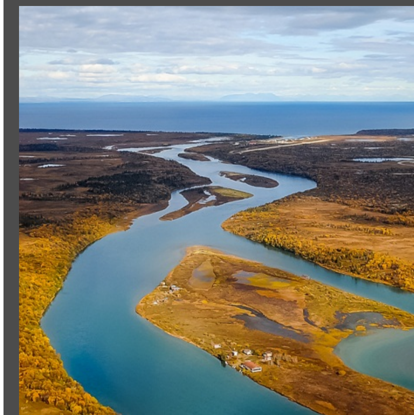
  

CHAPTER 9. TAILINGS DAM FAILURE

In this chapter, we describe risks to stream habitats and salmonid populations from potential failures of tailings storage facility (TSF) dams. Specifically, we consider tailings dam failures at TSF 1 and potential physical and toxicological effects on fish and fish habitat (Figure 9-1). Similar types of effects would occur following tailings dam failures at TSF 2 or TSF 3, or at TSF locations in other parts of the Bristol Bay watersheds, although the specifics of a failure at these locations would differ.

A breach of a TSF 1 dam would result in a flood wave and subsequent tailings deposition that would greatly alter the downstream channel and floodplain (Figure 9-1). The initial flood wave for the tailings dam failure scenarios modeled here would far exceed the typical flood event currently experienced in these watersheds. The flood itself would have the capacity to scour the channel and floodplain and alter the landscape, and the amount of tailings that could discharge from the TSF could bury the existing channel and floodplain system with meters of fine-grained tailings material. The existing channel and floodplain would be eliminated and a new channel form would develop in the resulting topography. Given the size of these new fine-grain deposits, sediment would be highly mobile under typical streamflow events and channel form would remain unstable. Sediment deposited on floodplains and remaining behind the breached dam would create a concentrated source of highly mobile material that does not currently exist in the mine scenario watersheds. Although a sediment transport study would be required to quantify the temporal and spatial extent of effects, it is likely that the sediment regime of the affected stream and downstream waters would be greatly altered, and that the existing and well-defined gravel-bed stream would be transformed to an unstable, silt- and sand-dominated channel.

Remediation is possible following a tailings spill, but the occurrence and effectiveness of these measures would be uncertain. A tailings spill would be flowing into a roadless area with streams and rivers that are too small to float a dredge, so the proper course of remediation is not obvious. The remediation process could be delayed by planning, litigation, and negotiation, particularly concerning the proper removal and disposal of excavated tailings. If the operator was no longer present at the site or was no longer in existence, the response would, at best, be delayed further. Once started, the building of a road and support facilities and the excavation, hauling, and disposal of tailings could take years, particularly given the long winter season. Therefore, the extent to which tailings exposure would be diminished by remediation cannot be estimated. Given this uncertainty, the assessment assumes that significant amounts of tailings would remain in the receiving watersheds for some time, and remediation would never be complete.

9.1 Tailings Dam Failures

9.1.1 Causes

A tailings dam failure occurs when a tailings dam loses its structural integrity and releases tailings material from the impoundment. Released tailings flow under the force of gravity as a fast-moving flood that contains a dense mixture of solids and liquids, often with catastrophic results. This flood can contain several million cubic meters of material traveling at speeds in excess of 60 km/hour. At dam heights ranging from 5 to 50 m—substantially less than the 92-m and 209-m tailings dam failures considered here—the flood wave can travel many kilometers over land and more than 100 km along waterways (Rico et al. 2008). There are many international examples of such failures (Box 9-1), involving dams that were significantly smaller than those considered in our mine scenarios.

BOX 9-1. EXAMPLES OF HISTORICAL TAILINGS DAM FAILURES

The examples below illustrate the characteristics and potential consequences of a tailings dam failure. Details of the design, construction, or operation of any tailings dams constructed for mines in the Bristol Bay watershed would not be the same as these mine tailings dams. However, these examples demonstrate that tailings dam failures can occur and illustrate how these failures may affect downstream areas. In addition, the dams in these failure examples were significantly smaller than the dams in our mine scenarios.

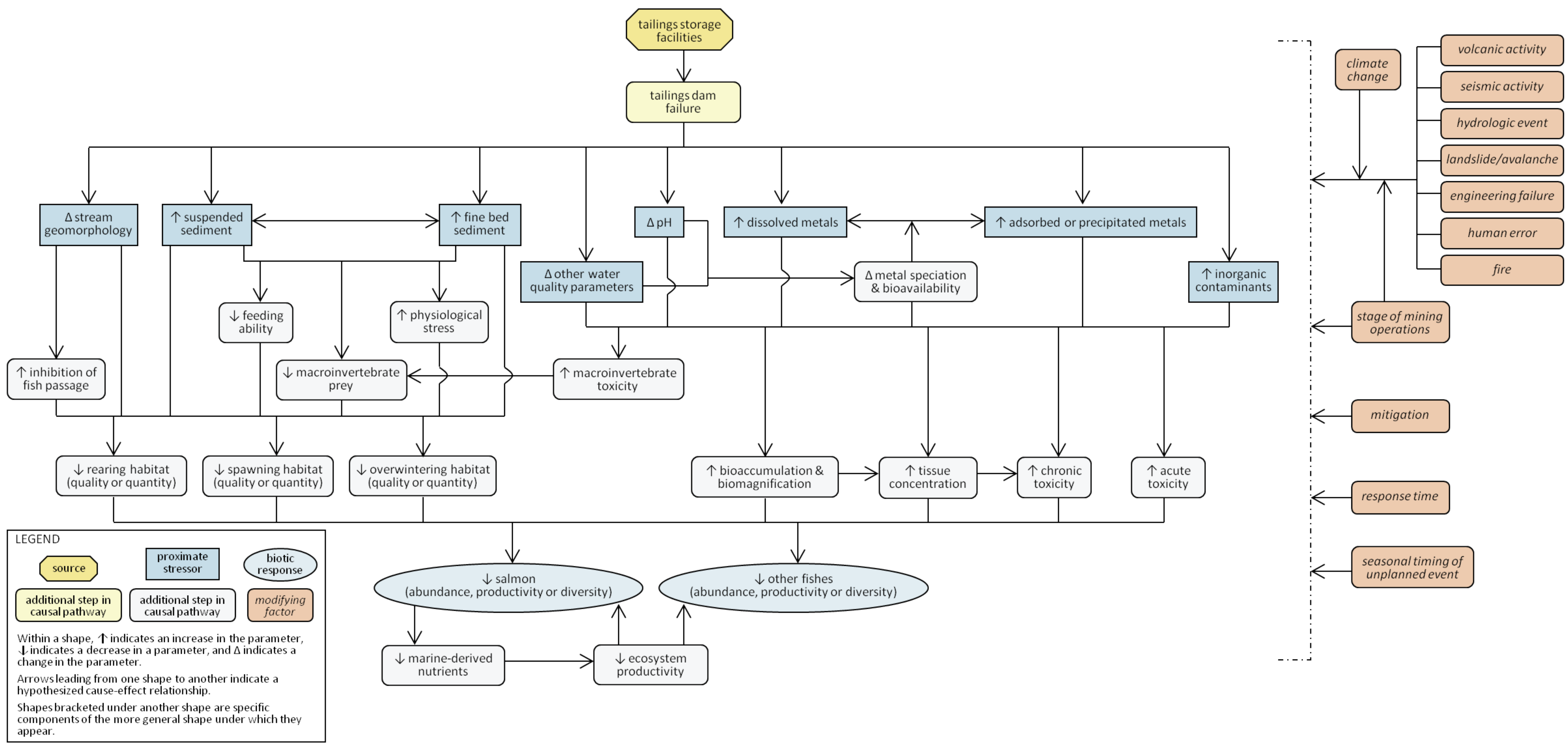
Stava, Italy, 1985. Two tailings impoundments were built, one upslope from the other, in the mountains of northern Italy. The upslope dam had a height of 29 m; the downslope dam had a height of 26 m. A stability failure of the upper dam released tailings, which then caused the lower dam to fail. The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

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

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

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

Figure 9-1. Conceptual model illustrating potential pathways linking tailings dam failure and effects on fish endpoints. Not all potential pathways are analyzed in this assessment.



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

- **Overtopping.** Overtopping occurs when sufficient freeboard (the distance between the top of a dam and the impounded water level) is not maintained and the water level behind a dam rises due to heavy rainfall, rapid snowmelt, flooding, or operator error.
- **Slope instability.** A slope instability failure occurs when shear stresses in a dam exceed the shear resistance of the dam material, most frequently resulting in a rotational or sliding failure of a portion of the downstream slope, leading to overtopping or breaching of the dam.
- **Earthquake.** Earthquake-induced shaking (Box 9-2; Section 3.6) causes additional shear forces on a dam that can lead to a slope instability failure.
- **Foundation failure.** Weak soil or rock layers and high pore pressures below the base of a dam can lead to shear failures in the foundation, causing the entire dam to slide forward or rotate out of position.
- **Seepage.** Seepage through an earthfill embankment increases interstitial pore pressures and reduces the intergranular effective stresses and shear resistance, potentially leading to a slope instability failure. Seepage can also cause internal erosion and piping within a dam leading to a hydraulic failure.
- **Structural failure.** Tailings dams often contain structural components such as drainage systems or spillways that, if they fail to function properly, can cause overtopping or slope instability failure.
- **Erosion.** Erosion, especially along the toe of a dam, can reduce slope stability to the point of failure. Erosion near the crest can reduce freeboard and increase the chance of overtopping.
- **Subsidence.** If a tailings dam is built on compressible soils or overlies cavities such as underground mining works (Box 4-4), subsidence can cause displacement or cracking of the dam. Cracking can lead to a direct hydraulic breach or to slope instability. Settlement can reduce freeboard and increase the chance of overtopping.

BOX 9-2. SELECTING EARTHQUAKE CHARACTERISTICS FOR DESIGN CRITERIA

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

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

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

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

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

The Initial Application Package for Approval to Construct a Dam submitted by Northern Dynasty Minerals (NDM) to the Alaska Department of Natural Resources (NDM 2006) included a seismic safety and design analysis prepared by Knight Piésold Consulting that identified the following design criteria for the tailings dams at the storage facility.

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

NDM used a deterministic evaluation to select the MDE and MCE, which were deemed equivalent for the preliminary safety design. In the application, NDM reports that the preliminary design incorporates additional safety factors, including design of storage facility embankments to withstand effects of the MDE and a distant magnitude 9.2 event (NDM 2006). Ghaffari et al. (2011) state that an MCE of magnitude 7.5 with 0.44g to 0.47g maximum ground acceleration was used in the stability calculations for the tailings dam design. Although the design specifications proposed by Ghaffari et al. (2011) exceed the minimum requirements for dams in Alaska, the deterministic dataset used is small and contains considerable uncertainties, which could lead to an underestimate of potential seismic risks.

A number of studies have attempted to analyze the historical record to determine proximate causes and probabilities of tailings dam failures (Davies et al. 2000, ICOLD 2001, Davies 2002, Rico et al. 2008, Chambers and Higman 2011). These efforts have been hindered by the lack of a worldwide inventory of tailings dams, incomplete reporting of tailings dam failures, and incomplete data for known failures. The National Inventory of Dams (2005) lists 1,448 tailings dams in the United States, and the worldwide total is estimated at over 3,500 (Davies et al. 2000). The International Commission on Large Dams compiled a database of 221 tailings dam incidents (events potentially leading to failure) and failures (events in which dams stop retaining tailings as designed) that occurred from 1917 through 2000 (ICOLD 2001). Causes of incidents and failures were reported for 220 of these, comprising 85 incidents and 135 failures. Causes of the 135 reported failures are summarized in Table 9-1.

Perhaps most noteworthy is the relatively high number of failures at active versus inactive tailings dams, primarily resulting from slope instability and failure (Table 9-1). This suggests that the stability of tailings dams and impoundments may increase with time, as dewatering and consolidation of tailings occurs and additional loads are no longer applied. However, failures do occur after operation. For example, rehabilitation of the Gull Bridge Mine in Newfoundland, Canada, occurred in 1999. In 2010, an inspection found that the tailings dam at the closed mine was deteriorating (Stantec Consulting 2011), and in 2012 the dam failed, leaving a 50-m gap the height of the dam (Fitzpatrick 2012). The primary cause of failure for inactive tailings dams is overtopping, which accounts for 80% of recorded failures with known causes (Table 9-1).

Table 9-1. Number and cause of tailings dam failures at active and inactive tailings dams.

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

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

9.1.2 Probabilities

It is difficult to estimate the probability of low-frequency events such as tailings dam failures, especially when each tailings dam is a unique structure subject to unique loading conditions. In addition, failure probabilities may be estimated and interpreted in different ways (Box 9-3).

BOX 9-3. INTERPRETATION OF DAM FAILURE PROBABILITIES

There are two fundamental types of probability interpretations: frequentist and subjectivist.

Frequentist probabilities are based on observed frequencies of past events. For example, based on the observed frequency of tailings dam failures (88 in 176,000 dam-years, where dam-year is the existence of one dam for one year), we estimate a frequency of 1 failure in 2,000 dam-years (or 0.00050 failures per dam-year). In conventional risk probabilities, this means the following.

- Each year, there is a 5×10^{-4} probability of failure per dam.
- Out of 200 dams, one fails each decade on average; out of 2,000 dams, one fails each year on average.

Strictly speaking, frequentist probabilities are properties of populations. However, by extension, if there is one dam and it is typical of the population, it would be expected to fail, on average, within a 2,000-year period. This does not mean it is expected to fail 2,000 years after it is built; a failure could occur during any year. Rather, it indicates that, after 2,000 years have passed, it is more likely than not that the dam would have failed (i.e., half of a population of such dams would have failed 2,000 years after they were built), although the actual failure could occur any year in that 2,000-year window.

