3.7 WATER QUALITY

SYNOPSIS

This section describes existing and projected surface water, groundwater, and sediment quality in the proposed project area, as well as the geochemistry of proposed mine facilities in relation to effects on these resources. The section discusses applicable laws and water quality criteria; then describes existing conditions in the potentially affected environment for each project component (Mine Site, Transportation Corridor, and Pipeline); followed by expected effects of the proposed action and alternatives.

EXISTING CONDITION SUMMARY

Mine Site – A study of natural background water quality was conducted in the mine area over a period of about 10 years. Several constituents are present naturally at concentrations in excess of the applicable standards. For example, naturally elevated mercury and arsenic levels are found sporadically in surface and groundwater within and surrounding the proposed Mine Site, which is common in mineralized areas. Arsenic and mercury are also both present in sediment samples above Sediment Quality Guidelines (SQGs), especially downgradient of mineralized areas. The more harmful form of mercury, methylmercury, is also present in existing surface water and sediment. By breaking up and exposing previously buried rock to air, water, and microbes, mining would increase natural rates of weathering and chemical dissolution of rocks and minerals, and may potentially release some of their constituents into the surrounding environment, including surface water, groundwater, and sediment resources.

Transportation Corridor – The primary body of water affected by proposed Transportation Corridor facilities would be the Kuskokwim River downstream from Crooked Creek. Water in the river is generally considered fit for all purposes, and several villages between Crooked Creek and Bethel draw drinking water directly from the Kuskokwim. However, there are points along the Kuskokwim, usually at confluences with tributaries that drain mineralized areas, where concentrations of mercury and other minerals are elevated above SQGs and naturally occurring ranges found elsewhere. Data describing baseline groundwater quality within the transportation corridor comes from village drinking water wells, where iron has sometimes exceeded drinking water standards. Sediment sampling along the Kuskokwim River between Crooked Creek and Bethel showed elevated metal levels, including arsenic and mercury, at all sampling sites.

Pipeline – Surface water field measurements in streams along the proposed pipeline indicate several areas of high turbidity from natural levels of glacial silt draining the Alaska Range. Groundwater quality in water supply wells near the east end of the pipeline route contains elevated aluminum, iron, and manganese levels above applicable criteria.

EXPECTED EFFECTS SUMMARY

Alternative 1 - No Action

Under the No Action Alternative, construction, operations, and closure activities associated with the proposed project would not take place and the required water quality-related permits would not be issued. Consequently, there would be no new effects, neither adverse nor beneficial, on these resources from implementation of the No Action Alternative.

Alternative 2 - Donlin Gold's Proposed Action

Mine Site – Impacts to water quality at the Mine Site could result from geochemical alteration of mined rock and its interaction with air and water, as well as mercury deposition from stacks and fugitive dust. Surface and groundwater quality within the Mine Site footprint in American and Anaconda creek watersheds would be irreversibly affected by the creation of the Waste Rock Facility (WRF), Tailings Storage Facility (TSF), and pit lake; however, due to perpetual water management and treatment, untreated water from these facilities would not leave the onsite watersheds. Effects from Mine Site waters on the environment outside the immediate mine area would meet water quality standards prior to discharge to Crooked Creek, as all contact water would be treated. Contaminated water from the pit would drain outward into localized bedrock for several years as the pit initially fills; this effect would be infrequent and limited to discrete portions of the project area, as the overall groundwater gradient would be towards the pit, maintaining hydraulic containment of affected groundwater. There is a low risk that impacts to Anaconda and Crooked creeks would be sufficient to exceed water quality standards (WQSs) in the event of SRS pump failure and overflow in post-Closure. There is also a possibility of contaminated groundwater migration from the South Overburden Stockpile towards Crooked Creek. Mitigation measures are described that would help reduce this potential impact.

Impacts to surface water quality resulting from atmospheric deposition of mercury would vary in intensity. Effects are likely to be within regulatory limits on average, but could vary above baseline conditions and EPA chronic criteria in certain tributary watersheds along Crooked Creek. Impacts to sediment quality in Crooked Creek, and increases in mercury and methylmercury concentrations in sediments, would be within the range of natural variation, and would be expected to decline in post-Closure. Impacts to sediment quality from surface disturbances would be limited to discrete portions of the project area by containment from BMPs.

Transportation Corridor and Pipeline – Barging in shallow sections of the Kuskokwim River may have effects on sediment and turbidity that would be intermittent and not last longer than the span of the project. Surface water quality could be temporarily and locally affected during pipeline Construction at stream crossings, but would be mitigated by BMPs, horizontal directional drilling (HDD) crossings of selected waterways, and winter trenching at other crossings.

OTHER ALTERNATIVES - This section discusses differences of note between Alternative 2 and the following alternatives, but does not include a comprehensive discussion of each alternative's impacts if they are the same as or similar to Alternative 2 impacts.

Alternative 3A would reduce fuel barging requirements, decreasing total barging on the Kuskokwim by 32 percent. Surface water effects such as increases in turbidity from barging would be reduced under Alternative 3A relative to Alternative 2.

Alternative 3B - Diesel Pipeline

Alternative 3B would eliminate fuel barging after the Construction Phase, reducing the risk of fuel spill impacts on the Kuskokwim River. However, this alternative would expose groundwater and surface water along the proposed pipeline corridor to the risk of spills and leaks from the diesel pipeline. There would also be an increased number of streams along the Tyonek and Port MacKenzie Option routes under this alternative that would be exposed to pipeline construction effects.

Alternative 4 - Birch Tree Crossing (BTC) Port

Alternative 4 would reduce the distance traveled by barge by 75 river miles, and would require a longer access road. Surface water quality impacts would be slightly increased due to stream crossings and runoff along the longer access road, while sediment impacts and spill risk exposure would be slightly decreased in the Kuskokwim River.

Alternative 5A - Dry Stack Tailings

Alternative 5A would exchange a dry stack and operating pond design for the wet tailings design of the Tailings Storage Facility under Alternative 2, and includes both unlined and lined drystack options, the latter having a pumped overdrain beneath the tailings. Where the TSF under Alternative 2 would be lined beneath the facility, dry stack tailings would be covered with a geomembrane liner at closure, but would be expected to leach contact water into underdrains (whether or not a bottom liner is used) in quantities that eventually would be similar to Alternative 2. This water would in turn be captured by a seepage recovery system (SRS) and pumped to the pit lake, where water would be treated before release. The main difference between the Mine Site alternatives and options with respect to surface water quality is the time it takes for the SRS water to meet State of Alaska ambient water quality criteria, allowing the SRS to be decommissioned. Those criteria will be met roughly 200 years after closure under the Unlined Option, and about 10 to 50 years after closure under the Lined Option and Alternative 2. Under either option, effects on downgradient water quality in Crooked Creek would be the same as Alternative 2, as the SRS water would be contained and conveyed to the open pit. The pit lake would fill in 41 to 46 years following closure (depending on whether the dry stack is lined) instead of 52 under Alternative 2, and more water would need to be treated on an ongoing basis prior to release. The SRS would potentially need to be monitored and maintained longer under the Unlined Option, until it was demonstrated to be suitable for release to the environment, and a failure of the system could lead to release of untreated water in a matter of weeks. Fugitive dust deposition from the dry stack would be slightly greater under this alternative, and like Alternative 2, could enter surface water within and beyond the footprint of the proposed mine during Operations.

Alternative 6A - Dalzell Gorge Route

Alternative 6A would change the alignment of the natural gas pipeline. The primary mechanisms of impact to water and sediment quality resulting from Construction Phase activities would be the same as Alternative 2. Specific locations of impacts would differ along the Dalzell Gorge segment; however, construction practices, stream crossing techniques, and BMPs would be the same.

3.7.1 APPLICABLE REGULATIONS

3.7.1.1 REGULATORY FRAMEWORK

The principal law governing surface water quality in the United States is the Federal Water Pollution Control Act, or Clean Water Act (CWA). The CWA establishes regulatory requirements that apply to industrial and municipal discharges, and includes a range of requirements for maintaining water quality. It provides EPA, the Corps, and the states with a variety of programs and tools to protect and restore the nation’s waters. These programs and tools generally rely either on water quality-based controls, such as water quality standards and water quality-based permit limitations, or technology-based controls such as effluent guidelines and technology-based permit limitations. Newly constructed facilities (new sources) that discharge to surface waters either directly or indirectly are governed by New Source Performance Standards and pretreatment standards for new sources. In developing New Source Performance Standards, the CWA requires that EPA determine the "best available demonstrated control technology" for the particular industrial category. The provisions described at 40 CFR Part 440, Ore Mining and Dressing Point Source Category, Subpart J, Copper, Lead, Zinc, Gold, Silver, and Molybdenum Ores Subcategory, would apply to discharge of mine drainage and net precipitation associated with the Donlin Gold Project, as long as certain requirements are met.

The CWA establishes a framework for federal-state partnerships in which the federal government sets the agenda and standards for pollution abatement, while states are generally responsible for implementation and enforcement (Copeland 2010). In particular, the CWA requires that a state must have the necessary legal authority to administer the National Pollutant Discharge Elimination System (NPDES) Program before EPA will approve a state's NPDES Program application. On May 1, 2008, the State of Alaska submitted a final application to the EPA for authority to administer the NPDES Program for wastewater discharges in Alaska, and on October 31, 2008, EPA approved the application. The ADEC authority over mining APDES permits began on October 31, 2010. ADEC assumed full authority to administer the wastewater and discharge permitting and compliance program for Alaska on October 31, 2012. The resulting program is called the Alaska Pollutant Discharge Elimination System (APDES) Program, which regulates point source discharges under Section 402 of the CWA.

The CWA contains requirements that water quality standards (WQS) must be established, and that water quality must be monitored to determine compliance relative to the established standards. ADEC is responsible for setting and regulating WQS for the State of Alaska. The EPA ensures that all WQS set by states are at least as stringent as the federal standards. Standards modified by the state are not effective for CWA purposes unless they are approved by EPA, and there is often a lag time between when a standard is proposed by a state and when it can be used for CWA purposes. If there is a discrepancy, only EPA-approved standards apply for CWA purposes.

The federal Safe Drinking Water Act (SDWA) requires actions to protect drinking water in rivers, lakes, and groundwater. Under the SDWA, EPA sets standards for drinking water quality, and implements various technical and financial programs to ensure drinking water safety (EPA 2017c). Alaska has primacy to implement SDWA in the state. The ADEC Drinking Water Program works with public water system operators to help them remain in compliance with state and federal drinking water regulations by reviewing public water system plans, providing oversight of contaminant testing, performing inspections, and conducting source water assessments to identify where public water supplies may be vulnerable to contamination (ADEC 2017d). The ADEC Drinking Water Program also provides oversight of well design and guidance for decommissioning water wells, in order to protect groundwater resources from surface contamination, and prevent wells and boreholes from acting as conduits for future contamination (ADEC 2017e).

ADEC source water assessments include mapping of the watershed or groundwater zone contributing to a public water system, inventory of potential and existing sources of contamination in the delineated protection area, and a vulnerability analysis to decide how likely a water supply may be to become contaminated. Communities can then use the information to manage identified contaminant sources, and prevent or minimize new threats in the source water assessment area. Source water protection areas for groundwater are provided for several zones upgradient of the public water supply well based on estimated groundwater migration time through the aquifer (ADEC 2017c).

ADEC Division of Water, Wastewater Discharge Authorization Program (WDAP) adopted General Permit AKG320000 – Statewide Pipelines (Statewide Pipeline GP) on June 14, 2017, which will authorize discharges associated with the gas pipeline construction and operations. The following discharges are proposed to be included in the Statewide Pipeline GP: Domestic Wastewater, Gravel Pit Dewatering, Excavation Dewatering, Hydrostatic Test Water, Fire Test Water, Secondary Containment, Mobile Spill Response, Horizontal Directional Drilling, and Storm Water.

The Statewide Pipeline GP will preclude the need for multiple Alaska Pollutant Discharge Elimination System permits which may include: AKG002000 – Excavation Dewatering, AKG003000 – Hydrostatic and Aquifer Pump Testing, AKR06000 – Multisector General Permit, AKR100000 – Construction General Permit, and AKG572000 – Secondary Treatment to Surface Water.

The Corps also has a role. Under Sections 301 and 502 of the CWA, any discharge of dredged or fill materials into waters of the U.S., including wetlands, is forbidden unless authorized by a permit issued by the Corps pursuant to Section 404. Essentially, all discharges of fill or dredged material affecting the bottom elevation of a jurisdictional water of the U.S. require a permit from the Corps. These permits play an essential role in protecting streams and wetlands. ADEC issues a Certificate of Reasonable Assurance under Section 401 of the Clean Water Act, and the Corps' 404 permit is not valid until ADEC issues a 401 Certification or waives its authority.

3.7.1.2 WATER QUALITY CRITERIA

3.7.1.2.1 SURFACE WATER

On February 27, 2004, EPA partially approved revisions to Alaska Water Quality Standards (WQS), while taking no action on Alaska's proposed acute and chronic freshwater aquatic life criteria for mercury and selenium. Therefore, the new aquatic life criteria for mercury and selenium will not be in effect for CWA purposes until a decision is made by EPA about whether these criteria can be approved. In the interim, the previously approved aquatic life criteria for mercury (2.4 µg/l acute and 0.012 µg/l chronic, both as total recoverable) and selenium (20 µg/l acute and 5 µg/l chronic, both as total recoverable) will remain the applicable CWA standards (65 FR 24643).

In instances where the State of Alaska WQS have been approved by EPA, water quality in this section is described in relation to Alaska WQS, which include use classifications, numeric and narrative water quality criteria, and an anti-degradation policy. The usage classification system designates the beneficial uses that each water body within the State of Alaska is expected to support. In the state of Alaska, all water bodies are designated for all protected water use classes unless otherwise stated (18 AAC 70.050). The water quality data presented in this section are compared to the most stringent applicable State of Alaska water quality standards (for all designated water uses).

For most parameters, the most stringent criteria are the aquatic life criteria for fresh water. However, the most stringent criteria for antimony, arsenic, nitrate, and sulfate are for drinking water (ADEC 2008a; ADEC 2012d). The most stringent criterion for manganese is based on human health for consumption of drinking water and aquatic organisms (ADEC 2008a). The most stringent criteria for boron and cobalt are based on irrigation use (ADEC 2008a).

Criteria for some dissolved metals, including cadmium, chromium, copper, lead, nickel, silver, and zinc, are hardness-dependent, meaning that the acceptable concentrations of these metals depend on the hardness of the water. Hardness is a measure of the concentration of polyvalent cations, such as calcium (Ca^{2+}) and magnesium (Mg^{2+}), in the water. The polyvalent cations that contribute to water hardness reduce the bioavailability of certain trace metals by competing with the trace metal ions for binding sites within organisms. The extent of this effect varies according to which dissolved metals are present and their oxidation states. To account for the influence of water hardness on the bioavailability and potential toxicity of certain dissolved metals, the numeric water quality criteria for those metals are calculated so that the allowable concentrations of the metals increase in proportion to the hardness of the water (ADEC 2008a). The hardness-dependent water quality criteria applied in this section were calculated according to ADEC (2008a) protocols using the 15th percentile hardness measurement for the population of samples from each category of surface water sites sampled as part of the Donlin Gold Water Quality Characterization Program. For evaluation of discharged water, this calculation would apply hardness values from the receiving water. For comparison to predicted concentrations of constituents in mine water management facilities, such as the TSF and pit lake, the hardness-dependent numeric water quality criteria were calculated based on predicted hardness values for each mine water management facility. Thus the numeric water quality criteria for hardness-dependent parameters vary depending upon the measured (or predicted) hardness value for the matrix water.

The ADEC numeric water quality standard for ammonia depends upon both the temperature and the pH of the matrix water, but not hardness. Because the measured or predicted temperature and pH of the water may vary over a wide range of values depending on time of year, sample location, and other factors, values representative of the proposed treated water discharge location on Crooked Creek were used to derive the ammonia criterion (2.99 mg/L; Donlin Gold 2017d; SRK 2017b) used as a basis for comparison to mine waters considered in this section.

3.7.1.2.2 GROUNDWATER

As specified in 18 AAC 70.050(a)(2), groundwater is protected for all uses in Class (1)(A), including drinking, culinary, and food processing; agriculture, including irrigation and stock watering; aquaculture; and industrial uses. While ADEC does not regulate groundwater directly, they do regulate contaminated sites and underground storage tanks which may affect groundwater quality, and establish water quality standards (WQS) that serve to protect groundwater.

Drinking water from groundwater sources is regulated by 18 AAC 80 (ADEC 2012c) and by EPA (2013k, 2017c). EPA sets standards for approximately 90 contaminants in drinking water. These standards include National Primary Drinking Water Regulations, which set legally enforceable Maximum Contaminant Levels (MCLs) that apply to public water systems. Primary standards protect public health by limiting the levels of contaminants in drinking water. Secondary Drinking Water Standards are unenforceable federal guidelines regarding taste, odor, color and certain other effects of drinking water. EPA MCLs for certain constituents (e.g., aluminum, chloride, iron, manganese, pH, sulfate, TDS, zinc) are Secondary Drinking Water Regulations that set non-mandatory water quality standards. EPA recommends them to the states as reasonable goals, but federal law does not require compliance with them. ADEC has established water quality standards to serve to protect groundwater, but the department does not regulate groundwater directly. Permits under 18 AAC 72 or 18 AAC 83 would be necessary to discharge to groundwater.

3.7.1.2.3 SEDIMENT

There are no regulations established for chemical concentrations in sediment. Sediment Quality Guidelines (SQGs) recommended by ADEC (2013d), as well as other sediment quality criteria used for comparison purposes in project studies are discussed further below.

3.7.1.2.4 IMPAIRED WATER BODIES

There are no water bodies listed as impaired under Section 303(d) of the CWA within the project components. The Kuskokwim River is listed as a Category 5 water body about 30 miles upriver of Crooked Creek at the mouth of Red Devil Creek because abandoned mining facilities contribute antimony, arsenic, and mercury to the water. The designation extends from 100 feet upriver of the confluence of Red Devil Creek and the Kuskokwim River to 900 feet downriver from the confluence. Category 5 refers to pollutant-caused impairments requiring a Total Maximum Daily Load (TMDL) technical analysis that calculates the reduction in pollutants necessary for the impaired water body to again meet WQS.

Dutch Harbor, which is part of the transportation corridor for this project, is listed as a Category 4 impaired water body for petroleum hydrocarbon pollution resulting from industrial uses and urban runoff. Category 4 refers to impaired water bodies that have moved beyond the planning phase of TMDL analysis towards the implementation phase of reducing point and non-point source pollutants.

Known contaminated sites within or near the proposed project footprint that could have a bearing on surface or groundwater quality are discussed in Section 3.2, Soils.

3.7.2 AFFECTED ENVIRONMENT

3.7.2.1 SURFACE WATER QUALITY

3.7.2.1.1 MINE SITE

Background and Setting

The proposed Mine Site is located entirely within the Crooked Creek watershed. All proposed mine-related facilities are located in areas that drain to tributaries of Crooked Creek, including American, Anaconda, Lewis, and Queen creeks, and Omega and Snow gulches, all of which flow west into Crooked Creek (Figure 3.7-1) (BGC Engineering, Inc. [BGC] 2005). Several unnamed first-order gullies that drain directly into Crooked Creek are also located within the Mine Site area. Crooked Creek is the main drainage out of the area, and flows southward for approximately 12 miles before joining the west-flowing Kuskokwim River at the village of Crooked Creek. Collectively, these relatively short, low-flow streams with small catchment areas comprise the existing surface water resources within the Mine Site.

Donlin Creek, located north of the proposed project area, and Crevice Creek, located immediately to the south of Anaconda Creek, are both tributaries of Crooked Creek. Although they are not located within the Mine Site, water quality data collected from Donlin and Crevice creeks provide background information about surface water quality in the areas immediately adjacent to the proposed project area. Additional water quality data have been collected on lower Crooked Creek several miles downstream of the proposed mine, and on two tributaries, Getmuna and Bell creeks, that join lower Crooked Creek in this area.

Baseline surface water conditions at the Mine Site, which are presently influenced by both natural and anthropogenic factors, are described in the following subsections. The predominant factor influencing existing surface water quality conditions in and around the Mine Site is the natural weathering of rocks associated with the mineralized area of interest. Spatial trends in existing surface water quality often correspond to the distribution of highly mineralized zones (Mueller et al. 2003; ARCADIS 2012b).

Surface waters within the Crooked Creek watershed are generally undisturbed and free from anthropogenic influences, with the exception of historic and current placer mining operations. Both upper Crooked Creek and Donlin Creek have been subject to placer mining activities in the past. Currently, an active placer operation mines gold from gravels in the upper reaches of Crooked Creek and its tributaries, including Donlin Creek, Snow Gulch, Queen Creek, and Lewis Creek. The placer mining area lies immediately to the north and upstream of the proposed facilities (SRK 2017b). The lower end of Snow Gulch has been extensively mined and sections of the stream have been rerouted, but the upper portions of Snow Gulch remain essentially undisturbed. The lower end of Queen Gulch has been mined and Queen Creek has been diverted into a ditch that discharges to Crooked Creek near American Creek. The principal water quality impacts resulting from placer mining operations are generally associated with increased turbidity and stream sedimentation (Weber 1986). The duration of such impacts is usually limited to the time period of active placer mining, and the intensity of the impacts generally decreases with distance downstream from mining input (Weber 1986). However, in some sulfide-rich areas, placer mining activities may result in acid drainage and increases in trace element concentrations in surface water (Madison 1981).

Donlin Gold Water Quality Characterization

Sampling and Analytical Program

Acquisition of baseline and background water quality data in the Crooked Creek watershed was initiated in 1996 in order to establish a reliable record of physical parameters and chemical constituents in surface water. Since that time, water quality parameters have been measured on an approximately quarterly basis at established stations in the Crooked Creek watershed, with the exception of 2001, when monitoring was curtailed for one year. Collectively, these data comprise the most extensive dataset describing baseline surface water quality within the proposed project area. The initial water quality data collection program included 14 water quality monitoring stations that were selected to characterize the upper portions of the Mine Site. As the water quality characterization program progressed, modifications were made to include additional analytes and monitoring stations (ARCADIS 2012b).

Although the Donlin Gold water quality characterization program data collected prior to 2005 provide some site-specific information, they were not collected according to the consistent procedures and data handling protocols specified in the quality management plan, which was implemented in 2005 and updated in 2012 (Barrick 2009; ARCADIS 2012b). As a result, the pre-2005 data include many outliers and systematic errors in recording (ARCADIS 2012b). Beginning in the first quarter of 2005, water quality sample collection and analysis has been conducted in accordance with a Quality Assurance Project Plan (QAPP) that incorporates Water Quality Monitoring, Sampling and Analysis Activity procedures. Groundwater sampling under the new QAPP began in the first quarter of 2005. Surface water sampling was not conducted during the first quarter of 2005, followed by a limited program in the second quarter, with the full surface water program beginning in the third quarter. For these reasons, the data collected after 2005 are the most appropriate for defining the background quality of surface waters at the Mine Site and adjacent areas. In this section, surface water quality data are summarized for the period from the third quarter of 2005 through the second quarter of 2015.

Since 2005, the Donlin Gold water quality characterization program has involved collection and analysis of samples from most of the sites described in Table 3.7-1 and shown on Figure 3.7-1 in yellow. Three additional monitoring sites (CCBB, BELL, and GETM2) were added to the water quality characterization program in June 2013 (shown on Figure 3.7-1 in purple), in order to establish parameters for baseline water quality in lower Crooked Creek below all proposed facilities, and to characterize baseline inputs of constituents to Crooked Creek from Getmuna and Bell creeks. Both Getmuna and Bell creek stations are located outside of the anticipated area of influence of the Mine Site, but Getmuna is located downstream of a proposed material borrow site along the potential route of the mine access road.

Sample collection methods and details of analyses are described in the Donlin Creek Project Water Quality Monitoring and Data Management Procedures Manual (Barrick 2009). Because the primary factors that affect existing water quality around the Mine Site are related to interactions of water with the mineralized area of interest, and to a lesser extent, the influences of current and historic placer mining activities, the surface water quality sampling sites were divided into three categories based on the location of each site relative to the mineralized area, and the potential for placer mining operations to affect water quality at each site. The three categories are described as follows:

Category 1: waters draining undisturbed areas and areas outside of the mineralized area of interest (background sites).

Category 2: waters draining areas of defined mineralized zones with no placer mining activities (background sites).

Category 3: waters draining areas of both placer mining and the mineralized zone of the proposed Donlin Gold Project (baseline sites).

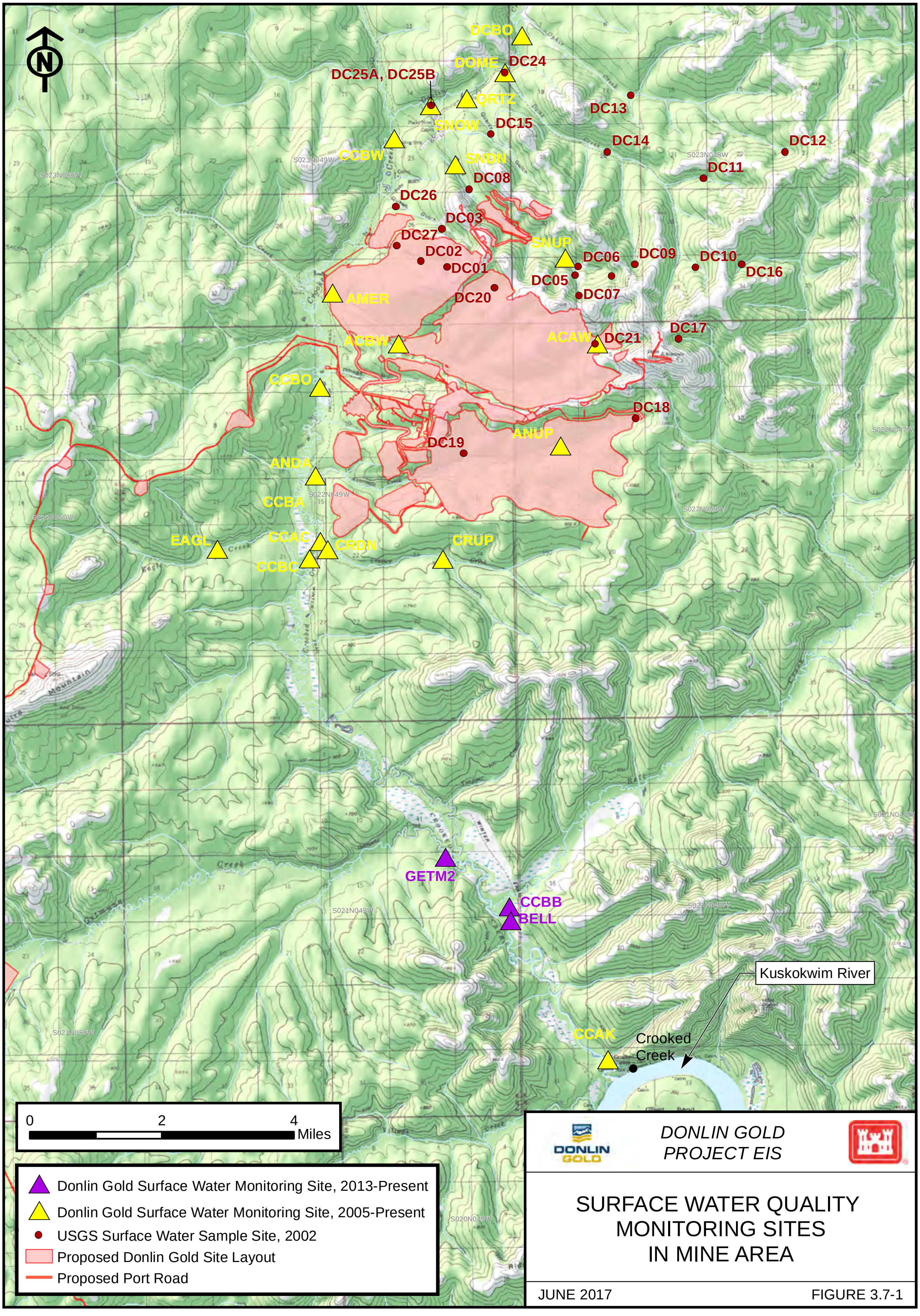
Study Results

Tables 3.7-2 through 3.7-4 summarize the cumulative results of the water quality characterization program for the Category 1, 2, and 3 surface waters. The data in these tables are organized according to the following groups of water quality parameters:

- Major cations (aluminum, barium, calcium, iron, magnesium, manganese, potassium, and sodium), in mg/L;
- Major anions (ammonia, bicarbonate, carbonate, chloride, fluoride, nitrite and nitrate, and sulfate), in mg/L;
- Nutrients (nitrate, nitrite, and ammonia), in mg/L;
- Trace elements (antimony, arsenic, beryllium, boron, cadmium, chromium, cobalt, copper, lead, lithium, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, zinc) and compounds (total and weak-acid dissociable [WAD] cyanide), in µg/L (except for mercury, which is in ng/L);
- General water quality parameters (hardness, total alkalinity, total dissolved solids (TDS), and total suspended solids), in mg/L; and
- Field parameters (conductivity, dissolved oxygen, pH, turbidity, and water temperature), in various units.

Averages

Due to limitations of chemical analysis procedures, small concentrations cannot be precisely measured. These concentrations are said to be below the limit of detection (LOD). In statistical analyses, these values are often censored and substituted with a constant value, such as half the LOD. The arithmetic average values reported in Tables 3.7-2 through 3.7-4 represent the averages of measured concentrations for a given analysis. Results that are below the reporting limit for the analysis are replaced with a value equivalent to one-half of the method detection limit for the purpose of calculating the average values reported in the tables.



