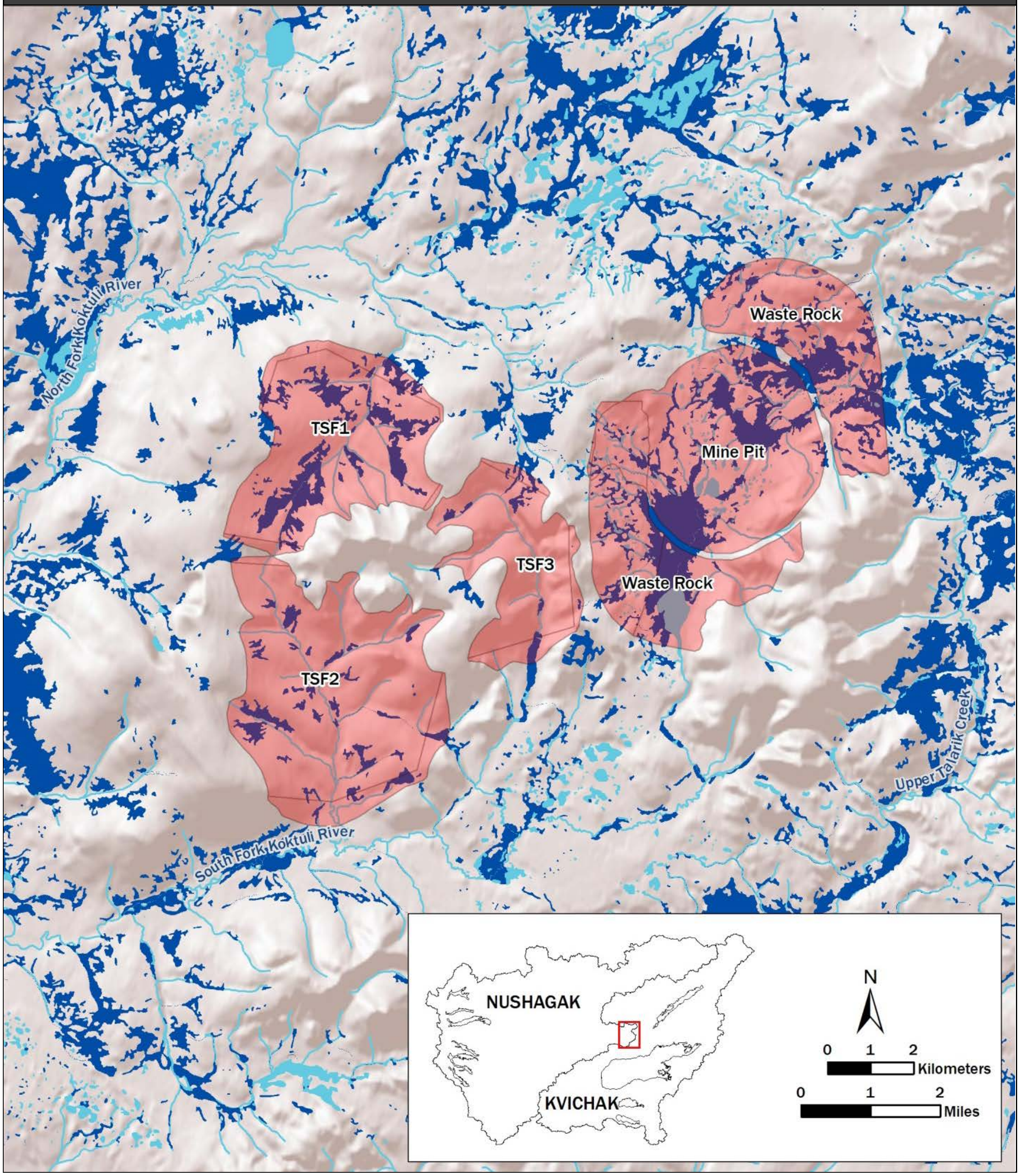


Figure 6-3. Footprint of the major mine components (mine pit, waste rock piles, and tailings storage facilities [TSFs]) in the Pebble 6.5 scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012a) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012).



6.1.1 Location

The mine scenarios considered in this assessment are sited at the Pebble deposit, in headwaters of the Nushagak and Kvichak River watersheds where the South and North Fork Koktuli Rivers and Upper Talarik Creek originate (Figure 2-5). The Pebble deposit represents the most likely site for near-term, large-scale mine development in the Bristol Bay watershed. Many other mineral exploration sites in the Nushagak and Kvichak River watersheds report findings consistent with a porphyry copper deposit similar to the Pebble deposit (see Table 13-1 and Figure 13-1 for other mineral prospects in the area). Non-porphyry copper deposits being explored in the area are likely to require similar mining facilities such as an open pit, a tailings impoundment, and waste rock dumps, and may produce acid-generating materials. Salmon and other fishes occur in streams throughout the Nushagak and Kvichak River watersheds (Chapter 5; Appendices A and B). Thus, much of our analysis is transferable to other portions of the two watersheds, in that a mining operation at any one of these sites could have qualitatively similar impacts to a mine operation at the Pebble deposit. However, we recognize that specific component placement could differ based on site-specific factors at each mine. Because our scenarios are located at the Pebble deposit, we refer to them throughout the text as Pebble 0.25, Pebble 2.0, and Pebble 6.5. This distinguishes the site of the analysis from other potential mine sites in the Nushagak and Kvichak River watersheds that are included in the evaluation of potential impacts of multiple mines (Chapter 13).

6.1.2 Mining Processes

6.1.2.1 Extraction

Ore associated with the western portion of the Pebble deposit is near-surface and, in our scenarios, would be mined via conventional open-pit mining methods of drilling and blasting. Pit depth and width would be increased progressively to recover the ore. Pit walls and benches would be constructed to stabilize slopes for safety and to direct runoff. Dusts would be controlled by wetting surfaces with site water and covering truck beds during transport of excavated rock. Groundwater flow into the pit would be managed by pumping to storage ponds or TSFs for later treatment or use in mine processes. Although our scenarios describe open pit mining, underground methods could be used, particularly for the deeper eastern portion of the ore body. Many of the impacts would be similar in type and magnitude to those of surface mining (Section 4.2.3.1).

6.1.2.2 Ore Processing

In the mine scenarios, an in-pit crusher would reduce the ore to particles below a maximum size and a conveyor would bring the crushed ore to processing facilities. Ore would be processed in a flotation circuit similar to that described in Section 4.2.3.3. The milling process would generate two tailings streams, one from the rougher flotation circuit (bulk tailings having undergone a single grind sequence) and another from the secondary cleaner circuit (cleaner scavenger tailings) (Figure 4-3). Selective flotation would be used to minimize the amount of potentially acid-generating (PAG) tailings. Copper (+gold) and molybdenum concentrates would be produced as described in Section 4.2.3.3, with the copper (+ gold) slurry concentrate pumped via pipeline to Cook Inlet and the final molybdenum concentrate dried, bagged, and trucked off site for processing. Gold associated with the copper minerals in the slurry concentrate would be recovered at an off-site smelter. Pyrite tailings would be directed either to the TSF for subaqueous disposal or to a vat leach cyanidation operation for removal of gold (Box 4-6), after which sulfide-rich tailings would be directed to the TSF for subaqueous disposal. A cyanide destruction unit would be used at the end of the leaching process (Box 4-6).

All chemical reagents used in ore processing (Box 4-5) would be transported to the mine site, then prepared and stored in areas with secondary containment and instrumentation to detect any spills or leaks. All pipelines would be designed to standards of the American Society of Mechanical Engineers (ASME), which include the use of liners to minimize abrasion and corrosion, freeze protection, secondary containment over water bodies, and leak monitoring and detection. Dusts would be controlled in the processing area through use of cartridges, wet scrubbers, and/or enclosures.

6.1.2.3 Waste Rock

Waste rock consisting of both PAG and non-acid-generating (NAG) materials would be stored around the mine pit, at least partially within the groundwater drawdown zone from mine pit dewatering. PAG waste rock would be stored separately from NAG waste rock. Over the life of the mine, PAG waste rock would be blended with processed ore to allow consistency in chemical usage and to remove material from surface storage prior to its expected time of acid generation (e.g., within 20 years of its excavation). Any PAG material remaining unprocessed at the end of mining would be processed separately prior to closure.

During operation, waste rock piles would be constructed with a 2:1 slope for structural stability and minimization of the amount of runoff requiring treatment. Waste rock piles would occupy approximately 2.3, 13.0, and 22.6 km^2 in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively (Table 6-2). Water quality of the leachate from waste rock is described in Tables 8-6 and 8-7. Monitoring and recovery wells and seepage cutoff walls would be placed downstream of waste rock piles to manage seepage, with seepage and contaminated groundwater directed either into collection ponds for use in mine processes or for treatment and release to the environment, or into the mine pit. Stormwater falling upslope of waste rock piles would be diverted around the piles and directed toward sedimentation ponds for settling of suspended solids prior to discharge to a nearby stream, or for treatment if determined to be contaminated. Embankments would be constructed above the seepage cutoff walls to contain any excess stormwater runoff that could not be contained in collection ponds. Water captured in these embankments would be released or directed to treatment as appropriate. Because the Tertiary volcanic rocks are classified as NAG (Ghaffari et al. 2011, PLP 2011), they may be useful for building purposes such as TSF construction. However, because of the potential for metals leaching, use would be appropriate only where leachate would be collected for treatment as necessary.

Table 6-2. Mine scenario parameters. These scenarios were developed by the U.S. Environmental Protection Agency for the purposes of this assessment, but draw heavily on specifics put forth by Ghaffari et al. (2011).

Parameter	Mine Scenario		
	Pebble 0.25	Pebble 2.0	Pebble 6.5
Amount of ore mined (billion metric tons)	0.23	1.8	5.9
Ore volume (million m^3)	86.9	697	2270
Approximate duration of mining	20 years	25 years	78 years
Ore processing rate (metric tons/day)	31,100	198,000	208,000
Tailings produced, dry (billion metric tons)	0.225	1.80	5.86
Tailings produced, volume (million m^3)	158	1,270	4,130
Mine Pit			
Surface area (km^2)	1.5	5.5	17.8
Depth (km)	0.30	0.76	1.24
Waste Rock Pile			
Surface area (km^2)	2.3	13.0	22.6
PAG waste rock (million metric tons)	86	580	4,700
PAG waste rock bulk density (metric tons/m^3)	2.08	2.08	2.08
PAG waste rock area (km^2)	0.55	1.79	6.77
NAG waste rock (million metric tons)	320	2,200	11,000
NAG waste rock bulk density (metric tons/m^3)	2.08	2.08	2.08
NAG waste rock area (km^2)	1.78	11.2	15.8
TSF 1[a]			
Capacity, dry weight (billion metric tons)	0.25	1.97	1.97
Surface area, interior (km^2)[b]	6.5	14.2	14.2
Surface area, exterior (km^2)	6.8	16.1	16.1
Maximum dam height (m)	92	209	209
Maximum number of dams	1	3	3
Capacity, volume (million m^3)	177	1,390	1,390
Tailings dry density (metric tons/m^3)[c]	1.42	1.42	1.42
NAG density, embankment (metric tons/m^3)[c]	2.31	2.31	2.31
TSF 2[a]			
Capacity, dry weight (billion metric tons)	NA	NA	3.69
Surface area, interior (km^2)[b]	NA	NA	20.1
Surface area, exterior (km^2)	NA	NA	22.7
Maximum dam height (m)	NA	NA	Not determined
Maximum number of dams	NA	NA	3
Capacity, volume (million m^3)	NA	NA	2,600
TSF 3[a]			
Capacity, dry weight (billion metric tons)	NA	NA	0.96
Surface area, interior (km^2)[b]	NA	NA	8.23
Surface area, exterior (km^2)	NA	NA	9.82
Maximum dam height (m)	NA	NA	Not determined
Maximum number of dams	NA	NA	2
Capacity, volume (million m^3)	NA	NA	680
Total TSF surface area, exterior (km^2)	6.8	16.1	48.6
Transportation Corridor			
Total length (km)	138	138	138
Length in assessment watersheds (km)	113	113	113

Notes:
[a] Final value when TSF is full.
[b] Area does not include TSF dams.
[c] Values are the same for TSF 2 and TSF 3, so not repeated under those TSFs.
NA = not applicable; TSF = tailings storage facility; PAG = potentially acid-generating; NAG = non-acid-generating.

6.1.2.4 Tailings Storage Facilities

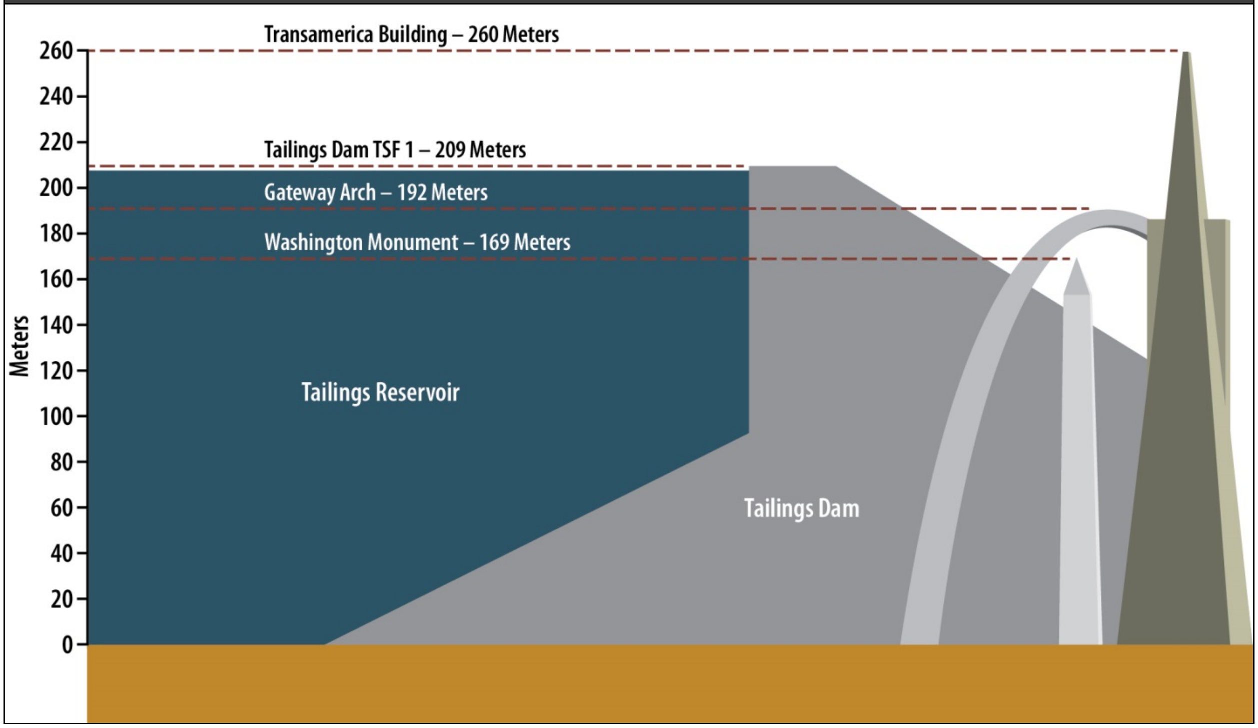
In the mine scenarios, TSF dam design would proceed as described by Ghaffari et al. (2011). The number and size of TSFs in each scenario (Figures 6-1 through 6-3) would be commensurate with tailings storage requirements. The water rights application submitted by Northern Dynasty Minerals to the State of Alaska in 2006 described several potential locations for TSFs (NDM 2006). Drawing on this information, and given site-specific geotechnical, hydrological, and environmental considerations, we assume that the higher mountain valleys similar to the site of TSF 1, on the flanks of Kaskanak Mountain, are the most plausible TSF sites for a mine at the Pebble deposit. This placement does not imply that these sites would not pose unacceptable environmental harm, or that they would be the least environmentally damaging practicable alternatives for purposes of Clean Water Act permitting. Permit-specific study, which is beyond the scope of this assessment, would determine if these or other sites met these criteria.

At each TSF, a rockfill starter dam would be constructed, with a liner (high-density polyethylene geomembrane on top of a geosynthetic clay liner) extending up the upstream dam face. Seepage capture and toe drain systems would be installed at the upstream toe, with perpendicular drains installed to direct seepage toward collection ponds. Each 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 4-4) (Ghaffari et al. 2011). At some point, dam construction would shift to the centerline method (Figure 4-4), and a new stage would be constructed as the capacity of each previous stage was approached. TSF 1 would require maximum dam heights of approximately 92 m for the Pebble 0.25 scenario and 209 m for both the Pebble 2.0 and Pebble 6.5 scenarios (Table 6-2, Figure 6-4).

Given the low grade of ore expected in the region, our mine scenarios would produce large amounts of tailings: approximately 99% of the mass of ore processed would be tailings, with 85% as NAG bulk tailings and 14% as PAG (pyritic) tailings (Ghaffari et al. 2011). Both types of tailings would be directed to TSFs (Figures 6-1 through 6-3). The discharge of bulk tailings would be managed such that the coarsest materials (fine sand) would be discharged 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. 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.

The capacity and dimensions of each TSF are listed in Table 6-2. Pebble 6.5, the largest size scenario considered, would require the construction of TSFs 1, 2, and 3, with a combined tailings capacity exceeding 6 billion metric tons. We estimate that these three TSFs would have a combined surface area of more than 48 km^2 (Table 6-2).

Figure 6-4. Height of the dam at tailing storage facility (TSF) 1 in the Pebble 2.0 and Pebble 6.5 scenarios, relative to U.S. landmarks.



During operation, water quality in TSF ponds would be similar to process water. At the end of mining, process water would no longer enter the TSF, so it is expected that, over time, dilution from precipitation would cause the composition of tailings pond water to approach that of local surface water. Seepage from the base of the tailings impoundment, either during operation or after closure, would be expected to be similar to water quality estimates based on pre-mining humidity-cell test results (Appendix H). The low solubility of oxygen in water (less than 15 mg/L) limits the access of oxygen to submerged unreacted sulfide minerals in the tailings, reducing dissolution reaction rates and thus the concentration of solutes. In addition, trace amounts of carbonate or silicate minerals may partially neutralize acid under anoxic conditions commonly encountered in sulfidic tailings, further limiting the solubility of metals and other trace elements (Blowes et al. 2003). However, a good deal of uncertainty exists because the humidity cell tests used to predict pore water chemistry represent a small sample of the ore body. Thus, actual water quality in the tailings impoundment may differ significantly from what is estimated (Appendix H). For example, lower concentrations of metals than those reported in humidity cells tests would likely be seen in TSF water if pH was buffered by reactions with carbonate and silicate minerals (see Section 8.1.1.1 for discussion of tailings leachate quality).

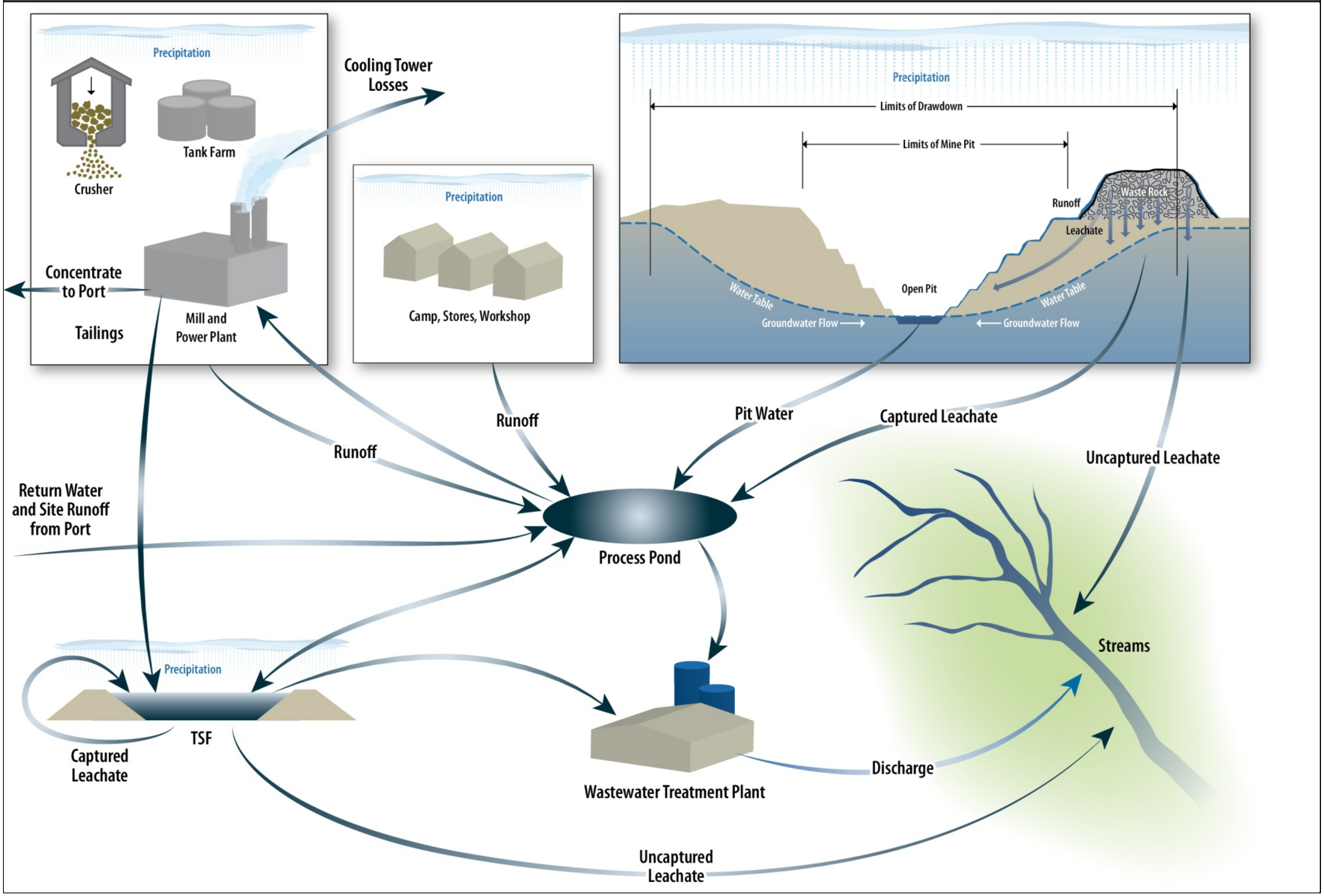
Well fields spanning the valley floor would be installed at the downstream base of all tailings dams to monitor groundwater flow down the valley, including potential uncaptured seepage from the TSF. If contaminated groundwater was detected, monitoring wells would be converted to collection wells or new recovery wells would be installed, and water from the well field would be pumped back into the TSF or treated and released to stream channels.

6.1.2.5 Water Management and Treatment

Water uses in the mine scenarios would include ore processing, tailings slurry transport, and transport of copper concentrate slurry in the product pipeline. In this section, we provide an overview of water management and treatment in the three mine scenarios. Figure 6-5 presents a schematic illustration of these components (note that, for clarity, diversions of stormwater around mine components are not shown on the schematic).

- **Stormwater runoff that did not contact potential contaminants** would be diverted around mine components (e.g., waste rock piles, processing facilities) in ditches directed toward sediment settling ponds.
- **Stormwater runoff from waste rock piles and water from pit dewatering** would be pumped to lined process water ponds; water reclaimed from tailings impoundments or tailings thickening also would be stored in the process water storage ponds for reuse in ore processing.
- **Stormwater falling onto TSFs** would be stored in the tailings impoundments and used in the process water cycle.
- **Seepage collected from waste rock piles and TSFs** would be directed to lined seepage collection ponds or TSFs for later treatment.
- **Seepage escaping the waste rock and TSF leachate collection systems** would be monitored with monitoring wells. If groundwater contamination was detected, wells would be converted to recovery wells or new recovery wells would be installed, and the groundwater pumped to either a TSF or a storage pond for later treatment.
- **Water reclaimed from the copper concentrate after transport to the port** would be returned to process water storage ponds via pipeline from the port.
- **Streams blocked by the mine pit or waste rock piles** would be diverted, where practicable, around and downstream of the mine. However, the zone of groundwater depression around the mine pit and the slow filling of the post-operation pit would likely dewater these streams for as long as it took the pit to fill, which could be hundreds of years.
- **Prior to being discharged, water would be treated** to meet effluent limits using chemical precipitation methods and/or reverse osmosis. Water would be discharged to the South and North Fork Koktuli Rivers according to permit conditions for composition, flow, and temperature. Sludge and brine from the treatment process would be disposed in the TSF.

Figure 6-5. Water management and water balance components for the three mine scenarios.



Water balances for both the operation and post-closure phases of our mine scenarios are discussed in detail in Section 6.2.2. Development of these water balances is important, because they estimate the amount of water available to contribute to downstream flows. Calculating these water balance components is challenging, however, and requires a number of assumptions (e.g., estimates of 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 evaporation, and the net balance of water to and from groundwater sources). Information exists to estimate precipitation and evaporation, 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 in these calculations—is the net balance of water from groundwater sources.

Mining operations would affect the quantity, quality, timing, and distribution of surface flows. Mining operations always consume some water, so there would be less water available in the landscape during active mining than before the mine was present. Major stream flow reductions during mine operation would result from the capture of precipitation falling on the mine pit, waste rock piles, and TSFs (Table 6-3, Figure 6-5). 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 6-5). Leachate recovery wells for any detected groundwater contamination downstream of the waste rock piles would extend the cone of depression. 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. Precipitation and other 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. Runoff at the port site would be pumped to the mine site in the return water pipeline, contributing to the mine's water supply and avoiding the need for treatment at the port.

Prior to active mining, but after the starter dam was built for TSF 1, site water would be diverted to TSF 1 to allow sufficient water for process plant startup. During mine operation, groundwater and precipitation would be pumped from the mine pit to prevent flooding of the mine workings (Figure 6-5). Water would be needed for the flotation mill, to operate the TSF, and to maintain concentrated slurry in the product pipeline.

Table 6-3. Summary of annual water balance flows (million m^3/year) during operations for the three mine scenarios.

Flow Component	Pebble 0.25	Pebble 2.0	Pebble 6.5
Captured at mine pit area	9.77	22.4	44.1
Captured at TSF 1	5.86	13.8	13.8
Captured at TSF 2	NA	NA	19.5
Captured at TSF 3	NA	NA	8.43
Captured at mill & other facilities	0.629	2.69	2.69
Potable water supply well(s)	0.031	0.124	0.124
Water in ore (3%)	0.340	2.17	2.27
Total Captured	**16.6**	**41.2**	**91.0**
Cooling tower losses	0.211	1.32	1.32
In concentrate to port	0.166	1.04	1.04
In concentrate return	−0.149	−0.934	−0.934
Runoff collected from port	−0.125	−0.251	−0.251
Stored in TSF as pore water	3.72	23.8	24.9
Stored in mine pit	0	0	0
Crusher use	0.113	0.722	0.758
Total Consumptive Losses	**3.93**	**25.7**	**26.8**
Returned to streams via wastewater treatment plant	10.9	10.3	51.0
Returned as NAG waste rock leachate	0.676	2.58	4.97
Returned as PAG waste rock leachate	0	0.216	1.03
Returned as TSF leakage	1.11	2.35	7.20
Total Reintroduced	**12.7**	**15.4**	**64.2**
Percent of Captured Water Reintroduced	**76.3%**	**37.5%**	**70.5%**

Notes:
TSF = tailings storage facility; NA = not applicable; NAG = non-acid-generating; PAG = potentially acid-generating.

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 that water is no longer available for reuse (Table 6-3, Section 6.2.2). Consumptive losses would be made up by withdrawing water stored in a TSF or by pumping directly from the mine pit. Some of this captured water (approximately 38 to 76%, Table 6-3) would not be needed at the mine site. This excess captured water would be treated to meet existing water quality standards and discharged to nearby streams (Figure 6-5), partially mitigating streamflow lost due to eliminated or blocked upstream reaches (Chapter 7).

6.1.3 Transportation Corridor

6.1.3.1 Roads

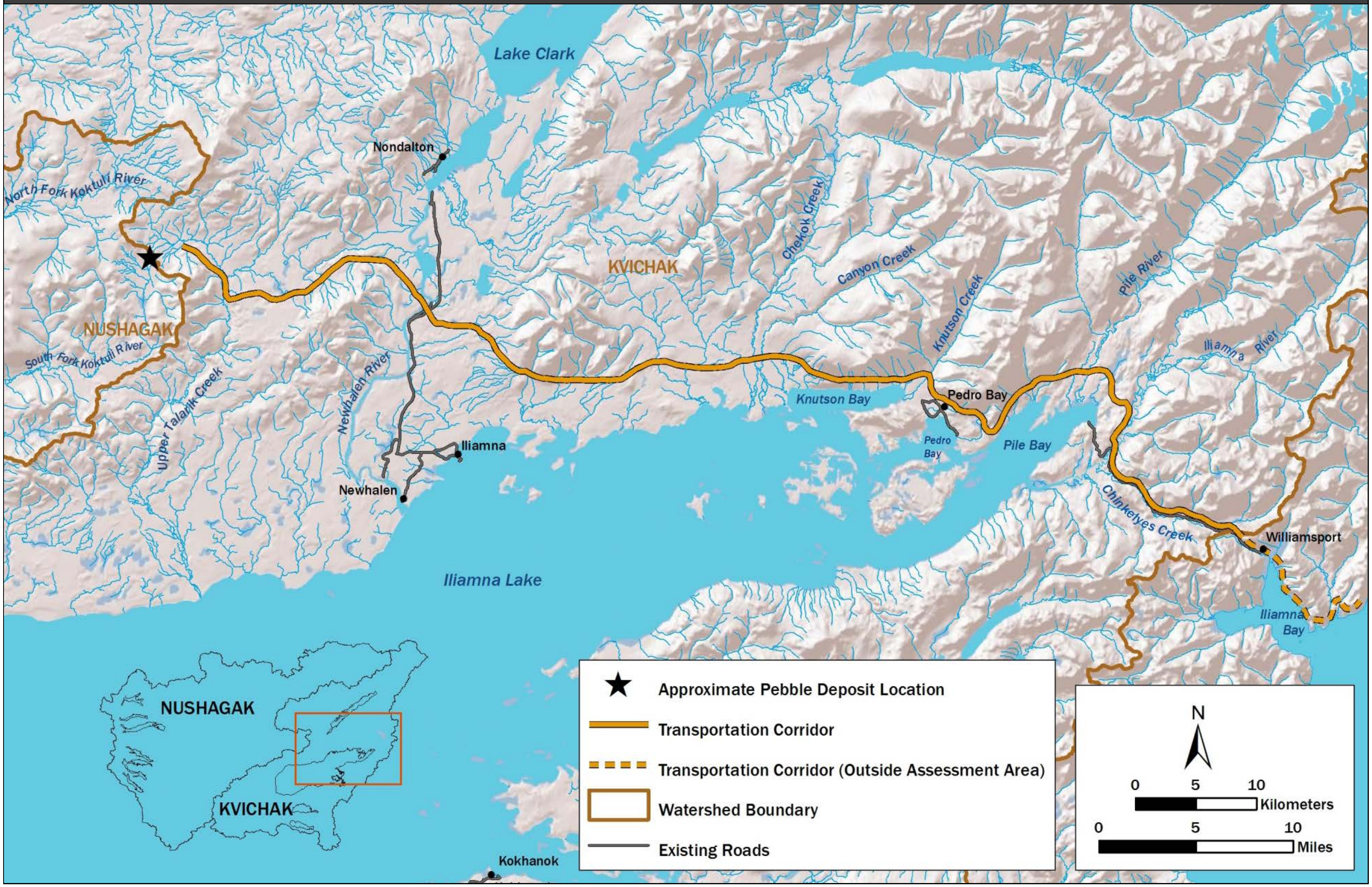
Development of any mine in the Bristol Bay watershed would require substantial expansion and improvement of the region's transportation infrastructure. The Bristol Bay watershed is located in one of the last remaining, virtually roadless regions in the United States. There are no improved federal or state highways, and no railroads, pipelines, or other major industrial transportation infrastructure. Roadways presently link Iliamna Lake (Pile Bay) to Cook Inlet (tidewater at Williamsport) and the Iliamna area (including the Iliamna airport) north to the site of a proposed bridge over the Newhalen River near the village of Nondalton. Two other short road segments link Dillingham to Aleknagik and Naknek to King Salmon (Figure 6-6). Local roads also exist in villages throughout the Nushagak and Kvichak River watersheds. Most people travel by air or boat during the ice-free season, and by air or snow machine in winter.

In our mine scenarios, a 138-km, two-lane (approximately 9-meter-wide), gravel surface, all-weather permanent access road would connect the mine site to a new deep-water port on Cook Inlet (Figure 6-6), from which concentrate would be shipped elsewhere for processing (Ghaffari et al. 2011). An estimated 113 km of this corridor would fall within the Kvichak River watershed (this distance does not include the portion of the road occurring within the potential mine site). This route would traverse highly variable terrain and variable subsurface soil conditions, including extensive areas of rock excavation in steep mountainous terrain.

The primary purpose of this road would be to transport freight by conventional highway tractor-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 (e.g., borrow and gravel pits and rock quarries) would be available at regular intervals along the road route. We assume state-of-the-art practices for design, construction, and operation of the road infrastructure, including design of bridges and culverts for fish passage. Permanent structures would be designed for a service life of 50 years. Because the access road would be kept open for ongoing care, maintenance, and environmental monitoring at the site post-closure, maintenance and resurfacing of the access road would necessarily be required over the same time period, which may extend in perpetuity.

The transportation corridor would cross many streams (including unmapped tributaries), rivers, wetlands, and extensive areas with shallow groundwater, all of which drain to Iliamna Lake (Figure 6-6, Section 10.1). We used a mean annual streamflow threshold of >0.15 m^3/s to designate stream crossings that would be bridged (this threshold was also used to separate small headwater streams from medium streams in broad-scale characterization of stream and river habitats; see Section 3.1.4.2). Bridges, with spans ranging from approximately 12 to 183 m, would be constructed over 12 known anadromous streams and seven additional streams likely to support salmonids. Culverts would be place at all remaining stream crossings. 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).

Figure 6-6. Transportation corridor connecting the Pebble deposit area to Cook Inlet.



Avalanche hazards exist in isolated locations along the alignment, but routing would attempt to avoid any avalanche chutes and runout areas. Because of steep mountain slopes and the 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 (Figure 6-6) may be especially susceptible to these runoff events, as demonstrated in late 2003 when storm runoff washed out several culverts on the state-maintained Pile Bay Road.

6.1.3.2 Pipelines

The transportation corridor would include four pipelines, which would carry copper (+gold) concentrate, return water, natural gas (to fuel a natural gas-fired power generating plant), and diesel fuel between the mine site and the Cook Inlet port (Table 6-4). All pipelines would be designed following ASME standards. 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, pipelines 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 (1.5 meters 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 6-4. Characteristics of pipelines in the mine scenarios.

Pipeline (number of pipes)	Route	Pipe Material	Nominal Diameter (cm)
Along Transportation Corridor			
Copper (+gold) concentrate (1)	Mine to port	HDPE-lined steel	20
Reclaimed water (1)	Port to mine	HDPE-lined steel	18
Natural gas (1)	Port to mine	Steel	5
Diesel fuel (1)	Port to mine	Steel	13
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
Reclaimed water (1)	TSF barge to TSF head tank	HDPE	107
Reclaimed 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; TSF = tailings storage facility; TBD = to be determined.
Source: Ghaffari et al. 2011.

On the mine site, pipelines would carry tailings slurry from the process plant to the TSFs and reclaimed water from the TSFs to the process water storage ponds (Table 6-4). There also 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 storage pond or the TSFs. Failures of these on-site pipelines could result in uncontrolled releases in 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.

6.2 Specific Mine Scenarios

In this assessment we evaluate three specific mine scenarios representing mines of different sizes. The smallest mine scenario, Pebble 0.25, represents a median-sized porphyry copper deposit of 250 million tons (230 million metric tons) (Singer et al. 2008). The second mine scenario, Pebble 2.0, is based largely on the 25-year, 2 billion tons (1.8 billion metric tons) case described by Ghaffari et al. (2011) for initial development at the Pebble deposit. The third mine scenario, Pebble 6.5, is based largely on the 78-year, 6.5 billion tons (5.9 billion metric tons) case described by Ghaffari et al. (2011) for further resource development at the Pebble deposit.

Pebble 2.0 and Pebble 6.5 reflect projects based on extensive exploration, assessment, and preliminary engineering, which are described by Ghaffari et al. (2011) as "economically viable, technically feasible and permittable." They are among the most likely to be developed in the Bristol Bay watershed and are site-specific to the Pebble deposit. For the purposes of this assessment, we have also placed the Pebble 0.25 scenario at the Pebble deposit because of the availability of site-specific information. If mines are developed at other exploration sites in the watershed (Figure 13-1), they are likely to have characteristics and impacts much closer to those of the Pebble 0.25 scenario. Table 6-2 provides detailed parameters for each of our three mine scenarios, and Figures 6-1 through 6-3 show the general layout of each scenario's major mine components.

6.2.1 Mine Scenario Footprints

The major mine components contributing to each mine scenario footprint are the mine pit, waste rock piles, and TSFs. Placement of these components for each of the scenarios is shown in Figures 6-1 through 6-3. In each case, these layouts represent one possible configuration for the mine. Other configurations are possible, but would be expected to have impacts of similar types and magnitudes. Each mine scenario footprint also includes two additional components: the groundwater drawdown zone, or the area over which the water table is lowered due to pit dewatering (Figure 6-5), and the area covered by plant and ancillary facilities (e.g., ore-crushing and screening areas, processing mill, storage and stockpile areas, workshops, roads within the mine site, pipeline corridors, and other disturbed areas). Summing these areas (mine pit, waste rock piles, TSFs, drawdown zone, and plant and ancillary facilities) and correcting for any overlap among them yields an estimate for total mine footprint area in each scenario (Tables 6-5 through 6-7).

6.2.1.1 Pebble 0.25 Footprint

Figure 6-1 shows the general layout of the mine pit, waste rock piles, and TSF for the Pebble 0.25 scenario. The TSF, identified as TSF 1, is located in a natural valley in a headwater tributary of the North Fork Koktuli River located to the west of the Pebble deposit. The valley would be closed off by the construction of a rockfill dam 92 m in height (Table 6-2). The waste rock pile area was determined by calculating the area that would be covered by the expected volume of waste rock, assuming approximately 100-m-high piles and taking advantage of natural landforms near the mine pit. In this scenario, separate PAG and NAG waste rock would be created during mine operation. PAG waste rock would be processed as mill conditions permit throughout the mine life, with the intent to process all of the PAG waste rock before mine closure. The area of the plant and ancillary facilities is estimated to account for approximately 4% of the total mine footprint area (Table 6-5). The drawdown zone (Table 6-5) includes the mine pit and the area beyond the mine pit perimeter, including some of the waste rock piles, up to the limit of the cone of depression (see Box 6-2 for discussion of mine pit drawdown calculations).

Table 6-5. Estimated areas for individual mine components in the Pebble 0.25 scenario.

Component	Area (km^2)
Drawdown zone	10.1
Mine pit	1.54
NAG waste rock in drawdown zone	0.49
PAG waste rock in drawdown zone	0.55
Other area in drawdown zone	7.49
NAG waste rock not in drawdown zone or TSFs	1.29
PAG waste rock not in drawdown zone	0.00
Cumulative plant and ancillary areas	0.73
TSFs[a]	6.82
TSF 1	6.82
TOTAL MINE FOOTPRINT	18.9

Notes:
[a] Exterior TSF area.
[b] NAG = non-acid-generating; PAG = potentially acid-generating; TSF = tailings storage facility.

6.2.1.2 Pebble 2.0 Footprint

Figure 6-2 depicts the general layout of the major mine components for the Pebble 2.0 scenario, including the mine pit, the waste rock piles, and the TSF. The TSF is located in the same valley as TSF 1 in the Pebble 0.25 scenario (Figure 6-2), but it is increased in size to accommodate the additional tailings expected with this larger mine size. Plant and ancillary facilities are estimated to account for approximately 7% of the total disturbed area (Table 6-6).

Waste rock piles are located around the perimeter of the mine pit, with separate areas designated for NAG and PAG waste rock. As in the Pebble 0.25 scenario, PAG and NAG waste rock would be stored in separate waste rock piles during mine operation, and the PAG rock would be processed as mill conditions permit throughout the mine life with the intent to process all of the PAG waste rock before mine closure. Dewatering of the mine pit would generate a cone of depression around the pit, and more than half of the area of the waste rock piles would fall within the resulting drawdown zone (Table 6-6).

Table 6-6. Estimated areas for individual mine components in the Pebble 2.0 scenario.

Component	Area (km^2)
Drawdown zone	21.4
Mine pit	5.50
NAG waste rock in drawdown zone	7.08
PAG waste rock in drawdown zone	1.29
Other area in drawdown zone	7.52
NAG waste rock not in drawdown zone or TSFs	4.14
PAG waste rock not in drawdown zone	0.50
Cumulative plant and ancillary areas	3.13
TSFs[a]	16.1
TSF 1	16.1
TOTAL MINE FOOTPRINT	45.3

Notes:
[a] Exterior TSF area.
[b] NAG = non-acid-generating; PAG = potentially acid-generating; TSF = tailings storage facility.

6.2.1.3 Pebble 6.5 Footprint

The general layout of the Pebble 6.5 scenario is similar to that of the Pebble 2.0 scenario, with major differences being a larger open pit, different and expanded areas for the waste rock piles, and the inclusion of two additional TSFs (TSF 2 and TSF 3) to store the increased tailings volume (Figure 6-3, Table 6-7). Placement of TSF 2 and TSF 3 in this scenario draws upon some of the TSF options presented in Northern Dynasty Minerals' water rights application (NDM 2006) and takes advantage of natural landforms in the Pebble deposit area.

The mine pit is located as shown by Ghaffari et al. (2011), based on evaluation of the Pebble deposit. Waste rock piles are located around the perimeter of the expanded mine pit, with some portion of the PAG waste rock stored in the mine pit to utilize storage within the drawdown zone prior to PAG waste rock being taken to the surface for processing. This practice would reduce the amount of PAG waste rock that must be stored outside the drawdown zone and, therefore, the amount of PAG leachate that could seep into the South Fork Koktuli River.

Areas of the plant and ancillary facilities are the same as those described for the Pebble 2.0 scenario; because production rates of the Pebble 2.0 and Pebble 6.5 scenarios are similar, no increase in these areas is needed for the larger mine scenario.

Table 6-7. Estimated areas for individual mine components in the Pebble 6.5 scenario.

Component	Area (km^2)
Drawdown zone	43.4
Mine pit	17.8
NAG waste rock in drawdown zone	10.3
PAG waste rock in drawdown zone	4.37
Other area in drawdown zone	10.9
NAG waste rock not in drawdown zone or TSFs	5.50
PAG waste rock not in drawdown zone or mine pit	2.40
Cumulative plant and ancillary areas	3.13
TSFs[a]	48.6
TSF 1	16.1
TSF 2	22.7
TSF 3	9.8
TOTAL MINE FOOTPRINT	103

Notes:
[a] Exterior TSF area.
[b] NAG = non-acid-generating; PAG = potentially acid-generating; TSF = tailings storage facility.

6.2.2 Water Balance

Many of the potentially significant impacts of large-scale mining relate to a mine's use of water and its impact on water resources. To understand potential impacts of water use in our mine scenarios, we developed an annual water balance for each scenario that accounts for major flows into and out of the mine area. Three major categories of flows make up each water balance estimate: water inputs, consumptive losses, and water outputs; these categories are discussed in detail in the following sections. These water balances focus on changes in flows entering or leaving the mine site, relative to pre-mining conditions. 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. Each water balance subtracts consumptive water losses within mine operations from water inputs to determine the water available for release. This water balance analysis does not attempt to describe or quantify internal flows among mine components, although some are mentioned when necessary to explain the analysis. The water balance analysis also does not attempt to quantify any flows that are recycled within the mine site, because these do not capture water from the environment or release water to it.

6.2.2.1 Water Inputs

Water inputs for each of the three scenarios are summarized in Table 6-3. These inputs are derived primarily from net precipitation (total precipitation minus any losses due to evapotranspiration) that falls on the mine footprints and is captured by water collection and management systems within the mine site. We assume that all captured flows would be available for use by the mine operator. Three gages surrounding the mine site were used to calculate net precipitation at the mine site: gage SK100B (USGS gage 15302200) on the South Fork Koktuli River, gage NK100A (USGS gage 15302250) on the North Fork Koktuli River, and gage UT100B (USGS gage 15300250) on Upper Talarik Creek. Net precipitation (or measured runoff) at each gage represents precipitation minus evapotranspiration, plus or minus interbasin storage, plus or minus internal groundwater storage. We assumed interbasin and groundwater storage were zero since we were averaging across the three watersheds. Therefore, the runoff measured at each gage represents net precipitation (precipitation minus evapotranspiration). Monthly mean flows for each gage were summed across the year, producing an area-weighted average of net runoff of 860 mm per year.

Water inputs resulting from the mine footprints are calculated as the product of footprint areas multiplied by annual net precipitation. For the TSFs, the volume of water captured is based on the interior area of the TSF, defined as the area within the dam crests and excluding the downstream faces of the rockfill dams.

Dewatering the mine pit would create a cone of depression around the mine extending beyond the limits of the mine pit. Because the mine pit would be located very close to the water divide between the South Fork Koktuli River, North Fork Koktuli River, and Upper Talarik Creek watersheds, we assume that there would be negligible net influx of groundwater from beyond the cone of depression. Most of the groundwater outside the cone of depression would flow away from the site. Therefore, the area of the cone of depression would be determined by matching net precipitation falling within the drawdown zone with the calculated groundwater inflow into the mine pit (Box 6-2).

Precipitation falling on areas outside of these disturbed footprints would infiltrate as groundwater or flow into streams without treatment. Flow in upstream tributaries blocked by the mine footprint would be piped or otherwise diverted around the footprint and discharged back into streams without treatment, where practicable. Because this diverted flow is not captured by the mine operations, it is not explicitly included in the water balance tabulations and is assumed to remain part of the background flow.

6.2.2.2 Consumptive Losses

Consumptive losses for each mine scenario are summarized in Table 6-3. To estimate the amount of water available for release, we subtracted consumptive losses associated with mining activities from the captured flows (Table 6-3). Consumptive losses would include water pumped to the port in the copper (+gold) concentrate pipeline minus return water, cooling tower evaporation and drift losses, interstitial water trapped in the pores of stored tailings, water used for dust suppression, and water stored in the mine pit after closure. The tailings pore water accounts for over 90% of consumptive loss during mine operations (Table 6-3). When the tailings settle, about 46% of the volume would consist of voids between solid particles; the water trapped in these pore spaces would no longer be available for use at the mine or release to streams.

BOX 6-2. MINE PIT DRAWDOWN CALCULATIONS

Groundwater flow into the mine pit was calculated using a simplified model based on the Dupuit-Forcheimer discharge formula for steady-state radial flow into a fully penetrating well in a phreatic aquifer with a diameter equal to the average mine pit diameter. The hydraulic conductivity data gathered in the Pebble deposit area during geologic investigations show significant scatter (Figure 6-7). 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 ($k = 1 \times 10^{-4}$ 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 analytical 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 in our simplified model was at the ground surface and assumed to be horizontal.

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 0.274 m^3/s (4,350 gpm), 0.584 m^3/s (9,250 gpm) and 1.19 m^3/s (18,800 gpm) for the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. The Pebble 2.0 mine inflow agrees closely with the estimate provided by Ghaffari et al. (2011).

The cone of depression was determined to extend 1,148 m, 1,222 m, and 1,260 m from the edge of the idealized circular mine pit in the Pebble 0.25, 2.0, and 6.5 scenarios, 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 mine pits to derive the drawdown zones presented in Tables 6-5 through 6-7.

The waste rock piles do not lie completely within the drawdown zones. This is important in assessing water quality because precipitation falling on the waste rock piles within the drawdown zone is presumed to be collected within the mine pit, whereas precipitation falling outside of the drawdown zone is presumed to migrate away from the mine pit. To assess more accurately the waste rock pile positions relative to the drawdown zones, we distorted the shape of the cone of depression by superimposing the drawdown zone on a uniform flow field with a southern gradient of 0.0354, approximately equal to the slope of the ground surface across the mine pit from north to south. The effect of this distortion is a shift in the boundaries of the cone of depression to the north, resulting in larger areas of waste rock outside of the drawdown zones.

Information on flows in the concentrate and return water pipelines and on cooling tower losses is reported by Ghaffari et al. (2011). The return water pipeline reduces consumptive losses by returning water 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 that area by the precipitation rate at the port (1,830 mm/year) to determine contributions from port site runoff (Table 6-3). We also included a consumptive loss at the crusher and screening site for dust control equal to 1% of the mass of the material being crushed.

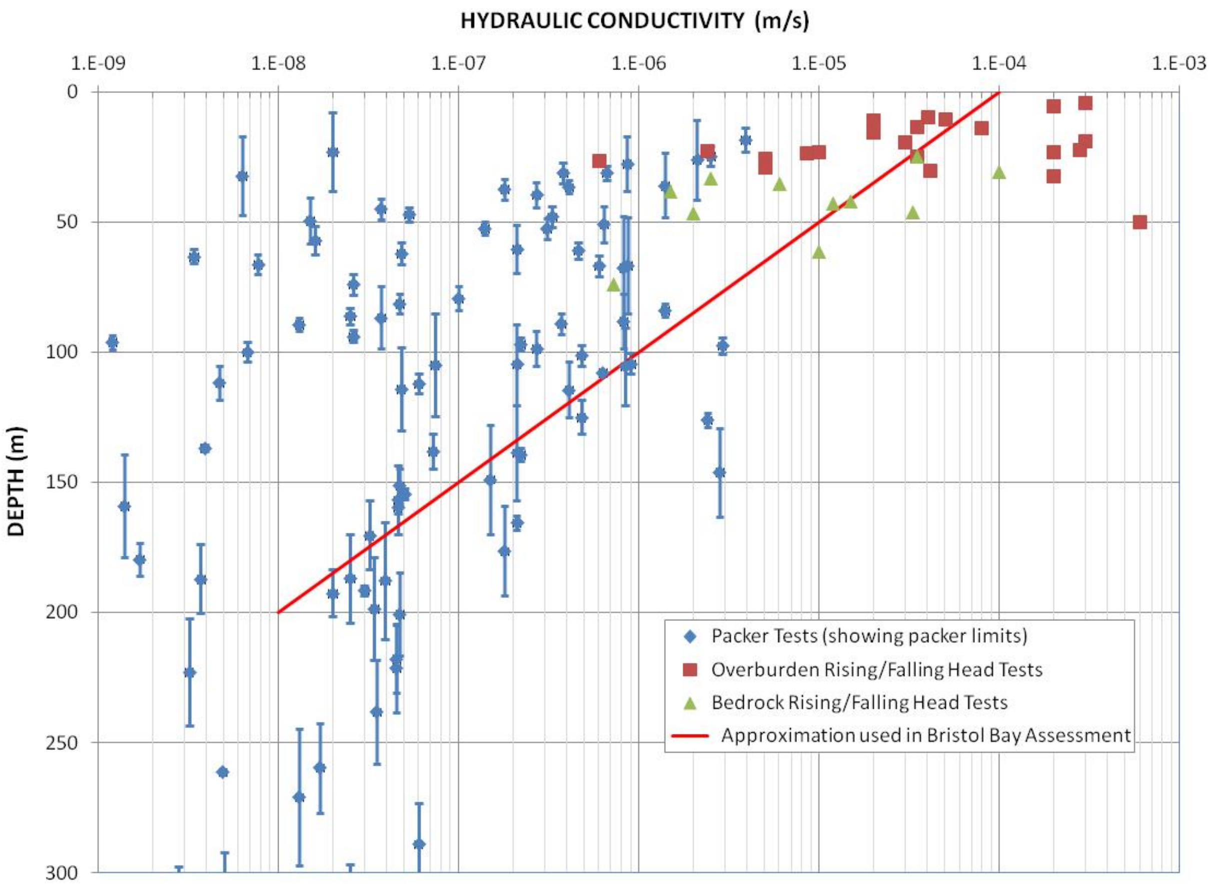
6.2.2.3 Water Outputs

When the amount of captured water exceeds consumptive losses, water would be available, after testing and treatment, for release into area streams. This released water may differ from natural stream water in chemistry and temperature, but would comply with permitted discharge requirements. Water may be reintroduced at locations, flow rates, or times of year that differ from baseline conditions.

The water deficit for each scenario—that is, the amount of water extracted from the environment and not returned to streams—is presented in Table 6-3. These water deficits equal the total consumptive losses of approximately 3.9 million m^3/year, 26 million m^3/year, and 27 million m^3/year for the Pebble

0.25, 2.0, and 6.5 scenarios, respectively. The percentage of water reintroduced to streams, including uncaptured leachate, would equal approximately 76, 38, and 71% of the total water captured in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively.

Figure 6-7. Hydraulic conductivity in the Pebble deposit area. Data are from three test types: bedrock packer (Lugeon) tests (blue diamonds, with error bars indicating upper and lower limits of zone tested) (PLP 2011: Chapter 8 and Appendix 8.1N); overburden rising or falling head tests (red squares) (PLP 2011: Chapter 8 and Appendix 8.1C); and bedrock rising or falling head tests (green triangles) (PLP 2011: Chapter 8 and Appendix 8.1C). Red line indicates values used in the assessment's mine pit drawdown and tailings storage facility leakage calculations.



6.2.2.4 Additional Water Balance Issues

During the early life of each mine, there is one other significant source of water that a mine operator would need to manage that is not considered in Table 6-3: the water obtained from dewatering the sandy and gravelly overburden overlying the waste rock and ore. Based on an average overburden thickness of 30.5 m and a porosity of 0.40, dewatering the overburden would produce one-time quantities of 19 million m^3, 67 million m^3, and 220 million m^3 of water over the mine pit areas in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. This water would be expected to be relatively clean and, if properly managed to control turbidity, could most likely be released without chemical treatment to maintain or augment stream flow.

Water treated at the wastewater treatment plant (WWTP) might not be discharged to the same streams that were dewatered. In accordance with the WWTP discharge points shown by Ghaffari et al. (2011), the WWTP is assumed to discharge to the South and North Fork Koktuli Rivers, but not to Upper Talarik Creek (Figures 6-8 through 6-11).

6.3 Closure and Post-Closure Site Management

As discussed in Section 4.2.4, the assessment examines potential impacts both during mine operations and after mining activities have ceased, either as planned or prematurely. In this section, we consider how the mine scenarios would be handled during and after closure of the mine.

We assume that the mine would be closed after all economically profitable ore was removed from the site, leaving behind the mine pit, NAG waste rock piles, and TSFs. Water at the site would require capture and treatment for as long as it did not meet water quality standards. Weathering of exposed waste rock and pit walls would release ions of potential concern, such as sulfates and metals. Weathering to the point where these contaminants decreased toward their pre-mining background concentrations would likely take hundreds to thousands of years, resulting in the need for monitoring and management of exposed materials and leachate over that time (Blight 2010). To minimize exposure of waste rock and pit walls to weathering, we assume that they would be reclaimed. We also assume that existing water management structures and the WWTP would be monitored and maintained as part of post-closure operations.

Seepage and leachate monitoring and collection systems, as well as the WWTP, might need to be maintained for hundreds to thousands of years. It is impossible to evaluate the success of such long-term collection and treatment systems for mines. No examples exist, because these timeframes exceed both existing systems and most human institutions. Throughout this section, we refer to the potential need for treatment over extended periods. The uncertainty that human institutions have the stability to apply treatment for these timeframes applies to all treatment options.

Figure 6-8. Water flow schematic for the Pebble 0.25 scenario. Flows include water from the non-acid-generating waste rock pile and tailings storage facility (TSF) 1 (dashed black arrows), discharge from the wastewater treatment plant (solid black arrows), flow along the stream channels (solid blue arrows), and known groundwater transfers (dashed blue arrow). For clarity, only flows greater than 5% of total outflows from the TSF and waste rock pile are shown. Gage locations are based on U.S. Geological Survey (2012b) and Pebble Limited Partnership (2011). Confluence points represent virtual gages that were created for analysis purposes (see Section 7.3 for additional details). Note that the spatial orientation of streams and mine components is for schematic purposes only and is not to scale (see Figure 6-11 for a spatially accurate map).

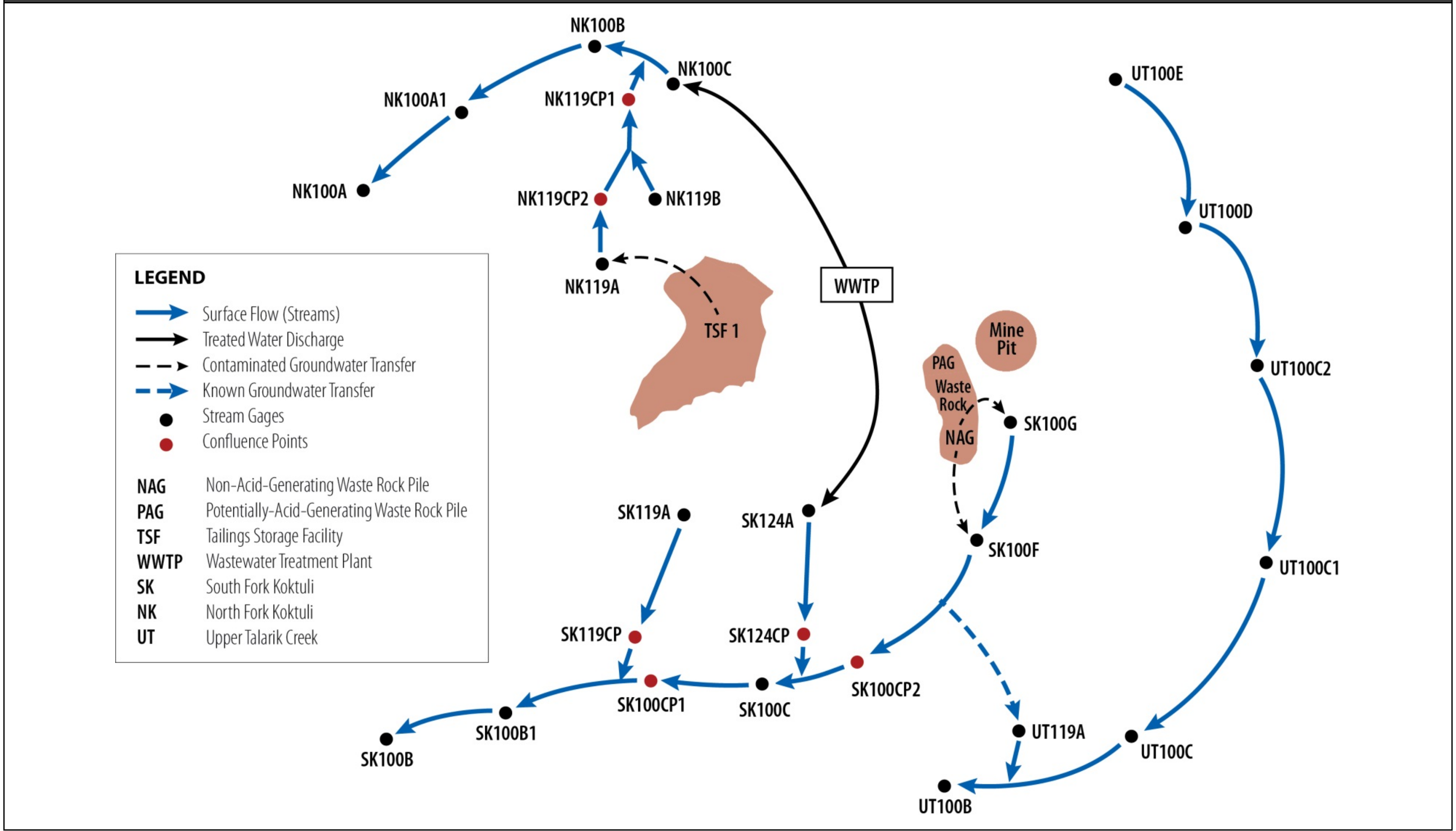


Figure 6-9. Water flow schematic for the Pebble 2.0 scenario. Flows include water from the potentially acid-generating and non-acid-generating waste rock piles and tailings storage facility (TSF) 1 (dashed black arrows), discharge from the wastewater treatment plant (solid black arrows), flow along the stream channels (solid blue arrows), and known groundwater transfers (dashed blue arrow). For clarity, only flows greater than 5% of total outflows from the TSF and waste rock pile are shown. Gage locations are based on U.S. Geological Survey (2012b) and Pebble Limited Partnership (2011). Confluence points represent virtual gages that were created for analysis purposes (see Section 7.3 for additional details). Note that the spatial orientation of streams and mine components is for schematic purposes only and is not to scale (see Figure 6-11 for a spatially accurate map).

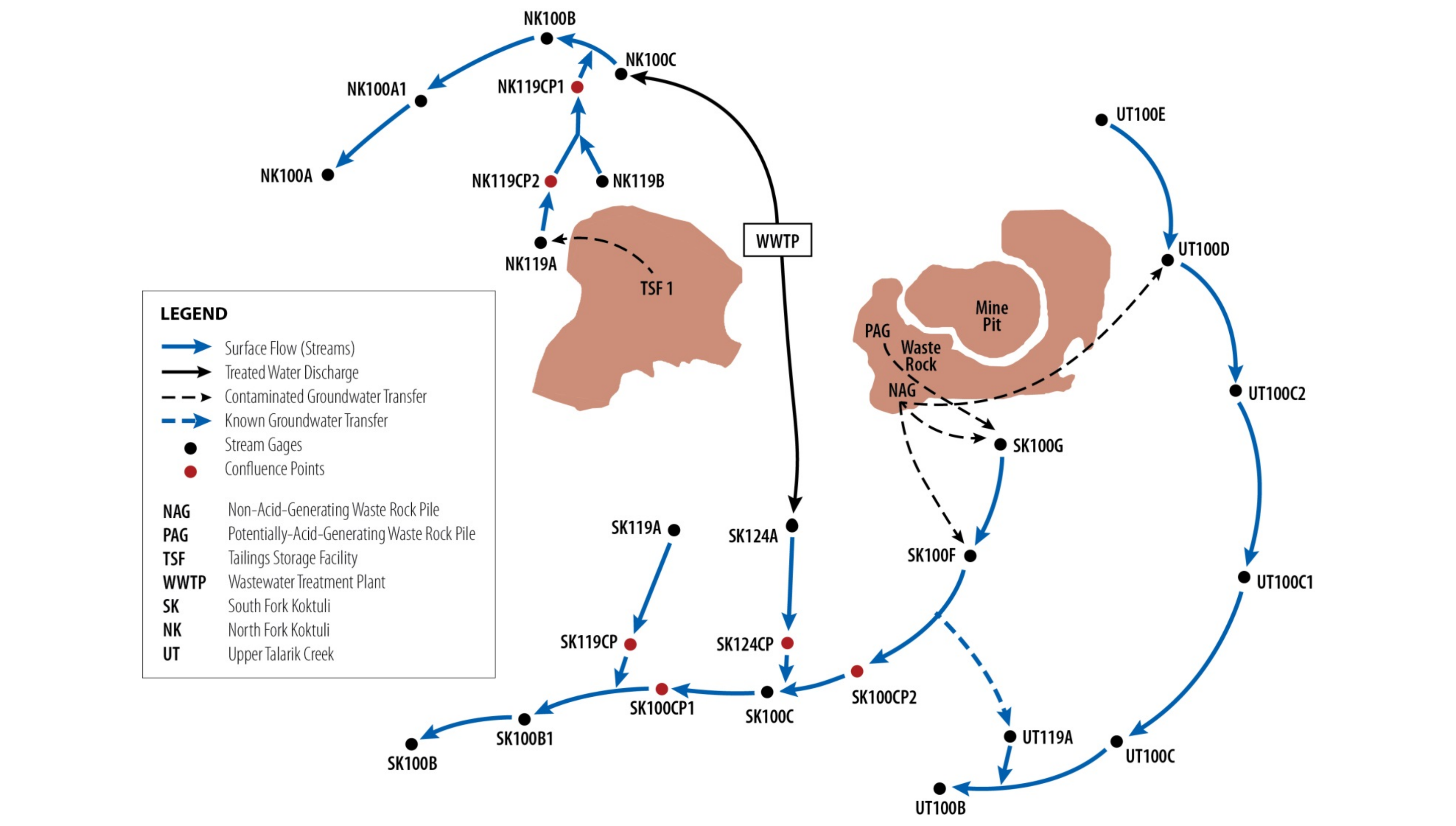


Figure 6-10. Water flow schematic for the Pebble 6.5 scenario. Flows include water from the potentially acid-generating and non-acid-generating waste rock piles and tailings storage facilities (TSFs) 1, 2, and 3 (dashed black arrows), discharge from the wastewater treatment plant (solid black arrows), flow along the stream channels (solid blue arrows), and known groundwater transfers (dashed blue arrow). For clarity, only flows greater than 5% of total outflows from the TSFs and waste rock piles are shown. Gage locations are based on U.S. Geological Survey (2012b) and Pebble Limited Partnership (2011). Confluence points represent virtual gages that were created for analysis purposes (see Section 7.3 for additional details). Note that the spatial orientation of streams and mine components is for schematic purposes

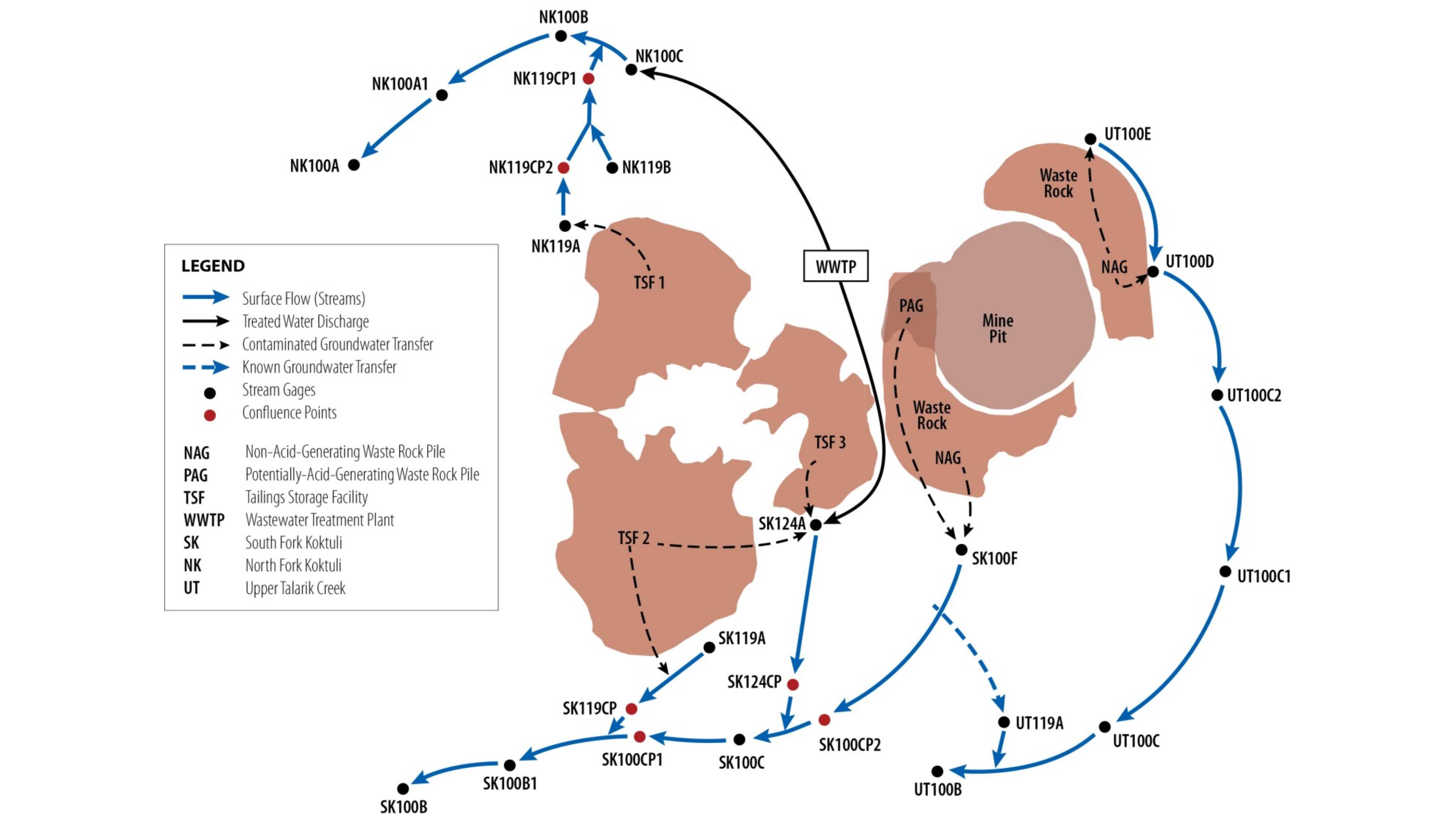
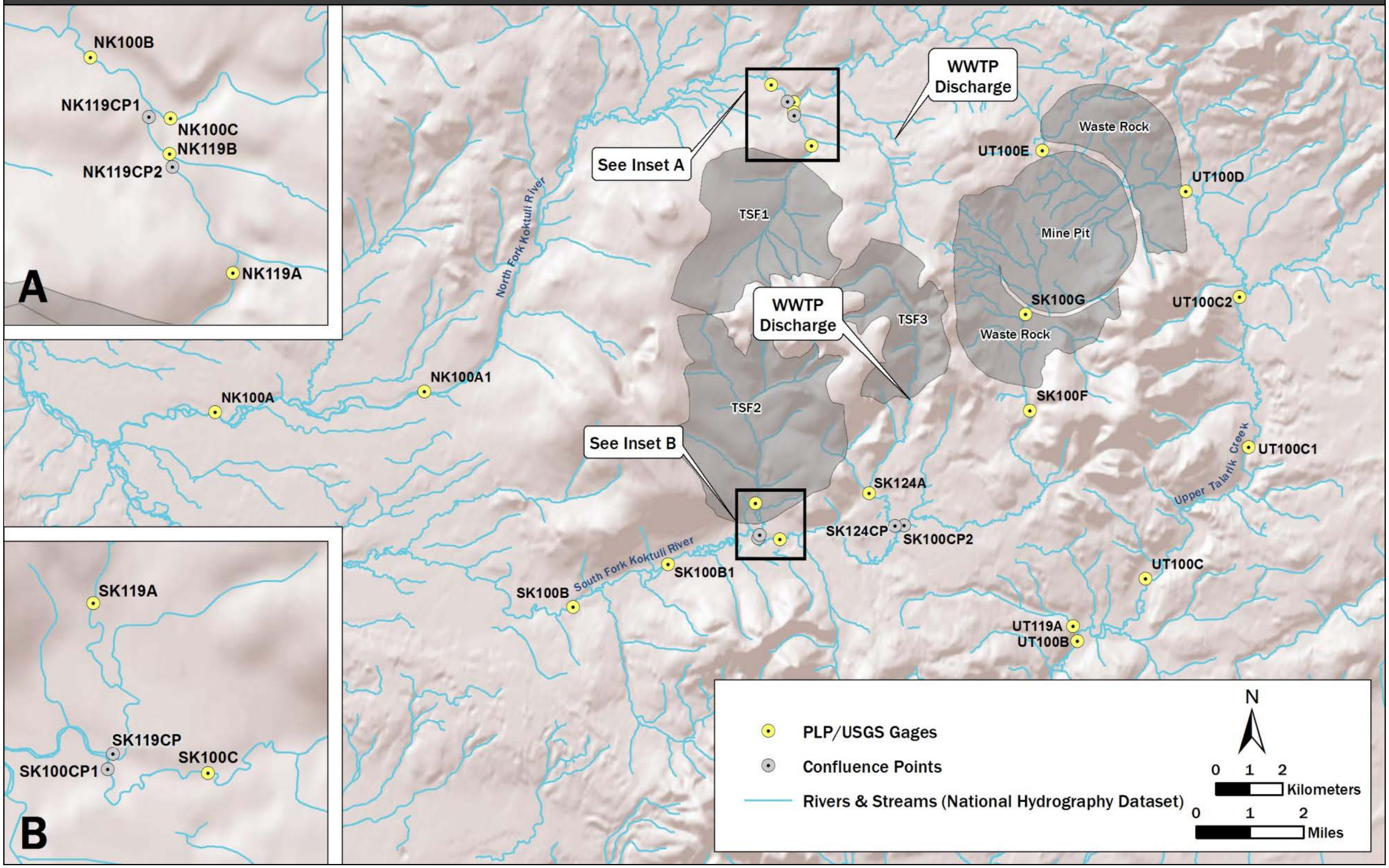


Figure 6-11. Approximate locations of stream gages and wastewater treatment plant (WWTP) discharges represented in Figures 6-8 through 6-10. Gages denoted with CP indicate confluence points, where virtual gages were created for analysis purposes. Footprint of the major mine components of the Pebble 6.5 scenario are shown for reference. Gage locations are based on U.S. Geological Survey (2012b) and Pebble Limited Partnership (2011).



6.3.1 Mine Pit

Upon mine closure, pit dewatering pumps would be turned off. The cone of depression would persist around the pit for a time, and groundwater would flow toward the pit in response to the local gradient. Eventually, the water level in the pit would recover toward equilibrium with the surrounding water table. Any water exiting the pit through surface channels or pumped from the pit would be tested, and treated if necessary, prior to discharge to surrounding surface waters. Based on our calculations for groundwater and precipitation inflows to the pit after operations have ceased, we estimate that the time required for the pit to fill ranges from approximately 20 years for the Pebble 0.25 scenario to more than 200 years for the Pebble 6.5 scenario. If additional runoff or TSF discharges were directed to the pit instead of allowed to flow into streams, these time frames would be considerably shorter (e.g., approximately 100 years for the Pebble 6.5 scenario).

Upper benches of the pit would be partially backfilled, regraded, covered with plant-growth medium, and vegetated. Some areas may be flattened to enable construction of wetlands for passive water treatment. At least portions of the pit walls, as well as rocks on the pit bottom or on side benches, would consist of mineralized rock that was not economical to mine. Any exposed rock containing sulfide minerals would likely be acid-generating for as long as it remained above the water surface in the pit, resulting in water with low pH and dissolved metals running down the sides of the pit into the water body at the bottom. As water level in the pit rose, 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. Surfaces anticipated to produce acidic drainage could be sealed against exposure to oxygen. However, this might not be effective for a pit of this size, since it might be difficult to seal all cracks and fissures in the pit walls. There could be degradation of sealants from exposure to sun and air, and freeze-thaw fracturing of rock could reduce acid-preventing efficacy over time. Predicting pit water quality has a high degree of uncertainty (Section 8.1.4; Appendix I) (Gammons et al. 2009), but water would need to be monitored and treated to meet effluent requirements prior to being discharged to streams, for as long as the water remained contaminated.

6.3.2 Tailings Storage Facilities

At closure, tailings beaches in the TSFs would be covered with NAG waste rock and a plant-growth medium, then vegetated with native species (Ghaffari et al. 2011). Embankments and crests also would be covered with a growth medium and vegetated. The tailings pond would be drawn down to prevent flooding and to maintain stability, but a pond of sufficient depth would be retained to keep the core of PAG tailings hydrated and minimize oxidation. Retaining water in the tailings maintains a higher potential for tailings dam failure than if the tailings were drained; however, draining the tailings to stabilize them could allow oxygen-rich water to percolate through the tailings and oxidize the sulfides. As long as a cover of water is maintained, oxygen movement into the tailings would be retarded, minimizing acid generation. Drawing down the water level 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. Additionally, wetlands might be included in reclamation to provide additional stormwater retention, passive water treatment, and significantly increased evapotranspiration (Reeve and Gracz 2008).

TSFs would require active management for hundreds to thousands of years (Blight 2010). A tailings dam is an engineered structure that requires 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, there appears to be little data available that document the magnitude of this stability gain. A recent analysis suggests that densification of oil sands tailings may stop after a period of time (Wells 2011). Although oil sands tailings are different from porphyry copper tailings, the principle is the same. Lack of data specific to porphyry copper tailings suggests a cautious approach, so we do not assume that tailings consolidate to a fully stable land form. Even if the tailings did consolidate over time, they would remain susceptible to erosion if the tailings dam were compromised. Thus, the system may require continued monitoring to ensure hydraulic and physical integrity in perpetuity.

6.3.3 Waste Rock

Some NAG waste rock would be used to cover tailings beaches, and some would be used to backfill upper portions of the mine pit. The remaining NAG waste rock would be sloped to a stable angle (e.g., less than 15 degrees [Blight and Fourie 2003]), covered with soil and plant-growth medium, and vegetated with native species. No PAG waste rock would remain on the surface, as it would have been processed either as blending material during operations or at the end of operations.

6.3.4 Water Management

Table 6-8 summarizes the flow components of the water balance after closure, both during the period in which the mine pit is filling and the steady state condition after the mine pit reaches its maximum water level. During the post-closure period, the mine would still capture water from precipitation over the mine pit, waste rock piles, and the TSFs. Groundwater would continue to flow into the mine pit, so precipitation over the cone of depression would continue to contribute to the captured water. Consumptive losses from operation would cease, but water stored in the mine pit would constitute a new consumptive loss until the mine pit water level reaches equilibrium with the surrounding groundwater level.

The footprint of the mine would be reduced as land occupied by production facilities is reclaimed. For purposes of estimating water inputs, we assume that 80% of the areas disturbed by the plant and ancillary facilities would be reclaimed, but that some facilities (e.g., the fuel depot, the WWTP, some pipelines, and part of the camp) would remain.

Table 6-8. Summary of annual water balance flows (million m^3/year) during the post-closure period for the Pebble 6.5 scenario.

Flow Component	During Mine Pit Filling	Post-Closure
Captured at mine pit area	39.7	26.1
Captured at TSF 1	13.0	13.0
Captured at TSF 2	18.4	18.4
Captured at TSF 3	7.76	7.76
Captured at mill & other facilities	0.538	0.538
Potable water supply well(s)	0	0
Water in ore (3%)	0	0
Total Captured	**79.4**	**65.8**
Cooling tower losses	0	0
Water in concentrate to port	0	0
Water in concentrate return	0	0
Runoff collected from port	0	0
Stored in TSF as pore water	0	0
Stored in mine pit	37.3	0
Crusher use	0	0
Total Consumptive Losses	**37.3**	**0**
Returned to streams via wastewater treatment plant	33.9	57.6
Returned as NAG waste rock leachate	0.947	0.947
Returned as PAG waste rock leachate	0	0
Returned as TSF leakage	7.20	7.20
Total Reintroduced	**42.1**	**65.8**
Percent of Captured Water Reintroduced	53.0%	100%

Notes:
TSF = tailings storage facility; NAG = non-acid-generating; PAG = potentially acid-generating.

As the mine pit fills, the cone of depression would shrink to the point that most or all of the waste rock would be outside of the drawdown zone. Runoff from the reclaimed NAG waste rock piles would either seep into the ground, travel as overland flow, or be diverted to streams. Some precipitation would be expected to infiltrate through the NAG waste rock cover, drain through the waste rock piles, and become groundwater. Runoff from the reclaimed NAG waste rock piles is not anticipated to require treatment, but would be monitored periodically to confirm this assumption.

The elevation of the north rim of the Pebble 6.5 mine pit would be over 100 m higher than the elevation of the south rim, so that even when the mine pit reaches its maximum water level there would still be seepage into the pit from the higher ground. For water balance purposes, we estimate that the post-closure cone of depression would extend an average of 100 m beyond the pit rim as a result of surface outflow or pumping.

Precipitation falling on the post-closure tailings would be monitored and discharged downstream or diverted for treatment in the WWTP, as necessary, to meet water quality standards. Stormwater diversions and collection systems from the operations phase would be maintained and water directed away from the TSF, or, if risk of contamination existed, toward the WWTP for treatment prior to discharge to streams. Interstitial water within the tailings would continue to seep into naturally fractured bedrock below the TSF. The well field placed downstream from the TSF during operations would be retained and monitored post-closure, with water pumped and treated if determined to be contaminated by leachate from the TSF. The pit water would be monitored and treated prior to being released to streams, for as long as concentrations of contaminants exceeded effluent limits.

6.3.5 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 have closed prematurely in Alaska.

Closure before originally planned—that is, premature closure—may occur for many reasons, including technical issues, project funding, deteriorating markets, operational issues, or 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 fully reclaimed or equivalent to those under a planned closure, may be severely contaminated and require extensive remediation, or may fall anywhere between these extremes. Environmental impacts associated with premature closure may be more significant than impacts 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 re-opening of the mine. For example, PAG waste rock in our mine scenarios would likely still be on the surface in the event of a premature closure. If the mine closed because of a drop in commodity price, there would be little economic incentive to incur the cost of moving or processing millions of metric tons of PAG waste rock, and water treatment systems might be insufficient to treat the volume of low pH water containing high metal concentrations from this previously unplanned source. Some method of financial assurance generally is required by state and federal agencies to ensure closure if a mine company defaults on its responsibility (Box 4-3). To be effective, financial assurance must be based on accurate estimates of reclamation costs. In the past, financial assurance often has not been adequate, and taxpayers have been left with substantial cleanup costs (USEPA 1997). This may be changing, as agencies update bonding requirements to reflect cleanup costs more accurately, but projecting these costs far into the future is a difficult task.

When a mine re-opens after premature closure, the owners might change the mining plan, implement different mitigation practices, or negotiate new effluent permits. An example is the Gibraltar copper mine in British Columbia. The Gibraltar mine began operations permitted as a zero-discharge operation. However, when it was re-opened under new ownership after having closed prematurely, the new permit allowed treated water to be discharged to the Fraser River with a 92-m dilution zone for copper and other metals.

6.4 Conceptual Models

The development of conceptual models is a key component of the problem formulation stage of an ecological risk assessment (USEPA 1998), and in Chapter 2 we introduced the use of conceptual models as tools to help structure ecological risk assessments. At the outset, we broadly define the scope of this assessment to be potential effects of a large-scale mine and a transportation corridor on freshwater habitats, resulting effects on fish, and consequent fish-mediated effects on wildlife and Alaska Native populations (Section 2.2.1, Figure 2-1). To conduct a risk analysis, this scope needs to be refined and the specific sources, stressors, and endpoints to be evaluated must be explicitly identified.

In this section, we summarize the specific sources, stressors, and endpoints considered in the assessment, as informed by the background information on the region, type of development, and endpoints of interest presented in the preceding chapters, and based on the mine scenarios described in this chapter. We then integrate these components into conceptual model diagrams that illustrate hypothesized cause-effect linkages among these sources, stressors, and endpoints.

6.4.1 Sources Evaluated

The two main sources considered in this assessment are the mine and the transportation corridor, each of which can be subdivided into several components. These components are summarized below, and discussed in greater detail in Section 6.1.

- The **mine infrastructure** includes the major mine components (open mine pit, waste rock piles, TSFs), the groundwater drawdown zone associated with the mine pit, and plant and ancillary facilities (e.g., water collection and storage facilities, a WWTP, ore-processing facilities, and chemical storage facilities).
- The **transportation corridor** comprises a road and four pipelines (one each for product slurry, diesel fuel, natural gas, and return water) connecting the mine site area to Cook Inlet.

6.4.2 Stressors Evaluated

As discussed above and in Chapter 4, large-scale mining is a complex process that typically involves both physical alteration of the environment and the release of pollutants. The specific stressors considered for inclusion in the assessment were identified based on their potential to significantly affect our primary endpoint of interest—the region's salmon resources—and their relevance to the U.S. Environmental Protection Agency's (USEPA's) regulatory authority and decision-making context. Stakeholders also identified potential stressors of concern, which were considered by the assessment team. These stressors are summarized in Table 6-9 and discussed in detail below. Those stressors that are analyzed in the assessment or are of particular concern to stakeholders are discussed in the following subsections.

Table 6-9. Stressors considered in the assessment and their relevance to the assessment's primary endpoint (salmonids) and the U.S. Environmental Protection Agency's regulatory authority.

Stressor	Description	Relevance to Salmonids	Relevance to Decision-Making
Excavation	Removal of streams and wetlands due to creation of the mine pit and other excavations.	Relevant	Directly relevant to Section 404 of the Clean Water Act
Filling	Filling in of streams and wetlands due to waste rock piles, tailings impoundments, and roads.	Relevant	Directly relevant to Section 404 of the Clean Water Act
Water diversion and withdrawal	Reduced flow in streams and wetlands due to removal of water.	Relevant	Consequence of excavation and filling
Water temperature	Changes in water temperature associated with discharges of treated water or reduced groundwater flows.	Relevant	Consequence of excavation and filling
Product metal (copper)	Copper occurring in the product concentrate, waste rock, or tailings could enter streams and wetlands.	Relevant	Consequence of excavation and filling
Other metals	Metals other than copper occurring in the product concentrate, waste rock, or tailings could enter streams and wetlands.	Relevant	Consequence of excavation and filling
pH	Oxidation of sulfides could result in acidification of waste and receiving waters.	Relevant	Consequence of excavation and filling
Process chemicals	Chemicals used in ore processing would occur in tailings and product concentrate and could spill.	Relevant	Consequence of excavation and filling
Nitrogen	Nitrogen compounds released during blasting would deposit on the landscape. Nitrates could also reach groundwater via leachate from waste rock piles.	Weakly relevant	Consequence of excavation and filling
Tailings and other fine sediment	Tailings, product concentrate, and other fine particles could fill streams or wetlands or, at lower concentrations, could change substrate texture and abrade fish gills.	Relevant	Directly relevant to Section 404 of the Clean Water Act (if particles act as fill) and consequence of excavation and filling
Diesel fuel	Spilled diesel fuel could enter streams and wetlands.	Relevant	Necessary for excavation and filling
Natural gas	Leaking natural gas could combust.	Not relevant	Peripheral to excavation and filling
Dust	Dust from blasting, tailings beaches, and vehicle traffic could deposit on the landscape and wash into streams.	Weakly relevant	Consequence of excavation and filling
Noise	Noise from blasting or other activities.	Not relevant	Consequence of excavation and filling
Rock slide	Slides from waste rock piles or roads.	Relevant	Consequence of excavation and filling
Blocked or perched culvert	Inhibition of fish passage due to malfunctioning culverts.	Relevant	Consequence of excavation and filling for a road
Washed out culvert	Downstream siltation or inhibition of fish passage due to washed out culverts.	Relevant	Consequence of excavation and filling for a road
Invasive plants	Changes in habitat quality due to invasion by plants carried by road traffic.	Weakly relevant	Peripheral to excavation and filling
Climate change	Altered risk of mine failures, and changes in marine and freshwater habitat quality and life history timing, associated with increased precipitation and temperature.	Indirectly relevant	Not related to excavation and filling, but modifies other consequences of excavation and filling

6.4.2.1 Physical Habitat Alteration

Large-scale mining in the Bristol Bay region would necessarily involve the destruction of streams and wetlands through excavation and filling associated with the mine pit, waste rock piles, TSFs, borrow pits, and the transportation corridor. This excavation and filling would directly affect anadromous and resident salmonid habitats and directly involve USEPA under Section 404 of the Clean Water Act.

Mining-related excavation and filling would also result in water diversion and withdrawal. Stream and overland flow must be diverted around the mine site to keep it dry and minimize erosion; the mine pit must be dewatered to continue excavation; and water must be obtained for use in ore processing, tailings and product transport, and other purposes. These diversions and withdrawals would redirect and reduce flow and plausibly affect fish via reduced habitat quality or quantity.

6.4.2.2 Water Temperature

Stream and wetland water temperatures could be affected by the capture, storage, use, treatment, and discharge of water throughout the mining process. Elevated temperatures could result from warm water discharges or, in summer, from reduced groundwater inputs. In winter, reduced groundwater inputs could result in reduced temperatures. Because water temperature affects fish development and habitat, any temperature changes could plausibly influence fish populations.

6.4.2.3 Chemical Contaminants

A range of chemical contaminants associated with mining may enter surface waters and pose risks to fish. These contaminants include rock-derived inorganic contaminants (metals and acidity), ore-processing chemicals, fuels, and nitrogen compounds.

Rock-Derived Inorganic Contaminants

Mines are developed because rocks at the site have high metal concentrations, which are further concentrated as ore is isolated from waste rock and as product concentrate is created from the ore. These metals may enter surface waters from uncollected leachate and runoff, from WWTP discharges, or from spills of product concentrate and its associated water. Metals are known to cause toxic effects on aquatic biota, including fish; however, when combined with low pH (acidity), metals become especially problematic. Acid rock drainage occurs when PAG rocks are present at the mine site. Acidity can be directly deleterious to aquatic biota, but it also increases the solubility of minerals, which results in increased concentrations of metals in solution.

Because copper is the major resource metal in the Pebble deposit and is particularly toxic to aquatic organisms, it is the metal most likely to cause toxic effects at this site. Copper toxicity also has been a primary concern of stakeholders, including the National Oceanic and Atmospheric Administration, the federal agency responsible for salmon management. Thus, copper criteria, standards, and toxicity are considered in detail in the assessment.

Other metals are considered if their concentrations in test leachates from the Pebble deposit indicate that they are potentially toxic, based on benchmark values. When possible, national ambient water quality criteria are used as screening benchmarks. Both criterion maximum concentrations (CMCs) and criterion continuous concentrations (CCCs) are used to account for acute and chronic exposures, respectively. When U.S. criteria are not available, the most similar available value is used (e.g., Canadian benchmarks, the lowest acute and chronic values from the USEPA's ECOTOX database, or the European Chemical Agency and Organization for Economic Cooperation and Development's eChemPortal) (Table 6-10).

Table 6-10. Screening benchmarks for metals with no national ambient water quality criteria.

Metal	Acute/Chronic Benchmarks (μg/L)	Source and Notes
B	29,000/1,500	Canadian acute and chronic guidelines based on SSDs (CCME 2009)
Ba	46,000/8,900	*Austroptamobius pallipes* 96-hour LC_{50} (Boutet and Chaisemartin 1973) and *Daphnia magna* 21-day reproductive EC_{50} (Biesinger and Christensen 1972)
Co	89/2.5	Acute value is the lowest acute test datum and the chronic value is the 5th centile of a chronic species sensitivity distribution (Environment Canada and Health Canada 2011)
Fe[a]	350/-	Chronic data were inadequate to set a value, but the Canadian authors believed that it would not be much lower than this acute value (BC 2008)
Mn	760/693	Hardness adjusted (for 20 mg/L) acute and chronic guidelines (BC 2001)
Mo	32,000/73	*Daphnia magna* 48-hour LC_{50} (Kimball 1978) and Canadian chronic guideline (CCME 1999)
Sb	14,400/1,600	Lowest acute and chronic values from a fathead minnow early life-stage test (USEPA 1980, Swedish Chemicals Inspectorate 2008, Environment Canada and Health Canada 2010)

Notes:
[a] The listed U.S. iron criterion, from the 1976 Red Book (USEPA 1976), is less reliable than this more recent benchmark.
SSD = species sensitivity distribution; LC_{50} = median lethal concentration; EC_{50} = median effective concentration.

Some metals, such as calcium, magnesium, and sodium, are not screened because of their low toxicity. Molybdenum is treated as a contaminant of concern because it is a specific product of the mine, even though it would not be retained based on the comparison of test leachates with benchmark values. Molybdenum concentrate would be trucked to the port, and spills of the sand-like material could occur. Gold is also a product, but is not evaluated because it has very low solubility and toxicity and would not be transported in a form likely to result in aqueous exposures.

Screening against tailings and waste rock leachates are presented in Tables 8-4 through 8-8. The metals of concern are aluminum, cadmium, cobalt, copper, manganese, nickel, lead, selenium, and zinc based on average concentrations exceeding either acute or chronic benchmarks for at least one leachate. However, most of the estimated total toxicity is due to copper.

Major Ions (Total Dissolved Solids)

Total dissolved solids (TDS) comprise all organic and inorganic materials dissolved in a water sample, which can be measured directly or estimated from conductivity measurements (specific conductance is the term for conductivity values that have been temperature-compensated to 25°C). Mining inevitably involves crushing rocks, and the leaching of crushed rock results in enhanced dissolution and elevated concentrations of dissolved major ions (calcium, magnesium, sodium, potassium, chlorine, sulfate, and bicarbonate). These major ions generally contribute the most mass to TDS measurements, especially sulfate in waters influenced by metal mining. Some metals, such as calcium, magnesium, potassium, and sodium are not screened because of their low toxicity, but they contribute to ionic stress. Thus, even if this mixture of TDS is not acidic, it can be toxic to aquatic biota, particularly in this region's waters, which have low ambient concentrations of these ions. Examples of toxicity due to leaching of major ions from mine-derived waste rock are discussed in USEPA (2011) and Chapman et al. (2000). Also, the history of TDS compliance problems at the Red Dog Mine near Kotzebue, Alaska, suggests that dissolved major ions should be a stressor of concern.

Ore-Processing Chemicals

Chemicals used to process the ore and separate product from tailings have the potential to enter the environment as a result of truck wrecks, on-site spills, tailings slurry spills, product concentrate slurry spills, or water collection and treatment failures. Tests of the Pebble deposit ore used alkaline flotation to separate product concentrate from tailings (Ghaffari et al. 2011). The collector was sodium ethyl xanthate, the frother was methyl isobutyl carbinol, and lime was used to adjust pH. Molybdenum separation also requires fuel oil as a collector (Box 4-5). Of these, xanthate is clearly a contaminant of concern because it is highly toxic to aquatic life (Hidalgo and Gutz 2001). Methyl isobutyl carbinol has been poorly tested but appears to have relatively low toxicity (acute lethality to African clawed frogs and goldfish requires a relatively high concentration, 360 to 656 mg/L [USEPA 2013]). Lime would contribute to the risk from major ions (TDS). Fuel oil use for this purpose would be small relative to its use as fuel.

In addition, cyanide might be used to recover gold from pyritic tailings (Box 4-6). It is expected that a cyanide destruction unit would be used at the end of the leaching process to achieve the acute and chronic water quality criteria for free cyanide of 22 and 5.2 μg/L, respectively. Cyanide in the TSF is likely to be rapidly diluted and degraded. Accidental releases and on-site spills, as recently occurred at the Fort Knox mine (ADEC 2012), are possible but are not judged to be as directly significant to our endpoints as other accidents considered. However, because cyanide is assumed to be transported as a solid, as is common at other mines, truck accidents could result in cyanide spills to streams.

Fuels

Both diesel oil and natural gas would be piped to the mine site and could enter the environment via pipeline leaks or failures. Diesel spills could enter surface waters and have been known to adversely affect aquatic biota, so diesel is considered in the assessment. Natural gas could combust, but a natural gas fire is unlikely to significantly affect salmon populations.

Nitrogen Compounds

Nitrogen compounds, expected to be predominantly nitrate due to combustion, would be released during the blasting associated with excavation. Some of these compounds would deposit on waste rock piles and the landscape and could enter surface water and groundwater. However, it is likely that these streams are phosphorus-limited, not nitrogen-limited (Goldman 1960, Moore and Schindler 2004), and the consequences of an increase in nitrogen/phosphorus ratio for salmonids are unknown but judged to be minimal. Thus, nitrogen residues are not considered in the assessment.

6.4.2.4 Fine Sediment

If tailings, product concentrate, unpaved road materials, or other fine particles are spilled or eroded, they could fill streams and wetlands, alter streambed substrates, or abrade the gills of fish.

6.4.2.5 Dust

Blasting and vehicle traffic, both at the mine site and along the transportation corridor, would generate dust. Exposed tailings beaches within the TSFs also could result in dust generation. This dust could contribute to the sedimentation of streams and, depending on the composition of the rock, could contribute toxic metals to surface waters. Dust from unpaved roads is known to affect streams, so it is included in this assessment. In contrast, the occurrence of dust from blasting and tailings beaches is poorly documented, highly site-specific, and its effects are unknown. We anticipate that much of the dust generated from blasting and tailings beaches would settle on the site and be collected with runoff water. Wind may carry dust off site, but would also disperse it across the landscape. We do not judge dust from blasting or tailings beaches to be an important contributor to risks to salmonids (although this judgment is uncertain), and do not consider it in the assessment.

6.4.2.6 Noise

Noise would be generated by blasting at the mine site and vehicle traffic along the transportation corridor. Although noise may directly affect wildlife, it is unlikely to affect salmonids and is not considered in the assessment.

6.4.2.7 Culverts

Blocked or perched culverts could significantly reduce fish passage, thereby reducing salmon migrations or movement among habitats by resident salmonids. Culverts also may wash out during floods, temporarily inhibiting fish movement and reducing habitat due to siltation by the deposited roadbed material. Culverts are a component of roads that fill wetlands and the floodplains of streams. They may significantly affect salmon in the surface waters they intersect and thus are considered in the assessment.

6.4.2.8 Invasive Species

Several dozen species of plants, animals, and micro-organisms are considered to be or have the potential to be invasive in Alaska (ADF&G 2013, Eddmaps 2013). Of those currently present, reed canarygrass (*Phalaris arundinacea*) is widespread on the Kenai Peninsula (HSWCD 2007) and elodea (*Elodea canadensis*) exists in Stormy Lake on the northern Kenai Peninsula (Etcheverry 2012). These plants have the potential to degrade salmon habitat (Merz et al. 2008). The improved and expanded road from Cook

Inlet may facilitate the spread of reed canary grass and elodea from the Kenai Peninsula to the Bristol Bay watershed, where they may adversely affect salmon habitat.

6.4.3 Endpoints Evaluated

In this assessment, the primary endpoint of interest is the region's key salmonid populations (Pacific salmon, rainbow trout, and Dolly Varden) in terms of abundance, productivity, or diversity. Given the importance of salmonids to the region's ecosystems and culture, we also consider the effects of potential changes in fish populations on wildlife abundance, productivity, or diversity and on Alaska Native culture. These endpoints are discussed in detail in Chapter 5.

6.4.4 Conceptual Model Diagrams

To frame the assessment, we developed conceptual model diagrams illustrating potential pathways linking the sources, stressors, and endpoints detailed above (see Box 2-1 for an overview of how the assessment's conceptual models are structured). These diagrams went through several iterations, from initial brainstorming of all potential pathways associated with large-scale mine development in the Bristol Bay region (both with the assessment team and other stakeholders) to focusing on those pathways considered both within the assessment's scope (Chapter 2) and likely to affect endpoints of interest.

Through this iterative process, we developed a series of three conceptual model diagrams illustrating hypothesized cause-effect relationships leading from mine-related sources to endpoints of interest. These diagrams illustrate potential effects of routine mine construction and operation on physical habitat (Figure 6-12), potential effects of routine mine construction and operation on water chemistry (Figure 6-13), and potential effects of unplanned events on physical habitat and water chemistry (Figure 6-14). Note that the distinction between physical habitat and water chemistry was made for presentation purposes, though we recognize that water chemistry can be an important component of the physical habitat. These diagrams provide a framework for the analysis sections of the assessment, and the relevant portions of these diagrams evaluated in each analysis section are highlighted throughout the remaining chapters of the assessment. Note that not all pathways included in each conceptual diagram are necessarily evaluated in the assessment. For example, in some cases, we hypothesized pathways that may be significant, but data were not sufficient for quantitative analysis.

We also developed three more general conceptual model diagrams for specific topics (wildlife, Alaska Native cultures, and cumulative effects of multiple mines) that were defined as outside of the assessment's scope but that are of key importance to stakeholders (Chapters 12 and 13).

Figure 6-12. Conceptual model illustrating potential effects of routine mine construction and operation on physical habitat.

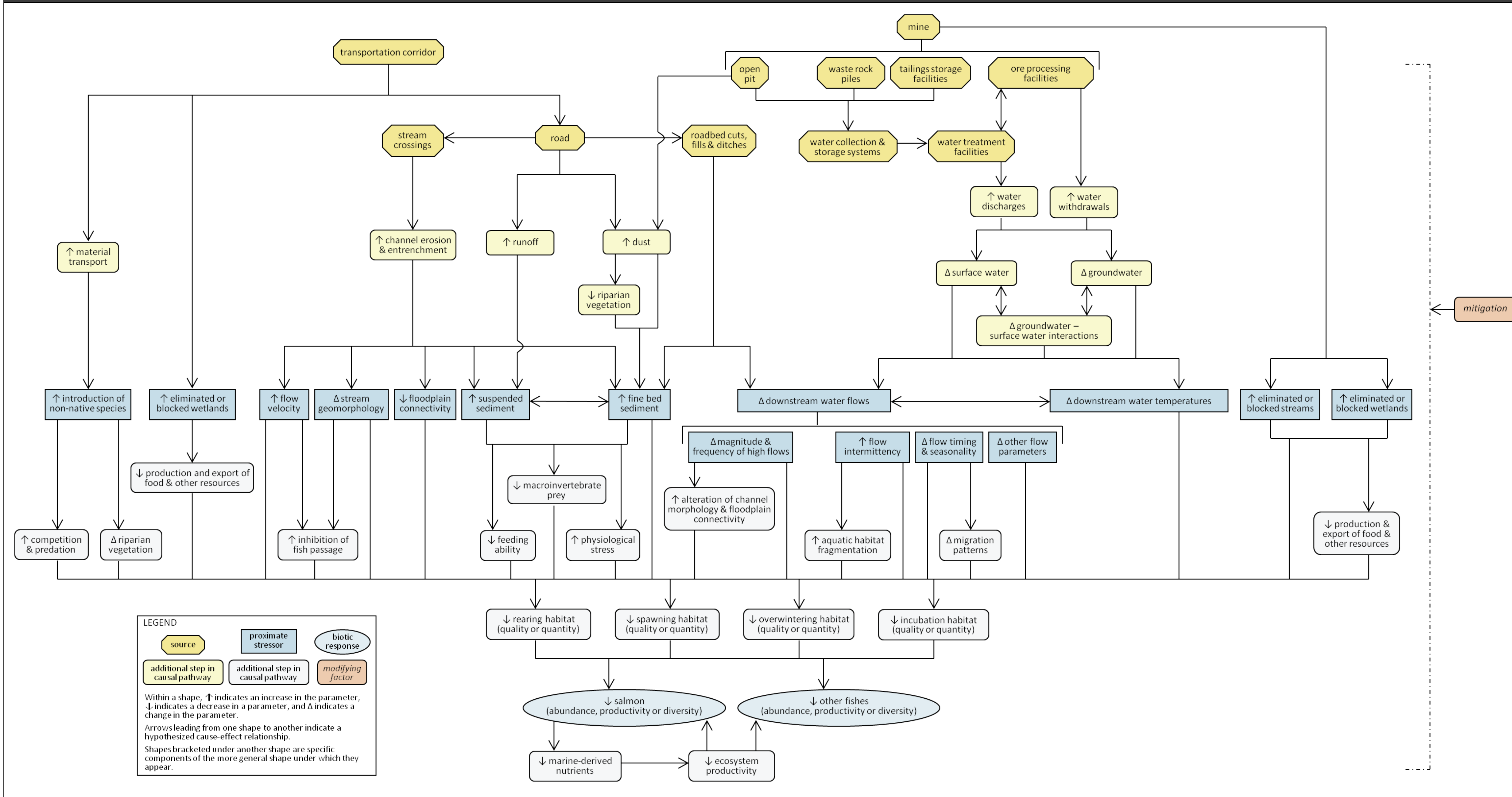


Figure 6-13. Conceptual model illustrating potential effects of routine mine construction and operation on water chemistry.

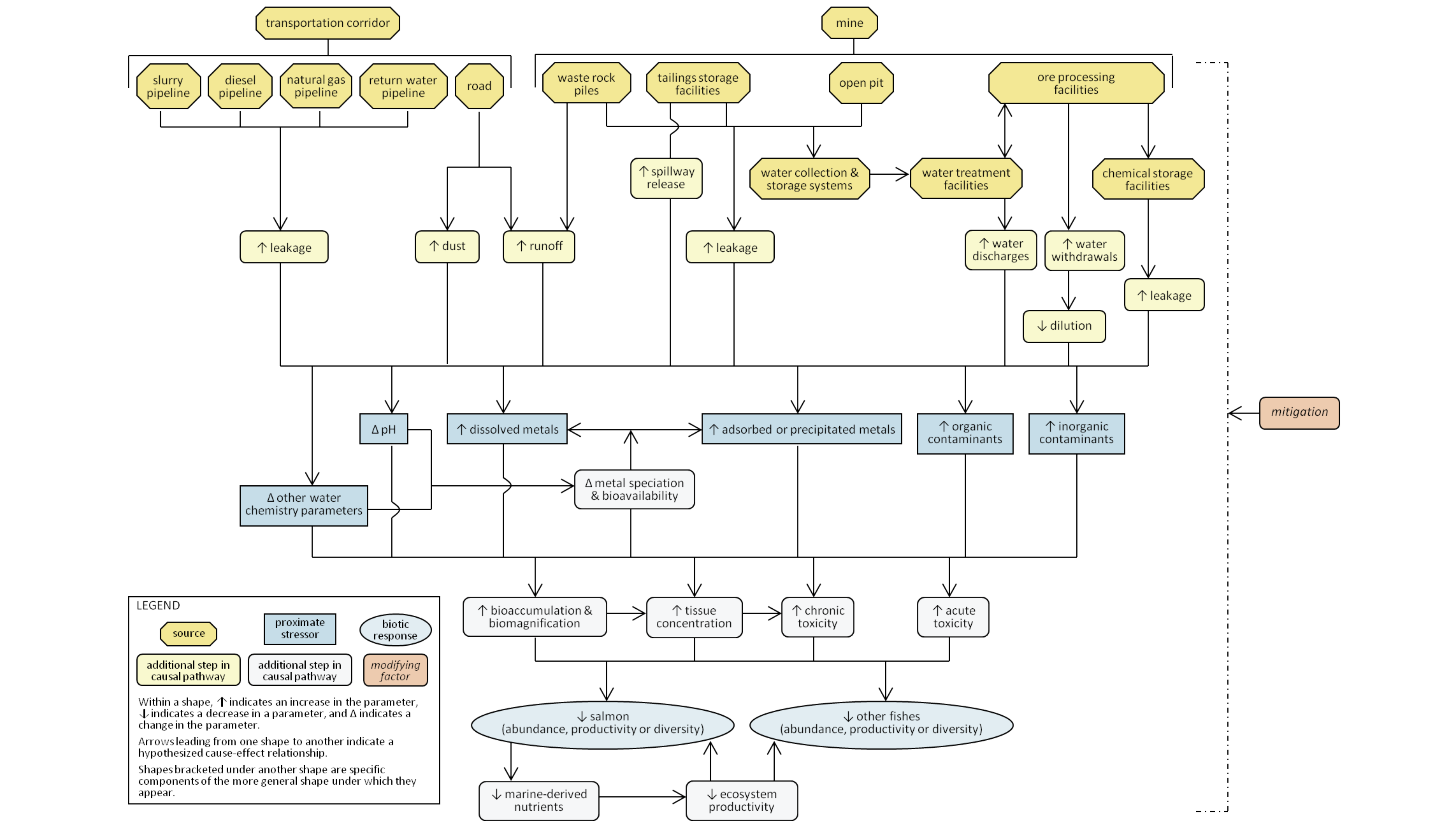
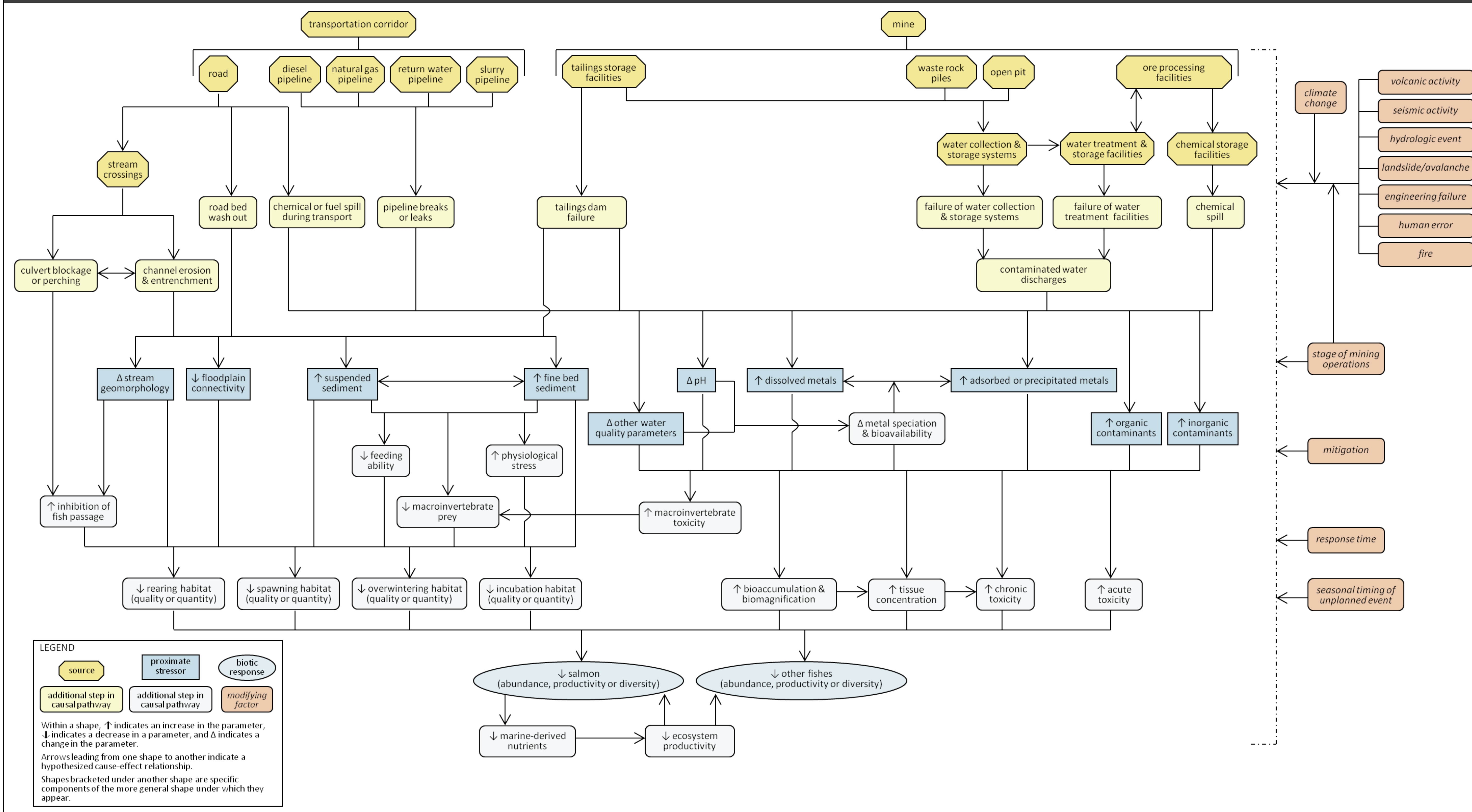


Figure 6-14. Conceptual model illustrating potential effects of unplanned events on physical habitat and water chemistry.





CHAPTER 7. MINE FOOTPRINT

This chapter addresses the stream habitat and streamflow risks associated with routine operations of the mine scenarios described in Chapter 6. It considers the unavoidable environmental effects associated with the footprint of each mine scenario, in the absence of failures of water collection or treatment facilities, tailings storage facilities (TSFs), the transportation corridor, or pipelines. This is not meant to suggest that the absence of failures is a realistic possibility, because accidents and failures do happen in complex and long-lasting operations. The risks and potential impacts of these failures are described in Chapters 8, 9, 10, and 11. In this chapter we evaluate the inevitable effects of the mine scenarios, rather than those that are the result of accidents and failures.

Potential pathways linking mine components, stream habitat and streamflow alterations, and biotic responses are highlighted in Figure 7-1. Key stressors associated with routine mine development and operation include elimination and modification of habitat (Section 7.2) and changes in downstream streamflow (Section 7.3), both of which can affect fish populations. The pathways associated with stream and wetland elimination highlighted in Figure 7-1 primarily reflect linkages occurring within the spatial extent of the mine footprint (Scale 4). Linkages and effects associated with streamflow alterations primarily operate from the edge of the footprint downstream to the extent of detectable streamflow changes (Scale 3). Effects on fish populations due to these modifications could extend beyond these geographic scales and into the larger Nushagak and Kvichak River watersheds (Scale 2), depending on the types and severity of impacts; these effects could not be quantified and are discussed qualitatively (see also Chapter 14). Routine effects of water collection, treatment, and discharge and the transportation corridor are discussed in Chapters 8 and 10, respectively.

7.1 Abundance and Distribution of Fishes in the Mine Scenario Watersheds

Potential effects of the mine footprint (addressed in this chapter) and of routine mine operations and failures (addressed in Chapters 8 through 11) on the assessment endpoints depend on the abundance and distribution of salmonids in the streams and rivers of the three watersheds draining the Pebble deposit area: the South Fork Koktuli River, North Fork Koktuli River, and Upper Talarik Creek watersheds (hereafter referred to as the mine scenario watersheds).

7.1.1 Fish Distribution

The mine scenario watersheds have been sampled extensively for summer fish distributions over several years. These data, collected by the Alaska Department of Fish and Game (ADF&G) and various consultants and non-profits, are captured in the *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region* (also known as the Anadromous Waters Catalog [AWC]) (Johnson and Blanche 2012) and the Alaska Freshwater Fish Inventory (AFFI) (ADF&G 2012). The AWC is the State of Alaska's official record of anadromous fish distributions and, if available, the life stages present (categorized as spawning, rearing, or present but life stage unspecified) in individual stream reaches. The AFFI includes all fish species, including resident fishes, found at specific sampling points. The catalogued distributions of the five Pacific salmon species (sockeye, coho, Chinook, chum, and pink), Dolly Varden (both anadromous and non-anadromous forms are present), and resident rainbow trout in the mine scenario watersheds are shown in Figures 7-2 through 7-8. In addition, Alaskan or Arctic brook lamprey, longnose sucker, northern pike, humpback whitefish, least cisco, round whitefish, Arctic char, Arctic grayling, burbot, threespine stickleback, ninespine stickleback, and slimy sculpin occur in these watersheds (ADF&G 2012). Details of these species, including information on distributions, abundances, habitats, life cycles, predator-prey relationships, and harvests, are provided in Appendix B. AWC stream reach designations and AFFI observation points should be interpreted with care, because not all streams could be sampled and there are potential errors associated with fish identification and mapping. Additional caveats and uncertainties concerning interpretation of AWC and AFFI data are discussed in Section 7.2.5.

Figure 7-1. Conceptual model illustrating potential linkages between sources associated with the mine scenario footprints, changes in physical habitat, and fish endpoints.

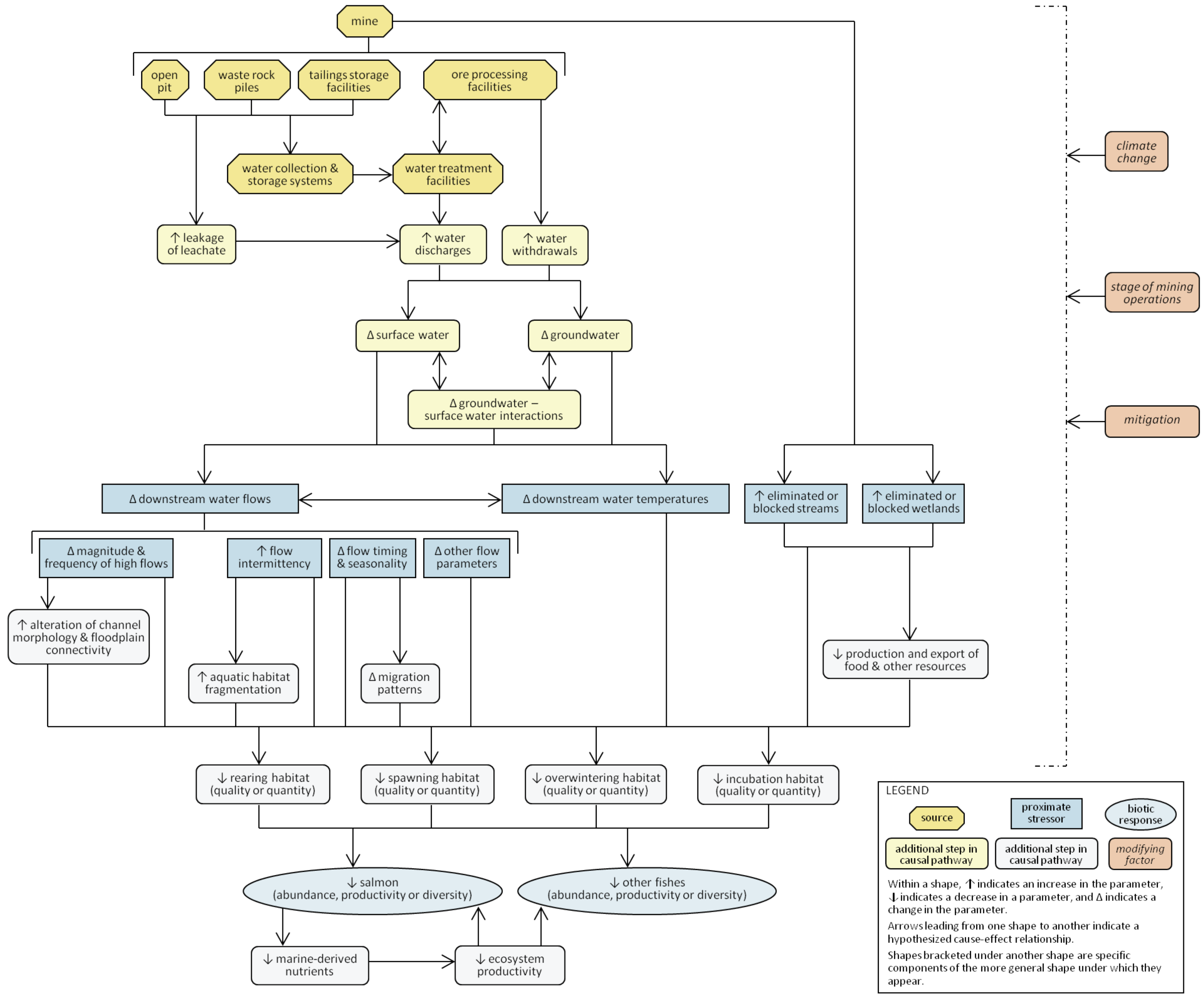


Figure 7-2. Reported sockeye salmon distribution in the mine scenario watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year (see Section7.2.5 for additional notes on interpretation of fish distribution data). Footprints of the major mine components for the three mine scenarios and the drawdown zone for the Pebble 6.5 scenario are shown for reference.

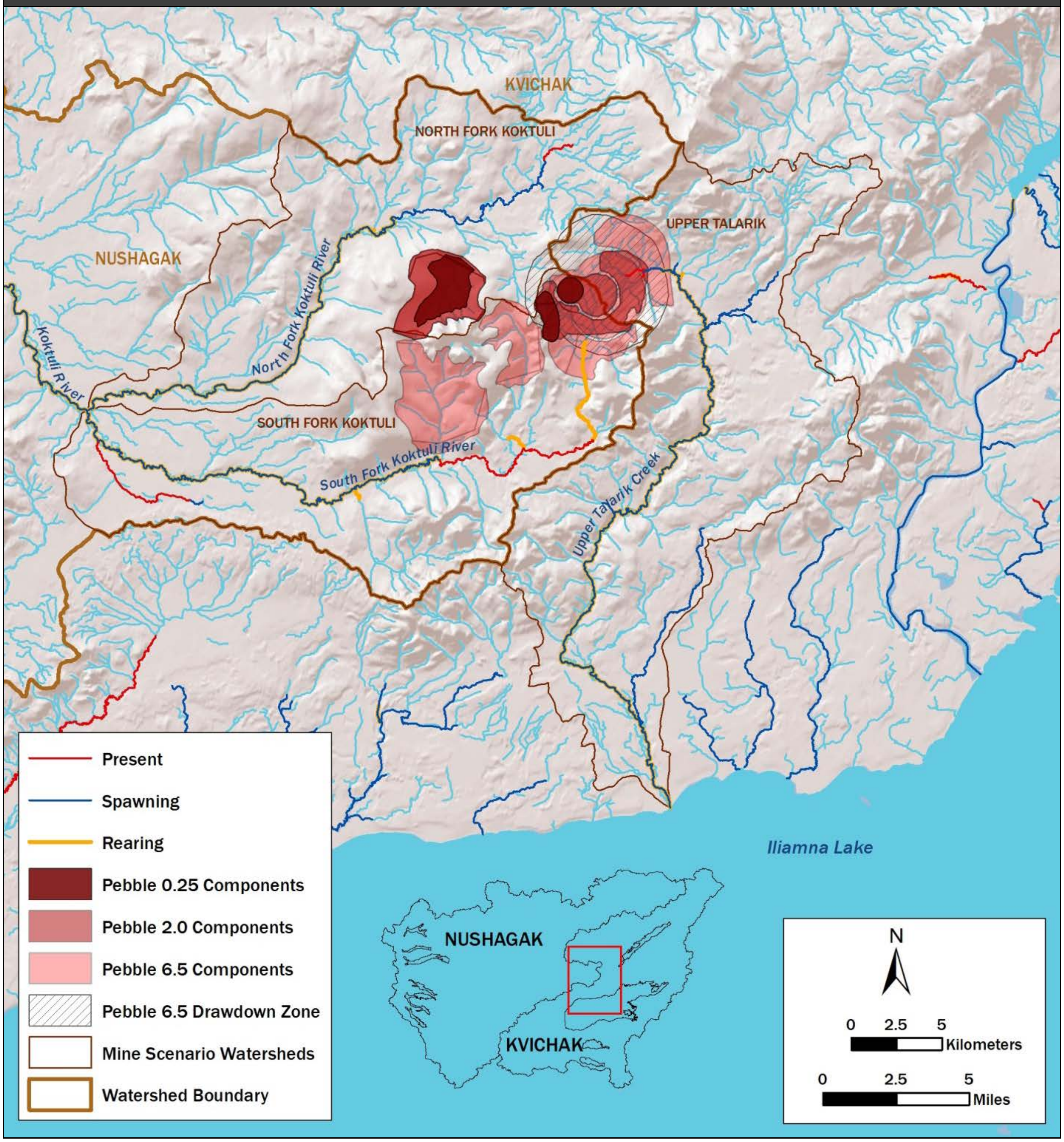


Figure 7-3. Reported coho salmon distribution in the mine scenario watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year (see Section 7.2.5 for additional notes on interpretation of fish distribution data). Footprints of the major mine components for the three mine scenarios and the drawdown zone for the Pebble 6.5 scenario are shown for reference.

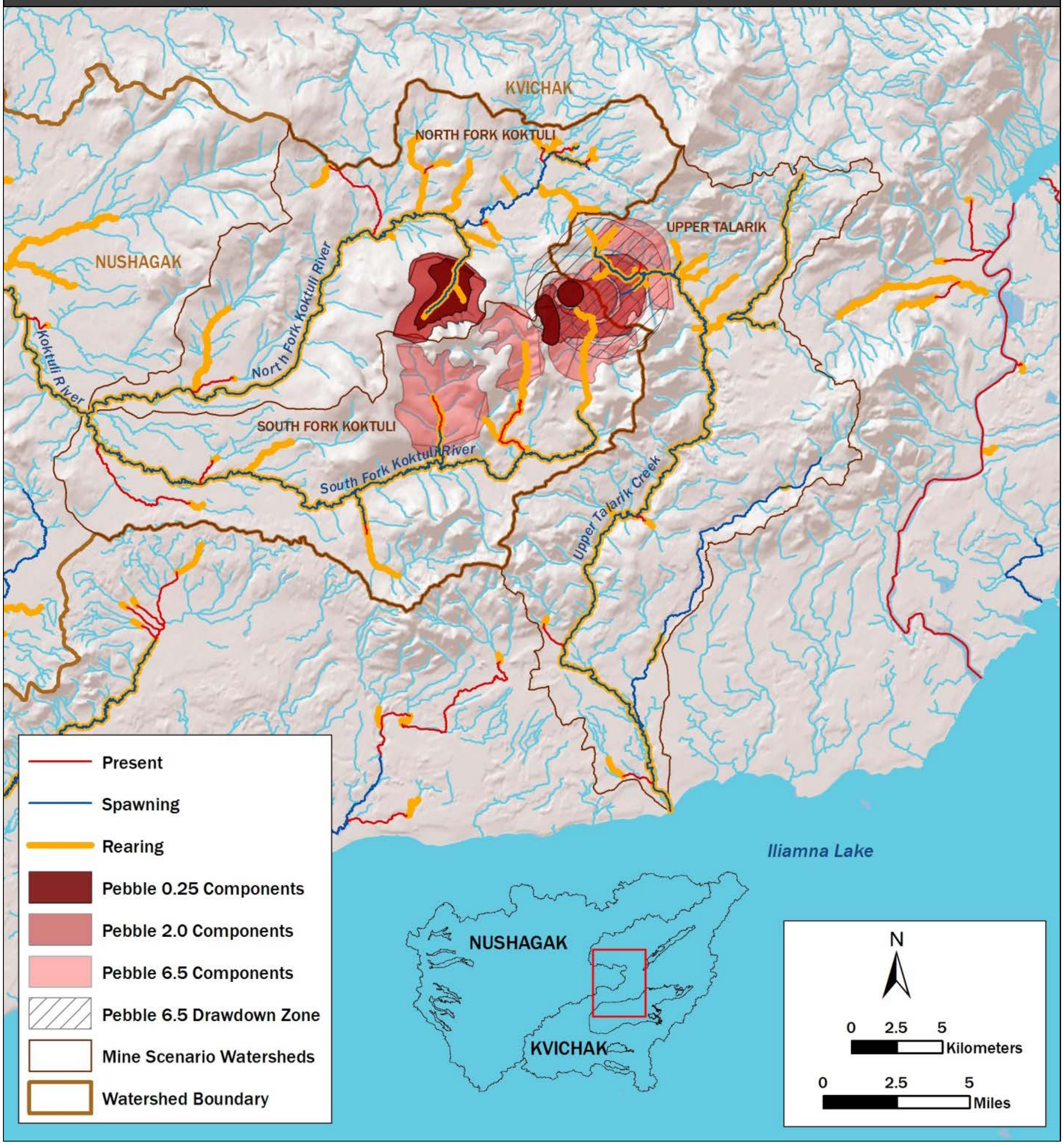


Figure 7-4. Reported Chinook salmon distribution in the mine scenario watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year (see Section 7.2.5 for additional notes on interpretation of fish distribution data). Footprints of the major mine components for the three mine scenarios and the drawdown zone for the Pebble 6.5 scenario are shown for reference.

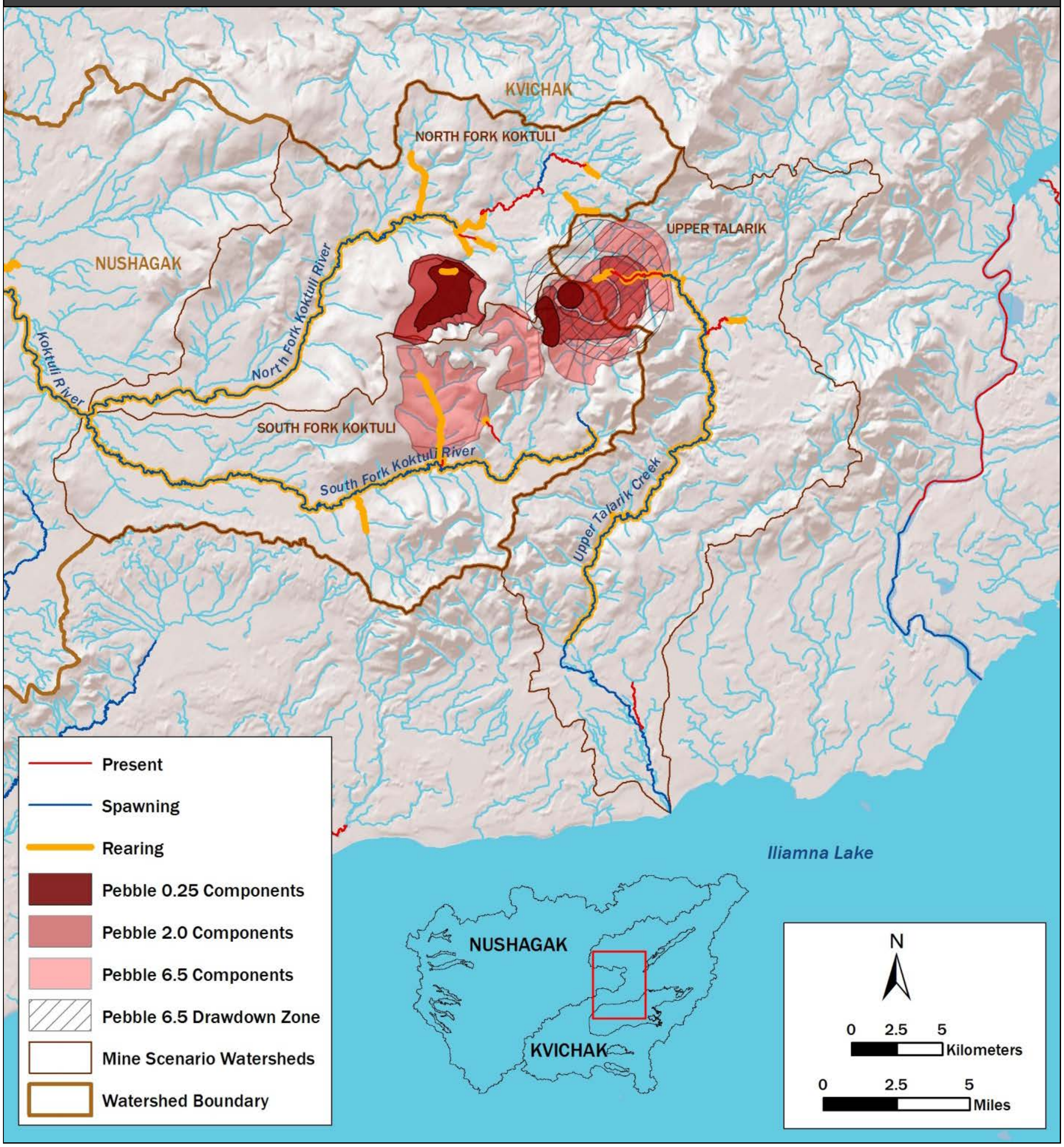


Figure 7-5. Reported chum salmon distribution in the mine scenario watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed; "rearing" indicates juveniles were observed. Present, spawning, and rearing designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year (see Section 7.2.5 for additional notes on interpretation of fish distribution data). Footprints of the major mine components for the three mine scenarios and the drawdown zone for the Pebble 6.5 scenario are shown for reference.

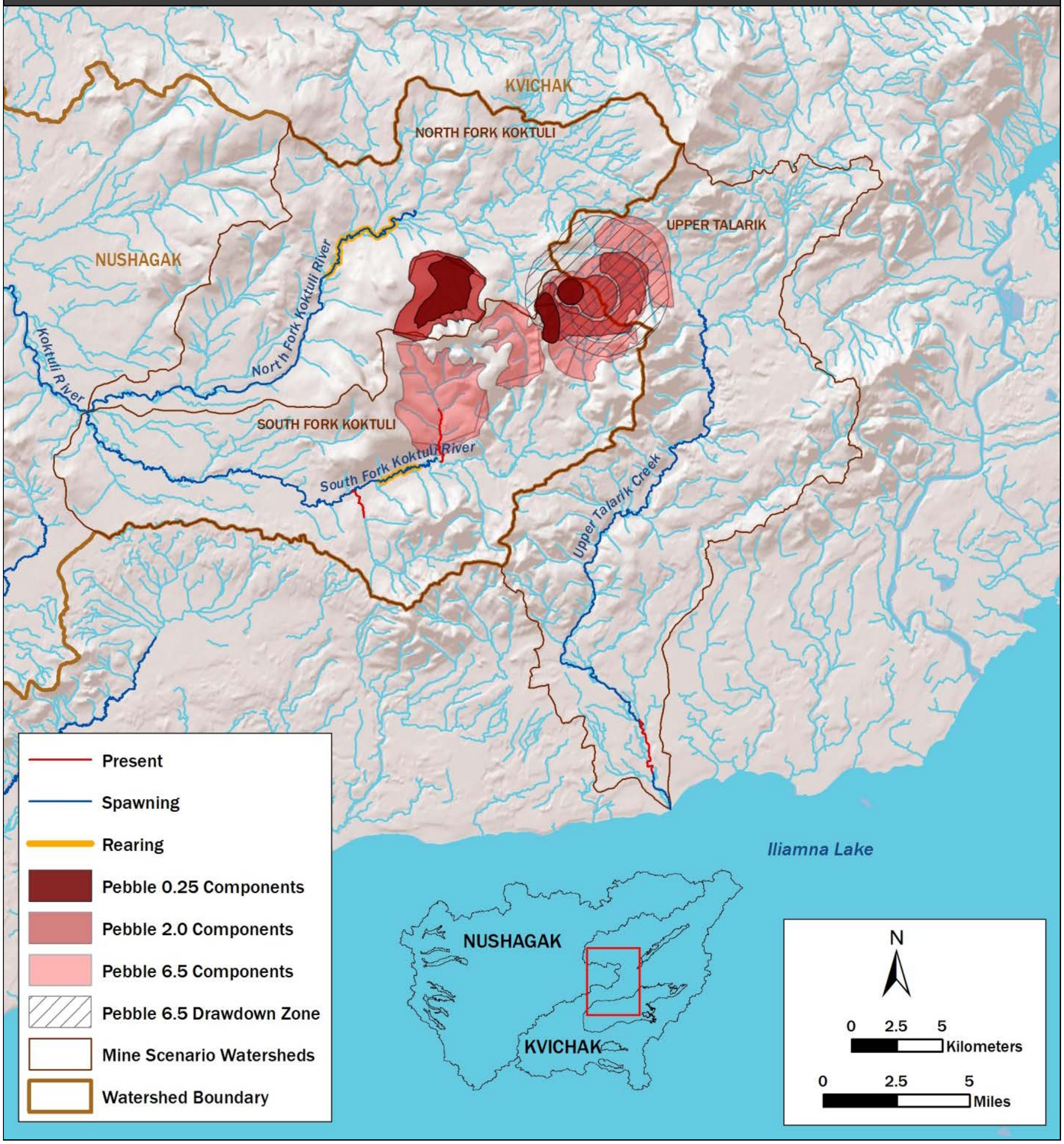


Figure 7-6. Reported pink salmon distribution in the mine scenario watersheds. "Present" indicates species was present but life-stage use was not determined; "spawning" indicates spawning adults were observed. Present and spawning designations are based on the Anadromous Waters Catalog (Johnson and Blanche 2012). Life-stage-specific reach designations are likely underestimates, given the challenges inherent in surveying all streams that may support life-stage use throughout the year (see Section 7.2.5 for additional notes on interpretation of fish distribution data). Footprints of the major mine components for the three mine scenarios and the drawdown zone for the Pebble 6.5 scenario are shown for reference.

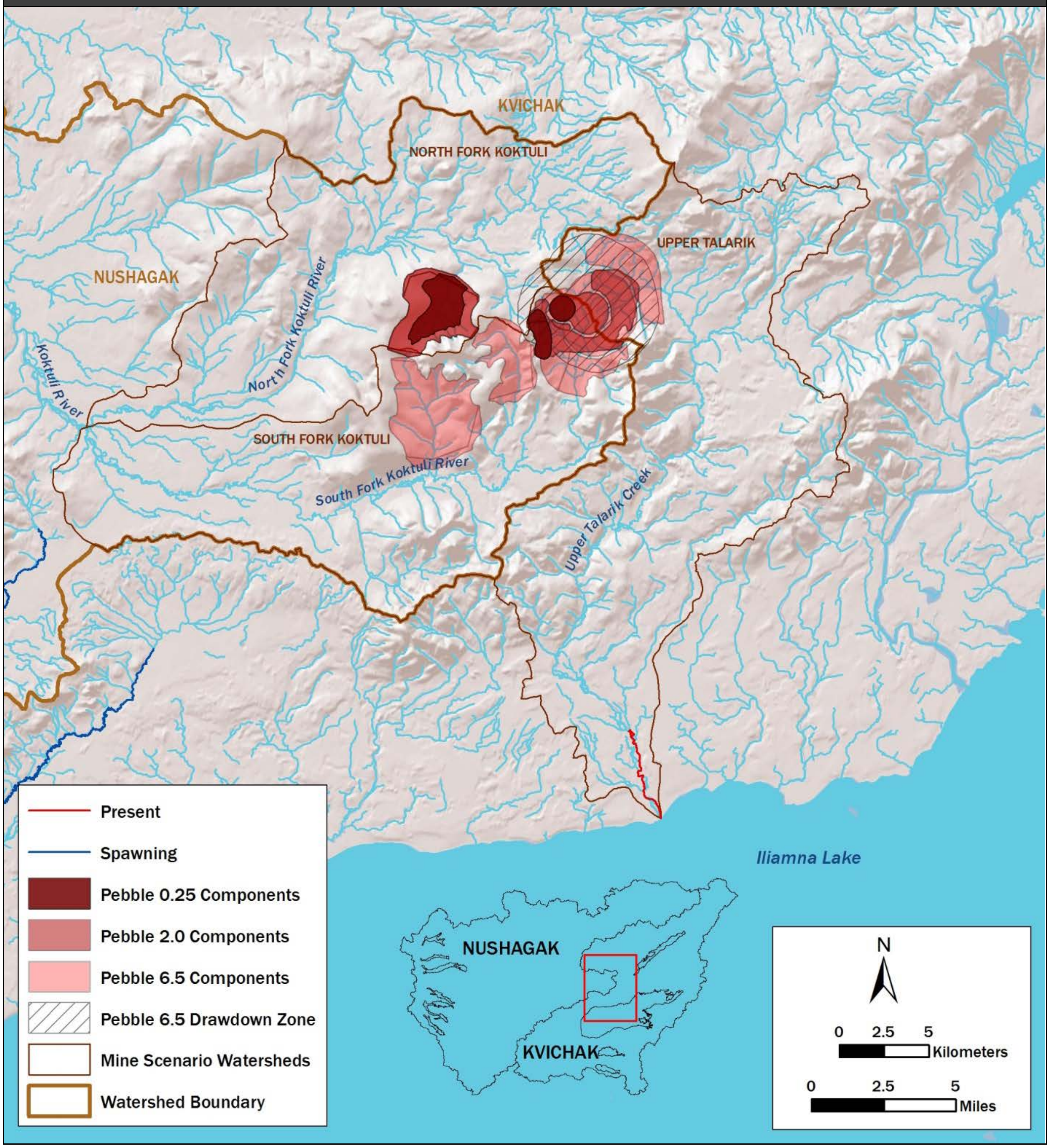


Figure 7-7. Reported Dolly Varden occurrence in the mine scenario watersheds. Designation of species presence is based on the Alaska Freshwater Fish Inventory (ADF&G 2012). Absence cannot be inferred from this map (see Section 7.2.5 for additional notes on interpretation of fish distribution data). Footprints of the major mine components for the three mine scenarios and the drawdown zone for the Pebble 6.5 scenario are shown for reference.

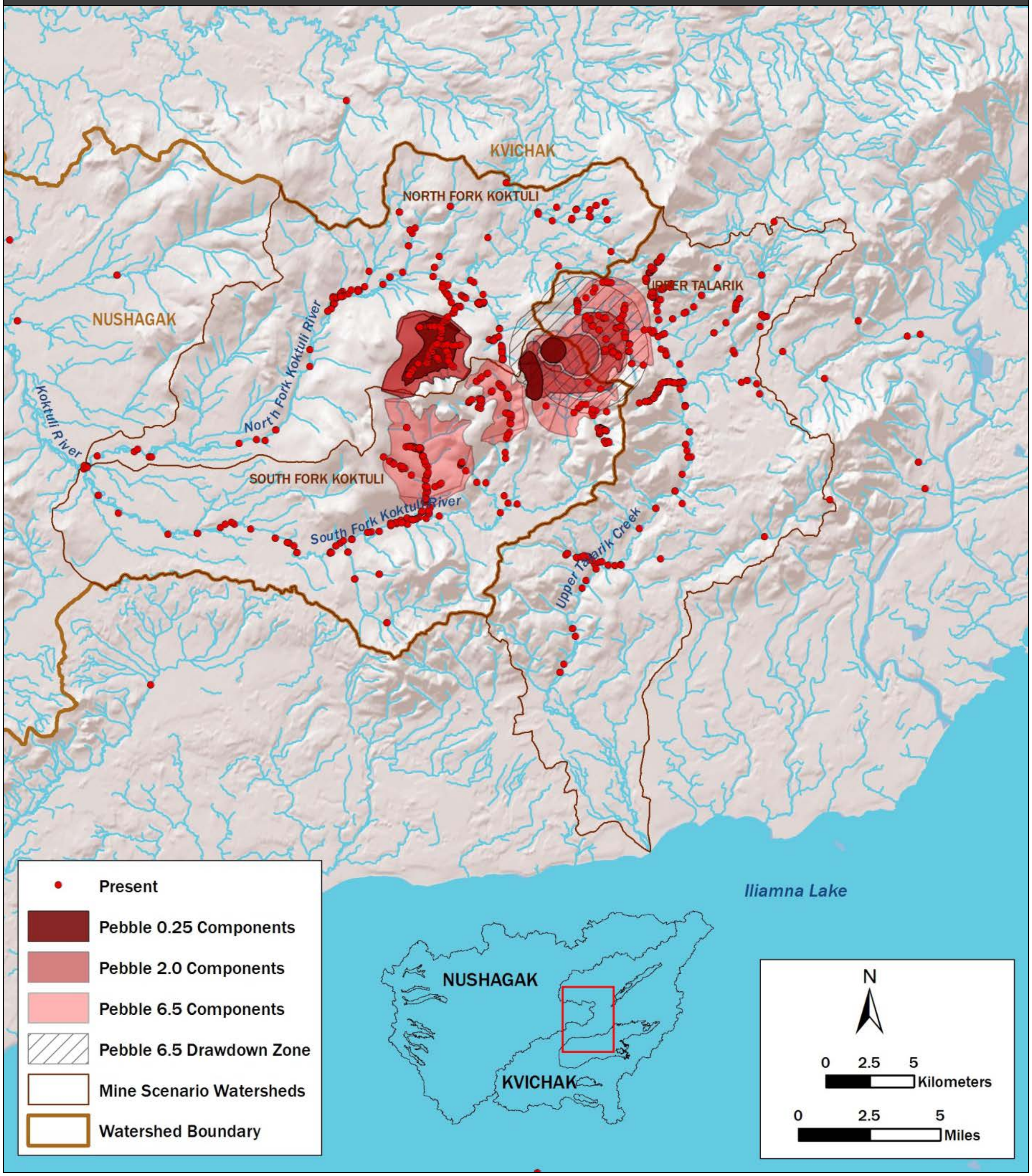
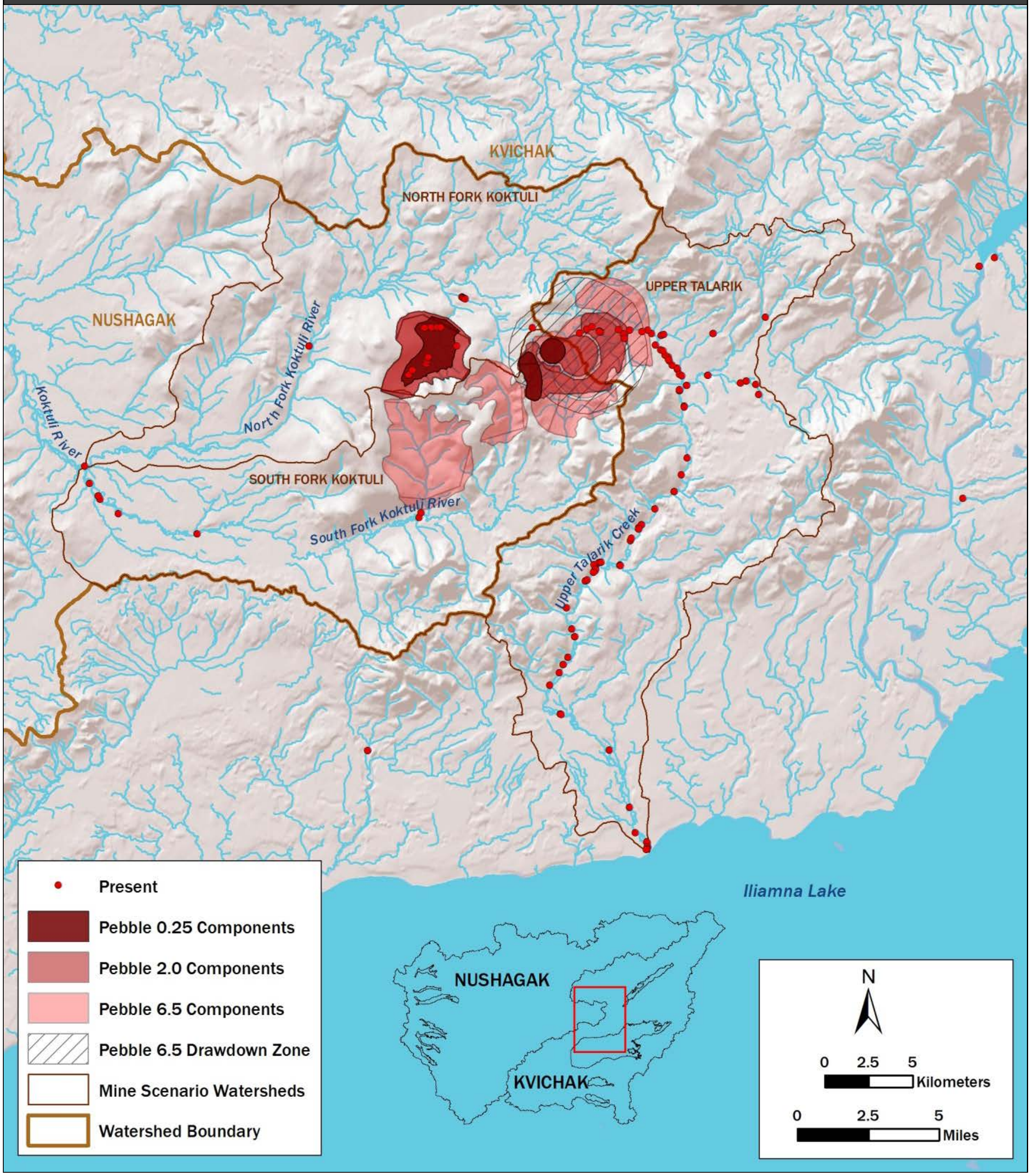


Figure 7-8. Reported rainbow trout occurrence in the mine scenario watersheds. Designation of species presence is based on the Alaska Freshwater Fish Inventory (ADF&G 2012). Absence cannot be inferred from this map (see Section 7.2.5 for additional notes on interpretation of fish distribution data). Footprints of the major mine components for the three mine scenarios and the drawdown zone for the Pebble 6.5 scenario are shown for reference.