Subjectivist probabilities are based on degree of belief. For example, if engineers design a dam using novel methods, they cannot make use of frequencies when estimating failure risks. They may, however, use a model or best professional judgment to support a statement that the annual probability of failure is some value (e.g., 1×10^{-6}, or 1 failure in a million dam-years). As with frequentist probabilities, this does not mean that the dam is expected to fail only after a million years have passed. Because subjective probabilities are not based on frequencies, they are typically described as equivalent to betting odds—that is, the engineers would be willing to accept a bet in which, if the dam stands for a year they win $1, but if it fails they pay $1 million. Rather than present subjective probabilities of failure, designs are more commonly said to conform to standard or best engineering practices.

Despite these difficulties, several studies have calculated the frequency of past tailings dam failures, resulting in the following failure frequencies.

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

Available data do not permit reliable estimation of failure rates for different causes of failure or stages of activity. Although most failures have occurred while the tailings dams were actively receiving tailings (Table 9-1), the dam inventories do not indicate whether the thousands of dams in the inventory are active or inactive and do not include years of operation. This prevents estimation of the proportion in each category and makes it impossible to calculate the number of active dam-years. Low failure frequencies and incomplete datasets also make any meaningful correlations between failure probabilities and dam height or other characteristics questionable. For example, although the 1,448 tailings dams listed in the National Inventory of Dams create a statistically large and fairly complete database that includes dam heights, the International Commission on Large Dams failure database includes only 49 U.S. tailings dam failures—too small a dataset to develop a meaningful correlation between dam height and failure probability. Very few existing rockfill dams approach the size of the structures in our mine scenarios, and none of these large dams have failed.

The historical frequencies of tailings dam failures presented above may be interpreted as an upper bound on the failure probability of a modern tailings dam. Morgenstern (2011), in reviewing data from Davies and Martin (2009), did not observe a substantial downward trend in failure rates over time. However, improvements in the understanding of dam behavior, dam design, construction techniques, construction quality control, dam monitoring, and dam safety assessment would be expected to reduce the probability of failure for dams designed, constructed, and operating using more modern or advanced engineering techniques.

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

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

The State of Alaska regulates its tailings dams under Alaska Administrative Code (AAC) Title 11, Chapter 93, Article 3, Dam Safety (11 AAC 93). Each dam is assigned to a class based on the potential hazards of a tailings dam failure (Table 9-2). Given that anadromous fish would be affected but no loss of human life is expected under the tailings dam failure scenarios, Class II would be applicable, although a mine operator might choose to exceed state requirements and meet Class I. Therefore, the tailings dams in the mine scenarios would be classified as either Hazard Class I or II, both of which require a detailed computer stability analysis with verification by other methods, and may require more sophisticated finite element analyses in special circumstances. This analysis considers the effects of earthquakes based on a site-specific evaluation of seismicity in the area (Section 3.6). Box 9-2 describes the selection of earthquake characteristics for design criteria.

Table 9-2. Summary of Alaska's classification of potential dam failure hazards.

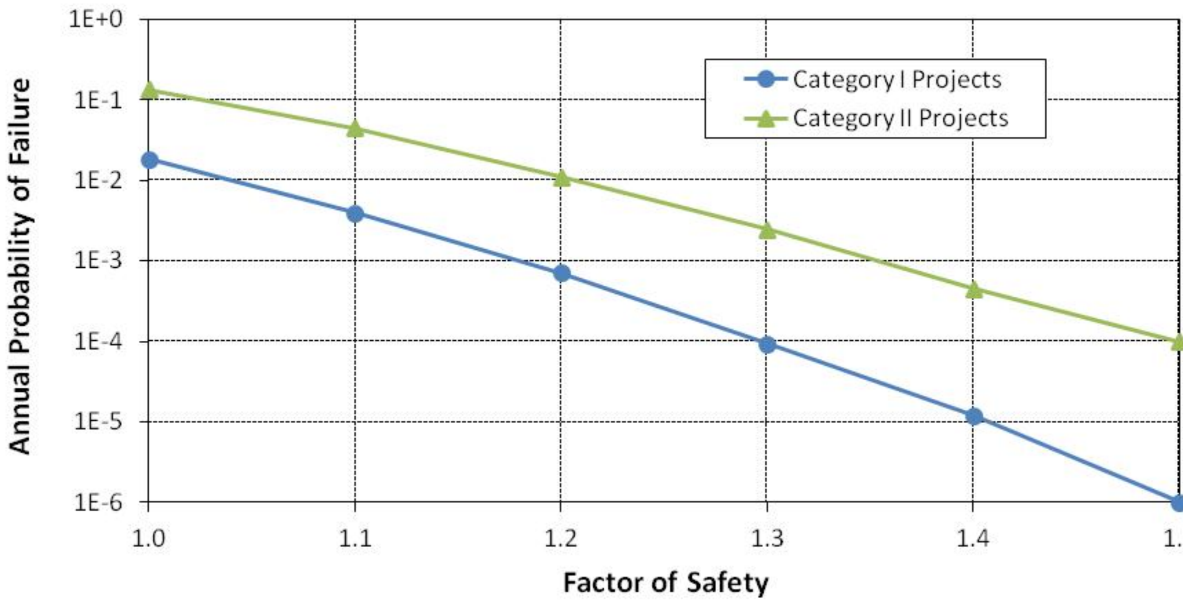
Hazard Class	Effect on Human Life	Effect on Property
I (High)	Probable loss of one or more lives	Irrelevant for classification, but may include the same losses indicated in Class II or III
II (Significant)	No loss of life expected, although a significant danger to public health may exist	Probable loss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

Notes:
Tailings dams in the mine scenarios would be classified as Hazard Class I or II.
AAC = Alaska Administrative Code.
Source: ADNR 2005.

The Guidelines for Cooperation with the Alaska Dam Safety Program (ADNR 2005) do not specify a minimum safety factor for dams, but rather allow the applicant to propose one. Guidelines suggest that the applicant follow accepted industry design practices such as those provided by the U.S. Army Corps of Engineers (USACE), the Bureau of Reclamation, the Federal Energy Regulatory Commission (FERC), and other agencies. Both USACE and FERC require a minimum safety factor of 1.5 for the loading condition corresponding to steady seepage from the filled storage facility (FERC 1991, USACE 2003). Based on the correlations among level of engineering, factor of safety, and slope failure probability derived from Silva et al. (2008), application of a 1.5 safety factor yields an expected annual probability of slope failure between 0.0001 (Category II) and 0.000001 (Category I) (Figure 9-2). This translates to one tailings dam failure due to slope failure every 10,000 to 1 million dam-years.

The upper bound of this range (0.0001) is lower than the historical average of 0.00050 (1 failure every 2,000 dam-years) for tailings dams. This is partly because slope failure is only one of several possible failure mechanisms, but it also suggests that some past tailings dams may have been designed for lower safety factors or designed, constructed, operated, or monitored to lower than Category II engineering standards. As shown in Table 9-1, slope failures only account for about 25% of all tailings dam failures with known causes. Thus, the probability of failure from all causes may be about four times higher than dam failures from slope instability alone (yielding an expected annual probability of failure between 0.0004 and 0.000004, or one tailings dam failure every 2,500 to 250,000 dam-years), although it is important to recognize that this small dataset may not be representative. Because 90% of tailings dam failures have occurred in active dams (Table 9-1), the probability of a tailings dam failure after TSF closure would be expected to be lower than the historical average for all tailings dams.

Figure 9-2. Annual probability of dam failure due to slope failure vs. factor of safety (modified from Silva et al. 2008).



These low probabilities are based on failure frequencies within categories of engineering practice and safety factors, but the authors describe results as "semiempirical" due to the judgment involved in categorizing the dams and creating the curves to describe the relationships (Silva et al. 2008). Modern, high earthen dams do not exist in large numbers and have not existed for long periods of time, and the frequencies and time courses of failures may differ from both the historical record and design goals. In particular, the failure rates of large earthen dams that are hundreds of years old are not known.

Given an annual probability of failure per dam-year, we can calculate the probability of the failure of any project dam over any number of years. The three mine scenarios have different numbers of dams and different operating lives: the Pebble 0.25 scenario has a single tailings dam and an operating life of 20 years; the Pebble 2.0 scenario has three tailings dams and an operating life of 25 years; and the Pebble 6.5 scenario has eight tailings dams and an operating life of 78 years. Using an upper bound annual probability of failure of 0.0004, the probability of dam failure would range from 0.8% to 22% over the operating life of each scenario (Table 9-3). This range decreases to 0.008% to 0.25% when a lower bound annual failure probability of 0.000004 is used (Table 9-3). If the tailings in the TSFs remain saturated (e.g., to keep the pyritic tailings covered with water), the potential for dam failure over a longer period needs to be considered. The probability that any of the dams would fail during a post-closure period of 1,000 years ranges from upper bounds of 33 to 96% to lower bounds of 0.4 to 3% across the three scenarios (Table 9-3).

Table 9-3. Summary of tailings dam failure probabilities in the three mine scenarios.				
		Probability of Failure		
Time Period	Annual Failure Probability	Pebble 0.25[a]	Pebble 2.0[b]	Pebble 6.5[c]
Operational life	0.0004	0.8	3	22
	0.000004	0.008	0.03	0.25
1,000 year post-closure period	0.0004	33	70	96
	0.000004	0.4	1.2	3

Notes:
[a] Operational life of 20 years; 1 tailings dam.
[b] Operational life of 25 years; 3 tailings dams.
[c] Operational life of 78 years, 8 tailings dams.

9.1.3 Uncertainties

The variability in published probabilities of tailings dam failure reflects the uncertainty inherent in these estimates. Much of this uncertainty is due to incomplete data. TSFs may remain in place for long periods. Most dams are created as water-holding dams with limited expected lifespans (generally about 50 years). TSFs can be drained after mine closure to reduce the probability and consequences of tailings dam failures, but draining a thick layer of fine-grained material can be difficult. In the mine scenarios, only 17 to 28% of net precipitation (depending on the TSF) would need to infiltrate into the tailings to maintain full saturation with steady-state downward flow, so draining the TSFs would require maintaining a high runoff percentage. Furthermore, if tailings ponds need to be maintained to keep pyritic tailings hydrated and isolated from oxidation, tailings dams must retain solid and liquid materials behind them in perpetuity—meaning that the dams must be maintained in perpetuity, in the face of uncertain seismic and weather events that may occur over thousands of years and have cumulative effects.

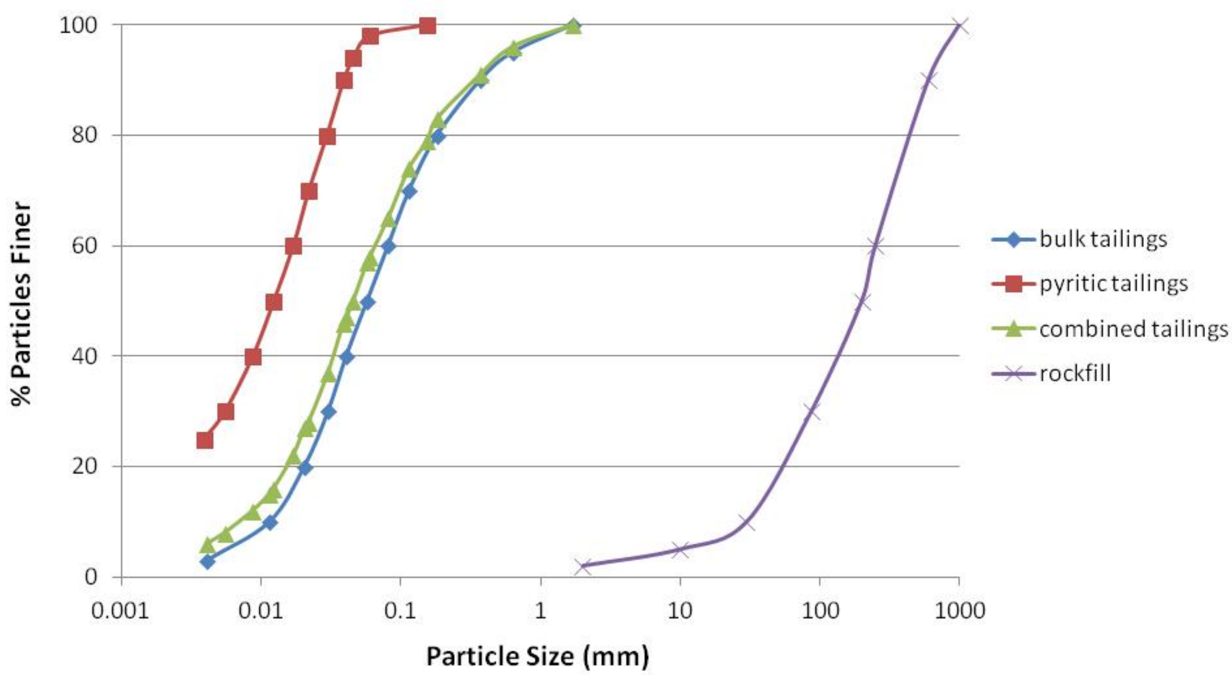
9.2 Material Properties

9.2.1 Tailings Dam Rockfill

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

Well-graded rock would have a coefficient of uniformity (D_{60}/D_{10}) greater than 4 and would have a coefficient of curvature ($D_{30}/ (D_{60}*D_{10})$) between 1 and 3. Combining these coefficients with Dawson and Morin's (1996) report of a D_{50} particle size greater than 200 mm for waste rock, one can generate a representative particle size distribution curve for the bulk of the tailings dam material (Figure 9-3).