DONLIN GOLD
PROJECT EIS

SURFACE WATER QUALITY
MONITORING SITES
IN MINE AREA

JUNE 2017

FIGURE 3.7-1

Table 3.7-1: Description of Surface Water Sampling Locations in Mine Area, Donlin Gold Baseline Water Quality Monitoring Program

Station Description	ID	Rationale and Purpose	Cat.	# of Sample Events	Easting[1] (meters)	Northing[1] (meters)
Donlin Creek below Ophir Creek	DCBO	Upstream of all proposed activity and above any disturbance from historic placer mining. Project Control.	1	40	543948	6885105
Snow Gulch Upstream of activity and mineralization trend	SNUP	Snow Gulch crosses both the mineralized trend and historic placer mining. This site is above both the mineralization trend and placer mining.	1	20	545024	6879725
Snow Gulch downstream of mineralization trend	SNDN	Snow Gulch below the mineralized trend and above historic placer tails.	2	28	542329	6881964
Snow Gulch above confluence with Crooked Creek	SNOW	Snow Gulch below mineralization and historic placer tails and above confluence with Crooked Creek.	3	36	541729	6883397
American Creek above waste rock	ACAW	American Creek upstream of proposed waste rock placement near upstream diversion of water. Also upstream of mineralization in American Creek. Placed to determine quality of diversion water that would be directed to Crooked Creek as non-mine water.	1	26	545787	6877604
American Creek below proposed waste rock storage	ACBW	American Creek Below waste rock and downstream of seepage collection pond. Designed as long-term monitoring station through reclamation and closure.	2	30	540937	6877606
American Creek above confluence with Crooked Creek	AMER	American Creek below all planned facilities and disturbance and above confluence with Crooked Creek.	2	49	539333	6878839
Crooked Creek Below Ophir	CCBO	Downstream of Ophir Creek, which drains from the camp area and airstrip	3	40	539021	6876552
Anaconda Creek up stream	ANUP	Anaconda Creek above any potential influence from diversions or other physical disturbance.	1	23	544880	6875121
Anaconda Creek above confluence with Crooked Creek	ANDA	Below all proposed facilities in Anaconda Creek and above Crooked Creek.	1	40	539055	6874441
Crooked Creek above Crevice Creek.	CCAC	Below all proposed facilities and potential impacts to Crooked Creek. This site replaces CCBA.	3	39	538972	6872889

Table 3.7-1: Description of Surface Water Sampling Locations in Mine Area, Donlin Gold Baseline Water Quality Monitoring Program

Station Description	ID	Rationale and Purpose	Cat.	# of Sample Events	Easting[1] (meters)	Northing[1] (meters)
Crooked Creek below Confluence with Crevice Creek	CCBC	Below all proposed facilities and potential impacts to Crevice Creek.	3	30	538772	6872368
Crooked Creek directly below Lyman Wash Plant	CCBW	Crooked Creek below influence of placer mining operation.	3	39	540832	6882598
Upper Crevice Creek	CRUP	Crevice Creek above any potential influence from Anaconda facilities.	1	24	542022	6872357
Lower Crevice Creek above confluence with Crooked Creek	CRDN	Crevice Creek draining unmineralized and undisturbed area.	1	36	539215	6872587
Dome Creek upstream of Donlin Creek	DOME	Dome Creek downstream of potential exploration activities.	1	10	543532	6884207
Quartz Gulch upstream of Donlin Creek	QRTZ	Quartz Gulch downstream of potential exploration activities.	1	6	542595	6883575
Eagle Creek above confluence with Crooked Creek	EAGL	Eagle Creek downstream of potential domestic wastewater outfall facilities.	3	9	563529	6872600
Getmuna Creek above confluence with Crooked Creek	GETM2	Getmuna Creek below any potential influence from Jungjuk Road material site, and upstream of any mine site influence on Crooked Creek.	3	5	542076	6865101
Bell Creek above confluence with Crooked Creek	BELL	Bell Creek upstream of potential mine influence on Crooked Creek.	1	8	543636	6863886
Crooked Creek downstream of confluence with Bell Creek	CCBB	Crooked Creek about 6 miles downstream of all proposed mine facilities and potential impacts to Crooked Creek.	3	8	543667	6863547
Crooked Creek upstream of confluence with Kuskokwim River	CCAK	Crooked Creek about 8 miles downstream of all proposed mine facilities and potential impacts to Crooked Creek, and upstream of historical mine influences on Kuskokwim River.	3	21	546040	6860167

Notes:

1 Universal Transverse Mercator (UTM), Zone 4, NAD83 Datum

Source: ARCADIS 2012b, Enos 2013b, Weglinski 2016, Donlin Gold 2016e

Table 3.7-2: Surface Water Quality Summary for Category 1 Locations – Background Sites Outside of Mineralized Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations[4]	Arithmetic Mean Concentration[3]	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Major Cations										
Calcium, Dissolved	mg/L	227/227	100%	0.1-2.00	0.031-0.62	6.48-88.9	26.1	ANUP	-	-
Magnesium, Dissolved	mg/L	227/227	100%	0.03-2.00	0.015-0.62	2.26-43.5	8.48	CRDN	-	-
Potassium, Dissolved	mg/L	222/227	98%	0.3-2.00	0.15-1.2	0.301-4.17	0.398	CRDN	-	-
Sodium, Dissolved	mg/L	227/227	100%	0.3-2.00	0.15-1.2	1.42-53.1	5.19	CRDN	-	-
Major Anions										
Total Alkalinity (as $CaCO_3$)	mg/L	226/226	100%	0.02-40.0	2.00-12.4	22.5-410	95.4	DCBO	20 (min) [2b]	no
Bicarbonate	mg/L	226/226	100%	0.02-40.0	2.00-12.4	22.5-410	95.4	DCBO	-	-
Fluoride	mg/L	204/209	98%	0.062-0.1	0.031-0.1	0.031-0.161	0.0374	ANDA	1[2e]	no
Sulfate	mg/L	227/227	100%	0.062-2.5	0.031-0.5	0.883-53.2	11.7	SNUP	250[1]	no
Chloride	mg/L	227/227	100%	0.062-0.5	0.031-0.5	0.163-7.63	0.650	ACAW	230[2b]	no
Nutrients										
Nitrite + Nitrate (as N)	mg/L	227/227	100%	0.02-0.1	0.0062-0.031	0.0415-3.73	0.450	ACAW	10[2d]	no
Ammonia (as N)	mg/L	82/227	36%	0.062-0.1	0.031-0.031	0.031-0.497	0.0354	CRDN	2.99[2h]	no
Cyanide										
Total Cyanide	µg/L	39/227	17%	3.00-5.00	1.5-3.0	1.5-17	0.949	ACAW	-	-
WAD Cyanide	µg/L	49/227	22%	3.00-5.00	1.5-3.0	1.5-17	0.889	ACAW	5.2[2g]	yes
Metals										
Mercury, Dissolved	ng/L	20/20	100%	1.00-1.00	0.5-0.5	0.551-20	2.30	CRDN	-	-
Mercury	ng/L	226/237	95%	1.00-10.0	0.5-5.00	0.54-170	8.11	CRUP	12[2b]	yes
Aluminum, Dissolved	µg/L	132/192	69%	1.00-20.0	1.00-31	6.00-2,380	30.9	CRDN	-	-
Aluminum, Total	µg/L	224/227	99%	1.00-500	1.00-155	7.05-25,400	818	DCBO	750[2a]	yes
Antimony, Dissolved	µg/L	19/191	10%	0.4-1.00	0.31-0.4	0.316-4.93	0.191	SNUP	-	-
Antimony, Total	µg/L	17/226	8%	0.4-5.00	0.31-1.55	0.322-4.47	0.191	SNUP	6[2d]	no
Arsenic, Dissolved	µg/L	12/192	6%	0.5-5.00	0.5-2.5	1.65-231	1.27	QRTZ	-	-
Arsenic, Total	µg/L	38/227	17%	0.5-25.0	0.5-12.5	0.8-194	2.26	QRTZ	10[2d]	yes
Barium, Dissolved	µg/L	192/192	100%	0.5-3.00	0.5-0.94	19.2-226	78.8	ANDA	-	-
Barium, Total	µg/L	227/227	100%	0.5-9.4	0.5-4.7	27.8-950	98.0	DCBO	2,000[2d]	no
Beryllium, Dissolved	µg/L	0/192	0%	0.1-0.4	0.1-0.13	-	0.2	-	-	-
Beryllium, Total	µg/L	19/227	8%	0.1-2.00	0.1-0.65	0.13-1.62	0.0805	DCBO	4[2d]	no
Boron, Dissolved	µg/L	31/192	16%	10.0-50.0	3.1-15.0	3.58-31.9	7.07	CRDN	-	-
Boron, Total	µg/L	66/227	29%	10.0-250	3.1-75.0	3.33-72.8	7.75	DCBO	750[2e]	no
Cadmium, Dissolved	µg/L	1/192	1%	0.1-0.5	0.1-0.15	0.267-0.321	0.0761	DCBO	-	-

Table 3.7-2: Surface Water Quality Summary for Category 1 Locations – Background Sites Outside of Mineralized Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations[4]	Arithmetic Mean Concentration[3]	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Cadmium, Total	µg/L	16/227	7%	0.1-1.5	0.00075-0.75	0.0725-3.00	0.100	DCBO	0.23[2ab]	yes
Chromium, Dissolved	µg/L	39/192	20%	0.5-2.00	0.31-0.62	0.621-3.49	0.355	CRDN	-	-
Chromium, Total	µg/L	96/226	43%	0.5-6.2	0.31-3.1	0.2-36.3	1.17	DCBO	100[2d]	no
Cobalt, Dissolved	µg/L	23/192	12%	0.05-4.00	0.05-1.2	0.16-11.7	0.962	CRUP	-	-
Cobalt, Total	µg/L	29/226	13%	0.05-12.0	0.05-6.00	0.1-23.7	0.932	DCBO	50[2e]	no
Copper, Dissolved	µg/L	166/192	86%	0.5-1.00	0.31-0.5	0.311-4.02	0.432	CRDN	-	-
Copper, Total	µg/L	203/226	90%	0.5-3.1	0.31-1.55	0.313-46.5	1.73	QRTZ	7.7[2ab]	yes
Iron, Dissolved	µg/L	145/192	76%	20.0-250	6.2-78.0	10.0-3,430	179	CRDN	-	-
Iron, Total	µg/L	205/225	91%	20.0-780	6.2-390	8.62-38,100	1,370	DCBO	1,000[2b]	yes
Lead, Dissolved	µg/L	44/192	23%	0.124-0.5	0.062-0.1	0.0639-1.31	0.0480	DCBO	-	-
Lead, Total	µg/L	146/227	64%	0.124-200	0.062-0.31	0.066-19.7	0.558	DCBO	2.4[2ab]	yes
Lithium, Dissolved	µg/L	14/192	7%	6.2-100	3.1-20.0	3.17-9.29	1.58	DCBO	-	-
Lithium, Total	µg/L	41/209	20%	6.2-100	3.1-20.0	3.14-35.4	2.13	DCBO	2,500[2e]	no
Manganese, Dissolved	µg/L	186/192	97%	0.5-1.00	0.31-0.5	0.328-868	57.3	CRDN	-	-
Manganese, Total	µg/L	225/226	100%	0.5-3.1	0.31-1.55	0.484-2,350	90.5	DCBO	50[2f]	yes
Molybdenum, Dissolved	µg/L	6/192	3%	0.5-10.0	0.5-3.1	0.707-1.21	1.39	DCBO	-	-
Molybdenum, Total	µg/L	15/227	7%	0.5-50.0	0.5-15.5	0.6-8.71	1.62	CRDN	10[2e]	no
Nickel, Dissolved	µg/L	162/191	85%	1.24-20.0	0.6-0.62	0.627-3.69	0.523	CRDN	-	-
Nickel, Total	µg/L	216/226	96%	1.24-20.0	0.6-3.1	0.63-44.4	1.65	DCBO	43[2ab]	yes
Selenium, Dissolved	µg/L	6/192	3%	0.3-5.00	0.1-1.5	0.1-3.46	0.750	ACAW	-	-
Selenium, Total	µg/L	12/227	5%	0.3-25.0	0.1-7.5	0.1-3.98	0.836	ACAW	5[2b]	no
Silver, Dissolved	µg/L	0/192	0%	0.3-1.00	0.05-0.31	-	0.5	-	-	-
Silver, Total	µg/L	7/227	3%	0.3-5.00	0.05-1.55	0.62-0.62	0.172	ANDA	2.6[2ac]	no
Thallium, Dissolved	µg/L	0/192	0%	0.5-1.00	0.1-0.31	-	0.5	-	-	-
Thallium, Total	µg/L	12/227	5%	0.5-1000	0.1-1.55	0.338-0.696	0.179	ANDA	1.7[2f]	no
Vanadium, Dissolved	µg/L	1/192	1%	1.00-20.0	0.2-6.2	1.2-6.55	3.12	ANDA	-	-
Vanadium, Total	µg/L	26/227	11%	1.00-100	0.2-31.0	0.6-65.3	4.16	DCBO	100[2e]	no
Zinc, Dissolved	µg/L	77/191	41%	5.00-5.00	2.00-2.5	2.55-15.6	2.37	SNUP	-	-
Zinc, Total	µg/L	134/227	59%	5.00-25.0	1.5-12.5	1.65-159	6.10	QRTZ	100[2ab]	yes
General Water Quality Parameters										
Hardness (CaCO3)	mg/L	253/253	100%	1.00-5.00	1.00-5.00	17.9-401	98.0	DCBO	-	-
Total Dissolved Solids	mg/L	226/226	100%	6.2-10.0	3.1-10.0	47-478	125	DCBO	500[1]	no
Total Suspended Solids	mg/L	224/226	99%	0.294-20.0	0.141-6.00	0.204-1,470	42.6	ANDA	-	-

Table 3.7-2: Surface Water Quality Summary for Category 1 Locations – Background Sites Outside of Mineralized Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations[4]	Arithmetic Mean Concentration[3]	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Field Parameters										
Conductivity (Lab)	µS/cm	228/228	100%	0.954-1.00	0.477-1.00	58.5-809	217	DCBO	-	-
Conductivity, Field	µS/cm	233/233	100%	-	-	0.105-755	124	ACAW	-	-
Dissolved Oxygen, Field	mg/L	219/219	100%	-	-	3.46-24.5	13.7	ANDA	-	-
Ox./Reduc. Pot.(ORP/eH), field	mV	231/231	100%	-	-	-262-1,121	93.0	ANDA	-	-
pH (lab)	pH units	227/227	100%	0.1-0.2	0.1-0.1	6.8-8.3	7.51	CRDN	6.5-8.5[1]	no
pH, field	pH units	206/206	100%	-	-	4.42-12.9	7.42	ANDA	-	-
Water Temperature, Field	degrees C	232/232	100%	-	-	-0.53-10.5	2.35	BELL	-	-
Turbidity, Field	NTU	191/192	99%	-	-	0.00-242	13.5	CRDN	5[1]	yes

Notes:

Most Stringent Applicable Water Quality Criteria are provided for the parameters for which water quality standards have been established. Standards have not been established for all parameters. Cases where there are no applicable standards are indicated by a dash (-).

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012.
2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. WQS for metals are expressed as 'total recoverable' concentrations.
2a Aquatic life for fresh water hardness-dependent criteria. Acute and chronic aquatic life numeric criteria for some metals (Cd, Cu, Pb, Ni, Zn) are hardness dependent. Values contained in this table were calculated using a hardness value of 80.55 mg/L as $CaCO_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d). For Al, if pH≥7.0 and hardness ≥50, then 750 ug/L.
2b Aquatic life for fresh water (chronic) criteria.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water criteria.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life criteria (chronic) for free cyanide, measured as weak acid dissociable (WAD) cyanide.
2h Aquatic life for fresh water (chronic) criteria for ammonia based on 85th percentile of pH (7.85) and temperature (6.83°C) measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).
3 Calculated for all observations using detected values and ½ of the method detection limit value when analyte not detected.
4 Determined for the minimum and maximum of all detected reportable values; negative values are included when no detections are observed.
Source: Enos (2013b), Weglinski (2016), SRK (2017b), Rieser (2017)

Table 3.7-3: Surface Water Quality Summary for Category 2 Locations - Background Sites in Mineralized Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations[4]	Arithmetic Average Concentration[3]	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Major Cations										
Calcium, Dissolved	mg/L	101/101	100%	0.1-1.00	0.031-0.2	15.6-55.5	29.2	SNDN	-	-
Magnesium, Dissolved	mg/L	92/92	100%	0.03-1.00	0.015-0.2	6.02-25.5	12.4	SNDN	-	-
Potassium, Dissolved	mg/L	98/101	97%	0.3-2.00	0.15-1.2	0.327-1.34	0.499	AMER	-	-
Sodium, Dissolved	mg/L	101/101	100%	0.3-2.00	0.15-1.2	1.32-13.9	3.46	AMER	-	-
Major Anions										
Total Alkalinity (as $CaCO_3$)	mg/L	101/101	100%	0.02-40.0	2.00-12.4	57-166	104	SNDN	20 (min)[2b]	no
Bicarbonate	mg/L	101/101	100%	0.02-40.0	2.00-12.4	57-166	104	SNDN	-	-
Fluoride	mg/L	86/86	100%	0.062-0.5	0.031-0.1	0.038-0.116	0.0710	AMER	1[2e]	no
Sulfate	mg/L	101/101	100%	0.062-3.00	0.031-0.5	7.81-51.3	21.3	SNDN	250[1]	no
Chloride	mg/L	101/101	100%	0.062-3.00	0.031-0.5	0.356-59.2	1.60	AMER	230[2b]	no
Nutrients										
Nitrite + Nitrate(as N)	mg/L	101/101	100%	0.062-0.12	0.02-0.0372	0.101-3.24	0.508	AMER	10[2d]	no
Ammonia (as N)	mg/L	19/68	28%	0.062-0.1	0.031-0.031	0.032-0.239	0.0402	ACBW	2.99[2h]	no
Cyanide										
Total Cyanide	µg/L	13/101	13%	3.00-10.0	1.5-3.00	1.5-4.1	1.02	SNDN	-	-
WAD Cyanide	µg/L	16/101	16%	3.00-10.0	1.5-3.00	1.5-3.4	1.05	AMER	5.2[2g]	no
Metals										
Mercury, Dissolved	ng/L	4/4	100%	1.00-1.00	0.5-0.5	0.804-2.57	1.27	AMER	-	-
Mercury	ng/L	102/105	97%	1.00-3.00	0.1-1.00	0.561-46.7	6.11	SNDN	12[2b]	yes
Aluminum, Dissolved	µg/L	51/77	66%	5.00-20.0	1.00-6.2	6.3-889	65.6	AMER	-	-
Aluminum	µg/L	77/77	100%	5.00-80.0	1.00-24.8	6.73-4,240	302	SNDN	750[2a]	yes
Antimony, Dissolved	µg/L	20/77	26%	0.62-2.00	0.31-0.4	0.311-1.05	2.26	SNDN	-	-
Antimony	µg/L	31/101	31%	0.62-2.00	0.31-0.4	0.313-7.17	1.85	AMER	6[2d]	yes
Arsenic, Dissolved	µg/L	54/77	70%	2.00-5.00	0.5-2.5	1.68-11.8	3.27	SNDN	-	-
Arsenic	µg/L	81/101	80%	2.00-20.0	0.5-2.5	1.51-150	6.91	SNDN	10[2d]	yes
Barium, Dissolved	µg/L	77/77	100%	1.88-3.00	0.5-0.94	37.3-151	54.9	AMER	-	-
Barium	µg/L	101/101	100%	1.88-3.00	0.5-0.94	39.9-167	60.6	SNDN	2,000[2d]	no
Beryllium, Dissolved	µg/L	0/77	0%	0.4-0.5	0.1-0.13	-	0.2	-	-	-
Beryllium	µg/L	5/101	5%	0.4-0.5	0.1-0.13	0.143-0.143	0.715	SNDN	4[2d]	no
Boron, Dissolved	µg/L	14/77	18%	10.0-50.0	3.1-15.0	3.16-25.2	105	SNDN	-	-
Boron	µg/L	32/101	32%	10.0-50.0	3.1-15.0	3.37-35.6	81.9	SNDN	750[2e]	no

Table 3.7-3: Surface Water Quality Summary for Category 2 Locations - Background Sites in Mineralized Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations[4]	Arithmetic Average Concentration[3]	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Cadmium, Dissolved	µg/L	1/77	1%	0.5-0.5	0.1-0.15	0.168-0.168	0.0762	AMER	-	-
Cadmium	µg/L	9/101	9%	0.5-0.5	0.05-0.15	0.071-3.00	0.11	AMER	0.23[2ab]	yes
Chromium, Dissolved	µg/L	8/76	11%	1.00-2.00	0.31-0.62	0.657-3.18	4.43	AMER	-	-
Chromium	µg/L	39/101	39%	1.00-2.00	0.31-0.62	0.485-7.14	3.38	SNDN	100[2d]	no
Cobalt, Dissolved	µg/L	10/77	13%	0.3-4.00	0.05-1.2	1.41-11.6	8.91	SNDN	-	-
Cobalt	µg/L	5/101	5%	0.3-4.00	0.05-1.2	1.39-3.74	6.63	SNDN	50[2e]	no
Copper, Dissolved	µg/L	66/77	86%	0.62-3.00	0.31-0.5	0.312-1.63	0.547	AMER	-	-
Copper	µg/L	91/101	90%	0.62-3.00	0.31-0.5	0.34-5.47	0.922	SNDN	7.7[2ab]	no
Iron, Dissolved	µg/L	69/77	90%	20.0-250	6.2-78.0	38.9-1190	179	AMER	-	-
Iron	µg/L	101/101	100%	20.0-250	6.2-78.0	24.8-7,930	679	AMER	1,000[2b]	yes
Lead, Dissolved	µg/L	21/77	28%	0.124-0.5	0.062-0.1	0.063-0.591	0.479	AMER	-	-
Lead	µg/L	60/101	59%	0.124-0.5	0.062-0.1	0.066-3.18	0.565	SNDN	2.4[2ab]	yes
Lithium, Dissolved	µg/L	15/77	19%	6.2-100	3.1-20.0	3.12-5.94	2.05	SNDN	-	-
Lithium	µg/L	19/86	22%	6.2-100	3.1-20.0	3.15-10.6	2.26	SNDN	2,500[2e]	no
Manganese, Dissolved	µg/L	77/77	100%	0.62-3.00	0.31-0.5	15.0-760	96.7	AMER	-	-
Manganese	µg/L	101/101	100%	0.62-3.00	0.31-0.5	0.531-751	107	SNDN	50[2f]	yes
Molybdenum, Dissolved	µg/L	0/77	0%	2.00-10.0	0.5-3.1	-	4.7	-	-	-
Molybdenum	µg/L	6/101	6%	2.00-10.0	0.5-3.1	0.895-6.2	17.0	SNDN	10[2e]	no
Nickel, Dissolved	µg/L	71/77	92%	1.24-3.00	0.6-0.62	0.663-2.58	1.07	AMER	-	-
Nickel	µg/L	99/101	98%	1.24-3.00	0.6-0.62	0.665-8.16	1.45	SNDN	43[2ab]	no
Selenium, Dissolved	µg/L	1/78	1%	0.3-5.00	0.1-1.5	1.68-1.68	0.762	SNDN	-	-
Selenium	µg/L	3/101	3%	0.3-5.00	0.1-1.5	1.66-3.00	0.804	SNDN	5[2b]	no
Silver, Dissolved	µg/L	0/77	0%	0.3-1.00	0.05-0.31	-	0.5	-	-	-
Silver	µg/L	2/101	2%	0.3-1.00	0.05-0.31	0.62-0.62	1.70	SNDN	2.6[2ac]	no
Thallium, Dissolved	µg/L	1/77	1%	0.5-1.00	0.1-0.31	0.364-0.364	2.17	ACBW	-	-
Thallium	µg/L	7/101	7%	0.5-1000	0.1-0.31	0.341-0.865	1.72	AMER	1.7[2f]	no
Vanadium, Dissolved	µg/L	0/77	0%	1.00-20.0	0.2-6.2	0.6-0.6	9.88	AMER	-	-
Vanadium	µg/L	5/101	5%	1.00-20.0	0.2-6.2	0.8-80,500	34.2	SNDN	100[2e]	yes
Zinc, Dissolved	µg/L	27/77	35%	1.00-5.00	2.00-2.5	2.58-12.4	18.6	ACBW	-	-
Zinc	µg/L	51/99	52%	5.00-5.00	1.5-2.5	1.63-26.8	16.2	SNDN	100[2ab]	no

Table 3.7-3: Surface Water Quality Summary for Category 2 Locations - Background Sites in Mineralized Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations[4]	Arithmetic Average Concentration[3]	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
General Water Quality Parameters										
Hardness (CaC0$_3$), Dissolved	mg/L	103/103	100%	5.00-7.00	1.00-5.00	22.4-206	120	SNDN	-	-
Total Dissolved Solids	mg/L	101/101	100%	6.2-50.0	3.1-10.0	76.0-274	141	AMER	500[1]	no
Total Suspended Solids	mg/L	100/101	99%	0.298-5.00	0.141-1.5	0.2-141	15.9	AMER	-	-
Field Parameters										
Conductivity (Lab)	µS/cm	101/101	100%	0.954-10.0	0.477-1.00	130-470	243	AMER	-	-
Conductivity, Field	µS/cm	100/100	100%	-	-	0.124-1,620	156	SNDN	-	-
Dissolved Oxygen, Field	mg/L	100/100	100%	-	-	3.36-122.7	14.3	ACBW	-	-
Ox./Reduc. Pot.(ORP/eH), Field	mV	100/100	100%	-	-	-292.7-271	73	ACBW	-	-
pH (Lab)	pH Units	101/101	100%	0.1-0.2	0.1-0.1	5.6-8.0	7.51	AMER	6.5-8.5[1]	yes
pH, Field	pH Units	100/100	100%	-	-	5.54-8.94	7.44	AMER	-	-
Water Temperature, Field	degrees C	100/100	100%	-	-	-0.51-9.4	2.22	AMER	-	-
Turbidity, Field	NTU	75/75	100%	-	-	0.81-97.2	9.41	AMER	5[1]	yes

Notes:

Most Stringent Applicable Water Quality Criteria are provided for the parameters for which water quality standards have been established. Standards have not been established for all parameters. Cases where there are no applicable standards are indicated by a dash (-).

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. WQS for metals are expressed as 'total recoverable' concentrations.
2a Aquatic life for fresh water hardness-dependent criteria. Acute and chronic aquatic life numeric criteria for some metals (Cd, Cu, Pb, Ni, Zn) are hardness dependent. Values contained in this table were calculated using a hardness value of 80.55mg/L as CaCO$_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d). For Al, if pH≥7.0 and hardness ≥50, then 750 µg/L.
2b Aquatic life for fresh water (chronic) criteria.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water criteria.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life criteria (chronic) for free cyanide, measured as weak acid dissociable (WAD) cyanide.
2h Aquatic life for fresh water (chronic) criteria for ammonia based on 85th percentile of pH (7.85) and temperature (6.83°C) measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).

3 Calculated for all observations using detected values and ½ of the method detection limit value when analyte not detected.

4 Determined for the minimum and maximum of all detected reportable values; negative values are included when no detections are observed.

Source: Enos (2013b), Weglinski (2016), SRK (2017b), Rieser (2017)

Table 3.7-4: Surface Water Quality Summary for Category 3 Locations – Baseline Sites Draining Proposed Mine and Placer Mining Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations^{4}	Arithmetic Average Concentration^{3}	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Major Cations										
Calcium, Dissolved	mg/L	214/214	100%	0.1-2.00	0.031-0.62	9.8-53.8	29.5	SNOW	-	-
Magnesium, Dissolved	mg/L	214/214	100%	0.03-2.00	0.015-0.62	3.97-30.5	12.4	SNOW	-	-
Potassium, Dissolved	mg/L	210/214	98%	0.3-2.00	0.15-0.3	0.312-1.45	0.424	EAGL	-	-
Sodium, Dissolved	mg/L	214/214	100%	0.3-2.00	0.15-0.3	1.45-18.3	4.44	EAGL	-	-
Major Anions										
Total Alkalinity (as $CaCO_3$)	mg/L	214/214	100%	0.02-40.0	2.00-12.4	37.5-260	116	CCBO	20 (min) ^{2b}	no
Bicarbonate	mg/L	214/214	100%	0.02-40.0	2.00-12.4	37.5-260	116	CCBO	-	-
Fluoride	mg/L	192/192	100%	0.062-0.1	0.031-0.1	0.033-0.93	0.0439	SNOW	1^{2e}	no
Sulfate	mg/L	215/215	100%	0.062-2.5	0.031-0.5	2.19-48.3	15.6	SNOW	250^{1}	no
Chloride	mg/L	215/215	100%	0.062-0.5	0.031-0.5	0.183-1.78	0.628	SNOW	230^{2b}	no
Nutrients										
Nitrite + Nitrate(as N)	mg/L	213/215	99%	0.062-0.1	0.02-0.031	0.033-1.4	0.329	CCAC	10^{2d}	no
Ammonia (as N)	mg/L	59/215	27%	0.062-0.1	0.031-0.031	0.0315-0.393	0.0281	CCBB	2.99^{2h}	no
Cyanide										
Total Cyanide	µg/L	41/215	19%	3.00-5.00	1.5-3.00	1.5-5.6	0.904	CCBB	-	-
WAD Cyanide	µg/L	48/215	22%	3.00-5.00	1.5-3.00	1.5-4.8	0.840	CCBW	5.2^{2g}	no
Metals										
Mercury, Dissolved	ng/L	20/21	95%	1.00-1.00	0.5-0.5	1.07-24.3	3.20	CCAK	-	-
Mercury	ng/L	217/222	98%	1.00-10.0	0.1-5.00	0.518-260	8.31	CCBW	12^{2b}	yes
Aluminum, Dissolved	µg/L	106/177	60%	1.00-100	1.00-31.0	3.00-898	71.5	CCAK	-	-
Aluminum	µg/L	206/215	96%	1.00-200	1.00-62.0	6.32-18,100	379	EAGL	750^{2a}	yes
Antimony, Dissolved	µg/L	32/176	18%	0.4-1.00	0.31-0.4	0.302-1.14	2.80	CCAK	-	-
Antimony	µg/L	23/213	11%	0.4-1.00	0.31-0.4	0.282-2.41	4.53	CCAC	6^{2d}	no
Arsenic, Dissolved	µg/L	34/176	19%	0.5-5.00	0.5-2.5	1.00-4.43	22.3	SNOW	-	-
Arsenic	µg/L	97/215	45%	0.5-5.00	0.5-2.5	1.2-32.1	36.8	EAGL	10^{2d}	yes
Barium, Dissolved	µg/L	177/177	100%	0.5-3.00	0.5-0.94	38.7-216	77.0	EAGL	-	-
Barium	µg/L	215/215	100%	0.5-3.00	0.5-0.94	39.8-488	84.6	EAGL	2,000^{2d}	no
Beryllium, Dissolved	µg/L	1/177	3%	0.1-0.4	0.1-0.13	0.132-0.132	1.17	CCBW	-	-
Beryllium	µg/L	6/215	0%	0.1-0.4	0.1-0.13	0.13-0.681	1.88	EAGL	4^{2d}	no
Boron, Dissolved	µg/L	29/176	16%	10.0-50.0	3.1-15.0	3.31-21.4	134	CCBC	-	-
Boron	µg/L	65/215	30%	10.0-50.0	3.1-15.0	3.25-88.3	217	SNOW	750^{2e}	no
Cadmium, Dissolved	µg/L	1/177	1%	0.1-0.5	0.1-0.15	0.266-0.266	1.35	CCBC	-	-

Table 3.7-4: Surface Water Quality Summary for Category 3 Locations – Baseline Sites Draining Proposed Mine and Placer Mining Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations^4	Arithmetic Average Concentration^3	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Cadmium	µg/L	13/215	6%	0.1-0.5	0.1-0.15	0.137-0.419	0.0832	EAGL	0.23^{2ab}	yes
Chromium, Dissolved	µg/L	32/175	18%	0.5-2.00	0.31-0.62	0.579-2.32	5.64	CCAK	-	-
Chromium	µg/L	73/213	34%	0.5-2.00	0.31-0.62	0.5-25.7	9.36	EAGL	100^{2d}	no
Cobalt, Dissolved	µg/L	24/177	14%	0.05-4.00	0.05-1.2	0.45-10.3	7.64	EAGL	-	-
Cobalt	µg/L	7/215	3%	0.05-4.00	0.05-1.2	0.15-11.7	17.5	EAGL	50^{2e}	no
Copper, Dissolved	µg/L	164/175	94%	0.5-1.00	0.31-0.5	0.144-2.32	3.10	CCAK	-	-
Copper	µg/L	203/214	95%	0.5-10.0	0.31-0.5	0.16-25.0	4.00	SNOW	7.7^{2ab}	yes
Iron, Dissolved	µg/L	157/176	89%	20.0-250	6.2-78.0	9.06-3,160	912	CCBW	-	-
Iron	µg/L	213/215	99%	20.0-250	6.2-78.0	49.1-27,100	1,000	EAGL	$1{,}000^{2b}$	yes
Lead, Dissolved	µg/L	36/177	20%	0.124-0.5	0.062-0.1	0.062-0.611	0.573	CCAK	-	-
Lead	µg/L	119/215	55%	0.124-200	0.062-0.1	0.0623-11.8	0.836	EAGL	2.4^{2ab}	yes
Lithium, Dissolved	µg/L	31/176	18%	6.2-100	3.1-20.0	3.1-5.66	28.0	SNOW	-	-
Lithium	µg/L	43/192	22%	6.2-100	3.1-20.0	3.12-19.6	50.2	EAGL	$2{,}500^{2e}$	no
Manganese, Dissolved	µg/L	177/177	100%	0.62-3.00	0.31-0.5	0.362-1,170	78.9	CCBW	-	-
Manganese	µg/L	215/215	100%	0.62-10.0	0.31-0.5	2.61-1,460	102	CCBW	50^{2f}	yes
Molybdenum, Dissolved	µg/L	4/177	2%	2.00-10.0	0.5-3.1	0.674-4.23	27.6	CCBO	-	-
Molybdenum	µg/L	5/215	2%	2.00-10.0	0.5-3.1	0.62-6.2	44.7	EAGL	10^{2e}	no
Nickel, Dissolved	µg/L	171/176	97%	1.24-3.00	0.6-0.62	0.651-2.69	4.03	EAGL	-	-
Nickel	µg/L	212/214	99%	1.24-3.00	0.6-0.62	0.659-26.2	9.79	EAGL	43^{2ab}	no
Selenium, Dissolved	µg/L	1/177	1%	0.3-5.00	0.1-1.5	0.2-1.88	13.4	CCBW	-	-
Selenium	µg/L	6/215	3%	0.3-5.00	0.1-1.5	0.2-3.00	0.781	CCBO	5^{2b}	no
Silver, Dissolved	µg/L	0/177	0%	0.3-1.00	0.05-0.31	-	0.5	-	-	-
Silver	µg/L	3/215	1%	0.3-1.00	0.05-0.31	0.62-0.62	4.48	CCAC	2.6^{2ac}	no
Thallium, Dissolved	µg/L	2/177	1%	0.5-1.00	0.1-0.31	0.486-0.6	2.78	SNOW	-	-
Thallium	µg/L	5/215	2%	0.5-1.00	0.1-0.31	0.31-0.62	4.49	CCAK	1.7^{2f}	no
Vanadium, Dissolved	µg/L	1/177	1%	1.00-20.0	0.2-6.2	6.53-6.53	55.6	CCBO	-	-
Vanadium	µg/L	8/215	4%	1.00-20.0	0.2-6.2	1.24-41.0	89.9	EAGL	100^{2e}	no
Zinc, Dissolved	µg/L	69/175	39%	1.00-5.00	2.00-2.5	0.518-20.5	20.5	SNOW	-	-
Zinc	µg/L	107/214	50%	5.00-5.00	1.5-2.5	0.374-75.3	38.4	EAGL	100^{2ab}	no
General Water Quality Parameters										
Hardness (CaC0$_3$), Dissolved	mg/L	215/215	100%	1.00-5.00	1.00-5.00	29.2-260	122	SNOW	-	-
Total Dissolved Solids	mg/L	214/214	100%	6.2-32.3	3.1-10.0	73-248	145	SNOW	500^1	no
Total Suspended Solids	mg/L	212/215	99%	0.298-5.00	0.141-2.5	0.2-896	20.3	EAGL	-	-

Table 3.7-4: Surface Water Quality Summary for Category 3 Locations – Baseline Sites Draining Proposed Mine and Placer Mining Areas

Analyte	Units	Frequency of Detection		Range of Detection Limits (Reporting)	Range of Method Detection Limits	Range of Detected Concentrations[4]	Arithmetic Average Concentration[3]	Maximum Sample Location	Most Stringent Applicable Water Quality Criterion	Max Detect > Criterion
Field Parameters										
Conductivity (Lab)	µS/cm	214/214	100%	0.954-1.00	0.477-1.00	89.1-465	250	SNOW	-	-
Conductivity, Field	µS/cm	217/217	100%	-	-	0.125-518	140	SNOW	-	-
Dissolved Oxygen, Field	mg/L	202/202	100%	-	-	2.14-22.8	12.8	CCBW	-	-
OX./Reduc. Pot.(ORP/eH), Field	mV	217/217	100%	-	-	-390-525	93.9	EAGL	-	-
pH (Lab)	pH Units	214/214	100%	0.1-0.2	0.1-0.1	6.7-8.1	7.53	CCBC	6.5-8.5[1]	no
pH, Field	pH Units	215/215	100%	-	-	4.6-9.8	7.40	CCAC	-	-
Water Temperature, Field	degrees C	215/215	100%	-	-	-0.5-13.0	3.30	CCAC	-	-
Turbidity, Field	NTU	172/172	100%	-	-	0.34-393	8.91	EAGL	5[1]	yes

Notes:

Most Stringent Applicable Water Quality Criterion are provided for the parameters for which water quality standards have been established. Standards have not been established for all parameters. Cases where there are no applicable standards are indicated by a dash (-).