Sockeye salmon use mainstem reaches of the mine scenario watersheds for spawning and rearing, including a portion of Upper Talarik Creek that is within the waste rock piles of the Pebble 2.0 and Pebble 6.5 scenarios (Figure 7-2). Coho salmon have the most widespread distribution of the five salmon species in the mine scenario watersheds, and make extensive use of mainstem and tributary habitats (Figure 7-3). Coho spawn and rear in headwater streams that would be eliminated, blocked, or dewatered by the mine pits, waste rock piles, and TSFs of the Pebble 0.25, 2.0, and 6.5 scenarios (Figure 7-3). Chinook salmon have been documented throughout mainstem reaches of the mine scenario watersheds (Figure 7-4). Chinook are known to use small streams for rearing habitat, and juveniles have been observed in streams that would be within the footprints of TSF 1 (North Fork Koktuli River), TSF 2 (South Fork Koktuli River), and the waste rock piles and mine pits (Upper Talarik Creek) (Figure 7-4). The distributions of chum and pink salmon are generally restricted to mainstem reaches where spawning and migration occur. Chum salmon have been found in all three mine scenario watersheds and in a stream within the footprint of TSF 2 (Figure 7-5). Pink salmon have only been reported 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 7-6, Figure 5-8). Dolly Varden are found throughout the mine scenario watersheds, and fish surveys indicate that they are commonly found in the smallest streams (i.e., first-order tributaries), including streams within the footprints of each of the TSFs (Figure 7-7). Rainbow trout have been collected at many mainstem locations, especially in Upper Talarik Creek, and their reported distribution extends upstream throughout the TSF 1 area and in the portions of Upper Talarik Creek within the waste rock pile footprints (Figure 7-8).

7.1.2 Spawning Salmon Abundance

Index estimates of relative spawning salmon abundance are available for sockeye, coho, Chinook, and chum salmon in the mine scenario watersheds. Aerial index counts of spawning salmon are available from ADF&G and the Pebble Limited Partnership (PLP). This type of survey is used primarily as an index to track variation in run size over time. We recognize that survey values tend to underestimate true abundance for several reasons. An observer in an aircraft is not able to count all fish in dense aggregations or those concealed under overhanging vegetation or undercut banks, and 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). Weather, water clarity, and other factors that influence fish visibility can also contribute to underestimates. In addition, surveys intended to capture peak abundance may not always do so. For example, aerial surveys counted, on average, only 44% of the pink salmon counted by surveyors walking the same Prince William Sound spawning streams (Bue et al. 1988). Peak aerial counts of pink salmon in southeastern Alaska are routinely multiplied by 2.5 to represent more accurately the number of fish present at the survey time (Jones et al. 2007). Helicopter surveys of Chinook salmon on the Kenai Peninsula's Anchor River over 5 years counted only 5 to 10% of the fish counted by a concurrent sonar/weir counting station (Szarzi et al. 2007).

ADF&G conducts aerial index counts that target peak sockeye salmon spawning periods on Upper Talarik Creek and peak Chinook salmon spawning periods on the Koktuli River system. Sockeye salmon counts have been conducted in most years since 1955 (Morstad 2003), and Chinook salmon counts in most years since 1967 (Dye and Schwanke 2009). Between 1955 and 2011, sockeye salmon counts in Upper Talarik Creek 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 system ranged from 240 to 10,620, with an average of 3,828 over 29 count periods (Dye and Schwanke 2009).

PLP (2011) provides aerial index counts for Chinook, chum, coho, and sockeye salmon adults in the mine scenario watersheds from 2004 to 2008. Surveys on the South and North Fork Koktuli Rivers began at the confluence and extended upward to the intermittent reach or Frying Pan Lake on the South Fork Koktuli River and upward to Big Wiggly Lake or river kilometer 56 on the North Fork Koktuli River. Surveys on Upper Talarik Creek ran from the mouth and extended upstream to Tributary 1.350 (just east of Koktuli Mountain) or to the headwaters. Multiple counts were usually made for each stream and species in a given year (Table 7-1). Repeat surveys of this type can be used to estimate the size of spawning populations using an area under the curve (AUC) approach if estimates of stream life (i.e., the number of days that salmon are present on the spawning grounds) and observer efficiency are available (Hilborn et al. 1999). However, PLP was unable to make reliable estimates of stream life and observer efficiency (PLP 2011: 15.1-14), a common shortcoming given the data-intensive demands of AUC estimates (Holt and Cox 2008). Mean index counts can be reliable indicators of spawning coho salmon abundance trends in simulation studies (e.g., Holt and Cox 2008), but optimum reliability is contingent on sampling date and frequency. Peak index counts have been used to monitor trends in spawner abundance, but these counts also have shortcomings associated with survey design and execution and require area- and species-specific expansion factors to allow escapement estimates (e.g., Parken et al. 2003). In addition, trend analysis needs to account for the high interannual variability in escapement estimates noted above, and likely requires many years of data. Streams or river segments lacking long-term survey data require a larger watershed and population context to approximate baseline conditions for those locations and populations.

Table 7-1 reports the highest of each year's index counts for each population, approximated from figures in PLP (2011: Chapter 15). We report peak index counts because only a portion of the spawning population is present on the spawning grounds on any given day. Thus, the highest index count is mathematically closer to the true abundance than is the average of multiple surveys, and it more closely matches ADF&G's index methods based on a single count that targets peak spawning. The highest peak index counts for coho and sockeye salmon were in Upper Talarik Creek, whereas the highest counts for Chinook and chum salmon were in the South and North Fork Koktuli Rivers (Table 7-1). The overall highest count was for sockeye salmon in Upper Talarik Creek and Tributary 1.60 in 2008, when approximately 82,000 fish were estimated (Table 7-1).

Table 7-1. Highest reported index spawner counts in the mine scenario watersheds for each year, 2004 to 2008.

Mine Scenario Watershed	Salmon Species	Highest Index Spawner Count Per Year (Number Of Counts)[a]				
		2004	2005	2006	2007	2008
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)
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)
Upper Talarik Creek	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]

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

The spatial distribution of spawner counts in the study streams during the 2008 return year was provided by PLP (2011). Spawner counts were summarized by individual stream reaches throughout the mainstem of each of the mine scenario watersheds. Data were reported for three reaches in the South Fork Koktuli River (A through C, extending from the confluence upstream to the intermittent reach), five reaches in the North Fork Koktuli River (A through E, extending from the confluence upstream to beyond Big Wiggly Lake), and seven reaches in Upper Talarik Creek (A through G, extending from the mouth to the headwaters) (Figure 15.1-2 in PLP [2011] illustrates the stream reaches; Table 7-2 provides river kilometer boundaries for each reach). Count data (approximated from figures in PLP [2011]) and location (in river kilometers) for each of these reaches are shown in Table 7-2 to demonstrate the relative spatial distribution of salmon during the 2008 spawning period.

Table 7-2. Average 2008 index spawner counts by stream reach[a].

Stream	Salmon Species	Stream Reach, Downstream to Upstream						
		A	B	C	D	E	F	G
South Fork Koktuli River	Reach Boundaries (river km)	0–24.9	24.9–34.3	34.3–51.7	-	-	-	-
	Chinook	200	70	0	-	-	-	-
	Chum	90	190	0	-	-	-	-
	Coho	200	250	8	-	-	-	-
	Sockeye	800	1,510	1	-	-	-	-
North Fork Koktuli River	Reach Boundaries (river km)	0–13.7	13.7–21.1	21.1–36.6	36.6–48.4	48.4–52.5	-	-
	Chinook	110	40	50	0	0	-	-
	Chum	50	50	320	0	0	-	-
	Coho	100	70	210	30	60	-	-
	Sockeye	530	<10	220	60	0	-	-
Upper Talarik Creek	Reach Boundaries (river km)	0–5.9	5.9–16.8	16.8–24.8	24.8–36.3	36.3–45.1	45.1–59.1	59.1–62.4
	Chinook	<10	<10	20	<10	20	<10	0
	Chum	<10	<10	<10	<10	<10	10	0
	Coho	100	50	40	180	280	180	<10
	Sockeye	10,000	4,500	3,000	3,000	500	47	0

Notes:
Dashes (-) indicate no applicable stream reach.
[a] Values likely underestimate true spawner abundance.
Source: PLP 2011.

7.1.3 Juvenile Salmon and Other Salmonid Abundance

PLP (2011) reports counts of juvenile salmon and other salmonids in the South and North Fork Koktuli Rivers and Upper Talarik Creek based on extensive sampling efforts from 2004 through 2008. 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 always possible to determine which survey methods generated which counts in PLP (2011). Raw field counts were frequently 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. They are very likely underestimates because of the extreme difficulty of observing or capturing all fish in complex habitats (Hillman et al. 1992). Density estimates with confidence bounds (e.g., mark-recapture or depletion estimates) were generated for some parts of the PLP (2011) studies (e.g., PLP 2011: Appendix 15.1D), but such efforts were uncommon as they are much more time-consuming and labor-intensive.

Reported fish densities summarized over the 5-year period 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 in the same stream, which is typical for fish in heterogeneous stream environments. Table 7-3 presents maximum fish densities in the mainstem of each mine scenario watershed, approximated from figures in PLP (2011), for species that rear for extended periods in the surveyed streams and for which data are available: 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). Highest reported densities were approximately 2,500 Arctic grayling and 1,600 coho salmon per 100 m from adjacent reaches on Upper Talarik Creek, and 1,400 coho salmon per 100 m from a reach on the North Fork Koktuli River.

Table 7-3. Highest index counts of selected stream-rearing fish species from mainstem habitats of the mine scenario watersheds.

	Highest Reported Density (count per 100 m)[a]				
Stream	Chinook Salmon	Coho Salmon	Arctic Grayling	Dolly Varden	Source
South Fork Koktuli River	450	600	275	55	Figure 15.1-52 (PLP 2011)
North Fork Koktuli River	500	1,400	40	40	Figure 15.1-23 (PLP 2011)
Upper Talarik Creek	400	1,600	2,500	10	Figure 15.1-82 (PLP 2011)

Notes:
[a] Values were approximated from figures listed in the source column.

7.2 Habitat Modification

The footprints of the major mine components (mine pit, waste rock piles, and TSFs) would directly modify the amount of habitat available to salmon, rainbow trout, and Dolly Varden by eliminating headwater streams and wetlands within and up-gradient of their footprints. Potential effects of this habitat modification are described for the three mine scenarios in Sections 7.2.2, 7.2.3, and 7.2.4, and uncertainties and assumptions are described in Section 7.2.5.

7.2.1 Stream Segment Characteristics in the Mine Scenario Watersheds

The mine scenario watersheds encompass an area of 925 km^2 and contain 930 km of stream channels mapped for this analysis (methods described in Section 3.4). In this section, we summarize stream segment characteristics in the mine scenario watersheds to better characterize stream environments in and downstream of the mine footprints. In Section 7.2.2, we summarize the characteristics of stream segments that would be lost to the footprints of the major mine components themselves. Stream segments for the entire Nushagak and Kvichak River watersheds (Scale 2) are characterized in Chapter 3. This characterization is provided to help readers understand variation in the relative size (mean annual streamflow), channel gradient, and floodplain potential (proportion of flatland in lowland) among stream segments in the mine scenario watersheds. Because these characteristics can strongly influence the quality and suitability of stream habitats as fish habitat, they provide a way to evaluate the coarse-scale characteristics of streams at risk of impacts at various scales. This characterization helps highlight the fact that not all stream kilometers in these watersheds are equal in their potential to support salmon carrying capacity or productivity.

Results from this analysis are presented in tables that summarize the proportion of stream channel length within each stream size, gradient, and floodplain potential category. To allow direct visual comparison of the distribution of stream characteristics across scales, we present cumulative frequency plots (e.g., Figure 7-9). These plots show a frequency curve for each attribute at different geographic scales. Attributes are grouped into meaningful classes (Chapter 3), denoted by the vertical red classification bars. For example, the lowest gradient streams are classified as having gradients of less than 1% (Table 7-4), as shown by the vertical classification bar at 1% in Figure 7-9B. Cumulative frequency plots can be interpreted by evaluating the height at which the frequency curve is intersected by the red vertical classification bar. In Figure 7-9B, the 1% gradient classification bar intersects the Scale 3 frequency curve (solid black line) at a cumulative frequency value of approximately 50%. Thus, approximately 50% of the stream kilometers in the mine scenario watersheds (Scale 3) have less than 1% gradient. In comparison, approximately 64% of the stream kilometers in the Nushagak and Kvichak River watersheds (Scale 2) have less than 1% gradient.

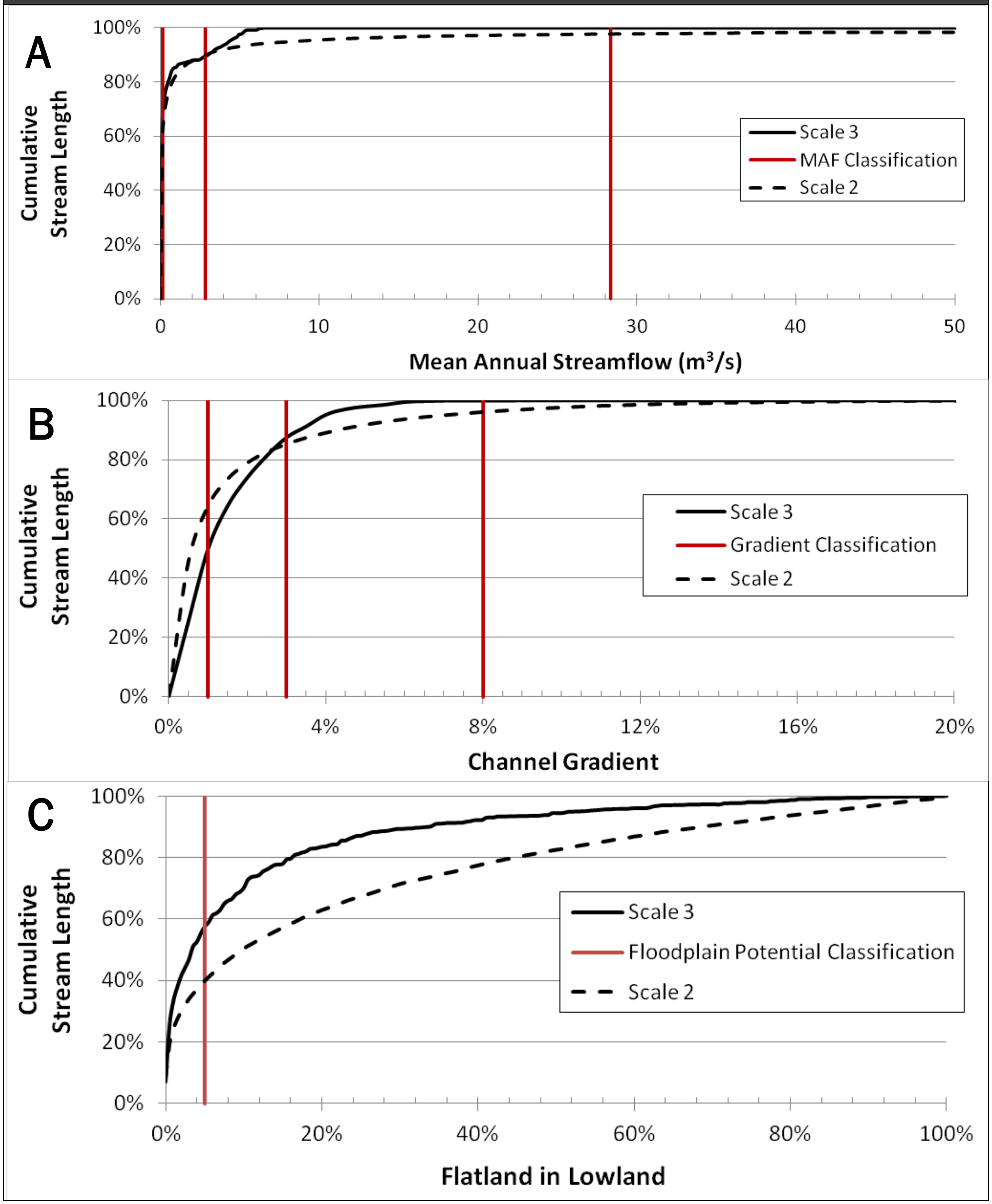
Table 7-4. Distribution of stream channel length classified by channel size (based on mean annual streamflow in m³/s), channel gradient (%), and floodplain potential (based on % flatland in lowland) for streams and rivers in the mine scenario watersheds. Gray shading indicates values greater than 5%; bold indicates values greater than 10%.

	Gradient							
	<1%		≥1% and <3%		≥3% and <8%		≥8%	
Channel Size	FP	NFP	FP	NFP	FP	NFP	FP	NFP
Small headwater streams[a]	**15%**	5%	5%	**28%**	0%	**12%**	0%	0%
Medium streams[b]	**14%**	6%	0%	3%	0%	1%	0%	0%
Small rivers[c]	8%	2%	0%	1%	0%	0%	0%	0%
Large rivers[d]	0%	0%	0%	0%	0%	0%	0%	0%

Notes:
[a] 0–0.15 m³/s; most tributaries in the mine footprints.
[b] 0.15–2.8 m³/s; upper reaches and larger tributaries of the South and North Fork Koktuli Rivers and Upper Talarik Creek.
[c] 2.8–28 m³/s; middle to lower portions of the South and North Fork Koktuli Rivers and Upper Talarik Creek, including mainstem Koktuli River.
[d] >28 m³/s; the Mulchatna River below the Koktuli confluence, the Newhalen River, and other large rivers.
FP = high floodplain potential (≥5% flatland in lowland); NFP = no or low floodplain potential (<5% flatland in lowland) (see Chapter 3 for additional explanation).

Similar to the larger Nushagak and Kvichak River watersheds (summarized in Table 3-3), streams in the mine scenario watersheds are generally low-gradient, with extensive flat floodplains or terraces in the larger valleys (Figure 7-9; also see PLP 2011: Chapter 15 and Appendix 15.1B). There are no large rivers (greater than 28 m³/s mean annual streamflow) in the mine scenario watersheds (Table 7-4). Compared to the larger Nushagak and Kvichak River watersheds, streams in the mine scenario watersheds have fewer very low gradient streams (mean gradient 0.7% versus 0.4%) and a higher proportion (58% versus 39%) of stream length flowing through valleys with low floodplain potential (i.e., less than 5% of flatland in lowland) (Table 7-4, Figure 7-9).

Figure 7-9. Cumulative frequency of stream channel length classified by (A) mean annual streamflow (MAF) (m^3/s), (B) channel gradient (%), and (C) floodplain potential (based on % flatland in lowland) for the mine scenario watersheds (Scale 3) versus the Nushagak and Kvichak River watersheds (Scale 2). See Section 3.4 for further explanation of MAF, gradient, and floodplain potential classifications.



Broadly classified, streams and rivers in the Nushagak and Kvichak River watersheds that are likely to provide high capacity and quality habitats for salmonids include streams with gradients less than 3% and of medium stream size (0.15 to 2.8 m^3/s mean annual streamflow) or greater. Such streams and rivers account for 36% of the stream network in the larger Nushagak and Kvichak River watersheds (Table 3-3), and account for 34% of the stream network in the mine scenario watersheds (Table 7-4). Smaller, steeper streams provide seasonal (and some year-round) habitat, and provide important provisioning services to downstream waters (Section 7.2.3). Although streams in the mine scenario watersheds are smaller and slightly steeper than streams and rivers throughout the entire Nushagak and Kvichak River watersheds, these results show the high proportion of stream channels in these basins with the broad geomorphic and hydrologic characteristics that support stream and river habitats highly suitable for fish species such as Pacific salmon, Dolly Varden, and rainbow trout.

7.2.2 Exposure: Habitat Lost to the Mine Scenario Footprints

For each mine scenario, the total mine footprint consists of the area devoted to the major mine components (mine pit, waste rock piles, and TSFs), the groundwater drawdown zone associated with the mine pit, and plant and ancillary facilities (e.g., ore-processing facilities and water collection and treatment facilities) (see Chapter 6 for additional details on each mine scenario). Portions of the mine scenario watersheds would be affected by mining activity in this footprint. Stream and wetland habitats would be lost within and upstream of the footprint (Figures 7-10 through 7-12), and downstream habitat would be degraded by the loss of the headwater streams and wetlands. Streams under or upstream of each mine footprint would be inaccessible by fish from downstream reaches because of the following factors.

- Elimination of streams and wetlands within the mine footprints, either due to removal (e.g., excavation of streams or wetlands in the mine pit area) or burial under a TSF or waste rock pile.
- Dewatering by capture into a groundwater drawdown zone associated with the pit. This effect is distinct from the effect of water removal and capture on streamflows downstream of the mine footprint, which is covered in Section 7.3.
- Blockage due to either of the above or channel diversion in a manner that prevents fish passage (e.g., via pipes or conveyances too steep for fish passage).

Streams and wetlands removed or altered via these various mechanisms are collectively referred to as “lost” in this assessment. Methods used to estimate these losses are described in Box 7-1.

Figure 7-10. Streams and wetlands lost (eliminated, blocked, or dewatered) in the Pebble 0.25 scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012a) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012). See Box 7-1 for definitions and methods used for delineation.

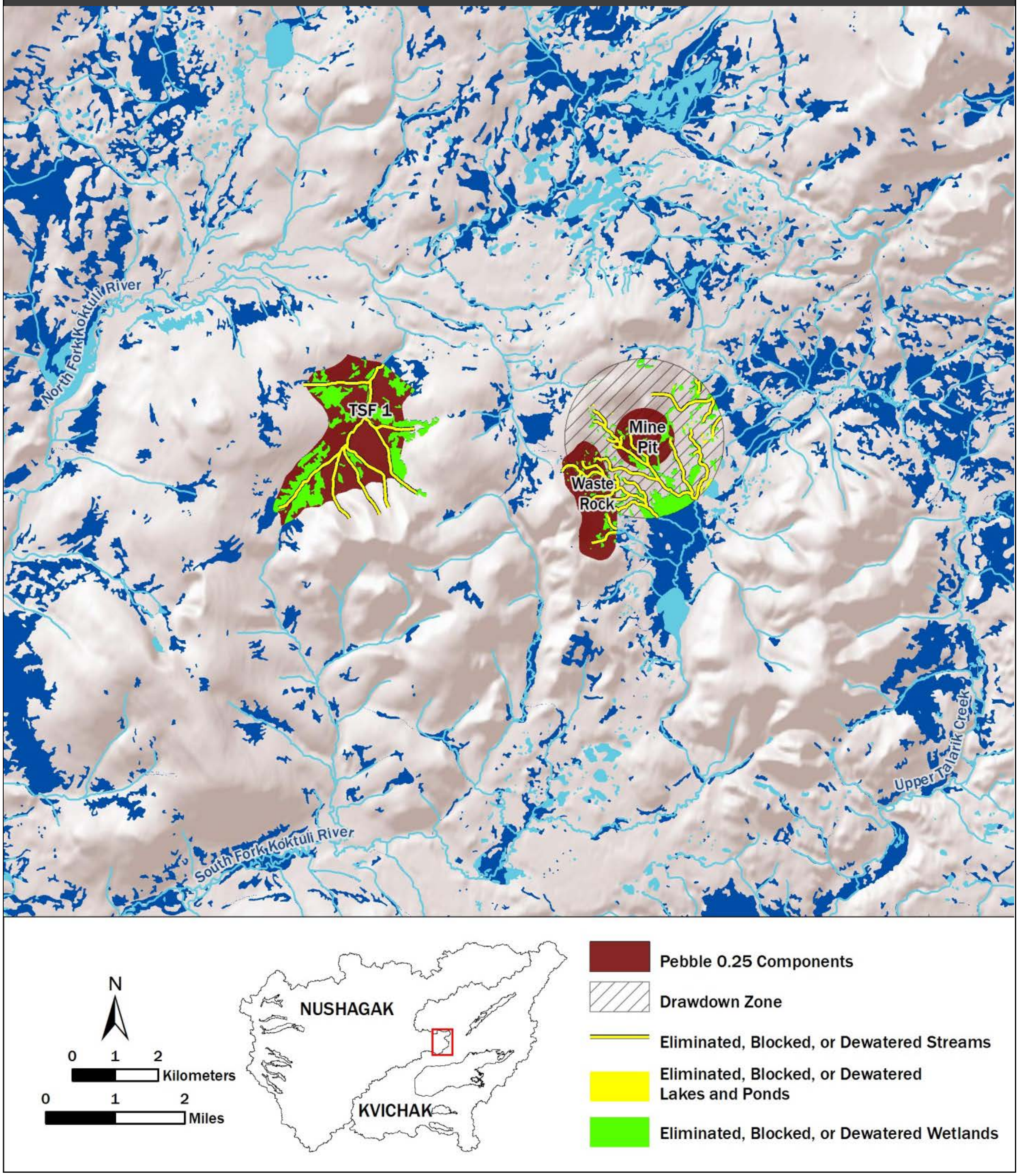


Figure 7-11. Streams and wetlands lost (eliminated, blocked, or dewatered) in the Pebble 2.0 scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012a) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012). See Box 7-1 for definitions and methods used for delineation.

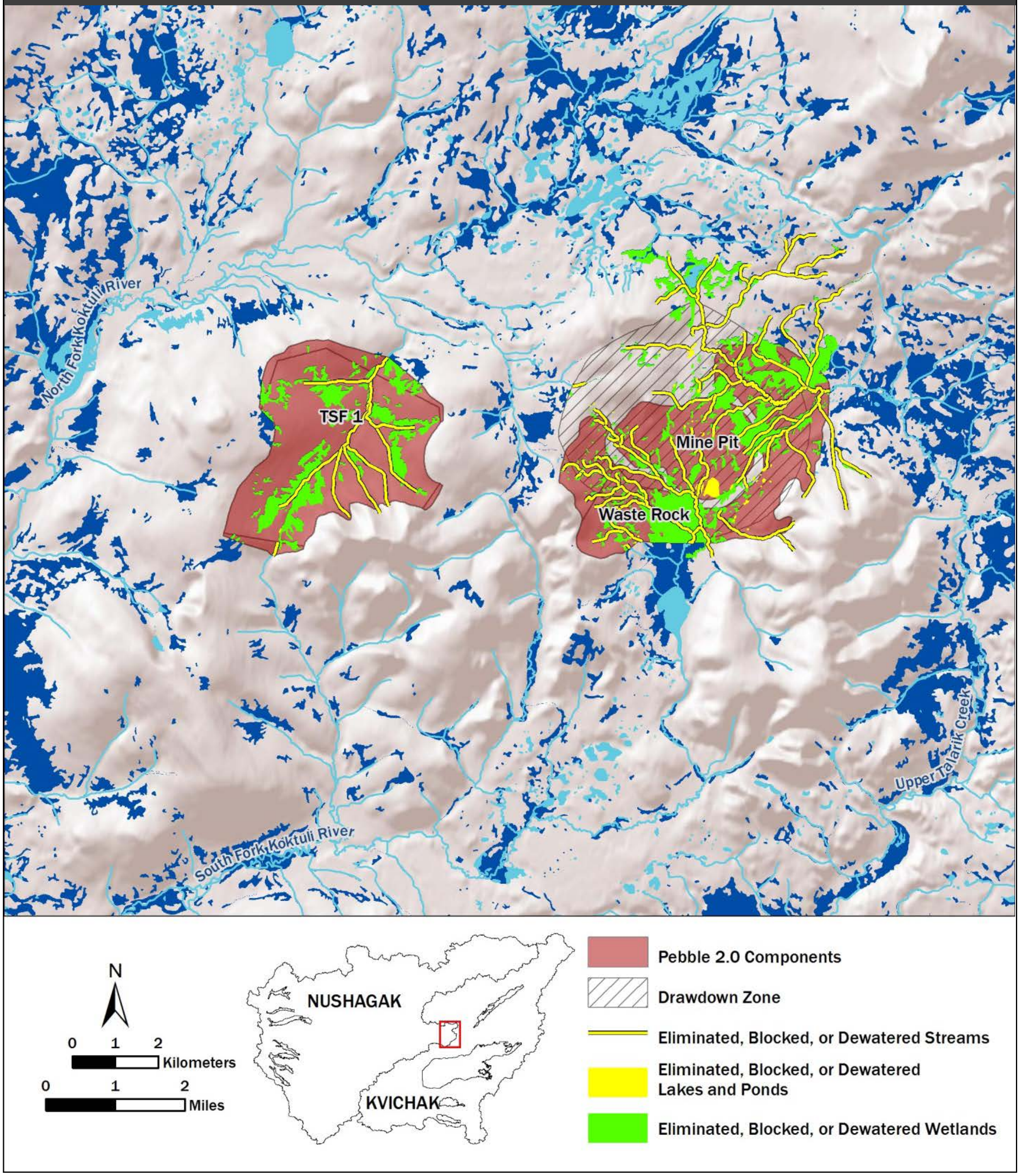
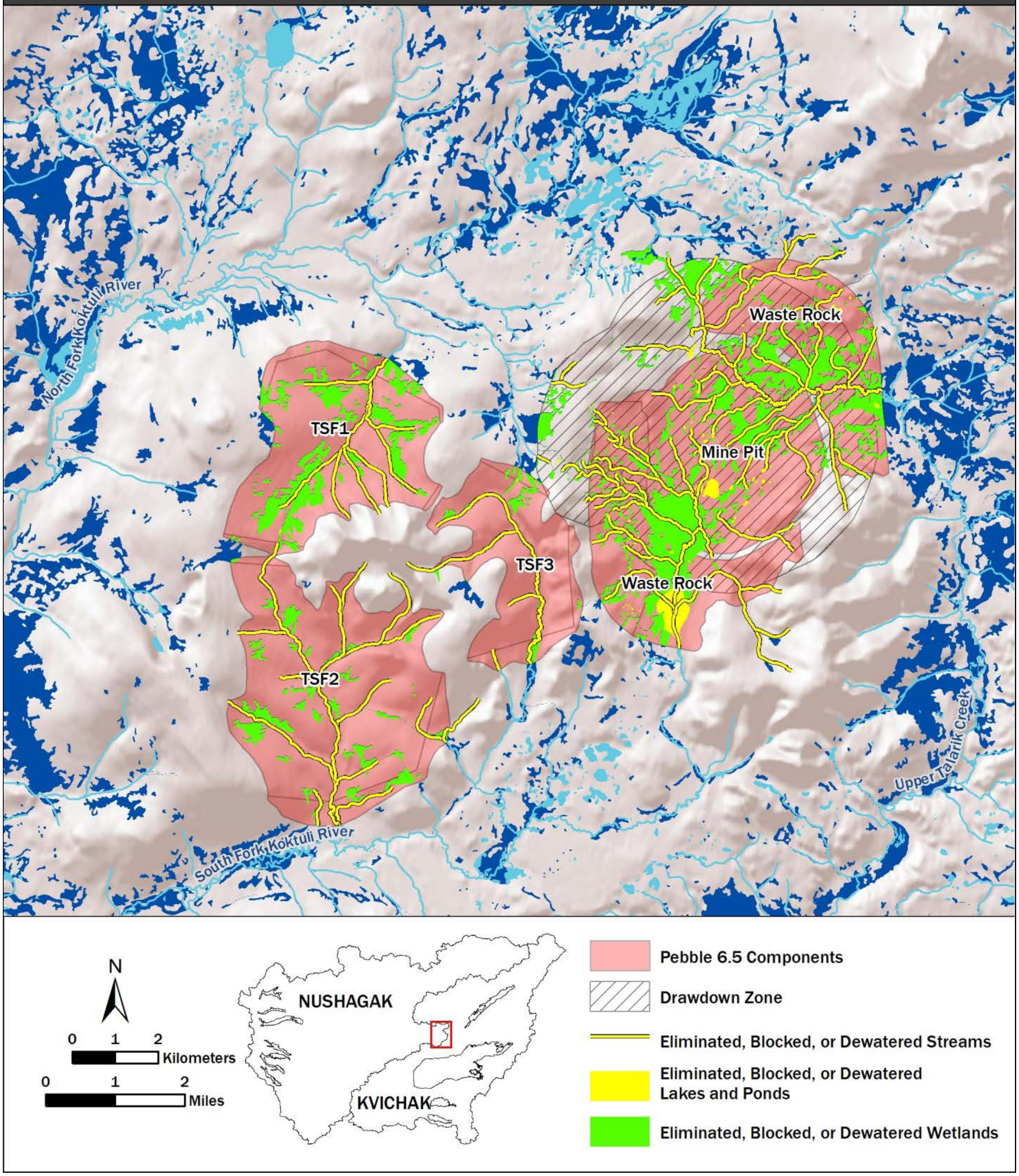


Figure 7-12. Streams and wetlands lost (eliminated, blocked, or dewatered) in the Pebble 6.5 scenario. Light blue areas indicate streams and rivers from the National Hydrography Dataset (USGS 2012a) and lakes and ponds from the National Wetlands Inventory (USFWS 2012); dark blue areas indicate wetlands from the National Wetlands Inventory (USFWS 2012). See Box 7-1 for definitions and methods used for delineation.



BOX 7-1. CALCULATION OF STREAMS AND WETLANDS AFFECTED BY MINE SCENARIO FOOTPRINTS

To calculate kilometers of streams eliminated, blocked, or experiencing streamflow alteration due to the footprints of the major mine components and the groundwater drawdown zone associated with the mine pit, we used the Alaska National Hydrography Dataset (NHD) (USGS 2012a). The scale of this dataset is 1:63,360. In this assessment, a stream segment is classified as eliminated if it falls within the boundaries of the mine pit, waste rock pile, or tailings storage facility (TSF). A segment is classified as blocked if it or a downstream segment it connects to directly intersects the mine pit, waste rock pile, or TSF. A stream segment not otherwise eliminated is classified as dewatered if it falls within the groundwater drawdown zone associated with the mine pit, or is classified as blocked and dewatered if it falls within the groundwater drawdown zone and a downstream segment it connects to directly intersects the mine pit. For calculation of stream kilometers either eliminated or blocked that are inhabited by anadromous and resident fish species, we used the Anadromous Waters Catalog (AWC) (Johnson and Blanche 2012) and the Alaska Freshwater Fish Inventory (AFFI) (ADF&G 2012). Stream lengths blocked, eliminated, or dewatered were summed across each classification for both NHD and AWC fish-inhabited stream segments (Table 7-5).

Estimates of wetland, pond, and lake areas eliminated, blocked, or dewatered by the mine scenario footprints were derived from the National Wetlands Inventory (NWI) (USFWS 2012). For the State of Alaska, the scale of this dataset is 1:63,360. In this assessment, wetland, pond, or lake area is classified as eliminated if it falls within the boundaries of the mine pit, waste rock pile, or TSF; dewatered if it falls within the groundwater drawdown zone associated with the mine pit; and blocked if it directly intersects a previously categorized blocked NHD stream. Wetland, pond, and lake areas blocked, eliminated, or dewatered were summed across each classification (Table 7-8).

The area covered by plant and ancillary facilities associated with mine site development (e.g., housing, crushing plant, wastewater treatment plant) is not considered in the calculation of eliminated and blocked streams and wetlands due to lack of knowledge about the orientation and placement of these structures on the landscape. Thus, the values reported in Tables 7-5 and 7-8 are conservative estimates, as additional development on the landscape would likely impact additional stream length and wetland area due to the abundance of aquatic habitats in this region.

It is important to note that estimates of stream length and wetland, pond, and lake areas affected represent a lower bound on the estimate. The NHD does not capture all stream courses and may underestimate channel sinuosity, resulting in underestimates of affected stream length. In addition, 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 Alaska Department of Fish and Game, in its on-line AWC database, states: “Based upon thorough surveys of a few drainages it is believed that this number represents less than 50% of the streams, rivers and lakes actually used by anadromous species” (ADF&G 2013). The characterization of wetland area is limited by resolution of the available NWI data product, which is not available for the full assessment area. Other investigations have revealed high spatial variability in wetland density across the region (e.g., Hall et al. 1994). Others have conducted enhanced wetland inventories. For example, the Pebble Limited Partnership (2011) used multiple sources of high resolution remote imaging and ground-truthing to map wetlands in their mine mapping area, which focused on the proposed mine working area and major stream valleys. They reported wetland densities of approximately 29% for the mapping area (PLP 2011: Table 14.1-3), whereas preliminary NWI mapping identified approximately 20% of this same area as wetland (PLP 2011: Table 14.1-1). Furthermore, the major mine components of the mine scenarios often bisected wetland, pond, or lake features, and areas falling outside the boundary were assumed to maintain their functionality. We were also unable to determine the effect that mine site development may have on wetlands with no direct surface connection to a blocked NHD stream segment, but with a potential connection via groundwater pathways. Given these limitations, these estimates could be enhanced with improved, higher-resolution mapping, increased sampling of possible fish-bearing waters, and ground-truthing.

7.2.2.1 Stream Losses

In the Pebble 0.25 scenario, 38 km of streams would be eliminated, blocked, or dewatered by the mine footprint (Table 7-5, Figure 7-10). In the Pebble 2.0 scenario, over 89 km of streams would be eliminated, blocked, or dewatered by the mine footprint (Table 7-5, Figure 7-11). In the Pebble 6.5 scenario, an additional 20 km of streams in the pit and waste rock pile areas and 41 km of streams under TSF 2 and TSF 3 would be eliminated or blocked, for a total of 151 km of streams lost to the mine footprint (Table 7-5, Figure 7-12). These scenarios represent 4, 8, and 14% of the total stream length within the mine scenario watersheds. Of the streams lost to the mine footprint in the Pebble 6.5 scenario, 82% are headwater streams (less than 0.15 m^3/s mean annual streamflow); 76% have less than 3% gradient, and 26% have less than 1% gradient (Table 7-6, Figure 7-13). The majority (74%) of smaller streams lost to the mine footprint in the Pebble 6.5 scenario flow through valleys with limited flatland (Table 7-6).

Compared to the larger Nushagak and Kvichak River watersheds, streams lost to the mine footprints are smaller: 9% of stream length in the Nushagak and Kvichak River watersheds exceeds 2.8 m^3/s mean annual streamflow (Table 3-3), whereas no streams lost to the mine footprints exceed this size (Figure 7-13A). Streams within the mine footprints also have a lower proportion of stream length with less than 1% gradient (26% versus 64% of stream length in the Nushagak and Kvichak River watersheds) (Figure 7-13B), and more stream length with low floodplain potential (74% versus 39%) (Figure 7-13C).

These results provide some indication of the relative size, steepness, and geomorphic setting of streams that would be lost to the mine footprints. The streams that would be lost include a range of stream types that provide a variety of habitat functions for salmon, including as year-round or seasonal habitat for salmonids or other fishes or as important sources of water, macroinvertebrates, and other materials to downstream waters (Section 7.2.3). Of the 151 km of streams lost to the Pebble 6.5 footprint, 36 km are currently cataloged as anadromous fish streams in the AWC (Johnson and Blanche 2012). Most of these cataloged anadromous streams are in the medium stream size class (0.15 to 2.8 m^3/s), with gradients less than 3%. These include the upper reaches and larger tributaries of the South and North Fork Koktuli Rivers and Upper Talarik Creek, including smaller streams with documented occurrence of coho salmon (Figure 7-3). Many of the smaller, steeper tributaries have been documented to contain Dolly Varden (Figure 7-7).

Figure 7-13. Cumulative frequency of stream channel length classified by (A) mean annual streamflow (m^3/s), (B) channel gradient (%), and (C) floodplain potential (based on % flatland in lowland) for the mine footprints (Scale 4) versus the Nushagak and Kvichak River watersheds (Scale 2).

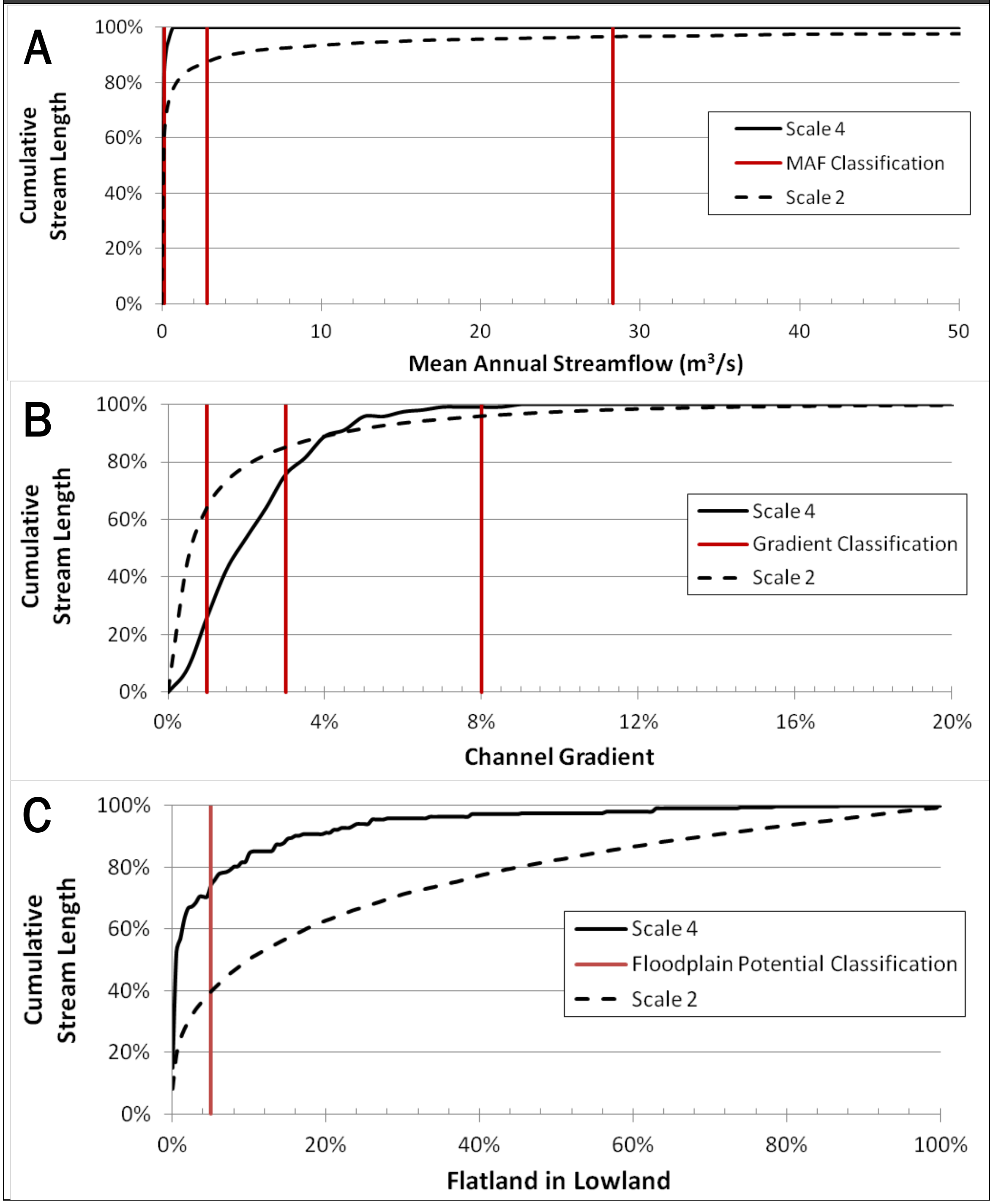


Table 7-5. Stream length (km) eliminated, blocked, or dewatered by the mine footprints in the Pebble 0.25, 2.0, and 6.5 scenarios. See Box 7-1 for methods.

Component	Stream Length[a]					AWC Stream Length[b]				Salmon Species Present in Lost[c] Streams	TOTAL AWC Stream Length Lost[c] to Footprint
	Eliminated by Footprint	Blocked by Footprint	Dewatered by Footprint	Blocked and Dewatered by Footprint	TOTAL Stream Length Lost[c] to Footprint	Eliminated by Footprint	Blocked by Footprint	Dewatered by Footprint	Blocked and Dewatered by Footprint		
Pebble 0.25											
Pit	3.0	0.0	13.4	1.4	17.8	0.0	0.0	1.5	0.0	Chinook, coho	1.5
Waste rock	5.1	<0.1	NA	NA	5.1	0.0	0.0	NA	NA		0.0
TSF 1[d]	12.3	2.7	NA	NA	15.0	6.1	0.0	NA	NA	Chinook, coho	6.1
TOTAL	**20.4**	**2.8**	**13.4**	**1.4**	**38.0**	**6.1**	**0.0**	**1.5**	**0.0**		**7.7**
Pebble 2.0											
Pit + waste rock	56.9	10.2	1.9	4.8	73.8	11.3	1.7	0.0	2.4	Chinook, coho, sockeye	15.4
TSF 1[d]	15.4	0.2	NA	NA	15.5	6.3	0.0	NA	NA	Chinook, coho	6.3
TOTAL	**72.3**	**10.4**	**1.9**	**4.8**	**89.4**	**17.6**	**1.7**	**0.0**	**2.4**		**21.7**
Pebble 6.5											
Pit + waste rock	76.9	5.9	3.4	7.7	93.9	18.7	0.0	0.0	1.6	Chinook, coho, sockeye	20.3
TSF 1	15.4	0.2	NA	NA	15.5	6.3	0.0	NA	NA	Chinook, coho,	6.3
TSF 2	28.3	2.2	NA	NA	30.5	6.1	0.0	NA	NA	Chinook, chum, coho	6.1
TSF 3	10.2	0.7	NA	NA	10.9	3.3	0.0	NA	NA	Coho	3.3
TOTAL	**130.8**	**9.0**	**3.4**	**7.7**	**150.9**	**34.4**	**0.0**	**0.0**	**1.6**		**36.0**

Notes:
[a] From the National Hydrography Dataset (USGS 2012a).
[b] From the Anadromous Waters Catalog (Johnson and Blanche 2012)
[c] Lost = eliminated + blocked + dewatered.
[d] TSF 1 expands in size in the Pebble 2.0 scenario.
TSF = tailings storage facility; AWC = Anadromous Waters Catalog; NA = not applicable as the mine pit dewatering zone does not overlap these individual components.

Table 7-6. Distribution of stream channel length classified by channel size (based on mean annual streamflow in m³/s), channel gradient (%), and floodplain potential (based on % flatland in lowland) for streams lost to the Pebble 6.5 mine footprint. Gray shading indicates proportions greater than 5%; bold indicates proportions greater than 10%.

Channel Size	Gradient							
	<1%		≥1% and <3%		≥3% and <8%		≥8%	
	FP	NFP	FP	NFP	FP	NFP	FP	NFP
Small headwater streams[a]	10%	4%	9%	**35%**	0%	**23%**	0%	1%
Medium streams[b]	6%	6%	1%	5%	0%	0%	0%	0%
Small rivers[c]	0%	0%	0%	0%	0%	0%	0%	0%
Large rivers[d]	0%	0%	0%	0%	0%	0%	0%	0%

Notes:
[a] 0–0.15 m^3/s; most tributaries in the mine footprints.
[b] 0.15–2.8 m^3/s; upper reaches and larger tributaries of the North and South Fork Koktuli Rivers and Upper Talarik Creek.
[c] 2.8–28 m^3/s; middle to lower portions of the North and South Fork Koktuli Rivers and Upper Talarik Creek, including the mainstem Koktuli River.
[d] >28 m^3/s; the Mulchatna River below the Koktuli confluence, the Newhalen River, and other large rivers.
FP = high floodplain potential (≥5% flatland in lowland); NFP = no or low floodplain potential (<5% flatland in lowland) (see Chapter 3 for additional details).

Table 7-7 provides a summary of the total documented anadromous fish stream length in the mine scenario watersheds (Johnson and Blanche 2012). Approximately 2, 7, and 11% of the total anadromous fish stream length in the mine scenario watersheds would be eliminated, blocked, or dewatered in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively (Table 7-5). In addition to these direct losses, loss of these headwater habitats would have indirect impacts on fishes and their habitats in downstream mainstem reaches of each watershed (Section 7.2.3).

Table 7-7. Total documented anadromous fish stream length and stream length documented to contain different salmonid species in the mine scenario watersheds.

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

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

7.2.2.2 Wetland, Pond, and Lake Losses

In addition to the stream losses detailed above, 4.5, 12, and 18 km^2 of wetlands and 0.41, 0.93, and 1.8 km^2 of ponds and lakes would be lost in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. (Table 7-8, Figures 7-10 through 7-12). Methods used to estimate these losses are described in Box 7-1.

7.2.3 Exposure-Response: Implications of Stream and Wetland Loss for Fish

7.2.3.1 Fish Occurrence in Streams and Wetlands Lost to the Mine Scenario Footprints

Tables 7-5 and 7-8 provide an estimate of salmon habitat directly affected by the mine footprint in the three mine scenarios. A total of 8, 22, and 36 km of documented anadromous fish streams would be eliminated, blocked, or dewatered by the mine footprints in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. The distribution of anadromous Dolly Varden in the Nushagak and Kvichak River watersheds is not fully known, making our estimate of the total anadromous fish habitat affected by the mine scenarios incomplete. Of the total wetland area eliminated, blocked, or dewatered by each 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 deposit area, due to differences in the duration and timing of surface water connectivity with stream habitats, distance from the main channel, and 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 deposit area, it is not possible to calculate the effects of lost wetland connectivity and abundance on stream fish populations.

Among the Pacific salmon species, coho salmon occupy the highest proportion of designated AWC stream segments in the mine scenario watersheds (Table 7-7). Spawning habitat for coho salmon would be lost in the South and North Fork Koktuli River watersheds under TSF 1 and TSF 2, respectively (Figure 7-3); sockeye and coho salmon spawning habitat would be lost in the Upper Talarik Creek watershed under the waste rock piles (Figures 7-2 and 7-3) (Johnson and Blanche 2012). In other regions, anadromous and resident forms of Dolly Varden have 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). Under the Pebble 6.5 footprint, 99% of stream kilometers are estimated to have gradients less than 8% and 76% are estimated to have gradients less than 3%, well within the range of gradients used by these species.

Table 7-8. Wetland, pond, and lake areas[a] (km²) eliminated, blocked, or dewatered by the mine footprints in the Pebble 0.25, 2.0, and 6.5 scenarios.

	Eliminated by Footprint				Blocked by Footprint				Dewatered by Footprint				Blocked and Dewatered by Footprint				Total Area Lost to Footprint			
Component	Wetland	Pond	Lake	Sum	Wetland	Pond	Lake	Sum	Wetland	Pond	Lake	Sum	Wetland	Pond	Lake	Sum	Wetland	Pond	Lake	Sum
Pebble 0.25																				
Pit	0.26	0.02	0.00	0.27	0.00	0.00	0.00	0.00	1.30	0.28	0.02	1.60	0.03	0.03	0.00	0.06	1.59	0.32	0.02	1.93
Waste rock	0.29	0.07	0.00	0.36	0.00	0.00	0.00	0.00	NA	NA	NA	NA	NA	NA	NA	NA	0.29	0.07	0.00	0.36
TSF1	2.33	<0.01	0.00	2.34	0.31	0.00	0.00	0.31	NA	NA	NA	NA	NA	NA	NA	NA	2.64	<0.01	0.00	2.64
Total	2.88	0.09	0.00	2.97	0.31	0.00	0.00	0.31	1.30	0.28	0.02	1.60	0.03	0.03	0.00	0.06	4.52	0.39	0.02	4.93
Pebble 2.0																				
Pit + waste rock	5.86	0.63	0.12	6.61	1.67	0.06	0.00	1.73	0.33	0.08	0.00	0.41	0.15	0.04	0.00	0.19	8.01	0.81	0.12	8.94
TSF1[b]	3.56	<0.01	0.00	3.57	0.00	0.00	0.00	0.00	NA	NA	NA	NA	NA	NA	NA	NA	3.56	<0.01	0.00	3.57
Total	9.42	0.64	0.12	10.18	1.67	0.06	0.00	1.73	0.33	0.08	0.00	0.41	0.15	0.04	0.00	0.19	11.57	0.82	0.12	12.51
Pebble 6.5																				
Pit + waste rock	10.16	0.88	0.70	11.74	0.24	0.01	0.00	0.26	0.73	0.18	0.00	0.91	0.72	0.03	0.00	0.75	11.86	1.10	0.70	13.66
TSF1	3.56	<0.01	0.00	3.57	0.00	0.00	0.00	0.00	NA	NA	NA	NA	NA	NA	NA	NA	3.56	<0.01	0.00	3.57
TSF2	1.94	<0.01	0.00	1.94	0.02	0.00	0.00	0.02	NA	NA	NA	NA	NA	NA	NA	NA	1.96	<0.01	0.00	1.96
TSF3	0.54	0.01	0.00	0.55	0.02	0.00	0.00	0.02	NA	NA	NA	NA	NA	NA	NA	NA	0.56	0.01	0.00	0.57
Total	16.20	0.90	0.70	17.80	0.28	0.01	0.00	0.30	0.73	0.18	0.00	0.91	0.72	0.03	0.00	0.75	17.94	1.11	0.70	19.77

Notes:
[a] Based on the National Wetlands Inventory (USFWS 2012).
[b] TSF 1 expands in size in the Pebble 2.0 scenario.
TSF = tailings storage facility; NA = not applicable as the mine pit dewatering zone does not overlap with the footprints of these individual components.

In addition to spawning, streams in each mine footprint provide rearing habitat for fish of the mine scenario watersheds. Species known to rear in habitats in and upstream of the mine footprints are sockeye salmon (Figure 7-2), coho salmon (Figure 7-3), Chinook salmon (Figure 7-4), chum salmon (Figure 7-5), Dolly Varden (Figure 7-7), rainbow trout (Figure 7-8), Arctic grayling, slimy sculpin, northern pike, and ninespine stickleback (ADF&G 2012, Johnson and Blanche 2012).

7.2.3.2 Importance of Headwater Stream and Wetland Habitats

The majority of streams lost to the footprint of the Pebble 6.5 scenario are classified as small headwater streams (less than 0.15 m^3/s mean annual streamflow) (Table 7-6). Because of their narrow width, headwater streams receive proportionally greater inputs of organic material from the surrounding terrestrial vegetation than larger stream channels (Vannote et al. 1980). This material is either used locally (Tank et al. 2010) or transported downstream as a subsidy to larger streams in the network (Wipfli et al. 2007). Consumers in headwater stream foodwebs, such as invertebrates and juvenile salmon, can rely heavily on terrestrial inputs that enter the stream (Doucett et al. 1996, Dekar et al. 2012). Headwater streams also encompass the upper limits of anadromous fish distribution. These streams may receive fewer or no marine-derived nutrients (MDN) from spawning salmon relative to downstream portions of the river network, making terrestrial nutrient sources relatively more important (Wipfli and Baxter 2010). 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 most extensive in late summer and early fall (Elliott and Finn 1983). This coincides with maximum growth periods for rearing juvenile salmon, as both stream temperatures and food availability increase (Quinn 2005).

Data on riparian vegetation communities specific to the mine footprints were not available, but vegetation in the deposit area is described generally by 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 are 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 foodwebs 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 deposit area.

In addition to providing summer rearing habitat, lower-gradient headwater streams and associated wetlands may also provide important habitat for stream fishes during other seasons. Loss of wetlands is a common result 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). Thermally diverse habitats in off-channel wetlands can 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 fish in the mine scenario watersheds given the relatively cold temperatures and long winters in the region.

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 stream fishes (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 by trapping organic materials and nutrients and creating structurally complex, large capacity pool habitats with potentially high macrophyte cover, low streamflow velocity, and/or moderate temperatures (Nickelson et al. 1992, Collen and Gibson 2001, Lang et al. 2006). Additionally, beaver dams, including ponds at a variety of successional stages, provide a mosaic of habitats for not just salmon but other fish and wildlife species.

An aerial survey of active beaver dams in the deposit area, conducted in October 2005 (PLP 2011: Chapter 16:16.2-8), mapped 113 active beaver colonies. The area surveyed did not include streams draining the TSF 1 area (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 areas. Beaver ponds provide important and relatively abundant habitat within the deposit area and may be particularly important for overwinter rearing of species such as coho salmon and for providing deeper pool habitats for additional species during low streamflow conditions (PLP 2011: Appendix 15.1D). 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.

Inputs of groundwater-influenced streamflow from headwater tributaries likely benefit fish by moderating mainstem temperatures and contributing to thermal diversity in downstream waters (Cunjak 1996, Power et al. 1999, Huusko et al. 2007, Armstrong et al. 2010, Brown et al. 2011). PLP (2011) collected temperature data from stream sampling sites using in-situ field meters (PLP 2011: Appendix 15.1-E). Maximum summer (June through August) water temperatures recorded at gage NK119A, which drains the TSF 1 area, 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 at gage SK119A, which drains the TSF 2 area and where recorded maximum summer water temperatures were 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 South and North Fork Koktuli River watersheds from August and October indicate that the mainstem reaches just downstream of the tributaries draining TSF 1 and TSF 2 experience significant summer cooling and winter warming compared to adjacent upstream reaches (PLP 2011: Figures 15.1-11 and 15.1-41). Such thermal diversity can be an important attribute of stream systems in the region, providing localized water temperature patches that may offer differing trade-offs for species bioenergetics. For example, salmon may select relatively cold-temperature sites—often associated with groundwater upwelling—for spawning, whereas juvenile salmon rearing in those same streams may take advantage of warm-temperature patches for optimal food assimilation (e.g., Armstrong and Schindler 2013). Headwater streams in the South and North Fork Koktuli River watersheds may provide a temperature-moderating effect and serve as sources of thermal heterogeneity, providing cooler temperatures in summer and warmer temperatures in winter.

It has long been recognized that, in addition to providing habitat for stream fishes, headwater streams and wetlands serve an important role in the stream network by contributing water, nutrients, organic material, macroinvertebrates, algae, and bacteria downstream to higher-order streams in the watershed (Vannote et al. 1980, Meyer et al. 2007). However, only recently have specific subsidies from headwater streams been extensively quantified (Wipfli and Baxter 2010). Headwater contributions to downstream systems result from the high density of headwaters in the dendritic stream network. Headwater streams can also have high instream rates of nutrient processing and storage, thereby influencing downstream water chemistry due to relatively large organic matter inputs, high retention capacity, high primary productivity, bacteria-induced decomposition, and/or extensive hyporheic zone interactions (Richardson et al. 2005, Alexander et al. 2007, Meyer et al. 2007).

Wipfli and Gregovich (2002) demonstrated that invertebrates and detritus are exported from headwaters to downstream reaches and provide an important energy subsidy for juvenile salmonids. Wipfli and Baxter (2010) describe how the relative importance of energy subsidies from headwaters, terrestrial inputs, benthic production, and marine sources varies within salmon watersheds based on spatial and temporal context. For example, foodwebs in small headwater streams of the mine scenario watersheds may be proportionally more dependent on local terrestrial energy subsidies, whereas stream communities in downstream waters may be more dependent on large seasonal fluxes of MDN. Small headwater streams can be important exporters of subsidies to downstream waters, but the relative value of this contribution will depend on the quantity and energy content of headwater-derived subsidies relative to other energy sources (e.g., MDN, benthic production) that vary in time and space (Wipfli and Baxter 2010).

The export value of headwater streams can be mediated by the surrounding vegetation. In southeastern Alaskan streams, 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). In south-central Alaskan streams on the Kenai Peninsula, grass-dominated headwater wetlands and associated vegetation can also be important sources of dissolved organic matter, particulate organic matter, and macroinvertebrate diversity, contributing to the chemical, physical, and biological condition of streams draining these landscapes (Shaftel et al. 2011, Dekar et al.

2012, King et al. 2012, Walker et al. 2012). Because of their crucial influence on downstream water flow, chemistry, and biota, impacts on headwaters reverberate throughout entire watersheds downstream (Freeman et al. 2007, Meyer et al. 2007).

7.2.4 Risk Characterization

Direct loss of streams and wetlands to the mine footprints would make these habitats unavailable to fishes. Such losses would be unavoidable for projects of the sizes described in our mine scenarios, due to the density of streams and wetlands in the deposit area (combined 33% of the mine mapping area [PLP 2011: Table 14.1-5 and Figure 14.1-3]). Stream blockage is not necessarily unavoidable, but would require appropriate engineering and maintenance. Indirect effects of headwater stream and wetland losses due to the mine footprints would include reduced inputs of organic material, nutrients, water, and macroinvertebrates to downstream reaches, but the relative effects of losses of upstream subsidies would be highly context-dependent (Section 7.2.3).

The net effects of headwater stream and wetland losses would reduce the capacity and productivity of stream habitats. Together, these reductions would result in adverse impacts on fish populations (Figure 7-1). These streams provide known spawning and rearing habitats for anadromous and resident fish species, and their watersheds support some of region's highest diversity of salmonid species (Figure 5-3). The lengths of streams lost directly to the Pebble 0.25, 2.0, and 6.5 mine footprints represent losses of approximately 2, 7, and 11%, respectively, of the total AWC length in the mine scenario watersheds (Table 7-7). Stream habitat losses leading to losses of local, unique populations would erode the population diversity that is crucial to the stability of the overall Bristol Bay salmon fishery (Hilborn et al. 2003, Schindler et al. 2010).

Impact avoidance and minimization measures would not eliminate all the footprint impacts associated with the mine scenarios, given the large extent and wide distribution of wetlands and streams in the watersheds, the substantial infrastructure needed to support porphyry copper mining in this vast undeveloped area, and the constraints that the ore body location puts on infrastructure siting options. Compensatory mitigation measures could offset some of the stream and wetland losses described here (Box 7-2), although the potential efficacy, applicability, and sustainability of these measures to successfully offset adverse impacts face substantial challenges. Appendix J presents a more detailed discussion of these compensatory mitigation issues.

BOX 7-2. COMPENSATORY MITIGATION

Compensatory mitigation refers to the restoration, establishment, enhancement, and/or preservation of wetlands, streams, or other aquatic resources. Compensatory mitigation regulations jointly promulgated by the U.S. Environmental Protection Agency (USEPA) and the U.S. Army Corps of Engineers (USACE) state that "the fundamental objective of compensatory mitigation is to offset environmental losses resulting from unavoidable impacts to waters of the United States authorized by [Clean Water Act Section 404 permits issued by the USACE]" (40 Code of Federal Regulations [CFR] 230.93(a)(1)). Compensatory mitigation enters the analysis only after a proposed project design has incorporated all appropriate and practicable means to avoid and minimize adverse impacts on aquatic resources (40 CFR 230.91(c)). Compensatory mitigation measures are usually not part of project design, but are considered necessary to maintain the integrity of the nation's waters. In addition, guidance issued by the USACE Alaska District in 2009 clarifies that fill placed in streams or in wetlands adjacent to anadromous fish streams in Alaska will require compensatory mitigation (USACE 2009). A 2011 supplement to the Alaska District's 2009 guidance further recommends that projects in "difficult to replace" wetlands, fish-bearing waters, or wetlands within 500 feet of such waters will also likely require compensatory mitigation, as will "large scale projects with significant aquatic resource impacts," such as "mining development" (USACE 2011).

The mine scenarios evaluated in this assessment identify that the mine footprints alone will result in the loss (i.e., filling, blocking or otherwise eliminating) of high-functioning wetlands and tens of kilometers of salmon-supporting streams. Appendix J provides an overview of Clean Water Act (CWA) Section 404 compensatory mitigation requirements for unavoidable impacts on aquatic resources and discusses the likely efficacy of these potential compensation measures at offsetting potential adverse impacts. Note that any formal determinations regarding compensatory mitigation can only take place in the context of a regulatory action. This assessment is not a regulatory action, and thus a complete evaluation of compensatory mitigation is outside the scope of this assessment.

Potential compensatory mitigation measures discussed in Appendix J include mitigation bank credits, in-lieu fee program credits, and permittee-responsible compensatory mitigation projects, such as aquatic resource restoration and enhancement within the South and North Fork Koktuli River and Upper Talarik Creek watersheds as well as more distant portions of the Nushagak and Kvichak River watersheds. As discussed in Appendix J, there are significant challenges regarding the potential efficacy, applicability, and sustainability of compensation measures for use in the Bristol Bay region, raising questions as to whether compensation measures could realistically address impacts of this type and magnitude.

7.2.5 Uncertainties

Losses of anadromous fish-bearing streams in the mine scenario watersheds (Table 7-5) are likely underestimated because of the difficulty of accurately capturing data on all streams that may support fish use throughout the year. We rely on the AWC (Johnson and Blanche 2012) and the AFFI (ADF&G 2012) for documentation of species distributions, but these records are 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 by 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 in the mine scenario watersheds may also be underestimated because of challenges associated with stream network mapping. Estimates of headwater stream extent were derived from the National Hydrography Dataset (NHD) for Alaska

(USGS 2012a), which does not capture all stream courses and may underestimate channel sinuosity, resulting in underestimates of stream length. A stream network map derived from a light detection and ranging (LiDAR) mapping system would likely yield substantially different results than those presented here. Similarly, actual wetland loss or blockage due to the mine footprints (Table 7-8) would likely be higher than estimated here, as the National Wetlands Inventory (USFWS 2012) is based on remotely-sensed imagery and generally underestimates wetland area. See Box 7-1 for additional discussion of uncertainties associated with stream and wetland mapping.

In the Bristol Bay region, hydrologically diverse riverine and wetland landscapes provide a variety of large river, floodplain, pond, and lake habitats for salmon spawning and rearing. Environmental conditions can be very different among habitats in close proximity. The spatial separation and unique spawning habitat features within the Bristol Bay watershed are associated with variation in life-history characteristics and body morphology (Blair et al. 1993), and 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 differences in traits, such as spawn timing, spawn site fidelity, and productivity, that are consistent with discrete populations (Quinn et al. 2012). In the Bristol Bay region, phenotypic variation with apparent adaptive significance has been illustrated for sockeye salmon egg size and spawning gravel size (Quinn et al. 1995), and for sockeye salmon body shape and predation risk from brown bears (Quinn et al. 2001). Olsen et al. (2003) proposed that the fine-scale genetic differentiation they observed in Alaskan coho salmon may be associated with local adaptation to locally diverse freshwater selective pressures, but they did not examine phenotypic variation. These results highlight the potential for fine-scale salmon population structure in the Bristol Bay watershed. Current monitoring approaches are inadequate to fully assess population-level trends across the Bristol Bay watershed (Rand et al. 2007). Additional genetic and ecological research is needed to clarify the spatial scale of this population structure and the varying vulnerabilities of populations across the landscape.

7.3 Streamflow Modification

7.3.1 Exposure: Streamflow

In this section, we describe projected changes in the hydrology of the mine scenario watersheds and associated effects on downstream flows that would result from mine development and operation. We assume that streams in and downstream of the mine footprints would experience streamflow alterations due to water collection, treatment, and discharge to streams via wastewater treatment plant (WWTP) outfalls; leakage from TSFs; and leachate from waste rock piles. See Chapter 6 for a full description of water flows through the mine facilities.

Streamflow alterations resulting from mine operations were estimated by reducing the streamflows recorded at existing stream gages in the mine scenario watersheds (Table 7-9, Figures 7-14 through 7-

16) by the percentage of expected surface area lost to each mine footprint and water yield efficiencies for each watershed. Reductions also included losses to the drawdown zone, caused by the cone of depression at the mine pit, or other locations of dewatering (Table 7-9, Section 6.2.2). Discharges through the WWTP resulted in streamflow additions. Net effects on resulting streamflows were mapped and summarized for individual stream and river segments (Figures 7-14 through 7-16).

Table 7-9. Stream gages and related characteristics for the South and North Fork Koktuli Rivers and Upper Talarik Creek.

Stream and Gage	Drainage Area (km^2)	Mean Annual Streamflow[a] (m^3/s)	Mean Annual Unit Runoff ($m^3/s*km^2$)
South Fork Koktuli River			
SK100G	14	0.4	0.026
SK100F	31	0.8	0.026
SK124A	22	0.5	0.024
SK100C	99	1.3	0.013
SK119A	28	1.0	0.036
SK100B1	141	3.7	0.026
SK100B[b]	179	5.1	0.029
North Fork Koktuli River			
NK119A	20	0.7	0.034
NK119B	11	0.1	0.012
NK100C	65	1.3	0.020
NK100B	99	2.4	0.024
NK100A1	222	5.8	0.026
NK100A[c]	279	7.0	0.025
Upper Talarik Creek			
UT100E	10	0.3	0.027
UT100D	31	0.8	0.025
UT100C2	133	2.9	0.022
UT100C1	159	3.4	0.022
UT100C	185	4.5	0.024
UT119A	10	0.8	0.079
UT100B[d]	222	6.2	0.028

Notes:
[a] Calculated from stream gage data from PLP 2011.
[b] USGS 15302200.
[c] USGS 15302250.
[d] USGS 15300250.

Figure 7-14. Stream segments in the mine scenario watersheds showing streamflow changes (%) associated with the Pebble 0.25 footprint. Streamflow modification class is shown for each stream segment to indicate degree and direction of change. These classes are assigned at a gage and extend upstream to the next gage, confluence point, or mine footprint. Channels and tributaries not classified are shown for informational purposes. Gage locations based on U.S. Geological Survey (2012b) and Pebble Limited Partnership (2011).

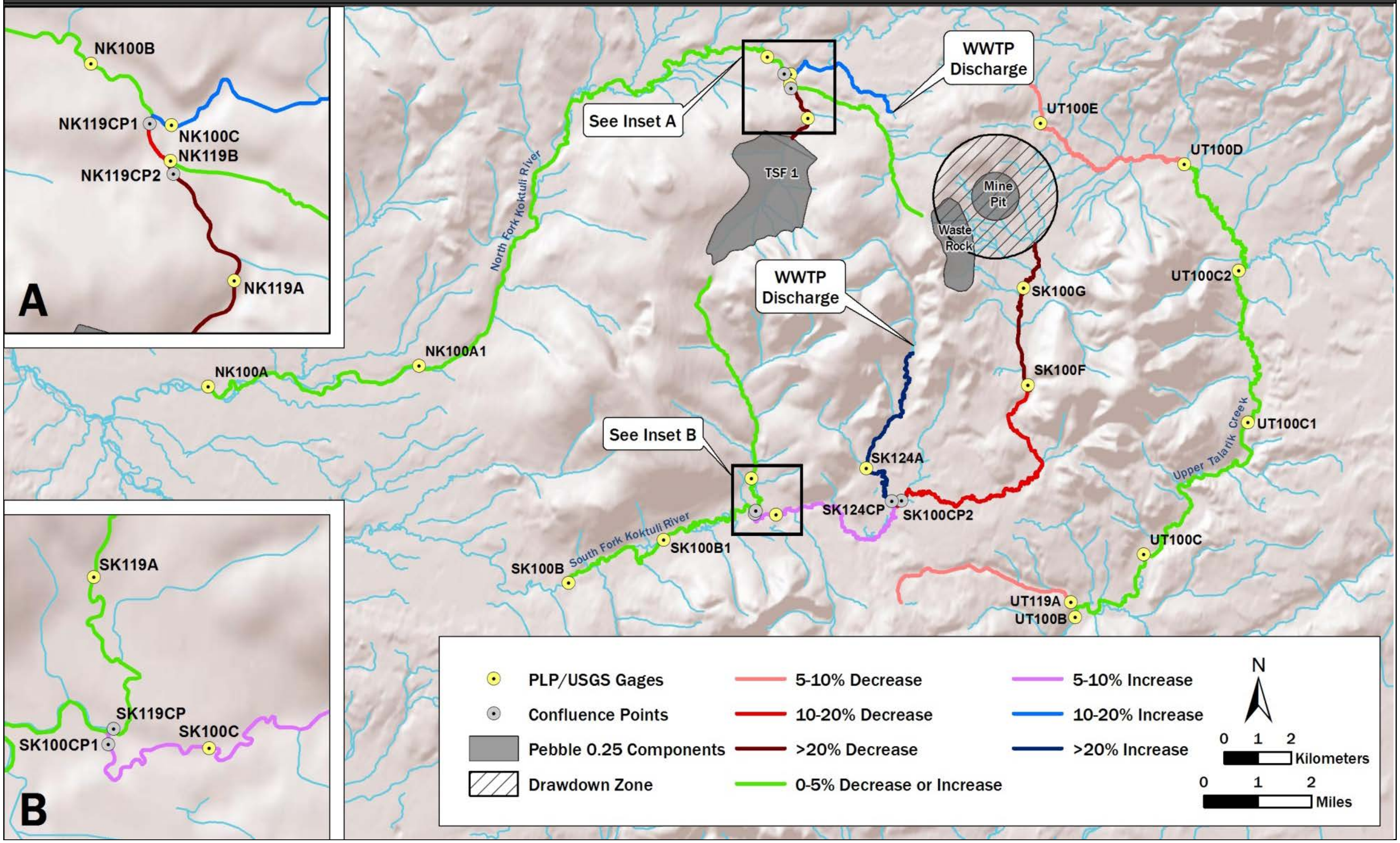


Figure 7-15. Stream segments in the mine scenario watersheds showing streamflow changes (%) associated with the Pebble 2.0 footprint. Streamflow modification class is shown for each stream segment to indicate degree and direction of change. These classes are assigned at a gage and extend upstream to the next gage, confluence point, or mine footprint. Channels and tributaries not classified are shown for informational purposes. Gage locations based on U.S. Geological Survey (2012b) and Pebble Limited Partnership (2011).

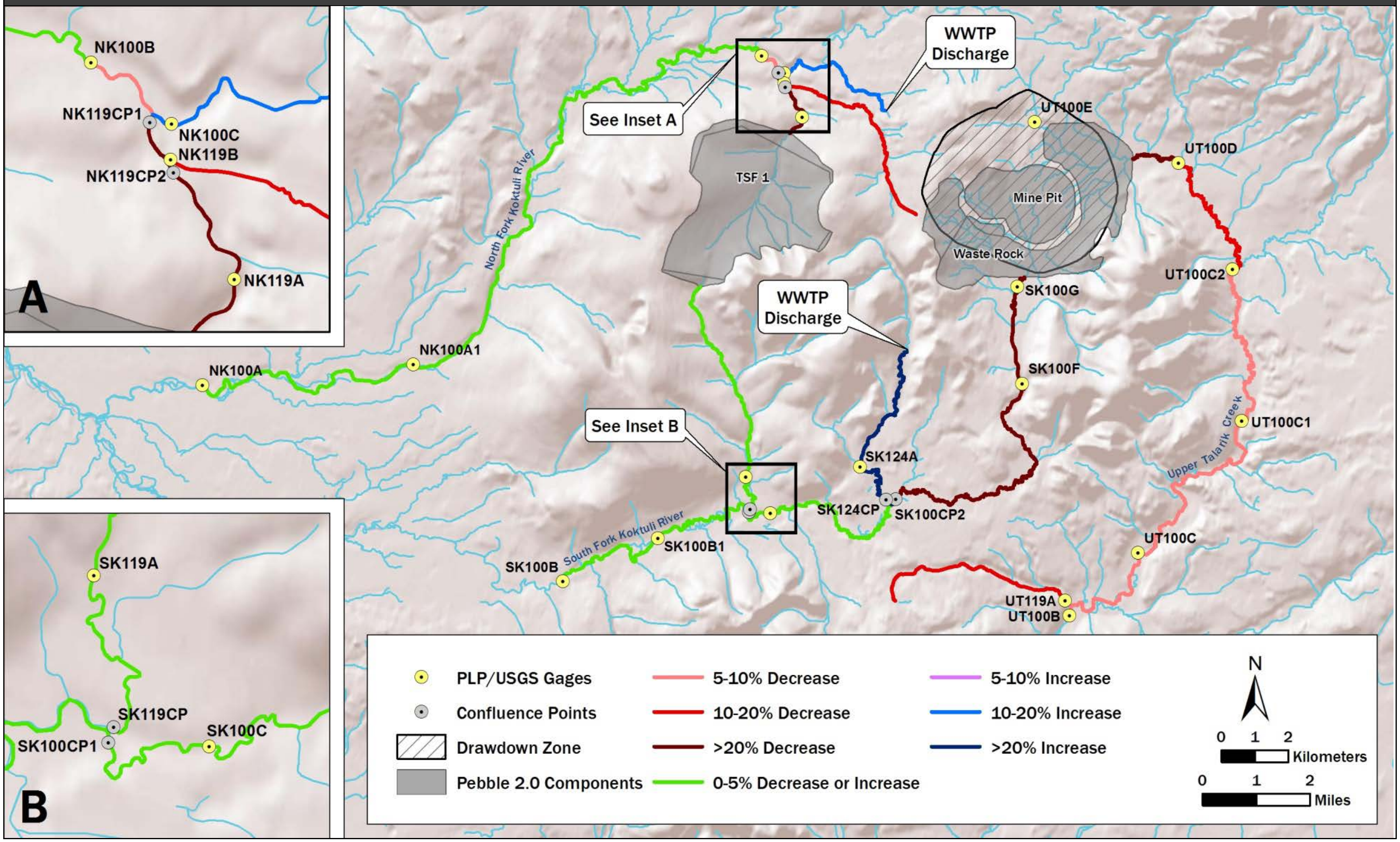
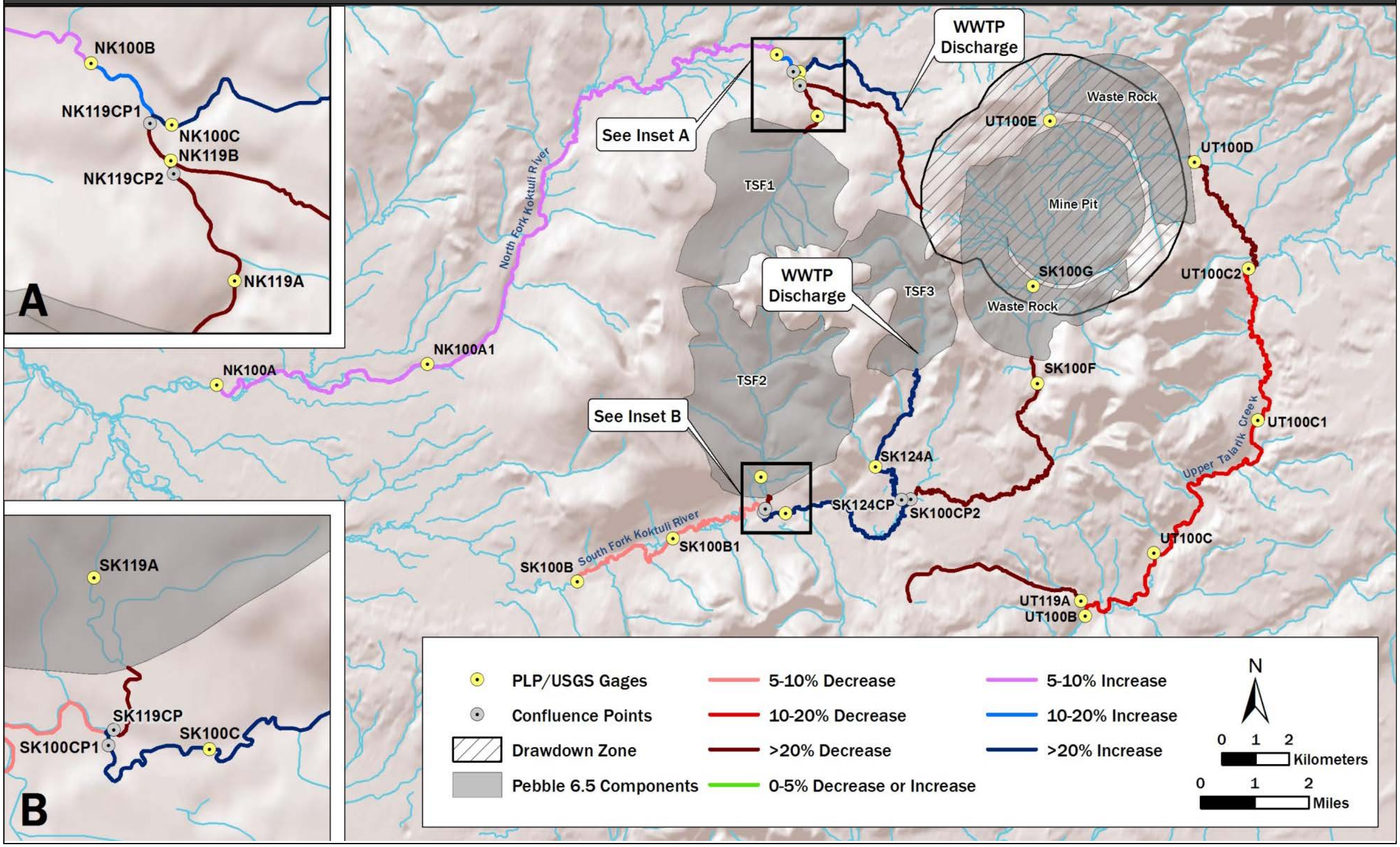


Figure 7-16. Stream segments in the mine scenario watersheds showing streamflow changes (%) associated with the Pebble 6.5 footprint. Streamflow modification class is shown for each stream segment to indicate degree and direction of change. These classes are assigned at a gage and extend upstream to the next gage, confluence point, or mine footprint. Channels and tributaries not classified are shown for informational purposes. Gage locations based on U.S. Geological Survey (2012b) and Pebble Limited Partnership (2011).



Daily streamflow data were obtained using data from seven gages in the South Fork Koktuli River, six gages in the North Fork Koktuli River, and seven gages in Upper Talarik Creek (Table 7-9) (PLP 2011). We calculated mean and minimum monthly streamflows for each gage under pre-mining baseline conditions (Tables 7-10 through 7-15, Figure 7-17). The periods of record varied for gages in the three mine scenario watersheds, but generally covered the period from 2004 to 2010.

In addition, we estimated streamflow at six confluence points where mining-related streamflow impacts were expected but where established stream gage records were lacking. This allowed for more discrete estimation of baseline streamflow, as well as expected streamflow modification in each mine scenario due to withdrawal, addition, or footprint loss. The tributary area to each stream gage or confluence point was calculated based on the National Elevation Dataset digital elevation model (Gesch et al. 2002, Gesch 2007, USGS 2013) in a geographic information system. We determined the area of each mine component (i.e., the mine pit, waste rock piles, plant and ancillary areas, and TSFs) (Tables 6-5 through 6-7) in each drainage basin (Tables 7-16 through 7-18), and calculated the percentage of watershed area covered by the mine components for each gage and confluence point subwatershed. Using the calculated percentage of watershed area covered by the mine components, mean annual streamflow records for each of the gages and confluence point subwatersheds were adjusted downward. Next, the annual volume of return streamflow expected to reach each gage was added back to the adjusted streamflow calculations based on the mine scenarios.

We assessed expected changes to surface water flows for the three mine scenarios (Tables 7-10 through 7-15). We also considered water balance issues for the post-closure period, but streamflow estimates were not assessed for this period. The Pebble 0.25 mine footprint consists of the mine pit, its drawdown zone (Section 6.2.2), one waste rock pile, plant and ancillary facilities, and TSF 1 (Table 6-5). The Pebble 2.0 footprint would add a second or expanded waste rock pile, larger areas for plant and ancillary facilities, an expanded TSF 1, and a larger drawdown zone from groundwater flow to the pit (Table 6-6). The Pebble 6.5 footprint would add effects associated with the fully expanded mine footprint (including TSF 2 and TSF 3) to accommodate expanded mine operations (Table 6-7). We assume that during the post-closure period, active dewatering of the pit would cease as the pit fills. Once the pit is filled, the water level would be maintained below equilibrium level by pumping or gravity drainage to maintain a gradient toward the pit. The pumped water would be treated for as long as it did not meet water quality standards. When treatment was no longer necessary, the pit would be allowed to have a natural outlet if the water level required one.

Table 7-10. Measured mean monthly pre-mining streamflow rates (m^3/s) and estimated mean monthly streamflow rates (m^3/s) in the Pebble 0.25, 2.0, and 6.5 scenarios, for gages along the South Fork Koktuli River.

	SK100G				SK100F				SK124A				SK100C				SK119A				SK100B1				SK100B[a]			
Month	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5
Jan	**0.23**	0.11	0.07	NA	**0.44**	0.32	0.28	0.17	**0.12**	0.16	0.16	0.26	**0.37**	0.39	0.38	0.51	**0.30**	0.30	0.30	NA	**1.54**	1.52	1.48	1.40	**2.47**	2.44	2.39	2.23
Feb	**0.14**	0.06	0.04	NA	**0.25**	0.18	0.16	0.09	**0.02**	0.03	0.03	0.05	**0.03**	0.03	0.03	0.04	**0.16**	0.16	0.15	NA	**0.79**	0.78	0.76	0.72	**1.40**	1.39	1.35	1.27
Mar	**0.11**	0.05	0.03	NA	**0.19**	0.14	0.12	0.07	**0.01**	0.01	0.01	0.01	**<0.01**	<0.01	<0.01	<0.01	**0.11**	0.11	0.11	NA	**0.57**	0.57	0.55	0.52	**1.09**	1.07	1.05	0.98
Apr	**0.18**	0.08	0.05	NA	**0.24**	0.18	0.16	0.09	**0.05**	0.06	0.06	0.10	**0.13**	0.13	0.13	0.18	**0.22**	0.22	0.22	NA	**0.80**	0.79	0.77	0.73	**1.41**	1.39	1.36	1.27
May	**0.72**	0.33	0.20	NA	**1.95**	1.44	1.26	0.74	**1.91**	2.54	2.49	4.10	**4.30**	4.52	4.37	5.87	**3.02**	3.02	2.97	NA	**10.75**	10.62	10.32	9.74	**12.70**	12.53	12.24	11.46
Jun	**0.50**	0.23	0.14	NA	**1.38**	1.02	0.90	0.53	**1.08**	1.43	1.41	2.32	**2.77**	2.90	2.81	3.77	**1.71**	1.71	1.69	NA	**6.67**	6.59	6.40	6.04	**8.56**	8.44	8.25	7.73
Jul	**0.29**	0.13	0.08	NA	**0.59**	0.44	0.38	0.23	**0.29**	0.39	0.38	0.63	**0.73**	0.77	0.75	1.00	**0.74**	0.74	0.73	NA	**2.56**	2.53	2.46	2.32	**3.85**	3.80	3.71	3.48
Aug	**0.42**	0.19	0.12	NA	**0.83**	0.62	0.54	0.32	**0.59**	0.78	0.77	1.27	**1.17**	1.23	1.19	1.60	**1.15**	1.15	1.13	NA	**4.05**	4.00	3.89	3.67	**5.92**	5.84	5.70	5.34
Sep	**0.55**	0.25	0.15	NA	**1.20**	0.89	0.78	0.46	**0.83**	1.10	1.08	1.79	**2.05**	2.15	2.08	2.79	**1.75**	1.75	1.73	NA	**5.18**	5.11	4.97	4.69	**7.75**	7.64	7.47	6.99
Oct	**0.64**	0.29	0.18	NA	**1.47**	1.08	0.95	0.56	**0.98**	1.29	1.27	2.10	**2.80**	2.93	2.84	3.81	**1.61**	1.61	1.59	NA	**6.12**	6.05	5.88	5.55	**9.08**	8.96	8.76	8.20
Nov	**0.35**	0.16	0.10	NA	**0.75**	0.55	0.48	0.28	**0.33**	0.43	0.43	0.70	**1.04**	1.09	1.06	1.42	**0.72**	0.72	0.70	NA	**2.84**	2.81	2.73	2.58	**4.44**	4.38	4.28	4.01
Dec	**0.28**	0.13	0.08	NA	**0.53**	0.39	0.35	0.20	**0.18**	0.24	0.23	0.38	**0.54**	0.57	0.55	0.74	**0.40**	0.40	0.39	NA	**1.92**	1.89	1.84	1.74	**3.02**	2.98	2.91	2.73

Notes:
[a] USGS 15302200.
NA = not applicable: SK100G would be eliminated by tailings storage facility (TSF) 2, and SK119A would be eliminated by TSF 3 in the Pebble 6.5 scenario.

Table 7-11. Measured mean monthly pre-mining streamflow rates (m^3/s) and estimated mean monthly streamflow rates (m^3/s) in the Pebble 0.25, 2.0, and 6.5 scenarios, for gages along the North Fork Koktuli River.

	NK119A				NK119B				NK100C				NK100B				NK100A1				NK100A[a]			
Month	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5
Jan	**0.15**	0.10	0.05	0.06	**0.03**	0.03	0.03	0.02	**0.71**	0.80	0.79	1.13	**1.04**	1.06	0.97	1.23	**2.08**	2.09	2.01	2.22	**2.85**	2.86	2.78	3.02
Feb	**0.10**	0.07	0.04	0.04	**0.01**	0.01	0.01	0.01	**0.48**	0.54	0.53	0.76	**0.67**	0.68	0.63	0.79	**1.44**	1.45	1.39	1.54	**1.88**	1.89	1.83	1.99
Mar	**0.08**	0.06	0.03	0.03	**<0.01**	<0.01	<0.01	<0.01	**0.39**	0.44	0.43	0.62	**0.54**	0.55	0.50	0.64	**1.23**	1.23	1.19	1.31	**1.55**	1.56	1.52	1.65
Apr	**0.21**	0.15	0.08	0.08	**0.03**	0.03	0.03	0.02	**0.54**	0.61	0.61	0.87	**0.88**	0.89	0.82	1.04	**2.17**	2.18	2.10	2.32	**2.66**	2.68	2.60	2.82
May	**2.28**	1.63	0.86	0.87	**0.54**	0.52	0.48	0.39	**3.48**	3.93	3.90	5.58	**7.03**	7.13	6.57	8.29	**16.57**	16.64	16.01	17.70	**20.10**	20.19	19.59	21.29
Jun	**1.15**	0.82	0.43	0.44	**0.20**	0.20	0.18	0.15	**1.91**	2.16	2.15	3.07	**3.64**	3.69	3.40	4.29	**9.48**	9.51	9.16	10.12	**11.39**	11.44	11.10	12.06
Jul	**0.55**	0.39	0.21	0.21	**0.05**	0.05	0.04	0.04	**1.11**	1.25	1.24	1.78	**2.04**	2.07	1.91	2.41	**5.13**	5.15	4.96	5.48	**5.88**	5.91	5.74	6.23
Aug	**0.71**	0.51	0.27	0.27	**0.10**	0.10	0.09	0.08	**1.24**	1.40	1.38	1.98	**2.44**	2.48	2.29	2.88	**6.21**	6.23	6.00	6.63	**7.40**	7.43	7.21	7.83
Sep	**1.10**	0.79	0.42	0.42	**0.20**	0.19	0.18	0.15	**1.75**	1.98	1.96	2.81	**3.31**	3.35	3.09	3.90	**7.98**	8.02	7.72	8.53	**9.35**	9.39	9.11	9.90
Oct	**1.10**	0.78	0.41	0.42	**0.26**	0.25	0.23	0.19	**2.20**	2.49	2.47	3.53	**4.01**	4.07	3.75	4.73	**9.40**	9.44	9.09	10.04	**11.14**	11.19	10.86	11.80
Nov	**0.52**	0.37	0.20	0.20	**0.09**	0.09	0.08	0.07	**1.24**	1.40	1.39	1.99	**2.12**	2.16	1.99	2.51	**4.79**	4.81	4.63	5.11	**5.95**	5.97	5.80	6.30
Dec	**0.24**	0.17	0.09	0.09	**0.04**	0.04	0.04	0.03	**0.88**	0.99	0.98	1.40	**1.35**	1.37	1.27	1.60	**2.89**	2.90	2.79	3.09	**3.84**	3.85	3.74	4.06

Notes:
[a] USGS 15302250.

Table 7-12. Measured mean monthly pre-mining streamflow rates (m^3/s) and estimated mean monthly streamflow rates (m^3/s) in the Pebble 0.25, 2.0, and 6.5 scenarios, for gages along Upper Talarik Creek.

	UT100E				UT100D				UT100C2				UT100C1				UT100C				UT119A				UT100B[a]			
Month	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5
Jan	**0.15**	0.14	NA	NA	**0.32**	0.32	0.29	0.18	**0.05**	1.32	1.29	1.18	**1.05**	1.74	1.71	1.59	**1.44**	2.45	2.41	2.27	**2.09**	0.76	0.69	0.67	**0.60**	3.62	3.54	3.34
Feb	**0.13**	0.13	NA	NA	**0.28**	0.28	0.25	0.16	**0.04**	1.15	1.13	1.03	**0.92**	1.55	1.52	1.41	**1.28**	2.25	2.22	2.08	**1.92**	0.75	0.69	0.66	**0.60**	3.31	3.23	3.05
Mar	**0.12**	0.11	NA	NA	**0.22**	0.22	0.20	0.12	**0.03**	0.93	0.91	0.84	**0.74**	1.28	1.26	1.17	**1.06**	1.98	1.95	1.83	**1.69**	0.74	0.67	0.65	**0.59**	2.88	2.81	2.66
Apr	**0.18**	0.17	NA	NA	**0.55**	0.55	0.50	0.31	**0.08**	2.06	2.02	1.85	**1.64**	2.51	2.47	2.30	**2.08**	3.44	3.38	3.18	**2.93**	0.78	0.71	0.69	**0.62**	4.79	4.68	4.42
May	**0.61**	0.57	NA	NA	**1.95**	1.95	1.77	1.09	**0.28**	6.64	6.50	5.96	**5.28**	7.43	7.30	6.79	**6.16**	9.11	8.97	8.43	**7.76**	0.88	0.80	0.77	**0.69**	12.80	12.49	11.81
Jun	**0.30**	0.28	NA	NA	**1.02**	1.02	0.93	0.57	**0.15**	4.04	3.96	3.63	**3.22**	4.29	4.22	3.93	**3.56**	5.63	5.55	5.21	**4.80**	0.82	0.75	0.72	**0.65**	7.40	7.22	6.83
Jul	**0.21**	0.19	NA	NA	**0.62**	0.62	0.56	0.34	**0.09**	2.40	2.35	2.16	**1.91**	2.76	2.72	2.53	**2.29**	3.77	3.72	3.49	**3.21**	0.80	0.72	0.70	**0.63**	5.13	5.00	4.73
Aug	**0.23**	0.22	NA	NA	**0.78**	0.78	0.71	0.43	**0.11**	2.81	2.75	2.52	**2.24**	3.30	3.25	3.02	**2.74**	4.38	4.32	4.06	**3.73**	0.81	0.74	0.72	**0.64**	6.48	6.32	5.97
Sep	**0.31**	0.29	NA	NA	**1.03**	1.03	0.94	0.58	**0.15**	4.21	4.12	3.78	**3.35**	4.67	4.59	4.27	**3.87**	6.09	6.00	5.63	**5.19**	0.86	0.78	0.76	**0.68**	7.82	7.63	7.21
Oct	**0.36**	0.34	NA	NA	**1.18**	1.18	1.07	0.66	**0.17**	4.69	4.59	4.21	**3.73**	5.26	5.17	4.81	**4.36**	6.67	6.57	6.18	**5.68**	0.88	0.81	0.78	**0.70**	9.08	8.86	8.37
Nov	**0.25**	0.23	NA	NA	**0.74**	0.74	0.68	0.41	**0.11**	2.98	2.91	2.67	**2.37**	3.67	3.60	3.35	**3.04**	4.59	4.52	4.25	**3.91**	0.84	0.76	0.74	**0.66**	6.33	6.18	5.84
Dec	**0.20**	0.19	NA	NA	**0.52**	0.52	0.47	0.29	**0.07**	2.05	2.01	1.84	**1.64**	2.61	2.57	2.39	**2.17**	3.37	3.31	3.12	**2.87**	0.80	0.73	0.71	**0.63**	5.00	4.88	4.61

Notes:
[a] USGS 15300250.
NA = not applicable: UT100E would be blocked by the waste rock pile in the Pebble 2.0 scenario (Figure 7-15), and by the mine pit in the Pebble 6.5 scenario (Figure 7-16).

Table 7-13. Measured minimum monthly pre-mining streamflow rates (m^3/s) and estimated minimum monthly streamflow rates (m^3/s) in the Pebble 0.25, 2.0, and 6.5 scenarios, for gages along the South Fork Koktuli River.

	SK100G				SK100F				SK124A				SK100C				SK119A				SK100B1				SK100B[a]			
Month	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5
Jan	**0.11**	0.05	0.03	NA	**0.20**	0.15	0.13	0.08	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.12**	0.12	0.11	NA	**0.60**	0.60	0.58	0.55	**1.13**	1.12	1.09	1.02
Feb	**0.08**	0.04	0.02	NA	**0.15**	0.11	0.10	0.06	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.08**	0.08	0.08	NA	**0.40**	0.40	0.39	0.37	**0.85**	0.84	0.82	0.77
Mar	**0.07**	0.03	0.02	NA	**0.11**	0.08	0.07	0.04	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.05**	0.05	0.05	NA	**0.27**	0.26	0.26	0.24	**0.65**	0.64	0.63	0.59
Apr	**0.04**	0.02	0.01	NA	**0.11**	0.08	0.07	0.04	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.05**	0.05	0.05	NA	**0.27**	0.26	0.26	0.24	**0.65**	0.64	0.63	0.59
May	**0.08**	0.04	0.02	NA	**0.14**	0.10	0.09	0.05	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.07**	0.07	0.07	NA	**0.38**	0.37	0.36	0.34	**0.79**	0.78	0.76	0.72
Jun	**0.20**	0.09	0.05	NA	**0.46**	0.34	0.30	0.18	**0.09**	0.12	0.11	0.19	**0.12**	0.12	0.12	0.16	**0.45**	0.45	0.45	NA	**1.51**	1.49	1.45	1.37	**2.49**	2.46	2.40	2.25
Jul	**0.08**	0.04	0.02	NA	**0.22**	0.16	0.14	0.08	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.23**	0.23	0.23	NA	**1.12**	1.10	1.07	1.01	**1.64**	1.62	1.58	1.48
Aug	**0.08**	0.04	0.02	NA	**0.16**	0.12	0.10	0.06	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.13**	0.13	0.13	NA	**0.67**	0.66	0.64	0.61	**1.25**	1.23	1.20	1.12
Sep	**0.06**	0.03	0.02	NA	**0.08**	0.06	0.05	0.03	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.09**	0.09	0.09	NA	**0.51**	0.50	0.49	0.46	**1.02**	1.01	0.98	0.92
Oct	**0.22**	0.10	0.06	NA	**0.63**	0.47	0.41	0.24	**0.23**	0.30	0.29	0.49	**0.71**	0.74	0.72	0.96	**0.45**	0.45	0.45	NA	**2.10**	2.07	2.01	1.90	**3.54**	3.49	3.41	3.19
Nov	**0.18**	0.08	0.05	NA	**0.34**	0.25	0.22	0.13	**0.04**	0.05	0.05	0.08	**0.12**	0.12	0.12	0.16	**0.23**	0.23	0.23	NA	**1.16**	1.14	1.11	1.05	**1.93**	1.90	1.86	1.74
Dec	**0.12**	0.05	0.03	NA	**0.21**	0.16	0.14	0.08	**0.00**	0.00	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.13**	0.13	0.13	NA	**0.66**	0.65	0.64	0.60	**1.22**	1.20	1.17	1.10

Notes:
[a] USGS 15302200.
NA = not applicable: SK100G would be eliminated by tailings storage facility (TSF) 2 and SK119A would be eliminated by TSF 3 in the Pebble 6.5 scenario.

Table 7-14. Measured minimum monthly pre-mining streamflow rates (m^3/s) and estimated minimum monthly streamflow rates (m^3/s) in the Pebble 0.25, 2.0, and 6.5 scenarios, for gages along the North Fork Koktuli River.

	NK119A				NK119B				NK100C				NK100B				NK100A1				NK100A[a]			
Month	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5
Jan	**0.08**	0.05	0.03	0.03	**0.00**	0.00	0.00	0.00	**0.33**	0.37	0.37	0.53	**0.43**	0.44	0.41	0.51	**0.93**	0.93	0.90	0.99	**1.10**	1.11	1.08	1.17
Feb	**0.07**	0.05	0.03	0.03	**0.00**	0.00	0.00	0.00	**0.34**	0.38	0.38	0.54	**0.44**	0.45	0.41	0.52	**0.95**	0.95	0.92	1.01	**1.13**	1.14	1.10	1.20
Mar	**0.06**	0.04	0.02	0.02	**0.00**	0.00	0.00	0.00	**0.23**	0.26	0.26	0.37	**0.33**	0.34	0.31	0.39	**0.80**	0.80	0.77	0.85	**0.91**	0.91	0.88	0.96
Apr	**0.04**	0.03	0.02	0.02	**0.00**	0.00	0.00	0.00	**0.13**	0.15	0.15	0.21	**0.18**	0.19	0.17	0.22	**0.84**	0.84	0.81	0.89	**0.96**	0.97	0.94	1.02
May	**0.01**	0.01	0.00	0.00	**0.00**	0.00	0.00	0.00	**0.38**	0.43	0.43	0.61	**0.54**	0.55	0.51	0.64	**1.16**	1.16	1.12	1.23	**1.44**	1.45	1.41	1.53
Jun	**0.30**	0.22	0.11	0.11	**0.00**	0.00	0.00	0.00	**0.71**	0.80	0.80	1.14	**1.31**	1.33	1.23	1.55	**3.75**	3.76	3.62	4.00	**4.27**	4.29	4.17	4.53
Jul	**0.21**	0.15	0.08	0.08	**0.00**	0.00	0.00	0.00	**0.54**	0.61	0.60	0.86	**1.04**	1.05	0.97	1.23	**2.57**	2.58	2.48	2.75	**2.35**	2.36	2.29	2.49
Aug	**0.14**	0.10	0.05	0.05	**0.00**	0.00	0.00	0.00	**0.46**	0.52	0.52	0.74	**0.96**	0.97	0.89	1.13	**2.02**	2.03	1.96	2.16	**1.93**	1.93	1.88	2.04
Sep	**0.12**	0.09	0.05	0.05	**0.00**	0.00	0.00	0.00	**0.37**	0.42	0.41	0.59	**0.91**	0.92	0.85	1.07	**1.89**	1.90	1.83	2.02	**1.76**	1.76	1.71	1.86
Oct	**0.20**	0.14	0.08	0.08	**0.00**	0.00	0.00	0.00	**0.99**	1.11	1.10	1.58	**1.53**	1.55	1.43	1.80	**3.19**	3.20	3.08	3.41	**4.39**	4.41	4.28	4.65
Nov	**0.12**	0.09	0.05	0.05	**0.00**	0.00	0.00	0.00	**0.50**	0.56	0.56	0.80	**0.71**	0.72	0.67	0.84	**1.51**	1.51	1.46	1.61	**1.98**	1.99	1.93	2.10
Dec	**0.10**	0.07	0.04	0.04	**0.00**	0.00	0.00	0.00	**0.41**	0.46	0.46	0.66	**0.57**	0.58	0.53	0.67	**1.21**	1.21	1.17	1.29	**1.53**	1.54	1.49	1.62

Notes:
[a] USGS 15302250.

Table 7-15. Measured minimum monthly pre-mining streamflow rates (m^3/s) and estimated minimum monthly streamflow rates (m^3/s) in the Pebble 0.25, 2.0, and 6.5 scenarios, for gages along Upper Talarik Creek.

	UT100E				UT100D				UT100C2				UT100C1				UT100C				UT119A				UT100B[a]			
Month	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5	Pre	0.25	2.0	6.5
Jan	**0.09**	0.09	NA	NA	**0.12**	0.11	0.07	0.02	**0.53**	0.52	0.47	0.42	**0.80**	0.78	0.73	0.66	**1.55**	1.53	1.44	1.32	**0.72**	0.65	0.63	0.57	**2.09**	2.04	1.93	1.78
Feb	**0.09**	0.08	NA	NA	**0.10**	0.10	0.06	0.02	**0.47**	0.46	0.42	0.37	**0.73**	0.71	0.66	0.60	**1.48**	1.46	1.37	1.26	**0.71**	0.65	0.63	0.56	**1.98**	1.93	1.83	1.68
Mar	**0.08**	0.07	NA	NA	**0.12**	0.11	0.07	0.02	**0.53**	0.52	0.47	0.42	**0.80**	0.78	0.73	0.66	**1.37**	1.35	1.27	1.17	**0.72**	0.65	0.63	0.57	**2.09**	2.04	1.93	1.78
Apr	**0.07**	0.06	NA	NA	**0.11**	0.10	0.06	0.02	**0.50**	0.48	0.44	0.39	**0.76**	0.75	0.70	0.63	**1.42**	1.40	1.32	1.21	**0.71**	0.65	0.63	0.56	**2.04**	1.99	1.88	1.73
May	**0.10**	0.10	NA	NA	**0.22**	0.20	0.12	0.03	**0.91**	0.89	0.81	0.72	**1.25**	1.23	1.14	1.04	**2.02**	1.99	1.87	1.72	**0.63**	0.58	0.56	0.50	**2.83**	2.76	2.61	2.40
Jun	**0.15**	0.14	NA	NA	**0.23**	0.21	0.13	0.03	**1.46**	1.43	1.31	1.16	**1.57**	1.54	1.44	1.30	**2.85**	2.81	2.64	2.43	**0.62**	0.56	0.54	0.49	**2.58**	2.51	2.38	2.19
Jul	**0.14**	0.13	NA	NA	**0.21**	0.19	0.12	0.03	**1.46**	1.43	1.31	1.17	**1.37**	1.35	1.25	1.14	**2.50**	2.46	2.31	2.13	**0.65**	0.59	0.57	0.51	**2.55**	2.49	2.35	2.16
Aug	**0.12**	0.11	NA	NA	**0.22**	0.20	0.12	0.03	**1.35**	1.32	1.21	1.07	**1.58**	1.55	1.44	1.31	**2.40**	2.36	2.22	2.04	**0.62**	0.57	0.55	0.49	**2.97**	2.90	2.74	2.52
Sep	**0.11**	0.10	NA	NA	**0.20**	0.18	0.11	0.03	**1.29**	1.26	1.15	1.02	**1.52**	1.49	1.39	1.26	**2.37**	2.33	2.19	2.02	**0.67**	0.61	0.59	0.53	**2.83**	2.76	2.61	2.40
Oct	**0.17**	0.16	NA	NA	**0.33**	0.30	0.18	0.05	**1.71**	1.67	1.53	1.36	**2.24**	2.20	2.05	1.86	**3.03**	2.98	2.80	2.58	**0.69**	0.62	0.60	0.54	**3.82**	3.73	3.52	3.24
Nov	**0.16**	0.15	NA	NA	**0.31**	0.28	0.17	0.04	**1.34**	1.32	1.21	1.07	**2.04**	2.00	1.86	1.69	**2.36**	2.32	2.18	2.01	**0.78**	0.71	0.68	0.61	**3.68**	3.59	3.39	3.12
Dec	**0.12**	0.12	NA	NA	**0.22**	0.20	0.12	0.03	**0.91**	0.89	0.81	0.72	**1.25**	1.23	1.14	1.04	**1.83**	1.80	1.69	1.56	**0.74**	0.67	0.65	0.58	**2.83**	2.76	2.61	2.40

Notes:
[a] USGS 15300250.
NA = not applicable: UT100E would be blocked by the waste rock pile in the Pebble 2.0 scenario (Figure 7-15) and by the mine pit in the Pebble 6.5 scenario (Figure 7-16).

Table 7-16. Pre-mining watershed areas, mine footprint areas, and flows in the mine scenario watersheds for the Pebble 0.25 scenario.

					Returned Flow in Each Pathway (%)					85.9	8.8	75.3	0.0				
	Pre-Mining				Volume from Water Balance (m³/yr)					10,909,000	1,113,000	676,000	0	Operational Flows			
Stream and Gage	Watershed Area (km²)	Mean Annual Unit Runoff (m³/s*km²)	Runoff per Unit Area (m³/yr*m²)	Mean Annual Runoff (m³/yr)	Total Mine Footprint Drainage Area (km²)	Mine Footprint other than TSF, NAG, or PAG (km²)	TSF 1 Footprint (km²)	NAG Waste Rock Footprint (km²)	PAG Waste Rock Footprint (km²)	Flow Volume Returned through WWTP (m³/yr)[a]	Flow Volume Returned as TSF Leakage (m³/yr)	Flow Volume Returned as NAG Waste Rock Leachate (m³/yr)	Flow Volume Returned as PAG Waste Rock Leachate (m³/yr)	Flow Volume Remaining Due to Mine Footprint with No Returns (m³/yr)	Captured Flow Volume Returned from Footprint (m³/yr)	Total Flow Volume in Stream During Operations (m³/yr)	Change in Average Annual Runoff (%)
South Fork Koktuli River																	
SK100G	14	0.026	0.82	11,618,000	8.0	7.5	-	0.5	-	-	-	207,000	-	5,080,000	207,000	5,287,000	−54
SK100F	31	0.026	0.83	25,842,000	8.8	<0.1	-	0.8	-	-	-	350,000	-	18,499,000	556,000	19,055,000	−26
SK100CP2[b] (total runoff)	54	0.026	0.83	44,681,000	8.8	-	-	-	-	-	-	-	-	-	-	-	-
SK100CP2[b] (losses to UTC)[c]	54	0.009	0.28	−14,894,000	8.8	-	-	-	-	-	-	-	-	−12,446,000	−185,000	-	-
SK100CP2[b] (net streamflow at gage)	54	0.018	0.55	29,788,000	8.8	-	-	-	-	-	-	-	-	24,892,000	371,000	25,263,000	−15
SK124A	22	0.024	0.76	16,811,000	0.0	-	-	-	-	5,454,000	-	-	-	16,811,000	5,454,000	22,265,000	32
SK124CP[b]	24	0.024	0.76	17,937,000	0.0	-	-	-	-	-	-	-	-	17,937,000	5,454,000	23,391,000	30
SK100C	99	0.013	0.42	41,858,000	8.8	-	-	-	-	-	-	-	-	38,117,000	5,825,000	43,942,000	5
SK100CP1[b]	99	0.013	0.42	42,029,000	8.8	-	-	-	-	-	-	-	-	38,288,000	5,825,000	44,113,000	5
SK119A	28	0.036	1.12	31,268,000	0.0	-	-	-	-	-	-	-	-	31,268,000	-	31,268,000	0
SK119CP[b]	30	0.036	1.12	33,124,000	0.0	-	-	-	-	-	-	-	-	33,124,000	-	33,124,000	0
SK100B1	141	0.026	0.82	115,110,000	8.8	-	-	-	-	-	-	-	-	107,911,000	5,825,000	113,737,000	−1
SK100B[d]	179	0.029	0.91	162,122,000	8.8	-	-	-	-	-	-	-	-	154,112,000	5,825,000	159,937,000	−1
North Fork Koktuli River																	
NK119A	20	0.034	1.08	21,515,000	6.8	<0.1	6.5	0.3	-	-	1,113,000	120,000	-	14,146,000	1,233,000	15,378,000	−29
NK119CP2[b]	22	0.034	1.08	24,155,000	6.9	0.1	-	-	-	-	-	-	-	16,691,000	1,233,000	17,923,000	−26
NK119B	11	0.012	0.38	4,081,000	0.3	0.3	-	-	-	-	-	-	-	3,975,000	-	3,975,000	−3
NK119CP1[b]	33	0.027	0.85	28,431,000	7.2	<0.1	-	-	-	-	-	-	-	22,279,000	1,233,000	23,512,000	−17
NK100C	65	0.020	0.64	41,853,000	0.0	<0.1	-	-	-	5,454,000	-	-	-	41,828,000	5,454,000	47,282,000	13
NK100B	99	0.024	0.77	76,408,000	7.2	<0.1	-	-	-	-	-	-	-	70,826,000	6,687,000	77,513,000	1
NK100A1	222	0.026	0.82	182,297,000	7.3	<0.1	-	-	-	-	-	-	-	176,335,000	6,687,000	183,022,000	<1
NK100A[e]	279	0.025	0.79	220,715,000	7.3	-	-	-	-	-	-	-	-	214,981,000	6,687,000	221,668,000	<1
Upper Talarik Creek																	
UT100E	10	0.027	0.84	7,996,000	0.6	0.6	-	-	-	-	-	-	-	7,474,000	-	7,474,000	−7
UT100D	31	0.025	0.78	24,201,000	2.8	2.2	-	-	-	-	-	-	-	22,008,000	-	22,008,000	−9
UT100C2	133	0.022	0.70	92,734,000	2.8	0.0	-	-	-	-	-	-	-	90,768,000	-	90,768,000	−2
UT100C1	159	0.022	0.68	107,971,000	2.8	-	-	-	-	-	-	-	-	106,050,000	-	106,050,000	−2
UT100C	185	0.024	0.76	141,213,000	2.8	-	-	-	-	-	-	-	-	139,053,000	-	139,053,000	−2
UT119A (local runoff)	10	0.033	1.04	10,655,000	0.0	-	-	-	-	-	-	-	-	10,655,000	-	-	-
UT119A (gains from SFK)[c]	10	0.046	1.45	14,894,000	0.0	-	-	-	-	-	-	-	-	12,446,000	185,000	-	-
UT119A (net streamflow at gage)	10	0.079	2.48	25,549,000	0.0	-	-	-	-	-	-	-	-	23,101,000	185,000	23,286,000	−9
UT100B[f]	222	0.028	0.88	196,182,000	2.8	-	-	-	-	-	-	-	-	191,238,000	185,000	191,423,000	−2

Notes:
Dashes (-) indicate that values are either not applicable or are equal to zero.
[a] WWTP discharges 50% of flow to South Fork Koktuli River, 50% of streamflow to North Fork Koktuli River (no WWTP flows are directed to Upper Talarik Creek).
[b] Confluence point where virtual gage was created because physical gage does not exist.
[c] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location. Interbasin transfer flows are represented by negative flow values from SK100CP2 (losses to UTC) and equivalent positive flow values for UT119A (gains from SFK).
[d] USGS 15302200.
[e] USGS 15302250.
[f] USGS 15300250.
TSF = tailings storage facility; PAG = potentially acid-generating; NAG = non-acid-generating; WWTP = wastewater treatment plant; UTC = Upper Talarik Creek; SFK = South Fork Koktuli.

Table 7-17. Pre-mining watershed areas, mine footprint areas, and flows in the mine scenario watersheds for the Pebble 2.0 scenario.

					Returned Flow in Each Pathway (%)					66.7	15.2	16.7	1.4				
	Pre-Mining				Volume from Water Balance (m^3/yr)					10,304,000	2,351,000	2,576,000	216,000	Operational Flows			
Stream and Gage	Watershed Area (km^2)	Mean Annual Unit Runoff (m^3/s*km^2)	Runoff per Unit Area (m^3/yr*m^2)	Mean Annual Runoff (m^3/yr)	Total Mine Footprint Drainage Area (km^2)	Mine Footprint other than TSF, NAG, or PAG (km^2)	TSF 1 Footprint (km^2)	NAG Waste Rock Footprint (km^2)	PAG Waste Rock Footprint (km^2)	Flow Volume Returned through WWTP (m^3/yr)[a]	Flow Volume Returned as TSF Leakage (m^3/yr)	Flow Volume Returned as NAG Waste Rock Leachate (m^3/yr)	Flow Volume Returned as PAG Waste Rock Leachate (m^3/yr)	Flow Volume Remaining Due to Mine Footprint with No Returns (m^3/yr)	Captured Flow Volume Returned from Footprint (m^3/yr)	Total Flow Volume in Stream During Operations (m^3/yr)	Change in Average Annual Runoff (%)
South Fork Koktuli River																	
SK100G	14	0.026	0.82	11,618,000	11.2	9.2	-	1.5	0.5	-	-	633,000	213,000	2,420,000	846,000	3,266,000	−72
SK100F	31	0.026	0.83	25,842,000	12.6	0.2	-	1.2	<0.01	-	-	507,000	3,000	15,389,000	1,356,000	16,745,000	−35
SK100CP2[b] (total runoff)	54	0.026	0.83	44,681,000	12.6	-	-	-	-	-	-	-	-	-	-	-	-
SK100CP2[b] (losses to UTC)[c]	54	0.009	0.28	−14,894,000	12.6	-	-	-	-	-	-	-	-	−11,409,000	−452,000	-	-
SK100CP2[b] (net streamflow at gage)	54	0.018	0.55	29,788,000	12.6	-	-	-	-	-	-	-	-	22,819,000	904,000	23,723,000	−20
SK124A	22	0.024	0.76	16,811,000	0.1	0.1	<0.1	0.1	-	5,152,000	2,000	22,000	-	16,702,000	5,175,000	21,878,000	30
SK124CP[b]	24	0.024	0.76	17,937,000	0.1	-				-	-	-	-	17,829,000	5,175,000	23,004,000	28
SK100C	99	0.013	0.42	41,858,000	12.7	<0.1	-	-	-	-	-	-	-	36,472,000	6,079,000	42,552,000	2
SK100CP1[b]	99	0.013	0.42	42,029,000	12.7	-				-	-	-	-	36,643,000	6,079,000	42,722,000	2
SK119A	28	0.036	1.12	31,268,000	0.6	0.1	0.1	0.4	-	-	21,000	151,000	-	30,602,000	172,000	30,774,000	−2
SK119CP[b]	30	0.036	1.12	33,124,000	0.6	-				-	-	-	-	32,458,000	172,000	32,630,000	−1
SK100B1	141	0.026	0.82	115,110,000	13.3	-	-	-	-	-	-	-	-	104,262,000	6,251,000	110,513,000	−4
SK100B[d]	179	0.029	0.91	162,122,000	13.3	-	-	-	-	-	-	-	-	150,051,000	6,251,000	156,302,000	−4
North Fork Koktuli River																	
NK119A	20	0.034	1.08	21,515,000	14.9	0.1	13.9	0.9	-	-	2,305,000	402,000	-	5,405,000	2,707,000	8,111,000	−62
NK119CP2[b]	22	0.034	1.08	24,155,000	15.3	0.4	<0.1	<0.1	-	-	1,000	13,000	-	7,627,000	2,720,000	10,347,000	−57
NK119B	11	0.012	0.38	4,081,000	1.2	1.1	-	<0.1	-	-	-	3,000	-	3,638,000	3,000	3,641,000	−11
NK119CP1[b]	33	0.027	0.85	28,431,000	16.5	-	-	-	-	-	-	-	-	14,346,000	2,723,000	17,069,000	−40
NK100C	65	0.020	0.64	41,853,000	0.2	0.2	-	-	-	5,152,000	-	-	-	41,753,000	5,152,000	46,905,000	12
NK100B	99	0.024	0.77	76,408,000	16.6	<0.1	-	-	-	-	-	-	-	63,577,000	7,875,000	71,452,000	−6
NK100A1	222	0.026	0.82	182,297,000	17.3	0.1	0.1	0.5	-	-	23,000	204,000	-	168,068,000	8,102,000	176,169,000	−3
NK100A[e]	279	0.025	0.79	220,715,000	17.3	-	-	-	-	-	-	-	-	207,031,000	8,102,000	215,132,000	−3
Upper Talarik Creek																	
UT100E	10	0.027	0.84	7,996,000	3.2	3.2	-	-	-	-	-	-	-	5,290,000	-	5,290,000	−34
UT100D	31	0.025	0.78	24,201,000	14.5	9.8	-	1.5	-	-	-	642,000	-	12,839,000	642,000	13,481,000	−44
UT100C2	133	0.022	0.70	92,734,000	14.6	0.1	-	-	-	-	-	-	-	82,573,000	642,000	83,215,000	−10
UT100C1	159	0.022	0.68	107,971,000	14.6	-	-	-	-	-	-	-	-	98,042,000	642,000	98,684,000	−9
UT100C	185	0.024	0.76	141,213,000	14.6	-	-	-	-	-	-	-	-	130,049,000	642,000	130,691,000	−7
UT119A (local runoff)	10	0.033	1.04	10,655,000	-	-	-	-	-	-	-	-	-	10,655,000	-	-	-
UT119A (gains from SFK)[c]	10	0.046	1.45	14,894,000	-	-	-	-	-	-	-	-	-	11,409,000	452,000	-	-
UT119A (net streamflow at gage)	10	0.079	2.48	25,549,000	-	-	-	-	-	-	-	-	-	22,064,000	452,000	22,516,000	−12
UT100B[f]	222	0.028	0.88	196,182,000	14.6	-	-	-	-	-	-	-	-	179,795,000	1,094,000	180,889,000	−8

Notes:
Dashes (-) indicate that values are either not applicable or are equal to zero. UT100E is blocked by the mine footprint in this scenario.
[a] WWTP discharges 50% of flow to South Fork Koktuli River, 50% of flow to North Fork Koktuli River (no WWTP flows are directed to Upper Talarik Creek).
[b] Confluence point where virtual gage was created because physical gage does not exist.
[c] 1/3 of total return flow from is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location. Interbasin transfer flows are represented by negative flow values from SK100CP2 (losses to UTC) and equivalent positive flow values for UT119A (gains from SFK).
[d] USGS 15302200.
[e] USGS 15302250.
[f] USGS 15300250.
TSF = tailings storage facility; PAG = potentially acid-generating; NAG = non-acid-generating; WWTP = wastewater treatment plant; UTC = Upper Talarik Creek; SFK = South Fork Koktuli.

Table 7-18. Pre-mining watershed areas, mine footprint areas, and flows in the mine scenario watersheds for the Pebble 6.5 scenario.

	Pre-Mining				Returned Flow in Each Pathway (%)							79.4	11.2	7.7	1.6	Operational Flows			
					Volume from Water Balance (m³/yr)							50,988,000	7,203,000	4,971,000	1,032,000				
Stream and Gage	Watershed Area (km²)	Mean Annual Unit Runoff (m³/s*km²)	Runoff per Unit Area (m³/yr*m²)	Mean Annual Runoff (m³/yr)	Total Mine Footprint Drainage Area (km²)	Mine Footprint other than TSF, NAG, or PAG (km²)	TSF 1 Footprint (km²)	TSF 2 Footprint (km²)	TSF 3 Footprint (km²)	NAG Waste Rock Footprint (km²)	PAG Waste Rock Footprint (km²)	Flow Volume Returned through WWTP (m³/yr)[a]	Flow Volume Returned as TSF Leakage (m³/yr)	Flow Volume Returned as NAG Waste Rock Leachate (m³/yr)	Flow Volume Returned as PAG Waste Rock Leachate (m³/yr)	Flow Volume Remaining Due to Mine Footprint with No Returns (m³/yr)	Captured Flow Volume Returned from Footprint (m³/yr)	Total Flow Volume in Stream During Operations (m³/yr)	Change in Average Annual Runoff (%)
South Fork Koktuli River																			
SK100G	14	0.026	0.82	11,618,000	14.0	14.0	-	-	-	-	-	-	-	-	-	95,000	-	-	-
SK100F	31	0.026	0.83	25,842,000	22.1	2.5	-	-	0.1	3.0	2.4	-	20,000	1,278,000	1,032,000	7,480,000	2,330,000	9,810,000	−62
SK100CP2[b] (total runoff)	54	0.026	0.83	44,681,000	22.1	-	-	-		<0.1	-	-	-	5,000	-	26,309,000	2,335,000	-	-
SK100CP2[b] (losses to UTC)[c]	54	0.009	0.28	−14,894,000	22.1	-	-	-	-	-	-	-	-	-	-	−8,770,000	−778,000	-	-
SK100CP2[b] (net flow at gage)	54	0.018	0.55	29,788,000	22.1	-	-	-	-	-	-	-	-	-	-	17,540,000	1,557,000	19,096,000	−36
SK124A	22	0.024	0.76	16,811,000	11.4	0.1	<0.1	1.8	7.8	1.7	-	25,494,000	1,626,000	713,000	-	8,216,000	27,833,000	36,049,000	114
SK124CP[b]	24	0.024	0.76	17,937,000	11.4	-	-	-	-	-	-	-	-	-	-	9,342,000	27,833,000	37,175,000	107
SK100C	99	0.013	0.42	41,858,000	33.6	-	-	<0.1	-	0.1	-	-	-	54,000	-	27,627,000	29,443,000	57,070,000	36
SK100CP1[b]	99	0.013	0.42	42,029,000	33.6	-	-	-	-	-	-	-	-	-	-	27,798,000	29,443,000	57,241,000	36
SK119A	28	0.036	1.12	31,268,000	18.0	0.1	0.1	17.2	-	0.6		-	2,930,000	242,000	-	11,091,000	3,171,000	-	-
SK119CP[b]	30	0.036	1.12	33,124,000	19.2	-	-	0.3	-	1.0		-	50,000	413,000	-	11,537,000	3,635,000	15,172,000	−54
SK100B1	141	0.026	0.82	115,110,000	54.3	<0.1	-	0.9	-	0.6	-	-	145,000	260,000	-	70,839,000	33,482,000	104,322,000	−9
SK100B[d]	179	0.029	0.91	162,122,000	54.3	-	-	-	-	-	-	-	-	-	-	112,863,000	33,482,000	146,346,000	−10
North Fork Koktuli River																			
NK119A	20	0.034	1.08	21,515,000	14.9	0.1	13.9	-	-	0.9	-	-	2,360,000	402,000	-	5,405,000	2,762,000	8,167,000	−62
NK119CP2[b]	22	0.034	1.08	24,155,000	15.3	0.4	<0.1	-	-	<0.1	-	-	1,000	13,000	-	7,627,000	2,775,000	10,402,000	−57
NK119B	11	0.012	0.38	4,081,000	3.3	2.7	-	-	0.3	0.3	-	-	48,000	144,000	-	2,812,000	192,000	3,004,000	−26
NK119CP1[b]	33	0.027	0.85	28,431,000	18.6	-	-	-	-	-	-	-	-	-	-	12,506,000	2,967,000	15,473,000	−46
NK100C	65	0.020	0.64	41,853,000	0.5	0.5	-	-	-	-	-	25,494,000	-	-	-	41,559,000	25,494,000	67,053,000	60
NK100B	99	0.024	0.77	76,408,000	19.1	-	-	-	-	-	-	-	-	-	-	61,683,000	28,461,000	90,144,000	18
NK100A1	222	0.026	0.82	182,297,000	19.8	0.1	0.1	-	-	0.5	-	-	23,000	204,000	-	166,049,000	28,688,000	194,738,000	7
NK100A[e]	279	0.025	0.79	220,715,000	19.8	-	-	-	-	-	-	-	-	-	-	205,090,000	28,688,000	233,778,000	6
Upper Talarik Creek																			
UT100E	10	0.027	0.84	7,996,000	7.4	6.6	-	-	-	0.8	-	-	-	346,000	-	1,779,000	346,000	2,125,000	−73
UT100D	31	0.025	0.78	24,201,000	27.8	18.7	-	-	-	1.7	-	-	-	739,000	-	2,398,000	1,085,000	3,482,000	−86
UT100C2	133	0.022	0.70	92,734,000	29.0	0.8	-	-	-	0.4	-	-	-	160,000	-	72,570,000	1,245,000	73,815,000	−20
UT100C1	159	0.022	0.68	107,971,000	29.0	-	-	-	-	-	-	-	-	-	-	88,266,000	1,245,000	89,511,000	−17
UT100C	185	0.024	0.76	141,213,000	29.0	-	-	-	-	-	-	-	-	-	-	119,058,000	1,245,000	120,303,000	−15
UT119A (local runoff)[c]	10	0.033	1.04	10,655,000	-	-	-	-	-	-	-	-	-	-	-	10,655,000	-	-	-
UT119A (gains from SFK)	10	0.046	1.45	14,894,000	-	-	-	-	-	-	-	-	-	-	-	8,770,000	778,000	-	-
UT119A (net flow at gage)	10	0.079	2.48	25,549,000	-	-	-	-	-	-	-	-	-	-	-	19,425,000	778,000	20,203,000	−21
UT100B[f]	222	0.028	0.88	196,182,000	29.1	-	-	-	-	-	-	-	-	-	-	164,453,000	2,023,000	166,476,000	−15

Notes:

Dashes (-) indicate that values are either not applicable or are equal to zero. UT100E is blocked and SK100G and SK119A are eliminated by the mine footprint in this scenario.

[a] WWTP discharges 50% of flow to South Fork Koktuli River, 50% of flow to North Fork Koktuli River (no WWTP flows are directed to Upper Talarik Creek).

[b] Confluence point where virtual gage was created because physical gage does not exist.

[c] 1/3 of total return flow from is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location. Interbasin transfer flows are represented by negative flow values from SK100CP2 (losses to UTC) and equivalent positive flow values for UT119A (gains from SFK).

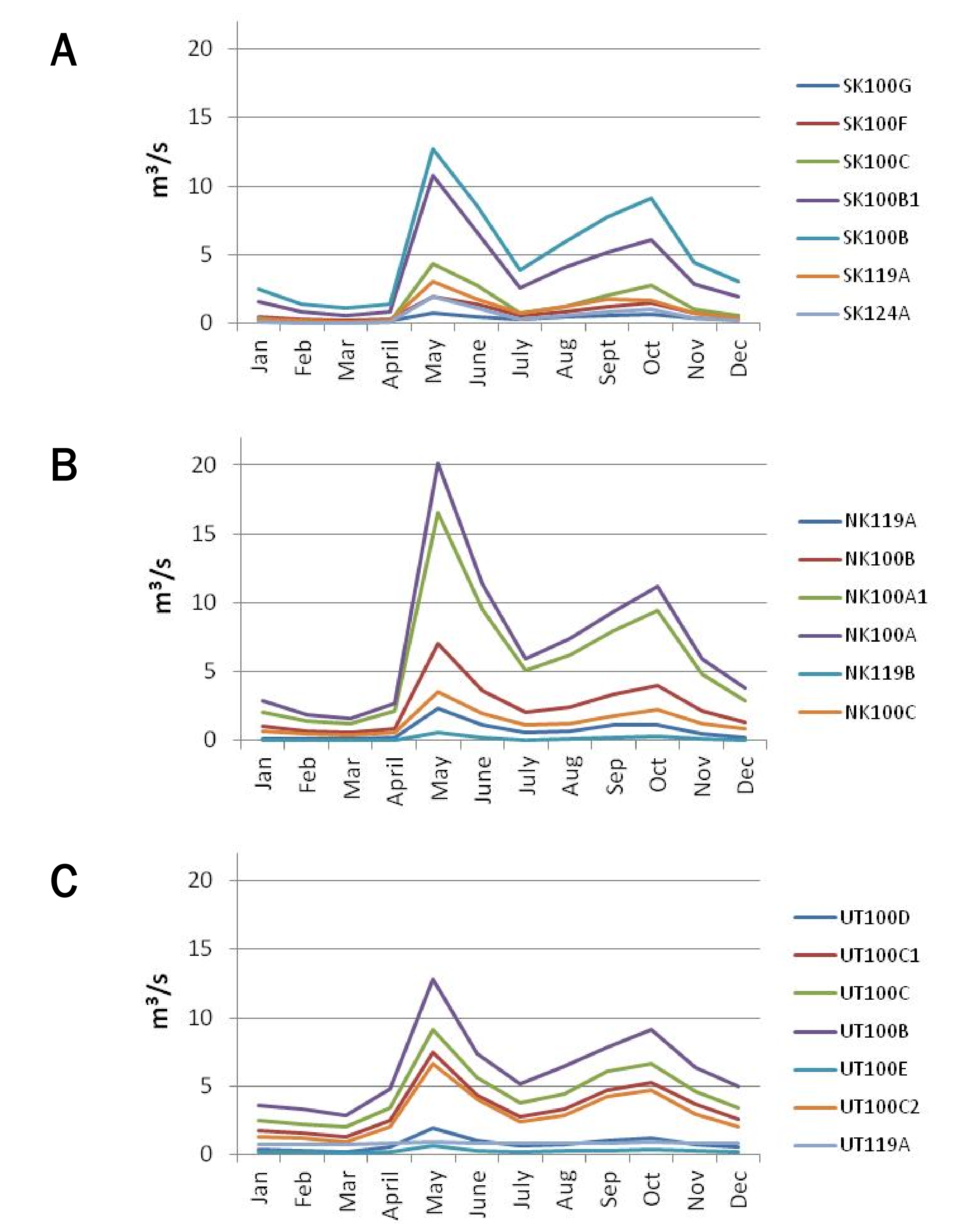
[d] USGS 15302200.

[e] USGS 15302250.

[f] USGS 15300250.

TSF = tailings storage facility; PAG = potentially acid-generating; NAG = non-acid-generating; WWTP = wastewater treatment plant; UTC = Upper Talarik Creek; SFK = South Fork Koktuli.

Figure 7-17. Monthly mean streamflows for stream gages in the (A) South Fork Koktuli River, (B) North Fork Koktuli River, and (C) Upper Talarik Creek watersheds, based on water years 2004 through 2010.



For the three mine scenarios, it was assumed that some water captured from each mine footprint would be treated and reintroduced to downstream areas. For the Pebble 0.25, 2.0, and 6.5 scenarios, we estimated that 76.3, 37.5, and 70.5% of the total water captured, respectively, would be reintroduced (Table 6-3). Figures 6-8 through 6-10 illustrate the various flowpaths expected in the three mine scenarios. For each of the watersheds, reintroduced flow was returned to the appropriate gage based on the expected flowpath as defined by the mine scenarios. Some upper tributaries would experience reduced streamflows from watershed area losses, whereas others would experience increased annual runoff from mining operation discharges.

Although some surface runoff might be collected, most of the precipitation in the drawdown zone would flow as groundwater into the mine pit and be removed by pumping to the WWTP. Much of the flow from components outside the drawdown zone, such as leachate from TSFs and waste rock piles, would be captured and directed to the WWTP, but some would escape the collection systems and flow back to the downstream receiving waters (Tables 7-16 through 7-18, Figures 6-8 through 6-10). It is important to note that the WWTP is assumed to discharge to the South and North Fork Koktuli River watersheds via the WWTP outfalls (after Ghaffari et al. 2011), so no treated flow would be reintroduced to streams in the Upper Talarik Creek watershed. An area of interbasin groundwater transfer has been observed between the South Fork Koktuli River and Upper Talarik Creek (PLP 2011: Chapter 7). This transfer was accounted for by allowing one-third of the flow at gage SK100F to transfer to gage UT119A (Tables 7-16 through 7-18, Figures 6-8 through 6-10). The spatial extent of these projected changes in streamflow and implications for fish and aquatic habitat are discussed in Section 7.3.2.

7.3.1.1 Pebble 0.25 Scenario

Water balance estimates for the Pebble 0.25 scenario considered an operational facility that intercepts precipitation from a footprint encompassing portions of the mine scenario watersheds (Table 7-16, Figure 7-14). Based on these conditions, we estimate that in each watershed the uppermost gages closest to the mine footprint would experience the most significant streamflow reductions. Overall, it is projected that 76.3% of captured watershed flows would be returned (Table 6-3), but the location of return would vary depending on mine needs for process water and the location of mine facilities and water treatment (Table 7-16). In the Upper Talarik Creek watershed in the Pebble 0.25 scenario, streamflow would be reduced by 7% at gage UT100E and 9% at gage UT100D due to capture in the mine footprint. The most significant streamflow reductions in the South Fork Koktuli River would be expected at gages SK100G (54%) and SK100F (26%) (Table 7-16). In the North Fork Koktuli River, the greatest changes would be expected at gage NK119A (29% reduction) (Table 7-16), as much of the watershed would be occupied by TSF 1 (Figure 7-14).

Streamflow reductions due to water capture in the mine footprint would be partially offset by water return via the WWTP, leakage through the TSF, and leaching through the waste rock piles. Water balance calculations for these water budget components are described in Chapter 6. Excess captured water would be treated at the WWTP and discharged upstream of gage SK124A in the South Fork Koktuli River and gage NK100C in the North Fork Koktuli River (Figure 7-14). It is assumed that the

WWTP would discharge equally to both outfalls, creating a 50/50 volume split for treated flows on an annual basis, but that on-site storage would allow management of environmental streamflows to match seasonal hydrographs to the degree possible. Flows from the WWTP outfalls would be projected to increase streamflows by 32% at gage SK124A, in a tributary to the South Fork Koktuli River. In the North Fork Koktuli River watershed, streamflows would be projected to increase by 13% at gage NK100C, downstream of the WWTP outfall. In the mainstem South and North Fork Koktuli Rivers downstream of these points, WWTP outfall flows (approximately 5.4 million m^3/year from each outfall) (Table 7-16), leakage from the TSF, and waste rock leaching would partially offset streamflow reductions expected from water capture within the mine footprint. Projected streamflow changes for gages farther downstream of the WWTP outfalls are within 5% of pre-project streamflows (Tables 7-16 and 7-19, Figure 7-14).

Because of the natural interbasin streamflow transfer from the South Fork Koktuli River watershed to the Upper Talarik Creek watershed (described above), decreased streamflows in the South Fork Koktuli River resulting from capture by the mine footprint would translate to decreased rates of interbasin transfer. As a result, there would be a projected 9% decrease in streamflow to the tributary of Upper Talarik Creek where the interbasin transfer flows emerge (gage UT119A) (Tables 7-16 and 7-19, Figure 7-14).

7.3.1.2 Pebble 2.0 Scenario

In the Pebble 2.0 scenario, area lost to the mine footprint would increase from the addition of a second or expanded waste rock pile that would occupy much of the Upper Talarik Creek valley between gages UT100E and UT100D (Figure 7-15). An expanded groundwater drawdown zone would develop around the larger mine pit and further reduce water flowing to surrounding streams, and TSF 1 would expand in size (Figure 7-15). Approximately 37.5% of the total water captured would be returned to the three watersheds (Table 6-3). However, as in the Pebble 0.25 scenario described above, flow returns in the upper watersheds via the WWTP outfalls would not necessarily be returned to their source stream reaches.

After accounting for water captured in the footprint, leakage, leachate, and reintroduced water, streamflow reductions in Upper Talarik Creek would be most severe for gage UT100D (44% reduction) (Tables 7-17 and 7-19). In the South Fork Koktuli River, gages SK100G, SK100F, and confluence point SK100CP2 would experience reductions of 72, 35, and 20%, respectively. In the North Fork Koktuli River, the most severe effects would be seen in the watershed occupied by TSF 1, with gages on this tributary predicted to experience streamflow reductions ranging from 40 to 62% (Tables 7-17 and 7-19, Figure 7-15). Contributions of the WWTP flow to the South Fork Koktuli River watershed would cause an increase in streamflow at gage SK124A (30%) and the associated confluence point SK124CP (28%). WWTP contributions to the North Fork Koktuli River watershed would cause a 12% streamflow increase at gage NK100C. At the lowermost gages in each watershed, projected reductions in streamflow would be 8% (Upper Talarik Creek), 4% (South Fork Koktuli River), and 3% (North Fork Koktuli River) (Tables 7-17 and 7-19).

Table 7-19. Estimated changes in streamflow (%) and subsequent stream lengths affected (km) in the mine scenario watersheds in the Pebble 0.25, Pebble 2.0, and Pebble 6.5 scenarios. Italics indicates changes greater than 10% (minor effects on salmon populations expected); bold indicates changes greater than 20% (moderate to major effects on salmon populations expected).

Stream and Gage	Pebble 0.25		Pebble 2.0		Pebble 6.5	
	Estimated Change in Streamflow	Stream Length Affected	Estimated Change in Streamflow	Stream Length Affected	Estimated Change in Streamflow	Stream Length Affected
South Fork Koktuli River—Mainstem						
SK100G	**−54**	1.9	**−72**	0.5	NA	NA
SK100F	**−26**	3.3	**−35**	3.3	**−62**	0.8
SK100CP2	*−15*	10.7	**−20**	10.7	**−36**	10.7
SK100C	5	6.3	2	6.3	**36**	6.3
SK100CP1	5	1.2	2	1.2	**36**	1.2
SK100B1	−1	4.3	−4	4.3	−9	4.3
SK100B[a]	−1	4.5	−4	4.5	*−10*	4.5
South Fork Koktuli River—Tributaries						
SK119A	0	7.0	−2	6.7	NA	NA
SK119CP	0	1.6	−1	1.6	**−54**	0.7
SK124A	**32**	5.0	**30**	5.0	**114**	4.2
SK124CP	**30**	2.6	**28**	2.6	**107**	2.6
North Fork Koktuli River—Mainstem						
NK100C[b]	*13*	4.5	*12*	4.5	**60**	4.5
NK100B	1	0.8	−6	0.8	*18*	0.8
NK100A1	0	20.4	−3	20.4	7	20.4
NK100A[c]	0	8.4	−3	8.4	6	8.4
North Fork Koktuli River—Tributaries						
NK119A	**−29**	0.8	**−62**	0.7	**−62**	0.7
NK119CP2	**−26**	1.3	**−57**	1.3	**−57**	1.3
NK119B	−3	6.8	*−11*	6.8	**−26**	6.5
NK119CP1	*−17*	0.4	**−40**	0.4	**−46**	0.4
Upper Talarik Creek—Mainstem						
UT100E	−7	2.3	NA	NA	NA	NA
UT100D	−9	7.1	**−44**	2.1	**−86**	0.3
UT100C2	−2	6.1	−10	6.1	**−20**	6.1
UT100C1	−2	6.9	−9	6.9	*−17*	6.9
UT100C	−2	7.5	−7	7.5	*−15*	7.5
UT100B[d]	−2	4.3	−8	4.3	*−15*	4.3
Upper Talarik Creek Tributary—Tributaries						
UT119A	−9	6.5	*−12*	6.5	**−21**	6.5

Notes:

Stream lengths are typically calculated from the gage upstream to the next gage or the mine footprint (but see below); stream lengths affected do not include portions of stream lost in the pit drawdown zone.

For gages UT100D, SK100G, SK100F, SK119A, SK124A, and NK119A, stream lengths include mainstem length upstream to edge of mine footprint only, and do not include upstream lengths, including tributaries, that would be blocked or eliminated by the mine footprint.

[a] USGS 15302200.

[b] Upstream to wastewater treatment plant outfall point.

[c] USGS 15302250.

[d] USGS 15300250.

NA = not applicable; the stream at the gage would be eliminated or blocked by the mine footprint

7.3.1.3 Pebble 6.5 Scenario

In the Pebble 6.5 scenario, area lost to the mine footprint would increase with inclusion of a larger pit and its associated drawdown zone, a substantially larger waste rock pile, and the development of TSF 2 on a tributary of South Fork Koktuli River upstream of gage SK100B1 and TSF 3 on a tributary upstream of gage SK124A (Table 7-18, Figure 7-16). Gage SK100G would be eliminated under the Pebble 6.5 waste rock piles, gage UT100E would be isolated upstream of the mine footprint, and gage SK119A would be buried under the TSF 2 dam (Figure 7-16). Although the larger mine footprint would result in the capture of much greater quantities of water in the Pebble 6.5 scenario, annual water consumption would not be appreciably higher than in the Pebble 2.0 scenario. Thus, an estimated 70.5% of the captured water would be available for reintroduction to streams (Table 6-3). The net effects of lost effective watershed area and the reintroduction of treated water would result in streamflow reductions that would be most severe for gages UT100D (86% reduction), SK100F (62% reduction), and NFK119A (62% reduction) (Tables 7-18 and 7-19).