Figure 9-3. Representative particle size distributions for tailings solids (bulk and pyritic tailings) and tailings dam rockfill. Tailings distributions are based on particle sizes specified by Ghaffari et al. (2011) and on typical tailings particle size distributions.



9.2.2 Tailings Solids and Liquids

The tailings solids would include both bulk and pyritic tailings. The bulk tailings would consist largely of sand and silt-sized particles (D_{80} = 200 μm) and have an average dry density of 1.36 metric tons/m^3. The pyritic tailings would consist of predominantly silt-sized particles (D_{80} = 30 μm) and have an average dry density of 1.76 metric tons/m^3. The mass of the bulk tailings and the pyritic tailings would equal 85 and 14% of the ore mass, respectively (Ghaffari et al. 2011). Representative particle size distribution curves for bulk, pyritic, and combined tailings are shown in Figure 9-3.

Given the dry density of the bulk tailings reported above and the specific gravity reported for the ore (2.63 for the solids) (Ghaffari et al. 2011), the bulk tailings would be 52% solids and 48% liquid by volume. The pyritic tailings, given the dry density reported above and a solids-specific gravity of 3.00 (Ghaffari et al. 2011), would be 59% solids and 41% water. Based on the proportions of bulk and pyritic tailings, the combined material in the TSF would be 53% solids and 47% water by volume, exclusive of any ponded water above the settled tailings. As the tailings consolidate, the bulk density of the deeper tailings would be expected to increase above the average density, although this consolidation may be limited (Section 6.3.2).

9.3 Modeling a Tailings Dam Failure

Although a tailings dam failure is a low-probability event, the probability is not zero. Should such an unlikely event occur, it is important to understand its potential impacts on the Bristol Bay watershed. In this assessment, we consider the effects of two potential dam failures at TSF 1: one at a volume approximating the complete Pebble 0.25 scenario (92-m dam height, with 158 million m^3 of tailings produced) and one at a volume approximating the complete Pebble 2.0 scenario (209-m dam height, with, 1,270 million m^3 of tailings produced). In both cases, we assumed 20% of the impounded tailings (solids and pore water) would be mobilized (Azam and Li 2010, Dalpatram 2011). Although it is reasonable to expect that 30 to 66% of the impounded tailings material could contribute to debris flow following a tailings dam failure, given the particle size distribution of the tailings (Browne 2011), we used a conservative estimate of 20% to account for the fact that the volume of material mobilized, the distance it travels downstream, and the amount of deposition can vary greatly based on numerous factors (e.g., dam height, material size distribution, material water content at time of failure) (Rico et al. 2008).

As detailed in Box 9-4, we used the USACE Hydrologic Engineering Center's River Analysis System (HEC-RAS) to model hydrologic characteristics of the dam failures. This tool requires the selection of one of two failure initiation mechanisms: overtopping or piping failure. We selected overtopping as the initiating event for final model runs for several reasons.

- The assessment TSF dam includes a liner (Section 6.1.2.4) that would reduce the risk of embankment failure due to seepage and piping (Section 9.1.1).
- Many of the failure mechanisms listed in Section 9.1.1 involve failure via breaching or overtopping, and thus are better approximated by the overtopping modeling approach in HEC-RAS.
- Overtopping could plausibly occur, for example, if storage freeboard was exceeded due to excessive precipitation, settlement over time, or a landslide or seismic event, or if any designed overflow spillway became blocked by ice or debris.

Although we modeled an overtopping failure, sensitivity analysis showed that model results were insensitive to initiation type relative to failure duration (Box 9-4)—that is, the mechanism of failure initiation did not significantly influence potential effects. The overtopping failure outputs were compared to similar piping outputs generated by subsequent HEC-RAS model runs. Comparison of peak discharges at the dam indicated that failure by overtopping generated the smallest expected flood wave peaks, and did not create a situation in which the selection of model assumptions overestimated the potential for flooding. Comparison of peak discharges was also reviewed by varying the time to full dam breach from 30 minutes to 4 hours (Gee 2008). Results indicated that magnitude of the peak flood wave was sensitive to breach development time, so we selected 2 hours as a reasonable time to full dam breach (Box 9-4).

BOX 9-4. METHODS FOR MODELING TAILINGS DAM FAILURES

We modeled hydrologic characteristics of tailings dam failures at tailings storage facility (TSF) 1 in the Pebble 0.25 and Pebble 2.0 scenarios using the U.S. Army Corps of Engineers Hydrologic Engineering Center's River Analysis System (HEC-RAS). Under both dam failure scenarios we modeled hydrologic conditions (e.g., water discharges, depths, and velocities) in the stream channel and floodplain during and immediately following dam failure, and then used these outputs to estimate tailings transport and deposition along the stream network. We limited the extent of the model to a 30-km reach downstream of the TSF (i.e., from the face of the TSF 1 tailings dam down the North Fork Koktuli River valley to the confluence of the South and North Fork Koktuli Rivers); extension of the simulation beyond this point would have introduced significant error and uncertainty associated with the contribution of the South Fork Koktuli River flows.

HEC-RAS inputs included geometry of an inline structure to simulate the dam cross-section and stream channel geometry data, both of which we derived from a 30-m digital elevation model. Flow calculations are completed between successive cross-sections in the model, balancing the hydraulic energy to determine the water surface elevations and flow velocity, and then moving to the next cross-section in the sequence and repeating the process. Because HEC-RAS is most often used to simulate clear water flows, it is appropriate to increase the channel roughness coefficient (i.e., Manning's *n* coefficient) to better emulate flow characteristics of the sediment-rich water released during a tailings dam failure; thus, we used a Manning's $n = 0.09$ for analyses.

We present model outputs for overtopping failures at both the Pebble 0.25 scenario dam (92-m dam height) and the Pebble 2.0 scenario dam (209-m dam height), assuming a 2-hour failure duration and the release of 20% of available tailings storage capacity in each failure. In HEC-RAS, options for initiating a dam failure are limited to overtopping or piping failure. Both initiation types were modeled to examine sensitivity to initiation conditions. In addition, a range of dam failure durations (30 minutes to 4 hours) was examined (Gee 2008). Results showed that peak flows during a failure were much more sensitive to failure duration than to initiation type. The 2-hour duration to full failure generated peak flows that fell within the middle range of potential peak flows to consider. Overtopping generated the smallest peaks in the 2-hour simulation group (Q_{max} = 39,100 m^3/s). Piping failure in the 2-hour group was tested for failures initiating near the base of the dam, at mid-elevation, and near the top of the dam face, generating Q_{max} values of 92,263 m^3/s, 85,747 m^3/s, and 48,868 m^3/s, respectively. The 30-minute simulation group average Q_{max} values were 222% greater, and the 4-hour simulation group average Q_{max} values were 38% lower.

We assumed a particle size distribution of 0.001- to 1.0-m diameter for the dam construction material, and less than 0.01- to just over 1.0-mm diameter for the impounded tailings material (Figure 9-3). We focused on transport and deposition of fine-grained (less than 1.0-mm diameter) tailings material, since larger dam construction material would likely deposit within the first few kilometers downstream of the failure. Based on the Hjulström curve—which estimates when a stream or river will erode, transport, or deposit sediment based on flow speed and sediment grain size—all mobilized tailings would remain in suspension at water velocities greater than 0.05 m/s. Thus, the channel would transport tailings under typical stormflow conditions, and deposited tailings from floodplain terraces could be suspended and transported during typical storm events following the failure.

Based on historical failures, we assumed that sediment deposition would be greatest near the dam, forming an initial "wedge" that would be deposited rapidly and extend from the lowest elevation of the breach. Given the potential mobility of the fine-grained tailings, we held the initial modeled slope to 1.6%, the valley slope near the dam. We determined this slope was a reasonable estimate based on comparison with a publicly available, simple tailings flow calculator that predicts flow depths for tailings with a variety of viscosities (WISE 2012). Extending this slope from the dam breach, calculated sediment depths ranged from 1 m to 20 m 1.4 km downstream of the dam for both failure scenarios. We modeled that, on average, approximately 1 m of deposited tailings would remain on valley surfaces (i.e., in the channel and on the floodplain) downstream of the dam following each failure; this created a conservative, uniform estimate of sediment deposition. Deposition at each cross-section at this 1 m meter depth was used to calculate the volume between modeled river sections, and this volume was subtracted from the volume released from the tailings dam failure. We assumed that the remaining sediment in the tailings dam failure flow was available to deposit at the next downstream section, and this logic was carried downstream until the end of the modeled river length was reached.

9.3.1 Hydrologic Characteristics

Model results for hydrologic characteristics of the Pebble 0.25 and Pebble 2.0 tailings dam failures are shown in Table 9-4. In both cases, estimated peak flows during a TSF dam failure would be much larger than streamflows typically experienced in this watershed. This is because the impounded tailings would create a flood wave far larger than any that could result from a precipitation event alone. The tailings dam failure and subsequent release of massive quantities of impounded tailings and associated pore water would produce a peak flood immediately downstream of the dam. Maximum depths of the flood wave would exceed 10 m and 25 m, with peak velocities of approximately 4 and 10 m/s (14 and 36 km/hr), for the Pebble 0.25 and Pebble 2.0 dam failures, respectively (Table 9-4).

Peak discharges would exceed 5,000 m^3/s for the Pebble 0.25 dam failure and 39,000 m^3/s for the Pebble 2.0 dam failure. If the failure occurred during an intense rainfall or rapid snowmelt event discharges would be negligibly higher, due to the small watershed area of the TSF 1 dam. A dam-break flood of this magnitude would dwarf the peak flows of even the largest rivers in the region. For example, a local flood event measured by a U.S. Geological Survey (USGS) gage on the Nushagak River located near the village of Ekwok, Alaska (Figure 2-4), experienced a record peak flood of 3,313 m^3/s. Peak flows predicted in the North Fork Koktuli River valley from the TSF dam failures would be more than 1.6 times (Pebble 0.25) and 11.8 times (Pebble 2.0) greater than the flood of record on the Nushagak River at Ekwok. Although we recognize that these are not analogous watersheds, this observed flood does provide a point of reference for the flood magnitudes that would result from tailings dam failures.

9.3.2 Sediment Transport and Deposition

Dam failure flood waves (Table 9-4) and post-failure recessional flows in the Pebble 2.0 failure scenario suggest that transport and deposition of tailings material could occur throughout (and beyond) the 30-km modeled reach (Table 9-5). Deposition in the Pebble 0.25 failure scenario could extend for over 29 km, to within 1 km of the confluence with the South Fork Koktuli (Table 9-5). After the initial deposition event, concentrated channel flows and floodplain conveyance areas would continue to transport sediment further downstream, as channel and valley morphology would re-establish in the newly deposited substrate.

Even with only 20% of impounded tailings mobilized, the flood wave and tailings deposition that would result from a tailings dam failure under either dam failure scenario could significantly alter the downstream channel and floodplain. The initial flood itself would have the capacity to scour the channel and floodplain, as the wave of tailings slurry would travel down the valley at velocities of up to approximately 10 m/s (Table 9-4). The quantity of mobilized sediments that could be released from the TSF would bury the existing channel and floodplain under meters of fine-grained sediment in an initial wedge near the dam; this material would move across the downstream valley as the flood wave receded and water velocities slowed (Box 9-4, Tables 9-4 and 9-5). The sediment regime of the affected stream and downstream waters would be greatly altered, with calculated sediment depths of up to 20 m (Pebble 2.0) and 1 m (Pebble 0.25) extending 1.4 km downstream of the dam.

Table 9-4. HEC-RAS model results for the Pebble 0.25 and Pebble 2.0 TSF dam failure analyses. Values were modeled for more than 39 river stations along a 30-km length of the North Fork Koktuli River; representative river stations along that length are shown here, listed by distance upstream from the confluence of the South and North Fork Koktuli Rivers (River Station 30 km = foot of the TSF 1 dam).

River Station (km)	Pebble 0.25 Dam Failure[a]					Pebble 2.0 Dam Failure[b]				
	Discharge (m^3/s)	Depth (m)	Velocity (m/s)			Discharge (m^3/s)	Depth (m)	Velocity (m/s)		
			LFP	CH	RFP			LFP	CH	RFP
30.0	5,270	11.2	1.6	3.1	1.4	39,100	23.6	4.1	7.2	4.2
29.0	5,270	10.7	1.2	4.2	1.4	39,100	19.8	5.2	10.5	5.8
25.0	4,990	9.8	0.4	0.8	0.6	39,000	20.2	1.3	2.1	1.4
20.4	4,190	7.5	0.5	0.9	0.5	38,000	19.0	1.5	2.5	1.5
15.5	3,610	10.0	0.8	1.3	0.7	34,900	25.8	1.5	2.6	1.8
10.3	2,940	12.3	0.8	1.6	0.7	29,800	27.2	2.5	4.2	2.3
4.8	2,650	4.1	0.3	0.5	<0.1	25,800	10.1	0.9	1.4	0.5
0.0	1,710	5.8	0.1	0.4	<0.1	18,600	10.6	0.5	0.9	<0.1

Notes:

[a] Dam height = 92 m, maximum volume of tailings and water expected to be stored = 158 million m^3.

[b] Dam height = 209 m, maximum volume of tailings and water expected to be stored = 1,270 million m^3.

HEC-RAS = Hydrologic Engineering Center's River Analysis System; LFP = left floodplain; CH = channel; RFP = right floodplain; TSF = tailings storage facility.