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012.
2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. WQS for metals are expressed as 'total recoverable' concentrations.
  2a Aquatic life for fresh water hardness-dependent criteria. Acute and chronic aquatic life numeric criteria for some metals (Cd, Cu, Pb, Ni, Zn) are hardness dependent. Values contained in this table were calculated using a hardness value of 80.55 mg/L as $CaCO_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d). For Al, if pH≥7.0 and hardness ≥50, then 750 µg/L.
  2b Aquatic life for fresh water (chronic) criteria.
  2c Aquatic life for fresh water (acute) criteria.
  2d Drinking water criteria.
  2e Irrigation water criteria.
  2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
  2g Aquatic life criteria (chronic)for free cyanide, measured as weak acid dissociable (WAD) cyanide.
  2h Aquatic life for fresh water (chronic) criteria for ammonia based on 85th percentile of pH (7.85) and temperature (6.83°C) measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).
3 Calculated for all observations using detected values and ½ of the method detection limit value when analyte not detected.
4 Determined for the minimum and maximum of all detected reportable values; negative values are included when no detections are observed.
Source: Enos (2013b), Weglinski (2016), SRK (2017b), Rieser (2017)

General Patterns

Tables 3.7-2 through 3.7-4 show the minimum, maximum, and average values measured for each parameter within each category of sites. The range of detection-limit values is also given for analytes that were not present at concentrations above the analytical detection limits. The location of the maximum measured value for each parameter is shown, and the maximum measured values are compared to the State of Alaska WQS (most stringent standard for all designated water uses) (ADEC 2008a, 2012d, SRK 2017b). The selection of the appropriate standards is discussed in Section 3.7.1.2.1.

The following analytes had maximum concentrations that exceeded the relevant standards (ADEC 2008a, 2012a).

- Category 1 (background sites outside of mineralized areas):
    - WAD Cyanide, aluminum (total), arsenic (total), cadmium (total), copper (total), iron (total), lead (total), manganese (total), mercury (total), nickel (total), zinc (total), and turbidity were above the relevant standards.
- Category 2 (waters draining areas of defined mineralized zones with no placer mining activities):
    - Aluminum (total), antimony (total), arsenic (total), cadmium (total), iron (total), lead (total), manganese (total), mercury (total), vanadium (total), pH, and turbidity exceeded or were outside of the relevant standards.
- Category 3 (sites draining proposed mine and placer mining areas):
    - Aluminum (total), arsenic (total), cadmium (total), copper (total), iron (total), lead (total), manganese (total), mercury (total), and turbidity were above the relevant standards.

For all categories of surface water sampling locations, total concentrations of metals such as copper, aluminum, arsenic, iron, lead, and zinc were often substantially higher than dissolved concentrations, indicating that the majority of these metals are present as particulate matter rather than dissolved aqueous species (ARCADIS 2012b, SRK 2017b). Dissolved metals are generally more bioavailable than those present in the particulate phase; therefore particulate phase metals are less likely to contribute to effects on biological resources than dissolved aqueous phase metals.

The calculated averages for cadmium exceeded the ADEC standard for Category 1, Category 2, and Category 3 sites. However, this result is likely an artifact of the high reporting level detection limits (0.1 to 1.5 µg/L), which sometimes also exceeds the applicable water quality standard for cadmium. For all three stream categories, only about 6 percent of the samples had detectable levels of cadmium.

In order to generally check the correctness of the analytical data presented in Table 3.7-2 through Table 3.7-4, the anion-cation balances were determined for each individual sample for which sufficient data were available. As an aqueous solution is always electrically neutral, the sum (in milliequivalents/liter) of the anions and the cations should always balance. For all three categories, nearly all of the samples (99 to 100 percent) had a charge balance error of less than 15 percent. The mean concentrations of selected analytes collected from each category of surface water locations (Categories 1, 2, and 3) were compared using the statistical program Pro-UCL version 4 (EPA 2009e). The surface water data were statistically compared to test the hypothesis that the mean concentrations of the key analytes in Category 1 waters (non-mineralized zones) were not statistically different (at a 95 percent confidence level) from the mean concentrations of the same analytes in Category 2 (mineralized areas, not impacted by historical placer mining) and Category 3 (mineralized areas impacted by historical placer mining, and encompassing the proposed project site) waters (ARCADIS 2012b). The comparison showed that at a 95 percent confidence level, there were no statistically significant differences between the categories, with the following exceptions:

- Barium- The mean concentration of barium was statistically higher in Category 2 waters versus Category 1 waters.
- Sulfate- The mean concentration of sulfate was statistically higher in Category 2 waters versus Category 1 waters.
- Manganese - The mean concentration of manganese was statistically higher in Category 3 waters versus Category 1 waters.

The statistically higher concentrations in Category 2 and 3 waters indicate that the mineralized locations, both mined and un-mined, have naturally elevated concentrations of these species relative to background, non-mineralized (Category 1) sample locations (ARCADIS 2012b).

The Donlin Gold Baseline Water Quality Monitoring Program includes three sites located within the Snow Gulch drainage: SNUP, above both the mineralization trend and placer mining (Category 1); SNDN, below the mineralized trend and above historic placer tails (Category 2); and SNOW, below mineralization and historic placer tails and above confluence with Crooked Creek (Category 3). Mean turbidity values at these three sites over the period from the third quarter of 2005 to through the second quarter of 2015 were 3.4 NTU at station SNUP, 7.3 NTU at station SNDN, and 3.1 NTU at station SNOW, all of which are below the mean turbidity values for each category of sites, suggesting that placer mining has not resulted in adverse water quality impacts in the Snow Gulch drainage.

U.S. Geological Survey Study of Crooked Creek Watershed

Sampling and Analytical Program

In addition to the data generated by the Donlin Gold water quality characterization program, the U.S. Geological Survey (USGS) conducted a study involving the analysis of 20 stream samples, 6 samples from seeps, and 2 well water samples collected from the Crooked Creek watershed in July 2002 (Mueller et al. 2003). The sites sampled for the USGS study include the headwaters, mid-stream, and lower reaches of Bell Creek, Anaconda Creek, American Creek, Lewis Gulch, Queen Gulch, Snow Gulch, Quartz Gulch, Dome Creek, and Ophir Creek. The locations of several of these sites correspond to Donlin Gold's surface water sampling stations (Figure 3.7-1). Thus in some instances, the USGS water quality data are comparable to those collected as part of the Donlin Gold Baseline Water Quality Monitoring Program; however, it should be noted that temporal variations in the water quality parameters and differences in the numbers of samples analyzed for each location may be responsible for differences in the reported values and descriptive statistics for water quality parameters reported as part of the USGS study and the Donlin Gold Baseline Water Quality Monitoring Program.

Samples were collected using trace-clean techniques and all analyses were performed at the Denver laboratories of the USGS, with the exception of the total organic carbon analyses, which were performed at the Applied Science and Engineering Technology laboratory, University of Alaska, Anchorage (UAA). Analyses for dissolved constituents were performed subsequent to filtration through 0.45 µm filters; unfiltered samples were also analyzed to determine concentrations of total recoverable constituents (Mueller et al. 2003).

Study Results – General Patterns

The USGS study generally characterizes stream and seep waters in the Crooked Creek watershed as neutral to slightly alkaline pH (6.9-7.9), with major element and anion abundances in the following order, from highest concentration to lowest: calcium, magnesium, sodium, potassium; and, bicarbonate, sulfate, nitrate, chlorine, and fluorine (Mueller et al. 2003). The dissolved oxygen (DO) concentrations range from 6.9 to 12 mg/L, and specific conductance values range from 89 to 531 µS/cm (Mueller et al. 2003). A statistical summary of the analyses performed on surface water samples is provided as Table 3.7-5.

One of the goals of the USGS study was to examine spatial relationships between water chemistry and mineralized areas within the Crooked Creek watershed, around and within the Mine Site. Analyses of the total recoverable fractions of constituents (as opposed to the dissolved fraction) were used to characterize spatial patterns in the distributions of certain constituents in the water. Two general spatial patterns were identified (Mueller et al. 2003):

- First, the locations of elevated concentrations of arsenic and antimony in samples collected from Lewis Gulch, Queen Gulch, and Quartz Gulch correspond to the orientation of the known mineralized zone (Section 3.1.1.2 Geology, Figure 3.1-3). Weathering of arsenopyrite and stibnite associated with the mineralized zone was identified as the most likely factor responsible for the anomalous concentrations. In addition, concentrations of aluminum, iron, and to a lesser extent manganese, were elevated at sites associated with the known mineralized zone, as well as at DC10 located in the upper reaches of the Dome Creek drainage approximately 2 miles east of the known mineralized zone (Figure 3.7-1). The elevated concentrations of aluminum, iron, and manganese may be related to weathering of minerals in the igneous rocks that are spatially associated with the gold deposit.
- The second spatial pattern involves elevated concentrations of calcium, magnesium, sulfate, and antimony relative to other surface water sites in the study area. This occurs at DC17 located in the upper part of the Bell Creek drainage east of the known mineralized zone, and at DC07, a bedrock seep on the south side of Snow Gulch. Mueller et al. (2003) suggest that the elevated concentrations[1] may be attributable to changes in lithology associated with local interbedding of limestone.

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[1] These include calcium (55 mg/L), magnesium (23 mg/L), and sulfate (130 mg/L).

Table 3.7-5: Summary of USGS Surface Water Quality Data from Proposed Mine Area

Parameter	Surface Water Samples									
	Filtered-acidified					**Unfiltered-acidified**				
	MAX	**MIN**	**MEDIAN**	**MEAN**	**n**	**MAX**	**MIN**	**MEDIAN**	**MEAN**	**n**
Temperature (ºC)	7.6	2	3.5	3.9	25	7.6	2	3.5	3.9	25
pH	8	7	7.5	7.5	25	8	7	7.5	7.50	25
Conductivity (µS/cm)	535	90	289	304	25	535	90	289	304	25
Dissolved Oxygen (mg/L)	13	7	12	11	25	13	7	12	11.00	25
Alkalinity (mg/L)	140	35	100	104	25	140	35	100	104	25
Total Alkalinity (mg/L)	170	43	122	126	25	170	43	122	126	25
Chlorine (mg/L)	0.7	0.4	0.5	0.6	26	0.7	0.4	0.5	0.6	26
Fluorine (mg/L)	0.10	0.08	0.10	0.09	24	0.1	0.08	0.10	0.09	24
Nitrate (mg/L)	4.4	0.7	2.5	2.5	26	4.4	0.7	2.45	2.51	26
Sulfate (mg/L)	130	3	19	26	26	130	3	19	26	26
Calcium (mg/L)	64	11	31	32.0	26	61	11	34	34	26
Magnesium (mg/L)	25	1.8	12	13.0	26	23	1.8	11	12.0	26
Sodium (mg/L)	5	0.7	1.6	1.8	26	5.1	0.78	1.5	1.7	26
Potassium (mg/L)	0.66	0.26	0.4	0.4	26	1	0.18	0.38	0.40	26
Silver (µg/L)	<3	N.A.	N.A.	N.A.	N.A.	<3	N.A.	N.A.	N.A.	N.A.
Aluminum (µg/L)	19	0.95	7.6	8.1	26	15300	17	132	1056	26
Arsenic (µg/L)	7.7	1	3.7	4.0	14	274	1	4	25	15
Boron (µg/L)	7.4	6.3	6.3	6.7	3	5.5	5.3	5.4	5.4	2
Barium (µg/L)	101	18	56	53	26	218	18	54	64	26
Beryllium (µg/L)	<0.05	N.A.	N.A.	N.A.	N.A.	0.5	0.07	0.14	0.21	4
Bismuth (µg/L)	0.08	0.005	0.03	0.038	12	0.11	0.007	0.02	0.030	8
Cadmium (µg/L)	<0.02	N.A.	N.A.	N.A.	N.A.	0.22	0.02	0.06	0.07	7
Cobalt (µg/L)	0.10	0.02	0.04	0.04	26	9.3	0.04	0.12	0.82	26
Chromium (µg/L)	6.4	1	5.1	4.8	26	16	1.7	3.5	4.1	26
Copper (µg/L)	1.2	0.5	0.7	0.8	7	16	0.5	0.8	2.7	14
Iron (µg/L)	74	2	16	20	23	18,000	20	180	1,600	25
Lithium (µg/L)	3.4	1.1	2.0	2.1	23	18	1.1	1.9	3.0	20
Manganese (µg/L)	35	0.04	2.0	6.8	26	790	0.4	12	62	26
Molybdenum (µg/L)	2.1	0.20	0.39	0.66	17	2.4	0.22	0.46	0.76	21
Nickel(µg/L)	0.7	0.3	0.4	0.5	26	21	0.6	1	2.6	26
Phosphorus (µg/L)	0.1	0.01	0.02	0.03	21	0.5	0.01	0.02	0.07	13

Table 3.7-5: Summary of USGS Surface Water Quality Data from Proposed Mine Area

	Surface Water Samples									
	Filtered-acidified					Unfiltered-acidified				
Parameter	MAX	MIN	MEDIAN	MEAN	n	MAX	MIN	MEDIAN	MEAN	n
Lead (µg/L)	0.05	0.05	0.05	0.05	1	12	0.05	0.2	1.3	17
Rubidium (µg/L)	0.53	0.05	0.25	0.25	26	7.9	0.19	0.33	0.78	26
Antimony (µg/L)	4.1	0.1	0.5	1.0	16	4.5	0.1	0.33	0.91	20
Selenium (µg/L)	4.2	1	1.5	2.0	6	4.4	1	1.4	1.7	16
Silicon (µg/L)	4.1	1.7	3	3	26	14	1.8	3.1	3.9	26
Strontium (µg/L)	383	36	130	134	26	366	34	125	126	26
Titanium (µg/L)	1	0.3	0.6	0.6	26	191	0.4	3.4	17	24
Uranium (µg/L)	1.3	0.01	0.25	0.27	26	1.3	0.03	0.25	0.34	24
Vanadium (µg/L)	1.9	0.5	1.5	1.4	26	28	0.6	1.35	3.3	26
Tungsten (µg/L)	3.8	0.1	0.3	0.6	26	13	0.03	0.2	0.39	24
Zinc (µg/L)	5.6	0.5	0.7	1.4	11	59	0.6	0.95	6.0	22
Zirconium (µg/L)	0.1	0.06	0.06	0.07	11	1.8	0.05	0.1	0.24	15

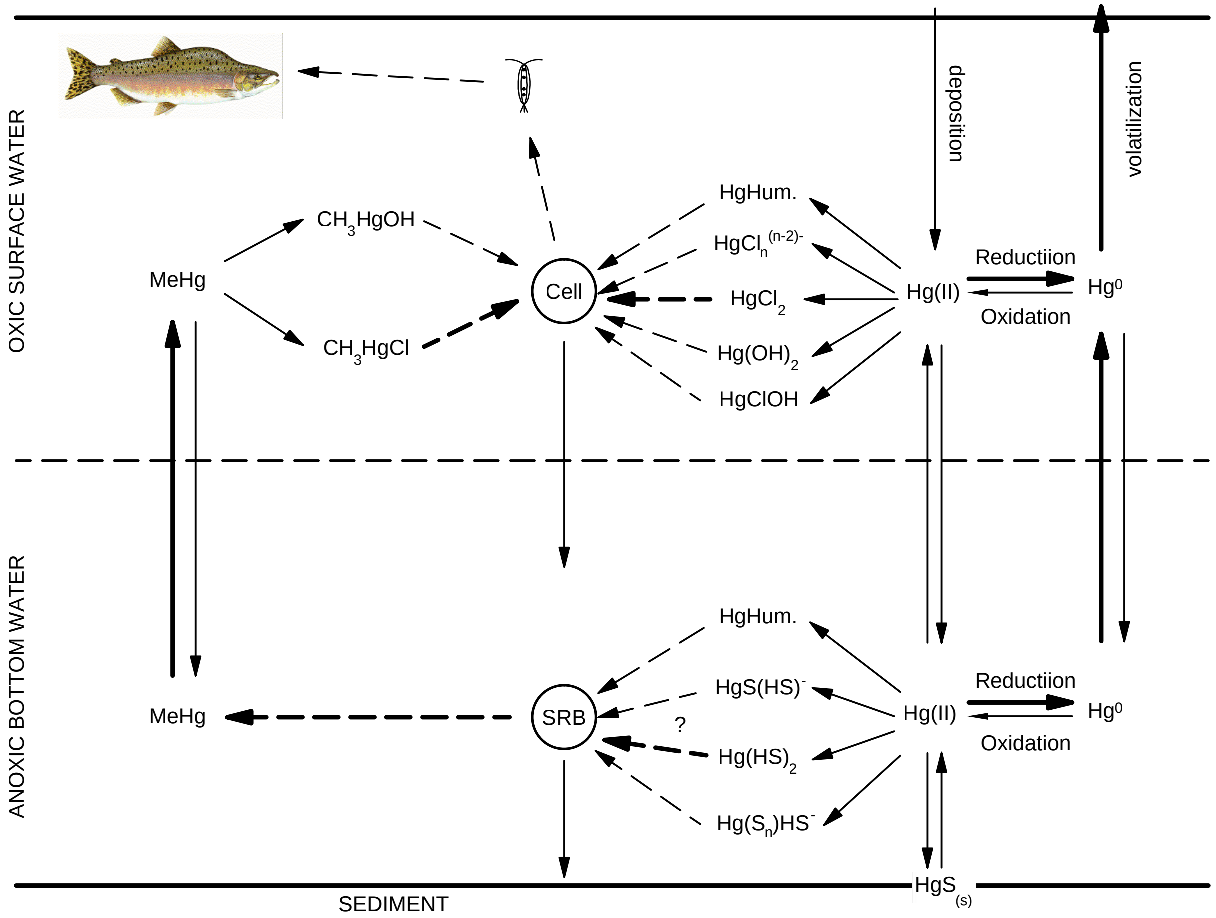
Source: Mueller et al. 2003

Note: Cells marked “N.A.” indicate parameters for which the maximum measured values did not exceed the lower limit of quantification for the analysis.

Discussion of Individual Constituents

Mercury and Methylmercury

Because mercury is potentially toxic and accumulates in organisms, including plants and fish, the distribution and speciation of mercury in the water are important water quality considerations for the proposed project. Besides elemental mercury, which is relatively insoluble in water, the major forms of mercury in water are ionic mercury, which forms complexes with chloride, sulfide, or organic acids; and organic mercury, which is mainly methylmercury (Morel et al. 1998). In aquatic environments, the ionic forms of mercury are often associated with particulate material or dissolved organic matter (e.g., humic acids and other forms of dissolved organic carbon). The processes that transform mercury between its elemental, ionic, and organic forms are described in detail by Morel et al. (1998), and are summarized in Figure 3.7-2.



Data Sources: Morel, et al. (1998)

1. Processes that transform mercury between elemental, ionic, and organic forms in oxic and anoxic waters. Sulfate-reducing bacteria (SRB) and iron-reducing bacteria in anoxic waters and sediments are responsible for converting the ionic mercury into methylmercury (MeHg).

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THE AQUATIC CYCLE OF MERCURY

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FIGURE 3.7-2

Oxidation of elemental mercury to ionic mercury occurs both in the atmosphere via photochemically initiated processes, and in the water as a result of both photo-oxidation and enzyme-catalyzed reactions (Morel et al. 1998; Lalonde et al. 2001; Siciliano et al. 2002). Aqueous ionic mercury in oxic waters interacts with chloride and hydroxide ions and organic acids to form complexes that are bioavailable, but do not bioaccumulate because they are readily excreted by organisms. In anoxic waters, however, ionic mercury exhibits an extremely high affinity for sulfide, such that the speciation of dissolved ionic mercury in anoxic waters is completely dominated by sulfide and bisulfide complexes[2], even when total sulfide concentrations are very low (Morel et al. 1998). Some neutrally charged mercury-sulfide complexes and mercury-organic ligand complexes are available for uptake by the sulfate-reducing and iron-reducing bacteria that are responsible for the conversion of ionic mercury to methylmercury in anoxic waters and sediments (Morel et al. 1998; Benoit et al. 1999; Schaefer et al. 2011; Hsu-Kim et al. 2013). Methylation occurs primarily in anoxic, low pH (acidic) environments with high concentrations of organic matter. Because methylmercury is more readily retained by higher trophic-level organisms than other mercury species, it is the primary form of mercury that biomagnifies in the food chain. The accumulation of methylmercury in higher trophic-level organisms results mainly from the ingestion of methylmercury-containing food rather than direct uptake of methylmercury from the water (Morel et al. 1998).

As part of the Donlin Gold water quality characterization program, concentrations of total mercury in surface water samples collected from the mine area have been analyzed using EPA Method 1631 (EPA 2002a). Concentrations of total mercury detected in 545 of 564 water samples collected between June 2005 and June 2015 ranged from 0.518 to 260 nanograms per liter (ng/L) (Rieser 2017). The applicable numeric criteria for mercury for CWA purposes are the EPA-approved aquatic life criteria: 2,400 ng/L acute and 12 ng/L chronic, both as total recoverable Hg (EPA 2013k). Total mercury concentrations did not exceed acute criterion in any of the samples. About 80 samples or 14 percent of samples from all three categories had total mercury concentrations in excess of the chronic criterion, and three samples had concentrations higher than 10 times the criterion. The highest of these were found in samples from tributaries such as American, Crevice, and Eagle creeks, as well as the main stem Crooked Creek. These data suggest that naturally elevated concentrations of mercury are found sporadically in surface water in the vicinity of the Mine Site, with occasional spikes possibly due to precipitation and localized rock weathering conditions.

Seasonal trend analysis of total mercury concentrations measured as part of the Donlin Gold water quality characterization program was performed using Seasonal Kendall and Mann-Kendall statistical tests (ARCADIS 2012b). No statistically significant temporal trends in the mercury concentrations were detected when data were analyzed on a quarter-by-quarter basis (ARCADIS 2012b). Correlation analysis to indicate the proportion of the variance in the analyte concentration that can be attributed to the variation in stream flow using least squares regression showed moderately strong relationships between mercury concentrations and stream flow at stations CCAC (r^2=0.575; p=0.026; n=15) and CCAK (r^2=0.659; p=0.107; n=7) (Tetra Tech 2013). In addition, analysis of the mercury concentrations measured during base flow, spring flow, and storm flow conditions indicate that the mercury concentrations are generally higher during spring flow and storm flow conditions relative to base flow (Tetra Tech

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[2] Sulfide and bisulfide complexes include $Hg(HS)_2$, $HgS(HS)^-$, and HgS_2^{2-}.

2013). Due to higher concentrations of mercury and higher flow rates, the mercury load in Crooked Creek increases during spring flow and storm flow conditions relative to base flow (Tetra Tech 2013). Higher stream discharge is usually associated with higher flow velocity, which entrains particulate material from the substrate (Tetra Tech 2013). Thus the data suggest that a substantial fraction of the total mercury load in the Crooked Creek watershed (measured as total mercury in the water) is associated with particles entrained from the substrate during high flow events. This particulate mercury is generally less bioavailable than mercury found in the dissolved phase.

During a USGS investigation of the spatial distribution of chemical constituents in the Kuskokwim River (Wang 1999), total mercury was detected at a concentration of 10.5 ng/L in a 1997 sample collected from Crooked Creek near its confluence with the Kuskokwim River. The USGS result is consistent with those from Donlin Gold station CCAK, where total mercury concentrations ranging from 1.59 to 96.9 ng/L (arithmetic mean = 10.8 ng/L; n=20) were measured in water samples collected between June 2005 and 2015.

Concentration of methylmercury in water is a key factor that determines concentrations and accumulation of this organic mercury constituent in biota (Morel et al. 1998). Methylmercury concentrations in water are often lower and more temporally variable than concentrations in sediment. In 2007, measurement of methylmercury was added to the Donlin Gold sediment monitoring program (Section 3.7.2.3.2), and in 2013 through 2015, methylmercury and dissolved mercury were added to the surface water quality characterization program. Concentrations of dissolved mercury in surface water samples from the Crooked Creek watershed are shown in Tables 3.7-2 through 3.7-4. Analysis of methylmercury in mine area water samples was conducted on unfiltered samples using EPA method 1630 (EPA 1998a).

Sixty-two (62) surface water samples from the Crooked Creek watershed were collected and analyzed for methylmercury during six separate sampling events in June 2013 through June 2015 as part of the Donlin Gold Water Quality Characterization Program (Table 3.7-6). The concentrations of methylmercury in the water ranged from below the detection limit (~0.020 ng/L) to 0.55 ng/L. Several samples with results close to detection limits have laboratory qualifiers indicating that methylmercury was also found in a blank. Methylmercury detections not associated with blanks were found in 34 of the 62 samples collected. About three-quarters of these were from Category 3 waters, some of which were located in lower Crooked Creek several miles below the Mine Site. The location with the highest concentration was collected from Category 3 site CCAC, located immediately downstream of the Mine Site. The mean of all samples in Table 3.7-6, assuming half the detection limit when results are non-detect, and excluding blank-flagged samples, is 0.074 ng/L.

Other data describing concentrations of methylmercury in surface water from the Crooked Creek watershed were collected in 1997 as part of the USGS investigation of the spatial distribution of chemical constituents in the Kuskokwim River (Wang 1999). The concentration of methylmercury was 0.49 ng/L in an unfiltered water sample collected from Crooked Creek near its confluence with the Kuskokwim River (Wang 1999). The Donlin Gold methylmercury results from the same location (site CCAK, Crooked Creek near its confluence with the Kuskokwim River) range from 0.047 ng/L (March 2014) to 0.32 ng/L (June 2014) as shown in Table 3.7-6.

Table 3.7-6: Methylmercury Concentrations in Mine Area Surface Water Samples, Crooked Creek Watershed

Station ID	Category[a]	Result[b] (ng/L)					
		June 2013	March 2014	June 2014	September 2014	December 2014	June 2015
AMER	2	0.038[B]	ND (<0.020)	0.066	0.079	ND (<0.020)	NS
ANDA	1	0.034[B]	ND (<0.020)	0.069	0.034[B]	ND (<0.020)	0.05
ANUP	1	NS	NS	NS	NS	NS	ND (<0.022)
BELL	1	NS	0.024[B]	0.093	0.078	0.039[B]	0.058
CCAC	3	NS	0.034[B]	0.55	0.082	0.053	0.085
CCAK	(3)	0.058	0.047[B]	0.32	0.083	0.059	0.074
CCBB	(3)	0.056	0.039[B]	0.065	0.103	0.047[B]	0.085
CCBC	3	ND (<0.020)	NS	NS	NS	NS	NS
CCBO	3	0.054	0.024[B]	0.13	0.095	0.052	0.076
CCBW	3	0.025[B]	0.024[B]	0.116	0.056	0.046	0.062
CRDN	1	ND (<0.020)	ND (<0.020)	ND (<0.114)	0.059	ND (<0.020)	ND (<0.020)
DCBO	1	0.031[B]	NS	0.071	0.067	0.033[B]	0.055[B]
GETM2	3	NS	0.033[B]	0.139[B]	0.095	NS	NS
SNOW	3	NS	NS	NS	NS	NS	0.071
SNUP	1	NS	NS	NS	NS	NS	0.027[B]

Notes:

a Categories:
1 = Background sites outside of mineralized areas
2 = Background sites in mineralized areas
3 = Baseline sites draining proposed mine and placer mining areas
(3) = Baseline sites draining proposed mine and placer mining areas, located 6-8 miles downstream
b Analyses performed on unfiltered samples.
B = Analyte found in blank associated with sample.
ND = not detected above method detection limit (MDL) shown in parentheses.
NS = not sampled

Source: Enos 2013b; Weglinski 2016

The recommended CWA Section 304(a) numeric water quality criterion for methylmercury is expressed as a fish and shellfish tissue concentration, not as a water column concentration (EPA 2010a). This criterion, 0.3 mg methylmercury per kg fish tissue (wet weight), describes the concentration of methylmercury in freshwater and estuarine fish and shellfish tissue that should not be exceeded to protect consumers of fish and shellfish among the U.S. population that consumes an average of about two fish meals per person per month. The 0.3 mg/kg criterion is based on a fish consumption rate of 17.5 g/day. It is important to note that the 0.3 mg/kg criterion is for fish that humans would consume. Because methylmercury bioaccumulates in biota, lower trophic level organisms will have generally lower methylmercury levels than higher trophic level organisms (the ones that humans are more likely to consume). It is possible for concentrations to increase by an order of magnitude between prey and higher level predatory fish. States and authorized tribes remain free to adjust EPA's recommended criterion, provided that their new or revised water quality criterion for methylmercury protect the designated uses and are based on scientifically defensible methodology. Concentrations of total mercury in fish tissue from the Crooked Creek drainage (Section 3.13.2.1.5, Fish and Aquatic Resources) range from 0.013 to 0.045 mg/kg wet weight (OtterTail 2012f). These values, which include both methylmercury and other forms of mercury, are well below the EPA criterion for methylmercury alone.

Arsenic

Arsenic is naturally occurring within the mineralization in the Donlin gold deposit. It exists primarily in the form of sulfide minerals including arsenopyrite, orpiment, and realgar. Because of its widespread occurrence at the project site, it is an important parameter for characterizing baseline water quality in the proposed project area. The most stringent Alaska Water Quality Standard numeric criterion for total arsenic is 10 µg/L (for drinking water) (ADEC 2008a). The numeric criteria for protection of aquatic life are less stringent than the criterion for drinking water; the acute criterion for protection of aquatic life is 340 µg/L (one-hour average), and the chronic criterion for the protection of aquatic life is 150 µg/L (4-day average). Unlike the applicable criterion for drinking water, which refers to the total concentration of arsenic in the water, the criteria for the protection of aquatic life refer to the concentrations of dissolved arsenic in the water. None of the applicable criteria distinguish between different oxidation states of arsenic, and no data are known to be available concerning whether the toxicities of the different redox species of arsenic to aquatic organisms are additive (ADEC 2008a).

Samples collected as part of the Donlin Gold water quality characterization program contain total arsenic concentrations greater than the most stringent applicable criterion (10 µg/L for protection of drinking water) in all three categories of site: the highest concentration in Category 1 waters was 194 µg/L at sample location QRTZ; in Category 2 waters, was 150 µg/L at sample location SNDN; and in Category 3 waters, was 32.1 µg/L at sample location CCBO (Enos 2013b; Weglinski 2015g). With regard to arsenic concentrations, creeks draining the known mineralized and placer-mined areas (e.g., Snow, Queen, Lewis, and American) have characteristics that set them apart from other waters. Because gold is associated with arsenopyrite, arsenic concentrations are generally greatest in the creeks draining the mineralized area of interest (Mueller et al. 2003; SRK 2012b).

Although placer tailings piles may contain concentrations of arsenopyrite and scorodite, dissolved arsenic concentrations in waters collected downstream from other placer mines in Alaska are highly variable, and often similar or less than those at background sites. For example, in the Fairbanks placer district, near the center of a heavily placer-mined stretch of Cleary Creek, dissolved arsenic concentrations were only 14 ppb. Yet on Fairbanks Creek in the same district, waters flowing through a heavily placer-mined area immediately above its junction with Too-Much-Gold Creek contained arsenic ranging from 41 to 44,000 ppb, and background arsenic concentrations in this area ranged from 5 to 260 ppb (Ray and others 1992, Goldfarb et al. 1997).

Arsenic can occur in the environment in several oxidation states (-3, 0, +3 and +5), but in natural waters is mostly found in inorganic form as oxyanions of trivalent arsenite or pentavalent arsenate (Smedley and Kinniburgh 2002). Arsenic speciation (distribution between different oxidation states) is driven primarily by the pH and oxidation-reduction potential of the aquatic environment. Under oxidizing conditions and at higher pH, the less mobile arsenate (pentavalent) form is favored, whereas more mobile trivalent arsenite ions dominate the speciation under reducing conditions and at lower pH (Smedley and Kinniburgh 2002). The Eh-pH diagram for arsenic in oxygenated water is shown on Figure 3.7-3. The only available measurements of arsenic speciation in the vicinity of the Mine Site suggest that the predominant arsenic species in the surface water are the oxidized, less mobile pentavalent species (Mueller et al. 2003).

