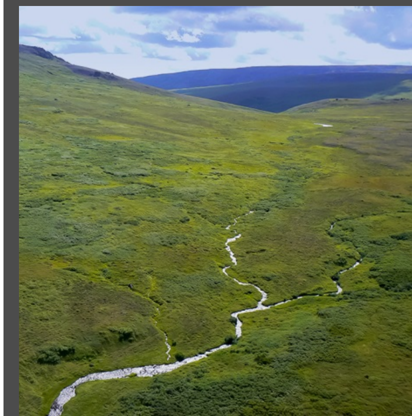
  

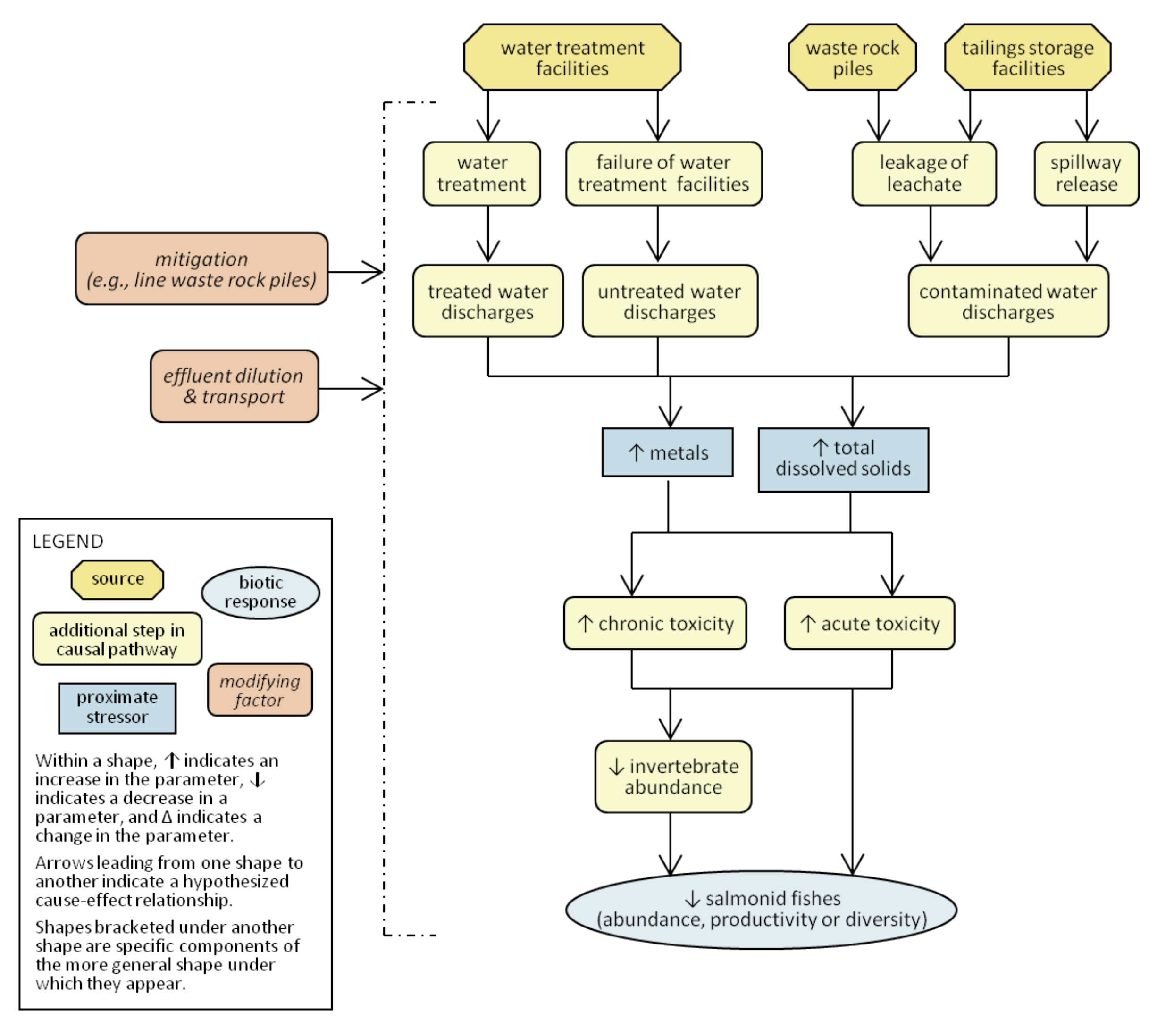
CHAPTER 8. WATER COLLECTION, TREATMENT, AND DISCHARGE

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:

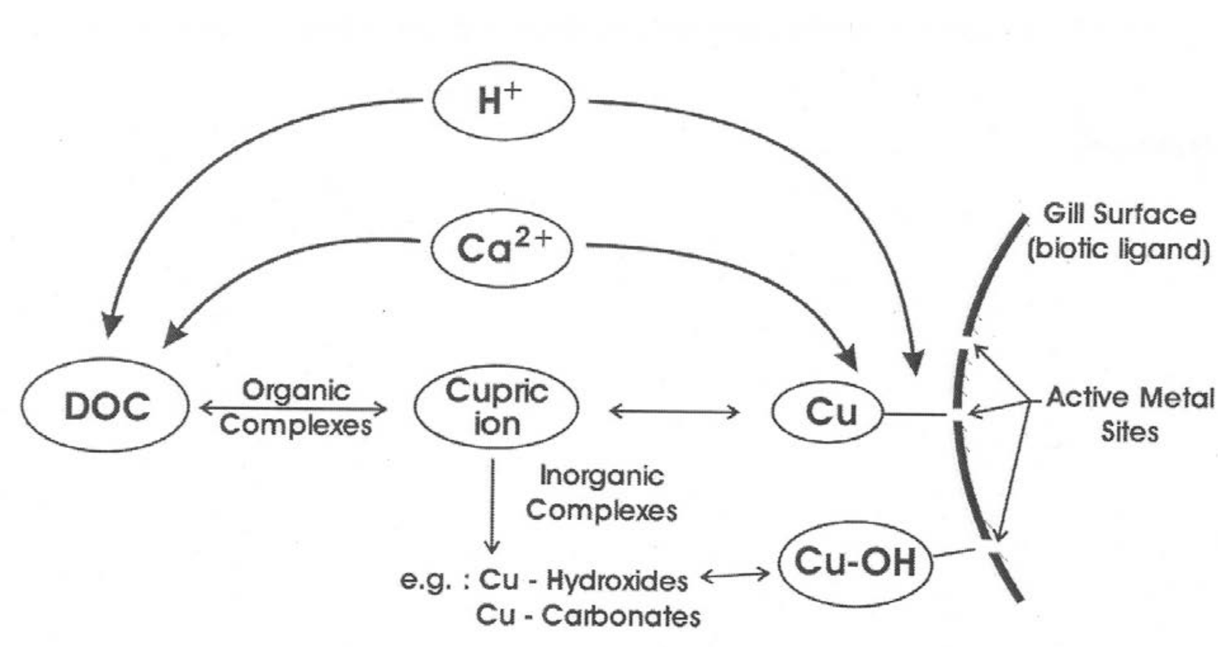
$$\text{Copper acute criterion} = e^{0.9422(\ln \text{hardness}) - 1.700} \times 0.96$$

$$\text{Copper chronic criterion} = e^{0.8545(\ln \text{hardness}) - 1.702} \times 0.96$$

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:

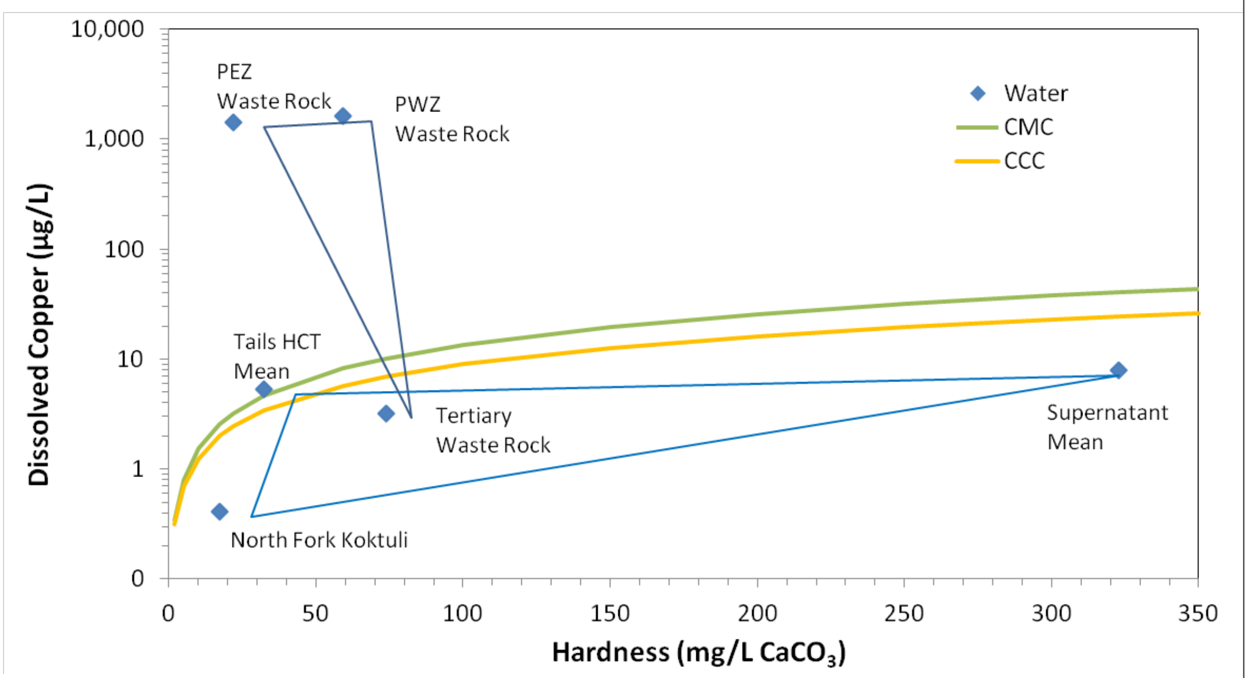
- $Q < 1$ = not overtly toxic
- $2 > Q \geq 1$ = marginally toxic
- $10 > Q \geq 2$ = moderately toxic
- $100 > Q \geq 10$ = highly toxic
- $Q \geq 100$ = extremely toxic

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.

Toxic Effect[a]	Length of Stream Potentially Affected (km)						
	Pebble 0.25		Pebble 2.0		Pebble 6.5		Pebble 2.0 and 6.5
	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.