Table 9-5. Tailings mobilized and deposited in the Pebble 0.25 and Pebble 2.0 dam failures analyses. The mobilized tailings include material within the dam cross-section that has failed, plus a percentage (5 to 20%) of the stored tailings material. See Box 9-4 for additional information on how the dam failures were modeled.

Failure Scenario	Volume of Tailings[a] (million m^3)	% Mobilized[b]	Mobilized Tailings (metric tons)	Tailings in Transport at Downstream Extent of Model[c] (metric tons)	Downstream Extent of Wedge (km)	Downstream Extent of Expected Deposition[d] (km)
Pebble 0.25	158	20	59,724,000	0	1.4	29
		15	44,793,000	0	1.4	27
		10	29,862,000	0	1.4	24
		5	14,931,000	0	1.4	9
Pebble 2.0	1,270	20	479,682,000	350,668,000	1.4	30 (+)
		15	359,761,500	241,756,000	1.4	30 (+)
		10	239,841,000	138,981,000	1.4	30 (+)
		5	119,920,500	39,767,000	1.4	30 (+)

Notes:

[a] Maximum volume of tailings and water expected to be stored, allowing for freeboard in tailings storage facility (TSF). This volume was used to estimate metric tons of stored tailings released in a TSF dam failure, using an average tailings total density of 1.89 metric tons/m^3 and an average tailings dry density of 1.42 metric tons/m^3.

[b] 20% value was used in model; values less than 20% are shown to illustrate how weight of mobilized tailings changes with % mobilized.

[c] Weight of mobilized tailings that would remain in transport assuming 1 m of deposition in the floodplain inundation area.

[d] Measured downstream from face of dam.

Downstream of this initial sediment wedge, deposition could occur in the channel and the floodplain as peak flood discharges decreased with increasing distance downstream of the dam, water velocities returned to baseflow levels, and the potential for tailings deposition increased. In the Pebble 0.25 failure scenario, release of 20% of tailings material was sufficient to fill the entire North Fork Koktuli River valley to within 1 km of the confluence with the South Fork Koktuli River with an average depth of 1 m of tailings material (Table 9-5). In the Pebble 2.0 failure scenario, over 350 million metric tons of sediment remained available for transport and subsequent deposition beyond the end of the modeled reach, indicating that tailings would extend into the mainstem Koktuli River (Table 9-5).

Most of the deposition would be very fine material that would be susceptible to resuspension and deposition with each subsequent natural flow event. Following the dam failure, the stream channel would seek equilibrium and could remain unstable over several flow events, potentially creating a braided system in the post-failure depositional zone. As the new valley fluvial geomorphology developed over time, newly deposited materials on the floodplain, material at the base of the dam, and material that remained behind the breached dam of the TSF (if not removed or contained by corrective action) would serve as concentrated sources of easily transportable, potentially toxic material (Section 9.4).

The two possible failure scenarios presented here are well within the range of reported case histories. For example, when the parameters for the Pebble 0.25 and Pebble 2.0 dam failures were applied to runout distance equations from Rico et al. (2008), expected runout distances reached the marine waters of Bristol Bay. In our analyses, we made a simple assumption that deposition depths averaged 1 m (Box 9-4). We emphasize that our tailings dam failure scenarios reflect a range of possible outcomes, but are not exhaustive. The depth of tailings deposition on the floodplain could be higher or lower, depending on the amount of tailings mobilized and the runout distance. Based on historical tailings dam failure data, potential runout distances can range from hundreds to thousands of kilometers (Box 9-1).

9.3.3 Uncertainties

In this chapter, we have evaluated two potential dam failure scenarios, both caused by overtopping. Although our sensitivity analyses indicate that the repercussions of failure were relatively insensitive to the initial cause of the failure (Box 9-4), it is important to note that overtopping represents only one of several potential failure mechanisms (Section 9.1.1).

Also, a significant amount of uncertainty surrounds potential sediment deposition depths and downstream distributions. Valley topography, rate of the dam failure, and ultimate make-up of the flood wave sediment concentration and viscosity can affect outcomes and complicate predictive efforts. Despite the uncertainty associated with the massive quantities of sediment available and the complexities of hydraulic forces that would act on this sediment, we present reasonable post-failure sediment deposition outcomes in the two dam failure scenarios. Other outcomes are possible, but all share the common reality that massive quantities (i.e., millions of cubic meters) of tailings fines would be deposited in downstream floodplains and channels (Table 9-5).

Use of a traditional sediment transport model would likely improve estimates of sediment movement and deposition, especially as the model is extended further downstream. In addition, tributary streams would input clean water at each confluence. Because of the site-specific data required to implement a sediment transport model, we limited our model to the 30 km above the confluence of the South and North Fork Koktuli Rivers (Box 9-4).

9.4 Scour, Sediment Deposition, and Turbidity

The Pebble 0.25 and Pebble 2.0 tailings dam failures described in the preceding section could have devastating effects on aquatic life and habitat (Figure 9-1). We identified several processes associated with a tailings dam failure that would pose risks to aquatic habitat. These processes include exposure to hydraulic scour and bed mobilization, deposition of tailings fines, and mobilization and suspension of tailings fines affecting downstream water and habitat quality. Effects of suspended sediments are discussed in Section 9.5.1, and effects associated with potential toxicity are discussed in Section 9.5.2.

Natural background conditions indicate the sediment levels that support the region's current productivity of salmonid populations, and two available sources provide data on substrate size distribution and fine sediment concentrations in the study area. Pebble Limited Partnership (PLP) (2011) reports concentrations of fine sediments from sieved bulk gravel samples collected at three known salmon spawning sites in the South and North Fork Koktuli Rivers and Upper Talarik Creek (sample locations are shown in report by PLP [2011: Figure 4 in Appendix 15.1F]). Average concentration of fines (less than 0.84 mm) was less than 6% for all streams and dates, except for the August sample from the uppermost South Fork Koktuli River site (gage SGSK3) (PLP 2011: Figure 27 in Appendix 15.1F), which had nearly 8% fines. The geometric mean grain size was greater than 15 mm at all sites for both sampling periods, except the uppermost Upper Talarik Creek site (gage SGUT3), where the mean grain size for both seasons was between 10 and 15 mm (PLP 2011: Figure 26 in Appendix 15.1F). These data led the authors to conclude that gravel quality was generally high and that, based on published criteria (Shirazi et al. 1981, Chapman and McLeod 1987, Kondolf 2000), salmonid survival to emergence would be high (presumably above 80%) at all sites except the uppermost Upper Talarik Creek site, where criteria predicted survival between 50 and 80% (PLP 2011).

Areal coverage of substrate sizes is also available for 77 wadeable stream sites around the Nushagak and Kvichak River watersheds, including one site each on the South and North Fork Koktuli Rivers and Upper Talarik Creek (Table 9-6). Substrate sampling at these study sites followed U.S. Environmental Protection Agency (USEPA) methodology (Peck et al. 2006), in which five particles are systematically selected across each of 21 evenly spaced transects (from each wetted margin of the channel and three locations in between). These data indicate that a mix of substrate sizes occurs in these streambeds, with cobble and gravel generally abundant (Table 9-6).

Table 9-6. Sediment size distributions surveyed at the South and North Fork Koktuli Rivers, Upper Talarik Creek, and 77 wadeable stream sites in the Nushagak and Kvichak River watersheds. Values represent percentage areal coverage based on 105 systematically selected particles at each site, following U.S. Environmental Protection Agency methods. All data were collected during June.

River or Stream(s)	Date	Latitude	Longitude	% Large Boulder (>1000 mm)	% Small Boulder (250–1000 mm)	% Cobble (64–250 mm)	% Coarse Gravel (16–64 mm)	% Fine Gravel (2–16 mm)	% Sand (0.06–2 mm)	% Fines (<0.06 mm)
South Fork Koktuli River	6/8/2010	59.83047	−155.27719	-	3	3	55	16	0	23
North Fork Koktuli River	6/6/2009	59.84033	−155.71272	-	-	17	49	24	10	-
Upper Talarik Creek	6/13/2011	59.91820	−155.27771	-	2	30	29	13	24	2
77 streams	2008 to 2011	-	-	0 (±1)	2 (±4)	13 (±13)	40 (±15)	17 (±12)	17 (±11)	-

Notes:
Dashes (-) indicate values equal to zero.
Sources: Rinella pers. comm., Peck et al. 2006.

9.4.1 Exposure through Sediment Transport and Deposition

The tailings dam failure scenarios evaluated here would result in intense scour and extensive deposition in the North Fork Koktuli River valley. Deposition would extend from the tailings dam downstream for many kilometers. Even with our conservative assumption that 20% of the tailings would be released, deposition would extend to within 1 km of or beyond the confluence with the South Fork Koktuli River, a distance of approximately 30 km (Table 9-5). This volume of available fine tailings material could result in many meters of deposition in a sediment wedge across the entire valley near the TSF dam, with lesser thicknesses of fines deposited to the confluence with the South Fork Koktuli River or beyond. Erosion and transport of fines would be expected to continue as the channel adjusted to the vastly increased fine sediment supply.

To translate these tailings dam failures into effects on aquatic habitat and biota, we assumed that the calculated velocities during the tailings dam failure flood event (Table 9-4) and the associated transport and deposition of tailings material and collected debris (Table 9-5) would result in a reworking and mobilization of the existing North Fork Koktuli River channel bed and banks downstream of the TSF. Given the volume of material that would be exported from the TSF, we assume that portions of the new valley floor would be predominantly tailings material, with 70% of the particle mass being finer than 0.1 mm. Following recession of the tailings dam failure flood event, we assume that the bed, margins, and floodplain would be primarily tailings material, with incorporated coarser dam fill and valley fill material accounting for less than 20%.

Immediately following either a Pebble 0.25 or Pebble 2.0 tailings dam failure, suitable spawning and rearing habitat for salmon and other native fishes would be eliminated in the North Fork Koktuli River downstream of the tailings dam. Tributaries of the North Fork Koktuli River, including portions of the watershed upstream of the confluence of the North Fork Koktuli River tributary containing the TSF, could also be adversely affected. Temporary flooding of tributary junctions during the tailings dam failure event, and subsequent sediment deposition at confluence zones causing local aggradation, steepening, or shallowing of tributary confluences, could make movement of resident and anadromous fish between tributaries and the mainstem difficult. Recovery of channel dimensions and substrate size distributions suitable for salmonid spawning and rearing habitat would be contingent upon rates of fine sediment export and recruitment of gravels and larger substrates from tributaries or pre-failure valley fill.

The type, magnitude, and frequency of channel adjustments that would occur in the North Fork Koktuli River valley following a tailings dam failure would depend on available sediment, channel slope, and discharge. Post-failure streams flowing across the depositional zone would have tremendous supplies of fine-grained sediments in the channel bed and banks available for transport. Channels would likely experience rapid channel incision with frequent bank failure, followed by periods of channel widening and aggradation interspersed with episodic channel avulsion. Given the volume and depth of deposition, stream channels would likely remain unstable and continue to contribute sediments to downstream reaches until equilibrium conditions were met. Recovery of suitable structural habitat in the North Fork

Koktuli River watershed would likely take decades, given the volume of sediment that could be delivered in the tailings dam failures considered here. Whether the benefits of removing spilled tailings fines would outweigh the risks of additional adverse impacts resulting from dredging and removal operations would depend on the nature and distribution of the tailings spill, the duration of risks, and existing technologies (e.g., Wenning et al. 2006).

The tailings dam failure scenarios evaluated here would have the potential to fill the North Fork Koktuli River valley with extensive deposits of tailings fines and, in some cases, still carry a substantial volume of fine sediments farther downstream. The mass of material remaining in transport at the confluence of the South and North Fork Koktuli Rivers following a Pebble 2.0 tailings dam failure, and thus available for deposition in the mainstem Koktuli, Mulchatna, and Nushagak Rivers could exceed 350 million metric tons (Table 9-5). In addition, some of the remaining stored tailings material could mobilize as pore water seeped from the exposed slopes immediately following the failure event, creating slides and smaller flow events. Fine sediment could also be mobilized during any subsequent precipitation or snow melt runoff events that would direct water across the tailings and down valley through the breach before it was repaired. The depth and distribution of fines in the mainstem Koktuli, Mulchatna, and Nushagak Rivers cannot be estimated at this time, but given the volume and grain size of these sediments, it is reasonable to expect that continued pulses of fine sediments would be transported through and transiently stored in these mainstem rivers during spring snow melt and fall rain events for many years (Knighton 1984).

9.4.2 Exposure-Response

9.4.2.1 Fish

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

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

9.4.2.2 Invertebrates

Aquatic macroinvertebrates are an important food source for Chinook and coho salmon, rainbow trout, Dolly Varden, Arctic grayling, and other fishes that rear in streams of the mine scenario watersheds (Nielsen 1992, Scheuerell et al. 2007). Two available data sources describe the existing macroinvertebrate communities for streams in the study area: PLP (2011: Chapter 15.2) and Bogan et al. (2012). Both documents describe broadly similar communities that are consistent with those reported from other regions of Alaska (Oswood 1989). Communities are reasonably diverse, with Bogan et al. (2012) reporting 137 taxa from 38 families, with 9 to 40 taxa occurring at a given site (Chironomidae were lumped at the family level). Communities are dominated by Diptera (true flies), primarily Chironomidae (midges), with lesser numbers of Ephemeroptera (mayflies) and Plecoptera (stoneflies) and relatively few Trichoptera (caddisflies). Macroinvertebrate densities were characteristically variable, ranging two orders of magnitude (102 to 11,371 organisms per m^2) (Bogan et al. 2012).