WWTP flows would be increased greatly over the Pebble 2.0 scenario and would create increased streamflow at SK124A (114%) and SK124CP (107%). This increase would continue to influence streamflows downstream to gage SK100C (36% increase), but the large reduction attributed to the TSF on the tributary measured by gage SK119A again creates streamflow deficits downstream at gages SK100B1 and SK100B relative to pre-mining conditions (9 and 10% reductions, respectively) (Tables 7-18 and 7-19, Figure 7-16). In the North Fork Koktuli River watershed, WWTP contributions would lead to streamflow increases of 60% at gage NK100C and increased streamflows at all downstream gages (Table 7-18). Upper Talarik Creek would experience streamflow reductions of 15% or more at all mainstem gages. Upper Talarik Creek tributary gage UT119A would experience a 21% decrease in streamflow due to reduced interbasin transfer resulting from streamflow losses in the South Fork Koktuli River watershed. At the lowermost gages in each watershed, projected streamflow changes would be a 15% reduction for Upper Talarik Creek, a 10% reduction for the South Fork Koktuli River, and a 6% increase for the North Fork Koktuli River (Tables 7-18 and 7-19).

7.3.1.4 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 WWTP would no longer be available for streamflow. This period is projected to last from about 20 years for the Pebble 0.25 scenario to over 200 years for the Pebble 6.5 scenario, after which the pit would approach equilibrium with surrounding groundwater. The pit water level could be controlled by pumping or gravity drainage to maintain a hydraulic gradient toward the pit for as long as water needed treatment. When treatment was no longer necessary and active control was abandoned, water from the filled mine pit would eventually discharge to down-gradient streams, ponds, and wetlands (Section 6.3) under steady-state flow conditions. Given uncertainties in the post-closure water balance, we have not attempted to estimate streamflows during that period.

7.3.1.5 Uncertainties and Assumptions

Our assessment of streamflow changes distributes losses according to the percentage of the area lost to the mine footprint in a given watershed. The analysis uses flow per unit area derived from stream gage data, and allocates water routing through the three mine scenarios based on decisions about mine processes that will consume and reintroduce water to the watersheds. We assume that water captured within the footprint and requiring treatment would be routed through the WWTP and discharged to the two locations specified by Ghaffari et al. (2011). We assume that reduced streamflows would follow the same spatial patterns of gaining or losing groundwater reaches as would initial (pre-mine) conditions. We acknowledge, however, that mine operations could alter the relative importance of groundwater flowpaths, and thus result in a different spatial distribution of streamflow changes than we have reported.

7.3.2 Exposure-Response: Streamflow

Water from streams originating upstream of the mine footprints (i.e., blocked streams) could be captured at the footprint for use or stored on site for eventual treatment and return to the stream downstream of the footprint, either directly or via the WWTP. Water from blocked streams could be returned to downstream stream segments via diversion channels or pipes. Habitat upstream of the footprint would no longer be accessible to fish downstream because of the inability of fish to move upstream through diversion channels or pipes.

7.3.2.1 Altered Streamflow Regimes

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 streamflow if withdrawals are intermittent (Curry et al. 1994, Cunjak 1996). Temporal variability in streamflows is a natural feature of stream ecosystems (Poff et al. 1997), although the degree of variability differs depending on hydrologic controls such as 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 on 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 based on the premise that natural streamflow variability, including the magnitude, frequency, timing, duration, rate of change, and predictability of streamflow events and the sequence of streamflow 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 natural streamflows, potentially affecting ecosystem function and structure. Guidelines for minimizing impacts of altered hydrologic regimes have been offered by several researchers (Poff et al. 1997 and 2009, Richter 2010). Determining the natural streamflow regime is a data-intensive process, but it is crucial to understanding how to manage streamflows within a system (Arthington et al. 2006).

Given the high likelihood of complex groundwater–surface water connectivity in the deposit area, predicting and regulating flows to maintain key ecosystem functions associated with groundwater–surface water exchange would be 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 these efforts will help identify relationships among surface-water flow, groundwater and surface-water temperatures, and instream fish habitat (Bartholow 2010). However, until linkages between biology, groundwater, surface water, and potential mining activities can be better evaluated, predicted, and understood, a protective approach is warranted to maintain surface-water and groundwater flows and natural streamflow regimes across the mine scenario watersheds.

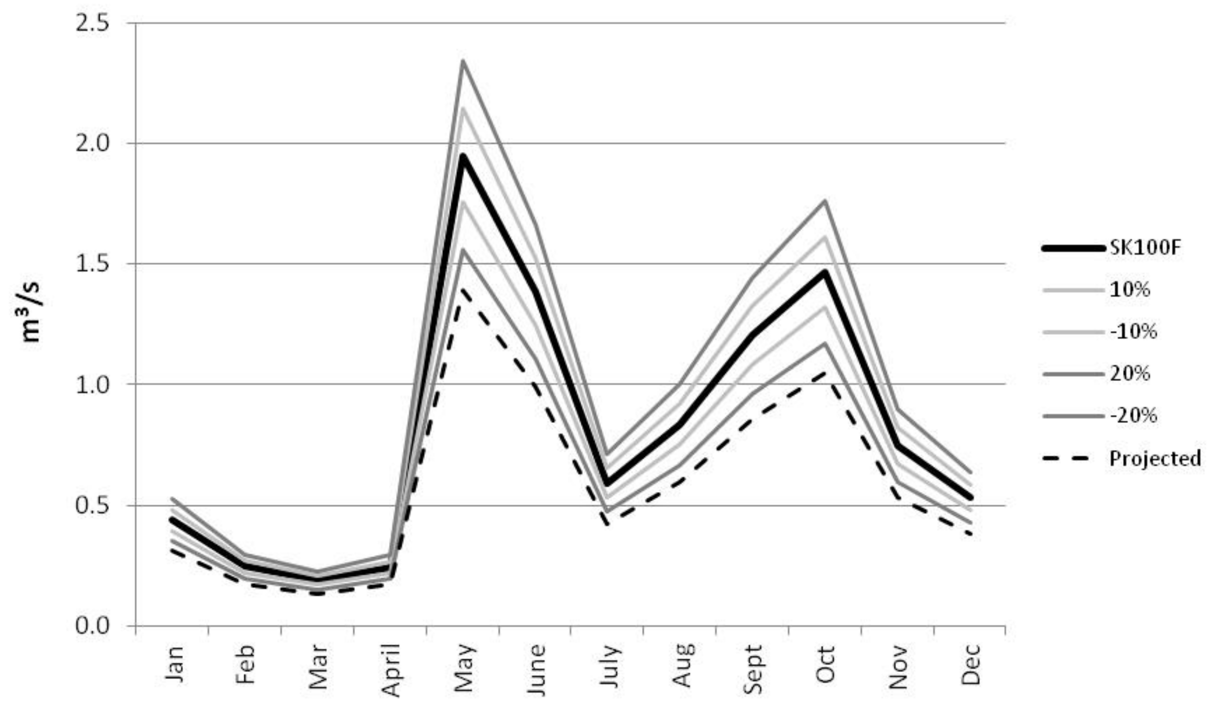
The sustainability boundary approach offers such a protective approach for balancing the maintenance of aquatic ecosystems with human demands (Richter et al. 2012). Under this approach, percentage-based deviations from natural conditions are used to set streamflow alteration limits. These percentages are based on the natural flow regime and do not focus on the more simplistic approach of setting a percentage based on a high-streamflow or low-streamflow event. Rather than a salmon-specific instream flow habitat model, this is a system-based approach targeting the entire aquatic ecosystem. Numerous case studies have tested this type of approach, and the percentage bounds of streamflow alteration around natural daily streamflow that caused measurable ecological harm were determined to be similar regardless of geographic location (Richter et al. 2012). Based on these studies, Richter et al. (2012) proposed that streamflow alteration be managed based on the following thresholds of daily percentage alteration.

- Streamflow alteration below 10% would cause minor impacts on the ecosystem with a relatively high level of ecosystem protection.
- Streamflow alteration of 11 to 20% would cause measurable changes in ecosystem structure and minor impacts on ecosystem function.
- Streamflow alteration greater than 20% would cause moderate to major changes in ecosystem structure and function. Increasing alteration beyond 20% would cause significant losses of ecosystem structure and function.

Losses of ecosystem structure and function could include reduced habitat availability for salmon and other stream fishes, particularly during low-streamflow periods (West et al. 1992, Cunjak 1996); reductions in macroinvertebrate production (Chadwick and Huryn 2007); and increased stream habitat fragmentation due to increased frequency and duration of stream drying. Increases in streamflow above background levels could result in altered sediment transport dynamics with increased scour and transport of gravels. Increased streamflows could also be associated with altered distributions of water velocities favorable for various fish life stages. These alterations, depending on magnitude, could significantly decrease salmon habitat quantity and quality in these watersheds (Figure 7-1).

We compared predicted streamflows for the Pebble 0.25, 2.0, and 6.5 scenarios (Tables 7-10 through 7-15) with the sustainability boundary limits of 10 and 20% streamflow alteration around mean monthly flow. As an example, mean monthly streamflows for the South Fork Koktuli River at gage SK100F during the pre-mining period, projected streamflows in the Pebble 0.25 scenario and the 10 and 20% sustainability boundaries for the baseline streamflow are shown in Figure 7-18.

Figure 7-18. Monthly mean pre-mining streamflow for South Fork Koktuli River gage SK100F (bold solid line), with 10 and 20% sustainability boundaries (gray lines) and projected monthly mean streamflows, in the Pebble 0.25 scenario (dashed line).



We used this sustainability boundary approach to evaluate risks associated with potential streamflow alterations throughout the mine scenario watersheds. To estimate the spatial extent of potential deleterious streamflow alterations, we calculated the length of stream network upstream of the uppermost stream gage to the edge of each mine footprint, and the length of each segment between stream gages in each mine scenario watershed. This stream length is in addition to the length of stream that would be eliminated or blocked by the mine footprint—that is, this and all subsequent references to stream lengths affected by flow modification reflect stream lengths downstream of the mine footprint for each scenario, and thus do not include stream lengths eliminated, blocked, or dewatered by each footprint (Section 7.2). Table 7-19 summarizes estimated percent changes in streamflow at each gage location, and the length of stream affected by each streamflow alteration in each mine scenario. Figures 7-14 through 7-16 illustrate the spatial extent and location of streamflow alterations in relation to gage sites. These estimates are for direct effects only. Stream sections throughout the stream network could be affected indirectly, via streamflow reductions 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 fish movement into those areas from downstream.

Pebble 0.25 Scenario

During operation of the Pebble 0.25 scenario, streamflow reductions exceeding 20% sustainability boundaries would occur in 7 km of streams beyond the mine footprint. Substantial reductions in fish habitat capacity and productivity could be expected for these streams. Streamflow increases greater than 20% are expected for 8 km of streams downstream of the WWTP outfall, and would likely lead to substantial changes in sediment dynamics and habitat suitability for fish. An additional 16 km of streams would experience streamflow alterations of 13 to 17%, with anticipated minor effects on ecosystem structure and function.

In the upper South Fork Koktuli River, gages SK100G and SK100F would experience 54 and 26% reductions in streamflow, respectively, affecting 5 km of streams (Table 7-19). The tributary to the South Fork Koktuli River receiving outfall from the WWTP would experience increased streamflows (28 to 30%), affecting 8 km of streams. In the North Fork Koktuli River, the tributary downstream of TSF 1 would experience 17 to 26% reductions in streamflow, affecting 2 km of streams (Table 7-19).

Several sections of the South Fork Koktuli River and tributaries below Frying Pan Lake are losing reaches (i.e., discharge decreases in a downstream direction), which under pre-mine conditions experience periods of zero minimum monthly discharge (e.g., gage SK100C and WWTP-receiving stream gage SK124A) (Table 7-13). We assumed that streamflow increases due to the WWTP would follow the natural hydrograph, reflecting the amount of precipitation and runoff that must be captured and treated. As a result, WWTP outfall flows would be lowest during periods when these streams typically go dry based on pre-mine baseline data, and would be highest during period of snowmelt runoff and fall storms.

Pebble 2.0 Scenario

In the Pebble 2.0 scenario, streamflow reductions exceeding 20% sustainability boundaries would occur in 19 km of streams downstream of the mine footprint. For these streams, substantial reductions in fish habitat capacity and productivity would be expected. Increases in streamflow of 28 to 30% would be expected for 8 km of streams downstream of the WWTP in the South Fork Koktuli River, and increases of 12% would be expected for 4 km of the WWTP-receiving tributary to the North Fork Koktuli River, leading to changes in sediment dynamics and habitat suitability for fish. An additional 6 km of streams in Upper Talarik Creek and 7 km of streams in the North Fork Koktuli River would experience flow reductions of 10 to 11%, with anticipated minor effects on ecosystem structure and function.

In the Pebble 2.0 scenario, the mine footprint captures 47% of the Upper Talarik Creek watershed above gage UT100D (Table 7-17). As a result, most of the total stream length in its upstream reaches, 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 7.2, Figure 7-15). Of this stream length, 2 km of mainstem downstream of the footprint would experience a significant loss of habitat and decline in habitat quality from the predicted 44% streamflow reduction at gage UT100D (Figure 7-15). Downstream of gage UT100D in Upper Talarik Creek, streamflow reductions would range from 8 to 10% (Table 7-19). Impacts on salmon habitat from streamflow reductions would be moderated by tributary and groundwater inputs 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 7.3.2.3). For instance, the groundwater-dominated Upper Talarik Creek tributary monitored at gage UT119A would experience a 12% streamflow reduction due to reduced flow in portions of the South Fork Koktuli River resulting from losses to the mine footprint. This was the only case of interbasin hydrologic connectivity explicitly modeled, but other undocumented connections are likely to occur.

In the South Fork Koktuli River, streamflow reductions would exceed the 20% sustainability threshold at gages SK100G, SK100F, and SK100CP2 (Table 7-19, Figure 7-15). In the South Fork Koktuli River mainstem and tributaries upstream of gage SK100G, the majority of stream length would be eliminated by the mine footprint (Figure 7-15), resulting in severe streamflow reductions at gages SK100G (72%) and SK100F (35%) (Table 7-19). Streamflows in the South Fork Koktuli River at gage SK100C would increase by 2% because of WWTP releases discharged at tributary gage SK124A, which would experience a 28% increase in streamflow at the confluence with the South Fork Koktuli River (Table 7-19, Figure 7-16).

In the North Fork Koktuli River, the majority of stream length above gage NK119A would be eliminated by construction of TSF 1 (Figure 7-15), resulting in substantial streamflow losses (62% reduction at gage NK119A) for approximately 2 km of streams between TSF 1 and the North Fork Koktuli River (Table 7-19, Figure 7-15). Approximately 7 km of streams in the tributary measured by gage NK119B would experience 11% reductions in streamflow. Increases in streamflow downstream of the WWTP discharge point would increase streamflows by 12% in 4 km of the North Fork Koktuli River upstream of gage NK100C (Table 7-19, Figure 7-15).

Pebble 6.5 Scenario

The Pebble 6.5 scenario would capture an even larger portion of the South and North Fork Koktuli Rivers and Upper Talarik Creek watersheds in its footprint. During operation of the Pebble 6.5 scenario, streamflow reductions exceeding 20% sustainability boundaries would occur in 34 km of streams. For these streams, reductions in fish habitat capacity and productivity could be expected. An additional 19 km of streams in Upper Talarik Creek would experience streamflow reductions exceeding 10%, with anticipated minor effects on ecosystem structure and function. Increases in streamflow exceeding 20% are expected for 14 km of streams downstream of the WWTP in the South Fork Koktuli River and for 4 km of the WWTP-receiving tributary to the North Fork Koktuli River, and would likely lead to substantial changes in sediment dynamics and habitat suitability for fish.

In the Upper Talarik Creek watershed, substantial streamflow reductions are projected at gages UT100D (86%) and UT100C2 (20%), affecting 6 km of streams. Streamflow alterations exceeding 10% would occur in an additional 19 km of streams at gages UT100C1, UT100C, and UT100B (Table 7-19). In the

South Fork Koktuli River, gages SK100G and SK119A would be buried under the expanded mine footprint. A 62% reduction in streamflow would be expected for 1 km of the upper South Fork Koktuli River downstream from the edge of the waste rock to gage SK100F (Table 7-19, Figure 7-16).

In the Pebble 6.5 scenario, the WWTP is estimated to discharge over 50 million m^3 of water per year (Table 7-18). This discharge would result in a 36% increase in streamflow for 8 km in the South Fork Koktuli River above gage SK100CP1, and a 107% increase in streamflow for 7 km of streams above gage SK124CP (Table 7-19, Figure 7-16). In the North Fork Koktuli River, WWTP outfalls would result in a 60% increase in streamflows for 4 km of streams above gage NK100C, and an 18% increase in streamflows for 1 km of streams upstream of gage NK100B.

Streamflow reductions and stream habitat losses of the magnitudes estimated in the Pebble 0.25, 2.0, and 6.5 scenarios 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 the mine scenario watersheds. Habitat quantity and quality would be significantly diminished by the loss of streamflow from the mine footprint, via multiple mechanisms such as direct reduction in habitat area and volume, the loss of channel to off-channel habitat connectivity, increased periods of zero streamflow, and reduced food production. Streamflow increases could alter channel morphologies, induce higher rates of sediment transport and erosion, and change the distribution of water velocities within habitats used by spawning and rearing salmon and other fishes. Although the loss of salmonid production has not been estimated, streamflow alterations greater than 20% would be expected to have substantial effects (Richter et al. 2012).

7.3.2.2 Connectivity, Timing, and Duration of Off-Channel Habitats

Losses of streamflow resulting from the mine footprints and potential water withdrawals described above would affect connectivity between the main channel and off-channel habitats important to juvenile salmonids. Losses of flood peaks could alter groundwater recharge rates and influence characteristics of floodplain percolation channels, seeps, or other expressions of the hyporheic zone (Hancock 2002). Rapid streamflow reductions that exceed recession rates typically experienced by fish in these systems could result in stranding or isolation of fish 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 sockeye salmon spawning habitats (Quinn 2005). Maintaining connectivity and the physical and chemical attributes of these habitats in conditions similar to baseline conditions would 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 footprint areas 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.

Flow regulation through the WWTP could be designed to somewhat approximate natural hydrologic regimes during periods when sufficient water and water storage capacity were available, which could provide appropriate timing and duration of connectivity with off-channel habitats. Channel cross-section data and gage data (PLP 2011) would provide useful insights into streamflow connectivity relationships and could help guide a streamflow management plan.

7.3.2.3 Changes in Groundwater Inputs and Importance to Fish

There is limited information describing potential surface water–groundwater interactions in the mine scenario watersheds, but groundwater is likely the dominant source of streamflow in these streams (Rains 2011) and can be very important locally. High baseflow levels in the monthly hydrographs of the mine scenario watersheds illustrate groundwater's important influence on these streams (Figure 3-10).

Aerial winter open-water surveys consistently suggest the presence of upwelling groundwater, which maintains ice-free conditions in portions of area streams and rivers. Highly permeable glacial outwash deposits create a complex mosaic within less permeable, clay to silt-dominated 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 not predicted in this assessment, but that could have significant impacts on fish. In our analyses of the water management regimes for the mine scenarios, we project increasing proportions of streamflow derived from water released from the WWTP as the mine develops. These 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 replace a portion of the groundwater that would otherwise be feeding stream systems, and could have substantially different chemical characteristics (Chapter 8).

Fish in the region are highly attuned to groundwater signals in the hydrologic and thermal regimes (Power et al. 1999). Spatial heterogeneity in streamflow 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 foodweb (Armstrong et al. 2010, Ruff et al. 2011).

Altered groundwater contributions to surface waters in the mine area could have profound effects on the thermal regimes and thermally cued life histories of aquatic biota. Curry et al. (1994) examined the influence of altered hydrologic regimes on groundwater–surface water interchange at brook trout spawning locations 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).

7.3.2.4 Stream Temperature

Projecting specific mine-associated changes to groundwater and surface water interactions and corresponding effects on surface water temperature 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 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. The duration of freezing and the extent and type of ice formation, including anchor ice, frazil, or surface ice (Slaughter 1990), can severely limit habitat availability during the winter and spring months.

Two aerial surveys of the mine scenario watersheds provide additional information on groundwater inputs to headwater streams and ice cover conditions in streams draining the mine footprints (PLP 2011, Woody and Higman 2011). PLP conducted aerial and foot surveys during late-winter low-flow conditions in 2006, 2007, and 2008 to determine the extent of open water and ice cover (PLP 2011: Appendix 7.2B). Open-water reaches were consistently observed in strongly gaining reaches in the South and North Fork Koktuli Rivers and Upper Talarik Creek. Open-water reaches corresponded to areas of relatively warm groundwater that helped keep portions of the river network relatively ice-free (PLP 2011: Appendix 15.1E). Aerial surveys documented by Woody and Higman (2011) in March 2011 showed broadly similar patterns of open water, suggesting that the general patterns reflect consistent areas of strong groundwater–surface water interaction. Maintaining winter groundwater connectivity may be critical for fish in such streams (Cunjak 1996, Huusko et al. 2007, Brown et al. 2011).

7.3.3 Risk Characterization

The water consumption predicted for our mine scenarios would require large volumes of water from surface streams or groundwater, inevitably resulting in alterations to streamflows. Streamflow alterations exceeding 20% would occur in 15, 27, and 53 km of streams in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively, leading to significant adverse effects on fish and other aquatic life. The seasonal timing and magnitude of streamflow alterations would be contingent on water storage and management systems and strategies, but would be constrained by the fundamental needs for water at specific times and locations in the mining process (Chapter 6). Impacts on fish habitat and fish populations would likewise depend on the magnitude and timing of streamflow changes, but would be most severe for streams close to the mine footprint.

The volume of water that would require treatment by the mine's WWTP would range from 11 million m^3/yr for Pebble 0.25 (Table 7-16) to over 50 million m^3/yr for Pebble 6.5 (Table 7-18). To avoid or minimize risks associated with altered streamflows in downstream effluent-receiving areas, water storage and release capacities would be required to maintain natural streamflow regimes or to maintain any minimum streamflows required by regulatory agencies. Application of the Instream Flow Incremental Methodology (IFIM) Physical Habitat Simulation (PHABSIM) system modeling approach (Bovee 1982, Bovee et al. 1998) is being used by PLP to assess streamflow-habitat relationships (PLP 2011: Chapter 15), and could provide additional guidance for establishing streamflow requirements (Estes 1998) beyond those identified in this document.

Maintenance of mine discharges, in terms of water quality, quantity, and timing, to avoid adverse impacts would require long-term monitoring and facility maintenance commitments. 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 potential accidents, failures, and human error would increase with time. In addition, climate change and projected changes in temperature and precipitation in the region (Section 3.8) would result in potential changes in streamflow magnitude and seasonality. These climate-related changes would interact with mining-related flow impacts (Box 14-2), requiring adaptation to potentially new streamflow regimes. We know of no precedent for the long-term management of water quality and quantity on this scale at an inactive mine.

7.3.4 Uncertainties and Assumptions

Projecting changes to groundwater–surface water interactions in the mine footprint 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 footprint 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 WWTP effluent. Given the high likelihood of complex groundwater–surface water connectivity in the mine area, predicting and regulating streamflows to maintain key ecosystem functions associated with groundwater–surface water exchange would be particularly challenging.

Our approach for assessing potential risks of streamflow alteration rests on simplifying assumptions regarding changes to the natural streamflow regime in the three mine scenarios (Section 7.3.2). The natural streamflow 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 alterations in flow magnitude. However, it is very likely that other aspects of the streamflow regime would be modified as well, depending on how flows respond to water management at the mine site. In addition, any changes in the duration of open-water freezing conditions associated with mining activities could alter seasonal streamflow regimes differently than we assume here. Our analysis does not account for these possibilities.

We assumed that streamflow modifications would follow the natural hydrograph, reflecting the amount of precipitation and runoff that was intercepted and thus must be captured and treated. As a result, WWTP outfall flows are lowest during periods when these streams typically go dry based upon pre-mine baseline data, and are highest during snowmelt runoff and fall storms. Alternative flow management strategies may be feasible, depending on the capacity to store and release flows to meet environmental streamflow objectives (see Appendix J for additional discussion).

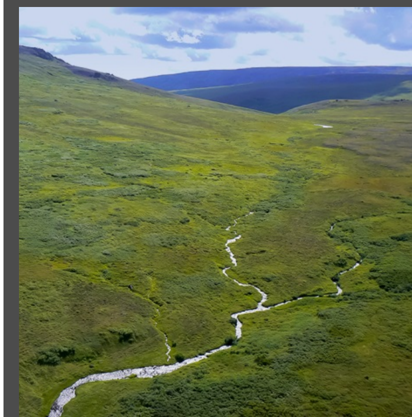
Additionally, we assume that larger deviations from the natural streamflow regime pose greater risks of ecological change. The scientific literature supports this assumption as a general trend (Poff et al. 2009, Poff and Zimmerman 2010, Richter et al. 2012). However, as pointed out by Poff and Zimmerman (2010), specific responses to changes in streamflow vary. Although all stream studies reviewed by Poff and Zimmerman (2010) showed declines in fish abundance, diversity, and demographic rates with any level of streamflow modification, other ecological responses (e.g., macroinvertebrate abundance, riparian vegetation metrics) sometimes increased. Responses of fish populations and other ecological metrics to streamflow modification would depend 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).

Potential impacts of the mine footprints discussed in this chapter do not explicitly take into account the effects of climate change. Over the time scale at which large-scale mining would potentially affect the assessment area, projected increases in temperature and precipitation may substantially change the physical environment (Section 3.8 and Box 14-2). Such changes could significantly alter the variability and magnitude of streamflows. Seasonal transitions between frozen and unfrozen conditions can strongly influence groundwater–surface water interactions and streamflow dynamics (Callegary et al. 2013). Duration of freezing conditions and timing of snowmelt may be highly sensitive to climate change, with significant implications for flow regimes. Increases in rain-on-snow events are likely, but the potential implications for flooding are unclear. Nevertheless, these changes in streamflow regime would likely lead to changes in sediment transport, bed stability, and channel morphology with potential adverse impacts to fish habitat and population genetic diversity and resiliency.

7.4 Summary of Footprint Effects

Streams eliminated, blocked, or dewatered by the mine footprints in the Pebble 0.25, 2.0, and 6.5 scenarios would result in the loss of 8, 22, or 36 km, respectively, of documented anadromous waters as defined in the AWC (Johnson and Blanche 2012). These lengths represent a loss of 2 to 11% of the total AWC length in the mine scenario watersheds (total AWC length = 322 km) (Johnson and Blanche 2012). An additional 30 to 115 km of headwater streams supporting habitat for non-anadromous fish species would be lost to the mine footprint in these scenarios. Loss of headwater streams to the footprints would alter groundwater–surface water hydrology, nutrient processing, and export rates of resources and materials to downstream aquatic ecosystems. Losses of wetlands would be 4.5, 12, and 18 km^2 in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. In addition, the Pebble 0.25, 2.0, and 6.5 scenarios would result in losses of 0.41, 0.93, and 1.8 km^2 of ponds and lakes, respectively. An unquantified area of riparian floodplain wetland habitat would either be lost or suffer substantial changes in hydrologic connectivity with streams because of reduced streamflow from the mine footprint.

Reduced streamflow resulting from water consumption in mine operations, ore processing, transport, and other processes, would further reduce the amount and quality of fish habitat downstream of the mine footprints. Changes in streamflow exceeding 20% would adversely affect habitat in an additional 15, 27, and 53 km of streams in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively, reducing production of coho salmon, sockeye salmon, Chinook salmon, rainbow trout, and Dolly Varden. Losses of stream habitat leading to losses of local, unique populations would erode the population diversity that is essential to the stability of the overall Bristol Bay salmon fishery (Schindler et al. 2010).



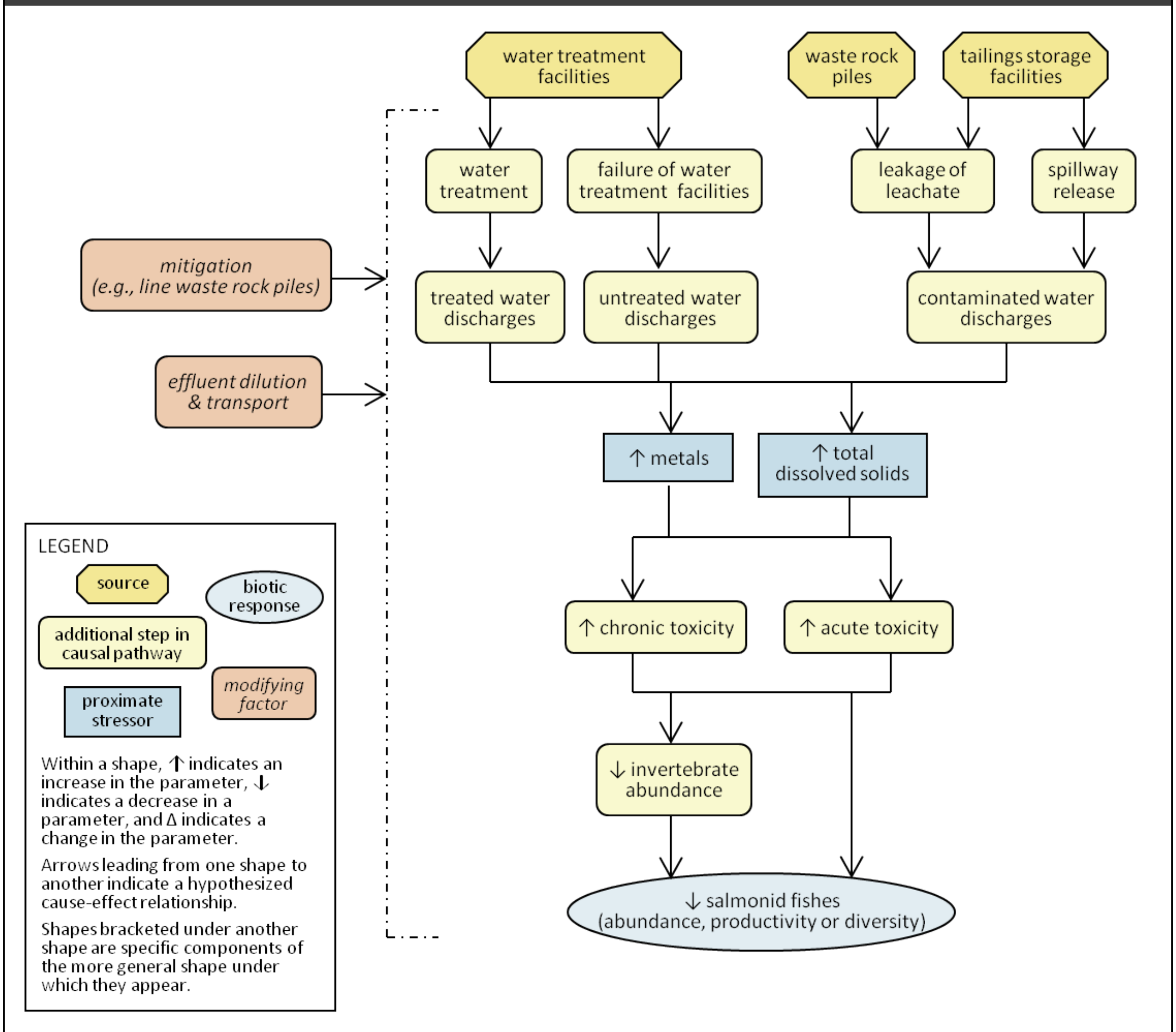
CHAPTER 8. WATER COLLECTION, TREATMENT, AND DISCHARGE

The water collection, treatment, and discharge scenarios presume that, under routine operations, runoff water, leachate, and wastewater would be collected and properly treated before release to meet state standards, federal criteria, and permit requirements. However, some leachate would escape collection, supernatant water may be spilled from tailings storage facilities (TSFs), and some treatment failures would be expected to occur. This chapter begins with a description of potential sources of contaminants (Section 8.1). It then describes potential routes and magnitudes of exposure to contaminated water and the exposure-response relationships used to screen leachate constituents (Section 8.2), with particular focus on the major contaminant of concern, copper. This section ends with a characterization of the potential risks from aqueous effluents and a discussion of potential additional remediation and uncertainties. Potential effects of water temperature changes associated with water collection, treatment, and discharge are discussed in Section 8.3. Figure 8-1 illustrates potential linkages between sources, stressors, and responses associated with water treatment, discharge, fate, and effects that are considered in this chapter.

8.1 Water Discharge Sources

Discharges were calculated for routine operations and wastewater treatment plant (WWTP) failure in the Pebble 0.25, 2.0, and 6.5 scenarios; post-closure discharges are discussed qualitatively. Sources of water discharge under routine operations associated with each mine scenario include effluents discharged from the WWTP, uncollected leachates from the TSFs and waste rock piles, and spillway releases from the TSFs. Other routine sources, including domestic wastewater, are outside the scope of this assessment and thus not analyzed here. In addition, we evaluate a WWTP failure scenario in which the system releases untreated wastewater. This failure represents one potential failure among many accidents and failures that could occur. We specify that under routine operations, the WWTP would meet permit limits. In the event of a complete treatment failure, flows would pass through the WWTP at the estimated influent concentrations. These two water collection, treatment, and discharge scenarios bound the likely range of water treatment operation, but do not encompass the worst case. For example, treatment might fail when wastewater composition is worse than average, or an extreme accident like dumping reverse-osmosis brine could occur.

Figure 8-1. Conceptual model illustrating the pathways linking water treatment, discharge, fate, and effects.



In addition to the discharge of treated water, water treatment generates wastes that are likely to be hazardous due primarily to the copper and other metals removed from the wastewaters. The treatment process is unspecified and it is unclear whether treatment wastes would be transported off site or deposited in the TSFs or another on-site facility. Therefore, this assessment does not include risks of water pollution resulting from spills of those waste materials and does not include them when estimating chemical concentrations in the TSF leakage or spillage.

Following the termination of mine operations, it is expected that water collection and treatment would continue for waste rock and tailings leachates. 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 in perpetuity—that is, until untreated water quality was acceptable or institutional failures ultimately resulted in abandonment of the system. If the mine operator abandons the site, the State of Alaska should assume operation of the treatment system; if both the mine operator and the State of Alaska abandon the site, untreated leachates would flow to streams draining the site.

The promulgated state water quality standards are enforceable numeric limits on the concentrations and durations of exposure for ambient waters, biotic communities, and associated designated uses. They would be applied to permits for the discharges discussed here. National ambient water quality criteria are contaminant limits that are recommended to the states. However, states such as Alaska may lag in adopting the latest criteria. In particular, the U.S. Environmental Protection Agency (USEPA) (2007) has published copper criteria based on the biotic ligand model (BLM), but Alaska still uses the hardness-based criteria for copper. We use the current USEPA copper criteria in this assessment based on the assumption that, before permitting a copper mine in the Bristol Bay watershed, Alaska would adopt those criteria at the state level or would apply them on a site-specific basis to any discharge permits.

8.1.1 Routine Operations

Under the mine scenarios, water in contact with tailings, waste rock, ore, product concentrate, or mine walls would leach minerals from those materials (Section 6.1.2.5). In addition, chemicals would be added to the water used in ore processing (Box 4-5). Most of the water used to transport tailings or concentrate or used in ore processing would be reused. Leachates collected from TSFs or waste rock piles would be stored in the TSF or treated for use or discharge, but leachate that escaped collection would flow to streams (Figure 6-5). Waste rock used in the construction of berms, roads and other mine structures would be leached by rain and snowmelt, but that source is assumed to be small relative to the waste rock piles and dams. Waste rock leachates are assumed to have the mean concentrations of reported humidity cell tests (Appendix H and PLP 2011). Mine pit water would also be used or treated for disposal. Surplus water on the site would be treated to meet applicable standards and other permit limits and discharged. Based on Alaskan Water Quality Standards defined in the Alaska Administrative Code, Title 18, Section 70, no mixing zones would be authorized for anadromous streams or spawning habitat for most game or subsistence fish species. Thus, it is expected that effluents would be required to meet state standards that are equivalent to national criteria and other permit limits (i.e., no exemptions would be granted).

During mine operations, water available on the site would exceed operational needs, and approximately 11 to 51 million m^3 of treated water would be discharged per year (Table 6-3). The mine scenarios specify that effluent would be discharged to the South and North Fork Koktuli Rivers as proposed by Ghaffari et al. (2011) (Tables 8-1 through 8-3). The effluent could contain treated tailings leachate, waste rock leachate, mine pit water, runoff, and excess transport or process waters. Tailings leachate would come from the TSFs as either excess water in the impoundment or leakage captured below the dams. The primary concern during routine operations would be waste rock leachate. Captured waste rock leachate would become more voluminous as the waste rock piles increased during operation. After mine closure, that leachate would be a major component of routinely generated wastewater, along with water pumped from the TSFs and the pit (after it has filled). In addition, because the waste rock piles and TSFs would not be lined, some leachates from both would not be captured and would flow to the three receiving streams.

Risk quotients are used to determine whether the leachates are potentially toxic and, if so, which constituents are most responsible (Tables 8-4 through 8-8). A risk quotient equals the exposure level divided by an ecotoxicological benchmark. For screening, the undiluted leachate concentration is treated as an exposure level. The benchmarks are national ambient water quality criteria or equivalent values (Section 6.4.2.3). These benchmarks are for either acute (the criterion maximum concentration, or CMC) or chronic (the criterion continuous concentration, or CCC) exposures—that is, CMCs are intended to be thresholds for significant lethality in short-term exposures, whereas CCCs are intended to be thresholds for significant lethal or nonlethal effects in long-term exposures. If the quotient is less than 1, the leachate or constituent can be eliminated as a chemical of potential concern because instream concentrations would not exceed the undiluted concentrations.

8.1.1.1 Tailings Leachate

Estimation of potential flow through the substrate located under and around proposed TSFs requires estimation of hydraulic conductivity. The hydraulic conductivity of the substrate material located near possible dam sites varies greatly with depth and location. Ghaffari et al. (2011) report a range from 10^{-6} to 10^{-5} m/s in the upper bedrock, with a general decrease with depth and a range on the order of 10^{-7} to 10^{-9} m/s in the lower portions of bedrock with some zones of higher hydraulic conductivity (Figure 6-7). In addition, the presence of fractured bedrock allows for localized discontinuities in the rate of groundwater movement that can greatly influence overall groundwater conveyance (Ghaffari et al. 2011).

We estimated leachate flow from the TSFs using a hydraulic conductivity of 10^{-6} m/s in the upper 100 m of overburden and bedrock, with no flow below that depth. We allowed vertical downward flow in the tailings and radial flow outward in all directions from the TSF, with the excess head dissipating over a horizontal distance of 1,200 m, comparable to the distance of the mine pit drawdown beyond the pit rim. The interior surface area of TSF 1 would be 6.5 km^2 for the Pebble 0.25 scenario and 14.2 km^2 for the Pebble 2.0 and 6.5 scenarios (Table 6-2). The Pebble 6.5 scenario would include two additional impoundments, with interior surface areas of 20.1 km^2 (TSF 2) and 8.2 km^2 (TSF 3) (Table 6-2).

Table 8-1. Annual effluent and receiving water flows at each gage in the Pebble 0.25 scenario. All values are presented in m^3/yr.

Stream and Gage	Flow Returned Through WWTP[a]	Flow Returned as TSF Leakage	Flow Returned as NAG Waste Rock Leachate	Flow Returned as PAG Waste Rock Leachate	Flow of Interbasin Transfer	TOTAL FLOW
South Fork Koktuli River						
SK100G	-	-	207,000	-	-	5,287,000
SK100F	-	-	350,000	-	-	19,055,000
SK100CP2[b,c]	-	-	−185,000	-	−12,446,000	25,263,000
SK124A	5,454,000	-	-	-	-	22,265,000
SK124CP[b]	-	-	-	-	-	23,391,000
SK100C	-	-	-	-	-	43,942,000
SK100CP1[b]	-	-	-	-	-	44,113,000
SK119A	-	-	-	-	-	31,268,000
SK119CP[b]	-	-	-	-	-	33,124,000
SK100B1	-	-	-	-	-	113,737,000
SK100B[d]	-	-	-	-	-	159,937,000
North Fork Koktuli River						
NK119A	-	1,113,000	120,000	-	-	15,378,000
NK119CP2[b]	-	-	-	-	-	17,923,000
NK119B	-	-	-	-	-	3,975,000
NK119CP1[b]	-	-	-	-	-	23,512,000
NK100C	5,454,000	-	-	-	-	47,282,000
NK100B	-	-	-	-	-	77,513,000
NK100A1	-	-	-	-	-	183,022,000
NK100A[e]	-	-	-	-	-	221,668,000
Upper Talarik Creek						
UT100E	-	-	-	-	-	7,474,000
UT100D	-	-	-	-	-	22,008,000
UT100C2	-	-	-	-	-	90,768,000
UT100C1	-	-	-	-	-	106,050,000
UT100C	-	-	-	-	-	139,053,000
UT119A[c]	-	-	185,000	-	12,446,000	23,286,000
UT100B[f]	-	-	-	-	-	191,423,000

Notes:
Dashes (-) indicate that values are either not applicable or are equal to zero.
[a] WWTP discharges 50% of flow to South Fork Koktuli River, 50% of flow to North Fork Koktuli River (no WWTP flows are directed to Upper Talarik Creek).
[b] Confluence point where virtual gage was created because physical gage does not exist.
[c] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location. Interbasin transfer flows are represented by negative flow values for SK100CP2 (losses to Upper Talarik Creek) and equivalent positive flow values for UT119A (gains from South Fork Koktuli).
[d] USGS 15302200.
[e] USGS 15302250.
[f] USGS 15300250.
WWTP = wastewater treatment plant; TSF = tailings storage facility; PAG = potentially acid-generating; NAG = non-acid-generating.

Table 8-2. Effluent and receiving water flows at each gage in the Pebble 2.0 scenario. All values are presented in m^3/yr.

Stream and Gage	Flow Returned Through WWTP[a]	Flow Returned as TSF Leakage	Flow Returned as NAG Waste Rock Leachate	Flow Returned as PAG Waste Rock Leachate	Flow of Interbasin Transfer	TOTAL FLOW
South Fork Koktuli River						
SK100G	-	-	633,000	213,000	-	3,266,000
SK100F	-	-	507,000	3,000	-	16,745,000
SK100CP2[b,c]	-	-	−380,000	−72,000	−11,409,000	23,723,000
SK124A	5,152,000	2,000	22,000	-	-	21,878,000
SK124CP[b]	-	-	-	-	-	23,004,000
SK100C	-	-	-	-	-	42,552,000
SK100CP1[b]	-	-	-	-	-	42,722,000
SK119A	-	21,000	151,000	-	-	30,774,000
SK119CP[b]	-	-	-	-	-	32,630,000
SK100B1	-	-	-	-	-	110,513,000
SK100B[d]	-	-	-	-	-	156,302,000
North Fork Koktuli River						
NK119A	-	2,305,000	402,000	-	-	8,111,000
NK119CP2[b]	-	1,000	13,000	-	-	10,347,000
NK119B	-	-	3,000	-	-	3,641,000
NK119CP1[b]	-	-	-	-	-	17,069,000
NK100C	5,152,000	-	-	-	-	46,905,000
NK100B	-	-	-	-	-	71,452,000
NK100A1	-	23,000	204,000	-	-	176,169,000
NK100A[e]	-	-	-	-	-	215,132,000
Upper Talarik Creek						
UT100E	-	-	-	-	-	5,290,000
UT100D	-	-	642,000	-	-	13,481,000
UT100C2	-	-	-	-	-	83,215,000
UT100C1	-	-	-	-	-	98,684,000
UT100C	-	-	-	-	-	130,691,000
UT119A[c]	-	-	380,000	72,000	11,409,000	22,516,000
UT100B[f]	-	-	-	-	-	180,889,000

Notes:

Dashes (-) indicate that values are either not applicable or are equal to zero.

[a] WWTP discharges 50% of flow to South Fork Koktuli River, 50% of flow to North Fork Koktuli River (no WWTP flows are directed to Upper Talarik Creek).

[b] Confluence point where virtual gage was created because physical gage does not exist.

[c] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location. Interbasin transfer flows are represented by negative flow values for SK100CP2 (losses to Upper Talarik Creek) and equivalent positive flow values for UT119A (gains from South Fork Koktuli).

[d] USGS 15302200.

[e] USGS 15302250.

[f] USGS 15300250.

WWTP = wastewater treatment plant; TSF = tailings storage facility; PAG = potentially acid-generating; NAG = non-acid-generating.

Table 8-3. Effluent and receiving water flows at each gage in the Pebble 6.5 scenario. All values are presented in m^3/yr.

Stream and Gage	Flow Returned Through WWTP[a]	Flow Returned as TSF Leakage	Flow Returned as NAG Waste Rock Leachate	Flow Returned as PAG Waste Rock Leachate	Flow of Interbasin Transfer	TOTAL
South Fork Koktuli River						
SK100G	-	-	-	-	-	95,000
SK100F	-	20,000	1,278,000	1,032,000	-	9,810,000
SK100CP2[b,c]	-	−7,000	−423,000	−344,000	−8,770,000	19,096,000
SK124A	25,494,000	1,626,000	713,000	-	-	36,049,000
SK124CP[b]	-	-	-	-	-	37,175,000
SK100C	-	-	54,000	-	-	57,070,000
SK100CP1[b]	-	-		-	-	57,241,000
SK119A		2,930,000	242,000			14,262,000
SK119CP[b]	-	50,000	413,000	-	-	15,172,000
SK100B1	-	145,000	260,000	-	-	104,322,000
SK100B[c]	-	-	-	-	-	146,346,000
North Fork Koktuli River						
NK119A	-	2,360,000	402,000	-	-	8,167,000
NK119CP2[b]	-	1,000	13,000	-	-	10,402,000
NK119B	-	48,000	144,000	-	-	3,004,000
NK119CP1[b]	-	-	-	-	-	15,473,000
NK100C	25,494,000	-	-	-	-	67,053,000
NK100B	-	-	-	-	-	90,144,000
NK100A1	-	23,000	204,000	-	-	194,738,000
NK100A[e]	-	-	-	-	-	233,778,000
Upper Talarik Creek						
UT100E	-	-	346,000	-	-	2,125,000
UT100D	-	-	739,000	-	-	3,482,000
UT100C2	-	-	160,000	-	-	73,815,000
UT100C1	-	-		-	-	89,511,000
UT100C	-	-		-	-	120,303,000
UT119A[c]	-	7,000	428,000	344,000	8,770,000	20,203,000
UT100B[f]	-	-	-	-	-	166,476,000

Notes:
Dashes (-) indicate that values are either not applicable or are equal to zero. SK100G and SK119A are eliminated by the mine footprint in this scenario.
[a] WWTP discharges 50% of flow to South Fork Koktuli River, 50% of flow to North Fork Koktuli River (no WWTP flows are directed to Upper Talarik Creek).
[b] Confluence point where virtual gage was created because physical gage does not exist.
[c] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location. Interbasin transfer flows are represented by negative flow values for SK100CP2 (losses to Upper Talarik Creek) and equivalent positive flow values for UT119A (gains from South Fork Koktuli).
[d] USGS 15302200.
[e] USGS 15302250.
[f] USGS 15300250.
WWTP = wastewater treatment plant; TSF = tailings storage facility; PAG = potentially acid-generating; NAG = non-acid-generating.

Table 8-4. Aquatic toxicological screening of tailings supernatant against acute (criterion maximum concentration) and chronic (criterion continuous concentration) water quality criteria or benchmark values. Values are in µg/L unless otherwise indicated. Average leachate values are from Appendix H.

Analyte	Average Value	CMC or equivalent	CCC or equivalent	Acute Quotient	Chronic Quotient
pH (standard units)	7.9	-	-	-	-
Alkalinity (mg/L $CaCO_3$)	75	-	-	-	-
Hardness (mg/L $CaCO_3$)	320	-	-	-	-
SO_4	320,000	-	-	-	-
Ag	0.018	24	-	0.0007	-
Al	72	750	87	0.096	0.82
As	17	340	150	0.051	0.11
Ca	116,000	-	-	-	-
Cd	<0.1	6.3	0.55	<0.012	<0.14
Co	<0.1	89	2.5	<0.0011	<0.040
Cr	<1.0	1,500	190	<0.0007	<0.0051
Cu[a]	7.8	40	24	0.19	0.32
Cu[b]	7.8	7.2	4.4	1.1	1.8
Fe	17	350	-	0.048	-
Hg	<0.037	1.4	0.77	<0.026	<0.048
K	26,000	-	-	-	-
Mg	8,000	-	-	-	-
Mn	72	760	690	0.095	0.10
Mo	70	32,000	73	0.0022	0.96
Na	44,000	-	-	-	-
Ni	<0.8	1,300	140	<0.0006	<0.0056
Pb	0.2	220	8.8	0.0010	0.026
Sb	6.0	14,000	1,600	0.0004	0.0038
Se	7.6	-	5	-	1.5
Tl	0.0	-	-	-	-
Zn	4.3	316	316	0.014	0.014
Sum of metals	-	-	-	0.50[a] : 1.4[b]	4.3[a] : 5.8[b]

Dashes (-) indicate that criteria are not available.

[a] Acute and chronic criteria from Alaska's hardness-based standard.

[b] Acute and chronic criteria from the national ambient water quality criteria based on the biotic ligand model.

CMC = criterion maximum concentration; CCC = criterion continuous concentration.

Table 8-5. Aquatic toxicological screening of tailings humidity cell leachates against acute (criterion maximum concentration) and chronic (criterion continuous concentration) water quality criteria or benchmark values. Values are in μg/L unless otherwise indicated. Average concentrations are from Appendix H.

Analyte	Average Value	CMC or equivalent	CCC or equivalent	Acute Quotient	Chronic Quotient
pH (standard units)	7.8	-	6.5–9	-	-
Alkalinity (mg/L $CaCO_3$)	60	-	-	-	-
Hardness (mg/L $CaCO_3$)	67	-	-	-	-
Cl	520	-	-	-	-
F	450	-	-	-	-
SO_4	17,000	-	-	-	-
Ag	0.01	1.6	-	0.0062	-
Al	24	750	87	0.031	0.27
As	5.5	340	150	0.016	0.036
B	11	29,000	1,500	0.0004	0.0071
Ba	9.2	46,000	8,900	0.0002	0.0010
Be	0.20	-	-	-	-
Bi	0.49	-	-	-	-
Ca	23,000	-	-	-	-
Cd	0.05	1.5	0.20	0.038	0.28
Co	0.19	89	2.5	0.0021	0.076
Cr	0.50	445	58	0.0012	0.0094
Cu[a]	5.3	10	6.9	0.58	0.84
Cu[b]	5.3	2.5	1.6	1.1	1.8
Fe	30	350	-	-	-
Hg	0.01	1.4	0.77	0.0071	0.013
K	4,000	-	-	-	-
Mg	2,500	-	-	-	-
Mn	44	760	693	0.058	0.064
Mo	33	32,000	73	0.0010	0.45
Na	2,100	-	-	-	-
Ni	0.54	360	40	0.0016	0.014
Pb	0.06	46	1.8	0.0015	0.039
Sb	1.8	14,000	1,600	-	-
Se	1.5	-	5.0	-	0.30
Sn	2.9	3,600	75	0.0008	0.039
Tl	0.05	-	0.8	-	-
V	0.78	1,370	120	0.0006	0.0065
Zn	3.2	91	91	0.038	0.038
Sum of metals	-	-	-	0.78[a] : 1.3[b]	2.5[a] : 3.4[b]

Notes:
Dashes (-) indicate that criteria are not available.
[a] Acute and chronic criteria from Alaska's hardness-based standard.
[b] Acute and chronic criteria from the national ambient water quality criteria based on the biotic ligand model.
CMC = criterion maximum concentration; CCC = criterion continuous concentration.

Table 8-6. Aquatic toxicological screening of test leachate from Tertiary waste rock in the Pebble deposit and quotients against acute (criterion maximum concentration) and chronic (criterion continuous concentration) water quality criteria or benchmark values. Values are in µg/L unless otherwise indicated. Average leachate concentrations are from Appendix H.

Parameter	Average Value	CMC or equivalent	CCC or equivalent	Acute Quotients	Chronic Quotients
pH	7.2	-	6.5–9	-	-
Alkalinity (mg/L $CaCO_3$)	66	-	-	-	-
Hardness (mg/L $CaCO_3$)	74	-	-	-	-
Cl	530	-	-	-	-
F	62	-	-	-	-
SO_4	28,000	-	-	-	-
Ag	0.011	1.9	-	0.0059	-
Al	80	750	87	0.11	0.92
As	2.7	340	150	0.0081	0.018
B	18	29,000	1,500	0.0006	0.012
Ba	57	46,000	8,900	0.0012	0.0064
Be	0.31	-	-	-	-
Bi	0.54	-	-	-	-
Ca	21,000	-	-	-	-
Cd	0.22	1.5	0.20	0.15	1.1
Co	3.9	89	2.5	0.044	1.6
Cr	0.55	445	58	0.0012	0.0094
Cu[a]	3.2	10	6.9	0.32	0.46
Cu[b]	3.2	2.5	1.6	1.3	2.0
Fe	140	350		0.40	
Hg	0.010	1.4	0.77	0.0073	0.013
K	1,900	-	-	-	-
Mg	5,100	-	-	-	-
Mn	100	760	693	0.13	0.15
Mo	6.3	32,000	73	0.0002	0.086
Na	7,200	-	-	-	-
Ni	4.4	360	40	0.012	0.11
Pb	0.12	46	1.8	0.0025	0.06
Sb	2.1	14,000	1,600	0.0002	0.0013
Se	1.9	-	5.0	-	0.38
Sn	1.3	3,600	75	0.0003	0.017
Tl	0.068	-	0.8	-	0.085
V	1.8	1,370	120	0.0013	0.15
Zn	16	91	91	0.17	0.17
Sum of metals	-	-	-	1.4[a] : 2.4[b]	5.3[a] : 6.8[b]

Notes:
Dashes (-) indicate that criteria are not available.
[a] Acute and chronic criteria from Alaska's hardness-based standard.
[b] Acute and chronic criteria from the national ambient water quality criteria based on the biotic ligand model.
CMC = criterion maximum concentration; CCC = criterion continuous concentration.

Table 8-7. Aquatic toxicological screening of test leachate from Pebble East pre-Tertiary waste rock and quotients against acute (criterion maximum concentration) and chronic (criterion continuous concentration) water quality criteria or benchmark values. Values are in µg/L unless otherwise indicated. Average leachate values are from Appendix H.

Parameter	Average Value	CMC or equivalent	CCC or equivalent	Acute Quotients	Chronic Quotients
pH (standard units)	4.8	-	6.5–9	-	-
Alkalinity (mg/L $CaCO_3$)	9.9	-	-	-	-
Hardness (mg/L $CaCO_3$)	22	-	-	-	-
Cl	910	-	-	-	-
F	110	-	-	-	-
SO_4	52,000	-	-	-	-
Ag	0.019	0.24		0.082	-
Al	380	750	87	0.51	4.4
As	8.0	340	150	0.023	0.053
B	13	29,000	1,500	0.0004	0.0084
Ba	4.5	46,000	8,900	0.0001	0.0005
Be	0.55	-	-	-	-
Bi	0.63	-	-	-	-
Ca	6,300	-	-	-	-
Cd	3.2	0.46	0.085	7.0	38
Co	9.7	89	2.5	0.11	3.9
Cr	1.6	160	21	0.0096	0.073
Cu[a]	1,400	3.20	2.4	440	580
Cu[b]	1,400	0.043	0.027	33,000	52,000
Fe	10,000	350	-	-	-
Hg	0.010	1.4	0.77	0.0072	0.013
K	960	-	-	-	-
Mg	1,500	-	-	-	-
Mn	340	760	693	0.44	0.49
Mo	4.3	32,000	73	0.0001	0.059
Na	2,100	-	-	-	-
Ni	10	130	14	0.081	0.73
Pb	0.35	12	0.47	0.029	0.75
Sb	0.78	14,000	1,600	0.0001	0.0005
Se	3.2	-	5.0	-	0.65
Sn	1.9	3,600	75	0.0005	0.024
Tl	0.088	-	0.8	-	0.110
V	2.4	1,370	120	0.0018	0.020
Zn	480	32	32	15	15
Sum of metals	-	-	-	460[a] : 33,000[b]	640[a] : 52,000[b]

Notes:
Dashes (-) indicate that criteria are not available.
[a] Acute and chronic criteria from Alaska's hardness-based standard.
[b] Acute and chronic criteria from the national ambient water quality criteria based on the biotic ligand model.
CMC = criterion maximum concentration; CCC = criterion continuous concentration.

Table 8-8. Aquatic toxicological screening of test leachate from Pebble West pre-Tertiary waste rock against acute (criterion maximum concentration) and chronic (criterion continuous concentration) water quality criteria or benchmark values. Values are in µg/L unless otherwise indicated. Average leachate values are from Appendix H.

Parameter	Average Value	CMC or equivalent	CCC or equivalent	Acute Quotients	Chronic Quotients
pH (standard units)	6.6	-	6.5–9	-	-
Alkalinity (mg/L $CaCO_3$)	18	-	-	-	-
Hardness (mg/L $CaCO_3$)	59	-	-	-	-
Cl	520	-	-	-	-
F	120	-	-	-	-
SO_4	61,000	-	-	-	-
Ag	0.027	1.3	-	0.021	-
Al	320	750	87	0.42	3.7
As	1.5	340	150	0.0044	0.0100
B	16	29,000	1,500	0.0005	0.011
Ba	14	46,000	8,900	0.0003	0.0015
Be	0.33	-	-	-	-
Bi	0.69	-	-	-	-
Ca	13,000	-	-	-	-
Cd	0.40	1.2	0.17	0.33	2.3
Co	7.0	89	2.5	0.079	2.8
Cr	0.69	370	48	0.0019	0.014
Cu[a]	1,600	8.2	5.7	190	280
Cu[b]	1,600	0.88	0.55	1,800	2,900
Fe	1,700	350	-	4.8	-
Hg	0.011	1.4	0.77	0.0076	0.014
K	1,400	-	-	-	-
Mg	6,700	-	-	-	-
Mn	730	760	690	0.96	1.1
Mo	1.8	32,000	73	0.0001	0.025
Na	2,100	-	-	-	-
Ni	6.8	300	33	0.023	0.20
Pb	0.17	36	1.4	0.0047	0.12
Sb	3.1	14,000	1,600	-	-
Se	3.8	-	5.0	-	0.76
Sn	0.14	3,600	75	0.00004	0.0019
Tl	0.41	-	0.8	-	0.52
V	0.68	1,370	120	0.0005	0.0057
Zn	56	75	75	0.74	0.74
Sum of metals	-	-	-	200[a] : 1,800[b]	290[a] : 2,900[b]

Notes:
Dashes (-) indicate that criteria are not available.
[a] Acute and chronic criteria from Alaska's hardness-based standard.
[b] Acute and chronic criteria from the national ambient water quality criteria based on the biotic ligand model.
CMC = criterion maximum concentration; CCC = criterion continuous concentration.

Total leakage amounts for the three mine scenarios are 1.1 x 10^6 m^3/yr (Pebble 0.25), 2.4 x 10^6 m^3/yr (Pebble 2.0), and 7.2 x 10^6 m^3/yr (Pebble 6.5) (Tables 8-1 through 8-3). These estimates are based on a simple assessment of seepage from the TSFs. Actual hydraulic conductivity would likely span several orders of magnitude, from rapid flow in large fractures to essentially no flow in tight formations. Even a small number of flowpaths with higher than expected hydraulic conductivity could significantly affect the direction and quantity of flow.

Two potential estimates of tailings leachate composition are presented in Tables 8-4 and 8-5. Tailings leachate from the humidity cell tests (Table 8-5) is judged to better represent effluent from a tailings impoundment than the supernatant (Table 8-4); thus, these values are used to represent leachate from the bottom of the TSFs and excess water from the TSFs routed to the WWTP.

The tailings slurry would also contain ore-processing chemicals. We use an estimated concentration of sodium ethyl xanthate, the primary ore-processing contaminant of concern, of 1.5 mg/L in the tailings slurry (NICNAS 1995). Process chemicals could enter the environment in TSF leachate or WWTP effluent. The potential for process chemicals in product concentrate slurry is considered in Chapter 11.

8.1.1.2 Waste Rock Leachate

Tertiary rock would be used for construction of tailings dams and berms and potentially other structures requiring fill, but most would be piled near the mine pit. It is classified as non-acid-generating (NAG) and its leachate is neutral (Table 8-6). Pre-Tertiary rock is classified as potentially acid-generating (PAG) and its leachate is acidic (Tables 8-7 and 8-8). PAG waste rock would be piled separately and blended with ore, as needed, to maintain consistent composition in the processing plant feed. Incomplete collection of pre-Tertiary waste rock leachate would result in acid mine drainage.

The mine scenarios (and the plan put forth for Northern Dynasty Minerals by Ghaffari et al. [2011]) do not include liners for the waste rock piles. Instead, leachate within the mine pit's drawdown zone would be captured in the pit and pumped to the WWTP. Outside the drawdown zone, we estimate that half the leachate would be captured by extraction wells or other means and the rest would flow to surface waters. This is considered reasonable given the likelihood that water would flow between wells and below their zones of interception in the relatively permeable overburden materials and upper bedrock. Wells would not catch all flows from the mine site given its geological complexity and the permeability of surficial layers. As a result, we estimate that 84% of PAG leachate and 82% of total waste rock leachate would be captured by the pit and the wells for the Pebble 2.0 scenario.

8.1.1.3 Mine Pit and Runoff Water

Water pumped from the mine pit would consist of captured waste rock leachate and leachate from the pit walls as precipitation passes over them and groundwater flows through them. The pit wall leachate is estimated from the maximum groundwater concentration at the mine site, because rainwater flowing through the ore body and rocks in its vicinity is assumed to be similar to rainwater flowing over the pit walls. The estimated concentration of the critical contaminant, 3.2 μg/L copper, is almost identical to the mean Tertiary (NAG) waste rock test leachate. Other constituent concentrations are 37 μg/L aluminum, 0.05 μg/L cadmium, 0.63 μg/L cobalt, 45 μg/L manganese, 3.2 μg/L nickel, 0.86 μg/L lead, 0.30 μg/L selenium, 7.9 μg/L zinc, 1600 mg/L total dissolved solids, and 5.6 pH. This means that the mine pit water is much cleaner than PAG pre-Tertiary leachate (e.g., copper in estimated pit wall leachate is only 0.2% of PAG waste rock leachate).

Runoff from the ore-crushing and screening area is assumed to have the composition of pre-Tertiary (PAG) waste rock test leachate (Table 8-7). All other plant and ancillary area runoff is assumed to have the composition of the maximum background stream water. All of these waters would be captured and routed to a TSF or the WWTP.

8.1.1.4 Wastewater Discharge

Under the three mine scenarios, the WWTP would be designed and sized to treat the expected volume and composition of inflow water based on estimated groundwater flow from the mine pit and runoff from other site areas (waste rock piles, TSFs, and plant and ancillary facilities). The WWTP would be fed by pipelines that pump water to the plant from the mine pit, crusher area, waste rock and TSF leachate collection systems, and other operating areas of the site. However, mine pit water represents the largest component of flow into the WWTP in our scenarios. The flow volume contributed by each mine component has been estimated for each scenario (Table 6-3). If the volume or composition of untreated water exceeded plant specifications, it could be stored temporarily in a TSF process pond or even the mine pit, and fed into the plant as needed to balance flows and meet permit effluent quality requirements.

We specify that the WWTP would operate under a permit that would require meeting all national criteria and Alaskan standards. We also assume that the Alaskan Pollutant Discharge Elimination System wastewater discharge permit for a mine would include requirements that all other potentially toxic contaminants be kept below concentrations equivalent to national chronic criteria. This use of non-standard benchmarks in permitting is not normal practice, but the importance of the aquatic resources and the degree of public concern would justify that action. The equivalent benchmark values used in this assessment for metals with no criteria or standards appear in Table 6-10. Assumed discharge concentrations are the minimum of the input water concentration and the chronic criterion, standard or benchmark value. Influent and effluent concentrations of contaminants of concern are presented in Table 8-9. WWTP discharge rates for the Pebble 0.25, 2.0, and 6.5 scenarios are estimated to be approximately 11, 10, and 51 million m^3/year, respectively, equally distributed to the South and North Fork Koktuli Rivers.

8.1.1.5 Sources of Total Dissolved Solids

Neither total dissolved solids (TDS) nor specific conductance data are available for waste rock or tailings leachates from the Pebble deposit. However, TDS can be estimated by summing the concentrations of leachate analytes after converting alkalinity to bicarbonate. Estimated TDS concentrations for the tailings leachates, waste rock leachates, and WWTP effluents are summarized in Table 8-9.

Table 8-9. Estimated concentration of contaminants of concern in effluents from the wastewater treatment plant, tailings, non-acid-generating waste rock, and potentially acid generating waste rock. Values are in μg/L unless otherwise indicated.

Contaminant	WWTP Influent and Failure Effluent[a]			WWTP Effluent[a]			Tailings Leachate	NAG Waste Rock Leachate	PAG Waste Rock Leachate
	0.25[a]	2.0[b]	6.5[c]	0.25[a]	2.0[b]	6.5[c]			
TDS (mg/L)	312	297	529	280[d]			123	145	100
Zn	17	26	33	17	23	23	3.2	16	270
Se	1.6	1.4	1.5	1.6	1.4	1.4	1.5	1.9	3.5
Pb	0.22	0.28	0.33	0.22	0.28	0.29	0.064	0.12	0.26
Ni	2.3	3.1	3.3	2.3	3.2	3.3	0.54	4.4	8.6
Mn	67	92	105	67	92	100	44	101	530
Co	0.99	2.2	2.1	0.99	2.2	2.0	0.19	3.9	8.4
Cd	0.14	0.22	0.26	0.064	0.064	0.064	0.052	0.22	1.8
Al	44	66	73	44	66	73	24	80	350
Cu	75	101	150	1.1[d,e]			5.3	3.2	1,500

Notes:
[a] Concentrations for the Pebble 0.25 scenario.
[b] Concentrations for the Pebble 2.0 scenario.
[c] Concentrations for the Pebble 6.5 scenario.
[d] When only one value is shown across all three scenarios, it means that the contaminant is above the chronic criterion and must be lowered to the criterion under all three scenarios.
[e] Chronic water quality criterion based on the biotic ligand model using mean North Fork Koktuli River water.
WWTP = wastewater treatment plant; TDS = total dissolved solids; NAG = non-acid-generating; PAG = potentially acid-generating.

8.1.2 Wastewater Treatment Plant Failure

There are innumerable ways in which wastewater treatment could fail in the mine scenarios, in terms of failure type (e.g., breakdown of treatment equipment, ineffective leachate collection, wastewater pipeline failure), location, duration, and magnitude (e.g., partial vs. no treatment). Box 8-1 presents an example wastewater collection failure, and mechanisms of treatment failure are discussed in Box 8-2. To bound the range of reasonable possibilities, we assess a serious failure in which the WWTP allows untreated water to discharge directly to streams. This type of failure could result from a lack of storage or treatment capacity or treatment efficacy problems. Chronic releases would occur during operation if a lengthy process were required to repair a failure. We evaluate potential effects of this type of failure using the following assumptions.

- The effluent is untreated water that is released to discharge points on tributaries to the South and North Fork Koktuli Rivers.
- Untreated water composition is a flow-weighted average of concentrations from multiple wastewater sources, including mine pit dewatering, waste rock leachates, runoff from crusher and ancillary areas, and TSF leachates.
- Discharge rates are based on the sum of component flow volumes from the wastewater sources, developed as part of the scenario water balances (Section 6.2.2).
- Discharge rates and concentrations were calculated for each of the three mine scenarios (Pebble 0.25, 2.0 and 6.5) and account for shifts in the relative contribution and concentration of different wastewater sources for different mine sizes.

- Duration of a release could range from a few hours to several months, depending on the nature of the failure and the difficulty of repair and replacement.

BOX 8-1. 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 mine is located on federal lands managed by the Bureau of Land Management. The mine operates under authorizations from the Bureau of Land Management, the Alaska Department of Natural Resources (ADNR), and the Alaska Department of Environmental Conservation (ADEC).

In January and February 2012, the tailings impoundment at the Nixon Fork Mine overtopped. Below is the chronology of events described by the mine operator that led to this event, based on a March 15, 2012 memo to the Alaska State Mine Safety Engineer from Mystery Creek Resources, Inc.

- Prior to October 25, 2011, mine staff monitored the freeboard in the tailings impoundment per requirements of agency authorizations.
- After October 25, 2011, 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 president of Mystery Creek Resources, Inc. noticed insufficient freeboard in the tailings pond. He notified the 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 (i.e., more water would be flowing to the tailings impoundment).
- On March 9, 2012, mine personnel noticed evidence of dam overtopping. The 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 dam inspection 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, such that the later spill overtopped the dam at another location not designed for overflow. This case illustrates the diversity of potential failures that can happen and suggests the practical impossibility of predicting all possible failure modes.

Water treatment also would generate sludges or brines containing material removed from the wastewaters plus materials added to the water, such as precipitating agents. These materials are expected to be deposited in the TSFs. Because the mine scenarios do not include a specific water treatment technology, no spill scenario for these wastes was developed. However, copper and other metal concentrations in these wastes would be high, so they likely would be significantly toxic if spilled into surface waters.

If a gold-processing facility was added at the site, a separate water treatment system would decompose or recycle the cyanide used in the separation of gold (Box 4-6). That system would have the potential to fail, releasing the cyanide solution to a stream or groundwater. Cyanide in the tailings would flow to a TSF, where it could degrade or combine with copper or other metals. However, a cyanide-processing system has not been described, and we do not consider a water treatment failure scenario for this potential source.

BOX 8-2. POTENTIAL FAILURES OF REVERSE OSMOSIS WASTEWATER TREATMENT PLANTS

Because the high-quality receiving waters in the mine scenario watersheds would require extremely low copper criteria and standards, reverse osmosis has been discussed as a potential treatment technology for wastewater at the Pebble site. Studies of wastewater treatment plant (WWTP) efficiency and design considerations show that reverse osmosis water treatment systems can be compromised by fouling and scaling from calcium, iron, barium, strontium, silica, microbial growth, and silt (Mortazavi 2008). The Bingham Canyon WWTP in Utah treats groundwater contaminated with sulfate and total dissolved solids from copper mining by reverse osmosis. Pilot tests and optimization studies have shown that the structural integrity of its reverse osmosis membranes can be damaged by abrasive materials (e.g., silt) or chlorine (ITRC 2010). Changes in water composition could increase the concentration of chlorine if the mine pit encounters a large flow of brine transmitted to the pit through deep fracture systems, or from localized areas of mineralized rock with anomalous water quality. An example of WWTP failure due to highly variable chemical composition of inflow wastewater has been documented at a copper mine in Chile: when silica concentrations exceeded the design range, the whole reverse osmosis system could not be operated and was therefore shut down until feed water quality improved (Shao et al. 2009).

8.1.3 Spillway Release

The spillway release scenario considered here involves the controlled release of water from TSF 1 to the North Fork Koktuli River. Spillway releases are not part of routine operations; however, because overflow is a sufficiently likely event, spillways are considered a routine feature of operating TSFs. This spillway release is not a worst-case spill, in that it does not involve overfilling of the TSF with wastewaters that would be diverted to the TSF during a WWTP shutdown or failure. It is, however, a severe case.

For this spillway release analysis, we assume that TSF 1 has reached its maximum interior area of 14.2 km^2. A spillway constructed in or near the dam on the north side of the TSF would discharge towards the North Fork Koktuli River. This spillway may be either a temporary construction spillway for emergency releases or the permanent spillway. We assume that the pond within the TSF has reached its maximum safe operating level for the current dam height and that any additional precipitation requires the release of a volume of water equal to the precipitation volume. We further assume that the volume of water released exceeds the capacity of the WWTP and the conveyance mechanisms to transfer water from the TSF to the mine pit or other on-site locations, resulting in all released water discharging directly into the stream with no treatment.

8.1.4 Post-Closure Wastewater Sources

The post-closure period includes two distinct phases with respect to water management (Section 6.3.4). The first phase would be from the time the mine ceased operations until the mine pit was effectively full of water. The second phase would be after the mine pit filled until treatment was no longer necessary. During both phases, the quality of the water captured at the mine site would be substantially better than the water captured during mine operations. During operations, leachate from the PAG waste piles would account for between 80% and 94% of the total copper load in the captured water, depending on the scenario. Since the mine scenarios specify that all of the PAG waste rock would be processed by the close of operations and the PAG areas rehabilitated during site closure, the remaining flows would carry a much lower concentration of copper. The expected reduction in copper concentration in the loading to the WWTP would be greater than 90%, with substantial reductions also expected for other metals.

During pit filling, the mine operator would potentially need to treat water captured from the surface of the TSFs, captured leachate from the TSFs, captured leachate from the NAG waste rock piles outside the drawdown zone, and runoff from remaining facility areas that support the ongoing water treatment. Based on our drawdown model, the drawdown zone would not begin to shrink until pit water level was within about 100 m of its final level. As the remainder of the pit filled, the drawdown zone would shrink until the pit reached its final level. If water in the pit required treatment, the final pit level would be maintained below the level that allowed natural outflow by pumping water to the WWTP. We assume that this drawdown would result in drainage toward the pit for about 100 m beyond the pit perimeter. If or when the pit water and other sources met the discharge criteria, all flows could be discharged without treatment and the pit water level would be allowed to rise until natural discharge was established at the low point of its perimeter.

Because post-closure water quality is expected to be better than water quality during operation, the assessment does not model or evaluate water quality during this period. In addition, post-closure conditions are much more uncertain than conditions during mine operations, so it is more difficult to defend a particular set of conditions and assumptions. It is important to note, however, that although post-closure water treatment failures would be less consequential, they also would be less likely to be promptly detected and corrected. In addition, because site hydrology and chemistry would change over time, particularly as the pit filled, treatment requirements would change and responses might be slow.

The pit lake is a novel feature of the post-closure period, and, because it has been a subject of stakeholder and reviewer concern, it requires more specific consideration. After closure, the time required for the mine pit to fill with water would range from approximately 20 years (in the Pebble 0.25 scenario) to more than 200 years (in the Pebble 6.5 scenario). Eventually, the pit water would be a source of leached minerals to streams, if it were not collected and treated. Precipitation on the pit walls, groundwater entering the pit, and water collected in the pit would dissolve metals and anions from the rock walls and any waste rock returned to the pit, resulting in leachate.

Leachate composition would be approximated by some mixture of the waste rock test leachates (Section 8.1.1.2), with some dilution by ambient water. These tests were run in oxidizing conditions, so they maximize leaching rates. Although 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, oxygen levels are expected to be lower in the pit than in the leachate tests. Flow of waste rock leachate to the pit after closure would contribute to the mixture in the pit. However, as the drawdown zone shrinks, most waste rock would be outside of the drawdown zone and much would be downgradient of the pit, so its leachate would flow away from the pit. Pit water composition cannot be predicted with any confidence, but some degree of leaching is inevitable. The experience with closed pit mines is quite variable, but some mine pit lakes (e.g., the Berkeley Pit in Montana) are acidic and have high metal concentrations. Water flowing out of the full pit would be expected to flow to Upper Talarik Creek, where it would mix with waste rock leachate and water diverted from upstream.

In sum, failure to collect and treat waters from the waste rock piles, TSFs, or mine pit could expose biota in the streams draining the post-closure mine site to contaminated water. There is little information on failure rates for post-closure wastewater management at mines. If the closure occurs as described in this assessment, toxic effects could occur but they are unlikely to be severe. However, premature closures of mines do occur and such closures are likely to leave acid-generating 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 actually carry out those functions in perpetuity.

8.1.5 Probability of Contaminant Releases

Water collection and treatment failures are a common feature of mines. A review of the 14 porphyry copper mines that have operated for at least 5 years in the United States found that all but one (93%) had experienced reportable aqueous releases (the definition of a reportable release is determined by local regulations and differs among mines), with the number of events ranging from three to 54 (Earthworks 2012). Mine water releases range from chronic releases of uncaptured leachate to acute events caused by equipment malfunctions, heavy rains, or power failures. The USEPA has observed that some operators continue to discharge when they know that treatment is ineffective and not meeting standards. Hence, the record of analogous mines indicates that releases of water contaminated beyond permit limits would be likely over the life of any mine at the Pebble deposit.

The probability of the specific WWTP failure analyzed here cannot be estimated. It is improbable in that it requires that wastewater not be treated and not be diverted to storage. However, it is plausible that such an event would result from equipment failures, inadequate storage or human errors. It is more likely that a partial failure (e.g., incomplete treatment) would occur, but any one of the innumerable incomplete treatment scenarios is also unlikely. Hence, the WWTP failure scenario analyzed here provides a reasonable upper bound.

8.2 Chemical Contaminants

8.2.1 Exposure

8.2.1.1 Effluent Dilution and Transport

Under the mine scenarios, treated wastewater discharges would be divided between the South and North Fork Koktuli Rivers. South Fork Koktuli River flows include interbasin transfer to Upper Talarik Creek. Tailings water leakage and any uncontrolled leachate from the NAG rockfill dams would discharge to the South Fork Koktuli River (except in the Pebble 0.25 scenario, in which no tailings would be placed in that watershed) and the North Fork Koktuli River from TSF 1 and TSF 3 and to the South Fork Koktuli River from TSF 2. Leachate from the waste rock piles that is not captured and treated would flow to Upper Talarik Creek (except in the Pebble 0.25 scenario, in which no waste rocks would be placed in that watershed) and the South Fork Koktuli River. NAG waste rock leachate would be the only direct source of wastewater to Upper Talarik Creek during routine operations in the Pebble 2.0 and Pebble 6.5 scenarios, and no wastewater would directly enter Upper Talarik Creek in the Pebble 0.25 scenario.

WWTP effluents would be released at the surface, entering receiving waters as a plume and gradually being diluted. Input of contaminated groundwater from waste rock or tailings leachates would be introduced via upwelling through cobble and gravel substrates (i.e., via hyporheic input). In either case, a gradient would occur between full-strength effluents and fully-mixed ambient waters.

Fully-mixed ambient concentrations for each scenario are calculated by diluting the estimated discharge (i.e., contributing loads) in the background receiving waters using ambient flows and concentrations from Pebble Limited Partnership (PLP) (2011), after adjusting baseflows for the reductions in watershed areas due to the mine footprints (Tables 8-1 through 8-3; note that constituent flows at a gage are less than total flows because mine-related flows from upstream are carried forward in the model). Concentrations of contaminants of concern in wastewater discharges, waste rock leachates, and tailings leachates are presented in Table 8-9. Discharge flow rates are based on the water balances described in Section 6.2.2, and include reduced streamflows due to water use in the mine scenarios and interbasin transfers. Contaminant flows were blended with adjusted ambient water flows and tracked downstream from one stream gage to the next.

Working from the upstream-most point in each mine scenario watershed, ambient contaminant mass flows were added to discharge contaminant mass flows and divided by total flow at each stream gage to determine the diluted concentration. Moving to the next downstream stream gage the process was repeated, each time adding the mining process flows at their expected concentrations, assuming that background concentrations at each stream gage would be capturing all concentration inputs other than mining inputs. This implies that mining processes cause no other degradation or metal contributions through other mechanisms, such as surface erosion or mobilization of metals from in situ minerals by acidic leachates.

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 Pebble deposit area has been extensively characterized (PLP 2011, Zamzow 2011). Streams in the mine scenario watersheds are neutral to slightly acidic with low conductivity, hardness, dissolved solids, suspended solids, and dissolved organic carbon (DOC) (Table 8-10). In this respect, they are characteristic of undisturbed streams. However, as would be expected for a metalliferous site, levels of sulfate and some metals (copper, molybdenum, nickel, and zinc) are elevated, particularly in the South Fork Koktuli River. PLP (2011) 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). Therefore, the stream reaches with significantly elevated copper concentrations would largely be destroyed by the mine footprints and by water diversions.

The chemical fate of metals in receiving streams may be complex. Aluminum, iron, and manganese are commonly precipitated in streams receiving acid mine drainage, diminishing or destroying stream habitats with deposited flocs but also reducing the aqueous toxicity of those metals. Acidic leachates would form from the PAG waste rocks, but concentrations of precipitating metals in PAG waste rock leachates are not particularly high (Tables 8-7 and 8-8). Other metals do not precipitate to a significant degree but may have reduced bioavailability due to receiving water chemistry. That issue is largely dealt with by use of the BLM for copper, which includes a metal speciation submodel.

Table 8-10. Means and coefficients of variation for background surface water characteristics of the mine scenario watersheds, 2004–2008.

Analyte	South Fork Koktuli River	North Fork Koktuli River	Upper Talarik Creek
TDS (mg/L)	44 (0.41)	37 (0.035)	51.2 (0.37)
pH (field)	7.0 (0.045)	6.74 (0.10)	6.99 (0.091)
DO (mg/L)	10.2 (0.21)	10.2 (0.2)	10.5 (0.19)
Temperature (°C)	4.77 (1.02)	4.39 (1.12)	4.04 (0.98)
Specific Conductivity (μS/cm)	ND	ND	73.4 (0.34)
TSS (mg/L)	2.21 (2.13)	1.39 (1.9)	2.52 (1.58)
Ca (mg/L)	6.34 (0.42)	5.09 (0.35)	8.77 (0.3)
Mg (mg/L)	1.41 (0.56)	1.32 (0.44)	2.12 (0.46)
Na (mg/L)	2.35 (0.35)	2.38 (0.23)	2.82 (0.32)
K (mg/L)	0.38 (0.47)	0.41 (0.39)	0.44 (0.43)
Alkalinity (mg/L)	17.4 (0.43)	20.5 (0.38)	31.8 (0.34)
SO_4 (mg/L)	8.78 (0.87)	2.26 (0.56)	5.48 (1.43)
Cl (mg/L)	0.69 (0.26)	0.66 (0.25)	0.7 (0.29)
F (mg/L)	ND	ND	ND
Hardness (mg/L)	19.6 (0.47)	14.4 (0.36)	26.5 (0.42)
Al (μg/L)	11 (0.68)	13 (0.82)	13 (1.1)
As (μg/L)	ND	ND	ND
Ba (μg/L)	4.1 (0.48)	3.1 (0.35)	5.5 (0.41)
Cd (μg/L)	ND	ND	ND
Cu (μg/L)	1.3 (0.88)	0.39 (0.84)	0.42 (0.89)
Fe (μg/L)	120 (1.01)	110 (0.68)	110 (0.83)
Mn (μg/L)	20 (1.14)	10 (1.67)	21 (1.09)
Mo (μg/L)	0.66 (0.98)	0.19 (1.2)	0.2 (0.51)
Ni (μg/L)	0.41 (0.61)	0.30 (1.14)	0.63 (1.04)
Pb (μg/L)	ND	ND	ND
Zn (μg/L)	2.7 (1.02)	1.8 (0.64)	2.0 (1.09)
CN (μg/L)	ND	ND	ND
DOC (mg/L)	1.36 (0.62)	1.5 (0.51)	1.57 (0.82)

Notes:
Filtered concentrations are used for hardness and trace elements.
ND = analytes detected in less than half of samples.
Source: PLP 2011.

8.2.1.2 Biological Exposures

Aquatic biota would be directly exposed to contaminants in discharged waters. Fish embryos and larvae (e.g., salmon eggs and alevins) would be exposed to benthic pore water, which would be provided by groundwater in areas of upwelling and otherwise by surface water. In this chapter we assume that sediments would not be contaminated by tailings, waste rock, or other mine-derived particles. Juvenile fish (e.g., salmon fry and parr) would be exposed to surface water. Adult resident salmonids would also be exposed to surface water, but unlike the early life stages, they would occur in the smallest streams only during spawning. Adult anadromous salmonids would have brief exposures to waters near the site. Aquatic insects would be exposed in all juvenile stages, which constitute most of their life cycles. They would be exposed to benthic pore water or surface water depending on their habits.

8.2.2 Exposure-Response

We screened potential contaminants against ecotoxicological benchmarks to identify the most potentially toxic constituents and indicated the degree of treatment that would be required and the types of effects that might occur due to mining emissions. Criteria and equivalent screening benchmarks are presented in Tables 8-4 through 8-8, and the sources of non-criteria screening benchmarks are presented in Table 6-10. Benchmarks were derived from the literature to be as similar to criteria as possible, given the available data (Section 6.4.2.3). Criteria for many of the metals are functions of hardness, and copper criteria are a function of multiple water properties. For those metals, criteria are calculated for each leachate based on its chemistry and for each receiving stream based on its background chemistry.

8.2.2.1 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's effects are likely to mitigate, to some extent, effects from co-occurring metals. For these reasons, we focus on copper criteria, standards, and toxicity in this assessment.

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 Alaska's acute value (CMC) and chronic value (CCC), in micrograms per liter and based on hardness in milligrams per liter, are:

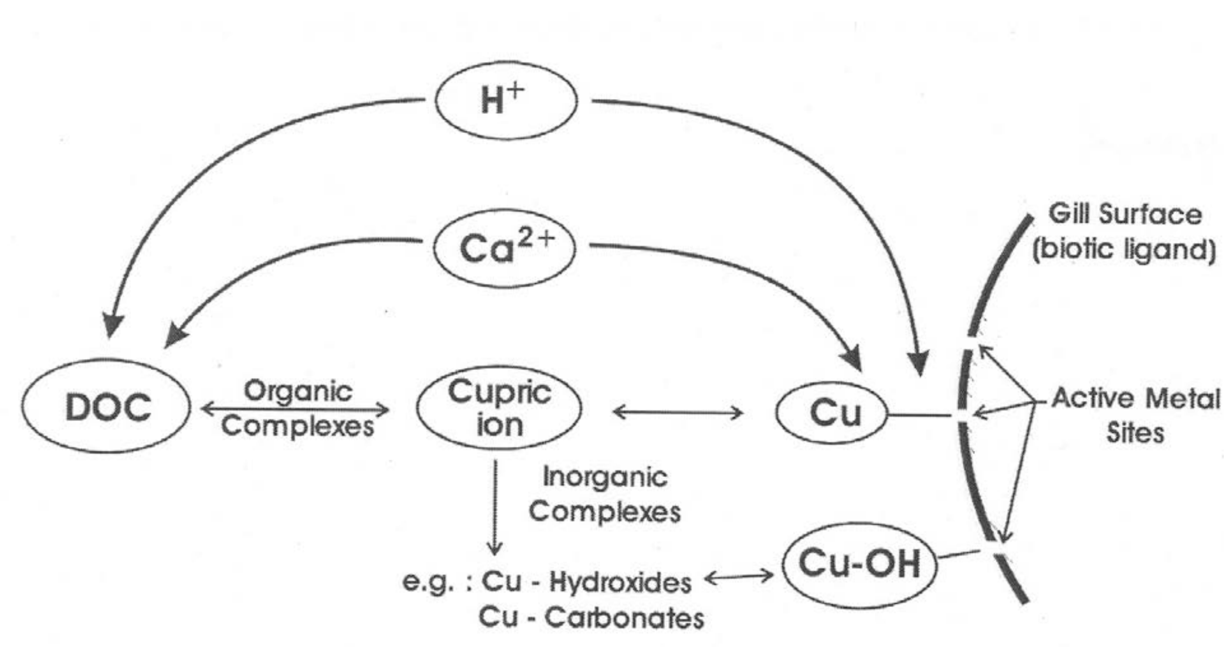
$$\text{Copper acute criterion} = e^{0.9422(\ln \text{hardness}) - 1.700} \times 0.96$$

$$\text{Copper chronic criterion} = e^{0.8545(\ln \text{hardness}) - 1.702} \times 0.96$$

Note that the formulae are similar and yield similar values—that is, when copper causes toxic effects, the effects occur relatively quickly. At 20 mg/L hardness (i.e., in soft water typical of the Bristol Bay region), Alaska's 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 (hereafter, criteria) for copper (USEPA 2007). These criteria are calculated using the 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 8-2). Copper competes for ligands with calcium and other cations. The competitive binding model for the biotic ligand requires input from a metal speciation submodel and user-input values for basic water chemistry parameters (i.e., pH, temperature, DOC, humic acid, calcium, magnesium, sodium, potassium, sulfate, sulfide, chloride, and alkalinity). 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 the binding of copper by dissolved organic matter. The BLM is freely available from USEPA (http://water.epa.gov) and from the model's developer Hydroqual Inc. (http://www.hydroqual.com/blm).

Figure 8-2. Processes involved in copper uptake as defined in the biotic ligand model (USEPA



The results of applying the BLM to mean water chemistries of the South and North Fork Koktuli Rivers and Upper Talarik Creek are presented in Table 8-11. These values are lower than Alaska's hardness-based values and the variance among streams is potentially significant.

Table 8-11. Results of applying the biotic ligand model to mean water chemistries in the mine scenario watersheds (Table 8-10) to derive acute (CMC) and chronic (CCC) copper criteria specific to receiving waters. Values are in μg/L.

Stream	CMC	CCC
South Fork Koktuli River	2.4	1.5
North Fork Koktuli River	1.7	1.1
Upper Talarik Creek	2.7	1.7

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

The results of applying the BLM to mean chemistries of the waste rock leachates are presented in Table 8-12. Model runs used mean water chemistries from the PLP tests (Appendix H). These effluent-specific values differ from each other and from the values for ambient waters due to differences in water chemistries.

Table 8-12. Results of applying the biotic ligand model to mean water chemistries in waste rock leachates (Appendix H) to derive effluent-specific acute (CMC) and chronic (CCC) copper criteria. Values are in μg/L.

Leachates	CMC	CCC
Pebble Tertiary	2.5	1.6
Pebble West pre-Tertiary	0.88	0.55
Pebble East pre-Tertiary	0.043	0.027

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

For both the background waters and the leachates, temperature was set to the mean from three streams on the site (4.5°C) (PLP 2011). For the leachates, DOC was set to 1 mg/L (a reasonable value given the absence of DOC in the leachate, which would mix with ambient water containing approximately 1.5 mg/L of DOC) and humic acid was set to the default value (10% of DOC).

Both the state standards and national criteria for copper are derived from the 5th centile of the aquatic genera sensitivity distribution. 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 fishes in 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 in acute tests (Chapman 1975, 1978). Acute and chronic values for rainbow trout can be derived for background water quality using the BLM (Table 8-13). BLM-estimated acute values could also be calculated for three cladoceran species in the three streams draining the mine scenario footprints: *Daphnia magna* (8.68 to 13.02 μg/L), *Daphnia pulex* (4.28 to 6.63 μg/L), and *Ceriodaphnia dubia* (5.99 to 9.13 μg/L). These zooplankters are less directly relevant to the receiving streams, but they are relevant to ponds and Iliamna Lake and they illustrate the sensitivity of aquatic arthropods to copper.

Table 8-13. Site-specific acute and chronic copper toxicity values for rainbow trout, derived by applying the biotic ligand model to mean water chemistries in the mine scenario watersheds (Table 8-10).

Stream	Acute Toxicity (LC_{50} in µg/L)	Chronic Toxicity (CV in µg/L)
South Fork Koktuli River	63	22
North Fork Koktuli River	59	21
Upper Talarik Creek	75	26

Notes:
LC_{50} = median lethal concentration; CV = chronic value, calculated using the species-specific acute to chronic ratio of 2.88.
Biotic ligand model source: USEPA 2007.

A test conducted with juvenile Chinook salmon showed greater sensitivity to subchronic copper exposures than is suggested by Table 8-13 (Mebane and Arthaud 2010). After 120 days, reductions in both length (5.6%) and weight (21%) were observed in salmon exposed to 7.4 µg/L copper. A BLM could not be developed for the salmon, but the test water chemistry was relevant to the Pebble site (hardness = 25.4 mg/L, pH = 7.32, DOC = 1.2 mg/L). Mebane and Arthaud (2010) applied these growth effects to a population demographic model for a threatened Chinook salmon population spawning in Idaho. They found that the observed reductions in individual growth would reduce population growth due to increased mortality of smaller out-migrating fish (Mebane and Arthaud 2010). However, it should be noted that the sensitivity of juvenile Chinook salmon was still less than that of sensitive invertebrates (USEPA 2007).

Alternative Copper Endpoints for Fish

Copper standards and criteria are based on conventional test endpoints of survival, growth, and reproduction. However, research has shown that salmonid olfactory systems are affected at low copper concentrations (Hecht et al. 2007). Salmon use olfaction to find their spawning streams, 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. 2011a).

Meyer and Adams (2010) applied the hardness-corrected criteria and the BLM to data from multiple laboratory tests for olfactory effects. They found that the BLM accounted well for variance among tests, and that BLM-based criteria were consistently protective of olfactory effects in the test systems. In contrast, hardness-corrected criteria were not consistently protective (Meyer and Adams 2010). DeForest et al. (2011b) extended those results by applying the same models to 133 ambient waters in the western United States (including Alaska) that 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 that the BLM-based chronic criteria were protective of this chronic effect in 100% of waters. Even the acute BLM-based criteria were protective of this chronic effect in 98% of waters, since 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.

Although the criteria are protective screening values for sensory effects of copper on salmonids, it is necessary to consider potential effects when criteria are exceeded. Meyer and Adams (2010) adapted the BLM to sensory data and derived IC_{20}:BLM factors that can be used to convert site-specific criteria into estimates of the copper concentration at which 20% of rainbow trout avoid the contaminated water or at which they experience 20% inhibition of their olfactory senses (Table 8-14). These values bracket the threshold for growth effects in juvenile Chinook salmon (7.4 μg/L copper) described above.

Table 8-14. Site-specific benchmarks for sensory effects in rainbow trout. Values are derived by applying IC_{20}:BLM ratios from Meyer and Adams (2010) to the acute values in Table 8-8.

Stream	Avoidance (IC_{20} in μg/L)	Sensory Inhibition (IC_{20} in μg/L)
South Fork Koktuli River	5.2	26
North Fork Koktuli River	3.8	19
Upper Talarik Creek	5.9	30

IC_{20} = 20% inhibitory concentration; BLM = biotic ligand model.

Avoidance cannot prevent severe toxic effects of copper on salmonid fish, unless they encounter low concentrations before high concentrations (e.g., if they are swimming up a concentration gradient). At concentrations sufficient to cause mortality or reproductive failure, copper damages the sensory organs and avoidance does not occur (Hansen et al. 1999).

Neurobehavioral effects may be responsible for findings that low-level exposures to copper reduce out-migration success. Lorz and McPherson (1977) pre-exposed coho salmon to 0, 5, 10, 20, or 30 μg/L of copper for between 6 and 165 days and released them into a coastal Oregon stream on four dates. Percent successful out-migration was reduced relative to controls by copper exposure at all concentrations and durations, with greater effects observed at higher exposures.

Dietary Copper Exposure-Response for Fish

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 indicated that macroinvertebrates can accumulate metals at levels that result in toxicity and reduced growth in fish that consume them (Farag et al. 1994, Woodward et al. 1994, Woodward et al. 1995, Farag et al. 1999). Although those effects were shown to be most correlated with exposure to copper, subsequent studies suggest that the effects were primarily caused by co-occurring arsenic (Hansen et al. 2004, Erickson et al. 2010).

Participants in a recent Pellston Workshop (a workshop series convened by the Society for Environmental Toxicology and Chemistry to examine environmental toxicological issues) 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 of 646 μg/g for rainbow trout. The resulting factor is 0.95, so the adjustment is small. 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 ambient water quality criteria and state standards, due to the relative insensitivity of fish. This result applies to aqueous-only exposures (i.e., it does not include contaminated sediment or allochthonous material). Because this dietary exposure factor has little influence on risks to fish from direct aqueous exposure and adds another source of uncertainty, it is not applied to the risk estimates in this chapter. However, dietary exposure of fish to copper in sediments, where direct aqueous exposures of post-larval fish may be minor, is considered in Chapter 9.

Copper and Algal Production

Although copper sulfate is used as an algicide, a relatively small amount of high-quality toxicity data is available for algae or other aquatic plants (USEPA 2007, European Copper Institute 2008). Freshwater algae and aquatic vascular plants are generally less sensitive than invertebrates or fish, with No Observed Effects Concentrations for growth ranging from 15.7 to 510.2 μg/L in high-quality data sources (European Copper Institute 2008); these values are for dissolved copper but are not corrected for water chemistry. However, a few whole ecosystem studies suggest that algal production may be reduced at lower copper concentrations and this may contribute to the sensitivity of insects to copper in the field (Hedtke 1984, Leland and Carter 1984, 1985, Leland et al. 1989, Brix et al. 2011). The effects of copper are complex and involve competition among algal taxa that vary in their sensitivities and changes in grazing intensity, so in some systems algal production is relatively resistant (Le Jeune et al. 2006, Roussel et al. 2007). It appears that criteria based on toxicity to invertebrates would also be protective of algal production, but the data are unclear. Risks to algal production from copper are not considered further, because the uncertainties are so large relative to risks to fish and invertebrates.

Copper 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 scenarios presume 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 predominantly 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, although 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). This result was supported by a study funded by Rio Tinto, which concluded that “aquatic insects are indeed very sensitive to some metals and in some cases may not be protected by existing WQC [water quality criteria]” (Brix et al. 2011). 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 for this assessment, and the hypothesized mechanisms for the greater sensitivity of field communities are supported by evidence from laboratory and field experiments (Brix et al. 2011). It also must be noted that the occurrence of biological effects below criteria 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).

Copper Exposure-Response Uncertainties

The copper criteria are based on a large body of data and a mechanistic model of exposure and effects. Hence, at least with respect to acute toxicity, 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 that 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 mayfly, stonefly, caddisfly, or other invertebrate species in the streams draining the mine footprints are more sensitive than 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-to-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 and some other metals in chronic relative to acute exposures and may not be protected by current criteria. Thus, the protectiveness of the chronic criterion is more uncertain than that of the acute criterion.

Based on the literature cited above, resolution of this uncertainty through additional research and testing is likely to lower the chronic criterion. Therefore, this uncertainty biases downward the estimated length of streams experiencing toxic effects and could change our conclusions with respect to relatively low toxicity materials such as tailings and NAG waste rock. The naturally elevated copper concentrations in the highest reaches of some of the South Fork Koktuli River tributaries further complicate the assessment of copper toxicity. Sensitive taxa may not occur in those reaches. Alternatively, the biota in those reaches may be somewhat resistant to copper additions, although studies in the Colorado metal belt (see previous subsection) suggest that significant adaptation does not occur. However, in the mine scenarios, the reaches with the highest natural copper levels would be destroyed and effluents and leachates would enter downstream or in other tributaries or watersheds, so this source of uncertainty is largely moot.

Another source of uncertainty is the assumption that the State of Alaska would adopt the national copper criterion as a state standard or apply it on a site-specific basis to any mine in the Bristol Bay watershed. If the state retains and applies the current standard, the effects of copper on salmon and other aquatic organisms in permitted effluents would be greater by a factor of approximately 1.3 to 2.0, based on differences among receiving streams (Table 8-11).

8.2.2.2 Other Metals

Chronic national ambient water quality criteria, state standards, or equivalent benchmarks were used to screen the constituents of tailings, waste rock, and product concentrate leachates (Section 8.1.1). Those that were retained in the screening were carried forward to release, transport, and dilution modeling. For hardness-dependent criteria, screening values were calculated for each receiving stream using mean hardness (Table 8-15). Finally, those potential contaminants that exceeded screening values in streams after dilution, or that were otherwise of concern, are discussed in more detail below.

Table 8-15. Hardness-dependent acute water quality criteria (CMC) and chronic water quality criteria (CCC) for the three potential receiving streams in the mine scenarios. All values are in µg/L.

Criteria	South Fork Koktuli River	North Fork Koktuli River	Upper Talarik Creek
Cd CMC	0.45	0.30	0.50
Cd CCC	0.085	0.064	0.089
Pb CMC	12	7.4	15
Pb CCC	0.46	0.29	0.58
Ni CMC	130	91	150
Ni CCC	14	10	17
Zn CMC	32	23	38
Zn CCC	32	23	38

Notes:
CMC = criterion maximum concentration; CCC = criterion continuous concentration.

Aluminum

The environmental chemistry and resulting toxicity of aluminum are complex (Gensemer and Playle 1999). Aluminum is more soluble at acidic and alkaline pHs and less soluble at circumneutral pH. It occurs in a variety of forms, including soluble complexes with common anions and humic and fulvic acids, but in most streams soluble and insoluble hydroxide compounds dominate. Free ionic aluminum is expected to be a small component of dissolved aluminum at the circumneutral pHs found in the streams draining the mine footprints, unless mine drainage acidifies them. Aluminum is most toxic in mixing zones where acidic waters mix with neutral or basic ambient waters, apparently due to precipitation at the gill surface. In general, fish are more sensitive to aluminum than invertebrates.

Cadmium

Cadmium is an uncommon but highly toxic divalent metal (Mebane 2010). A series of rainbow trout acute median lethal concentration (LC_{50}) values for cadmium, at hardness values from 7 to 32 mg/L, ranged from 0.34 to 1.3 µg/L (Mebane 2010, Mebane et al. 2012). A 53-day early-life-stage test of rainbow trout (at 21 mg/L hardness) gave a chronic value for survival and growth of 0.88 µg/L, but the test was interrupted prior to completion due to quality control issues (Mebane et al. 2008). A later test in the same series (but without those quality control issues and at 29 mg/L hardness) gave a higher rainbow trout chronic value of 1.6 µg/L. Acute tests with mayflies, stoneflies, and caddisflies all resulted in values that were much higher than the trout values (Mebane et al. 2012). The tests by Mebane et al. (2008, 2012) were conducted for the State of Idaho to support the derivation of site-specific criteria for the Coeur d'Alene River. BLM-derived acute values for *Ceriodaphnia dubia* were 37 to 51 µg/L for the three streams draining the mine footprints. This is consistent with the relative insensitivity of invertebrates to acute lethality. Although these and other tests in the literature show fish to be more sensitive to cadmium than invertebrates in acute exposures, invertebrates were more sensitive in chronic exposures (Mebane 2010). In particular, mortality of the amphipod *Hyalella azteca* increased at 0.16 µg/L cadmium at relevant hardness (17 mg/L) (Mebane 2010).

Cobalt

Current studies of the aquatic toxicity of cobalt can be found in a recent review (Environment Canada and Health Canada 2011). Acutely lethal concentrations range from 89 to 585,800 µg/L. Chronically toxic concentrations for invertebrates range from 2.9 to 155 µg/L (with the exception of a 1972 4-day test for rotifers, which resulted in 59,000 µg/L). Only three fish species have been tested for chronic effects, yielding values of 340 to 2,171 µg/L; the least sensitive of these three species was rainbow trout. In experimental studies, Chinook salmon avoided waters with cobalt concentrations of 24 µg/L but rainbow trout were less sensitive, avoiding concentrations of 188 µg/L (Hansen et al. 1999). It is expected that the same water quality parameters that modify copper toxicity also affect cobalt toxicity, but existing data are insufficient to perform adjustments.

Lead

Lead is a divalent metal with national criteria and state standards based on water hardness (USEPA 1986, Eisler 2000). A BLM is available that estimates acute LC_{50} values for fathead minnows in the South and North Fork Koktuli Rivers as 382 and 383 µg/L, respectively. In comparison, a rainbow trout test at hardness similar to that in the South and North Fork Koktuli Rivers (20 mg/L) resulted in an LC_{50} of 120 µg/L; the closely related cutthroat trout produced an LC_{50} as low as 47 µg/L at a hardness of 11 mg/L (Mebane et al. 2012). Tests at similar hardness levels for mayflies, stoneflies, caddisflies, and chironomid midges gave higher LC_{50s} (253 to more than 1,255 µg/L) (Mebane et al. 2012). This indicates that, for acute lethality, trout species are more sensitive to lead than aquatic insect larvae, which is consistent with BLM-derived acute values of 523 to 748 µg/L for *Daphnia magna*. Chronic tests gave values for reduced rainbow trout weight and length of 36.0 and 12.1 µg/L at 21 and 29 mg/L hardness, respectively, and for the midge *Chironomus tentans* of 65.4 µg/L at 32 mg/L hardness (Mebane et al.

2008). Note that we use tests performed for the State of Idaho (Mebane et al. 2008, 2012) for cadmium, lead, and zinc, because they are high-quality tests that use species and water chemistries relevant to the Bristol Bay watershed.

Manganese

The toxicity of manganese is strongly related to water hardness. Acutely lethal concentrations for manganese in soft water range from 0.8 to 4.83 mg/L for invertebrates and 2.4 to 3,350 mg/L for fish. The most sensitive acutely tested fish was coho salmon. In four soft-water tests of rainbow and brown trout, chronic values ranged from 0.79 to 14.6 mg/L. More details can be found in recent reviews (Reimer 1999, IPCS 2004).

Selenium

Selenium is a bioaccumulative and moderately biomagnifying element. Dissolved oxyanions of selenate (Se^{+4}) and selenite (Se^{+6}) are taken up by microbes, algae, and plants and converted to organic forms. In streams, periphyton growing on rocks and woody debris are the primary community that performs this conversion and conversion rates are relatively low. Selenium causes deformities and death in fish embryos and larvae, which are exposed to selenium accumulated by their mothers. Therefore, potential selenium toxicity is of concern for resident but not anadromous fish. Effects of selenium on salmonids have been studied below mines in British Columbia. For example, cutthroat trout embryos from a pond with selenium concentrations of 93 µg/L at a coal mine in British Columbia showed effects ranging from larval deformities to mortality (Rudolph et al. 2008). The probability of mortality was correlated with selenium concentrations in the embryos. Invertebrates are less sensitive to selenium than fish.

The complex dynamics of selenium and its various forms have led to complex water quality criteria (USEPA 2004). The acute national criteria, based on the proportions of selenite and selenate, are185.9 µg/L selenite and 12.82 µg/L selenate. The chronic criterion is 5.0 µg/L total selenium. However, because the transformations and bioaccumulative processes are so complex, a chronic criterion for fish tissue concentrations (7.91 µg/g whole body dry weight) has been proposed based on juvenile bluegill sunfish mortality. The genus mean chronic value for rainbow trout is a little higher (9.32 µg/g dry weight). These tissue-based values are believed to be more accurate than benchmarks based on water concentrations. However, implementing the criterion or using the dietary toxicity test and field data that were used in its derivation would require a model of selenium bioaccumulation that is applicable to streams and lakes in the Bristol Bay watershed. No such model is currently available.

Zinc

Zinc, like copper, is a divalent metal and trace nutrient that is a common aquatic toxicant. The national criteria and state standard are based on water hardness (USEPA 1987), but a BLM is available that provides more accurate predictions of acute toxicity, at least for some test species (DeForest and Van Genderen 2012). The BLM-based LC_{50} estimates for rainbow trout in the South and North Fork Koktuli Rivers are 64 and 63 µg/L, respectively. In comparison, results of a series of 17 rainbow trout LC_{50} tests (at hardnesses of 7 to 71 mg/L) ranged from 20 to 289 µg/L (Mebane et al. 2012). Acute tests at

14 mg/L hardness for two mayfly species and a caddisfly species resulted in values greater than 2,926 μg/L, and a stonefly species test resulted in values greater than 1,526 μg/L (Mebane et al. 2012). These results suggest that an endpoint fish species is considerably more sensitive to zinc than relevant stream invertebrates in acute exposures. BLM-derived acute values for *Daphnia magna* were 407 to 502 μg/L for the three receiving streams, which is consistent with the relative insensitivity of invertebrates. The chronic value (20% effective concentration [EC_{20}] for survival) from a 69-day, early-life-stage test of rainbow trout in 21 mg/L hardness water was 147 μg/L (Mebane et al. 2012).

8.2.2.3 Total Dissolved Solids

The Alaskan Water Quality Standard for Growth and Propagation of Fish, Shellfish, Other Aquatic Life and Wildlife states: "TDS may not exceed 1,000 mg/L. A concentration of TDS may not be present in water if that concentration causes or reasonably could be expected to cause an adverse effect on aquatic life" (ADEC 2011). Meeting the state standard for TDS proved difficult at the Red Dog zinc and lead mine (USEPA 1998, 2008). Laboratory tests of synthetic TDS for effluents from Red Dog and Kensington Mines caused no statistically significant effects on rainbow trout embryo viability or fry survival or weight, but did show statistically significant effects on chironomid larvae at 2,089 mg/L (Red Dog) and 1,750 mg/L (Kensington) (Chapman et al. 2000). However, the toxicity of TDS depends on the specific mixture composition, and chironomids are relatively tolerant of major ion mixtures (USEPA 2011). Also, the rainbow trout tests did not include the State of Alaska's endpoint of concern, egg fertilization.

8.2.2.4 Whole Leachates and Effluents

Metals and other aqueous contaminants have combined toxic effects that may be concentration additive, effects additive, or more or less than additive. The assumption of concentration additivity is considered to provide the best general approximation of combined metals effects. If, as in this assessment, the number of metals potentially discharged is large, less than and more than additive interactions may roughly average out. However, pairwise laboratory tests of defined metal mixtures indicate the complexity of potential interactions. For example, Chinook salmon avoided a mixture of 0.9 μg/L cobalt and 1.0 μg/L copper, which suggests that cobalt has no effect on copper avoidance at low levels (Hansen et al. 1999). However, at overtly toxic copper levels (43 μg/L) cobalt did increase avoidance (Hansen et al. 1999).

As discussed above with respect to copper (Section 8.2.2.1), field studies of streams contaminated by copper and other metals indicate that laboratory-based criteria are not fully protective of aquatic communities. Hence, the screening of metal mixtures by applying an additive model to criteria and equivalent benchmarks probably does not overestimate effects on aquatic communities.

8.2.2.5 Ore-Processing Chemicals

Of the proposed ore-processing chemicals, sodium ethyl xanthate is the primary contaminant of concern (Section 6.4.2.3). An assessment by Environment Australia generated a predicted no effect concentration of 1 μg/L (NICNAS 2000). Australia and New Zealand have established a trigger value of 0.05 μg/L to protect aquatic life (ANZECC 2000). However, because relatively little testing has been done, this is a "low reliability" value that "may not protect the most sensitive species." Rainbow trout appear to be relatively tolerant of sodium ethyl xanthate, with lethal concentrations ranging from 1 to 50 mg/L depending on test conditions (Fuerstenau et al. 1974, Webb et al. 1976). Other fishes had median lethal concentrations of 0.01 to 10 mg/L (emerald shiner) and 0.32 to 3.2 mg/L (fathead minnow) (NICNAS 1995). Aquatic invertebrates are represented by only *Daphnia magna*, which has a median effective concentration (EC_{50}) of 0.35 mg/L (Xu et al. 1988).

8.2.3 Risk Characterization

Risk characterization was performed in stages. First, screening was performed against mean concentrations in tailings and waste rock leachates to determine whether leachates pose a potential risk and which constituents contribute to risks. Second, contaminants of concern from the initial screening were screened against estimated ambient concentrations for routine operations and for WWTP failure. The implications of potential toxic effects are discussed in terms of their spatial distributions.

Contaminants were screened for risks to aquatic biota by comparing exposure levels to criteria or other ecotoxicological benchmarks using a risk quotient (Box 8-3). This conventional approach was used to determine which contaminants and materials are likely to be toxic (Tables 8-4 through 8-8).

BOX 8-3. USE OF RISK QUOTIENTS TO ASSESS TOXICOLOGICAL EFFECTS

A risk quotient (Q) equals the exposure level divided by an ecotoxicological benchmark. If the quotient exceeds 1, the effect implied by the benchmark is expected to occur, but with some uncertainty (see below). Quotients much larger than 1 suggest larger effects than those that define the benchmark, with greater confidence that an adverse effect would occur. Quotients much smaller than 1 suggest that even small effects are unlikely. The acute criterion, or criterion maximum concentration (CMC), estimates a concentration at which 5% of aquatic species experience some mortality among later life-stages in short-term exposures. The chronic criterion, or criterion continuous concentration (CCC), estimates a concentration at which 5% of aquatic species experience decreased survival, growth, or reproduction in longer-term exposures. The criteria, or equivalent numbers when criteria are not available, are relatively well-accepted as approximate thresholds for significant effects. Thus, these values are the ecotoxicological benchmarks used as the divisor for calculating quotients in the screening portion of this assessment.

To describe the results of screening using chronic criteria or equivalent benchmarks in a consistent manner (acute criteria and less protective benchmarks would be interpreted differently), the following scale was developed:

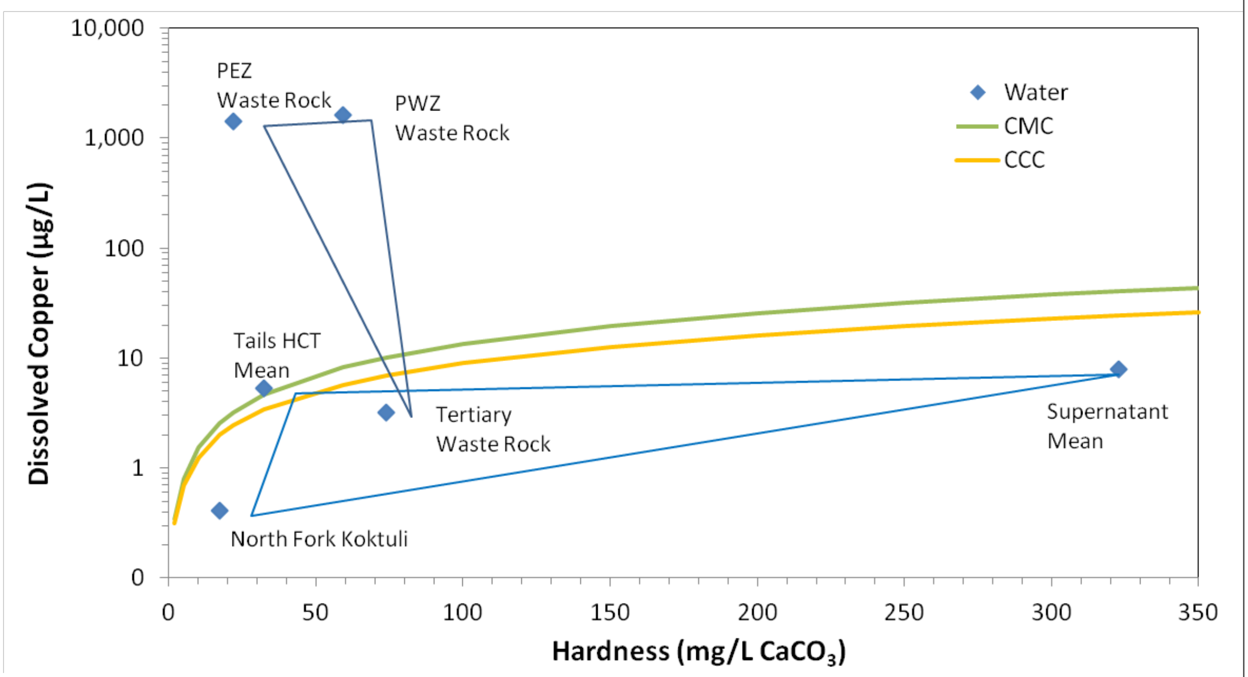
- $Q < 1$ = not overtly toxic
- $2 > Q \geq 1$ = marginally toxic
- $10 > Q \geq 2$ = moderately toxic
- $100 > Q \geq 10$ = highly toxic
- $Q \geq 100$ = extremely toxic

8.2.3.1 Screening Leachate Constituents

The results of screening inorganic leachate constituents from tailings and waste rock tests are presented in Tables 8-4 through 8-8. All have at least moderate chronic toxicity based on their estimated total toxicity (sum of chronic toxic units), and all are predicted to be acutely toxic if the BLM-based copper criterion is used instead of the state standard. In all cases, copper is the dominant source of toxicity. The acidic pre-Tertiary leachate is estimated to be extremely toxic, with copper concentrations thousands of times higher than the chronic criterion. Figure 8-3 shows the copper concentrations of leachates and ambient water in relation to state standards.

Tailings slurry would also contain processing chemicals, particularly sodium ethyl xanthate. The predicted concentration in the slurry (1.5 mg/L) is above or within the range of acute lethality for fish and well above the level for *Daphnia magna*. Therefore, the aqueous phase of the slurry delivered to the TSF would be moderately toxic due to xanthate alone.

Figure 8-3. Comparison of copper concentrations in leachates and background water to state hardness-based acute (CMC) and chronic (CCC) water quality criteria for copper. North Fork Koktuli River = background water; Tails HCT = leachate from humidity tests of tailings; Supernatant = leachate from column tests of tailings; PWZ = Pebble West pre-Tertiary; PEZ = Pebble East pre-Tertiary; CMC = criterion maximum concentration; CCC = criterion continuous concentration. 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 sources: Appendix H and PLP 2011.



8.2.3.2 Screening Contaminants in Receiving Waters

Concentrations of contaminants of concern were calculated at the gages on all three receiving streams for each mine scenario (Pebble 0.25, 2.0, and 6.5), and those concentrations were screened against chronic criteria and benchmarks. Because the Pebble 6.5 scenario resulted in the highest concentrations, screening results for this mine size are presented in Tables 8-16 and 8-17 to show which contaminants remain of concern after dilution. In the Pebble 6.5 scenario under routine operations, copper is estimated to exceed chronic water quality criteria at all stations on the South Fork Koktuli River, two of six stations on the North Fork Koktuli River, and three of seven stations on Upper Talarik Creek (Table

8-16). The pattern of exceedance is the same for the Pebble 6.5 scenario with WWTP failure, except that all stations on the North Fork Koktuli River exceed the copper criterion (Table 8-17). Cadmium and zinc also exceed chronic criteria, but at fewer stations and by much smaller magnitudes. No other metal exceeded a criterion or benchmark.

Concentrations of major ions are a particular concern at mine sites because of the leaching of large volumes of crushed rock. However, estimates of TDS, both with and without WWTP failure, are within state standards (Tables 8-16 and 8-17). Without toxicity information on the dissolved solids mixtures that would occur at the site, we must assume that the standard is protective.

The concentration of sodium ethyl xanthate was not estimated in the receiving streams. Although the aqueous phase of the tailings slurry would be toxic due to xanthate, we expect that xanthate would occur at non-toxic levels in ambient waters below TSFs due to degradation and dilution (Xu et al. 1988).

8.2.3.3 Screening Total Metal Toxicity in Receiving Waters

Table 8-18 presents the sums of quotients across the nine metals of concern (excluding selenium, which has a different mode of exposure and toxicity), for all three mine scenarios and gage locations. In addition, the sums of quotients for background water are presented. As is expected for streams draining a surficial ore body, background metal concentrations are elevated. Although total metal toxicities are estimated to be significantly higher than any individual metal, copper is responsible for most of the estimated toxicity. Therefore, copper concentrations in contributing loads and ambient waters, as well as quotients with respect to chronic criteria for the receiving waters, are presented for all three mine scenarios and gage locations in Table 8-19. The same information but with WWTP failure is presented in Table 8-20.

8.2.3.4 Screening for Severity of Effects

Tables 8-19 and 8-20 indicate that water management, assuming both routine operations and WWTP failure, results in exceedance of the BLM-based chronic copper criteria at several locations in the mine scenario watersheds. However, they do not provide an indication of the severity of effects. For that purpose, we estimated copper concentrations in individual stream reaches. The reaches are defined by flow gages and major confluences for each of the receiving streams (Table 8-21). Those fully-mixed copper concentrations were screened against a series of benchmarks of increasing severity, beginning with the national criteria, as follows.

- **Invertebrate chronic (IC).** The BLM-derived chronic ambient water quality criterion (Table 8-11), based on toxicity to sensitive aquatic invertebrates in extended exposures. It implies reduced survival, growth, or reproduction of copper-sensitive invertebrates.
- **Invertebrate acute (IA).** The BLM-derived acute ambient water quality criterion (Table 8-11), based on lethality to sensitive aquatic invertebrates in short-term exposures. It implies greatly reduced survival of copper-sensitive invertebrates.

- **Fish avoidance (FA).** The BLM-derived concentration at which 20% of rainbow trout avoid the contaminated water (Table 8-14). It implies loss of habitat due to aversion.
- **Fish sensory (FS).** The BLM-derived concentration at which the olfactory sensitivity of rainbow trout is reduced by 20% (Table 8-14). It implies an inability to identify natal streams, reduced predator avoidance, and other behavioral effects.
- **Fish reproduction (FR).** The BLM-derived chronic value for rainbow trout, the concentration at which their fecundity or the survival and growth of their larvae are reduced (Table 8-13). It implies partial or complete reproductive failure of salmonids.
- **Fish kill (FK).** The BLM-derived rainbow trout LC_{50}, the concentration at which half of adults and juveniles are killed in short-term exposures (Table 8-13). It implies a fish kill and, in the long term, local extirpation of fish populations.

These copper benchmarks were applied to streams reaches (Table 8-21) rather than the point values in the prior screening assessment. The combinations of stream reaches and mine scenarios at which these copper benchmarks are exceeded are shown in Tables 8-22 and 8-23. The range of effect severity extends from no overt effects expected (-) through the full range of effects up to numerous dead post-larval salmonids (IC/IA/FA/FS/FR/FK).

Table 8-16. Estimated concentrations of contaminants of concern and associated risk quotients for the Pebble 6.5 scenario, assuming routine operations, at locations in the mine scenario watersheds. See Box 8-3 for a description of how risk quotients were calculated.

Stream and Gage	Copper		Aluminum		Cadmium		Cobalt		Manganese		Nickel		Lead		Selenium		Zinc		Total Dissolved Solids	
	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	mg/L	Quotient
South Fork Koktuli River																				
SK100G	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
SK100F	160	150	57	0.65	0.23	3.6	1.5	0.58	98	0.14	1.8	0.18	0.11	0.38	0.73	0.15	33	1.4	52	0.05
SK100CP2[a,b]	56	52	28	0.32	0.09	1.4	0.55	0.22	58	0.08	0.89	0.090	0.10	0.33	0.35	0.07	13	0.58	62	0.06
SK124A	1.4	1.3	56	0.65	0.06	0.91	1.6	0.63	82	0.12	2.5	0.25	0.23	0.80	1.2	0.24	18	0.79	49	0.05
SK124CP[a,c]	1.4	1.3	55	0.63	0.23	3.6	1.5	0.62	80	0.12	2.5	0.25	0.23	0.78	1.2	0.24	18	0.78	390	0.39
SK100C	20	18	46	0.52	0.07	1.1	1.2	0.47	62	0.09	1.9	0.19	0.19	0.65	0.88	0.18	15	0.67	260	0.26
SK100CP1[a]	20	18	45	0.52	0.07	1.1	1.2	0.47	61	0.09	1.9	0.19	0.19	0.64	0.88	0.17	15	0.67	260	0.26
SK119A	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
SK119CP[a]	1.5	1.4	18	0.21	0.03	0.44	0.29	0.12	15	0.02	0.49	0.05	0.08	0.29	0.49	0.10	3.3	0.14	57	0.06
SK100B1	11	10	28	0.32	0.04	0.69	0.69	0.28	35	0.05	1.2	0.12	0.12	0.39	0.61	0.12	9.5	0.41	170	0.17
SK100B[d]	7.9	7.3	22	0.25	0.03	0.54	0.50	0.20	26	0.04	0.89	0.09	0.10	0.35	0.47	0.09	7.6	0.33	120	0.12
North Fork Koktuli River																				
NK119A	1.9	1.8	23	0.26	0.03	0.52	0.30	0.12	23	0.03	0.59	0.06	0.07	0.23	0.62	0.12	2.7	0.12	63	0.06
NK119CP2[a]	1.6	1.5	22	0.25	0.03	0.45	0.26	0.10	19	0.03	0.54	0.05	0.07	0.23	0.52	0.10	2.5	0.11	57	0.06
NK119B	0.63	0.59	20	0.23	0.02	0.31	0.22	0.09	8.0	0.01	0.46	0.05	0.06	0.20	0.24	0.05	2.2	0.09	38	0.04
NK119CP1[a]	1.2	1.1	21	0.24	0.02	0.38	0.22	0.09	15	0.02	0.49	0.05	0.06	0.21	0.41	0.08	2.3	0.10	49	0.05
NK100C[c]	0.62	0.58	34	0.39	0.03	0.53	0.83	0.33	49	0.07	1.5	0.15	0.16	0.54	0.66	0.13	9.9	0.43	230	0.23
NK100B	0.74	0.69	30	0.34	0.03	0.45	0.65	0.26	37	0.05	1.2	0.12	0.16	0.55	0.57	0.11	8.0	0.35	180	0.18
NK100A1	0.74	0.69	18	0.20	0.02	0.29	0.32	0.13	18	0.03	0.66	0.07	0.07	0.24	0.35	0.071	5.0	0.22	100	0.10
NK100A[e]	0.54	0.51	20	0.23	0.02	0.29	0.28	0.11	21	0.03	0.63	0.06	0.09	0.29	0.31	0.06	4.2	0.18	92	0.09
Upper Talarik Creek																				
UT100E	0.81	0.75	19	0.22	0.05	0.72	0.67	0.27	20	0.03	1.1	0.11	0.06	0.22	0.43	0.09	4.3	0.19	75	0.08
UT100D	1.3	1.3	35	0.40	0.08	1.2	1.3	0.52	60	0.09	1.8	0.18	0.07	0.24	0.69	0.14	6.3	0.27	89	0.09
UT100C2	0.38	0.36	13	0.15	0.01	0.19	0.14	0.06	24	0.03	0.50	0.05	0.04	0.14	0.19	0.04	1.9	0.09	48	0.05
UT100C1	0.34	0.32	9.7	0.11	0.01	0.18	0.11	0.04	14	0.02	0.45	0.05	0.04	0.13	0.18	0.04	1.3	0.06	49	0.05
UT100C	0.41	0.38	11	0.13	0.01	0.18	0.09	0.03	10	0.02	0.44	0.04	0.04	0.14	0.18	0.04	1.5	0.06	48	0.05
UT119A[b]	27	25	17	0.20	0.05	0.72	0.27	0.11	28	0.04	0.61	0.06	0.08	0.26	0.24	0.05	7.0	0.30	49	0.05
UT100B[f]	3.6	3.3	10	0.12	0.02	0.29	0.11	0.05	17	0.03	0.46	0.05	0.07	0.24	0.17	0.03	2.4	0.10	47	0.05

Notes:
[a] Confluence point where virtual gage was created because physical gage does not exist.
[b] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location.
[c] Wastewater treatment plant discharges 50% of its flow at this site.
[d] USGS 15302200.
[e] USGS 15302250.
[f] USGS 15300250.
NA = not applicable; the stream at the gage would be destroyed.

Table 8-17. Estimated concentrations of contaminants of concern and associated risk quotients for the Pebble 6.5 scenario, assuming wastewater treatment plant failure, at locations in the mine scenario watersheds. Upper Talarik Creek would be unchanged from Table 8-16. See Box 8-3 for a description of how risk quotients were calculated.

Stream and Gage	Copper		Aluminum		Cadmium		Cobalt		Manganese		Nickel		Lead		Selenium		Zinc		Total Dissolved Solids	
	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	µg/L	Quotient	mg/L	Quotient
South Fork Koktuli River																				
SK100G	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
SK100F	160	150	57	0.65	0.23	3.6	1.5	0.58	98	0.14	1.8	0.18	0.11	0.38	0.73	0.15	33	1.4	62	0.06
SK100CP2[a,b]	56	52	28	0.32	0.09	1.3	0.55	0.22	58	0.08	0.90	0.09	0.10	0.33	0.35	0.07	13	0.58	49	0.05
SK124A	110	100	56	0.65	0.20	3.1	1.6	0.64	82	0.12	2.5	0.25	0.26	0.90	1.2	0.24	26	1.1	390	0.39
SK124CP[a,c]	100	97	56	0.63	0.19	3.0	1.5	0.62	80	0.12	2.5	0.25	0.26	0.89	1.2	0.24	25	1.1	380	0.38
SK100C	86	80	46	0.52	0.16	2.4	1.2	0.47	62	0.09	1.9	0.19	0.21	0.71	0.88	0.18	20	0.88	260	0.26
SK100CP1[a]	86	80	45	0.52	0.15	2.4	1.2	0.47	61	0.09	1.9	0.19	0.21	0.71	0.88	0.18	20	0.88	260	0.26
SK119A	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
SK119CP[a]	1.5	1.4	18	0.21	0.03	0.44	0.29	0.12	15	0.02	0.49	0.05	0.08	0.29	0.49	0.10	3.3	0.14	57	0.06
SK100B1	47	44	28	0.32	0.09	1.4	0.70	0.28	35	0.05	1.2	0.12	0.13	0.43	0.61	0.12	12	0.52	170	0.17
SK100B[d]	34	32	22	0.25	0.07	1.1	0.50	0.20	26	0.04	0.89	0.09	0.11	0.37	0.47	0.09	9.5	0.41	120	0.12
North Fork Koktuli River																				
NK119A	0.07	0.23	0.63	0.12	0.03	0.52	0.30	0.12	23	0.03	0.59	0.06	0.07	0.23	0.62	0.12	2.7	0.12	63	0.06
NK119CP2[a]	0.07	0.23	0.52	0.10	0.03	0.45	0.26	0.10	19	0.03	0.54	0.05	0.07	0.23	0.52	0.10	2.5	0.11	57	0.06
NK119B	0.06	0.20	0.24	0.05	0.02	0.31	0.22	0.09	8.0	0.01	0.46	0.05	0.06	0.20	0.24	0.05	2.2	0.09	38	0.04
NK119CP1[a]	1.2	1.1	21	0.24	0.02	0.38	0.22	0.09	15	0.02	0.49	0.05	0.06	0.21	0.41	0.08	2.3	0.10	49	0.05
NK100C[c]	57	54	34	0.39	0.11	1.7	0.83	0.33	49	0.07	1.5	0.15	0.17	0.60	0.66	0.13	14	0.60	230	0.23
NK100B	43	40	30	0.34	0.08	1.3	0.65	0.26	37	0.05	1.2	0.12	0.17	0.60	0.57	0.11	11	0.48	180	0.18
NK100A1	20	19	18	0.20	0.04	0.69	0.32	0.13	18	0.03	0.66	0.07	0.08	0.26	0.35	0.07	6.3	0.27	100	0.10
NK100A[e]	17	16	20	0.23	0.04	0.63	0.28	0.11	21	0.03	0.63	0.06	0.09	0.31	0.31	0.06	5.3	0.23	92	0.09

Notes:

[a] Confluence point where virtual gage was created because physical gage does not exist.

[b] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location.

[c] Wastewater treatment plant discharges 50% of its flow at this site.

[d] USGS 15302200.

[e] USGS 15302250.

NA = not applicable; the stream at the gage would be destroyed.

Table 8-18. Estimated total toxicity of metals of concern for each mine scenario, under routine operations and with wastewater treatment plant failure, at locations in the mine scenario watersheds. Values are the sums of the toxic quotients for the metals of concern.

Stream and Gage	Background	Pebble 0.25		Pebble 2.0		Pebble 6.5	
		Routine Operations	WWTP Failure	Routine Operations	WWTP Failure	Routine Operations	WWTP Failure
South Fork Koktuli River							
SK100G	3.1	3.4	3.4	100	100	NA	NA
SK100F	2.4	2.7	2.7	22	22	160	157
SK100CP2[a,b]	-	2.6	2.6	12	12	55	55
SK124A	2.5	3.2	20	3.4	26	5.7	107
SK124CP[a,c]	-	3.1	20	3.4	25	5.6	104
SK100C	2.3	2.7	11	7.9	20	22	86
SK100CP1[a]	-	2.7	11	7.9	19	22	86
SK119A	1.2	1.2	1.2	1.3	1.3	2.6	2.6
SK119CP[a]	-	1.2	1.2	1.3	1.3	2.8	2.8
SK100B1	1.2	1.3	4.7	3.3	7.8	12	47
SK100B[d]	1.1	1.2	3.6	2.6	5.8	9.2	34
North Fork Koktuli River							
NK119A	1.0	1.6	1.6	3.2	3.2	3.2	3.2
NK119CP2[a]	-	1.5	1.5	2.8	2.8	2.8	2.8
NK119B	1.0	1.0	1.0	1.1	1.1	1.6	1.6
NK119CP1[a]	-	1.4	1.4	2.1	2.1	2.3	2.3
NK100C[b]	1.1	1.6	9.7	1.7	12	3.2	57
NK100B	1.2	1.6	6.6	1.8	8.8	2.9	43
NK100A1	1.1	1.3	3.4	1.4	4.2	1.9	21
NK100A[e]	1.1	1.2	3.0	1.3	3.6	1.8	17
Upper Talarik Creek							
UT100E	0.93	1.0	0.96	1.0	1.0	2.6	2.6
UT100D	1.3	1.3	1.3	1.8	1.8	4.3	4.3
UT100C2	0.90	0.93	0.93	1.0	1.0	1.1	1.1
UT100C1	0.76	0.80	0.80	0.86	0.9	0.9	0.94
UT100C	0.89	0.92	0.92	1.0	1.0	1.0	1.0
UT119A[b]	0.75	1.8	1.8	6.5	6.5	27	27
UT100B[f]	0.99	1.1	1.1	1.8	1.8	4.2	4.2

Notes:

[a] Confluence point where virtual gage was created because physical gage does not exist; dash (-) indicates that no background value is available.

[b] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location.

[c] Wastewater treatment plant discharges 50% of its flow at this site.

[d] USGS 15302200.

[e] USGS 15302250.

[f] USGS 15300250.

WWTP = wastewater treatment plant; NA = not applicable, because the stream at the gage would be destroyed.

Table 8-19. Background copper concentrations and, for each mine scenario, copper concentrations in contributing loads and ambient waters (fully mixed reaches below each gage) and associated risk quotients, assuming routine operations. See Box 8-3 for a description of how risk quotients were calculated. All concentrations are in µg/L.

Stream and Gage	Background	Pebble 0.25			Pebble 2.0			Pebble 6.5		
		CL	AW	Quotient	CL	AW	Quotient	CL	AW	Quotient
South Fork Koktuli River										
SK100G	2.4	3.2	2.4	2.2	380	100	94	-	NA	NA
SK100F	1.6	3.2	1.7	1.6	12	21	20	670	160	150
SK100CP2[a,b]	1.6	1.7	1.7	1.5	11	11	10	56	56	52
SK124A	1.4	1.1	1.3	1.2	1.1	1.3	1.2	1.4	1.4	1.3
SK124CP[a,c]	1.4	-	1.3	1.2	-	1.3	1.2	-	1.4	1.3
SK100C	1.4	-	1.4	1.3	-	6.5	6.1	3.2	20	18
SK100CP1[a]	1.4	-	1.4	1.3	-	6.5	6.0	-	20	18
SK119A	0.42	-	0.42	0.39	3.5	0.43	0.41	5.2	1.5	1.4
SK119CP[a]	0.42	-	0.42	0.39	-	0.43	0.41	3.4	1.5	1.4
SK100B1	0.62	-	0.54	0.51	-	2.5	2.4	4.0	11	10
SK100B[d]	0.54	-	0.47	0.44	-	1.9	1.8	-	7.9	7.3
North Fork Koktuli River										
NK119A	0.31	5.1	0.70	0.65	5.0	1.9	1.8	5.0	1.9	1.8
NK119CP2[a]	0.31	-	0.64	0.60	3.3	1.5	1.4	3.3	1.6	1.5
NK119B	0.41	-	0.41	0.39	3.2	0.42	0.39	3.7	0.63	0.58
NK119CP1[a]	0.33	-	0.58	0.55	-	1.1	1.0	-	1.2	1.1
NK100C[c]	0.35	1.1	0.43	0.41	1.1	0.43	0.40	1.1	0.62	0.58
NK100B	0.40	-	0.52	0.49	-	0.62	0.58	-	0.74	0.69
NK100A1	0.61	-	0.66	0.61	3.4	0.70	0.65	3.4	0.74	0.69
NK100A[e]	0.41	-	0.45	0.42	-	0.49	0.46	-	0.54	0.51
Upper Talarik Creek										
UT100E	0.34	-	0.34	0.32	-	0.34	0.32	3.2	0.81	0.75
UT100D	0.50	-	0.50	0.47	3.2	0.63	0.59	3.2	1.3	1.3
UT100C2	0.34	-	0.34	0.31	-	0.36	0.33	3.2	0.38	0.36
UT100C1	0.30	-	0.30	0.28	-	0.32	0.30	-	0.34	0.32
UT100C	0.38	-	0.38	0.35	-	0.39	0.37	-	0.41	0.38
UT119A[b]	0.21	1.7	0.99	0.93	11	5.8	5.4	56	27	25
UT100B[f]	0.34	-	0.43	0.40	-	1.04	0.97	-	3.6	3.3

Notes:
NA = not applicable, because stream at gage location would be destroyed. Dashes (-) indicate there are no contributing loads at that gage under that scenario.
[a] Confluence point where virtual gage was created because physical gage does not exist.
[b] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location.
[c] Wastewater treatment plant discharges 50% of its flow at this site.
[d] USGS 15302200.
[e] USGS 15302250.
[f] USGS 15300250.
CL = contributing loads; AW = ambient waters; quotient = predicted/criterion.

Table 8-20. Background copper concentrations and, for each mine scenario, copper concentrations in contributing loads and ambient waters (fully-mixed reaches below each gage) and associated risk quotients, assuming wastewater treatment plant failure. Upper Talarik Creek would be unchanged from Table 8-19. See Box 8-3 for a description of how risk quotients were calculated. All concentrations are in µg/L.

Stream and Gage	Background	Pebble 0.25			Pebble 2.0			Pebble 6.5		
		CL	AW	Quotient	CL	AW	Quotient	CL	AW	Quotient
South Fork Koktuli River										
SK100G	2.4	3.2	2.4	2.2	380	100	94	-	NA	NA
SK100F	1.6	3.2	1.7	1.6	12	21	20	670	160	150
SK100CP2[a,b]	1.6	1.7	1.7	1.5	11	11	10	56	56	52
SK124A	1.4	75	20	18	100	25	23	140	110	100
SK124CP[a,c]	1.4	-	19	17	-	24	22	-	100	97
SK100C	1.4	-	11	9.9	-	19	17	3.2	86	80
SK100CP1[a]	1.4	-	11	9.9	-	19	17	-	86	80
SK119A	0.42	-	0.42	0.39	3.5	0.43	0.41	5.2	1.5	1.4
SK119CP[a]	0.42	-	0.42	0.39	-	0.43	0.41	3.4	1.5	1.4
SK100B1	0.62	-	4.1	3.8	-	7.2	6.7	4.0	47	44
SK100B[d]	0.54		3.0	2.8	-	5.2	4.8	-	34	32
North Fork Koktuli River										
NK119A	0.31	5.1	0.70	0.65	5.0	1.9	1.8	5.0	1.9	1.8
NK119CP2[a]	0.31	-	0.64	0.60	3.3	1.5	1.4	3.3	1.6	1.5
NK119B	0.41	-	0.41	0.39	3.2	0.42	0.39	3.7	0.63	0.58
NK119CP1[a]	0.33	-	0.58	0.55	-	1.1	1.0	-	1.2	1.1
NK100C[c]	0.35	75	9.0	8.4	100	11	11	150	57	54
NK100B	0.40	-	5.8	5.4	-	7.8	7.3	-	43	40
NK100A1	0.61	-	2.9	2.7	3.4	3.6	3.4	3.4	20	19
NK100A[e]	0.41	-	2.3	2.1	-	2.9	2.7	-	17	16

Notes:
NA = not applicable, because stream at gage location would be destroyed. Dashes (-) indicate there are no contributing loads at that gage under that scenario.
[a] Confluence point where virtual gage was created because physical gage does not exist.
[b] 1/3 of total return flow is transferred from SK100CP2 to UT119A to represent interbasin transfer at this location.
[c] Wastewater treatment plant discharges 50% of its flow at this site.
[d] USGS 15302200.
[e] USGS 15302250.
CL = contributing loads; AW = ambient waters; quotient = predicted/criterion.

Table 8-21. Description of stream reaches affected in the mine scenarios and sources of the concentration estimates applied to the stream reaches.

Reach Designation[a]	Reach Description[b]	Length (km)[c]	Concentration Assigned and Qualifiers[d]
South Fork Koktuli River—Mainstem			
SK100B	SK100B to confluence of the South and North Fork Koktuli Rivers	23	SK100B, overestimates lower end due to dilution
SK100B1	SK100B1 to SK100B	4.5	SK100B1, small overestimate of lower end due to dilution
SK100CP1/ SK119CP	SK100CP1/ SK119CP confluence to SK100B1	4.3	Mixed SK100CP1 and SK119CP, little dilution downstream
SK100C	SK100C to SK100CP1	1.2	SK100C, negligible further dilution in short reach
SK100CP2/ SK124CP	SK100CP2/SK124CP confluence to SK100C	6.4	Mixed SK100CP2 and SK124CP, little dilution downstream
SK100F	SK100F to SK100CP2	11	Mean SK100F and SK100CP2 due to significant dilution
SK100G	SK100G to SK100F (not Pebble 6.5)	3.3/3.3/NA	Mean SK100G and SK100F due to significant dilution
SK Rock	Waste rock to SK100F (Pebble 6.5 only)	NA/NA/0.83	SK 100F, assuming input near base of rock pile and short reach
SK Halo/Rock	Dewatering halo and rock pile to SK100G (Pebble 0.25 and 2.0)	1.87/0.54/NA	SK 100G, assuming input near base of rock pile and short reach
South Fork Koktuli River—Tributaries			
SK Headwaters	Headwaters to SK119A (Pebble 0.25)	7.0	Background for Pebble 0.25 scenario
SK TSF1	TSF1 to SK119A (Pebble 2.0)	6.8	SK119A, significant dilution so underestimate
SK119A	SK119A to SK119CP	1.6/1.6/1.5	SK119A, for Pebble 0.25 and Pebble 2.0 scenarios, dilution is minimal; SK119CP for remnant reach in Pebble 6.5 scenario, when SK119A destroyed
SK124A	SK124A to SK124CP	2.6	SK124A, no dilution in this reach within precision
SK WWTP	WWTP to SK124A	5.0	SK124A, underestimate of upper end from dilution of WWTP and, in Pebble 6.5 scenario, TSF 3 leachate
North Fork Koktuli River—Mainstem			
NK100A	NK100A to confluence of the South and North Fork Koktuli Rivers	4.7	NK100A, little dilution
NK100A1	NK100A1 to NK100A	8.4	N100A1, which has a small contributing load in the Pebble 2.0 and Pebble 6.5 scenarios, so small overestimate
NK100B	NK100B to NK100A1	20	NK100B, approximately two times dilution over long reach in Pebble 0.25 scenario but no change in Pebble 2.0 and Pebble 6.5 scenarios due to balance of dilution by tailings leachate
NK100CP1/NK100C	NK100CP1/NK100C confluence to NK100B	0.79	Mixed NK100CP1 and NK100C, little dilution downstream
NK100C	NK100C to confluence NK119A stream	0.19	NK100C, negligible further dilution in tiny reach
NK WWTP	WWTP discharge to NK100C	4.3	NK100C, underestimate of upper end, but assuming negligible dilution

Table 8-21. Description of stream reaches affected in the mine scenarios and sources of the concentration estimates applied to the stream reaches.

Reach Designation[a]	Reach Description[b]	Length (km)[c]	Concentration Assigned and Qualifiers[d]
North Fork Koktuli River—Tributaries			
NK119B/NK100CP2	NK119B/NK100CP2 confluence to NK119CP1	0.43	Mixed NK119B and NK100CP2, little dilution downstream
NK119A	NK119A to NK119CP2	1.3	NK119A, little dilution downstream
NK TSF1	TSF 1 to NK119A	0.6	NK119A, assuming input near toe of dam and short reach
NK Headwaters	Headwaters or dewatering halo to NK119B	6.8/6.8/6.6	NK119B, which has a small contributing load in the Pebble 6.5 scenario from tailings leachate at its upper end, so small underestimate
Upper Talarik Creek—Mainstem			
UT100B	UT100B to Iliamna Lake	23	UT100B, considerable dilution would occur in this long reach, so only the upper end would not be overestimate
UT100C	UT100C to UT119 confluence	4.3	UT100C, some unquantified dilution at lower end so overestimate there
UT100C1	UT100C1 to UT100C	7.6	UT100C1, minimal change in concentration
UT100C2	UT100C1 to UT100C2	6.9	UT100C2, dilution and loading balance in reach
UT100D	UT100D to UT100C2	6.1	Mean of UT100D and UT100C2 because of significant dilution in the reach
UT100E	UT100E to UT100D (Pebble 0.25 only)	7.1/NA/NA	UT100E flows at background concentrations for the Pebble 0.25 scenario
UT Rock	Waste rock to UT100D (not Pebble 0.25)	NA/2.1/0.15	UT100D, assuming input near base of rock pile and short reach
Upper Talarik Creek—Tributaries			
UT Headwaters	Headwaters to UT119A	6.5	UT119A receives interbasin transfer; assumed along nearly all of length but overestimates at upper end

Notes:

[a] Reaches are designated by the gage or other feature at their heads. Designations in the form G1/G2 indicate the confluence of a stream and tributary with gages G1 and G2 above the confluence.

[b] Upper and lower bounds of the reach.

[c] Lengths that differ among mine sizes are presented as Pebble 0.25/Pebble 2.0/Pebble 6.5.

[d] Concentrations are point estimates at upstream gages from Table 8-20, flow-weighted mixtures of concentrations at upstream gages, or means of upstream and downstream gages. Qualifiers explain the possibility of over or underestimation.

WWTP = wastewater treatment plant; NA = not applicable

Table 8-22. Copper concentrations and benchmarks exceeded in ambient waters in each reach and for each mine scenario, assuming routine operations. Reaches are described in Table 8-21.