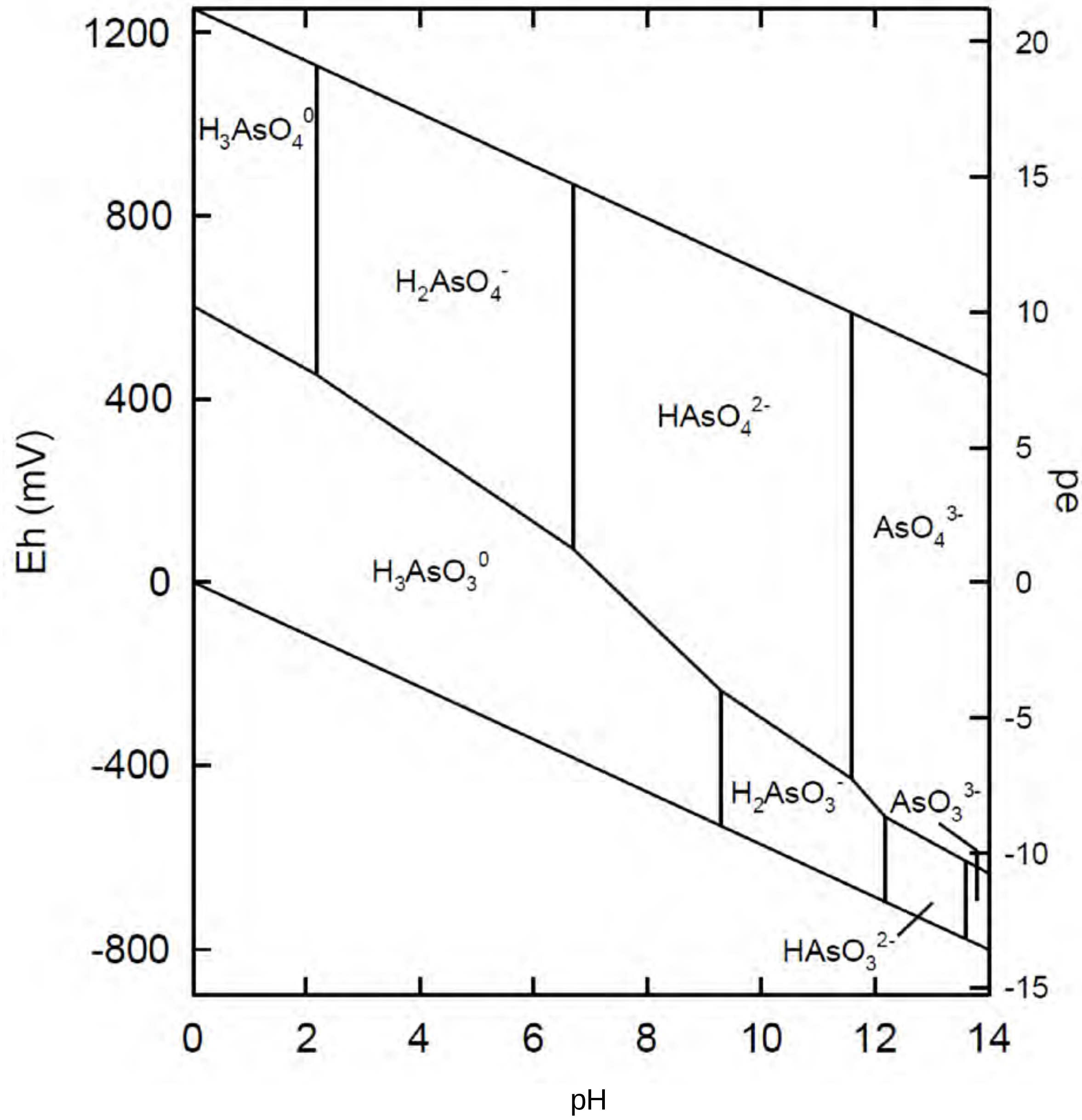
Antimony

Like arsenic, antimony is a potentially toxic redox active-metalloid that occurs primarily as compounds of trivalent antimonite and pentavalent antimonate in natural environments. Stibnite is the most common form of antimony in sulfide deposits. Stibnite may dissolve gradually in stream waters with circumneutral pH, with the antimony mobilized into solution and subsequently precipitated into hydrated iron oxides, clays, organic material, and sulfide compounds. In the fluvial environment, antimony from stibnite deposits can be mobilized and cycled through aquatic ecosystems and riparian vegetation (Ashley et al. 2005).

Of 529 surface water samples analyzed for total antimony as part of the Donlin Gold water quality characterization program between June 2005 and June 2015, 70 had concentrations of antimony above the detection limits (Enos 2013b; Weglinski 2015g). The sample with the highest concentration of antimony (7.17 µg/L) was collected from location AMER on American Creek, below all proposed facilities and disturbance and above its confluence with Crooked Creek. The most stringent applicable numeric water quality criterion for antimony in surface water is the State of Alaska criterion for drinking water, which specifies a maximum concentration of 6 µg/L total antimony. Only one of the surface water samples (from AMER in 2012) had a reportable antimony concentration in excess of this criterion.

Cyanide

Cyanide is a chemical group consisting of one atom of carbon connected to one atom of nitrogen by a triple molecular bond. Cyanides are compounds (substances formed by the joining of two or more atoms) that contain a cyanide group. Cyanides can occur naturally as products of anabolism in some plants, bacteria, and fungi, or they can be manmade (CDC 2013). Unlike elemental constituents of surface waters (such as arsenic and mercury), cyanides can be destroyed by oxidation reactions, which predominantly yield less-toxic cyanate as a reaction product that can be subsequently hydrolyzed, ultimately yielding ammonium and bicarbonate ions. Such reactions may occur both naturally and as a result of anthropogenic processes. Because many cyanide compounds are potentially toxic, and because cyanide would be used in the gold extraction process at the proposed mine, the baseline concentrations of cyanides in surface waters are an important water quality consideration.



Notes:

1. Shown for the system $As–O_2–H_2O$ at 25ºC and 1 bar pressure, with total [As] = $1*10^{-5}$ mol/L
2. Predominant[DN1] forms in mine site surface water are oxidized, less mobile As(V) species (Mueller et al. 2003).

Data Sources: Smedley and Kinniburgh 2002

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Eh-pH DIAGRAM[1] OF AQUEOUS
ARSENIC SPECIES[2] IN
OXYGENATED WATER

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FIGURE 3.7-3

Since 2005, concentrations of weak acid dissociable (WAD) cyanide have been measured in 529 surface water samples as part of the Donlin Gold water quality characterization program. The WAD method provides an indication of the cyanide forms that could be potentially toxic to humans and animals. Frequency of detection in these samples ranged from 17 percent in Category 2 waters, to 23 percent in both Category 1 and Category 3 waters (Tables 3.7-2 through 3.7-4). Only two of the samples had concentrations in excess of the most stringent Alaska Water Quality Standard numeric criterion (5.2 µg/L, measured as WAD cyanide or equivalent) (Enos 2013b). Both of these samples were collected from Category 1 background locations outside of mineralized areas (17 µg/L at ACAW and 5.9 µg/L at ANDA), and may represent naturally occurring forms of cyanide described above. None of the samples collected from Category 2 and 3 sites had WAD cyanide concentrations greater than the most stringent criterion.

Crooked Creek Watershed Loading Study

The term “loading” is defined as the mass of a chemical constituent or element that is discharged per unit time. It is calculated by multiplying the concentration of the chemical in the water by the measured volumetric discharge rate of water in a stream or river, or through a discharge structure (Tetra Tech 2013). As part of the baseline characterization of surface water quality in the Crooked Creek watershed, loading of several metals and other chemical constituents was analyzed based on concurrent water quality and flow measurements made between 2005 and 2010. The Baseline Water Quality Loading Analysis (Tetra Tech 2013) provides a characterization of the mass loading for selected chemical constituents at established sites within the Crooked Creek watershed. Key findings from the report are summarized below.

At the Donlin Gold project site, stream flow changes throughout the year, occurring as one of three main flow regimes: base flow, storm-generated flow, and spring breakup and runoff. Consistently high flows can be expected during spring breakup and runoff. Hydrographs from area gages indicate that relatively high stream flows can occur from late April through June, depending on the year. Lower base flows occur in the summer and fall after spring runoff subsides, and in the winter under the snow pack. Storm events can occur anytime during the summer and fall months; however, they are generally more prominent in August and September (Tetra Tech 2013).

Both the concentrations of chemical constituents in the water, and the total mass loading at particular stations, are affected by seasonal changes in stream flow. Concentrations of some metals and other analytes often increase concomitant with stream flow, especially when the analyte of interest is present primarily in the particulate phase, associated with sediments or naturally occurring minerals. Higher stream discharge is usually associated with higher flow velocity, which entrains sediments from the substrate, increasing the total concentration of the analyte in the water (Tetra Tech 2013). Positive correlations between flow rate and the concentrations of total aluminum, iron, and manganese have been noted in area streams (ARCADIS 2012b; Tetra Tech 2013). In contrast, concentrations of other analytes, for example sulfates, may decrease with increasing stream flow as a result of dilution. Tetra Tech (2013) considered the correlations between stream flow and both total and dissolved phase concentrations of cadmium, copper, nickel, and zinc. Overall, the positive correlations between stream flow and total concentrations of these elements were consistent with what would be expected for analytes entrained from sediment and other solid material by high stream flow.

Loading records were calculated for all events where both stream flow and water quality data were concurrently measured (Table 3.7-1). The following general observations were made based on the individual event data (Tetra Tech 2013):

- In Snow Gulch, a Category 2 drainage, loads increase for most analytes as the drainage travels through mineralized zones, with up to an order of magnitude increase in load often observed between stations SNUP and SNDN. This trend is more prominent during spring runoff and storm-generated flow than during base flow. In contrast, loading increases only slightly near the mouth of the drainage (between stations SNDN and SNOW). For example, the average load of total iron during storm events increases substantially from 0.91 kg/day at Station SNUP to 7.0 kg/day at station SNDN, and increases only slightly more between there and station SNOW near the mouth (to 7.5 kg/day).
- Loads of most analytes are similar between Station DCBO on Donlin Creek, and Station CCBW just downstream of its confluence with Crooked Creek. This observation is generally true during all three flow regimes: base flow, storm flow, and spring breakup. Loading from Snow Gulch appears to have little influence on the load at Station CCBW, probably because of the relatively lower stream flow in Snow Gulch compared to Crooked Creek.
- American Creek was the smallest drainage evaluated. The data indicate that the range of flow values at Station AMER was small in relation to the other stations. Total daily loads of arsenic at AMER ranged from 0.11 to 0.32 kg/day, which represents approximately 17 percent of the total daily arsenic load in Crooked Creek at Station CCBO, located downstream of the confluence with American Creek.
- Similar to Snow Gulch, the loads for most analytes in American Creek increase as the creek flows through the mineralized zones. This trend is evident for all three flow regimes (base flow, storm flow, and spring breakup), but is most apparent during spring runoff and storm flow events. Mean total iron loads increase substantially between Stations CCBW and CCBO, located on either side of the American Creek confluence, for all three flow regimes.
- Loads of analytes generally increase in Anaconda Creek from the upper part of the drainage to the mouth of the drainage across all three flow regimes. The increase in average load can be up to an order of magnitude for total aluminum, iron, and mercury. This trend is most prominent during spring runoff and storm events. For example, the average load of total iron during spring runoff increases from 16 kg/day at station ANUP to 103 kg/day at Station ANDA.
- Similar to Anaconda Creek, loads of most analytes in Crevice Creek generally increase from the upper part of the drainage to the mouth, and the trend is most noticeable during spring runoff and storm events.
- The loads of most analytes increase approximately two to four times on Crooked Creek between Station CCBC below the proposed mine area and Station CCAK near the mouth of Crooked Creek. Evaluation of the observed analyte concentrations and flow data indicate that such increases are caused in part by large increases in flow between the two stations.

3.7.2.1.2 TRANSPORTATION CORRIDOR

Kuskokwim River

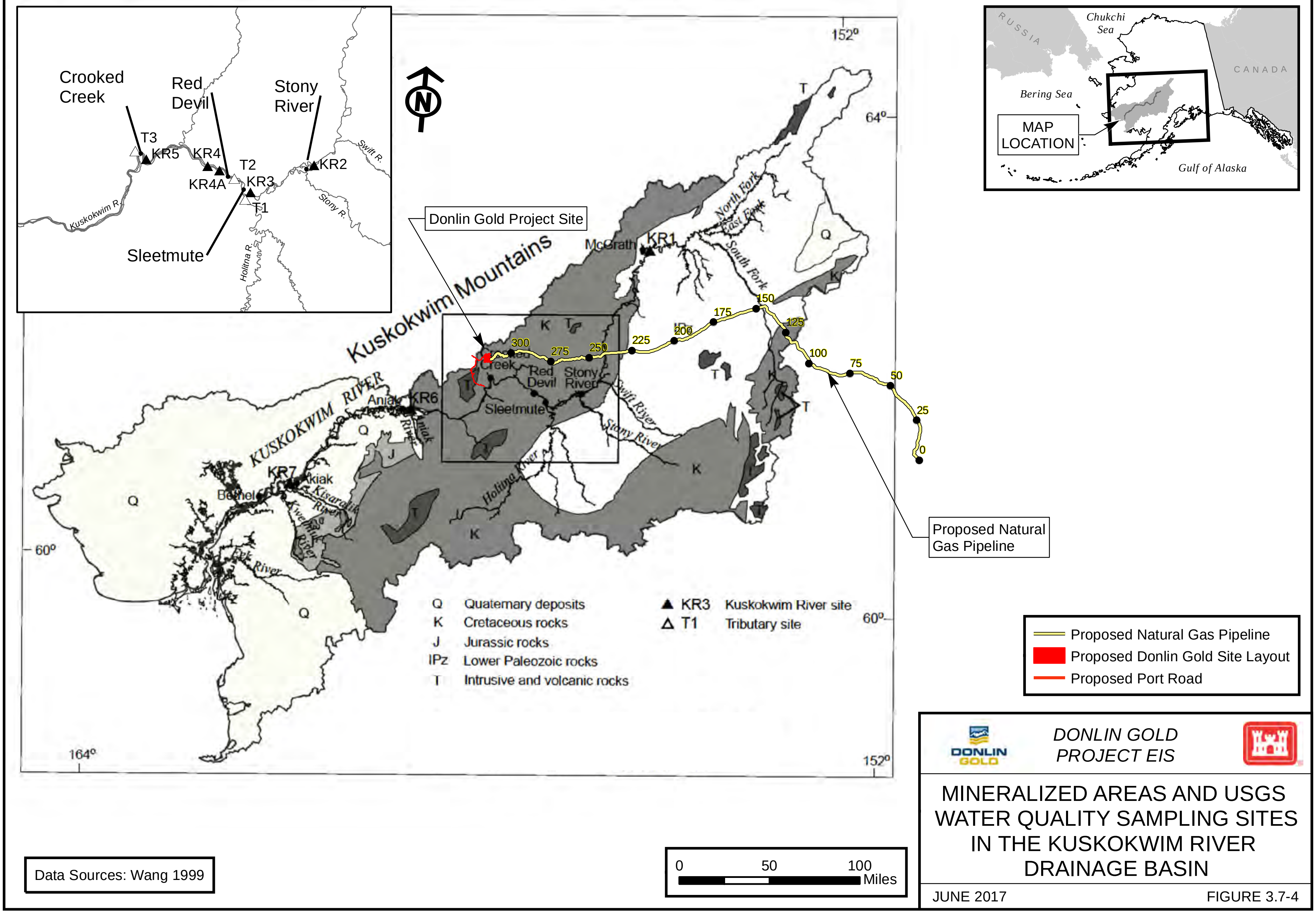
The primary body of water potentially affected in this component is the Kuskokwim River, which drains a basin of over 50,000 square miles along its 700-mile course from Interior Alaska to the Bering Sea (Wang 1999). Water quality in the Kuskokwim River downstream from its confluence with Crooked Creek (the area considered within the transportation corridor for this analysis) is generally protected for all uses under the Alaska Water Quality Standards. Several villages between Crooked Creek and Bethel get their drinking water directly from the river (Section 3.5, Surface Water Hydrology).

The middle section of the Kuskokwim River runs through a highly mineralized region of Alaska that contains mercury, antimony, gold, silver, and polymetallic deposits (Szumigala and Weakland 2012). Although the Kuskokwim River basin is largely undeveloped, mining for placer gold and mercury have occurred in the basin. Remnant waste rock and ore from historic mining activities are still present at some sites and influence the distribution of chemical constituents in the Kuskokwim River (Wang 1999; BLM 2012e). Mineralized areas are primarily associated with Cretaceous-age sedimentary rock and Tertiary-age intrusives as shown on Figure 3.7-4. The Kuskokwim River is not industrially polluted, but it does have anomalous mercury content near naturally mineralized sources and historic mine sites (Nelson et al. 1977; BLM 2012e). The highest individual mercury anomalies occur near the largest mineralized sources of tributary streams, and the mercury content is rapidly diluted downstream (Nelson et al. 1977; Wang 1999). It is likely that some mercury from allochthonous sources (e.g., Asian coal) is present in the Kuskokwim River as a result of atmospheric transport and deposition; however, the spatial distributions of mercury concentrations in Kuskokwim River watersheds strongly suggest that local minerals are the predominant source of the mercury in the water.

Background water quality in the Kuskokwim River is described in the USGS Water-Resources Investigations Report 99-4177: Spatial Distribution of Chemical Constituents in the Kuskokwim River (Wang 1999), incorporated here by reference. The USGS has collected water quality data from the Kuskokwim River at the Crooked Creek confluence since 1951. Historical water quality data for the Kuskokwim River at Crooked Creek are summarized below and in Table 3.7-7.

In general, Kuskokwim River water, like Crooked Creek water, is classified as calcium to calcium-magnesium-bicarbonate water. Calcium and magnesium concentrations decrease down the main stem of the Kuskokwim, as do the bicarbonate and sulfate concentrations. However, the total loads of calcium, magnesium, bicarbonate, and sulfate generally increase at downstream locations as a result of increased volumetric flow rates.

Dissolved iron concentrations increase downstream from McGrath and peak at Crooked Creek. Dissolved iron concentrations in Crooked Creek are about four times greater than those in the Kuskokwim River, however the contribution of Crooked Creek to the dissolved iron load in the Kuskokwim River is relatively minor, due to the much greater discharge of the Kuskokwim River (about 40,000 cubic feet per second [cfs]) relative to Crooked Creek (about 200 cfs).



Data Sources: Wang 1999

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MINERALIZED AREAS AND USGS WATER QUALITY SAMPLING SITES IN THE KUSKOKWIM RIVER DRAINAGE BASIN

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FIGURE 3.7-4

Dissolved manganese concentrations are relatively uniform throughout the main stem of the Kuskokwim River, and are higher in Crooked Creek than in the main stem of the Kuskokwim River.

The highest concentrations of dissolved strontium and barium in Kuskokwim River water were measured upstream of the confluence with Crooked Creek, and concentrations of both dissolved strontium and barium decrease with distance downriver.

Total mercury concentrations in water samples from the main stem of the Kuskokwim River range from 1.9 to 9.7 ng/L, which are comparable to average total mercury concentrations in each of the three location categories in the Crooked Creek watershed (5.77 to 8.93 ng/L) measured as part of the Donlin Gold water quality characterization program (Tables 3.7-2 through 3.7-4). Elevated concentrations of total mercury have been measured in Red Devil Creek; a tributary to the Kuskokwim River located about 30 miles upstream of Crooked Creek and not considered part of the proposed project area. In Red Devil Creek, total mercury exceeds the concentration at which the EPA indicates that aquatic life is affected; however, the elevated concentrations of total mercury in Red Devil Creek water do not substantially affect total mercury concentrations in the main stem of the Kuskokwim River because Red Devil Creek is small (discharge about 1 cfs) relative to the Kuskokwim River (about 40,000 cfs) (Wang 1999, BLM 2012e). A description of mercury transport at a regional scale by Nelson et al. (1977) suggests that mercury is concentrated in hotspots near lode sources, the influences of which decrease over short spatial scales due to the effects of dilution and mixing driven by water and sediment discharge in the main stem of the Kuskokwim River.

Nutrient concentrations are generally low throughout the lower Kuskokwim River system, and virtually all of the nitrogen present is in the form of nitrate (Wang 1999). The historical median suspended sediment concentration at the Kuskokwim River confluence with Crooked Creek during open water is 134 mg/L, with an inner quartile range from 87 to 223 mg/L (Table 3.7-7) (Wang 1999). The historical mean suspended sediment concentration at the Kuskokwim River confluence with Crooked Creek is 188 mg/L (Table 3.7-7) (Wang 1999). As a basis for comparison, the mean concentration of total suspended solids measured at Category 3 sites as part of the Donlin Gold Baseline Water Sampling Program was 25.7 mg/L. Thus, the mean concentration of suspended sediment in the Kuskokwim River at its confluence with Crooked Creek is more than eight times greater than the mean value recorded for the Category 3 sites, suggesting that concentrations of suspended sediment in the Kuskokwim River are generally much higher than those measured in Crooked Creek and other tributaries located downstream from the proposed mine location.

Turbidity data were collected in July 2014 at three locations in the Kuskokwim River: from upriver to downriver, these include Holokuk, Birch Tree Crossing, and Kalskag. Turbidity measurements at these locations averaged 150, 101, and 84 Nephelometric Turbidity Units (NTU), respectively (Donlin Gold 2015i). While these measurements exhibit a declining trend from upriver to downriver, turbidity is likely to be variable depending on season, river stage, and precipitation occurring at different locations along the river. Overall, the turbidity levels are very high compared to smaller streams throughout the middle Kuskokwim region (Figure 3.7-5), and are likely related to flow rate and naturally high levels of glacial silt and fine-grained soils in banks throughout the river system.

Table 3.7-7: Historical Water Quality Data from Kuskokwim River at Crooked Creek

Property or constituent	Unit	Period of record	Mean	Median	Q1a	Q3b	Reporting Limit	Most Stringent Applicable Water Quality Criterion
Suspended sediment	mg/L	1967-97	188	134	87	223	1.0	200[1]
Specific conductance	S/cm	1951-98	171	176	152	193	1.0	-
pH	pH units	1951-98	7.6	7.6	7.3	7.8	0.1	6.5-8.5
Dissolved oxygen	mg/L	1975-97	10.6	10.3	9.9	11.25	0.1	7 (min)
Alkalinity	mg/L as $CaCO_3$	1951-97	70	73	64	80	1.0	20 (min)
Calcium	mg/L as Ca	1951-96	24.6	25	21	28	0.1	-
Magnesium	mg/L as Mg	1951-96	5.6	5.5	4.6	6.4	0.1	-
Sulfate	mg/L as SO_4	1951-96	16.7	17	13	20	0.1	250
Silica	mg/L as SiO_2	1951-96	8.3	8	7.1	9.45	0.1	-
Arsenic, total	µg/L as As	1975-82	4.4	4	3	5	1	10
Barium, dissolved	µg/L as Ba	1980-96	35.9	35	30	41	1	2,000[2]
Iron, total	µg/L as Fe	1975-82	5,440	4,500	3,700	7,300	10	1,000
Iron, dissolved	µg/L as Fe	1975-96	227	225	140	300	10	-
Manganese, total	µg/L as Mn	1975-82	130	130	80	170	10	50
Manganese, dissolved	µg/L as Mn	1975-96	12.2	6.5	5	9	4	-
Strontium, dissolved	µg/L as Sr	1975-96	130	140	99	160	10	-

Notes:

Samples collected during the open water period, mid-May to mid-October.

a 25th percentile

b 75th percentile

1 ADEC (2012d) specifies no measurable increase in concentration of settleable solids above natural conditions; for irrigation or water spreading, may not exceed 200 mg/l for an extended period of time. In addition, the water quality standard requires no measureable increase in concentration of settleable solids above natural conditions, as measured by the volumetric Imhoff cone method.

2 ADEC (2008a) WQS for barium expressed as total concentration.

Source: Wang 1999.

In addition to the USGS data describing the spatial distribution of chemical constituents in the Kuskokwim River, the Georgetown Tribal Council has conducted baseline water quality monitoring and analyses describing surface water quality at a site located in the main stem of the Kuskokwim River adjacent to Georgetown, Alaska, beginning in 2008 (Schaberg 2014). The Georgetown Tribal Council baseline water quality sampling project was initiated due to impending development in the area, including the Donlin Gold Project. The primary goal of water quality monitoring was to document existing conditions in Georgetown prior to development in the area (Schaberg 2014). The water quality analyses conducted by the Georgetown Tribal Council from 2008 through 2013 indicate that pH values at the Kuskokwim River site were within acceptable pH range, according to ADEC standards. The average pH at the Kuskokwim River site was 7.7 ± 0.5 (Schaberg 2014). Likewise, measured levels of dissolved oxygen were within the range specified by the most stringent applicable water quality standards. The average dissolved oxygen concentration at the Kuskokwim River site was 12.7 mg/L ± 2.8 and concentrations of dissolved oxygen do not appear to be trending upward or downward in any way (Schaberg 2014). Arsenic, barium, chromium, lead, and vanadium measured at the Kuskokwim River site were present at very low levels, well below the thresholds specified by the applicable water quality criteria (Schaberg 2014). Cadmium was never detected at the Kuskokwim River site used for the Georgetown Tribal Council Water Quality Monitoring Program (Schaberg 2014). During a single sampling event in September 2013, the analysis for semi-volatile organic compounds (SVOCs) detected bis-2-ethylhexyl phthalate, also known as DEHP, in a sample of the Kuskokwim River water at a concentration of 0.010 mg/L (Schaberg 2014). The EPA has established an MCL of 0.006 mg/L for this contaminant. Although the concentration of DEHP measured in the Kuskokwim River water in September 2013 exceeds the MCL, it is possible that sample contamination was responsible for the elevated concentration of DEHP and ongoing testing conducted by the Georgetown Tribal Council will continue to monitor levels of SVOCs and other potential contaminants of concern in the Kuskokwim River at Georgetown (Schaberg 2014).

Angyaruaq (Jungjuk) and Birch Tree Crossing Roads

Most of the proposed mine access road to the potential Jungjuk Port, and the initial section of the proposed BTC Road (Alternative 4), would be located in the Crooked Creek drainage basin. The southern portion of the proposed mine access road would be located in the Jungjuk Creek drainage, a separate tributary to the Kuskokwim River. The proposed airstrip, some material sites, a short northern section of the mine access road, and the central portion of the proposed BTC Road would be located in the headwaters of the Iditarod River, which flows northwest into the Yukon River. Additional tributaries to the Kuskokwim River, such as the Owhat River, would be crossed by the western portion of the BTC Road (see Section 3.5, Surface Water Hydrology).

Surface water quality in the Crooked Creek watershed is described in detail in Section 3.7.2.1.1. Sampling location GETM2 provides baseline water quality data for the Getmuna Creek, a major tributary to Crooked Creek that would be crossed by the mine access road. Factors affecting water quality in Crooked Creek, such as trends correlated to mineralized areas, may be similar for Jungjuk Creek, and the Iditarod and Owhat rivers. While the specific prospect targeted by the proposed mine does not extend south of American Creek, bedrock types and mineralization similar to those that host the Donlin Gold prospect extend throughout the proposed mine access road and about half of the potential BTC Road (Figure 3.7-4) (Bundtzen and Laird 1991).

Jungjuk Creek has clear water, a moderate gradient, and a gravel/cobble-dominated substrate. Beaver activity is present in much of the drainage basin (OtterTail Environmental Inc. 2012f, Recon 2011b). Likewise, Owhat River has a gravel and cobble streambed. The Iditarod River and a number of smaller drainages crossed by the western part of the BTC route contain mucky stream bottoms in permafrost thermokarst terrain (RECON 2007c).

Dutch Harbor Fuel Storage Facility

As described in Section 3.2, Soils, a number of ADEC contaminated sites have been documented in Dutch Harbor near potential locations of the proposed fuel storage facility expansion. These are related to numerous historical spills in and near the harbor from decades of fuel shipping and handling in the area.

3.7.2.1.3 PIPELINE

The proposed pipeline would be located in a sparsely populated portion of Alaska and as such, limited water quality data are available to characterize surface water quality in the vicinity of the potential pipeline corridor. Under all options, the proposed corridor crosses more than 400 individual streams ranging in size from small, intermittent channels to large glacially fed river systems (CH2MHill 2011c). Multiple data sources were queried for surface water quality information to provide a generalized characterization for the area, including the USGS National Water Information System (NWIS).

Nine NWIS surface water monitoring stations occur within watersheds crossed by the proposed pipeline route. These stations are located in the following watersheds (Section 3.5, Surface Water Hydrology, Figures 3.5-17 and 3.5-18):

- Susitna River–Frontal Cook Inlet
- Lower Skwentna River
- George River
- Crooked Creek

The NWIS data span from 1959 to 2007, and are a compilation of many discrete water quality analyses. Therefore, not all of the same water quality data are available for all of the sampling locations. Similarly, not all areas were sampled during the same time periods. In addition, most of the NWIS sampling locations were not located adjacent to the proposed pipeline corridor, and the data are not necessarily representative of surface water quality along the potential pipeline route. For these reasons, the NWIS water quality data are not considered further as a source of information about baseline water quality along the proposed pipeline route.

Water quality field measurements (including pH, temperature, conductivity, color, and turbidity) were collected during the Donlin Gold aquatic resources surveys at about half of the stream crossings along the proposed pipeline route in 2010. The results are summarized in (Table 3.7-8) and described below from OtterTail (2012c).

Values of pH ranged widely from 3.6 to 9.6 for the 394 measurements. The arithmetic average was 7.55, and the pH value equivalent to the $-\log_{10}$ arithmetic average of hydrogen ion concentrations was 6.05. A review of the individual pH data indicate that the highest and lowest values are likely data outliers, as 95 percent of the measured pH values are between 6.0

and 9.0. The most stringent ADEC WQS (fresh water supply – aquaculture) for pH are 6.5-8.5 or not more than 0.5 pH unit from natural conditions; 74 percent of the measured values are within the range of 6.5-8.5.

The average of 434 temperature measurements was 45.7°F (7.6°C). ADEC WQS (fresh water supply – aquaculture) for temperature range from a maximum of 13°C for spawning and incubation areas, to no more than 20°C at any time. Of the total number of temperature measurements collected, 32 (or 7.4 percent) exceeded the most stringent WQS of 13°C, and none exceeded 20°C.

Baseline turbidity measurements along the proposed pipeline route ranged from 0.06 to 500 Nephelometric Turbidity Units (NTU). The most stringent ADEC WQS for fresh water streams (water supply for drinking and contact recreation) specifies that turbidity may not exceed 5 NTU above natural conditions when the natural turbidity is 50 NTU or less, and may not have more than 10 percent increase in turbidity when the natural turbidity is more than 50 NTU, not to exceed a maximum increase of 25 NTU.

Some of the highest turbidity values were measured in the Middle Fork Kuskokwim River and in two of its tributaries (Figure 3.7-5) with baseline turbidity values ranging from 58.4 to 165 NTU, values which are probably related to naturally high levels of glacial silt or fine-grained soils in the area. Surficial deposits and soil/bank erosion conditions that provide an indication of possible sources of naturally high turbidity levels are described in Sections 3.1, Geology and 3.2, Soils, respectively.

There are no WQS for conductivity. Color was recorded at some stream crossings based on qualitative descriptors that are not comparable to WQS color units.

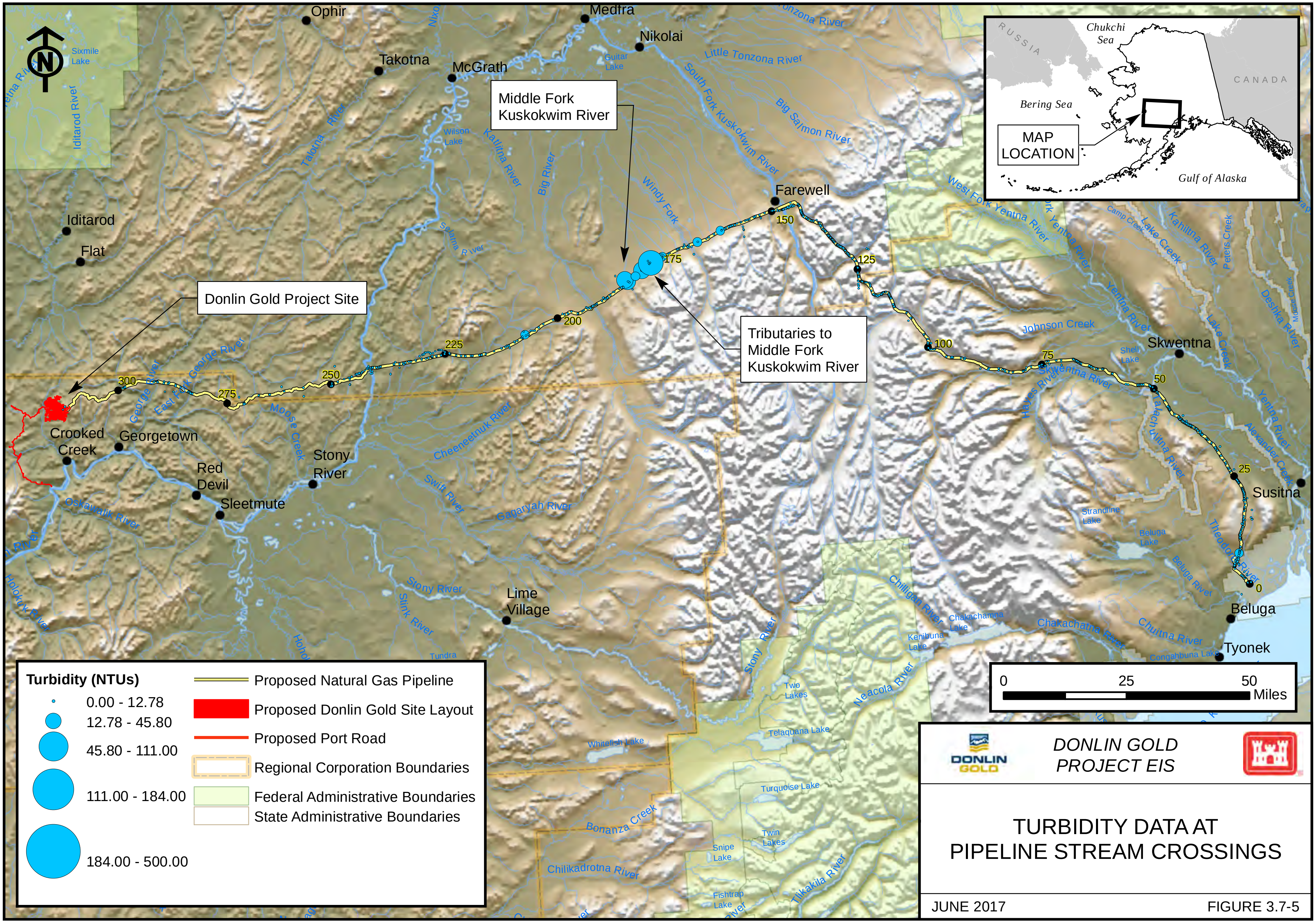
Table 3.7-8: Summary of Surface Water Quality Field Measurements along Proposed Pipeline Route

Parameter	Units	Number of Measurements (n)	Range of Measured Values	Arithmetic Average
pH	pH units	394	3.6 – 9.6	7.55[a]
Turbidity	NTU	182	0.06 - 500	12.2
Conductivity	µS/cm	447	1 - 2000	244
Water Temperature	Degrees C	434	1.17 – 16.6	7.62

Notes:

[a] average – $\log_{10}[H^+]$ = 6.05

Source: OtterTail 2012c.