In addition to the direct impacts on fish described in Section 9.4.2.1, catastrophic sedimentation associated with tailings dam failure also would likely affect fish populations through habitat-related reductions in macroinvertebrate food resources (see Section 9.5 for discussion of toxicity-related effects). Sedimentation can affect benthic macroinvertebrates through abrasion, burial, and reduction of living space, oxygen supply, and food availability (Jones et al. 2011). Deleterious effects of sedimentation have been reviewed thoroughly (Wood and Armitage 1997, Jones et al. 2011). Sedimentation typically leads to reductions in density and taxonomic diversity (Wagener and LaPerriere 1985, Culp et al. 1986, Quinn et al. 1992, Milner and Piorkowski 2004), even at sediment loads substantially lower than those modeled in the tailings dam failure scenarios (Wood and Armitage 1997, Jones et al. 2011). The conversion of a stable streambed dominated by gravel and cobble to a highly unstable one composed entirely of fine sediments, as would occur in the tailings dam failures considered here, would certainly lead to reductions in the biomass and diversity of macroinvertebrate prey available to fish populations.

9.4.3 Risk Characterization

The complete loss of suitable salmonid habitat in the North Fork Koktuli River in the short term (less than 10 years), along with the likelihood of very low-quality spawning and rearing habitat in the long term (decades), would likely result in near-complete loss of North Fork Koktuli River fish populations downstream of the tailings dam. These impacts would persist for multiple salmon life cycles, so salmon cohorts that are at sea during the tailings dam failure would eventually return to find degraded spawning and rearing habitat. The North Fork Koktuli River provides complex, low-gradient, high-quality habitats that currently support spawning and rearing populations of sockeye, Chinook, and coho salmon, and spawning populations of chum salmon (Figures 7-2 through 7-5) (Johnson and Blanche 2012). For example, aerial index surveys in the North Fork Koktuli River documented roughly 3,000 Chinook salmon (surveyed in 2005), 2,100 sockeye salmon (surveyed in 2004), 1,750 coho salmon (surveyed in 2008), and 1,400 chum salmon (surveyed in 2008) (values inferred from figures in report by PLP [2011: Chapter 15]). The North Fork Koktuli River also supports rearing Dolly Varden and rainbow trout (Figures 7-7 and 7-8) (ADF&G 2012). The Koktuli River watershed has been recognized as an important producer of Chinook salmon for the greater Nushagak River Management Zone (Dye and Schwanke 2009). Total Chinook salmon run-size estimates for the Nushagak River include estimates of harvest plus escapement of spawners. Estimates based on a variety of techniques, including sonar, averaged over 190,000 Chinook salmon from 2002 through 2011 (Buck et al. 2012), making the Nushagak the largest producer of Chinook salmon for the Bristol Bay region. Of all the Chinook salmon tallied during annual aerial index counts in the Nushagak River watershed between 1969 and 1985 (years that all reported spawning areas were surveyed), on average 29% (range 21 to 37%) were counted in the Koktuli River system (Dye and Schwanke 2009) (see Section 7.1.2 for a discussion of the limitations of abundance estimates based on aerial counts). The Mulchatna River accounts for another 12% (range 9 to 17%) of the Nushagak Chinook salmon count, and the Stuyahok River (which drains to the Mulchatna downstream of the Koktuli River) represents another 18% (range 10 to 27%). Hence, Chinook salmon production could be significantly degraded by loss of habitat downstream of the tailings dam, particularly if effects extended downstream into the Koktuli and Mulchatna Rivers and beyond.

Sockeye are the most abundant salmon returning to the Nushagak River watershed, with annual runs averaging more than 1.9 million fish between 2001 and 2010 (Jones et al. 2012). Spatially extensive sockeye salmon spawner data are not available for the Nushagak River watershed, so it is impossible to estimate what proportion of the population spawns in the Koktuli River system. The Nushagak River watershed supports two genetically and ecologically distinct groups of sockeye salmon (Dann et al. 2011): those that rear in, and spawn in and near lakes (lake-type, as in Semko 1954), and those that spawn and rear, at least briefly, in rivers and streams (sea-type and river-type, as in Semko 1954, collectively called riverine-type here). Sockeye salmon in much of the Mulchatna River system, including the Koktuli River and adjacent Stuyahok River, are riverine-type, and are more closely related to riverine-type sockeye salmon of the Kuskokwim River drainage than to Nushagak River watershed lake-type sockeye salmon (Dann et al. 2011). It is likely that these population groups share a similar life history pattern. Riverine-type sockeye in Kuskokwim River tributaries preferentially rear in off- and side-channel habitats within floodplain-prone stream reaches (Ruggerone et al. 2011). From 1995 to 2006, an estimated 528,000 adult sockeye salmon annually migrated to spawning areas in the Nushagak and Mulchatna River systems upstream of the Wood River system (Jones et al. 2012). Of these, approximately 70% (an annual average of 363,000) appear to be riverine-type sockeye salmon based on the proportion of sockeye that escaped to the Nushagak/Mulchatna portion of the basin.

Spawning and rearing riverine-type sockeye salmon habitats occur throughout the South and North Fork Koktuli Rivers downstream to and beyond the confluence of the Mulchatna and Nushagak Rivers

(ADF&G 2012). The tailings dam failures considered here would likely affect sockeye salmon production throughout the Koktuli River system, but the proportion of the total Nushagak River sockeye salmon production that would potentially be affected is unknown (see Section 7.1 for additional information on fish abundance).

Populations of resident and anadromous fishes present in North Fork Koktuli River headwaters upstream of TSF 1 or in tributaries at the time of a tailings dam failure would not immediately suffer habitat losses, but would suffer indirect effects resulting from alteration of the North Fork Koktuli River valley. Many species in the region's rivers, including resident non-anadromous species, undergo extensive seasonal migrations (West et al. 1992). Such movements are important for juveniles moving from natal areas to overwintering habitats, for smolts emigrating to sea, for adult spawning migrations, or, in the case of resident species, for migration between spawning, foraging, and overwintering areas. Sediment deposition at tributary mouths in the North Fork Koktuli River valley could adversely affect passage of juvenile and adult fish into and out of these tributaries. For several years, access to mainstem river habitats upon which many tributary fishes depend for portions of their life history could be blocked or severely degraded.

Successful re-colonization of the North Fork Koktuli River by resident fish would depend on whether unimpaired tributary habitats or downstream areas could function as suitable refugia and source areas for re-colonization of the North Fork Koktuli River. Resident fish would require sufficient tributary habitat to complete their entire life history, as it is likely that downstream habitat would be unusable for multiple generations. Re-colonization of salmon from tributary refugia or downstream areas would require suitable passage at tributary junctions and suitable migratory corridors throughout the mainstem. Aquatic macroinvertebrate food resources would also likely be adversely affected in the main river channel, limiting rearing potential for insectivorous fishes such as juvenile salmonids. Given estimates of fine-sediment deposition and the unstable, silt and sand bed channels that would likely form across the valley floor, as well as metal concentrations in these tailing substrates that could inhibit migratory behavior (Section 9.5.2.1), successful migratory conditions seem unlikely for at least several years after a tailings dam failure.

The near-complete loss of North Fork Koktuli River fish populations downstream of the TSF and long-term transport of fine sediment to downstream locations would have significant adverse effects on the Koktuli and Nushagak River salmon, Dolly Varden, and rainbow trout populations, affecting downstream fisheries, including subsistence users (Figure 5-2). Spawning and rearing habitat would be eliminated or impaired by deposition of transported sediment and/or reductions in the invertebrate prey base. Direct loss of habitat in the North Fork Koktuli River, and impairments further downstream because of transport and deposition of sediment, could adversely affect a substantial portion of Chinook salmon returning to the Nushagak River watershed. Assuming that Alaska Department of Fish and Game (ADF&G) aerial survey counts reflect the proportional distribution of Chinook salmon in the Nushagak River watershed, habitat destruction of the North Fork Koktuli River valley, downstream transport of sediment to the Koktuli River mainstem, and subsequent loss of access to or inhibition of migration into the South Fork Koktuli River would affect the entire Koktuli River component of the Nushagak Chinook run. If the deposited tailings material is of sufficient quantity and toxicity (Section 9.5.2) to have effects on aquatic life and fish migratory behavior in the lower Koktuli, Mulchatna, and Stuyahok Rivers, much greater proportions of the Nushagak Chinook populations and other resident and anadromous fish populations could be affected. Adult salmon returning to these rivers could potentially seek other tributaries for spawning, but successful recruitment of displaced spawners would require access to and comparable use of spawning and rearing capacity elsewhere in the Nushagak River watershed.

9.4.4 Uncertainties

It is certain that a tailings dam failure such as those evaluated here would have devastating effects on aquatic habitat and biota, but the distribution and magnitude of effects is uncertain. Uncertainties associated with the initial events, including the likelihood of dam failures and sediment transport and deposition processes are discussed in Sections 9.1.3 and 9.3.3. Uncertainties associated with the timing, feasibility, and effectiveness of remediation of a tailings spill are discussed in Section 9.6.2. Other uncertainties related to the time frame for geomorphic recovery, the longitudinal extent and magnitude of habitat impacts downstream of our modeled 30-km reach of the North Fork Koktuli River, and the fish populations affected are discussed in this section.

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

The tailings dam failure simulations (Section 9.3) were restricted to approximately 30 km of the North Fork Koktuli River, from the face of the TSF 1 dam downstream to the confluence of the South and North Fork Koktuli Rivers. Extension of the simulations beyond this confluence would introduce significant error and uncertainty associated with the contribution of South Fork Koktuli River flows, and would require a more sophisticated sediment transport model. As a result, we were unable to quantify sediment transport and deposition in the mainstem Koktuli, Mulchatna, and Nushagak Rivers. However, given the high volume of tailings fines that could be transported beyond the confluence of the South and North Fork Koktuli Rivers (Table 9-5), it is highly likely that impacts on fish habitat estimated for the North Fork Koktuli River would extend some significant distance down the mainstem Koktuli River.

We estimate that the combined effects of direct habitat losses in the North Fork Koktuli River, downstream in the mainstem Koktuli River, and beyond, as well as impacts on macroinvertebrate prey for salmon, could adversely affect 25% or more of Chinook salmon returning to spawn in the Nushagak

River watershed. If the Koktuli River, Stuyahok, and Mulchatna portion of the Nushagak runs are impacted via downstream transport of tailings fines, the tailings dam failure may affect nearly 60% of the Chinook run (on average, 59% of the aerial survey counts were from these three watersheds [range = 48% to 75%]). Uncertainty around this estimate is associated with the downstream extent of habitat impacts (described above) and the variable and imprecise estimates of the relative abundance of Chinook salmon in the Nushagak, Mulchatna, and Koktuli River systems. We based our estimate of proportions on long-term (1969 to 1970 and 1974 to 1985) aerial counts of Chinook salmon collected and interpreted by ADF&G (Dye and Schwanke 2009), but aerial counts can substantially underestimate true abundance (Jones et al. 1998).

Because long-term abundance data are lacking for most other fish species and locations in the mine scenario watersheds, losses caused by a tailings dam failure are not quantified for other species. Our analysis focuses on a few endpoint species, and does not incorporate considerations of metacommunity dynamics, which are poorly understood for the region but may be critical to understanding species responses to environmental change (Westley et al. 2010). Information documenting known occurrence of non-endpoint fish species in the region's rivers and major streams is available (Johnson and Blanche 2012, ADF&G 2012), but information on their abundances, productivities, and limiting factors is not currently available.

9.5 Post-Tailings Spill Water Quality

9.5.1 Suspended Tailings Particles

9.5.1.1 Exposure

During a tailings dam failure, aquatic biota would be exposed to a slurry of suspended tailings moving at up to 10 m/s (Table 9-4). In the Pebble 2.0 scenario, much of this material would still be flowing 30 km downstream, at the mouth of the North Fork Koktuli River (the limit of the model) (Table 9-5).

For years after a tailings dam failure, settled tailings would be resuspended and carried downstream. At first, this process would be frequent if not continuous (except when and where the substrate is frozen), as channel and floodplain structure is established by erosional processes resuspending the tailings (Section 9.4.1). Gradually, as the tailings flow downstream, a substrate consisting of gravel and cobble embedded in tailings fines would become established, and the flow velocities necessary to suspend sediment would increase until they resembled those of an undisturbed stream.

Studies at other tailings-contaminated sites do not usefully address suspended tailings, as they typically have been carried out long after the spills occurred, are based on events that differ from the one large spill that would result from a tailings dam failure, and focus on toxic properties of the tailings (Section 9.5.2.3).

9.5.1.2 Exposure-Response

Suspended sediment has a variety of effects on fish that are similar to effects of toxic chemicals. Like chemical effects, the severity of effects increases with concentration and duration of exposure (Newcombe and Jensen 1996). At low levels, suspended sediment causes physiological and behavioral effects; at higher levels it causes death. Salmonids avoid turbid waters when possible, which may result in loss or underutilization of traditional spawning habitats (Bisson and Bilby 1982, Newcombe and Jensen 1996). However, salmonids must withstand brief periods of high suspended sediment concentrations associated with spring floods (Rowe et al. 2003). Empirically derived effective exposures for lethal and sublethal effects (i.e., reduced abundance or growth or delayed hatching) on juvenile and adult salmonids may be summarized as follows (derived from Newcombe and Jensen 1996):

- 22,026 mg/L for 1 hour
- 2,981 mg/L for 3 hours
- 1,097 mg/L for 7 hours
- 148 mg/L for 1 to 2 days
- 55 mg/L for 6 days
- 7 mg/L for 2 weeks
- 3 mg/L for 7 weeks to 11 months

However, salmon may adapt to migrate through high levels of suspended sediment. For example, during mid-May to early August, when adult salmon migrate upstream through the lower Copper River (El Mejjati et al. 2010), suspended sediment concentrations range from 750 to 1780 mg/L (Brabets 1992).