Reach Designation[a]	Pebble 0.25		Pebble 2.0		Pebble 6.5	
	Copper (μg/L)	Effects	Copper (μg/L)	Effects	Copper (μg/L)	Effects
South Fork Koktuli River—Mainstem						
SK100B	<0.47	-	<1.9	IC	<7.9	IC/IA/FA
SK100B1	0.54	-	2.5	IC/IA	11	IC/IA/FA
SK100CP1/ SK119CP	0.95	-	3.9	IC/IA	16	IC/IA/FA
SK100C	1.4	-	6.5	IC/IA/FA	20	IC/IA/FA
SK100CP2/SK124CP	1.5	IC	6.1	IC/IA/FA	20	IC/IA/FA
SK100F	1.7	IC	16	IC/IA/FA	110	IC/IA/FA/FS/FR/FK
SK100G	2.0	IC	61	IC/IA/FA/FS/FR	NA	NA
SK Rock	NA	NA	NA	NA	>160	IC/IA/FA/FS/FR/FK
SK Halo/Rock	>2.4	IC/IA	>100	IC/IA/FA/FS/FR/FK	NA	NA
South Fork Koktuli River—Tributaries						
SK Headwaters	0.42	-	NA	NA	NA	NA
SK TSF1	NA	NA	>0.44	-	NA	NA
SK119A	0.42	-	0.44	-	1.5	-
SK124A	1.3	-	1.3	-	1.4	-
SK WWTP	1.3	-	1.3	-	1.3	-
North Fork Koktuli River—Mainstem						
NK100A	0.45	-	0.44	-	0.54	-
NK100A1	0.66	-	0.70	-	0.74	-
NK100B	0.52	-	0.62	-	0.74	-
NK119CP1/NK100C	0.48	-	0. 61	-	0.74	-
NK100C	0.44	-	0.43	-	0.62	-
NK WWTP	>0.44	-	>0.43	-	>0.62	-
North Fork Koktuli River—Tributaries						
NK119B/NK119CP2	0.60	-	1.1	IC	1.4	IC
NK119A	0.70	-	1.8	IC/IA	1.9	IC/IA
NK TSF1	>0.70	-	>1.8	IC/IA	>1.9	IC/IA
NK Headwaters (NK119B)	>0.41	-	>0.42	-	>0.63	-
Upper Talarik Creek—Mainstem						
UT100B	<0.42	-	<1.0	-	<3.6	IC/IA
UT100C	0.38	-	0.39	-	0.41	-
UT100C1	0.30	-	0.32	-	0.34	-
UT100C2	0.34	-	0.36	-	0.38	-
UT100D	0.42	-	0.49	-	0.86	-
UT100E	<0.34	-	NA	NA	NA	NA
UT Rock	NA	NA	0.63	-	1.5	-
Upper Talarik Creek—Tributaries						
UT Headwaters (119A)	>0.98	-	>5.8	IC/IA	>27	IC/IA/FA

Notes:

Dashes (-) indicate that no effects are expected.

[a] Reaches are designated by the gage or other feature at their heads. Designations in the form G1/G2 indicate the confluence of a stream and tributary with gages G1 and G2 above the confluence.

IC = invertebrate chronic; IA = invertebrate acute; FA = fish avoidance; FS = fish sensory; FR = fish reproduction; FK = fish kill;
NA = not applicable.

Table 8-23. Copper concentrations and benchmarks exceeded in ambient waters in each reach and for each mine scenario, assuming a wastewater treatment plant failure. Reaches are described in Table 8-21.

Reach Designation[a]	Pebble 0.25		Pebble 2.0		Pebble 6.5	
	Copper (µg/L)	Effects	Copper (µg/L)	Effects	Copper (µg/L)	Effects
South Fork Koktuli River—Mainstem						
SK100B	<3.0	IC/IA	<5.1	IC/IA/FA	<34	IC/IA/FA/FS/FR
SK100B1	4.1	IC/IA	7.2	IC/IA/FA	47	IC/IA/FA/FS/FR
SK100CP1/SK119CP	6.2	IC/IA/FA	11	IC/IA/FA	68	IC/IA/FA/FS/FR/FK
SK100C	11	IC/IA/FA	19	IC/IA/FA	86	IC/IA/FA/FS/FR/FK
SK100CP2/SK124CP	9.8	IC/IA/FA	17	IC/IA/FA	87	IC/IA/FA/FS/FR/FK
SK100F	1.7	IC	16	IC/IA/FA	110	IC/IA/FA/FS/FR/FK
SK100G	1.9	IC	60	IC/IA/FA/FS/FR/FK	NA	NA
SK Rock	NA	NA	NA	NA	>160	IC/IA/FA/FS/FR/FK
SK Halo/Rock	>2.4	IC/IA	>100	IC/IA/FA/FS/FR/FK	NA	NA
South Fork Koktuli River—Tributaries						
SK Headwaters	>0.42	-	NA	NA	NA	NA
SK TSF1	NA	NA	>0.43	-	NA	NA
SK119A	0.42	-	0.43	-	1.5	IC
SK124A	19	IC/IA/FA	25	IC/IA/FA/ FR	110	IC/IA/FA/FS/FR/FK
SK WWTP	>19	IC/IA/FA	>25	IC/IA/FA/ FR	>110	IC/IA/FA/FS/FR/FK
North Fork Koktuli River—Mainstem						
NK100A	2.3	IC/IA	2.9	IC/IA	17	IC/IA/FA
NK100A1	2.9	IC/IA	3.6	IC/IA	20	IC/IA/FA/FS
NK100B	5.8	IC/IA/FA	7.8	IC/IA/FA	43	IC/IA/FA/FS/FR
NK119CP1/NK100C	6.2	IC/IA/FA	8.7	IC/IA/FA	47	IC/IA/FA/FS/FR
NK100C	9.0	IC/IA/FA	11	IC/IA/FA	57	IC/IA/FA/FS/FR
NK WWTP	>9.0	IC/IA/FA	>11	IC/IA/FA	>57	IC/IA/FA/FS/FR
North Fork Koktuli River—Tributaries						
NK119B/NK119CP2	0.60	-	1.3	IC	1.4	IC
NK119A	0.70	-	1.9	IC/IA	1.9	IC/IA
NK TSF1	>0.70	-	>1.9	IC/IA	>1.9	IC/IA
NK Headwaters	>0.41	-	>0.41	-	>0.63	-

Notes:
Dashes (-) indicate that no effects are expected.
[a] Reaches are designated by the gage or other feature at their heads. Designations in the form G1/G2 indicate the confluence of a stream and tributary with gages G1 and G2 above the confluence.
IC = invertebrate chronic; IA = invertebrate acute; FA = fish avoidance; FS = fish sensory; FR = fish reproduction; FK = fish kill; NA = not applicable.

8.2.3.5 Dilution Zones

Analyses in Sections 8.2.3.3 and 8.2.3.4 dealt with risks from concentrations in fully-mixed locations or reaches. Prior to achieving full mixing, the effluent plume would create a gradient from undiluted to fully diluted, within which exposures would be higher than the fully-mixed concentrations. This should not be an issue for a plume of properly treated wastewater, but could result in locally high exposures under WWTP failure. The untreated wastewater concentrations of copper alone (Table 8-9) would be sufficient to cause lethality in trout and other toxic endpoint effects among survivors, in both receiving streams under all three mine scenarios.

For waste rock and tailings leachates, effluent can enter a stream below the waste rock pile or TSF dam. Leachate that drains to shallow aquifers would enter a stream through its cobble and gravel substrate. Where leachates enter, benthic invertebrates and fish eggs and larvae could be exposed to a range of concentrations, from undiluted to highly diluted leachate (the scenarios include significant dilution by groundwater). Undiluted concentrations of metals of concern are listed in Table 8-9. NAG leachate, which would enter Upper Talarik Creek in the Pebble 2.0 and Pebble 6.5 scenarios at 3.2 μg/L copper, would be sufficient to cause invertebrate mortality unless it was significantly diluted by groundwater first. The NAG and PAG leachate, which would enter the South Fork Koktuli River in the Pebble 2.0 scenario at 395 μg/L copper, would be more than six times the acute lethal concentration for trout. The Pebble 6.5 scenario, at 735 μg/L copper, would require dilution by more than a factor of 10 to avoid acute lethality of trout. At the more sensitive end of the spectrum, invertebrates would require dilution of NAG and PAG leachate in the Pebble 6.5 scenario by a factor of 490 to avoid chronic toxicity.

Tailings leachates would enter tributaries of the South and North Fork Koktuli Rivers at an undiluted copper concentration of 5.3 μg/L unless they were significantly diluted by groundwater first. This is sufficient to kill invertebrates and to cause avoidance by trout. It would require dilution by factors of 3.5 to 5 to avoid chronic toxicity to invertebrates.

8.2.3.6 Spatial Distribution of Estimated Effects

The results of screening for total metals and copper and the analysis of severity are presented in Tables 8-18 through 8-23 and summarized below. They are best understood by consulting the maps of the three mine scenario footprints showing streams and gages in Figures 7-14 through 7-16.

Pebble 0.25 Scenario—Routine Operations

- **South Fork Koktuli River**. Copper loading from NAG waste rock in reaches SK Halo/Rock, SK100G, and SK100 F would slightly increase the naturally high levels of copper and other metals and would increase estimated concentrations to chronically toxic levels for invertebrates in the first 22 km. WWTP effluent would enter the SK124 tributary. That effluent would slightly decrease copper concentration due to treatment to achieve criteria. Concentrations would decline downstream to SK100B due to dilution.
- **North Fork Koktuli River.** Input of TSF 1 leachate to the tributary above NK119A would increase copper levels from background, but no copper criteria or benchmarks would be exceeded. Input of water treatment effluent at NK100C would increase metal concentrations over background such that, although copper would meet the criterion, the total metal risk quotient would rise to 1.7. At the confluence of the TSF- and WWTP-influenced streams (NK100B), copper concentrations would be below criteria and total metal toxicity would be marginal and decline downstream.
- **Upper Talarik Creek.** Copper loading would come entirely from interbasin transfer to the UT119 tributary and copper would not reach toxic levels.

Pebble 2.0 Scenario—Routine Operations

- **South Fork Koktuli River.** Input of NAG and PAG waste rock leachate entering below the waste rock pile (reach SK Halo/Rock) would raise copper concentrations to levels sufficient to kill trout and other salmonids and would be sufficient to inhibit reproduction for another 3.3 km (reach SK100G). Levels at SK100F and for 18 km downstream would be sufficient to cause avoidance by trout and severely deplete invertebrates. Levels sufficient to cause acute lethality to invertebrates would extend another 8.9 km and chronic toxicity would extend for some distance beyond that. However, levels would be relatively low in the SK124 tributary due to dilution by the WWTP effluent.
- **North Fork Koktuli River.** The pattern of input would be the same as for the Pebble 0.25 scenario, but copper and total metals would be highly toxic to invertebrates in the NK119A tributary because the larger TSF would release more leachate. Concentrations would decrease below the confluence of the tributary and the mainstem below NK100C due to WWTP effluent and background water, so that by NK100A concentrations would be close to background.
- **Upper Talarik Creek.** Metals from NAG waste rock leachate would enter at UT100D and raise the naturally marginally toxic total metal levels but not copper. Concentrations would decline downstream to non-toxic levels in the mainstem. Interbasin transfers would raise copper and total metal concentrations to levels that would be highly toxic to invertebrates in the UT119 tributary (reach UT Headwaters).

Pebble 6.5 Scenario—Routine Operations

- **South Fork Koktuli River**. SK100G would be buried by waste rock and SK119A would be buried by tailings. SK100F would exceed the copper criterion by more than 100-fold due to NAG and PAG waste rock leachate, achieving levels sufficient to kill juvenile and adult trout and other salmonids for 12 km. For another 16 to 39 km, aversion and acute toxicity to invertebrates would occur. On the SK119 and SK124 tributaries toxicity would be low despite TSF leakage and WWTP effluent.
- **North Fork Koktuli River.** TSF leakage would enter both the NK119A and NK119B tributaries, resulting in copper and total metal toxicity to invertebrates for 2.4 km. Due to the WWTP, no copper toxicity would occur at or below NK100C but total metal toxicity to invertebrates would occur.
- **Upper Talarik Creek.** Due to interbasin transfer from the South Fork Koktuli River, copper in the UT119 tributary would be highly toxic to invertebrates and aversive to trout. Below the confluence of that tributary, the mainstem would be toxic to invertebrates. NAG waste rock leachate entering the stream from the base of the expanded waste rock pile would increase copper concentrations but would not be expected to cause toxicity.

Wastewater Treatment Plant Failure

The WWTP failure scenario would turn the WWTP effluent from a diluent for tailings leachates to a toxic input that would be diluted by tailings leachate. The effects of releasing untreated wastewater would, of course, be greatest at the points of release (on the SK124 tributary of the South Fork Koktuli River below the tailings dam location and at the head of the North Fork Koktuli River above gage NK100C)

(Table 8-17). Under the Pebble 6.5 scenario, the copper quotient at SK124A would increase from 1.3 (marginal toxicity) with routine operation to 100 (high toxicity) with the WWTP failure (Table 8-20), resulting in levels sufficient to cause a fish kill extending down the South Fork Koktuli mainstem (Table 8-23). Untreated wastewater input above gage NK100C would increase the copper risk quotient from 0.58 to 54 (Table 8-20), resulting in early-life-stage toxicity to trout and other salmonids. The effects would increase as mine size increases. The most severe effects on trout in the SK124 tributary are estimated to be aversion (FA), early-life-stage toxicity (FR), and lethality to all life stages (FK) for the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. The most severe effects on trout below the North Fork Koktuli outfall are estimated to be aversion for the Pebble 0.25 and Pebble 2.0 scenarios and early-life-stage toxicity for the Pebble 6.5 scenario. That implies a shift in severity from a depleted invertebrate community (which would reduce fish production) and fish aversion to loss of fish reproduction and death of all fish (Table 8-23). Toxicities for total metals are slightly higher (Table 8-18).

Toxic effects are functions of both duration of exposure and concentration. Because concentrations would be so high, toxic effects on salmonids for the Pebble 2.0 and Pebble 6.5 scenarios with WWTP failure would be severe in the South Fork Koktuli River, even if the failure was of short duration. However, in the North Fork Koktuli or downstream of the area analyzed, the effects of WWTP failure would depend on the duration of exposure for the Pebble 0.25 scenario. The WWTP failure described in this chapter could last from hours to months depending on the mechanics of the failure and whether replacement of components would be required. Alternatively, WWTP 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 Weber Scannell 1994, USEPA 1998, 2008). In that case, the failure could continue for years, until a new or upgraded treatment system could be designed, approved, and constructed. However, such failures would be much less severe than the upper bound failure scenario evaluated here.

Spillway Release

For the spillway release scenario, we assume that the TSF pond is deep relative to the amount of precipitation so that no appreciable dilution occurs within the TSF, and that the released water has the same chemical characteristics as the TSF supernatant (Table 8-4). Dilution would occur downstream due to runoff from the watersheds along the North Fork Koktuli River (Table 8-24). We assume that precipitation is uniform over the area and that all precipitation results in runoff to the streams. We also assume that runoff would not contribute any additional metal concentrations. The amount of dilution would be proportional to the areas of the contributing watersheds compared to the interior area of TSF 1 and would be independent of the amount or intensity of precipitation.

Table 8-24. Results of the spillway release scenario in terms of copper concentrations at North Fork Koktuli stream gages downstream of TSF 1, estimated effects, and the length of the associated reaches.

Stream Gage[a]	Copper Concentration (μg/L)	Effects	Reach Length (km)
NK100A	0.4	-	4.7
NK100A1	0.5	-	8.4
NK100B	1.1	IC	20
NK119CP1	3.3	IC/IA	0.79
NK119CP2	4.9	IC/IA/FA	0.43
NK119A	5.5	IC/IA/FA	1.5
NK TSF 1	7.8	IC/IA/FA	0.64

Notes:
Dashes (-) indicate that no effects are expected.
[a] Stream reaches and associated gages are described in Table 8-21.
TSF = tailings storage facility; IC = invertebrate chronic; IA = invertebrate acute; FA = fish avoidance.

Of the measured tailings supernatant constituents, only copper concentrations are estimated to exceed water quality criteria or equivalent benchmarks (Table 8-4). The spilled supernatant immediately below the dam (NK TSF 1) would be lethal to invertebrates and would cause avoidance by salmonids. Those effects would continue downstream for approximately 2.6 km through the reach below NK119CP1. Below that, invertebrate lethality would continue for another 0.79 km. The chronic criterion for the North Fork (1.1 μg/L) would be equaled at gage NK100B, so effects would not be expected to extend far down the 20 km reach due to dilution by tributaries. Note, however, that these estimates are based on the assumption that the spillway release would be the only source. If a spillway release was added to routine releases (Table 8-22), exceedance of chronic water quality criteria and chronic toxicity to invertebrates would be likely in all reaches.

8.2.3.7 Analogous Mines

Water quality degradation has been commonly associated with mining in the United States and elsewhere. In particular, the phenomenon known as acid mine or acid rock drainage has severely damaged many streams due to high acidity and dissolved metals and, as the effluent is neutralized, the formation of aluminum, iron, and manganese oxide precipitates. Pre-Tertiary waste rock at the Pebble deposit could produce such effluents (Table 8-8). Although published studies have emphasized the severe effects of acidic waters, it should not be assumed that neutral or alkaline leachates, such as would be expected from the Pebble deposit Tertiary rock (Table 8-6), would have no effects.

Water quality degradation at metal mines in the United States has been reviewed and summarized in a recent report (Earthworks 2012). Earthworks (2012) reviewed the 14 porphyry copper mines operating in the United States and found that all but one had reported failures to collect and treat seepage that resulted in water quality degradation. Such degradation has not been uncommon at mines due to various factors, including inadequate pre-mining data, poor prediction of mitigation needs, inadequate design, improper operation, and equipment failure (Earthworks 2012). Although past frequencies of water quality degradation are not predictive of future frequencies due to changes in engineering practices, they do provide a reasonable upper bound.

Unfortunately, biological or ecological monitoring has not been routinely conducted at operating mines, so ecological consequences are not reported by Earthworks (2012). Where biological monitoring has occurred, acid drainage has been shown to eliminate fish and invertebrates from streams and, after dilution, to reduce abundance, production, and diversity in stream and river ecosystems (Marchand 2002, Jennings et al. 2008). For example, acid drainage from an abandoned copper mine in Britannia Creek, British Columbia, resulted in pH levels below 6 and spring copper concentrations greater than 1,000 μg/L (Barry et al. 2000). 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 cages in the creek. In addition, sustained discharges have resulted in the loss of habitat through precipitation of metal hydroxides. This case illustrates the sensitivity of salmon to acid drainage from a copper mine.

The Fraser River watershed in British Columbia has been recommended as an example of how salmon can coexist with metal mining, and therefore suggested as a model for potential mining in the Bristol Bay watershed. However, a long and dramatic decline in Fraser River sockeye salmon led to an official investigation of causes, with inconclusive results (Box 8-4). In any case, it is clear that the Fraser River is not a good analogue because, unlike potential Bristol Bay development, Fraser River mines are located away from salmon spawning and rearing habitats. The many other activities occurring in the Fraser River watershed confound efforts to pinpoint specific causes of salmon population decline, and the dramatic variability in Fraser River sockeye abundance is not an example that would reassure Alaskans accustomed to the more productive and stable Bristol Bay sockeye salmon fishery.

BOX 8-4. THE FRASER RIVER

The Fraser River watershed, which supports sockeye and other salmon and contains multiple copper mines, could serve as an analogue for proposed mine development in the Bristol Bay watershed. 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 watershed's rivers. In addition, the fishery has been closed in some recent years and most of its 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 commissioned scientific projects to investigate 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 development in the watershed, because there are only five active metal mines and only one (Endako) was near sockeye rearing habitat. Other developments in the Fraser River watershed that potentially affect habitat include logging; pulp, paper, and other wood product manufacturing; coal, placer, and gravel mining; urbanization; hydroelectricity generation; oil and gas drilling; agriculture; and water withdrawal. Although the authors argued that acid and metal drainage from closed mines pose a risk to salmon, they did not analyze those exposures. They concluded that, based on sedimentation of stream habitats, mining was a plausible contributor but not the major contributor to declines in sockeye salmon.

Another Cohen Commission report that addressed contaminants listed mine-related contaminants, but could not specifically quantify the effects of mines (MacDonald et al. 2011). 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. The final report concluded that contaminants could be a secondary contributor, but data were insufficient (Cohen 2012).

In light of this information, Cohen Commission reports on the Fraser River do not provide evidence that mining and salmon co-exist. The fishery declined from 1990 to 2007 and has fluctuated widely since. Recent fluctuations have been associated with marine conditions, but available evidence is insufficient to conclude whether harvesting, habitat degradation, or contaminants have been significant contributors.

Neither the Cohen Commission nor the U.S. Environmental Protection Agency's contractor, ICF International, was able to assess the effects of metal mines in the Fraser River watershed, because compliance documents are not readily available and monitoring data are insufficient. 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 during reclamation activities at the Pinchi Lake Mine in 2004, releasing tailings and leachate to Pinchi Lake. This accident, along with prior releases, resulted in a fish consumption advisory related to mercury bioaccumulation.

In sum, mines in the Fraser River watershed are not located in salmon habitat (Cohen 2012, Gustafson 2012) and other development activities in the watershed obscure any effects of mines at the watershed scale. This diverse and relatively intensive development and the spatial discontinuity between mining and salmon habitat make the Fraser River watershed a poor analogue for potential mine development in the Bristol Bay watershed.

8.2.3.8 Summary

The risks to salmon, rainbow trout, Arctic grayling, and Dolly Varden can be summarized in terms of the total stream kilometers likely to experience different types of effects (Table 8-25). Based on toxicity to rainbow trout, the endpoint salmonids are estimated to be at risk of mortality at all life stages in 0.54 km in the Pebble 2.0 scenario and 12 km in the Pebble 6.5 scenario, assuming routine operations. The waters would be aversive for a much greater length. It is not clear how much resident fish might acclimate to the copper, but newly arriving salmon would not be acclimated and would lose spawning habitat. Hence, salmon could lose 24 km (Pebble 2.0) and 34 to 57 km (Pebble 6.5) of spawning habitat due to copper contamination, assuming that they are as sensitive as rainbow trout. Additional habitat would be lost in tributaries that would not be accessed due to aversion.

Table 8-25. Length of stream in which copper concentrations would exceed levels sufficient to cause toxic effects, assuming routine operations, wastewater treatment plant failure, and spillway release, for each of the three mine scenarios. Intervals account for the unknown but apparently significant dilution in reach SK100B.

	Length of Stream Potentially Affected (km)						
	Pebble 0.25		Pebble 2.0		Pebble 6.5		Pebble 2.0 and 6.5
Toxic Effect[a]	Routine Operations	WWTP Failure	Routine Operations	WWTP Failure	Routine Operations	WWTP Failure	Spillway Release[b]
Invertebrate chronic	21	78–100	40–62	80–100	60–82	78–100	3.4–23
Invertebrate acute	1.9	65–87	39	79–100	59–82	76–99	3.4
Fish avoidance	-	27	24	64–87	34–57	74–97	2.6
Fish sensory	-	-	3.8	27	12	70–92	-
Fish reproduction	-	-	3.8	11	12	61–84	-
Fish kill	-	-	0.54	3.8	12	31	-

Notes:
[a] Effects are defined in Section 8.2.3.4.
[b] Spillway releases are independent routine releases.
Dashes (-) indicate that no stream lengths would likely be affected.
Intervals account for the unknown but apparently significant dilution in reach SK100B.
WWTP = wastewater treatment plant.

The effects of a WWTP failure would depend on its timing and duration. If it occurred during the period of salmon return, more than 64 km (Pebble 2.0) and 74 km (Pebble 6.5) of habitat could be lost due to aversion alone. Mortality of all fish life stages would occur in 3.8 km (Pebble 2.0) and 31 km (Pebble 6.5). Mortality or inhibited development of early fish life stages would occur in 11 km (Pebble 2.0) and 61 to 84 km (Pebble 6.5), where the interval distances account for dilution in the SK 100B reach by excluding and including its 23 km length.

Under routine operations, toxic effects from copper on aquatic invertebrates would occur in 21 km (Pebble 0.25), 40 to 62 km (Pebble 2.0), and 60 to 82 km (Pebble 6.5) of streams (Table 8-25). These effects are highly relevant to protecting salmon and other valued fishes. 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. These estimated effects are based on metal concentrations in fully mixed reaches. Locally, in mixing zones below outfalls or in areas of upwelling of contaminated water, effects would be more severe.

Because available data do not quantify fish production in the potentially affected reaches, it is not possible to estimate the lost production of salmon, trout, Arctic grayling, or Dolly Varden. However, the semi-quantitative surveys performed by PLP (2011) and summarized in Section 7.1 provide some indication of the relative amounts of fish potentially affected. The focal species are those that rear for extended periods in the receiving streams: Chinook salmon, coho salmon, Arctic grayling, and Dolly Varden.

The South Fork Koktuli River, which would be the most severely affected stream, has the lowest reported density of focal species that rear for extended periods in the receiving streams and for which data are available (roughly 14,000 fish/km for Chinook and coho salmon, Arctic grayling and Dolly Varden) (Table 7-3), as well as chum and sockeye salmon. Because 28 to 50 km of the South Fork Koktuli River would have copper levels sufficient to directly affect fish in the Pebble 6.5 scenario, more than a half million individuals of the focal species would be exposed to copper levels sufficient to cause aversion, sensory inhibition, inhibited development, or death. In the Pebble 2.0 scenario, copper levels in 22 km of the South Fork Koktuli River would have direct effects on more than 300,000 individuals of the focal species. Direct effects on fish would not be expected in the Pebble 0.25 scenario.

The North Fork Koktuli River has a focal species density of roughly 20,000 fish/km (Table 7-3), plus unenumerated rainbow trout and chum and sockeye salmon. Since 2.4 km of the North Fork Koktuli River would have copper levels sufficient to be toxic to invertebrates in the Pebble 2.0 and Pebble 6.5 scenarios, more than 47,000 individuals of the focal species would experience reduced food resources.

Upper Talarik Creek has the highest density of the focal species at 45,000 fish/km (Table 7-3) plus unenumerated rainbow trout and sockeye and chum salmon. The 6.5 km of the tributary receiving South Fork Koktuli River interbasin transfers would be expected to have avoidance effects on fish in the Pebble 6.5 scenario and reduced invertebrates in the Pebble 2.0 scenario. In the mainstem below the confluence of that tributary, less than 23 km of stream would experience effects on invertebrates.

For the WWTP failure in the Pebble 2.0 and Pebble 6.5 scenarios, as under routine operations, 40 to 50 km of the South Fork Koktuli River would have copper levels sufficient to directly affect more than a half million of the focal fish. However, effects would be more severe than under routine operations and include acute lethality to all life stages in most reaches. For the Pebble 0.25 scenario, 20 km would experience aversive effects on fish and, in 40 to 62 km, toxicity to invertebrates would result in reduced food resources for more than a half million of the focal fishes.

Due to the uncertainties in the fish density data and the compounding uncertainties in exposure and toxicity, these effects estimates are rough. However, it appears that the number of fish experiencing death or an equivalent effect, such as loss of habitat, would be between 10,000 and 1 million for the Pebble 2.0 and Pebble 6.5 scenarios.

For the WWTP failure in the Pebble 0.25, 2.0, and 6.5 scenarios, 27, 64 to 87, and 74 to 97 km of streams, respectively, would have copper concentrations sufficient to directly affect fish (Table 8-25). Toxicity would result in reduced survival or inhibited development for early salmonid life stages in 61 to 84 km in the Pebble 6.5 scenario, potentially affecting more than a half million fish, depending on the season. Sensory inhibition or aversion would affect 600,000 to 1.4 million individuals of the focal fish species in the three mine scenarios until the failure was corrected.

For the spillway release in the Pebble 2.0 and 6.5 scenarios, copper concentrations would be sufficient to cause avoidance by fish in 2.6 km in the North Fork Koktuli River (Table 8-25). Effects on invertebrate survival would be expected in more than 3.4 km, depending on dilution by tributaries in the lowest reach (Table 8-25).

8.2.4 Additional Mitigation of Leachates

The high metal concentrations in the South Fork Koktuli River due to PAG waste rock leachate suggest that mitigation measures beyond those described in the scenarios or the preliminary Northern Dynasty mining case (Ghaffari et al. 2011) should be considered. Although that design may be sufficient for a typical porphyry copper mine (e.g., equivalent to the Pebble 0.25 scenario), it likely is sufficient for not the massive Pebble 2.0 and 6.5 mine sizes. To avoid exceeding copper criteria, a leachate barrier or collection system for the Pebble 6.5 scenario would require more than 99% effectiveness. Wells, trenches, or walls are not likely to achieve that. Lining the PAG waste rock piles might be effective, but liners have some leakage due to imperfect installation, punctures, and deterioration. An alternative mitigation measure would be to ensure that all PAG waste rock is stored within the drawdown zone for the mine pit. In that way, most acidic and high-metal leachate would be collected and treated before discharge. If PAG waste rock was processed before or at closure, the risk of an acidic pit lake would be minimized (Section 8.1.4). Moving all PAG waste rock near the pit would mean an increase in NAG waste rock leachate leakage to streams as NAG waste rock is moved out of the drawdown zone. If all of the leakage of waste rock leachate for the Pebble 6.5 mine were NAG and if mining did not affect the background copper concentration, the copper concentration would be approximately 1.5 μg/L. That would be a great improvement, but would still equal the chronic criterion for the stream and affect sensitive invertebrates. Hence, it would also be necessary to improve the 50% efficiency of leachate capture assumed here. The magnitude and extent of these predicted effects suggest the need for additional mitigation measures to reduce the input of copper and other metals, beyond the conventional practices assumed in the scenarios. Simply improving capture well efficiency, making the cutoff walls more extensive, or adding a trench is unlikely to achieve water quality criteria under those scenarios. Additional measures might include lining the waste rock piles, reconfiguring the piles, or processing more of the waste rock as it is produced.

8.2.5 Uncertainties

Although it is highly likely that mine operations would adversely affect water quality at the mine site, several factors make it difficult to predict the level of effects and consequent risks to fish.

One component of this uncertainty is associated with the likelihood of water collection and treatment failure. Water collection and treatment failures have been documented at 13 of 14 porphyry copper mines in the United States (Earthworks 2012). These 13 cases represent instances in which engineering uncertainties led to prediction failures, despite the fact that mine permits included mitigation measures intended to prevent such occurrences. These results indicate that failures are not uncommon at modern

U.S. copper mines; however, they cannot be used to quantitatively predict the likelihood of water collection and treatment failures in this or future assessments.

Even in the absence of failures, predicting the effects of mining on water quality is difficult and results are uncertain. Further, the effects of water quality changes on aquatic communities are uncertain. The following factors contribute to these uncertainties.

- The range of potential failures is wide and the probability of occurrence for any of them cannot be estimated from available data. Therefore, we can only state that, based on the record of the mining industry, treatment failures of some sort are likely to occur.
- The waste rock leachate concentrations used in the assessment are from humidity cell tests. Because these tests involve repeated flushing of rock under oxic conditions, they may reasonably represent waste 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. This uncertainty may be minimally estimated by comparing the humidity cell tests with barrel tests conducted in the field with more realistic rock sizes and test conditions (PLP 2011: Section 11.7.1). These tests give qualitatively similar results, but the initial flush of high leachate concentrations in the barrel tests seems to be persisting past the date of data compilation (PLP 2011). Therefore, the concentrations reported for humidity cells are used because they are likely to be closer to long-term values, and the magnitude of uncertainty cannot be estimated.
- The tailings leachate concentrations are also from laboratory tests. Such tests of relatively small samples are imperfect models of the processes in tailings slurries, TSF surface water, near surface–deposited tailings, deeply buried tailings, and leakage into groundwater in the field. It is not clear whether these tests tend to over or underestimate leachates in the field or how large the discrepancy might be.
- The tailings test data do not include pyritic tailings, which are strongly acid-generating. This would tend to underestimate the metal content of tailings leachate, but the effects on leachates from a TSF are likely to be small due to the relatively small proportion of pyritic tailings.
- The available leach testing appears to be preliminary and should be augmented with additional and more realistic testing if mine planning proceeds.
- The surface-water and groundwater hydrology of the potential mine site is complex and the hydrological models used to estimate exposures are inevitably simplifications. This is one of the greatest sources of uncertainty for the water quality risks. More information is needed concerning the movement of water from precipitation to groundwater and surface water, including seasonality and storm and melt events.
- The water quality models assume that mining would not affect background water quality. That is unlikely, but any changes could not be estimated. This assumption is expected to result in overestimation of copper levels, particularly in the South Fork Koktuli River. However, as mining

reduces background levels, it would increase levels from leachate input even more. It could change the expected effects at the margins of toxicity but would not significantly affect the conclusions.

- The use of average receiving water flows neglects the potential consequences of low dilution during low-flow periods. This would be difficult to model, because low streamflows would be associated with low leachate formation and low groundwater levels. This consideration suggests that the low dilution and low leaching rates might balance to some extent, but the degree of balance is unknown.
- Chemical criteria and other single chemical benchmarks do not address interactions or combined effects of individual constituents. The additivity model used here is a reasonable default, but the lack of test data for the actual mixture adds uncertainty. This is a concern of some reviewers but is judged to be a relatively small contributor to uncertainty. Strong interactions tend to occur when all constituents are at or near toxic levels. Only cadmium and zinc reach toxic levels and only in the WWTP failure scenario. Given the overwhelming dominance of copper toxicity, this uncertainty appears to be relatively minor.
- 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. The implications of this uncertainty are discussed at the end of Section 8.2.2.1.
- Criteria for chemicals other than copper either do not address site water chemistry or address it in a simple way (e.g., via hardness normalization). Hence, they may be inaccurate estimates of threshold concentrations for toxic effects in these highly pure waters. The example of copper suggests that the criteria and screening benchmarks could be too high by a factor of 2 (see discussion below).
- Some leachate and process water constituents have water quality criteria, standards or benchmarks for aquatic life that are based on old or sparse literature. Additional data are likely to reveal more sensitive species or responses. This would result in lower benchmarks and criteria and higher risks for the poorly studied chemicals. However, this is unlikely to affect conclusions, because the relatively well-studied metal copper dominates the toxicity.
- If the State of Alaska uses its standards in effluent permitting rather than the national criteria, and if the toxicity of chemicals with no state standards is not considered in the permit, toxicity of the effluents would be significantly higher than estimated in this assessment. This could result in an underestimate of effects by more than a factor of 2, primarily due to using the hardness-adjusted copper standard.
- The concentrations of xanthate and other ore-processing chemicals in ambient waters are roughly estimated to be below toxic levels, but studies in the laboratory or at mine sites are insufficient to determine whether that would actually be the case. If xanthate does not degrade rapidly in the tailings, the estimate that it would not leach into streams at toxic concentrations could be incorrect.
- If the tested rock and tailings samples are not representative, other wastewater constituents may be of concern. Some waste rocks or tailings may have high levels of elements other than those identified in the screening analysis for mean concentrations. For example, selenium concentrations

are not high on average but are well above criteria in some individual leachate samples. This uncertainty might be estimated from statistical analyses of sampling results and modeling of waste rock piles with variance in concentrations among locations, but that is beyond the scope of this assessment.

- The separation of PAG and NAG into separate waste rock piles will inevitably be imperfect. Ghaffari et al. (2011) estimate that 5% of rock in the NAG piles would be PAG. The humidity cell tests were not reported to include any PAG, so humidity cell tests on NAG mixed with 5% PAG would be expected to have higher leachate concentrations. This causes underestimation of risks.
- Although Alaska has a standard for TDS from any source, the toxicity of mixtures of major ions depends on the constituents. Most studies of TDS are based on sodium chloride, which is less toxic than mining leachates that have been studied. Hence, the degree of protection provided by the state standard is uncertain. The toxicity of different salt mixtures may vary by at least a factor of 3. However, estimated TDS levels are not high enough for this to be a major uncertainty.

One method for quantifying uncertainty is provided by comparing the benchmarks for copper toxicity that might be used as thresholds for minimum risk (Section 8.2.2). The national ambient water quality criteria for copper are based on the BLM, which better accounts for the influence of water quality on bioavailability than the hardness-derived state standard. These two benchmarks differ by a factor of 1.7 for acute values and 2.1 for chronic values. Four other metals of concern (cadmium, lead, nickel, and zinc) have hardness-dependent standards but no BLM-based criteria. If they also are too high by a factor of approximately 2, then those metals and total metal toxicity are more of a concern than suggested by the screening assessment. The same applies to relevant conventional thresholds for acute and chronic toxicity in salmonids (the LC_{50} and chronic value for rainbow trout), which are BLM-corrected for copper (Table 8-13) but not for other metals. However, the threshold for avoidance of copper exposures (IC_{20}) is 12 to 28 times lower than the LC_{50} and 4.2 to 5.4 times lower than the chronic value (Table 8-14). Hence, the concentration at which a stream would no longer be suitable for trout is considerably underestimated by conventional endpoints. The effects thresholds for less well-studied metals are likely to be equivalently underestimated. Even copper toxicity is likely to be underestimated, due to the absence of tests for sensitive insect species—much less tests of the most sensitive responses of those species (Section 8.2.2.1).

8.3 Temperature

Changes in water temperature associated with mine development activities are a concern given the importance of suitable water temperatures for Pacific salmon. This section begins with a description of current thermal regimes in the mine scenario watersheds and potential alterations due to WWTP discharges under routine operations (Section 8.3.1). It then describes exposure-response relationships for temperature (Section 8.3.2). It ends with a characterization of potential risks associated with the thermal regime of water treatment effluents (Section 8.3.3) and a discussion of uncertainties (Section 8.3.4).

8.3.1 Exposure

8.3.1.1 Thermal Regimes in the Mine Scenario Watersheds

Water temperature data collected by PLP (2011: Appendix 15.1E, Attachment 1) indicate significant spatial variability in thermal regimes. Average monthly stream temperatures in the Pebble deposit area in July or August can range from 6°C to 16°C. 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 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. The range of spatial and temporal variability in temperatures provided by PLP (2011) is consistent with streams influenced by a variety of thermal modifiers, including upstream lakes, groundwater, or tributary contributions (Mellina et al. 2002, Armstrong et al. 2010). Longitudinal profiles of temperature indicate that summertime stream temperatures in the Pebble deposit area do not uniformly increase with decreasing elevation, often due to substantial inputs of cooler water from tributaries or groundwater inputs (PLP 2011). An example of combined tributary and groundwater inflow contributing to significant cooling in summer mainstem temperatures is the South Fork Koktuli River downstream of gage SK100C (Figure 7-14). This is the section of the South Fork Koktuli River fed by the tributary gaged by SK119A, on which a WWTP outfall would be located and to which a portion of the WWTP flows would be directed. As reported by PLP (2011: Appendix 15.1E), combined groundwater and tributary contributions between gages SK100C and SK100B1, including contributions from the tributary gaged by SK119A, contributed to a cooling of 5.4°C, with a gain in flow of 1.36 m^3/s on August 24, 2007. Other examples of spatial variability in summer temperatures are detailed by PLP (2011: Appendix 15.1E, Attachment 1).

Winter water temperatures are also spatially variable, as indicated by instream temperature monitoring data (PLP 2011). The same reach of the South Fork Koktuli River that was cooled in August by groundwater and tributary inflows experienced warming in October. Contributions of relatively warmer groundwater were observed to maintain ice-free conditions in some areas, as revealed by patchiness in ice cover seen in aerial surveys (PLP 2011, Woody and Higman 2011).

8.3.1.2 Thermal Regime Alterations

Mine development and operation would result in alteration of surface-water and groundwater flows and water collection, treatment, and discharge, all of which would affect water temperatures. Some streams would experience increased streamflows, whereas others would experience significant reductions (Table 7-19). Increased streamflows due to additions of effluent from the WWTP would alter temperatures significantly, depending on effluent temperature and quantity. Changes in the source of water supplying streams would also influence thermal responses. For example, reductions in the proportion of thermally-moderated groundwater inputs would result in surface-water temperatures that would be warmer in summer and colder in winter. Conversely, active thermal management (i.e., heating or cooling of effluent) and timed releases from the WWTP could be used to attempt to compensate for mine-related thermal modifications. However, the plan for a Pebble mine outlined by Ghaffari et al. (2011) does not include temperature control by the planned WWTP. The mine scenarios include temperature control to meet state standards, but not to match natural water temperature regimes.

Treated water would be released to tributaries of the South and North Fork Koktuli Rivers and would influence streamflows and water temperatures in downstream reaches. Thermal effects of WWTP effluent would be greatest in the receiving tributaries. Effects would moderate with distance from the WWTP outfall, due to mixing with surface-water and groundwater inputs and heat exchange. Due to the substantial increases in discharge over baseline levels associated with the WWTP (up to 114% increases in monthly mean flow depending on mine scenario and location; Table 7-19), the thermal loads attributable to WWTP discharges would potentially influence temperatures downstream in the South and North Fork Koktuli Rivers. For example, WWTP discharge is expected to comprise 11 to 38% of mean annual flow in the North Fork Koktuli River at gage NK100C (calculated from Tables 8-1 through 8-3). Sensitivities of downstream reaches to WWTP outfall temperatures were not evaluated in this assessment due to uncertainties in the timing and temperature of WWTP discharges and heat exchange processes in downstream reaches. Managing treated water temperatures to maintain baseline thermal regimes would be most protective of fish populations adapted to local thermal regimes, but would require temperature and hydrologic modeling informed by baseline monitoring and the ability to control temperatures and quantities of discharged flows to meet temperature targets. Baseline data collected by PLP contractors (PLP 2011) for the purposes of developing and applying surface-water temperature models would be useful for managing flows and temperatures to minimize impacts on aquatic life.

8.3.2 Exposure-Response

Water temperature controls the metabolism and behavior of salmon, and, if temperatures are stressful, fish can be more vulnerable to disease, competition, predation, or death (McCullough et al. 2009). Recognizing the importance of water temperature to healthy salmon populations, the State of Alaska requires that maximum water temperatures not exceed 20°C at any time, with specific maximum temperatures for migration routes and rearing areas (15°C) and spawning areas and egg and fry incubation (13°C). For all other waters, the weekly average temperature may not exceed site-specific requirements needed to preserve normal species diversity or to prevent the appearance of nuisance organisms (ADEC 2012).

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 (Section 3.8) (Bryant 2009). Elevated summer temperatures are a management concern due to potential adverse effects including increased risk of direct mortality, disease, elevated metabolic costs, and altered community interactions. Sockeye salmon are particularly sensitive to high temperatures during spawning, being limited to temperatures between 2°C and 7°C (Weber Scannell 1991). Summer, however, is not the only period during which salmon are sensitive to temperatures

(Poole et al. 2004). Salmon and other native fishes in the mine scenario watersheds rely on suitable temperature regimes to successfully complete their life cycles (Quinn 2005). The period of salmon egg incubation in gravels can be particularly sensitive to temperature changes, and changes of just a few degrees Celsius in winter mean temperature can change emergence timing of young salmon by months (Figure 3-19) (Brannon 1987, Beacham and Murray 1990, 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).

8.3.3 Risk Characterization

Stream temperatures in the mine scenario watersheds could be substantially altered due to changes in streamflow, sources of streamflow (e.g., relative importance of groundwater versus WWTP contributions), or other changes to the heat balance of WWTP discharges. We expect treated water returned to streams would have different thermal characteristics than water derived from groundwater sources (the dominant water source prior to mining). The extent and duration of temperature effects would depend not only on source water temperatures, but also on the quantity and timing of water contributed from various additional sources, such as tributaries and groundwater inputs. Simple mixing models can be used to estimate stream temperatures below the confluence of multiple sources with known temperatures and discharges. However, we do not use such models here, because we cannot account for all sources of heat transfer. In the absence of models, we have relied on available literature to identify the most likely risks to fish associated with deviations from current thermal regimes in the Pebble deposit area.

Interception of groundwater that is collected then released as a point-source through a WWTP would alter the ways in which groundwater feeds stream channels through dispersed and complex pathways. Groundwater–surface water interactions in streams can create thermal heterogeneity, enhancing the diversity of habitats available to fish (Power et al. 1999). Migration, spawning, and incubation timing are closely tied to seasonal water temperatures. Diversity of thermal habitats can allow 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 salmon populations from climatic events or other environmental changes that may adversely affect a particular run (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). Depending on the degree to which adaptation and compensatory strategies may mitigate thermal effects on life-history development and spawning timing, deviations from the thermal regime to which local populations of salmon may be adapted could have serious population-level consequences (Angilletta et al. 2008).

The volume of water that would require treatment ranges from roughly 10 to 51 million m^3/yr across the three mine scenarios (Tables 8-1 through 8-3). To avoid or minimize risks associated with altered thermal regimes in downstream effluent-receiving areas, capacity for thermal control of effluent would be required to maintain natural thermal regimes or temperatures required by regulatory agencies. Water temperature modeling is being used by PLP to assess thermal characteristics of streams in the Pebble deposit area (PLP 2011: Chapter 15, Appendix 15.1E) and could provide additional guidance for establishing a temperature management plan for the WWTP.

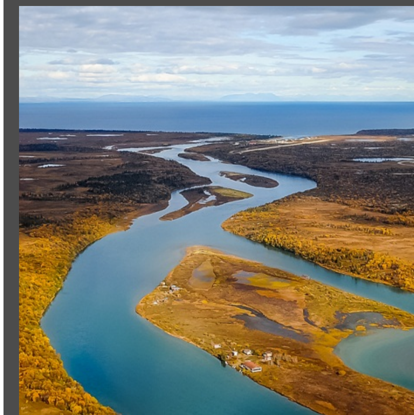
8.3.4 Uncertainties

The temperature of waters discharged from the mine, whether directly from the WWTP or indirectly through changes in groundwater or surface-water runoff, would be influenced by a number of factors controlling heat exchange that cannot be known with confidence at this point. Likewise, the influence of these discharges on stream temperatures 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 temperature due to changes in groundwater–surface water interactions in the mine area was not attempted for this assessment. Local geology and stream hydrographs are indicative of systems that are largely driven by groundwater. Disruptions or changes to groundwater flowpaths and mechanisms of thermal exchange 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 WWTP effluent with a novel thermal regime. Given the high likelihood of complex groundwater–surface water connectivity in the mine area, predicting and regulating temperatures to maintain key ecosystem functions associated with groundwater–surface water exchange would be particularly challenging.

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 large, and the cumulative risks (and likely instantaneous consequences) of facility accidents, failures, and human error would increase with time. Additionally, climate change and the predicted increases in water surplus for the region (Section 3.8) will result in potential changes in streamflow magnitude and seasonality, requiring adaptation to potentially new water management regimes for the water processing facilities. We know of no precedent for the long-term management of water temperature on this scale at a mine.

Finally, whereas the bioenergetics of the endpoint fish species are relatively well known, how these species would respond to changes to thermal regimes is poorly understood—particularly with regard to sublethal effects, behavior, adaptation, effects of fitness on the population, and other effects ranging from the molecular to the ecosystem level (McCullough et al. 2009). The existing information consists largely of field studies of salmonid distributions with respect to temperatures, supplemented by laboratory studies of development, growth, and survival at controlled temperatures. Monitoring studies to help confirm relationships between temperature alterations of various magnitudes and durations and population consequences are desirable.



CHAPTER 9. TAILINGS DAM FAILURE

In this chapter, we describe risks to stream habitats and salmonid populations from potential failures of tailings storage facility (TSF) dams. Specifically, we consider tailings dam failures at TSF 1 and potential physical and toxicological effects on fish and fish habitat (Figure 9-1). Similar types of effects would occur following tailings dam failures at TSF 2 or TSF 3, or at TSF locations in other parts of the Bristol Bay watersheds, although the specifics of a failure at these locations would differ.

A breach of a TSF 1 dam would result in a flood wave and subsequent tailings deposition that would greatly alter the downstream channel and floodplain (Figure 9-1). The initial flood wave for the tailings dam failure scenarios modeled here would far exceed the typical flood event currently experienced in these watersheds. The flood itself would have the capacity to scour the channel and floodplain and alter the landscape, and the amount of tailings that could discharge from the TSF could bury the existing channel and floodplain system with meters of fine-grained tailings material. The existing channel and floodplain would be eliminated and a new channel form would develop in the resulting topography. Given the size of these new fine-grain deposits, sediment would be highly mobile under typical streamflow events and channel form would remain unstable. Sediment deposited on floodplains and remaining behind the breached dam would create a concentrated source of highly mobile material that does not currently exist in the mine scenario watersheds. Although a sediment transport study would be required to quantify the temporal and spatial extent of effects, it is likely that the sediment regime of the affected stream and downstream waters would be greatly altered, and that the existing and well-defined gravel-bed stream would be transformed to an unstable, silt- and sand-dominated channel.

Remediation is possible following a tailings spill, but the occurrence and effectiveness of these measures would be uncertain. A tailings spill would be flowing into a roadless area with 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 removal and disposal of excavated tailings. If the operator was no longer present at the site or was no longer in existence, the response would, at best, be delayed further. 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 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 would never be complete.

9.1 Tailings Dam Failures

9.1.1 Causes

A tailings dam failure occurs when a tailings dam loses its structural integrity and releases tailings material from the impoundment. Released tailings flow under the force of gravity as a fast-moving flood that contains a dense mixture of solids and liquids, often with catastrophic results. This flood can contain several million cubic meters of material traveling at speeds in excess of 60 km/hour. At dam heights ranging from 5 to 50 m—substantially less than the 92-m and 209-m tailings dam failures considered here—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 9-1), involving dams that were significantly smaller than those considered in our mine scenarios.

BOX 9-1. EXAMPLES OF HISTORICAL TAILINGS DAM FAILURES

The examples below illustrate the characteristics and potential consequences of a tailings dam failure. Details of the design, construction, or operation of any tailings dams constructed for mines in the Bristol Bay watershed would not be the same as these mine tailings dams. However, these examples demonstrate that tailings dam failures can occur and illustrate how these failures may affect downstream areas. In addition, the dams in these failure examples were significantly smaller than the dams in our mine scenarios.

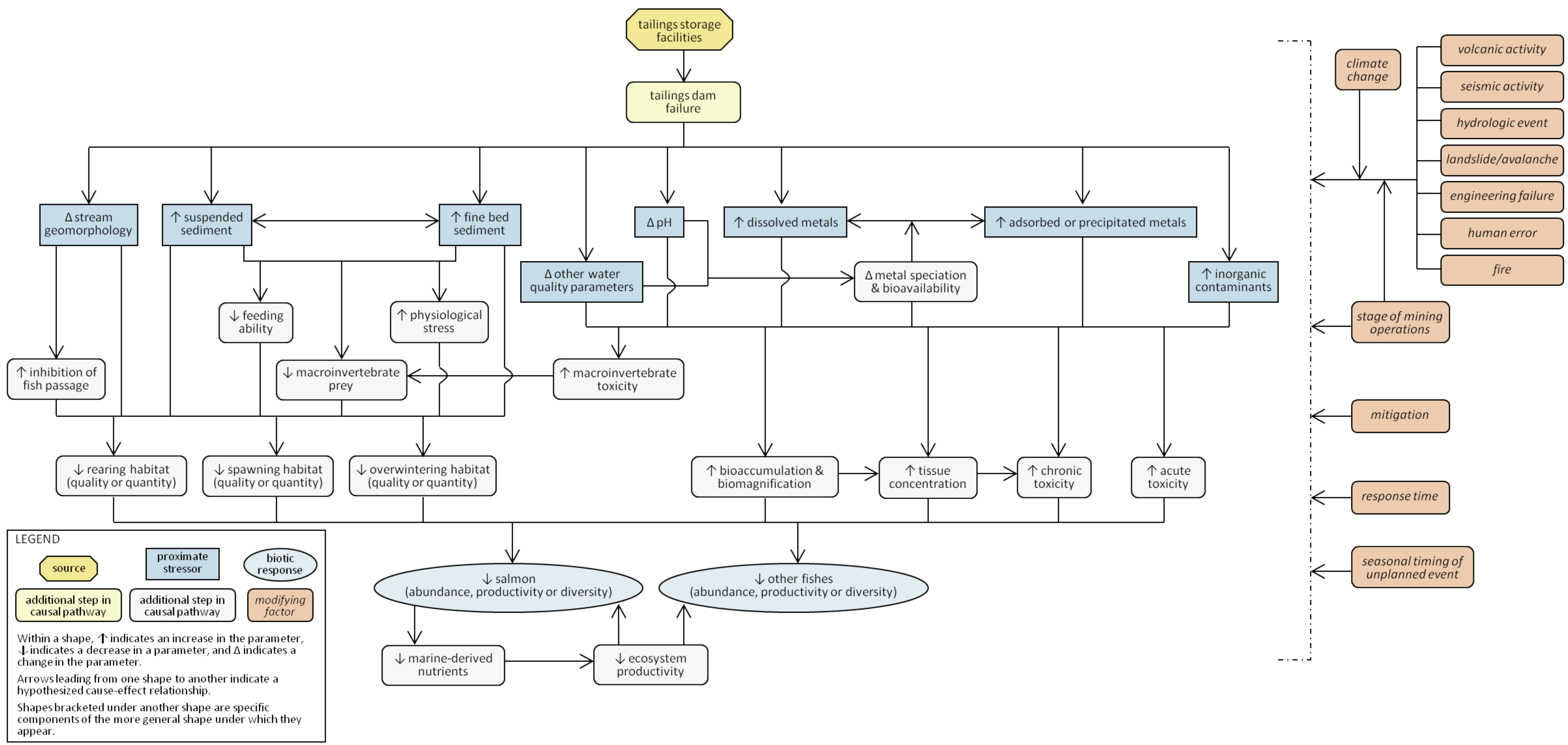
Stava, Italy, 1985. Two tailings impoundments were built, one upslope from the other, in the mountains of northern Italy. The upslope dam had a height of 29 m; the downslope dam had a height of 26 m. A stability failure of the upper dam released tailings, which then caused the lower dam to fail. The 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 approximately 5 minutes. The failure killed 269 people (ICOLD 2001).

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).

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 cyanide and heavy metal concentrations. Heavy rains and a sudden thaw caused overtopping of the embankment, cutting a 20- to 25-m breach and releasing 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 fish kill 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, USA, 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 9-1. Conceptual model illustrating potential pathways linking tailings dam failure and effects on fish endpoints. Not all potential pathways are analyzed in this assessment.



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

- **Overtopping.** Overtopping occurs when sufficient freeboard (the distance between the top of a dam and the impounded water level) is not maintained and the water level behind a dam rises due to heavy rainfall, rapid snowmelt, flooding, or operator error.
- **Slope instability.** A slope instability failure occurs when shear stresses in a 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.** Earthquake-induced shaking (Box 9-2; Section 3.6) causes additional shear forces on a dam that can lead to a slope instability failure.
- **Foundation failure.** Weak soil or rock layers and high pore pressures below the base of a dam can lead to shear failures in the foundation, causing the entire dam to slide forward or rotate out of position.
- **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 chance of overtopping.
- **Subsidence.** If a tailings dam is built on compressible soils or overlies cavities such as underground mining works (Box 4-4), 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 chance of overtopping.

BOX 9-2. 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. This 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 of 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. For a dam in Alaska with a Class II hazard potential, 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 would result. All critical structures such as tailings dams must be designed to resist the effects of the MDE, so underestimating the MDE could increase the risk of a catastrophic tailings dam failure. 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.

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 seismologic and geologic evidence and interpretations. Design engineers sometimes use the MCE to represent a floating earthquake (i.e., an earthquake not associated with a known fault) located directly under a critical structure.

The return periods stated in Alaska dam safety guidance are inconsistent with the expected lifetime of a tailings dam for a porphyry copper mine developed in the Bristol Bay watershed, and represent a minimal margin of safety. The mine scenarios evaluated in this assessment represent approximately 25 to 78 years of mineral extraction, with additional long-term operations likely required for closeout and maintenance of the mine. This period is barely within the minimum OBE return period for Class II dams. The MDE analysis presents a potentially greater chance of underestimating the size of a characteristic earthquake. Tailings storage facilities would 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 during the period over which the tailings dams would be in place.

The Initial Application Package for Approval to Construct a Dam submitted by Northern Dynasty Minerals (NDM) to the Alaska Department of Natural Resources (NDM 2006) included a seismic safety and design analysis prepared by Knight Piésold Consulting that identified the following design criteria for the tailings dams at the 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. In the application, NDM reports that the preliminary design incorporates additional safety factors, including design of storage facility embankments to withstand effects of the MDE and a distant magnitude 9.2 event (NDM 2006). Ghaffari et al. (2011) state that an MCE of magnitude 7.5 with 0.44g to 0.47g maximum ground acceleration was used in the stability calculations for the tailings dam design. Although the design specifications proposed by Ghaffari et al. (2011) exceed the minimum requirements for dams in Alaska, the deterministic dataset used is small and contains considerable uncertainties, which could lead to an underestimate of potential seismic risks.

A number of studies have attempted to analyze the historical record to determine proximate causes and probabilities of tailings dam failures (Davies et al. 2000, ICOLD 2001, Davies 2002, 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. The National Inventory of Dams (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 incidents (events potentially leading to failure) and failures (events in which dams stop retaining tailings as designed) that occurred from 1917 through 2000 (ICOLD 2001). Causes of incidents and failures were reported for 220 of these, comprising 85 incidents and 135 failures. Causes of the 135 reported failures are summarized in Table 9-1.

Perhaps most noteworthy is the relatively high number of failures at active versus inactive tailings dams, primarily resulting from slope instability and failure (Table 9-1). This suggests that the stability of tailings dams and impoundments may increase with time, as dewatering and consolidation of tailings occurs and additional loads are no longer applied. However, failures do occur after operation. For example, rehabilitation of the Gull Bridge Mine in Newfoundland, Canada, occurred in 1999. In 2010, an inspection found that the tailings dam at the closed mine was deteriorating (Stantec Consulting 2011), and in 2012 the dam failed, leaving a 50-m gap the height of the dam (Fitzpatrick 2012). The primary cause of failure for inactive tailings dams is overtopping, which accounts for 80% of recorded failures with known causes (Table 9-1).

Table 9-1. Number and cause 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 failures for which causes were reported, from 1917 to 2000.
Source: ICOLD 2001.

9.1.2 Probabilities

It is difficult to estimate the probability of low-frequency events such as tailings dam failures, especially when each tailings dam is a unique structure subject to unique loading conditions. In addition, failure probabilities may be estimated and interpreted in different ways (Box 9-3).

BOX 9-3. INTERPRETATION OF DAM FAILURE PROBABILITIES

There are two fundamental types of probability interpretations: frequentist and subjectivist.

Frequentist probabilities are based on observed frequencies of past events. For example, based on the observed frequency of tailings dam failures (88 in 176,000 dam-years, where dam-year is the existence of one dam for one year), we estimate a frequency of 1 failure in 2,000 dam-years (or 0.00050 failures per dam-year). In conventional risk probabilities, this means the following.

- Each year, there is a 5×10^{-4} probability of failure per dam.
- Out of 200 dams, one fails each decade on average; out of 2,000 dams, one fails each year on average.

Strictly speaking, frequentist probabilities are properties of populations. However, by extension, if there is one dam and it is typical of the population, it would be expected to fail, on average, within a 2,000-year period. This does not mean it is expected to fail 2,000 years after it is built; a failure could occur during any year. Rather, it indicates that, after 2,000 years have passed, it is more likely than not that the dam would have failed (i.e., half of a population of such dams would have failed 2,000 years after they were built), although the actual failure could occur any year in that 2,000-year window.

Subjectivist probabilities are based on degree of belief. For example, if engineers design a dam using novel methods, they cannot make use of frequencies when estimating failure risks. They may, however, use a model or best professional judgment to support a statement that the annual probability of failure is some value (e.g., 1×10^{-6}, or 1 failure in a million dam-years). As with frequentist probabilities, this does not mean that the dam is expected to fail only after a million years have passed. Because subjective probabilities are not based on frequencies, they are typically described as equivalent to betting odds—that is, the engineers would be willing to accept a bet in which, if the dam stands for a year they win $1, but if it fails they pay $1 million. Rather than present subjective probabilities of failure, designs are more commonly said to conform to standard or best engineering practices.

Despite these difficulties, several studies have calculated the frequency of past tailings dam failures, resulting in the following failure frequencies.

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

Available data do not permit reliable estimation of failure rates for different causes of failure or stages of activity. Although most failures have occurred while the tailings dams were actively receiving tailings (Table 9-1), the dam inventories do not indicate whether the thousands of dams in the inventory are active or inactive and do not include 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 failure probabilities and dam height or other characteristics questionable. For example, although the 1,448 tailings dams listed in the 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. Very few existing rockfill dams approach the size of the structures in our mine scenarios, and none of these large dams have failed.

The historical frequencies of tailings dam failures presented above may be interpreted as an upper bound on the failure probability of a modern tailings dam. Morgenstern (2011), in reviewing data from Davies and Martin (2009), did not observe a substantial downward trend in failure rates over time. However, improvements in the understanding of dam behavior, dam design, construction techniques, construction quality control, dam monitoring, and dam safety assessment would be expected to reduce the probability of failure for dams designed, constructed, and operating using more modern or advanced engineering techniques.

Silva et al. (2008) reported on over 75 earthen dams, tailings dams, natural and cut slopes, and some earth-retaining structures to illustrate the relationship between the level of engineering, the annual probability of slope failure in earthen structures, and factors of safety. They grouped projects into the following four categories based on the level of engineering applied to 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 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 9-2). Given that anadromous fish would be affected but no loss of human life is expected under the tailings dam failure scenarios, Class II would be applicable, although a mine operator might choose to exceed state requirements and meet Class I. Therefore, the tailings dams in the mine scenarios would be classified as either Hazard Class I or 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 (Section 3.6). Box 9-2 describes the selection of earthquake characteristics for design criteria.

Table 9-2. Summary of Alaska's classification of potential dam failure hazards.

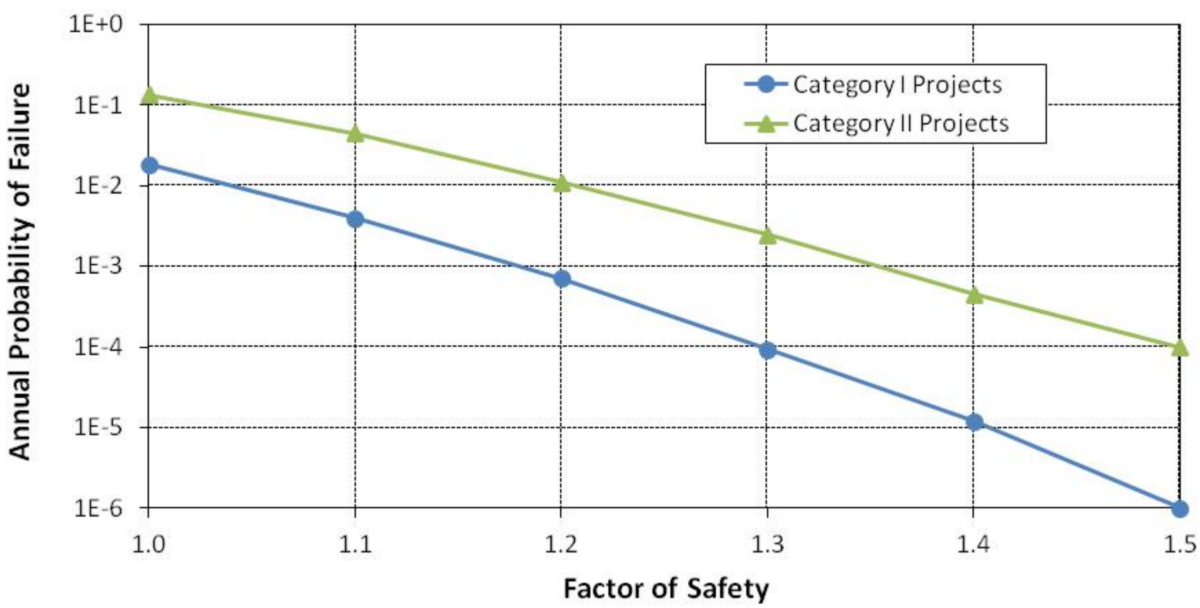
Hazard Class	Effect on Human Life	Effect on Property
I (High)	Probable loss of one or more lives	Irrelevant for classification, but may include the same losses indicated in Class II or III
II (Significant)	No loss of life expected, although a significant danger to public health may exist	Probable loss 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:
Tailings dams in the mine scenarios 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 the U.S. Army Corps of Engineers (USACE), the Bureau of Reclamation, the Federal Energy Regulatory Commission (FERC), and other agencies. Both USACE and FERC require a minimum safety factor of 1.5 for the loading condition corresponding to steady seepage from the filled storage facility (FERC 1991, USACE 2003). Based on the correlations among level of engineering, factor of safety, and slope failure probability derived from Silva et al. (2008), application of a 1.5 safety factor yields an expected annual probability of slope failure between 0.0001 (Category II) and 0.000001 (Category I) (Figure 9-2). This translates to one tailings dam failure due to slope failure every 10,000 to 1 million dam-years.

The upper bound of this range (0.0001) is lower than the historical average of 0.00050 (1 failure every 2,000 dam-years) for tailings dams. This is partly because slope failure is only one of several possible failure mechanisms, but it also suggests that some past tailings dams may have been designed for lower safety factors or designed, constructed, operated, or monitored to lower than Category II engineering standards. As shown in Table 9-1, slope failures only account for about 25% of all tailings dam failures with known causes. Thus, the probability of failure from all causes may be about four times higher than dam failures from slope instability alone (yielding an expected annual probability of failure between 0.0004 and 0.000004, or one tailings dam failure every 2,500 to 250,000 dam-years), although it is important to recognize that this small dataset may not be representative. Because 90% of tailings dam failures have occurred in active dams (Table 9-1), the probability of a tailings dam failure after TSF closure would be expected to be lower than the historical average for all tailings dams.

Figure 9-2. Annual probability of dam failure due to slope failure vs. factor of safety (modified from Silva et al. 2008).



These low probabilities are based on failure frequencies within categories of engineering practice and safety factors, but the authors describe results as "semiempirical" due to the judgment involved in categorizing the dams and creating the curves to describe the relationships (Silva et al. 2008). Modern, high earthen dams do not exist in large numbers and have not existed for long periods of time, and the frequencies and time courses of failures may differ from both the historical record and design goals. In particular, the failure rates of large earthen dams that are hundreds of years old are not known.

Given an annual probability of failure per dam-year, we can calculate the probability of the failure of any project dam over any number of years. The three mine scenarios have different numbers of dams and different operating lives: the Pebble 0.25 scenario has a single tailings dam and an operating life of 20 years; the Pebble 2.0 scenario has three tailings dams and an operating life of 25 years; and the Pebble 6.5 scenario has eight tailings dams and an operating life of 78 years. Using an upper bound annual probability of failure of 0.0004, the probability of dam failure would range from 0.8% to 22% over the operating life of each scenario (Table 9-3). This range decreases to 0.008% to 0.25% when a lower bound annual failure probability of 0.000004 is used (Table 9-3). If the tailings in the TSFs remain saturated (e.g., to keep the pyritic tailings covered with water), the potential for dam failure over a longer period needs to be considered. The probability that any of the dams would fail during a post-closure period of 1,000 years ranges from upper bounds of 33 to 96% to lower bounds of 0.4 to 3% across the three scenarios (Table 9-3).

Table 9-3. Summary of tailings dam failure probabilities in the three mine scenarios.				
		Probability of Failure		
Time Period	Annual Failure Probability	Pebble 0.25[a]	Pebble 2.0[b]	Pebble 6.5[c]
Operational life	0.0004	0.8	3	22
	0.000004	0.008	0.03	0.25
1,000 year post-closure period	0.0004	33	70	96
	0.000004	0.4	1.2	3

Notes:
[a] Operational life of 20 years; 1 tailings dam.
[b] Operational life of 25 years; 3 tailings dams.
[c] Operational life of 78 years, 8 tailings dams.

9.1.3 Uncertainties

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. TSFs may remain in place for long periods. Most dams are created as water-holding dams with limited expected lifespans (generally about 50 years). TSFs can be drained after mine closure to reduce the probability and consequences of tailings dam failures, but draining a thick layer of fine-grained material can be difficult. In the mine scenarios, only 17 to 28% of net precipitation (depending on the TSF) would need to infiltrate into the tailings to maintain full saturation with steady-state downward flow, so draining the TSFs would require maintaining a high runoff percentage. Furthermore, if tailings ponds need to be maintained to keep pyritic tailings hydrated and isolated from oxidation, tailings dams must retain solid and liquid materials behind them in perpetuity—meaning that the dams must be maintained in perpetuity, in the face of uncertain seismic and weather events that may occur over thousands of years and have cumulative effects.

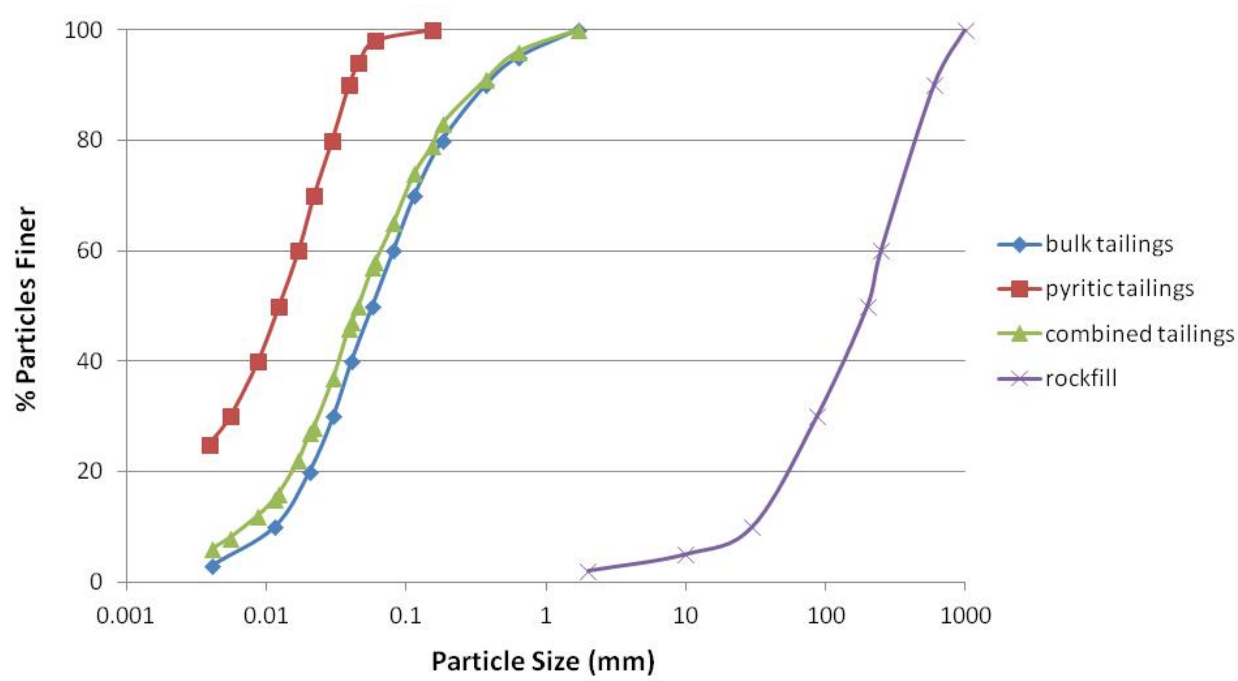
9.2 Material Properties

9.2.1 Tailings Dam Rockfill

In the mine scenarios, TSFs would be enclosed by rockfill dams constructed primarily of well-graded, non-acid-generating (NAG) 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 used to construct tailings dams typically range from sand to large boulders (Blight 2010). For a large rockfill dam with a high or significant hazard potential, lift thickness would be expected to be limited to 1.5 m to guarantee adequate compaction, which limits the maximum particle size to about 1 m (Breitenbach 2007).

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 9-3).

Figure 9-3. Representative particle size distributions for tailings solids (bulk and pyritic tailings) and tailings dam rockfill. Tailings distributions are based on particle sizes specified by Ghaffari et al. (2011) and on typical tailings particle size distributions.



9.2.2 Tailings Solids and Liquids

The tailings solids would include both bulk and pyritic tailings. The bulk tailings would consist largely of sand and silt-sized particles (D_{80} = 200 μm) and have an average dry density of 1.36 metric tons/m^3. The pyritic tailings would consist of predominantly silt-sized particles (D_{80} = 30 μm) and have an average dry 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 ore mass, respectively (Ghaffari et al. 2011). Representative particle size distribution curves for bulk, pyritic, and combined tailings are shown in Figure 9-3.

Given the dry density of the bulk tailings reported above and the specific gravity reported for the ore (2.63 for the solids) (Ghaffari et al. 2011), the bulk tailings would be 52% solids and 48% liquid by volume. The pyritic tailings, given the dry density reported above and a solids-specific gravity of 3.00 (Ghaffari et al. 2011), would be 59% solids and 41% water. Based on the proportions of bulk and pyritic tailings, the combined material in the TSF would be 53% solids and 47% 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 above the average density, although this consolidation may be limited (Section 6.3.2).

9.3 Modeling a Tailings Dam Failure

Although a tailings dam failure is a low-probability event, the probability is not zero. Should such an unlikely event occur, it is important to understand its potential impacts on the Bristol Bay watershed. In this assessment, we consider the effects of two potential dam failures at TSF 1: one at a volume approximating the complete Pebble 0.25 scenario (92-m dam height, with 158 million m^3 of tailings produced) and one at a volume approximating the complete Pebble 2.0 scenario (209-m dam height, with, 1,270 million m^3 of tailings produced). In both cases, we assumed 20% of the impounded tailings (solids and pore water) would be mobilized (Azam and Li 2010, Dalpatram 2011). Although it is reasonable to expect that 30 to 66% of the impounded tailings material could contribute to debris flow following a tailings dam failure, given the particle size distribution of the tailings (Browne 2011), we used a conservative estimate of 20% to account for the fact that the volume of material mobilized, the distance it travels downstream, and the amount of deposition can vary greatly based on numerous factors (e.g., dam height, material size distribution, material water content at time of failure) (Rico et al. 2008).

As detailed in Box 9-4, we used the USACE Hydrologic Engineering Center's River Analysis System (HEC-RAS) to model hydrologic characteristics of the dam failures. This tool requires the selection of one of two failure initiation mechanisms: overtopping or piping failure. We selected overtopping as the initiating event for final model runs for several reasons.

- The assessment TSF dam includes a liner (Section 6.1.2.4) that would reduce the risk of embankment failure due to seepage and piping (Section 9.1.1).
- Many of the failure mechanisms listed in Section 9.1.1 involve failure via breaching or overtopping, and thus are better approximated by the overtopping modeling approach in HEC-RAS.
- Overtopping could plausibly occur, for example, if storage freeboard was exceeded due to excessive precipitation, settlement over time, or a landslide or seismic event, or if any designed overflow spillway became blocked by ice or debris.

Although we modeled an overtopping failure, sensitivity analysis showed that model results were insensitive to initiation type relative to failure duration (Box 9-4)—that is, the mechanism of failure initiation did not significantly influence potential effects. The overtopping failure outputs were compared to similar piping outputs generated by subsequent HEC-RAS model runs. Comparison of peak discharges at the dam indicated that failure by overtopping generated the smallest expected flood wave peaks, and did not create a situation in which the selection of model assumptions overestimated the potential for flooding. Comparison of peak discharges was also reviewed by varying the time to full dam breach from 30 minutes to 4 hours (Gee 2008). Results indicated that magnitude of the peak flood wave was sensitive to breach development time, so we selected 2 hours as a reasonable time to full dam breach (Box 9-4).

BOX 9-4. METHODS FOR MODELING TAILINGS DAM FAILURES

We modeled hydrologic characteristics of tailings dam failures at tailings storage facility (TSF) 1 in the Pebble 0.25 and Pebble 2.0 scenarios using the U.S. Army Corps of Engineers Hydrologic Engineering Center's River Analysis System (HEC-RAS). Under both dam failure scenarios we modeled hydrologic conditions (e.g., water discharges, depths, and velocities) in the stream channel and floodplain during and immediately following dam failure, and then used these outputs to estimate tailings transport and deposition along the stream network. We limited the extent of the model to a 30-km reach downstream of the TSF (i.e., from the face of the TSF 1 tailings dam down the North Fork Koktuli River valley to the confluence of the South and North Fork Koktuli Rivers); extension of the simulation beyond this point would have introduced significant error and uncertainty associated with the contribution of the South Fork Koktuli River flows.

HEC-RAS inputs included geometry of an inline structure to simulate the dam cross-section and stream channel geometry data, both of which we derived from a 30-m digital elevation model. Flow calculations are completed between successive 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. Because HEC-RAS is most often used to simulate clear water flows, it is appropriate to increase the channel roughness coefficient (i.e., Manning's *n* coefficient) to better emulate flow characteristics of the sediment-rich water released during a tailings dam failure; thus, we used a Manning's $n = 0.09$ for analyses.

We present model outputs for overtopping failures at both the Pebble 0.25 scenario dam (92-m dam height) and the Pebble 2.0 scenario dam (209-m dam height), assuming a 2-hour failure duration and the release of 20% of available tailings storage capacity in each failure. In HEC-RAS, options for initiating a dam failure are limited to overtopping or piping failure. Both initiation types were modeled to examine sensitivity to initiation conditions. In addition, a range of dam failure durations (30 minutes to 4 hours) was examined (Gee 2008). Results showed that peak flows during a failure were much more sensitive to failure duration than to initiation type. The 2-hour duration to full failure generated peak flows that fell within the middle range of potential peak flows to consider. Overtopping generated the smallest peaks in the 2-hour simulation group (Q_{max} = 39,100 m^3/s). Piping failure in the 2-hour group was tested for failures initiating near the base of the dam, at mid-elevation, and near the top of the dam face, generating Q_{max} values of 92,263 m^3/s, 85,747 m^3/s, and 48,868 m^3/s, respectively. The 30-minute simulation group average Q_{max} values were 222% greater, and the 4-hour simulation group average Q_{max} values were 38% lower.

We assumed a particle size distribution of 0.001- 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 9-3). We focused on transport and deposition of fine-grained (less than 1.0-mm diameter) tailings material, since larger dam construction material would likely deposit within the first few kilometers downstream of the failure. 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 mobilized tailings would remain in suspension at water velocities greater than 0.05 m/s. Thus, the channel would transport tailings under typical stormflow conditions, and deposited tailings from floodplain terraces could be suspended and transported during typical storm events following the failure.

Based on historical failures, we assumed that sediment deposition would be greatest near the dam, forming an initial "wedge" that would be deposited rapidly and extend from the lowest elevation of the breach. Given the potential mobility of the fine-grained tailings, we held the initial modeled slope to 1.6%, the valley slope near the dam. We determined this slope was a reasonable estimate based on comparison with a publicly available, simple tailings flow calculator that predicts flow depths for tailings with a variety of viscosities (WISE 2012). Extending this slope from the dam breach, calculated sediment depths ranged from 1 m to 20 m 1.4 km downstream of the dam for both failure scenarios. We modeled that, on average, approximately 1 m of deposited tailings would remain on valley surfaces (i.e., in the channel and on the floodplain) downstream of the dam following each failure; this created a conservative, uniform estimate of sediment deposition. Deposition at each cross-section at this 1 m meter 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. We assumed that the remaining sediment in the tailings dam failure flow was available to deposit at the next downstream section, and this logic was carried downstream until the end of the modeled river length was reached.

9.3.1 Hydrologic Characteristics

Model results for hydrologic characteristics of the Pebble 0.25 and Pebble 2.0 tailings dam failures are shown in Table 9-4. In both cases, estimated peak flows during a TSF dam failure would be much larger than streamflows typically experienced in this watershed. This is because the impounded tailings would create a flood wave far larger than any that could result from a precipitation event alone. The tailings dam failure and subsequent release of massive quantities of impounded tailings and associated pore water would produce a peak flood immediately downstream of the dam. Maximum depths of the flood wave would exceed 10 m and 25 m, with peak velocities of approximately 4 and 10 m/s (14 and 36 km/hr), for the Pebble 0.25 and Pebble 2.0 dam failures, respectively (Table 9-4).

Peak discharges would exceed 5,000 m^3/s for the Pebble 0.25 dam failure and 39,000 m^3/s for the Pebble 2.0 dam failure. If the failure occurred during an intense rainfall or rapid snowmelt event discharges would be negligibly higher, due to the small watershed area of the TSF 1 dam. A dam-break flood of this magnitude would dwarf the peak flows of even the largest rivers in the region. For example, a local flood event measured by a U.S. Geological Survey (USGS) gage on the Nushagak River located near the village of Ekwok, Alaska (Figure 2-4), experienced a record peak flood of 3,313 m^3/s. Peak flows predicted in the North Fork Koktuli River valley from the TSF dam failures would be more than 1.6 times (Pebble 0.25) and 11.8 times (Pebble 2.0) greater than the flood of record on the Nushagak River at Ekwok. Although we recognize that these are not analogous watersheds, this observed flood does provide a point of reference for the flood magnitudes that would result from tailings dam failures.

9.3.2 Sediment Transport and Deposition

Dam failure flood waves (Table 9-4) and post-failure recessional flows in the Pebble 2.0 failure scenario suggest that transport and deposition of tailings material could occur throughout (and beyond) the 30-km modeled reach (Table 9-5). Deposition in the Pebble 0.25 failure scenario could extend for over 29 km, to within 1 km of the confluence with the South Fork Koktuli (Table 9-5). After the initial deposition event, concentrated channel flows and floodplain conveyance areas would continue to transport sediment further downstream, as channel and valley morphology would re-establish in the newly deposited substrate.

Even with only 20% of impounded tailings mobilized, the flood wave and tailings deposition that would result from a tailings dam failure under either dam failure scenario could significantly alter the downstream channel and floodplain. The initial flood itself would have the capacity to scour the channel and floodplain, as the wave of tailings slurry would travel down the valley at velocities of up to approximately 10 m/s (Table 9-4). 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 in an initial wedge near the dam; this material would move across the downstream valley as the flood wave receded and water velocities slowed (Box 9-4, Tables 9-4 and 9-5). The sediment regime of the affected stream and downstream waters would be greatly altered, with calculated sediment depths of up to 20 m (Pebble 2.0) and 1 m (Pebble 0.25) extending 1.4 km downstream of the dam.

Table 9-4. HEC-RAS model results for the Pebble 0.25 and Pebble 2.0 TSF dam failure analyses. Values were modeled for more than 39 river stations along a 30-km length of the North Fork Koktuli River; representative river stations along that length are shown here, listed by distance upstream from the confluence of the South and North Fork Koktuli Rivers (River Station 30 km = foot of the TSF 1 dam).

River Station (km)	Pebble 0.25 Dam Failure[a]					Pebble 2.0 Dam Failure[b]				
	Discharge (m^3/s)	Depth (m)	Velocity (m/s)			Discharge (m^3/s)	Depth (m)	Velocity (m/s)		
			LFP	CH	RFP			LFP	CH	RFP
30.0	5,270	11.2	1.6	3.1	1.4	39,100	23.6	4.1	7.2	4.2
29.0	5,270	10.7	1.2	4.2	1.4	39,100	19.8	5.2	10.5	5.8
25.0	4,990	9.8	0.4	0.8	0.6	39,000	20.2	1.3	2.1	1.4
20.4	4,190	7.5	0.5	0.9	0.5	38,000	19.0	1.5	2.5	1.5
15.5	3,610	10.0	0.8	1.3	0.7	34,900	25.8	1.5	2.6	1.8
10.3	2,940	12.3	0.8	1.6	0.7	29,800	27.2	2.5	4.2	2.3
4.8	2,650	4.1	0.3	0.5	<0.1	25,800	10.1	0.9	1.4	0.5
0.0	1,710	5.8	0.1	0.4	<0.1	18,600	10.6	0.5	0.9	<0.1

Notes:

[a] Dam height = 92 m, maximum volume of tailings and water expected to be stored = 158 million m^3.

[b] Dam height = 209 m, maximum volume of tailings and water expected to be stored = 1,270 million m^3.

HEC-RAS = Hydrologic Engineering Center's River Analysis System; LFP = left floodplain; CH = channel; RFP = right floodplain; TSF = tailings storage facility.

Table 9-5. Tailings mobilized and deposited in the Pebble 0.25 and Pebble 2.0 dam failures analyses. The mobilized tailings include material within the dam cross-section that has failed, plus a percentage (5 to 20%) of the stored tailings material. See Box 9-4 for additional information on how the dam failures were modeled.

Failure Scenario	Volume of Tailings[a] (million m^3)	% Mobilized[b]	Mobilized Tailings (metric tons)	Tailings in Transport at Downstream Extent of Model[c] (metric tons)	Downstream Extent of Wedge (km)	Downstream Extent of Expected Deposition[d] (km)
Pebble 0.25	158	20	59,724,000	0	1.4	29
		15	44,793,000	0	1.4	27
		10	29,862,000	0	1.4	24
		5	14,931,000	0	1.4	9
Pebble 2.0	1,270	20	479,682,000	350,668,000	1.4	30 (+)
		15	359,761,500	241,756,000	1.4	30 (+)
		10	239,841,000	138,981,000	1.4	30 (+)
		5	119,920,500	39,767,000	1.4	30 (+)

Notes:

[a] Maximum volume of tailings and water expected to be stored, allowing for freeboard in tailings storage facility (TSF). This volume was used to estimate metric tons of stored tailings released in a TSF dam failure, using an average tailings total density of 1.89 metric tons/m^3 and an average tailings dry density of 1.42 metric tons/m^3.

[b] 20% value was used in model; values less than 20% are shown to illustrate how weight of mobilized tailings changes with % mobilized.

[c] Weight of mobilized tailings that would remain in transport assuming 1 m of deposition in the floodplain inundation area.

[d] Measured downstream from face of dam.

Downstream of this initial sediment wedge, deposition could occur in the channel and the floodplain as peak flood discharges decreased with increasing distance downstream of the dam, water velocities returned to baseflow levels, and the potential for tailings deposition increased. In the Pebble 0.25 failure scenario, release of 20% of tailings material was sufficient to fill the entire North Fork Koktuli River valley to within 1 km of the confluence with the South Fork Koktuli River with an average depth of 1 m of tailings material (Table 9-5). In the Pebble 2.0 failure scenario, over 350 million metric tons of sediment remained available for transport and subsequent deposition beyond the end of the modeled reach, indicating that tailings would extend into the mainstem Koktuli River (Table 9-5).

Most of the deposition would be very fine material that would be susceptible to resuspension and deposition with each subsequent natural flow event. Following the dam failure, the stream channel would seek equilibrium and could remain unstable over several flow events, potentially creating a braided system in the post-failure depositional zone. As the new valley fluvial geomorphology developed over time, newly deposited materials on the floodplain, material at the base of the dam, and material that remained behind the breached dam of the TSF (if not removed or contained by corrective action) would serve as concentrated sources of easily transportable, potentially toxic material (Section 9.4).

The two possible failure scenarios presented here are well within the range of reported case histories. For example, when the parameters for the Pebble 0.25 and Pebble 2.0 dam failures were applied to runout distance equations from Rico et al. (2008), expected runout distances reached the marine waters of Bristol Bay. In our analyses, we made a simple assumption that deposition depths averaged 1 m (Box 9-4). We emphasize that our tailings dam failure scenarios reflect a range of possible outcomes, but are not exhaustive. The depth of tailings deposition on the floodplain could be higher or lower, depending on the amount of tailings mobilized and the runout distance. Based on historical tailings dam failure data, potential runout distances can range from hundreds to thousands of kilometers (Box 9-1).

9.3.3 Uncertainties

In this chapter, we have evaluated two potential dam failure scenarios, both caused by overtopping. Although our sensitivity analyses indicate that the repercussions of failure were relatively insensitive to the initial cause of the failure (Box 9-4), it is important to note that overtopping represents only one of several potential failure mechanisms (Section 9.1.1).

Also, a significant amount of uncertainty surrounds potential sediment deposition depths and downstream distributions. Valley topography, rate of the dam failure, and ultimate make-up of the flood wave sediment concentration and viscosity can affect outcomes and complicate predictive efforts. Despite the uncertainty associated with the massive quantities of sediment available and the complexities of hydraulic forces that would act on this sediment, we present reasonable post-failure sediment deposition outcomes in the two dam failure scenarios. Other outcomes are possible, but all share the common reality that massive quantities (i.e., millions of cubic meters) of tailings fines would be deposited in downstream floodplains and channels (Table 9-5).

Use of a traditional sediment transport model would likely improve estimates of sediment movement and deposition, especially as the model is extended further downstream. 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 South and North Fork Koktuli Rivers (Box 9-4).

9.4 Scour, Sediment Deposition, and Turbidity

The Pebble 0.25 and Pebble 2.0 tailings dam failures described in the preceding section could have devastating effects on aquatic life and habitat (Figure 9-1). We identified several processes associated with a tailings dam failure that would pose risks to aquatic habitat. These processes include exposure to hydraulic scour and bed mobilization, deposition of tailings fines, and mobilization and suspension of tailings fines affecting downstream water and habitat quality. Effects of suspended sediments are discussed in Section 9.5.1, and effects associated with potential toxicity are discussed in Section 9.5.2.

Natural background conditions indicate the sediment levels that support the region's current productivity of salmonid populations, and two available sources provide data on substrate size distribution and fine sediment concentrations in the study area. Pebble Limited Partnership (PLP) (2011) reports concentrations of fine sediments from sieved bulk gravel samples collected at three known salmon spawning sites in the South and North Fork Koktuli Rivers and Upper Talarik Creek (sample locations are shown in report by 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 River site (gage SGSK3) (PLP 2011: Figure 27 in Appendix 15.1F), which had nearly 8% fines. The geometric mean grain size was greater than 15 mm at all sites for both sampling periods, except the uppermost Upper Talarik Creek site (gage SGUT3), where the mean grain size for both seasons was between 10 and 15 mm (PLP 2011: Figure 26 in Appendix 15.1F). 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 also available for 77 wadeable stream sites around the Nushagak and Kvichak River watersheds, including one site each on the South and North Fork Koktuli Rivers and Upper Talarik Creek (Table 9-6). Substrate sampling at these study sites followed U.S. Environmental Protection Agency (USEPA) methodology (Peck et al. 2006), in which five particles are systematically selected across each of 21 evenly spaced transects (from each wetted margin of the channel and three locations in between). These data indicate that a mix of substrate sizes occurs in these streambeds, with cobble and gravel generally abundant (Table 9-6).

Table 9-6. Sediment size distributions surveyed at the South and North Fork Koktuli Rivers, Upper Talarik Creek, and 77 wadeable stream sites in the Nushagak and Kvichak River watersheds. Values represent percentage areal coverage based on 105 systematically selected particles at each site, following U.S. Environmental Protection Agency methods. All data were collected during June.

River or Stream(s)	Date	Latitude	Longitude	% Large Boulder (>1000 mm)	% Small Boulder (250–1000 mm)	% Cobble (64–250 mm)	% Coarse Gravel (16–64 mm)	% Fine Gravel (2–16 mm)	% Sand (0.06–2 mm)	% Fines (<0.06 mm)
South Fork Koktuli River	6/8/2010	59.83047	−155.27719	-	3	3	55	16	0	23
North Fork Koktuli River	6/6/2009	59.84033	−155.71272	-	-	17	49	24	10	-
Upper Talarik Creek	6/13/2011	59.91820	−155.27771	-	2	30	29	13	24	2
77 streams	2008 to 2011	-	-	0 (±1)	2 (±4)	13 (±13)	40 (±15)	17 (±12)	17 (±11)	-

Notes:
Dashes (-) indicate values equal to zero.
Sources: Rinella pers. comm., Peck et al. 2006.

9.4.1 Exposure through Sediment Transport and Deposition

The tailings dam failure scenarios evaluated here would result in intense scour and extensive deposition in the North Fork Koktuli River valley. Deposition would extend from the tailings dam downstream for many kilometers. Even with our conservative assumption that 20% of the tailings would be released, deposition would extend to within 1 km of or beyond the confluence with the South Fork Koktuli River, a distance of approximately 30 km (Table 9-5). This volume of available fine tailings material could result in many meters of deposition in a sediment wedge across the entire valley near the TSF dam, with lesser thicknesses of fines deposited to the confluence with the South Fork Koktuli River or beyond. Erosion and transport of fines would be expected to continue as the channel adjusted to the vastly increased fine sediment supply.

To translate these tailings dam failures into effects on aquatic habitat and biota, we assumed that the calculated velocities during the tailings dam failure flood event (Table 9-4) and the associated transport and deposition of tailings material and collected debris (Table 9-5) would result in a reworking and mobilization of the existing North Fork Koktuli River channel bed and banks downstream of the TSF. Given the volume of material that would be exported from the TSF, we assume that portions of the new valley floor would be predominantly tailings material, with 70% of the particle mass being finer than 0.1 mm. Following recession of the tailings dam failure flood event, we assume that the bed, margins, and floodplain would be primarily tailings material, with incorporated coarser dam fill and valley fill material accounting for less than 20%.

Immediately following either a Pebble 0.25 or Pebble 2.0 tailings dam failure, suitable spawning and rearing habitat for salmon and other native fishes would be eliminated in the North Fork Koktuli River downstream of the tailings dam. Tributaries of the North Fork Koktuli River, including portions of the watershed upstream of the confluence of the North Fork Koktuli River tributary containing the TSF, could also be adversely affected. Temporary flooding of tributary junctions during the tailings dam failure event, and subsequent sediment deposition at confluence zones causing local aggradation, steepening, or shallowing of tributary confluences, could make movement of resident and anadromous fish between tributaries and the mainstem difficult. Recovery of channel dimensions and substrate size distributions suitable for salmonid 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.

The type, magnitude, and frequency of channel adjustments that would occur in the North Fork Koktuli River valley following a tailings dam failure would depend on available sediment, channel slope, and discharge. Post-failure streams flowing across the depositional zone would have tremendous supplies of fine-grained sediments in the channel bed and banks available for transport. Channels would likely experience rapid channel incision with frequent bank failure, followed by periods of channel widening and aggradation interspersed with episodic channel avulsion. Given the volume and depth of deposition, stream channels would likely remain unstable and continue to contribute sediments to downstream reaches until equilibrium conditions were met. Recovery of suitable structural habitat in the North Fork

Koktuli River watershed would likely take decades, given the volume of sediment that could be delivered in the tailings dam failures considered here. Whether the benefits of removing spilled tailings fines would outweigh the risks of additional adverse impacts resulting from dredging and removal operations would depend on the nature and distribution of the tailings spill, the duration of risks, and existing technologies (e.g., Wenning et al. 2006).

The tailings dam failure scenarios evaluated here would have the potential to fill the North Fork Koktuli River valley with extensive deposits of tailings fines and, in some cases, still carry a substantial volume of fine sediments farther downstream. The mass of material remaining in transport at the confluence of the South and North Fork Koktuli Rivers following a Pebble 2.0 tailings dam failure, and thus available for deposition in the mainstem Koktuli, Mulchatna, and Nushagak Rivers could exceed 350 million metric tons (Table 9-5). In addition, some of the remaining stored tailings material could mobilize as pore water seeped from the exposed slopes immediately following the failure event, creating slides and smaller flow events. Fine sediment could also be mobilized during any subsequent precipitation or snow melt runoff events that would direct water across the tailings and down valley through the breach before it was repaired. The depth and distribution of fines in the mainstem Koktuli, Mulchatna, and Nushagak Rivers cannot be estimated at this time, but given the volume and grain size of these sediments, it is reasonable to expect that continued pulses of fine sediments would be transported through and transiently stored in these mainstem rivers during spring snow melt and fall rain events for many years (Knighton 1984).

9.4.2 Exposure-Response

9.4.2.1 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 small amounts of fines (exceeding 5% fines or 13% sands and fines) in streambed sediments were associated with declines in sediment-sensitive aquatic vertebrates, including salmonids. The tailings dam failures evaluated here would completely scour and transport or bury existing substrates in the North Fork Koktuli River valley under 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 fine sediments during precipitation events, providing new inputs of fines during spawning and egg incubation. Thus, exceedance of fine sediment standards in the entire North Fork Koktuli River would be a likely outcome for years to decades.

Interstitial spaces used by juvenile salmonids for overwintering and concealment 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 initially would be eliminated by the tailings dam failure, and then subject to continued high levels of embeddedness as new channels eroded into the new valley fill composed of tailing fines. The new sediment regime in the North Fork Koktuli River and associated transport and storage of massive quantities of fine sediments would essentially eliminate interstitial habitat for years to decades, if not longer. 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). Because of these habitat changes, suitable spawning environments and overwintering habitats for salmonids would be greatly diminished in this watershed, likely leading to severe declines in salmonid spawning success and juvenile survival (Wood and Armitage 1997).

9.4.2.2 Invertebrates

Aquatic macroinvertebrates are an important food source for Chinook and coho salmon, rainbow trout, Dolly Varden, Arctic grayling, and other fishes that rear in streams of the mine scenario watersheds (Nielsen 1992, Scheuerell et al. 2007). Two available data sources describe the existing macroinvertebrate communities for streams in the study area: 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, with Bogan et al. (2012) reporting 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 (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 to 11,371 organisms per m^2) (Bogan et al. 2012).

In addition to the direct impacts on fish described in Section 9.4.2.1, catastrophic sedimentation associated with tailings dam failure also would likely affect fish populations through habitat-related reductions in macroinvertebrate food resources (see Section 9.5 for discussion of toxicity-related effects). Sedimentation can affect benthic macroinvertebrates through abrasion, burial, and reduction of living space, oxygen supply, and food availability (Jones et al. 2011). Deleterious effects of sedimentation have been reviewed thoroughly (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 in the tailings dam failure scenarios (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 would occur in the tailings dam failures considered here, would certainly lead to reductions in the biomass and diversity of macroinvertebrate prey available to fish populations.

9.4.3 Risk Characterization

The complete loss of suitable salmonid habitat in the North Fork Koktuli River in the short term (less than 10 years), along with the likelihood of very low-quality spawning and rearing habitat in the long term (decades), would likely result in near-complete loss of North Fork Koktuli River 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 provides complex, low-gradient, high-quality habitats that currently support spawning and rearing populations of sockeye, Chinook, and coho salmon, and spawning populations of chum salmon (Figures 7-2 through 7-5) (Johnson and Blanche 2012). For example, aerial index surveys in the North Fork Koktuli River documented roughly 3,000 Chinook salmon (surveyed in 2005), 2,100 sockeye salmon (surveyed in 2004), 1,750 coho salmon (surveyed in 2008), and 1,400 chum salmon (surveyed in 2008) (values inferred from figures in report by PLP [2011: Chapter 15]). The North Fork Koktuli River also supports rearing Dolly Varden and rainbow trout (Figures 7-7 and 7-8) (ADF&G 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). Total Chinook salmon run-size estimates for the Nushagak River include estimates of harvest plus escapement of spawners. Estimates based on a variety of techniques, including sonar, averaged over 190,000 Chinook salmon from 2002 through 2011 (Buck et al. 2012), making the Nushagak the largest producer of Chinook salmon for the Bristol Bay region. Of all the Chinook salmon tallied during annual aerial index counts in the Nushagak River watershed between 1969 and 1985 (years that all reported spawning areas were surveyed), on average 29% (range 21 to 37%) were counted in the Koktuli River system (Dye and Schwanke 2009) (see Section 7.1.2 for a discussion of the limitations of abundance estimates based on aerial counts). The Mulchatna River accounts for another 12% (range 9 to 17%) of the Nushagak Chinook salmon count, and the Stuyahok River (which drains to the Mulchatna downstream of the Koktuli River) represents another 18% (range 10 to 27%). Hence, Chinook salmon production could be significantly degraded by loss of habitat downstream of the tailings dam, particularly if effects extended downstream into the Koktuli and Mulchatna Rivers and beyond.

Sockeye are the most abundant salmon returning to the Nushagak River watershed, with annual runs averaging more than 1.9 million fish between 2001 and 2010 (Jones et al. 2012). 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 system. The Nushagak River watershed supports two genetically and ecologically distinct groups of sockeye salmon (Dann et al. 2011): those that rear in, and spawn in and near lakes (lake-type, as in Semko 1954), and those that spawn and rear, at least briefly, in rivers and streams (sea-type and river-type, as in Semko 1954, collectively called riverine-type here). Sockeye salmon in much of the Mulchatna River system, including the Koktuli River and adjacent Stuyahok River, are riverine-type, and are more closely related to riverine-type sockeye salmon of the Kuskokwim River drainage than to Nushagak River watershed lake-type sockeye salmon (Dann et al. 2011). It is likely that these population groups share a similar life history pattern. Riverine-type sockeye in Kuskokwim River tributaries preferentially rear in off- and side-channel habitats within floodplain-prone stream reaches (Ruggerone et al. 2011). From 1995 to 2006, an estimated 528,000 adult sockeye salmon annually migrated to spawning areas in the Nushagak and Mulchatna River systems upstream of the Wood River system (Jones et al. 2012). Of these, approximately 70% (an annual average of 363,000) appear to be riverine-type sockeye salmon based on the proportion of sockeye that escaped to the Nushagak/Mulchatna portion of the basin.

Spawning and rearing riverine-type sockeye salmon habitats occur throughout the South and North Fork Koktuli Rivers downstream to and beyond the confluence of the Mulchatna and Nushagak Rivers

(ADF&G 2012). The tailings dam failures considered here would likely affect sockeye salmon production throughout the Koktuli River system, but the proportion of the total Nushagak River sockeye salmon production that would potentially be affected is unknown (see Section 7.1 for additional information on fish abundance).

Populations of resident and anadromous fishes present in North Fork Koktuli River headwaters upstream of TSF 1 or in tributaries at the time of a tailings dam failure would not immediately suffer habitat losses, 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 smolts emigrating to sea, for adult spawning migrations, or, in the case of resident species, for migration between spawning, foraging, and overwintering areas. 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, access to mainstem river habitats upon which many tributary fishes depend for portions of their life history could be blocked or severely degraded.

Successful re-colonization of the North Fork Koktuli River by resident fish would depend on whether unimpaired tributary habitats or downstream areas could function as suitable refugia and source areas for re-colonization of the North Fork Koktuli River. Resident fish 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 salmon from tributary refugia or downstream areas would require suitable passage at tributary junctions and suitable migratory corridors throughout the mainstem. Aquatic macroinvertebrate food resources would also likely be adversely affected in the main river channel, limiting rearing potential for insectivorous fishes such as juvenile salmonids. Given estimates of fine-sediment deposition and the unstable, silt and sand bed channels that would likely form across the valley floor, as well as metal concentrations in these tailing substrates that could inhibit migratory behavior (Section 9.5.2.1), successful migratory conditions seem unlikely for at least several years after a tailings dam failure.

The near-complete loss of North Fork Koktuli River fish populations downstream of the TSF and long-term transport of fine sediment to downstream locations would have significant adverse effects on the Koktuli and Nushagak River salmon, Dolly Varden, and rainbow trout populations, affecting downstream fisheries, including subsistence users (Figure 5-2). Spawning and rearing habitat would be eliminated or impaired by deposition of transported sediment and/or reductions in the invertebrate prey base. Direct loss of habitat in the North Fork Koktuli River, and impairments further downstream because of transport and deposition of sediment, could adversely affect a substantial portion of Chinook salmon returning to the Nushagak River watershed. Assuming that Alaska Department of Fish and Game (ADF&G) 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 River mainstem, and subsequent loss of access to or inhibition of migration into the South Fork Koktuli River would affect the entire Koktuli River component of the Nushagak Chinook run. If the deposited tailings material is of sufficient quantity and toxicity (Section 9.5.2) to have effects on aquatic life and fish migratory behavior in the lower Koktuli, Mulchatna, and Stuyahok Rivers, much greater proportions of the Nushagak Chinook populations and other resident and anadromous fish populations could be affected. Adult salmon returning to these rivers could potentially seek other tributaries for spawning, but successful recruitment of displaced spawners would require access to and comparable use of spawning and rearing capacity elsewhere in the Nushagak River watershed.

9.4.4 Uncertainties

It is certain that a tailings dam failure such as those evaluated here would have devastating effects on aquatic habitat and biota, but the distribution and magnitude of effects is uncertain. Uncertainties associated with the initial events, including the likelihood of dam failures and sediment transport and deposition processes are discussed in Sections 9.1.3 and 9.3.3. Uncertainties associated with the timing, feasibility, and effectiveness of remediation of a tailings spill are discussed in Section 9.6.2. Other uncertainties related to the time frame for geomorphic recovery, the longitudinal extent and magnitude of habitat impacts downstream of our modeled 30-km reach of the North Fork Koktuli River, and the fish populations affected are discussed in this section.

We estimate that recovery of suitable structural habitat in the North Fork Koktuli River and off-channel areas would likely take years to decades, given the scouring action of the flood wave and the volume of fine-grain sediment that would potentially be delivered under a tailings dam failure. However, the 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 sections of the North Fork Koktuli River valley to bedrock, which would then be buried under massive deposits of tailings fines. Recruitment of gravels and coarser substrates to the North Fork Koktuli River valley could be delayed by low supplies and/or low rates of transport from tributaries or unaffected upstream sources. Recovery may also be delayed if riparian vegetation does not recover because the tailings are toxic to plants (although this causal pathway is not considered in this assessment).

The tailings dam failure simulations (Section 9.3) were 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 South and North Fork Koktuli Rivers. Extension of the simulations beyond this confluence would introduce significant error and uncertainty associated with the contribution of South Fork Koktuli River flows, and 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. However, given the high volume of tailings fines that could be transported beyond the confluence of the South and North Fork Koktuli Rivers (Table 9-5), it is highly likely that impacts on fish habitat estimated for the North Fork Koktuli River would extend some significant distance down the mainstem Koktuli River.

We estimate that the combined effects of direct habitat losses in the North Fork Koktuli River, downstream in the mainstem Koktuli River, and beyond, as well as impacts on macroinvertebrate prey for salmon, could adversely affect 25% or more of Chinook salmon returning to spawn in the Nushagak

River watershed. If the Koktuli River, Stuyahok, and Mulchatna portion of the Nushagak runs are impacted via downstream transport of tailings fines, the tailings dam failure may affect nearly 60% of the Chinook run (on average, 59% of the aerial survey counts were from these three watersheds [range = 48% to 75%]). Uncertainty around this estimate is associated with the downstream extent of habitat impacts (described above) and the variable and imprecise estimates of the relative abundance of Chinook salmon in the Nushagak, Mulchatna, and Koktuli River systems. We based our estimate of proportions on long-term (1969 to 1970 and 1974 to 1985) aerial counts of Chinook salmon collected and interpreted by ADF&G (Dye and Schwanke 2009), but aerial counts can substantially underestimate true abundance (Jones et al. 1998).

Because long-term abundance data are lacking for most other fish species and locations in the mine scenario watersheds, losses caused by a tailings dam failure are not quantified for other species. Our analysis focuses on a few endpoint species, and does not incorporate considerations of metacommunity dynamics, which are poorly understood for the region but may be critical to understanding species responses to environmental change (Westley et al. 2010). Information documenting known occurrence of non-endpoint fish species in the region's rivers and major streams is available (Johnson and Blanche 2012, ADF&G 2012), but information on their abundances, productivities, and limiting factors is not currently available.

9.5 Post-Tailings Spill Water Quality

9.5.1 Suspended Tailings Particles

9.5.1.1 Exposure

During a tailings dam failure, aquatic biota would be exposed to a slurry of suspended tailings moving at up to 10 m/s (Table 9-4). In the Pebble 2.0 scenario, much of this material would still be flowing 30 km downstream, at the mouth of the North Fork Koktuli River (the limit of the model) (Table 9-5).

For years after a tailings dam failure, settled tailings would be resuspended and carried downstream. At first, this process would be frequent if not continuous (except when and where the substrate is frozen), as channel and floodplain structure is established by erosional processes resuspending the tailings (Section 9.4.1). Gradually, as the tailings flow downstream, a substrate consisting of gravel and cobble 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 typically 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 9.5.2.3).

9.5.1.2 Exposure-Response

Suspended sediment has a variety of effects on fish that are similar 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 higher 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 follows (derived from Newcombe and Jensen 1996):

- 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
- 3 mg/L for 7 weeks to 11 months

However, salmon may adapt to migrate through high levels of suspended sediment. For example, during mid-May to early August, when adult salmon migrate upstream through the lower Copper River (El Mejjati et al. 2010), suspended sediment concentrations range from 750 to 1780 mg/L (Brabets 1992).

9.5.1.3 Risk Characterization

During and immediately after a tailings spill, exposure to suspended sediment would be far higher than any of the effects thresholds listed above. Fish could be literally smothered and buried in the slurry. Because the standard of 1,000 mg/L of suspended sediment is exceeded by ordinary events such as erosion of construction sites and tilled fields, erosion of tailings from the re-formation of the channel and floodplain would likely exceed that standard for days at a time, over a period of years. Fish would be 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 during these periods, although salmon have adapted to migration corridors with high suspended sediment levels. The potential for tailings to be more aversive or toxic than natural suspended sediment is unknown. Exposure levels would gradually decline over time as tailings are carried downstream, channel stability increases, and the floodplain becomes revegetated. Rates of these processes are unknown, but it is reasonable to assume that decades would be required for suspended sediment loads in the Koktuli and Mulchatna Rivers to drop to levels that occur with normal high flows in stable channels of the Bristol Bay watershed.

9.5.1.4 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 reduce 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.

9.5.2 Tailings Constituents

Although the most dramatic effect of a tailings dam failure would be habitat destruction and modification due to the flow of tailings slurry downstream, exposures to potentially toxic materials in the slurry would also occur. The toxic effects of a tailings dam failure can be assessed using the composition of the tailings and of experimental tailings leachates, as well as experience with tailings spills at other sites (Box 9-5).

9.5.2.1 Exposure

Aqueous Exposures to Impoundment Waters

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 would include 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 (i.e., a failure under “dry” conditions), undiluted pore water and supernatant water would be released. If the dam was eroded or overtopped by a flooding event (i.e., a failure under “wet” conditions), the pore and surface water could be diluted by fresh water. However, this dilution would be trivial relative to the volume of pore water in the tailings.

A spill would have two phases in our tailings dam failure scenarios; other scenarios could differ in timing and magnitude. At first, tailings slurry would pour through the breach for approximately 2 hours based on the specified rates of dam erosion and slurry flow. Pore water would then drain from the tailings that are not released but are above the elevation of the breach. This latter process would be slow and could continue until the dam was repaired. If a tailings dam failure occurred after the mine site was abandoned and no corrective action was taken or was delayed, an equilibrium would be achieved in which rain, snow, and upstream flows were balanced by outflow of leachate through the breach.

BOX 9-5. BACKGROUND ON RELEVANT ANALOGOUS TAILINGS SPILL SITES

Past deliberate or accidental spills of metal mine tailings into salmonid streams and rivers have occurred by mechanisms and mining practices other than those evaluated in this assessment. However, these spills provide evidence concerning the fate of tailings and the nature of exposures to aquatic biota once the tailings are in streams and floodplains. In the United States, some of these sites are relatively well studied because their observed effects have led to classification as Superfund sites. Other tailings spills have 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 potential tailings dam failures 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 wastes. 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 River were largely limited to movement of previously released solids. It became a Superfund site in 1983. Contaminants of concern were 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 fishes continue to exhibit low growth and abundance, and intermittent fish kills have followed metal pulses from rainstorms or rapid snow melt. However, sedimentation was also thought to contribute to effects on fish populations through habitat degradation. Detailed information can be found in the responsible party's remedial investigation (ARCO 1998) and in U.S. Environmental Protection Agency (USEPA) documents (USEPA 2012a).

Coeur d'Alene River, Idaho. The Coeur d'Alene River 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 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 their 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, which was covered and seeded. In 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 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).

Once in the stream, potentially toxic constituents dissolved in the water would not settle out. Because the potentially toxic constituents are not degradable or volatile, they would eventually flow to Bristol Bay, although they would be diluted along the way. In the Pebble 2.0 tailings dam failure, the peak flow of mobilized tailings at the confluence of the North and South Fork Koktuli Rivers is estimated to be approximately 18,600 m^3/s (Table 9-4). The Nushagak River at Ekwok (Figure 2-4) would be the first downstream gaging station at which 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), concentrations of dissolved chemicals in the Nushagak River would be 96 and 94% of those in the spill. Minimum flow is not considered, because a failure is believed to be less likely during freezing conditions.

We used the tailings humidity cell test results to estimate the composition of the bulk of the aqueous phase. However, those values are uncertain, because none of the tests performed by PLP represent the leaching conditions in a tailings impoundment, material other than bulk tailings would be added to the TSFs, and no model exists to mathematically simulate the leaching process. However, some mixture of tailings supernatant, which represents the source water for the impoundment (Table 8-4); humidity cell leachate, which represents aqueous leaching from tailings under oxidizing conditions (Table 8-5); and a small amount of local water (Table 8-10) can be used to approximate aqueous phase composition.

During mine operation, 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 8-4) with some dilution by precipitation. However, those results do not include process chemicals (e.g., xanthates and cyanide) that may be associated with the supernatant but that are not quantified in this assessment. Supernatant water would be slightly diluted by rain and snow onto the surface of the impoundment, but peripheral berms should generally prevent dilution by runoff.

The waters released from a tailings spill during mine operation 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 8-4 and 8-5). Pore water from deep 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.

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 and rivers. Benthic organisms, or aquatic insects and other invertebrates that burrow into or crawl upon substrates, would be most exposed. Eggs and larvae (fry) of any salmon, trout, or char that spawned in the contaminated substrate also would be exposed. In either case, the bioavailable contaminants would be 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 waters. Shallow, turbulent water is typically near oxygen saturation. Bedload sediment bounces and slides downstream during high flows. At high enough flows sediment is suspended, exposing it to oxygen. Water also 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 (Tables 8-4 and 8-5). Ideally, a leaching test would be performed that simulated conditions in a streambed, but no such test results are available. In theory, 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 upwelling or downwelling areas, and during high-flow periods such as spring runoff and floods. However, high flows would be expected to increase leaching rates, resulting in complex dynamics (Nagorski et al. 2003).

Although we assume that spilled tailings would be mixed and would have average metal compositions (Table 9-7), stream processes would be expected to sort them. In Soda Butte Creek (Box 9-5), 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).

After a spill, aquatic biota would also be indirectly exposed to tailings deposited on land, primarily in the floodplains. Erosion of these floodplain-deposited tailings would result in additional 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 8-5 may roughly estimate leachate composition from floodplain-deposited tailings.

Table 9-7. Comparison of average metal concentrations of tailings (Appendix H) to threshold effect concentration and probable effect concentration values for freshwater sediments and sums of the quotients (∑ TU). Values are in mg/kg dry weight.

Tailings Constituents	Average	TEC[a]	TEC Quotient	PEC[a]	PEC Quotient
Ag	0.7	-	-	-	-
As	25	9.8	2.6	33	0.76
Ba	30	-	-	-	-
Be	0.3	-	-	-	-
Bi	0.6	-	-	-	-
Cd	0.1	0.99	0.10	5.0	0.02
Co	8.1	-	-	-	-
Cr	150	43	3.5	110	1.3
Cu	680	32	21	150	4.5
Hg	0.1	0.18	0.56	1.1	0.09
Mn	360	630	0.57	1200	0.30
Mo	52	-	-	-	-
Ni	68	23	2.9	49	1.4
Pb	15	36	0.41	130	0.12
Sb	1.0	-	-	-	-
Se	1.8	-	-	-	-
Tl	0.3	-	-	-	-
U	0.4	-	-	-	-
V	87	-	-	-	-
Zn	87	120	0.72	460	0.19
Sum	-	-	32	-	8.7

Notes:
Dashes (-) indicate that criteria are not available.
[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-day tests from (Ingersoll et al. 1996).
TEC = threshold effect concentration; PEC = probable effect concentration; TEL = threshold effect level; PEL = probable effect level.

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); or 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 to 8,380 μg/L) (Nimik and Moore 1991, ARCO 1998). Concentrations from spills in the Bristol Bay watershed would probably be lower than for the acidic Clark Fork tailings and salt accumulation on the surface would be less because of greater precipitation, but the same processes would occur. Dilution of leachate that moves into the stream would be highly location- and condition-specific. Once in a stream, leached metals are likely to remain dissolved because of the highly dilute water chemistry in the region, but some precipitation or sorption to clays or organic matter would occur, depending on the conditions that moved the leachate into the stream.

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 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 to 36 µg/L in depositional areas and 3 to 22 µg/L in riffles (ARCO 1998). Concentrations would differ for tailings from the Bristol Bay watershed, 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 could 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 9-7). During and after a tailings spill, there may be some sorting of the tailings by size or density that would result in locally higher metal compositions, but this variability cannot currently be quantified. Although 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. 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, 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 volume of tailings deposited in the watershed would be so large and the watershed is nearly undisturbed except for potential mine facilities. The background sediment load (1.4 to 2.5 mg/L total suspended solids, Table 8-10) is miniscule compared to the meters of tailings that would be deposited (Table 9-5). 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 8.2.2, 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 in that section. It may be applied to cases in which both direct aqueous and dietary exposures may occur, such as flow into a stream through floodplain tailings or from upwelling through tailings. Dietary exposures with respect to sediment levels may also be estimated. In such cases, 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:

Collector-Gatherer Copper = 0.294 x

Scraper-Grazer Copper = 1.73 x

where x is sediment concentration and copper 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 an average bioaccumulation factor of 1.0 (Goodyear and McNeill 1999). This implies copper concentrations in invertebrates are equal to those in sediments, which in this case are tailings with an average copper concentration of 680 mg/kg (Table 9-7).

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, this method requires measurements of AVS and SEM within 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, 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 metal 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 9-5) (Marcus et al. 2001). Sediment was still characterized by high copper concentrations after 48 years despite two 100-year floods, indicating that some tailings were retained by streams and maintained high metal levels even 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, which caused extensive damage to the watershed (Box 9-5) (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 that entered the river until 1968 (Hoiland et al. 1994, NRC 2005). At least as late as 2000, metal (cadmium, lead, and zinc) concentrations were elevated in caddisflies and were more highly correlated with sediment concentrations than with surface water concentrations, suggesting that deposited tailings were the primary source of exposure (Maret et al. 2003).

A new study has modeled future decline in sediment metals concentrations for the Clark Fork River (Box 9-5), assuming an exponential decay in concentrations over time due to loss and dilution (Moore and Langner 2012). Although there was no significant decline over time (1991 to 2009) in downstream concentrations (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 Langner (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 was predicted to reach the threshold effect concentration (TEC) in 163 years. In the Bristol Bay watershed, dilution with clean sediment would likely be slowed by denser vegetation and less land disturbance. Lower gradients in the Bristol Bay receiving streams relative to Montana would also tend to slow recovery, as recovery is primarily achieved by tailings transport downstream. It also should be noted that these time estimates are not from the date of a spill, but rather from a date decades later, when channel structure had stabilized and much of the tailings had been carried downstream.

9.5.2.2 Exposure-Response

Aqueous Chemicals

The toxic effects of exposure to a tailings spill can be estimated from aquatic toxicity data. Ambient water quality criteria and equivalent benchmarks are used to screen the metals in the two types of tailings leachates (Tables 8-4 and 8-5). Copper is the dominant contaminant in tailings leachates, and criteria values based on the biotic ligand model (BLM) (described in Section 8.2.2.1) are used as benchmarks (Table 9-8). 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 9-8. Results of applying the biotic ligand model to mean water chemistries of tailings leachates and supernatants to derive effluent-specific copper criteria.

Stream	Acute Criterion (CMC in μg/L)	Chronic Criterion (CCC in μg/L)
Tailings humidity cell leachates	4.8	3.0
Tailings supernatants	7.2	4.5

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

Note that these criteria are calculated for the water chemistry of the tailings 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, these criteria would be too high for situations in which significant dilution occurs, because uncontaminated water has lower hardness and lower levels of chemicals that reduce copper binding to biotic ligands. Hence, dilution by a factor of 2 would not quite reduce toxicity of metal-contaminated water by a factor of 2.

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 TECs and 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. Out of 347 total sediments from 17 rivers and lakes, that validation study found toxic effects in 17.7% of sediments with copper concentrations less than the TEC, in 64% of sediments with copper concentrations between the TEC and PEC, and in 91.8% of sediments with copper concentrations above the PEC (MacDonald et al. 2000). The consensus TECs and PECs are used to evaluate the potential toxicity of tailings should they become sediment following a spill, because they have the best scientific support.

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 8.2.2.1) 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 concentration survival and growth are observed to decline in multiple studies.

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 (Box 9-5), both toxicity and observed field effects on fish and invertebrates 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 cannot reasonably be applied to the tailings dam failures analyzed here. Nevertheless, the qualitative relationships are applicable.

9.5.2.3 Risk Characterization

Toxicological risks are usually judged by comparing exposure levels to a criterion or other ecotoxicological benchmark using a risk quotient (Box 8-3), which equals the exposure level divided by the ecotoxicological benchmark. If the quotient exceeds 1, the effect implied by the benchmark is expected to occur, but with some uncertainty (Box 8-3). Quotients much larger than 1 suggest larger effects than those that define the benchmark, with greater confidence that an adverse effect would occur. Quotients much smaller than 1 suggest that even small effects are unlikely. The criterion maximum concentration (CMC or acute criterion) and criterion continuous concentration (CCC or chronic criterion), and equivalent numbers when criteria are not available, are the primary ecotoxicological benchmarks used in this assessment for aqueous exposures, because they are relatively well-accepted as thresholds for significant effects. The CMC estimates a concentration at which 5% of aquatic species experience some mortality among developed life stages in short-term exposures. The CCC estimates a concentration at which 5% of aquatic species experience decreased survival, growth, or reproduction in longer-term exposures. Other benchmarks are used to indicate direct toxicity to salmonids (Tables 8-13 and 8-14).

Acute Toxic Risks

At sites closest to a failed TSF, acutely toxic effects of a tailings spill would be indistinguishable from the concurrent effects of being smothered by tailings particles. Aquatic life in 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 (Tables 8-4 and 8-5), which suggests that they would kill invertebrates. However, even the minimal dilution in the Nushagak River at Ekwok would dilute leachate from the maximum spill to the national criterion or below. 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 trout in acute exposures (93 and 188 μg/L for the humidity cell and supernatant, respectively). Hence, in a tailings dam failure, acute exposure to dissolved copper immediately downstream of the TSF would be sufficient to kill sensitive invertebrates but not salmonids, but those effects would be eclipsed by the physical effects. Downstream, where physical effects would be minimal, toxic effects would be reduced or eliminated by dilution.

Chronic Toxic Risks for Aqueous Exposure

Potential effects of chemicals leaching from tailings in streambed and riverbed sediments and associated floodplains are addressed by dividing leachate concentrations by chronic water quality criteria and standards to derive hazard quotients (exposure concentrations divided by effects concentrations). These hazard quotients can be interpreted as relative degrees of toxicity of leachate constituents or as indicators of the degree of dilution required to avoid significant toxic effects. The two estimates of tailings leachate composition give similar results (Tables 8-4 and 8-5). Undiluted leachate of both types would be expected to exceed the chronic national criterion, but not the Alaskan standard, for copper. If combined toxic effects of metals are considered (see the *Sum of Metals* line in Tables 8-4 and 8-5), 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 (CCC) imply that dilution by a factor of 2 to 4 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.

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 9-7 shows that tailings would be expected to cause severe toxic effects on the organisms that live in or on them. Notably, copper concentration would be approximately 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 8-10) and the time required to achieve that degree of dilution would be very long.

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. 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, but the result is marginal and certainly within the range of uncertainty.

Chronic Toxic Risks—Analogous Sites

Some well-documented cases indicate that adverse effects of chronic toxicity on aquatic communities in general, and on 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 in which the ecological consequences of a major metal mine tailings spill to a stream or river were studied, extensive and long-lasting toxic effects were observed. However, these tailings are likely to be more toxic than future tailings due to more efficient metal extraction in modern ore processing.

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 9-5). In the Soda Butte Creek case, the copper content of macroinvertebrates was positively correlated (r^2 = 0.80) and their taxa richness was inversely correlated (r^2 = 0.48) with sediment copper (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 for 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), and 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).

9.5.2.4 Uncertainties

All of the lines of evidence concerning risks to aquatic communities from the toxic properties of spilled tailings have notable uncertainties. In particular, the estimates from test leachates and whole test tailings underestimate risks because they do not include pyritic tailings.

Toxic Risks from Aqueous Exposures

The use of leachate and supernatant concentrations to estimate effects of a tailings spill is uncertain primarily because of issues 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 bulk tailings but not pyritic tailings. Finally, the degree of leachate dilution in the field would be highly variable and could be roughly estimated, at best. These uncertainties concerning exposure are significant in terms of both their potential size (at least an order of magnitude uncertainty) and in terms of their implications (leachates from the spilled tailings may be non-toxic or severely toxic given this uncertainty in exposure).

The exposure-response relationships for this line of evidence are also uncertain. As noted in Section 8.2.2, 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 field concentrations are much higher than test leachate and supernatant concentrations, so fish toxicity is not an important uncertainty. The uncertainty concerning exposure-response relationships is smaller than the uncertainty concerning exposure.

Toxic Risks from Sediment

Although the consensus TECs and PECs are the best available effects benchmarks for sediment, their applicability to tailings in Bristol Bay habitats is uncertain. The studies from which the values are derived include lakes, reservoirs, and other systems that differ from 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 for copper 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 (680 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, and reductions in four different community metrics occurred below the sum of TEL values (Griffith et al. 2004). This result suggests that toxicity would be even more severe than the TECs and PECs suggest, but it may be somewhat confounded by mine drainage.

Dietary Risks

Effects of dietary exposures would 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 from the PLP (2011) tests, 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, 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 range 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. Finally, the variation in results among dietary toxicity studies for copper is large, and the factors controlling dietary toxicity are poorly known.

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 9-5). A large source of uncertainty when evaluating effects at those sites is the composition of the tailings. In general, the Pebble test tailings are 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 mainstem Koktuli River receiving waters would make biota of the receiving streams more susceptible to metals than at the analogous sites. Although these cases are highly uncertain sources of information concerning the potential toxicity of spilled tailings, they can be used with confidence to identify or confirm important modes of exposure and the processes leading to exposure. They also demonstrate the persistence of tailings and the leaching of their metals for multiple decades.

9.5.3 Weighing Lines of Evidence

This risk assessment is based on weighing multiple lines of evidence, and evidence for the various routes of exposure (summarized in Table 9-9) is complex. 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). Evidence scored as positive (+) supports the case for adverse effects, whereas evidence scored as negative (–) weakens the case for adverse effects. A zero (0) score indicates no or ambiguous evidence.

- 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. For example, if the predicted concentration of copper is twice the median lethal concentration (LC_{50}) for rainbow trout, that is evidence of acute lethality; if it is 10 times as high, that is stronger evidence. In Table 9-9, zero signifies a low quotient, + a moderate quotient, and ++ a high quotient.

- **Quality** is a complex concept that includes conventional data quality issues, but in this case, the primary determinant is the relevance of the evidence to the mine scenarios. Because this is a predictive assessment, none of the evidence is based on observations of an actual spill at the site of concern. Hence, the evidence is based on the tailings dam failure scenarios, 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 9-9 are not a substitute for the actual evidence, but rather are intended to remind the reader what evidence is available and summarize the strength and quality of the different lines of evidence.

Table 9-9. Summary of evidence concerning risks to fish from the toxic effects of 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 one or more +, 0, or - symbols) on three attributes: logical implication, strength, and quality. Here, all lines of evidence have the same logical implication, since 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 the mine scenarios.

Route of Exposure Sources of Evidence (Exposure/E-R)	Logical Implication	Strength	Quality		Results
			Exposure	E-R	
Suspended sediment Assumption/synthesis of laboratory and field studies	+	++	0	+	Adverse effects on fish are certain. Although the exposure level is unknown, it would clearly be at effective levels.
Acute aqueous exposure Leachate measurements/laboratory-based criteria	+	0	0	+	Acute lethality to invertebrates close to tailings storage facilities, but not downstream and not to fish.
Chronic aqueous exposure Leachate measurements/laboratory-based criteria	+	0	0	+	Chronic toxic effects on invertebrates due to *in situ* leachate, but effects would end after some years if diluted by clean sediment.
Chronic sediment exposure Tailings measurements/sediment guidelines	+	++	+	0	High likelihood of toxic effects on invertebrates or fish based on a summary of field studies.
Chronic dietary exposure Tailings measurements and bioaccumulation factors/mean of laboratory-based effects levels	+	0	+	0	Marginal dietary copper toxicity to trout would be eliminated by minimal dilution.
All routes in the field Exposure and effects at analogous sites	+	++	+	0	Tailings spilled to streams have persisted and caused severe effects, but the toxicity of the tailings is likely to be higher in those cases.
Summary weight of evidence	+	+	0	0	All lines of evidence are consistent with toxic effects of tailings. Despite the ambiguous quality and marginal strength of some lines of evidence, the overall strength is positive.

Notes:
E-R = exposure-response relationship.

9.6 Summary of Risks

9.6.1 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 30-km distance downstream. 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. In addition, the flow velocity of the receiving rivers, particularly in the spring, would readily transport the fine tailings particles farther downstream. The mouth of the Koktuli River is 63.6 km from the confluence of the South and North Fork Koktuli Rivers. 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, so contamination of the entire downstream system would be likely soon after a spill.

We did not explicitly model failures of TSF 2 or TSF 3. The types of risks and effects that would occur with a dam failure at TSF 2 or TSF 3 would be generally similar to those described for a failure at TSF 1. The content and toxicity of TSF 2 and TSF 3 are assumed to be similar to that of TSF 1, and the magnitude and extent of risks would be largely dependent on the volume of material released. One important distinction, however, should be noted. The South Fork Koktuli River and Upper Talarik Creek are hydrologically connected via groundwater transfer. In the event of a dam failure at TSF 3, transfer of contaminated water leached from tailings fines and deposited in the South Fork Koktuli River valley to the Upper Talarik Creek watershed and Iliamna Lake would be expected.

9.6.2 Remediation of a Tailings Spill

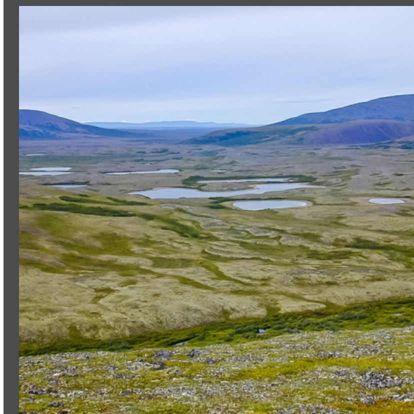
Although streams typically recover from aqueous effluents in less than a decade, the effects of tailings deposition in streams and floodplains persist for as long as they have been monitored at analogous sites. For that reason, many tailings-contaminated aquatic habitats in the United States have been or will be dredged, riprapped, or redirected under the federal Superfund or state cleanup programs. The tailings dam failure scenarios evaluated here do not consider any mitigating effects of remediation efforts by the mine operator or other parties. 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 fish habitat complexity and quality for fish by channelizing the stream or river (Schmetterling et al. 2001).

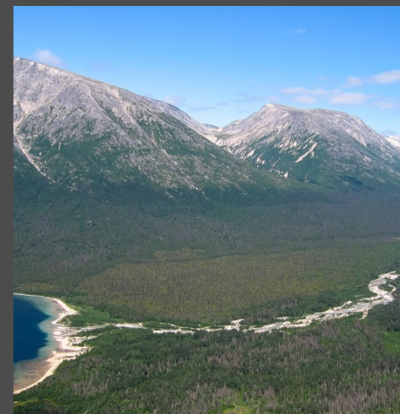
Remediation under the tailings dam failure scenarios considered here would be particularly difficult and damaging, because the area of the spill is almost entirely unaffected by other development. One or more roads would need to be built into this roadless area to bring in equipment and haul out the tailings. At the upper end of the affected area, the process of removing tailings would do little additional damage, since the structure of the watershed would have been destroyed. If tailings removal extended to streams that were not scoured in the initial tailings release, removal would destroy those streams and associated wetlands. If removal was not undertaken, the substrate of the streams would consist of tailings until high flows carried them downstream. It may be technically impractical to recover tailings fines that have been transported past the point of confluence with larger rivers.

In the Pebble 0.25 tailings dam failure scenario considered here, an estimated 45 million metric tons of tailings solids would be deposited in the North Fork Koktuli River valley (calculated from Table 9-5, assuming a dry density of 1.42 metric tons/m^3). Complete removal of this material would require a substantial earth-moving effort (e.g., including over 3 million round trips by 20-ton dump trucks). Recovery and removal would be even more challenging in the Pebble 2.0 tailings dam failure scenario, in which an estimated 97 million metric tons of tailings solids would be deposited in the North Fork Koktuli River valley, and an additional 263 million metric tons of tailings solids would be transported beyond the confluence with the South Fork Koktuli River and into the mainstem Koktuli River (Table 9-5). Material not deposited on the floodplains would be carried downstream; material deposited in the floodplains, if not recovered, would be remobilized by future precipitation and would wash downstream. It is unlikely that tailings in river channels would be recovered, because the fine material would be rapidly transported by the relatively high flow velocities of the rivers.

Remediation of prior tailings dam failures can serve as case studies. Failures are numerous, but the degree to which remediation results in restoration of natural resources has not been well documented. The 1998 failure of the Aznalcóllar tailings dam at Los Frailes Mine in Seville, Spain (Box 9-1) has been described as a case of substantial remediation. However, this kind of successful removal of tailings would be difficult to replicate in the Bristol Bay watershed. The Aznalcóllar area has a drier and warmer climate, flatter topography, better access from existing roads, and more readily available equipment and labor compared to the Bristol Bay region. The goal of the Aznalcóllar remediation was restoration of land use, which would not be the primary goal in the Bristol Bay watershed. In addition, potential releases from TSF 1 would be much larger than the release at Aznalcóllar (Box 9-1).

Emergency plans for metal mines in Alaska that have been provided to USEPA do not address remediation or restoration after a tailings spill has occurred. In fact, no tailings spill has been reported in Alaska, so it is not clear what remediation or restoration might be required. Given the uncertain toxicity of the tailings, the difficulty and expense of remediation and restoration, and the damage that would be done by remediation, it is possible that a spill would be left to be restored by natural processes.





CHAPTER 10. TRANSPORTATION CORRIDOR

10.1 Introduction

Because the Bristol Bay watershed is located in one of the last remaining virtually roadless areas in the United States, development of any mine in the Bristol Bay watershed would require substantial expansion and improvement of the region's transportation infrastructure. There are few existing roadways, no improved federal or state highways, and no railroads, pipelines, or other major industrial transportation infrastructure (Figure 6-6). As described in Section 6.1.3, the mine scenarios evaluated in this assessment include a 138-km gravel surface, all-weather permanent access road (Figure 6-6) connecting the mine site to a new deep-water port on Cook Inlet (Ghaffari et al. 2011). This length does not include road sections within the mine site itself. Approximately 113 km of this corridor would fall within the Kvichak River watershed.

The transportation corridor area considered in the assessment comprises 32 subwatersheds draining to Iliamna Lake (Figure 2-7). These subwatersheds, referred to as the corridor subwatersheds, encompass approximately 2,340 km^2 and contain nearly 1,900 km of streams mapped for this analysis (see Chapter 3 for a description of these methods). The seven largest subwatersheds are, from west to east, the headwaters of Upper Talarik Creek, the headwaters of the Newhalen River, Chekok Creek, Canyon Creek, Knutson Creek, Pile River, and the Iliamna River. The Newhalen River is the largest river crossed by the corridor, draining Sixmile Lake and Lake Clark. Sockeye return to spawn in the Newhalen River and tributaries to Sixmile Lake and Lake Clark. The transportation corridor would cross the Newhalen River and parallel the north shore of Iliamna Lake (Figure 6-6). It would traverse rolling, glaciated terrain for approximately 60 road km until reaching steeper hillsides northwest of the village of Pedro Bay and the shoreline of Knutson Bay. After crossing gentler terrain around the northeast end of Iliamna Lake (Pedro Bay and Pile Bay), the corridor would cross the Chigmit Mountains (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.

Although this route is not necessarily the only option for corridor placement, the assessment of potential environmental risks would not be expected to change substantially with minor shifts in road alignment. Along most feasible routes, the proposed transportation corridor would cross many streams (including unmapped tributaries), rivers, wetlands, and extensive areas with shallow groundwater, all draining to Iliamna Lake (Figures 10-1 and 10-2).

In this chapter, we consider the risks to fish habitats and populations associated with the transportation corridor, as illustrated in a conceptual model showing potential linkages among the corridor, associated sources and stressors, and assessment endpoints (Figure 10-3). We begin with a discussion of fish habitats and populations along the corridor. We then consider potential impacts on these habitats and populations resulting from its construction and operation. Although the transportation corridor would include the road and adjacent pipelines (Section 6.1.3), we focus primarily on the road component; potential pipeline failures are considered in Chapter 11.

Best management practices (BMPs) or mitigation measures would be used along the transportation corridor to minimize potential risks to salmonids and the ecosystems that support them. Relevant BMPs, and their likely effectiveness, are discussed in text boxes throughout the chapter.

Figure 10-1. Streams, wetlands, ponds, and lakes along the transportation corridor. Streams and rivers are from the National Hydrography Dataset (USGS 2012); wetlands, lakes, and ponds are from the National Wetlands Inventory (USFWS 2012).

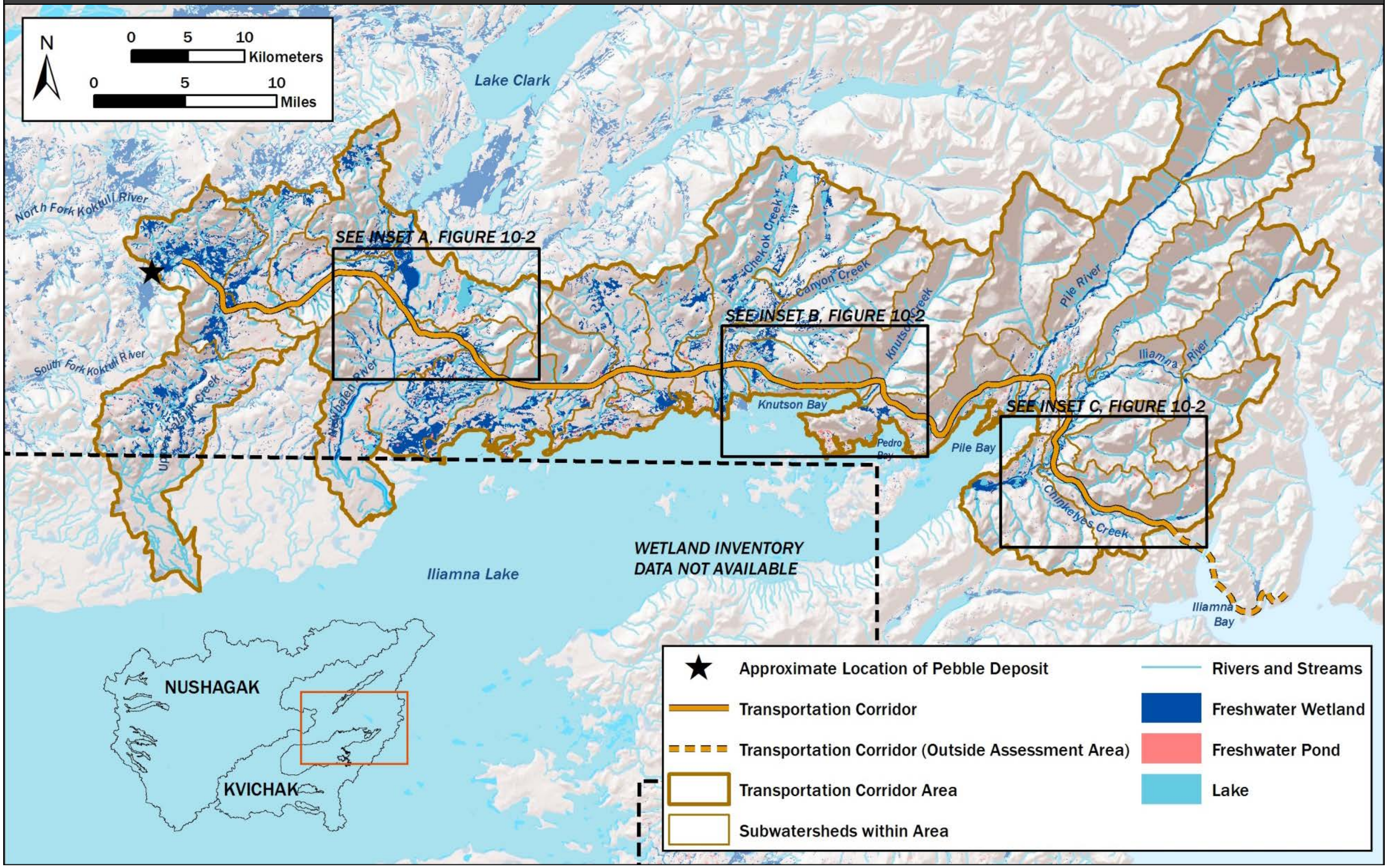


Figure 10-2. High-impact areas along the transportation corridor. Streams and rivers are from the National Hydrography Dataset (USGS 2012); wetlands, lakes, and ponds are from the National Wetlands Inventory (USFWS 2012). Image source: ESRI 2013. See Figure 10-1 for location of these areas along the transportation corridor.