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TURBIDITY DATA AT PIPELINE STREAM CROSSINGS

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FIGURE 3.7-5

3.7.2.2 GROUNDWATER QUALITY

3.7.2.2.1 MINE SITE

Background

Groundwater in the vicinity of the Mine Site occurs in three main hydrogeologic environments: an alluvium unit, a colluvial layer, and an underlying bedrock unit (ARCADIS 2012b). The alluvium unit is primarily associated with Crooked Creek (BGC 2007a). The relatively thin colluvial layer is present along gently sloping valley walls and valley bottoms, and the bedrock unit consists primarily of siltstones, shales, graywacke, and intrusives (ARCADIS 2012b). A complete description of geology and groundwater occurrence at the Mine Site, including discussion of geologic ages and aquifer properties of the alluvium, colluvium, and bedrock units, is provided in Section 3.1, Geology and Section 3.6, Groundwater Hydrology.

The main sources of recharge to the groundwater system are rainfall and snowmelt, and to a lesser extent, infiltration from stream beds. Groundwater discharge occurs through creeks, streams, and gulches, and also through evapotranspiration. In general, the level of the water table mimics the surface topography, with the greatest depths to groundwater occurring in upland areas and shallowest depths to groundwater in the valley bottoms and low-lying areas (BGC 2007a).

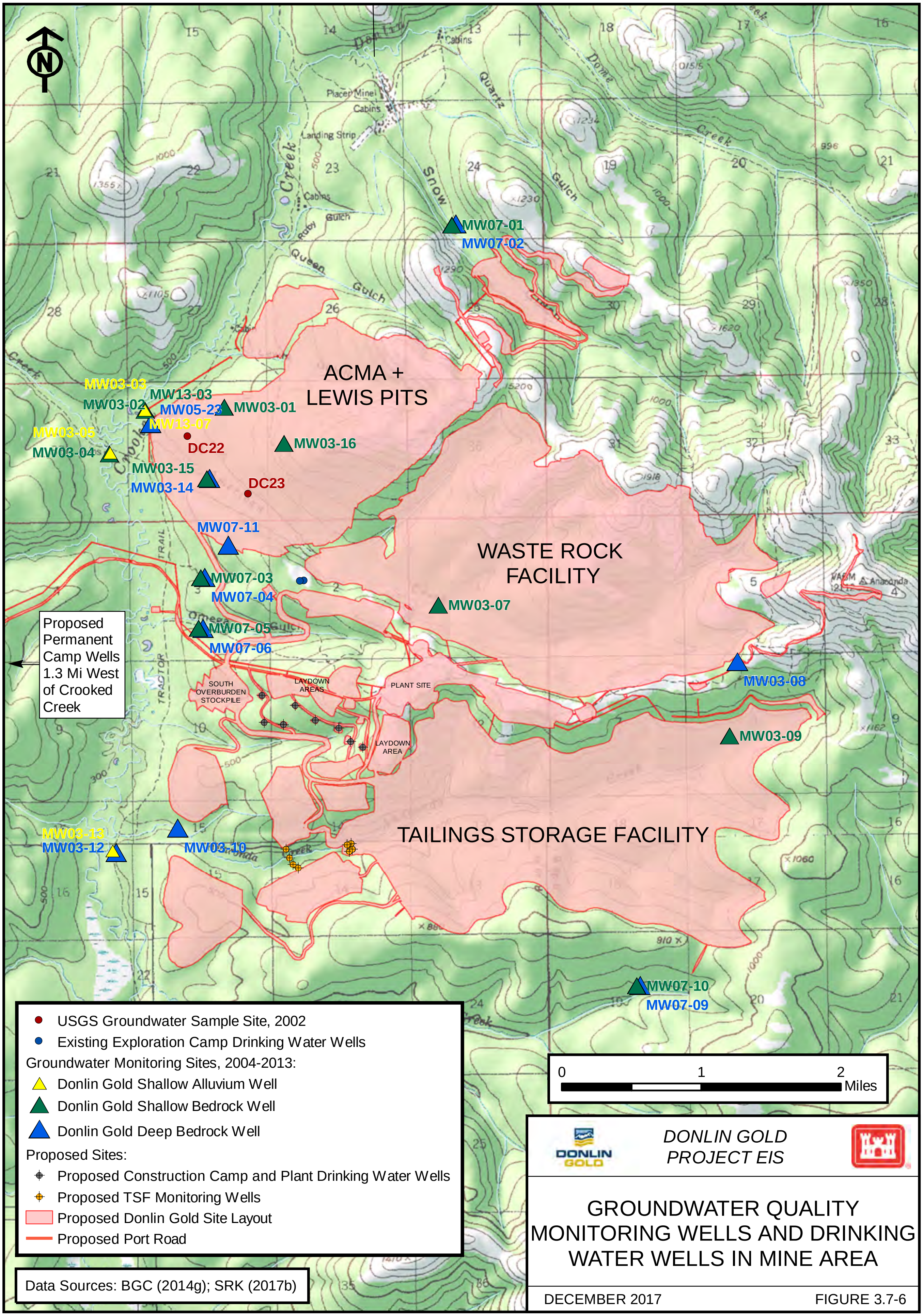
Donlin Gold Groundwater Quality Characterization Program

Sampling and Analytical Program

A total of 27 wells comprise the groundwater monitoring network at the Donlin Gold proposed project site. Monitoring locations were established to characterize the groundwater system both upgradient and downgradient of each major proposed mine facility. Sixteen monitoring wells (MW03 series) were installed in late 2003, and 14 of these were initially sampled during the first quarter of 2004 and quarterly thereafter. Ten additional wells (MW07 series) were installed in late 2007, and one well was not sampled due to an ice plug. These MW07 series wells have been sampled quarterly beginning in 2008 (ARCADIS 2012b). Additional wells, designated as MW05 and MW13 series, were completed as observation points for pumping tests and sampled during the tests (BCG 2014f). Groundwater quality monitoring data were collected through the third quarter of 2013 (SRK 2017b).

For the purposes of this analysis, wells with quarterly monitoring data were grouped into alluvial and bedrock categories. For bedrock wells, these have been further subdivided into shallow and deep wells, and those that are located within the pit area compared to outside the pit area. Shallow wells include those less than about 75 feet deep; these range in depth from roughly 15 to 30 feet for the alluvial wells, and 20 to 75 feet for the shallow bedrock wells. Deep wells include those completed in bedrock greater than 75 feet deep; these range in depth from 75 to more than 500 feet (BGC 2007a, 2011d, SRK 2017b).

Table 3.7-9 summarizes the well locations, related mine facilities, and sampling information for the monitoring wells that are part of Donlin Gold's groundwater quality sampling program, and Figure 3.7-6 shows the well locations.



Data Sources: BGC (2014g); SRK (2017b)

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GROUNDWATER QUALITY MONITORING WELLS AND DRINKING WATER WELLS IN MINE AREA

DECEMBER 2017

FIGURE 3.7-6

Table 3.7-9: Groundwater Quality Monitoring Well Locations

Well ID	UTM Zone 4, NAD83		Category	Bedrock Wells: Within or Outside of Pit Area	Target	Number of Sample Events
	Easting	Northing				
MW03-01	539999	6879366	Shallow Bedrock	Within	Downgradient from mineralized zone	32
MW03-02	539144	6879164	Shallow Bedrock	Within	Downgradient from pit / upstream from American Creek	33
MW03-03	539163	6879178	Alluvium	-	Downgradient from pit / upstream from American Creek	30
MW03-04	538680	6878830	Shallow Bedrock	Within	Downgradient from pit / downstream from American Creek	33
MW03-05	538685	6878841	Alluvium	-	Downgradient from pit / downstream from American Creek	28
MW03-07	542470	6877084	Shallow Bedrock	Outside	Downgradient from the proposed waste rock facilities	32
MW03-08	545875	6876430	Deep Bedrock	Outside	Upgradient from the proposed waste rock facilities	35
MW03-09	545829	6875574	Shallow Bedrock	Outside	Upgradient from the proposed tailings facility	33
MW03-10	539423	6874503	Deep Bedrock	Outside	Downgradient from proposed tailings facility	33
MW03-12	538714	6874223	Deep Bedrock	Outside	Downgradient from proposed tailings / downstream from Anaconda Creek	30
MW03-13	538719	6874245	Alluvium	-	Downgradient from proposed tailings / downstream from Anaconda Creek	30
MW03-14	539782	6878537	Deep Bedrock	Within	Existing groundwater quality in the mineralized zone	14
MW03-15	539797	6878539	Shallow Bedrock	Within	Existing groundwater quality in the mineralized zone	29
MW03-16	540692	6878952	Shallow Bedrock	Within	Upgradient from the proposed pit	21
MW05-23	539151	6879175	Deep Bedrock	Within	Downgradient from pit	3
MW07-01	542627	6881469	Shallow Bedrock	Outside	Groundwater in Snow Gulch watershed	25
MW07-02	542627	6881477	Deep Bedrock	Outside	Groundwater in Snow Gulch watershed	25
MW07-03	539729	6877401	Shallow Bedrock	Outside	Downgradient from process facility – Omega North	21
MW07-04	539734	6877400	Deep Bedrock	Outside	Downgradient from process facility – Omega North	25
MW07-05	539703	6876817	Shallow Bedrock	Outside	Downgradient from process facility – Omega Gulch	21
MW07-06	539714	6876812	Deep Bedrock	Outside	Downgradient from process facility – Omega Gulch	23
MW07-07	540910	6872313	Deep Bedrock	Outside	Downgradient of potential water diversion from Anaconda to Crevice Creek	24
MW07-09	544752	6872692	Deep Bedrock	Outside	Upgradient of potential water diversion from Anaconda to Crevice Creek	25
MW07-10	544752	6872692	Shallow Bedrock	Outside	Upgradient of potential water diversion from Anaconda to Crevice Creek	25
MW07-11	540044	6877772	Deep Bedrock	Within	Upgradient of pit and American Creek	1
MW13-03	539086	6879331	Deep Bedrock	Within	Within Crooked Creek floodplain	3
MW13-07	539087	6879333	Alluvium	-	Within Crooked Creek floodplain	3

Source: BGC (2007a, 2014g); ARCADIS (2012b); SRK (2017b)

Study Results

A summary of groundwater quality data collected during Donlin Gold's groundwater quality sampling program is provided in Tables 3.7-10 and 3.7-11 for the alluvial and bedrock wells, respectively. Table 3.7-11 further subdivides the bedrock data by location and depth. The tables summarize the groundwater quality data as a range of arithmetic means for individual wells in each category. These are compared to the most stringent of ADEC WQS (2008a, 2012d) for both aquatic life and drinking water uses, as groundwater at the Mine Site would be used both for potable water supply and treated dewatering discharge.

The following analytes occurred at average concentrations above the ADEC WQS in at least one of the alluvial wells:

- Aluminum (total), arsenic (total), iron (total), lead (total), and manganese (total).

The following analytes occurred at average concentrations above the ADEC WQS in at least one of the bedrock wells:

- Aluminum (total), antimony (total), arsenic (total), barium (total), copper (total), iron (total), lead (total), manganese (total), mercury (total), TDS, fluoride, and pH.

As noted above, there are a number of analyte exceedances of WQS that occur in bedrock groundwater, but not alluvial groundwater. There are also differences in groundwater composition between bedrock wells in the vicinity of the ore body compared to bedrock wells outside this zone (BGC 2007a, SRK 2017b). For example, aluminum, arsenic, barium, and zinc are notably higher in the vicinity of the proposed pit than outside the mineralized area (Table 3.7-11). In addition, within the vicinity of the ore body, there is some variation of water quality with depth. For example, arsenic is notably higher in the deep bedrock wells (highest mean $[As]_{total}$ = 1.97 mg/L) than the shallow alluvial or bedrock wells (highest mean $[As]_{total}$ = 0.212 mg/L).

U.S. Geological Survey Study of Crooked Creek Watershed

Two groundwater samples, DC22 and DC23, were analyzed during the USGS investigation in 2002 (Mueller et al. 2003). The samples were collected from wells on the lower southwestern slope of the ridge that separates Lewis Gulch from American Creek (Figure 3.7-6). The two samples are notably different from the USGS surface and seep waters in that they have higher pH (7.6-8.4), lower dissolved oxygen (0.02 to 0.49 mg/L), and higher specific conductance (384 to 770 µS/cm) relative to samples collected from surface waters and seeps.

Table 3.7-10: Groundwater Quality Summary for Alluvial Mine Site Wells

Analyte	Units	Range of Average Concentrations[3] for Individual Wells	Maximum Detected Sample Location	Alaska Most Stringent Applicable Water Quality Criterion	Max Detects Exceeds Alaska Criterion
Calcium, Dissolved	mg/L	22.1-41.3	MW03-05	-	-
Magnesium, Dissolved	mg/L	4.90-10.2	MW03-03	-	-
Potassium, Dissolved	mg/L	0.632-0.907	MW03-03	-	-
Sodium, Dissolved	mg/L	4.37-11.0	MW03-03	-	-
Total Alkalinity (as $CaCO_3$)	mg/L	98.4-133	MW03-03	20 (min)[2b]	No
Bicarbonate	mg/L	98.4-133	MW03-03	-	-
Fluoride	mg/L	<0.031-0.102	MW03-03	1[2e]	No
Sulfate	mg/L	3.89-18.7	MW13-07	250[1]	No
Chloride	mg/L	0.614-1.33	MW03-03	230[2b]	-
Nitrite + Nitrate(as N)	mg/L	<0.031-0.202	MW03-13	10[2d]	No
Ammonia (as N)	mg/L	0.124-1.04	MW03-03	2.99[2h]	No
WAD Cyanide	µg/L	0.955-1.28	MW03-13	5.2[2g]	No
Mercury	ng/L	0.359-6.57	MW03-13	12[2b]	No
Aluminum, Dissolved	µg/L	6.14-208	MW03-13	-	-
Aluminum	µg/L	3.69-1,740	MW03-13	750[2a]	Yes
Antimony, Dissolved	µg/L	<0.31-<1.55	ND	-	-
Antimony	µg/L	0.219-1.23	MW03-05	6[2d]	No
Arsenic, Dissolved	µg/L	16.8-218	MW03-03	-	-
Arsenic	µg/L	16.6-212	MW03-03	10[2d]	Yes
Barium, Dissolved	µg/L	291-1,210	MW03-13	-	-
Barium	µg/L	287-1,400	MW03-13	2,000[2d]	No
Beryllium, Dissolved	µg/L	<0.13-0.0949	MW03-13	-	-
Beryllium	µg/L	0.0671-0.288	MW03-13	4[2d]	No
Boron, Dissolved	µg/L	7.35-10.8	MW03-03	-	-
Boron	µg/L	7.47-16.3	MW03-03	750[2e]	No
Cadmium, Dissolved	µg/L	<0.15-<0.75	ND	-	-
Cadmium	µg/L	0.0727-0.101	MW03-13	0.23[2a]	No
Chromium, Dissolved	µg/L	0.474-1.44	MW03-13	-	-
Chromium	µg/L	<0.62-2.77	MW03-13	100[2d]	No
Cobalt, Dissolved	µg/L	0.913-1.03	MW03-13	-	-
Cobalt	µg/L	0.649-0.722	MW03-13	50[2e]	No
Copper, Dissolved	µg/L	0.163-0.333	MW03-13	-	-
Copper	µg/L	-<0.31-1.82	MW03-05	7.7[2a]	No
Iron, Dissolved	µg/L	6,100-40,400	MW03-03	-	-
Iron	µg/L	6,130-42,200	MW03-13	1,000[2b]	Yes

Table 3.7-10: Groundwater Quality Summary for Alluvial Mine Site Wells

Analyte	Units	Range of Average Concentrations[3] for Individual Wells	Maximum Detected Sample Location	Alaska Most Stringent Applicable Water Quality Criterion	Max Detects Exceeds Alaska Criterion
Lead, Dissolved	µg/L	<0.062-0.514	MW03-13	-	-
Lead	µg/L	0.0586-2.47	MW13-07	2.4[2a]	Yes
Lithium, Dissolved	µg/L	2.11-2.89	MW03-13	-	-
Lithium	µg/L	2.29-3.43	MW03-13	2,500[2e]	No
Manganese, Dissolved	µg/L	1,320-1,830	MW13-07	-	-
Manganese	µg/L	1,220-1,800	MW13-07	50[2f]	Yes
Molybdenum, Dissolved	µg/L	<3.1-<15.5	ND	-	-
Molybdenum	µg/L	<3.1-<15.5	ND	10[2e]	No
Nickel, Dissolved	µg/L	0.517-1.18	MW13-07	-	-
Nickel	µg/L	0.789-1.86	MW03-13	43[2a]	No
Selenium, Dissolved	µg/L	<1.5-<7.5	ND	-	-
Selenium	µg/L	<1.5-<7.5	ND	5[2b]	No
Thallium, Dissolved	µg/L	<0.31-<1.55	ND	-	-
Thallium	µg/L	<0.31-<1.55	ND	1.7[2f]	No
Vanadium, Dissolved	µg/L	<6.2-<31	ND	-	-
Vanadium	µg/L	<6.2-4.70	MW03-13	100[2e]	No
Zinc, Dissolved	µg/L	2.57-18.0	MW13-07	-	-
Zinc	µg/L	5.10-39.2	MW03-13	100[2a]	No
Hardness ($CaCO_3$), Dissolved	mg/L	93.9-131	MW03-16	-	-
Total Dissolved Solids	mg/L	160-171	MW03-02	500[1]	No
Conductivity, Field	µS/cm	167-213	MW03-13	-	-
Dissolved Oxygen, Field	mg/L	3.6-5.5	MW03-02	-	-
Ox./Reduc. Pot.(ORP/eH), Field	mV	-59 - -36	MW07-10	-	-
pH, Field	pH Units	6.6-7.2	MW03-15	6.5-8.5[1]	No

Notes:

Most Stringent Applicable Water Quality Criteria are provided for the parameters for which water quality standards have been established. Standards have not been established for all parameters. Cases where there are no applicable standards are indicated by a dash (-).

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008.
2a Aquatic life for fresh water hardness-dependent criteria using a hardness value of 80.55 mg/L as $CaCO_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d). For Al, if pH≥7.0 and hardness ≥50, then 750 µg/L.
2b Aquatic life for fresh water (chronic) criteria.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water criteria.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life criteria (chronic)for free cyanide, measured as weak acid dissociable (WAD) cyanide.
2h Aquatic life for fresh water (chronic) criteria for ammonia based on 85th percentile of pH (7.85) and temperature (6.83°C) measured at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).

3 Arithmetic averages calculated for all observations using detected values and ½ of the method detection limit value when analyte not detected.

Source: SRK (2017b, Table A-7)

Table 3.7-11: Groundwater Quality Summary for Bedrock Mine Site Wells

Analyte	Units	Range of Average Concentrations^{3} for Individual Wells				Maximum Detected Sample Location	Alaska Most Stringent Water Quality Criterion	Max Detects Exceeds Alaska Criterion
		Within Pit Area		Outside Pit Area				
		Shallow	Deep	Shallow	Deep			
Major Cations								
Calcium, Dissolved	mg/L	9.57-40.2	2.31-63.4	15.0-40.5	7.89-81.9	MW03-12	-	-
Magnesium, Dissolved	mg/L	6.54-15.9	1.01-27.6	3.97-13.0	2.24-15.2	MW03-16	-	-
Potassium, Dissolved	mg/L	0.772-1.67	1.30-1.54	0.141-0.995	0.260-2.18	MW03-12	-	-
Sodium, Dissolved	mg/L	4.47-105	13.7-222	2.17-26.1	2.36-82.4	MW03-14	-	-
Major Anions								
Total Alkalinity (As $CaCO_3$)	mg/L	130-287	202-448	56.1-203	75.5-308	MW03-14	20 (min)^{2b}	-
Bicarbonate	mg/L	130-286	202-429	56.1-203	75.5-308	MW03-14	-	-
Fluoride	mg/L	0.0736-0.147	<0.031-2.22	0.0348-0.124	0.0334-0.171	MW03-14	1^{2e}	Yes
Sulfate	mg/L	4.20-24.9	8.90-85.3	1.79-60.5	0.706-44.6	MW03-16	250^{1}	No
Chloride	mg/L	0.564-0.797	0.834-5.88	0.525-0.720	0.464-0.709	MW05-23	230^{2b}	No
Nutrients								
Nitrite + Nitrate(as N)	mg/L	0.0253-0.0708	0.0205-0.0271	0.0253-0.639	0.0287-0.294	MW07-10	10^{2d}	No
Ammonia (as N)	mg/L	0.0790-1.05	0.208-0.850	0.0243-0.335	0.0205-0.814	MW03-02	2.99^{2h}	No
Cyanide								
WAD Cyanide	µg/L	0.934-1.08	0.946-1.04	0.810-1.02	0.812-1.01	MW03-02	5.2^{2g}	No
Metals								
Mercury	ng/L	0.406-3.44	<0.5-40.6	1.92-24.2	0.393-3.87	MW07-01	12^{2b}	Yes
Aluminum, Dissolved	µg/L	<6.2-5.80	3.95-310	<6.2-35.2	3.11-13.6	MW05-23	-	-
Aluminum	µg/L	3.38-532	<6.2-5,050	12.6-965	14.2-454	MW05-23	750^{2a}	Yes
Antimony, Dissolved	µg/L	<0.31-0.211	0.237-55.2	0.190-35.0	0.214-30.9	MW05-23	-	-
Antimony	µg/L	0.183-0.215	<0.31-30.0	0.181-35.8	0.220-31.6	MW07-01	6^{2d}	Yes
Arsenic, Dissolved	µg/L	10.8-223	<2.5-1,870	1.16-174	2.15-143	MW03-16	-	-
Arsenic	µg/L	11.7-212	<2.5-1,970	1.17-181	1.18-146	MW03-16	10^{2d}	Yes
Barium, Dissolved	µg/L	39.2-1,300	33.2-1,270	45.4-265	36.5-6,170	MW03-02	-	-
Barium	µg/L	38.8-1,370	40.1-1,290	45.8-265	38.2-6,350	MW03-12	$2{,}000^{2d}$	Yes
Beryllium, Dissolved	µg/L	<0.13-0.0837	<0.13-<0.13	<0.13-0.0705	<0.13-0.0836	MW03-02	-	-
Beryllium	µg/L	<0.13-0.0907	0.0917-0.467	0.0678-0.0904	0.0718-0.0952	MW05-23	4^{2d}	No
Boron, Dissolved	µg/L	10.5-119	13.7-185	0.590-13.0	6.20-72.1	MW03-14	-	-
Boron	µg/L	10.8-121	19.5-186	5.84-17.7	6.11-51.5	MW03-14	750^{2e}	No
Cadmium, Dissolved	µg/L	<0.15-<0.15	<0.05-<0.15	<0.15-0.0818	<0.15-0.0961	MW07-04	-	-
Cadmium	µg/L	<0.15-0.0790	<0.05-0.167	0.0811-0.0849	0.0753-0.100	MW03-10	0.23^{2a}	No
Chromium, Dissolved	µg/L	<0.62-<0.62	<0.31-5.85	0.325-0.536	<0.62-0.475	MW05-23	-	-

Table 3.7-11: Groundwater Quality Summary for Bedrock Mine Site Wells

Analyte	Units	Range of Average Concentrations[3] for Individual Wells				Maximum Detected Sample Location	Alaska Most Stringent Water Quality Criterion	Max Detects Exceeds Alaska Criterion
		Within Pit Area		Outside Pit Area				
		Shallow	Deep	Shallow	Deep			
Chromium	µg/L	0.374-1.30	<0.31-1.49	0.485-1.19	0.329-0.930	MW03-14	100[2d]	No
Cobalt, Dissolved	µg/L	<1.2-0.917	<1.2-7.49	0.630-0.813	<1.2-0.949	MW05-23	-	-
Cobalt	µg/L	<1.2-0.854	0.663-2.68	0.625-1.15	0.622-1.18	MW05-23	50[2e]	No
Copper, Dissolved	µg/L	0.180-0.213	<0.31-4.73	0.164-1.47	0.175-0.773	MW03-14	-	-
Copper	µg/L	0.168-1.84	<0.31-9.13	0.285-6.31	0.254-2.24	MW05-23	7.7[2a]	Yes
Iron, Dissolved	µg/L	38.7-5,270	49.7-878	23.7-4,940	24.5-5,280	MW07-02	-	-
Iron	µg/L	272-5,330	<78-2,120	129-5,010	41.6-5,480	MW07-02	1,000[2b]	Yes
Lead, Dissolved	µg/L	0.0439-0.0512	0.0703-1.76	0.0327-0.0474	0.0327-0.0671	MW05-23	-	-
Lead	µg/L	0.0608-1.09	0.164-12.8	0.0549-1.16	0.0484-0.510	MW05-23	2.4[2a]	Yes
Lithium, Dissolved	µg/L	2.76-70.1	28.5-168	2.83-24.6	1.74-40.2	MW03-14	-	-
Lithium	µg/L	3.04-15.2	28.3-164	2.84-25.0	1.96-40.6	MW03-14	2,500[2e]	No
Manganese, Dissolved	µg/L	6.90-1,390	12.1-78	1.07-502	7.09-512	MW03-04	-	-
Manganese	µg/L	14.4-1,380	12.2-80.3	8.17-470	8.80-509	MW03-04	50[2f]	Yes
Molybdenum, Dissolved	µg/L	<3.1-<3.1	1.81-7.75	<3.1-<3.1	<3.1-1.64	MW05-23	-	-
Molybdenum	µg/L	<3.1-1.70	<3.1-9.48	<3.1-1.64	<3.1-1.64	MW05-23	10[2e]	No
Nickel, Dissolved	µg/L	0.487-0.617	<0.62-4.49	0.332-4.78	0.347-2.98	MW07-01	-	-
Nickel	µg/L	0.592-1.50	<0.62-11.6	0.508-4.90	0.489-3.79	MW05-23	43[2a]	No
Selenium, Dissolved	µg/L	<1.5-<1.5	<1.5-2.13	<1.5-<1.5	<1.5-<1.5	MW05-23	-	-
Selenium	µg/L	<1.5-0.787	0.790-1.68	<1.5-<1.5	0.8-1.00	MW05-23	5[2b]	No
Thallium, Dissolved	µg/L	<0.31-<0.31	<0.31-0.593	<0.31-<0.31	<0.31-<0.31	MW05-23	-	-
Thallium	µg/L	<0.31-0.202	<0.31-0.207	0.161-0.174	0.162-0.170	MW03-16	1.7[2f]	No
Vanadium, Dissolved	µg/L	<6.2-<6.2	<6.2-8.35	<6.2-<6.2	<6.2-<6.2	MW05-23	-	-
Vanadium	µg/L	<6.2-<6.2	4.00-12.2	<6.2-3.29	<6.2-3.37	MW05-23	100[2e]	No
Zinc, Dissolved	µg/L	1.92-7.53	4.67-519	1.66-20.2	1.81-38.8	MW05-23	-	-
Zinc	µg/L	1.42-8.39	3.69-1,400	1.95-27.7	2.21-50.1	MW05-23	100[2a]	Yes
General Water Quality Parameters								
Hardness ($CaCO_3$), Dissolved	mg/L	63.7-148	10.2-274	53.3-145	35.9-268	MW03-16	-	-
Total Dissolved Solids	mg/L	152-305	219-584	65.6-207	86.3-345	MW03-14	500[1]	Yes
Field Parameters								
Conductivity, Field	µS/cm	164-315	44-785	66-175	94-323	MW05-23	-	-
Dissolved Oxygen, Field	mg/L	3.1-5.5	0.1-4.2	2.7-8.7	2.5-9.9	MW03-08	-	-
Ox./Reduc. Pot.(ORP/eH), Field	mV	-40 - 24	-65 - -20	4.5-109	-20 - 108	MW07-10	-	-
pH, Field	pH Units	7.0-8.1	7.5-8.7	6.7-7.6	7.1-8.2	MW03-14	6.5-8.5[1]	Yes

Table 3.7-11: Groundwater Quality Summary for Bedrock Mine Site Wells

Analyte	Units	Range of Average Concentrations[3] for Individual Wells				Maximum Detected Sample Location	Alaska Most Stringent Water Quality Criterion	Max Detects Exceeds Alaska Criterion
		Within Pit Area		Outside Pit Area				
		Shallow	Deep	Shallow	Deep			

Notes:

Most Stringent Applicable Water Quality Criteria are provided for the parameters for which water quality standards have been established. Standards have not been established for all parameters. Cases where there are no applicable standards are indicated by a dash (-).

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used.

2a Aquatic life for fresh water hardness-dependent criteria using a hardness value of 80.55 mg/L as $CaCO_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d). For Al, if pH≥7.0 and hardness ≥50, then 750 µg/L.
2b Aquatic life for fresh water (chronic) criteria.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water criteria.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life criteria (chronic)for free cyanide, measured as weak acid dissociable (WAD) cyanide.
2h Aquatic life for fresh water (chronic) criteria for ammonia based on 85th percentile of pH (7.85) and temperature (6.83°C) measured at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).

3 Arithmetic averages calculated for all observations using detected values and ½ of the method detection limit value when analyte not detected.

Source: SRK (2017b, Tables A-5 and A-6)

Drinking Water Supply Wells

Existing camp drinking water supply wells at the mine site are described in Section 3.6.1.5.1 (Groundwater Use), and are shown on Figures 3.6-2 and 3.7-6. The camp wells were temporarily closed in March 2015, but are noted as seasonally active in the ADEC (2017f) Drinking Water Watch system as of Summer 2017. These wells draw from the deep bedrock aquifer, and the water is treated prior to use.

As described in Section 3.6.1.5.1 (Groundwater Use), the drinking water source protection area identified for these wells extends southeast beneath the ridge between American Creek and Omega Gulch (ADEC 2017g). The ADEC source water assessment for this system indicates a rating of medium susceptibility to contamination from the upgradient aquifer, and very high susceptibility to contamination at the wellhead / surface intake.

Existing groundwater quality in these wells prior to treatment is similar to data listed in Table 3.7-11 for deep monitoring wells outside of the pit area. This groundwater exceeds drinking water standards for several constituents: antimony, arsenic, barium, iron, and manganese. As described in Section 3.7.1.1 (Regulatory Framework), ADEC provides oversight of treatment and testing of the water to help maintain compliance with the SDWA. Based on ADEC (2017f) records from 1999 to 2015 of samples taken from points of use (e.g., treatment plant or distribution system), the water system has had several violations of coliform standards, and occasional violations of other constituents (volatile organics, metals, nitrate, fluoride, cyanide), with documentation in the record of these issues having been resolved and return to compliance achieved.

3.7.2.2.2 TRANSPORTATION CORRIDOR

Kuskokwim River

Groundwater along the Kuskokwim River exists in alluvial deposits that may be hundreds of feet thick. Surface water drainage is into the Kuskokwim River, which flows southwest, and on a regional scale, the ground water system tends to mimic the drainage patterns observed for surface water (Dorava 1994). The groundwater generally flows in the direction of topographic gradients to the valley bottoms, where it emerges in streams (Dorava 1994). Adjacent to the Kuskokwim River, shallow groundwater flows into and out of the riverbanks as the elevation of water in the river rises and falls (Dorava 1994). This flow of water into and out of the aquifer in response to changing stage of the river is termed "bank storage effects" (Figure 3.7-7). Such interactions between the groundwater and surface water have the potential to influence groundwater quality, such that the quality of the groundwater in areas adjacent to the Kuskokwim River is linked to water quality in the main stem of the river.