9.5.1.3 Risk Characterization

During and immediately after a tailings spill, exposure to suspended sediment would be far higher than any of the effects thresholds listed above. Fish could be literally smothered and buried in the slurry. Because the standard of 1,000 mg/L of suspended sediment is exceeded by ordinary events such as erosion of construction sites and tilled fields, erosion of tailings from the re-formation of the channel and floodplain would likely exceed that standard for days at a time, over a period of years. Fish would be likely to avoid these streams or experience lethality, reduced growth, or reduced abundance. Avoidance could also block migrating salmon and other fish from their spawning areas in upstream tributaries during these periods, although salmon have adapted to migration corridors with high suspended sediment levels. The potential for tailings to be more aversive or toxic than natural suspended sediment is unknown. Exposure levels would gradually decline over time as tailings are carried downstream, channel stability increases, and the floodplain becomes revegetated. Rates of these processes are unknown, but it is reasonable to assume that decades would be required for suspended sediment loads in the Koktuli and Mulchatna Rivers to drop to levels that occur with normal high flows in stable channels of the Bristol Bay watershed.

9.5.1.4 Uncertainties

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

9.5.2 Tailings Constituents

Although the most dramatic effect of a tailings dam failure would be habitat destruction and modification due to the flow of tailings slurry downstream, exposures to potentially toxic materials in the slurry would also occur. The toxic effects of a tailings dam failure can be assessed using the composition of the tailings and of experimental tailings leachates, as well as experience with tailings spills at other sites (Box 9-5).

9.5.2.1 Exposure

Aqueous Exposures to Impoundment Waters

During a tailings dam failure, aquatic biota would be exposed to water that had been in contact with tailings during processing and in the TSF. This water would include pore water associated with the deposited tailings and water overlying the tailings. If the spill was caused by flow through a fault in the dam or by a seismically induced tailings dam failure (i.e., a failure under “dry” conditions), undiluted pore water and supernatant water would be released. If the dam was eroded or overtopped by a flooding event (i.e., a failure under “wet” conditions), the pore and surface water could be diluted by fresh water. However, this dilution would be trivial relative to the volume of pore water in the tailings.

A spill would have two phases in our tailings dam failure scenarios; other scenarios could differ in timing and magnitude. At first, tailings slurry would pour through the breach for approximately 2 hours based on the specified rates of dam erosion and slurry flow. Pore water would then drain from the tailings that are not released but are above the elevation of the breach. This latter process would be slow and could continue until the dam was repaired. If a tailings dam failure occurred after the mine site was abandoned and no corrective action was taken or was delayed, an equilibrium would be achieved in which rain, snow, and upstream flows were balanced by outflow of leachate through the breach.

BOX 9-5. BACKGROUND ON RELEVANT ANALOGOUS TAILINGS SPILL SITES

Past deliberate or accidental spills of metal mine tailings into salmonid streams and rivers have occurred by mechanisms and mining practices other than those evaluated in this assessment. However, these spills provide evidence concerning the fate of tailings and the nature of exposures to aquatic biota once the tailings are in streams and floodplains. In the United States, some of these sites are relatively well studied because their observed effects have led to classification as Superfund sites. Other tailings spills have caused extensive fish kills and other significant effects but have not generated useful long-term monitoring data. These brief descriptions provide background information and support the use of evidence from these cases in analyzing risks from potential tailings dam failures in the Bristol Bay watershed.

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

The primary source of exposure is tailings deposited on the floodplains, resulting in aquatic pollution through erosion and leaching. Large areas with acidic tailings (both acidic and neutral tailings were deposited) are barren of plant life due to metal toxicity, which contributes to erosion and leaching. The river was fishless from the late 1800s to the 1950s, but has begun to recover. Trout and other fishes continue to exhibit low growth and abundance, and intermittent fish kills have followed metal pulses from rainstorms or rapid snow melt. However, sedimentation was also thought to contribute to effects on fish populations through habitat degradation. Detailed information can be found in the responsible party's remedial investigation (ARCO 1998) and in U.S. Environmental Protection Agency (USEPA) documents (USEPA 2012a).

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

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

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

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

Once in the stream, potentially toxic constituents dissolved in the water would not settle out. Because the potentially toxic constituents are not degradable or volatile, they would eventually flow to Bristol Bay, although they would be diluted along the way. In the Pebble 2.0 tailings dam failure, the peak flow of mobilized tailings at the confluence of the North and South Fork Koktuli Rivers is estimated to be approximately 18,600 m^3/s (Table 9-4). The Nushagak River at Ekwok (Figure 2-4) would be the first downstream gaging station at which most of the tailings would have settled out and dilution could be estimated. Using the annual average and highest monthly average flows (668 and 1,215 m^3/s, respectively), concentrations of dissolved chemicals in the Nushagak River would be 96 and 94% of those in the spill. Minimum flow is not considered, because a failure is believed to be less likely during freezing conditions.

We used the tailings humidity cell test results to estimate the composition of the bulk of the aqueous phase. However, those values are uncertain, because none of the tests performed by PLP represent the leaching conditions in a tailings impoundment, material other than bulk tailings would be added to the TSFs, and no model exists to mathematically simulate the leaching process. However, some mixture of tailings supernatant, which represents the source water for the impoundment (Table 8-4); humidity cell leachate, which represents aqueous leaching from tailings under oxidizing conditions (Table 8-5); and a small amount of local water (Table 8-10) can be used to approximate aqueous phase composition.

During mine operation, tailings impoundment surface waters would consist of water used to transport the tailings (supernatant) and any other waters stored in the impoundments prior to reuse or treatment and discharge. Hence, the surface water is expected to resemble the PLP's test supernatant (Table 8-4) with some dilution by precipitation. However, those results do not include process chemicals (e.g., xanthates and cyanide) that may be associated with the supernatant but that are not quantified in this assessment. Supernatant water would be slightly diluted by rain and snow onto the surface of the impoundment, but peripheral berms should generally prevent dilution by runoff.

The waters released from a tailings spill during mine operation could consist of surface water, surficial pore water, and a much larger volume of deep pore water. The surficial tailings pore water would be generated by leaching tailings in the presence of some oxygen. The composition and concentrations of constituents in that water may be roughly similar to a mixture of those observed in the supernatant and humidity cell tests (Tables 8-4 and 8-5). Pore water from deep anoxic tailings would have begun primarily as supernatant, but may have lower metal content due to chemical precipitation under anoxic conditions. Leachate flowing from an abandoned and failed impoundment would be more oxidized because the cover water and much of the pore water would have drained away.

Aqueous Exposures from Deposited Tailings

After a tailings dam failure, aquatic biota would be exposed to potentially toxic tailings that covered the substrate of streams and rivers. Benthic organisms, or aquatic insects and other invertebrates that burrow into or crawl upon substrates, would be most exposed. Eggs and larvae (fry) of any salmon, trout, or char that spawned in the contaminated substrate also would be exposed. In either case, the bioavailable contaminants would be those that are dissolved in the pore water of the deposited tailings.

Hence, exposure is determined by the rate of leaching of the tailings and the rate of dilution of the leachate, which depend on hydrological conditions. Unlike the lakes and estuaries that are the usual sites of sediment pollution studies, streams have a high level of interaction between substrates and surface waters. Shallow, turbulent water is typically near oxygen saturation. Bedload sediment bounces and slides downstream during high flows. At high enough flows sediment is suspended, exposing it to oxygen. Water also flows longitudinally and laterally through bed and floodplain sediments and vertically between groundwater and surface water.

Because the biologically active zone is oxidized, the tailings leachate to which biota would be exposed could resemble leachates from the supernatants and humidity cells (Tables 8-4 and 8-5). Ideally, a leaching test would be performed that simulated conditions in a streambed, but no such test results are available. In theory, leachate composition could be estimated using a mechanistic model, but no such model is available. Dilution of the leachate would be minimal in low-flow areas such as pools and backwaters and during low-flow periods. Dilution would be greatest in high-flow and turbulent locations such as riffles, in groundwater upwelling or downwelling areas, and during high-flow periods such as spring runoff and floods. However, high flows would be expected to increase leaching rates, resulting in complex dynamics (Nagorski et al. 2003).

Although we assume that spilled tailings would be mixed and would have average metal compositions (Table 9-7), stream processes would be expected to sort them. In Soda Butte Creek (Box 9-5), copper concentrations in riffles and glides gradually decreased downstream from the tailings spill site. However, fine sediments in pools had higher copper concentrations than the high-energy segments, and some of the highest copper concentrations were found in fine pool sediments more than 10 km downstream (Nimmo et al. 1998).

After a spill, aquatic biota would also be indirectly exposed to tailings deposited on land, primarily in the floodplains. Erosion of these floodplain-deposited tailings would result in additional deposition in streams, potentially replacing tailings lost through streambed erosion (Marcus et al. 2001). In addition, rain and snowmelt would run across and percolate through tailings deposited on floodplains, leaching metals and carrying them into the stream. Leachate would also form during lateral groundwater movement through tailings, particularly where tailings deposited in wetlands. Floodplain-deposited tailings are leached in the presence of oxygen, with episodes of saturation and drainage (ARCO 1998). Hence, humidity cell leachates would be more relevant to this exposure route than to others, and leachate concentrations in Table 8-5 may roughly estimate leachate composition from floodplain-deposited tailings.

Table 9-7. Comparison of average metal concentrations of tailings (Appendix H) to threshold effect concentration and probable effect concentration values for freshwater sediments and sums of the quotients (∑ TU). Values are in mg/kg dry weight.

Tailings Constituents	Average	TEC[a]	TEC Quotient	PEC[a]	PEC Quotient
Ag	0.7	-	-	-	-
As	25	9.8	2.6	33	0.76
Ba	30	-	-	-	-
Be	0.3	-	-	-	-
Bi	0.6	-	-	-	-
Cd	0.1	0.99	0.10	5.0	0.02
Co	8.1	-	-	-	-
Cr	150	43	3.5	110	1.3
Cu	680	32	21	150	4.5
Hg	0.1	0.18	0.56	1.1	0.09
Mn	360	630	0.57	1200	0.30
Mo	52	-	-	-	-
Ni	68	23	2.9	49	1.4
Pb	15	36	0.41	130	0.12
Sb	1.0	-	-	-	-
Se	1.8	-	-	-	-
Tl	0.3	-	-	-	-
U	0.4	-	-	-	-
V	87	-	-	-	-
Zn	87	120	0.72	460	0.19
Sum	-	-	32	-	8.7

Notes:
Dashes (-) indicate that criteria are not available.
[a] TECs and PECs are consensus values from (MacDonald et al. 2000), except for Mn which are the TEL and PEL for *Hyalella azteca* 28-day tests from (Ingersoll et al. 1996).
TEC = threshold effect concentration; PEC = probable effect concentration; TEL = threshold effect level; PEL = probable effect level.

This leachate could have three fates: it could move upward during dry periods and deposit on the surface as soluble salts (e.g., hydrated metal sulfates); it could move down into buried soils and deposit as weak acid-extractable compounds (e.g., metal sulfides); or it could sorb to organic matter or move laterally to the surface channel as dissolved metal ions (Nimik and Moore 1991, ARCO 1998). Runoff from tailings-contaminated floodplains of the Clark Fork River had high copper levels (67.8 to 8,380 μg/L) (Nimik and Moore 1991, ARCO 1998). Concentrations from spills in the Bristol Bay watershed would probably be lower than for the acidic Clark Fork tailings and salt accumulation on the surface would be less because of greater precipitation, but the same processes would occur. Dilution of leachate that moves into the stream would be highly location- and condition-specific. Once in a stream, leached metals are likely to remain dissolved because of the highly dilute water chemistry in the region, but some precipitation or sorption to clays or organic matter would occur, depending on the conditions that moved the leachate into the stream.

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

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

Solid Phase Exposure to Deposited Tailings

Although the most bioavailable metals in sediment are those dissolved in pore water, it is useful to consider the whole sediment as a source of exposure. This approach avoids uncertainties associated with using leaching tests to represent field processes. It is reasonable to consider the average tailings composition to represent stream sediment to which biota downstream of a spill would be exposed (Table 9-7). During and after a tailings spill, there may be some sorting of the tailings by size or density that would result in locally higher metal compositions, but this variability cannot currently be quantified. Although the material in the failed dam would dilute the tailings initially, particles in the dam would be larger than the tailings and would settle out in the first few kilometers downstream. Some soil would be scoured from the receiving stream, but that would be associated with the first wave of the slurry. Hence, given the volume spilled, tailings in most of the initial depositional area would be effectively undiluted. After the spill, the tailings sediment would be diluted by clean sediment from tributaries, but that process would be slow because the volume of tailings deposited in the watershed would be so large and the watershed is nearly undisturbed except for potential mine facilities. The background sediment load (1.4 to 2.5 mg/L total suspended solids, Table 8-10) is miniscule compared to the meters of tailings that would be deposited (Table 9-5). The washing of tailings from floodplains into streams and rivers would be more important for many years, so the sediments in streams and rivers below a tailings spill would resemble average tailings.

Dietary Exposures

As discussed in Section 8.2.2, dietary exposures of fish to metals have been an issue of concern at mine sites. An adjustment factor for rainbow trout to account for a dietary component to aqueous exposures

(0.95) is presented in that section. It may be applied to cases in which both direct aqueous and dietary exposures may occur, such as flow into a stream through floodplain tailings or from upwelling through tailings. Dietary exposures with respect to sediment levels may also be estimated. In such cases, direct aqueous exposures of fish may be negligible, but invertebrates, particularly metal-tolerant insects such as chironomids, may accumulate metals, carry them out of the sediment, and then serve as sources of dietary exposure. This phenomenon has been documented in both the Clark Fork and Coeur d'Alene River basins (Kemble et al. 1994, Farag et al. 1999).