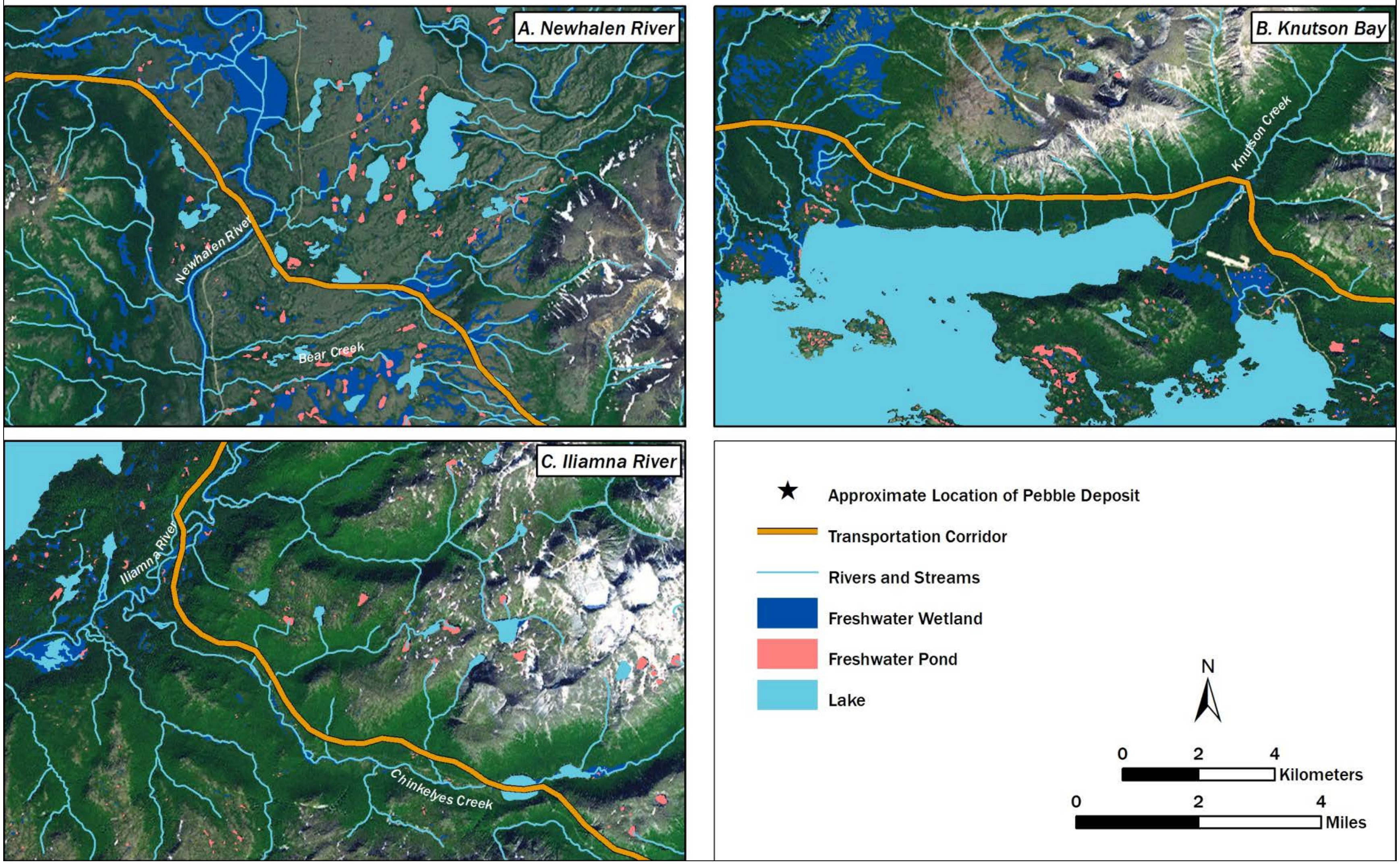
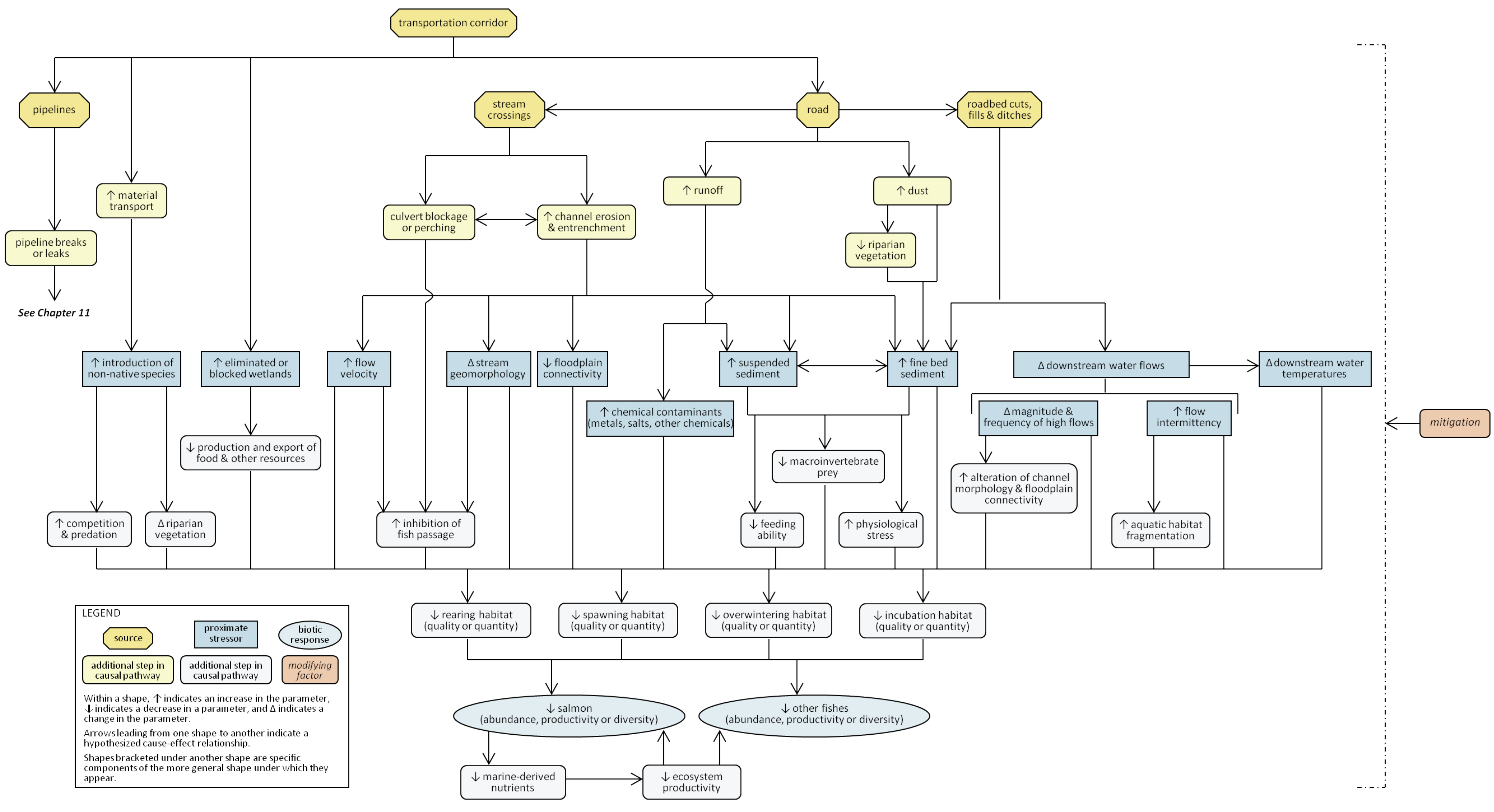


Figure 10-3. Conceptual model showing potential pathways linking the transportation corridor and related sources to stressors and assessment endpoints.



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10.2 Fish Habitats and Populations along the Transportation Corridor

In Chapter 3, we characterized stream segments in the Nushagak and Kvichak River watersheds by relative size (mean annual streamflow), channel gradient, and an index of the degree of channel constraint to describe floodplain potential (proportion of flatland in lowland, where stream segments with greater than 5% flatland in lowland in each reach's adjacent drainage basin are likely to be unconstrained and to exhibit floodplain potential). These attributes were selected because they represent fundamental aspects of the physical and geomorphic stream setting and provide context for stream and river habitat development and consequent fish habitat suitability (Burnett et al. 2007). Table 10-1 summarizes the proportion of stream channel lengths in the corridor subwatersheds (Scale 5), classified according to stream size, channel gradient, and floodplain potential. To allow direct visual comparison of the distribution of stream characteristics in the corridor subwatersheds relative to those in the entire Nushagak and Kvichak River watersheds (Scale 2), we present cumulative frequency plots in Figure 10-4. These plots show a frequency curve for each attribute at each geographic scale. Attributes are grouped into meaningful classes (Chapter 3), denoted by the vertical red classification bars. For example, the lowest gradient streams are classified as having gradients of less than 1% (Table 10-1), as shown by the vertical classification bar at 1% in Figure 10-4B. Cumulative frequency plots can be interpreted by evaluating the height at which the frequency curve is intersected by the red vertical classification bar. In Figure 10-4B, the 1% gradient classification bar intersects the Scale 5 frequency curve (solid black line) at a cumulative frequency value of approximately 32%. Thus, approximately 32% of the stream kilometers in the corridor subwatersheds (Scale 5) are less than 1% gradient. In comparison, approximately 64% of the stream kilometers in the Nushagak and Kvichak River watersheds (Scale 2) are less than 1% gradient.

Streams along the transportation corridor have not been sampled as extensively as streams near the Pebble deposit. Small to large rivers (2.8 m^3/s mean annual streamflow and larger) that would be crossed by the corridor (Table 10-1) provide spawning and rearing habitat, and are important routes for adult salmonid migration to upstream spawning areas and juvenile salmonid migration downstream to Iliamna Lake. Large and small streams with low to moderate gradients (3% or less) provide important high-quality spawning habitats, primarily for sockeye salmon. These streams also likely provide high-quality seasonal and some year-round habitats for resident Dolly Varden and rainbow trout. Dolly Varden are distributed across a much wider range of stream gradients (ADF&G 2012). The majority of stream length in the corridor subwatersheds consists of small headwater (58%) and medium (31%) streams, whereas small and large rivers make up 10 and 2% of stream length, respectively (Table 10-1). A majority (62%) of stream length in the corridor subwatersheds is classified as low to moderate gradient (32% at less than 1% gradient, and 30% at 1 to 3% gradient) (see Box 3-1 for discussion on how gradient was calculated). However, the corridor streams are generally steeper and have higher proportions of stream length without floodplain potential (i.e., less than 5% of flatland in lowland adjacent to stream) relative to those in the larger Nushagak and Kvichak River watersheds (Table 10-1,

Figure 10-4). Streams and rivers with high proportions of length with floodplain potential are more likely to be unconstrained and to develop complex off-channel habitats that provide a diversity of channel habitat types and create favorable conditions, particularly for salmonid rearing. However, the corridor streams are unique within the Nushagak and Kvichak River watersheds in that many of them are short and originate within the corridor subwatersheds. In addition, they all flow into Iliamna Lake, which provides high-quality habitat suitable for salmonid rearing and migration among streams.

Table 10-1. Proportion of stream channel length in stream subwatersheds intersected by the transportation corridor (Scale 5) classified according to stream size (based on mean annual discharge in m³/s), channel gradient (%), and floodplain potential (based on % flatland in lowland). Gray shading indicates proportions greater than 5%; bold indicates proportions greater than 10%.

	Gradient							
	<1%		≥1% and <3%		≥3% and <8%		≥8%	
Mean annual discharge	FP	NFP	FP	NFP	FP	NFP	FP	NFP
Small headwater and Iliamna Lake tributary streams[a]	**11%**	3%	4%	**12%**	1%	**17%**	0%	10%
Medium streams[b]	7%	2%	1%	10%	1%	7%	0%	3%
Small rivers[c]	5%	2%	0%	3%	0%	0%	0%	0%
Large rivers[d]	2%	0%	0%	0%	0%	0%	0%	0%

Notes:

[a] 0–0.15 m^3/s; headwater tributaries of streams crossing the transportation corridor and small streams flowing directly to Iliamna Lake (e.g., Eagle Bay and Chekok Creeks).

[b] 0.15–2.8 m^3/s; upper reaches and larger tributaries of streams crossing the transportation corridor, and medium streams flowing directly into Iliamna Lake (e.g., Chinkelyes and Knutson Creeks).

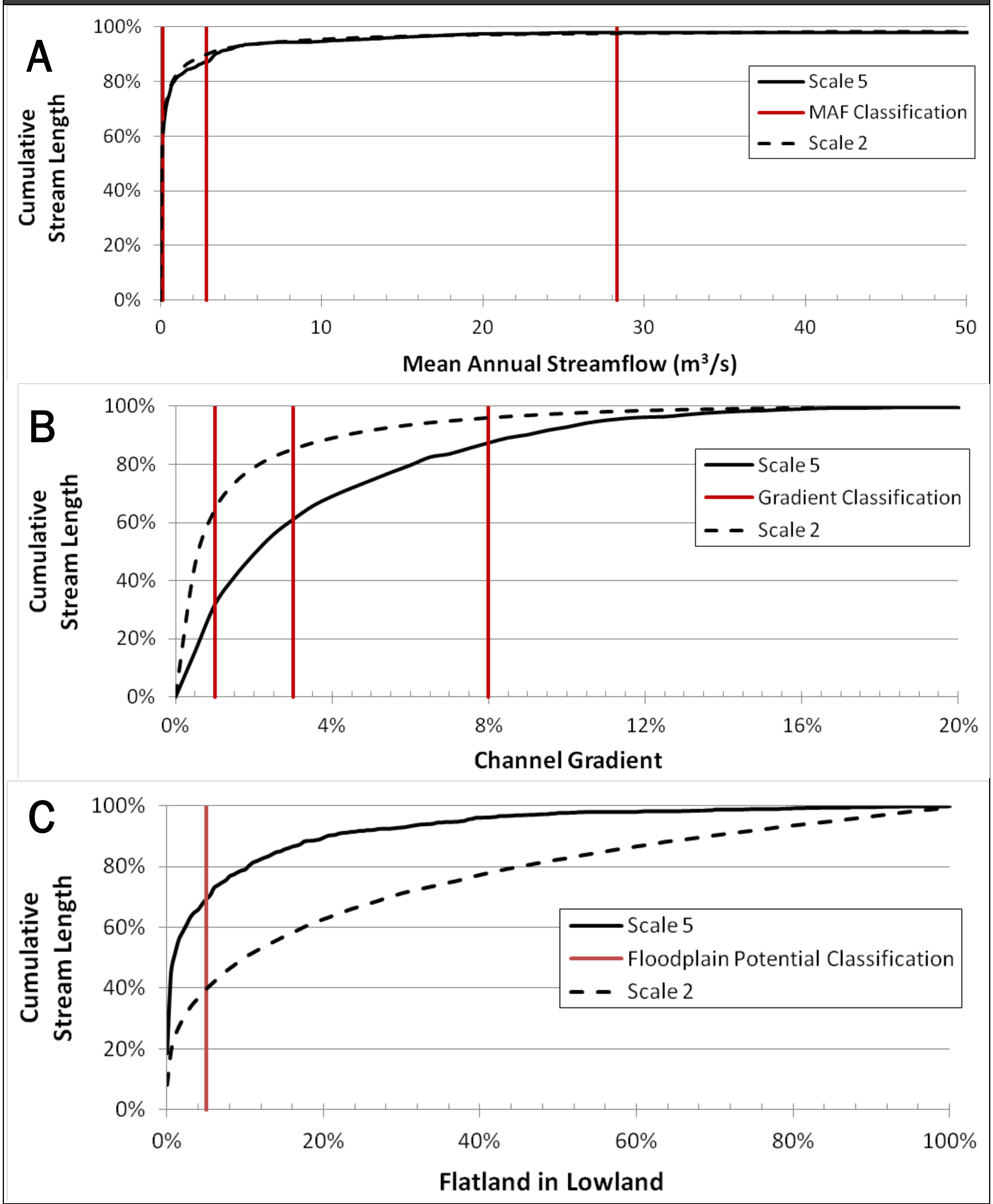
[c] 2.8–28 m^3/s; middle to lower portions of the Iliamna and Pile Rivers.

[d] >28 m^3/s; the Newhalen River.

FP = high floodplain potential (≥5% flatland in lowland); NFP = no or low floodplain potential (<5% flatland in lowland) (see Chapter 3 for additional explanation).

At the scale of the Nushagak and Kvichak River watersheds, 85% of stream length is classified as less than 3% gradient (64% at less than 1% gradient and 21% at 1 to 3% gradient), versus 62% in the corridor subwatersheds. Sixty percent of total stream length in the Nushagak and Kvichak River watersheds is classified as exhibiting floodplain potential, versus 31% in the corridor subwatersheds (Figure 10-4). These differences stem in large part from the large portions of the unconfined, low-gradient lower Nushagak River watershed. Percent of stream length less than 3% gradient is 73 and 91% in the Kvichak and Nushagak River watersheds, respectively; the percent of stream length classified as floodplain prone is 50% across the Kvichak River watershed and 65% across the Nushagak River watershed. Thus, stream characteristics in the transportation corridor area are generally more similar to those in the Kvichak River watershed. Characterization of stream segments for the entire Nushagak and Kvichak River watersheds, as well as the methods used, are described in Chapter 3.

Figure 10-4. Cumulative frequency of stream channel length classified by (A) mean annual streamflow (MAF) (m³/s), (B) channel gradient (%), and (C) floodplain potential (based on % flatland in lowland) for stream subwatersheds intersected by the transportation corridor (Scale 5) versus the Nushagak and Kvichak River watersheds (Scale 2). See Section 3.4 for further explanation of MAF, gradient, and floodplain potential classifications.



These low- to moderate-gradient streams provide important spawning habitat for sockeye. 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 diverse 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. For example, sockeye salmon that use spring-fed ponds and streams approximately 1 km apart exhibit differences in traits that are consistent with discrete populations, such as spawn timing, spawn site fidelity, and productivity (Quinn et al. 2012).

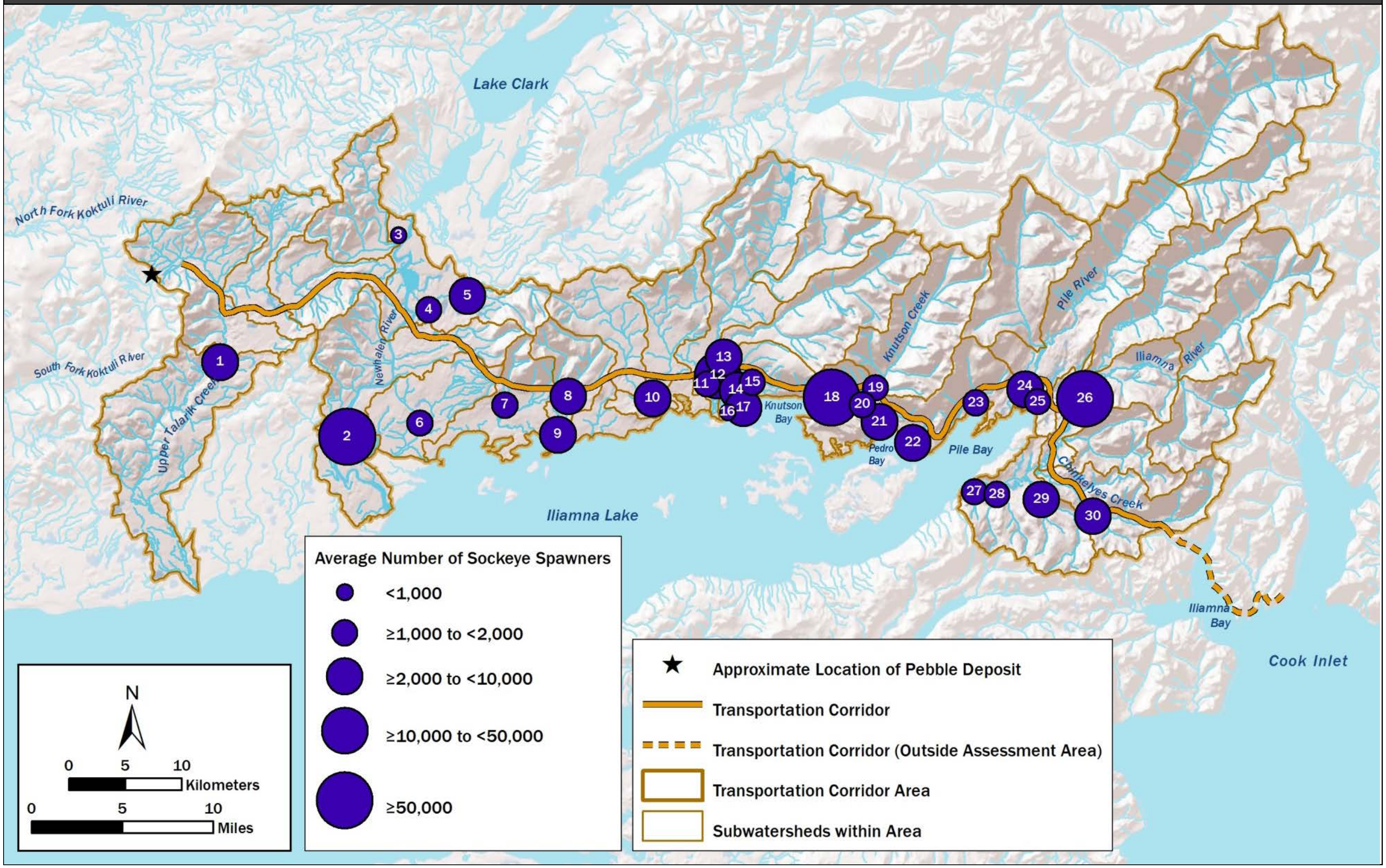
Sockeye spawning has been observed at 30 locations along the transportation corridor (Demory et al. 1964). The Alaska Department of Fish and Game (ADF&G) has conducted aerial index counts of sockeye salmon spawning abundance at these locations in most years since 1955 (Morstad 2003). We recognize that survey values tend to underestimate true abundance for many reasons. An observer in an aircraft is not able to count all fish in dense aggregations, and 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). Surveys intended to capture peak abundance may not always do so. Weather, water clarity, and other factors influencing fish visibility can also contribute to underestimates. Finally, spawning locations along the corridor occur across a variety of habitats, including mainland beaches, small ponds, streams, and larger rivers. Aerial survey-based indices of sockeye salmon spawning abundance vary considerably. Sockeye index counts are highest in the Iliamna River (averaging over 100,000 spawners), the Newhalen River (averaging over 80,000 spawners), and on beaches in Knutson Bay (averaging over 70,000 spawners) (Table 10-2, Figure 10-5). In some years, these counts can be very large, as illustrated by the 1960 survey for Knutson Bay that reported 1 million adults (Demory et al. 1964). Sockeye spawning is associated with upwelling groundwater areas on beaches along the north and east shores of Knutson Bay, adjacent to the transportation corridor. In addition, sockeye use of spring-fed ponds has been observed 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 adapted to the unique abiotic features of ponds (Quinn et al. 2012).

Table 10-2. Average number of spawning adult sockeye salmon at locations near the transportation corridor. See Figure 10-5 for the locations of these areas.

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	Northwest Eagle Bay Creek	Stream	1,649	32	0–17,562
8	North East	Northeast Eagle Bay Creek/Ponds	Stream	3,416	38	0–18,175
9	North East	Northeast Eagle Bay Creek 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.
Sources: Morstad 2003, Morstad pers. comm.

Figure 10-5. Location of sockeye salmon surveys and number of spawners observed along the transportation corridor. Numbers refer to map points listed in Table 10-2.

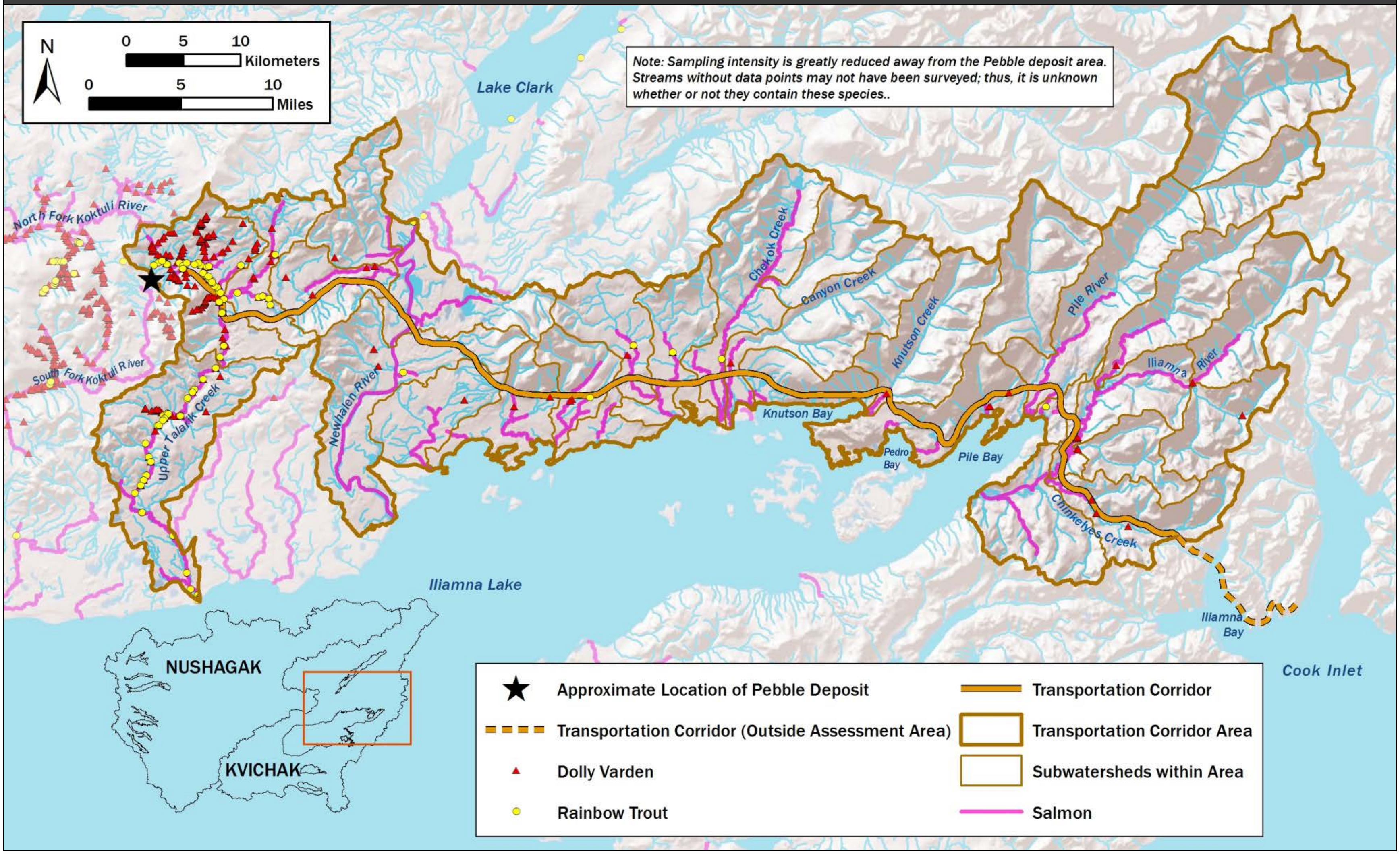


Less is known about the occurrence or abundance of other salmon species in streams and rivers crossing or adjacent to the transportation corridor. Chinook, coho, pink, and chum salmon are present in the Kvichak River watershed, but data for their spatial occurrences are for isolated points in the system (Johnson and Blanche 2012). In streams intersected by the transportation corridor, sockeye salmon are in all streams included in the *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region* (also known as the Anadromous Waters Catalog [AWC]) (Johnson and Blanche 2012) (Figure 10-6). Working from west to east along the corridor, streams with salmon species in addition to sockeye are as follows: Upper Talarik Creek (Chinook, coho, chum, and pink salmon), the Newhalen River (Chinook and coho salmon), Youngs Creek (East and West Branches), Chekok and Tomkok Creeks (coho salmon), Swamp Creek (a tributary to Pile Bay) (Chinook salmon), and the Iliamna River (Chinook, coho, chum, and pink salmon).

Dolly Varden and rainbow trout distributions have not been surveyed as extensively as salmon distributions along the transportation corridor (ADF&G 2012). Dolly Varden have been documented in nearly every sockeye salmon-bearing stream that would be crossed by or adjacent to the corridor, as well as in locations upstream of reported anadromous salmon use (Figure 10-6). Rainbow trout presence along the corridor is reported for only a few streams, including Upper Talarik Creek, the Newhalen River, an unnamed tributary to Eagle Bay, Youngs Creek, Tomkok Creek, and Swamp Creek (ADF&G 2012). Rainbow trout have also been documented in the Iliamna River (Russell 1977) and Chinkelyes Creek (Berejikian 1992).

The distributions of both Dolly Varden and rainbow trout along the transportation corridor are likely much more extensive than reported in the Alaska Freshwater Fish Inventory (AFFI) resident fish database (ADF&G 2012), which does not account for seasonal movements or low sampling effort. Sockeye salmon provide an important food subsidy to Dolly Varden and rainbow trout. For example, Denton et al. (2009) reported Dolly Varden movement into multiple ponds used by spawning sockeye next to the Pedro Bay village, where they feed on sockeye salmon fry, eggs, and carcass-associated blowflies. Information on rainbow trout movement between Iliamna Lake and streams intersected by the corridor is not available, but these movements are likely to occur. Movements between lakes and tributary streams in response to feeding and spawning opportunities have been documented elsewhere in Iliamna Lake (Russell 1977), the Alagnak River system (Meka et al. 2003), and in the Wood River lake system (Ruff et al. 2011).

Figure 10-6. Reported salmon, Dolly Varden, and rainbow trout distributions along the transportation corridor. Salmon presence data are from the Anadromous Waters Catalog (Johnson and Blanche 2012); Dolly Varden and rainbow trout presence data are from the Alaska Freshwater Fish Inventory (ADF&G 2012). Note that rainbow trout have also been documented in the Iliamna River (Russell 1977) and Chinkelyes Creek (Berejikian 1992), although these points are not indicated on this map.



10.3 Potential Risks to Fish Habitats and Populations

Only rarely has it been possible to build roads that have no negative effects on streams (Furniss et al. 1991). Roads modify natural drainage networks and accelerate erosion processes, which 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. Road construction can increase the frequency of slope failures by orders of magnitude, depending on variables such as soil type, slope steepness, bedrock type and structure, and presence of subsurface water. These slope failures can result in episodic sediment delivery to streams and rivers, potentially for decades after roads are built (Furniss et al. 1991). All of these potential changes can have important biological consequences for anadromous and resident fishes by negatively affecting food, shelter, spawning habitat, water quality, and access for upstream and downstream migration (Appendix G) (Furniss et al. 1991).

In the Bristol Bay region, risks to fish from construction and operation of the transportation corridor would be complex and potentially significant, largely because of hydrological issues. Field observations in the mine area (Hamilton 2007, Woody and O'Neal 2010) indicate terrain with abundant near-surface groundwater and a high incidence of seeps and springs associated with complex glaciolacustrine, alluvial, and slope till deposits (Appendix G). The abundance of mapped wetlands (Figures 10-1 and 10-2) further demonstrates the pervasiveness of shallow subsurface flows and high connectivity between groundwater and surface-water systems in the areas traversed by the transportation corridor (Appendix G). As noted in Section 3.3, the strong connection between groundwater and surface waters helps to moderate water temperatures and streamflows, and this moderation can be critical for fish populations. The construction and operation of the transportation corridor could fundamentally alter connections between shallow aquifers and surface channels and ponds by intercepting shallow groundwater flowpaths, leading to impacts on surface water hydrology, water quality, and fish habitat (Darnell et al. 1976, Stanford and Ward 1993, Forman and Alexander 1998, Hancock 2002).

The lengths of the transportation corridor and their proximities to National Hydrography Dataset (NHD) streams (USGS 2012) and National Wetlands Inventory (NWI) wetlands, ponds, and small lakes (USFWS 2012) are shown in Tables 10-3 and 10-4, respectively (see Box 10-1 for a description of methods used to estimate these values). In sum, the length of road within 200 m of NHD streams or NWI aquatic habitats would be approximately 67 km (Table 10-5). These lengths do not encompass the section of corridor outside of the Kvichak River watershed (i.e., the watersheds flowing into Cook Inlet). The 200-m road buffer was derived from an estimate of the road-effect zone for secondary roads (Forman 2000). The largest impact on sockeye salmon would likely occur where the road would run parallel to the Iliamna River and Chinkelyes Creek, sites at which many sockeye salmon spawn (Figure 10-2, Inset C). Other high-impact areas include where the road would run parallel to Knutson Bay, intersecting many small streams and where groundwater upwelling supports spawning for hundreds of thousands of salmon (Figure 10-2, Inset B), and where the road crosses wetlands north of Iliamna Lake (Figure 10-2, Inset A).

In the following sections, we consider potential risks to fish habitats and populations resulting from construction and operation of the transportation corridor. We focus on risks related to filling and alteration of wetlands, stream crossings, fine sediments, dust deposition, runoff contaminants, and invasive species.

BOX 10-1. CALCULATION OF STREAM LENGTHS AND WETLAND AREAS AFFECTED BY TRANSPORTATION CORRIDOR DEVELOPMENT

We used the National Hydrography Dataset (NHD) (USGS 2012), the National Wetlands Inventory (NWI) (USFWS 2012), the Alaska Anadromous Waters Catalog (AWC) (Johnson and Blanche 2012), and the Alaska Freshwater Fish Inventory (AFFI) (ADF&G 2012) to evaluate potential effects of the transportation corridor on hydrologic features and fish populations.

The length of stream downstream of each crossing was estimated from NHD flowlines. Stream length by subwatershed, based on 12-digit hydrologic unit codes, was calculated as the total distance from each crossing to Iliamna Lake. In the multiple instances where stream crossings were tributaries to a single main channel, the mainstem length was only counted once (Table 10-3). Downstream lengths reported in Table 10-6 include mainstem lengths downstream of tributary crossings. In cases where the corridor crossed tributaries of a mainstem channel, the mainstem length is included in both crossings.

Mean annual streamflow of NHD streams upstream of the transportation corridor was estimated using methods described in Box 3-2.

The channel gradient of NHD stream segments intersected by and upstream of the corridor was estimated using a 30-m National Elevation Dataset digital elevation model (DEM) (Gesch 2007, Gesch et al. 2002, USGS 2013). A drainage network was developed from a flow analysis using the DEM and slope was estimated using this drainage network. The DEM-based drainage network paralleled the NHD stream flowlines and therefore, using the toolset in the spatial analyst extension in ArcGIS, slope from the drainage network was transferred to NHD reach segments. A 12% slope was used to calculate stream length likely to support fish (Table 10-6). Stream length upstream of the corridor with less than 12% slope was based on the NHD stream length to the first instance where slope was greater than 12%. The analysis of upstream fish habitat was extended to include streams in subwatersheds in the Headwaters Newhalen River, Tomkok Creek, Pile River, and Iliamna River.

For the analysis of road length intersecting and within 100 or 200 m of either a stream or wetland (Tables 10-3 through 10-5), each stream (NHD) or wetland (NWI) was buffered to a distance of 100 m and 200 m and the length of corridor within these ranges was summed. Similarly, for the area of wetlands, ponds, and small lakes within 100 m and 200 m of the road corridor, the road corridor was buffered and the area of wetlands, ponds, and small lakes within that buffered area was summed across the length of road. For the area of wetlands, ponds, and small lakes directly filled by the road corridor, we assumed a road width of 9.1 m.

The characterization of both stream length and wetland, pond, and small lake area affected is likely a conservative estimate. The NHD may 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 due to limited sampling along the corridor. The characterization of wetland, pond, and small lake area is limited by the resolution of the available NWI data product. In this analysis, the transportation corridor often bisects wetland features and the wetland area falling outside the 200-m boundary was assumed to maintain its functionality. We were also unable to determine the effect that the transportation corridor may have on wetlands that had no direct surface water connection, but that may be hydrologically connected via groundwater pathways. Together, these limitations likely make our calculations an underestimate of the effect that transportation corridor development would have on hydrologic features in this region. These estimates could be improved with enhanced, higher-resolution mapping, increased sampling of possible fish-bearing waters, and ground-truthing of surface-water and groundwater connections.

Table 10-3. Proximity of the transportation corridor to National Hydrography Dataset streams (USGS 2012).

HUC-12 Name or Description	HUC-12 Digit	Proximity to Streams			
		Not nearby (km)	<100 m (km)	100–200 m (km)	Total (km)
Headwater, Upper Talarik Creek	190302060702	5.4	0.8	1.2	7.4
Upper tributary stream to Upper Talarik Creek	190302060701	4.3	0.2	0.1	4.6
Tributary to Newhalen River portion of corridor	190302051404	7.8	1.9	1.2	10.9
Headwaters, Newhalen River	190302051405	2.6	0.4	0.4	3.4
Outlet, Newhalen River	190302051406	4.2	1.5	0.8	6.5
Roadhouse Creek	190302060907	0.8	1.2	1.3	3.3
Iliamna Lake	190302060914	29.3	4.3	4.1	37.7
Eagle Bay Creek	190302060905	3.1	0.5	0.8	4.4
Youngs Creek Mainstem (Roadhouse Mountain HUC)	190302060903	3.0	0.1	0.2	3.4
Youngs Creek East Branch	190302060904	1.4	1.0	0.6	3.0
Chekok Creek	190302060302	1.8	0.3	0.3	2.5
Canyon Creek	190302060902	1.1	0.1	0.2	1.4
Knutson Creek	190302060901	1.2	0.3	0.4	2.0
Outlet, Pile River	190302060104	2.1	0.6	0.7	3.4
Middle Iliamna River	190302060205	4.5	1.1	0.7	6.4
Chinkelyes Creek	190302060206	9.6	0.8	2.1	12.5
Total length across all HUCs		**82.1**	**15.3**	**15.2**	**113**
Percentage across all HUCs		73%	14%	13%	100%

Notes:
HUC = hydrologic unit code.

Table 10-4. Proximity of the transportation corridor to National Wetlands Inventory wetlands, ponds, and small lakes (USFWS 2012).

HUC-12 Name or Description	HUC-12 Digit	Proximity to Wetlands				
		Not nearby (km)	Intersects (km)	<100 m (km)	100–200 m (km)	Total (km)
Headwater, Upper Talarik Creek	190302060702	0.2	1.9	4.0	1.2	7.4
Upper tributary stream to Upper Talarik Creek	190302060701	1.7	0.3	1.4	1.2	4.6
Tributary to Newhalen River portion upstream of corridor	190302051404	4.0	0.4	3.9	2.6	10.9
Headwaters, Newhalen River	190302051405	2.3	0.1	0.4	0.5	3.4
Outlet, Newhalen River	190302051406	1.1	2.4	1.7	1.4	6.5
Roadhouse Creek	190302060907	0.7	0.3	1.8	0.5	3.3
Iliamna Lake	190302060914	28.3	1.8	3.9	3.7	37.7
Eagle Bay Creek	190302060905	1.3	0.7	1.7	0.8	4.4
Youngs Creek Mainstem (Roadhouse Mountain HUC)	190302060903	0.9	0.2	1.1	1.2	3.4
Youngs Creek East Branch	190302060904	0.3	0.5	0.8	1.5	3.0
Chekok Creek	190302060302	1.8	0.2	0.3	0.2	2.5
Canyon Creek	190302060902	0.8	0.0	0.2	0.3	1.4
Knutson Creek	190302060901	1.0	0.1	0.6	0.3	2.0
Outlet, Pile River	190302060104	0.3	1.2	1.5	0.5	3.4
Middle Iliamna River	190302060205	2.7	0.6	1.7	1.3	6.4
Chinkelyes Creek	190302060206	7.7	1.4	1.9	1.5	12.5
Total length across all HUCs		**55.0**	**12.2**	**27.0**	**18.5**	**113**
Percentage across all HUCs		49%	11%	24%	16%	100%

Notes:
HUC = hydrologic unit code.

Table 10-5. Proximity of the transportation corridor to water, in terms of the length occurring within 200 m of National Hydrography Dataset streams (USGS 2012) or National Wetlands Inventory wetlands, ponds, and small lakes (USFWS 2012).

HUC-12 Name or Description	HUC-12 Digit	Proximity to Streams, Wetlands, Ponds, and Small Lakes		
		Not nearby (km)	Within 200 m (km)	Total (km)
Headwater, Upper Talarik Creek	190302060702	0.1	7.3	7.4
Upper tributary stream to Upper Talarik Creek	190302060701	1.5	3.1	4.6
Tributary to Newhalen River portion upstream of corridor	190302051404	3.8	7.0	10.9
Headwaters, Newhalen River	190302051405	2.3	1.1	3.4
Outlet, Newhalen River	190302051406	1.1	5.4	6.5
Roadhouse Creek	190302060907	0.0	3.3	3.3
Iliamna Lake	190302060914	22.1	15.5	37.7
Eagle Bay Creek	190302060905	0.9	3.5	4.4
Youngs Creek Mainstem (Roadhouse Mountain HUC)	190302060903	0.7	2.7	3.4
Youngs Creek East Branch	190302060904	0.3	2.8	3.0
Chekok Creek	190302060302	1.5	1.0	2.5
Canyon Creek	190302060902	0.8	0.5	1.4
Knutson Creek	190302060901	0.7	1.2	2.0
Outlet, Pile River	190302060104	0.3	3.1	3.4
Middle Iliamna River	190302060205	1.9	4.5	6.4
Chinkelyes Creek	190302060206	7.3	5.2	12.5
Total length across all HUCs		**45.4**	**67.3[a]**	**113**
Percentage across all HUCs		40%	60%	100%

Notes:
HUC = hydrologic unit code.
[a] Reported length is the sum of the road length within 200 m of a National Hydrography Dataset stream or National Wetlands Inventory wetland reported in Tables 10-3 and 10-4, respectively. In cases where the same section of road is near both types of water bodies, section is only reported once. Therefore total length is less than sum of lengths in Tables 10-3 and 10-4.

10.3.1 Filling and Alteration of Wetlands, Ponds, and Small Lakes

10.3.1.1 Exposure

Approximately 10% (12 km) of the transportation corridor would intersect mapped wetlands, ponds, and small lakes (Table 10-4). An additional 24% (27 km) would be located within 100 m of these habitats, and another 16% (19 km) would be located within 100 to 200 m (Table 10-4). In total, approximately 51% (58 km) of the corridor would fill or otherwise alter wetlands, ponds, and small lakes. These habitats encompass 2.3 km^2 (1.6, 0.1, and 0.6 km^2 of wetlands, ponds, and small lakes, respectively), or nearly 11% of the total area within 100 m of the transportation corridor. The area of NWI-mapped aquatic habitats within 200 m of the corridor would be 4.7 km^2 (3.3, 0.2, and 1.2 km^2 of wetlands, ponds, and small lakes, respectively). These areas do not include NWI-mapped aquatic habitats that would be covered by the mine footprints in the mine scenarios (Chapter 7). The area of these habitats filled by the roadbed would be 0.11 km^2 (i.e., approximately 12 km of road, assuming a road width of 9 m).

10.3.1.2 Exposure-Response

The distribution of salmonids in wetlands, ponds, and small lakes along the transportation corridor is not known. However, these aquatic habitat losses can result in the loss of resting habitat for adult salmonids and of spawning and rearing habitat in ponds and riparian side channels. These habitats can provide refuge habitats (Brown and Hartman 1988) and important rearing habitats for juvenile salmonids by providing hydraulically and thermally diverse conditions. In addition, by damming and diverting surface flow and inhibiting subsurface flow, road construction could block or limit access by fish to important habitats. Beaver ponds associated with small streams, ponds, and wetlands can be important winter refugia for coho salmon (Nickelson et al. 1992, Cunjak 1996). Beaver ponds provide high-quality habitat for salmon rearing, because they provide macrophyte cover, low-flow velocity, and increased temperatures and trap organic materials and nutrients (Nickelson et al. 1992, Collen and Gibson 2001, Lang et al. 2006).

These habitats can also provide enhanced foraging opportunities (Sommer et al. 2001). Floodplain wetlands and ponds can be an important contributor to the abundance and diversity of food (and foodwebs) upon which salmon depend (Opperman et al. 2010). Within aquatic habitats that are not blocked and are still accessible, the road bed could alter hydrology and flow paths from these habitats to the stream network. These alterations could mobilize minerals and stored organic carbon, and expose soils to new wetting, drying, and leaching regimes, thereby leading to changes in vegetation, nutrient and salt concentrations, and water quality (Ehrenfeld and Schneider 1991). These changes in wetland dynamics and structure could affect the availability of these habitats to fish and the contribution of nutrients, organic material, and a diverse array of macroinvertebrates from headland wetlands to higher order streams in the watershed (i.e., streams receiving wetland drainage) and downstream waters (Shaftel et al. 2011, Dekar et al. 2012, King et al. 2012, Walker et al. 2012).

10.3.1.3 Risk Characterization

Filling wetlands would eliminate habitat for salmonids and would indirectly alter wetlands in ways that could reduce the quality, quantity, and accessibility of habitat for fish. Effects on fish production cannot be estimated given available data; however, the loss of long riparian side channels to culvert or bridge crossings that do not span the entire floodplain could be locally significant. These wetlands provide important spawning and rearing habitats and resting areas for migrating adults. Other wetlands such as shallow ponds may also provide habitat, but all wetlands serve to moderate variation in flow and maintain water quality.

10.3.2 Stream Crossings

The transportation corridor would cross approximately 64 streams in the Kvichak River watershed. Of these streams, 20 are listed as supporting anadromous fish in the AWC (Johnson and Blanche 2012) at the crossing (Table 10-6). An additional 35 are likely to support salmonids (Table 10-6), and a number of these are anadromous downstream of the crossing. In total, the transportation corridor would cross 55 streams known or likely to support salmonids.

The physical effects of roads on streams and rivers often propagate long distances from actual stream crossings, because of the energy associated with moving water (Richardson et al. 1975). Thus, alteration of hydrology and sediment deposition by road crossings can change channels or shorelines many kilometers away. The transportation corridor could affect 272 km of stream between its road crossings and Iliamna Lake (Table 10-7). Fish may also be affected in the approximately 780 km of streams upstream of the transportation corridor that are likely to support salmonids (based on surveys and stream gradients less than 12%, Table 10-8). In this assessment, we assume streams with segment gradients less than 12% both downstream and upstream of the corridor-stream crossing are likely to support salmonids (i.e., salmon, rainbow trout, or Dolly Varden). The amount of upstream length that may be salmonid habitat is calculated as stream length to the first reach segment with a gradient greater than 12%. This criterion is used as an upstream limit for salmonid habitat, as Dolly Varden can be dispersed across a wide range of channel gradients (Wissmar et al. 2010) and have been observed in higher-gradient reaches (average 12.9% gradient) throughout the year in southeastern Alaska (Bryant et al. 2004).

Table 10-6. Road-stream crossings along the transportation corridor, upstream lengths of streams of different sizes likely to support salmonids (based on stream gradients of less than 12%), and downstream lengths to Iliamna Lake. Bold reach codes are those assumed to be bridged.

HUC-12 Name or Description	NHD Reach Code at Road-Stream Crossing	AWC (*Salmonid Potential)	Upstream Fish Habitat Length (km)					Downstream Length to Iliamna Lake (km)
			Small Headwater Streams[a]	Medium Streams[a]	Small Rivers[a]	Large Rivers[a]	Total	
Headwaters Upper Talarik Creek	19030206007354	Y *	3.5	0.0	0.0	0.0	3.5	57.6
	19030206007015	Y *	97.4	37.6	0.0	0.0	134.9	57.0
	19030206007159	Y *	1.4	0.0	0.0	0.0	1.4	55.6
Upper Tributary to Upper Talarik Creek[b]	19030206007175	N *	3.7	0.0	0.0	0.0	3.7	66.0
Tributary to Newhalen River[c]	19030205007587	N *	5.7	0.0	0.0	0.0	5.7	45.9
	19030205007593	N *	3.8	0.0	0.0	0.0	3.8	41.7
	19030205007598	N *	3.6	0.0	0.0	0.0	3.6	44.5
	19030205007606	Y *	6.8	0.0	0.0	0.0	6.8	37.2
	19030205007602	Y *	2.8	0.0	0.0	0.0	2.8	34.8
Headwaters Newhalen River	19030205007615	N *	3.1	0.0	0.0	0.0	3.1	29.4
	19030205000002	Y *	67.7	45.2	0.0	13.1	126.1	26.4
Outlet Newhalen River	19030205013069	N	0.0	0.0	0.0	0.0	0.0	1.1
	19030205013055	N *	6.2	2.6	0.0	0.0	8.8	1.3
	19030205013057	N *	1.8	0.0	0.0	0.0	1.8	3.7
	19030205013041	N *	3.2	0.0	0.0	0.0	3.2	3.7
Roadhouse Creek	19030206010623	N *	0.7	0.0	0.0	0.0	0.7	2.4
	19030206010628	N *	0.4	0.0	0.0	0.0	0.4	3.6
	19030206010629	N *	0.7	0.0	0.0	0.0	0.7	2.2
	19030206006712	N	0.0	0.0	0.0	0.0	0.0	15.7
Iliamna Lake–Eagle Bay	**19030206006678**	Y *	0.9	1.5	0.0	0.0	2.4	9.6
	19030206006677	N	0.0	0.0	0.0	0.0	0.0	10.3
	19030206006644	N *	1.6	0.0	0.0	0.0	1.6	11.1
Eagle Bay Creek	**19030206006671**	N *	0.4	5.5	0.0	0.0	5.9	6.4
	19030206006663	Y *	11.3	0.0	0.0	0.0	11.3	6.3
	19030206006654	Y *	4.0	0.0	0.0	0.0	4.0	6.4
Youngs Creek Mainstem (Roadhouse Mountain HUC)	**19030206006598**	Y *	25.7	16.3	0.0	0.0	42.0	10.4

Table 10-6. Road-stream crossings along the transportation corridor, upstream lengths of streams of different sizes likely to support salmonids (based on stream gradients of less than 12%), and downstream lengths to Iliamna Lake. Bold reach codes are those assumed to be bridged.

HUC-12 Name or Description	NHD Reach Code at Road-Stream Crossing	AWC (*Salmonid Potential)	Upstream Fish Habitat Length (km)					Downstream Length to Iliamna Lake (km)
			Small Headwater Streams[a]	Medium Streams[a]	Small Rivers[a]	Large Rivers[a]	Total	
Youngs Creek East Branch[d]	**19030206006553**	Y *	32.9	12.4	0.0	0.0	45.3	9.0
Chekok Creek	19030206006533	Y *	5.8	0.0	0.0	0.0	5.8	5.0
	19030206032854	Y *	36.1	42.5	7.9	0.0	86.6	8.4
Canyon Creek	**19030206006359**	Y *	0.0	1.2	8.6	0.0	9.8	12.1
Iliamna Lake–Knutson Bay	19030206006336	N *	4.4	0.0	0.0	0.0	4.4	3.8
	19030206006337	N *	0.3	0.0	0.0	0.0	0.3	3.6
	19030206006236	N *	1.0	0.0	0.0	0.0	1.0	3.4
	19030206006331	N *	0.6	0.0	0.0	0.0	0.6	4.2
	19030206006329	N *	0.6	0.0	0.0	0.0	0.6	3.9
	19030206006327	N *	0.2	0.0	0.0	0.0	0.2	1.9
	19030206006325	N *	0.8	0.0	0.0	0.0	0.8	2.6
	19030206006322	N	0.0	0.0	0.0	0.0	0.0	0.1
	19030206006320	N *	0.1	0.0	0.0	0.0	0.1	0.7
	19030206006321	N *	0.5	0.0	0.0	0.0	0.5	0.7
	19030206006318	N	0.0	0.0	0.0	0.0	0.0	0.8
	19030206006317	N	0.0	0.0	0.0	0.0	0.0	0.9
	19030206006316	N *	0.5	0.0	0.0	0.0	0.5	0.5
	19030206006315	N *	0.7	0.0	0.0	0.0	0.7	0.6
	19030206006314	N *	0.7	0.0	0.0	0.0	0.7	0.7
	19030206006251	N *	0.6	0.0	0.0	0.0	0.6	1.7
Knutson Creek	**19030206006255**	Y *	0.1	3.2	1.9	0.0	5.2	4.4
	19030206006280	N *	0.4	0.0	0.0	0.0	0.4	4.4
Iliamna Lake–Pedro Bay	19030206006239	N	0.0	0.0	0.0	0.0	0.0	2.5
	19030206006248	N *	0.3	0.0	0.0	0.0	0.3	4.7
Iliamna Lake–Pile Bay	19030206006231	N	0.0	0.0	0.0	0.0	0.0	0.6
	19030206006230	N	0.0	0.0	0.0	0.0	0.0	0.4
	19030206006228	Y *	0.0	0.3	0.0	0.0	0.3	1.5
	19030206006227	N *	0.0	0.9	0.0	0.0	0.9	3.0

Table 10-6. Road-stream crossings along the transportation corridor, upstream lengths of streams of different sizes likely to support salmonids (based on stream gradients of less than 12%), and downstream lengths to Iliamna Lake. Bold reach codes are those assumed to be bridged.

HUC-12 Name or Description	NHD Reach Code at Road-Stream Crossing	AWC (*Salmonid Potential)	Upstream Fish Habitat Length (km)					Downstream Length to Iliamna Lake (km)
			Small Headwater Streams[a]	Medium Streams[a]	Small Rivers[a]	Large Rivers[a]	Total	
Outlet Pile River	**19030206006222**	N *	0.0	3.4	0.0	0.0	3.4	6.3
	19030206000474	Y *	34.1	24.9	50.0	0.0	109.0	5.7
	19030206010632	Y *	4.2	0.0	0.0	0.0	4.2	0.9
	324-10-10150-2343-3006[e]	Y *	NO NHD DATA					1.0
Middle Iliamna River	**19030206000032**	Y *	27.9	36.5	40.6	0.0	104.9	10.2
Chinkelyes Creek	19030206005773	N *	0.3	0.0	0.0	0.0	0.3	13.4
	19030206005761	N *	0.5	2.7	0.0	0.0	3.2	14.5
	19030206005759	N *	0.4	0.0	0.0	0.0	0.4	18.0
	19030206005754	N *	0.7	1.0	0.0	0.0	1.7	21.6
	19030206005737	N *	0.0	8.5	0.0	0.0	8.5	22.1

Notes:

Values (lengths) are arranged by 12-digit HUC from west (top) to east (bottom) along the transportation corridor. Each upstream value is a sum of NHD stream segment lengths in the HUCs between the crossing and upper extent of salmonid habitat potential based on 12% gradient. Each downstream value is a sum of stream segment lengths in the HUCs between the crossing and Iliamna Lake. Because the lengths at each crossing represent contiguous lengths, a portion of stream may be included in more than one crossing.

[a] Small headwater streams = 0–0.15 m^3/s; medium streams = 0.15–2.8 m^3/s; small rivers = 2.8–28 m^3/s; large rivers = >28 m^3/s.

[b] 190302060701.

[c] 190302051404.

[d] 190302060904.

[e] AWC stream code used, because no corresponding NHD stream code (and no upstream habitat data) available.

NHD = National Hydrography Dataset; AWC = Anadromous Waters Catalog; HUC = hydrologic unit code.

Source: AWC data from Johnson and Blanche (2012); NHD data from USGS (2012).

Table 10-7. Stream lengths downstream of road-stream crossings, classified by stream size. Stream size was based on mean annual streamflow; downstream length was measured from the road-stream crossing to Iliamna Lake.

HUC-12 Name or Description	Downstream Length (km)				
	Small Headwater Streams[a]	Medium Streams[a]	Small Rivers[a]	Large Rivers[a]	Total
Headwaters Upper Talarik Creek	2.1	9.0	36.5	0.0	47.6
Upper Tributary to Upper Talarik Creek[b]	0.8	8.3	0.0	0.0	9.1
Tributary to Newhalen River[c]	4.1	14.5	0.0	0.0	18.6
Headwaters Newhalen River	0.9	0.0	0.0	8.3	9.2
Outlet Newhalen River	3.0	1.3	0.0	23.7	28.0
Roadhouse Creek	11.4	11.4	0.0	0.0	22.8
Iliamna Lake–Eagle Bay	4.4	11.9	0.0	0.0	16.3
Eagle Bay Creek	2.8	8.1	0.0	0.0	10.9
Youngs Creek Mainstem (Roadhouse Mountain HUC)	0.0	4.2	0.0	0.0	4.2
Youngs Creek East Branch[d]	0.8	8.0	0.0	0.0	8.7
Chekok Creek	2.9	0.0	5.8	0.0	8.7
Canyon Creek	4.8	0.0	6.5	0.0	11.3
Iliamna Lake–Knutson Bay	16.0	2.9	0.0	0.0	18.9
Knutson Creek	1.8	0.0	2.9	0.0	4.6
Iliamna Lake–Pedro Bay	6.8	5.5	0.0	0.0	12.3
Iliamna Lake–Pile Bay	3.5	4.5	0.0	0.0	8.0
Outlet Pile River	1.2	0.7	3.2	0.0	5.2
Middle Iliamna River	0.0	0.7	10.2	0.0	10.9
Chinkelyes Creek	1.3	4.4	10.7	0.0	16.4
Total length across all HUCS	**68.6**	**95.4**	**75.7**	**32.0**	**272**
Percentage across all HUCS	25%	35%	28%	12%	100%

Notes:

Values (lengths) are arranged by 12-digit HUC, from west (top) to east (bottom) along the transportation corridor. Downstream values are the sum of National Hydrography Dataset stream segment lengths in the HUCs between the crossing and Iliamna Lake.

[a] Small headwater streams = 0–0.15 m^3/s; medium streams = 0.15–2.8 m^3/s; small rivers = 2.8–28 m^3/s; large rivers = >28 m^3/s.

[b] 190302060701.

[c] 190302051404.

[d] 190302060904.

HUC = hydrologic unit code.

Table 10-8. Lengths of different stream sizes that occur upstream of road-stream crossings and are likely to support salmonids (based on stream gradients of less than 12%).

HUC-12 Name or Description	Upstream Fish Habitat Length (km)				
	Small Headwater Streams[a]	Medium Streams[a]	Small Rivers[a]	Large Rivers[a]	Total
Headwaters Upper Talarik Creek	69.5	17.8	0.0	0.0	87.4
Upper Tributary to Upper Talarik Creek[b]	36.5	19.7	0.0	0.0	56.2
Tributary to Newhalen River[c]	37.7	15.9	0.0	0.0	53.6
Headwaters Newhalen River	55.8	29.3	0.0	13.1	98.2
Outlet Newhalen River	11.9	2.6	0.0	0.0	14.5
Roadhouse Creek	1.7	0.0	0.0	0.0	1.7
Iliamna Lake–Eagle Bay	2.4	1.5	0.0	0.0	4.0
Eagle Bay Creek	15.6	5.5	0.0	0.0	21.2
Youngs Creek Mainstem (Roadhouse Mountain HUC)	25.7	16.3	0.0	0.0	42.0
Youngs Creek East Branch[d]	32.9	12.4	0.0	0.0	45.3
Chekok Creek	41.9	42.5	7.9	0.0	92.3
Canyon Creek	0.0	1.2	8.6	0.0	9.8
Iliamna Lake–Knutson Bay	11.0	0.0	0.0	0.0	11.0
Knutson Creek	0.6	3.2	1.9	0.0	5.7
Iliamna Lake–Pedro Bay	0.3	0.0	0.0	0.0	0.3
Iliamna Lake–Pile Bay	0.0	1.2	0.0	0.0	1.2
Outlet Pile River	38.3	28.3	50.0	0.0	116.6
Middle Iliamna River	27.9	36.5	40.6	0.0	104.9
Chinkelyes Creek	1.8	12.2	0.1	0.0	14.1
Total length across all HUCS	**411.7**	**246.2**	**109.1**	**13.1**	**780.1**
Percentage across all HUCS	53%	31%	14%	2%	100%

Notes:

Values (lengths) are arranged by 12-digit HUC, from west (top) to east (bottom) along the transportation corridor. Each upstream value is a sum of National Hydrography Dataset stream segment lengths in the HUCs between the crossing and upper extent of salmonid habitat potential based on 12% gradient.

[a] Small headwater streams = 0–0.15 m^3/s; medium streams = 0.15–2.8 m^3/s; small rivers = 2.8–28 m^3/s; large rivers = >28 m^3/s.

[b] 190302060701.

[c] 190302051404.

[d] 190302060904.

HUC = hydrologic unit code.

10.3.2.1 Exposure

Based on the assumption that crossings over streams with mean annual streamflows greater than 0.15 m^3/s would be bridged (Section 6.1.3), the transportation corridor would include 19 bridges, 12 over known anadromous streams and 7 over streams likely to support salmonids (Table 10-6). Mean annual streamflow at a crossing in the Eagle Bay Creek hydrologic unit code (HUC)-12 (reach code 19030206006663) was 0.14 m^3/s, but we assumed that this crossing would be bridged because the stream is anadromous and contains 11.3 km of upstream fish habitat. Culverts would be placed at all other stream crossings. Given that the transportation corridor would cross a total of 55 streams and rivers known or likely to support migrating or resident salmonids, culverts would be constructed on 36 presumed salmonid streams.

Bridges would generally have fewer impacts on salmon than culverts, but could result in the loss of long riparian side channels if they did not span the entire floodplain. Approximately 500,000 bridges listed in the National Bridge Inventory are built over streams, and many of these, especially those on more active streams, will experience problems with aggradation, degradation, bank erosion, and lateral channel shift during their useful life (FHWA 2012).

Where flow restrictions such as culverts are placed in stream channels, stream power increases. This can lead to increased channel scouring and down-cutting, streambank erosion, and undermining of the road. Salmonids and other riverine fishes 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 funneling flow from the entire floodplain through the culvert and into the main channel. High water velocities in a stream channel may result from storm and snowmelt flows being forced through a culvert rather than spreading across the floodplain. Higher velocities cause scouring and down-cutting of the channel downstream of the culvert. This downstream erosion can result in perched culverts, impairing fish access to upstream reaches. In addition, it can hydrologically isolate the floodplain from the channel and block fish access to floodplain habitat. Entrenchment of the channel also prevents fish from reaching slow-water refugia during high-flow events and reduces nutrient and sediment cycling processes between the stream channel and the floodplain. Lastly, channel entrenchment may cause a change in the water table and the extent of the hyporheic zone, with consequences for floodplain water-body connectivity and water temperatures in the floodplain habitat.

Culverts are deemed to have failed if fish passage is blocked (e.g., by debris, ice, beaver activity, or culvert perching) or if streamflow exceeds culvert capacity and results in overtopping and road washout. Reported culvert failure frequencies vary in the literature but are generally high. Values of 30% (Price et al. 2010), 53% (Gibson et al. 2005), and 61% (Langill and Zamora 2002) have been reported, for an average culvert failure estimate of 48% (i.e., culvert surveys indicate that, on average, 48% block or inhibit fish passage at any given time).

When culverts are plugged by debris or overtopped by high flows, road damage, channel realignment, and severe sedimentation often result (Furniss et al. 1991). Changes in sediment load due to culvert failures can change stream hydraulics and geomorphic pressures. Generally, habitat value in the stream is diminished as the channel becomes wider and shallower and silt is deposited in the streambed. Stream crossing failures that divert streamflow outside of stream channels are particularly damaging and persistent (Weaver et al. 1987).

Free access to spawning and early rearing habitat in headwater streams is critical for a number of fish species, and culverts are common migration barriers. Culvert blockages are usually caused by woody debris and sometimes by woody material used by beavers to block a culvert and create a pond. In addition, aufeis—an ice feature that forms when water in or adjacent to a stream channel rises above the level of an existing ice cover and gradually freezes to produce a thickened ice cover (Slaughter 1982)—can completely fill culverts. When this occurs, water will run over the roadway unless flow is initiated through the culvert (Kane and Wellan 1985). The ice also reduces the cross-sectional area of flow so that high headwater conditions (and higher velocities than indicated by the culvert design) are produced during periods of peak flow. In some cases, considerable ice remains after the breakup period, particularly upstream of the culvert in the channel and floodplain (Kane and Wellan 1985).

Blockages could persist for as long as the intervals between culvert inspections. We assume that the transportation corridor would receive daily inspection and maintenance during operation of the mine. The level of surveillance along the corridor can be expected to affect the frequency of culvert failure detection. Driving inspections would likely identify a single erosional failure of a culvert that damaged the road or debris blockage sufficient to cause water to pool about the road, and in such cases temporary repairs would be made to protect the road. However, long-term fixes may not be possible until conditions are suitable to replace a culvert or bridge crossing. Further, multiple failures such as might occur during an extreme precipitation event would likely take longer to repair. These fixes may not fully address fish passage, which may be reduced or blocked for longer periods. Also, some failures that would reduce or block fish passage (e.g., gradual downstream channel erosion resulting in a perched culvert) might not be noticed by a driving inspection. Thus, blockage of migration could persist for an extended period. Extended blockage of migration would be less likely if daily road inspections included stops to inspect both ends of each culvert.

After mine operations end, traffic would decrease to that which is necessary to maintain any residual operations on the site, and inspections and maintenance would likely decrease. If the road was adopted by the state or local government, the frequency of inspections and quality of maintenance would likely 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.

10.3.2.2 Exposure-Response

Blockage of a culvert by debris or downstream erosion would inhibit the upstream and downstream 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 adult migration periods and persisted for several days. It could cause the loss of a year class of salmon from a stream if it occurred during juvenile migration periods and persisted for several days or more.

Erosional failure of a road resulting from failure of a culvert would create suspended sediment that would be carried and deposited downstream. Relationships between the concentration and duration of elevated sediment concentrations and effects on fish and invertebrates are presented in Section 9.4.2.1.

10.3.2.3 Risk Characterization

The mine scenarios specify that culverts would be installed along the transportation corridor with adequate size for normal flows of 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. Although culverts would be designed to certain specifications (Box 10-2), they are not always installed correctly or do not stand up to the rigors of a harsh environment, as indicated by the failure frequencies cited in Section 10.3.2.1. The transportation corridor would traverse varied terrain and subsurface soil conditions, including extensive areas of rock excavation in steep, mountainous terrain where storm runoff can rapidly accumulate and result in intense local runoff conditions (Ghaffari et al. 2011). Although the road design, including placement and sizing of culverts, would take into account seasonal drainage and spring runoff requirements, culvert failures would still be expected. For example, heavy rains in late September 2003 washed out sections of the Williamsport–Pile Bay Road (Lake and Peninsula Borough 2009), and culverts on this road have been washed out on numerous occasions (PLP 2011: Appendix 7.3A).

Culverts are not always built to specifications and the behavioral responses of migrating salmonid life stages to culvert-induced changes in flow are not always anticipated correctly. Standards for culvert installation on fish-bearing streams in Alaska mainly consider fish passage (ADF&G and ADOT 2001). Additional factors unrelated to fish passage, such as the physical structure of the stream or habitat quality, are addressed on a project-specific basis during preparation of the Alaska Department of Transportation and Public Facilities environmental document. Culvert capacities are allowed to be less than channel capacity (ADF&G and ADOT 2001). In most cases culvert width must be greater than 90% of the ordinary high-water channel width, but where channel slope is less than 1.0% culvert width must only be greater than 75% of the ordinary high-water channel width. During flood flows, this reduced channel width results in slower than normal velocities upstream of the culvert and higher water velocities exiting the culvert. This could result in scoured downstream channel beds, altered channel dynamics, and disassociated channels and floodplains. These processes would reduce the capacity of downstream reaches to support salmonids. High flows in and immediately downstream of the culvert, as well as the structure of the culvert itself, could inhibit fish passage even if movement is not blocked. Downstream erosion could result in perched culverts that, if they were not inspected and maintained, would inhibit and ultimately block fish passage. Floodplain habitat and floodplain/channel ecosystem processes could be disrupted by channel entrenchment resulting from culvert-induced erosion. These potential reductions in downstream habitat quality and inhibited fish passage could occur in any of the 36 culverted streams that likely support salmonids.

BOX 10-2. CULVERT MITIGATION

Bridge or culvert installation and maintenance activities in fish-bearing water bodies require a fish habitat permit. Permit application information requirements for culvert installations in fish streams are detailed in a memorandum of agreement (MOA) between the Alaska Department of Fish and Game (ADF&G) and the Alaska Department of Transportation and Public Facilities (ADOT) (ADF&G and ADOT 2001). The MOA provides guidance to project designers and permitting staff to ensure that culverts are designed and installed to provide efficient fish passage and to ensure statewide consistency in Title 16 permitting of culvert related work. Title 16 is the statute by which the ADF&G performs Fish Habitat and Special Area permitting.

Fish habitat regulations under Title 16 include the Anadromous Fish Act and the Fishway (or Fish Passage) Act.

- The **Anadromous Fish Act** (AS 16.05.871-.901) requires that an individual or government agency provide prior notification and obtain permit approval from ADF&G before altering or affecting "the natural flow or bed" of a specified water body or fish stream. All activities within or across a specified anadromous water body—including construction; road crossings; gravel removal; mining; water withdrawals; the use of vehicles or equipment in the waterway; stream realignment or diversion; bank stabilization; blasting; and the placement, excavation, deposition, or removal of any material—require approval from ADF&G's Division of Habitat.
- The **Fishway (or Fish Passage) Act** (AS 16.05.841), requires that an individual or government agency notify and obtain authorization from the ADF&G's Division of Habitat for activities within or across a stream used by fish if it is determined that such uses or activities could represent an impediment to the efficient passage of resident or anadromous fish.

The MOA describes the procedures, criteria and guidelines used for permitting culvert related work in fish-bearing waters; these criteria augment but do not replace ADOT's standard design criteria presented in the Alaska Highway Drainage Manual (ADOT 1995). Culverts are designed and permitted using one of the following design approaches.

- **Tier I—Stream Simulation Design** (developed by the U.S. Department of Agriculture, Forest Service [FSSSWG 2008]). The Tier 1 approach most clearly replicates natural stream conditions, and is applicable in stream gradients less than 6%. Using this design, culverts are sized larger than culverts sized hydraulically for floodwater conveyance alone. The culvert width at the ordinary high water (OHW) stage waterline must be greater than 90% of the OHW width. The culvert grade should approximate the channel slope, but in no instance should it deviate more than 1% from the natural grade. In stream channels with slopes less than 1%, culverts may be installed at slopes less than 0.5% with culvert widths greater than 75% of the OHW width.
- **Tier II—FISHPASS Program Design.** Under this approach, culverts are designed using a combination of traditional hydraulic engineering methods and the Alaska Interagency Fish Passage Task Force's 1991 "FISHPASS" computer modeling program (Behlke et al. 1991). The FISHPASS program evaluates component hydraulic forces in a culvert against a fish's available power and energy capabilities.
- **Tier III—Hydraulic Engineering Design.** The Tier III approach is used where site-specific conditions preclude use of Tier I and Tier II designs. Under this approach, professionally recognized hydraulic engineering methods are used to ensure appropriate fish passage characteristics in the culvert.

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. 2005). In a study of 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 10-6 shows that, of the 55 known or likely salmonid-supporting streams that would be crossed by the transportation corridor, 39 contain less than 5.5 km of habitat (stream length) upstream of the proposed road crossings. These 39 stream crossings contain a total of 68 km of upstream habitat and 493 km of downstream habitat. Seven of these crossings would be bridged, leaving 32 with culverts. Assuming typical maintenance practices after mine operations, roughly 48% of these streams, or 15 streams, would be entirely or partially blocked at any one time. As a result, these streams would likely not be able to support long-term populations of resident species such as rainbow trout or Dolly Varden.

The risk of culvert failures is somewhat uncertain due to the paucity of literature on culvert failures both in Alaskan taiga and tundra and for modern mining roads crossing salmonid habitat. The most relevant studies on potential effects of roads, particularly as they relate to salmon, are from forest and rangeland roads. These roads may differ in important ways from mining roads. Forested streams inevitably carry more woody debris that could block culverts. However, forested vegetation types represent 68% of the potential transportation corridor area mapped by Pebble Limited Partnership (PLP) (2011: Chapter 13). Mine roads carry much heavier loads than logging roads, but would likely be better engineered. For example, the transportation corridor in this assessment would be designed to support 190-ton haul truck travel on the road surface (Ghaffari et al. 2011), compared to an average gross legal weight limit of approximately 44 tons per log truck (Mason et al. 2008). In any case, the culvert failure frequencies cited in this assessment are from modern roads and not restricted to forest roads, and represent the most relevant data available.

10.3.3 Chemical Contaminants

In this section we address three sources of potentially toxic chemicals related to the transportation corridor: traffic residues, road construction, and road treatment and chemical cargos.

During runoff events, traffic residues produce a contaminant mixture of metals (e.g., lead, zinc, copper, chromium, and cadmium), oil, and grease that can get washed into streams and accumulate in sediments (Van Hassel et al. 1980) or disperse into groundwater (Van Bohemen and Van de Laak 2003). It is unclear if the transportation corridor would have sufficient traffic to contaminate runoff with significant amounts of metals or oil (although stormwater runoff from roads at the mine site itself is more likely to contain metal concentrations sufficient to affect stream water quality). Therefore, this risk is not considered further.

Road construction involves the crushing of minerals for the road fill and bed and the exposure of rock surfaces at road cuts, which leads to leaching of minerals and increased dissolved solids. Fish mortality in streams, with effects on populations recorded as far as 8 km downstream, has been related to high concentrations of aluminum, manganese, copper, iron, or zinc from highway construction activities in geological formations containing pyritic materials (Morgan et al. 1983). Because it is not clear where materials for the road will come from or their composition, this risk is not considered further.

Two potentially significant contaminants of aquatic habitats may occur along the transportation corridor: chemicals released during spills from truck accidents and stormwater runoff of salts or other materials used for winter road treatment. It should also be noted that increased runoff associated with roads may increase 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 of stormwater runoff are not assessed, however, because they are highly location-specific and not quantifiable given available data. Increases in sediment associated with stormwater runoff are addressed in Section 10.3.4.

10.3.3.1 Exposure

Many chemical reagents would be used to process ore (Box 4-5), and these chemicals would be transported by road to the mine site. Truck accidents along the transportation corridor could spill reagents into wetlands or streams. To estimate how much reagent and thus how many transport trucks would be needed for the mine scenarios, we extrapolated from the number of trucks required to transport reagents at a smaller gold mine (175 trucks per year at Pogo Mine) to the mine scenarios, based on the relative annual ore production at the two mines. Assuming 20 tons of reagent per truck and expected annual production rates of 3,000 tons per day at Pogo Mine (USEPA 2003a) and 200,000 tons per day in the mine scenarios (Ghaffari et al. 2011), we estimate that transport of reagents would require approximately 11,725 truck trips per year.

The length of the transportation corridor within the Kvichak River watershed would be 113 km. The probability of truck accidents and releases was reported as 1.9×10^{-7} spills per mile of travel for a rural two-lane road (Harwood and Russell 1990). Based on this rate, the number of spills over the roughly 25-year life of the Pebble 2.0 scenario would be 3.9—that is, approximately 4 spills from truck accidents would be expected during mine operations. Over the roughly 78-year life of the Pebble 6.5 scenario, 12 spills would be expected. Only one-way travel is considered, because return trips from the mine would be with empty trucks or with a load other than process reagents. Because conditions on the mine road would be different from those for which the statistics were developed (e.g., more difficult driving and road conditions), this calculation provides an order of magnitude estimate. The reasonableness of this estimate is suggested by an assessment of the Cowal Gold Project in Australia, which estimated that a truck wreck would occur every 1 to 2 years, resulting in a spill every 3 to 6 years (NICNAS 2000).

For 14% of its length (15 km), the transportation corridor would be within 100 m of a stream or river (Table 10-3), and for 24% of its length it would be within 100 m of a mapped wetland (Table 10-4). If the probability of a chemical spill is independent of location, and if it is assumed that liquid spills within 100 m of a stream could flow to that stream, a spill would have a 14% probability of entering a stream within the Kvichak River watershed. This would result in roughly 0.5 stream-contaminating spills over the 25-year life of the Pebble 2.0 scenario or up to 2 stream-contaminating spills over the 78-year life of the Pebble 6.5 scenario. Similarly, a spill would have a 24% probability of entering a wetland, resulting in an estimate of 1 wetland-contaminating spill in the Pebble 2.0 scenario or 3 wetland-contaminating spills in the Pebble 6.5 scenario. A portion of those wetlands would be ponds or backwaters that support fish. It should be noted that the risk of spills could be somewhat mitigated by using spill-resistant containers.

Cyanide for gold processing would be transported as a solid. We assume containment equivalent to that at the Pogo mine (i.e., dry sodium cyanide pellets inside plastic bags inside wooden boxes inside metal shipping containers). Hence, even in a truck wreck, a cyanide spill is an unquantifiable but low probability occurrence. A spill on land could be collected, but during periods of rain or snowmelt it would rapidly dissolve and wash into surface or groundwater. A spill of pellets into a stream or wetland would rapidly dissolve and dissociate into free ions or, depending on the pH, hydrogen cyanide. Pellets spilled into a stream would be transported downstream as described for the copper concentrate (Section 11.3), but, rather than slurry water and solids, the transported material would consist of dissolving pellets and increasing cyanide or hydrogen cyanide solution.

In addition to process chemicals, the molybdenum concentrate (primarily molybdenum sulfide) would be transported by truck. The concentrate would be a dewatered fine granular material contained in bags packed in shipping containers. Thus, as with cyanide, a spill of molybdenum concentrate is an unquantifiable but low probability occurrence. A spill on land could be collected. A spill into water would be transported by streamflow as described for the copper concentrate (Section 11.3). Settled concentrate would oxidize, forming acidic pore water with dissolved molybdenum to which benthic invertebrates and fish eggs and larvae could be exposed.

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 calcium chloride levels in runoff or streams from roads treated in this way.

10.3.3.2 Exposure-Response

A principle processing chemical of concern would be sodium ethyl xanthate (Section 6.4.2.3). A risk assessment by Environment Australia estimated that a spill of as little as 10% of a 25-metric-ton-capacity truck carrying sodium ethyl xanthate into a stream would require a "650000:1 dilution before the potential hazard is considered acceptable" and that the spill could not be mitigated (NICNAS 2000).

Cyanide has acute and chronic U.S. ambient water quality criteria for freshwater of 22 and 5.2 µg free cyanide per liter. The geometric mean of 30 median lethal concentration (LC_{50}) values from acute tests of rainbow trout is 55.7 µg/L (USEPA 1985, 2013). In a 2-hour exposure to 10 µg/L cyanide, swimming speed of coho salmon was reduced (USEPA 1985). Unlike metals, cyanide is not more toxic to invertebrates than fish. Standard acute endpoints for invertebrates range from 17 to 210,000 µg/L (USEPA 1985, 2013).

Molybdenum's aquatic toxicity is relatively poorly characterized. The most directly relevant values are 28-day LC_{50} values for rainbow trout eggs of 730 and 790 µg/L (Birge 1978, Birge et al. 1979). The mean of two acute lethality tests with rainbow trout is 1,060,000 µg/L (USEPA 2013). Acute and chronic values for *Daphnia* are 206,800 and 4,500 µg/L (USEPA 2013). Hence, molybdenum appears to be much less toxic than copper. However, the small body of test data and lack of information on the influence of water chemistry on toxicity make judgments about the effects of aqueous molybdenum much more uncertain than copper or many other metals. Also unlike copper, there are no whole sediment benchmarks for molybdenum.

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). Based on that study, the toxicity of the calcium chloride commonly used in Alaska would be expected to be a little greater than the more studied sodium chloride, based on total chlorine concentrations. Alaska acute and chronic water quality standards for chloride are 860 and 230 mg/L, respectively (ADEC 2003). However, these values may not provide adequate protection from calcium salts. In addition, exceedances of the acute criterion could affect many species, because freshwater biota have a narrow range of acute susceptibilities to chloride (ADEC 2003). These standards and the associated federal criteria also may not be adequately protective due to the absence of tests of critical life stages (e.g., egg fertilization).

Rainwater tends to leach out the highly soluble chlorides (Withycombe and Dulla 2006), which can degrade nearby vegetation, surface water, groundwater, and aquatic species (Environment Canada 2005). Salmonids are sensitive to salinity, particularly at fertilization (Weber-Scannell and Duffy 2007). According to Bolander and Yamada (1999), application of chloride salts should be avoided within at least 8 m of water bodies (including shallow groundwater, if significant migration of chloride would reach the groundwater table), and restricted if low salt-tolerant vegetation occurs within 8 m of the treated area. 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).

10.3.3.3 Risk Characterization

Given the liquid form and toxicity of sodium ethyl xanthate (Section 8.2.2.5), it is expected that a spill of this compound into a stream along the transportation corridor would cause a fish kill. Runoff or groundwater transport from a more distant spill would cause effects that would depend on the amount of dilution or degradation occurring before the spilled material entered a stream. Although other process chemicals would also be used, xanthate is representative of the chemicals estimated to result in roughly two stream-contaminating spills over the 78-year life of the Pebble 6.5 scenario.

Cyanide pellets spilled by a truck wreck into a stream would be carried by the current but would rapidly dissolve into a cyanide solution and would ultimately disperse, volatilize, and degrade in Iliamna Lake. Spills into a wetland would dissolve in place. Spills on land would be collected unless they occurred during rain or snowmelt, in which case spilled pellets would dissolve and flow to surface or groundwater. Data needed to derive a cyanide spill scenario and quantify risks are unavailable, but given the toxicity of cyanide and its rapid action, effects on invertebrates and fish, including death, would be likely if a substantial spill into a stream or wetland occurred.

Molybdenum concentrate spilled by a truck wreck into a stream would be carried by the current and deposited in pools and backwaters and ultimately in Iliamna Lake. Compared to copper concentrate, relatively little is known about molybdenum concentrate. The solubility of the molybdenum in the Aitik copper concentrate is undefined but appears to be relatively low (Appendix H: Tables H-8 and H-9), and molybdenum is much less toxic than copper. The frequency of truck passages is also unknown, so the spill risk is unquantified. Therefore, the ecological risk from a molybdenum spill is unquantifiable but appears to be low relative to the risk from a copper concentrate spill (Section 11.3).

Risks to salmonids from de-icing salts and dust suppressants could be locally significant, but would depend on the amount and frequency of application. The transportation corridor would intersect 55 streams and rivers known or likely to support salmonids, and there would be approximately 272 km of streams between road crossings and Iliamna Lake (Table 10-7). Additionally, approximately 12 km of roadway would intersect wetlands within and beyond those mapped by NWI. Runoff from these road segments could have significant effects on fish and the invertebrates that they consume, particularly if sensitive life stages are present.

10.3.4 Fine Sediment

10.3.4.1 Exposure

During rain and snowmelt, soil eroded from road cuts, borrow areas, road surfaces, shoulders, cut-and-fill surfaces, and drainage ditches (as well as road dust deposited on vegetation; see Section 10.3.5), would be washed into streams and other water bodies. Erosion and siltation are likely to be greatest during road construction. The main variables determining surface erosion are the inherent erodibility of the soil, slope steepness, surface runoff, slope length, and ground cover. Mitigation measures for fine sediments are discussed in Box 10-3. It is worth noting that improvements have been proposed for the road between Iliamna and Nondalton, in part to alleviate erosion and sedimentation problems at some areas along the road (ADOT 2001).

BOX 10-3. STORMWATER RUNOFF AND FINE SEDIMENT MITIGATION

The Alaska Department of Environmental Conservation (ADEC) administers Alaska Pollutant Discharge Elimination System (APDES) stormwater general permits for construction activities and multi-sector general permits for industrial operation activities. ADEC also approves stormwater pollution prevention plans (SWPPPs) that include stormwater best management practices (BMPs).

A permittee covered under the APDES stormwater general permit for construction activities (ADEC 2011a) must comply with control measures that are determined by site-specific conditions. ADEC developed the *Alaska Storm Water Guide* (ADEC 2011b) to assist permittees with selecting, installing, and maintaining control measures that may be used for projects in Alaska. Erosion and sediment control measures covered under the stormwater general permit for construction activities (ADEC 2011a) are summarized below.

Erosion Control Measures

- Delineate the site, specifically the location of all areas where land disturbing activities will occur and areas that will be left undisturbed (e.g., boundaries of sensitive areas or established buffers).
- Minimize the amount of soil exposed during construction activity by preserving areas of native topsoil on the site where feasible and sequencing or phasing construction activities to minimize the extent and duration of exposed soils.
- Maintain natural buffer areas.
- Control stormwater discharges and flow rates, via the following mechanisms:
  - Diversion of stormwater around the site.
  - Slow down or containment of stormwater that collects and concentrates at the site.
  - Avoidance of structural control measure placement in active floodplains, to the degree practicable and achievable.
  - Placement of velocity dissipation devices (e.g., check dams, sediment traps, or riprap) along conveyance channels and where discharges from conveyance channels join water courses.
- Protect steep slopes, via the following mechanisms:
  - Design and construction of cut-and-fill slopes to minimize erosion.
  - Diversion of concentrated stormwater flows away from and around the disturbed slopes, using interceptor dikes, swales, grass-lined channels, pipe slope drains, surface drains, and check dams.
  - Stabilization of exposed slope areas.

Sediment Control Measures

Sediment control measures (e.g., sediment ponds, traps, filters) should be functional before other land-disturbing activities take place. These measures may include:

- Storm drain inlet protection measures (e.g., filter berms, perimeter controls, temporary diversion dikes), that minimize the discharge of sediment prior to entry into the inlet for storm drain inlets located on site or immediately downstream.
- Water body protection measures (e.g., velocity dissipation devices) that minimize the discharge of sediment prior to its entry into water bodies located on site or immediately downstream.
- Down-slope sediment controls (e.g., silt fences, temporary diversion dikes) for any portion of the down- and side-slope perimeters where stormwater would be discharged from disturbed areas of the site.
- Establishment and stabilization of construction vehicle access and exit points, limited to one route if possible.
- Minimization of dust generation through the application of water or other dust suppression techniques prior to vehicle exit.
- Stabilization or coverage of soil stockpiles, protection with sediment trapping measures, and, where possible, location away from storm drain inlets, water bodies, and conveyance channels.
- Design of sediment detention basins to capture runoff or conveyed stormwater and reduce water velocity to allow sediments to settle out before they can enter streams or other water bodies. Storm flows eventually pass through an outflow structure leaving the sediment (i.e., solids that can settle) in the basin. There are important design and management considerations for sediment detention basins for hard rock mining (USEPA 2003b: Appendix H, Section 6.1.6).

Soil Stabilization

All disturbed areas of the site should be stabilized to minimize on-site erosion and sedimentation and the resulting discharge of pollutants according to the requirements in ADEC (2011a). Existing vegetation should be preserved wherever possible.

Many of the BMPs for industrial operations associated with metal mining focus on sediment and erosion control and are similar to BMPs used in the construction industry (USEPA 2006). Some of these BMPs pertain specifically to haul and/or access roads (USEPA 2006).

- Construction of haul roads should be supplemented by BMPs that divert runoff from road surfaces, minimize erosion, and direct flow to appropriate channels for discharge to treatment areas. Examples of these BMPs include:
  - Dikes, curbs, and berms for discharge diversions.
  - Conveyance systems such as channels, gutters, culverts, rolling dips and road sloping, and/or roadway water deflectors.
  - Check dams, rock outlet protection, level spreaders, stream alternation, and drop structures for runoff dispersion.
  - Gabions, riprap, native rock retaining walls, straw bale barriers, sediment traps/catch basins, and vegetated buffer strips for sediment control and collection.
  - Vegetation to stabilize soils.
- Roads should be placed as far as possible from natural drainage areas, lakes, ponds, wetlands, and floodplains.
- Width and grade of roads should be as small as possible to meet regulatory requirements and designed to match the area's natural contours.

All stabilization and structural erosion control measures should be inspected frequently and all necessary maintenance and repairs should be performed.

10.3.4.2 Exposure-Response

Sediment loading from roads can severely affect streams downstream of the roadbed (Furniss et al. 1991). Salmonids are adapted to episodic exposures to suspended sediment, but survival and growth can be affected as concentrations or durations of exposure increase (Section 9.4.2.1). Increased deposition of fine sediment decreases the abundance and production of fish and benthic invertebrates (Section 9.4.2.2). Fine sediments have been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, increased predation on fish, and reduced benthic organism populations and algal production (Newcombe and MacDonald 1991, 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 and render it useless for spawning, and sediment deposited after spawning may smother eggs and alevins. Excessive stream sediment loading 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 reduced quality and quantity of available spawning habitat (Furniss et al. 1991).

Increased runoff associated with roads may increase rates and extent of erosion, reduce percolation and aquifer recharge rates, alter channel morphology, and increase stream discharge rates (Forman and Alexander 1998). During high-discharge events and in high velocity streams, accumulated sediment tends to be flushed out and re-deposited in larger water bodies (Forman and Alexander 1998). Because streams crossed by the transportation corridor connect downstream to Iliamna Lake and ponds, accelerated sedimentation could have an impact on the concentrated sockeye spawning populations in these habitats. Accelerated sedimentation could also have a localized impact on the clarity and chemistry of Iliamna Lake, affecting the photic zone (the depth of light penetration sufficient for photosynthesis) and thereby primary production and zooplankton abundance, which is critical to juvenile sockeye salmon.

10.3.4.3 Risk Characterization

Suspended and deposited sediment washed from roads, shoulders, ditches, cuts, and fills would likely 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 (Section 9.4) indicate that even relatively small amounts of additional sediment could have locally significant effects on reproductive success of salmonids and production of aquatic invertebrates. Potential mitigation measures for stormwater runoff, erosion, and sedimentation are discussed in Box 10-3.

10.3.5 Dust

Dust results from traffic operating on unpaved roads in dry weather, grinding and breaking down road materials into fine particles (Reid and Dunne 1984). These fines are either transported aerially in the dry season or mobilized by water in the wet season. Dust particles may also include trace contaminants, including de-icing salts, hydrocarbons, and metals. Following initial suspension by vehicle traffic, aerial transport by wind spreads dust over long distances, so that it can reach surface waters that are otherwise buffered from sediment delivery via aqueous overland flow (Appendix G). Dust control agents such as calcium chloride have been shown to reduce the generation of road dust by 50 to 70% (Bader 1997), but these agents may cause toxic effects when they run off and enter surface waters (Section 10.3.3).

10.3.5.1 Exposure

The amount of dust derived from a road surface is a function of many variables, including composition and moisture state of the surface, amount and type of vehicle traffic, and speed. An Iowa Highway Research Board project (Hoover et al. 1973) that quantified dust sources and emissions created by traffic on unpaved roads found that one vehicle, traveling 1 mile of unpaved road once a day every day for 1 year, would result in the deposition of 1 ton of dust within a 1,000-foot corridor centered on the road (i.e., traffic would annually deposit 1 ton of dust per mile per vehicle).

To estimate truck traffic required by the mine scenarios, we extrapolated from vehicle use at a smaller gold mine (Pogo Mine) based on the rate of ore production at Pogo relative to the mine scenarios. Estimated production rate at Pogo is 3,000 tons per day (USEPA 2003a), versus 200,000 tons per day in the mine scenarios (Ghaffari et al. 2011). Overall mine-related vehicle use at Pogo averages between 10 and 20 round trips per day (USEPA 2003a). Approximately 175 truck trips per year (0.5 round trip per day) are required at Pogo to transport reagents, leaving 19.5 round trips per day for other purposes. The number of truck trips required for transport of reagents is assumed to be roughly proportional to ore production, resulting in an estimate of 33 round trips per day to transport reagents in the assessment mine scenarios. The number of daily round trips for purposes other than reagent transport was estimated at 19.5 round trips per day, for a total daily traffic estimate of 52.5 round trips in the mine scenarios. This value is likely an underestimate, as it does not account for potential effects of size differences between Pogo Mine and the mine scenarios or the number of trips for purposes other than reagent transport.

The length of the transportation corridor within the Kvichak River watershed would be 113 km. Based on the estimate from Hoover et al. (1973), the average amount of dust (in tons) generated per mile of road per year along the transportation corridor within the Kvichak River watershed would be equivalent to the daily average number of vehicles passing along the corridor (one vehicle making a round-trip constituting two passages). Using this method, the mine scenarios would generate approximately 105 tons of dust per mile (59 metric tons per km) annually or approximately 6,700 metric tons annually for the entire length of road within the Kvichak River watershed. This value may be an underestimate because smaller vehicles typically use rural roads in Iowa, or an overestimate if roads in Iowa are drier or if dust suppression is effective. Regardless, it indicates that dust production along the transportation corridor could be substantial.

10.3.5.2 Exposure-Response

Walker and Everett (1987) evaluated the effects of road dust generated by traffic on the Dalton Highway and Prudhoe Bay Spine Road in northern Alaska. Dust deposition altered the albedo of snow cover, causing earlier (and presumably more rapid) snowmelt up to 100 m from the road margin and increased depth of thaw in roadside soils. Dust was also associated with loss of lichens, sphagnum, and other mosses and reduced plant cover (Walker and Everett 1987). Loss of near-roadway vegetation has important implications for water quality, as that vegetation helps to filter sediment from road runoff. Thus, dust deposition can contribute to stored sediment that can mobilize in wet weather, and deposition can reduce the capacity of roadside landscapes to filter that sediment.

In a study of road effects in Arctic tundra at acidic (soil pH less than 5.0) and less acidic (soil pH at least 5.0) sites, Auerbach et al. (1997) found that vegetation effects were more pronounced at the acidic site. Permafrost thaw was deeper next to than away from the road at both sites, and could affect road structure detrimentally. Vegetation biomass of most taxa was reduced near the road at both sites. Species richness in acidic tundra next to the road was less than half the richness at 100 m away from the road. Sphagnum mosses, dominant in acidic low arctic tussock tundra, were virtually eliminated near the road. According to PLP (2011: Chapter 5), approximately 72% of the mine area is composed of well-drained acidic soils (58% strongly acidic); approximately 34% of the transportation corridor is composed of well-drained acidic soils (3.5% strongly acidic).

10.3.5.3 Risk Characterization

The main impact of dust from the transportation corridor on salmonids likely would be reduced habitat quality due to a reduction in riparian vegetation and subsequent increase in suspended sediment and fine bed sediment, especially during road construction. Potential effects of increased sediment loading are discussed in Section 10.3.4. Loss of riparian vegetation would also occur at the mine site, but there the main impact of dust would be a direct increase in fine bed sediment due to mine construction and operation.

10.3.6 Invasive Species

10.3.6.1 Exposure

Construction and operation of the transportation corridor would increase the probability that new terrestrial and aquatic species would be transported to and could establish themselves in the Bristol Bay region. Roads can facilitate introductions via contaminated soil or gravel used in road construction and maintenance, or via contaminated vehicles, equipment, cargo, and people that travel those roads. For example, road fill appears to be the mode of introduction and spread for invasive sweetclover (*Melilotus alba*) in central and southeast Alaska (Wurtz et al. 2010). Elsewhere, road maintenance further spreads invasive plants along suitable roadside habitat (Christen and Matlack 2009). Vehicles can carry contaminated equipment and cargo. Over the 2-year construction of a research station in Antarctica, an estimated 5,000 seeds from 14 different plant families were introduced on almost 15,000 m^3 of cargo (Lee and Chown 2009). Once docked, seeds on cargo could disperse at almost any location along the transportation corridor. Finally, people unintentionally introduce and spread invasive species in Alaska and other Arctic environments on their shoes (Bella 2011, Ware et al. 2012).

Once established along or near the transportation corridor, terrestrial species that thrive in riparian and floodplain areas could spread to salmon-bearing habitat at any of the points where the road crosses a river, stream, or other aquatic habitat. In a survey of 2,865 km (1,780 miles) of major highways in interior and south-central Alaska, 64 of 192 sampled bridge crossings (over 30%) were found to have sweetclover adjacent to them, and sweetclover had spread to downstream floodplains at 17 of these bridge crossings (Wurtz et al. 2010). This survey likely underestimates the number of floodplain invasions, because it did not sample numerous stream crossings serviced by culverts or other locations along streams where fill had been placed.

Aquatic invasive species, including macrophytes, shellfish, and salmonid pathogens and parasites can also be introduced along the transportation corridor on equipment that has come into contact with contaminated waters. Most literature emphasizes recreation equipment (Johnson et al. 2001, Arsan and Bartholomew 2008); little or no information exists about the incidence of aquatic or riparian species introductions specifically on construction or mining equipment. Transported equipment contaminated with aquatic invaders could spread those species to salmon-bearing habitat via direct contact with anadromous waters during stream crossing construction or during mining activity. Aquatic invaders could also be carried by water to other salmon-bearing habitats downstream of the initial introduction locations, including into Iliamna Lake and other parts of the Kvichak River watershed.

The likelihood that an aquatic invasive species will establish and spread successfully can depend heavily on environmental requirements. For instance, *Myxobolus cerebralis*, a cnidarian parasite that causes whirling disease, has already been detected in an Anchorage, Alaska, trout hatchery. This parasite has very specific abiotic and biotic conditions under which it infects salmonids. If the pathogen is introduced to a new area, susceptible genetic variants of the secondary host (the oligochaete worm *Tubifex tubifex*) must be present, seasonal water temperatures must exceed 10°C with approximately 1,500 degree-days, and susceptible salmonid species and life-stages must co-occur with the secondary host (Arsan and Bartholomew 2008). In addition to the hatchery location where whirling disease has already been found, favorable conditions exist for parasite establishment in two tributaries of Cook Inlet near Anchorage (Arsan and Bartholomew 2008). However, conditions for whirling disease establishment are not known for the Bristol Bay region.

10.3.6.2 Exposure-Response

Invasive species can drastically alter the composition of riparian and floodplain vegetation adjacent to salmon habitats. Invasive sweetclover, purple loosestrife (*Lythrum salicaria*), and giant knotweed (*Polygonum sachalinense*)—all current invaders in Alaska—can replace native riparian species (Blossey et al. 2001, Urgenson et al. 2009, Spellman and Wurtz 2011). In general, it has been difficult to show a direct effect of riparian vegetation alteration on fish diversity, abundance, or biomass (Smokorowski and Pratt 2007), but indirect effects on salmon via aquatic foodwebs have been documented (Wipfli and Baxter 2010). Giant knotweed was shown to release nitrogen-poor litter into a tributary of the salmon-bearing Skagit River in Washington, which can have cascading, negative effects on fish by altering their invertebrate food sources (Urgenson et al. 2009). Purple loosestrife was found to decompose four times faster than native sedge in the Fraser River, making detritus available in fall rather than winter and spring, when it was usually used by invertebrates that support salmon production (Grout et al. 1997).

Links between aquatic invaders, particularly macrophytes, and fish performance have been made in lentic, but rarely in lotic, habitats (Smokorowski and Pratt 2007). Effects of invasive macrophytes range from increased native fish abundance, to no effect, to detrimental effects on fish and their food sources via exuded toxic compounds, depending on the invasive species and fish species of interest (Schultz and Dibble 2012). Streambed coverage of several aquatic macrophyte species, both native and introduced (including the recent Alaska invader *Elodea canadensis*), reduced the number of Chinook salmon redds and the percentage of available spawners observed using infested habitat in northern California (Merz et al. 2008). This is significant in the regulated, low-flow Mokelumne River in California, where spawning habitat is considered a limiting resource.

Evidence of the effects of other aquatic invaders on salmonids also exists. Didymo (*Didymosphenia germinata*) is a colonial diatom capable of covering stream substrates with thick, slippery mats. Documented effects of didymo on salmonids vary with location and fish species. Effects of didymo on the invertebrate communities that serve as fish food sources could ultimately affect salmonid growth and abundance (Whitton et al. 2009). The aquatic invader that causes whirling disease (*M. cerebralis*) has had devastating effects on several wild fisheries in the United States intermountain west (Nehring and Walker 1996). The disease can cause lesions, neurological defects, skeletal deformities, and death. Both sockeye salmon and rainbow trout fry are highly susceptible to whirling disease, should conditions be right for infection.

10.3.6.3 Risk Characterization

The spread of aquatic, riparian, and floodplain invasive species along roads and into salmon-bearing habitats could occur during construction and operation of the proposed transportation corridor, although mitigation measures can lower the likelihood of invasion (Box 10-4). Invasion of riparian and floodplain species is occurring in Alaska via the use of contaminated gravel road fill. In the case of invasive sweetclover, subsequent dispersal to almost 9% of floodplains downstream of bridges along one major highway was observed (Wurtz et al. 2010). Assuming similar rates of invasion along both the transportation corridor and bridges and culverts, 9% of the 64 streams and rivers—5 to 6 streams—crossed by the corridor in the Kvichak River watershed would experience invasion. Given that 55 of the 64 streams crossed by the transportation corridor are known or likely to support salmonids, alteration of salmon habitats would be expected in approximately 5 streams. However, this is almost assuredly an underestimate because it is based on rate of invasion for only one species and assumes that the spread of that species has reached equilibrium.

BOX 10-4. MITIGATION FOR INVASIVE SPECIES

The use of contaminated gravel road fill in Alaska has fostered the invasion of nonnative riparian and floodplain plant species. In some cases, the species are subsequently dispersed to floodplains downstream of road-stream crossings. Introduction and invasion of nonnative riparian and floodplain species may also occur via contaminated cargo, equipment, and boots. The following steps can help to mitigate the introduction and spread of invasive species.

- Purchase of fill from existing or new gravel pits certified by the Alaska Department of Natural Resources Division of Agriculture as weed-free (ADNR 2013).
- Proper and thorough inspection and de-contamination of cargo, equipment, and boots, at the port and at the mine site.
- Use of new equipment, where possible.
- Use of a process for cleaning, draining, and drying equipment previously used at another site (including personal gear worn by workers) that is advocated by the Alaska Department of Fish and Game for recreational equipment.

Should sweetclover, purple loosestrife, giant knotweed or other species invade riparian areas and floodplains adjacent to salmon-bearing streams and wetlands in the Bristol Bay region, they could change organic matter inputs into those streams and affect salmon food sources (Blossey et al. 2001, Urgenson et al. 2009, Spellman and Wurtz 2011). The extent to which salmon growth, diversity, or abundance would be altered would depend on the extent and intensity of infestation. Once initiated, these invasions would be difficult to reverse.

Invasions by aquatic species seem less likely but cannot be quantified. The most likely vector is believed to be construction equipment that has been used at stream crossings in a prior project. Such equipment could carry microbes or propagules in mud that could be transferred when constructing road and pipeline crossings in the Bristol Bay watershed.

The spread of invasive species is highly stochastic and there are no good, relevant models for risk estimation. Therefore, it is not as clear a threat as other issues considered in this assessment. However, the introduction and spread of invasive species has been a major cause of environmental degradation in the United States, and mitigation measures could reduce the risks (Box 10-4).

10.4 Overall Risk Characterization for the Transportation Corridor

Risks to salmonids from filling of wetlands, hydrologic modifications, spillage or runoff of contaminants and fine sediment, dust deposition, and introduction of invasive species are likely to diminish the production of anadromous and resident salmonids in many of the 55 streams known or likely to support salmonids that would be crossed by the transportation corridor. Salmonid spawning migrations and other movements may be impeded by culverts in 36 streams, 32 of which contain restricted (less than 5.5 km) upstream habitat. Assuming typical maintenance practices after mine operations, approximately 15 of these 32 streams would be entirely or partly blocked at any time. As a result, salmonid passage—and ultimately production—would be reduced in these streams, and they would likely not be able to support long-term populations of resident species such as rainbow trout or Dolly Varden. Approximately 272 km of streams downstream of road crossings also could be affected.

The migratory barriers and degradation of stream habitat discussed herein could also reduce the high genetic diversity among sockeye populations reported by Gomez-Uchida et al. (2011) and Quinn et al. (2012). This loss in diversity may decrease the long-term viability of sockeye salmon and would negatively affect localized watershed food webs.

Truck accidents may spill xanthates, cyanide, or molybdenum concentrate into streams crossed by the road. Xanthate and cyanide are highly toxic and could kill fish and invertebrates in the receiving streams and, depending on the size of the spill, portions of Iliamna Lake. Molybdenum concentrate is much less toxic and unlikely to cause severe effects.

The exact magnitudes of changes in fish productivity, abundance, and diversity cannot be estimated at this time, but the species, abundances, and distributions that could be affected are summarized below.

- Sockeye salmon spawning has been observed at 30 locations along the transportation corridor. Highest average abundances are in the Iliamna River (100,000 spawners), the Newhalen River (80,000 spawners), and Knutson Bay (70,000 spawners), although abundances can be much higher (e.g., 1 million adults were reported in 1960 survey of Knutson Bay).
- Chinook, coho, pink and chum salmon have been reported at isolated points in the Kvichak River watershed, and all four species have been observed in Upper Talarik Creek and the Iliamna River.
- Dolly Varden have been reported in nearly every sockeye salmon-bearing stream that would be crossed by or adjacent to the corridor, as well as in locations upstream of sites with reported anadromous salmon use.

- Rainbow trout have been reported in Upper Talarik Creek, the Newhalen River, an unnamed tributary to Eagle Bay, Youngs, Tomkok, and Swamp Creeks, the Iliamna River, and Chinkelyes Creek.

10.5 Uncertainties

In this chapter we evaluated the risks to salmonid habitats and populations associated with the transportation corridor (Figure 10-3). A number of uncertainties are inherent in assessing these risks, which are summarized below (uncertainties related to the effectiveness of mitigation measures are discussed in Box 10-5).

- **Characterization of streams and wetlands affected by the transportation corridor**. The NWI, NHD, AWC, and AFFI were used to evaluate the effects of the transportation corridor on hydrological features and fish populations (Box 10-1). These datasets include the following limitations.
    - Underestimation of the number of stream crossings and degree of channel sinuosity, resulting in underestimates of affected stream lengths.
    - Underestimation of fish-bearing streams due to limited sampling.
    - Potential undercharacterization of wetland area due to limited resolution of available NWI data.
    - Underestimation of potential impacts on wetlands bisected by the transportation corridor, because wetland area outside the 200-m boundary was assumed to maintain functionality.

  Overall, these uncertainties likely result in a moderate underestimation of risks to fish.
- **Estimation of dust production from the transportation corridor.** Our dust production estimate is based on a study that quantified dust sources and emissions created by traffic on unpaved roads. Extrapolating that study to the transportation corridor does not take into account variables such as composition and moisture of the road surface, number and width of tires, and speed. In addition, road dust generation may be reduced by 50 to 70% by the application of dust control agents such as calcium chloride. Overall, these uncertainties likely have a negligible effect on risks to fish, but a moderate effect on our dust production calculations.
- **Estimation of chemical spill frequency due to truck accidents**. Extrapolation of truck accident probability from a study of rural two-lane roads does not take into account specific, generally more difficult road and weather conditions prevalent in the area of the Pebble deposit. However, the risk of spills could be at least partially mitigated by using spill-resistant containers. Overall, these uncertainties likely result in a moderate underestimation of risk to fish because of effects on spill frequency calculations. Frequencies of cyanide and molybdenum concentrate spills were not estimated due to uncertainties in the mining scenarios.
- **Estimation of risks to salmonids from spills.** A spill of cyanide, xanthate, or molybdenum concentrate could occur in various ways and at various locations. The sparse literature on the aquatic chemistry and toxicology of xanthates and molybdenum makes the consequences of these

events particularly uncertain. Given its high toxicity, we are confident that toxic effects would occur following a xanthate spill into a stream; we are simply uncertain of the magnitude and extent of effects.

- **Estimation of culvert failure frequencies.** These frequencies, derived from the literature, assume that culverts are designed to specifications but are not always installed correctly and/or do not stand up to the rigors of a harsh environment. This uncertainty likely has a moderate effect on risks to fish, with unclear direction. Nonetheless, this does not change overall conclusions reached with respect to reduction of passage and ultimately production of salmonids or the viability of long-term populations of resident species.
- **Risks from invasive species.** Roads serve as corridors for the spread of weeds, pathogens, and other invasive species. However, the list of potential invaders is ill-defined and the rate of their spread along an industrial road is unknown.
- **Climate change effects.** The potential impacts of road construction and operation discussed in this chapter do not take into account potential effects of climate change. Over the timeframe considered in this assessment (approximately 80 years), the physical environment of the Bristol Bay watershed is likely to change substantially as a result of increases in temperature and precipitation (Section 3.8). Increases in rain-on-snow events are likely to increase flood frequency. Such changes could undermine the structure of the transportation corridor and its stream crossings. The variability and magnitude of streamflows could also enhance other impacts described in this chapter, including channel entrenchment and the loss of water-body connectivity. Collectively, these impacts would likely further reduce the diversity of fish habitat, causing a loss of population genetic diversity over time that would reduce the resiliency of salmon stocks to environmental fluctuations related to climate change. Overall, these climate-related uncertainties result in a moderate underestimation of risk to fish.

BOX 10-5. LIKELY EFFECTIVENESS OF MITIGATION MEASURES

Environmental characteristics along the transportation corridor would likely render the effectiveness of standard or even state-of-the-art mitigation measures highly uncertain.

- Subarctic extreme temperatures and frozen soil conditions could complicate planning for remediation, with uncertain outcomes due to variable conditions and spill material characteristics.
- Subarctic climatic conditions could limit the lushness and rapidity of vegetation growth or re-growth following ground disturbance, reducing the effectiveness of vegetated areas as sediment and nutrient filtration buffers.
- Widespread and extensive areas of near-surface groundwater and seasonally or permanently saturated soils could limit the potential for absorption or trapping of road runoff, and increase likelihood of its delivery to surface waters.
- The likelihood of ice flows and drives during thaws could make water crossing structures problematic locations for jams and plugging.
- The region is seismically active (Section 3.6), and even a small increment of ground deformation could easily disturb engineered structures and alter patterns of surface and subsurface drainage in ways that render engineered mitigations inoperative or harmful.
- Remote locations that are not frequented by humans mean that mitigation failures and accidents could go undetected until substantial harm to waters has occurred unless frequent inspections are conducted.

Although many possible mitigation measures can be identified and listed in a mitigation plan, they cannot all be ideally applied in every instance. Mitigation measures are often mutually limiting or offsetting when applied in the field. As a salient example for the transportation corridor, choosing a road location that minimizes crossings of streams, wetlands, and areas of shallow groundwater in a landscape that is rich in those hydrologic features could result in a tortuous alignment that is excessively long and curved to accommodate the upland terrain. This alignment would greatly increase the total ground area disturbed, and increased road curvature in either horizontal and vertical dimensions may increase risk of traffic accidents and consequent spills. It would also increase the length and structural complexity of the road-parallel pipelines (Chapter 11). Thus, avoidance of sensitive habitat features could elevate other environmental risks.

CHAPTER 11. PIPELINE FAILURES

As described in Section 6.1.3, the mine scenarios include four pipelines along the transportation corridor—one each for natural gas, diesel, product concentrate, and return water—and various pipelines on the mine site. Any of these pipelines could fail and release their contents to the environment. The risks from failure of product concentrate (Figure 11-1), return water (Figure 11-2), and diesel (Figure 11-3) pipelines are considered particularly high. These failure scenarios are evaluated in the following sections. Other pipelines are discussed briefly below.

On the mine site, the largest pipelines would carry tailings slurry from the mill to the tailings storage facilities (TSFs) and reclaimed water from the TSFs to the mill (Table 6-4). Smaller pipelines would convey water for processing and other uses and wastewater for treatment or storage. Other pipelines would carry diesel and natural gas from storage tanks to points of use. On-site pipeline spills have occurred at porphyry copper mines in the United States and some have resulted in significant aquatic exposures (Earthworks 2012). Such spills are possible at a future mine and could result in uncontrolled releases within the mine site; however, these spills are more likely to be contained or controlled without significant environmental effects than pipeline spills along the transportation corridor. In this assessment, we decided that leakage from on-site pipelines would be captured and controlled by the mine's drainage system and either treated prior to discharge or pumped to the process water pond or TSF.

Natural gas is lighter than air, so any release due to a natural gas pipeline failure 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 nearby environment. During dry periods, a wildfire could result. Such failures were considered to pose relatively low risks to the assessment endpoints and are not evaluated further in this assessment.

Figure 11-1. Conceptual model illustrating potential stressors and effects resulting from a concentrate pipeline failure.

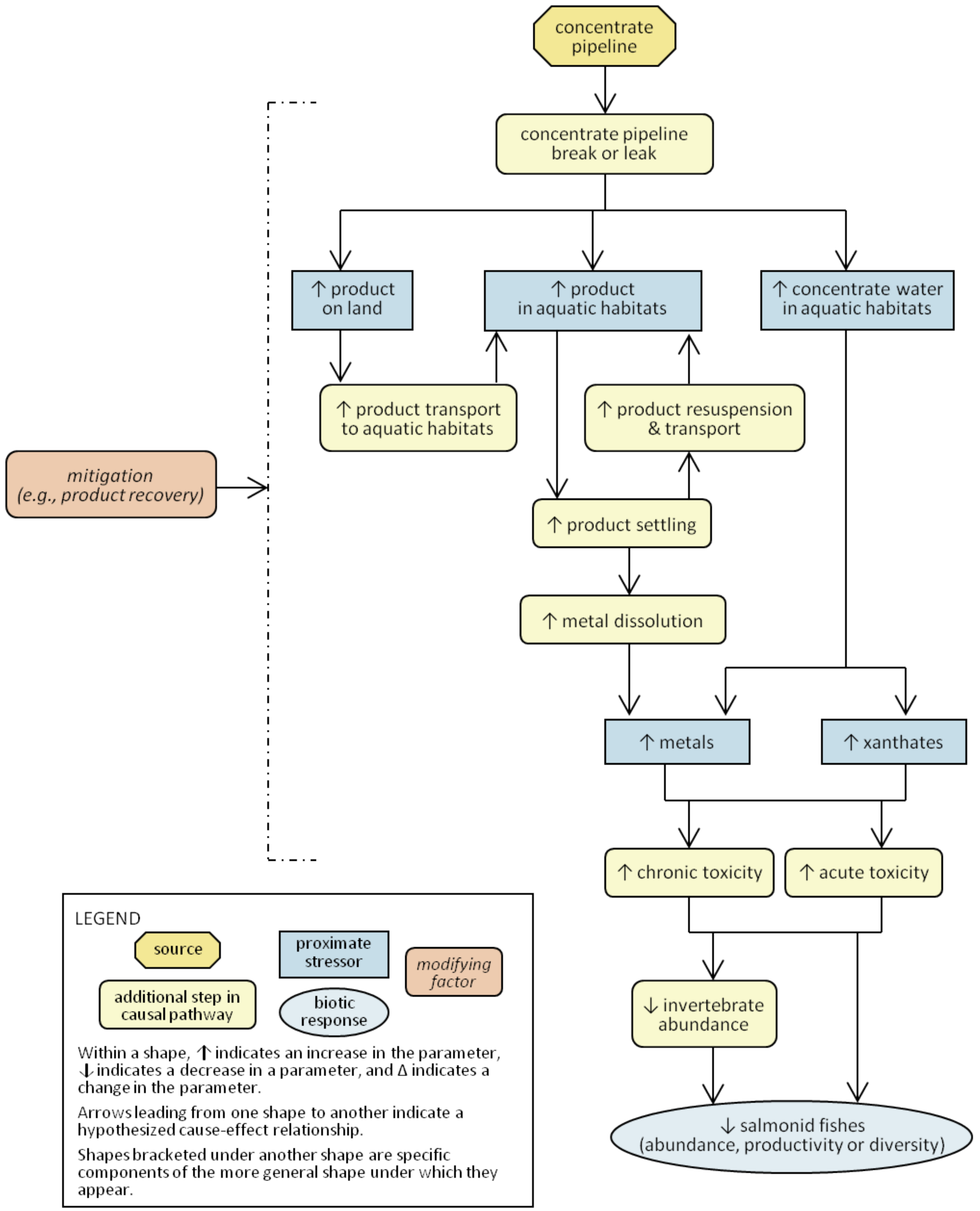


Figure 11-2. Conceptual model illustrating potential stressors and effects resulting from a return water pipeline failure.

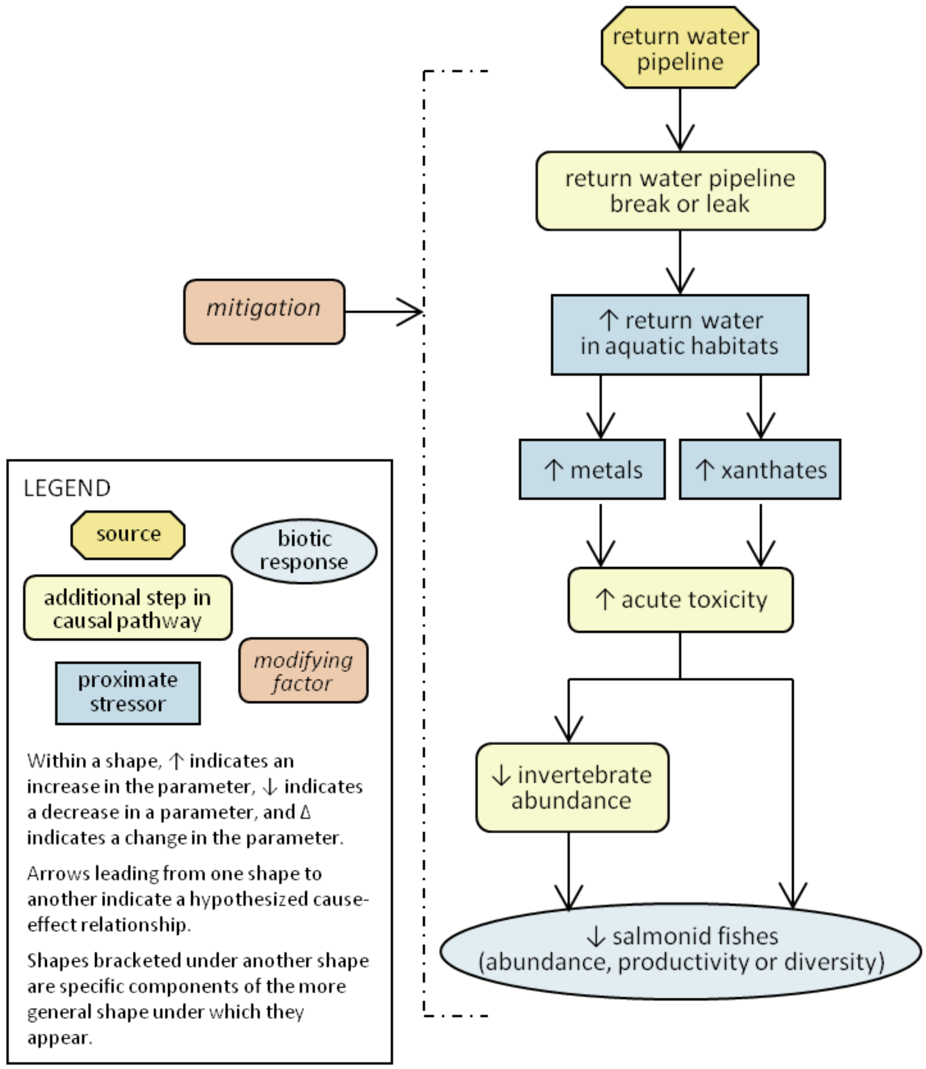
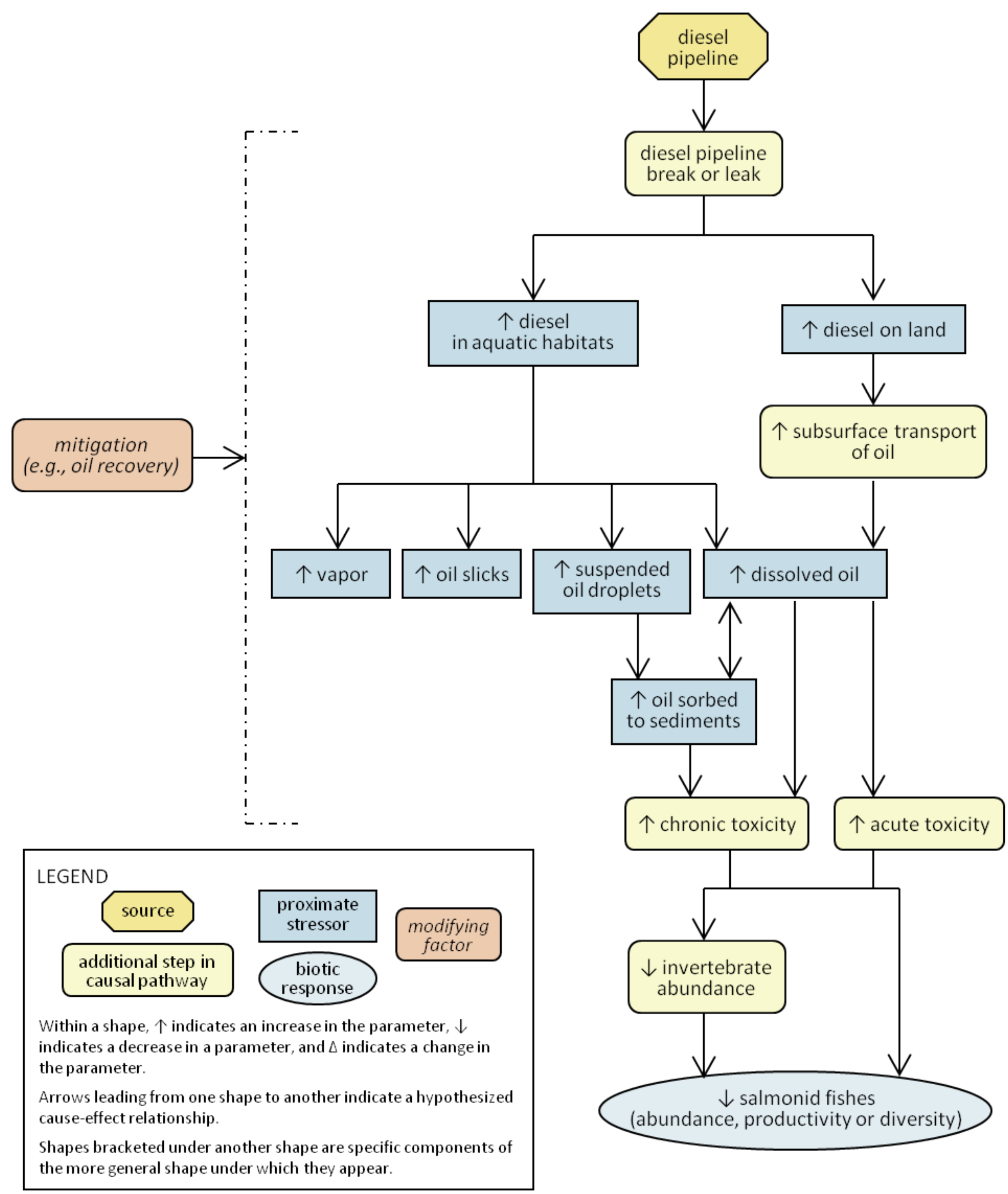


Figure 11-3. Conceptual model illustrating potential stressors and effects resulting from a diesel pipeline failure.



11.1 Causes and Probabilities of Pipeline Failures

The U.S. transportation system includes more than 4 million km of pipeline, of which more than 3.8 million km are gas transmission or natural gas distribution mains and more than 280,000 km carry hazardous liquids, primarily petroleum products (PHMSA 2012). The principal causes of failures along these pipelines are external corrosion and mechanical damage such as impacts by excavating equipment. Internal corrosion and material breakdown also may cause pipeline failures, but are less common. The failure rate from impacts, such as can occur during road, pipeline, or bridge maintenance, tends to be steady over the lifetime of a pipeline, whereas corrosion failures tend to increase with age of the pipe.

Pipeline failures include both leaks and ruptures. Leaks are small holes and cracks that result in product loss but do not immediately prevent the functioning of the pipeline. Ruptures are larger holes or breaks that render the pipeline inoperable. A study of over 2 million km-yr of pipelines in Canada indicated that leaks account for 87% of failures and ruptures account for 13% (EUB 1998). A rupture could result in the immediate release of a significant amount of pipeline material. A leak would allow pipeline material to escape more slowly than a rupture, but a leak could remain undetected for a much longer time, ultimately releasing quantities comparable to or exceeding a rupture.

The most extensive pipeline failure statistics are derived from oil and gas industry data (Table 11-1). The industry's record of pipeline failures is directly relevant to the oil and gas pipelines considered in the pipeline failure scenarios. The failure rate of metal concentrate slurry pipelines is unknown, because few such pipelines are in operation and no published failure rates are available for those that are in operation. Oil pipeline failure rates are used as the best available estimate, although it is possible that the erosive or corrosive nature of the product concentrate slurry would increase pipeline failure rates.

Although the range of published annual failure rates for U.S. oil and gas pipelines spans more than one order of magnitude (0.000046 to 0.0011 per km-yr) (URS 2000), the range for pipelines most similar to the assessment pipelines along the transportation corridor is much narrower. For example, the failure rate is 0.0010 failure/km-yr for pipelines less than 20 cm in diameter (OGP 2010), 0.0015 failure/km-yr for pipelines in a climate similar to Alaska (Alberta, Canada) (ERCB 2013), and 0.00062 failure/km-yr for pipelines run by small operators (those operating total pipeline lengths less than 670 km) (URS 2000). The geometric mean of these three values yields a failure probability of 0.0010 failure/km-yr.

This overall estimate of annual failure probability, coupled with the 113-km length of each pipeline as it runs along the transportation corridor within the Kvichak River watershed, results in an 11% 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 Pebble 2.0 scenario (i.e., approximately 25 years) would be 95% for each pipeline. The expected number of failures in each pipeline would be about 2.2, 2.8, and 8.6 over the life of the mine in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. The chance of a large rupture in each of the three pipelines over the life of the mine would exceed 25%, 30%, and 67% in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. In each of the three scenarios, there would be a greater than

99.9% chance that at least one of the three pipelines carrying liquid would fail during the project lifetime.

Table 11-1. Studies that examined 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	1970–2004	0.00041
		2000–2004	0.00017
Caleyo 2007	34,595	Mexican gas pipelines	0.0030
	28,270	Mexican oil pipelines	0.0052
URS 2000 (56 U.S. oil pipeline operators)	1,268,370	Highest failure rate	0.0011
		Average failure rate	0.00028
		Minimum failure rate	0.000046
		10 smallest operators (<670 km)	0.00062
		10 largest operators (>6,900 km)	0.00020
ERCB 2013	285,000	2000, Alberta, Canada	0.0033
	380,331	2007, Alberta, Canada	0.0022
	395,479	2008, Alberta, Canada	0.0021
	386,930	2009, Alberta, Canada	0.0016
	398,253	2010, Alberta, Canada	0.0015
	406,974	2011, Alberta, Canada	0.0015

Although data are insufficient to determine failure probabilities specific to the metal mining industry, the record suggests that pipeline failures at mines are not uncommon. A review of 14 operating porphyry copper mines in the United States (including all operating U.S. porphyry copper mines except two that have been operating for less than 5 years) found that all had experienced pipeline spills or accidental releases and that pipeline failures have continued into 2012 (Earthworks 2012).

It may be argued that engineering can reduce pipeline failures rates below historical levels, but improved engineering has little effect on the rate of human errors. Many pipeline failures, such as the cyanide water spill at the Fort Knox mine (Fairbanks, Alaska) that resulted from a bulldozer ripper blade hitting the pipeline (ADEC 2012), are due to human errors. Perhaps more important, human error can negate safety systems. For example, on July 25 and 26, 2010, crude oil spilled into the Kalamazoo River, Michigan, from a pipeline operated by Enbridge Energy. A series of in-line inspections had showed multiple corrosion and crack-like anomalies at the river crossing, but no field inspection was performed (Barrett 2012). When the pipeline failed, more than 3 million L (20,000 barrels) of oil spilled over 2 days as operators repeatedly overrode the shut-down system and restarted the line (Barrett 2012). The spill was finally reported by a local gas company employee who happened to witness the leak. The spill may have been prevented if repairs had been made when defects were detected, and the release could have been minimized if operators had promptly shut down the line.

11.2 Potential Receiving Waters

The transportation corridor pipelines evaluated in the assessment would cross approximately 64 streams and rivers in the Kvichak River watershed, 55 of which are believed to support salmonids and all of which could convey contaminants to Iliamna Lake. This number of crossings is much larger than the number of hydrologic units presented in Tables 10-3 through 10-5, because hydrologic units may contain multiple watersheds and each watershed may include crossings of multiple tributaries.

For approximately 14% of their length (15 km), these pipelines would be within 100 m of a stream or river (Table 10-3), and for 24% of their length (27 km) they would be within 100 m of a mapped wetland, pond, or small lake (Table 10-4). This proximity would create the potential for spilled slurry to flow into surface waters either directly or via overland flow. Some of the affected ponds support salmonids, but the number and distribution of salmonids in the area's wetlands, ponds, and small lakes are unknown. Approximately 272 km of streams, as well as Iliamna Lake, are downstream of these pipeline crossings (Table 10-7).

Although exposure pathways for all failure locations are considered, the quantitative analysis addressed two stream crossings along the assessment's transportation corridor: Chinkelyes Creek and Knutson Creek. Channel velocities for these creeks were calculated to estimate the time it would take for a spill to reach Iliamna Lake. Information from the Pebble Limited Partnership (PLP) (2011: Chapter 15.3) was used to develop channel width and depths. Streamflows were calculated from precipitation models used to determine mean annual runoff for the assessment's stream culvert analysis (Section 10.3.2). These mean annual streamflows applied to the basic channel geometry yielded channel velocities and thus travel times from each crossing to Iliamna Lake.

From the Chinkelyes Creek crossing, the creek flows 14 km to a confluence with the Iliamna River that continues for 7.6 km to Iliamna Lake. Lake levels can be seasonally high and create a backwater effect in the lower 3.5 km of the Iliamna River; however, most of the year the river flows freely for the entire distance to the lake shore. From the Knutson Creek crossing, the creek flows 2.6 km to Iliamna Lake. As Knutson Creek approaches the lake, the creek is steeper than the Iliamna River and it flows freely into the lake year-round. Total travel times to Iliamna Lake are estimated to be 170 minutes and 19 minutes for a Chinkelyes Creek and a Knutson Creek spill, respectively (Table 11-2). More details concerning these and other stream crossings are presented in Section 10.3.2.

Table 11-2. Parameters for concentrate pipeline spills to Chinkelyes Creek and Knutson Creek.

Parameter	Spill into Chinkelyes Creek: Chinkelyes Creek	Spill into Chinkelyes Creek: Iliamna River	Spill into Knutson Creek: Knutson Creek
Water Flow			
Discharge (m^3/s)	1.8	22	3.4
Velocity (m/s)	2.2	2.0	2.2
Channel Length (km)	14	7.6	2.6
Pipeline Drainage and Dilution			
Flow rate while draining (m^3/s)	0.11	-	0.07
Flow rate while pumping (m^3/s)	0.04	-	0.04
Release time—draining (minutes)	9.3	-	5.6
Release time—pumping (minutes)	5.0	-	5.0
Volume of slurry spilled (L)	75,000	-	37,000
Mass of concentrate solids spilled (metric tons)	66	-	32
Volume of aqueous phase spilled (L)	58,000	-	28,000
Maximum fully mixed dissolved copper concentration (µg/L)	37	3.3	16
Quotient[a], acute copper criterion	13	1.3	5.9
Quotient[a], chronic copper criterion	22	2.1	9.6
Travel time to confluence (minutes)[b]	110	64	19
Pipeline and Slurry Specifications			
Length from top of nearest hill to valve (m)	2100	-	810
Elevation drop (m)	150	-	25
Viscosity of slurry (cP)	9.5		
Density of slurry (metric tons/m^3)	1.7		

Notes:
Dashes (-) indicate that spill is not directly into Iliamna River, which receives flow from Chinkelyes Creek.
[a] See Box 8-3 for a description of how risk quotients were calculated.
[b] Confluence with Iliamna River for Chinkelyes Creek; confluence with Iliamna Lake for the Iliamna River and Knutson Creek.

11.3 Concentrate Pipeline Failure Scenarios

11.3.1 Sources

A full pipeline break or a defect of equivalent size in the copper (+gold) concentrate pipeline (Table 6-4) at the Chinkelyes Creek or Knutson Creek crossing would release slurry into these water bodies. This kind of failure could result from mechanical failure of the pipe due to ground movement, vehicle impact, maintenance error, or material failure. Parameters for the concentrate pipeline failure scenarios are summarized in Table 11-2.

In the concentrate pipeline failure scenarios, a single complete break of the pipeline would occur at the edge of the stream, just upstream of an isolation valve. These valves would be placed on either side of major crossings (Ghaffari et al. 2011) and could be remotely activated. Pumping would continue for 5 minutes until the alarm condition was assessed and an operator shut down the pumps. The estimated total slurry volume draining to the stream would equal the pumped flow rate times 5 minutes, plus the volume between the break and local high point in the pipeline (i.e., the nearest watershed boundary)

(Table 11-2). During the entire spill, gravity drainage would govern the flow rate based on calculations for free-flowing pipes.

The product concentrate would have a density of 3.8 metric tons/m^3 and would sink rapidly if released into a water body at low flows. The slurry water would have a density near 1.0 metric ton/m^3 and would mix readily with surface waters. No analyses of product concentrate or concentrate transport water are available for the Pebble deposit or any other ore body in the region. To estimate the concentration of metals and other constituents in the concentrate, we used analyses from the Aitik (Sweden) porphyry copper mine as described in Appendix H.

The fine particles of product concentrate would, like spilled tailings (Section 9.3), degrade habitat quality for fish and benthic invertebrates. However, these potential physical effects would be much lower in magnitude than for a tailings dam failure because of the much lower volume of material, and would be less important than potential toxic effects. Thus, we focus on toxic effects rather than effects of sediment deposition on habitat.

11.3.2 Exposure

In these concentrate pipeline failure scenarios, 66 metric tons of product concentrate would be released into Chinkelyes Creek or 32 metric tons into Knutson Creek. Based on its size and the well-established relationship between particle size and particle mobilization and transport (commonly represented by the Hjulström diagram), the concentrate would be transported in suspension by streamflows greater than approximately 20 cm/s and would be transported as bedload between approximately 1 and 20 cm/s. Estimated mean velocities of the streams (2.2 m/s for Chinkelyes Creek and Knutson Creek and 2.0 m/s for the Iliamna River) are consistent with those described for these streams (PLP 2011) and are well above the transport velocities. Therefore, the fine sand-sized concentrate would be carried downstream during typical or high flows, even given that the concentrate is denser (3.8 metric tons/m^3) than typical rock (2.8 metric tons/m^3 for granite) and would move less readily. Concentrate would be deposited in any backwaters, pools, or other low-flow locations. If the spill occurred during a period of high flow, it would be carried downstream immediately, reaching Iliamna Lake within 3 hours (via Chinkelyes Creek and Iliamna River) or 0.5 hour (via Knutson Creek). Because flood flows are a potential cause of pipeline failure at stream crossings, this is a reasonable possibility. If the spill occurred during low flows, concentrate that is not collected would be spread downstream by erosion during subsequent typical or high-flow periods, eventually entering Iliamna Lake. Concentrate that entered the lake could mix into sand and gravel beaches used by spawning sockeye salmon. These transport and deposition processes cannot be quantified with existing data and modeling resources.

The estimated annual failure rate of one per 1,000 km per year (Section 11.1) results in an estimated failure rate of 0.11 per year for the 113 km of concentrate pipeline within the Kvichak River watershed. 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 14% probability of entering a stream within the Kvichak River watershed. This would result in an estimate of 0.015 stream-contaminating concentrate spills per year, or 1.2 stream-contaminating concentrate spills over the duration of the

Pebble 6.5 scenario (approximately 78 years). In other words, we expect roughly 1 such spill in the Pebble 6.5 scenario. Similarly, a spill would have a 24% probability of entering a wetland, resulting in an estimate of 0.026 wetland-contaminating spills per year or 2 wetland-contaminating spills in the Pebble 6.5 scenario. A portion of those wetlands would be ponds or backwaters that support fish.

Spills from the pipeline failure would contaminate 2.6 km of Knutson Creek or 14 km of Chinkelyes Creek and 7.6 km of the Iliamna River with product concentrate and leachate (the slurry water that has leached ions from the product concentrate) before entering Iliamna Lake. The potential extent of wetland, pond, and small lake contamination cannot be readily estimated.

As with a tailings spill (Chapter 9), toxicologically relevant exposures could occur via multiple routes following a concentrate pipeline spill. During and immediately following a spill, organisms would be acutely exposed to leachate and suspended particles. After a spill, product concentrate deposited on a stream or lake bed would result in chronic aqueous exposures to pore water and acute aqueous exposures during resuspension events. Unlike the tailings spill, which would inevitably enter a stream and its floodplain, a slurry spill might directly enter a stream, pond, or wetland; it might flow over land to a nearby water body; or it might flow across the landscape without reaching water. Terrestrial slurry deposits are likely to be collected by the operator, so rain and snowmelt are unlikely to leach those concentrate deposits and significantly contaminate streams. However, spilled leachate from the pipeline slurry could enter a stream, wetland, pond, or lake by overland or groundwater flow. Contaminated groundwater could upwell through the gravels and cobbles of streams or deltaic gravels and sands in Iliamna Lake, and benthic invertebrates and fish eggs and larvae could be exposed to toxic concentrations if sufficient dilution did not occur.

11.3.2.1 Aqueous Phase Chemical Constituents

The concentrate slurry is estimated to contain 77% water by volume, with dissolved constituents that include dissolved salts of the product and trace metals, as well as process chemicals. Copper is the primary ecotoxicological concern, because it is the principal product and is highly toxic to aquatic life. Analyses of aqueous filtrate from samples taken on 3 different days, from a concentrate pipeline at a porphyry copper mine with a separation process similar to that considered in the mine scenarios (Section 6.1.2), reported copper concentrations of 500, 664, and 800 μg/L (Adams pers. comm.). The mean of these values (655 μg/L) was used as the estimated copper concentration in this assessment.

Due to its relatively high toxicity, sodium ethyl xanthate is the highest risk ore-processing chemical that could occur in the product concentrate slurry. We were unable to find an estimate of process chemical concentrations in the concentrate slurry, but xanthate concentration would be 1.5 mg/L if we assume that it occurs in the concentrate slurry at the same concentration as in tailings slurry (NICNAS 1995). Unlike the metals, xanthate would degrade, but because its environmental half-life is approximately 260 hours (at pH 7 and 25°C) (NICNAS 2000) it could persist long enough to cause significant exposures until diluted in Iliamna Lake.

Flows in the potential receiving streams vary considerably. Measurements in streams along the transportation corridor in 2004 and 2005 yielded a maximum observed flow of 58,000 L/s in the Iliamna River and a minimum observed flow of 2.8 L/s in an unnamed stream (PLP 2011). Thus, full mixing of spilled leachate could result in as much as a 33-fold dilution, but in smaller streams dilution effectively would not occur. Of 12 monitored streams along the transportation corridor, only two had observed flows in August 2004 (an estimate of summer low flow) that were greater than the estimated flow of the aqueous phase of the slurry (PLP 2011: Table 7.3-10).

11.3.2.2 Solid Phase Chemical Constituents

If spilled product concentrate entered a stream, wetland, pond, or small lake directly or by overland flow or erosion, it would flow for some distance, settle, and become substrate for invertebrates and possibly salmon eggs and fry. In streams, it would be carried downstream by the current and would collect in pools, behind debris, and in other localized low-flow areas. Some would settle into the cobble substrate until high flows mobilized the bed. Much of the product concentrate could wash into Iliamna Lake, where it could contribute to the substrate for spawning sockeye salmon.

Metal concentrations in the solid phase are expected to be similar to those of the Aitik product concentrate (Table 11-3). Settled concentrate would be leached, resulting in direct aqueous exposure of benthic invertebrates and fish eggs and larvae that inhabit the substrate to concentrations similar to leachates from the Aitik product concentrate (Table 11-4). Local accumulation in streams could result in local exposures to nearly pure concentrate and leachate. However, concentrate in Iliamna Lake would be distributed and diluted to an extent that could not be estimated. Dietary exposure of fish is not considered, because invertebrate abundance would be greatly diminished due to sediment toxicity, even with considerable dilution by clean sediment.

11.3.3 Exposure-Response

Acute water quality criteria (criterion maximum concentrations [CMCs]), chronic criteria (criterion continuous concentrations [CCCs]), and equivalent benchmark values are used as thresholds for aqueous toxicity. Consensus sediment quality guidelines are used as thresholds for sediment solids toxicity. These aqueous and sediment benchmark values are discussed in Section 8.2 and Section 9.5, respectively. The biotic ligand model (BLM) generates low acute and chronic water quality criteria and other toxicity values because of the extreme water chemistry of the leachate and receiving waters (Section 8.2.2). However, the parameters are all within calibration range of the model (except for alkalinity and dissolved organic carbon, which were set to minimum values because they were absent from the leachate; this slightly raises criteria values) (HydroQual 2007).

In addition to the product concentrate and its dissolved constituents, the slurry would contain process chemicals. Sodium ethyl xanthate is sufficiently toxic that it has been used as a pesticide (NICNAS 2000). Exposure-response information for xanthates is summarized in Section 8.2.2.

Table 11-3. Comparison of mean metal concentrations in product concentrate from the Aitik (Sweden) porphyry copper mine (Appendix H) to threshold effect concentration and probable effect concentration values for fresh water. Values are in mg/kg dry weight.

Concentrate Constituents	Concentrations	TEC[a]	TEC Quotient[b]	PEC[a]	PEC Quotient[b]
Ag	>10	-	-	-	-
As	12	9.8	1.2	33	0.36
Ba	59	-	-	-	-
Bi	45	-	-	-	-
Cd	2.4	0.99	2.4	5.0	0.48
Co	54	-	-	-	-
Cu	>10,000	32	>310	150	>67
Ga	0.88	-	-	-	-
In	2.4	-	-	-	-
Mn	345	630	0.55	1,200	0.29
Mo	1,100	-	-	-	-
Ni	72	23	3.1	49	1.5
Pb	65	36	1.8	130	0.50
Sb	43	-	-	-	-
Te	4.1	-	-	-	-
Th	1.5	-	-	-	-
Tl	0.2	-	-	-	-
U	2.2	-	-	-	-
V	23	-	-	-	-
Zn	2,200	120	18	460	4.8
Sum of metals	-	-	>340	-	>75

Notes:
Dashes (-) indicate that values are not available.
[a] TECs and PECs are consensus values from MacDonald et al. (2000), except for Mn values, which are the TEL and PEL for *Hyalella azteca* 28-day tests from Ingersoll et al. (1996).
[b] See Box 8-3 for a description of how risk quotients were calculated.
TEC = threshold effect concentration; PEC = probable effect concentration; TEL = threshold effect level; PEL = probable effect level.

11.3.4 Risk Characterization

Toxicological risk characterization is performed primarily by calculating risk quotients based on the ratios of exposure concentrations to aquatic toxicological benchmarks (Box 8-3). However, it also includes consideration of actual concentrate spills, the potential for remediation, and site-specific factors.

11.3.4.1 Concentrate Pipeline Failure Scenarios

The concentrate pipeline failure scenarios and resulting spills would release 58,000 L of leachate to Chinkelyes Creek or 28,000 L to Knutson Creek (Table 11-2). Risks to aquatic biota would result from direct exposure to the aqueous phase of the slurry, the deposited concentrate, and *in situ* leachate from the concentrate.

Table 11-4. Aquatic toxicological screening of leachates from Aitik (Sweden) product concentrate (Appendix H) based on acute (criterion maximum concentration) and chronic (criterion continuous concentration) water quality criteria or equivalent benchmarks, and quotients of concentrations divided by benchmark values. Values are in µg/L unless otherwise specified.

Analyte	Concentrations	Acute/Chronic Benchmarks	Quotients[a]
pH (standard units)	5.4	6.5–9	-
Spec. conductivity (µS/cm)	260	-	-
Alkalinity (mg/L)	0	-	-
SO_4 (mg/L)	120	-	-
SiO_2 (mg/L)	59	-	-
Ag	<1	0.90[b]/-	<1/-
Al	840	750/87	1.1/9.7
As	<1	340/150	<0.0029/<0.0067
Ba	38	46,000/8,900	0.0008/0.0043
Ca	27,000	-	-
Cl	800	19/11	42/73
Cd	3.5	1.7/0.22[b]	2.0/16
Co	140	89/2.5	1.5/54
Cr	<1	500/65[b]	<0.002/<0.007
Cu	8,400	12/7.9[b] 0.05/0.03[c]	720/1,100 180,000/290,000
F	1,600	-	-
Fe	210	350/-	0.60/-
K	4,000	-	-
Mg	4,500	-	-
Mn	640	760/690	0.85/0.93
Mo	<2	32,000/73	<0.0001/<0.03
Na	890	-	-
Ni	480	410/46[b]	1.2/10
Pb	11	54/2.1[b]	0.20/5.0
Sb	13	14,000/1,600	0.0009/0.008
Se	7.3	-/5.0	-/1.5
U	11	33/15	0.32/0.70
Zn	1,300	100/100[b]	13/13
Sum of metals	-	-	740[b]/1,300[b] 180,000[c]/290,000[c]

Notes:
Dashes (-) indicate that benchmarks are not available or, in the case of pH, that the value is not applicable.
[a] See Box 8-3 for a description of how risk quotients were calculated.
[b] Hardness-based criterion or standard based on hardness of 85.5 estimated from 2.5 Ca + 4.1 Mg in mg/L.
[c] From the national ambient water quality criterion for copper based on the biotic ligand model and leachate chemistry.

The estimated dissolved copper concentration in the aqueous phase of the slurry is 655 µg/L, which is roughly 240 times the acute water quality criterion and 390 times the chronic criterion for Upper Talarik Creek, the nearest stream with complete water quality data (Table 8-11). Clearly, this would be sufficient to cause severe toxic effects in small streams, large streams at low flow, and wetlands. The dilution provided by the receiving waters considered here would not be enough to prevent acute, much less chronic, toxicity based on the copper criteria. These criteria are based on toxicity to sensitive invertebrates, so the food base for salmonids could be severely reduced.

In all three streams, the diluted values are below the BLM-derived acute lethal levels for rainbow trout, so a fish kill would not be expected (Table 8-13). Therefore, copper is not predicted to cause a kill of adult salmonids in the receiving streams once mixing has occurred, but localized mortality could occur in the mixing zone in the absence of avoidance behavior. However, fully diluted concentrations in Chinkelyes Creek are above the chronic toxicity value for rainbow trout, suggesting that fry would be affected. Concentrations at mean flow in Knutson Creek are a little below the rainbow trout chronic toxic level (22 versus 26 μg/L), suggesting that effects on fry could occur at low flows.

Sodium ethyl xanthate, after fully mixing in Chinkelyes and Knutson Creeks, would occur at approximately 0.1 and 0.07 mg/L. These values are at the low end of observed acutely lethal concentrations for aquatic biota and below the observed median lethal concentrations for rainbow trout (Section 8.2.2.5). Hence, the processing chemicals could contribute to acute toxicity in sensitive species.

The occurrence of acute toxicity depends on the exposure duration relative to the concentration. The 5.6- to 9.3-minute exposure duration (Table 11-2) may be sufficient to cause acute injury or lethality to invertebrates or fish in receiving streams, given the high concentrations of copper (the rate of toxic response is a function of the concentration) and given that the chronic effects of copper on fish include lethality to fry. However, it would be more likely to cause acute effects in backwaters and ponds that retained spilled water, and those areas are important rearing habitat for salmon (Appendix A).

Where the 32 to 66 metric tons of concentrate settled, sediment and benthic invertebrates and fish eggs and fry would be exposed. The Aitik concentrate exceeds the sediment probable effect concentration (PEC) for copper by more than a factor of 67 (Table 11-3). Hence, based on experience with other high-copper sediments, any product concentrate from the Pebble deposit 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), the chronic leaching of copper from deposited product concentrate may prevent returning salmon from using a contaminated stream or river.

Exposure to pore water in sediments consisting of spilled product concentrate would be chronic. The screening assessment performed here on Aitik concentrate leachate suggests that spilled concentrate would cause severe toxic effects (Table 11-4). The 8,400 μg/L of dissolved copper in leachate would be sufficient to kill benthic or epibenthic invertebrates and fish eggs and fry.

At mine closure, concentrate and return water pipelines would be removed. Therefore, these risks would be limited to the approximately 78-year maximum operational life of the mine in the Pebble 6.5 scenario.

11.3.4.2 Analogous Mines

No Alaskan mine has a product concentrate pipeline, but 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 the 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 control required 2 hours, and attribute the failure to “an existing outer mark on the pipe” (Minera Alumbrera 2004). Other pipeline failures with concentrate spills were reported in 2006 and 2007, but not in other years (Minera Alumbrera 2004, 2005, 2006, 2007, 2008, 2009, 2010). They claimed that those releases were small due to automatic shutoff, that concentrate did not reach water, and that “no hazard is involved in concentrate handling since it is a harmless product consisting of ground rock” (Minera Alumbrera 2006). Composition of this ground rock included 28% copper and 32% sulfur (Minera Alumbrera 2006).