The use of groundwater resources for community water supply in villages along the Kuskokwim River is described in Section 3.6, Groundwater Hydrology. Water quality data for community wells in Aniak provide some regional context for these resources. Analyses of samples from more than 30 wells drilled in Aniak in 1980 indicated an iron content averaging 0.30 mg / L, a silica content ranging from 7 to 20 mg / L, and alkalinity as $CaCO_3$ ranging from 100 to 200 mg / L (Dorava 1994). The iron content in Aniak groundwater is often higher than the 0.3 mg / L secondary maximum contaminant level (MCL) specified by EPA for drinking water (EPA 2013j).

Water quality data from community water supply wells, tested from points of use (i.e., following treatment), is tracked by the ADEC (2017f) Drinking Water Watch program. Information on the Crooked Creek village water supply system, which draws from a well located about 10 miles downstream from the mine site, is provided in Section 3.6.1.5 (Groundwater Use). The drinking water source protection area identified by ADEC (2017g) for this well extends northwest and upslope of the hill west of the Crooked Creek airstrip (Figure 3.6-6). The ADEC source water assessment for this system indicates a rating of medium susceptibility to contamination from the upgradient aquifer, and very high susceptibility to contamination at the wellhead / surface intake (ADEC 2017f).

Based on ADEC (2017f) records from 1999 to 2017, water quality data from treatment and distribution points in the Crooked Creek village water system indicate a number of violations, most of which are documented as having been resolved and return to compliance achieved. Constituents that have exceeded standards in the past include PCBs, pesticides, nitrate, trihalomethanes, volatile organics, E. coli, coliform, chlorine, metals, radionuclides, fluoride, manganese, cyanide, and sampling and reporting requirements. Currently there are unresolved violations (from January-February 2017) only for PCBs / pesticides and E. coli.

Angyaruaq (Jungjuk) and Birch Tree Crossing Roads

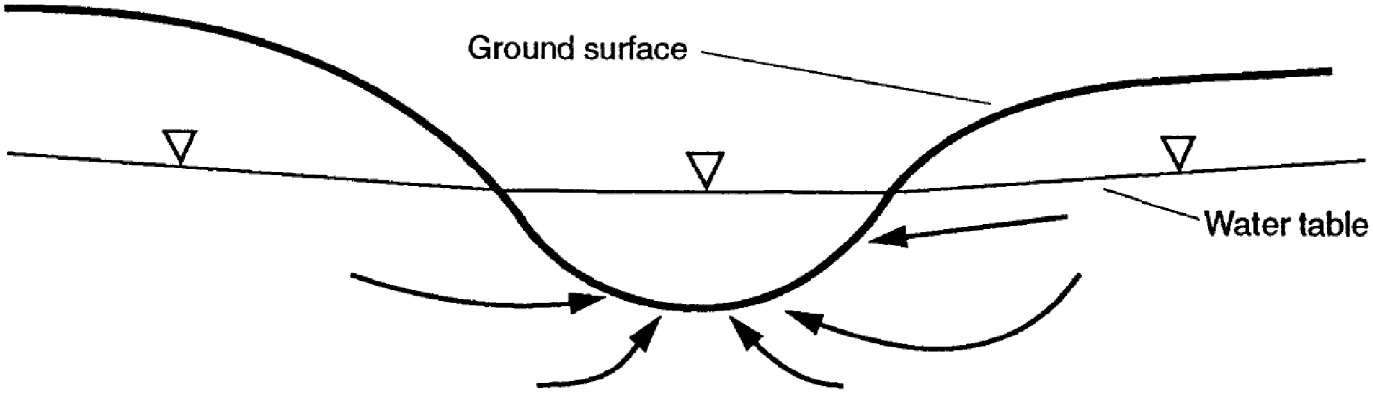
The location of the proposed Angyaruaq (Jungjuk) and BTC roads relative to drainage basins in the areas is described in Section 3.5, Surface Water Hydrology. In general, baseline groundwater quality throughout the area of the Angyaruaq (Jungjuk) Road and most of the BTC Road is expected to be very similar to groundwater quality in the Crooked Creek drainage basin due to similar settings and bedrock types, which is described in Section 3.7.2.2.1.

Dutch Harbor Fuel Storage Facility

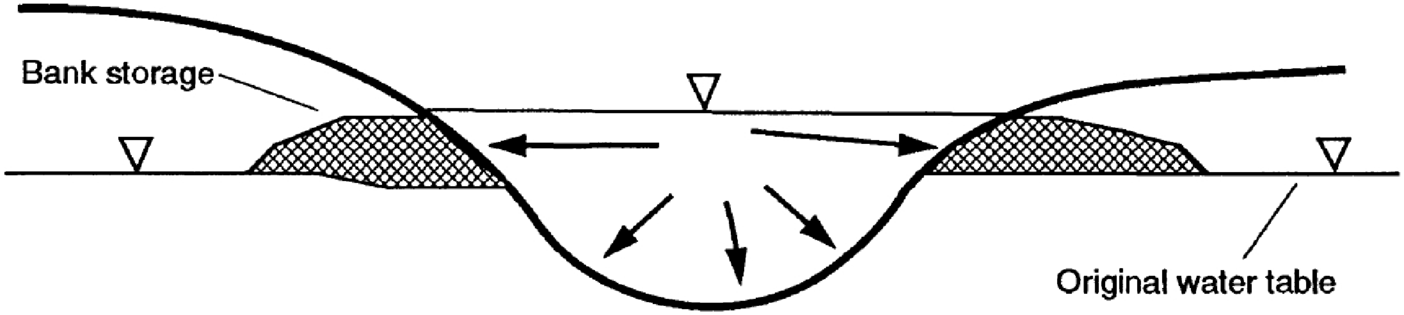
As described in Section 3.2, Soils, a number of ADEC contaminated sites have been documented in Dutch Harbor related to historical spills from existing tank farms and fuel handling in the area. Additional information on the location and footprint of the proposed facility would be needed to further assess site-specific groundwater quality conditions.

3.7.2.2.3 PIPELINE

The installation and operation of the proposed natural gas pipeline is not anticipated to affect groundwater quality outside the immediate vicinity of the trench; therefore, project-specific groundwater quality data were not collected in the proposed pipeline corridor.



LOW-FLOW CONDITIONS---FLOW IS INTO STREAM



HIGH-WATER CONDITIONS---FLOW IS OUT OF STREAM

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GROUNDWATER/SURFACE WATER INTERACTIONS DUE TO BANK STORAGE EFFECTS[1] ALONG KUSKOKWIM RIVER

JUNE 2017

FIGURE 3.7-7

Data Sources: Dorava (1994)

[1] Shallow groundwater flowing into and out of aquifer and riverbanks as water elevation in river rises and falls.

Known and potential uses of groundwater resources along the proposed pipeline are summarized in Section 3.6, Groundwater Hydrology. Groundwater wells for commercial, community, and private water supply are located near the proposed diesel pipeline route (Alternative 3B) in the Beluga to Tyonek area. Detailed USGS groundwater quality data are not available for the west side of Cook Inlet, and groundwater quality data are not publically available for wells associated with the Chugach Electric Association Power Plant at Beluga. Baseline groundwater quality in the vicinity of the eastern terminus of the proposed pipeline corridor has been characterized by analyzing samples from wells located at Ladd Landing, located approximately 5 miles south of Beluga (Riverside 2010). Groundwater quality from the glacial drift hydrostratigraphic unit in the Ladd Landing area is characterized by circumneutral pH values (average = 6.8), low hardness values (average = 20 mg/L as $CaCO_3$), and relatively low concentrations of total dissolved solids (average = 71 mg/L) (Riverside 2010). Selected groundwater samples from wells in the Ladd Landing area often exceeded the most stringent water quality criteria for total aluminum (average = 119µg/L), total iron (average = 1,000µg/L), and total manganese (average = 90 µg/L) (Riverside 2010). These exceedances did not correspond to elevated turbidity or TSS concentrations, though most exceedances were associated with dissolved aluminum below the detection limits of the analyses, indicating that aluminum in the samples was particulate (Riverside 2010).

3.7.2.3 SEDIMENT

There are no regulations established for chemical concentrations in sediment. Sediment Quality Guidelines (SQGs) recommended by ADEC (2013b) for use at contaminated sites include Threshold Effects Levels (TELs) and Probable Effects Levels (PELs) published in NOAA Screening Quick Reference Tables for both fresh and marine water sediment (Buchman 2008). TELs are concentrations below which adverse effects of benthic organisms are expected to occur rarely, and PELs represent concentrations above which effects are expected to be frequent.

Other sediment quality criteria used for comparison purposes in project studies (RWJ Consulting Inc. 2010a) include Canadian Interim Sediment Quality Guidelines (ISQG) for protection of freshwater aquatic life (Canadian Council of Ministers of the Environment [CCME] 2014), Washington State (2013) marine sediment quality criteria, and ADEC (2000, 2012a) cleanup levels for soils.

3.7.2.3.1 MINE SITE

The Donlin Gold environmental studies program has conducted several studies to characterize sediment quality in the vicinity of the Mine Site. In 1999 and 2006 through 2008, sediment samples were collected and analyzed to establish baseline concentrations of metals in sediment from the Crooked Creek watershed. Stream sediment grab samples were collected at nine sampling stations within the Crooked Creek watershed, several of which correspond to sample locations used for the collection of surface water samples. The sediment sample locations are shown in Figure 3.7-8, and reasons for the selection of each location are provided below:

> DCBO is located on Donlin Creek above all historic placer mining, and past, current, and proposed activities in the Crooked Creek drainage, and serves as a background control site.

FLAT is located on Flat Creek above the confluence with Crooked Creek. FLAT is outside the known mineralized area and serves as a background control site.

CCBW, located on Crooked Creek, is downstream of current placer mining activities, but upstream of the Mine Site.

GRSE is located on Grouse Creek upstream of the confluence with Crooked Creek. GRSE is outside the known mineralized area and serves as a background control site.

AMER, located on American Creek, drains the known mineralized area.

CCBO is below the known mineralized area on Crooked Creek upstream of the proposed tailing impoundment.

ANDA, located on Anaconda Creek, drains the proposed tailing impoundment area.

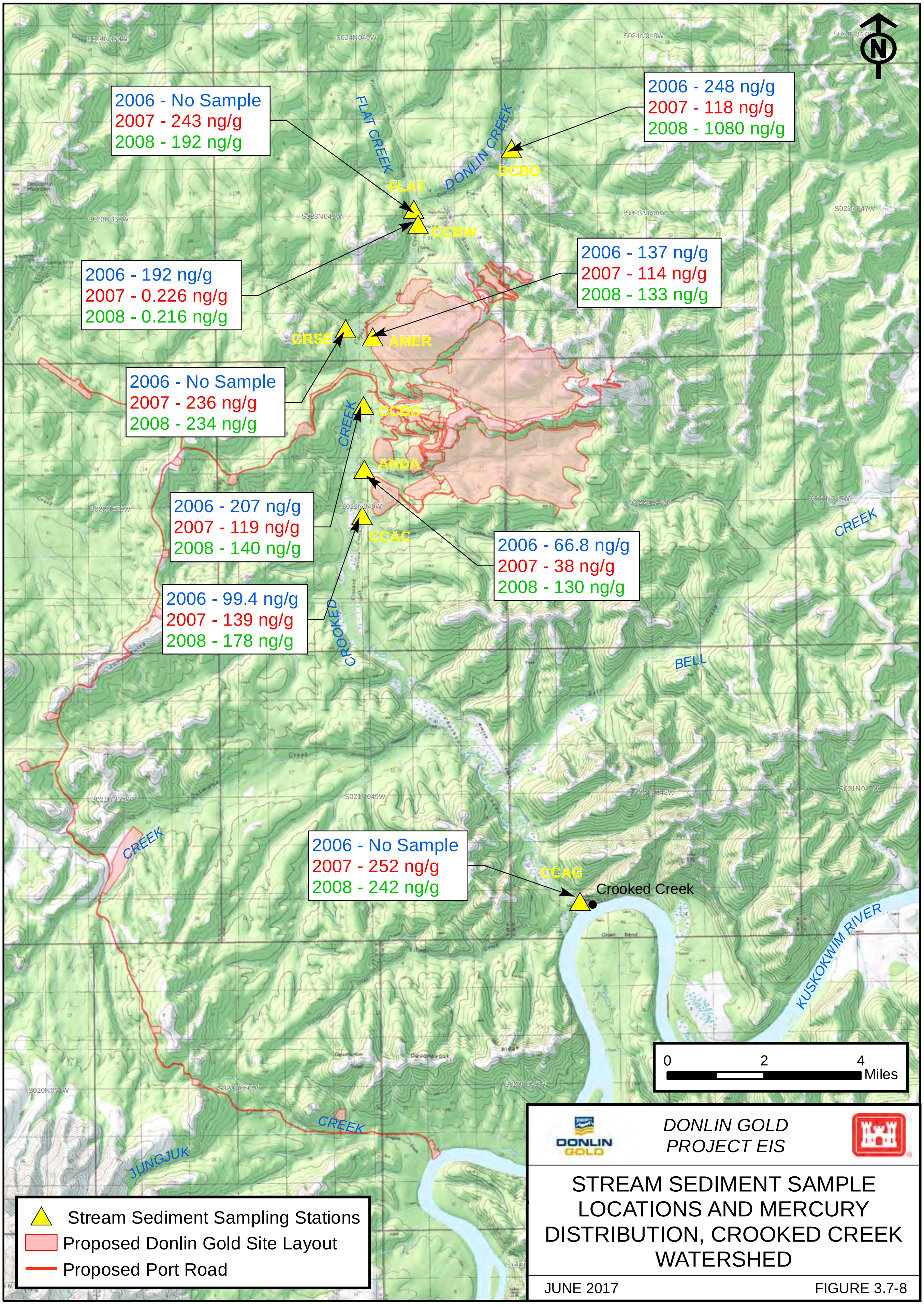
CCAC is located on Crooked Creek, downstream of all past, current, and proposed mining activity and known mineralization.

CCAG located on Crooked Creek just upstream of the USGS gauging station at the Village of Crooked Creek.

Stream sediment samples were collected from undisturbed stream sediment using one-time-use LEXAN sample tubes. All samples were collected in active water channels with the lowest velocity, at a depth of 0 to 6 inches below the stream bottom. Sediment samples were homogenized and digested using EPA Method SW3050B. Total metals were analyzed using EPA Method SW6020. Mercury and methylmercury samples were analyzed using EPA Methods 1631 and 1630, respectively, using cold vapor atomic fluorescence spectrometry (CVAFS) (ARCADIS 2008a).

The sample collection, handling, and analysis methods for the stream sediment samples were generally consistent among all sampling events, with the following minor deviations. Due to high water conditions during the 2007 sample event, sample locations GRSE and CCBW were collected downstream of the designated sampling location at the first favorable deposition location. Four metals analytes were added in 2008 that were not tested in 2007: bismuth, molybdenum, tin, and vanadium (ARCADIS 2008a).

Analytical results for total metals from the 2008 sediment sampling event are shown in Table 3.7-12. Concentrations of bismuth, cadmium, molybdenum, sodium, and tin were all below the detection limits (ARCADIS 2008a). While there are no regulatory criteria established for sediment (Section 3.7.1.2.3), TELs and PELs recommended for use by ADEC in its Contaminated Sites program (2013b) are provided in Table 3.7-12 for comparison purposes. Arsenic and nickel exceeded the TEL level in all samples, and mercury exceeded it in six of nine samples. Arsenic also exceeded the PEL level in five of nine samples, mercury exceeded it in one sample (DCBO) and nickel slightly exceeded it in one sample (CCAG).



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STREAM SEDIMENT SAMPLE LOCATIONS AND MERCURY DISTRIBUTION, CROOKED CREEK WATERSHED

JUNE 2017

FIGURE 3.7-8

Table 3.7-12: Summary of Total Metals Results for Proposed Mine Area Stream Sediment Samples, 2008

Station Sample ID[1]	DCBO	FLAT	GRSE	CCAG	CCAC	CCBW	ANDA	CCBO	AMER	TEL/PEL[2]
Analyte										
Aluminum (mg/Kg)	14,500	15,500	16,300	19,900	13,200	14,400	16,300	9,920	19,000	–
Antimony (mg/Kg)	ND	0.189	ND	0.269	0.287	0.199	ND	1.04	0.5	–
Arsenic (mg/Kg)	9.64	9.44	10.3	21.5	30.3	17.9	12.1	47.8	132	5.9/17
Barium (mg/Kg)	174	280	400	293	143	193	406	140	296	–
Beryllium (mg/Kg)	0.426	0.519	0.645	0.714	0.537	0.422	0.664	0.602	0.538	–
Bismuth (mg/Kg)	ND	ND	ND	ND	ND	ND	ND	ND	ND	–
Cadmium (mg/Kg)	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.596/3.53
Calcium (mg/Kg)	2,600	2,090	2,060	2,320	2,970	2,320	5,990	2,640	3,360	–
Chromium (mg/Kg)	26.9	19.9	21.9	30.3	26.9	22.9	20.9	21.8	24	37.3/90
Cobalt (mg/Kg)	14.3	15	15.5	15.5	14.2	11.8	13.9	13.1	14.4	–
Copper (mg/Kg)	13.3	17.3	20.7	22.8	16.8	15	25.4	26.1	22.4	35.7/197
Iron (mg/Kg)	34,800	19,000	16,000	21,200	35,000	27,800	27,300	40,600	37,500	–
Lead (mg/Kg)	5.78	4.98	4.89	5.55	6.86	5.9	8.55	6.82	9.88	35/91.3
Magnesium (mg/Kg)	6,240	2,870	2,030	3,580	5,000	4,750	4,090	3,860	4,770	–
Manganese (mg/Kg)	760	449	654	442	651	507	1,050	1,100	1,000	–
Mercury (ng/g)	1,080	192	234	242	178	216	130	140	133	174/486
Molybdenum (mg/Kg)	ND	ND	ND	ND	ND	ND	ND	ND	ND	–
Nickel (mg/Kg)	35.5	35.6	31.9	36.6	35.6	30.9	27.9	33.8	34.1	18/36
Potassium (mg/Kg)	769	597	569	759	773	646	861	644	965	–
Selenium (mg/Kg)	ND	ND	ND	ND	ND	ND	ND	ND	1.18	–
Silver (mg/Kg)	ND	ND	ND	ND	ND	ND	ND	ND	0.291	–
Sodium (mg/Kg)	ND	ND	ND	ND	ND	ND	ND	ND	ND	–
Thallium (mg/Kg)	0.0456	0.0577	0.0703	0.077	0.0408	0.0574	0.0961	0.0406	0.106	–
Tin (mg/Kg)	ND	ND	ND	ND	ND	ND	ND	ND	ND	–

Table 3.7-12: Summary of Total Metals Results for Proposed Mine Area Stream Sediment Samples, 2008

Station Sample ID[1]	DCBO	FLAT	GRSE	CCAG	CCAC	CCBW	ANDA	CCBO	AMER	TEL/PEL[2]
Solids (%)	68.7	56.4	51.5	43.8	74.6	62.6	36.3	77.9	50.7	–
Vanadium (mg/Kg)	40.5	38.5	40.7	44.1	37.9	33.5	43.5	36	39	–
Zinc (mg/Kg)	88.9	91.5	96.3	103	86.8	78.3	81.4	82.7	95.1	123/315

Notes:

1 Samples collected from Crooked Creek watershed at locations shown on Figure 3.7-8 in September 2008.
2 NOAA SQuiRT values for freshwater sediment (ADEC 2013d; Buchman 2008).

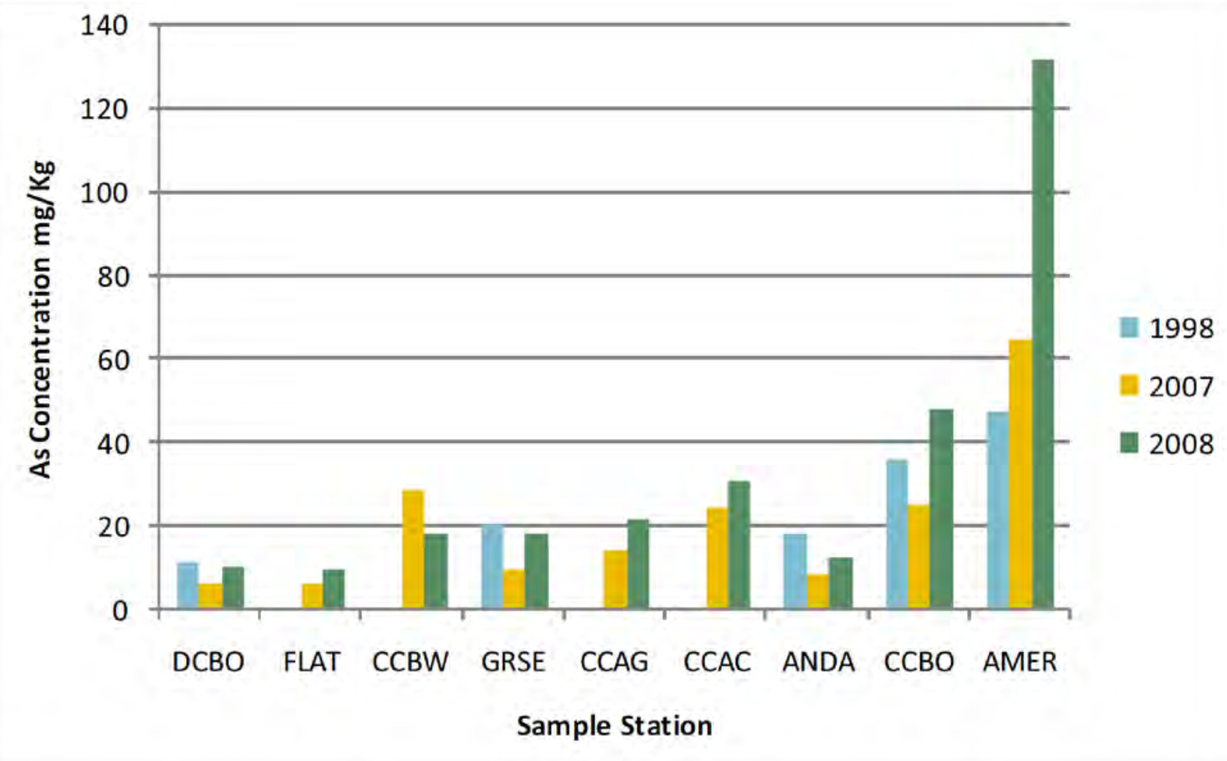
Abbreviations:

– = No value established
mg/Kg = milligrams per kilogram
ND = Not detected
ng/g = nanograms per gram (same as micrograms per kilogram (ug/Kg))
NOAA = National Oceanic and Atmospheric Administration
PEL = Probable Effects Level
TEL = Threshold Effects Level
SQuiRT = Screening Quick Reference Tables
Shaded values exceed TEL.

Source: ARCADIS 2008a.

The mean concentration of arsenic in the sediment was 32.3 mg/kg, and the highest concentration of arsenic was measured in the American Creek sample at a concentration of 132 mg/kg, almost an order of magnitude higher than the PEL (Table 3.7-12). A summary of arsenic results for three years (1998, 2007, and 2008), shown on Figure 3.7-9, indicates that concentrations are greatest in streams that drain the known mineralized area (ARCADIS 2007b, 2008). Elevated concentrations of arsenic at locations CCBO and AMER are likely attributable to the proximity of sampling sites to the ore body, and for the Crooked Creek samples, may also be related to the influence of past placer mining activities upstream (ARCADIS 2008a). In addition, variability is likely caused by the heterogeneous nature of the sediments, as the sediment data are based on analysis of bulk sediment samples and no size fractionation was performed prior to analysis. Sediment concentrations are known to vary with grain size, with fine-grained deposits typically exhibiting higher concentrations.

Concentrations of total mercury measured in sediment samples from the Crooked Creek watershed during the 2006-2008 sampling events ranged from 38 to 1,080 ng/g dry weight, as shown on Figure 3.7-8 and Figure 3.7-10. The highest of these was located at station DCBO on Donlin Creek between the Ophir and Dome creek confluences upstream of historical placer mining in the area. It is possible that mineralization trends to the northeast of the Mine Site are contributing to the elevated mercury concentrations at this location (see Section 3.1, Geology). Based on a review of available published data, the average mercury concentration in regional stream sediment from the USGS Sleetmute quadrangle, which covers an area of about 7,000 square miles south and east of the Mine Site, was reported to be 100 ng/g (Gray et al. 1997b; Miller et al. 1998; Theodorakos et al., 1992). Most of the proposed mine area results are higher than this regional average (ARCADIS 2007c).



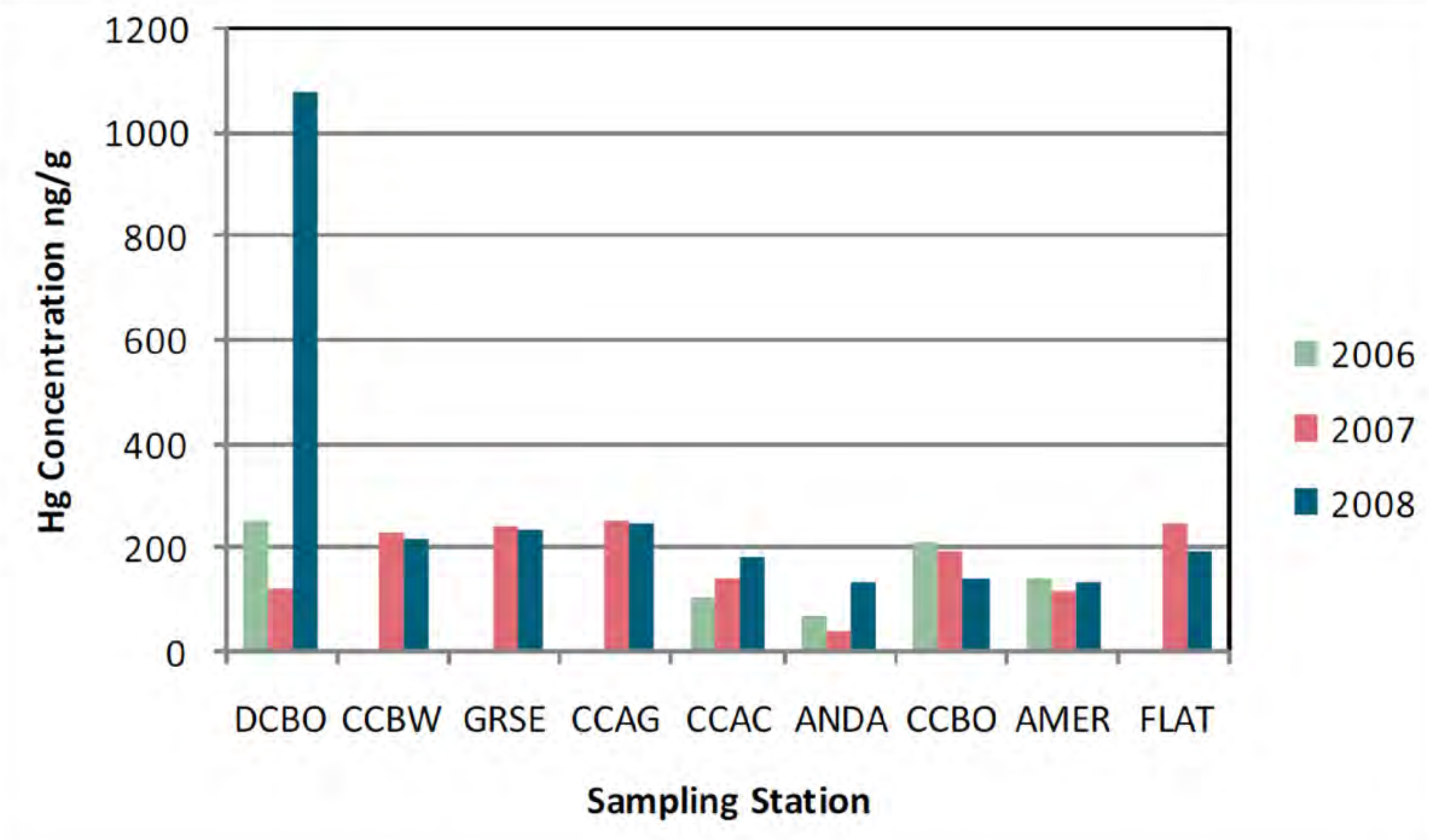
Data Sources: Arcadis 2007a, 2008

DONLIN GOLD
PROJECT EIS

ARSENIC CONCENTRATIONS IN STREAM SEDIMENT, CROOKED CREEK WATERSHED

JUNE 2017

FIGURE 3.7-9



Data Sources: Arcadis 2007b, 2008

Hg = mercury
ng/g = nanograms per gram (same as ug/Kg)

DONLIN GOLD PROJECT EIS

MERCURY CONCENTRATIONS IN STREAM SEDIMENT, CROOKED CREEK WATERSHED

JUNE 2017

FIGURE 3.7-10

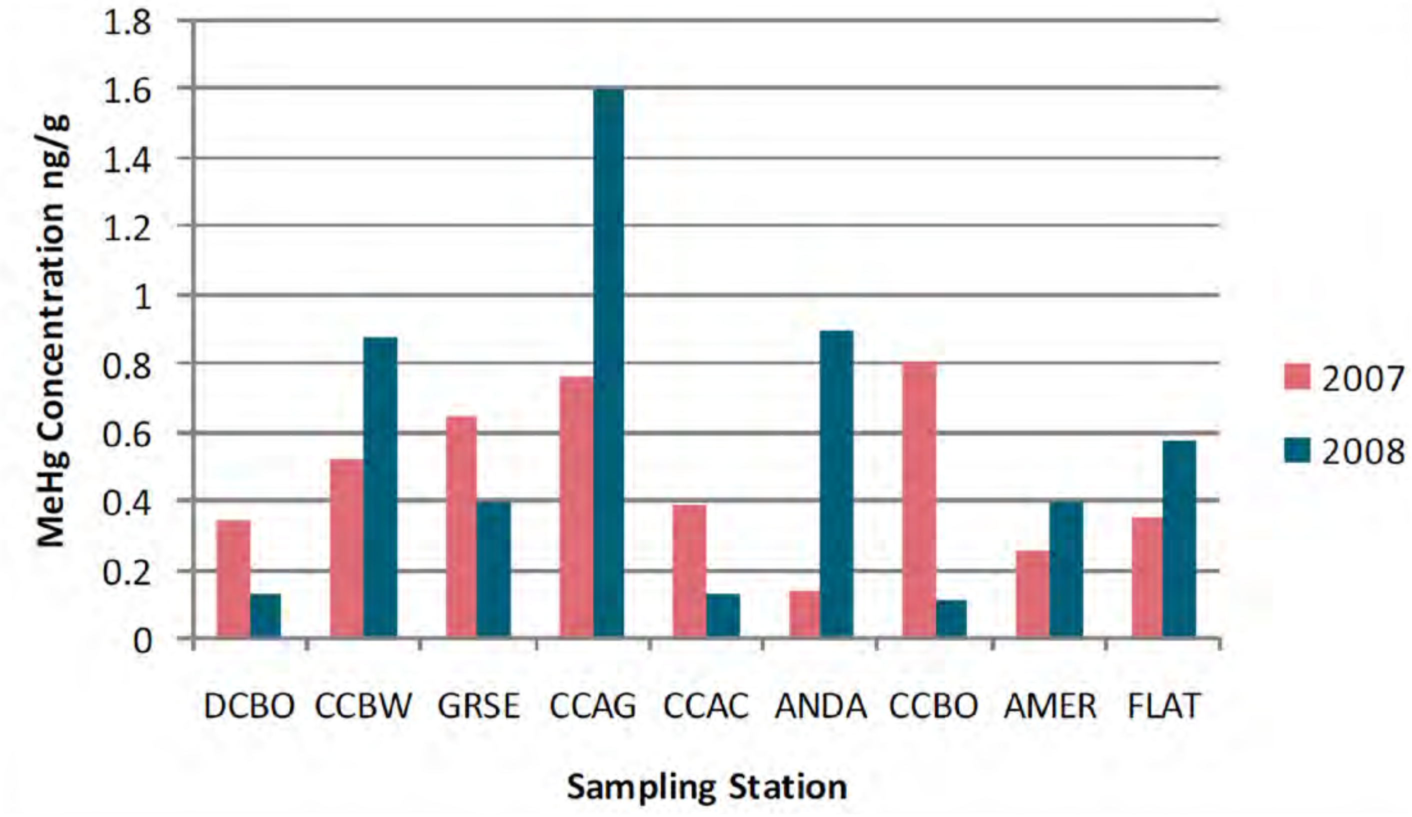
Methylmercury was measured in sediment samples collected from the Crooked Creek watershed during the 2007 and 2008 sediment studies. Methylmercury is more toxic than elemental mercury due to its ability to cross cell membranes and interact with biological systems (Section 3.7.2.1.1 and Figure 3.7-2). Methylmercury concentrations ranged from 0.111 ng/g to 1.60 ng/g dry weight in samples collected during the 2007 and 2008 sampling events (Figure 3.7-11). Differences in methylmercury concentrations between the two sampling events were most pronounced at CCAG and ANDA which more than doubled in 2008, and at CCBO which dropped substantially in 2008. As described above, no size fractionation was performed prior to analysis; thus, the wide range of methylmercury results may be indicative of grain size heterogeneity in the samples (ARCADIS 2008a). There are no sediment quality criteria specific to methylmercury.