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

Collector-Gatherer Copper = 0.294 x

Scraper-Grazer Copper = 1.73 x

where x is sediment concentration and copper is tissue concentration, both in μg/g dry weight (Goodyear and McNeill 1999). Studies of the Soda Butte Creek tailings spill found that copper concentrations in mixed invertebrates were slightly lower than sediment concentrations (Marcus et al. 2001). Studies of the Clark Fork River give bioaccumulation factors for copper and river invertebrates ranging from 0.18 to 1.62, with factors generally rising as sediment concentrations declined (calculated from Brumbaugh et al. 1994, Ingersoll et al. 1994). Equivalent studies in the Coeur d'Alene River give very similar factors (0.15 to 1.77) (calculated from Farag et al. 1998). These results support the use of an average bioaccumulation factor of 1.0 (Goodyear and McNeill 1999). This implies copper concentrations in invertebrates are equal to those in sediments, which in this case are tailings with an average copper concentration of 680 mg/kg (Table 9-7).

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

Persistence of Exposures

Evidence that tailings persist in streams as sources of metal exposures is provided by prior tailings releases. A review by Miller (1997) found persistence of high metal content sediment in streams after 10 to 100 years. One well-documented case is provided by a tailings dam failure in Soda Butte Creek, Montana, in 1950 (Box 9-5) (Marcus et al. 2001). Sediment was still characterized by high copper concentrations after 48 years despite two 100-year floods, indicating that some tailings were retained by streams and maintained high metal levels even after decades of leaching. Similarly, the Coeur d'Alene River basin was contaminated by direct discharge of tailings to floodplains, tailings dam failures, and mine drainage, which caused extensive damage to the watershed (Box 9-5) (NRC 2005). Treatment of the mine drainage improved biotic communities, but they were still impaired, apparently as a result of metals leaching from deposited tailings that entered the river until 1968 (Hoiland et al. 1994, NRC 2005). At least as late as 2000, metal (cadmium, lead, and zinc) concentrations were elevated in caddisflies and were more highly correlated with sediment concentrations than with surface water concentrations, suggesting that deposited tailings were the primary source of exposure (Maret et al. 2003).

A new study has modeled future decline in sediment metals concentrations for the Clark Fork River (Box 9-5), assuming an exponential decay in concentrations over time due to loss and dilution (Moore and Langner 2012). Although there was no significant decline over time (1991 to 2009) in downstream concentrations (which one would expect as tailings wash downstream), concentrations did decline over time at three individual sites. Based on regression for each of those sites, Moore and Langner (2012) estimated that average copper concentrations would decline below the probable effect concentration (PEC) in less than 85 years. At the most contaminated of the three sites, copper was predicted to reach the threshold effect concentration (TEC) in 163 years. In the Bristol Bay watershed, dilution with clean sediment would likely be slowed by denser vegetation and less land disturbance. Lower gradients in the Bristol Bay receiving streams relative to Montana would also tend to slow recovery, as recovery is primarily achieved by tailings transport downstream. It also should be noted that these time estimates are not from the date of a spill, but rather from a date decades later, when channel structure had stabilized and much of the tailings had been carried downstream.

9.5.2.2 Exposure-Response

Aqueous Chemicals

The toxic effects of exposure to a tailings spill can be estimated from aquatic toxicity data. Ambient water quality criteria and equivalent benchmarks are used to screen the metals in the two types of tailings leachates (Tables 8-4 and 8-5). Copper is the dominant contaminant in tailings leachates, and criteria values based on the biotic ligand model (BLM) (described in Section 8.2.2.1) are used as benchmarks (Table 9-8). Acutely lethal levels for rainbow trout exposed to the humidity cell leachate and supernatant are estimated to be 93 and 188 μg/L, respectively, based on the BLM.

Table 9-8. Results of applying the biotic ligand model to mean water chemistries of tailings leachates and supernatants to derive effluent-specific copper criteria.

Stream	Acute Criterion (CMC in µg/L)	Chronic Criterion (CCC in µg/L)
Tailings humidity cell leachates	4.8	3.0
Tailings supernatants	7.2	4.5

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

Note that these criteria are calculated for the water chemistry of the tailings supernatant and leachate. This is clearly appropriate for the acute exposures immediately following a tailings dam failure, when the slurry volume would greatly exceed natural flows. It would also be appropriate for situations like sediment pore water, where dilution is minimal. However, these criteria would be too high for situations in which significant dilution occurs, because uncontaminated water has lower hardness and lower levels of chemicals that reduce copper binding to biotic ligands. Hence, dilution by a factor of 2 would not quite reduce toxicity of metal-contaminated water by a factor of 2.

Sediment Chemicals

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

Dietary Chemicals

Effects may also be estimated from dietary exposures. If the primary source of exposure is dissolved copper in the water column (e.g., if significant upstream and floodplain leaching occurs), then the 0.95 adjustment factor (Section 8.2.2.1) is applicable. However, if sediment is the primary source of exposure, a dietary value is needed for consumption of benthic invertebrates. A dietary chronic value for rainbow trout derived from multiple studies is 646 µg/g (micrograms of copper per gram of dry diet) (Borgmann et al. 2005), at which concentration survival and growth are observed to decline in multiple studies.

Analogous Sites

The effects of exposure to leachate from tailings can also be estimated from effects at analogous sites. In the Clark Fork River, Coeur d'Alene River, and Soda Butte Creek (Box 9-5), both toxicity and observed field effects on fish and invertebrates have been associated with deposited tailings. However, the magnitude and nature of effects are so site-specific that quantitative empirical exposure-response models from these sites cannot reasonably be applied to the tailings dam failures analyzed here. Nevertheless, the qualitative relationships are applicable.

9.5.2.3 Risk Characterization

Toxicological risks are usually judged by comparing exposure levels to a criterion or other ecotoxicological benchmark using a risk quotient (Box 8-3), which equals the exposure level divided by the ecotoxicological benchmark. If the quotient exceeds 1, the effect implied by the benchmark is expected to occur, but with some uncertainty (Box 8-3). Quotients much larger than 1 suggest larger effects than those that define the benchmark, with greater confidence that an adverse effect would occur. Quotients much smaller than 1 suggest that even small effects are unlikely. The criterion maximum concentration (CMC or acute criterion) and criterion continuous concentration (CCC or chronic criterion), and equivalent numbers when criteria are not available, are the primary ecotoxicological benchmarks used in this assessment for aqueous exposures, because they are relatively well-accepted as thresholds for significant effects. The CMC estimates a concentration at which 5% of aquatic species experience some mortality among developed life stages in short-term exposures. The CCC estimates a concentration at which 5% of aquatic species experience decreased survival, growth, or reproduction in longer-term exposures. Other benchmarks are used to indicate direct toxicity to salmonids (Tables 8-13 and 8-14).

Acute Toxic Risks

At sites closest to a failed TSF, acutely toxic effects of a tailings spill would be indistinguishable from the concurrent effects of being smothered by tailings particles. Aquatic life in the range of the tailings slurry would be devastated by its physical effects. Dissolved components of the spill would continue to flow to Bristol Bay, beyond the extent of significant particle deposition. Undiluted leachates of both types would be expected to exceed the acute national criterion for copper (Tables 8-4 and 8-5), which suggests that they would kill invertebrates. However, even the minimal dilution in the Nushagak River at Ekwok would dilute leachate from the maximum spill to the national criterion or below. Even copper in undiluted tailings leachates (5.3 and 7.8 μg/L for the humidity cell and supernatant, respectively) would be well below levels required to kill post-larval trout in acute exposures (93 and 188 μg/L for the humidity cell and supernatant, respectively). Hence, in a tailings dam failure, acute exposure to dissolved copper immediately downstream of the TSF would be sufficient to kill sensitive invertebrates but not salmonids, but those effects would be eclipsed by the physical effects. Downstream, where physical effects would be minimal, toxic effects would be reduced or eliminated by dilution.

Chronic Toxic Risks for Aqueous Exposure

Potential effects of chemicals leaching from tailings in streambed and riverbed sediments and associated floodplains are addressed by dividing leachate concentrations by chronic water quality criteria and standards to derive hazard quotients (exposure concentrations divided by effects concentrations). These hazard quotients can be interpreted as relative degrees of toxicity of leachate constituents or as indicators of the degree of dilution required to avoid significant toxic effects. The two estimates of tailings leachate composition give similar results (Tables 8-4 and 8-5). Undiluted leachate of both types would be expected to exceed the chronic national criterion, but not the Alaskan standard, for copper. If combined toxic effects of metals are considered (see the *Sum of Metals* line in Tables 8-4 and 8-5), chronic toxicity would be expected with both the hardness-based and BLM-based copper criteria, and acute lethality would be expected with the BLM-based copper criterion. However, direct aqueous exposures of fish to copper are unlikely to be toxic unless concentrations in the actual field leachates are much higher than the tailings test leachate concentrations.

The quotients with respect to chronic criteria (CCC) imply that dilution by a factor of 2 to 4 would be sufficient to render leachate nontoxic. Low dilutions would be expected in the years immediately after a spill, when flows would pass through large volumes of tailings. After tailings have eroded and a more normal channel and floodplain are established, low dilution of tailings could occur in sediments during normal flows and in locations where water contaminated by floodplain tailings feeds a stream. In those situations, sensitive invertebrates could be reduced or eliminated.

Chronic Toxic Risks from Sediment Chemicals

Sediment quality guidelines provide another line of evidence to assess risks from tailings after a tailings dam failure. Table 9-7 shows that tailings would be expected to cause severe toxic effects on the organisms that live in or on them. Notably, copper concentration would be approximately 4.5 times the PEC; chromium and nickel concentrations would also exceed their PECs. The sum of TEC quotients of 32 implies that tailings would need to be diluted by 32 parts clean sediment to one part tailings before toxic effects would be unlikely (below the TEC). Because the Bristol Bay watershed is relatively undisturbed, background levels of total suspended solids are low (Table 8-10) and the time required to achieve that degree of dilution would be very long.

Chronic Toxic Risks from Dietary Chemicals

The most relevant estimate of fish dietary exposure to tailings is provided by bioaccumulation factors with respect to sediment. The best estimate bioaccumulation factor of 1 implies copper concentrations in invertebrates of 683 mg/kg. Dividing this concentration by a consensus dietary chronic value for rainbow trout of 646 μg/g (micrograms of copper per gram of dry diet) (Borgmann et al. 2005) results in a quotient of 1.1. This implies that the undiluted tailings would produce toxic prey for fish, but the result is marginal and certainly within the range of uncertainty.

Chronic Toxic Risks—Analogous Sites

Some well-documented cases indicate that adverse effects of chronic toxicity on aquatic communities in general, and on salmonids in particular, can occur in streams and rivers that receive tailings spills. These cases have shown that effects continue indefinitely, but that the nature and magnitude of those effects vary among sites. In every case in which the ecological consequences of a major metal mine tailings spill to a stream or river were studied, extensive and long-lasting toxic effects were observed. However, these tailings are likely to be more toxic than future tailings due to more efficient metal extraction in modern ore processing.

The most relevant case appears to be Soda Butte Creek in Montana, where a tailings spill from a porphyry gold and copper mine occurred in 1950 (Box 9-5). In the Soda Butte Creek case, the copper content of macroinvertebrates was positively correlated ($r^2 = 0.80$) and their taxa richness was inversely correlated ($r^2 = 0.48$) with sediment copper (Marcus et al. 2001). Although copper concentrations generally decreased downstream, sediments and sediment pore waters were toxic to the amphipod *Hyalella azteca* for the full 28-km length of the study area (Nimmo et al. 1998). Macroinvertebrate community effects persisted for at least 40 years after the spill. These effects were attributed to sediment toxicity (Nimmo et al. 1998), but habitat effects of deposited tailings also may have contributed. Although they were less well studied, it was clear that trout were also affected: only two trout were found in the 300-m reach downstream of the spill site in 1993, although prior to mining Soda Butte Creek was known for “fast fishing and large trout” (Nimmo et al. 1998).

In the Coeur d’Alene River and its tributaries, elevated metals concentrations and effects on both benthic invertebrates and fish persisted for more than 30 years after tailings releases ended and after treatment of mine drainage. Some fish species were absent; others were reduced in abundance and experienced toxic effects from both aqueous and dietary exposures (Farag et al. 1999, Maret and MacCoy 2002, Maret et al. 2003). Returning Chinook salmon avoided the more contaminated South Fork in favor of the North Fork (Goldstein et al. 1999), and macroinvertebrate communities and taxa were also impaired (Hoiland et al. 1994, Maret et al. 2003).

In the Clark Fork River, a sediment quality triad approach demonstrated that tailings-containing sediments had high metal levels, were toxic to the amphipod *Hyalella azteca*, and shifted the macroinvertebrate community to generally metal-tolerant Oligochaeta (worms) and Chironomidae (midges) (Canfield et al. 1994). Rainbow and brown trout abundances were low in contaminated reaches of the Clark Fork, fish kills occurred apparently due to metals washing from floodplain tailings deposits, and metals in invertebrates were sufficient to cause toxic effects in laboratory tests of trout (Kemble et al. 1994, Pascoe et al. 1994, ARCO 1998).

9.5.2.4 Uncertainties

All of the lines of evidence concerning risks to aquatic communities from the toxic properties of spilled tailings have notable uncertainties. In particular, the estimates from test leachates and whole test tailings underestimate risks because they do not include pyritic tailings.