Operators subsequently built collection pits at pumping stations, monitored streams at pipeline crossings, and brought water into the community of Amanao in part to mitigate effects of “potential pipeline failure” (Minera Alumbrera 2008, 2010). They stated based on monitoring that pipeline crossings of streams have no adverse effects on biodiversity, but they do not report monitoring to 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 notable that, despite International Organization for Standardization 14001 certification of the pipeline, it failed and released concentrate in 3 of 7 years.

More recently (July 25, 2012), a joint broke on the product slurry pipeline for the Antamina copper and zinc mine in Peru (Briceno and Bajak 2012, Taj and Cespedes 2012). It released 45 metric tons of slurry over 2 hours, of which 3 metric tons escaped the containment area. Local villagers intervened to stop the flow of slurry to the nearby Rio Fortenza. A mine spokesperson stated that the river showed no signs of contamination and the material was only an irritant, although a company document called the concentrate very toxic (Taj and Cespedes 2012). An Associated Press photo shows workers in white protective suits apparently cleaning a channel. News reports and Minera Antamina’s press releases on the event emphasized human health effects: 210 people received medical treatment and 45 were hospitalized, apparently due to inhalation of aerosolized slurry. People reported a strong pesticide odor, which suggests significant concentrations of a xanthate collector chemical, but no analyses have been reported. Ecological effects are unknown. Antamina is a modern mine (operation began October 1, 2001) where sustainability is said to be given a higher priority than cost or profitability (Caterpillar Global Mining 2009). As in the mine scenarios evaluated here, the pipeline is buried except at bridges and is monitored using a parallel fiber optic system.

Product concentrate spills from pipeline failures have also occurred at the Bingham Canyon mine in Utah. Between May 31 and June 2, 2003, operators reported to the U.S. Coast Guard's National Response Center a spill of 70 tons (63.5 metric tons) of product concentrate from a pipeline failure. On October 2, 2009, they reported a pipeline leak that spilled 1,400 gallons (5,300 L) of copper concentrate.

Although the Alumbrera, Antamina, and Bingham Canyon cases do not provide evidence concerning the ecological effects of a concentrate spill, they do support the plausibility of pipeline failures leading to concentrate spills. Our estimated pipeline failure rate of one per 1,000 km-year (Section 11.1) implies a failure rate of 0.32 per year for the 316-km Alumbrera pipeline, which is similar to the 0.43 observed rate at Alumbrera from 2004 to 2010. These cases indicate that concentrate pipeline failures do occur at modern copper mines operated by large international mining companies, and that they can result in spills that are potentially larger than our assumptions indicate.

11.3.4.3 Concentrate Spill Remediation

Remediation of a product concentrate spill would be less problematic than remediation of a tailings spill. The concentrate is valuable, it would be spilled near a road, and the volume would be much smaller than a potential tailings spill. Hence, remediation would likely occur relatively quickly if the spill occurred on land or in a wetland, by excavating or dredging the concentrate and trucking it back to the mine. However, because concentrate would be carried downstream by high or typical flows in the receiving streams, substantial recovery of material spilled into a stream is unlikely except possibly during low-flow periods (less than one-ninth of mean flows). The proportion recovered by dredging would depend on the circumstances, the rapidity of response, and the balance between the desire to minimize habitat damage and to reduce potential toxic effects. If the spill was associated with high flows, it is likely that little of the material would be recovered from a stream even if the entire stream was dredged. Dredging in Iliamna Lake might be feasible if concentrate was not too dispersed or diluted by other sediment.

11.3.4.4 Weighing and Summarizing the Evidence

Past experience with pipelines in general, and with the Alumbrera, Antamina, and Bingham Canyon product concentrate pipeline failures in particular, suggests that pipeline failures and product spills would be likely in the Pebble 6.5 scenario. A concentrate spill into a stream is likely to kill invertebrates and early fish life stages immediately. If it is not remediated (and remediation of streams may not be possible), it 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 in the receiving streams for many years. Ultimately, this settled concentrate would reach 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. The length of streams affected in the scenarios would be 14 km of Chinkelyes Creek and 7.6 km of the Iliamna River for a release to Chinkelyes Creek, or 2.6 km of Knutson Creek for a release there. The area of the lake that would experience toxic effects cannot be estimated at this time.

The weighing of these lines of evidence is summarized in Table 11-5. For each route of exposure, 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). For logical implication, possible scores are indicated as (+) for results supportive of adverse effects on the endpoint populations, (−) for results contrary to adverse effects on assessment endpoints, and (0) for neutral or ambiguous results. In this case, the logical implication is that the concentrate pipeline failure scenario evaluated here 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): a low quotient is indicated as (0), a moderate quotient as (+), and a high quotient as (++). Quality is a more complex concept. It includes conventional data quality issues, but in this case the primary determinant is the relevance of the evidence to the mine scenario. Separate quality scores are provided for the exposure estimate and for the exposure-response relationship. The scores are intended to remind the reader what evidence is available and show the pattern of strength and quality of the several lines of evidence and to transparently present our weighing process and results.

Table 11-5. Summary of evidence concerning risks to fish from a product concentrate spill. The risk characterization is based on weighing four lines of evidence for different routes of exposure. All evidence is qualitatively weighed (using one or more +, 0, - symbols) on three attributes: logical implication, strength, and quality. Here, all lines of evidence have the same logical implication—that is, all suggest a concentrate spill 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 the spill scenario.

Route of Exposure Source of Evidence (Exposure/E-R)	Logical Implication	Strength	Quality		Results
			Exposure	E-R	
Dissolved copper Measurements from analogous mine and dilution model/Laboratory-based benchmarks	+	++	+	++	Lethality to invertebrates is certain and sensitive larval fish may also be killed.
Concentrate particles Undiluted concentration from analogous mine/Field–based benchmarks	+	++	0	+	The concentrate would clearly form toxic sediment but its distribution in unclear.
Concentrate leachate Leachate from analogous mine/ Laboratory-based benchmarks	+	++	0	++	Invertebrates and fish in sediment would experience toxic effects unless the concentrate was highly diluted.
Actual spills Amount spilled/None	0	0	0	0	The record indicates that concentrate spills occur but exposure and effects have not been studied.
Summary weight of evidence	+	++	0	+	A spill is likely to occur and toxicity to aquatic biota is highly likely.

Notes:
E-R = exposure-response relationship.

Overall, available lines of evidence for effects of a concentrate spill are positive (i.e., supportive of the hypothesis that acute and chronic toxic effects would occur) (Table 11-5). The quality of the exposure-response information is good, but the quality of the exposure information for the deposited concentrate and its leachate is uncertain because of the uncertain potential for dispersal in streams. The analogous spills provide no information on exposure or effects beyond confirming that concentrate spills do occur. However, this evidence supplements the more extensive experience with oil pipelines (Section 11.1), which suggests that a spill is likely.

If the spill could be remediated, some fraction of the concentrate (but none of the leachate) could be recovered and the extent of chronic (but not acute) toxic effects would be diminished. The proportion of concentrate recovered would depend on spill location, time of year, diligence of the operator, and the amount of physical damage due to remediation that is considered acceptable. Concentrate spilled into streams would be unlikely to be recovered unless streamflows were particularly low. Recovery of the concentrate would require excavation of streambeds, wetlands, or uplands, depending of the location of the spill. When determining how thoroughly to excavate and, in particular, how far downstream to dredge the stream, reduction in toxicity would need to be balanced against habitat destruction.

The effects of a spill on salmonid populations would depend on the receiving waters. Streams along the transportation corridor that might receive a spill (described in Section 10.1) are quite variable. Chinkelyes Creek receives an average of more than 9,000 spawning sockeye salmon and flows to the Iliamna River, which receives an average of more than 100,000 sockeye spawners (Table 10-2). Knutson Creek receives an average of roughly 1,500 sockeye spawners and flows to Knutson Bay, which receives an average of 73,000 beach spawning sockeye (Table 10-2). Not all of those salmon spawn below stream crossings, but copper leaching from concentrate spills could be aversive to salmon and thereby reduce spawning production along the entire stream lengths. Also, the concentrate deposited in Knutson Bay would persist and could render a considerable area unsuitable for spawning and rearing for years. In any case, these values indicate that a non-trivial number of spawners and potential salmon production would be at risk.

Potential effects on those salmon and other fishes in the receiving waters would include the following.

- Reduced production of salmon fry and parr and all life stages of other salmonids from the loss of invertebrate prey due to extensive acute lethality during the spill and persistent chronic toxicity in areas where the concentrate deposited.
- Loss of a year-class of salmon and other salmonids due to direct acute toxicity during and immediately following a concentrate spill.
- Loss of salmon spawning habitat due to avoidance of copper in areas of deposition and possibly in the entire stream, if aqueous concentrations from leaching concentrate were sufficiently high.
- Persistent chronic toxicity to salmonid eggs and fry in areas of concentrate deposition, where it is not aversive to spawning adults.

11.3.5 Uncertainties

Based on multiple lines of evidence, it is certain that a spill from a product concentrate pipeline into a stream would cause toxic effects. However, there are uncertainties regarding individual pieces of evidence, which are summarized below.

- 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. Copper concentrations in North and South American copper concentrates generally fall in the 200 to 340 mg/kg range, so variance of a factor of 2 is a reasonable estimate for potential variance in Pebble deposit concentrate from Aitik concentrate. Hence, uncertainty concerning the major source of toxicity is not large, and therefore it is implausible that the concentrate and its leachate would be nontoxic to aquatic biota. An informal internet search for copper concentrate compositions suggests that minor metals differ by an order of magnitude among copper concentrates. Thus, it is possible that metals other than copper may be significant contributors to toxicity.
- The copper concentration of the aqueous fraction of the slurry is also based on analyses from an existing mine. However, estimates based on the existing mine are realistic, given that the ore type and processing are believed to be very similar and that the leachate was formed during actual operations rather than in a test. Therefore, this uncertainty is estimated to be at least a factor of 2 but no more than 5. Effects on invertebrates are certain, but effects on fish may not occur or may be more severe than estimated.
- The composition of the aqueous fraction of the slurry is unknown for constituents other than copper. Although it is certain that copper is by far the most toxic metal in the slurry, the composition of other constituents is unknown. Sodium ethyl xanthate is highly toxic and might increase the toxicity of a spill. Combined metal toxicity would make some difference but is unlikely to change the qualitative conclusions.
- The 5-minute time to shut-off is uncertain, and this estimate appears to be conservative. For example, Trans Canada's risk assessment for the Keystone XL pipeline assumed that the time to detection would range from 90 days for a small leak (1.5% of pumping volume) to 9 minutes for a large leak (50% of pumping volume), and that an additional 2.5 minutes would be required for the shutdown sequence (DNV Consulting 2006, O'Brien's Response Management 2009). This suggests that a large spill like the one assessed here would leak for 11.5 minutes based on a state-of-practice design from an experienced company, which is more than twice our assumed duration.
- The 5-minute time to shut-off depends on successful operation of a remote shutoff system. The potential for a larger spill if the shutoff failed (e.g., if an earthquake damaged the pipeline and the shutoff system) or was overridden by the operators is unknown. There are precedents for large spills but not enough data to quantify the risk.
- The frequency and location of spills are also uncertain. The extensive experience with oil and gas pipelines provides probabilistic estimates, but these estimates vary considerably among studies. The more directly relevant experiences with concentrate pipelines at Alumbrera, Antamina, and Bingham Canyon mines suggest that estimates based on oil and gas pipeline failure rates are consistent with mining-related pipeline failures.

11.4 Return Water Pipeline Failure Scenarios

A spill from a return water pipeline would result in an acute aqueous exposure (Table 11-6), as discussed above for a product concentrate spill. The return water is expected to be the same as the aqueous phase of the concentrate slurry (i.e., it would not be treated at the port), although estimated flow rates would differ. Hence, copper concentration in the return water is assumed to be the mean of analyses of the aqueous phase of slurry from a Rio Tinto mine (655 μg/L). Both acute and chronic criteria would be exceeded, but because of the short spill duration and the absence of a persistent solid phase, toxic effects would not be expected to be so severe as a product concentrate spill. Effects would be most likely in low-flow habitats such as backwaters, ponds, and bays. We know of no analogous return water pipeline failures that might be used to assess this risk; 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 11.1).

Table 11-6. Parameters for return water pipeline spills to Chinkelyes and Knutson Creeks.

	Spill into Chinkelyes Creek		Spill into Knutson Creek
Parameter	Chinkelyes Creek	Iliamna River	Knutson Creek
Water Flow			
Discharge (m^3/s)	1.8	22	3.4
Velocity (m/s)	2.2	2.0	2.2
Channel Length (km)	14	7.6	2.6
Pipeline Drainage and Dilution			
Flow rate while draining (m^3/s)	0.09	-	0.06
Flow rate while pumping (m^3/s)	0.03	-	0.03
Release time—draining (minutes)	8.6	-	5.1
Release time—pumping (minutes)	5.0	-	5.0
Volume spilled (L)	56,000	-	27,000
Maximum concentration dissolved copper (μg/L)	39	3.5	17
Travel time to confluence (minutes)[a]	110	64	19
Pipeline and Return Water Specifications			
Length from top of nearest hill to valve (m)	2100	-	810
Elevation drop (m)	150	-	25
Viscosity of return water (cP)	1		
Density of return water (metric tons/m^3)	1		

Notes:
Dashes (-) indicate that spill is not directly into Iliamna River, which receives flow from Chinkelyes Creek.
[a] Confluence with Iliamna River for Chinkelyes Creek; confluence with Iliamna Lake for the Iliamna River and Knutson Creek.

11.5 Diesel Pipeline Failure Scenarios

As with the product concentrate pipeline, effects of a diesel pipeline failure would depend on many factors, including pipeline design, location of the pipeline failure along the transportation corridor, and time of year at which the pipeline failure occurred. Parameters for the diesel pipeline failure scenarios are presented in Table 11-7.

Table 11-7. Parameters for diesel pipeline spills to Chinkelyes and Knutson Creeks.

	Spill into Chinkelyes Creek		Spill into Knutson Creek
Parameter	Chinkelyes Creek	Iliamna River	Knutson Creek
Water Flow			
Discharge (m^3/s)	1.8	22	3.4
Velocity (m/s)	2.2	2.0	2.2
Channel Length (km)	14	7.6	2.6
Pipeline Drainage and Dilution			
Flow rate while draining (m^3/s)	0.035	-	0.023
Flow rate while pumping (m^3/s)	0.005	-	0.005
Release time—draining (minutes)	13	-	7.9
Release time—pumping (minutes)	5	-	5
Volume—total (m^3)	30	-	12
Volume % diesel to water in stream at spill	2.2%	-	0.83%
Mass of diesel in stream at input (mg/L)	17,000	1,500	6,500
Maximum concentration dissolved diesel (mg/L)	1.9–7.8	1.7–7.2	1.9–7.8
Distance traveled during release (km)	1.7		1.1
Travel time to confluence (minutes)[a]	110	64	19
Pipeline and Diesel Specifications			
Length from top of nearest hill to valve (m)	2100	-	810
Elevation drop (m)	150	-	25
Viscosity of diesel at 15°C (cP)	2		
Density of diesel at 15°C (metric tons/m^3)	0.85		

Notes:
Dashes (-) indicate that spill is not directly into Iliamna River, which receives flow from Chinkelyes Creek.
[a] Confluence with Iliamna River for Chinkelyes Creek; confluence with Iliamna Lake for the Iliamna River and Knutson Creek.

11.5.1 Sources

11.5.1.1 Pipeline Failure

The volume of material released from a pipeline leak would depend on the type of failure, rate of loss from the pipe, pumping rate, leak duration, pipe diameter, distance to the nearest shutoff valves, and time until those valves are closed. For the purposes of this assessment, we evaluate a full break or a defect of equivalent size in the diesel pipeline that occurs at a stream crossing, thereby releasing fuel into that aquatic ecosystem. This could occur as a result of mechanical failure of the pipe from ground movement, vehicle impact, material failure or other cause. Characteristics of the pipeline are described in Table 6-4. We analyzed spills to two streams that would be crossed by the transportation corridor, Chinkelyes Creek and Knutson Creek (Section 11.2).

11.5.1.2 Diesel Fuel Composition

In the diesel pipeline failure scenarios, the pipeline would contain fuel from one of the Alaskan refineries and would have a composition similar to those presented by Geosphere and CH2M Hill (2006). Diesel fuel is a mixture of many hydrocarbon compounds, and its composition is a function of the petroleum feedstock source and the refining process. The type and amount of water-soluble hydrocarbons in the diesel determine the dissolved aqueous concentration when mixed with water. The most soluble compounds in diesel are the volatile aromatic hydrocarbons benzene, toluene, ethylbenzene and xylene (together, BTEX). Most diesel fuels have a low proportion of these soluble compounds and therefore have low solubilities. The bulk of diesel fuel is made up of heavier hydrocarbons that are essentially insoluble. A study of the composition of four diesel fuels from Alaskan refineries showed that the fuels had less than 2% BTEX and resulting diesel solubilities of 1.89 to 7.81 mg/L.

In the analysis of concentrations and solubilities, we incorporate all hydrocarbon compounds in the diesel samples and calculate the solubility based on Raoult's Law to account for effects of the mixture on the solubility of individual compounds.

11.5.2 Exposure

11.5.2.1 Background

A failure of the diesel pipeline in these scenarios could occur in the buried or above-ground portions. An above-ground failure would occur at a bridged stream or river crossing. An underground failure would result in diesel leaking into the soil and flowing down-gradient (e.g., as in the Trans-Alaska pipeline failure described in Section 11.5.3.3). If the underground failure occurred below a stream, it would float upward and into the surface water. An above-ground failure would release diesel directly to a river or stream, a wetland, or upland soil.

The behavior of diesel fuel in fresh water is less well-studied than the behavior of crude oil or diesel in marine environments. Diesel fuel has a density of less than 1.0 metric ton/m^3 and floats on water. It typically dissolves or evaporates within a day. In turbulent stream reaches, diesel would form small droplets suspended in the water column.

The soluble fraction would mix into the streamflow, be transported by advection and dispersion, and flow with the water. Solubility decreases with temperature, so in colder temperatures a smaller amount is dissolved in the stream. The soluble fraction is attenuated through dilution (advection and dispersion), biological activity, photodegradation, and aeration in turbulent streams, but is renewed by dissolution from the floating oil. The soluble compounds are also susceptible to evaporation from the floating oil, which typically occurs at a faster rate than dissolution. The soluble fraction compounds have relatively short residence times in water and sediments (Hayes et al. 1992) and can be reduced to below detection levels in a few days or weeks, depending on site-specific conditions.

Diesel components that are lighter than water and have low solubility tend to spread on the surface and form a thin film or sheen less than 0.1 mm thick. As the diesel spreads, it is more susceptible to destruction by evaporation, dissolution, and photodegradation but is also more likely to contact and attach to suspended sediments and shorelines. Most of the spilled diesel would flow with the stream until it reached Iliamna Lake and dissipated. The pour point of diesel (the temperature below which the oil will not flow) is approximately -7°C (20°F); thus, if the spill occurs during cold weather, the diesel would be less likely to spread and would instead form globs or strings and become suspended within the water column. For example, a 1999 cold-weather diesel spill in the Delaware River resulted in more than 90% of the diesel forming globules that were not visible from the surface (Overstreet and Galt 1995).

Oil dispersed in the water column can adhere to fine-grained suspended sediments that settle and deposit on stream edges and bottoms in low-energy areas. Depending on the source of the diesel, there may be a significant portion of compounds that are heavier than water and therefore sink, sorb to sediments, and persist longer than the dissolved fraction. In wetlands or pools and slack water areas of streams, a large percentage of spilled diesel can be deposited in the sediments.

When spilled on ice, diesel is viscous and forms tar-like accumulations on the surface. Lighter diesel components can penetrate the ice, become trapped within the ice structure, and be released as the ice melts. If the spill is trapped below the ice, as is more likely with buried pipelines, it would spread and stick to the underside of the ice in thin layers. Because cold temperatures reduce the solubility of diesel components, less would be dissolved in the stream water (NOAA and API 1994). As the ice breaks up and melts, the diesel would be released from the ice and mix with the stream water.

Because of its low viscosity (except in cold weather), diesel spilled onto the land tends to be rapidly absorbed by soil so that an above-ground spill on land could soon resemble an underground spill. In this area, where the groundwater surface tends to be shallow, spilled diesel would flow on top of the groundwater and a fraction would dissolve in that groundwater. It would then flow down-gradient to any nearby stream, possibly passing through wetlands on the way. Upon reaching a stream, it could pass into the channel through the gravels in which salmon, trout, and Dolly Varden spawn. In some locations, it might flow to Iliamna Lake and pass through a deltaic spawning beach used by sockeye salmon. Diesel-contaminated soil could episodically contaminate water when the water table rises following rain or snow melt. The extent to which fish eggs or fry are exposed by this route would depend on the specific structure of the spill site. Given the abundance of streams, wetlands, and shallow groundwater in the area crossed by the diesel pipeline, some variant of this exposure route is likely. However, saturated soils and particularly those that are frozen could result in overland rather than groundwater flow of diesel fuel.

The primary cause of toxicity to aquatic organisms in oil spills is direct exposure to the dissolved fraction. Exposure via this route would occur immediately following a direct spill to a stream or wetland as the oil dissolved, resulting in an acute exposure. Longer exposures to dissolved oil could result from slow releases of oil from terrestrial spills, flows from oiled wetlands, or the gradual dissolution of oil sorbed to sediments or plant materials. Oil spills can indirectly expose aquatic organisms to low dissolved oxygen as microbes decompose the oil.

Which of these transport and exposure processes would occur in a diesel spill depends on the spill location. The number and nature of water body crossings are the same as for the other pipelines (Section 11.2).

11.5.2.2 Transport and Fate

In the diesel pipeline failure scenarios, a pipeline failure would result in release of diesel directly into either Chinkelyes or Knutson Creek at mean streamflows (Table 11-7). The spill at Knutson Creek would release 12,000 L of diesel into approximately 1.6 million L of stream water, resulting in a 1:130 dilution. At Chinkelyes Creek, the spill would release approximately 30,000 L of diesel into 1.5 million L of stream water, resulting in a 1:49 dilution. At a typical diesel density of 850 g/L, this would result in 6,500 and 17,000 mg diesel/L water in Knutson and Chinkelyes Creeks, respectively. Both of these dilutions are less than the minimum aqueous volume required to get below the saturation of the diesel, if the dissolved hydrocarbons are well-mixed. This conclusion is based on calculation of the minimum volume of water required for diluting each component to a concentration below saturation. For benzene, the minimum volume of water required for dilution below saturation is 169 to 225 L benzene/L diesel; all other components would require higher dilutions. Thus, it is reasonable to assume that at both spill locations the diesel would be at saturation (i.e., at concentrations between 1.89 and 7.81 mg/L) in the receiving waters. Concentrations in the Iliamna River would be lower due to depletion of benzene. The benzene concentration would fall below its Raoult's saturation limit, resulting in a diesel concentration of 1.7 to 7.3 mg/l and a saturation of 92 to 94%.

11.5.3 Exposure-Response

Diesel is considered to be one of the most acutely toxic petroleum products (NOAA 2006), but its composition is variable. Although a model exists for estimating the acute aquatic toxicity of petroleum products from their chemical composition (Redman et al. 2012), the composition of diesel that would be piped to the mine is unknown. For example, the compositions of water-soluble fractions of two brands of Alaskan diesel fuel were found to be C4–C6 non-aromatic hydrocarbons (0.4–1.2 mg/L), benzene (0.03–0.2 mg/L), toluene (0.03–0.2 mg/L), and C2 benzenes (0.005–0.1 mg/L) (Guard et al. 1983). Given this variance in composition, data from laboratory tests and field studies of various whole diesel oils are used in this section to indicate the range of toxic effects observed in response to different exposures.

11.5.3.1 State Standards

According to Alaska water quality standards (ADEC 2011), total aqueous hydrocarbons in the water column may not exceed 15 µg/L and total aromatic hydrocarbons in the water column may not exceed 10 µg/L. The standards state (ADEC 2011): "There may be no concentrations of petroleum hydrocarbons, animal fat, or vegetable oils in shoreline or bottom sediments that cause deleterious effects to aquatic life. Surface waters and adjoining shorelines must be virtually free from floating oil, film, sheen, or discoloration."

11.5.3.2 Laboratory Tests

Laboratory tests of the toxicity of petroleum and its derivative fuels to aquatic organisms are performed with either an oil-water dispersion or a dissolved solution, called the water-soluble fraction. Dispersions are created by adding oil to water at prescribed ratios and mixing. The vigor and duration of mixing is variable, ranging from gentle mixing with a stirring rod to extended mixing with a magnetic stirrer. The resulting dispersion may have an oil layer on the surface as well as suspended oil droplets, although most tests attempt to avoid suspended material. The oil layer may be left in the test container, but more often the aqueous material is drawn off for the test. Results may be expressed as mg diesel/L or volume percent diesel. Water-soluble fractions are created by mixing oil and water to create a nominally saturated solution. The aqueous solution is drawn off and should be filtered to remove any suspended oil droplets. It is then diluted in water to create the test media. Results may be expressed as mg hydrocarbons/L or percent water-soluble fraction. In theory, one could also use toxicity data for each of the component chemicals in diesel fuel and estimate the combined effect based on individual effects, but that approach was judged to be impractical given uncertainties about diesel fuel composition in the scenarios and the paucity of toxicity data.

Potentially relevant results of tests of diesel dispersions and water-soluble fractions are summarized in Table 11-8. Results range over 4 orders of magnitude, and are highly variable even within an individual species or test type. This range results from differences in test procedures and diesel fuel compositions. Tests with biodiesel, synthetic diesel, sub-organismal endpoints, salt water, and dispersants were not included.

11.5.3.3 Analogous Spills: Diesel in Streams

Diesel spills into streams and wetlands are not uncommon, but their biological effects are seldom determined and published. Relevant diesel spill case studies are summarized in Table 11-9 and discussed in the text below. None of these studies were conducted in the Bristol Bay region, so they provide only a general indication of the nature and duration of effects expected from an instream diesel spill. We found no publications describing biological effects of diesel spills in relevant wetland habitats.

Multiple diesel spills have been associated with construction of the Trans-Alaska Pipeline, but biological effects were studied only for a 1972 spill from a broken underground pipeline that released 3,750 L to Happy Valley Creek (during spring streamflows of 14 m^3/s). Biological effects of the spill were studied downstream in the Sagavanirktok River (Nauman and Kernodle 1975, Alexander and VanCleve 1983). Invertebrate abundance declined by 89% after the spill (Nauman and Kernodle 1975), and stonefly and caddisfly nymphs were eliminated from the stream (Alexander and VanCleve 1983). Recovery was not reported.

A pipeline spill into Camas Creek, Montana, of oil that “most strongly resembled diesel fuel” resulted in low abundance and low richness of the invertebrate community with few mayfly, stonefly, and caddisfly taxa (Van Derveer et al. 1995). After remediation that included stream diversion and extensive removal of contaminated soil below the spill and recovery for approximately 1 year, taxa richness and abundance at the spill site were 60 to 70% of the upstream reference site, whereas at sites farther downstream from the remediation activities taxa richness and abundance were less than 15% and 10% of the reference site levels, respectively.

A tanker truck wreck in Trinity County, California, resulted in the flow of approximately half of a 15,000-L tank of diesel fuel into Hayfork Creek, a tributary of the Trinity River (Bury 1972). The oil was spilled on land and reached the stream after 36 hours. An area 1 to 2.5 miles below the spill was surveyed, because it had been previously studied. Numerous dead organisms were collected, including 4,469 vertebrates (rainbow trout and other fishes, tadpoles, snakes, turtles, and a bird) and uncounted thousands of macroinvertebrates. Recovery was not monitored.

A 1980 pipeline break released 340 m^3 of Number 2 fuel oil to a small tributary of Mine Run Creek, which ultimately flows to the Rapidan River, Virginia (Bass et al. 1987). The operator reported collecting 240 m^3 of oil. Monitoring was initiated 4 months after the spill, so acute effects were not observed. Standing crop, density, and diversity of macroinvertebrates were reduced in Mine Run Creek downstream of the tributary, and caddisflies were particularly affected. Effects were still observed at 16 months, when the study ended.

Table 11-8. Toxicity of diesel fuel to freshwater organisms in laboratory tests.

Species	Life Stage[a]	Test Endpoint	Concentration	Source—Notes
Water-Soluble Fraction				
Rainbow trout	Free-swimming embryos	9-day LC_{50}	8 mg/L	Schein et al. 2009—total dissolved hydrocarbon concentration
Rainbow trout	2 months after yolk resorption	48-hour LC_{50}	2.43 mg/L	Lockhart et al. 1987—total hydrocarbon concentration
Daphnia magna	1st instar	48-hour EC_{50}	6.7%	Giddings et al. 1980—percent water soluble fraction
Microcystis aeruginosa	Culture	4-hour carbon fixation	100%	Giddings et al. 1980—significant inhibition as percent water soluble fraction
Selenastrum capricornutum	Culture	4-hour carbon fixation	100%	Giddings et al. 1980—significant inhibition as percent water soluble fraction
Pseudokirchneriella subcapitata	Cultures	96-hour IC_{50}	58.7%	Pereira et al. 2012—inhibition of growth as percent water soluble fraction
Aqueous Dispersion				
Coho salmon	Juvenile	96-hour LC_{50}	10,299 mg/L	Wan et al. 1990—soft water
Coho salmon	Fry	96-hour TLm	2,870 mg/L	Hébert and Kussat 1972
Pink salmon	Juvenile	96-hour LC_{50}	74 mg/L	Wan et al. 1990—soft water
Rainbow trout	Juvenile	96-hour LC_{50}	3,017 mg/L	Wan et al. 1990—soft water
Rainbow trout	Fry	14-day LC_{50}	44.9 mg/L	Mos et al. 2008
Rainbow trout	Swim-up fry	72-hour LC_{50}	133.52 mg/L	Khan et al. 2007
Rainbow trout	Juvenile	96-hour LC_{50}	31 (6.6–65) mg/L	API 2003—mean and range of three tests
Fathead minnow	Juvenile	96-hour LC_{50}	57 mg/L	API 2003
Daphnia magna	Juvenile	24-hour LC_{50}	1.78 mg/L	Khan et al. 2007
Daphnia magna	Unspecified	96-hour LC_{50}	20.0 mg/L	Das and Konar 1988
Daphnia magna	Juvenile	48-hour LC_{50}	36 (2–210) mg/L	API 2003—mean and range of 12 tests
Chironomidae	Larvae	96-hour LC_{50}	346 mg/L	Das and Konar 1988
Selenastrum capricornutum	Culture	72-hour EL_{50}	20 (1.8–78) mg/L	API 2003—mean and range of seven results from three endpoints (inhibition of cell density, biomass, or growth) and three tests

Notes:

[a] As described by the authors.

LC_{50} =median lethal concentration; EC_{50} = median effective concentration; IC_{50} = median inhibitory concentration; TLm = equivalent to LC_{50}; EL_{50} = median effective level.

Table 11-9. Cases of diesel spills into streams. For comparison, the diesel pipeline failure scenarios evaluated here would release 30 and 8 m^3 of diesel into receiving streamflows of 1.8 and 3.4 m^3/s for spills into Chinkelyes Creek and Knutson Creek, respectively.

Case	Diesel Released (m^3)	Receiving Streamflow (m^3/s)	Observed Effects
Happy Valley Creek, AK	3.7	14	Significant declines in the abundance and species richness of invertebrates
Camas Creek, MT	Unknown	0.42	Low invertebrate abundance and richness
Hayfork Creek, CA	15	4.1	Large kill of vertebrates and invertebrates
Mine Run Creek, VA	240	1.2	Reduced invertebrate abundance and diversity
Reedy River, SC	3,600	6.4	Near-complete fish kill
Cayuga Inlet, NY	26	1.8	Fish kill and reduced abundance, reduced invertebrate abundance and species composition
Westlea Brook, UK	9.8	1.34	Fish kill, invertebrates severely affected
Hemlock Creek, NY	0.5	0.76	No significant effects on invertebrates

Notes:
[a] Mean flow from NHDPlus v2; others as reported by the authors.

In 1996, a pipeline ruptured and released 22,800 barrels (3.6 million L) of diesel into the Reedy River, South Carolina (Kubach et al. 2011). That spill resulted in a severe fish kill for 37 km downstream to the confluence with a reservoir. Recovery of the fish community, based on non-metric multidimensional scaling, occurred after 52 months.

In 1997, a train wreck spilled an estimated 26,500 L of diesel into Cayuga Inlet, a tributary stream of Cayuga Lake, New York (Lytle and Peckarsky 2001). Despite containment efforts, a kill occurred, which reduced fish (including rainbow trout) abundance by 92% and invertebrate abundance by 90%. Invertebrate density recovered within 1 year, but species composition had not recovered after 15 months.

In 2005, 9,800 L of diesel spilled into Westlea Brook in Wiltshire, UK (Smith et al. 2010). Due to its urban location, response was rapid, and approximately 7,000 L were recovered. However, the spill killed approximately 2,000 fish and a few frogs and birds. Invertebrate surveys showed that macroinvertebrates were severely affected and impacts were discernible for 4 km. Recovery occurred within the 13.5-month sampling period for all but the most affected site.

A tank of home heating oil (described as similar to diesel) leaked 500 L and an unknown amount entered Hemlock Creek, New York (Coghlan and Lund 2005). Three days after the spill, a survey of benthic invertebrates below the spill site found no significant reduction in the Hilsenhoff index (Coghlan and Lund 2005). The authors concluded that their techniques were sufficiently sensitive and no significant effects resulted from this small spill.

11.5.3.4 Analogous Spills: Crude Oil in Salmon Spawning Streams

The Exxon Valdez oil spill infiltrated the beaches of tidal Alaskan streams that provide spawning habitat for pink salmon (Rice et al. 2007). Water draining over the buried oil dissolved hydrocarbons, exposing salmon eggs and resulting in embryo histopathology and mortality for at least 2 years after the spill. The type of oil spilled and the circumstances of the spill are different from the diesel pipeline failure scenarios, but the studies described by Rice et al. (2007) demonstrate that oil buried near spawning habitats can be a source of potentially toxic exposures for years.

11.5.4 Risk Characterization

Toxicological risk characterization is performed primarily by calculating risk quotients based on the ratios of exposure levels to aquatic toxicological benchmarks (Box 8-3). However, it also includes consideration of actual diesel spills, the potential for remediation and recovery, site-specific factors, and the overall weight of evidence.

To characterize risks from a potential diesel spill, we weighed four lines of evidence based on different exposure estimates and sources of exposure-response relationships. The first two lines of evidence relate modeled estimates of dissolved hydrocarbon concentrations to laboratory test results for dissolved fractions of diesel oil and to state water quality standards. Because the diesel pipeline failure scenarios are sufficient to saturate the two potential receiving streams, we assume dissolved concentrations equal the solubilities of the Alaskan diesels (1.9 and 7.8 mg/L). Estimated concentrations in the Iliamna River are a little lower (1.7 and 7.2 mg/L) due to limited concentrations of soluble chemicals in diesel. These exposure levels are similar to the two median lethal concentration (LC_{50}) toxicity values for rainbow trout (2.43 and 8 mg/L) (Table 11-8) and far higher than the state standard (0.015 mg/L). Based on these estimates of soluble hydrocarbon concentrations, invertebrate kills would be highly likely and some salmonid mortality would be expected in the diesel pipeline failure scenario at either location.

The next line of evidence relates exposure (expressed as the amount of oil added to the stream) to laboratory test results for diesel dispersed in water. This line of evidence is based on the assumption that diesel added to a flowing stream is equivalent to diesel added to water and stirred. Exposure levels within the receiving water would be 17,000 mg/L for Chinkelyes Creek, 1,500 mg/L for the Iliamna River, and 6, 500 mg/L for Knutson Creek (Table 11-7). The laboratory LC_{50} tests for diesel dispersions are shown in Table 11-8, and strongly suggest that an oil spill would result in acute lethality of fish and invertebrates, even if turbulent mixing in a stream is not as efficient as stirring. In addition, tests of the alga *Selenastrum capricornutum* found that multiple growth and production endpoints were reduced by 50% at 20 mg/L (API 2003), which is also well below the estimated exposure.

The published history of freshwater diesel spills provides the final line of evidence. Diesel spill volumes at the two locations considered in these diesel pipeline failure scenarios—30 m^3 at Chinkelyes Creek and 12 m^3 at Knutson Creek—fall within the range of the cases described in Table 11-9 that caused effects on stream and river biotic communities. In addition, the sizes of the receiving streams in these failure scenarios and those in the case studies are similar. If we calculate a crude index of exposure by dividing the amount of diesel spilled by streamflow, values for the two scenarios (17 and 3.5) fall in the middle of the range of cases (0.26 to 560).

Only the case of a very small spill (less than 500 L into Hemlock Creek, NY) caused no significant biological effects. Other diesel spills caused fish and invertebrate kills and reduced invertebrate abundance and diversity. Invertebrate community effects persisted for several months to more than 3 years. Exposures and effects may be more persistent in Alaska's cold climate, but the only Alaskan study did not monitor recovery. Based on past diesel spills in streams, the diesel spills evaluated in this assessment—and any other spill that released more than a trivial amount of diesel to a stream—would be expected to cause an immediate loss of fish and invertebrates. The community would be likely to recover within 3 years, but the time to recovery in Bristol Bay streams is uncertain.

11.5.4.1 Weighing and Summarizing the Lines of Evidence

The diesel pipeline failure probability used in this assessment is based on one line of evidence, the record of actual oil pipelines. However, the predicted effects of a diesel spill are based on four lines of evidence. All lines of evidence lead to the conclusion that a diesel spill into a stream would result in an invertebrate and fish kill and reductions in abundance and diversity (Table 11-10). In the diesel pipeline failure scenarios evaluated here, the lengths of affected stream would be roughly 22 km (Chinkelyes Creek and the Iliamna River) or 2.6 km (Knutson Creek). Because these distances are short relative to oil degradation rates, effects would be likely to extend to Iliamna Lake. Effects in the lake are not estimated here, but are unlikely to extend far beyond the area of input due to dilution. In Knutson Creek, however, flow to Knutson Bay could result in mortality of congregated spawning salmon, their eggs, and other fish attracted by salmon eggs as a food source (Appendices A and B). Based on the monitoring of diesel spills in streams, effects on stream communities would be likely to persist for one to several years. Although each line of evidence has associated uncertainties and weaknesses (Section 11.5.5), they all support these general conclusions.

The weighing of these lines of evidence is summarized in Table 11-10, using the same methods described in Section 11.3.4.4. Overall, available lines of evidence for effects of a diesel spill are supportive of the hypothesis that acute toxic effects would occur following a diesel pipeline failure (Table 11-10). The quality of the exposure-response information is good (+) for all routes of exposure based on reported observations in case studies, because the information is realistic; the quality of information is considered good (+) for exposure via dissolved and hydrocarbons based on laboratory acute tests, because the information reflects multiple tests. The quality of the exposure information for the dissolved and dispersed hydrocarbons is considered ambiguous (0) because of the uncertain relationship between the laboratory preparations and modeled stream exposures. The quality of the exposure-response information is considered very good (++), because it is based on the Alaska water quality standard, an official standard. The analogous spills, as a whole, are considered very strong (++) evidence that a diesel spill would cause toxic effects in streams.

Table 11-10. Summary of evidence concerning risks to fish from a diesel spill. The risk characterization is based on weighing four lines of evidence for different routes of exposure. All evidence is qualitatively weighed (using one or more +, 0, - symbols) on three attributes: logical implication, strength, and quality. Here, all lines of evidence have the same logical implication—that is, all suggest a diesel spill would have adverse effects. Strength refers to how strongly the line of evidence indicates effects, and quality refers to the quality of the evidence sources (i.e., data quality and relevance to the diesel pipeline failure scenario).

Route of Exposure Source of Evidence (Exposure/E-R)	Logical Implication	Strength	Quality		Result
			Exposure	E-R	
Dissolved hydrocarbons Model/laboratory acute tests	+	+	0	+	Modeled dissolved diesel concentrations are clearly lethal to invertebrates and approximately lethal to trout.
Dissolved hydrocarbons Model/laboratory–based standard	+	++	0	++	Modeled dissolved diesel concentrations greatly exceed State standard.
Dispersed hydrocarbons Diesel-to-water ratio/laboratory acute tests	+	++	0	+	Diesel oil/water ratios in the spills and in tests suggest lethality to invertebrates and trout.
All routes in actual spills Amount spilled/observed effects	+	++	+	+	Diesel spills in other streams cause acute biological effects.
Summary Weight of Evidence	+	++	0	+	The effects by four lines of evidence are consistent and the observed effects are strong. The greatest uncertainty is the relation of laboratory to field exposures.

Notes:
E-R = exposure-response relationship.

The specific effects of a diesel spill on salmonid populations would depend on the individual receiving waters. Streams along the transportation corridor that could receive a spill are described in Section 10.1. Chinkelyes Creek receives on average roughly 9,000 spawning sockeye salmon and flows to the Iliamna River, which receives on average more than 100,000 sockeye spawners (Table 10-2). Knutson Creek receives 1,500 sockeye spawners and flows to Knutson Bay, which receives an average of 73,000 beach spawning sockeye (Table 10-2). Not all of those salmon spawn below the stream crossing, but these values indicate that a non-trivial number of spawners and their potential production are at risk. In these scenarios, a spill would likely disrupt spawning if it occurred during the spawning season and would potentially kill adults. In other seasons, it would likely kill fry, and would certainly kill invertebrates on which salmon fry and all stages of other salmonids depend.

11.5.4.2 Duration of Risks

Diesel and natural gas pipelines would be retained after mine closure as long as fuel was needed at the mine site (e.g., for monitoring, water treatment, and site maintenance). Therefore, the diesel pipeline risks would continue indefinitely.

11.5.4.3 Remediation

Remediation of freshwater oil spills is discussed in detail in a review by the National Oceanic and Atmospheric Administration (NOAA) and American Petroleum Institute (API) (1994). For diesel spills in small rivers and streams, remediation via booms, skimming, vacuum, berms, and sorbents results in the least environmental impact. Diesel is difficult to remediate by conventional techniques because its components seep into soil, dissolve in water, or evaporate relatively quickly, making it is less containable than typical crude oil. Also, booms, although useful, are imperfect tools for containing floating oil. Booms were deployed after the diesel spill in Cayuga Inlet (Table 11-9), but within 24 hours a slick was reported on Cayuga Lake, 16 km downstream (Lytle and Peckarsky 2001). Even when recovery of diesel fuel was rapid and approximately 70% effective, as in the Westlea Brook spill (Table 11-9), the rapidly dissolved component was sufficient to cause severe acute effects (Smith et al. 2010).

There has been relatively little study on remediation of oil spills in freshwater wetlands. For diesel, the NOAA and API (1994) review recommends natural recovery, sorbents, flooding, and low-pressure cold-water flushing as least adverse options. Wetlands also have been remediated by burning, which can remove floating oil and destroy oiled vegetation that is likely to die from effects of the oil. Burning can cause severe but localized and short-term air pollution and, if improperly controlled, can result in fires that spread beyond the oiled area. However, burning does not destroy the dissolved fraction, which would move to streams or the lake and is primarily responsible for aquatic toxicity.

Cold winter weather complicates remediation of diesel spills (NOAA and API 1994). Spills into water at temperatures below the oil's pour point can result in the formation of viscous tar-like particles that are difficult to recover. Ideally, a spill onto ice could congeal on the surface where it might be relatively easily recovered if action is prompt; however, diesel oil can penetrate ice, and solar absorption by the oil can result in freeze-thaw cycles that create a complex material. Spills that flow under ice deposit on the undersurface. Standard procedures for oil remediation do not address those conditions.

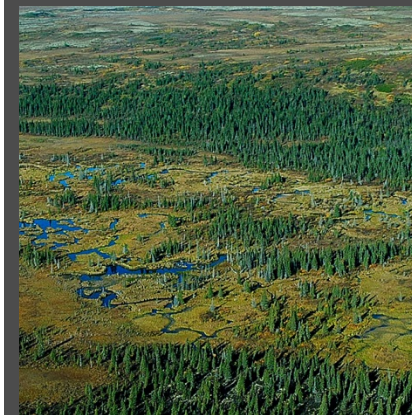
11.5.5 Uncertainties

Based on weighing multiple lines of evidence, it is certain that a diesel pipeline spill into a stream would cause acute toxic effects. However, the following uncertainties apply to individual pieces of evidence.

- The composition of diesel oil is highly variable. As a result, the fate and toxicity of diesel spills are inherently uncertain unless the specific source is known and analyzed; the source does not change over time; and any physical, chemical, and biological tests are performed with that specific oil. This uncertainty cannot be resolved without case-specific studies of a sort that are not normally performed. This and other uncertainties concerning test results could cause errors of at least one order of magnitude in the risks estimated from laboratory toxicology.
- Measurement of petroleum hydrocarbons in water is performed using a variety of methods. Because the results of hydrocarbon analysis are method-specific, significant uncertainty can be introduced when these results are compared to benchmarks generated using different analytical methods. This contributes to the overall uncertainty of toxicity test results.
- Invertebrate and fish losses are likely if a diesel spill occurs at a stream, but the magnitude and nature of these losses would be highly uncertain. Some mortality would occur for some species, but the species and number of organisms affected cannot be specified. This uncertainty would take a

major case-specific research program to resolve. The inability to exactly define the expected ecological effects occurs in all risk assessments, but is worse for the diesel spill than for other contaminants such as copper.

- The ability of the laboratory toxicity tests to predict responses to diesel in the field is highly uncertain due to the variety of preparation methods, the simplicity of laboratory exposures relative to the complexity of oil spills in streams, and the lack of field validation studies.
- Variation in sensitivity to diesel among species appears to be high relative to other aquatic pollutants. Remarkably, even in the same test series, different salmon species range in sensitivity over two orders of magnitude (Table 11-8).
- Spills into wetlands are likely to have severe and persistent effects due to low rates of flow, but no relevant studies of diesel spills in freshwater wetlands are available to confirm even that very general hypothesis.
- The applicability of previous diesel spills considered in Table 11-9 to streams in the Bristol Bay region is uncertain, given that all of the spills occurred elsewhere. However, the effects observed in the one Alaskan case are not dissimilar from those in temperate regions. The most likely differences are slower loss of oil and longer recovery times. Therefore, effects are likely to be more severe in Alaska than in the temperate cases.
- The principle uncertainty in this analysis is the number and location of spills into aquatic ecosystems, given the probability of a pipeline failure. We can say with some certainty that a diesel spill of a non-trivial volume into a stream would have adverse ecological effects. We can also say that a spill is likely, based on the record of oil pipelines in general and large recent spills from oil pipelines (e.g., into the Kalamazoo River, as described in Section 11.1). However, we cannot predict with any certainty where such as spill may occur.
- Although the diesel spill cases suggest that streams are likely to recover within 3 years, time to recovery is seldom reported. Where it has been reported, it apparently depends on the conditions and the recovery metric used, and ranges from a year to several years.



CHAPTER 12. FISH-MEDIATED EFFECTS

Large-scale mining, as described in the mine scenarios (Table 6-1), could have both direct and indirect effects on wildlife and Alaska Native cultures (Figures 12-1 and 12-2). In this chapter, we primarily consider indirect effects, focusing on how wildlife and Alaska Native cultures may be affected by any mining-associated changes in salmon resources. Direct effects on these endpoints—defined here as effects that are independent of impacts on fish populations—could be significant, and would need to be fully evaluated as part of a comprehensive environmental impact statement for any proposed future development. However, these direct effects are generally considered outside the scope of the current assessment (Chapter 2) and are only mentioned briefly here (Box 12-1). Potential cumulative effects that multiple mines in the region may have on wildlife and Alaska Native cultures are discussed in Chapter 13.

12.1 Effects on Wildlife

As discussed in Chapters 7 through 11, a large-scale mine and its associated transportation corridor would likely affect the abundance, productivity, and diversity of Pacific salmon. These changes in salmon resources could stem from direct habitat losses and downstream flow alterations resulting from the mine footprint, or from changes in the physical and chemical habitat characteristics resulting from mine operations and potential accidents or failures. Wildlife species in the Nushagak and Kvichak River watersheds that depend on salmon could be affected by decreases in salmon abundance. Interactions between salmon and other fish and wildlife, and the potential for disruption of these interactions, are complex (Section 5.2.5). In this section, we qualitatively consider how a decrease in salmon abundance may affect wildlife—that is, salmon-mediated effects on wildlife—via the loss of salmon as a food source and the loss of marine-derived nutrients (MDN) as a source of productivity.

BOX 12-1. POTENTIAL DIRECT EFFECTS OF MINING

The salmon-mediated effects considered in this assessment represent only one component of potential large-scale mining impacts on wildlife and Alaska Natives. Both wildlife and Alaska Natives would likely experience direct impacts, the magnitude and extent of which could be significant. For example, direct impacts on wildlife would include loss of terrestrial and aquatic habitat, reduced habitat effectiveness (e.g., in otherwise suitable habitats adjacent to mine area), habitat fragmentation, increased stress and avoidance due to noise pollution, and increased conditioning on human food (Figure 12-1).

Direct effects of large-scale mining on Alaska Native populations could result from multiple stressors, including noise pollution, air emissions, changes to water supply and quality, an influx of new residents, and induced development. Mine construction and operation also would have direct economic and social effects, both positive and negative, on Alaska Native cultures. For example, 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. A shift from part-time to full-time wage employment in mining or mine-associated jobs would provide additional employment opportunities and income, but would affect subsistence-gathering capabilities by reducing the time available to harvest and process subsistence resources.

At this time, it is difficult to determine what, if any, effects routine operations at the Pebble deposit would have on drinking water sources in the Nushagak and Kvichak River watersheds. Private wells are a primary drinking water source for many residents of the Nushagak and Kvichak River watersheds, and communities also rely on groundwater for their public water supply. The extent to which surface water influences the quality or quantity of the groundwater source for these wells is unknown. There are also communities in the area that rely on surface water sources, which may be more susceptible to mine-related contamination.

Although a thorough evaluation of potential direct effects of large-scale mining on wildlife and Alaska Native populations is beyond the scope of this assessment, these examples illustrate just a few of the complex ways in which wildlife and Alaska Natives could be affected by large-scale mine development.

Lower salmon production would likely reduce the abundance and production of wildlife in the mine area and presumably in the range areas of the affected species, but the magnitude of those effects cannot be quantified. The Bristol Bay region is home to a complex foodweb that includes salmon and salmon predators and scavengers (Box 5-3, Figure 12-1). Annual salmon runs provide food for brown bears, wolves, bald eagles, other land birds, and water birds, and it is likely that these species would be directly affected by a reduction in salmon abundance. Waterfowl prey on salmon eggs, parr, and smolts and scavenge on carcasses. Salmon carcasses are an important food source for bald eagles, water birds, other land birds, other freshwater fish, and other terrestrial mammals. Aquatic invertebrate larvae also benefit from carcasses and are an important food source for water birds and land birds.

Figure 12-1. Conceptual model illustrating potential effects on wildlife resulting from effects on salmon.

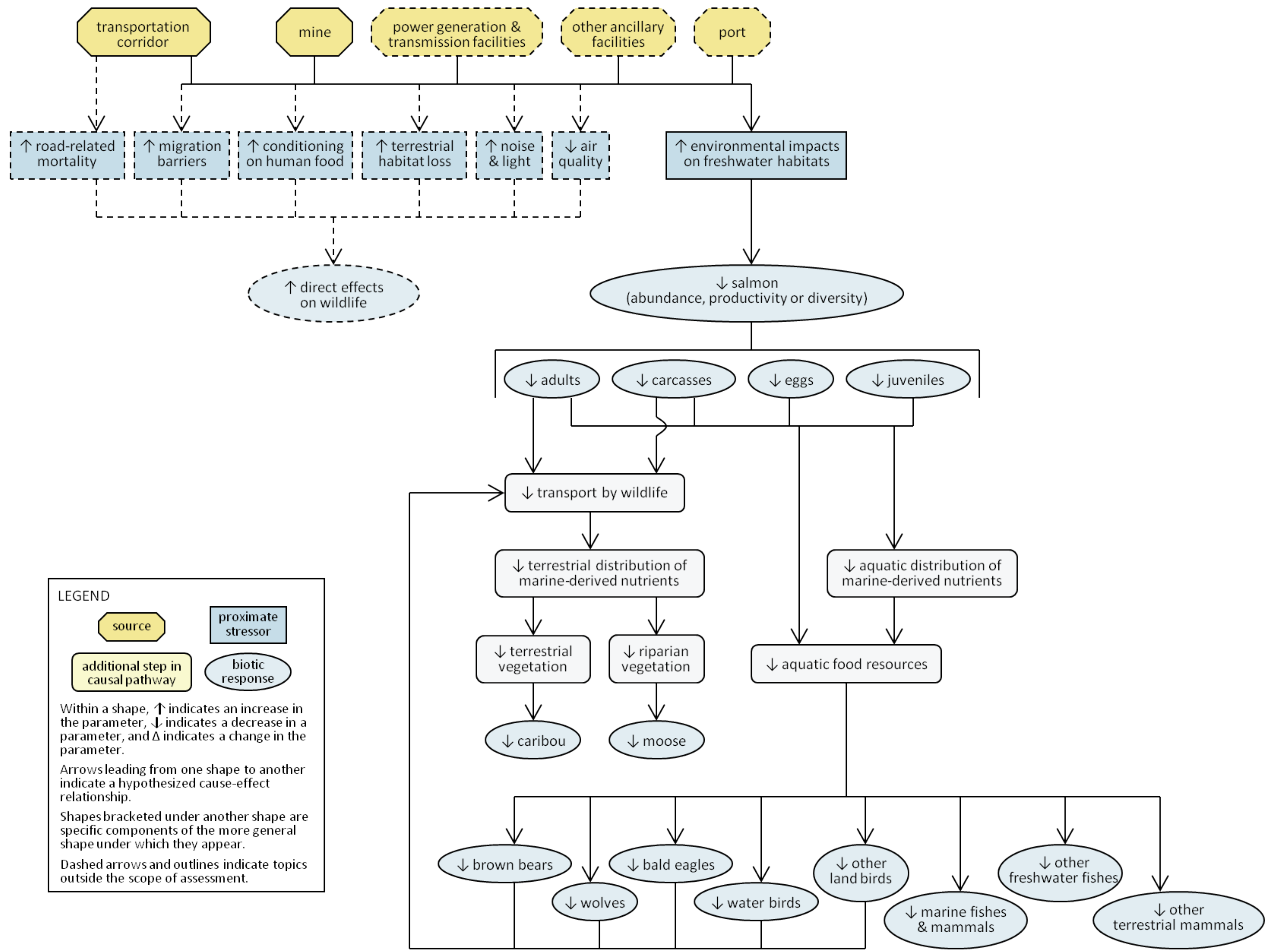
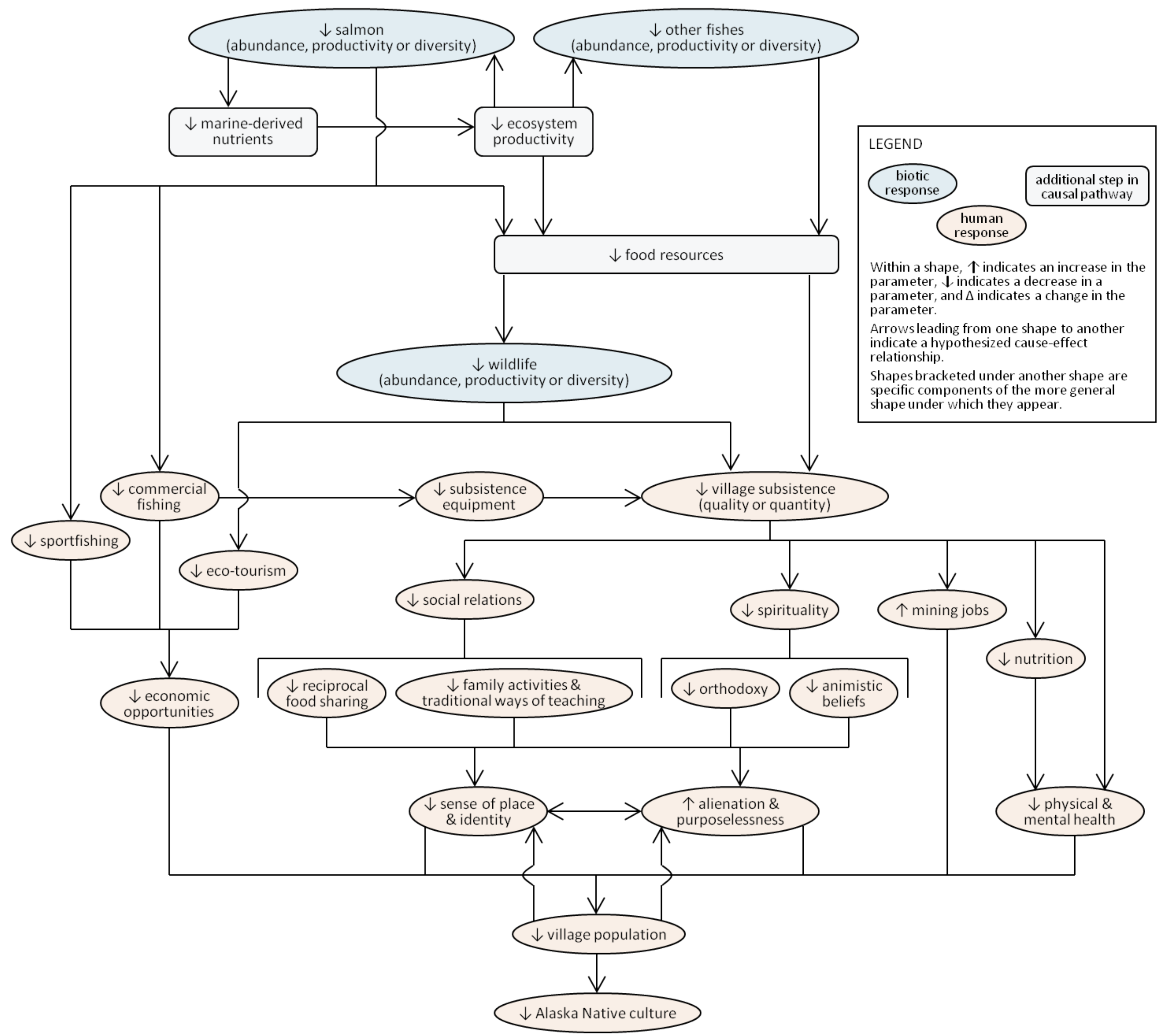


Figure 12-2. Conceptual model illustrating potential effects on Alaska Native cultures resulting from effects on salmon and other fishes.



Salmon predators and scavengers then deposit MDN on the landscape, as either carcasses or excreta. These nutrients contribute to the plant production that supports caribou, birds, and other terrestrial wildlife. Caribou are, in turn, prey species for wolves and brown bears. The link between increased vegetation and MDN distributed by brown bears has been documented (Hilderbrand et al. 1999, Helfield and Naiman 2006), but additional research is needed to confirm and quantify the links between moose, caribou, and MDN.

Factors such as the magnitude, seasonality, duration, and location of salmon losses would influence the specific wildlife species affected and the magnitude of effects. Generally, the loss of salmon as food resources in any area of the mine scenario watersheds would be expected to create displacement or loss of wildlife species dependent on those food resources. If the loss were of a sufficient magnitude and duration, there may be additional indirect effects, such as loss of vegetation from lack of MDN and consequent loss of food resources for species such as moose and caribou. Should riparian vegetation be reduced by long-term loss of MDN, there would be decreased food resources for moose, particularly in the Nushagak and Mulchatna River systems, which have large riparian zones (Brna and Verbrugge 2013).

Seasonality of salmon resources is also important for wildlife species. Brown bears, wolves, bald eagles, and other species depend on salmon for a large fraction of their summer diet. Mine failures that reduced or eliminated a salmon substock would be expected to reduce or displace wildlife species that depend on those particular salmon nutrients during important life-history periods such as breeding, nesting, and pre-winter feeding.

Alaska Natives have expressed concerns that wildlife may be affected by consuming contaminated fish. Two potential contaminants of concern are copper and selenium. The primary aquatic contaminant of concern from a porphyry copper mine is copper, which can cause both acute and chronic toxicity to salmon and other fishes (Chapter 8). However, copper is relatively weakly accumulated by fish in both aqueous and dietary exposures and does not bioaccumulate. 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).

Data for copper toxicity to wildlife are not available, because direct toxicity has not been a problem—the indirect effects of reduced aquatic prey are likely to be greater than direct toxic effects. However, the dietary maximum for pigs and poultry of 200 mg/kg dry weight can be used as a surrogate benchmark (Eisler 2000). If we use the highest reported fish bioconcentration factor (290 for fathead minnows, from an unpublished manuscript cited in USEPA 1985) as a conservative value, we obtain a safe water concentration of 690 μg/L. This safe level for wildlife is much higher than both the toxic levels for aquatic biota (Section 8.2.2.1) and the estimated instream concentrations for a wastewater treatment plant failure (Table 8-20). It is a little higher than the estimated concentration in the concentrate transport water and return water (655 μg/L, Section 11.3.2), but is not a concern because of dilution and the short duration of exposures to spills. The copper concentration in the product concentrate leachate is 12 times that value. However, a product spill would be localized at the mine site or along the transportation corridor. Most important, if copper concentrations were so high that the leachate was not diluted by a factor of 12, there would not be a sufficient aquatic community remaining after a spill to attract species such as mink, river otters, or belted kingfishers, which may forage in a particular stream or river for fishes such as trout, char, or salmon fry. Hence, copper toxicity to fish-eating wildlife is improbable.

Selenium is a well-characterized avian toxicant that has been a concern for waste rock leachate at other mines (USEPA 2011). It does biomagnify and the primary route of exposure for fish is diet (Chapman et al. 2010). However, selenium bioaccumulation depends on biogeochemical conditions that occur in slowly flowing (lentic) ecosystems such as ponds and wetlands, and is much less prevalent in the streams that are the likely receptors for effluents with elevated selenium levels. In addition, the selenium concentrations in wastes identified in the mine scenarios would be relatively low. Mean selenium (the appropriate measure for a biomagnifying chemical) expected in waste rock leachates is below the water quality criterion, and leachates from tailings and product concentrate are only 1.5 times the criterion concentration (Section 8.2.2.2). Hence, minimal dilution would bring these concentrations down to safe levels. Although fish and birds are sensitive to selenium, mammals are not. The body burdens of adult salmon are almost entirely due to marine exposures; because salmonid eggs take up contaminants relatively slowly, body burdens of eggs and larvae would also be expected to reflect marine sources. Local sources of selenium are therefore not relevant, and bioaccumulation of selenium would only be a concern for wildlife consuming resident fish or older salmon fry. Fish-eating birds feeding on resident fish also may forage in more than one stream or other water body, potentially providing further dilution. For these reasons, aqueous selenium is unlikely to pose a risk to wildlife via fish consumption.

12.2 Effects on Alaska Natives

As discussed in Chapters 7 through 11, routine development and operation of a large-scale mine, as well as potential mine accidents or failures, would likely affect salmon resources in the Nushagak and Kvichak River watersheds. The importance of salmon to Alaska Native cultures is well documented (Section 5.4, Appendix D). Because these cultures are so intimately related to the local landscape and the resources it provides, any changes to salmon or other subsistence resources would likely result in changes to the cultures. The magnitude of these changes could be assumed to be dependent on the magnitude and duration of both the loss of subsistence resources and the disruption to the landscape itself. Changes in salmon resources may affect indigenous health, welfare, and cultural stability in several ways (Appendix D).

- Because the traditional diet is heavily dependent on wild foods, particularly salmon, diets would move from highly nutritious wild foods to increased reliance on purchased processed foods.
- Social networks are highly dependent on procuring and sharing salmon and wild food resources, so the current social support system would be degraded.

- The transmission of cultural values, language learning, and family cohesion would be affected because meaningful family-based work takes place in fish camps or similar settings for traditional ways of life.
- Values and belief systems are represented by interaction with the natural world through salmon practices, clean water practices, and symbolic rituals. Thus, core beliefs would be challenged by a loss of salmon resources, potentially resulting in a breakdown of cultural values, mental health degradation, and behavioral disorders.
- The region exhibits a high degree of cultural uniformity tied to shared traditional and customary practices, so significant change could provoke increased tension and discord both between villages and among villagers.

Human health and cultural effects related to potential decreases in salmon resources would depend on 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.

Salmon-mediated effects from potential accidents and failures associated with large-scale mining would likely have much greater effects on human welfare and Alaska Native cultures than the effects from routine operations. It should be assumed that any negative impact on salmon quantity or quality resulting from mine failures or accidents would affect human health and welfare, both from loss of or change in food resources and from cultural disruption. Because all aspects of Alaska Native cultures in the Nushagak and Kvichak River watersheds are closely tied to salmon and other fishes, cultural vulnerability to long-term environmental disruption is very high (Appendix D). A major failure or accident that resulted in long-term disruption of salmon habitat and ongoing toxicity to salmon or their food would significantly affect both subsistence resources and cultural identity. Potential causes of salmon-mediated effects on Alaska Native cultures would differ across the two watersheds. For example, villages near the transportation corridor could be negatively affected by road and culvert failures (Chapter 10) or pipeline spills (Chapter 11). Villages downstream of the mine would be more affected by any water collection, treatment, and discharge failures (Chapter 8), and impacts from these failures would likely be much greater than impacts from routine operations.

This assessment focuses on potential effects on Alaska Native cultures, but other groups would also be particularly vulnerable to mining-associated impacts on salmon. Many of the non-Alaska Natives that reside in the area practice a subsistence way of life and have strong cultural ties to the landscape that go back generations (Box 12-2). Many seasonal commercial anglers and cannery workers also depend on these resources and have strong, multi-generational cultural connections to the region.

In this section, we discuss the range of potential salmon-mediated effects on Alaska Native cultures from large-scale mining in the Nushagak and Kvichak River watersheds. We also reference key impacts that other mining, oil, and gas development activities in Alaska—including northwest Alaska's Red Dog Mine and oil and gas development on Alaska's North Slope—have had on Alaska Native cultures, especially in terms of losses of or changes to subsistence resources. Although not directly applicable to large-scale mining, information about oil and gas extraction activities provides insight into potential effects of large-scale mining on Alaska Native culture in the Nushagak and Kvichak River watersheds.

BOX 12-2. TESTIMONY ON POTENTIAL EFFECTS OF MINING ON ALASKA NATIVE CULTURES

The U.S. Environmental Protection Agency (USEPA) held a series of public meetings to collect input on the May 2012 draft of this assessment. Many Alaska Natives, including tribal Elders and other tribal leaders, provided testimony on concerns about potential effects of large-scale mining in the Bristol Bay watershed, as well as the desire for economic development. The following are selected quotes representative of this testimony. To view the full public meeting transcripts, visit www.epa.gov/bristolbay.

- "Salmon has been part of our Native spiritual food; and without the food and waters we will die slowly, we'll be here existing, but our spirit will be gone."
- "I urge you to pay especially close attention to the voices of our Elders across Bristol Bay. They have instilled in them the deepest of our roots, and our God given way of life; our culture that has been slowly fading away. It is the adaptation to modern civilization that we have embraced so far that is causing our cultures to become lost."
- "We support the science of this document as it in turn supports what the Elders of this area and their traditional knowledge have said all along. We are preparing our boats, we are mending our nets, we are cleaning our smokehouses and we are sharpening our knives. However, by testifying at these meetings and missing out on one day or maybe several days of preparation for those who attended multiple meetings here in the region, we hope that this will prevent, with the help of the EPA, missing out on a lifetime of salmon and missing out on a way of life that we have treasured for thousands of years."
- "Bristol Bay is much different. Everyone who lives here has a deep and strong sense of place. There is a powerful connection to the lands and waters and resources of Bristol Bay. It is a connection that starts before birth. It is genetic. It is handed down through the generations and it is also learned from a very young age. A connection told in stories from parents and Elders and experienced firsthand. Toddlers accompany parents and grandparents fishing, hunting, berry picking. They participate at home to store that food and save it. It is part of the family experiencing for anyone who grows up in Bristol Bay, and as a result, this land its water and its resources become a part of who you are. This is a connection without a price tag and it cannot be replaced. If it lost, it is lost forever."
- "My family does subsistence. We like our fish. But still, nobody is going to give my boys jobs. Nobody is going to pay my bills. I'm not for or against. I want clean water, but we need jobs around here. Who is going to pay for my bills?"
- "I work for Pebble. I have a big family who loves the outdoors and enjoy their subsistence way of life. Subsistence is good, but it is not paying for my bills and does not clothe my kids."

12.2.1 Subsistence Use

As discussed in Chapter 5, subsistence foods make up a substantial proportion of the human diet in the Nushagak and Kvichak River watersheds and likely contribute a disproportionately high amount of protein and certain nutrients. The percentage of salmon harvest in relation to all subsistence resources ranges from 29 to 82% in the villages (Appendix D).

The mine scenario footprints would have some effects on subsistence resources. Although no subsistence salmon fisheries are documented directly in any of the mine scenario footprints, other fish are harvested in these locations, and the areas are identified as being important for the health and abundance of subsistence resources (PLP 2011: Chapter 23). Negative impacts on downstream fisheries from headwater disturbance (Section 7.2) could affect subsistence salmon resources beyond the mine footprints. Those residents using the mine area and immediate areas downstream of the mine pit and tailings storage facilities (TSFs) for subsistence harvests would be most affected (Figure 5-2). Access to subsistence resources is also important. A reduction in downstream seasonal water levels caused by mine-related withdrawals during and after mine operation could pose obstacles for subsistence users who are dependent on water for transportation to fishing, hunting, or gathering areas.

There could also be effects from the footprint of the transportation corridor. A review of Alaska Department of Fish and Game data (Appendix D: Table 13) indicates that some residents use the area along the transportation corridor considered in the assessment for subsistence salmon harvest. Of the villages in these watersheds, reliance on salmon is highest in Pedro Bay, where salmon provides 82% of per-capita subsistence harvest. The estimated annual per-capita subsistence harvest for Pedro Bay, the village closest to the transportation corridor considered in this assessment, was 306 pounds in 2004. Thus, this village is particularly vulnerable to losses of salmon resources.

The effects of the transportation corridor on subsistence resources would be complex and unpredictable. Based on the analysis in Chapter 10, we anticipate that routine transportation operations would have some negative effects on salmon habitat in streams along and downstream from the transportation corridor. Some subsistence users in these areas could be affected. The corridor also would increase accessibility of the area, which could increase subsistence use of nearby streams but also create greater competition for resources.

The initial effect of a mine 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 the geographic extent of disruption to subsistence resources. However, this assessment provides examples of the potential magnitude of salmon impacts from failures. One such example is the potential effect of a tailings dam failure on Chinook salmon in the Nushagak River. As described in Chapter 9, a tailings dam failure at TSF 1 could significantly affect Koktuli River Chinook runs, which constitute up to 29% of the larger Nushagak River Chinook runs. Stuyahok River and Mulchatna River Chinook runs, which constitute up to 17 and 10% of the Nushagak River Chinook runs, respectively, could also be affected. The Alaska Native villages on the Nushagak River (Koliganek, New Stuyahok, Ekwok, and Dillingham) (Figure 2-4) are culturally and nutritionally dependent on Chinook salmon. Thus, a tailings dam failure would have negative and potentially significant effects on the ability of subsistence users to harvest salmon downstream of the mine area.

It is not possible to predict the magnitude of effects from the loss of salmon as a subsistence food, 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 consequent substitution of purchased foods would have negative effects on individual and public health (Appendix D). Salmon is especially valued around the world for nutrition and disease prevention. Dietary transition away from subsistence foods in rural Alaska carries a high risk of increased consumption of processed simple carbohydrates and saturated fats. This has occurred in 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 and are certainly not as healthy. Compounding the shift to a less healthy diet, the physical benefits of engaging in a subsistence lifestyle would be reduced (Appendix D).

In addition to the salmon-mediated effects of large-scale mining considered here, there could be effects from the loss of non-salmon subsistence resources, such as land mammals, birds, and other fishes. Subsistence use of the mine area is high and centers on hunting caribou and moose and trapping small mammals (Braund and Associates 2011 in PLP 2011). Because no subsistence salmon fisheries are documented in the mine scenario footprints, direct loss of non-salmon subsistence food resources likely would represent a greater direct effect than loss of salmon harvest areas in the mine footprints. Tribal Elders have expressed concerns about ongoing mine exploration activities directly affecting wildlife resources, especially the caribou herd range (Appendix D).

Experience with existing development in Alaska supports the contention that development of a large-scale mine operation would directly affect subsistence resources within and around the mine scenario footprints during routine operations and in perpetuity, from both loss of habitat and disturbance related to routine operations. For example, the supplemental environmental impact statement for the Red Dog Mine (USEPA 2009) documented multiple subsistence impacts, including reduced harvest of beluga by Kivalina harvesters, likely related to port activities. Related to transportation corridors, traffic along the Delong Mountain Regional Transportation System road was found to cause “limited, localized” effects on caribou movement and distribution, and nine caribou fatalities occurred because of traffic collisions. Kivalina harvesters and harvest data also indicated that traffic along the road has likely resulted in fewer caribou harvested by Kivalina harvesters than would otherwise be the case.

A study of the cumulative environmental effects of oil and gas activities on Alaska’s North Slope included a summary of hearings held with North Slope residents, who are predominantly Alaska Natives. Community members provided testimony on both positive and negative effects of these activities. North Slope residents recognize that oil production in the region has brought benefits such as money to spend on community facilities, schools, modern water and sewer systems, village clinics, child emergency shelters, and behavioral outpatient and residential programs that provide mental health care and counseling for substance abuse and domestic violence. However, they also reported that traditional subsistence hunting areas have been reduced, the behavior and migratory patterns of key subsistence species have changed, and there is increased incidence of cancer and diabetes and disruption of traditional social systems.

Residents also reported experiencing significant increases in the time, effort, and funding necessary to respond politically and administratively to the increased number of projects proposed in their communities (NRC 2003). The stress of integrating a new way of life with generations of traditional teachings and the associated impacts of rapid modernization and loss of tradition is known as acculturative stress. This stress has been linked to a wide variety of health outcomes, ranging from impaired mental health and social pathology (such as substance abuse, violence, and suicide) to cardiovascular disease and diabetes. For the Inupiat on the North Slope of Alaska, the greatest defense against acculturative stress is the continued practice of the bowhead whale hunt, which involves the entire community (NMFS 2013).

Changes in diet and nutrition are common potential effects of oil and gas exploration and production activities where populations rely on subsistence resources. These changes can lead to a number of important public health outcomes. For example, a traditional diet has been shown to be strongly protective against chronic diseases for indigenous populations. A shift away from subsistence diets is associated with food insecurity, or the inability to secure sufficient healthy food for a family. Studies of food insecurity and health have found a variety of detrimental health impacts, including obesity, poor psychological function among children, poor cardiovascular health outcomes, and lower physical and mental health ratings (NMFS 2013). The high cost of store-bought food, the costs associated with harvesting of subsistence resources, and the year-to-year variation in subsistence resource availability are all implicated in the high food insecurity rates experienced by many northern indigenous populations.

Alaska Native residents also report subtle changes in species harvested by subsistence hunters, including changes in color, texture, and taste of the flesh and skin of several subsistence species. Transportation corridors associated with resource extraction activities can also increase competition for local subsistence resources. For example, hunting by non-local residents along the Dalton highway has been reported to have increased after the development (and later public opening) of the road (NRC 2003).

The experiences of subsistence users near Red Dog Mine and Alaska's North Slope indicate that localized changes in resource movement can affect that resource's availability and predictability to subsistence users, even when the overall pattern or abundance of the resource may not be affected by development activities. From a biological standpoint, changes in caribou related to the Red Dog Mine may be viewed as minimal. However, because residents rely on only a portion of the expansive range of the Western Arctic caribou herd to harvest caribou, small and localized changes in caribou availability can have large effects on subsistence uses. Subsistence users have observed changed or diverted migration routes, reduced harvests of caribou, decreased size of caribou individuals and groups, and increased disease and infection since mine operations began, and cite both mine-related and other causes (USEPA 2009).

The Exxon Valdez oil spill also resulted in reduced subsistence activities (Palinkas et al. 1993). These reductions resulted from the closure of many areas to subsistence activities, local concerns over subsistence food safety, voluntary abstinence from consumption after the spill, and reduced time for subsistence activities by Alaska Natives who participated in cleanup efforts.

12.2.2 Perception of Food Security

Even a negligible reduction in salmon quantity or quality related to mining could decrease use of salmon resources, 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). Aside from actual exposure to environmental contamination, the perception of exposure to contamination is also linked to known health consequences, including stress and anxiety about the safety of subsistence foods and avoidance of subsistence food sources (Joyce 2008, CEAA 2010, Loring et al. 2010), with potential changes in nutrition-related diseases as a result. These health results arise regardless of whether there is contamination at a level that could induce toxicological effects in humans—rather, the effects are linked to the perception of contamination (NMFS 2011).

Literature on impacts from oil and gas development on Alaska's North Slope and ongoing operations at Red Dog Mine demonstrates that even perceived contamination could have a real effect on subsistence harvesters. In a recent survey, 44% of Inupiat village residents reported concern that fish and wildlife may be unsafe to eat (Poppel et al. 2007, NMFS 2011). Residents of Kivalina and Noatak, the communities closest to Red Dog Mine, also have expressed concerns about food safety, potential contamination of subsistence resources, and corresponding changes in subsistence foraging (USEPA 2009). Kivalina residents are concerned about potential contamination of the Wulik River, which is used both for subsistence and as the drinking water source for the village. These concerns persist even though studies by the Alaska Department of Health and Social Services found that heavy metal concentrations in drinking water were low and did not pose a risk (USEPA 2009).

12.2.3 Economic Impacts

Alaska Natives, as well as other local residents, participate in the salmon-based market economy, primarily via commercial fishing and tourism. Subsistence harvests also represent a significant economic value to local residents. A decrease in salmon that affected either of these sectors would be particularly burdensome to local residents dependent on the commercial fishery for income and the subsistence fishery for food. The necessity of purchasing expensive foods from outside the region, in conjunction with more limited opportunities to obtain paid seasonal employment in the region, could be extremely difficult for families. In many cases, income from commercial and recreational fishing provides money to purchase equipment for subsistence fishing, so lost or reduced income from commercial fishing would affect subsistence harvests even if subsistence fishing remains possible. For those able to benefit economically from mining and induced development, there would be increased cash resources to purchase equipment and supplies, resulting in more efficient subsistence activities. However, increased full-time employment could decrease the time available for subsistence activities and thus the social relationships based on these activities. Some residents have expressed a desire for jobs and development related to large-scale mining and a market economy, whereas other residents have expressed concerns that this type of economic shift would be detrimental to their culture (Box 12-2, Appendix D).

Although large-scale mining would inject some market-based economic benefits for some period of time, resource extraction experiences in other rural Alaska areas suggest it would likely have only modest direct employment benefits in the local region (Goldsmith 2007). At the Red Dog Mine, ownership of the resource empowered the NANA Regional Corporation, Inc. (NANA) to negotiate a development agreement with strong protections and benefits to Northwest Alaska Natives (Storey and Hamilton 2004). NANA shareholders account for approximately 56% of the mine's 464 full-time employees and 91% of its 78 part-time employees. Although first preference in hiring and most of the training slots go to shareholders, shareholders disproportionately occupy the mine's lower-skilled positions (Storey and Hamilton 2004). Additionally, the supplemental environmental impact statement showed that employment at the Red Dog Mine may have facilitated the relocation of community residents to Anchorage for lifestyle or economic reasons (Storey and Hamilton 2004, USEPA 2009).

A disproportionately low number of Inupiat people are employed by the oil and gas industry on Alaska's North Slope, although this may partially result from the large percentage of young people in the population (NRC 2003). The Alaska Department of Labor reported that, of the 7,432 people who reported working in the oil and gas sector on the North Slope in 1999 (and worked for companies that collected and reported residency information), only 64 lived in the state's Northern Region (i.e., the Nome, North Slope, and Northwest Arctic boroughs). A variety of factors affected both male Inupiat willingness to work in the oil fields and the desire of companies in Prudhoe Bay to hire them (Kruse et al. 1983, NRC 2003).

There may be decreased participation in a subsistence way of life for those benefiting from any employment opportunities. The cash economy and the subsistence economy are intertwined, and subsistence is a full-time job for those fully engaged in it. However, it is necessary to supplement subsistence with cash from part-time wage labor or commercial fishing to defray the costs of subsistence activities (Appendix D). Despite differences in the types of subsistence and traditional cultural practices between the people of the Nushagak and Kvichak River watersheds and the people of the North Slope, studies from the North Slope region can provide some insights. A study of Alaska's North Slope Inupiat people found that there is an inverse relationship between active subsistence harvesting and wage labor time for the individual worker, but that cash from employment is often used for subsistence inputs (e.g., gasoline, boats, ammunition) (Kerkvliet and Nebesky 1997).

One of the mitigation measures that can address the impact of full-time employment on subsistence activities is the implementation of subsistence leave policies. For the development of the Red Dog Mine supplemental environmental impact statement, interviews were conducted that included questions asking Noatak and Kivalina residents about their employment history related to the Red Dog Mine and their employer's subsistence leave policies. Responses were mixed regarding whether or not interviewees were aware of a subsistence leave policy and whether or not the policy worked. Some of the companies did not have subsistence leave policies, so workers conducted subsistence activities during their weeks off or would take personal time. Where the companies did have policies for subsistence leave, an average of 46% of respondents were unsure whether or not the policy worked (USEPA 2009).

The creation of mining-related jobs for local residents and attendant increases in the region's cash economy are often mentioned as potential benefits of large-scale mining development. However, increases in personal income may not be the best measure of benefits in a subsistence-based culture and should be considered over the long-term, as oil, gas, or mineral resources are exhausted and future opportunities—including subsistence resources—are potentially damaged. These types of damages persist, even when resource extraction ceases (NRC 2003).

12.2.4 Social, Cultural, and Spiritual Impacts

The inability to harvest salmon from portions of these watersheds would result in some degree of cultural disruption, which 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."

On a cultural level, a significant loss of salmon would result in negative stress on a culture that is highly reliant on 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 are integrated with the natural world, especially salmon. Of particular importance is the sharing and passing along of traditional knowledge to future generations. This knowledge transfer occurs in several ways but one critical component is fish camp. According to Boraas and Knott (Appendix D):

> Families typically view fish camp as a good time when they can renew bonds of togetherness by engaging in the physical work of catching and processing salmon. Family members who don't live in the villages often schedule vacation time to return home to fish camp, not just for the salmon but for family. The importance of sharing in vigorous, meaningful work cannot be overestimated. It creates cross-generational bonds between children, their parents, aunts, uncles, and/or grandparents that today are rare in Western culture because there are so few instances in which meaningful, multi-generational work occurs.
>
> Some interviewees expressed fear of the future, that a traditional prophecy of "bad times" told by Elders might be coming true due to economic development resulting in cultural loss characterized as "anomie," the loss of meaningfulness, sense of belonging, and direction in life. Anomie increases cultural and individual risk for social ills such as depression and suicide, alcoholism and drug abuse, domestic violence, and aggressive behaviors. Healing practices can include those used for trauma and post-traumatic stress disorders, including traditional practices that reconnect the individual to society and the natural environment through meditative rituals. Culture camps and other methods of cultural revitalization can be both preventative and healing for children and adults of indigenous cultures.

Acculturation is a commonly used concept to describe the psychological and cultural impacts of rapid modernization and loss of tradition. Identity and involvement in cultural activities provide numerous benefits to Alaska Natives. Participation in subsistence activities and consumption of subsistence foods include cultural, traditional, and spiritual activities that involve the entire community. One of the greatest risks to the Alaska Native communities in the Nushagak and Kvichak River watersheds with respect to acculturation would arise from a major and persistent decline in the subsistence salmon fishery. For the people on the Nushagak River who consider themselves the "King Salmon" people, any impact on the Chinook salmon fishery would stress their community and the cultural traditions that bind them together.

Studies on disruption to Alaska Native cultures from resource extraction industries illustrate the potential social and cultural impacts of large-scale mining on a key subsistence resource in the Nushagak and Kvichak River watersheds. Land use by Alaska Natives on the North Slope has been mostly non-intensive, leaving few traces on the landscape outside the established villages. In contrast, oil development has altered the landscape in ways that will persist long after resource extraction activities have ceased. Testimony repeatedly cited “scars on the land” that result from industrial development, and indicated that these scars have altered both the physical and spiritual elements of the landscape and thus the very basis of Alaska Native cultures on the North Slope (NRC 2003).

Alterations to the North Slope physical environment have had aesthetic, cultural, and spiritual effects on human populations (NRC 2003). These alterations have resulted primarily from the construction of roads, pipelines, buildings, and power lines and from off-road travel. Hunters report that they do not hunt in the oil fields for aesthetic reasons. North Slope residents have reported that the imposition of a huge industrial complex on the Arctic landscape was offensive to the people and an affront to the spirit of the land.

North Slope residents report that there has been a vast increase in the time, effort, and funding necessary to respond politically and administratively to the ever-multiplying number of projects proposed in the region (NRC 2003). Local residents must attend industry-related meetings and hearings and review documents, because they believe that decisions will be made that can significantly affect both their daily lives and future generations. Additionally, North Slope residents stated that increasing anxiety about offshore and onshore development is widespread in North Slope communities. Hunters worry about contamination of the food they consume and know that their health will suffer if they are unable to eat as their ancestors did. They worry about not being able to provide for their families, or about the added risk and expense if essential and traditional foods are harder to find. Elders who are no longer able to provide for themselves worry about the challenges younger hunters face. Families worry about the safety of hunters who must travel farther and more often if game is not easily accessible (NRC 2003).

According to the National Research Council (2003), increased alcoholism, drug abuse, and child abuse have resulted from the stresses inherent in integrating traditional and new ways of life. Health effects also are apparent, as the incidence of diabetes has increased with higher consumption of non-subsistence foods (NRC 2003). The North Slope Borough bears the costs of these social stresses and provides services such as counseling, substance abuse treatment, public assistance, crisis lines, shelters, and other social service programs. It also supports search and rescue services and the police force that respond to domestic violence and other situations arising when communities are subjected to long-term and persistent stress. The borough supports biologists, planners, and other specialists who review and offer recommendations on lease sale, exploration, and development project documents that are produced each year, and bears the expense of traveling to Fairbanks, Anchorage, Juneau, Seattle, and Washington, DC, where agencies with permitting authority make decisions that affect their way of life (NRC 2003).

The goal of a more recent study on the effects of oil and gas development on subsistence harvesters on the North Slope (Braund and Associates 2009, Braund and Kruse 2009) was to enhance benefits and mitigate impacts of development. This study reported that, despite raising concerns about oil development as early as 1975, the Inupiat have, until recently, been successful in maintaining their subsistence lifestyle. Since 2003, North Slope active harvesters have been experiencing impacts of oil development at higher rates and report that their wellbeing has declined. This has led to social problems, including higher rates of drug and alcohol abuse and suicide.

A study that looked at the social, cultural, and psychological impacts of the Exxon Valdez oil spill determined that the psychosocial impacts of contamination were as significant as the physical impacts on the environment (Palinkas et al. 1993). Reported issues included declines in traditional social relations with family members, friends, neighbors, and coworkers; perceived increases in the amount of and problems associated with drinking, drug abuse, and domestic violence; a decline in perceived health status; an increase in the number of physician-verified medical conditions; and increased post-spill rates of generalized anxiety disorder, post-traumatic stress disorder, and depression (Palinkas et al. 1993).

Community-wide pre-occupation with the spill and cleanup affected traditional social relations and resulted in conflicts and divisiveness, arguments about environmental effects of the spill, issues of fault and responsibility, issues of whether or not to work on the cleanup, and related monetary and employment issues (Palinkas et al. 1993). There were pervasive fears and increased fundamental concerns about cultural survival for many residents in the affected Alaska Native villages.

Palinkas et al. (1993) documented the profound impact that exposure to the oil spill had on social relations, traditional subsistence activities, the prevalence of psychiatric disorders, community perceptions of alcohol and drug abuse and domestic violence, and the physical health of Alaskan Native and non-native residents of the affected communities. Although the specifics of the Exxon Valdez oil spill may be quite different, a large-scale or long-term failure of mine waste collection, treatment, or containment systems would produce a similar reduction of subsistence activities, and similar social and cultural effects could be expected.

12.2.5 Mitigation and Adaptation

It is not likely that any direct or indirect loss of subsistence use areas resulting from the mine footprints could be avoided. In the mine scenarios, the mine pit, waste rock piles, and TSFs would remain on the landscape in perpetuity and thus represent permanent habitat loss for salmon and other subsistence resources. Some measures could be put in place to prevent and respond to accidents and spills. Small spills and releases that are contained in a timely manner may not affect the salmon subsistence resource. However, large-scale releases, even with active remediation, would have long-term effects on the salmon subsistence resource and Alaska Native cultures. Because the Alaska Native cultures in this area have significant ties to specific land and water resources that have evolved over thousands of years, it would not be possible to replace the value of lost subsistence use areas elsewhere, or to relocate residents and their cultures, making compensatory mitigation infeasible (Appendix J).

The ability of Alaska Native cultures to adapt to losses of subsistence use areas or to the larger impacts of a mine failure or accident is unknown. Several studies have considered adaptation related to subsistence resources. Holen (2009) studied the adaptations related to the Nondalton subsistence fishery and identified two major socio-cultural factors that could potentially affect the long-term resilience of the fishery: children and young adults are not actively participating in subsistence salmon fishing as they have in the past, and because summer is often when seasonal employment is available, some residents miss the subsistence fishing season because of work obligations. These factors interrupt the inter-generational transfer of existing knowledge and wisdom and suggest that permanent cultural change can result from cultural disruption. On Alaska's North Slope, the issue has not been a question of whether Alaska Natives adapt to oil and gas development, but rather the consequences of that adaptation (NRC 2003). There are two potential problems: the loss—sometimes quickly—of traditional languages, patterns of behavior, economic activities, skills and capital improvements that are no longer relevant; and the use of human and financial capital and non-renewable resources by the new development (NRC 2003).

As the cash economy develops and Alaska Natives become involved in discussions about how changes associated with oil and gas development affect their cultures, they increasingly must use English as their primary language. They lose fluency in their native language and the traditional ecological knowledge embedded in that language. Many North Slope residents expressed concern about the loss of their traditional way of life, while at the same time enjoying the benefits of the cash economy (NRC 2003). However, over-adaptation can also occur, leaving communities less able to survive in their environments when extraction activities decline or stop. The significant tax revenues that oil and gas development have provided North Slope Borough residents are now declining, and the current standard of living for North Slope residents will be impossible to maintain unless significant external sources of local revenue are found. If borough revenues decline, residents may face lower standards of living, or be forced to find other sources of economic activity or migrate to different areas (NRC 2003).

Offshore exploration and development and the announcement of offshore sales have resulted in perceived risks to Inupiat culture that are widespread and intense. People of the North Slope have a centuries-old nutritional and cultural relationship with the bowhead whale, and most view offshore industrial activity as a threat to bowheads and thus their cultural survival.

12.3 Uncertainties

The preceding sections provide a qualitative overview of how wildlife and Alaska Natives may be affected by mining-associated changes in salmon resources. Because we mention but do not evaluate direct effects of mining on wildlife and Alaska Natives (Box 12-1), this assessment represents a conservative estimate of how these endpoints could be affected by mine development and routine operations. We focused on a limited suite of wildlife species (Section 5.3), but additional species also could be affected by changes in salmon resources. We also did not consider mining-related changes to all subsistence species.

In addition to these scope-related limitations, there are several uncertainties inherent in our consideration of fish-mediated effects on wildlife and Alaska Natives.

- The magnitude of salmon-mediated effects on wildlife, subsistence resources, and indigenous cultures is uncertain and cannot be quantified at this time. Ultimately, the magnitude of overall impacts will depend on many factors, including the location and temporal scale of effects, cultural resilience, the degree and consequences of cultural adaptation, and the availability of alternative subsistence resources.
- Interactions between salmon and other wildlife species are complex and reciprocal, and the assessment did not comprehensively evaluate all potential linkages between endpoints. Many of these linkages have not been well-documented or researched (e.g., potential relationships between MDN, riparian vegetation, and moose and caribou), but may be significant. Therefore, this assessment likely underestimates salmon-mediated risks to wildlife.
- The magnitude of effects on Alaska Native cultures resulting from any mining-associated changes in salmon resources is unknown, but other studies related to resource extraction industries (North Slope, Red Dog Mine) or environmental contamination (Exxon Valdez) in Alaska confirm that there certainly would be changes in human health and Alaska Native cultures.
- The cumulative effects of mining and climate change represent a significant uncertainty in the region (Section 3.8, Box 14-2). Residents of the Kvichak River watershed have observed that social and cultural changes are occurring in an environment where they are also seeing rapid climate changes (Holen 2009). These changes, which include climate variability and unpredictable weather, make it difficult to plan for subsistence activities (Appendix D). On Alaska's North Slope, climate change and oil and gas development together result in greater cumulative effects on the environment and Inupiat cultural traditions (Braund and Associates 2009). The cumulative effects of climate change and potential effects on subsistence resources from large-scale mining are unknown.

Despite these uncertainties, the inability to mitigate or replace subsistence resources or cultural values lost to effects of large-scale mining is certain because of the significant and long-standing ties that Alaska Native cultures have to specific land and water resources in these watersheds.



CHAPTER 13. CUMULATIVE RISKS OF MULTIPLE MINES

Thus far, this assessment has focused on the potential effects of a single mine, described by a range of mine scenarios. Although the Pebble deposit represents the most imminent and likely site of mine development in the Nushagak 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 13-1), and active exploration of deposits is occurring in a number of claim blocks (Figure 13-1). If the infrastructure for one mine is built, it would likely facilitate the development of additional mines. Thus, the potential exists in these watersheds for the development of a mining district that could include a number of mines, their associated infrastructure, and resulting induced development. In this chapter, we briefly consider potential cumulative effects of the establishment of large-scale mining in the Nushagak and Kvichak River watersheds on Pacific salmon. In addition to addressing the potential impacts of multiple mines, we briefly consider induced development and potential increases in the accessibility of the watersheds' currently roadless areas.

13.1 Cumulative and Induced Impacts

13.1.1 Definition

National Environmental Policy Act regulations define cumulative impacts as "the impact on the environment [that] results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time." Assessing the cumulative impacts of multiple mines requires considering the impacts of their combined footprints, as well as the cumulative risks of leaks, spills, and other accidents and failures associated with each individual mine.

Induced effects contribute to the cumulative effects of an action, and are those effects that are "caused by the action and are later in time or farther removed in distance, but are still reasonably foreseeable" (43 CFR 1508.8(b)).

Figure 13-2 illustrates how cumulative and induced effects could follow the initiation of large-scale mining in the Nushagak and Kvichak River watersheds. The original mine—with its associated transportation corridor, port, power generation facilities, and other infrastructure—likely would initiate the accumulation of impacts across the watersheds. Mineralized areas in the region (Figure 13-1) are currently without development infrastructure (e.g., roads, utilities, and airports), which creates an expensive barrier to development. Thus, it is reasonably foreseeable that infrastructure development for an initial mine could make mining cost-effective for other, smaller mineral deposits, facilitating further accumulation of impacts. In addition, the initial and subsequent mines would increase accessibility of the region, causing other induced development and associated impacts.

As environmental effects on freshwater habitats accumulate, the magnitude of total impact on the region's fisheries would increase. Increased spatial dispersion of effects, both within and across watersheds, means that individual effects may go unnoticed but still cumulatively affect the greater Bristol Bay salmon fishery. Cumulative effects associated with multiple mines would potentially reduce biodiversity of the overall salmon population (Section 5.2.4) and its resilience to natural and anthropogenic disturbances, thereby exacerbating total effects on salmon.

13.1.2 Vulnerability of Salmonids to Cumulative Impacts

Throughout the range of Pacific salmon, most ecosystems outside of the Bristol Bay watershed face the cumulative effects of multiple land and water uses within and across watersheds, resulting in a variety of stressors that occur in combination. Anadromous and 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 fish in a river system. The effect of each stressor accumulates regardless of whether factors occur at the same time, or even in temporal proximity. Because 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 result of these cumulative effects has been the reduction and even extinction of many salmonid populations.

Historical salmon losses have resulted from the cumulative impacts of many land use activities over a time span of 150 years (NRC 1996). Salmon depend on adequate supplies of clean, cool water throughout the freshwater portions of their lifecycles. In addition, well-aerated streambed gravels are essential for spawning. These instream conditions depend on the overall health of the entire watershed (NRC 1996). Human development in the watershed can adversely affect these conditions by increasing sedimentation, raising water temperature, degrading water quality, changing water flow, and reducing water depths (NRC 1996).

In the Pacific Northwest, the four principle factors responsible for the degradation of salmon stocks have been referred to colloquially as the “four H’s”: habitat degradation and loss, hydroelectric dams and other impoundments, harvest practices, and hatchery propagation (Ruckelshaus et al. 2002). Of these factors, habitat degradation and loss is the most likely to affect salmon stocks in the Nushagak and Kvichak River watersheds. In the Pacific Northwest, habitat degradation and loss related to human land use have obviously been a major factor in salmon declines by reducing population productivity, adult densities, and early-life-stage production over large geographic areas (Ruckelshaus et al. 2002).

Table 13-1. Mining prospects (in addition to the Pebble deposit) with more than minimal recent exploration in the Nushagak and Kvichak River watersheds. See Figure 13-1 for prospect locations.

Prospect	Resource	References
AUDN	Porphyry copper	Millrock Resources 2011 Millrock Alaska 2012a
Big Chunk North	Porphyry copper	U5 Resources 2010
Big Chunk South	Porphyry copper	AHEA 2012 Big Chunk Corp. 2012 Liberty Star 2012a
Fog Lake	Gold, copper	Alix Resources 2008 USGS 2008
Groundhog	Porphyry copper	Kennecott Exploration Co. 2011 Szumigala et al. 2011
Humble	Porphyry copper	Millrock Resources 2011 Millrock Alaska 2012b
Iliamna	Porphyry copper	Bristol Exploration Co., Inc. 2011
Kamishak	Porphyry copper	AERI 2008
Kaskanak	Porphyry copper	Full Metal Minerals 2008, 2012
Kisa	Gold	Golden Lynx 2009
Northern Bonanza	Gold	Northern Bonanza Trust 2011
Shotgun	Gold	TNR Gold Corp. 2011, 2012 ADNR 2012a
Sleitat	Tin, tungsten	Thor Gold Alaska, Inc. 2011
Pebble South/PEB (38/308 Zones/BOO)	Porphyry copper	Full Metal Minerals 2008, 2012 PLP 2011 Szumigala et al. 2011
Stuy	Porphyry copper	Stuy Mines, LLC 2010

Figure 13-1. Claim blocks with more than minimal recent exploration in the Nushagak and Kvichak River watersheds.

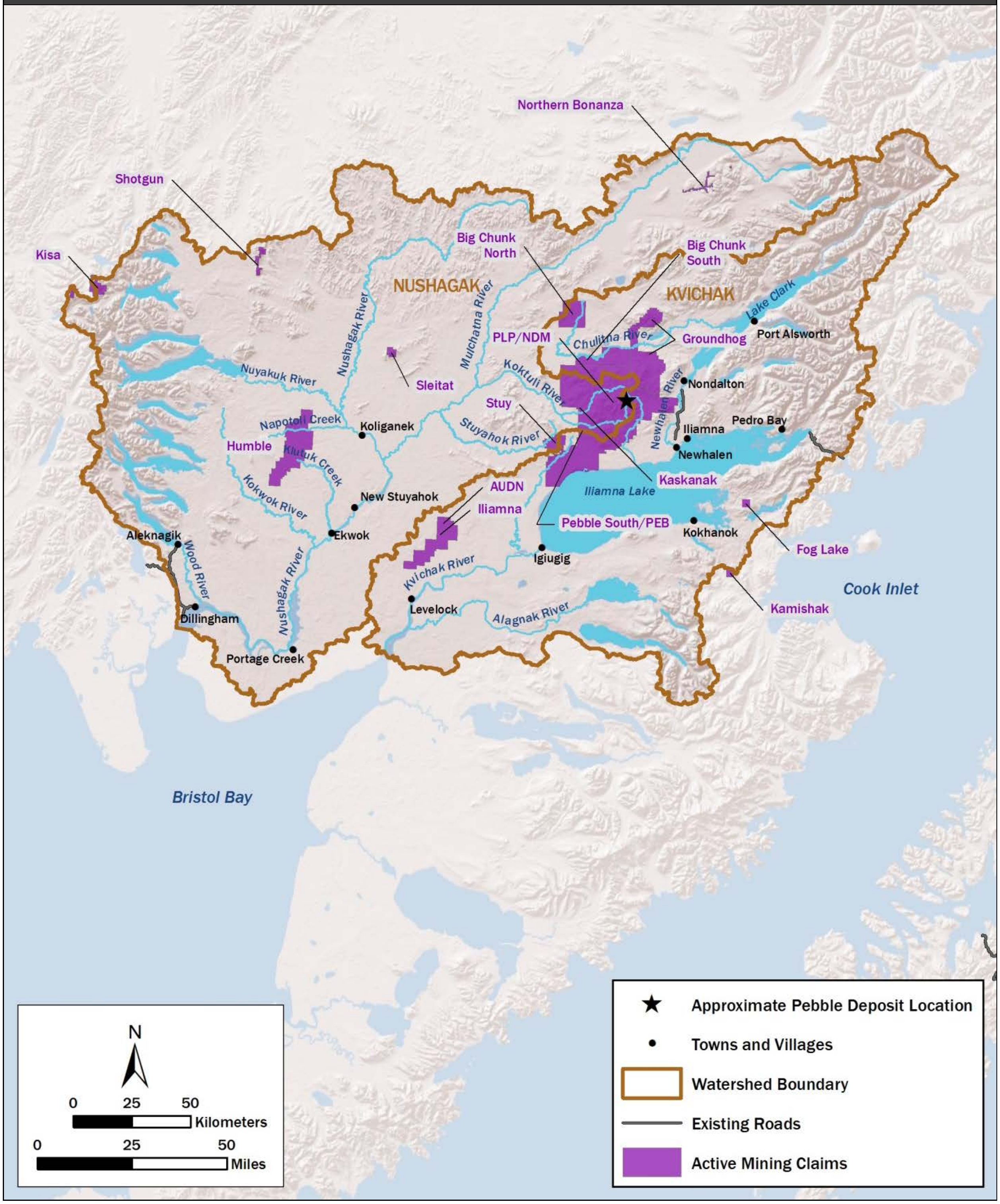
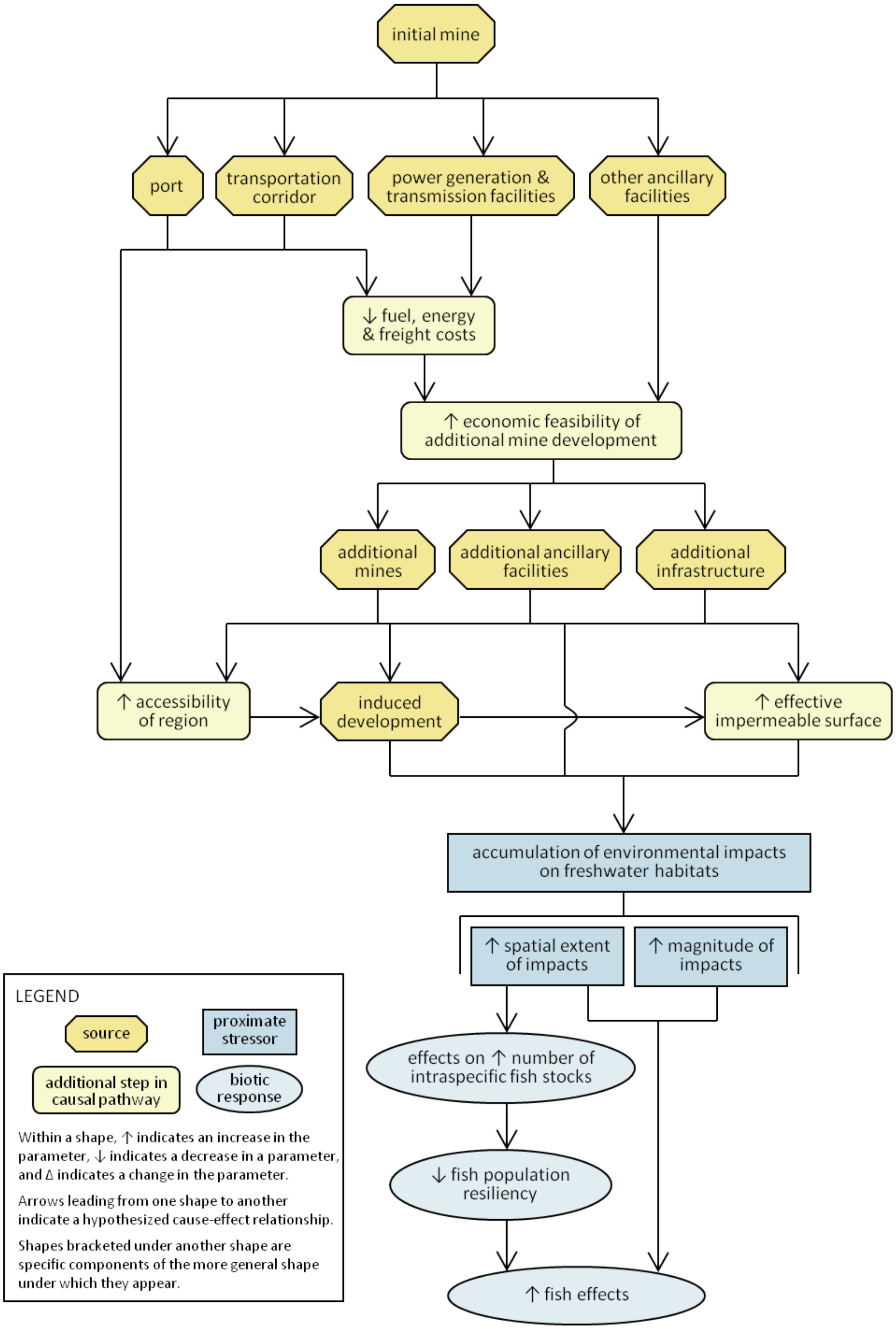


Figure 13-2. Conceptual model illustrating potential cumulative effects of multiple mines.



13.1.3 Nature and Extent of Past, Present, and Future Impacts

In cumulative impact analyses, the U.S. Environmental Protection Agency (USEPA) generally evaluates any past, present, and reasonably foreseeable future actions that are spatially and temporally linked and thus can act in combination on the resource(s) of interest. In the Bristol Bay watershed, the contribution of past or present actions to degradation of salmon habitat and populations is minimal. To date, the Nushagak and Kvichak River watersheds have experienced minimal cumulative stresses associated with human activity, and their ecosystems are relatively undisturbed by significant human development (Section 3.7). Large-scale, human-caused modification of the landscape—a factor contributing to the extinction risk for many native salmon populations in the Pacific Northwest (Nehlsen et al. 1991)—is absent from these watersheds, and development consists of only a small number of towns, villages, and roads. There are no hydroelectric dams and reservoirs and no salmon hatcheries. In fact, harvesting is the only one of the four H's responsible for devastating salmon stocks of the Pacific Northwest that is applicable to the Nushagak and Kvichak River watersheds. There have been periods of poor harvesting practices and overfishing in Bristol Bay in the past. However, when Alaska achieved statehood, the Alaska Department of Fish and Game (ADF&G) assumed management of the fishery and established the primary objective of restoring the runs to their former abundance (Box 5-4, Appendix A). The ADF&G management strategy, based on maximum sustainable yield, is considered a success in maintaining sustainable salmon harvests (Hilborn et al. 2003, Hilborn 2006). Indeed, since the late 1970s, sockeye salmon catch, spawning stock, and total returns have been at record levels. Although there has been some concern that harvest of returning salmon has reduced ecosystem productivity in this region, Schindler et al. (2005) found that paleoecological analysis of returns to Lake Nerka in the Wood River system did not suggest decreased salmon production due to commercial fishing.

Reasonably foreseeable future development can be predicted based on project approvals, planning documents, and data on local trends. The Bristol Bay Area Plan for State Lands (BBAP) (ADNR 2005) provides information on reasonably foreseeable mining in the Nushagak and Kvichak River watersheds. Although the Pebble deposit represents the most imminent and likely site of mine development in the watersheds, the development of several mines of varying sizes is plausible in this region. Several known mineral deposits with potentially significant resources are located in the two watersheds, and active exploration of deposits is occurring in a number of claim blocks (Figure 13-1, Table 13-1). Reasonably foreseeable non-mining development is discussed in Section 13.3.

13.2 Cumulative Impacts from Multiple Mines

Construction of mining and transportation infrastructure at and for the Pebble deposit would substantially reduce development costs for surrounding prospects and could facilitate creation of a mining district. Based on planning documents and current patterns of mineral exploration in the Nushagak and Kvichak River watersheds, it is possible to identify a scenario for potential mine development in the region over the next 50 to 100 years. Although this scenario is plausible given available information, it should be kept in mind that it is impossible to predict with certainty what mining activities will occur in the region in the future, the order in which mines will be developed, or the specific impacts of those mines.

The BBAP assigns land use designations to discrete areas of state-owned or selected lands called management units. These designations represent the uses and resources for which the Alaska Department of Natural Resources will manage the units. In the Nushagak and Kvichak River watersheds, the BBAP assigns the land use designation “Mi” (Minerals) to seven management units (Shotgun, Sleitat, Kemuk, Fog Lake, and units 06-23, 06-24, and 10-02 in the Pebble deposit area), which total more than 1,300 km^2 (ADNR 2005). The Mi designation applies to areas “associated with significant resources, either measured or inferred, that may experience minerals exploration or development during the [BBAP’s] planning period” of 20 years. The BBAP also allows for mining on units with other designations (e.g., General Use, Public Recreation and Tourism-Dispersed) that far exceed the area of those designated Mi, although the BBAP does not describe them as having the same known potential for mining.

Since 2008, there has been exploration in all of the BBAP Mi-designated management units, as well as in claim blocks with several other designations. Table 13-1 lists mine prospects experiencing more than minimal recent exploration activity. These target areas could be future mine sites, if exploration identifies marketable quantities of metals. Other mineral claim blocks exist in the watersheds, but at the time of this writing they had experienced limited exploration in recent years (ADNR 2012b, 2012c, 2012d).

Any potential mine site would presumably include a mine pit and an adjacent waste rock disposal area (Chapter 6). For analysis purposes, we assume the size of other ore bodies in the area would be more typical of worldwide porphyry copper deposits than the Pebble deposit (Table 4-2). We used the Pebble 0.25 scenario, which is comparable to a median-size porphyry copper mine, to characterize the footprints of the major mine components (mine pit, waste rock piles, and tailings storage facilities [TSFs]) for additional mines. The Pebble 0.25 scenario represents 250 million tons (230 million metric tons) of ore, resulting in a mine pit and waste rock disposal area of approximately 1.5 and 2.3 km^2, respectively (Table 6-2). Mines affiliated with or close to an existing mine (e.g., a mine at the Pebble deposit) may be able to use the TSFs, mill, and other infrastructure constructed for that mine. Mines not affiliated with or more distant from a previously developed mine would require one or more TSFs (an additional 6.8 km^2 in the Pebble 0.25 scenario) (Table 6-2), as well as a mill and other operational infrastructure as described in Box 6-1. Thus, for potential mines distant from Pebble, we calculated the footprints of the major mine components both with and without TSFs. Any additional mines would also require construction of transportation infrastructure, including access roads, pipelines, and possibly port facilities.

To examine the potential scope of cumulative impacts from large-scale mining, we consider development of additional mines at six potential sites where there was notable activity and/or investment in drilling or other exploration in 2011 to 2012: Pebble South/PEB (PLP/NDM claim block), Big Chunk South, Big Chunk North, Groundhog, AUDN/Iliamna, and Humble prospects (Figure 13-1).

This list does not include four other prospects designated as Mi in the BBAP: Shotgun, Sleitat, Kemuk, and Fog Lake. Kemuk is an older name for the Humble prospect; the other three prospects are not porphyry copper deposits. Because exploration of the six selected prospects began approximately 15 to 25 years later than exploration at the Pebble deposit, proposals to develop mines at these sites could be 20 years or more in the future (Millrock Resources 2011, ADNR 2012e, 2012f, and 2012g, Liberty Star 2012a). We describe the waters, fishes, and subsistence resources that could be affected by mines at these locations (Tables 13-2 through 13-7). The sources of information for these tables are the Alaska Anadromous Waters Catalog (Johnson and Blanche 2012), the Alaska Freshwater Fish Inventory (ADF&G 2012), and a series of technical papers from ADF&G on subsistence harvest and use by villages in the Nushagak and Kvichak River watersheds (Fall et al. 1986, Schichnes and Chythlook 1991, Fall et al. 2006, Krieg et al. 2009). We also estimate the stream lengths and wetland areas that could be eliminated by the footprint of the major mine components at each site. Box 13-1 describes the methodology for estimating these impacts; results across the six potential sites are summarized in Table 13-8. It is important to note that we did not estimate the size of the groundwater drawdown zones around dewatered pits at the six additional mines as we did at the Pebble site, so our estimates of habitat loss are conservative. Inclusion of the drawdown zone in the Pebble 0.25 scenario increases stream and wetland losses by roughly 50%. Similar increases in habitat loss estimates, subject to variations in the local geology, would be expected at each of the other mine sites.

13.2.1 Pebble South/PEB

13.2.1.1 Description

The Pebble South/PEB prospect, which is part of the PLP/NDM claim block, includes the 38 and 308 Zone prospects, approximately 15 km southwest of the Pebble deposit (Ghaffari et al. 2011). The BOO prospect is 4 km south of 308 Zone on claims held by Full Metal Minerals (USA), Inc. (Ghaffari et al. 2011, Full Metal Minerals 2012). Full Metal entered into an option agreement with PLP/NDM; if completed, the option will result in at least 60% PLP/NDM interest in the claims (Full Metal Minerals 2012).

Due to its proximity to the Pebble deposit, we assume that any future mines at Pebble South/PEB would use the TSFs, mill, and other operational infrastructure initially built for mining at the Pebble deposit. Thus, we anticipate that the primary additional development associated with this prospect would be a mine pit, waste rock areas, and a transportation corridor to existing operational infrastructure.

BOX 13-1. METHODS FOR ESTIMATING IMPACTS OF OTHER MINES

To estimate the extent of aquatic habitat that each mine would eliminate, we overlaid typical footprints of the major mine components (mine pit, waste rock piles, and tailings storage facilities) onto the stream, water body, and, where available, wetland densities for each entire claim block. In this way the analysis is less affected by uncertainty about the precise location of a potential mine within the claim block. Since the Pebble South/PEB prospects are associated with the PLP/NDM claims, we used the aquatic area densities for the PLP/NDM claims to assess that potential mine.

We derived the boundaries of the prospects from the Alaska Department of Natural Resources' State Mining Claims dataset (ADNR 2012h). We then determined stream and water body density using the National Hydrography Dataset (NHD) for Alaska (USGS 2012). For wetland density, we used the National Wetlands Inventory (NWI) coverage for the Groundhog, AUDN/Iliamna, and Pebble South/PEB (part of the PLP/NDM claim block) prospects (USFWS 2012). The NWI covered 95, 69, and 58% of these prospects, respectively. For these three areas, we compared NHD water body density to NWI wetland density to ascertain the efficacy of using the former as a surrogate for wetland density in areas where NWI coverage was lacking. This analysis revealed that the NHD water body dataset severely underestimates wetland density in areas of overlap: NWI wetland density was roughly 10 to 14 times the NHD water body density. Thus, for the three prospects with no NWI coverage (Big Chunk South, Big Chunk North, and Humble), we calculate a range of wetland impacts, using NHD water body density as a lower bound and roughly 14 times that density as an upper bound. We also provide a range of wetland impacts for the three claim blocks with NWI coverage, because the NHD water body density in the NWI-covered area of all three prospects was lower than for the claim block as a whole. For those three claim blocks (Pebble South/PEB, Groundhog, and AUDN/Iliamna), the higher estimate applies the wetland-to-water body differential from the area of NWI/NHD overlap to the full claim block's higher water body density.

For Pebble South/PEB, we used a direct impact area that represents only the typical mine pit and waste rock disposal area, based on our assumption that any mine at that site would use the mill, tailings storage facility, and other facilities at an initial mine at the Pebble deposit (Section 13.2.1). We applied a similar assumption as a lower bound for Big Chunk South, Big Chunk North, and Groundhog: for all three, the upper bound represents a stand-alone mine, with no shared facilities (Sections 13.2.2 through 13.2.4). We assume no sharing of mine facilities for AUDN/Iliamna or Humble, based on their more remote locations (Sections 13.2.5 and 13.2.6).

13.2.1.2 Potentially Affected Waters, Fishes, and Subsistence Uses

Table 13-2 summarizes information on the waters, fishes, and subsistence uses potentially affected by a mine at the Pebble South/PEB prospect. The 38 and 308 Zones occur near the south edge of the South Fork Koktuli River watershed, within the Mulchatna River watershed of the Nushagak River watershed. The upper reaches of some streams on Sharp Mountain likely are too steep to provide fish habitat and there have been few fish surveys in this area to date, even in the lower reaches of those streams. Drilling on the BOO prospect has been just south of the divide between the Nushagak and Kvichak River watersheds, in the uppermost portion of the Lower Talarik Creek watershed, which flows to Iliamna Lake. BOO is located approximately 2 km southwest of the fish-bearing stream that drains the south side of Sharp Mountain. No fish survey data are available for the immediate area of the BOO prospect, and the streams may be spatially intermittent.

Connecting to infrastructure at an existing mine at the Pebble deposit likely would involve following the South Fork Koktuli River upstream. This route would presumably involve crossing the river in addition to a number of tributaries, water bodies, and wetlands.

Waters of both the Nushagak and Kvichak River watersheds could be affected at this site, although no information is available on fish resources in the tributaries of the Kvichak River that would be affected by mining of the BOO prospect. The tributaries of the Nushagak River that would be affected by mining of the 38 and 308 Zone prospects are known to contain four Pacific salmon species and Dolly Varden. People from seven Alaska Native villages hunt for bear, moose, caribou, other mammals, and birds in these areas, but no subsistence fishing has been reported (Table 13-2). Based on the average stream density in the area of the prospect, the footprint of the major mine components would eliminate 4.1 km of streams and between 0.71 and 1.2 km^2 of wetlands (Table 13-8).

13.2.2 Big Chunk South

13.2.2.1 Description

The Big Chunk South prospect may be of the same geologic origin as the Pebble deposit. The claim block abuts the north edge of the PLP/NDM claim block, approximately 20 km north of the potential Pebble mill site (AHEA 2012, Big Chunk Corp. 2012) and approximately 24 km northwest of the village of Nondalton.

Based on its proximity to the Pebble deposit, a future mine at Big Chunk South may use the TSFs, mill, and other facilities built for potential mining at the Pebble deposit, under a joint venture or other agreement with PLP/NDM. In late 2012, Big Chunk partner Liberty Star settled debt and terminated joint-venture negotiations with NDM (Liberty Star 2012b). For the purposes of this assessment, we consider Big Chunk South under two scenarios: one in which it shares some facilities built for a mine at the Pebble deposit and the other in which it operates as a fully separate, stand-alone mine, with no shared facilities other than the transportation corridor connecting the Pebble deposit site to Cook Inlet (Table 13-8). In this chapter, we refer to this transportation corridor as the assessment corridor. A mine at Big Chunk South presumably would connect to the roads and pipelines of the assessment corridor somewhere near its western terminus, currently estimated to be approximately 14 km south of the Big Chunk South claim block.

Table 13-2. Waters, fishes, and subsistence uses potentially affected by a mine at the Pebble South/PEB prospect.

Location	Affected Waters								Fish Species[b]																			Subsistence Uses[c]									
	Watersheds and Named Subwatersheds																												Target Species/Group								
	Nushagak River	Mulchatna River	Koktuli River	South Fork Koktuli River	Kvichak River	Iliamna Lake	Lower Talarik Creek	Waters[a]	Unknown Lamprey	Longnose Sucker	Northern Pike	Pond Smelt	Round Whitefish	Unknown Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Chum Salmon	Pink Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Arctic Grayling	Burbot	Ninespine Stickleback	Threespine Stickleback	Slimy Sculpin	Village	Salmon	Other Salmonids	Other Fishes	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
38/308 Zones and BOO Prospect	x	x	x	x				14 unnamed streams							A-Sp, J	J	J	A				A, J	J		A, J		A, J	Dillingham						x			
								2 unnamed lakes and several ponds																				Ekwok						x			
																												Igiugig					x		x		
								Unknown extent of wetlands																				Iliamna					x	x			
								4 unnamed first-order streams																				Newhalen				x	x	x	x		
					x	x	x	Numerous ponds	Unknown																			Nondalton				x	x	x	x		
								Unknown extent of wetlands																				Port Alsworth						x			x
Transportation Corridor to Pebble Site[d]								South Fork Koktuli River																				Dillingham						x			
																												Ekwok						x			
																												Igiugig							x		
		x	x	x				≥4 unnamed tributaries			A, J			x	A-Sp, J	A-Sp, J	A-Sp, J	A-Sp, J		x	A, J	A, J	A, J	x	A, J	A	A, J	Iliamna					x	x			
																												Newhalen					x	x	x		x
								Unknown extent of wetlands																				Nondalton					x	x	x		x
																												Port Alsworth						x	x	x	
Downstream of Mine(s) and Transportation Corridor	x	x	x					South Fork Koktuli River			A, J		A, J						x									Igiugig							x		
								Lower Talarik Creek																				Iliamna		x	x					x	x
																												Kokhanok		x				x			
					x	x	x		A-Sp	A-Sp	A-Sp	A-Sp	A, J		A-Sp, J	J	A-Sp, J	x	x	A, J	A-Sp, J	x	A-Sp, J		A, J	A-Sp, J	A, J	Newhalen	x	x	x	x				x	
																												New Stuyahok		x				x			
																												Nondalton								x	

Notes:

[a] "Tributary" indicates that the channel flows into the stream listed above it; "stream" indicates that the receiving water is offsite as identified in the columns to the left. For waters downstream of the mine and transportation corridor, fishes and subsistence uses are noted only where different from the waters' prior listing.

[b] A = adult; Sp = spawning; J = juvenile; x = unknown life stage. "Unknown" indicates apparent lack of surveys.

[c] Subsistence uses are not separated by watershed. Uses are noted only for areas of direct impacts (i.e., mine and/or transportation corridor), if they occur there; otherwise, they are noted for areas downstream. "Downstream" applies only to the drainage immediately downstream (e.g., Lower Talarik Creek, but not Iliamna Lake or the Kvichak River). Data for Dillingham and Ekwok are less detailed and more dated than for other villages, so use patterns may have changed.

[d] Assumes existing mine infrastructure at the Pebble deposit; hypothetical routing of the new corridor minimizes distance, topographic gradients, and stream crossings and assumes water body crossings would be avoided.

Sources: Siedelman et al. 1973, Russell 1974, Russell 1975, Fall et al. 1986, Schichnes and Chythlook 1991, Fall et al. 2006, Krieg et al. 2009, ADF&G 2012, Johnson and Blanche 2012.

Table 13-3. Waters, fishes, and subsistence uses potentially affected by a mine at the Big Chunk South prospect.

	Affected Waters											Fish Species[b]																			Subsistence Uses[c]									
	Watersheds and Named Subwatersheds																															Target Species/Group								
Location	Nushagak River	Mulchatna River	Koktuli River	North Fork Koktuli River	Kvichak River	Iliamna Lake	Newhalen River	Lake Clark	Chulitna River	Upper Talarik Creek	Waters[a]	Northern Pike	Longnose Sucker	Humpback Whitefish	Least Cisco	Pygmy Whitefish	Round Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Chum Salmon	Pink Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Arctic Grayling	Burbot	Ninespine Stickleback	Threespine Stickleback	Slimy Sculpin	Village	Salmon	Other Salmonids	Other Fishes	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
Claim Block					x	x	x	x	x		>66 unnamed streams		J			J									A, J	A, J		A, J		A, J	Dillingham						x			
											Nikabuna Lakes																				Ekwok						x			
											130 other lakes and ponds																				Newhalen					x	x			
																															Nondalton			x	x	x	x	x	x	
											Extensive wetlands																				Port Alsworth						x			
Transportation Corridor to Pebble Site[d]	x	x	x	x							North Fork Koktuli River							A-Sp, J	J	A					A, J			A		A, J	Dillingham						x			
											>7 unnamed tributaries																				Ekwok						x			
																															Iliamna					x				
											Unknown extent of wetlands																				Newhalen				x	x	x	x	x	
					x	x				x	Upper Talarik Creek							A-Sp, J	J	x				J	A, J	J				A, J	Nondalton				x	x	x	x		
											>3 unnamed tributaries																				Port Alsworth						x	x		x
Downstream of Mine(s) and Transportation Corridor					x	x	x	x	x		Chulitna River	A, J	A	A	A, J					x			x				A, J				Igiugig							x		
																															Iliamna							x	x	x
	x	x	x	x							North Fork Koktuli River	A, J			A, J		J		A-Sp	A-Sp, J	A-Sp					A, J			A, J		Newhalen	x		x					x	
																															New Stuyahok					x	x			
					x	x				x	Upper Talarik Creek								A-Sp, J	A-Sp, J	A-Sp, J	x	x	A		A		A, J	A		Nondalton			x						x
																															Port Alsworth					x				x

Notes:

[a] "Tributary" indicates that the channel flows into the stream listed above it; "stream" indicates that the receiving water is off-site, as identified in the columns to the left. For waters downstream of the mine and transportation corridor, fishes and subsistence uses are noted only where different from the waters' prior listing.

[b] A = adult; Sp = spawning; J = juvenile; x = unknown life stage.

[c] Subsistence uses are not separated by watershed. Uses are noted only for areas of direct impacts (i.e., mine and/or transportation corridor), if they occur there; otherwise, they are noted for areas downstream. "Downstream" applies only to the drainage immediately downstream (e.g., the Chulitna River, but not Lake Clark or the Newhalen River). Data for Dillingham and Ekwok are less detailed and more dated than for other villages, so use patterns may have changed.

[d] Assumes existing mine infrastructure at the Pebble deposit; hypothetical routing minimizes distance, topographic gradients, and stream crossings and assumes water body crossings would be avoided.

Sources: Fall et al. 1986, Schichnes and Chythlook 1991, Stickman et al. 2003, Fall et al. 2006, Woody and Young 2006, Krieg et al. 2009, ADF&G 2012, Johnson and Blanche 2012.

Table 13-4. Waters, fishes, and subsistence uses potentially affected by a mine at the Big Chunk North prospect.

Affected Waters and Fish Species[b]

Location	Nushagak River	Mulchatna River	Keefer Creek	Koktuli River	Swan River	North Fork Swan River	Kvichak River	Iliamna Lake	Newhalen River	Lake Clark	Chulitna River	Waters[a]	Arctic-Alaskan Brook Lamprey	Northern Pike	Longnose Sucker	Humpback Whitefish	Least Cisco	Pygmy Whitefish	Round Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Chum Salmon	Arctic Char	Dolly Varden	Arctic Grayling	Burbot	Ninespine Stickleback	Slimy Sculpin
Claim Block							x	x	x	x	x	Chulitna River ≥70 tributaries ≥100 lakes and ponds Unknown extent of wetlands	Unknown																
Claim Block	x	x	x									Keefer Creek headwaters ≥7 tributaries ≥35 lakes and ponds Unknown extent of wetlands	Unknown																
Claim Block	x	x		x	x	x						Unnamed stream	Unknown																
Transportation Corridor to Big Chunk South[d]							x	x	x	x	x	Chulitna River ≥4 tributaries Unknown extent of wetlands	Unknown																
Downstream of Mine(s) and Transportation Corridor							x	x	x	x	x	Chulitna River		A, J	A, J	A	A, J	J	J			x		x	A, J	A, J	A, J	A, J	A, J
Downstream of Mine(s) and Transportation Corridor	x	x	x									Keefer Creek								A-Sp	A-Sp	x	A-Sp	x					
Downstream of Mine(s) and Transportation Corridor	x	x		x	x	x						North Fork Swan River	A, J							A-Sp, J	J	A-Sp			J	J			A, J

Subsistence Uses[c]

Location	Village	Salmon	Other Salmonids	Other Fishes	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
Claim Block	Dillingham						x			
Claim Block	Ekwok						x			
Claim Block	Nondalton				x	x	x			
Claim Block	Port Alsworth						x	x		
Transportation Corridor to Big Chunk South[d]	Dillingham						x			
Transportation Corridor to Big Chunk South[d]	Ekwok						x			
Transportation Corridor to Big Chunk South[d]	Nondalton				x	x	x	x		
Transportation Corridor to Big Chunk South[d]	Port Alsworth						x	x		
Downstream of Mine(s) and Transportation Corridor	Iliamna							x	x	
Downstream of Mine(s) and Transportation Corridor	Newhalen			x		x	x	x		
Downstream of Mine(s) and Transportation Corridor	New Stuyahok							x	x	
Downstream of Mine(s) and Transportation Corridor	Nondalton			x				x	x	x
Downstream of Mine(s) and Transportation Corridor	Port Alsworth									x

Notes:

[a] "Tributary" indicates that the channel flows into the stream listed above it; "stream" indicates that the receiving water is off-site, as identified in the columns to the left. For waters downstream of the mine and transportation corridor, fishes and subsistence uses are noted only where different from the waters' prior listing.

[b] A = adult; Sp = spawning; J = juvenile; x = unknown life stage. "Unknown" indicates an apparent lack of surveys.

[c] Subsistence uses not separated by watershed. Uses noted only for areas of direct impacts (i.e., mine and/or transportation corridor), if they occur there; otherwise, noted for areas downstream. "Downstream" applies only to the drainage immediately downstream (e.g., North Fork Swan River, but not the Swan or Koktuli Rivers). Data for Dillingham and Ekwok are less detailed and more dated than for other villages, so use patterns may have changed (see Section 13.3.6.2).

[d] Waters, fishes, and subsistence uses limited to those upstream of the Big Chunk South claim block. See Table 13-3 for resources potentially affected by connecting a transportation corridor through and beyond that block. Hypothetical routing minimizes distance, topographic gradients, and stream crossings and assumes water body crossings would be avoided.

Sources: Fall et al. 1986, Schichnes and Chythlook 1991, Stickman et al. 2003, Fall et al. 2006, Woody and Young 2006, Krieg et al. 2009, ADF&G 2012, Johnson and Blanche 2012.

Table 13-5. Waters, fishes, and subsistence uses potentially affected by a mine at the Groundhog prospect.

Location	Affected Waters																															Subsistence Uses[c]									
	Watersheds and Named Subwatersheds																Fish Species[b]																Target Species/Group								
	Nushagak River	Mulchatna River	Koktuli River	North Fork Koktuli River	Kvichak River	Iliamna Lake	Upper Talarik Creek	Newhalen River	Lake Clark	Chulitna River	Rock Creek	Groundhog Creek	Long Lake	Koksetna River	Black Creek	Waters[a]	Northern Pike	Longnose Sucker	Humpback Whitefish	Least Cisco	Round Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Arctic Grayling	Burbot	Ninespine Stickleback	Slimy Sculpin	Village	Salmon	Other Salmonids	Other Fishes	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
																Chulitna River																									
																≥72 unnamed tributaries																Dillingham						x			
					x	x		x	x	x						Joe Nort Lake															A, J										
																58 unnamed lakes and ponds																									
																Extensive wetlands																									
																Rock Creek																									
					x	x		x	x	x	x					≥39 unnamed tributaries												J	A, J		A, J	Ekwok						x			
																9 unnamed lakes and ponds																									
																Extensive wetlands																									
																Groundhog Creek																									
					x	x		x	x	x	x	x				≥20 unnamed tributaries								Unknown								Kokhanok						x			
																30 unnamed ponds and lakes																									
																Unknown wetland extent																									
Claim Block					x	x		x	x	x			x			≥5 unnamed tributaries								Unknown																	
																Unknown wetland extent																Newhalen				x	x	x	x		
					x	x		x	x	x				x		Unknown wetland extent								Unknown																	
																Black Creek																									
					x	x		x	x	x				x	x	≥7 unnamed tributaries								Unknown																	
																one unnamed lake																									
																Unknown wetland extent																Nondalton				x	x	x	x	x	
					x	x		x	x	x						Unknown wetland extent								Unknown																	
																≥29 unnamed tributaries																									
					x	x	x	x	x	x						42 unnamed ponds and lakes						A-Sp, J					A, J				A, J										
																Unknown wetland extent																									
																≥13 unnamed tributaries																Port Alsworth						x	x		x
	x	x	x	x												7 unnamed ponds							J				A, J														
																Unknown wetland extent																									
																																Dillingham						x			
																≥1 unnamed tributary																Ekwok						x			
Transportation Corridor to Pebble Site[d]					x	x	x																									Kokhanok						x			
																																Newhalen				x	x	x	x		
																Unknown extent of wetlands																Nondalton				x	x	x	x	x	
																																Port Alsworth						x	x		x

Table 13-5. Waters, fishes, and subsistence uses potentially affected by a mine at the Groundhog prospect.

Location	Affected Waters																																Subsistence Uses[c]									
	Watersheds and Named Subwatersheds																Fish Species[b]																Target Species/Group								
	Nushagak River	Mulchatna River	Koktuli River	North Fork Koktuli River	Kvichak River	Iliamna Lake	Upper Talarik Creek	Newhalen River	Lake Clark	Chulitna River	Rock Creek	Groundhog Creek	Long Lake	Koksetna River	Black Creek	Waters[a]	Northern Pike	Longnose Sucker	Humpback Whitefish	Least Cisco	Round Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Arctic Grayling	Burbot	Ninespine Stickleback	Slimy Sculpin	Village	Salmon	Other Salmonids	Other Fishes	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
Downstream of Mine(s) and Transportation Corridor					x	x		x	x	x						Chulitna River	A, J	A, J	A	A, J	J			x	x			A, J	A, J			Igiugig							x		
					x	x		x	x	x	x					Rock Creek		J			A, J		A					A				Iliamna					x	x	x	x	x
					x	x		x	x	x			x			Unnamed tributaries		J												A, J											
					x	x		x	x	x				x		Koksetna River		A, J			A, J		A, J					A, J	J		A, J	Newhalen	x	x						x	
					x	x	x									Unnamed tributaries							J	A-Sp		A, J		J				Nondalton			x						x
	x	x	x	x												Unnamed tributaries						A-Sp, J									A, J	Port Alsworth					x			x	

Notes:

[a] "Tributary" indicates that the channel flows into the stream listed above it; "stream" indicates that the receiving water is off-site, as identified in the columns to the left. For waters downstream of the mine and transportation corridor, fishes and subsistence uses are noted only where different from the waters' prior listing.

[b] A = adult; Sp = spawning; J = juvenile; x = unknown life stage. "Unknown" indicates an apparent lack of surveys.

[c] Subsistence uses not separated by watershed. Uses are noted only for areas of direct impacts (i.e., mine and/or transportation corridor), if they occur there; otherwise, they are noted for areas downstream. "Downstream" applies only to the drainage immediately downstream (e.g., Lower Talarik Creek, but not Iliamna Lake or the Kvichak River). Data for Dillingham and Ekwok are less detailed and more dated than for other villages, so use patterns may have changed.

[d] Assumes existing mine infrastructure at the Pebble deposit; hypothetical routing minimizes distance, topographic gradients and stream crossings and assumes water body crossings would be avoided.

Sources: Fall et al. 1986, Schichnes and Chythlook 1991, Stickman et al. 2003, Fall et al. 2006, Woody and Young 2006, Krieg et al. 2009, ADF&G 2012, Johnson and Blanche 2012.

Table 13-6. Waters, fishes, and subsistence uses potentially affected by a mine at the AUDN/Iliamna prospect.

Location	Affected Waters: Watersheds and Named Subwatersheds: Kvichak River	Coffee Creek	Alagnak River	Levelock Creek	Jensen Creek	Yellow Creek	Ole Creek	Pecks Creek	Iliamna Lake	Lower Talarik Creek	Upper Talarik Creek	Pete Andrews Creek	Newhalen River	Waters[a]	Fish Species[b]: Arctic-Alaskan Brook Lamprey	Longnose Sucker	Northern Pike	Alaska Blackfish	Rainbow Smelt	Humpback Whitefish	Round Whitefish	Unknown Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Chum Salmon	Pink Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Lake Trout	Arctic Grayling	Burbot	Ninespine Stickleback	Threespine Stickleback	Unknown Stickleback	Coastrange Sculpin	Slimy Sculpin	Unknown Sculpin
Claim Block	x				x									Jensen Creek >4 unnamed tributaries Numerous unnamed lakes and ponds Unknown extent of wetlands	Unknown								x	A-Sp	A-Sp	A-Sp		x	Unknown										
Claim Block	x					x								Yellow Creek >70 unnamed tributaries Numerous unnamed lakes and ponds Extensive wetlands			J	A, J							A-Sp	A-Sp		x				J		A, J				A, J	
Claim Block	x													Unnamed stream 3 unnamed ponds Unknown extent of wetlands	Unknown									A-Sp		A-Sp			Unknown										
Transportation Corridor to Levelock[d]	x			x										>1 unnamed tributary Unknown extent of wetlands	Unknown																								
Transportation Corridor to Naknek[e]	x													Kvichak River Unknown extent of wetlands					A			x	A	x	A	A	x	A	A	x		A							
Transportation Corridor to Naknek[e]	x		x											Alagnak River >1 unnamed tributary Unknown extent of wetlands			A	A, J	A			x	A-Sp, J	A-Sp	A	A-Sp	A	x	A	A		A						A, J	
Transportation Corridor to Naknek[e]	x													Coffee Creek >4 unnamed tributaries Unknown extent of wetlands	Unknown																								
Transportation Corridor to Newhalen[e]	x													Kvichak River (>4 crossings) >8 unnamed tributaries Unknown extent of wetlands		A	A	A	A	A	A	x	A	A	A	A	A	x	A	A		A	A						
Transportation Corridor to Newhalen[e]	x						x							Ole Creek Unknown wetland extent								x	x	x	A-Sp	x		x											
Transportation Corridor to Newhalen[e]	x							x						Pecks Creek Unknown extent of wetlands									x	A-Sp	x	A-Sp		x											
Transportation Corridor to Newhalen[e]	x								x					>15 unnamed tributaries Unknown extent of wetlands								x	x		A-Sp			x											

Location	Subsistence Uses[c]: Village	Target Species/Group: Salmon	Other Salmonids	Other Fishes	Beluga Whale	Seal	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
Claim Block	Dillingham								x			
Claim Block	Ekwok								x			
Claim Block	Igiugig									x		
Claim Block	Kokhanok									x		
Claim Block	Koliganek								x			
Claim Block	Levelock							x	x	x		x
Claim Block	New Stuyahok								x			
Transportation Corridor to Levelock[d]	Ekwok								x			
Transportation Corridor to Levelock[d]	Levelock				x	x		x	x	x	x	x
Transportation Corridor to Levelock[d]	New Stuyahok								x			
Transportation Corridor to Naknek[e]	Ekwok								x			
Transportation Corridor to Naknek[e]	Igiugig				x							
Transportation Corridor to Naknek[e]	Koliganek										x	
Transportation Corridor to Naknek[e]	Levelock	x	x	x				x	x	x	x	x
Transportation Corridor to Newhalen[e]	Ekwok								x			
Transportation Corridor to Newhalen[e]	Igiugig	x	x	x	x	x	x	x	x	x	x	x
Transportation Corridor to Newhalen[e]	Iliamna	x	x	x				x	x		x	
Transportation Corridor to Newhalen[e]	Kokhanok		x					x	x		x	
Transportation Corridor to Newhalen[e]	Koliganek										x	

Table 13-6. Waters, fishes, and subsistence uses potentially affected by a mine at the AUDN/Iliamna prospect.

Location	Kvichak River	Coffee Creek	Alagnak River	Levelock Creek	Jensen Creek	Yellow Creek	Ole Creek	Pecks Creek	Iliamna Lake	Lower Talarik Creek	Upper Talarik Creek	Pete Andrews Creek	Newhalen River	Waters[a]	Arctic-Alaskan Brook Lamprey	Longnose Sucker	Northern Pike	Alaska Blackfish	Rainbow Smelt	Humpback Whitefish	Round Whitefish	Unknown Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Chum Salmon	Pink Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Lake Trout	Arctic Grayling	Burbot	Ninespine Stickleback	Threespine Stickleback	Unknown Stickleback	Coastrange Sculpin	Slimy Sculpin	Unknown Sculpin
	Affected Waters: Watersheds and Named Subwatersheds														Fish Species[b]																								
Transportation Corridor to Newhalen[e] (Continued)	x								x	x				Lower Talarik Creek >3 unnamed tributaries Wetlands at least along stream			A				A, J		J	x	A-Sp, J			A	A, J	A		A		A, J	A, J	A		A, J	
Transportation Corridor to Newhalen[e] (Continued)	x								x		x			Upper Talarik Creek Wetlands at least along streams							A		A-Sp, J	A-Sp	A-Sp, J	A-Sp	x	x	A			A				A, J		A, J	
Transportation Corridor to Newhalen[e] (Continued)	x											x		Pete Andrews Creek >1 unnamed tributary Wetlands at least along stream											A-Sp			x											
Transportation Corridor to Newhalen[e] (Continued)	x												x	Newhalen River Unknown extent of wetlands		J	A			A	A, J		x	A-Sp	A-Sp, J	A		A	A, J	A	A	A, J							x
Downstream of Mine(s) and Transportation Corridor	x				x									Jensen Creek									A																

Subsistence Uses[c] — Target Species/Group

Location	Village	Salmon	Other Salmonids	Other Fishes	Beluga Whale	Seal	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
Transportation Corridor to Newhalen[e] (Continued)	Levelock	x	x	x	x	x	x	x	x	x	x	x
Transportation Corridor to Newhalen[e] (Continued)	Newhalen	x	x	x			x	x	x	x	x	x
Transportation Corridor to Newhalen[e] (Continued)	New Stuyahok										x	
Transportation Corridor to Newhalen[e] (Continued)	Nondalton								x		x	
Transportation Corridor to Newhalen[e] (Continued)	Port Alsworth	x							x			
Downstream of Mine(s) and Transportation Corridor	Levelock	x										

Notes:

[a] "Tributary" indicates the channel flows into the stream listed above it; "stream" indicates the receiving water is off-site, as identified in the columns to the left. For waters downstream of the mine and transportation corridor, fishes and subsistence uses are noted only where different from the waters' prior listing.

[b] A = adult; Sp = spawning; J = juvenile; x = unknown life stage. "Unknown" indicates an apparent lack of surveys.

[c] Subsistence uses are not separated by watershed. Uses are noted only for areas of direct impacts (i.e., mine and/or transportation corridor), if they occur there; otherwise, they are noted for areas downstream. "Downstream" applies only to the drainage immediately downstream (e.g., Yellow Creek for the mine, but not the Kvichak River). Data for Dillingham and Ekwok are less detailed and more dated than for other villages, so use patterns may have changed.

[d] Hypothetical routing minimizes distance, topographic gradients and stream crossings and assumes water body crossings would be avoided.

[e] Hypothetical routing follows that shown in the *Southwest Alaska Transportation Plan* (ADOT 2004), but avoids stream and water body crossings, where possible. Waters and fishes for the transportation corridor to Naknek are limited to those in the Kvichak River watershed (i.e., Coffee Creek and north); subsistence uses do not include those by villages outside the watershed.

Sources: Levelock Village Council 2005, Fall et al. 2006, Krieg et al. 2009, ADF&G 2012, Johnson and Blanche 2012.

Table 13-7. Waters, fishes, and subsistence uses potentially affected by a mine at the Humble prospect.