The impact of mercury on aquatic systems may depend on the amount that is methylated. Mercury methylation requires inorganic mercury and methylating bacteria. The predominant (though not exclusive) methylators of mercury are sulfate-reducing bacteria, which require the presence of anoxic conditions, sulfate, and an organic carbon source. Sediment concentrations of methylmercury are often greater than those measured in aqueous samples because sediments are more likely to provide the anoxic conditions and electron acceptors requisite for the activity of sulfate-reducing bacteria responsible for mercury methylation; however, concentrations of methylmercury in the water, not sediment, are generally accepted as the key factor that determines the concentrations and accumulation of mercury in biota (Morel et al. 1998).

3.7.2.3.2 TRANSPORTATION CORRIDOR

Donlin Gold Baseline Sediment Sampling Program

The primary source of information related to sediment quality in the Kuskokwim River transportation corridor is the baseline sediment sampling program conducted by Donlin Gold, which involved collection at eight stations located along the Kuskokwim River, from above Crooked Creek to the mouth of the Kuskokwim, during August 2007 and June 2010 (RWJ Consulting Inc. [RWJ] 2008a, 2010a). Comparable stations were used for both sampling events, located (from downriver to upriver) at Helmick Point, Bethel Dock, upriver of Bethel (near Straight Slough), Birch Tree Crossing, Aniak, Jungjuk Creek, Crooked Creek, and George River (Figure 3.7-12). As described in RWJ 2010a, these sites are located in the main channel of the Kuskokwim River. The “Crooked Creek” site is located in the Kuskokwim River at its confluence with Crooked Creek, and so it is not comparable to Crooked Creek tributary sampling summarized in the previous section. A total of 299 and 390 samples were collected in 2007 and 2010, respectively (including QA samples). All samples were analyzed for trace metals, hydrocarbons, and various organics.



Data Sources: Arcadis 2007a, 2008

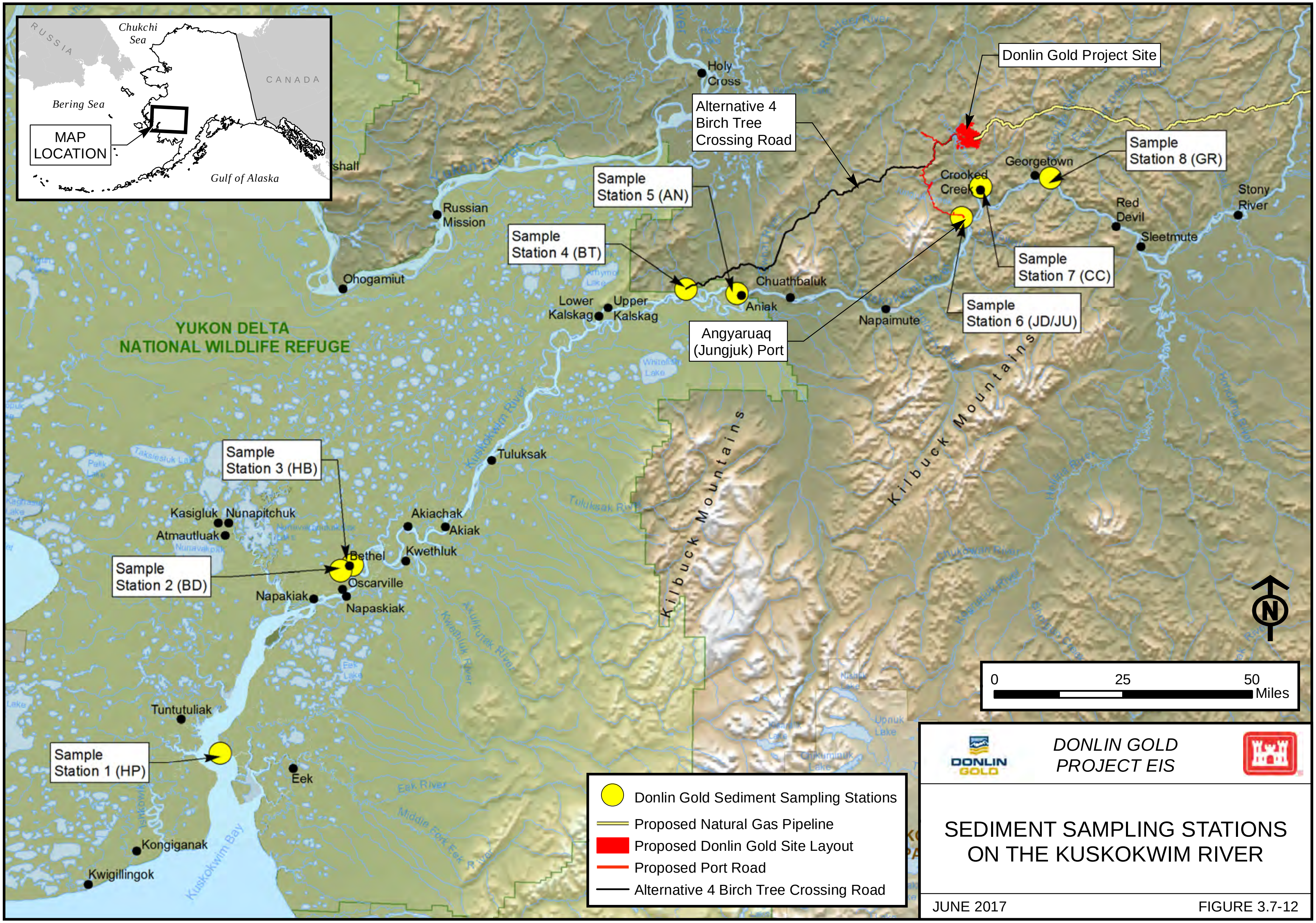
MeHg = methylmercury
ng/g = nanograms per gram (same as ug/Kg)

DONLIN GOLD PROJECT EIS

METHYLMERCURY CONCENTRATIONS IN STREAM SEDIMENT, CROOKED CREEK WATERSHED

JUNE 2017

FIGURE 3.7-11



DONLIN GOLD PROJECT EIS

SEDIMENT SAMPLING STATIONS ON THE KUSKOKWIM RIVER

JUNE 2017

FIGURE 3.7-12

Mean concentrations of trace metals are shown in Tables 3.7-13 and 3.7-14. RWJ (2008a, 2010a) interpreted the results compared to the most stringent of several different sediment quality criteria: Canadian ISQGs (CCME 2014), NOAA TELs/PELs recommended for use by ADEC (2013d), Washington State marine sediment quality criteria (WMSQC), and ADEC soil cleanup levels. Six trace metals (arsenic, chromium, copper, lead, mercury, and nickel) exceeded these criteria for at least one of the statistical summary measures at all stations in 2007; and four metals (arsenic, copper, mercury and nickel) were also elevated in 2010. Mean arsenic was elevated relative to the ISQG at all stations sampled, and all samples collected from the Crooked Creek and George River stations exceeded the ISQG for arsenic (RWJ 2010a). One Crooked Creek sample also contained a slightly elevated level of cadmium (0.709 mg/Kg) relative to the ISQG of 0.6 mg/kg. For copper, the only two samples that exceeded the ISQG were offshore of the Bethel Dock in the slowest moving portion of the river on the inside bend where several inches of soft material had accumulated (RWJ 2008a).

In both 2007 and 2010, the furthest upriver sampling stations (Crooked Creek and George River) showed generally higher concentrations for trace metals relative to the lower river stations. Sediment samples collected from Crooked Creek and George River, known to be in some of the most highly mineralized portions of the river drainage, contained the highest concentrations of arsenic (RWJ 2008a). Data from the 2007 sampling event indicate that samples collected from the Kuskokwim River at the George River location exceeded the ISQG and TEL for mercury of 0.17 mg/kg with concentrations up to 0.34 mg/kg, which generally decrease in a downstream direction.

Overall, the Kuskokwim River sediment results suggested that background trace metal concentrations were relatively high for at least six metals when compared to the most stringent criteria, at levels not unexpected for an area rich in metallic mineral deposits. For values that exceeded the ISQG and/or PEL, the results indicate that such levels could cause biological effects for some organisms exposed to them (RWJ 2010a).

For pesticides measured as part of the RWJ (2010a) study, results indicated that none of the samples exceeded Sediment Quality Guidelines (SQGs) recommended by ADEC (2013b) for use at contaminated sites. Gasoline range organics (GRO) were not detected in 2010 at any of the eight stations sampled. Diesel range organics (DRO) and residual range organics (RRO) representing anthropogenic hydrocarbons, were detected at concentrations lower than guideline levels (Table 3.7-15) (RWJ 2010a).

Table 3.7-13: Summary of Total Metals Results[1] in Kuskokwim River Sediment, 2007 Sampling Event

Trace Metal (Units)	Eek Island (n=2)	Helmick Point (n=6)	Bethel Dock (n=3)	Upriver of Bethel (n=5)	Birch Tree Crossing (n=5)	Aniak (n=2)	Crooked Creek (n=5)	George River (n=3)	TEL/PEL[2]
Aluminum (%)	7.80	7.23	10.17	7.93	7.95	6.91	11.37	11.46	-
Antimony (mg/Kg)	0.06	0.07	0.18	0.15	0.21	0.15	0.38	0.29	-
Arsenic (mg/Kg)	9.47	6.82	10.2	6.99	10.4	6.66	18.5	11.3	5.9/17
Barium (mg/Kg)	57.9	68.0	106	88.2	121	82.0	181	193	-
Beryllium (mg/Kg)	0.24	0.22	0.30	0.29	0.31	0.23	0.44	0.42	-
Bismuth (mg/Kg)	0.05	0.05	0.12	0.07	0.10	0.08	0.20	0.09	-
Boron (mg/Kg)	5.22	4.00	3.48	3.00	2.98	2.61	4.18	4.89	-
Cadmium (mg/Kg)	0.32	0.23	0.34	0.25	0.33	0.25	0.48	0.40	0.596/3.53
Calcium (mg/Kg)	3,860	3,510	5,680	2,910	4,970	4,200	8,900	5,390	-
Chromium (mg/Kg)	17.8	15.7	18.8	18.8	19.9	15.2	25.7	25.1	37.3/90
Cobalt (mg/Kg)	7.29	6.97	7.35	7.42	7.32	6.16	10.3	10.5	-
Copper (mg/Kg)	17.2	10.2	16.8	12.8	16.9	11.8	26.0	22.5	35.7/197
Iron (%)	17.1	14.6	18.6	16.3	17.4	15.8	25.0	24.1	-
Lead (mg/Kg)	4.85	3.95	6.17	4.83	5.39	3.33	9.36	7.38	35/91.3
Magnesium (mg/Kg)	4,300	3,963	4,720	4,460	4,610	4,490	5,360	4,980	-
Manganese (mg/Kg)	374	266	361	296	525	461	640	852	-
Mercury (µg /Kg)	30.5	24.9	46.8	48.0	36.3	20.9	165	341	174/486

Table 3.7-13: Summary of Total Metals Results[1] in Kuskokwim River Sediment, 2007 Sampling Event

Trace Metal (Units)	Eek Island (n=2)	Helmick Point (n=6)	Bethel Dock (n=3)	Upriver of Bethel (n=5)	Birch Tree Crossing (n=5)	Aniak (n=2)	Crooked Creek (n=5)	George River (n=3)	TEL/PEL[2]
Methylmercury (µg/Kg)	0.03	0.04	0.11	0.01	0.13	0.01	0.99	0.40	-
Molybdenum (mg/Kg)	0.24	0.23	0.62	0.21	0.21	0.38	0.58	0.45	-
Nickel (mg/Kg)	19.5	18.7	21.9	21.3	21.5	17.4	28.0	28.8	18/36
Potassium (mg/Kg)	777	618	953	717	941	626	1,358	945	-
Selenium (mg/Kg)	0.31	0.29	0.53	0.24	0.27	0.10	0.42	0.44	-
Silver (mg/Kg)	0.08	0.05	0.10	0.06	0.09	0.12	0.19	0.12	-
Sodium (mg/Kg)	287	227	181	153	173	114	221	170	-
Thallium (mg/Kg)	0.07	0.07	0.08	0.03	0.04	0.07	0.13	0.09	-
Tin (mg/Kg)	0.89	0.84	1.07	0.99	1.10	0.97	1.44	1.37	-
Vanadium (mg/Kg)	28.8	26.1	28.5	27.1	27.9	25.9	35.9	36.2	-
Zinc (mg/Kg)	52.6	46.3	61.8	54.5	59.7	48.3	89.2	80.1	123/315

Notes:

1 Mean concentrations on a dry weight basis.
2 NOAA SQuiRT values for freshwater sediment (ADEC 2013d; Buchman 2008).
Shaded values indicate concentrations in excess of the sediment Quality Guidelines (SQGs) recommended by ADEC (2013b) for use at contaminated sites including Threshold Effects Levels (TELs) and Probable Effects Levels (PELs) published in NOAA Screening Quick Reference Tables for both fresh and marine water sediment (Buchman 2008).

Abbreviations:

mg/kg = milligrams per kilogram
µg/kg = micrograms per kilogram (equivalent to nanograms per gram [ng/g])
n = number of discrete samples collected

Source: RWJ 2010a.

Table 3.7-14: Summary of Total Metals Results[1] in Kuskokwim River Sediment, 2010 Sampling Event

Trace Metal (Units)	Helmick Point (n=6)	Bethel Dock (n=6)	Upriver of Bethel (n=6)	Birch Tree Crossing (n=6)	Aniak (n=5)	Jungjuk Creek		Crooked Creek (n=5)	George River (n=3)	TEL/PEL[2]
						JD (n=4)	JU (n=3)			
Aluminum (%)	11.48	10.26	8.66	9.78	8.41	9.55	9.87	10.38	7.94	-
Antimony (mg/Kg)	0.02	0.02	0.02	0.02	0.02	0.11	0.08	0.04	0.05	-
Arsenic (mg/Kg)	10.4	11.6	8.72	9.13	9.53	13.95	13.12	12.67	12.3	5.9/17
Barium (mg/Kg)	97	125	104	132	138	156	141	160	144	-
Beryllium (mg/Kg)	0.33	0.38	0.29	0.34	0.33	0.30	0.31	0.33	0.34	-
Bismuth (mg/Kg)	0.04	0.05	0.04	0.12	0.10	0.09	0.10	0.08	0.20	-
Cadmium (mg/Kg)	0.26	0.33	0.24	0.24	0.24	0.24	0.25	0.21	0.22	0.596/3.53
Calcium (mg/Kg)	4,380	4,246	3,034	3,586	3,213	4,070	5,133	4,055	5,120	-
Cobalt (mg/Kg)	9.60	8.52	8.60	7.49	7.68	8.33	7.73	8.18	9.41	-
Chromium (mg/Kg)	21.8	22.3	20.7	22.5	21.6	20.9	19.2	19.9	22.7	37.3/90
Copper (mg/Kg)	20.3	21.5	16.5	17.6	16.8	15.0	16.6	14.7	18.7	35.7/197
Iron (%)	21.8	20.9	19.0	20.6	19.5	20.8	21.5	20.0	18.8	-
Lead (mg/Kg)	5.66	6.43	35.8	5.38	4.44	5.31	5.69	5.25	6.80	35/91.3
Magnesium (mg/Kg)	5,450	5,100	4,630	4,860	4,333	4,520	4,233	4,068	4,770	-
Manganese (mg/Kg)	524	422	373	397	554	777	618	622	561	-
Mercury (µg/Kg)	63.6	61.2	34.1	34.5	32.0	43.7	105	80.1	65.6	174/486
Methylmercury (µg/Kg)	0.05	0.11	0.04	0.09	0.01	0.05	0.10	0.30	0.11	-

Table 3.7-14: Summary of Total Metals Results[1] in Kuskokwim River Sediment, 2010 Sampling Event

Trace Metal (Units)	Helmick Point (n=6)	Bethel Dock (n=6)	Upriver of Bethel (n=6)	Birch Tree Crossing (n=6)	Aniak (n=5)	Jungjuk Creek		Crooked Creek (n=5)	George River (n=3)	TEL/PEL[2]
						JD (n=4)	JU (n=3)			
Molybdenum (mg/Kg)	0.22	0.27	0.26	0.49	0.55	0.61	0.56	0.42	0.60	-
Nickel (mg/Kg)	24.0	25.4	25.4	25.2	25.6	24.9	24.3	23.6	30.8	18/36
Phosphorus (mg/Kg)	740.40	644.60	601.60	584.80	566.00	597.00	543.33	516.50	513.50	-
Potassium (mg/Kg)	980.00	945.40	718.20	916.40	915.50	867.00	1,005.33	839.25	986.00	-
Selenium (mg/Kg)	0.28	0.49	0.30	0.40	0.31	0.50	0.44	0.47	0.59	-
Silver (mg/Kg)	0.12	0.12	0.07	0.09	0.08	0.08	0.11	0.08	0.16	-
Sodium (mg/Kg)	608	178	134	172	171	126	137	110	102	-
Thallium (mg/Kg)	0.11	0.10	0.08	0.09	0.09	0.10	0.12	0.09	0.10	-
Tin (mg/Kg)	0.22	0.21	0.39	0.92	0.81	0.99	1.03	0.98	1.25	-
Vanadium (mg/Kg)	35.4	32.2	29.0	30.4	31.9	27.7	25.9	26.7	30.1	-
Zinc (mg/Kg)	70.3	72.0	62.1	65.2	55.6	62.9	63.0	60.4	73.2	123/315

Notes:

1 Mean concentrations on a dry weight basis.
2 NOAA SQuiRT values for freshwater sediment (ADEC 2013d; Buchman 2008).
Shaded values indicate concentrations in excess of the Sediment Quality Guidelines (SQGs) recommended by ADEC (2013) for use at contaminated sites including Threshold Effects Levels (TELs) and Probable Effects Levels (PELs) published in NOAA Screening Quick Reference Tables for both fresh and marine water sediment (Buchman 2008).

Abbreviations:

mg/kg = milligrams per kilogram
µg/kg = micrograms per kilogram (equivalent to nanograms per gram [ng/g])
n = number of discrete samples collected

Source: RWJ 2010a.

Table 3.7-15: Summary of Non-Metal Analytes in Sediments on the Main Channel of the Kuskokwim River from Helmick Point to Georgetown

	Units	Helmick Point	Bethel Dock	Upriver of Bethel	Birch Tree Crossing	Aniak	Jungjuk Creek		Crooked Creek	George River	Sediment Quality Guideline
							JD	JU			
n		6	3	5	5	2	5	5	5	3	
Grain Size											
Percent gravel	%	0	0	0	20.6	25	30.25	10.67	30	49	-
Percent sand	%	17.8	50.14	60.22	51.72	54.75	62.2	56.87	55.975	40.35	-
Percent sand/silt or mud	%	82.2	49.86	39.78	27.68	20.25	7.55	32.47	14.025	10.65	-
Percent finer than 0.02 mm	%	26	18.8	12	10.2	4.75	3.75	10.67	5.25	1.5	-
Inorganics											
Cyanide	mg/Kg	0.04	0.03	0.04	0.03	0.03	0.06	0.05	0.06	0.05	-
Organics											
Diesel Range Organics	mg/Kg	17.20	24.24	13.74	13.34	44.10	38.00	20.65	65.27	25.70	100[1]
Residual Range Organics	mg/Kg	134.32	147.24	62.34	81.6	36.60	176.00	98.65	192.40	23.05	2,000[1]
Total organic carbon	%	1.01	0.96	0.54	0.58	0.28	0.90	0.86	1.00	0.56	-
BTEX											
Toluene	µg/Kg	ND	13.32	5.92	ND	ND	ND	ND	ND	ND	-
Pesticides											
4,4'-DDT	µg/Kg	ND	0.64	ND	ND	ND	ND	ND	ND	ND	1.19/4.77[2]
4,4'-DDD	µg/Kg	ND	0.65	ND	ND	ND	ND	ND	ND	ND	3.54/8.51[2]

Notes:

1 SQGs for organics are the most stringent ADEC recommended clean-up levels for contaminated soils in non-arctic zones (ADEC 2000).
2 SQGs for pesticides are TEL / PEL from Buchman (2008).

Abbreviations:

mg/kg = milligrams per kilogram
µg/kg = micrograms per kilogram (equivalent to nanograms per gram [ng/g])
n = number of discrete samples collected

Previous Studies

Several previous studies have measured mercury levels in the sediments of the Kuskokwim River and some of the surrounding tributaries (Bailey and Gray 1997; Gray et al. 1991, 1994; Nelson et al. 1977). Nelson et al. (1977) analyzed mercury in 299 bulk bottom sediment samples collected from throughout the Kuskokwim River and tributary system as part of a study to determine patterns of mercury dispersal from lode sources in the Kuskokwim River drainage. Throughout areas identified as mercury source areas in the upper Kuskokwim River and tributaries, median background concentrations of mercury were reported as 0.1 mg/kg in bulk sand and silt bottom sediments. The median background concentrations of mercury in bulk bottom sediment generally decreased with distance downriver, and were reported as 0.06 mg/kg in bulk bottom sediment collected from the lower Kuskokwim River and Kuskokwim Bay. This finding suggests that mercury-bearing sediment eroded from upstream sources does not substantially enrich the mercury content of downstream bottom sediments at a watershed scale (Nelson et al. 1977). In addition, a comprehensive study of the Kuskokwim River conducted for the USGS in June 1997 included measurements of the spatial distribution of mercury in sediments of the Kuskokwim River watershed (Wang 1999). This study found similar results: values between 0.01 and 1 mg/kg, with total mercury concentrations in the Kuskokwim River decreasing downstream from McGrath. These data trends are similar to those described above for the project area, where mercury concentrations in sediments are highest between George River and Jungjuk, and generally decrease downstream.

In suspended sediment and filtered water samples, the highest recorded mercury concentrations were measured in samples collected near the largest mineralized areas; concentrations declined substantially with distance downstream (Nelson et al. 1977). The authors suggest that most mercury enters the river system as particulate cinnabar in bottom sediment and that the diluting effects of water discharge and suspended sediment discharge are sufficient to disperse the influence of mercury-enriched bottom sediments over short distances. The distribution of total mercury in Kuskokwim River sediments can be described as a series of stationary, localized, persistent hotspots that diminish over short spatial scales due to the diluting and dispersing effects of river discharge (Nelson et al. 1977).

3.7.2.4 GEOCHEMICAL CHARACTERIZATION

As stated on the Alaska Department of Natural Resources (ADNR) website: “Mining 101 – rock chemistry drives water quality and mine design.” In the natural environment, rocks are broken down into soil through exposure to air and water in a process called weathering. During weathering, minerals react with air and water to release some of their constituents (ions) into the surrounding environment, aqueous and otherwise. In many cases, the primary minerals are transformed into secondary residual minerals during this process. The ions that go into solution may be transported away by runoff, streams, and groundwater, and therefore have a large influence on water quality. If a mineralized deposit is buried beneath other rocks and soil, it naturally weathers very slowly. However, when a mineralized deposit is excavated during mining, the weathering process can increase substantially because previously unexposed rocks are broken up and exposed to rain, snow, and air at the surface.

Both the ore and non-ore rocks in many mineral deposits contain minerals (mainly sulfides) that can produce acid during weathering. The resulting acidic water is known as acid rock drainage

(ARD). Metals and other potentially hazardous constituents can also be released during weathering in a process called metal leaching (ML). Most metals are released more rapidly in acidic water. However, some other constituents, including metalloids such as arsenic, and salts such as sulfate, can be released into the environment even if the water draining the rock has a neutral or basic pH (Smith, 2007).

The task of geochemical characterization at a Mine Site is to identify the potential of the rocks in and surrounding the mineralized deposit to produce ARD and/or ML that could affect water quality in surface water and/or groundwater. The characterization process involves studies of the mineralogy of the rocks, the quantities of minerals with potential to generate or neutralize acid, the amounts of potentially hazardous constituents in the rocks, and the rates of weathering/release of these minerals and constituents expected during mining. Several studies have been undertaken over a number of years to evaluate the potential for ARD and/or ML for the proposed project. A brief summary of these studies follows.

In some mineralized deposits, rock type alone can be a good indicator of whether a rock will potentially produce ARD and/or ML. Gold mineralization at the Mine Site occurs mainly within the sulfide minerals pyrite (FeS_2) and arsenopyrite (FeAsS), which are contained in clastic sedimentary rocks (mainly greywackes and shales) that have been intruded by rhyodacite dikes and sills (SRK 2007). The geology of the deposit is described in detail in Section 3.1, Geology, and shown on Figure 3.1-3. Because the hot, ore-forming fluids that produced mineralization at the Donlin deposit have already chemically reacted with the surrounding rocks, it is not sufficient to only use rock type as the indicator of which rocks are potentially acid generating (PAG). Extensive drilling was conducted to determine ore zones. Rock material from the drill holes was then used for testing. Initial geochemical studies (MDAG 2006) found substantial variability in both the composition of rock material and the way the material reacts over time to weathering conditions.

Because of the geochemical variability in the rocks, the selected approach was to look at the material that would be mined each year. The annual area mined was estimated by developing a block model. (The block model is a computer model that shows the three-dimensional location of each type of rock and the likely order of mining.) Based on the block model, over the life of the proposed mine, there would be about 556 million tons of ore, 3 billion tons of waste rock, and about 50 million tons of overburden mined from the proposed open pit. Rock material was assessed based on whether the material would take a long time to react – developing acid or losing neutralizing potential – and whether the material would be processed to end up in tailings or set aside as waste rock, in order to design waste management strategies.

Based on the results obtained from integrating the geochemical studies with the block model, the majority of the rocks that would be mined from the Donlin Gold deposit do not have the potential for acid generation and could be considered substantially acid neutralizing. However, most rocks do have the potential to leach certain constituents, mainly arsenic and sulfate. These results will be discussed in more detail in the next few sections.

3.7.2.4.1 WASTE ROCK

Acid Rock Drainage

Typically, rocks are predicted to be non-acid generating (NAG) or potentially acid generating (PAG) based on the relative amounts of constituent minerals that can form acid and those that can neutralize acid. The most typical and strongest acid-generating mineral is pyrite. It forms sulfuric acid (H_2SO_4) when it reacts with oxygen and water during weathering:

$$FeS_2(s) + 15/4\ O_2 + 7/2\ H_2O = 2\ H_2SO_4 + Fe(OH)_3(s)$$

Other sulfides and a few other types of minerals can also form acid, but to a lesser extent. The acid-generation potential (AP) of a rock is the total capacity of that rock to generate acid if all of its acid-generating minerals react to completion during weathering.

The best and most common acid neutralizing mineral is calcite, one of the polymorphs of calcium carbonate ($CaCO_3$):

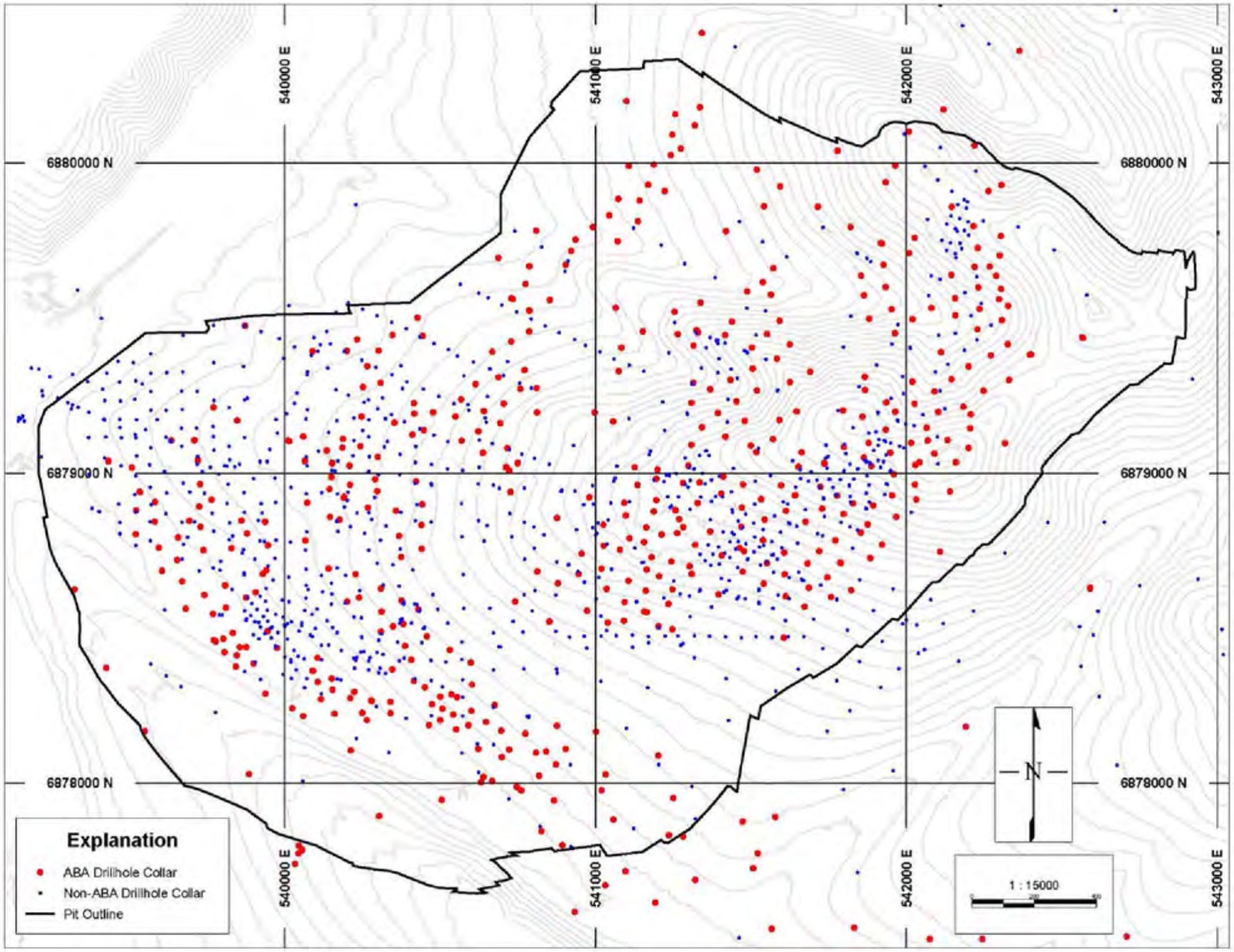
$$CaCO_3(s) + H_2SO_4 = Ca^{2+} + SO_4^{2-} + H_2O + CO_2(g)$$

Other carbonates, including dolomite [$CaMg(CO_3)_2$] and ankerite [$Ca(Fe,Mg,Mn)(CO_3)_2$], can also neutralize acid to varying extents. In contrast, siderite ($FeCO_3$), is not considered to have neutralization potential, due to its iron content, unless it has magnesium or calcium substituting for some of the iron. Similar to the definition of AP, the acid-neutralization potential (NP) of a rock is its total capacity to neutralize acid if all its carbonate minerals react to completion.

Acid-Base Accounting (ABA) is the series of laboratory tests designed to estimate a rock's AP and NP. Both AP and NP are expressed in units of tons of calcium carbonate equivalent per 1,000 tons of material (t $CaCO_3$/kt) to allow direct comparisons. Corrections must be made when the respective minerals are not all pyrite or calcite.

For the proposed project, the NP and AP terms were developed through extensive study of the chemistry and mineralogy of the various rock types at the site. This included ABA using the standard Sobek et al. (1978) method on more than 2,300 rock samples collected from more than 170 drill holes blanketing the proposed mine area (Figure 3.7-13; SRK 2007, 2011). The mineralogical characterization included analysis of 40 samples by thin section, Rietveld XRD, and/or ion microprobe to determine carbonate minerals, including detailed analysis of 617 carbonate mineral grains, as well as investigation of 132 sulfide mineral grains from 16 samples (SRK 2011). Based on mineralogical analysis, the NP was found to be mainly ankerite and dolomite, followed by calcite. Siderite (with about 20-25 percent magnesium substituting for iron) was found to be prevalent, especially in sediments. The AP was found to be mainly pyrite and arsenopyrite. Corrections were made to the standard ABA calculations to account for these minerals (SRK 2007, 2011).

The most common measure of whether a rock is NAG or PAG is its ratio of NP to AP. That ratio is referred to as NP*/AP in the Donlin geochemical studies, where the * denotes that corrections were made to the standard NP measurement to account for the relative amounts and different neutralization potentials of each of the various types of carbonate minerals found at the site (SRK 2007, 2011).