Toxic Risks from Aqueous Exposures

The use of leachate and supernatant concentrations to estimate effects of a tailings spill is uncertain primarily because of issues concerning test relevance to leaching in the field. Leaching of tailings in the impoundment, streambeds, and floodplains would occur under very different conditions than in humidity cell tests. In addition, it is possible that tailings could become more acidic over time, as their acid-neutralizing capacity is consumed or as acid-neutralizing chemicals are dissolved, resulting in increased metal concentrations. Test leachates are available for bulk tailings but not pyritic tailings. Finally, the degree of leachate dilution in the field would be highly variable and could be roughly estimated, at best. These uncertainties concerning exposure are significant in terms of both their potential size (at least an order of magnitude uncertainty) and in terms of their implications (leachates from the spilled tailings may be non-toxic or severely toxic given this uncertainty in exposure).

The exposure-response relationships for this line of evidence are also uncertain. As noted in Section 8.2.2, the water quality criteria and standards used in this assessment may not be protective of all macroinvertebrate taxa that are important prey for fish. However, direct aqueous exposures of fish to copper are unlikely to be toxic unless field concentrations are much higher than test leachate and supernatant concentrations, so fish toxicity is not an important uncertainty. The uncertainty concerning exposure-response relationships is smaller than the uncertainty concerning exposure.

Toxic Risks from Sediment

Although the consensus TECs and PECs are the best available effects benchmarks for sediment, their applicability to tailings in Bristol Bay habitats is uncertain. The studies from which the values are derived include lakes, reservoirs, and other systems that differ from rivers and streams in the Bristol Bay watershed. However, the Clark Fork River (a tailings-contaminated salmonid stream) was one of the confirmation sites for the TECs and PECs, which suggests that they are relevant to this type of situation.

Because the TECs and PECs are geometric means of prior sediment guidelines, the range of guidelines provides an estimate of uncertainty. Alternate threshold values for copper range from 16 to 70 mg/kg and probable effect values range from 86 to 390 mg/kg (MacDonald et al. 2000). The average copper concentration of tailings (680 mg/kg) is well above all of these values, so this uncertainty is immaterial.

Some evidence suggests that these sediment guidelines may not be fully protective. When quotients of sediment concentrations/TELs (one of the sources of the TECs and a numerically similar value) were summed to address the combined toxicity of cadmium, copper, lead, and zinc, that value was not a threshold for effects on stream invertebrates in the Colorado mining belt, and reductions in four different community metrics occurred below the sum of TEL values (Griffith et al. 2004). This result suggests that toxicity would be even more severe than the TECs and PECs suggest, but it may be somewhat confounded by mine drainage.

Dietary Risks

Effects of dietary exposures would depend on the tailings composition, the copper bioaccumulation factor for aquatic invertebrates, and the chronic toxic threshold for dietary exposures of rainbow trout. Tailings composition may differ in practice from the PLP (2011) tests, but that uncertainty is unknown. Ecological uncertainties are likely to be larger. Bioaccumulation factors for invertebrates and sediment range from 0.15 to 1.77, even in a single river, which translates to invertebrate body burdens of 102 to 1,210 μg/g. That range encompasses the seven available estimates for the copper toxic dietary threshold in rainbow trout, which range from 458 to 895 μg/g (Borgmann et al. 2005). This range of bioaccumulation factors is not surprising given the differences in feeding habits, morphology, and physiology among invertebrates. Finally, the variation in results among dietary toxicity studies for copper is large, and the factors controlling dietary toxicity are poorly known.

Analogous Sites

The analogous sites for a potential tailings spill are all salmonid streams or rivers that received large deposits of tailings from metal mines and that were well studied over an extended period (Box 9-5). A large source of uncertainty when evaluating effects at those sites is the composition of the tailings. In general, the Pebble test tailings are less acidic and contain less copper. On that basis, the nature and magnitude of effects are likely to be less. However, the setting is different in ways that might increase effects. For example, low hardness and low levels of dissolved materials in the mainstem Koktuli River receiving waters would make biota of the receiving streams more susceptible to metals than at the analogous sites. Although these cases are highly uncertain sources of information concerning the potential toxicity of spilled tailings, they can be used with confidence to identify or confirm important modes of exposure and the processes leading to exposure. They also demonstrate the persistence of tailings and the leaching of their metals for multiple decades.

9.5.3 Weighing Lines of Evidence

This risk assessment is based on weighing multiple lines of evidence, and evidence for the various routes of exposure (summarized in Table 9-9) is complex. For each route, sources of the exposure estimate and the exposure-response relationship are indicated. All evidence is qualitatively weighed based on three attributes: its logical implication, its strength, and its quality (Suter and Cormier 2011). Evidence scored as positive (+) supports the case for adverse effects, whereas evidence scored as negative (–) weakens the case for adverse effects. A zero (0) score indicates no or ambiguous evidence.

- The **logical implication** is the same for all lines of evidence: they all suggest that a spill from a tailings dam failure would have adverse effects.
- The **strength** of the evidence is based primarily on the magnitudes of the hazard quotients. For example, if the predicted concentration of copper is twice the median lethal concentration (LC_{50}) for rainbow trout, that is evidence of acute lethality; if it is 10 times as high, that is stronger evidence. In Table 9-9, zero signifies a low quotient, + a moderate quotient, and ++ a high quotient.

- **Quality** is a complex concept that includes conventional data quality issues, but in this case, the primary determinant is the relevance of the evidence to the mine scenarios. Because this is a predictive assessment, none of the evidence is based on observations of an actual spill at the site of concern. Hence, the evidence is based on the tailings dam failure scenarios, laboratory studies, or field studies at other sites where tailings have spilled into streams or rivers or where biota were exposed to other sediments with high copper levels. Separate quality scores are provided for the exposure estimate and for the exposure-response relationship.

The scores indicated in Table 9-9 are not a substitute for the actual evidence, but rather are intended to remind the reader what evidence is available and summarize the strength and quality of the different lines of evidence.

Table 9-9. Summary of evidence concerning risks to fish from the toxic effects of a tailings dam failure. The risk characterization is based on weighing multiple lines of evidence for different routes of exposure. All evidence is qualitatively weighed (using one or more +, 0, or - symbols) on three attributes: logical implication, strength, and quality. Here, all lines of evidence have the same logical implication, since all suggest a tailings dam failure would have adverse effects. Strength refers to the overall strength of the line of evidence, and quality refers to the quality of the evidence sources in terms of data quality and relevance of evidence to the mine scenarios.

Route of Exposure Sources of Evidence (Exposure/E-R)	Logical Implication	Strength	Quality		Results
			Exposure	E-R	
Suspended sediment Assumption/synthesis of laboratory and field studies	+	++	0	+	Adverse effects on fish are certain. Although the exposure level is unknown, it would clearly be at effective levels.
Acute aqueous exposure Leachate measurements/laboratory-based criteria	+	0	0	+	Acute lethality to invertebrates close to tailings storage facilities, but not downstream and not to fish.
Chronic aqueous exposure Leachate measurements/laboratory-based criteria	+	0	0	+	Chronic toxic effects on invertebrates due to *in situ* leachate, but effects would end after some years if diluted by clean sediment.
Chronic sediment exposure Tailings measurements/sediment guidelines	+	++	+	0	High likelihood of toxic effects on invertebrates or fish based on a summary of field studies.
Chronic dietary exposure Tailings measurements and bioaccumulation factors/mean of laboratory-based effects levels	+	0	+	0	Marginal dietary copper toxicity to trout would be eliminated by minimal dilution.
All routes in the field Exposure and effects at analogous sites	+	++	+	0	Tailings spilled to streams have persisted and caused severe effects, but the toxicity of the tailings is likely to be higher in those cases.
Summary weight of evidence	+	+	0	0	All lines of evidence are consistent with toxic effects of tailings. Despite the ambiguous quality and marginal strength of some lines of evidence, the overall strength is positive.

Notes:
E-R = exposure-response relationship.

9.6 Summary of Risks

9.6.1 Tailings Spill

Following a tailings spill, fish in the receiving stream and the invertebrates on which they depend would be exposed to deposited tailings, suspended tailings, and tailings leachates. The fine texture of deposited tailings would make them unsuitable for salmonid spawning and development, and a poor substrate for the invertebrates that serve as food for developing salmon and resident trout and char. Suspended tailings would have lethal and sublethal physical effects on fish and invertebrates immediately following the spill, which are likely to continue with gradually diminishing intensity for years thereafter. The most toxic constituent of the leachate and tailings would be copper, and exposures would be both direct and through diet. Copper in leachate and in food is mildly toxic for fish, but copper and other constituents in the tailings themselves would be moderately toxic to benthic invertebrates and potentially toxic to fish eggs and larvae spawned in tailings-contaminated streams.

The physical and chemical effects of tailings on fish and invertebrates would be extensive in both space and time. Elevated levels of suspended tailings would last for years. Deposited tailings and their leachate would persist at toxic levels for decades. The acute effects of a tailings spill would extend far beyond the modeled 30-km distance downstream. Based on data from other sites, tailings deposition from a spill would extend for more than 100 km downstream, resulting in chronic exposures and effects. In addition, the flow velocity of the receiving rivers, particularly in the spring, would readily transport the fine tailings particles farther downstream. The mouth of the Koktuli River is 63.6 km from the confluence of the South and North Fork Koktuli Rivers. From there, the mouth of the Mulchatna River is another 66.5 km, and the mouth of the Nushagak River at Dillingham is another 170.5 km, so contamination of the entire downstream system would be likely soon after a spill.

We did not explicitly model failures of TSF 2 or TSF 3. The types of risks and effects that would occur with a dam failure at TSF 2 or TSF 3 would be generally similar to those described for a failure at TSF 1. The content and toxicity of TSF 2 and TSF 3 are assumed to be similar to that of TSF 1, and the magnitude and extent of risks would be largely dependent on the volume of material released. One important distinction, however, should be noted. The South Fork Koktuli River and Upper Talarik Creek are hydrologically connected via groundwater transfer. In the event of a dam failure at TSF 3, transfer of contaminated water leached from tailings fines and deposited in the South Fork Koktuli River valley to the Upper Talarik Creek watershed and Iliamna Lake would be expected.

9.6.2 Remediation of a Tailings Spill

Although streams typically recover from aqueous effluents in less than a decade, the effects of tailings deposition in streams and floodplains persist for as long as they have been monitored at analogous sites. For that reason, many tailings-contaminated aquatic habitats in the United States have been or will be dredged, riprapped, or redirected under the federal Superfund or state cleanup programs. The tailings dam failure scenarios evaluated here do not consider any mitigating effects of remediation efforts by the mine operator or other parties. Although such remedial actions have net benefits, they create long-term impacts on aquatic habitats. For example, riprapping reduces downstream exposure to tailings and associated metals by reducing erosion of floodplain tailings, but it also reduces fish habitat complexity and quality for fish by channelizing the stream or river (Schmetterling et al. 2001).

Remediation under the tailings dam failure scenarios considered here would be particularly difficult and damaging, because the area of the spill is almost entirely unaffected by other development. One or more roads would need to be built into this roadless area to bring in equipment and haul out the tailings. At the upper end of the affected area, the process of removing tailings would do little additional damage, since the structure of the watershed would have been destroyed. If tailings removal extended to streams that were not scoured in the initial tailings release, removal would destroy those streams and associated wetlands. If removal was not undertaken, the substrate of the streams would consist of tailings until high flows carried them downstream. It may be technically impractical to recover tailings fines that have been transported past the point of confluence with larger rivers.

In the Pebble 0.25 tailings dam failure scenario considered here, an estimated 45 million metric tons of tailings solids would be deposited in the North Fork Koktuli River valley (calculated from Table 9-5, assuming a dry density of 1.42 metric tons/m^3). Complete removal of this material would require a substantial earth-moving effort (e.g., including over 3 million round trips by 20-ton dump trucks). Recovery and removal would be even more challenging in the Pebble 2.0 tailings dam failure scenario, in which an estimated 97 million metric tons of tailings solids would be deposited in the North Fork Koktuli River valley, and an additional 263 million metric tons of tailings solids would be transported beyond the confluence with the South Fork Koktuli River and into the mainstem Koktuli River (Table 9-5). Material not deposited on the floodplains would be carried downstream; material deposited in the floodplains, if not recovered, would be remobilized by future precipitation and would wash downstream. It is unlikely that tailings in river channels would be recovered, because the fine material would be rapidly transported by the relatively high flow velocities of the rivers.

Remediation of prior tailings dam failures can serve as case studies. Failures are numerous, but the degree to which remediation results in restoration of natural resources has not been well documented. The 1998 failure of the Aznalcóllar tailings dam at Los Frailes Mine in Seville, Spain (Box 9-1) has been described as a case of substantial remediation. However, this kind of successful removal of tailings would be difficult to replicate in the Bristol Bay watershed. The Aznalcóllar area has a drier and warmer climate, flatter topography, better access from existing roads, and more readily available equipment and labor compared to the Bristol Bay region. The goal of the Aznalcóllar remediation was restoration of land use, which would not be the primary goal in the Bristol Bay watershed. In addition, potential releases from TSF 1 would be much larger than the release at Aznalcóllar (Box 9-1).

Emergency plans for metal mines in Alaska that have been provided to USEPA do not address remediation or restoration after a tailings spill has occurred. In fact, no tailings spill has been reported in Alaska, so it is not clear what remediation or restoration might be required. Given the uncertain toxicity of the tailings, the difficulty and expense of remediation and restoration, and the damage that would be done by remediation, it is possible that a spill would be left to be restored by natural processes.