	Affected Waters														Fish Species[b]																						Subsistence Uses[c]										
	Watersheds and Named Subwatersheds																																					Target Species/Group									
Location	Nushagak River	Nuyakuk River	Napotoli Creek	Klutuk Creek	Kokwok River	Kenakuchuk Creek	Nameless Creek	Koggiung Creek	Iowithla River	Wood River	Muklung River	Kvichak River	Bear Creek	Waters[a]	Arctic-Alaskan Brook Lamprey	Longnose Sucker	Northern Pike	Alaska Blackfish	Rainbow Smelt	Round Whitefish	Unknown Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Chum Salmon	Pink Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Arctic Grayling	Burbot	Ninespine Stickleback	Threespine Stickleback	Coastrange Sculpin	Slimy Sculpin	Unknown Sculpin	Village	Salmon	Other Salmonids	Other Fishes	Seal	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
Claim Block	x	x												Unnamed stream and ≥6 tributaries 17 unnamed lakes and ponds Wetlands at least along main stem								J	J	J					A, J					A		A, J	Dillingham							x			
	x	x												Unnamed stream and ≥2 tributaries 8 unnamed lakes and ponds Wetlands at least along streams								J														A, J											
	x													Unnamed stream and ≥3 tributaries 2 unnamed ponds Unknown wetland extent								J													A	A, J	Ekwok						x	x			
	x		x											Napotoli Creek ≥43 unnamed tributaries 48 unnamed lakes and ponds Wetlands at least along main stem and primary tributary	J					A, J		A-Sp, J	A-Sp, J		A-Sp		x		A, J	A, J		A, J		A, J	A, J												
	x			x										Klutuk Creek ≥17 unnamed tributaries Numerous unnamed lakes and ponds Unknown wetland extent								A-Sp, J	A-Sp	A-Sp	A-Sp		x					A, J		A	A, J	A, J	Koliganek					x		x	x		
	x				x	x								Kenakuchuk Creek ≥3 unnamed tributaries 7 unnamed lakes and ponds Unknown wetland extent	Unknown																																
	x				x									Unnamed stream Approximately 1.3 km² (300 ac) of wetlands	Unknown																						New Stuyahok							x	x		x
	x				x									Unnamed stream and ≥17 tributaries Numerous unnamed lakes and ponds								J	J						A, J					A, J	A, J												
Transportation Corridor to Aleknagik[d]	x				x									Kokwok River Extensive wetlands	Unknown							A-Sp	A-Sp	A-Sp	A-Sp	A-Sp	x	Unknown									Aleknagik						x				
	x				x		x							Nameless Creek ≥3 tributaries Extensive wetlands								J										A, J		A, J	A, J		Ekwok		x	x			x				
	x									x	x			Muklung River ≥1 tributary Extensive wetlands								A-Sp	A-Sp	A-Sp			x										New Stuyahok						x	x		x	

Table 13-7. Waters, fishes, and subsistence uses potentially affected by a mine at the Humble prospect.

	Affected Waters														Fish Species[b]																						Subsistence Uses[c]										
	Watersheds and Named Subwatersheds																																					Target Species/Group									
Location	Nushagak River	Nuyakuk River	Napotoli Creek	Klutuk Creek	Kokwok River	Kenakuchuk Creek	Nameless Creek	Koggilung Creek	Iowithla River	Wood River	Muklung River	Kvichak River	Bear Creek	Waters[a]	Arctic-Alaskan Brook Lamprey	Longnose Sucker	Northern Pike	Alaska Blackfish	Rainbow Smelt	Round Whitefish	Unknown Whitefish	Coho Salmon	Chinook Salmon	Sockeye Salmon	Chum Salmon	Pink Salmon	Arctic Char	Rainbow Trout	Dolly Varden	Arctic Grayling	Burbot	Ninespine Stickleback	Threespine Stickleback	Coastrange Sculpin	Slimy Sculpin	Unknown Sculpin	Village	Salmon	Other Salmonids	Other Fishes	Seal	Brown Bear	Moose	Caribou	Other Mammals	Waterfowl	Other Birds
Transportation Corridor to Levelock[e]														Kokwok River																																	
	x				x									≥2 unnamed tributaries	Unknown							A-Sp	A-Sp	A-Sp	A-Sp	A-Sp	x	Unknown									Dillingham		x	x				x			
														Extensive wetlands																																	
	x								x					≥10 unnamed streams				A, J				A-Sp, J	A-Sp									A, J		A, J			Ekwok	x	x	x				x		x	
														Unknown wetland extent																																	
	x													Unnamed stream								J								J				A, J													
														Unknown wetland extent																																	
														Nushagak River (≥1 channel)																							Koliganek						x	x			
	x													≥2 unnamed tributaries		J	J				x	A-Sp, J	A-Sp, J	A-Sp	A-Sp	A-Sp	x				J				A												
														Unknown wetland extent																																	
														Koggilung Creek																																	
	x							x						≥3 unnamed tributaries	A, J	J						Unknown								J		A, J				J	Levelock						x	x	x	x	x
														Extensive wetlands																																	
														Bear Creek																																	
												x	x	≥3 unnamed tributaries								Unknown										A, J					New Stuyahok	x	x	x			x	x	x	x	x
														Unknown wetland extent																																	
Downstream of Mine(s) and Transportation Corridor	x	x												Unnamed tributary		A, J				A, J		A-Sp	A-Sp	A-Sp	A-Sp	J				A, J	A, J						Aleknagik				x						
	x		x											Napotoli Creek	A																																
	x			x										Klutuk Creek	J	A, J		A, J		A, J			J	J				A, J	J	A, J			A, J	J			Dillingham	x								x	
	x				x									Kokwok River	J	A, J	A, J			J										J	J				A, J												
	x				x	x								Kenakuchuk Creek								A-Sp	A-Sp	A-Sp	A-Sp	A-Sp	x																				
	x				x									Unnamed tributary	Unknown																						Koliganek	x	x	x				x		x	x
	x				x									Unnamed tributary											A-Sp	A-Sp				A, J																	
	x													Nushagak River	A, J	A	A		x	J				J						J			A, J				Newhalen							x			
	x							x						Koggilung Creek			A, J	A, J		J										A					A, J												
	x								x					Lowithla River	J	A, J				A, J		J	J	J				A, J	A, J	A, J	J				A, J		Portage Creek	x	x								
												x	x	Bear Creek																																	

Notes:

[a] "Tributary" indicates that the channel flows into the stream listed above it; "stream" indicates that the receiving water is off-site, as identified in the columns to the left. For waters downstream of the mine and transportation corridor, fishes and subsistence uses are noted only where different from the waters' prior listing.

[b] A = adult; Sp = spawning; J = juvenile; x = unknown life stage. "Unknown" indicates an apparent lack of surveys.

[c] Subsistence uses are not separated by watershed. Uses are noted only for areas of direct impacts (i.e., mine and/or transportation corridor), if they occur there; otherwise, they are noted for areas downstream. "Downstream" applies only to the drainage immediately downstream (e.g., Napotoli Creek, but not the Nushagak River). Data for Dillingham and Ekwok are less detailed and more dated than for other villages, so use patterns may have changed.

[d] Hypothetical routing minimizes distance, topographic gradients and stream crossings and assumes water body crossings would be avoided. Subsistence uses noted do not include those by residents of villages outside the Nushagak River watershed.

[e] Hypothetical routing follows that shown in the *Southwest Alaska Transportation Plan* (ADOT 2004), but avoids stream and water body crossings, where possible. Waters, fishes, and subsistence uses are limited to those west of Levelock.

Sources: Fall et al. 1986, Schichnes and Chythlook 1991, Fall et al. 2006, Portage Creek Village Council and Residents of Portage Creek 2006, Krieg et al. 2009, ADF&G 2012, Holen et al. 2012, Johnson and Blanche 2012.

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Table 13-8. Streams, water bodies, and wetlands potentially eliminated by additional large-scale mines in the Nushagak and Kvichak River watersheds.

Mine	Claim Block Size (km^2)	Mine Area[a] (km^2)	Streams		Water Bodies		Wetlands	
			Density (km/km^2)	Length Eliminated[b] (km)	Density (%)	Area Eliminated (km^2)	Density[c] (%)	Area Eliminated (km^2)
Pebble South/PEB[d]	1,380	3.87	1.07	4.1	3.14	0.12	18.3	0.71
							30.5	1.18
Big Chunk South[e]	142	3.87	1.18	4.6	6.11	0.24	6.1	0.24
							83.5	3.23
		10.7		12.6		0.65	6.1	0.65
							83.5	8.93
Big Chunk North	119	3.87	1.45	5.6	4.18	0.16	4.2	0.16
							57.2	2.21
		10.7		15.5		0.45	4.2	0.45
							57.2	6.11
Groundhog	317	3.87	1.23	4.8	1.24	0.05	15.8	0.61
							17.0	0.66
		10.7		13.2		0.13	15.8	1.69
							17.0	1.82
AUDN/Iliamna	183	10.7	1.19	12.7	6.01	0.64	57.3	6.13
							75.3	8.05
Humble	280	10.7	1.07	11.4	0.66	0.07	0.7	0.07
							9.1	0.97
TOTALS	2,420	36.9		43.2		1.28		7.9
		57.4		69.5		2.06		27.1

Notes:

[a] Mine area is based on the Pebble 0.25 scenario and includes footprint of major mine components (mine pit, waste rock piles, and tailings storage facility). Where two values are presented for a mine, the small value represents the footprint assuming the mine uses an existing tailings storage facility at the Pebble deposit, whereas the larger value represents the footprint assuming the mine uses its own tailings storage facility.

[b] Length eliminated = footprint of major mine components x stream density.

[c] For claim blocks with NWI coverage (i.e., Pebble South/PEB, Groundhog, and AUDN/Iliamna), minimum density = NWI wetland density and maximum density = (differential between National Wetlands Inventory (NWI) wetland density and National Hydrography Dataset (NHD) water body density in area of NWI wetland coverage) x NHD water body density for entire claim block. For claim blocks with no NWI coverage, minimum density = NHD water body density and maximum density = (maximum differential between NWI wetland density and NHD water body density) x NHD water body density.

[d] Claim block size for entire PLP/NDM block; water body density includes portion of Iliamna Lake.

[e] Water body density includes portions of Nikabuna Lakes.

13.2.2.2 Potentially Affected Waters, Fishes, and Subsistence Uses

Table 13-3 summarizes information on the waters, fish, and subsistence uses potentially affected by a mine at Big Chunk South prospect. The 142-km^2 Big Chunk South claim block is entirely within the drainage of the Chulitna River, which flows into Lake Clark National Park and Preserve and then into the lake itself, at Turner Bay, 40 km northeast of the block. A segment of the strongly meandering river, including the Nikabuna Lakes system, runs along the entire 27-km north boundary of the block. Current National Wetlands Inventory (NWI) mapping does not include the Big Chunk South claim block; however, based on aerial photography and U.S. Geological Survey (USGS) topographic mapping, extensive wetlands appear to be associated with the river, extending upstream along several tributaries. Stream density at Big Chunk South is 1.18 km/km^2, and the majority of streams in the block are headwater tributaries. Water body density—more than 6% of the block—is the highest of all the potential mine sites, due in part to the Nikabuna Lakes. The relatively flat valley of the Chulitna River occupies most of the claim block. Elevations range from approximately 90 m along the river to 320 m in the more rugged south-central part of the block.

To connect to Cook Inlet, a road and pipelines from the Big Chunk South claim block would likely ascend the valley of an unnamed Chulitna River tributary and then cross the headwaters of the North Fork Koktuli River to join the end of the assessment corridor, approximately 14 km south of the block.

To date, very few known fish surveys have been conducted in either the Big Chunk South claim block or the middle or upper Chulitna River (Table 13-3). Fall et al. (2006) describe the Chulitna River valley as one of the most important subsistence areas for Nondalton. Among the salmonids included as assessment endpoints, only Dolly Varden are found in the claim block. One village subsistence fishes and hunts for bear, moose, caribou, other mammals, and waterfowl in the claim block; four other villages hunt for caribou; and one other village hunts for moose. As an average across the claim block, stream loss would range from 4.6 to 12.6 km, and wetland area eliminated would range from 0.24 to 8.9 km^2, depending on whether or not a TSF would be constructed on site (Table 13-8).

13.2.3 Big Chunk North

13.2.3.1 Description

The Big Chunk North prospect is approximately 11 km northwest of the Big Chunk South prospect, 34 km northwest of the Pebble deposit, approximately 48 km northwest of Nondalton and 96 km northeast of Koliganek. A mine at Big Chunk North would potentially use TSFs, mill, and other facilities at either Big Chunk South or Pebble. We consider the impacts of both a mine with shared facilities and one without (Table 13-8). In both cases, we anticipate that a mine at Big Chunk North would connect to the assessment corridor.

13.2.3.2 Potentially Affected Waters, Fishes, and Subsistence Uses

Table 13-4 summarizes information on the waters, fishes, and subsistence uses potentially affected by a mine at Big Chunk North. Like the Pebble deposit, the 119-km^2 Big Chunk North claim block straddles the drainage divide between the Nushagak and Kvichak River watersheds. Northwest of Buck Mountain, in the northwest corner, the block contains ponds and streams that are part of the headwaters of Keefer Creek, a tributary of the Mulchatna River, as well as the uppermost reaches of the North Fork Swan River in the Koktuli River watershed. Nearly 90% of the block, though, is a high-density network of streams, ponds, and wetlands that form the headwaters of the Chulitna River, which rises immediately north of the block and flows into Lake Clark National Park and Preserve, and ultimately Lake Clark itself. The system departs the block along its south boundary. National Hydrography Dataset (NHD) stream density in the Big Chunk North claim block (1.45 km/km^2) is higher than in any of the other areas of potential mines we consider, and more than one-third higher than in the PLP/NDM claim block (USGS 2012). Water body density (>4%) is also fairly high compared to other mine sites, but lower than at Big Chunk South and AUDN/Iliamna.

A transportation corridor to service a mine in the Big Chunk North claim block would presumably follow the Chulitna River valley south and eastward to the Big Chunk South block, where it would link up to the corridor described in Section 13.2.2.1.

To date, no known fish surveys have been conducted in the Big Chunk North claim block or along its potential transportation corridor, nor any freshwater fish surveys anywhere in Keefer Creek. Table 13-4 summarizes what little information is available on fish presence in the waters potentially affected by a mine in this claim block. As in the south block, we do not have data on population sizes. At least one of the stream systems—the North Fork Swan River—has an abundance of beaver dams, indicating it may provide important overwintering and rearing habitat (Johnson and Blanche 2012). Four villages hunt in the claim block but no subsistence fishing is reported (Table 13-4). Across the claim block, stream loss would range from 5.6 to 15.5 km, and wetland loss would range from 0.16 to 6.1 km^2, depending on whether a TSF would be constructed on site (Table 13-8).

13.2.4 Groundhog

13.2.4.1 Description

The 317-km^2 Groundhog claim block abuts the northeast corner of the PLP/NDM block, approximately 10 km west of Nondalton and 20 km north-northwest of Iliamna. At present there does not appear to be a relationship between the Groundhog claim holders and PLP/NDM. Nevertheless, given its proximity to the potential Pebble facility (approximately 6 km), we consider it both as a separate, stand-alone mine and as one that shares some facilities associated with potential mine development at the Pebble deposit (Table 13-8), including the assessment corridor. The current route for such a corridor is approximately 4 km south of the claim block and 13 km from the recent target area for exploration drilling at Groundhog (AHEA 2011). A connector would presumably follow one of the Upper Talarik Creek tributaries down to the corridor.

13.2.4.2 Potentially Affected Waters, Fishes, and Subsistence Uses

Table 13-5 summarizes information on the waters, fishes, and subsistence uses potentially affected by a mine at Groundhog. Similar to Big Chunk North, nearly 90% of the Groundhog prospect lies in the drainage of the Chulitna River, which passes through the narrow, central part of the block and flows into the Lake Clark National Park and Preserve and ultimately Lake Clark itself. South of Groundhog Mountain, the claim includes a number of headwater tributaries to Upper Talarik Creek and the North Fork Koktuli River. A very small portion of the block—less than 1 km^2 in the southeast corner—drains to tributaries of the Newhalen River, which connects Sixmile Lake, below Lake Clark, to Iliamna Lake. A road from the claim block to the assessment corridor would presumably follow one of the Upper Talarik Creek tributaries.

Based on NHD mapping, stream density in the Groundhog block (1.23 km/km^2) is the second highest of those we consider; water body density (>1%) is much lower than at all other sites except Humble (Table 13-8). The NWI maps extensive wetlands along the Chulitna River and Rock Creek, as well as along lower Groundhog Creek, the drainage in which the most recent exploration of the prospect has occurred (AHEA 2011, USFWS 2012).

Surveys of fish use in waters of the Chulitna River drainage have been limited to date (Table 13-5). There is more information for the headwater tributaries in the Upper Talarik Creek and North Fork Koktuli River watersheds in the southern part of the claim block, which both support salmonids. The Upper Talarik Creek tributary system originates in the same series of lakes and ponds as Groundhog Creek, in the Chulitna River watershed, at an elevation of approximately 460 m. Two Pacific salmon species and Dolly Varden have been reported in the claim block, and one village hunts there (Table 13-5). On average across the claim block, stream loss would range from 4.8 to 13.2 km, and wetland loss would range from 0.61 to 1.8 km^2, depending on whether or not a TSF would be constructed on site (Table 13-8).

13.2.5 AUDN/Iliamna

13.2.5.1 Description

The AUDN/Iliamna prospect is approximately 35 km west of Iliamna Lake and 90 km southwest of the Pebble deposit. It is in the vicinity of the native villages of Levelock, New Stuyahok, Ekwok, and Igiugig (Figure 13-1). The bulk of the claims associated with this prospect were newly established in 2012 by Millrock Alaska, the same company that owns the Humble claims. Millrock began exploration of the prospect in 2012 and describes it as being part of the porphyry copper–gold belt that includes the Pebble deposit (Millrock Resources 2011). Their AUDN claims surround others staked earlier and still held by a partnership that includes TNR Gold Corp., the owner of the Shotgun claims. TNR Gold Corp. calls their project "Iliamna."

In light of the higher development costs associated with this prospect's distance from other potential mines, we assume that a future mine at this prospect would require a joint venture involving both claim blocks and would be self-contained (i.e., it would not share any facilities with other mines) (Table 13-8). The closest potential port site would be on the lower, tidally influenced reach of the Kvichak River, at or near Levelock. Naknek, approximately 56 km south of the prospect, already operates as a port and could be an alternative location, should establishing a port on the Kvichak River prove infeasible.

Development of a mine at AUDN/Iliamna could trigger the involvement of the Alaska Department of Transportation and Public Facilities (ADOT) in building the Levelock-to-Naknek portion of the Cook Inlet-to-Bristol Bay (CIBB) and Dillingham/Bristol Bay (DBB) corridors described in the *Southwest Alaska Transportation Plan* (SWATP) (ADOT 2004). The SWATP anticipates that such construction would not occur within its 20-year planning period; however, it noted that “changing circumstances” such as “discovery of high-value resource that could potentially be accessed economically through development of [such roads could] trigger consideration of an earlier implementation” schedule. A mine at AUDN/Iliamna could be such a trigger, and ADOT involvement in road construction could defray mine development costs. ADOT is currently investigating such a project (the Ambler Mining District Access, in northwestern Alaska) as part of the state’s “Roads to Resources” program (ADOT 2011a). For similar reasons, Cook Inlet could be another alternative location for a port, using the Levelock-to-Newhalen portion of the CIBB and DBB corridors; shipping distances to Canada and the lower 48 states appear to be shorter from Cook Inlet than from Naknek. For the purposes of this assessment, we consider all three possible transportation corridors.

13.2.5.2 Potentially Affected Waters, Fishes, and Subsistence Uses

Table 13-6 summarizes information on the waters, fish, and subsistence uses potentially affected by a mine at AUDN/Iliamna. The 183 km^2 AUDN/Iliamna claim block occupies a low, relatively flat area of the Kvichak River watershed, including a portion of the glacial outwash plain at the western edge of the end moraine that originally formed Iliamna Lake (Detterman and Reed 1973). The Yellow Creek system drains approximately 90% of the claim block and flows into the Kvichak River approximately 15 km upstream of Levelock. Jensen Creek drains the southern 11 km^2 of the block and enters the Kvichak River 10 km above Levelock; some maps incorrectly identify it as Yellow Creek (Levelock Village Council 2005). Both stream systems are strongly meandering. The remainder of the claim flows to a third tributary system that enters the Kvichak River above Yellow Creek. Residents of Levelock use the AUDN/Iliamna area most for subsistence, but other villages also fish or hunt in the area or along the potential transportation corridors (Table 13-6).

Stream density at AUDN/Iliamna is higher than at PLP/NDM and Humble, but lower than at most of the other blocks. Despite its lack of any large lakes (the largest is less than 0.5 km^2), water body density at AUDN/Iliamna (>6% of the block) is nearly twice that of the PLP/NDM claim block (which encompasses 4.9 km^2 of Iliamna Lake) and almost the same as at Big Chunk South (which includes 2.7 km^2 of the Nikabuna Lakes). NWI mapping, which covers almost 70% of the block, also shows extensive wetlands encompassing approximately 57% of the block. Four species of Pacific salmon are cataloged in numerous streams on the claim block and seven villages hunt there for a variety of mammals and birds (Table 13-6). Across the claim block, stream loss would average 12.7 km, and wetland loss would range from 6.1 to 8.1 km^2 (Table 13-8).

The short transportation corridor from AUDN/Iliamna to Levelock would potentially involve only one stream crossing, of an unnamed tributary to Levelock Creek. To connect to Naknek or Cook Inlet (via the conceptual CIBB route and the assessment corridor), the road would first have to cross the Kvichak

River at or near Levelock. To reach Naknek, it would also have to cross the Alagnak River and Coffee Creek, as well as several tributaries. Instead of crossing the Alagnak River, the route to the assessment corridor would follow it briefly, before crossing Ole and Pecks Creeks, as well as crossing the Kvichak River at least one more time (at or near Igiugig). From there, it would pass the west and north shores of Iliamna Lake, crossing Lower and Upper Talarik Creeks, at least 16 other streams, and the Newhalen River, before joining the assessment corridor near Iliamna.

13.2.6 Humble

13.2.6.1 Description

Accounts characterize the Humble prospect as geologically and geochemically similar to the Pebble deposit (Szumigala et al. 2011, Millrock Alaska 2012b). It is approximately 135 km southwest of the Pebble deposit, 60 km northwest of AUDN/Iliamna, and 20 to 30 km west to northwest of the villages of Koliganek, New Stuyahok, and Ekwok. Wood-Tikchik State Park, the largest state park in the United States, is approximately 13 km northwest and 29 km west of the claim block.

Due to Humble's distance from the Pebble deposit and the AUDN/Iliamna prospect, we do not anticipate sharing of facilities between the mines (Table 13-8). The Dillingham-Aleknagik Road (46 km to the southwest) is the closest link to existing road infrastructure and port facilities, the latter of which are another 40 km further at Dillingham. ADOT is currently pursuing federal permits for construction of a bridge over the Wood River, which presumably would be the southernmost link in a transportation corridor from Humble to Aleknagik (ADOT 2011b, USCG 2012). Alternatively, development of a mine at Humble could trigger the involvement of ADOT in building all or portions of the CIBB and DBB corridors, which could serve to connect Humble to a port on Cook Inlet. A road from the Humble claim block to the conceptual DBB route presented in the SWATP would be shorter than a mine road connecting to Aleknagik (ADOT 2004).

13.2.6.2 Potentially Affected Waters, Fishes, and Subsistence Uses

Table 13-7 summarizes information on the waters, fishes, and subsistence uses potentially affected by a mine at Humble. The 280-km^2 claim block's east- and south-flowing streams are entirely within the Nushagak River watershed (Figure 13-1), and the mainstem channels are all strongly meandering. More than 40% of the block drains to Napotoli Creek, which enters the Nushagak River approximately 14 km upstream of Koliganek. The northernmost 36 km^2 of the block flow to one unnamed tributary of the Nushagak River and two unnamed tributaries of the Nuyakuk River, which connects Tikchik Lake, in Wood-Tikchik State Park, to the Nushagak River.

In the south, nearly one-quarter of the block is in the Klutuk Creek watershed, which flows into the Nushagak River immediately downstream of Ekwok. The remainder of the block—approximately 20%—drains to Kenakuchuk Creek and other tributaries of the Kokwok River, which enters the Nushagak River 8 km downstream of Ekwok. Stream density in the claim block is approximately the same as at the Pebble deposit (1.07 km/km^2), which is lower than at all of the other sites considered here (Table 13-8).

NHD water body density at Humble is less than 25% of that at PLP/NDM and only slightly more than half of that at Groundhog, the next most similar prospect (USGS 2012). A long band of ponds occupies the divide between Klutuk Creek and the Kokwok River tributaries. Current NWI mapping does not extend to the Humble block. At minimum there appear to be wetlands along most of the larger stream corridors (e.g., Naptoli, Klutuk, and Kenakuchuk Creeks) and in an approximately 4-km^2 area in the southwest corner of the block, based on aerial photography and USGS topographic mapping. Four Pacific salmon species and Dolly Varden are present in many streams in the claim block, and four villages hunt for a variety of mammals (Table 13-7). On average across the claim block, stream loss would be 11.4 km, and wetland loss would range from 0.07 to 0.97 km^2 (Table 13-8).

A potential route for a transportation corridor from the Humble claim block to Aleknagik that minimizes distance and topographic gradient would cross the Kokwok River, just downstream of Kenakuchuk Creek, ascend the valley of Nameless Creek (a Kokwok tributary), skirt Wood-Tikchik State Park, and then follow the Muklung River (a tributary of the Wood River) before turning north of Marsh Mountain for Aleknagik. The overland route to Cook Inlet would presumably cross the Kokwok River further downstream from the route to Aleknagik, connecting to the Dillingham/Bristol Bay corridor near the Iowithla River, another Nushagak River tributary. The SWATP's conceptual corridor would cross the Nushagak River downstream of Ekwok. After crossing Koggilung Creek, the route would pass into the Kvichak River watershed, reaching the river at or near Levelock. Section 13.2.5.2 describes the route from Levelock to the assessment corridor.

Despite its relatively low stream density, a large number of fish-bearing streams traverse the Humble claim block (Table 13-7). Information on local population sizes is not available. The Napotoli and Klutuk Creek systems contain numerous beaver complexes, as well as frequent seeps and springs, that may provide important overwintering habitat for juvenile salmonids (Johnson and Blanche 2012: nomination forms 04-158, 04 160, 04-171, 04-890, 06 753, 06-754, 11-369–11-–372, 11-381, 11-382, and 11-384–11-386). Residents of Aleknagik, Dillingham, Ekwok, Koliganek, and New Stuyahok have historically used the claim block, the potential transportation corridor area west of Levelock, and/or downstream areas for subsistence fishing, hunting, and/or gathering (Table 13-7). Information for Aleknagik, Dillingham, and Ekwok is less detailed than for the other villages. The reports for Dillingham and Ekwok are more than 20 years old, so their subsistence use patterns may have changed, particularly for caribou hunting, since that species' population and migration routes have shifted (Brna and Verbrugge 2013).

13.2.7 Potential Impacts of Multiple Mines

In the preceding sections, we examined the waters, fishes, and subsistence uses that could be affected by the footprints of the major mine components at six prospects that could be developed after initial development of a large mine at the Pebble deposit. For the purposes of this assessment, we consider the cumulative impacts of these six mines—that is, potential effects on assessment endpoints resulting from the establishment of these six additional mines and their associated transportation corridors. These influences would likely accumulate over time and space, potentially having widespread and extensive effects on the region's populations of fish, wildlife, and human residents.

13.2.7.1 Habitat Eliminated

Table 13-8 summarizes direct losses of aquatic habitat to the footprints of the potential six additional mines. Total stream length eliminated by these footprints would range from approximately 43 to 70 km, and total water body and wetland area lost would range from approximately 1.3 to 2.1 km^2 and 7.9 to 27 km^2, respectively. Loss of these areas to mine footprints would result in extensive losses of floodplain, riparian habitat, and wetland areas. Waters on these claim blocks include the Chulitna River and Rock, Jensen, Yellow, Napotoli, Klutuk and Kenakuchuk Creeks, as well as over 250 unnamed tributaries and over 50 unnamed lakes and ponds. Although not all support salmon, many do. Elimination of substantial habitat across the watersheds would contribute to diminishing the biological complexity of salmon stocks and the portfolio effect, which would likely increase annual variability in the size of Bristol Bay salmon runs (Section 5.2.4).

13.2.7.2 Flow Alteration

Water Withdrawal and Retention

Routine operations at additional mines would also likely degrade or destroy downstream habitat due to water withdrawal and management of precipitation at the mine facilities (Chapters 6 and 7). Mines require water for mill operation and transport of tailings and concentrate. The required withdrawal and retention of surface water and groundwater would effectively reduce the size of the watershed contributing to downstream flow. Mine pit dewatering would further reduce the contributing watershed by creating a cone of depression (Section 6.2.2). Streams, wetlands, and ponds within this cone of depression that receive water through groundwater would dry up, discontinuing any contributions to downstream waters. Groundwater flow down the valley would also be disrupted, potentially affecting spawning and wintering habitat downstream (Chapter 7).

Increased Effective Impervious Surface

Research in other areas has shown that even low levels of industrial, commercial, and residential land uses can cause significant degradation and reduce ecological function in downstream water bodies (Booth and Jackson 1997). Comparisons between stream condition and level of development have consistently demonstrated a correlation between stream degradation and watershed imperviousness (Booth et al. 2002). Greater frequency and intensity of floods, erosion of streambeds, displacement of sediments, poor water quality, increased water temperature, and reductions in channel and habitat structure all have been associated with increases in impervious surface (Booth and Jackson 1997, Booth et al. 2002). In humid regions, approximately 10% effective impervious surface area (i.e., impervious surface area that is directly connected to stream channels) generally causes demonstrable loss of aquatic ecosystem function (Booth and Jackson 1997).

Additional mines and potential transportation corridors at each of the prospects would convert additional land to impervious surface, which could increase total effective impervious surface area above 10% in several subwatersheds in the region (e.g., Groundhog, Jensen, and Napotoli Creeks and the Nikabuna Lakes). Mine and road operators and village governments may be able to limit or even eliminate downstream damage from impervious surface. If operators could moderate discharges to simulate natural flows and prevent higher than natural stormflows, downstream channels could be maintained in relatively stable condition. Achieving this goal would require sufficient on-site water retention capacities to allow a slow meting out of storm-generated flows, rather than contemporaneous discharge of all surface runoff generated by storm events.

Road Crossings

In addition to increasing runoff rates due to their impervious surfaces, the transportation corridors associated with additional mines and other induced development would increase the likelihood of flow regime changes, including channel modification, wherever they crossed streams, wetlands, and other water bodies (Chapter 10, Appendix G). Such alterations frequently affect salmonids and other fishes by blocking access to habitat and/or physically degrading the habitat itself. Cumulatively, these additional transportation corridors would add at least five river crossings (South and North Fork Koktuli, Chulitna, Kokwok, and Nushagak or Muklung) and a minimum of 27 smaller stream crossings (including at least one of Upper Talarik Creek) (Tables 13-2 through 13-7). Overland access to AUDN/Iliamna from a port at Naknek or Cook Inlet (rather than near Levelock) would add at least one and possibly as many as four crossings of the Kvichak River, at least one of another river (Alagnak or Newhalen), and a minimum of six at other streams. These crossings would substantially increase the potential for hydraulic alterations resulting in upstream and downstream habitat degradation (Chapter 10).

13.2.7.3 Water Quality Degradation

Chapters 8 through 11 discuss the potential water quality impacts resulting from water treatment and discharge, tailings dam failure, road construction and operation, and pipeline spills at a single mine at the Pebble site. Additional mines and transportation corridors would have these same potential impacts.

Routine Operations

Routine operations at six additional mines would result in approximately 37 to 57 km^2 of ground disturbance (Table 13-8) depending on whether or not TSFs would be built at each of the mines. Rivers and streams in which water quality could be affected include the Chulitna River and Rock, Jensen, Yellow, Napotoli, Klutuk, and Kenakuchuk Creeks (Tables 13-2 through 13-7). The transportation corridors for these mines would potentially span the width of the Nushagak and Kvichak River watersheds from Cook Inlet to Dillingham and potentially affect the Alagnak, Kvichak, Kokwok, and Nushagak Rivers and Coffee and Bear Creeks. Salmon are reported in streams in the areas of all of these prospects except Big Chunk North and Big Chunk South.

Accidents and Failures

Chapters 8 through 11 describe the probabilities and consequences of a variety of accidents and failures in the mine scenarios. Although the probability of such failures at an individual facility at any given time is low, the cumulative probability of failures increases as the number of facilities increases. For example, historical data suggest a greater than 99% cumulative probability of failure in one of the four pipelines over the life of the Pebble 2.0 scenario (Section 11.1). Additional pipelines at additional mines would increase the overall probability of failure at some location in the Nushagak and Kvichak River watersheds each year. Similarly, the chances of a road failure with significant consequences for downstream waters would be substantial with development of a single mine, and would increase as the length of road in the watersheds increases (Section 10.3.2).

Although failure of any single TSF dam would be a low-probability event, it could be catastrophically damaging to fisheries in the receiving waters if it were to occur. The presence of multiple large-scale mines would increase the probability of at least one TSF dam failure occurring in the watersheds over the mine lifetimes and post-closure periods roughly in proportion to the total number of dams, and thus increase the chance of long-term adverse downstream effects. A TSF dam failure at one the additional mines would likely be similar in nature to the Pebble 0.25 dam failure scenario described in Section 9.3, although the magnitude of adverse impacts would vary with location, TSF size, and the degree of failure. Salmon-bearing waters into which slurry from TSF dam failures at additional mine sites could flow would include the North Fork and mainstem Koktuli Rivers; the Kvichak, Nushagak, Kokwok, and Chulitna Rivers; Upper Talarik, Rock, Napotoli, Klutuk, and Yellow Creeks; Iliamna, Nikabuna, and Long Lakes; and Lake Clark.

Another potential source of pollutant discharges results from human errors in characterizing the mining environment (e.g., its geochemistry or hydrology) and/or anticipating long-term needs for pollutant control (Box 13-2). Human error, as well as mechanical failure, can also result in water bypassing a treatment system. Similar unintended failures in human judgment could result in unanticipated discharges of pollutants from mine sites in the Nushagak and Kvichak River watersheds, particularly as the number of additional sites increases. The cumulative effect of such incidents in the Nushagak and Kvichak River watersheds would likely be a steady decline in productivity in these systems as the affected reaches increase in length and number.

As discussed in Section 14.1.2.5, common mode failures—that is, multiple failures with a common cause—could result from incidents such as earthquakes or severe storms. The potential for common mode failures would be compounded if a mining district were created, increasing the chance that a single severe event could result in multiple failures and adversely affect multiple salmon-bearing waters at one time.

BOX 13-2. EXAMPLES OF MINE CHARACTERIZATION ERRORS

Errors in mine site characterization or anticipation of long-term needs for pollutant control can contribute to mine-related pollutant discharges. Examples at existing mines include the following incidents.

- At the Red Dog Mine in northwest Alaska, treatment of waste rock runoff for metals elevated dissolved solids in runoff to the point that it had to be directed to the tailings storage facility (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 from the TSF were necessary to prevent dam overtopping.
- At the Greens Creek Mine in southeast Alaska, plans included reclamation of the dry stack TSF to prevent acid drainage. However, mine life exceeded the anticipated timeframe, delaying reclamation and resulting in acid drainage from the tailings. A new understanding of site geochemistry indicates that perpetual water treatment will be necessary even after reclamation, a substantial change from the original design. Moreover, mine operators have discovered that local wetland chemistry resulted in a treatment system that redissolves metals before discharge, requiring construction of a new water treatment facility to address this unanticipated source of pollution.
- Human error resulted in an uncontrolled discharge from a TSF at the Nixon Fork Mine, in interior Alaska, in January 2012 (see Box 8-1 for a description of events). This unpermitted discharge does not appear to have reached nearby streams at the time of this writing and may have caused no environmental harm.

The passage of time would be another component influencing cumulative impacts of large-scale mining in these watersheds. Although time could enable the recovery of streams affected by accidents or failures, it could also increase the likelihood that a particular accident or failure would occur. We assume that post-closure site management considerations (Section 6.3) would generally apply to each additional mine, although the specifics would be based on design and operational assumptions of each mine and thus differ from site to site. Closure at each mine would typically require hundreds to thousands of years of monitoring, maintenance, and treatment of any water flowing off-site. Given the magnitude of these timeframes, we would expect multiple and more frequent system failures in future years. In light of the relatively ephemeral nature of human institutions over these timeframes, we would expect that monitoring, maintenance, and treatment would eventually cease, leading to increased release of contaminated waters downstream.

13.3 Cumulative Impacts from Induced Development

Induced development, or development resulting from the introduction of industry, roads, and infrastructure associated with a specific activity to a region, is an iterative phenomenon. Opportunities for employment at mines or in mine-related services would contribute to growth in nearby communities and would increase demand for housing, community infrastructure, and amenities such as recreation in and around those localities. Independent of growth due to mine-related employment, improved accessibility due to road and port construction would reduce the cost of shipping fuel and freight to areas near such infrastructure. Reduced shipping costs would make construction, business operation, recreation/tourism, and general cost of living more affordable, which would facilitate increased growth.

Induced development following the advent of large-scale mining would bring welcome economic opportunities to the region. The potential road systems described in Section 13.2, which could extend completely across the Nushagak and Kvichak River watersheds (approximately 250 miles), would be a major driver of induced development. Currently, access to sites in the Nushagak and Kvichak River watersheds is by air, boat, snow machine, or foot and is typically facilitated and regulated by the tourism industry. Transportation corridors associated with large-scale mines likely would increase vehicle access throughout the two watersheds, thereby increasing both lawful and unlawful unmanaged access to currently remote sites. Access by all-terrain-vehicles and snow machines would be greatly enhanced by a road system, and areas in the two watersheds that are essentially never visited by humans would become accessible. This increased access would extend fishing and hunting pressure and make areas along any transportation corridor more susceptible to trespassing, poaching, and illegal dumping (ADOT 2001).

13.4 Potential Effects on Assessment Endpoints

13.4.1 Fishes

Based on the general locations of likely development and the fish species present in these areas (Tables 13-2 through 13-7), we can estimate some potential impacts on fish, such as the extent of direct habitat losses to typical footprints of the major mine components (43 to 70 km of streams), the approximate number of streams crossed by transportation corridors (5 rivers and 27 streams), and the potential for additional habitat loss or degradation that would result from accidents or failures (Section 13.2.7.3). By identifying general areas where development is reasonably foreseeable (Tables 13-2 through 13-7), we can also, to some extent, consider those losses by habitat type (e.g., headwater streams) and fish species.

In addition to the effects associated with the sheer quantity of lost or degraded habitat, the impacts of large-scale mining could cumulatively threaten biological complexity of the Nushagak-Kvichak salmonid stock complex (Section 5.2.4). Impacts on genetically distinct populations of salmon across the watersheds can reduce biological complexity (i.e., the portfolio effect) and lead to salmon population declines. As described in this chapter, reasonably foreseeable development during an 80-year timeframe could span the Nushagak and Kvichak River watersheds, encompassing many geographically and hydrologically distinct waters. For anadromous fish, the potentially affected waters would include at least 10 different rivers and 20 feeder stream systems. As described by Schindler et al. (2010), each river stock includes tens to hundreds of locally adapted populations distributed among tributaries and lakes. Given the extent of stream losses and habitat degradation, it is reasonable to assume that losses of genetic and life-history diversity would occur with the development of multiple large-scale mines. Even non-anadromous fish would be subject to genetic diversity losses wherever development blocked movement to different spawning, rearing, and overwintering habitats within a stream and/or stream-lake system, thereby isolating portions of the population (Appendix B) (Charles et al. 2000).

Although we can estimate potential habitat losses and anticipate resulting losses of population diversity, we cannot quantify specific fish losses or their significance in terms of overall populations given the current lack of abundance data and uncertainty about the precise locations and magnitude of future developments.

13.4.2 Wildlife and Alaska Native Culture

As the extent of development in the Nushagak and Kvichak River watersheds increased, so would development-related effects on wildlife and on Alaska Native culture. The six additional mines would affect a wide range of wildlife, including both resident and highly migratory species (Tables 13-2 through 13-7). As with fish, data are insufficient to predict wildlife population impacts.

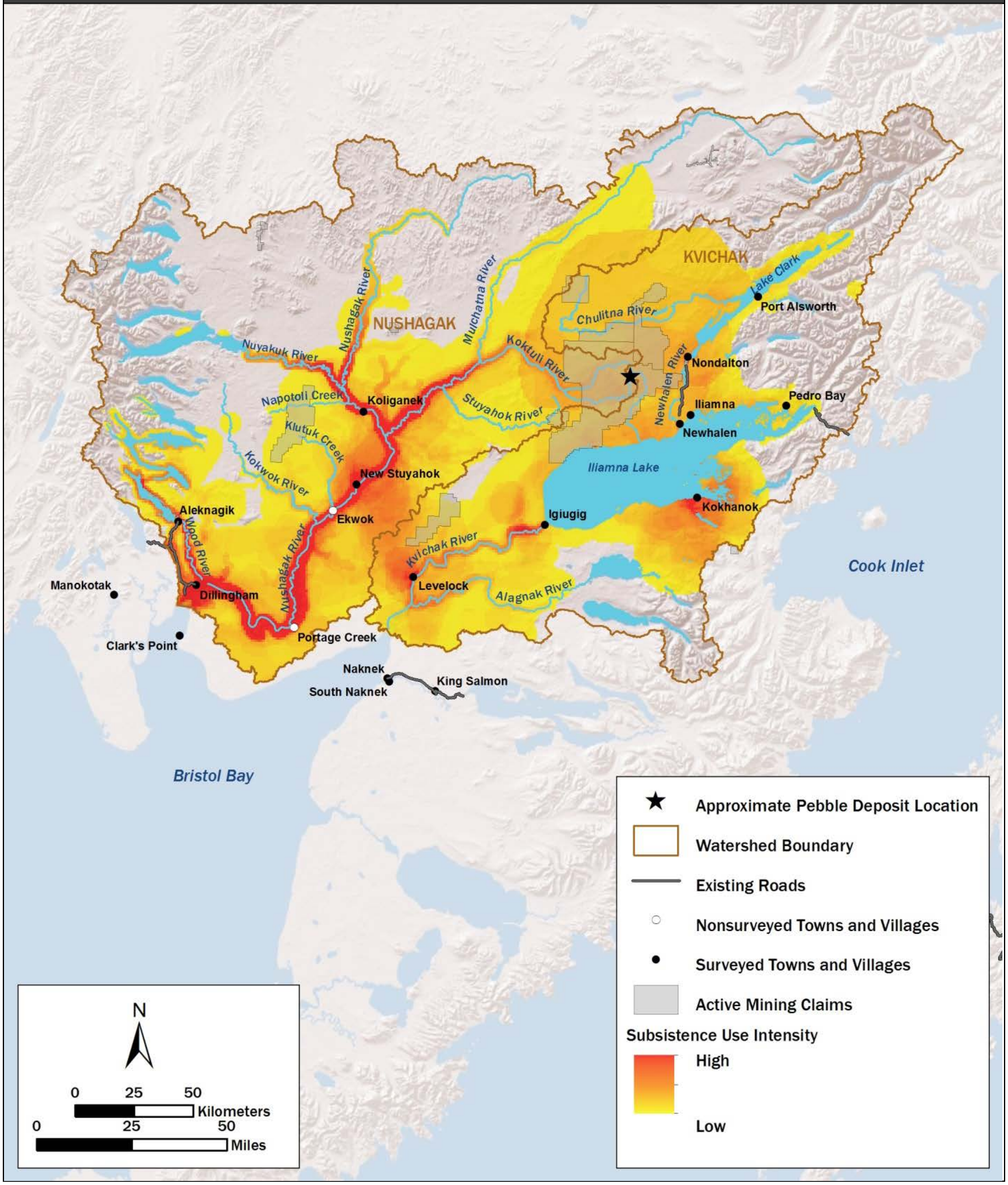
As for Alaska Natives, 13 of the 14 villages in the watersheds would experience some impact on traditional subsistence use areas from additional development (Figure 13-3). Levelock and Igiugig would experience the most impacts, in that additional development would have direct and/or indirect impacts on all categories of their subsistence resources (due primarily to proximity of the potential AUDN/Iliamna mine and a transportation corridor from that mine to Newhalen). Iliamna, Koliganek, Newhalen, New Stuyahok, and Nondalton would also have a large number of subsistence resource categories affected.

Development of a mining district could have broader cultural impacts related to the diminishing role of subsistence in village life, via any reductions in the areas or fish and wildlife populations available for subsistence activities (Chapter 12, Appendix D). Employment opportunities and reduced cost of living resulting from large-scale mining could address some current concerns associated with subsistence, chiefly the cost of fuel needed to access these resources. At the same time, loss of subsistence areas or populations would likely be accompanied by increased westernization resulting from increased access, development, tourism, and prosperity. Both of these factors could erode the current subsistence cultures, at least to some extent.

13.5 Summary

The industrial complex and transportation corridor associated with potential mine development at the Pebble deposit would constitute the second largest human population center in the Bristol Bay region and its longest road system. Figure 13-2 illustrates how cumulative impacts from an initial mine, multiple subsequent mines, and induced development could result from the introduction of large-scale mining in the region.

Figure 13-3. Location of claim blocks in relation to subsistence use intensity for salmon, other fishes, wildlife, and waterfowl in the Nushagak and Kvichak River watersheds. See Box 5-2 for discussion of subsistence use methodology.



Infrastructure (i.e., roads, airports, ports, and utilities) is virtually absent from the mineralized areas of the Nushagak and Kvichak River watersheds. The transportation corridor, deep-water port, power generation facilities, and other infrastructure that would be required for mining the Pebble deposit likely would increase the economic feasibility of developing and operating other, smaller mines. Over the approximately 25-year life of the Pebble 2.0 scenario mine, and certainly over the approximately 78-year life of the Pebble 6.5 scenario mine, it is reasonably foreseeable that a number of prospects recently under active exploration (Pebble South/PEB, Big Chunk South, Big Chunk North, Groundhog, AUDN/Iliamna, and Humble) could be developed. Each of the mines would need road access to a port facility and airport. Some of the additional mines could use the assessment corridor, with only relatively short extensions, but more distant sites such as AUDN/Iliamna and Humble would require development of extensive additional pipeline, road, or railroad systems. The mines and transportation corridors described herein are not certain, but the roads are part of state planning documents—and a large-scale mine could easily be the trigger that starts this pattern of development in motion.

Mines at these sites would cause their own direct impacts, which would accumulate over a much greater portion of the Nushagak and Kvichak River watersheds and increase the number of distinct salmon populations affected. These effects could cumulatively threaten the biological complexity of the Nushagak-Kvichak salmon stocks and the portfolio effect, potentially contributing to salmon population declines. The genetic and life-history diversity within and among Bristol Bay salmon stocks will likely be critical for maintaining the resiliency of the population under a future environment characterized by climate change. Thus, the potential effects of additional mines on salmon genetic diversity could exacerbate the impacts of climate change on salmon populations in the watershed.

It is reasonably foreseeable that the infrastructure, particularly the transportation corridors, associated with large-scale mining could induce further development in the region. Existing communities, the tourism industry, and the recreational housing market could benefit if large-scale mining expanded through the watersheds. Unmanaged access to currently roadless wilderness areas also could expand. Improved access would increase hunting and fishing pressure, as well as competition with existing subsistence users; increase damage from off-road vehicle, boat, and foot traffic in currently inaccessible areas; facilitate poaching, dumping, trespassing, and other illegal activities; and lead to scattered development in the watersheds.



CHAPTER 14. INTEGRATED RISK CHARACTERIZATION

This chapter summarizes the risk analysis results, organized by assessment endpoint, for a potential mine at the Pebble deposit. 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 scenarios and from knowledge of analogous mining operations. Limitations and uncertainties in the risk characterization are also summarized. Finally, these results are extrapolated to the cumulative effects of multiple mines. See Chapters 7 through 13 for the derivation of these conclusions.

14.1 Overall Risk to Salmon and Other Fishes

14.1.1 Routine Operation

During routine operations, mining would be conducted according to modern conventional practices, including common mitigation measures at the mine site and along the transportation corridor. Toxic effects would be minimized by collection of nearly all water from the site and treatment of collected water to meet state standards and national criteria before discharge. However, toxic effects would still occur, primarily due to the inevitable leakage of leachates. In addition, habitat loss and modification would occur due to destruction of streams and wetlands and water withdrawals. As a result, local populations of salmonids would decline in abundance and production. Compensatory mitigation of these losses in the Bristol Bay watershed would be problematic at best (Appendix J).

14.1.1.1 Mine Footprint

Even in the absence of accidents or failures, the development of a mine at the Pebble deposit would result in the destruction or modification of streams, wetlands, and ponds. Local habitat loss would be significant, because losses of stream habitat leading to losses of local, unique populations would erode the population diversity key to the stability of the overall Bristol Bay salmon fishery (Schindler et al. 2010).

- In the Pebble 0.25, 2.0, and 6.5 scenarios, **38, 89, and 151 km of streams, respectively, would be lost** to (eliminated, blocked, or dewatered by) each mine footprint (the area covered by the mine pit, waste rock piles, tailings storage facilities [TSFs], drawdown zone, and plant and ancillary facilities). This translates to losses of 8, 22, and 36 km of streams known to provide spawning or rearing habitats for coho salmon, sockeye salmon, Chinook salmon, and Dolly Varden.
- Altered streamflow resulting from retention and discharge of water used in mine operations, ore processing, transport, and other processes would reduce the amount and quality of fish habitat. **Streamflow alterations exceeding 20% would adversely affect habitat in an additional 15, 27, and 53 km of streams** in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively, reducing production of coho salmon, sockeye salmon, Chinook salmon, rainbow trout, and Dolly Varden. Reduced streamflows would also result in the loss or alteration of an unquantified area of riparian floodplain wetland habitat due to loss of hydrologic connectivity with streams.
- Off-channel habitats for salmon and other fishes would be reduced due to **losses of 4.5, 12, and 18 km^2 of wetlands and 0.41, 0.93, and 1.8 km^2 of ponds and lakes** to the Pebble 0.25, 2.0, and 6.5 mine footprints, respectively. These losses would reduce availability of and access to hydraulically and thermally diverse habitats that provide foraging opportunities and important rearing habitats for juvenile salmon.
- **Indirect effects of stream and wetland losses** would include reductions in the quality of downstream habitat in the three headwater streams draining the mine footprints, affecting the same species as the direct effects. Modes of action for these effects would include the following.
    - A reduction in food resources would result from the loss of organic material and drifting invertebrates exported from the 38 to 151 km of streams lost to the mine footprints.
    - The balance of surface water and groundwater inputs to downstream reaches would change. Shifting from groundwater to surface-water sources is expected to reduce winter habitat (i.e., unfrozen stream reaches) and make streams less suitable for spawning and rearing.
    - Water treatment and discharge, resulting in reduced passage through groundwater flowpaths, are expected to alter summer and winter water temperatures and make streams less suitable for Pacific salmon, rainbow trout, and Dolly Varden.

These indirect effects on the abundance and production of salmonids cannot be quantified due to lack of data. However, it is expected that one or more of these mechanisms would diminish fish production downstream of the mine footprints in each watershed.

14.1.1.2 Water Collection, Treatment, and Discharge

Water in contact with tailings, waste rock, or the pit walls would leach copper and other metals. Our assessment evaluates the discharge of treated wastewater and the realistic expectation that leachate would escape the waste rock pile and TSF water collection systems in the three mine size scenarios. Routine discharges from the wastewater treatment plant (WWTP) to the South and North Fork Koktuli Rivers should be non-toxic due to treatment to achieve permit requirements. However, they may be somewhat toxic due to combined effects of multiple chemicals, poorly known and unregulated contaminants, and untested species in the receiving waters.

The retention and collection of leachates are inevitably incomplete. In our routine operations scenario, leakage in the Pebble 2.0 and Pebble 6.5 scenarios would be sufficient to cause toxic levels of copper and, to a much lesser extent, other metals in the streams draining the mine footprints. The most severe effects, including death of salmonids, would occur in the South Fork Koktuli River, which would receive leachate from the acid-generating waste rock. Upper Talarik Creek would experience death of invertebrates only below the station at which it receives interbasin transfer from the South Fork Koktuli River. The North Fork Koktuli River would experience death of invertebrates below TSF 1. Death or inhibited reproduction of aquatic invertebrates, which are food for fish, is estimated to occur in 21, 40 to 62, and 60 to 82 km of streams in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively. Avoidance of streams by salmonids would occur in 24 and 34 to 57 km of streams in the Pebble 2.0 and Pebble 6.5 scenarios, respectively. Death or reduced reproduction of salmonids would occur in 3.8 and 12 km of streams in the Pebble 2.0 and Pebble 6.5 scenarios, respectively.

The magnitude and extent of these predicted effects suggest the need for mitigation measures beyond the conventional practices assumed in the routine operations scenario to reduce the input of leached copper and other metals. A design based on conventional practices may be sufficient for a typical porphyry copper mine (i.e., equivalent to the Pebble 0.25 scenario), but not the massive Pebble 2.0 and 6.5 scenarios. Simply improving the efficiency of the capture wells or adding a larger wall or trench is unlikely to achieve water quality criteria in those scenarios. Additional measures might include lining the waste rock piles, reconfiguring the piles, or processing the acid-generating waste rock as it is produced.

In the event of TSF 1 overfilling, supernatant water would be released via a spillway. If the water was equivalent to the test tailings supernatant, 2.6 km of stream would be avoided by fish and 3.4 to 23 km would be toxic to invertebrates, independent of other sources.

14.1.1.3 Road Construction and Operation

The assessment's transportation corridor, including a road and four pipelines, would cross approximately 64 streams and rivers, of which 55 are known or likely to support migrating and resident salmonids. Nearly 272 km of streams between the road and Iliamna Lake would be affected. Risks to salmonids from the construction and operation of the transportation corridor are as follows.

- Loss and alteration of habitat through filling of wetlands for the road.

- Increased suspended and deposited sediment washed from the road, shoulders, ditches, cuts, and fills.
- Increased stormwater runoff leading to increased suspended sediment, fine-bed sediment, salts, and, at the mine site, metals.
- Increased dust leading to a direct increase in fine-bed sediment in the mining area, and an indirect increase along the entire transportation corridor via reduced riparian vegetation.
- Possible introduction of invasive species, particularly plants and fish pathogens.

All of the above sources and stressors would likely lead to degraded or reduced habitat for salmon and other fish.

14.1.2 Accidents and Failures

Any complex activity such as the mine described in this assessment inevitably experiences accidents and failures. The number of ways in which failures and accidents can occur—their magnitudes, their locations, and the circumstances of their occurrence—are effectively infinite. Hence, a complete and specific assessment of risks from potential accidents and failures is not possible. Rather, a few failure scenarios are presented, which emphasize the consequences of failures rather than the means by which they are initiated. These scenarios address potential failures that could occur during mine operations or after mine closure in perpetuity: failure to treat contaminated water; tailings dam failure; failures of roads and culverts; wreck of a truck carrying a process chemical; and failure of diesel, product concentrate, or return water pipelines. Many other potential failures are not analyzed, including failures of the on-site pipelines, spills of ore-processing chemicals on site, failures of tailings dams on streams other than the North Fork Koktuli River, wildfires, waste rock slides, or failures at the port.

The probabilities and consequences of the failures analyzed in the assessment are summarized in Table 14-1. The derivation of these estimates is discussed in Box 14-1, and the interpretation of failure probabilities is discussed in Box 9-3. Probabilities of occurrence were estimated using the best available information. Some estimates are qualitative, because no applicable data are available. Those that are quantitative are somewhat uncertain and their interpretation is not straightforward. For example, the range of annual probabilities of a tailings dam failure is based on design expectations rather than actual performance data, which are unavailable for recently constructed large earthen dams. The actual observed frequency of tailings dam failures is near the upper end of that range, which suggests that the range is reasonable at that bound. However, the lower bound (1 in 250,000 per year) is purely aspirational, in that it has no empirical basis.

Table 14-1. Probabilities and consequences of potential failures in the mine scenarios.

Failure Type	Probability[a]	Consequences
Tailings dam	4×10^{-4} to 4×10^{-6} per dam-year = recurrence frequency of 2,500 to 250,000 years[b]	More than 29 km of salmonid stream would be destroyed or degraded for decades.
Product concentrate pipeline	10^{-3} per km-year = 95% 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	1.5×10^{-2} per year = 1 stream-contaminating spill 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	2.6×10^{-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 spill	Same as product concentrate pipeline	Fish and invertebrates would experience acute exposure to toxic water if return water spilled to a stream or wetland.
Diesel pipeline spill	Same as product concentrate pipeline	Acute toxicity would reduce the abundance and diversity of invertebrates and possibly cause a fish kill if diesel spilled to a stream or wetland.
Culvert, operation	Low	Frequent inspections and regular maintenance would result in few impassable culverts, but for those few, blockage of migration could persist for a migration period, particularly for juvenile fish.
Culvert, post-operation	3×10^{-1} to ~6×10^{-1} per culvert; instantaneous = 11 to 22 culverts	In surveys of road culverts, 30 to 61% are impassable to fish at any one time. This would result in 11 to 22 salmonid streams blocked at any one time. In 10 to 19 of the 32 culverted streams with restricted upstream habitat, salmon spawning may fail or be reduced and the streams would likely not be able to support long-term populations of resident species.
Truck accidents	1.9×10^{-7} spills per mile of travel = 4 accidents in 25 years and 2 near-stream spills in 78 years	Accidents that spill processing chemicals into a stream or wetland could cause a fish kill. A spill of molybdenum concentrate may also be toxic.
Water collection and treatment, operation	0.93 = proportion of recent U.S. porphyry copper mines with reportable water collection and treatment failures	Water collection and treatment failures could result in exceedance of standards potentially including death of fish and invertebrates. However, these failures would not necessarily be as severe or extensive as estimated in the failure scenario, which would result in toxic effects from copper in more than 60 km of stream habitat.
Tailings storage facility spillway release	No data, but spills are known to occur and are sufficiently frequent to justify routine spillway construction	Spilled supernatant from the tailings storage facility could result in toxicity to invertebrates and fish avoidance for the duration of the event.
Water collection and treatment, managed post-closure	Somewhat higher than operation	Post-closure collection and treatment failures are very likely to result in release of untreated or incompletely treated leachates for days to months, but the water would be less toxic due to elimination of potentially acid-generating waste rock.
Water collection and treatment, after site abandonment	Certain, by definition	When water is no longer managed, untreated leachates would flow to the streams. However, the water may be less toxic.

[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 5×10^{-4} per year or a recurrence frequency of 2,000 years).

BOX 14-1. FAILURE PROBABILITIES

Table 14-1 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, but they do convey the likelihood of occurrence. More details can be found in Chapters 8 through 11.

Tailings dam failure. 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 and the Federal Energy Regulatory Commission. Both regulatory agencies require a minimum factor of safety of 1.5 for the loading condition corresponding to steady seepage at 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 would perform as expected. Slope instability is only one type of failure; other failure modes, such as overtopping during a flood, would increase overall failure rates. Slope stability failures account for about one-fourth of tailings dam failures, so the probability of failure from all causes could be estimated to be 1 in 250,000 (Category I) to 1 in 2,500 (Category II). The mine scenarios include up to three tailings storage facilities (TSFs), two with multiple dams, so the annual probability of any dam failing would be approximately equal to the annual probability of a single dam failure times the number of dams.

Pipeline failure. 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 year. 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 113-km length of each pipeline within the Kvichak River watershed, results in a 0.11 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 0.14 probability of entering a stream within the Kvichak River watershed. This would result in an estimate of 0.015 stream-contaminating spills per year or 1 stream-contaminating spill over the duration of the Pebble 6.5 scenario (approximately 78 years). Similarly, a spill would have a 0.24 probability of entering a wetland, resulting in an estimate of 0.026 wetland-contaminating spills per year or 2 wetland-contaminating spills over the duration of the Pebble 6.5 scenario.

Water collection and treatment failure. During mine operation, collection or treatment of leachate from mine tailings, pit walls, or waste rock piles would be incomplete and could fail in various ways. In the routine operations scenario, leachate from the unlined TSFs and waste rock piles would not be fully collected. Equipment and operation failures and inadequate designs would also result in failures to avoid toxic emissions. Reviews of mine records found that 93% of operating porphyry copper mines in the United States reported a water collection or treatment failure (Earthworks 2012). Improved design and practices should result in lower failure rates, but given this record it is unlikely that failure rates would be lower than 10% over the life of a mine. During operation, failures should be brief (less than 1 week) unless they involve a faulty system design or parts that are difficult to replace. After a mine is abandoned (potentially many years after closure), water management would end and the discharge of untreated water would become inevitable but may not be problematic.

TSF spillway release. Releases of supernatant water from TSFs through spillways are unintended but are not uncommon (e.g., the release at Nixon Fork Mine described in Box 8-1). However, data on the frequency of such releases are unavailable. They are apparently sufficiently common that inclusion of a spillway in a tailings dam is a standard practice. Hence, it is judged likely that a release would occur over the 78-year life of the mine in the Pebble 6.5 scenario.

Culvert failure. 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 0.61 (3 to 6×10^{-1}) per culvert. In the Kvichak River watershed, 35 streams that are believed to support salmonids (salmon, trout, or Dolly Varden) have culverts, so at any time 11 to 22 culverted streams would be expected to have blocked fish passage at the published frequencies. The proportion of failed culverts during mine operation should be much lower.

It is important to remember that this is an assessment of mine scenarios. It is based on modern conventional mining practices, especially the plan proposed for the Pebble site by Northern Dynasty Minerals (Ghaffari et al. 2011). However, like any predictive assessment, it is hypothetical. Although the major features of the scenarios will undoubtedly be correct (e.g., a pit at the location of the ore body, waste rock deposited near the pit, the generation of a large volume of tailings), some specifics would inevitably differ. This would be true of any scenario, including a mining plan submitted for permitting or even a plan approved by the state. All plans are scenarios, and although each new plan is expected to be closer to actual operations than the ones before, unforeseen circumstances and events and new technologies inevitably compel changes in practice.

14.1.2.1 Tailings Dam Failure

Failure of a tailings dam would have a one in 2,500 to one in 250,000 probability of occurrence per year for each TSF. Probability of a tailings dam failure increases with an increase in the number of dams. The Pebble 0.25 and Pebble 2.0 scenarios include one TSF, and the Pebble 6.5 scenario includes three. Two of these TSFs would have multiple dams. However, the probability of a spill from these TSFs would not increase in proportion to the number of dams for an individual TSF, because failures would not be independent events. The failure of one dam on a TSF would relieve pressure on others, reducing the probability of multiple failures; conversely, common mode failures could occur, increasing the probability of multiple failures. The dam failure analyses in this assessment simulated the release of 20% of the tailings (a conservative estimate) from the failure of a 92-m (Pebble 0.25) and a 209-m (Pebble 2.0) 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 salmonid habitat in the North Fork Koktuli River (29 km of habitat in the Pebble 0.25 scenario and more than 30 km, our model limit, in the Pebble 2.0 scenario) in the short-term (less than 10 years). The high likelihood of very low-quality spawning and rearing habitat in the long-term (decades) would result in the nearly complete loss of mainstem North Fork Koktuli River fish populations below the dam. 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 farther down the Koktuli River, and probably into the Mulchatna and Nushagak Rivers, causing degraded spawning habitat and reduced food resources. Fish anywhere in the flowpath below a tailings dam failure would be killed or forced downstream. Fish migrating into tributaries of affected rivers would be inhibited 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 resuspended 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 streamflow would be expected to suspend sufficient concentrations of tailings to cause avoidance, reduced growth and fecundity, and possibly even death of fish. Migration to and from any affected tributaries would be impeded if streamflow from the tributaries was not sufficient to adequately dilute suspended sediment concentrations, meaning that fish would not reach spawning grounds, winter refugia, or seasonal feeding habitats.

Deposited tailings would degrade habitat quality for both fish and the invertebrates they eat. Pacific salmon, Dolly Varden, and rainbow 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 detectable 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 was 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 affect much of the watershed downstream of the failed dam.

In addition to degrading fish habitat, deposited tailings would be potentially toxic. Based largely on their copper content, deposited tailings would be toxic to benthic macroinvertebrates, although existing data concerning fish toxicity is less clear. Estimated pore water concentrations are below 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 salmonids 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 salmonids. 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 salmonids. 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 River fish populations. Deposition, resuspension, and redeposition 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 portion of the tailings into the Nushagak River. Effects would be qualitatively the same for both the Pebble 0.25 and Pebble 2.0 dam failures, although effects from the Pebble 2.0 dam failure would extend farther 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 average annual escapement of nearly 190,000 Chinook salmon from 2002 through 2011 (Buck et al. 2012). Assuming Alaska Department of Fish and Game 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 29% of that run due to loss of the Koktuli River salmon population; 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 annual runs averaging more than 1.9 million fish. However, the proportion of sockeye and other salmon species that originates in the Koktuli and Mulchatna River watersheds is unknown. Similarly, populations of rainbow trout and Dolly Varden of unknown size would be lost for decades.

Remediation of a tailings spill would be difficult and problematic. The affected area is roadless, and the rivers are too small to float a dredge. If the spill occurred after mine closure, people and equipment to repair the dam and begin remediation would be absent. Remediation may be slow to start due to the need to develop a plan, create a facility to receive the recovered tailings, build roads, and bring in personnel and equipment. Even in the Pebble 0.25 dam failure, complete removal of this material would require a substantial earth-moving effort, including over 3 million round trips by 20-ton dump trucks. Dredging tailings from rivers and streams would cause considerable habitat damage.

The dam failures evaluated in the assessment used TSF 1 as a plausible location. Failure of the other tailings dams at TSF 2 and TSF 3 were not modeled, but would have similar types of effects in the South Fork Koktuli River and downstream.

14.1.2.2 Wastewater Treatment Plant Failure

In the WWTP failure scenario, untreated wastewater would be discharged. The most severe effects, including lethality to invertebrates and fish, would occur in the South Fork Koktuli River where untreated effluent would mix with toxic waste rock leachate. The North Fork Koktuli River, where the untreated waste would mix with tailings leachate, would experience lethality to invertebrates and, depending on the season, reduced growth or survival of early fish life stages. In this scenario, Upper Talarik Creek would receive no wastewater discharge and would experience no additional effects. The WWTP failure is estimated to result in lethality or reduced reproduction of invertebrates in 78 to 100 km of streams in all three mine sizes. For salmonids, it is estimated to cause avoidance of 74 to 97 km of streams, sensory inhibition in 70 to 92 km, reduced reproduction in 61 to 84 km, and mortality in 31 km in the Pebble 6.5 scenario. Direct effects on fish would be less extensive in the Pebble 2.0 scenario, with avoidance in 64 to 87 km, sensory inhibition in 27 km, reduced reproduction in 11 km, and kills in 3.8 km—and would be limited to avoidance in 27 km of streams in the Pebble 0.25 scenario.

14.1.2.3 Culvert Failure

The most likely serious failure associated with the potential transportation corridor would be blockage or failure of culverts. Culverts commonly fail to allow fish passage. They can become blocked by debris or ice that may not stop water flow but that create a barrier to fish movement. Fish passage also may be blocked or inhibited by erosion below a culvert that “perches” the culvert and creates a waterfall, by shallow water caused by a wide culvert and periodic low streamflows, or by excessively high channel gradients. If blockages occurred during adult salmon immigration or juvenile salmon emigration and were not cleared for several days, production of a year-class (i.e., fish spawned in the same year) would be lost from or diminished in the stream above the culvert.

Culverts can also fail to convey water due to landslides or, more commonly, floods that wash out undersized or improperly installed culverts. In such failures, the stream would be temporarily impassible to fish until the culvert is repaired or until erosion re-establishes the channel. If the failure occurs during a critical period in salmon migration, 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 sediment. This could cause returning salmonids 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 salmonid eggs and larvae, if they were present, and would degrade the downstream habitat for salmonids and the invertebrates that they eat. It would also change stream hydraulics and channel morphology, generally diminishing habitat value.

Blockage of fish passage at road crossings would be infrequent during operation, because our scenarios assume daily inspection and maintenance. However, after mine operations end, the road may be maintained less carefully or maintenance may be transferred to a state or a local 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 61% inhibit fish passage at any time) (Langill and Zamora 2002, Gibson et al. 2005, Price et al. 2010). Of the 45 culverts that would be required, 36 would be on streams that are believed to support salmonids. Hence, 11 to 22 streams would be expected to lose passage of salmon or resident trout or Dolly Varden and some proportion of those would have degraded downstream habitat resulting from sedimentation caused by road washout.

Of the 36 culverted salmonid streams, 32 contain restricted (less than 5.5 km) upstream habitat. Assuming typical maintenance practices after mine operations, approximately 10 to 19 of the 32 streams would be entirely or partially blocked at any time. As a result, isolation of resident species such as rainbow trout or Dolly Varden in such short stream segments would likely result in failure of the populations, if that isolation was sustained.

It should be noted that high streamflows in and immediately downstream of a culvert and the structure of the culvert may inhibit fish passage even if movement is not blocked. Culvert-induced erosion could cause channel entrenchment, disrupting floodplain habitat and floodplain/channel ecosystem processes.

14.1.2.4 Truck Accidents

Trucks would carry ore-processing chemicals to the mine site and molybdenum product concentrate to the port. Truck accident records indicate that truck accidents near streams are likely over the long period of mine operation. These accidents could release sodium ethyl xanthate, cyanide, other process chemicals, or molybdenum product concentrate to streams or wetlands, resulting in toxic effects on invertebrates and fish. However, the risk of spills could be mitigated by using impact-resistant containers.

14.1.2.5 Pipeline Failure

The primary product of the mine would be a copper concentrate that would be pumped as a slurry in a pipeline to a Cook Inlet shipping facility. 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 metal concentrate pipelines in particular, one near-stream failure and two near-wetland failures of each of these pipelines would be expected to occur over the duration of the Pebble 6.5 scenario (approximately 78 years). In either case, metal-contaminated water would be released, potentially killing fish and invertebrates in the affected stream over a relatively brief period. The aqueous phase of the concentrate slurry would be lethal to sensitive invertebrates and potentially to fish larvae, but a kill of adult fish is not expected. If the concentrate pipeline spilled into a stream, concentrate would, depending on streamflows, settle and form bed sediment, be carried downstream and deposited in low-velocity areas, or be carried to Iliamna Lake and deposited near the shore. Deposited concentrate is predicted to be highly toxic based on its high copper content and the acidity of its leachate. Unless the receiving stream was dredged, causing physical damage, this sediment would persist for decades before ultimately being washed into Iliamna Lake. Potential concentrations in the lake could not be predicted; however, near the pipeline route, Iliamna Lake contains important beach spawning areas for sockeye salmon that could be exposed to a spill. Sockeye also spawn in the lower reaches of streams that could be directly contaminated by a spill.

Spills from a diesel pipeline are estimated to have the same probability of occurrence as concentrate spills. Based on multiple lines of evidence, a spill in the diesel pipeline failure scenario would be sufficient to kill invertebrates and possibly fish. Remediation is expected to have little success, but recovery would likely occur within 3 years.

14.1.2.6 Common Mode Failures

Multiple failures could result from a common event, such as an earthquake or a severe storm with heavy precipitation (particularly heavy rain on snow). Failures resulting from such an event could include multiple 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 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 the tailings, mine pit, road, and waste rock piles 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.

14.2 Overall Loss of Wetlands, Ponds, and Lakes

Wetlands are a dominant feature of the landscape in the Pebble deposit area and are important habitats for salmon and other fish. Ponds and riparian wetlands provide spawning, rearing, and refuge habitat for both anadromous and resident fish. Other wetlands moderate streamflows and water quality, and can influence downstream delivery of dissolved organic matter, particulate organic matter, and aquatic macroinvertebrates that supply energy sources to fish. In the Pebble 0.25, 2.5, and 6.5 scenarios, 4.5, 12, and 18 km^2 of wetlands and 0.41, 0.93, and 1.8 km^2 of ponds and lakes, respectively, would be filled or excavated. 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 0.11 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 influence approximately 4.7 km^2 of wetlands, ponds, and lakes occurring within 200 m of the roadbed. Finally, a diesel or product concentrate spill could damage wetlands and eliminate or degrade their capacity to support fish.

14.3 Overall Fish-Mediated Risk to Wildlife

Interactions between salmon and wildlife and the potential for disruption of these interactions are complex. Annual salmon runs provide food for brown bears, bald eagles, other land birds, and wolves. In addition, wildlife abundance and production are enhanced by the marine-derived 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 nutrients on the landscape, fertilizing the vegetation and increasing the abundance and production of moose, caribou, and other wildlife.

The effects of reduced Pacific salmon, Dolly Varden, and rainbow trout production on wildlife would be complex, may not be linearly proportional, and cannot be quantified at this time. Factors such as the magnitude, seasonality, duration, and location of salmon losses would determine the specific species affected and the magnitude of effects. However, some degree of reduction in wildlife would be expected due to the mine footprint and routine operations in each mine size scenario. Because salmon provide a food source for brown bears, wolves, bald eagles, and other birds, it is likely these species would be directly affected by a reduction in salmon abundance. Indirect effects on water birds and land birds through a loss of aquatic invertebrates and on moose and caribou through a loss of marine-derived nutrients to vegetation are likely, but research is needed to document those linkages.

Fish-eating wildlife species are also potentially exposed to contaminants bioaccumulated by fish. However, analyses based on the concentrations of metals in waste rock, tailings, and product concentrate leachates suggest that toxic effects to wildlife via this route of exposure are unlikely.

14.4 Overall Fish-Mediated Risk to Alaska Native Cultures

Alaska Natives are particularly vulnerable to any changes in the quantity or quality of wild salmon resources, due to the importance of salmon in terms of both subsistence and cultural identity. Any change in salmon resources would likely change the diet, social networks, cultural cohesion, and spiritual well-being of the Alaska Native cultures in the region. These changes could, in turn, result in the following.

- Effects on human health from loss of a highly nutritious subsistence food and the physical and mental benefits of a subsistence way of life.
- Degradation of a social support system based on food sharing.
- Decrease in family cohesion and cultural continuity from a loss of family-based subsistence work.
- Mental health degradation from the disruption of spiritual practices and beliefs centered on salmon and clean water.

Human health and cultural effects related to decreases in salmon resources would vary with the magnitude of these reductions and cannot be predicted quantitatively. Some fish-mediated effects on Alaska Native cultures are likely due to the mine footprint or routine operations in any of the mine size scenarios considered. At minimum, there would be a loss of subsistence use areas and the risk of decreased use of fish because of a perceived change in quality of the fish due to mine operations. Along the transportation corridor, complex and unpredictable changes to subsistence use would result from increased access (by both Alaska Natives and others) and possible habitat changes. If significant failures of water treatment or other infrastructure that greatly affect salmon resources occur during or after mine operation, large-scale impacts on both subsistence food resources and the cultural, social, and spiritual cohesion of the local indigenous cultures would occur.

Because the Alaska Native cultures in the Bristol Bay watershed have significant ties to specific land and water resources that have evolved over thousands of years, it is not possible to replace the value of any subsistence use areas lost to mine operations elsewhere. As a result, compensatory mitigation, restoration, or replacement in the case of a failure would be difficult, if not impossible.

It should be noted that, although this assessment focuses on potential effects on Alaska Native cultures, many of the non-Alaska Natives that reside in the area also practice a subsistence way of life and have strong long-term cultural ties to the landscape that go back generations. In addition, a large group of seasonal commercial fishers and cannery workers depend on these resources and have strong, multi-generational cultural connections to the region. These groups also would be vulnerable to negative impacts on salmon.

14.5 Summary of Uncertainties and Limitations in the Assessment

This assessment makes various reasonable assumptions about the mining, processing, and transporting of the porphyry copper resources in the Pebble deposit and elsewhere in the Nushagak and Kvichak River watersheds. If those resources are mined in the future, actual events would not be identical to the mine scenarios considered here. This is not treated as a source of uncertainty, because it 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.

As discussed in the individual chapters, this assessment does have uncertainties and limitations in the extent to which the potential effects of these scenarios can be estimated. Major 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 range of probabilities.
- 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 30-km limit of the model in the Pebble 2.0 dam failure scenario.
- The ultimate fate of spilled and deposited tailings in the event of a dam failure could not be quantified. From principles of geohydrology and review of analogous cases, we know that slurry would erode from areas of initial deposition and move downstream over more than a decade. However, the data needed to model that process and the resources to develop the model are not currently available.
- It is uncertain whether and how a tailings spill into a remote roadless area would be remediated, how long it would take to remediate, and to what extent remediation could reduce effects downstream of the initial slurry runout.
- The effects of mining on fish populations could not be quantified because of the lack of quantitative information concerning Pacific salmon, Dolly Varden, and rainbow trout populations and their responses. The occurrence of salmonid species in the region's 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. Salmon populations naturally vary in size because of a great many factors that vary among locations and years, and collecting sufficient data to establish reliable

salmon population estimates takes many years. Thus, we used estimated effects of mining on habitat as a reasonable 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 on their floodplains, and waste rocks 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, alevins, or sheltering fry 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 evidence suggest 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 salmonids and consequent indirect effects of diminished fish resources on wildlife and people. Direct effects of mining on humans, wildlife, and terrestrial ecosystems, as well as induced development associated with mine-related activities, are not evaluated in this assessment.
- Some sources, such as air pollution from a power plant, were not addressed because they are less related to the Clean Water Act or because they were judged to pose less risk to salmonids.
- Climate change will affect both the probability and magnitude of mine-related failures, as well as change the habitat quality and biology of salmonids. These climate effects are highly uncertain, but their likely qualitative influences are described in Box 14-2.

BOX 14-2. CLIMATE CHANGE AND POTENTIAL RISKS OF LARGE-SCALE MINING

Climate change in the Bristol Bay region (Section 3.8) will likely result in changes in snowpack and the timing of snowmelt, a greater chance for rain-on-snow events, and an increase in flooding. These changes are likely to affect multiple aspects of any large-scale mining in the area, including mine infrastructure, the transportation corridor, water treatment and discharge, and post-closure management (Pearce et al. 2011).

Mine infrastructure (e.g., buildings, waste rock piles, tailings storage facilities [TSFs], and water retention facilities) and the transportation corridor likely would be affected by extreme weather events resulting in increased flooding (Instanes et al. 2005, Pearce et al. 2011). These components would need to be designed for potential increases in flood frequency and magnitude and changes in storm patterns, because these changes could weaken structural integrity, increase embankment instability, and accelerate erosion (Instanes et al. 2005, Pearce et al. 2011).

Water management would be a major challenge at the mine site, and changes resulting from climate change could exacerbate the challenge. Climate change would contribute to future changes in temperature, precipitation, evapotranspiration, hydrology, and seasonal flooding and drying patterns. Changes in water availability and groundwater recharge would affect the amount and timing of water available and the hydrologic gradients of groundwater in and around the mine site, thereby requiring changes to water management in the mine pit and other areas of the mine site.

Under future climate conditions, the return intervals of various sized storms could change (e.g., medium-sized storms could become more frequent or the frequency of current 100-year storms could change). Possible increases in flood magnitudes would require the need to plan for larger and more frequent flood events at the mine site. This in turn may affect the likelihood of a tailings dam failure, overtopping of ponds, and/or flooding of water management facilities. Failure to plan for these conditions could result in unintended environmental releases. For example, Minto Mine, a copper-gold mine in Canada, was forced to release untreated water into the Yukon River system in 2008, due to torrential rains and the mine's inability to manage this increased water (Pearce et al. 2011).

Mine infrastructure would need to be designed to account for projected climatic changes, such that its structural integrity can be maintained in perpetuity, even under potentially more extreme climatic conditions. In addition, any mine reclamation plan would need to consider changing climate conditions and how those changes will directly affect fish and wildlife populations. The following list includes infrastructure and operations design, maintenance, and management that would need to consider climate change.

- **Mine footprint.** Climate change may affect water availability both within and across seasons (e.g., via changes in snowmelt patterns; amount, type, and timing of precipitation; frequency of large storms; and groundwater inputs). Water processing associated with these changes would alter flow and temperature in downstream water bodies.
- **Water treatment and discharge.** Climate change might result in greater volumes of water requiring treatment, changes in the dilution provided by receiving streams, changes in temperature that would affect management of discharged water, and potential overload due to lack of storage and treatment capacity.
- **TSF failure.** TSFs may be exposed to greater volumes of water, and the probability of dam failure due to overtopping may increase with changes in precipitation patterns and/or rapid snowmelt.
- **Transportation corridor.** Greater flood frequencies and an increase in erosion and sedimentation are likely to affect streams and wetlands along the transportation corridor.
- **Culvert, pipeline, and bridge failures.** These failures may be more likely due to changes in precipitation patterns, rapid snowmelt, larger water volumes, debris issues, and sedimentation.
- **Cumulative effects of multiple mines.** Climate change issues are likely to affect any mining operation in the area. As the number of mines increases, the likelihood of having mine and climate change interactions might increase.

Climate change could obscure or complicate efforts to monitor habitat and fish population responses to mine-related activities. Survey and monitoring designs would need to take potential climate change effects into account through strategic measurement of stream and lake temperatures, precipitation, water flow, and fish populations throughout the Bristol Bay watershed. Monitoring design could be aided by models able to downscale climate effects and project changes at watershed scales. Population monitoring should take salmon adaptation and metapopulation dynamics into account, and be cognizant of the many interacting processes influencing populations. Protecting salmon sustainability in an uncertain future will require adaptability of both management and monitoring strategies (Schindler et al. 2008).

14.6 Summary of Uncertainties in Mine Design and Operation

In addition to uncertainties in the assessment, uncertainties are inherent in planning, designing, constructing, operating, and closing a mine. Such uncertainties are inherent in any complex enterprise, particularly when it involves an incompletely characterized natural system. However, the large scales and long durations of any effort to mine the Pebble deposit make these inherent uncertainties more prominent.

- Mines are complex systems requiring skilled engineering, design, and operation. The uncertainties facing mining and geotechnical engineers include unknown geological features, uncertain values in geological properties, limited knowledge of mechanisms and processes at the site, and human error in design, construction, and operation. 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 reduce system failures, seemingly logical decisions about how to respond to a given situation can have unexpected consequences resulting from human error—for example, the January 2012 overtopping of the tailings dam at the Nixon Fork Mine near McGrath, Alaska (Box 8-1). 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 wastes would require management for centuries or even in perpetuity. Engineered mine waste storage systems have only been in existence for about 50 years, so their long-term behavior is not known. The performance of modern technology in the construction of tailings dams is untested and unknown in the face of centuries of extreme events such as earthquakes and major storms.
- 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). Similarly, governments that are expected to assume responsibility when mining companies fail may not appropriately manage mine sites or the funds in performance bonds.

14.7 Summary of Risks in the Mine Scenarios

Even if the mining and mitigation practices described in the mine scenarios were performed perfectly, an operation of this size would inevitably destroy or degrade habitat of salmonids. The mine scenario footprints would eliminate, block, or dewater streams known to support spawning and rearing habitat for coho, Chinook, and sockeye salmon and Dolly Varden (Table 14-2). Wetlands would be filled or excavated in 4.5, 12, and 18 km^2 of the mine footprints in the Pebble 0.25, 2.0, and 6.5 scenarios, respectively; an additional 0.41, 0.93, and 1.8 km^2 of ponds and lakes would also be lost. Altered streamflows resulting from water use would significantly degrade additional stream reaches (Table 14-2) and an unquantifiable area of wetland habitat. Leachates and other wastewater would be collected and treated to meet standards, but leakage would be sufficient to cause direct toxic effects to fish in up to 57 km of streams and indirect effects due to loss of invertebrate food species in up to 82 km of streams (Table 14-2). In addition, the temperature and distribution of effluents could further degrade habitat. Streams between the transportation corridor and Iliamna Lake would receive silt and deicing chemicals, which would reduce habitat quality.

Table 14-2. Summary of estimated stream lengths potentially affected in the three mine size scenarios, assuming routine operations.

Effect	Stream Length Affected (km)		
	Pebble 0.25	Pebble 2.0	Pebble 6.5
Eliminated, blocked, or dewatered	38	89	151
Eliminated, blocked, or dewatered—anadromous	8	22	36
>20% flow alteration[a]	15	27	53
Direct toxicity to fish[a]	0	24	34–57
Direct toxicity to invertebrates[a]	21	40–62	60–82
Downstream of transportation corridor	272		

Notes:
[a] Stream reaches with streamflow alterations partially overlap those with toxicity.

This assessment considered failures of a tailings dam; product concentrate, return water, and diesel pipelines; roads and culverts; and water collection and treatment. Tailings dam failures are improbable in that they have a low rate of occurrence, but some sort of failure becomes likely in the extremely long-term. A tailings dam failure could destroy salmonid habitat in more than 30 km of the North Fork Koktuli River and associated wetlands for years to decades. Product concentrate and diesel pipeline failures near streams would be expected to occur during the life of a mine. Both would cause acute lethal effects on invertebrates and fish, and the concentrate could create highly toxic sediment. A truck wreck near a stream could introduce highly toxic chemicals causing acute lethality to fish and invertebrates. Culvert failures would be common, unless a more rigorous than usual maintenance program were maintained, and could block fish passage and 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 highly toxic to fish and invertebrates.

14.8 Summary of Cumulative Risks of Multiple Mines

To provide realism and detail, this assessment largely addresses the potential effects of a single mine, at three different sizes, on the Pebble deposit. However, the development of multiple mines of various sizes is plausible in the Nushagak and Kvichak River watersheds. Several known mineral deposits with potentially significant resources are located in the two watersheds, and active exploration is underway at a number of claim 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 fish populations, wildlife, and Alaska Native villages distributed across these watersheds.

Outside of the Bristol Bay watershed, 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 regional-scale effects, because they require suitable habitat in spawning areas, rearing areas, and along 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 these species. 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 and Kvichak River watersheds are relatively undisturbed, and their ecosystems have not yet experienced these cumulative stresses associated with human activity. 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 of a discrete portion of habitat, but these fluctuations are compensated for 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 fish populations. Such effects may remove component populations permanently or for long periods of time, weakening the overall population's ability to resist and rebound from disturbance.

To examine the potential cumulative risks of multiple mines, we consider development of additional mines at the Pebble South/PEB, Big Chunk South, Big Chunk North, Groundhog, AUDN/Iliamna, and Humble prospects. The AUDN/Iliamna and Humble prospects are located approximately 90 and 135 km, respectively, southwest of the Pebble deposit. All of the other prospects are within 25 km of the Pebble deposit and may be of the same geological 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.

Impacts from the footprint of major mine components and associated accidents and failures would be similar to those projected in the mine scenarios. The footprints of the major mine components would eliminate substantial amounts of stream and wetland habitats, both directly and through dewatering. Total stream length eliminated by these components would range from 43 to 70 km, and wetland area lost would range from 7.9 to 27 km^2. Further habitat loss and degradation would result from flow alteration. Each additional mine would increase flow alteration from water removal and retention, increased impervious surface, and road crossings.

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-generating and non-acid-generating, some of the waste rock and a portion of the tailings at any of these additional mines would be reasonably likely to 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 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. Extreme natural events such as earthquakes and floods could cause failures of dams, roads, pipelines, or WWTPs at multiple mines.

Induced development is that which results from the introduction of industry, roads, and infrastructure. It is reasonably foreseeable that infrastructure from large-scale mining in the Nushagak and Kvichak River watersheds, particularly the transportation corridors, would induce further development in the region. Existing communities, the tourism industry, and the recreational housing market could benefit if large-scale mining expanded throughout the watersheds. Unmanaged access to currently roadless wilderness areas also could expand. Improved access would increase hunting and fishing pressure, as well as competition with existing subsistence users; increase damage from off-road vehicle, boat, and foot traffic in currently inaccessible areas; facilitate poaching, dumping, trespassing, and other activities; and lead to scattered development in the watersheds.



CHAPTER 15. REFERENCES

15.1 References by Chapter

15.1.1 Executive Summary

ADF&G (Alaska Department of Fish and Game). 2012. *Alaska Freshwater Fish Inventory Database*. Available: http://www.adfg.alaska.gov/index.cfm?adfg=ffinventory.main.

Johnson, J., and P. Blanche. 2012. *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region, Effective June 1, 2012*. Special Publication No. 12-08. Anchorage, AK: Alaska Department of Fish and Game.

Ruggerone, G. T., R. M. Peterman, and B. Dorner. 2010. Magnitude and trends in abundance of hatchery and wild pink salmon, chum salmon, and sockeye salmon in the North Pacific Ocean. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science* 2:306–328.

USFWS (U.S. Fish and Wildlife Service). 2012. *National Wetlands Inventory*. Available: http://www.fws.gov/wetlands/Wetlands-Mapper.html. Accessed: December 11, 2012.

USGS (U.S. Geological Survey). 2012. *National Hydrography Dataset.* High Resolution, Alaska. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

Personal Communications

Baker, T. Area Fishery Research Biologist, Bristol Bay Salmon Program. Alaska Department of Fish and Game, Anchorage, AK. July 8, 2011—email to Rebecca Shaftel.

15.1.2 Chapter 1—Introduction

Chambers, D., R. Moran, and L. Trasky. 2012. *Bristol Bay's Wild Salmon Ecosystems and the Pebble Mine: Key Considerations for a Large-Scale Mine Proposal.* January. Wild Salmon Center and Trout Unlimited.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska*. February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company), Vancouver, BC.

PLP (Pebble Limited Partnership). 2013. About Pebble. Available: http://corporate.pebblepartnership.com/about. Accessed: April 1, 2013.

USEPA (U.S. Environmental Protection Agency). 2002a. *Clinch and Powell Valley Watershed Ecological Risk Assessment*. EPA/600/R-01/050. Washington, DC: Office of Research and Development, National Center for Environmental Assessment. Available http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=15219.

USEPA (U.S. Environmental Protection Agency). 2002b. *Ecological Risk Assessment for the Middle Snake River, Idaho*. EPA/600/R-01/017. Washington, DC: Office of Research and Development, National Center for Environmental Assessment. Available: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=29097&partner=ORD-NCEA.

USEPA (U.S. Environmental Protection Agency). 2002c. *Waquoit Bay Watershed Ecological Risk Assessment: The Effect of Land-Derived Nitrogen Loads on Estuarine Eutrophication*. EPA/600/R-02/079. Washington, DC: Office of Research and Development, National Center for Environmental Assessment. Available: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=15221.

USEPA (U.S. Environmental Protection Agency). 2008. *Predicting Future Introductions of Nonindigenous Species to the Great Lakes (Final Report)*. EPA/600/R-08/066F. Washington, DC: Office of Research and Development, National Center for Environmental Assessment. Available: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=190305.

USEPA (U.S. Environmental Protection Agency). 2011. *The Effects of Mountaintop Mines and Valley Fills on Aquatic Ecosystems of the Central Appalachian Coalfields*. EPA/600/R-09/138F. Washington, DC: Office of Research and Development, National Center for Environmental Assessment. Available: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=225743.

15.1.3 Chapter 2—Overview of Assessment

Chambers, D. M., and B. Higman. 2011. *Long-Term Risks of Tailings Dam Failure*. Bozeman, MT: Center for Science in Public Participation. Available: http://www.csp2.org/files/reports/Long%20Term%20Risks%20of%20Tailings%20Dam%20Failure%20-%20Chambers%20%26%20Higman%20Oct11-2.pdf.

Earthworks. 2012. *U.S. Copper Porphyry Mines Report: The Track Record of Water Quality Impacts Resulting from Pipeline Spills, Tailings Failures and Water Collection and Treatment Failure*. Washington, DC.

Forman, R. T. T., and L. E. Alexander. 1998. Roads and their major ecological effects. *Annual Review of Ecology and Systematics* 29:207–231.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company), Vancouver, BC.

PLP (Pebble Limited Partnership). 2011. *Environmental Baseline Document 2004 through 2008.* Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

USEPA (U.S. Environmental Protection Agency). 1998. *Guidelines for Ecological Risk Assessment.* EPA/630/R-95/002F. April. Washington, DC: Office of Research and Development.

USFWS (U.S. Fish and Wildlife Service). 2012. *National Wetlands Inventory.* Available: http://www.fws.gov/wetlands/Wetlands-Mapper.html. Accessed: December 11, 2012.

USGS (U.S. Geological Survey). 2012. *National Hydrography Dataset.* High Resolution, Alaska. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

Woody, C. A., and B. Higman. 2011. *Groundwater as Essential Salmon Habitat in Nushagak and Kvichak River Headwaters: Issues Relative to Mining*. Unpublished report. Available: http://www.fish4thefuture.com/pdfs/Groundwater_and_SalmonFINAL27Aug11.pdf.

15.1.4 Chapter 3—Region

ACIA (Arctic Climate Impact Assessment). 2004. *Impacts of a Warming Arctic: Arctic Climate Impact Assessment*. Cambridge, UK: Cambridge University Press.

ADF&G (Alaska Department of Fish and Game). 2012. *Alaska Freshwater Fish Inventory Database*. Available: http://www.adfg.alaska.gov/index.cfm?adfg=ffinventory.main.

Araki, H., B. Cooper, and M. S. Blouin. 2009. Carry-over effect of captive breeding reduces reproductive fitness of wild-born descendants in the wild. *Biology Letters* 5(5):621–624.

Armstrong, J. B., D. E. Schindler, K. L. Omori, C. P. Ruff, and T. P. Quinn. 2010. Thermal heterogeneity mediates the effects of pulsed subsidies across a landscape. *Ecology* 91:1445–1454.

Barnett, T. P., J. C. Adam, and D. P. Lettenmaier. 2005. Potential impacts of a warming climate on water availability in snow-dominated regions. *Nature* 438:303–309.

Battin, J., M. W. Wiley, M. H. Ruckelshaus, R. N. Palmer, E. Korb, K. K. Bartz, and H. Imaki. 2007. Projected impacts of climate change on salmon habitat restoration. *Proceedings of the National Academy of Sciences of the United States of America* 104(16):6720–6725.

Beer, W. N., and J. J. Anderson. 2001. Effect of spawning day and temperature on salmon emergence: interpretations of a growth model for Methow River chinook. *Canadian Journal of Fisheries and Aquatic Sciences* 58:943–949.

Berggren, T. J., and M. J. Filardo. 1993. An analysis of variables influencing the migration of juvenile salmonids in the Columbia River basin. *North American Journal of Fisheries Management* 13:48–63.

Bjornn, T. C., and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. Pages 83–138 in W. R. Meehan (ed.), *Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats*. Bethesda, MD: American Fisheries Society.

Bryant, M. 2009. Global climate change and potential effects on Pacific salmonids in freshwater ecosystems of southeast Alaska. *Climatic Change* 95:169–193.

Burgner, R. L. 1991. Life history of sockeye salmon (*Oncorhynchus nerka*). Pages 1–118 in C. Groot and L. Margolis, (eds.), *Pacific Salmon Life Histories*. Seattle, WA: University of Washington Press.

Burnett, K. M., G. H. Reeves, D. J. Miller, S. Clarke, K. Vance-Borland, and K. Christiansen. 2007. Distribution of salmon-habitat potential relative to landscape characteristics and implications for conservation. *Ecological Applications* 17(1):66–80.

Christensen, J. H., B. Hewitson, A. Busuioc, A. Chen, X. Gao, I. Held, R. Jones, R. K. Kolli, W.-T. Kwon, R. Laprise, V. Magaa Rueda, L. Mearns, C. G. Menndez, J. Risnen, A. Rinke, A. Sarr, and P. Whetton. 2007. Regional Climate Projections. In Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.), *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UKC and New York, NY: Cambridge University Press.

CRU (Climatic Research Unit). 2012. *Data*. University of East Anglia, UK. Available: http://www.cru.uea.ac.uk/data.

Demory, R. L., R. L. Orrell, and D. R. Heinle. 1964. *Spawning Ground Catalog of the Kvichak River System, Bristol Bay, Alaska*. Issue 488 of Special Scientific Report—Fisheries. Washington, DC: U.S. Department of Interior, Bureau of Commercial Fisheries.

Detterman, R. L. 1986. Glaciation of the Alaska Peninsula. Pages 151–170 in Hamilton, T. D., K. M. Reed, and R. M. Thorson (eds.), *Glaciation in Alaska*. Alaska Geological Society.

Detterman, R. L., and B. L. Reed. 1973. *Surficial Deposits of the Iliamna Quadrangle, Alaska.* Geological Survey Bulletin 1368-A. U.S. Geological Survey. Available: http://www.dggs.alaska.gov/webpubs/usgs/b/text/b1368a.pdf. Accessed: December 17, 2012.

Eaton, J. G., and R. M. Scheller. 1996. Effects of climate warming on fish thermal habitat in streams of the United States. *Limnology and Oceanography* 41:1109–1115.

Eiler, J. H., B. D. Nelson, and R. F. Bradshaw. 1992. Riverine spawning by sockeye salmon in the Taku River, Alaska and British Columbia. *Transactions of the American Fisheries Society* 121(6):701–708.

Feddema, J. J. 2005. A revised Thornthwaite-type global climate classification. *Physical Geography* 26:442–466.

Gallant, A. L., E. F. Binnian, J. M. Omernik, and M. B. Shasby. 1995. *Ecoregions of Alaska.* U.S. Geological Survey Professional Paper 1567. Washington, DC.

Gesch, D. 2007. Chapter 4–The National Elevation Dataset. Pages 99–118 in Maune, D. (ed.), *Digital Elevation Model Technologies and Applications: The DEM Users Manual*. 2nd edition. Bethesda, MD: American Society for Photogrammetry and Remote Sensing.

Gesch, D., M. Oimoen, S. Greenlee, C. Nelson, M. Steuck, and D. Tyler. 2002. The National Elevation Dataset. *Photogrammatic Engineering and Remote Sensing* 68(1):5–11.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company), Vancouver, BC.

Haeussler, P. J., and R. W. Saltus. 2004. *26 km Offset on the Lake Clark Fault Since Late Eocene Time*. Studies by the U.S. Geological Survey in Alaska, 2004. U.S. Geological Survey Professional Paper 1709-A.

Haeussler, P. J., and C. F. Waythomas. 2011. *Review of the Origin of the Braid Scarp Near the Pebble Prospect, Southwestern Alaska*. U.S. Geological Survey Open File Report 2011-1028.

Hamon, W. R. 1961. Estimating potential evapotranspiration. *Journal of the Hydraulics Division*, *Proceedings of the American Society of Civil Engineers* 87:107–120.

Hauer, F. R., J. S. Baron, D. H. Campbell, K. D. Fausch, S. W. Hostetler, G. H. Leavesley, P. R. Leavitt, D. M. McKnight, and J. A. Stanford. 1997. Assessment of climate change and freshwater ecosystems of the Rocky Mountains, USA and Canada. *Hydrological Processes* 11:903–924.

Healey, M. 2011. The cumulative impacts of climate change on Fraser River sockeye salmon (*Oncorhynchus nerka*) and implications for management. *Canadian Journal of Fisheries and Aquatic Sciences* 68:953–953.

Heming, T. A., J. E. McInerney, and D. F. Alderdice. 1982. Effect of temperature on initial feeding in alevins of Chinook salmon (*Oncorhynchus tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 39:1554–1562.

Hilborn, R., T. Quinn, D. Schindler, and D. Rogers. 2003. Biocomplexity and fisheries sustainability. *Proceedings of the National Academy of Sciences of the United States of America* 100:6564–6568.

Hinzman, L. D., N. D. Bettez, W. R. Bolton, F. S. Chapin, M. B. Dyurgerov, C. L. Fastie, B. Griffith, R. D. Hollister, A. Hope, H. P. Huntington, A. M. Jensen, G. J. Jia, T. Jorgenson, D. L. Kane, D. R. Klein, G. Kofinas, A. H. Lynch, A. H. Lloyd, A. D. McGuire, F. E. Nelson, W. C. Oechel, T. E. Osterkamp, C. H. Racine, V. E. Romanovsky, R. S. Stone, D. A. Stow, M. Sturm, C. E. Tweedie, G. L. Vourlitis, M. D. Walker, D. A. Walker, P. J. Webber, J. M. Welker, K. S. Winker, and K. Yoshikawa. 2005. Evidence and implications of recent climate change in Northern Alaska and other Arctic regions. *Climatic Change* 72:251–298.

Hughes, N. F., and L. M. Dill. 1990. Position choice by drift-feeding salmonids: model and test for Arctic grayling (*Thymallus arcticus*) in subarctic mountain streams, Interior Alaska. *Canadian Journal of Fisheries and Aquatic Sciences* 47:2039–2048.

Irvine, J. R., M. Fukuwaka, T. Kaga, J. H. Park, K. B. Seong, S. Kang, V. Karpenko, N. Klovach, H. Bartlett, and E. Volk. 2009. *Pacific Salmon Status and Abundance Trends*. Document 1199, Rev. 1. North Pacific Anadromous Fish Commission.

Johnson, J., and P. Blanche. 2012. *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region, Effective June 1, 2012*. Special Publication No. 12-08. Anchorage, AK: Alaska Department of Fish and Game.

Kisslinger, C. 1976. A review of theories of mechanisms of induced seismicity. *Engineering Geology* 10:85–98.

Koehler, R. D., and R. D. Reger. 2011. *Reconnaissance Evaluation of the Lake Clark Fault, Tyonek Area, Alaska*. Preliminary Interpretative Report 2011-1. Alaska Division of Geological & Geophysical Surveys.

Kovach, R. P., A. J. Gharrett, and D. A. Tallmon. 2012. Genetic change for earlier migration timing in a pink salmon population. *Proceedings of the Royal Society B-Biological Sciences* 279:3870–3878.

Kyle R. E., and T. P. Brabets. 2001. *Water Temperature of Streams in the Cook Inlet Basin, Alaska, and Implications of Climate Change*. Water Resources Investigations Report 2001-4109. U.S. Geological Survey.

Lawson, P. W., E. A. Logerwell, N. J. Mantua, R. C. Francis, and V. N. Agostini. 2004. Environmental factors influencing freshwater survival and smolt production in Pacific Northwest coho salmon (*Oncorhynchus kisutch*). *Canadian Journal of Fisheries and Aquatic Sciences* 61:360–373.

Lea, P. D., and C. F. Waythomas. 1990. Late-Pleistocene eolian sand sheets in Alaska. *Quaternary Research* 34:269–281.

Lea, P.D., S. A. Elias, and S. K. Short. 1991. Stratigraphy and paleoenvironments of Pleistocene nonglacial deposits in the southern Nushagak lowland, southwestern Alaska, USA. *Arctic and Alpine Research* 23:375–391.

Lisi, P. J., D. E. Schindler, K. T. Bentley, and G. R. Pess. 2013. Association between geomorphic attributes of watersheds, water temperature, and salmon spawn timing in Alaskan streams. *Geomorphology* 185:78–86.

Lisle, T. 1989. Sediment transport and resulting deposition in spawning gravels, North Coastal California. *Water Resources Research* 25:1303–1319.

Liston, G. E., and C. A. Hiemstra. 2011. The changing cryosphere: pan-Arctic snow trends (1979–2009). *Journal of Climate* 24(21):5691–5712.

Luck, M., N. Maumenee, D. Whited, J. Lucotch, S. Chilcote, M. Lorang, D. Goodman, K. McDonald, J. Kimball, and J. Stanford. 2010. Remote sensing analysis of physical complexity of North Pacific Rim rivers to assist wild salmon conservation. *Earth Surface Processes and Landforms* 35:1330–1343.

Mantua, N., I. Tohver, and A. Hamlet. 2010. Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. *Climatic Change* 102:187–223.

Markon, C. J., S. F. Trainor, and F. S. Chapin III (eds.) 2012. *The United States National Climate Assessment—Alaska Technical Regional Report*. Circular 1379. Reston, VA: U.S. Geological Survey.

Martins, E. G., S. G. Hinch, S. J. Cooke, and D. A. Patterson. 2012. Climate effects on growth, phenology, and survival of sockeye salmon (*Oncorhynchus nerka*): a synthesis of the current state of knowledge and future research directions. *Reviews in Fish Biology and Fisheries* 22:887–914.

Mellina, E., R. D. Moore, S. G. Hinch, J. S. Macdonald, and G. Pearson. 2002. Stream temperature responses to clearcut logging in British Columbia: the moderating influences of groundwater and headwater lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 59:1886–1900.

Mohseni O., and H. G. Stefan. 1999. Stream temperature/air temperature relationship: a physical interpretation. *Journal of Hydrology* 218(3–4):128–141.

Montgomery, D. R., and J. M. Buffington. 1997. Channel-reach morphology in mountain drainage basins. *Bulletin of the Geological Society of America* 109(5):596–611.

Montgomery, D. R., E. M. Beamer, G. R. Pess, and T. P. Quinn. 1999. Channel type and salmonid spawning distribution and abundance. *Canadian Journal of Fisheries and Aquatic Sciences* 56(3):377–387.

Montgomery, D. R., J. M. Buffington, N. P. Peterson, D. Schuett-Hames, and T. P. Quinn. 1996. Stream-bed scour, egg burial depths, and the influence of salmonid spawning on bed surface mobility and embryo survival. *Canadian Journal of Fisheries and Aquatic Sciences* 53(5):1061–1070.

Mote, P. W., E. Parson, A. F. Hamlet, W. S. Keeton, D. Lettenmaier, N. Mantua, E. L. Miles, D. Peterson, D. L. Peterson, R. Slaughter, and A. K. Snover. 2003. Preparing for climatic change: the water, salmon, and forests of the Pacific Northwest. *Climatic Change* 61:45–88.

Nehlsen, W., J. Williams, and J. Lichatowich. 1991. Pacific salmon at the crossroads—stocks at risk from California, Oregon, Idaho, and Washington. *Fisheries* 16:4–21.

Parks, B., and R. J. Madison. 1985. *Estimation of selected flow and water-quality characteristics of Alaskan streams*. U.S. Geological Survey.

Peacock, S. 2012. Projected twenty-first century change in temperature, precipitation, and snow cover over North America in CCSM4. *Journal of Climate* 25(13):4405–4429.

Pess, G. R., T. P. Quinn, D. E. Schindler, and M. C. Liermann. 2013. Freshwater habitat associations between pink (*Oncorhynchus gorbuscha*), chum (*O. keta*) and Chinook salmon (*O. tshawytscha*) in a watershed dominated by sockeye salmon (*O. nerka*) abundance. *Ecology of Freshwater Fish.* doi: 10.1111/eff.12088.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008*. Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Power, G., R. S. Brown, and J. G. Imhof. 1999. Groundwater and fish—Insights from northern North America. *Hydrological Processes* 13:401–422.

PRISM Climate Group. 2012. Oregon State University. Available: www.prism.oregonstate.edu.

Quinn, T. P. 2005. *The Behavior and Ecology of Pacific Salmon and Trout*. Seattle, WA: University of Washington Press.

Quinn, T. P., H. B. J. Rich, D. Gosse, and N. Schtickzelle, and J. Grant. 2012. Population dynamics and asynchrony at fine spatial scales: a case history of sockeye salmon (*Oncorhynchus nerka*) population structure in Alaska, USA. *Canadian Journal of Fish and Aquatic Science* 69:297–306.

Rains, M. C. 2011. Water sources and hydrodynamics of closed-basin depressions, Cook Inlet Region, Alaska. *Wetlands* 31:377–387.

Rand, P. S., B. A. Berejikian, A. Bidlack, D. Bottom, J. Gardner, M. Kaeriyama, R. Lincoln, M. Nagata, T. N. Pearsons, M. Schmidt, W. W. Smoker, L. A. Weitkamp, and L. A. Zhivitovsky. 2012. Ecological interactions between wild and hatchery salmonids and key recommendations for research and management actions in selected regions of the North Pacific. *Environmental Biology of Fishes* 94:343–358.

RAP (Riverscape Analysis Project). 2011. Available: http://rap.ntsg.umt.edu. Accessed: December 2011.

Rich, H. B., Jr., T. P. Quinn, M. D. Scheuerell, and D. E. Schindler. 2009. Climate and intraspecific competition control the growth and life history of juvenile sockeye salmon (*Oncorhynchus nerka*) in Iliamna Lake, Alaska. *Canadian Journal of Fisheries and Aquatic Sciences* 66(2):238–246.

Rogers, L. A., and D. E. Schindler. 2011. Scale and the detection of climatic influences on the productivity of salmon populations. *Global Change Biology* 17:2546–2558.

Ruff, C. P., D. E. Schindler, J. B. Armstrong, K. T. Bentley, G. T. Brooks, G. W. Holtgrieve, M. T. McGlauflin, C. E. Torgersen, and J. E. Seeb. 2011. Temperature-associated population diversity in salmon confers benefits to mobile consumers. *Ecology* 92:2073–2084.

Salathé, E. P. 2006. Influences of a shift in North Pacific storm tracks on western North America precipitation under global warming. *Geophysical Research Letters* 33(19):L19820.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. *Nature* 465:609–612.

Selkregg, L. L. 1974. *Southwest Alaska, Alaska Regional Profiles*. Anchorage, AK: University of Alaska, Arctic Environmental Information and Data Center.

Serreze, M. C., J. E. Walsh, F. S. Chapin III, T. Osterkamp, M. Dyurgerov, V. Romanovsky, W. C. Oechel, J. Morison, T. Zhang, and G. Barry. 2000. Observational evidence of recent change in the northern high-latitude environment. *Climatic Change* 46(1-2):159–207.

Shallin Busch, D., M. Sheer, K. Burnett, P. McElhany, and T. Cooney. 2011. Landscape-level model to predict spawning habitat for lower Columbia River fall Chinook salmon (*Oncorhynchus tshawytscha*). *River Research and Applications* 29(3):297–312.

SNAP (Scenarios Network for Alaska and Arctic Planning). 2012. *Historical and Projected Decadal Averages of Annual Total and Annual Mean Temperature, Precipitation, and Evapotranspiration.* Available: http://www.snap.uaf.edu/data.php. Accessed: February 7, 2013.

Stafford, J. M., G. Wendler, and J. Curtis. 2000. Temperature and precipitation of Alaska: 50 year trend analysis. *Theoretical and Applied Climatology* 67(1-2):33–44.

Stanford, J. A., M. S. Lorang, and F. R. Hauer. 2005. The shifting habitat mosaic of river ecosystems. *International Association of Theoretical and Applied Limnology—Proceedings* 29:123–136.

Steen, R. P., and T. P. Quinn. 1999. Egg burial depth by sockeye salmon (*Oncorhynchus nerka*): implications for survival of embryos and natural selection on female body size. *Canadian Journal of Zoology* 77(5):836–841.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2004. Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. *Climatic Change* 62:217–232.

Stilwell, K. B., and D. S. Kaufman. 1996. Late Wisconsin glacial history of the northern Alaska Peninsula. *Arctic and Alpine Research* 28:475–487.

Tang, J., M. D. Bryant, and E. L. Brannon. 1987. Effect of temperature extremes on the mortality and development rates of coho salmon embryos and alevins. *Progressive Fish-Culturist* 49:167–174.

Taylor, E. B. 1990. Environmental correlates of life-history variation in juvenile Chinook salmon, *Oncorhynchus tshawytscha* (Walbaum). *Journal of Fish Biology* 37:1–17.

USGS (U.S. Geological Survey). 2010. *Significant United States Earthquakes, 1568–2009*. June. Available: http://nationalatlas.gov/atlasftp.html?openChapters=chpgeol#chpgeol. File (quksigx020.tar.gz). Accessed: April 13, 2012.

USGS (U.S. Geological Survey). 2012. *National Hydrography Dataset*. High Resolution, Alaska. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

USGS (U.S. Geological Survey). 2013. National Elevation Dataset (2 arc second), ArcGrid. August 26. [n59w155.zip, n59w156.zip, n59w157.zip, n59w158.zip, n60w154.zip, n60w155.zip, n60w156.zip, n60w157.zip, n60w158.zip, n60w159.zip, n60w160.zip, n61w153.zip, n61w154.zip, n61w155.zip, n61w156.zip, n61w157.zip, n61w158.zip, n61w159.zip, n61w160.zip] Available: http://viewer.nationalmap.gov/viewer/.

Wahrhaftig, C. 1965. *Physiographic Divisions of Alaska*. Geological Survey Professional Paper 482. U.S. Geological Survey.

Walsh, J. E., W. L. Chapman, V. Romanovsky, J. H. Christensen, and M. Stendel. 2008. Global climate model performance over Alaska and Greenland. *Journal of Climate* 21:6156–6174.

Webb B. W., and F. Nobilis. 1997. Long-term perspective on the nature of the air-water temperature relationship: a case study. *Hydrological Processes* 11(2):137–147.

Wendler, G., L. Chen, and B. Moore. 2012. The first decade of the new century: a cooling trend for most of Alaska. *The Open Atmospheric Science Journal* 6.

Winter, T. C. 2001. The concept of hydrologic landscapes. *Journal of the American Water Resources Association* 37:335–349.

Wissmar R. C., R. K. Timm, and M. D. Bryant. 2010. Radar-derived digital elevation models and field-surveyed variables to predict distributions of juvenile coho salmon and Dolly Varden in remote streams of Alaska. *Transactions of the American Fisheries Society* 139:288–302.

Zamzow, K. 2011. *Investigations of Surface Water Quality in the Nushagak, Kvichak, and Chulitna Watersheds, Southwest Alaska 2009–2010*. Sutton, AK: Center for Science in Public Participation.

Personal Communications

Wiedmer, M. Malma Consulting. September 2011—conversation with Joe Ebersole regarding distribution of salmon across the Nushagak and Kvichak River watersheds and factors controlling that distribution.

15.1.5 Chapter 4—Type of Development

Alto, K., S. Broderius, and L. L. Smith, Jr. 1977. *Toxicity of Xanthates to Freshwater Fish and Invertebrates*. Prepared for State of Minnesota Environmental Quality Council Regional Copper Nickel Study.

Anderson, E. D., R. G. Eppinger, and K. D. Kelley. 2009. Using regional geochemistry, geology, aeromagnetics, Landsat, and digital elevation models (DEM) to define favourable areas for porphyry-style mineralization in southwestern Alaska. Pages 345–348 in D. R. Lentz, K. G. Thorne, K. L. Beal (eds.), *24th International Applied Geochemistry Symposium, Fredericton, New Brunswick, Canada*. Association of Applied Geochemists.

Blight, G. E. 2010. *Geotechnical Engineering for Mine Waste Storage Facilities.* Boca Raton, FL: CRC Press.

Borden, R. K. 2003. Environmental geochemistry of the Bingham Canyon porphyry copper deposit, Utah. *Environmental Geology* 43(7):752–758.

Brodie, M. J., L. M. Broughton, and A. M. Robertson. 1991. *A Conceptual Rock Classification System for Waste Management and a Laboratory Method for ARD Prediction from Rock Piles*. Preprint for the 2nd International Conference on the Abatement of Acid Drainage, Montreal, Quebec, September 16–18, 1991.

Commonwealth of Australia. 2007. *Tailings Management – Leading Practice Sustainable Development Program for the Mining Industry*. Australian Government, Department of Industry, Tourism and Resources.

Davies, M. P. 2002. Tailings impoundment failures: are geotechnical engineers listening? *Geotechnical News.* September: 31–36.

Davies, M. P. 2011. Filtered dry stacked tailings – the fundamentals. Pages 189–197 in D. van Zyl (ed.), *Tailings and Mine Waste 2011: Proceedings of the 15th International Conference on Tailings and Mine Waste*, *Vancouver, Canada, November 6–9, 2011*.

Fuerstenau, M. C. 2003. *Principles of Mineral Processing.* Littleton, CO: Society for Mining, Metallurgy & Exploration, Incorporated.

Fuerstenau, M. C., G. Jameson, and R. H. Yoon (eds.). 2007. *Froth Flotation: A Century of Innovation*. Littleton, CO: Society of Mining, Metallurgy, and Exploration.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company), Vancouver, BC.

ICOLD (International Commission on Large Dams). 2001. *Tailings Dams, Risk of Dangerous Occurrences, Lessons Learnt from Practical Experiences*. United Nations Environmental Programme, Bulletin 121.

John, D. A., R. A. Ayuso, M. D. Barton, R. J. Blakely, R. J. Bodnar, J. H. Dilles, F. Gray, F. T. Graybeal, J. C. Mars, D. K. McPhee, R. R. Seal, R. D. Taylor, and P. G. Vikre. 2010. *Porphyry Copper Deposit Model.*Scientific Investigations Report 2010-5070-B. Reston, VA: U.S. Geological Survey.

Kesler, S. E., S. L. Chryssoulis, and G. Simon. 2002. Gold in porphyry copper deposits: its abundance and fate. *Ore Geology Reviews* 21(1–2):103–124.

Khoshdast, H., and A. Sam. 2011. Flotation frothers: review of their classifications, properties and preparation. *The Open Mineral Processing Journal* 2011(4):25–44.

Koerner, R. M., Y. G. Hsuan, and G. R. Koerner. 2011. *GRI White Paper #6 on Geomembrane Lifetime Prediction: Unexposed and Exposed Conditions, Updated 2011.* Available: http://www.geosynthetic-institute.org/papers/paper6.pdf. Accessed: April 20, 2012.

Levit, S., and D. M. Chambers. 2012. C*omparison of the Pebble Mine with Other Alaska Large Hard Rock Mines*. Sutton, AK: Center for Science in Public Participation.

Lipman, P. W., and D. A. Sawyer. 1985. Mesozoic ash-flow caldera fragments in southeastern Arizona and their relation to porphyry copper deposits. *Geology* 13:652–656.

Logsdon, M. J., K. Hagelstein, and T. I. Mudder. 1999. *The Management of Cyanide in Gold Extraction*. Ottawa, ON: International Council on Metals and the Environment.

Lusty, P. A. J., and S. D. Hannis. 2009. *Commodity Profile: Copper*. Internal Report OR/09/041. Keyworth, Nottingham: British Geological Survey.

Marsden, J. O., and C. I. House. 2006. *The Chemistry of Gold Extraction.* 2nd edition. Littleton, CO: Society of Mining, Metallurgy, and Exploration.

Martin, T. E., M. P. Davies, S. Rice, T. Higgs, and P. C. Lighthall. 2002. *Stewardship of Tailings Facilities*. Commissioned by the MMSD project of the International Institute for the Environment and Development (IIEA) and the World Business Council for Sustainable Development (WBCSD), London, England.

McLeod, H., and L. Murray. 2003. *Tailings Dam Versus a Water Dam, What Is the Design Difference?* ICOLD Symposium on Major Challenges in Tailings Dams, June 15, 2003.

National Inventory of Dams. 2005. *2005 United States National Inventory of Dams Database*. Available: http://geo.usace.army.mil/pgis/f?p=397:1:0::NO. Accessed: October 6, 2005.

Novagold. 2012. *Donlin Creek Gold Project Technical Report.* January 20.

O'Kane, M., and C. Wels. 2003. Mine Waste Cover System Design – Linking Predicted Performance to Groundwater and Surface Water Impacts. Pages 341–349 in *Proceedings of the 6th Annual International Conference on Acid Rock Drainage*, *Cairns, Queensland, Australia, July 12–18, 2003.*

Porter, K. E., and D. I. Bleiwas. 2003. *Physical Aspects of Waste Storage from a Hypothetical Open Pit Porphyry Copper Operation*. Open File Report 03-143, Version 1.0. U.S. Geological Survey.

Price, W. A. 2009. *Prediction Manual for Drainage Chemistry from Sulphidic Geologic Materials.* 579. Report prepared by CANMET – Mining and Mineral Sciences Laboratories for the Mine Environment Neutral Drainage (MEND) Program, Natural Resources Canada.

Price, W. A., and J. C. Errington. 1998. *Guidelines for Metal Leaching and Acid Rock Drainage at Mine Sites in British Columbia*. British Columbia Ministry of Energy and Mines.

Rico, M. G., G. Benito, and A. Diez-Herrero. 2008. Floods from tailings dam failures. *Journal of Hazardous Materials* 154(1-3):79–87.

Rowe, R. K. 2005. Long-term performance of contaminant barrier systems. *Geotechnique* 55(9):631–678. Available: http://www.jcigs.org/8ICG/SL/Geotechnique%20Rankine%20V55%20N9%20631-678%20Rowe%202005.pdf. Accessed: October 12, 2012.

Scott Wilson Mining. 2005. *Guelb Moghrein Copper Gold Project, Environmental and Social Impact Assessment, Akjoujt, Mauritania*.

Singer, D. A., V. I. Berger, and B. C. Moring. 2008. *Porphyry Copper Deposits of the World: Database and Grade and Tonnage Models, 2008*. Open File Report 2008-1155. U.S. Geological Survey.

Singer, D. A., W. D. Menzie, D. M. Sutphin, D. L. Mosier, and J. D. Bliss. 2001. *Mineral Deposit Density – An Update*. U.S. Geological Survey Professional Paper 1640-A. Reston, VA: U.S. Geological Survey.

Slaughter, C. B., G. W. Freethey, and L. E. Spangler. 1995. Hydrology of the North Fork of the Right Fork of Millar Creek, Carbon Country, Utah, before, during, and after underground coal mining. Water Resources Investigation Report 95-4025. Denver, CO: U.S. Geological Survey. Available: http://pubs.usgs.gov/wri/1995/4025/report.pdf.

SME (Society for Mining, Metallurgy, and Exploration). 2011. *SME Mining Engineering Handbook.* 3rd edition. Darling, P. (ed.). Denver, CO.

Stange, W. 1999. The process design of gold leaching and carbon-in-pulp circuits. *The Journal of the South African Institute of Mining and Metallurgy* January/February (1999):13–25.

State of Idaho. 1992. *Mill Tailings Impoundments*. Pages 16–23 in *Best Management Practices for Mining in Idaho*. Idaho Department of Lands. Available: http://www.idl.idaho.gov/Bureau/Minerals/bmp_manual1992/p16-ch4.pdf. Accessed: September 18, 2011.

USACE (U.S. Army Corps of Engineers). 2011. Permit # POA-1992-574-M19, Fish Creek. Issued March 4, 2011.

USDA (U.S. Department of Agriculture). 1995. *Anatomy of a Mine from Prospect to Production*. General Technical Report INT-GTR-35 revised. Ogden, UT: Forest Service, Intermountain Research Station. Available: http://www.fs.fed.us/geology/anatomy_mine.pdf.

USEPA (U.S. Environmental Protection Agency). 1994a. *Technical Resource Document: Extraction and Beneficiation of Ores and Minerals, Volume 4, Copper*. EPA 530-R-94-031

USEPA (U.S. Environmental Protection Agency). 1994b. *Technical Report: Design and Evaluation of Tailings Dams*. EPA 530-R-94-038. Washington, DC: Office of Solid Waste.

Vick, S. G. 1983. *Planning, Design, and Analysis of Tailings Dams.* New York, NY: John Wiley & Sons.

Vigneault, B., M. Desforges, and J. Geer. 2009. *Mining Reagents and By-Products (e.g. Thiosalts) as Potential Toxicants in Mine Effluents*. Metal Mining Investigation of Cause Workshop, Session 2: Potential Causes of Effects. Available: https://www.ec.gc.ca/esee-eem/default.asp?lang=En&n=495426D1-1&offset=6&toc=show.

Whittaker, B. N., and D. J. Reddish. 1989. *Subsidence—Occurrence, Prediction and Control*. Amsterdam, Netherlands: Elsevier.

15.1.6 Chapter 5—Endpoints

ADF&G (Alaska Department of Fish and Game). 2011. Moose Species Profile. Juneau, AK.

ADF&G (Alaska Department of Fish and Game). 2012. *Alaska Freshwater Fish Inventory Database*. Available: http://www.adfg.alaska.gov/index.cfm?adfg=ffinventory.main.

Alaska Department of Health and Social Services. 2005. The Prevalence of Diabetes Among Adult Alaskans, 2002–2004. *Epidemiology Bulletins* 1–8.

Alaska Federation of Natives. 2010. *The Right to Subsist—Federal Protection of Subsistence in Alaska*. Available: http://www.nativefederation.org/wp-content/uploads/2012/10/2010-april8-the-right-to-subsistence.pdf.

Armstrong, R. H., and J. E. Morrow. 1980. The Dolly Varden charr, *Salvelinus malma*. Pages 99–140 in E. K. Balon (ed.), *Charrs: Salmonid Fishes of the Genus* Salvelinus. Springer.

Augerot, X. 2005. *Atlas of Pacific Salmon: The First Map-Based Status Assessment of Salmon in the North Pacific*. Portland, OR.

Baker, T. T., L. F. Fair, R. A. Clark, and J. J. Hasbrouck. 2009. *Review of Salmon Escapement Goals in Bristol Bay, Alaska, 2009*. Alaska Department of Fish and Game, Division of Sport Fish, Research and Technical Services.

Ballew, C., A. Ross, R. S. Wells, V. Hiratsuka, K. J. Hamrick, E. D. Nobmann, and S. Bartell. 2004. *Final Report on the Alaska Traditional Diet Survey*. Anchorage, AK.

Bartz, K. K., and R. J. Naiman. 2005. Effects of salmon-borne nutrients on riparian soils and vegetation in Southwest Alaska. *Ecosystems* 8:529–545.

Bersamin, A., S. Zidenberg-Cherr, J. Stern, and B. Luick. 2007. Nutrient intakes are associated with adherence to a traditional diet among Yup'ik Eskimos living in remote Alaska native communities: the CANHR Study. *International Journal of Circumpolar Health* 66:62–70.

Blair, G. R., D. E. Rogers, and T. P. Quinn. 1993. Variation in life history characteristics and morphology of sockeye salmon in the Kvichak River system, Bristol Bay, Alaska. *Transactions of the American Fisheries Society* 122:550–559.

Brna, P. J., and L. A. Verbrugge (eds.). 2013. *Wildlife Resources of the Nushagak and Kvichak River Watersheds, Alaska*. Final Report. Anchorage, AK: Anchorage Fish and Wildlife Field Office, U.S. Fish and Wildlife Service.

Burger, C. V., and L. A. Gwartney. 1986. *A Radio Tagging Study of Naknek Drainage Rainbow Trout*. Anchorage, AK: U.S. National Park Service, Alaska Regional Office.

Butler, L. G. 2004. Unit 9 moose management report. Pages 113–120 in C. Brown, (ed.), *Moose Management Report of Survey and Inventory Activities: 1 July 2001–30 June 2003*. Juneau, AK: Alaska Department of Fish and Game.

Butler, L. G. 2009. Unit 9 and 10 Wolf Management Report. Pages 66–70 in P. Harper (ed.), *Wolf Management Report of Survey and Inventory Activities 1 July 2005–30 June 2008*. Juneau, AK: Alaska Department of Fish and Game.

Cederholm C. J., M. D. Kunze, T. Murota, and A. Sibatani. 1999. Pacfic salmon carcasses: essential contributors of nutrients and energy for aquatic and terrestrial ecosystems. *Fisheries* 24:6–15.

Chan, H., K. Fediuk, S. Hamilton, L. Rostas, H. Caughey, G. Egeland, and E. Loring. 2006. Food security in Nunavut, Canada: barriers and recommendations. *International Journal of Circumpolar Health* 65:416–431.

Christie, K. S., and T. E. Reimchen. 2008. Presence of salmon increases passerine density on Pacific Northwest streams. *Auk* 125:51–59.

Colombi, B. J., and J. F. Brooks. 2012. *Keystone Nations: Indigenous Peoples and Salmon Across the North Pacific*. Santa Fe, NM: School for Advanced Research.

Crane, P., M. Lisac, B. Spearman, E. Kretschmer, C. Lewis, S. Miller, and J. Wenburg. 2003. *Development and Application of Microsatellites to Population Structure and Mixed-Stock Analyses of Dolly Varden from the Togiak River Drainage, Final Report* (Study No. 00-011). Anchorage, AK: U.S. Fish and Wildlife Service, Conservation Genetics Laboratory.

Cunningham, C. J., R. Hilborn, J. Seeb, and T. Branch. 2012. *Reconstruction of Bristol Bay Sockeye Salmon Returns Using Age and Genetic Composition of Catch*. AFS-UW-1202. Seattle, WA: University of Washington, School of Aquatic and Fishery Sciences.

Darimont, C. T., T. E. Reimchen, and P. C. Paquet. 2003. Foraging behavior by gray wolves on salmon streams in coastal British Columbia. *Canadian Journal of Zoology* 81:349–353.

DeCicco, A. L. 1992. Long-distance movements of anadromous Dolly Varden between Alaska and the USSR. *Arctic* 45(2):120–123.

DeCicco, A. L. 1997. Movements of postsmolt anadromous Dolly Varden in northwestern Alaska. *American Fisheries Symposium* 19:175–183.

Denton, K. P., H. B. Rich, Jr., and T. P. Quinn. 2009. Diet, movement, and growth of Dolly Varden in response to sockeye salmon subsidies. *Transactions of the American Fisheries Society* 138(6):1207–1219.

Dewailly, E., C. Blanchet, S. Lemieux, L. Sauve, S. Gingras, P. Ayotte, and B. J. Holub. 2001. n-3 Fatty acids and cardiovascular disease risk factors among the Inuit of Nunavik. *The American Journal of Clinical Nutrition* 74:464–473.

Dewailly, E., C. Blanchet, S. Gingras, S. Lemieux, and B. J. Halub. 2002. Cardiovascular disease risk factors and n-3 fatty acid status in the adult population of James Bay Cree. *The American Journal of Clinical Nutrition* 76:85–92.

Din, J. N., D. E. Newby, and A. D. Flapan. 2004. Omega 3 fatty acids and cardiovascular disease—fishing for a natural treatment. *British Medical Journal* 328:30–35.

Dittman, A. H., and T. P. Quinn. 1996. Homing in Pacific salmon: mechanisms and ecological basis. *Journal of Experimental Biology* 199:83–91.

Duhaime, G., and N. Bernard (eds.). 2008. *Arctic Food Security*. Canadian Circumpolar Institute (CCI) Press, University of Alberta and Centre inter-universitaire d'etudes et de recherché autochtones (CIERA), Université Laval.

Dye, J. E., and C. J. Schwanke. 2009. *Report to the Alaska Board of Fisheries for the Recreational Fisheries of Bristol Bay, 2007, 2008, and 2009*. Alaska Department of Fish and Game, Division of Sport Fish, Research and Technical Services.

Dye, J. E., C. J. Schwanke, and T. A. Jaecks. 2006. *Report to the Alaska Board of Fisheries for the Recreational Fisheries of Bristol Bay, 2004, 2005, and 2006*. Special Publication No. 06-29. Alaska Department of Fish and Game, Divisions of Sport Fish and Commercial Fisheries.

Ebbesson, S. O. E., and M. T. Tejero. 2007. Fatty acid consumption and metabolic syndrome components: the GOCADAN study. *Journal of the Cardiometabolic Syndrome* 2(4):244–249.

Eliason, E., T. Clark, M. Hague, L. Hanson, Z. Gallagher, K. Jeffries, M. Gale, D. Patterson, S. Hinch, and A. Farrell. 2011. Differences in thermal tolerance among sockeye salmon populations. *Science* 332:109–112.

Fair, L. F. 2003. Critical elements of Kvichak River sockeye salmon management. *Alaska Fishery Research Bulletin* 10:95–103.

Fair, L. F., C. E. Barzil, X. Zyhang, R. A. Clark, and J. W. Erickson. 2012. *Review of Salmon Escapement Goals in Bristol Bay, Alaska, 2012*. Fishery Manuscript Series No. 12-0. Anchorage, AK: Alaska Department of Fish and Game.

Fall, J. A., D. L. Holen, B. Davis, T. Krieg, and D. Koster. 2006. *Subsistence Harvests and Uses of Wild Resources in Iliamna, Newhalen, Nondalton, Pedro Bay, and Port Alsworth, Alaska 2004*. Technical Paper No. 302. Juneau, AK: Alaska Department of Fish and Game, Division of Subsistence.

Fall, J. A., T. M. Krieg, and D. Holen. 2009. *Overview of the Subsistence Fishery of the Bristol Bay Management Area*. Special Publication No. BOF 2009-07. Anchorage, AK: Alaska Department of Fish and Game, Division of Subsistence.

Fancy, S. G. 1983. Movements and activity budgets of caribou near oil drilling sites in the Sagavanirktok River Floodplain, Alaska. *Arctic* 36:193–197.

Gende, S. M., and M. F. Willson. 2001. Passerine densities in riparian forests of Southeast Alaska: potential effects of anadromous spawning salmon. *Condor* 103:624–629.

Gende, S. M., R. T. Edwards, M. F. Willson, and M. S. Wipfli. 2002. Pacific salmon in aquatic and terrestrial ecosystems. *BioScience* 52(10):917–928.

Gerrard J. M., P. Gerrard, W. J. Maher, and D. W. A Whitfield. 1975. Factors influencing nest site selection of Bald Eagles in northern Saskatchewan and Manitoba. *Blue Jay* 33:169–176.

Goldsmith, D., A. Hill, T. Hull, M. Markowski, and R. Unswort. 1998. *Economic Assessment of Bristol Bay Area National Wildlife Refuges: Alaska Peninsula, Becherof, Izembek, Togiak*. U.S. Department of the Interior, Fish and Wildlife Service.

Gresh, T., J. Lichatowich, and P. Schoonmaker. 2000. An estimation of historic and current levels of salmon production in the northwest Pacific ecosystem: evidence of a nutrient deficit in the freshwater systems of the Pacific Northwest. *Fisheries* 25(1):15–21.

Gustafson, R. G., R. S. Waples, J. M. Myers, L. A. Weitkamp, G. J. Bryant, O. W. Johnson, and J. J. Hard. 2007. Pacific salmon extinctions: quantifying lost and remaining diversity. *Conservation Biology* 21:1009–1020.

Gwartney, L. A. 1985. *Naknek Drainage Rainbow Trout Study in the Katmai National Park and Preserve*. King Salmon, AK: Alaska Department of Fish and Game, Division of Sport Fish, and U.S. Department of Interior, National Park Service.

Hansen, A. J. 1987. Regulation of bald eagle reproductive rates in southeast Alaska. *Ecology* 68:1387–1392.

Hartman, W. L., and R. L. Burgner. 1972. Limnology and fish ecology of sockeye salmon nursery lakes of the world. *Journal of the Fisheries Research Board of Canada* 29:699–715.

Heard, W. R. 1991. Life history of pink salmon (*Oncorhynchus gorbuscha*). Pages 119–230 in C. Groot and L. Margolis (eds.), *Pacific Salmon Life Histories*. Vancouver, BC: UBC Press

Helfield, J. M., and R. J. Naiman. 2006. Keystone interactions: salmon and bear in riparian forests of Alaska. *Ecosystems* 9:167–180.

Hilborn, R. 2006. Fisheries success and failure: the case of the Bristol Bay salmon fishery. *Bulletin of Marine Science* 78(3):487–498.

Hilborn, R., T. Quinn, D. Schindler, and D. Rogers. 2003. Biocomplexity and fisheries sustainability. *Proceedings of the National Academy of Sciences of the United States of America* 100:6564–6568.

Holen, D., and T. Lemons. 2010. *Subsistence Harvests and Uses of Wild Resources in Lime Village, Alaska, 2007*. Technical Paper No. 355. Anchorage, AK: Alaska Department of Fish and Game, Division of Subsistence.

Holen, D., T. M. Krieg, and T. Lemons. 2011. *Subsistence Harvests and Uses of Wild Resources in King Salmon, Naknek, and South Naknek, Alaska, 2007*. Technical Paper No. 360. Anchorage, AK: Alaska Department of Fish and Game, Division of Subsistence.

Holen, D., J. Stariwat, T. M. Krieg, and T. Lemons. 2012. *Subsistence Harvest and Uses of Wild Resources in Aleknagik, Clark's Point, and Manokotak, Alaska, 2008*. Technical Paper No. 368. Anchorage, AK: Alaska Department of Fish and Game, Division of Subsistence.

Janetski, D. J., D. T. Chaloner, S. D. Tiegs, and G. A. Lamberti. 2009. Pacific salmon effects on stream ecosystems: a quantitative synthesis. *Oecologia* 159(3):583–595.

Johnson, J., and P. Blanche. 2012. *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region, Effective June 1, 2012*. Special Publication No. 12-08. Anchorage, AK: Alaska Department of Fish and Game.

Johnson, S. W., J. F. Thedinga, and A. S. Feldhausen. 1994. Juvenile salmonid densities and habitat use in the main-stem Situk River, Alaska, and potential effects of glacial flooding. *Northwest Science* 68(4):319–325.

Johnson, J. S., E. Nobmann, E. Asay, and A. Lanier. 2009. Dietary intake of Alaska Native people in two regions and implications for health: the Alaska Native Dietary and Subsistence Food Assessment Project. *International Journal of Circumpolar Health* 68:109–122.

King, R. S., C. M. Walker, D. F. Whigham, S. Baird, and J. A. Back. 2012. Catchment topography and wetland geomorphology drive macroinvertebrate community structure and juvenile salmonid distributions in southcentral Alaska headwater streams. *Freshwater Science* 31(2):341–364.

Kline T. C., J. J. Goering, O. A. Mathisen, P. H. Poe, P. L. Parker, and R. S. Scalan. 1993. Recycling of elements transported upstream by runs of Pacific salmon. 2. δ15N and δ13C evidence in the Kvichak River watershed, Bristol Bay, southwestern Alaska. *Canadian Journal of Fisheries and Aquatic Sciences* 50(11):2350–2365.

Krieg, T., M. Chythlook, P. Coiley-Kenner, D. Holen, K. Kamletz, and H. Nicholson. 2005. *Freshwater Fish Harvest and Use in Communities of the Kvichak watershed, 2003*. Technical Data Report No. 297. Juneau AK: Alaska Department of Fish and Game, Division of Subsistence.

Krieg, T. M., J. A. Fall, M. B. Chythlook, R. LaVine, and D. Koster. 2007. *Sharing, Bartering, and Cash Trade of Subsistence Resources in the Bristol Bay Area, Southwest Alaska*. Technical Paper No. 326. Juneau, AK: Alaska Department of Fish and Game, Division of Subsistence.

Krieg, T. M., D. L. Holen, and D. Koster. 2009. S*ubsistence Harvests and Uses of Wild Resources in Igiugig, Kokhanok, Koliganek, Levelock, and New Stuyahok, Alaska 2005*. Technical Paper No. 322. Dillingham, AK: Alaska Department of Fish and Game, Division of Subsistence.

Krueger, C. C., M. J. Lisac, S. J. Miller, and W. H. Spearman. 1999. *Genetic Differentiation of Rainbow Trout (*Oncorhynchus mykiss*) in the Togiak National Wildlife Refuge, Alaska*. Alaska Fisheries Technical Report No. 55. Anchorage, AK: U.S. Fish and Wildlife Service, Fish Genetics Laboratory.

Larkin, G. A., and P. A. Slaney. 1997. Implications of trends in marine-derived nutrient influx to South Coastal British Columbia salmonid production. *Fisheries* 22:16–24.

Lindley, S. T., C. B. Grimes, M. S. Mohr, W. Peterson. J. Stein, J. T. Anderson, L. W. Botsford, D. L. Bottom, C. A. Busack, T. K. Collier, J. Ferguson, J. C. Garza, A. M. Grover, D. G. Hankin, R. G. Kope, P. W. Lawson, A. Low, R. B. MacFarlane, K. Moore, M. Palmer-Zwahlen, F. B. Schwing, J. Smith, C. Tracy, R. Webb, B. K. Wells, and T. H. Williams. 2009. *What Caused the Sacramento River Fall Chinook Stock Collapse?* NOAA-TM-NMFS-SWFSC-447. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Lisac, M. J. 2009. *Seasonal Distribution and Biological Characteristics of Dolly Varden in the Goodnews River, Togiak National Wildlife Refuge, Alaska, 2005–2006*. Alaska Fisheries Technical Report No. 103. Dillingham, AK: U.S. Fish and Wildlife Service, Togiak National Wildlife Refuge.

Lisac, M. J., and R. D. Nelle. 2000. *Migratory Behavior and Seasonal Distribution of Dolly Varden* Salvelinus malma *in the Togiak River Watershed, Togiak National Wildlife Refuge*. Dillingham, AK: U.S. Fish and Wildlife Service.

McGlauflin, M. T., D. E. Schindler, L. W. Seeb, C. T. Smith, C. Habicht, and J. E. Seeb. 2011. Spawning habitat and geography influence population structure and juvenile migration timing of sockeye salmon in the Wood River lakes, Alaska. *Transactions of the American Fisheries Society* 140:763–782.

Meka, J. M., E. E. Knudsen, D. C. Douglas, and R. B. Benter. 2003. Variable migratory patterns of different adult rainbow trout life history types in a southwest Alaska watershed. *Transactions of the American Fisheries Society* 132:717–732.

Miller, S. D., and D. W. McCollum. 1994. *Nonresident Hunters: Their Characteristics and Attitudes toward Wildlife*. Anchorage, AK: Alaska Department of Fish and Game.

Miller, S. D., G. C. White, R. A. Sellers, H. V. Reynolds, J. W. Schoen, K. Titus, and V. G. Barnes. 1997. Brown and black bear density estimation in Alaska using radiotelemetry and replicated mark resight techniques. *Wildlife Monographs* 133:1–55.

Milner, A. M., and R. G. Bailey. 1989. Salmonid colonization of new streams in Glacier Bay National Park, Alaska, USA. *Aquaculture and Fisheries Management* 20:179–192.

Minard, R. E., M. Alexandersdottir, and S. Sonnichsen. 1992. *Estimation of Abundance, Seasonal Distribution, and Size and Age Composition of Rainbow Trout in the Kvichak River, Alaska, 1986–1991*. Fishery Data Series No. 92-51. Anchorage, AK: Alaska Department of Fish and Game, Division of Sport Fish.

Moore, J., and D. Schindler. 2004. Nutrient export from freshwater ecosystems by anadromous sockeye salmon (*Oncorhynchus nerka*). *Canadian Journal of Fisheries and Aquatic Sciences* 61:1582–1589.

Morstad S., and C. E. Brazil. 2012. *Kvichak River Sockeye Salmon Stock Status and Action Plan, 2012: A Report to the Alaska Board of Fisheries*. Anchorage, AK: Alaska Department of Fish and Game, Division of Sport Fish, Research and Technical Services.

Murphy, N. J., C. D. Schraer, E. Boyko, L. Bulkow, B. Doty, and A. Lanier. 1995. Dietary change and obesity associated with glucose intolerance in Alaska Natives. *Journal of the American Dietetic Association* 95:676–682.

Nehlsen, W., J. Williams, and J. Lichatowich. 1991. Pacific salmon at the crossroads—stocks at risk from California, Oregon, Idaho, and Washington. *Fisheries* 16:4–21.

Nickelson, T. E., J. D. Rodgers, S. L. Johnson, and M. F. Solazzi. 1992. Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. *Canadian Journal of Fisheries and Aquatic Sciences* 49:783–789.

Nobmann, E. D., R. Ponce, C. Mattil, R. Devereux, B. Dyke, S. O. E. Ebbesson, S. Laston, J. MacCluer, D. Robbins, T. Romenesko, G. Ruotolo, C. R. Wenger, and B. V. Howard. 2005. Dietary intakes vary with age among Eskimo adults of northwest Alaska in the GOCADAN study, 2000–2003. *The Journal of Nutrition* 135:856–862.

NRC (National Research Council). 1996. *Upstream: Salmon and Society in the Pacific Northwest*. Washington, DC.

Olson T., and J. Putera. 2007. *Refining Techniques to Survey Harvested Brown Bear Populations in Katmai National Park and Preserve and Lake Clark National Park and Preserve*. January.

Ostberg, C. O., S. D. Pavlov, and L. Hauser. 2009. Evolutionary relationships among sympatric life history forms of Dolly Varden inhabiting the landlocked Kronotsky Lake, Kamchatka, and a neighboring anadromous population. *Transactions of the American Fisheries Society* 138(1):1–14.

Pinsky, M. L., D. B. Springmeyer, M. N. Goslin, and X. Augerot. 2009. Range-wide selection of catchments for Pacific salmon conservation. *Conservation Biology* 23:680–691.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008.* Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Quinn, T. P. 2005. *The Behavior and Ecology of Pacific Salmon and Trout*. Seattle, WA: University of Washington Press.

Quinn T. P., H. B. Rich, D. Gosse, and N. Schtickzelle, and J. Grant. 2012. Population dynamics and asynchrony at fine spatial scales: a case history of sockeye salmon (*Oncorhynchus nerka*) population structure in Alaska, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 69(2):297–306.

Rahr, G., J. Lichatowich, R. Hubley, and S. Whidden. 1998. Sanctuaries for native salmon: a conservation strategy for the 21st century. *Fisheries* 23:6–36.

Reynolds, J. B. 2000. *Life History Analysis of Togiak River Char through Otolith Microchemistry*. Fairbanks, AK: Alaska Cooperative Fish and Wildlife Research Unit, University of Alaska Fairbanks.

Ruggerone, G. T., R. M. Peterman, and B. Dorner. 2010. Magnitude and trends in abundance of hatchery and wild pink salmon, chum salmon, and sockeye salmon in the North Pacific Ocean. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science* 2:306–328.

Russell, R. 1977. Rainbow Trout Life History Studies in Lower Talarik Creek - Kvichak Drainage. Anchorage, AK: Alaska Department of Fish and Game.

Russell, R. 1980. *A Fisheries Inventory of Waters in the Lake Clark National Monument Area*. King Salmon, AK: Alaska Department of Fish and Game, Division of Sport Fish.

Salo, E. O. 1991. Life history of chum salmon (*Oncorhynchus keta*). Pages 231–310 in C. Groot and L. Margolis (eds.), *Pacific Salmon Life Histories*. Vancouver BC: UBC Press.

Salomone, P., S. Morstad, T. Sands, M. Jones, T. Baker, G. Buck, F. West, and T. Kreig. 2011. *Bristol Bay Area Management Report*. Anchorage, AK.

Scanlon, B. 2000. *The Ecology of Arctic Char and the Dolly Varden in the Becharof Lake Drainage, Alaska*. Fairbanks, AK: University of Alaska.

Scheuerell, M., J. Moore, D. Schindler, and C. Harvey. 2007. Varying effects of anadromous sockeye salmon on the trophic ecology of two species of resident salmonids in southwest Alaska. *Freshwater Biology* 52:1944–1956.

Schindler, D. E., P. R. Leavitt, C. S. Brock, S. P. Johnson, and P. D. Quay. 2005. Marine-derived nutrients, commercial fisheries, and production of salmon and lake algae in Alaska. *Ecology* 86:3225–3231.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. *Nature* 465:609–612.

Sepulveda, A. J., D. S. Rutz, S. S. Ivey, K. J. Dunker, and J. A. Gross. 2013. Introduced northern pike predation on salmonids in southcentral Alaska. *Ecology of Freshwater Fish* 22(2):268–279.

Smith, R. J., K. A. Hobson, H. N. Koopman, and D. M. Lavigne. 1996. Distinguishing between populations of freshwater and saltwater harbor seals (*Phoca vitulina*) using stable isotope ratios and fatty acid profiles. *Canadian Journal of Fisheries and Aquatic Sciences* 53:272–279.

Solazzi, M., T. Nickelson, S. Johnson, and J. Rodgers. 2000. Effects of increasing winter rearing habitat on abundance of salmonids in two coastal Oregon streams. *Canadian Journal of Fisheries and Aquatic Sciences* 57:906–914.

U.S. Census Bureau. 2010. *Alaska QuickFacts from the United States Census Bureau*. Available: http://quickfacts.census.gov/qfd/states/02000.html.

USEPA (U.S. Environmental Protection Agency). 1998. *Guidelines for Ecological Risk Assessment.* EPA/630/R-95/002F. April. Washington, DC: Office of Research and Development.

Valkenburg, P., R. A. Sellers, R. C. Squibb, J. D. Woolington, A. R. Aderman, and B. W. Dale. 2003. Population dynamics of caribou herds in southwestern Alaska. *Rangifer Special Issue* 14:131–142.

Walsh, P., J. Reynolds, G. Collins, B. Russell, M. Winfree, and J. Denton. 2010. Application of a double-observer aerial line-transect method to estimate brown bear population density in southwestern Alaska. *Journal of Fish and Wildlife Management* 1:47–58.

Wentworth, C. 2007. *Subsistence Migratory Bird Harvest Survey: Bristol Bay 2001–2005*. Unpublished. Anchorage, AK: U.S. Fish and Wildlife Service.

Willson, M. F., and K. C. Halupka. 1995. Anadromous fish as keystone species in vertebrate communities. *Conservation Biology* 9:489–497.

Willson, M. F., S. M. Gende, and B. H. Marston. 1998. Fishes and the forest: expanding perspectives on fish-wildlife interactions. *Bioscience* 48:455–462.

Wipfli, M. S., and C. V. Baxter. 2010. Linking ecosystems, food webs, and fish production: subsidies in salmonid watersheds. *Fisheries* 35(8):373–387.

Wolfe, R. J., and R. J. Walker. 1987. Subsistence economies in Alaska: productivity, geography, and development impacts. *Arctic Anthropology* 24(2):56–81.

Wong D., and C. Wentworth. 1999. *Subsistence Waterfowl Harvest Survey, Bristol Bay 1995–1998*. Unpublished. Anchorage, AK: U.S. Fish and Wildlife Service.

Woody, C. A., and S. L. O'Neal. 2010. *Fish Surveys in Headwater Streams of the Nushagak and Kvichak River Drainages, Bristol Bay, Alaska, 2008–2010*. Anchorage, AK: Fisheries Research and Consulting.

Woolington, J. D. 2004. Unit 17 moose management report. Pages 246–266 in C. Brown (ed.), *Moose Management Report of Survey and Inventory Activities: 1 July 2001–30 June 2003*. Juneau, AK: Alaska Department of Fish and Game.

Woolington, J. D. 2009. Mulchatna caribou management report, Units 9B, 17, 18 south, 19A & 19B. Pages 11-31 in P. Harper (ed.), *Caribou Management Report Of Survey And Inventory Activities, 1 July 2006–1 July 2008*. Juneau, AK: Alaska Department of Fish and Game.

Personal Communications

Baker, T. Area Fishery Research Biologist, Bristol Bay Salmon Program, Alaska Department of Fish and Game, Anchorage, AK. July 8, 2011—email to Rebecca Shaftel.

15.1.7 Chapter 6—Mine Scenarios

ADEC (Alaska Department of Environmental Conservation). 2012. *Situation Report: Fairbanks Gold Mining, Inc. Cyanide Water Spill*. Anchorage, AK: Alaska Department of Environment and Conservation, Division of Spill Prevention and Response, Prevention and Emergency Response Program.

ADF&G (Alaska Department of Fish and Game). 2013. *Invasive Species. Overview*. http://www.adfg.alaska.gov/index.cfm?adfg=invasive.main. Accessed: March 4, 2013.

BC (Province of British Columbia, Canada). 2001. *Ambient Water Quality Guidelines for Manganese*. Ministry of Environment.

BC (Province of British Columbia, Canada). 2008. *Ambient Water Quality Guidelines for Iron*. Ministry of Environment.

Biesinger, K. E., and G. M. Christensen. 1972. Effects of various metals on survival, growth, reproduction, and metabolism of *Daphnia magna*. *Journal of the Fisheries Board of Canada* 39:1691–1700.

Blight, G. E. 2010. *Geotechnical Engineering for Mine Waste Storage Facilities.* Boca Raton, FL: CRC Press.

Blight, G. E., and A. B. Fourie. 2003. A Review of Catastrophic Flow Failures of Deposit of Mine Waste and Municipal Refuse. In L. Picarelli (ed.), *Proceedings of the International Workshop on Occurrence and Mechanisms of Flow-like Landslides in Natural Slopes and Earthfills, Sorrento, Italy, May 14–16, 2003*.

Blowes, D. W., C. J. Ptacek, and J. Jurjovec. 2003. Mill tailings: hydrogeology and geochemistry. Pages 95–116 in J. L. Jambar, D. W. Blowes, A. I. M. Ritchie (eds.), *Environmental Aspects of Mine Wastes: Mineralogical Association of Canada Short Course Series*. Volume 31.

Boutet, C., and C. Chaisemartin. 1973. Specific toxic properties of metallic salts. *Comptes Rendus Des Séances De La Societé De Biologie Et De Ses Filiales* 167(12):1933–1938.

CCME (Canadian Council of Ministers of the Environment). 1999. *Canadian Water Quality Guidelines for the Protection of Aquatic Life: Molybdenum*. Winnipeg, MB.

CCME (Canadian Council of Ministers of the Environment). 2009. *Canadian Water Quality Guidelines for the Protection of Aquatic Life: Boron*. Winnipeg, MB.

CEA (Chugach Electric Association). 2011. *Facilities*. Available: http://www.chugachelectric.com/inside-chugach/the-company/facilities. Accessed: January 3, 2013.

Chapman, P.M., H. Bailey, and E. Canaria. 2000. Toxicity of total dissolved solids associated with two mine effluents to chironomid larvae and early life stages of rainbow trout. *Environmental Toxicology and Chemistry* 19:210–214.

Eddmaps. 2013. Species Information. *Alaska Exotic Plant Information Clearinghouse*. Available: http://www.eddmaps.org/alaska/species/. Accessed: March 4, 2013.

Environment Canada and Health Canada. 2010. *Screening Assessment for the Challenge: Antimony trioxide (antimony oxide)*. Ottawa, ON. Available: http://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=9889ABB5-1.

Environment Canada and Health Canada. 2011. *Screening Assessment for the Challenge: Cobalt, Cobalt Chloride, Sulfuric Acid, Cobalt (2+) Salt (1:1), and Sulfuric Acid, Cobalt Salt.* Ottawa, ON. Available: http://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=8E18277B-1.

Etcheverry, D. 2012. *Aquatic Invasive Plants: A New Focus for Alaskan Invasive Species Management*. Presentation at the 13th Annual Invasive Species Conference, University of Alaska Fairbanks. http://www.uaf.edu/files/ces/cnipm/annualinvasivespeciesconference/13thAnnualMeetingProceedings/AISC-Elodea-D.Etcheverry_2012.pdf. Accessed: March 4, 2013.

Gammons, C. H., L. N. Harris, J. M. Castro, P. A. Cott, and B. W. Hanna. 2009. Creating lakes from open pit mines: processes and considerations, with emphasis on northern environments. *Canadian Technical Report of Fisheries and Aquatic Sciences* 2826.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Goldman, C. 1960. Primary productivity and limiting factors in three lakes of the Alaska Peninsula. *Ecological Monographs* 30:207–230.

Hidalgo, P., and I. G. R. Gutz. 2001. Determination of low concentrations of the flotation reagent ethyl xanthate by sampled DC polarography and flow injection with the amperometric detection. *Talanta* 54:403–409.

HSWCD (Homer Soil and Water Conservation District). 2007. *Managing Invasive Plants in Wetlands of the Kenai Peninsula: Developing a Management Program of Reed Canarygrass Infestations, FY2007 Progress Report Summary*. Homer, AK.

Laurence, D. 2011. Establishing a sustainable mining operation: an overview. *Journal of Cleaner Production* 19(2-3):278–284.

Marsik, T. 2009. *Basic Study of Renewable Energy Alternatives for Electricity Generation in Dillingham/Aleknagik Region*. Prepared for Nushagak Electric and Telephone Cooperative, Inc., by Bristol Bay Environmental Science Lab, University of Alaska Fairbanks, Bristol Bay Campus.

Merz, J. E., J. R. Smith, M. L. Workman, J. D. Setka, and B. Mulchaey. 2008. Aquatic macrophyte encroachment in Chinook salmon spawning beds: lessons learned from gravel enhancement monitoring in the Lower Mokelumne River, California. *North American Journal of Fisheries Management* 28:1568–1577.

ML&P (Municipal Light and Power). 2012. *About ML&P*. Available: http://www.mlandp.com/redesign/about_mlp.htm. Accessed: January 3, 2013.

Moore, J., and D. Schindler. 2004. Nutrient export from freshwater ecosystems by anadromous sockeye salmon (*Oncorhynchus nerka*). *Canadian Journal of Fisheries and Aquatic Sciences* 61:1582–1589.

NDM (Northern Dynasty Minerals). 2006. *Pebble Project – Water Right Applications Submitted to the Alaska Department of Natural Resources.* Available: http://dnr.alaska.gov/mlw/mining/largemine/pebble/water-right-apps/index.cfm. Accessed: October 12, 2012.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008*. Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Reeve, A. S., and M. B. Gracz. 2008. Simulating the hydrogeologic setting of peatlands in the Kenai Peninsula Lowlands, Alaska. *Wetlands* 28:92–106.

Singer, D. A., V. I. Berger, and B. C. Moring. 2008. *Porphyry Copper Deposits of the World: Database and Grade and Tonnage Models, 2008*. Open File Report 2008-1155. U.S. Geological Survey.

Swedish Chemicals Inspectorate. 2008. *European Union Risk Assessment Report: Diantimony Trioxide Risk Assessment*. Luxembourg: Office for Official Publications of the European Communities.

USEPA (U.S. Environmental Protection Agency). 1976. *Quality Criteria for Water*. Washington, DC.

USEPA (U.S. Environmental Protection Agency). 1980. *Ambient Water Quality Criteria for Antimony*. EPA/440/5-80-020. Washington, DC: Office of Water.

USEPA (U.S. Environmental Protection Agency). 1997. *EPA Can Do More to Minimize Hardrock Mine Liabilities.* EiDMF6-08-0016-7100223. June 11. Washington, DC: Office of the Inspector General.

USEPA (U.S. Environmental Protection Agency). 1998. *Guidelines for Ecological Risk Assessment.* EPA/630/R-95/002F. April. Washington, DC: Office of Research and Development.

USEPA (U.S. Environmental Protection Agency). 2011. *A Field-Based Aquatic Life Benchmark for Conductivity in Central Appalachian Streams (Final Report)*. EPA/600/R-10/023F. Washington, DC.

USEPA (U.S. Environmental Protection Agency). 2013. ECOTOX User Guide: ECOTOXicology Database System. Version 4.0. Available: http://www.epa.gov/ecotox. Accessed: October 2013.

USFWS (U.S. Fish and Wildlife Service). 2012. *National Wetlands Inventory.* Available: http://www.fws.gov/wetlands/Wetlands-Mapper.html. Accessed: December 11, 2012.

USGS (U.S. Geological Survey). 2012a. *National Hydrography Dataset.* High Resolution, Alaska. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

USGS (U.S. Geological Survey). 2012b. Water Resources of Alaska. USGS Alaska Science Center Water Resources Office. Last updated: December 12, 2012. Available: http://alaska.usgs.gov/science/water/index.php. Accessed: April 17, 2013.

Wells, P. S. 2011. Long term in situ behaviour of oil sands fine tailings in Suncor's Pond 1A. *Proceedings from the Tailings and Mine Waste 2011 Conference, Vancouver, British Columbia, November 6–9, 2011*.

15.1.8 Chapter 7—Mine Footprint

ADF&G (Alaska Department of Fish and Game). 2012. *Alaska Freshwater Fish Inventory Database*. Available: http://www.adfg.alaska.gov/index.cfm?adfg=ffinventory.main.

ADF&G (Alaska Department of Fish and Game). 2013. *Anadromous Waters Catalog, Overview*. Available: http://www.adfg.alaska.gov/sf/SARR/AWC/. Accessed: April 9, 2013.

Alexander, R., E. Boyer, R. Smith, G. Schwarz, and R. Moore. 2007. The role of headwater streams in downstream water quality. *Journal of the American Water Resources Association* 43:41–59.

Anderson, K. E., A. J. Paul, E. McCauley, L. J. Jackson, J. R. Post, and R. M. Nisbet. 2006. Instream flow needs in streams and rivers: the importance of understanding ecological dynamics. *Frontiers in Ecology and the Environment* 4:309–318.

Armstrong, J. B., and D. E. Schindler. 2013. Going with the flow: spatial distributions of juvenile coho salmon track an annually shifting mosaic of water temperature. *Ecosystems* 16(8):1429–1441.

Armstrong, J. B., D. E. Schindler, K. L. Omori, C. P. Ruff, and T. P. Quinn. 2010. Thermal heterogeneity mediates the effects of pulsed subsidies across a landscape. *Ecology* 91:1445–1454.

Arthington, A. H., S. E. Bunn, N. L. Poff, and R. J. Naiman. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications* 16:1311–1318.

Bartholow, J. M. 2010. Constructing an interdisciplinary flow regime recommendation. *Journal of the American Water Resources Association* 46:892–906.

Beechie, T. J., E. Beamer, and L. Wasserman. 1994. Estimating coho salmon rearing habitat and smolt production losses in a large river basin, and implications for habitat restoration. *North American Journal of Fisheries Management* 14:797–811.

Blair, G. R., D. E. Rogers, and T. P. Quinn. 1993. Variation in life history characteristics and morphology of sockeye salmon in the Kvichak River system, Bristol Bay, Alaska. *Transactions of the American Fisheries Society* 122(4):550–559.

Bovee, K. D. 1982. *A Guide to Stream Habitat Analysis Using the Instream Flow Incremental Methodology*. Instream Flow Group Information Paper No. 12, FWS/OBS-2/26. Fort Collins, CO: U.S. Fish and Wildlife Service.

Bovee, K. D., B. L. Lamb, J. M. Bartholow, C. D. Stalnaker, J. Taylor, and J. Henrikson. 1998. *Stream Habitat Analysis Using the Instream Flow Incremental Methodology*. Information and Technical Report USGS/BRD-1998-0004. U.S. Geological Survey, Biological Resources Division.

Bradford, M. J., G. C. Taylor, J. A. Allan, and P. S. Higgins. 1995. An experimental study of the stranding of juvenile coho salmon and rainbow trout during rapid flow decreases under winter conditions. *North American Journal of Fisheries Management* 15:47–479.

Brannon, E. 1987. Mechanisms stabilizing salmonid fry emergence timing. *Canadian Special Publication of Fisheries and Aquatic Sciences* 96:120–124.

Brown, T. G., and G. F. Hartman. 1988. Contribution of seasonally flooded lands and minor tributaries to the production of coho salmon in Carnation Creek, British Columbia. *Transactions of the American Fisheries Society* 117:546–551.

Brown, R. S., W. A. Hubert, and S. F. Daly. 2011. A primer on winter, ice, and fish: what fisheries biologists should know about winter ice processes and stream-dwelling fish. *Fisheries* 36:8–26.

Bryant, M. D., N. D. Zymonas, and B. E. Wright. 2004. Salmonids on the fringe: abundance, species composition, and habitat use of salmonids in high-gradient headwater streams, southeast Alaska. *Transactions of the American Fisheries Society* 133:1529–1538.

Bue, B. G., S. M. Fried, S. Sharr, D. G. Sharp, J. A. Wilcock, and H. J. Geiger. 1988. Estimating salmon escapement using area-under-the-curve, aerial observer efficiency, and stream-life estimates: the Prince William Sound pink salmon example. *North Pacific Anadromous Fish Commission Bulletin* 1:240–250.

Callegary, J., C. Kikuchi, J. Koch, M. Lilly, and S. Leake. 2013. Review: groundwater in Alaska (USA). *Hydrogeology Journal* 21(1):25–39.

Chadwick, M. A., and A. D. Huryn. 2007. Role of habitat in determining macroinvertebrate production in an intermittent-stream system. *Freshwater Biology* 52:240–251.

Collen, P., and R. J. Gibson. 2001. The general ecology of beavers (*Castor* spp.), as related to their influence on stream ecosystems and riparian habitats, and the subsequent effects on fish—A review. *Reviews in Fish Biology and Fisheries* 10:439–461.

Compton, J. E., M. R. Church, S. T. Larned, and W. E. Hogsett. 2003. Nitrogen export from forested watersheds in the Oregon Coast Range: the role of N-2-fixing red alder. *Ecosystem*s 6:773–785.

Cunjak, R. A. 1996. Winter habitat of selected stream fishes and potential impacts from land-use activity. *Canadian Journal of Fisheries and Aquatic Sciences* 53 (Supplement 1):267–282.

Curry, R. A., J. Gehrels, D. L. G. Noakes, and R. Swainson. 1994. Effects of river flow fluctuations on groundwater discharge through brook trout, *Salvelinus fontinalis*, spawning and incubation habitats. *Hydrobiologia* 277:121–134.

Dekar, M. P., R. S. King, J. A. Back, D. F. Whigham, and C. M. Walker. 2012. Allochthonous inputs from grass-dominated wetlands support juvenile salmonids in headwater streams: evidence from stable isotopes of carbon, hydrogen, and nitrogen. *Freshwater Science* 31:121–132.

Doucett, R. R., G. Power, D. R. Barton, R. J. Drimmie, and R. A. Cunjak. 1996. Stable isotope analysis of nutrient pathways leading to Atlantic salmon. *Canadian Journal of Fisheries and Aquatic Sciences* 53:2058–2066.

Dye, J. E., and C. J. Schwanke. 2009. *Report to the Alaska Board of Fisheries for the Recreational Fisheries of Bristol Bay, 2007, 2008, and 2009*. Alaska Deptartment of Fish and Game, Division of Sport Fish, Research and Technical Services.

Elliott, G. V., and J. E. Finn. 1983. *Fish Utilization of Several Kenai River Tributaries: 1982 Field Report*. Anchorage, AK: U.S. Fish and Wildlife Service, Special Studies.

Estes, C. C. 1998. *Annual Summary of Instream Flow Reservations and Protection in Alaska*. Alaska Department of Fish and Game, Division of Sport Fish, Research and Technical Services.

Ferreira, V., V. Gulis, and M. A. S. Graca. 2006. Whole-stream nitrate addition affects litter decomposition and associated fungi but not invertebrates. *Oecologia* 149:718–729.

Freeman, M. C., C. M. Pringle, and C. R. Jackson. 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. *Journal of the American Water Resources Association* 43(1):5–14.

Gesch, D. 2007. Chapter 4–The National Elevation Dataset. Pages 99–118 in Maune, D. (ed.), *Digital Elevation Model Technologies and Applications: The DEM Users Manual*. 2nd edition. Bethesda, MD: American Society for Photogrammetry and Remote Sensing.

Gesch, D., M. Oimoen, S. Greenlee, C. Nelson, M. Steuck, and D. Tyler. 2002. The National Elevation Dataset. *Photogrammatic Engineering and Remote Sensing* 68(1):5–11.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Gomez-Uchida, D., J. E. Seeb, M. J. Smith, C. Habicht, T. P. Quinn, and L. W. Seeb. 2011. Single nucleotide polymorphisms unravel hierarchical divergence and signatures of selection among Alaskan sockeye salmon (*Oncorhynchus nerka*) populations. *BMC Evolutionary Biology* 11(1):48.

Hall, J. V., W. E. Frayer, and B. O. Wilen. 1994. *Status of Alaska Wetlands*. U.S. Fish and Wildlife Service, Alaska Region, Anchorage, AK. Available: http://www.fws.gov/wetlands/Documents/Status-of-Alaska-Wetlands.pdf.

Hancock, P. J. 2002. Human impacts on the stream-groundwater exchange zone. *Environmental Management* 29:763–781.

Hendry, A. P., J. E. Hensleigh, and R. R. Reisenbichler. 1998. Incubation temperature, developmental biology, and the divergence of sockeye salmon (*Oncorhynchus nerka*) within Lake Washington. *Canadian Journal of Fisheries and Aquatic Sciences* 55:1387–1394.

Henning, J. A., R. E. Gresswell, and I. A. Fleming. 2006. Juvenile salmonid use of freshwater emergent wetlands in the floodplain and its implications for conservation management. *North American Journal of Fisheries Management* 26:367–376.

Hilborn, R., B. G. Bue, and S. Sharr. 1999. Estimating spawning escapements from periodic counts: a comparison of methods. *Canadian Journal of Fisheries and Aquatic Sciences* 56(5):888–896.

Hilborn, R., T. Quinn, D. Schindler, and D. Rogers. 2003. Biocomplexity and fisheries sustainability. *Proceedings of the National Academy of Sciences of the United States of America* 100:6564–6568.

Hillman, T. W., J. W. Mullan, and J. S. Griffith. 1992. Accuracy of underwater counts of juvenile Chinook salmon, coho salmon, and steelhead. *North American Journal of Fisheries Management* 12:598–603.

Hodgson, S., and T. P. Quinn. 2002. The timing of adult sockeye salmon migration into fresh water: adaptations by populations to prevailing thermal regimes. *Canadian Journal of Zoology* 80:542–555.

Holt, K. R., and S. P. Cox. 2008. Evaluation of visual survey methods for monitoring Pacific salmon (*Oncorhynchus* spp.) escapement in relation to conservation guidelines. *Canadian Journal of Fisheries and Aquatic Sciences* 65(2):212−226.

Huusko, A., L. Greenberg, M. Stickler, T. Linnansaari, M. Nykänen, T. Vehanen, S. Koljonen, P. Louhi, and K. Alfredsen. 2007. Life in the ice lane: the winter ecology of stream salmonids. *River Research and Applications* 23:469–491.

Irons J. G., III, S. R. Ray, L. K. Miller, and M. W. Oswood. 1989. Spatial and seasonal patterns of streambed water temperatures in an Alaskan subarctic stream. *Proceedings of the Symposium on Headwaters Hydrology* June:381–390. American Water Resources Association. As cited in Reynolds, J. B. 1997. Ecology of overwintering fishes in Alaskan freshwaters. Pages 281–302 in A. M. Milner and M. W. Oswood (eds.), *Freshwaters of Alaska: Ecological Syntheses*. New York, NY: Springer.

Johnson, J., and P. Blanche. 2012. *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes – Southwestern Region, Effective June 1, 2012*. Special Publication No. 12-08. Anchorage, AK: Alaska Department of Fish and Game.

Jones, E. L., III, S. Heinl, and K. Pahlke. 2007. Aerial counts. Pages 399–410 in D. H. Johnson, B. M. Shrier, J. S. O'Neal, J. A. Knutzen, X. Augerot, T. A. O'Neil, and T. N. Pearsons, (eds.), *Salmonid Field Protocols Handbook: Techniques for Assessing Status and Trends in Salmon and Trout Populations*. Bethesda, MD: American Fisheries Society.

King, R. S., C. M. Walker, D. F. Whigham, S. Baird, and J. A. Back. 2012. Catchment topography and wetland geomorphology drive macroinvertebrate community structure and juvenile salmonid distributions in southcentral Alaska headwater streams. *Freshwater Science* 31(2):341–364.

Lang, D. W., G. H. Reeves, J. D. Hall, and M. S. Wipfli. 2006. The influence of fall-spawning coho salmon (*Oncorhynchus kisutch*) on growth and production of juvenile coho salmon rearing in beaver ponds on the Copper River Delta, Alaska. *Canadian Journal of Fisheries and Aquatic Sciences* 63:917–930.

Lytle, D. A., and N. L. R. Poff. 2004. Adaptation to natural flow regimes. *Trends in Ecology & Evolution* 19:94–100.

Matthews, W. J., and E. Marsh-Matthews. 2003. Effects of drought on fish across axes of space, time and ecological complexity. *Freshwater Biology* 48:1232–1253.

Meyer, J. L., D. L. Strayer, J. B. Wallace, S. L. Eggert, G. S. Helfman, and N. E. Leonard. 2007. The contribution of headwater streams to biodiversity in river networks. *Journal of the American Water Resources Association* 43(1):86−103.

Morstad, S. 2003. *Kvichak River Sockeye Salmon Spawning Ground Surveys, 1955–2002*. Regional Information Number 2A02-32. Anchorage, AK: Alaska Department of Fish and Game.

Nickelson, T. E., J. D. Rodgers, S. L. Johnson, and M. F. Solazzi. 1992. Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. *Canadian Journal of Fisheries and Aquatic Sciences* 49:783–789.

Olsen, J. B., S. J. Miller, W. J. Spearman, and J. K. Wenburg. 2003. Patterns of intra-and inter-population genetic diversity in Alaskan coho salmon: implications for conservation. *Conservation Genetics* 4(5):557–569.

Parken, C. K., R. E. Bailey, and J. R. Irvine. 2003. Incorporating uncertainty into area-under-the-curve and peak count salmon escapement estimation. *North American Journal of Fisheries Management* 23(1):78–90.

Pess, G. R., D. R. Montgomery, E. A. Steel, R. E. Bilby, B. E. Feist, and H. M. Greenberg. 2002. Landscape characteristics, land use, and coho salmon (*Oncorhynchus kisutch*) abundance, Snohomish River, Wash., U.S.A. *Canadian Journal of Fisheries and Aquatic Sciences* 59:613–623.

Pess, G. R., S. A. Morley, J. L. Hall, and R. K. Timm. 2005. Monitoring Floodplain Restoration. Pages 127–166 in P. Roni, (ed.), *Monitoring Stream and Watershed Restoration*. Bethesda, MD: American Fisheries Society.

Piccolo, J. J., and M. S. Wipfli. 2002. Does red alder (*Alnus rubra*) in upland riparian forests elevate macroinvertebrate and detritus export from headwater streams to downstream habitats in southeastern Alaska? *Canadian Journal of Fisheries and Aquatic Sciences* 59:503–513.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008.* Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime: a paradigm for river conservation and restoration. *Bioscience* 47:769–784.

Poff, N., J. Olden, D. Pepin, and B. Bledsoe. 2006. Placing global stream flow variability in geographic and geomorphic contexts. *Regulated Rivers: Research & Management* 22:149–166.

Poff, N., B. Richter, A. Arthington, S. Bunn, R. Naiman, C. Apse, E. Kendy, A. Warner, R. Jacobson, and K. Rogers. 2009. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology* 55:147–170.

Poff, N. L., and J. K. H. Zimmerman. 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology* 55:194–205.

Pollock, M. M., M. Heim, and D. Werner. 2003. Hydrologic and geomorphic effects of beaver dams and their influence on fishes. *The Ecology and Management of Wood in World Rivers* 37:213–233.

Pollock, M. M., G. R. Pess, and T. J. Beechie. 2004. The importance of beaver ponds to coho salmon production in the Stillaguamish River basin, Washington, USA. *North American Journal of Fisheries Management* 24:749–760.

Postel, S., and B. Richter. 2003. *Rivers for Life: Managing Water for People and Nature.* Washington, DC: Island Press.

Power, G., R. S. Brown, and J. G. Imhof. 1999. Groundwater and fish—Insights from northern North America. *Hydrological Processes* 13:401–422.

Quinn, T. P. 2005. *The Behavior and Ecology of Pacific Salmon and Trout.* Seattle, WA: University of Washington Press.

Quinn, T. P., A. P. Hendry, and G. B. Buck. 2001. Balancing natural and sexual selection in sockeye salmon: interactions between body size, reproductive opportunity and vulnerability to predation by bears. *Evolutionary Ecology Research* 3(8):917−937.

Quinn, T. P., A. P. Hendry, and L. A. Wetzel. 1995. The influence of life history trade-offs and the size of incubation gravels on egg size variation in sockeye salmon (*Oncorhynchus nerka*). *Oikos* (1995):425−438.

Quinn, T. P., H. B. Rich Jr, D. Gosse, N. Schtickzelle, and J. Grant. 2012. Population dynamics and asynchrony at fine spatial scales: a case history of sockeye salmon (*Oncorhynchus nerka*) population structure in Alaska, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 69(2):297−306.

Rains, M. C. 2011. Water sources and hydrodynamics of closed-basin depressions, Cook Inlet Region, Alaska. *Wetlands* 31:377–387.

Rand, P. S., C. Kellopn, X. Augerot, M. Goslin, J. R. Irvine, and G. T. Ruggerone. 2007. Comparison of sockeye salmon (*Oncorhynchus nerka*) monitoring in the Fraser River Basin, British Columbia, Canada and Bristol Bay, Alaska, USA. *North Pacific Anadromous Fish Commission Bulletin* 4:271−284.

Reynolds, J. B. 1997. Ecology of overwintering fishes in Alaskan freshwaters. Pages 281–302 in A. M. Milner and M. W. Oswood (eds.), *Freshwaters of Alaska: Ecological Syntheses*. New York, NY: Springer.

Richardson, J. S., R. J. Naiman, F. J. Swanson, and D. E. Hibbs. 2005. Riparian communities associated with Pacific Northwest headwater streams: assemblages, processes, and uniqueness. *Journal of the American Water Resources Association* 41(4):935−947.

Richter, B. D. 2010. Re-thinking environmental flows: from allocations and reserves to sustainability boundaries. *River Research and Applications* 26:1052–1063.

Richter, B., M. Davis, C. Apse, and C. Konrad. 2012. A presumptive standard for environmental flow protection. *River Research and Applications* 28:1312–1321.

Rogers, L. A., and D. E. Schindler. 2008. Asynchrony in population dynamics of sockeye salmon in southwest Alaska. *Oikos* 117:1578–1586.

Ruff, C. P., D. E. Schindler, J. B. Armstrong, K. T. Bentley, G. T. Brooks, G. W. Holtgrieve, M. T. McGlauflin, C. E. Torgersen, and J. E. Seeb. 2011. Temperature-associated population diversity in salmon confers benefits to mobile consumers. *Ecology* 92:2073–2084.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. *Nature* 465:609–612.

Shaftel, R., R. King, and J. Back. 2011. Breakdown rates, nutrient concentrations, and macroinvertebrate colonization of bluejoint grass litter in headwater streams of the Kenai Peninsula, Alaska. *Journal of the North American Benthological Society* 30:386–398.

Shaftel, R., R. King, and J. Back. 2012. Alder cover drives nitrogen availability in Kenai lowland headwater streams, Alaska. *Biogeochemistry* 107:135-148.

Slaughter, C. W. 1990. Aufeis formation and prevention. Pages 433–458 in W. L. Ryan and R. D. Crissman (eds.), *Cold Regions Hydrology and Hydraulic*s. New York, NY: American Society of Civil Engineers.

Solazzi, M., T. Nickelson, S. Johnson, and J. Rodgers. 2000. Effects of increasing winter rearing habitat on abundance of salmonids in two coastal Oregon streams. *Canadian Journal of Fisheries and Aquatic Sciences* 57:906–914.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences* 58:325–333.

Szarzi, N. J, S. J. Fleischman, R. A. Clark, and C. M. Kerkvliet. 2007. *Stock status and recommended escapement goal for Anchor River Chinook salmon*. Fisheries Manuscript No. 07-05. Alaska Department of Fish and Game, Divisions of Sport Fish and Commercial Fisheries.

Tank, J. L., E. J. Rosi-Marshall, N. A. Griffiths, S. A. Entrekin, and M. L. Stephen. 2010. A review of allochthonous organic matter dynamics and metabolism in streams. *Journal of the North American Benthological Society* 29:118–146.

USACE (U.S. Army Corps of Engineers). 2009. Alaska District Regulatory Guidance Letter, RGL ID No. 09-01.

USACE (U.S. Army Corps of Engineers). 2011. *Memorandum for Regulatory Division, Alaska District, 3 March 2011*.

USFWS (U.S. Fish and Wildlife Service). 2012. *National Wetlands Inventory.* Available: http://www.fws.gov/wetlands/Wetlands-Mapper.html. Accessed: December 11, 2012.

USGS (U.S. Geological Survey). 2012a. *National Hydrography Dataset*. High Resolution, Alaska. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

USGS (U.S. Geological Survey). 2012b. Water Resources of Alaska. USGS Alaska Science Center Water Resources Office. Last updated: December 12, 2012. Available: http://alaska.usgs.gov/science/water/index.php. Accessed: April 17, 2013.

USGS (U.S. Geological Survey). 2013. National Elevation Dataset (2 arc second), ArcGrid. August 26. [n59w155.zip, n59w156.zip, n59w157.zip, n59w158.zip, n60w154.zip, n60w155.zip, n60w156.zip, n60w157.zip, n60w158.zip, n60w159.zip, n60w160.zip, n61w153.zip, n61w154.zip, n61w155.zip, n61w156.zip, n61w157.zip, n61w158.zip, n61w159.zip, n61w160.zip] Available: http://viewer.nationalmap.gov/viewer/.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37:130–137.

Walker, C. M., R. S. King, D. F. Whigham, and S. J. Baird. 2012. Landscape and wetland influences on headwater stream chemistry in the Kenai Lowlands, Alaska. *Wetlands* 32(2):301−310.

Webster, J. R., and E. F. Benfield. 1986. Vascular plant breakdown in fresh-water ecosystems. *Annual Review of Ecology and Systematics* 17:567–594.

West, R. L., M. W. Smith, W. E. Barber, J. B. Reynolds, and H. Hop. 1992. Autumn migration and overwintering of Arctic grayling in coastal streams of the Arctic National Wildlife Refuge, Alaska. *Transactions of the American Fisheries Society* 121:709–715.

Wipfli, M. S., and C. V. Baxter. 2010. Linking ecosystems, food webs, and fish production: subsidies in salmonid watersheds. Fisheries 35(8):373–387.

Wipfli, M. S., and D. P. Gregovich. 2002. Export of invertebrates and detritus from fishless headwater streams in southeastern Alaska: implications for downstream salmonid production. *Freshwater Biology* 47:957–969.

Wipfli, M. S., and J. Musslewhite. 2004. Density of red alder (*Alnus rubra*) in headwaters influences invertebrate and detritus subsidies to downstream fish habitats in Alaska. *Hydrobiologia* 520:153–163.

Wipfli, M. S., J. S. Richardson, and R. J. Naiman. 2007. Ecological linkages between headwaters and downstream ecosystems: transport of organic matter, invertebrates, and wood down headwater channels. *Journal of the American Water Resources Association* 43:72–85.

Woody, C. A., and B. Higman. 2011. *Groundwater as Essential Salmon Habitat in Nushagak and Kvichak River Headwaters: Issues Relative to Mining*. Unpublished report. Available: http://www.fish4thefuture.com/pdfs/Groundwater_and_SalmonFINAL27Aug11.pdf.

Personal Communications

Morstad, S. Fishery Biologist III, ADF&G. September 2011—email of unpublished data to Rebecca Shaftel.

15.1.9 Chapter 8—Water Collection, Treatment, and Discharge

ADEC (Alaska Department of Environmental Conservation). 2011. *Water Quality Standards: Alaska Department of Environmental Conservation*. Report 18–AAC–70.

ADEC (Alaska Department of Environmental Conservation). 2012. *Water Quality Standards Amended as of April 8, 2012*. Juneau, AK: Alaska Department of Environmental Conservation.

ADNR (Alaska Department of Natural Resources). 2012. *Nixon Fork Mine*. Available: http://dnr.alaska.gov/mlw/mining/largemine/nixonfork/. Accessed: April 11, 2012.

Angilletta, M. J., E. Ashley Steel, K. K. Bartz, J. G. Kingsolver, M. D. Scheuerell, B. R. Beckman, and L. G. Crozier. 2008. Big dams and salmon evolution: changes in thermal regimes and their potential evolutionary consequences. *Evolutionary Applications* 1:286–299.

ANZECC (Australia and New Zealand Environment and Conservation Council). 2000. Australian and New Zealand guidelines for fresh and marine water quality. Volume 2. *Aquatic Ecosystems: Rationale and Background Information*. Canberra, Australia: Australia and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand.

Armstrong, J. B., D. E. Schindler, K. L. Omori, C. P. Ruff, and T. P. Quinn. 2010. Thermal heterogeneity mediates the effects of pulsed subsidies across a landscape. *Ecology* 91:1445–1454.

Barry, K. L., J. A. Grout, C. D. Levings, B. H. Nidle, and G. E. Piercey. 2000. Impacts of acid mine drainage on juvenile salmonids in an estuary near Britannia Beach in Howe Sound, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences* 57:2032–2043.

Beacham, T. D., and C. B. Murray. 1990. Temperature, egg size, and development of embryos and alevins of five species of Pacific salmon: a comparative analysis. *Transactions of the American Fisheries Society* 119:927–945.

Borgmann, U., C. Janssen, R. Blust, K. Brix, R. Dwyer, R. Erickson, L. Hare, S. Luoma, P. Paquin, C. Roberts, and W.-X. Wang. 2005. Incorporation of dietborne metals exposure into regulatory frameworks. Pages 153–190 in J. Meyer, W. Adams, K. Brix, S. Luoma, D. Mount, W. Stubblefield, and C. Wood, editors. *Toxicity of Dietborne Metals to Aquatic Organisms*. Pensacola, FL: SETAC Press.

Brannon, E. 1987. Mechanisms stabilizing salmonid fry emergence timing. *Canadian Special Publication of Fisheries and Aquatic Sciences* 96:120–124.

Brix, K. V., D. K. DeForest, and W. J. Adams. 2011. The sensitivity of aquatic insects to divalent metals: a comparative analysis of laboratory and field data. *Science of the Total Environment* 409:4187–4197.

Bryant, M. 2009. Global climate change and potential effects on Pacific salmonids in freshwater ecosystems of southeast Alaska. *Climatic Change* 95:169–193.

Buchwalter, D. B., D. J. Cain, W. H. Clements, and S. N. Luoma. 2008. Using biodynamic models to reconcile differences between laboratory toxicity tests and field biomonitoring with aquatic insects (vol. 41, pg 4821, 2007). *Environmental Science & Technology* 42:3117–3117.

Chapman, G. 1975. Toxicity of copper, cadmium and zinc to Pacific Northwest salmonids. Corvallis, OR: U.S. Environmental Protection Agency, Western Fish Toxicology Station, National Water Quality Laboratory.

Chapman, G. A. 1978. Toxicities of cadmium, copper, and zinc to 4 juvenile stages of Chinook salmon and steelhead. *Transactions of the American Fisheries Society* 107:841–847.

Chapman, P. M., H. Bailey, and E. Canaria. 2000. Toxicity of total dissolved solids associated with two mine effluents to chironomid larvae and early life stages of rainbow trout. *Environmental Toxicology and Chemistry* 19:210–214.

Clements, W. H., D. M. Carlisle, J. M. Lazorchak, and P. C. Johnson. 2000. Heavy metals structure benthic communities in Colorado mountain streams. *Ecological Applications* 10:626–638.

Cohen, B. I. 2010. *Fraser River Sockeye Salmon: Past Declines. Future Sustainability?* Interim Report. Vancouver, BC: Commission of Inquiry into the Decline of Sockeye Salmon in the Fraser River.

Cohen, B. I. 2012. *The Uncertain Future of Fraser River Sockeye. Volume 2: Causes of the Decline*. Final Report CP32-93/2012E-2. Ottawa, ON: Commission of Inquiry into the Decline of Sockeye Salmon in the Fraser River.

Courtney, L. A., and W. H. Clements. 2002. Assessing the influence of water and substratum quality on benthic macroinvertebrate communities in a metal-polluted stream: an experimental approach. *Freshwater Biology* 47:1766–1778.

DeForest, D., J. Meyer, R. Genseer, B. Shephard, W. Adams, R. Dwyer, J. Gorsuch, and E. VanGenderen. 2011a. Are ambient water quality criteria for copper protective of olfactory impairment in fish? *Integrated Environmental Assessment and Management* 7:145–146.

DeForest, D., R. Gensemer, E. Van Genderen, and J. Gorsuch. 2011b. Protectiveness of water quality criteria for copper in western United States waters relative to predicted olfactory responses in juvenile Pacific salmon. *Integrated Environmental Assessment and Management* 7:336–347.

DeForest, D. K., and E. J. Van Genderen. 2012. Application of U.S. EPA guidelines in a bioavailability-based assessment of ambient water quality criteria for zinc in freshwater. *Environmental Toxicology and Chemistry* 31:1264-1272.

Earthworks. 2012. *U.S. Copper Porphyry Mines Report: The Track Record of Water Quality Impacts Resulting from Pipeline Spills, Tailings Failures and Water Collection and Treatment Failure*. Washington, DC.

Eisler, R. 2000. *Handbook of Chemical Risk Assessment: Health Hazards to Humans, Plants, and Animals. Volume 1: Metals*. Boca Raton, FL: Lewis Publisher.

Environment Canada and Health Canada. 2011. *Screening Assessment for the Challenge: Cobalt; Cobalt Chloride; Sulfuric Acid, Cobalt (2+) Salt (1:1); and Sulfuric Acid, Cobalt Salt*. Ottawa, ON. Available: http://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=8E18277B-1.

Erickson, R. J., D. R. Mount, T. L. Highland, J. R. Hockett, E. N. Leonard, V. R. Mattson, T. D. Dawson, and K. G. Lott. 2010. Effects of copper, cadmium, lead, and arsenic in a live diet on juvenile fish growth. *Canadian Journal of Fisheries and Aquatic Sciences* 67:1816–1826.

European Copper Institute. 2008. *Voluntary Risk Assessment of Copper, Copper II Sulphate Pentahydrate, Copper(I)oxide, Copper(II)oxide, Dicopper Chloride Trihydroxide*. Brussels, Belgium: European Copper Institute for the European Commission.

Farag, A. M., C. J. Boese, D. F. Woodward, and H. L. Bergman. 1994. Physiological changes and tissue metal accumulation in rainbow trout exposed to foodborne and waterborne metals. *Environmental Toxicology and Chemistry* 13:2021–2029.

Farag, A. M., D. E. Woodward, W. Brumbaugh, J. N. Goldstein, E. MacConnell, C. Hogstrand, and F. T. Barrows. 1999. Dietary effects of metals-contaminated invertebrates from the Coeur d'Alene River, Idaho, on cutthroat trout. *Transactions of the American Fisheries Society* 128:578–592.

Fuerstenau, M. C., B. M. Wakawa, R. K. Price, and R. D. Wellik. 1974. Toxicity of selected sulfhydryl collectors to rainbow trout. *Transactions of the Society of Mining Engineers* 256:337–341.

Gensemer, R. W., and R. C. Playle. 1999. The bioavailability and toxicity of aluminum in aquatic environments. *Critical Reviews in Environmental Science & Technology* 29:315–450.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Griffith, M. B., J. M. Lazorchak, and A. T. Herlihy. 2004. Relationships among exceedences of metals criteria, the results of ambient bioassays, and community metrics in mining-impacted streams. *Environmental Toxicology and Chemistry* 23:1786–1795.

Gustafson, O. 2012. *Active Metal Mines of the Fraser River Basin and Fish - Case Studies*. Technical Paper Series No. 1, White Paper No. 3. Pebble Limited Partnership.

Hansen, J. A., J. Lipton, P. G. Welsh, D. Cacela, and B. MacConnell. 2004. Reduced growth of rainbow trout (*Oncorhynchus mykiss*) fed a live invertebrate diet pre-exposed to metal-contaminated sediments. *Environmental Toxicology and Chemistry* 23:1902–1911.

Hansen, J. A., J. C. A. Marr, J. Lipton, D. Cacela, and H. L. Bergman. 1999. Differences in neurobehavioral response of Chinook salmon (*Oncorhynchus tshawytscha*) and rainbow trout (*Oncorhynchus mykiss*) exposed to copper and cobalt: behavioral avoidance. *Environmental Toxicology and Chemistry* 18:1972–1978.

Hecht, S., D. Baldwin, C. Mebane, T. Hawkes, S. Gross, and N. Scholz. 2007. *An Overview of Sensory Effects on Juvenile Salmonids Exposed to Dissolved Copper: Applying a Benchmark Concentration Approach to Evaluate Sublethal Neurobehavioral Toxicity*. Seattle, WA: National Oceanic and Atmospheric Administration, Northwest Fisheries Science Center.

Hedtke, S. F. 1984. Structure and function of copper-stressed aquatic microcosms. *Aquatic Toxicology* 5:227–244.

Hodgson, S., and T. P. Quinn. 2002. The timing of adult sockeye salmon migration into fresh water: adaptations by populations to prevailing thermal regimes. *Canadian Journal of Zoology* 80:542–555.

IPCS (International Programme on Chemical Safety). 2004. *Concise International Chemical Assessment Document 63. Manganese and its Compounds: Environmental Aspects*. Geneva, Switzerland: World Health Organization.

ITRC (Interstate Technology & Regulatory Council). 2010. *Bingham Canyon Water Treatment Plant, Kennecott South Zone*. Mining Waste Treatment Technology Selection Web. Washington, DC: Interstate Technology & Regulatory Council, Mining Waste Team. Available: http://www.itrcweb.org/miningwaste-guidance/cs48_kennecott_south.htm.

Jennings, S., D. Neuman, and P. Blicker. 2008. *Acid Mine Drainage and Effects on Fish Health and Ecology: A Review*. Bozeman, Montana: Reclamation Research Group, LLC.

Jensen, A., and B. Johnsen. 1999. The functional relationship between peak spring floods and survival and growth of juvenile Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*). *Functional Ecology* 13:778–785.

Joling, D. 2011. Mine Executive Says Pebble, Fishing Can Co-Exist. *Anchorage Daily News* (Anchorage, AK). March 4, 2011.

Kiffney, P. M., and W. H. Clements. 1996. Size-dependent response of macroinvertebrates to metals in experimental streams. *Environmental Toxicology and Chemistry* 15:1352–1356.

Le Jeune, A. H., M. Charpin, V. Deluchat, and J.-F. Briand. 2006. Effect of copper sulphate treatment on natural phytoplanktonic communities. *Aquatic Toxicology* 80:267–280.

Leland, H. V., and J. L. Carter. 1984. Effects of copper on species composition of periphyton in a Sierra Nevada, California, stream. *Freshwater Biology* 14:281–296.

Leland, H. V., and J. L. Carter. 1985. Effects of copper on production of periphyton, nitrogen-fixation and processing of leaf litter in a Sierra Nevada, California, stream. *Freshwater Biology* 15:155–173.

Leland, H. V., S. V. Fend, T. L. Dudley, and J. L. Carter. 1989. Effects of copper on species composition of benthic insects in a Sierra Nevada, California, stream. *Freshwater Biology* 21:163–179.

Lorz, H., and B. McPherson. 1977. *Effects of Copper and Zinc on Smoltification of Coho Salmon*. Corvallis, OR: U.S. Environmental Protection Agency, Environmental Research Laboratory.

Luoma, S. N., and P. S. Rainbow. 2005. Why is metal bioaccumulation so variable? Biodynamics as a unifying concept. *Environmental Science & Technology* 39:1921–1931.

MacDonald, D., J. Sinclair, M. Crawford, H. Principe, and M. Meneghetti. 2011. *Potential Effects of Contaminants on Fraser River Sockeye Salmon*. Cohen Commission Technical Report 2. Vancouver, BC: MacDonald Environmental Sciences, Ltd.

Marchand, E. A. 2002. Minerals and mine drainage. *Water Environment Research* 74:746–772.

McCullough, D. A., J. M. Bartholow, H. I. Jager, R. L. Beschta, E. F. Cheslak, M. L. Deas, J. L. Ebersole, J. S. Foott, S. L. Johnson, K. R. Marine, M. G. Mesa, J. H. Petersen, Y. Souchon, K. F. Tiffan, and W. A. Wurtsbaugh. 2009. Research in thermal biology: burning questions for coldwater stream fishes. *Reviews in Fisheries Science* 17:90–115.

McIntyre, J. K., D. H. Baldwin, J. P. Meador, and N. L. Scholz. 2008a. Chemosensory deprivation in juvenile coho salmon exposed to dissolved copper under varying water chemistry conditions. *Environmental Science & Technology* 42:1352–1358.

McIntyre, J. K., D. H. Baldwin, J. P. Meador, and N. L. Scholz. 2008b. Chemosensory deprivation in juvenile coho salmon exposed to dissolved copper under varying water chemistry conditions (42:1352-1358, 2008). *Environmental Science & Technology* 42:6774-6775 (Addition or correction).

Mebane, C. A. 2010. *Cadmium Risks to Freshwater Life: Derivation and Validation of Low-Effect Criteria Values Using Laboratory and Field Studies*. Investigations Report 2006-5245, Version 1.2. Reston VA: U.S. Geological Survey.

Mebane, C. A., and D. L. Arthaud. 2010. Extrapolating growth reductions in fish to changes in population extinction risks: copper and Chinook salmon. *Human and Ecological Risk Assessment* 16:1026–1065.

Mebane, C. A., F. S. Dillon, and D. P. Hennessy. 2012. Acute toxicity of cadmium, lead, zinc, and their mixtures to stream-resident fish and invertebrates. *Environmental Toxicology and Chemistry* 31:1334–1348.

Mebane, C. A., D. P. Hennessy, and F. S. Dillon. 2008. Developing acute-to-chronic toxicity ratios for lead, cadmium, and zinc using rainbow trout, a mayfly, and a midge. *Water, Air, and Soil Pollution* 188(1–4):41–66. Available: http://dx.doi.org/10.1007/s11270-007-9524-8.

Mellina, E., R. D. Moore, S. G. Hinch, J. S. Macdonald, and G. Pearson. 2002. Stream temperature responses to clearcut logging in British Columbia: the moderating influences of groundwater and headwater lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 59:1886–1900.

Meyer, J. S., and W. J. Adams. 2010. Relationship between biotic ligand model-based water quality criteria and avoidance and olfactory responses to copper by fish. *Environmental Toxicology and Chemistry* 29:2096–2103.

Meyer, J., W. Adams, K. Brix, D. Mount, W. Stubblefield, and C. Wood. 2005. *Toxicity of Dietborne Metals to Aquatic Organisms*. Pensacola, FL: SETAC Press.

Mortazavi, S. 2008. *Application of Membrane Separation Technology to Mitigation of Mine Effluent and Acidic Mine Drainage*. Mine Environment Neutral Drainage Program (MEND) Report 3.15.1.

Nelitz, M., M. Porterm, K. Wieckowski, D. Marmorek, K. Bryan, A. Hall, and D. Abraham. 2011. *Evaluating the Status of Fraser River Sockeye Salmon and Role Of Freshwater Ecology in Their Decline*. Cohen Commission Technical Report 3. Vancouver, BC: ESSA Technologies, Ltd.

NICNAS (National Industrial Chemicals Notification and Assessment Scheme). 1995. *Sodium Ethyl Xanthate, Priority Existing Chemical No. 5*. Full public report. Canberra, Australia: Australian Government Publishing Service.

NICNAS (National Industrial Chemicals Notification and Assessment Scheme). 2000. Sodium Ethyl Xanthate, Priority Existing Chemical Secondary Notification Assessment. Report No. 5S. Canberra, Australia: Australian Government Publishing Service.

O'Neal, S., and C. A. Woody. 2011. *Canada's Fraser River: Reasons for Sockeye Salmon Declines with a Comparison to Bristol Bay*. Anchorage, AK: Fisheries Research and Consulting.

Ott, A. G., and P. Weber Scannell. 1994. *Fish Monitoring Study, Red Dog Mine in the Wulik River Drainage, Emphasis on Dolly Varden (*Salvelinus malma*)*. Summary Report 1990–1993. Anchorage, AK: Alaska Department of Fish and Game.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008.* Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Poole, G. C., J. B. Dunham, D. M. Keenan, S. T. Saulter, D. A. McCoullough, C. Mebane, J. C. Lockwood, D. A. Essig, M. P. Hicks, D. J. Sturdevant, E. J. Materna, S. A. Spalding, J. Risely, and M. Deppman. 2004. The case for regime-based water quality standards. *BioScience* 54:155-161.

Power, G., R. S. Brown, and J. G. Imhof. 1999. Groundwater and fish – Insights from northern North America. *Hydrological Processes* 13:401–422.

Quinn, T. P. 2005. *The Behavior and Ecology of Pacific Salmon and Trout.* Seattle, WA: University of Washington Press.

Quinn, T. P., and D. J. Adams. 1996. Environmental changes affecting the migratory timing of American shad and sockeye salmon. *Ecology* 77:1151–1162.

Reimer, P. S. 1999. Environmental effects of manganese and proposed freshwater guidelines to protect aquatic life in British Columbia. M.S. thesis, University of British Columbia, Department of Chemical and Bio-Resource Engineering.

Roussel, H., L. Ten-Hage, S. Joachim, R. Le Cohu, L. Gauthier, and J.-M. Bonzom. 2007. A long-term copper exposure on freshwater ecosystem using lotic mesocosms: primary producer community responses. *Aquatic Toxicology* 81:168–182.

Rudolph, B. L., I. Andreller, and C. J. Kennedy. 2008. Reproductive success, early life stage development, and survival of westslope cutthroat trout (*Oncorhynchus clarki lewisi*) exposed to elevated selenium in an area of active coal mining. *Environmental Science & Technology* 42:3109–3114.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. *Nature* 465:609–612.

Schmidt, T. S., W. H. Clements, K. A. Mitchell, S. E. Church, R. B. Wanty, D. L. Fey, P. L. Verplanck, and C. A. San Juan. 2010. Development of a new toxic-unit model for the bioassessment of metals in streams. *Environmental Toxicology and Chemistry* 29:2432–2442.

Shao, E., J. Wei, A. Yo, and R. Levy. 2009. *Application of ultrafiltration and reverse osmosis for mine waste water reuse*. Water in Mining Conference, Perth, WA, September 9, 2009.

USEPA (U.S. Environmental Protection Agency). 1985a. *Ambient Water Quality Criteria for Copper - 1984.* EPA 440/5-84-031. Washington, DC.

USEPA (U.S. Environmental Protection Agency). 1985b. *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*. Washington, DC: Office of Research and Development, Environmental Research Laboratories.

USEPA (U.S. Environmental Protection Agency). 1986. *Ambient Aquatic Life Water Quality Criteria for Lead*. Washington, DC: Office of Water, Regulations and Standards, Criteria and Standards Division.

USEPA (U.S. Environmental Protection Agency). 1987. *Ambient Water Quality Criteria for Zinc – 1987*. EPA-440/6-87-003. Washington, DC: Office of Water, Regulations and Standards, Criteria and Standards Division.

USEPA (U.S. Environmental Protection Agency). 1998. *Alaska's Red Dog Mine Required to Reduce Pollutant Discharges*. Available: http://yosemite.epa.gov/opa/admpress.nsf/0000000000000000000000000000000/d0d8349388619c61852570cb0075e166!opendocument.

USEPA (U.S. Environmental Protection Agency). 2004. *Draft Aquatic Life Water Quality Criteria for Selenium – 2004*. EPA-822-D-04-001. Washington, DC: Office of Water.

USEPA (U.S. Environmental Protection Agency). 2007. *Aquatic Life Ambient Freshwater Quality Criteria – Copper, 2007 Revision*. Washington, DC.

USEPA (U.S. Environmental Protection Agency). 2008. *The U.S. Environmental Protection Agency (EPA) Plans to Re-Issue a Wastewater Discharge Permit to: Tech Cominco Alaska, Inc., Red Dog Mine Near Kotze, Alaska, and the State of Alaska Proposes to Certify the Permit*. Anchorage, AK.

USEPA (U.S. Environmental Protection Agency). 2011. *A Field-Based Aquatic Life Benchmark for Conductivity in Central Appalachian Streams (Final Report)*. EPA/600/R-10/023F. Washington, DC.

Webb, M., H. Ruber, and G. Leduc. 1976. The toxicity of various mining flotation reagents to rainbow trout (*Salmo gairdneri*). *Water Research* 10:303–306.

Weber Scannell, P. K. 1991. *Influence of Temperature on Freshwater Fishes: A Literature Review with Emphasis on Species in Alaska*. Juneau, AK: Alaska Department of Fish and Game.

Woodward, D. F., W. G. Brumbaugh, A. J. Delonay, E. E. Little, and C. E. Smith. 1994. Effects on rainbow trout fry of a metals-contaminated diet of benthic invertebrates from the Clark Fork River, Montana. *Transactions of the American Fisheries Society* 123:51–62.

Woodward, D. F., A. M. Farag, H. L. Bergman, A. J. Delonay, E. E. Little, C. E. Smith, and F. T. Barrows. 1995. Metals-contaminated benthic invertebrates in the Clark Fork River, Montana—Effects on age-0 brown trout and rainbow trout. *Canadian Journal of Fisheries and Aquatic Sciences* 52:1994–2004.

Woody, C. A., and B. Higman. 2011. *Groundwater as Essential Salmon Habitat in Nushagak and Kvichak River Headwaters: Issues Relative to Mining*. Unpublished report. Available: http://www.fish4thefuture.com/pdfs/Groundwater_and_SalmonFINAL27Aug11.pdf.

Xu, Y., J. P. Lay, and F. Korte. 1988. Fate and effects of xanthates in laboratory freshwater systems. *Bulletin of Environmental Contamination and Toxicology* 41:683–689.

Zamzow, K. 2011. *Investigations of Surface Water Quality in the Nushagak, Kvichak, and Chulitna Watersheds, Southwest Alaska 2009–2010*. Sutton, AK: Center for Science in Public Participation.

15.1.10 Chapter 9—Tailings Dam Failure

ADEC (Alaska Department of Environmental Conservation). 2011. *Water Quality Standards: Alaska Department of Environmental Conservation*. Report 18–AAC–70.

ADF&G (Alaska Department of Fish and Game). 2012. *Alaska Freshwater Fish Inventory Database*. Available: http://www.adfg.alaska.gov/index.cfm?adfg=ffinventory.main.

ADNR (Alaska Department of Natural Resources). 2005. *Guidelines for Cooperation with the Alaska Dam Safety Program.* Dam Safety and Construction Unit, Water Resources Section, Division of Mining, Land, and Waters.

AECOM. 2009. *Root Cause Analysis of TVA Kingston Dredge Pond Failure on December 22, 2008*.

Ankley, G. T., D. M. DiToro, D. J. Hansen, and W. J. Berry. 1996. Technical basis and proposal for deriving sediment quality criteria for metals. *Environmental Toxicology and Chemistry* 15:2056–2066.

ARCO (Atlantic Richfield Company). 1998. *Milltown Reservoir Sediments NPL Site, Clark Fork River Operable Unit, Final Draft Remedial Investigation Report.* Anaconda, MT.

Azam, S., and Q. Li. 2010. Tailings dam failures: a review of the last one hundred years. *Geotechnical News* December: 50–53.

Bisson, P., and R. Bilby. 1982. Avoidance of suspended sediment by juvenile coho salmon. *North American Journal of Fisheries Management* 2:371–374.

Blight, G. E. 2010. *Geotechnical Engineering for Mine Waste Storage Facilities.* Boca Raton, FL: CRC Press.

Bogan, D., D. Rinella, and R. Shaftel. 2012. *Baseline Macroinvertebrate and Diatom Surveys in Wadeable Streams of the Kvichak and Nushagak Watersheds, Bristol Bay, Alaska*. Anchorage, AK: Alaska Natural Heritage Program, University of Alaska. Prepared for The Nature Conservancy, Anchorage, AK.

Borgmann, U., C. Janssen, R. Blust, K. Brix, R. Dwyer, R. Erickson, L. Hare, S. Luoma, P. Paquin, C. Roberts, and W.-X. Wang. 2005. Incorporation of dietborne metals exposure into regulatory frameworks. Pages 153–190 in J. Meyer, W. Adams, K. Brix, S. Luoma, D. Mount, W. Stubblefield, and C. Wood, editors. *Toxicity of Dietborne Metals to Aquatic Organisms*. Pensacola, FL: SETAC Press.

Brabets, T. P. 1992. *Hydrologic Data for the Lower Copper River, Alaska, May to September 1991*. Open-File Report 92-89. Anchorage, AK: U.S. Geological Survey.

Breitenbach, A. 2007. *Rockfill for Mines*. Infomine E-Book. Available: http://www.infomine.com/publications/docs/E-Book_Rockfill_for_Mines.pdf.

Brown, R. S., W. A. Hubert, and S. F. Daly. 2011. A primer on winter, ice, and fish: what fisheries biologists should know about winter ice processes and stream-dwelling fish. *Fisheries* 36:8–26.

Browne, R. 2011. *Tailings Dam Break Model in Nevada – A Practitioner's Perspective*. PowerPoint presentation, August 24, 2011, Golder Associates. Available: http://www.infomine.com/publications/docs/Browne2011.pdf.

Brumbaugh, W. G., C. G. Ingersoll, N. E. Kemble, T. W. May, and J. L. Zajicek. 1994. Chemical characterization of sediments and pore-water from the upper Clark Fork River and Milltown Reservoir, Montana. *Environmental Toxicology and Chemistry* 13:1971–1983.

Bryce, S. A., G. A. Lomnicky, and P. R. Kaufmann. 2010. Protecting sediment-sensitive aquatic species in mountain streams through the application of biologically based streambed sediment criteria. *Journal of the North American Benthological Society* 29:657–672.

Buck, G. B., C. E. Brazil, F. West, L. F. Fair, X. Zhang, and S. L. Maxwell. 2012. *Stock Assessment of Chinook, Sockeye, and Chum Salmon in the Nushagak River*. Fishery Manuscript Series No. 12-05. Anchorage, AK: Alaska Department of Fish and Game.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). *Journal of the Fisheries Research Board of Canada* 32:667–680.

Canfield, T. J., N. E. Kemble, W. G. Brumbaugh, F. J. Dwyer, C. G. Ingersoll, and J. F. Fairchild. 1994. Use of benthic invertebrate community structure and the sediment quality triad to evaluate metal-contaminated sediment in the upper Clark Fork River, Montana. *Environmental Toxicology and Chemistry* 13:1999–2012.

Chambers, D. M., and B. Higman. 2011. *Long-Term Risks of Tailings Dam Failure*. Bozeman MT: Center for Science in Public Participation. Available: http://www.csp2.org/files/reports/Long%20Term%20Risks%20of%20Tailings%20Dam%20Failure%20-%20Chambers%20%26%20Higman%20Oct11-2.pdf.

Chapman, D. W., and K. P. McLeod. 1987. *Development of Criteria for Fine Sediment in Northern Rockies Ecoregion*. Battelle Columbus Laboratories. Prepared for EPA Region 10, EPA 910/9-87-162.

Culp, J., F. Wrona, and R. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. *Canadian Journal of Zoology* 64:1345–1351.

Cunjak, R. A. 1996. Winter habitat of selected stream fishes and potential impacts from land-use activity. *Canadian Journal of Fisheries and Aquatic Sciences* 53 (Supplement 1):267–282.

Dalpatram, A. 2011. *Estimation of Tailings Dam Break Discharges*. USSD Workshop on Dam Break Analysis Applied to Tailings Dam, Denver, CO, August 24–26, 2011,.

Dann, T. H., C. Habicht, H. A. Hoyt, T. T. Baker, and F. W. West. 2011. *Genetic Stock Composition of the Commercial Harvest of Sockeye Salmon in Bristol Bay, Alaska, 2009*. Fishery Data Series 11-21. Alaska Department of Fish and Game, Division of Sport Fish, Research and Technical Services.

Davies, M. P. 2002. Tailings impoundment failures: are geotechnical engineers listening? *Geotechnical News.* September: 31–36.

Davies, M., and T. Martin. 2009. Mining market cycles and tailings dam incidents. Pages 3–14 in *Tailings and Mine Waste '09, Proceedings of the 13th International Conference on Tailings and Mine Waste*. Edmonton, AB: University of Alberta Geotechnical Center.

Davies, M. P., T. E. Martin, and P. Lighthall. 2000. Mine tailings dams: when things go wrong. Pages 261–273 in *Tailings Dams 2000*. Las Vegas, NV: Association of State Dam Safety Officials, U.S. Committee on Large Dams.

Dawson, R. F., and K. A. Morin. 1996. *Acid Mine Drainage in Permafrost Regions: Issues, Control Strategies and Research Requirements*. MEND Project 1.61.2. Report CG25047. Prepared for the Department of Indian and Northern Affairs Canada, Ottawa, ON. Available: http://mend-nedem.org/wp-content/uploads/2013/01/1.61.2.pdf.

Dye, J. E., and C. J. Schwanke. 2009. *Report to the Alaska Board of Fisheries for the Recreational Fisheries of Bristol Bay, 2007, 2008, and 2009*. Alaska Deptartment of Fish and Game, Division of Sport Fish, Research and Technical Services.

El Mejjati, S., J. Bell, J. Botz, A. Faulkner, and S. Maxwell. 2010. Using hydroacoustic methods to enumerate migrating salmon in the Copper River, Miles Lake sonar project 2007. *Fishery Data Series* No. 10-98. Anchorage, AK: Alaska Department of Fish and Game, Division of Commercial Fisheries.

Farag, A. M., D. F. Woodward, J. N. Goldstein, W. Brumbaugh, and J. S. Meyer. 1998. Concentrations of metals associated with mining waste in sediments, biofilm, benthic macroinvertebrates, and fish from the Coeur d'Alene River Basin, Idaho. *Archives of Environmental Contamination and Toxicology* 34:119–127.

Farag, A. M., D. E. Woodward, W. Brumbaugh, J. N. Goldstein, E. MacConnell, C. Hogstrand, and F. T. Barrows. 1999. Dietary effects of metals-contaminated invertebrates from the Coeur d'Alene River, Idaho, on cutthroat trout. *Transactions of the American Fisheries Society* 128:578–592.

FERC (Federal Energy Regulatory Commission). 1991. Embankment Dams. Chapter 4 in *Engineering Guidelines for the Evaluation of Hydropower Projects.* Available: http://www.ferc.gov/industries/hydropower/safety/guidelines/eng-guide/chap4.pdf.

Fitzpatrick, A. 2012. Update: tailings dam collapse at former Gullbridge mine site. *The Telegraph* (St. John's, Newfoundland). December 17, 2012.

Gee, D. M. 2008. *Comparison of Dam Breach Parameter Estimators*. Davis, CA: U.S. Army Corps of Engineers, Hydrology Engineering Center.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Goldstein, J. N., D. F. Woodward, and A. M. Farag. 1999. Movements of adult Chinook salmon during spawning migration in a metals-contaminated system, Coeur d'Alene River, Idaho. *Transactions of the American Fisheries Society* 128:121–129.

Goodyear, K. L., and S. McNeill. 1999. Bioaccumulation of heavy metals by aquatic macro-invertebrates of different feeding guilds: a review. *Science of the Total Environment* 229:1–19.

Griffith, M. B., J. M. Lazorchak, and A. T. Herlihy. 2004. Relationships among exceedences of metals criteria, the results of ambient bioassays, and community metrics in mining-impacted streams. *Environmental Toxicology and Chemistry* 23:1786–1795.

Hancock, P. J. 2002. Human impacts on the stream–groundwater exchange zone. *Environmental Management* 29:763–781.

Hoiland, W. K., F. W. Rabe, and R. C. Biggam. 1994. Recovery of macroinvertebrate communities from metal pollution in the South Fork and mainstem of the Coeur d'Alene River, Idaho. *Water Environment Research* 66:84–88.

Huusko, A., L. Greenberg, M. Stickler, T. Linnansaari, M. Nykänen, T. Vehanen, S. Koljonen, P. Louhi, and K. Alfredsen. 2007. Life in the ice lane: the winter ecology of stream salmonids. *River Research and Applications* 23:469–491.

ICOLD (International Commission on Large Dams). 2001. *Tailings Dams, Risk of Dangerous Occurrences, Lessons Learnt from Practical Experiences*. United Nations Environmental Programme, Bulletin 121.

Ingersoll, C. G., W. G. Brumbaugh, F. J. Dwyer, and N. E. Kemble. 1994. Bioaccumulation of metals by *Hyalella azteca* exposed to contaminated sediments from the upper Clark Fork River, Montana. *Environmental Toxicology and Chemistry* 13:2013–2020.

Ingersoll, C. G., P. S. Haverland, E. L. Brunson, T. J. Canfield, F. J. Dwyer, C. E. Henke, N. E. Kemble, D. R. Mount, and R. G. Fox. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. *Journal of Great Lakes Research* 22:602–623.

Johnson, J., and P. Blanche. 2012. *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region, Effective June 1, 2012*. Special Publication No. 12-08. Anchorage, AK: Alaska Department of Fish and Game.

Jones, E. L., T. J. Quinn, and B. W. Van Alen. 1998. Observer accuracy and precision in aerial and foot survey counts of pink salmon in a southeast Alaska stream. *North American Journal of Fisheries Management* 18:832–846.

Jones, J. I., J. F. Murphy, A. L. Collins, D. A. Sear, P. S. Naden, and P. D. Armitage. 2011. The impact of fine sediment on macro-invertebrates. *River Research and Applications* 28:1055–1071.

Jones, M., T. Sands, S. Morstad, P. Salomone, G. Buck, F. West, T. Baker, and T. Krieg. 2012. *2011 Bristol Bay Area Annual Management Report*. Fishery Management Report No. 12-21. Anchorage, AK: Alaska Department of Fish and Game.

Kemble, N. E., W. G. Brumbaugh, E. L. Brunson, F. J. Dwyer, C. G. Ingersoll, D. P. Monda, and D. F. Woodward. 1994. Toxicity of metal-contaminated sediments from the Upper Clark Fork River, Montana, to aquatic invertebrates and fish in laboratory exposures. *Environmental Toxicology and Chemistry* 13:1985–1997.

Knighton, D. 1984. *Fluvial Forms and Processes: A New Perspective*. London: Edward Arnold.

Kondolf, G. 2000. Assessing salmonid spawning gravel quality. *Transactions of the American Fisheries Society* 129:262–281.

MacDonald, D. D., C. G. Ingersoll, and T. A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology* 39:20–31.

Marcus, W. A., G. A. Meyer, and D. R. Nimmo. 2001. Geomorphic control of persistent mine impacts in a Yellowstone Park stream and implications for the recovery of fluvial systems. *Geology* 29:355–358.

Maret, T. R., and D. E. MacCoy. 2002. Fish assemblages and environmental variables associated with hard-rock mining in the Coeur d'Alene River Basin, Idaho. *Transactions of the American Fisheries Society* 131:865–884.

Maret, T. R., D. J. Cain, D. E. MacCoy, and T. M. Short. 2003. Response of benthic invertebrate assemblages to metal exposure and bioaccumulation associated with hard-rock mining in northwestern streams, USA. *Journal of the North American Benthological Society* 22:598–620.

Miller, J. R. 1997. The role of fluvial geomorphic processes in the dispersal of heavy metals from mine sites. *Journal of Geochemical Exploration* 58:101–118.

Milner, A., and R. Piorkowski. 2004. Macroinvertebrate assemblages in streams of interior Alaska following alluvial gold mining. *River Research and Applications* 20:719–731.

Moore, J., and H. Langner. 2012. Can a river heal itself? Natural attenuation of metal contamination in river sediment. *Environmental Science & Technology* 46:2616–2623.

Morgenstern, N. R. 2011. Improving the safety of mine waste impoundments. Pages 3–10 in *Tailings and Mine Waste '10: Proceedings of the 14th International Conference on Tailings and Mine Waste, Vail, CO*. October 17–20, 2010. Boca Raton FL: CRC Press.

Nagorski, S. A., J. N. Moore, and T. E. McKinnon. 2003. Geochemical response to variable streamflow conditions in contaminated and uncontaminated streams. *Water Resources Research* 39:1044–1058.

National Inventory of Dams. 2005. *2005 United States National Inventory of Dams database*. Available: http://geo.usace.army.mil/pgis/f?p=397:1:0::NO. Accessed: October 6, 2005.

NDM (Northern Dynasty Minerals). 2006. *Pebble Project Tailings Impoundment G – Initial Application Report.* Prepared by Knight Piésold Ltd.

Newcombe, C., and J. Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. *North American Journal of Fisheries Management* 16:693–727.

Nielsen, J. 1992. Microhabitat-specific foraging behavior, diet, and growth of juvenile coho salmon. *Transactions of the American Fisheries Society* 121:617–634.

Nimik, D., and J. Moore. 1991. Prediction of water soluble metal concentrations in fluvially deposited tailings sediments, Upper Clark Fork Valley, Montana, U.S.A. *Applied Geochemistry* 6:635–646.

Nimmo, D. R., M. J. Willox, T. D. Lafrancois, P. L. Chapman, S. F. Brinkman, and J. C. Greene. 1998. Effects of metal mining and milling on boundary waters of Yellowstone National Park, USA. *Environmental Management* 22:913–926.

NRC (National Research Council). 2005. *Superfund and Mining Megasites*. Washington, DC.

Oswood, M. 1989. Community structure of benthic invertebrates in interior Alaskan (USA) streams and rivers. *Hydrobiologia* 172:97–110.

Pascoe, G. A., R. J. Blanchet, G. Linder, D. Palawski, W. G. Brumbaugh, T. J. Canfield, N. E. Kemble, C. G. Ingersoll, A. Farag, and J. A. Dalsoglio. 1994. Characterization of ecological risks at the Milltown Reservoir Clark Fork River sediments Superfund site, Montana. *Environmental Toxicology and Chemistry* 13:2043–2058.

Peck, P. 2007. Avoiding tailings dam failures: good practice in prevention*. Workshop on the Safety of Tailings Management Facilities, November 12, 2007*. Yerevan, Armenia. Communication on Progress to the United Nations Convention of the Transboundary Effects of Industrial Accidents.

Peck, D. V., A. T. Herlihy, B. H. Hill, R. M. Hughes, P. R. Kaufmann, D. J. Klemm, J. M. Lazorchak, F. H. McCormick, S. A. Peterson, P. L. Ringold, T. Magee, and M. R. Cappaert. 2006. *Environmental Monitoring and Assessment Program – Surface Waters Western Pilot Study: Field Operations Manual for Wadeable Streams*. EPA 600/R-06/003. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008*. Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Quinn, J., R. Daviescolley, C. Hickey, M. Vickers, and P. Ryan. 1992. Effects of clay discharges on streams. 2. Benthic invertebrates. *Hydrobiologia* 248:235–247.

Rico, M. G., G. Benito, and A. Diez-Herrero. 2008. Floods from tailings dam failures. *Journal of Hazardous Materials* 154(1-3):79–87.

Rowe, M., D. Essig, and B. Jessup. 2003. *Guide to Selection of Sediment Targets for Use in Idaho TMDLs*. Boise, ID: Idaho Department of Environmental Quality.

Ruggerone, G. T., B. A. Agler, S. E. Gilk, D. B. Molyneaux, D. J. Costello, D. E. Orabutt Jr., and D. B. Young. 2011. Habitat and growth of river-type sockeye salmon in the Kuskokwim watershed, Alaska. Pages 49–89 in S. E. Gilk, D. B. Molyneaux, and Z. W. Liller (eds.), *Kuskokwim River Sockeye Salmon Investigations, 2006 and 2007*. Fishery Manuscript Series No. 11-04. Anchorage, AK: Alaska Department of Fish and Game.

Scheuerell, M., J. Moore, D. Schindler, and C. Harvey. 2007. Varying effects of anadromous sockeye salmon on the trophic ecology of two species of resident salmonids in southwest Alaska. *Freshwater Biology* 52:1944–1956.

Schmetterling, D. A., C. G. Clancy, and T. M. Brandt. 2001. Effects of riprap bank reinforcement on stream salmonids in the western United States. *Fisheries* 26:6–23.

Semko, R. S. 1954. The stocks of west Kamchatka and their commercial utilization. *Izvestiia Tikhookeanskovo Nauchno-Issledovatelskovo Instituta Rybnovo Khozyaistva i Okeanografii* 41:3–109. Translated in 1960, Fisheries Research Board of Canada Translation Series 288.

Shirazi, M. A., W. K. Seim, and D. H. Lewis. 1981. Characterization of spawning gravel and stream system evaluation*. Proceedings of the Conference on Salmon-Spawning Gravel.* Pullman, WA: Washington Water Resources Research Center.

Silva, F. T., T. W. Lambe, and W. A. Marr. 2008. Probability and risk of slope failure. *Journal of Geotechnical and Geoenvironmental Engineering* 134:1691–1699.

Stantec Consulting. 2011. *Dam safety review, various former mine sites, former Gullbridge Copper Mine, NL*. Final Inspection Report. Prepared for Newfoundland Department of Natural Resources, File No. 121612646, St. John's, Newfoundland.

Suter, G. W., II, and S. M. Cormier. 2011. Why and how to combine evidence in environmental assessments: weighing evidence and building cases. *Science of the Total Environment* 409:1406–1417.

USACE (U.S. Army Corps of Engineers). 2003. *Slope Stability Engineer Manual*. EM 1110-2-1902. Available: http://140.194.76.129/publications/eng-manuals/EM_1110-2-1902_sec/toc.htm.

USEPA (U.S. Environmental Protection Agency). 2001. *Final Coeur d'Alene Basin Ecological Risk Assessment*. Seattle, WA: Region 10.

USEPA (U.S. Environmental Protection Agency). 2007. *Aquatic Life Ambient Freshwater Quality Criteria – Copper, 2007 Revision*. Washington, DC.

USEPA (U.S. Environmental Protection Agency). 2012a. *Clark Fork River Operable Unit*. Available: http://www2.epa.gov/region8/milltown-reservoir-sediments. Accessed: May 2, 2012.

USEPA (U.S. Environmental Protection Agency). 2012b. *Bunker Hill Mining and Metallurgical Superfund Site (Coeur d'Alene Basin)*. Available: http://yosemite.epa.gov/r10/cleanup.nsf/sites/bh. Accessed: May 2, 2012.

USGS (U.S. Geological Survey). 2004. *Integrated Investigations of Environmental Effects of Historical Mining in the Basin and Boulder Mining Districts, Boulder River Watershed, Jefferson County, Montana*. Reston, VA: U.S. Geological Survey.

USGS (U.S. Geological Survey). 2005. *Stream-Sediment Geochemistry in Mining-Impacted Streams: Sediment Mobilized by Floods in the Coeur d'Alene-Spokane River System, Idaho and Washington*. Washington, DC: U.S. Geological Survey.

USGS (U.S. Geological Survey). 2007. *Integrated Investigations of Environmental Effects of Historical Mining in the Animas River Watershed, San Juan County, Colorado.* Reston, VA: U.S. Geological Survey.

Wagener, S. M., and J. D. LaPerriere. 1985. Effects of placer mining on the invertebrate communities of interior Alaska streams. *Freshwater Invertebrate Biology* 4:208–214.

Wenning, R. J., M. Sorensen, and V. S. Magar. 2006. Importance of implementation and residual risk analyses in sediment remediation. *Integrated Environmental Assessment and Management* 2:59–65.

West, R. L., M. W. Smith, W. E. Barber, J. B. Reynolds, and H. Hop. 1992. Autumn migration and overwintering of Arctic grayling in coastal streams of the Arctic National Wildlife Refuge, Alaska. *Transactions of the American Fisheries Society* 121:709–715.

Westley, P. A. H., D. E. Schindler, T. P. Quinn, G. T. Ruggerone, and R. Hilborn. 2010. Natural habitat change, commercial fishing, climate, and dispersal interact to restructure an Alaskan fish metacommunity. *Oecologia* 163:471-484.

WISE (World Information on Service Energy). 2012. *Uranium Project website*. Last updated: June 12, 2012. Available: http://www.wise-uranium.org/ctfs.html.

Wood, P., and P. Armitage. 1997. Biological effects of fine sediment in the lotic environment. *Environmental Management* 21:203–217.

Personal Communications

Rinella, D. J. University of Alaska, Anchorage. Unpublished data.

15.1.11 Chapter 10—Transportation Corridor

ADEC (Alaska Department of Environmental Conservation). 2003. *Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances.* May.

ADEC (Alaska Department of Environmental Conservation). 2011a. *Alaska Pollutant Discharge Elimination System General Permit for Discharges from Large and Small Construction Activities.* Permit Number AKR100000. Anchorage, AK.

ADEC (Alaska Department of Environmental Conservation). 2011b. *Alaska Storm Water Guide*. December. Anchorage, AK: Division of Water.

ADF&G (Alaska Department of Fish and Game). 2012. *Alaska Freshwater Fish Inventory Database*. Available: http://www.adfg.alaska.gov/index.cfm?adfg=ffinventory.main.

ADF&G and ADOT (Alaska Department of Fish and Game, Alaska Department of Transportation and Public Facilities). 2001. *Memorandum of Agreement Between Alaska Department of Fish and Game and Alaska Department of Transportation and Public Facilities for the Design, Permitting, and Construction of Culverts for Fish Passage, Anchorage, AK*.

ADNR (Alaska Department of Natural Resources). 2013. *Invasive Plants and Agricultural Pest Management, Weed Free Gravel.* Palmer, AK: Division of Agriculture, Plant Materials Center. Available: http://dnr.alaska.gov/ag/akpmc/invasives/weed-free-gravel.htm. Accessed: March 22, 2013

ADOT (Alaska Department of Transportation and Public Facilities). 1995. *Alaska Highway Drainage Manual.* Revised June 2006. Available: http://www.dot.state.ak.us/stwddes/desbridge/pop_hwydrnman.shtml.

ADOT (Alaska Department of Transportation and Public Facilities). 2001. *Iliamna-Nondalton Road Improvements. Revised Environmental Assessment and Finding of No Significant Impact.* Project No. STP-0214(3)/51951. Prepared for Federal Highway Administration.

Angermeier, P. L., A. P. Wheeler, and A. E. Rosenberger. 2004. A conceptual framework for assessing impacts of roads on aquatic biota. *Fisheries* 29:19–29.

Arsan, E. L., and J. L. Bartholomew. 2008. Potential for dissemination of the nonnative salmonid parasite Myxobolus cerebralis in Alaska. *Journal of Aquatic Animal Health* 20:136–149.

Auerbach, N. A., M. D. Walker, and D. A. Walker. 1997. Effects of roadside disturbance on substrate and vegetation properties in Arctic tundra. *Ecological Applications* 7:218–235.

Bader, C. D. 1997. Controlling Dust. *Erosion Control* 4: July/August.

Behlke, C. E., D. L. Kane, R. F. McLean, and M. D. Travis. 1991. *Fundamentals of Culvert Design for Passage of Weak-Swimming Fish.* Final Report No. FHWA-AK-RD-90-10. Prepared for Alaska Department of Transportation and Public Facilities in cooperation with U.S. Department of Transportation, Federal Highway Administration.

Bella, E. M. 2011. Invasion prediction on Alaska trails: distribution, habitat, and trail use. *Invasive Plant Science and Management* 4:296–305.

Berejikian, B. A. 1992. Feeding ecology of rainbow trout with comparisons to Arctic char in Iliamna Lake, Alaska. M.S. thesis, University of Washington.

Birge, W. J. 1978. Aquatic toxicology of trace elements of coal and fly ash. In J. H. Thorp and J. W. Gibbons (eds.), *Energy and Environmental Stess in Aquatic Ecosystems*. U.S. Department of Energy.

Birge, W. J., J. A. Black, A. G. Westerman, and J. E. Hudson. 1979. Aquatic toxicity tests on inorganic elements occurring in oil shale. In C. Gale (ed.), *Oil Shale Symposium: Sampling, Analysis and Quality Assurance*. Cincinnati, OH: U.S. Environmental Protection Agency.

Blossey, B., L. C. Skinner, and J. Taylor. 2001. Impact and management of purple loosestrife (*Lythrum salicaria*) in North America. *Biodiversity and Conservation* 10:1787–1807.

Bolander, P., and A. Yamada. 1999. *Dust Palliative Selection and Application Guide.* U.S. Department of Agriculture, U.S. Forest Service. Available: http://www.dot.state.ak.us/stwddes/research/assets/pdf/dust_sag.pdf.

Brown, T. G., and G. F. Hartman. 1988. Contribution of seasonally flooded lands and minor tributaries to the production of coho salmon in Carnation Creek, British Columbia. *Transactions of the American Fisheries Society* 117:546–551.

Bryant, M. D., N. D. Zymonas, and B. E. Wright. 2004. Salmonids on the fringe: abundance, species composition, and habitat use of salmonids in high-gradient headwater streams, southeast Alaska. *Transactions of the American Fisheries Society* 133:1529–1538.

Bue, B. G., S. M. Fried, S. Sharr, D. G. Sharp, J. A. Wilcock, and H. J. Geiger. 1988. Estimating salmon escapement using area-under-the-curve, aerial observer efficiency, and stream-life estimates: the Prince William Sound pink salmon example. *North Pacific Anadromous Fish Commission Bulletin* 1:240–250.

Burnett, K. M., G. H. Reeves, D. J. Miller, S. Clarke, K. Vance-Borland, and K. Christiansen. 2007. Distribution of salmon-habitat potential relative to landscape characteristics and implications for conservation. *Ecological Applications* 17(1):66–80.

Christen, D. C., and G. R. Matlack. 2009. The habitat and conduit functions of roads in the spread of three invasive plant species. *Biological Invasions* 11:453–465.

Collen, P., and R. J. Gibson. 2001. The general ecology of beavers (*Castor* spp.), as related to their influence on stream ecosystems and riparian habitats, and the subsequent effects on fish—A review. *Reviews in Fish Biology and Fisheries* 10:439–461.

Copp, G. H. 1989. The habitat diversity and fish reproductive function of floodplain ecosystems. *Environmental Biology of Fishes* 26:1–27.

Cunjak, R. A. 1996. Winter habitat of selected stream fishes and potential impacts from land-use activity. *Canadian Journal of Fisheries and Aquatic Sciences* 53(Supplement 1):267–282.

Darnell, R. M., W. E. Pequegnat, F. J. Benson, and R. A. Defenbaugh. 1976. *Impacts of Construction Activities in Wetlands of the United States*. Washington, DC: U.S. Environmental Protection Agency.

Dekar, M. P., R. S. King, J. A. Back, D. F. Whigham, and C. M. Walker. 2012. Allochthonous inputs from grass-dominated wetlands support juvenile salmonids in headwater streams: evidence from stable isotopes of carbon, hydrogen, and nitrogen. *Freshwater Science* 31:121–132.

Demory, R. L., R. L. Orrell, and D. R. Heinle. 1964. *Spawning Ground Catalog of the Kvichak River System, Bristol Bay, Alaska*. Issue 488 of Special Scientific Report—Fisheries. Washington, DC: U.S. Department of Interior, Bureau of Commercial Fisheries.

Denton, K. P., H. B. Rich, Jr., and T. P. Quinn. 2009. Diet, movement, and growth of Dolly Varden in response to sockeye salmon subsidies. *Transactions of the American Fisheries Society* 138(6):1207–1219.

Ehrenfeld, J. G., and J. P. Schneider. 1991. *Chamaecyparis thyoides* wetlands and suburbanization: effects on hydrology, water quality and plant community composition. *Journal of Applied Ecology* 28:467–490.

Environment Canada. 2005. *Profile of Chloride-based Dust Suppressants Used in Canada.* Gatineau, Quebec: Government of Canada, Road Salts Working Group. Available: http://www.ec.gc.ca/nopp/roadsalt/reports/en/profile.cfm.

ESRI (Environmental Systems Research Institute). 2013. *World Imagery.* ArcMap 10.0.

FHWA (Federal Highway Administration). 2012. *Stream Stability at Highway Structures.* 4th edition. Publication No. FHWA-HIF-12-004 HEC-20. U.S. Department of Transportation.

Forman, R. T. 2000. Estimate of the area affected ecologically by the road system in the United States. *Conservation Biology* 14:31–35.

Forman, R. T. T., and L. E. Alexander. 1998. Roads and their major ecological effects. *Annual Review of Ecology and Systematics* 29:207–231.

FSSSWG (Forest Service Stream-Simulation Working Group). 2008. *Stream Simulation: An Ecological Approach to Providing Passage for Aquatic Organisms at Road-Stream Crossings.* U.S. Department of Agriculture, U.S. Forest Service.

Furniss, M. J., T. D. Roelofs, and C. S. Yee. 1991. Road construction and maintenance. Pages 29–323 *in* W. R. Meehan (ed.), *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*. American Fisheries Society Special Publication 19.

Gesch, D. 2007. Chapter 4–The National Elevation Dataset. Pages 99–118 in Maune, D. (ed.), *Digital Elevation Model Technologies and Applications: The DEM Users Manual*. 2nd edition. Bethesda, MD: American Society for Photogrammetry and Remote Sensing.

Gesch, D., M. Oimoen, S. Greenlee, C. Nelson, M. Steuck, and D. Tyler. 2002. The National Elevation Dataset. *Photogrammatic Engineering and Remote Sensing* 68(1):5–11.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Gibson, R. J., R. L. Haedrich, and C. M. Wernerheirn. 2005. Loss of fish habitat as a consequence of inappropriately constructed stream crossings. *Fisheries* 30:10–17.

Gomez-Uchida, D., J. E. Seeb, M. J. Smith, C. Habicht, T. P. Quinn, and L. W. Seeb. 2011. Single nucleotide polymorphisms unravel hierarchical divergence and signatures of selection among Alaskan sockeye salmon (*Oncorhynchus nerka*) populations. *BMC Evolutionary Biology* 11(1):48.

Grout, J. A., C. D. Levings, and J. S. Richardson. 1997. Decomposition rates of purple loosestrife (*Lythrum salicaria*) and Lyngbyei's sedge (*Carex lyngbyei*) in the Fraser River estuary. *Estuaries* 20:96–102.

Gucinski, H., M. J. Furniss, R. R. Ziemer, and M. H. Brookes. 2001. *Forest Roads: A Synthesis of Scientific Information*. General Technical Report, PNW-509. Portland, OR: U.S. Department of Agriculture.

Hamilton, T. D. 2007. *Surficial Geologic Map of the Pebble Project*. Report C-1. Anchorage, AK: The Pebble Limited Partnership.

Hancock, P. J. 2002. Human impacts on the stream–groundwater exchange zone. *Environmental Management* 29:763–781.

Harwood, D. W., and E. R. Russell. 1990. *Present Practices of Highway Transportation of Hazardous Materials, August 86 – November 89.* Kansas City, MO: Midwest Research Institute; Washington, DC: Federal Highway Administration.

Hastings, K. 2005. *Long-Term Persistence of Isolated Fish Populations in the Alexander Archipelago*. Missoula, MT: University of Montana.

Hilderbrand, R. H., and J. L. Kershner. 2000. Conserving inland cutthroat trout in small streams: how much stream is enough? *North American Journal of Fisheries Management* 20:513–520.

Hoover, J. 1981. *Emission-Oriented Dust Control and Surface Improvement Processes for Unpaved Roads*. Project 1308. Ames, IA: Iowa State University.

Hoover, J. M., K. L. Bergeson, D. E. Fox, C. K. Denny, and R. L. Handy. 1973. *Surface Improvement and Dust Palliation of Unpaved Secondary Roads and Streets.* Final report. Iowa Highway Research Board Project HR-151.

Johnson, J., and P. Blanche. 2012. *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes—Southwestern Region, Effective June 1, 2012*. Special Publication No. 12-08. Anchorage, AK: Alaska Department of Fish and Game.

Johnson, L. E., A. Ricciardi, and J. T. Carlton. 2001. Overland dispersal of aquatic invasive species: a risk assessment of transient recreational boating. *Ecological Applications* 11:1789–1799.

Jones, E. L., III, S. Heinl, and K. Pahlke. 2007. Aerial counts. Pages 399–410 in D. H. Johnson, B. M. Shrier, J. S. O'Neal, J. A. Knutzen, X. Augerot, T. A. O'Neil, and T. N. Pearsons, (eds.), *Salmonid Field Protocols Handbook: Techniques for Assessing Status and Trends in Salmon and Trout Populations*. Bethesda, MD: American Fisheries Society.

Kane, D. L., and P. M. Wellen. 1985. *Hydraulic Evaluation of Fish Passage through Roadway Culverts in Alaska.* Report No. FHWA-AK-RD-85-24 & 85-24A. Water Center/Institute of Northern Engineering, Fairbanks AK, in cooperation with U.S. Department of Transportation, Federal Highway Administration.

King, R. S., C. M. Walker, D. F. Whigham, S. Baird, and J. A. Back. 2012. Catchment topography and wetland geomorphology drive macroinvertebrate community structure and juvenile salmonid distributions in southcentral Alaska headwater streams. *Freshwater Science* 31(2):341–364.

Lake and Peninsula Borough. 2009. *Multi-Hazard Mitigation Plan.* February.

Lang, D. W., G. H. Reeves, J. D. Hall, and M. S. Wipfli. 2006. The influence of fall-spawning coho salmon (*Oncorhynchus kisutch*) on growth and production of juvenile coho salmon rearing in beaver ponds on the Copper River Delta, Alaska. *Canadian Journal of Fisheries and Aquatic Sciences* 63:917–930.

Langill, D. A., and P. J. Zamora. 2002. *An Audit of Small Culvert Installations in Nova Scotia: Habitat Loss and Habitat Fragmentation*. 2422. Dartmouth, Nova Scotia: Canadian Department of Fisheries and Oceans, Habitat Management Division.

Lee, J. E., and S. L. Chown. 2009. Quantifying the propagule load associated with the construction of an Antarctic research station. *Antarctic Science* 21:471–475.

Mason C. L., K. L. Casavant, B. R. Lippke, D. K. Nguyen, and E. Jessup. 2008. *The Washington Log Trucking Industry: Costs and Safety Analysis. University of Washington Rural Technology Initiative and Washington State University Transportation Research Group*. Report to the Washington State Legislature.

Meka, J. M., E. E. Knudsen, D. C. Douglas, and R. B. Benter. 2003. Variable migratory patterns of different adult rainbow trout life history types in a southwest Alaska watershed. *Transactions of the American Fisheries Society* 132:717–732.

Merz, J. E., J. R. Smith, M. L. Workman, J. D. Setka, and B. Mulchaey. 2008. Aquatic macrophyte encroachment in Chinook salmon spawning beds: lessons learned from gravel enhancement monitoring in the Lower Mokelumne River, California. *North American Journal of Fisheries Management* 28:1568–1577.

Morgan, E. L., W. F. Porak, and J. A. Arway. 1983. Controlling acidic-toxic metal leachates from southern Appalachian construction slopes: mitigating stream damage. *Transportation Research Record* 948:10–16.

Morstad, S. 2003. Kvichak River Sockeye Salmon Spawning Ground Surveys, 1955–2002. Regional Information Number 2A02-32. Anchorage, AK: Alaska Department of Fish and Game.

Mount, D. R., D. D. Gully, J. R. Hockett, T. D. Garrison, and J. M. Evans. 1997. Statistical models to predict the toxicity of major ions to *Ceriodaphnia dubia*, *Daphnia magna* and *Pimephales promelas* (fathead minnow). *Environmental Toxicolology and Chemistry* 16:2009–2019.

Nehring, R. B., and P. G. Walker. 1996. Whirling disease in the wild: the new reality in the intermountain west. *Fisheries* 21:28–30.

Newcombe, C. P., and D. D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. *North American Journal of Fisheries Management* 11:72–82.

Nickelson, T. E., J. D. Rodgers, S. L. Johnson, and M. F. Solazzi. 1992. Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. *Canadian Journal of Fisheries and Aquatic Sciences* 49:783–789.

NICNAS (National Industrial Chemicals Notification and Assessment Scheme). 2000. *Sodium ethyl xanthate, priority existing chemical secondary notification assessment*. Canberra, Australia: Australian Government Publishing Service.

Opperman, J. J., R. Luster, B. A. McKenney, M. Roberts, and A. W. Meadows. 2010. Ecologically functional floodplains: connectivity, flow regime, and scale. *Journal of the American Water Resources Association* 46:211–226.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008.* Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Price, D. M., T. Quinn, and R. J. Barnard. 2010. Fish passage effectiveness of recently constructed road crossing culverts in the Puget Sound region of Washington State. *North American Journal of Fisheries Management* 30:1110–1125.

Quinn, T. P., H. B. J. Rich, D. Gosse, and N. Schtickzelle, and J. Grant. 2012. Population dynamics and asynchrony at fine spatial scales: a case history of sockeye salmon (*Oncorhynchus nerka*) population structure in Alaska, USA. *Canadian Journal of Fish and Aquatic Science* 69:297–306.

Reid, L. M., and T. Dunne. 1984. Sediment production from forest road surfaces. *Water Resources Research* 20:1753–1761.

Richardson, E. V., B. Simons, S. Karaki, M. Mahmood, and M. A. Stevens. 1975. *Highways in the River Environment: Hydraulic and Environmental Design Considerations Training and Design Manual*. Washington, DC: U.S. Department of Transportation.

Ruff, C. P., D. E. Schindler, J. B. Armstrong, K. T. Bentley, G. T. Brooks, G. W. Holtgrieve, M. T. McGlauflin, C. E. Torgersen, and J. E. Seeb. 2011. Temperature-associated population diversity in salmon confers benefits to mobile consumers. *Ecology* 92:2073–2084.

Russell, R. 1977. *Rainbow Trout Life History Studies in Lower Talarik Creek - Kvichak Drainage.* Anchorage, AK: Alaska Department of Fish and Game.

Schultz, R., and E. Dibble. 2012. Effects of invasive macrophytes on freshwater fish and macroinvertebrate communities: the role of invasive plant traits. *Hydrobiologia* 684:1–14.

Shaftel, R., R. King, and J. Back. 2011. Breakdown rates, nutrient concentrations, and macroinvertebrate colonization of bluejoint grass litter in headwater streams of the Kenai Peninsula, Alaska*. Journal of the North American Benthological Society* 30:386–398.

Slaughter, C. W. 1982. Occurrence and recurrence of aufeis in an upland taiga catchment. Pages 182–188 in *Proceedings of the 4th Canadian Permafrost Conference*. National Research Council of Canada, Ottawa, ON.

Smokorowski, K. E., and T. C. Pratt. 2007. Effect of a change in physical structure and cover on fish and fish habitat in freshwater ecosystems: a review and meta-analysis. *Environmental Reviews* 15:15–41.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001. Floodplain rearing of juvenile Chinook salmon: evidence of enhanced growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences* 58:325–333.

Spellman, B. T., and T. L. Wurtz. 2011. Invasive sweetclover (*Melilotus alba*) impacts native seedling recruitment along floodplains of interior Alaska. *Biological Invasions* 13:1779–1790.

Stanford, J. A., and J. V. Ward. 1993. An ecosystem perspective of alluvial rivers - connectivity and the hyporheic corridor. *Journal of the North American Benthological Society* 12:48–60.

Suttle, K., M. Power, J. Levine, and C. McNeely. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. *Ecological Applications* 14:969–974.

Urgenson, L. S., S. H. Reichard, and C. B. Halpern. 2009. Community and ecosystem consequences of giant knotweed (*Polygonum sachalinense*) invasion into riparian forests of western Washington, USA. *Biological Conservation* 142:1536–1541.

USEPA (U.S. Environmental Protection Agency). 1985. Ambient Water Quality Criteria for Cyanide - 1984. Washington, DC.

USEPA (U.S. Environmental Protection Agency). 2003a. *Final Environmental Impact Statement, Pogo Gold Mine Project.* Seattle, WA: Region 10.

USEPA (U.S. Environmental Protection Agency). 2003b. *EPA and Hardrock Mining: A Source Book for Industry in the Northwest and Alaska.* Appendix H. Erosion and Sedimentation. Seattle, WA: Region 10.

USEPA (U.S. Environmental Protection Agency). 2006. *Industrial Stormwater Fact Sheet Series. Sector G: Metal Mining (Ore Mining and Dressing) Facilities.* EPA-833-F-06-022. Office of Water.

USEPA (U.S. Environmental Protection Agency). 2013. *ECOTOX User Guide: ECOTOXicology Database System*. Version 4.0. Available: http://www.epa.gov/ecotox. Accessed: October 2013.

USFWS (U.S. Fish and Wildlife Service). 2012. *National Wetlands Inventory.* Available: http://www.fws.gov/wetlands/Wetlands-Mapper.html. Accessed: December 11, 2012.

USGS (U.S. Geological Survey). 2012. *National Hydrography Dataset.* High Resolution, Alaska. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

USGS (U.S. Geological Survey). 2013. National Elevation Dataset (2 arc second), ArcGrid. August 26. [n59w155.zip, n59w156.zip, n59w157.zip, n59w158.zip, n60w154.zip, n60w155.zip, n60w156.zip, n60w157.zip, n60w158.zip, n60w159.zip, n60w160.zip, n61w153.zip, n61w154.zip, n61w155.zip, n61w156.zip, n61w157.zip, n61w158.zip, n61w159.zip, n61w160.zip] Available: http://viewer.nationalmap.gov/viewer/.

Van Bohemen, H. D., and W. H. J. Van de Laak. 2003. The influence of road infrastructure and traffic on soil, water, and air quality. *Environmental Management* 31:50–68.

Van Hassel, J. H., J. J. Hey, and D. L. Garling. 1980. Heavy metals in a stream ecosystem at sites near highways. *Transactions of the American Fisheries Society* 109:636–643.

Walker, D. A., and K. R. Everett. 1987. Road dust and its environmental impact on Alaskan taiga and tundra. *Arctic and Alpine Research* 19:479–489.

Walker, C. M., R. S. King, D. F. Whigham, and S. J. Baird. 2012. Landscape and wetland influences on headwater stream chemistry in the Kenai Lowlands, Alaska. *Wetlands* 32(2):301–310.

Ware, C., D. M. Bergstrom, E. Muller, and I. G. Alsos. 2012. Humans introduce viable seeds to the Arctic on footwear. *Biological Invasions* 14:567–577.

Weaver, W.E., D.K. Hagans, and M.A. Madej. 1987. Managing forest roads to control cumulative erosion and sedimentation effects. Pages 119–124 in *Proceedings of the California Watershed Management Conference, 18–20 November, 1986, West Sacramento, California*. Report 11. Berkeley, CA: Wildland Resources Center, University of California.

Weber-Scannell, P. K., and L. K. Duffy. 2007. Effects of total dissolved solids on aquatic organisms: a review of literature and recommendation for salmonid species. *American Journal of Environmental Sciences* 3:1–6.

Whitton, B. A., N. T. W. Ellwood, and B. Kawecka. 2009. Biology of the freshwater diatom *Didymosphenia*: a review. *Hydrobiologia* 630:1–37.

Wipfli, M. S., and C. V. Baxter. 2010. Linking ecosystems, food webs, and fish production: subsidies in salmonid watersheds. *Fisheries* 35(8):373–387.

Wissmar, R. C., R. K. Timm, and M. D. Bryant. 2010. Radar-derived digital elevation models and field-surveyed variables to predict distribution of juvenile coho salmon and Dolly Varden in remote streams of Alaska. *Transactions of the American Fisheries Society* 139:288–302.

Withycombe, E., and R. Dulla. 2006. *Alaska Rural Dust Control Alternatives.* Report No. SR2006-03-03. Prepared by Sierra Research, Inc. for Alaska Department of Environmental Conservation.

Woody, C. A., and S. L. O'Neal. 2010. *Fish Surveys in Headwater Streams of the Nushagak and Kvichak River Drainages, Bristol Bay, Alaska, 2008–2010*. Anchorage, AK: Fisheries Research and Consulting.

Wurtz, T. L., M. J. Macander, and B. Spellman, T. 2010. Spread of invasive plants from roads to river systems in Alaska: a network model. Pages 699−708 in J. M. Pye, H. M. Rauscher, Y. Sands, D. C. Lee, and J. S. Beatty (eds.), *Advances in Threat Assessment and Their Application to Forest and Rangeland Management.* PNW-GTR-802. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest and Southern Research Stations.

Young, M. K., P. M. Geunther-Gloss, and A. D. Ficke. 2005. Predicting cutthroat trout (*Oncorhynchus clarki*) abundance in high-elevation streams: revisiting a model of translocation success. *Canadian Journal of Fisheries and Aquatic Sciences* 62:2399–2408.

Personal Communications

Morstad, S. Fishery Biologist III, ADF&G. September 2011—email of unpublished data to Rebecca Shaftel.

15.1.12 Chapter 11—Pipeline Failures

ADEC (Alaska Department of Environmental Conservation). 2011. Water Quality Standards. 18–AAC–70.

ADEC (Alaska Department of Environmental Conservation). 2012. *Situation Report: Fairbanks Gold Mining, Inc. Cyanide Water Spill*. Anchorage, AK: Alaska Department of Environment and Conservation, Division of Spill Prevention and Response, Prevention and Emergency Response Program.

Alexander, V., and K. VanCleve. 1983. The Alaska pipeline: a success story. *Annual Review of Ecology and Systematics* 14:443–463.

API (American Petroleum Institute). 2003. *Robust Summary of Information on Distillate Fuels*. Report No. 201-14835B2. Washington, DC: American Petroleum Institute.

Barrett, D. 2012. Notice of Probable Violation and Proposed Civil Penalty: Letter to Mr. Richard Adams, Vice President, U.S. Operations, Enbridge Energy, Limited Partnership. Kansas City, MO: U.S. Department of Transportation, Pipeline and Hazardous Materials Safety Administration.

Bass, M. L., J. R. Voshell, and R. W. Young. 1987. Assessment of damage to aquatic communities resulting from a refined oil product spill. Pages 353–378 in J. H. Vandermeulen and S. E. Hrudey (eds.), *Oil in Freshwater: Chemistry, Biology, Countermeasure Technology*. New York: Pergamon Press.

Briceno, F., and F. Bajak. 2012. Townsfolk sickened after Peru toxic spill. *Businessweek*. Available: http://bigstory.ap.org/article/townsfolks-sickened-after-peru-toxic-spill.

Bury, R. B. 1972. The effects of diesel fuel on a stream fauna. *California Fish and Game* 58:291–295.

Caleyo, F. 2007. *Statistical and tolerance uncertainties in pipeline failure rate estimates*. Villahermosa, Mexico: Noveno Congreso y Expo Internacional de Ductos.

Caterpillar Global Mining. 2009. Minera Antamina: setting the standard, driving the future. *Viewpoint: Perspectives on Modern Mining* 2009:12–19.

Clap, L. 2004. *Xstrata's "Best Practice" in Argentina*. MAC: Mines and Communities. Available: http://www.minesandcommunities.org/article.php?a=5750&1=1.

Coghlan, S. M., Jr., and J. L. Lund. 2005. Rapid assessment of benthic faunal responses to a small petroleum spill in a headwaters stream. *Journal of Freshwater Ecology* 20:777–779.

Das, P. K. M. K., and S. K. Konar. 1988. Acute toxicity of petroleum products, crude oil and oil refinery effluent on plankton, benthic invertebrates and fish. *Environment and Ecology* 6:885–891.

DNV Consulting. 2006. *Frequency-Volume Study of Keystone Pipeline*. Report No. 70015849-2 rev 1. Prepared for TransCanada Pipelines, Ltd.

Earthworks. 2012. *U.S. Copper Porphyry Mines Report: The Track Record of Water Quality Impacts Resulting from Pipeline Spills, Tailings Failures and Water Collection and Treatment Failure*. Washington, DC.

ERCB (Energy Resources Conservation Board). 2013. *ST57-2012: Field Surveillance and Operations Branch – Field Operations Provincial Summary 2011*. Calgary, AB.

EUB (Energy and Utilities Board). 1998. *Pipeline Performance in Alberta 1980–1997*. Calgary, AB: Alberta Energy and Utilities Board.

Geosphere and CH2M Hill. 2006. *Hydrocarbon Characterization for Use in the Hydrocarbon Risk Calculator and Example Characterizations of Selected Alaska Fuels, Technical Background Document*. Draft report. Prepared for the Alaska Statement of Cooperation Working Group.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Giddings, J., B. Parkhurst, C. Gehrs, and R. Millemann. 1980. Toxicity of a coal liquefaction product to aquatic organisms. *Bulletin of Environmental Contamination and Toxicology* 25:1–6.

Goldstein, J. N., D. F. Woodward, and A. M. Farag. 1999. Movements of adult Chinook salmon during spawning migration in a metals-contaminated system, Coeur d'Alene River, Idaho. *Transactions of the American Fisheries Society* 128:121–129.

Guard, H. E., N. G. James, and R. B. Laughlin, Jr. 1983. *Characterization of Gasolines, Diesel Fuels and Their Water Soluble Fractions*. Oakland, CA: Naval Biosciences Laboratory, Naval Supply Center.

Hayes, M. O., R. Hoff, J. Michel, D. Scholz, and G. Shigenaka. 1992. *Introduction to Coastal Habitats and Biological Resources for Oil Spill Response*. NMRAD-92-4. Seattle, WA: National Oceanic and Atmospheric Administration, Coastal Monitoring and Bioeffects Assessment Division.

Hébert, G. W., and R. H. Kussat. 1972. *A Laboratory Evaluation of the Toxicity of Certain Oils and Chemical Oil Dispersants to Juvenile Coho Salmon and Staghorn Sculpins*. Technical Report 1972-6. Canada Department of the Environment, Fisheries Service, Pacific Region, Northern Operations Branch.

HydroQual. 2007. *Biotic Ligand Model Windows Interface, Version 2.2.3, User's Guide and Reference Manual*. Mahwah, NJ: HydroQual, Inc.

Ingersoll, C. G., P. S. Haverland, E. L. Brunson, T. J. Canfield, F. J. Dwyer, C. E. Henke, N. E. Kemble, D. R. Mount, and R. G. Fox. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the midge *Chironomus riparius*. *Journal of Great Lakes Research* 22:602–623.

Khan, N., M. A. Warith, and G. Luk. 2007. A comparison of acute toxicity of biodiesel, biodiesel blends, and diesel on aquatic organisms. *Journal of the Air and Waste Management Association* 57:286–296.

Kubach, K. M., M. C. Scott, and J. S. Bulak. 2011. Recovery of a temperate riverine fish assemblage from a major diesel oil spill. *Freshwater Biology* 56:503–518.

Lockhart, W. L., R. W. Danell, and D. A. Murray. 1987. Acute toxicity bioassays with petroleum products: influence of exposure conditions. Pages 335–344 in J. H. Vandermeulen and S. E. Hrudey (eds.), *Oil in Freshwater: Chemistry, Biology, Countermeasure Technology*. New York: Pergamon Press.

Lytle, D. A., and B. L. Peckarsky. 2001. Spatial and temporal impacts of a diesel fuel spill on stream invertebrates. *Freshwater Biology* 46:693–704.

MacDonald, D. D., C. G. Ingersoll, and T. A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology* 39:20–31.

Meyer, J. S., and W. J. Adams. 2010. Relationship between biotic ligand model-based water quality criteria and avoidance and olfactory responses to copper bv fish. *Environmental Toxicology and Chemistry* 29:2096–2130.

Minera Alumbrera. 2004. *Minera Alumbrera Sustainability Report for 2004.* Catamarca, Argentina: Minera Alumbrera Ltd. and Xstrata Copper.

Minera Alumbrera. 2005. *Minera Alumbrera Sustainability Report 2005*. Catamarca, Argentina: Minera Alumbrera YMAD-UTE and Xstrata Copper. Available: http://www.alumbrera.com.ar/download/articulos/Sustainability_Rep%20_Def.pdf.

Minera Alumbrera. 2006. *Minera Alumbrera Sustainability Report 2006*. Catamarca, Argentina: Minera Alumbrera and Xstrata Copper. Available: http://www.goldcorp.com/files/docs_sustainability/alumbrera_sustainability_report2006-en.pdf.

Minera Alumbrera. 2007. *Minera Alumbrera Sustainability Report 2007*. Catamarca, Argentina: Minera Alumbrera and Xstrata Copper.

Minera Alumbrera. 2008. *Minera Alumbrera Sustainability Report 2008*. Catamarca, Argentina: Minera Alumbrera YMAD-UTE and Xstrata Copper. Available: http://www.alumbrera.com.ar/download/articulos/Al_Sustainability_Report_2008.pdf.

Minera Alumbrera. 2009. *Minera Alumbrera Sustainability Report 2009.* Catamarca, Argentina: Minera Alumbrera YMAD-UTE and Xstrata Copper.

Minera Alumbrera. 2010. *Minera Alumbrera Sustainability Report 2010*. Catamarca, Argentina: Minera Alumbrera YMDA-UTE, and Xstrata Copper. Available: http://www.alumbrera.com.ar/download/articulos/sustainability_report_2010.pdf.

Mining Watch Canada. 2005. *Wheaton River Profits from Destruction of Bajo La Alumbrera, Argentina*. August 12. Available: http://www.miningwatch.ca/wheaton-river-profits-destruction-bajo-la-alumbrera-argentina.

Mos, L., G. A. Cooper, K. Serben, M. Cameron, and B. F. Koop. 2008. Effects of diesel on survival, growth, and gene expression in rainbow trout (*Oncorhynchus mykiss*) fry. *Environmental Science & Technology* 42:2656–2662.

Nauman, J. W., and D. R. Kernodle. 1975. The effect of a fuel oil spill on benthic invertebrates and water quality on the Alaskan Arctic Slope, Happy Valley Creek near Sagwon, Alaska. *Journal of Research of the U.S. Geological Survey* 3:495–500.

NICNAS (National Industrial Chemicals Notification and Assessment Scheme). 1995. *Sodium Ethyl Xanthate, Priority Existing Chemical No. 5*. Full Public Report. Canberra, Australia: Australian Government Publishing Service.

NICNAS (National Industrial Chemicals Notification and Assessment Scheme). 2000. *Sodium Ethyl Xanthate, Priority Existing Chemical Secondary Notification Assessment.* Report No. 5S. Canberra, Australia: Australian Government Publishing Service.

NOAA (National Oceanic and Atmospheric Administration). 2006. *Small diesel spills (50–5,000 gallons)*. Seattle, WA: NOAA Office of Response and Restoration.

NOAA (National Oceanic and Atmospheric Administration) and API (American Petroleum Institute). 1994. *Options for Minimizing Environmental Impacts of Freshwater Spill Respons*e. NOAA Hazardous Materials Response and Assessment Division and API.

O'Brien's Response Management. 2009. *TransCanada Emergency Response Plan: Worst Case Discharge Analysis and Scenarios*. Appendix B, Version 21.0.1. O'Brien's Response Management Inc.

OGP (International Association of Oil & Gas Producers). 2010. *Risk Assessment Data Directory, Riser & pipeline release frequencies.* Report No. 434-4. March 17.

Overstreet, R., and J. A. Galt. 1995. *Physical Processes Affecting the Movement and Spreading of Oils in Inland Environments*. HMRAD 95-7. Seattle, WA: National Oceanic and Atmospheric Administration, Hazardous Materials Response and Assessment Division.

Pereira. S., V. Araújo, M Reboucas, F. Vieira, M. de Almeida, F. Chinalia, and I. Nascimento. 2012. Toxicity of biodiesel, diesel, and biodiesel/diesel blends: comparative sub-lethal effects of water-soluble fractions to microalgae species. *Bulletin of Environmental Contamination and Toxicology* 88:234–238.

PHMSA (Pipeline and Hazardous Materials Safety Administration). 2012. *Pipeline Basics*. Pipeline Safety Stakeholder Communications, U.S. Department of Transportation. Available: http://primis.phmsa.dot.gov/comm/PipelineBasics.htm?nocache=7789. Accessed: February 21, 2012.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008.* Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Redman, A. D., T. F. Parkerton, J. A. McGrath, and D. M. Di Toro. 2012. PETROTOX: an aquatic toxicity model for petroleum substances. *Environmental Toxicology and Chemistry* 31:2498–2506.

Rice, S., J. Short, M. Carls, A. Moles, and R. Spies. 2007. The Exxon Valdez oil spill. Pages 419–520 in R. Spies, editor. *Long-Term Ecological Change in the Northern Gulf of Alaska*. Amsterdam, Netherlands: Elsevier B.V.

Schein, A., J. A. Scott, L. Mos, and P. V. Hodson. 2009. Oil dispersion increases the apparent bioavailability and toxicity of diesel to rainbow trout (*Oncorhynchus mykiss*). *Environmental Toxicology and Chemistry* 28:595–602.

Smith, P., D. Snook, A. Muscatt, and A. Smith. 2010. Effects of a diesel spill on freshwater macroinvertebrates in two urban watercourses, Wiltshire, UK. *Water and Environment Journal* 24:249–260.

Suter, G. W., II, and S. M. Cormier. 2011. Why and how to combine evidence in environmental assessments: weighing evidence and building cases. *Science of the Total Environment* 409:1406–1417.

Taj, M., and T. Cespedes. 2012. *Peru's Antamina mine faces fallout from spill*. Reuters.com. Accessed: August 20, 2012.

URS (URS Corporation). 2000. *Final Longhorn Pipeline Environmental Assessment, Volume 4 Responsiveness Summary, Appendix D, Leak Frequencies of Other Pipeline Operations*.

Van Derveer, W. D., R. J. Nadeau, and G. L. Case. 1995. A screening-level evaluation of impacts to a Montana lotic macroinvertebrate community from a fuel oil spill. Pages 391–397 in *Proceedings of the 14th Bien, API et al. Oil Spill (Achieving and Maintaining Preparedness) Internal Conference*. Long Beach, CA. February 27–March 2, 1995.

Wan, M. T., R. G. Watts, and D. J. Moul. 1990. Acute toxicity to juvenile Pacific salmonids and rainbow trout of butoxyethyl esters of 2,4-D, 2,4-DP and their formulated product: Weedone CB and its carrier. *Bulletin of Environmental Contamination and Toxicology* 45:604–611.

Personal Communications

Adams, W. General Manager Legacy, Rio Tinto. January 17, 2013—email to Glenn Suter.

15.1.13 Chapter 12—Fish-Mediated Effects

ARCO (Atlantic Richfield Company). 1998. *Milltown Reservoir Sediments NPL Site, Clark Fork River Operable Unit, Final Draft Remedial Investigation Report*. Anaconda, MT.

Bersamin, A., B. Luick, E. Ruppert, J. Stern, and S. Zidenberg-Cherr. 2006. Diet quality among Yup'ik Eskimos living in rural communities is low: the center for Alaska native health research pilot study. *Journal of the American Dietetic Association* 106:1055–1063.

Braund, S. R., and Associates. 2009. *Impacts of Oil and Gas Development to Barrow, Nuiqsut, Wainwright, and Atqasuk harvesters*. Report prepared for North Slope Borough, Department of Wildlife Management.

Braund, S. R., and Associates. 2011. Chapter 23: Subsistence Uses and Traditional Knowledge, Bristol Bay Drainages. In Pebble Limited Partnership. 2011. *Pebble Project Environmental Baseline Document 2004 through 2008*. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Braund, S. R., and J. Kruse (eds). 2009. *Synthesis: Three Decades of Research on Socioeconomic Effects Related to Offshore Petroleum Development in Coastal Alaska*. MMS OCS Study 2009-006.

Brna, P. J., and L. A. Verbrugge (eds.). 2013. *Wildlife Resources of the Nushagak and Kvichak River Watersheds, Alaska*. Final report. Anchorage, AK: Anchorage Fish and Wildlife Field Office, U.S. Fish and Wildlife Service.

CEAA (Canadian Environmental Assessment Agency). 2010. *Report of the Federal Review Panel Established by the Minister of the Environment: Taseko Mines Limited's Prosperity Gold-Copper Mine Project*. Ottawa, ON. Available: http://publications.gc.ca/site/eng/371768/publication.html. Accessed: May 4, 2011.

Chapman, P. M., W. J. Adams, M. L. Brooks, C. G. Delos, S. N. Luoma, W. A. Maher, H. M. Ohlendorf, T. S. Presser, and D. P. Shaw. 2010. *Ecological Assessment of Selenium in the Aquatic Environment*. Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC).

Eisler, R. 2000. *Handbook of Chemical Risk Assessment: Health Hazards to Humans, Plants, and Animals. Volume 1: Metals*. Boca Raton, FL: Lewis Publishers.

Goldsmith, S. 2007. *The Remote Rural Economy of Alaska*. Institute of Social and Economic Research, University of Alaska Anchorage. April 12. Available: http://www.iser.uaa.alaska.edu/Publications/u_ak/uak_remoteruraleconomyak.pdf.

Helfield, J. M., and R. J. Naiman. 2006. Keystone interactions: salmon and bear in riparian forests of Alaska. *Ecosystems* 9:167–180.

Hilderbrand, G. V., T. A. Hanley, C. T. Robbins, and D. D. Schwartz. 1999. Role of brown bears (*Ursus arctos*) in the flow of marine nitrogen into a terrestrial ecosystem. *Oecologia* 121:546–550.

Holen, D. 2009. The dynamic context of cultural and social sustainability of communities in southwest Alaska. *Journal of Enterprising Communities* 3:306–316.

Joyce, A. 2008. Risk and opportunity in British Columbia shellfisheries: the role of limited property rights in aquaculture development. PhD thesis, University of British Columbia, Vancouver, BC. Available: https://circle.ubc.ca/handle/2429/7569.

Kerkvliet, J., and W. Nebesky. 1997. Whaling and wages on Alaska's North Slope: a time allocation approach to natural resource use. *Economic Development and Cultural Change* 45(3):651–665.

Kruse, J. A., M. Baring-Gould, W. Schneider, J. Gross, G. Knapp, and G. Sherrod. 1983. *A Description of the Socioeconomics of the North Slope Borough*. Technical Report No. 85. Prepared by the Institute of Social and Economic Research, University of Alaska for the Minerals Management Service, Alaska Outer Continental Shelf Region, Anchorage, AK.

Kuhnlein, H., O. Receveur, and H. Chan. 2001. Traditional food systems research with Canadian indigenous peoples. *International Journal of Circumpolar Health* 60:112–112.

Loring, P. A., L. K. Duffy, and M. S. Murray. 2010. A risk-benefit analysis of wild fish consumption for various species in Alaska reveals shortcomings in data and monitoring needs. *Science of the Total Environment* 408:4532–4541.

NMFS (National Marine Fisheries Service). 2011. *Effects of Oil and Gas Activities in the Arctic Ocean, Draft Environmental Impact Statement*. United States Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources. Available: http://www.nmfs.noaa.gov/pr/pdfs/permits/arctic_deis_volume3.pdf.

NMFS (National Marine Fisheries Service). 2013. *Effects of Oil and Gas Activities in the Arctic Ocean, Draft Supplemental Environmental Impact Statement*. March. Office of Protected Resources. Available: http://www.nmfs.noaa.gov/pr/permits/eis/arctic.htm.

NRC (National Research Council). 2003. *Cumulative Environmental Effects of Oil and Gas Activities on Alaska's North Slope*. Washington, DC: The National Academies Press.

Palinkas, L. A., M. A. Downs, J. S. Petterson, and J. Russell. 1993. Social, cultural, and psychological impacts of the Exxon Valdez oil spill. *Human Organization* 52:1–12.

PLP (Pebble Limited Partnership). 2011. *Pebble Project Environmental Baseline Document 2004 through 2008.* Anchorage, AK. Available: http://www.arlis.org/docs/vol2/Pebble/2004-2008EBDIndex.pdf.

Poppel, B., J. Kruse, G. Duhaime, and L. Abryutina. 2007. *Survey of Living Conditions in the Arctic (SLiCA) Results*. Anchorage, AK: Institute of Social and Economic Research, University of Alaska Anchorage. Available: http://www.arcticlivingconditions.org. Accessed: May 4, 2011.

Storey, K., and L. C. Hamilton. 2004. Planning for the impacts of megaprojects: two North American examples. Pages 28–302 in R. O. Rasmussen and N. E. Koroleva (eds.), *Social and Economic Impacts in the North*. Dordrecht, Netherlands: Kluwer Academic Publishers.

USEPA (U.S. Environmental Protection Agency). 1985. *Ambient Water Quality Criteria for Copper – 1984*. EPA 440/5-84-031. Washington, DC.

USEPA (U.S. Environmental Protection Agency). 2009. *Red Dog Mine Extension Aqqaluk Project, Final Environmental Impact Statement*. October.

USEPA (U.S. Environmental Protection Agency). 2011. *A Field-Based Aquatic Life Benchmark for Conductivity in Central Appalachian Streams (Final Report)*. EPA/600/R-10/023F. Washington, DC: Office of Research and Development, National Center for Environmental Assessment.

15.1.14 Chapter 13—Cumulative Risks of Multiple Mines

ADF&G (Alaska Department of Fish and Game). 2012. *Alaska Freshwater Fish Inventory Database*. Available: http://www.adfg.alaska.gov/index.cfm?adfg=ffinventory.main.

ADNR (Alaska Department of Natural Resources). 2005. *Bristol Bay Area Plan (BBAP) for State Lands. Resource Assessment and Development Section, Division of Mining, Land, and Water*. Available: http://dnr.alaska.gov/mlw/planning/areaplans/bristol/. Accessed: December 27, 2012.

ADNR (Alaska Department of Natural Resources). 2012a. *Temporary Water Use Permit F2012-275 for TNR Gold Corp (Shotgun Prospect)*. Available: http://dnr.alaska.gov/mlw/mapguide/water/twup_start_tok.cfm. Accessed: November 29, 2012.

ADNR (Alaska Department of Natural Resources). 2012b. *Case Abstract, ADL 656311, James W. Caswell*. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=656311. Accessed: December 10, 2012.

ADNR (Alaska Department of Natural Resources). 2012c. *Case Abstracts ADL 701294 et al., Alaska Earth Resources, Inc., Kody and Koksetna claims*. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=701294. Accessed: December 10, 2012.

ADNR (Alaska Department of Natural Resources). 2012d. *Case Abstracts ADL 644576 et al., TNR Gold Corp., Winchester claims*. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=644576. Accessed: December 10, 2012.

ADNR (Alaska Department of Natural Resources). 2012e. *Case Abstracts, ADL 552871 et al., Full Metal Minerals (USA), Inc., South Pebble claims*. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=552871. Accessed: December 10, 2012.

ADNR (Alaska Department of Natural Resources). 2012f. *Case Abstracts, ADL 648509 et al., Alaska Earth Services, Inc., Groundhog and Nika claims*. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=648509. Accessed: December 10, 2012.

ADNR (Alaska Department of Natural Resources). 2012g. *Case Abstracts ADL 709821 et al., Millrock Alaska, LLC, AUDN claims*. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=709821. Accessed: December 10, 2012.

ADNR (Alaska Department of Natural Resources). 2012h. *State Mining Claims Dataset, Alaska State Geo-Spatial Data Clearinghouse*. Available: http://www.asgdc.state.ak.us. Accessed: November 19, 2012.

ADOT (Alaska Department of Transportation and Public Facilities). 2001. *Revised Environmental Assessment and Finding of No Significant Impact, Iliamna-Nondalton Road Improvements Project*. Project No. STP-0214(3)/51951. Available: http://www.iliamna-nondaltonroad.com/documents.shtml. Accessed: December 18, 2012

ADOT (Alaska Department of Transportation and Public Facilities). 2004. *Southwest Alaska Transportation Pla*n. Anchorage, AK. Available: http://www.dowlhkm.com/projects/SWAKTP/new_website/documents.html#. Accessed: December 19, 2012.

ADOT (Alaska Department of Transportation and Public Facilities). 2011a. *Ambler Mining District Access*. Available: http://www.dot.alaska.gov/roadstoresources/projects.shtml. Accessed: December 6, 2012.

ADOT (Alaska Department of Transportation and Public Facilities). 2011b. *Aleknagik: Wood River Bridge*. Available: http://www.brooks-alaska.com/aleknagik. Accessed: December 6, 2012.

AERI (Alaska Earth Resources, Inc.). 2008. *Affidavit of Annual Labor for Mining (Kamishak claims).* 2008-000296-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20080002960&District=320. Accessed: November 29, 2012.

AHEA (Alaska Hardrock Exploration Application). 2011. Alaska Hardrock Exploration Application A113099 by Alaska Earth Sciences, Inc., for Groundhog prospect. January 31.

AHEA (Alaska Hardrock Exploration Application). 2012. Alaska Hardrock Exploration Application A122793 by Big Chunk Corp., July 11, 2012.

Alix Resources. 2008. *News Release*. Available: http://www.mineweb.com/mineweb/content/en/mineweb-fast-news?oid=58213&sn=Detail. Accessed: January 3, 2013

Big Chunk Corp. 2012. *Affidavit of Annual Labor for Mining, Big Chunk Corporation/Liberty Star*. Doc. No. 2012-000351-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20120003510&District=320. Accessed: November 29, 2012

Booth, D. B., and C. R. Jackson. 1997. Urbanization of qquatic systems—Degradation thresholds, stormwater detention, and the limits of mitigation. *Journal of the American Water Resources Association* 33(5):1077–1090.

Booth, D., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. *Journal of the American Water Resources Association* 38(3):835–845.

Bristol Exploration Co., Inc. 2011. *Affidavit of Annual Labour for Mining (HSS Claims)*. Doc. No. 2011-000463-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20110004630&District=310. Accessed: November 29, 2012.

Brna, P. J., and L. A. Verbrugge (eds.). 2013. *Wildlife Resources of the Nushagak and Kvichak River Watersheds, Alaska*. Final Report. Anchorage, AK: U.S. Fish and Wildlife Service, Anchorage Fish and Wildlife Field Office.

Charles, S., R. Bravo De La Parra, J. P. Mallet, H. Persat, and P. Auger. 2000. Annual spawning migration in modeling brown trout population dynamics inside an arborescent river network. *Ecological Modelling* 133:15–31.

Detterman, R. L., and B. L. Reed. 1973. *Surficial Deposits of the Iliamna Quadrangle, Alaska*. Geological Survey Bulletin 1368-A. U.S. Geological Survey. Available: http://www.dggs.alaska.gov/webpubs/usgs/b/text/b1368a.pdf. Accessed: December 17, 2012.

Fall, J. A., D. L. Holen, B. Davis, T. Krieg, and D. Koster. 2006. *Subsistence Harvests and Uses of Wild Resources in Iliamna, Newhalen, Nondalton, Pedro Bay, and Port Alsworth, Alaska 2004*. Technical Paper No. 302. Juneau, AK: Alaska Department of Fish and Game, Division of Subsistence.

Fall, J. A., J. C. Schichnes, M. Chythlook, and R. J. Walker. 1986. *Patterns of Wild Resources Use in Dillingham: Hunting and Fishing in an Alaskan Regional Center*. Technical Paper No. 135. Alaska Department of Fish and Game, Division of Subsistence.

Full Metal Minerals, Inc. 2008. *Affidavit of Annual Labor for Mining (South Pebble claims)*. Doc. No. 2008-000298-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20080002980&District=320. Accessed: November 29, 2012.

Full Metal Minerals. 2012. *Pebble South*. Available: http://www.fullmetalminerals.com/properties/pebble-south/. Accessed: November 29, 2012.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska*. February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Golden Lynx, LLC. 2009. *Affidavit of Annual Labor for Mining (Kisa prospect)*. Doc. No. 2009-000419-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20090004190&District=307. Accessed: November 29, 2009.

Hilborn, R. 2006. Fisheries success and failure: the case of the Bristol Bay salmon fishery. *Bulletin of Marine Science* 78(3):487–498.

Hilborn, R., T. Quinn, D. Schindler, and D. Rogers. 2003. Biocomplexity and fisheries sustainability. *Proceedings of the National Academy of Sciences of the United States of America* 100:6564–6568.

Holen, D., J. Stariwat, T. M. Krieg, and T. Lemons. 2012. *Subsistence Harvest and Uses of Wild Resources in Aleknagik, Clark's Point, and Manokotak, Alaska, 2008*. Technical Paper No. 368. Anchorage, AK: Alaska Department of Fish and Game, Division of Subsistence.

Johnson, J., and P. Blanche. 2012. *Catalog of Waters Important for Spawning, Rearing, or Migration of Anadromous Fishes – Southwestern Region, Effective June 1, 2012*. Special Publication No. 12-08. Anchorage, AK: Alaska Department of Fish and Game.

Kennecott Exploration Co. 2011. *State of Alaska Affidavit of Annual Labor for Mining Groundhog and Nika Group of 484 Claims*. Doc. No. 2011-000223-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20110002230&District=320. Accessed: November 29, 2012.

Krieg, T. M., D. L. Holen, and D. Koster. 2009. *Subsistence Harvests and Uses of Wild Resources in Igiugig, Kokhanok, Koliganek, Levelock, and New Stuyahok, Alaska 2005*. Technical Paper No. 322. Dillingham, AK: Alaska Department of Fish and Game, Division of Subsistence.

Levelock Village Council. 2005. *Levelock Watershed Community Planning Project, Final Summary Report*. Available: http://www.commerce.state.ak.us/dca/plans/Levelock-LUP-2005.pdf. Accessed: December 14, 2012.

Liberty Star. 2012a. *Big Chunk Super Project*. Available: http://www.libertystaruranium.com/projects/big-chunk-super-project/. Accessed: November 29, 2012.

Liberty Star. 2012b. *Gold Copper Explorer Liberty Star Signs New Agreement with Northern Dynasty Minerals Ltd, Southwest Alaska*. NR 144. Available: http://www.libertystaruranium.com/2012/11/14/nr-144-gold-copper-explorer-liberty-star-signs-new-agreement-with-northern-dynasty-minerals-ltd-southwest-alaska/. Accessed: December 3, 2012.

Millrock Alaska. 2012a. *Affidavit of Annual Labor for State Mining Claims (AUDN), Millrock Alaska LLC.* Doc. No. 2012-000236-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20120002360&District=310. Accessed: November 29, 2012.

Millrock Alaska. 2012b. A*ffidavit of Annual Labor for State Mining Claims (KM), Millrock Alaska LLC. Doc. No. 2012-000470-0*. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20120004700&District=307. Accessed: November 29, 2012.

Millrock Resources. 2011. *Millrock Provides Exploration Update.* Available: http://www.millrockresources.com/news/millrock_provides_exploration_update. Accessed: November 29, 2012.

Nehlsen, W., J. Williams, and J. Lichatowich. 1991. Pacific salmon at the crossroads—stocks at risk from California, Oregon, Idaho, and Washington. *Fisheries* 16:4–21.

Northern Bonanza Trust. 2011. Affidavit of Annual Labor for Mining. Doc. No. 2011-000248-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20110002480&District=320. Accessed: November 30, 2012.

NRC (National Research Council). 1996. *Upstream: Salmon and Society in the Pacific Northwest*. Washington, DC.

PLP (Pebble Limited Partnership). 2011. *Affidavit of Annual Labor for Mining*. Doc. No. 2011-000252-0. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=638795. Accessed: January 3, 2013.

Portage Creek Village Council and Residents of Portage Creek. 2006. *Portage Creek Comprehensive Plan*. Available: http://www.commerce.state.ak.us/dca/plans/PortageCreek-CP-2006.pdf.

Ruckelshaus, M. H., P. Levin, J. B. Johnson, and P. M. Kareiva. 2002. The Pacific salmon wars: what science brings to the challenge of recovering species. *Annual Review of Ecology Systematics* 33:655–706.

Russell, R. B. 1974. *Rainbow Trout Life History Studies in Lower Talarik Creek-Kvichak Drainage*. Annual Performance Report, 1973-1974. Project F-9-6(Vol. 15), Job No. G-II-E. Juneau, AK: Alaska Department of Fish and Game.

Russell, R. B. 1975. *Rainbow Trout Life History Studies in Lower Talarik Creek-Kvichak drainage*. Annual Performance Report, 1973-1974. Project F-9-7(Vol. 16), Job No. G-II-E. Juneau, AK: Alaska Department of Fish and Game.

Schichnes, J., and M. Chythlook. 1991. *Contemporary Use of Fish and Wildlife in Ekwok, Koliganek, and New Stuyahok, Alaska*. Technical Paper No. 185. Alaska Department of Fish and Game, Division of Subsistence.

Schindler, D. E., P. R. Leavitt, C. S. Brock, S. P. Johnson, and P. D. Quay. 2005. Marine-derived nutrients, commercial fisheries, and production of salmon and lake algae in Alaska. *Ecology* 86:3225–3231.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. *Nature* 465:609–612.

Siedelman, D. L., P. B. Cunningham, and R. B. Russell. 1973. *Life History Studies of Rainbow Trout in the Kvichak Drainage of Bristol Bay*. Annual Performance Report, 1972-1972. Project F-9-5(Vol. 14), Job No. G-II-E. Juneau, AK: Alaska Department of Fish and Game.

Stickman, K., A. Balluta, M. McBurney, and D. Young. 2003. *K'ezghlegh, Nondalton Traditional Ecological Knowledge of Freshwater Fish*. Fisheries Project 01-075. U.S. Fish and Wildlife Service, Fisheries Information Services. Available: http://www.kuskokwimcouncil.org/documents/TEK/TEK%20Nondalton%20indians%20on%20freshwater%20fish.pdf. Accessed: December 11, 2012.

Stuy Mines, LLC. 2010. *Affidavit of Annual Labor for Mining*. Doc. No. 2010-000602-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20100006020&District=320. Accessed: November 29, 2012.

Szumigala, D. J., L. A. Harbo, and J. N. Adleman. 2011. *Alaska's Mineral Industry 2010*. Special Report 65. Fairbanks, AK: Alaska Department of Natural Resources, Division of Geological and Geophysical Surveys.

Thor Gold Alaska, Inc. 2011. *Affidavit of Annual Labor for Mining (Sleitat Mountain prospect)*. Doc. No. 2011-000443-0. Available: http://dnr.alaska.gov/projects/las/Case_Abstract.cfm?FileType=ADL&FileNumber=650085. Accessed: November 29, 2012.

TNR Gold Corp. 2011. *Affidavit of Annual Labour for Mining (Shotgun prospect)*. Doc. No. 2011-000202-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20110002020&District=403. Accessed: November 29, 2012.

TNR Gold Corp. 2012. *Shotgun, Alaska*. Available: http://www.tnrgoldcorp.com/index.php/projects/shotgun-alaska. Accessed: November 29, 2012.

U5 Resources, Inc. 2010. *Affidavit of Annual Labor for State Mining Claims (Big Chunk claims)*. Doc. No. 2010-000-591-0. Available: http://dnr.alaska.gov/ssd/recoff/sag/DocDisplay.cfm?SelectedDoc=20100005910&District=320. Accessed: January 3, 2013.

USCG (U.S. Coast Guard). 2012. *Public Notices of Bridge Projects—Public Notice 17-02-12*.

USFWS (U.S. Fish and Wildlife Service). 2012. *National Wetlands Inventory.* Available: http://www.fws.gov/wetlands/Wetlands-Mapper.html. Accessed: December 11, 2012.

USGS (U. S. Geological Survey). 2008. *Alaska Resource Data File, Fog Lake Prospect*. Available: http://mrdata.usgs.gov/ardf/show.php?labno=IL030. Accessed: January 3, 2013.

USGS (U.S. Geological Survey). 2012. *National Hydrography Dataset.* High Resolution, Alaska. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

Woody, C. A., and D. Young. 2006. *Life History and Essential Habitats of Humpback Whitefish in Lake Clark National Park, Kvichak River Watershed, Alaska*. Annual Report FIS 05-403. Available: http://www.fish4thefuture.com/pdfs/05-403-Lake_Clark_Whitefish_Annual_2005Feb07.pdf. Accessed: December 11, 2012.

15.1.15 Chapter 14—Integrated Risk Characterization

Blight, G. E. 2010. *Geotechnical Engineering for Mine Waste Storage Facilities.* Boca Raton, FL: CRC Press.

Buck, G. B., C. E. Brazil, F. West, L. F. Fair, X. Zhang, and S. L. Maxwell. 2012. *Stock Assessment of Chinook, Sockeye, and Chum Salmon in the Nushagak River*. Fishery Manuscript Series No. 12-05. Anchorage, AK: Alaska Department of Fish and Game.

Caldwell, J. A., and L. Charlebois. 2010. Tailings impoundment failures, black swans, incident avoidance and checklists. Pages 3–39 in *Tailings and Mine Waste 2010: Proceedings of the 14th International Conference on Tailings and Mine Waste, Vail, CO. October 17–20, 2010*. Boca Raton, FL: CRC Press.

Dye, J. E., and C. J. Schwanke. 2009. *Report to the Alaska Board of Fisheries for the Recreational Fisheries of Bristol Bay, 2007, 2008, and 2009*. Alaska Department of Fish and Game, Division of Sport Fish, Research and Technical Services.

Earthworks. 2012. *U.S. Copper Porphyry Mines Report: The Track Record of Water Quality Impacts Resulting from Pipeline Spills, Tailings Failures and Water Collection and Treatment Failure*. Washington, DC.

Ghaffari, H., R. S. Morrison, M. A., deRuijeter, A. Živković, T. Hantelmann, D. Ramsey, and S. Cowie. 2011. *Preliminary Assessment of the Pebble Project, Southwest Alaska.* February 15. Document 1056140100-REP-R0001-00. Prepared for Northern Dynasty Minerals Ltd., by WARDROP (a Tetra Tech Company). Vancouver, BC.

Gibson, R. J., R. L. Haedrich, and C. M. Wernerheirn. 2005. Loss of fish habitat as a consequence of inappropriately constructed stream crossings. *Fisheries* 30:10−17.

Instanes, A., O. Anisimov, L. Brigham, D. Goering, L. N. Khrustalev, B. Ladanyi, and J. O. Larsen. 2005. Infrastructure: buildings, support systems, and industrial facilities. Pages 907−944 in *Arctic Climate Impact Assessment*. New York, NY: Cambridge University Press.

Langill, D. A., and P. J. Zamora. 2002. *An Audit of Small Culvert Installations in Nova Scotia: Habitat Loss and Habitat Fragmentation*. 2422, Dartmouth, Nova Scotia: Canadian Department of Fisheries and Oceans, Habitat Management Division.

Pearce, T. D., J. D. Ford, J. Prno, F. Duerden, J. Pittman, M. Beaumier, L. Berrany-Ford, and B. Smit. 2011. Climate change and mining in Canada. *Mitigation and Adaptation Strategies for Global Change* 16:347–368.

Price, D. M., T. Quinn, and R. J. Barnard. 2010. Fish passage effectiveness of recently constructed road crossing culverts in the Puget Sound region of Washington State. *North American Journal of Fisheries Management* 30:1110–1125.

Schindler, D. E., X. Augerot, E. Fleishman, N. J. Mantua, B. Riddell, M. Ruckelshaus, J. Seeb, and M. Webster. 2008. Climate change, ecosystem impacts, and management for Pacific salmon. Fisheries 33:502-506.

Schindler, D. E., R. Hilborn, B. Chasco, C. P. Boatright, T. P. Quinn, L. A. Rogers, and M. S. Webster. 2010. Population diversity and the portfolio effect in an exploited species. *Nature* 465:609–612.

Vick, S. G. 2002. *Degrees of Belief: Subjective Probability and Engineering Judgement*. Reston, VA: American Society of Civil Engineers.

15.2 GIS Base Map Citations

Alaska Department of Natural Resources. 2006. Alaska Legislatively Designated Areas—State Parks. Anchorage, AK: Information Resource Management Section. Available: http://www.asgdc.state.ak.us. Accessed: February 7, 2013.

Alaska Department of Natural Resources. 1998. Populated Alaska Places. Anchorage, AK: Information Resource Management Section. Available: http://www.asgdc.state.ak.us. Accessed: February 7, 2013.

Alaska Department of Transportation. 2012. Department of Transportation Road System [DOT_RoadSystem_061212.zip].

Alaska Department of Transportation. 2012. Public Airports in Alaska [Public_Airports_Oct2012.zip].

Environmental Systems Research Institute. 2013. World Base—Light Gray Canvas. ArcMap 10.0.

Environmental Systems Research Institute. 2013. World Shaded Relief. ArcMap 10.0.

Environmental Systems Research Institute. 2007. U.S. States (Generalized). ESRI ArcMap 9.3 USA Census Data.

Environmental Systems Research Institute. 2007. U.S. and Canada Water Polygons. ESRI ArcMap 10.0 Streetmap.

National Atlas of the United States. 2005. Federal Lands of the United States. Reston, VA. Available: http://nationalatlas.gov/atlasftp.html. Accessed: February 7, 2013

U.S. Geological Survey. *National Hydrography Dataset*. High Resolution, Alaska [NHDH_AK_931v210.zip]. Available: ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution. Accessed: October 16, 2012.

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