Data Source: SRK 2011, Figure 1

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DRILLHOLES SAMPLED FOR ACID-BASE ACCOUNTING

JUNE 2017

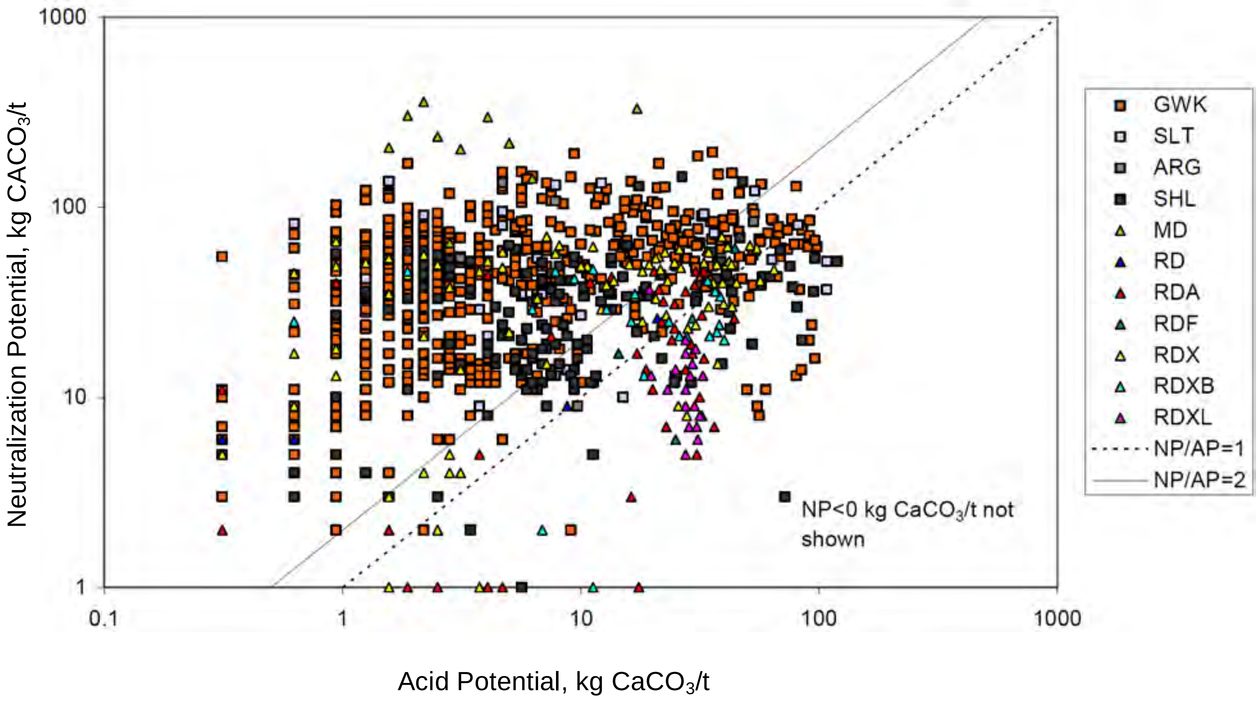
FIGURE 3.7-13

For the proposed project waste rock, the NP and AP showed substantial variation both within and across rock types (Figure 3.7-14). The AP distributions of greywackes (GWK) and shales (SHL) were both bimodal (i.e., each rock type had two local maxima in the probability distribution curve) with good separation between modes. The lower AP mode for shales was higher than that for greywackes, probably due to the presence of non-mineral-related diagenetic pyrite. Unlike AP, NP distributions for sedimentary rocks were more complex, not easily separable into modal groups.

The rhyodacite AP was typically higher than that for sedimentary rocks, but it also spanned a large range depending on the form of the rhyodacite. Unlike that of the sedimentary rocks, the rhyodacite NP was separable into two populations at an NP value of 34.5 t $CaCO_3$/kt. Aphanitic rhyodacite (RDA) and lathe-rich rhyodacite (RDXL) tended to have higher AP. Due to low NP, they were mostly PAG (NP/AP <1). Mafic dykes had the highest NP/AP of all the rock types (SRK 2007).

The block model mentioned in Section 3.7.2.4 was used in combination with the geochemical and mineralogical studies to estimate ARD potential during each proposed mining year. Each block in the block model was assigned a sulfur content value and a neutralizing carbonate content value, the combination of which were used to determine whether or not that block is PAG or NAG material. The sulfur and carbonate content values in the block model were assigned by geostatistically interpolating sulfur and carbonate values from drill core assays. If the carbonate content in a particular block was more than sufficient to neutralize any acid generating potential, the block was generally characterized as NAG.

Based on integrating the block model with the geochemical and mineralogical studies, the tonnage-weighted average NP*/AP ratio of the waste rock expected to be mined during the lifetime of the mine is estimated to be 5.5 (Enos 2013c). This means that the waste rock as a whole has the capacity to neutralize 5.5 times more acid than those same rocks can generate. The red line in Figure 3.7-15 shows how the ratio is expected to vary from year to year as different rocks are mined, with the lowest average ratio in the first and last years of operation. However, in all cases the annual average NP*/AP is greater than 2.6. Different states and countries have set different criteria for considering rocks to be non-acid generating. Typically a NP/AP ratio of 2 is used unless a less stringent ratio can be shown to be valid for a particular site. As an example, California at one time used the more stringent ratio of 3 as the criterion to divide NAG and PAG rock. Even based on the more stringent criterion applied there, the majority of waste rock at the Donlin Gold Project site as a whole would be considered non-acid generating.



Rock Types:
ARG = Argillite, GWK = greywacke, MD = mafic dykes, RDA = aphanitic rhyodacite, RDF = fault-zone rhyodacite,
RDX = crowded crystalline rhyodacite, RDXB = blue rhyodacite, RDXL = lathe-rich rhyodacite, SHL = shale, SLT = siltstone

Data Sources: SRK 2007

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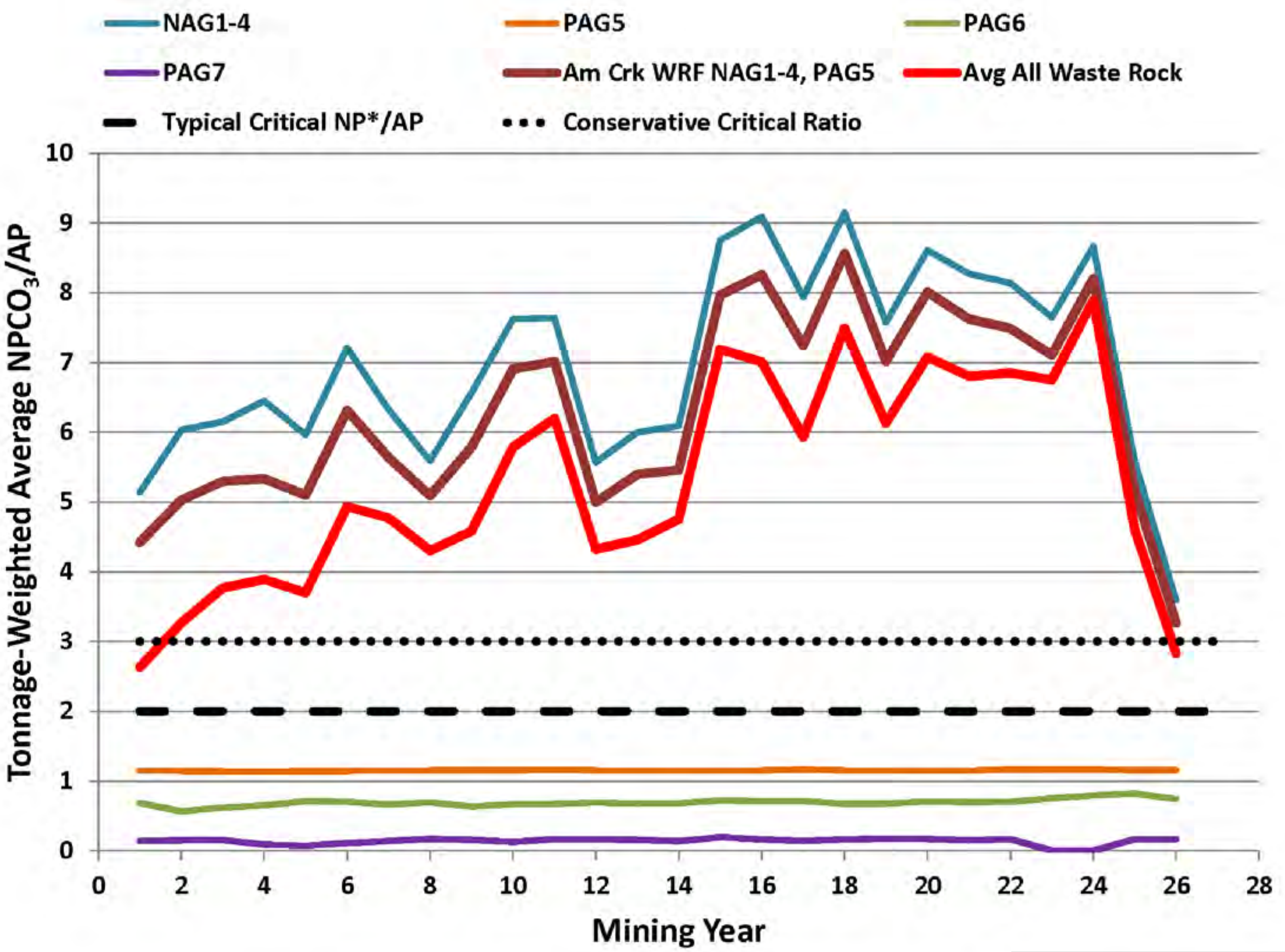
ACID- NEUTRALIZATION POTENTIAL VS. ACID-GENERATION POTENTIAL BY ROCK TYPE

JUNE 2017

FIGURE 3.7-14



Data Sources: BGC 2011b; Enos 2013c

Tonnage-Weighted Annual Averages for the Ratio of Neutralizing Potential (NP) to Acid Producing Potential (AP) for Waste Rock of Various Categories

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TONNAGE-WEIGHTED ANNUAL AVERAGE OF NP/AP FOR WASTE ROCK CATEGORIES

JANUARY 2018

FIGURE 3.7-15

Because some rocks were found to be PAG, Donlin further classified the rocks into several additional waste rock management categories (WRMCs) based on their NP*/AP and their tendency to leach arsenic. Table 3.7-16 (SRK 2007, 2011, 2012b) shows the classification scheme, which was developed in two stages. The definitions of different categories of NAG are based on both acid generating potential and concentrations of arsenic. The only differences between categories NAG 1 and NAG 2, and between NAG 3 and NAG 4, are the arsenic content. (Categories NAG 1 and NAG 3 were not used.) The development process is discussed in some detail later in this section. The classification scheme was then used to design an appropriate waste rock management strategy for each category of rock from low-reactivity mostly NAG rocks (NAG 1-4, with NP*/AP > 1.3) to rocks with high potential to generate acid in a short time period (PAG 7, with NP*/AP ≤ 0.2). (The annual average NP*/AP values for each of these classifications are shown as the thinner lines on Figure 3.7-15). The NAG 1-4 rocks constitute about 92.5 percent of the total waste rock. Of these, over 90 percent are NAG 1 and NAG 2 rocks, and less than 10 percent are NAG 3 or NAG 4. The most reactive PAG 7 rocks constitute less than 0.08 percent of the total waste rock (SRK 2007). The waste rock facility (WRF) and water treatment plant were designed based on the conclusions of the waste characterization reports regarding the percentage of waste rock that is PAG, as well as the leachability of metals and metalloids such as arsenic (BGC 2011b).

Table 3.7-16: Preliminary and Revised Waste Rock Management Categories for Proposed Donlin Gold Project

		SRK 2007 (Preliminary)		Revised 2011
Waste Rock Management Category	**Category Description**	**NP*/AP Range and AP**	**As/S (As in mg/kg and S in %)**	**NP*/AP Range and AP**
NAG 1	Very unlikely to generate ARD and "low" arsenic leaching	AP<3 kg CaCO3/t or NP*/AP >2	As/S<196 and As<260	Not Used
NAG 2	Very unlikely to generate ARD and arsenic leaching potentially significant	AP<3 kg CaCO3/t or NP*/AP > 2	As/S>196 or AS>250	AP<3 kg CaCO3/t or NP*/AP > 2
NAG 3	Unlikely to generate ARD and "low" arsenic leaching	1.4<NP*/AP<2	As/S<196 and As<250	Not Used
NAG 4	Unlikely to generate ARD and arsenic leaching potentially significant	1.4 <NP*/AP <2	As/S>196 or As>250	1.3 <NP*/AP <2
PAG 5	PAG but with very long delays (several decades) to onset of ARD	1.0 <NP*/AP <1.4	All	1.0 <NP*/AP <1.3
PAG 6	PAG in the life of the mine (possibly less than a decade)	0.2 <NP*/AP <1.0	All	0.2 <NP*/AP <1.0
PAG 7	PAG but with shorter delays to onset (less than a few years)	NP*/AP <0.2	All	NP*/AP <0.2

Notes:

* Revised site-specific $NPCO_3$ = 0.76 • NP + 4.8 (SRK 2011).

Source: SRK 2016d.

The ABA test is called a static test because it measures the total amount of acid-generating material and acid-neutralizing material and assumes that both react to completion. It gives no indication of relative reaction rates. In order to investigate relative rates, kinetic geochemical tests are used to simulate weathering. For the proposed project, kinetic tests included 40 laboratory humidity cell tests (HCTs) and 10 on-site barrel tests.

HCTs mimic weathering in the laboratory by exposing a sample to a weekly weathering cycle consisting of 3 days of wet air, 3 days of dry air, and a day in which the sample is rinsed with water to simulate rain. The leachate from the water rinse is collected each week. The Donlin HCTs were conducted on samples of individual rock types, in two phases: Phase 1 consisted of 16 HCTs and Phase 2 consisted of 24 additional HCTs. Twelve of the Phase 1 HCTs were continued for 84 weeks; while 3 were continued for over 8 years and 1 is still continuing after almost 12 years. The three longer-term Phase 1 HCTs included two intrusives with NP/AP values of <0.1 (sample 63762m PAG 7) and 0.3 (sample 674322, PAG 6); and one sedimentary sample with a NP/AP of 2.3 (sample 213138, PAG 2), respectively, on Figure 3.7-16). The continuing sample is number 216377, from a fault zone in sediment. It has a NP/AP of 1.0 (PAG 5). Three of the Phase 1 HCTs (one PAG 6 and two PAG 7 samples) produced acidic leachates throughout the test. The PAG 6 sample 674322 began producing acidic leachate after 200 weeks.

In 2008, 24 additional HCTs were started to test a broader range of geochemical characteristics that matched the seven preliminary WRMCs shown in Table 3.7-16. The Phase 2 HCTs were evaluated after approximately two years (SRK 2011). Most of these HCTs were continued for another two years thereafter, and seven are still continuing. Information on the geology and partial geochemistry of the samples used for the HCTs is given in Table 3.7-17, from SRK 2011. Leachate pH and sulfate for these samples are shown in Figure 3.7-17. Only one of the Phase 2 samples produced acidic leachate in almost seven years of tests. The pH has started to decrease in another sample (shale sample 683340, PAG 5), but is still neutral. The sulfate release rates (and thus sulfide oxidation rates) from the Phase 1 and 2 tests were found to be highly correlated to the sulfide content of the samples. This correlation allowed estimates of the delay to acid formation for various classes of PAG rock, as given in Table 3.7-16—from less than a few years for PAG 7 rocks to several decades for PAG 5 rocks.

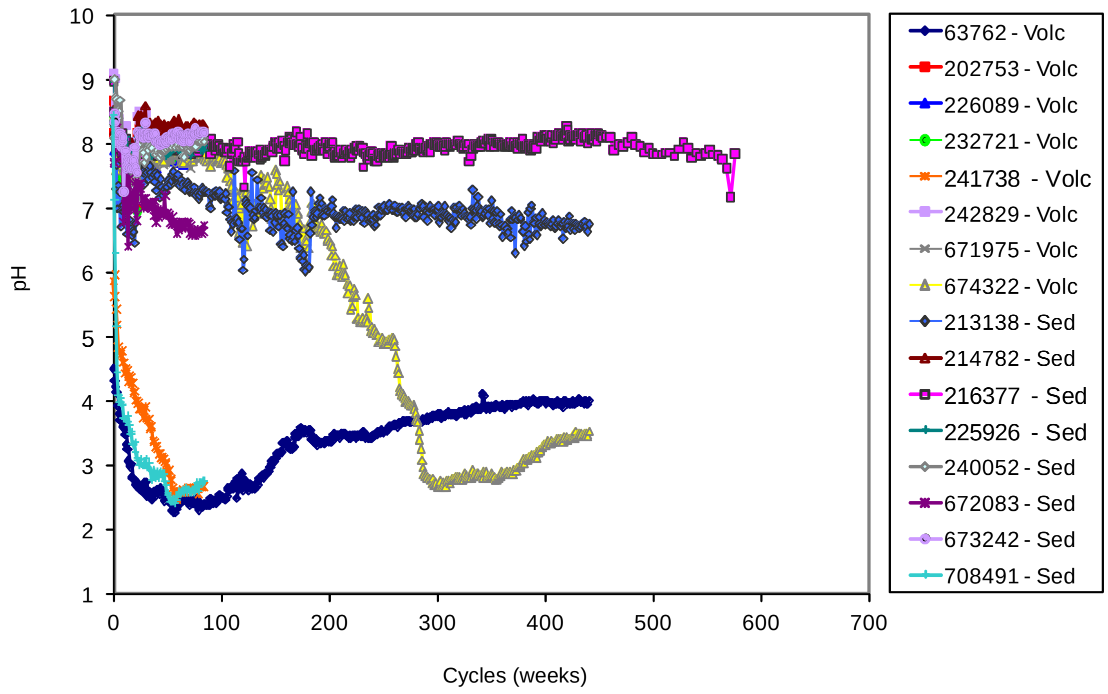

Figure 3.7-16A: Selected Phase 1 Humidity Cell Test Results: pH

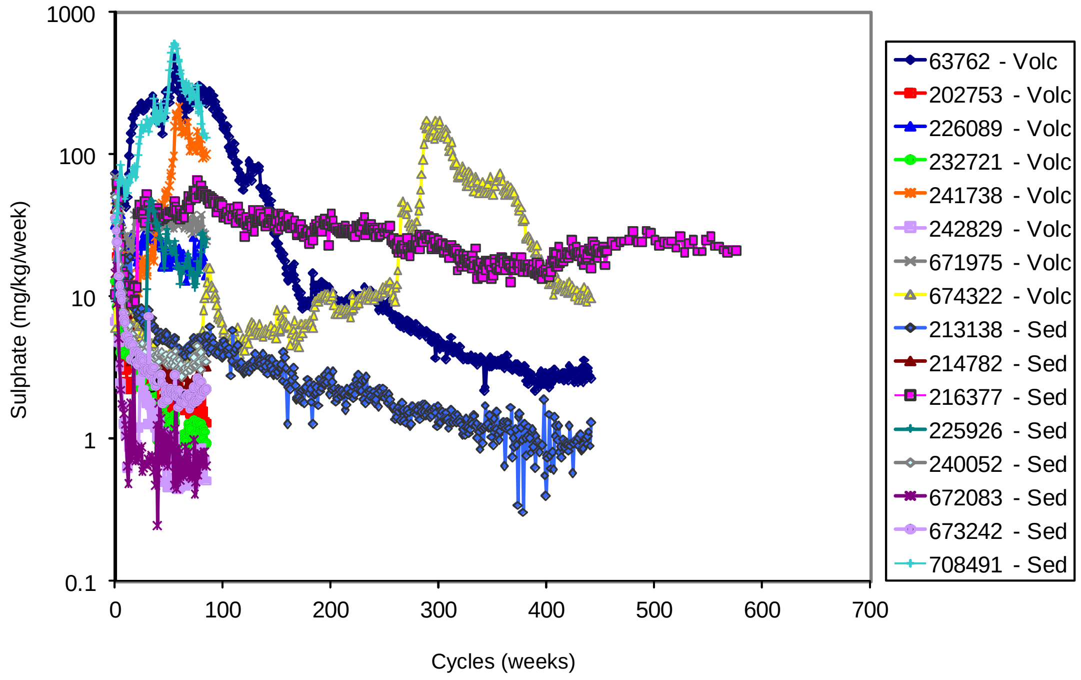

Figure 3.7-16B: Selected Phase 1 Humidity Cell Test Results: Sulfate Loading

Data Source: SRK 2015e; Weglinski 2015c

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PHASE 1 HUMIDITY
CELL TEST RESULTS
FOR pH AND SULFATE

JUNE 2017 FIGURE 3.7-16

Table 3.7-17: Characteristics of Rock Samples in Humidity Cell Tests

Phase	HCT	Date Started	Date Stopped	Logged Geology	WRMC	Paste pH	Total S %, S	Sulfate %, S	AP kg $CaCO_3$/t	NP* kg $CaCO_3$/t	NP*/AP	Arsenic mg/kg
1	216377	7/14/2004	Continuing	Fault Zone	5	7.8	1.75	0.03	55	54.2	1.0	2,690
1	63762	7/14/2004	12/26/2012	RDA	7	3.6	0.94	0.12	29	-2	-0.1	1,825
1	202753	7/14/2004	2/22/2006	RDXB	2	8.2	0.06	-0.01	2	39.76	21.2	53.4
1	226089	7/14/2004	2/22/2006	RDX	6	7.8	1.44	0.02	45	35.2	0.8	2,860
1	232721	7/14/2004	2/22/2006	RDXB	6	5.6	0.22	0.01	7	2	0.3	1,555
1	241738	7/14/2004	2/22/2006	RDA	7	4.5	0.56	0.03	18	1	0.1	519
1	242829	7/14/2004	2/22/2006	RDX	2	8.7	0.07	-0.01	2	47.4	21.7	17.4
1	671975	7/14/2004	2/22/2006	RDX	6	7.9	1.38	0.02	43	27.6	0.6	5,630
1	674322	7/14/2004	12/26/2012	RDA	6	7.7	0.73	0.01	23	7	0.3	812
2	678622	10/2/2008	12/27/2012	RDA	1	8.1	0.58	0.01	18.1	22	1.2	7.39
2	683388	10/2/2008	12/27/2012	RDXL	4	8.8	0.68	0.06	21	36.0	1.7	582
2	725741	10/4/2008	Continuing	RDX	5	8.5	1.58	0.03	49	48.9	1.0	1,940
2	733611	10/4/2008	1/5/2013	RDXL	7	7.7	0.93	0.02	29	4	0.1	1,250
2	746703	10/2/2008	12/27/2012	RDX	1	8.5	0.03	<0.01	1	24. 6	26.2	10.8
2	747402	10/2/2008	12/27/2012	RDA	4	8	1.05	0.02	33	36.0	1.1	265
2	747406	10/2/2008	Continuing	RDA	2	8.5	0.47	0.03	15	41.3	2.8	270
2	748001	10/4/2008	1/5/2013	RDX	5	8.3	1.37	<0.01	43	43.6	1.0	2,590
2	751417	10/2/2008	Continuing	RDX	1	9.1	0.15	0.01	5	69.4	14.8	20.6
2	758527	10/4/2008	1/5/2013	RDX	5	8.2	1.1	0.02	34	32.2	0.9	2,340
2	758533	10/2/2008	Continuing	RDX	4	8.2	0.86	0.03	27	43.6	1.6	809
1	213138	7/14/2004	12/26/2012	GWK	2	7.7	0.11	-0.01	3	8	2.3	1,520
1	214782	7/14/2004	2/22/2006	ARG	2	8	0.25	-0.01	8	86.9	11.1	188
1	225926	7/14/2004	2/22/2006	GWK	4	8	0.89	0.01	28	45.8	1.6	1,190

Table 3.7-17: Characteristics of Rock Samples in Humidity Cell Tests

Phase	HCT	Date Started	Date Stopped	Logged Geology	WRMC	Paste pH	Total S %, S	Sulfate %, S	AP kg $CaCO_3$/t	NP* kg $CaCO_3$/t	NP*/AP	Arsenic mg/kg
1	240052	7/14/2004	2/22/2006	SHL	6	7.6	0.79	0.02	25	12	0.5	168
1	672083	7/14/2004	2/22/2006	SHL	6	5.7	0.18	0.07	6	1	0.02	256
1	673242	7/14/2004	2/22/2006	SLT	1	7.6	0.32	0.01	10	25.3	2.5	59.6
1	708491	7/14/2004	2/22/2006	GWK	7	6.7	1.79	0.03	56	8	0.1	4,720
2	680671	10/2/2008	12/27/2012	GWK	2	7.8	0.07	0.01	2	18	8.2	134
2	681142	10/2/2008	12/27/2012	SHL	1	8.2	0.25	0.02	8	21	2.7	34
2	683340	10/4/2008	Continuing	SHL	5	7	0.25	<0.01	8	8	1.0	149
2	745975	10/4/2008	1/5/2013	SHL	5	7.9	0.33	0.02	10	19	1.8	356
2	751720	10/2/2008	12/27/2012	GWK	1	9.2	0.1	0.02	3	28.4	9.1	14.4
2	754502	10/2/2008	Continuing	GWK	4	8.5	2.24	0.02	70	95.2	1.4	1,580
2	760063	10/4/2008	1/5/2013	SHL	4	8.3	3.25	0.01	102	137	1.3	1,550
2	760101	10/2/2008	Continuing	GWK	3	8.3	1.69	0.02	53	74.7	1.4	549
2	779151	10/4/2008	1/5/2013	GWK	5	8.7	0.03	0.04	9	13	1.4	30.9

Notes:

* Revised site-specific $NPCO_3$ = 0.76 NP + 4.8 (SRK 2011)
ARG = Argillite
GWK = greywacke
MD = mafic dykes
RDA = aphanitic rhyodacite
RDX = crowded crystalline rhyodacite
RDXB = blue rhyodacite
RDXL = lathe-rich rhyodacite
SHL = shale
SLT = siltstone

Source: SRK 2011, 2013d.

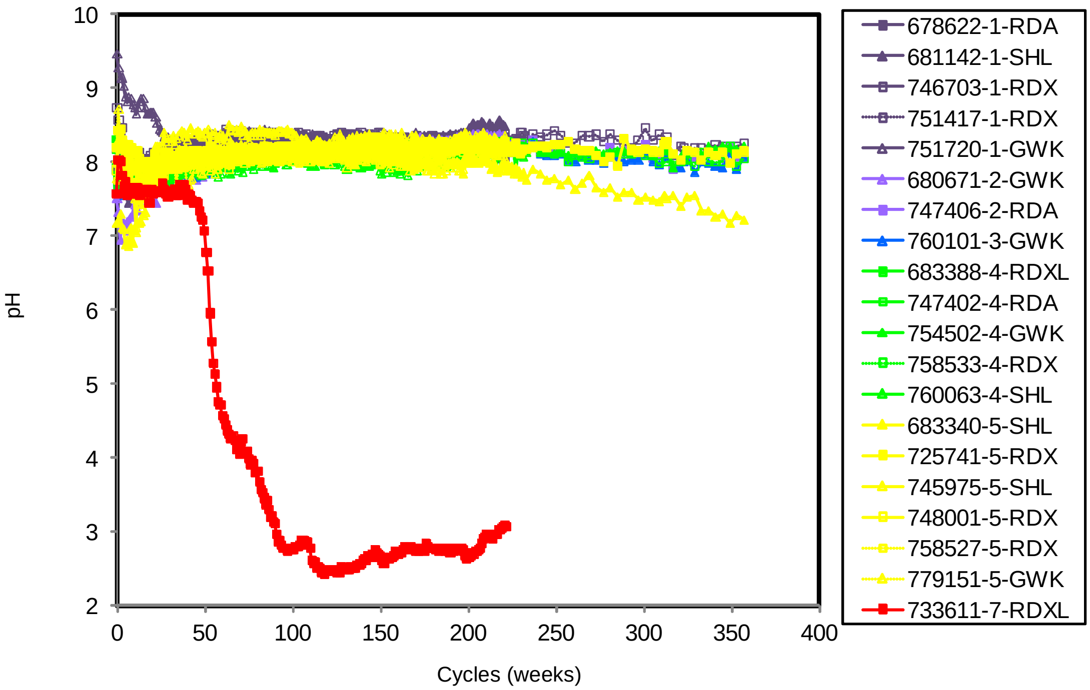

Figure 3.7-17A: Selected Phase 2 Humidity Cell Test Results: pH

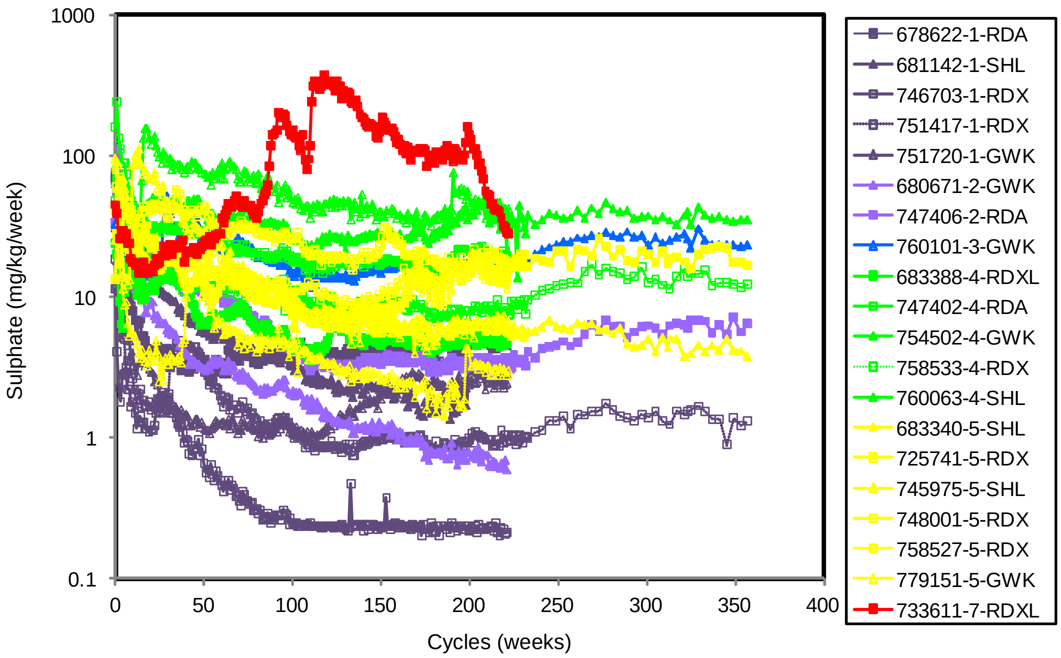

Figure 3.7-17B: Selected Phase 2 Humidity Cell Test Results: Sulfate Loading

Data Source: SRK 2015e; Weglinski 2015c

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PHASE 2 HUMIDITY
CELL TEST RESULTS
FOR pH AND SULFATE

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FIGURE 3.7-17

Four on-site/field barrel tests were begun in 2006. Each of these contained about 550 to 600 pounds of rock in a barrel exposed to the onsite weathering conditions. The tests included two composite samples of sedimentary rock and two of rhyodacite —one sample of each rock type had high sulfur and high arsenic contents; the other had low sulfur and low arsenic contents (Table 3.7-18). Six additional barrel tests (three volumes: 200, 400, and 600 pounds each for two composite samples) were conducted for a flow path experiment beginning in 2008. One of the composite samples had a tonnage-weighted average NP/AP value of 5.4 and a low arsenic value of 31.6 mg/kg. The other was a PAG 6 composite with a NP/AP value of 0.9 and an arsenic value of 1,020 mg/kg. (This arsenic value is below average for rhyodacite rocks, but above average for sedimentary rock).

All of the barrel test leachates have had circumneutral pH values throughout the several-year duration of the tests. However, most barrels show a decrease of pH over time and there appears to be a seasonal pattern with a pH minimum in late summer in all cases (Figure 3.7-18). (In 2009, only one sample was collected due to exceptionally dry conditions at the site.) There is some correlation in the results between arsenic and sulfur at higher sulfur concentrations, but not at low sulfur concentrations (low AP) (see SRK 2007, Table 2-11).

The preliminary NAG and PAG classifications (Table 3.7-16) were based on about 70 to 140 weeks of data on the 16 HCTs that began in 2004. The HCTs indicated that oxidation of sulfide minerals was strongly positively correlated with the sulfur content of the rock. For rock samples with the highest total sulfur content, the molar ratio of the release of calcium plus magnesium (the carbonate neutralization components of the rocks) to sulfur [(Ca+Mg)/SO_4] tended toward a value of 1.3:1. This ratio became the basis for the revised criterion separating NAG from PAG rock in Table 3.7-16. Selection of this ratio was further justified by the (Ca+Mg)/SO_4 results to-date from the on-site barrel tests, which take into account actual on-site temperature and precipitation conditions (Figure 3.7-19). Modeling of oxidation rates from the HCTs suggests that PAG 5 waste rock would potentially generate acidic conditions after a long time delay on the order of several decades; whereas PAG 6 is potentially acid-generating within the life of the mine (possibly less than a decade), and PAG 7 waste rock could potentially generate acidic conditions within less than a few years (SRK 2007).

Table 3.7-18: Characteristics of Samples Used in On-Site Barrel Tests

Barrel	Sample Description	Rock Type	Description	Paste pH	Total Sulfur % S	Sulfate Sulfur	AP kg $CaCO_3$/t	NP kg $CaCO_3$/t	NP/AP	Arsenic mg/kg
ARD-1	761001 +2mm 761001 -2mm	Sedimentary Rock	NAG 1	8.6 8.4	0.24 0.23	<0.01 <0.01	7.5 7.2	68 74	9.1 10.3	90.8 109
ARD-2	761002 +2mm 761002 -2mm	Rhyodacite	NAG 2	8.7 8.5	0.43 0.55	<0.01 0.01	13.4 16.9	33 33	2.5 2.0	686 725
ARD-3	761003 +2mm 761003 -2mm	Sedimentary Rock	PAG 5	8.3 8.1	1.57 1.68	0.02 0.03	48.4 51.6	60 68	1.2 1.3	3,810 4,000
ARD-4	761004 +2mm 761004 -2mm	Rhyodacite	NAG 4	8.9 8.7	0.62 0.8	0.01 0.01	19.1 24.7	34 34	1.8 1.4	2,650 2,590
WRMC 1	884003	Composite	NAG 1	9.0	0.35	0.01	11	59.5	5.4	31.6
WRMC 6	884004	Composite	PAG 6	8.2	1.05	0.02	33	28.4	0.9	1,020

Source: SRK 2007.

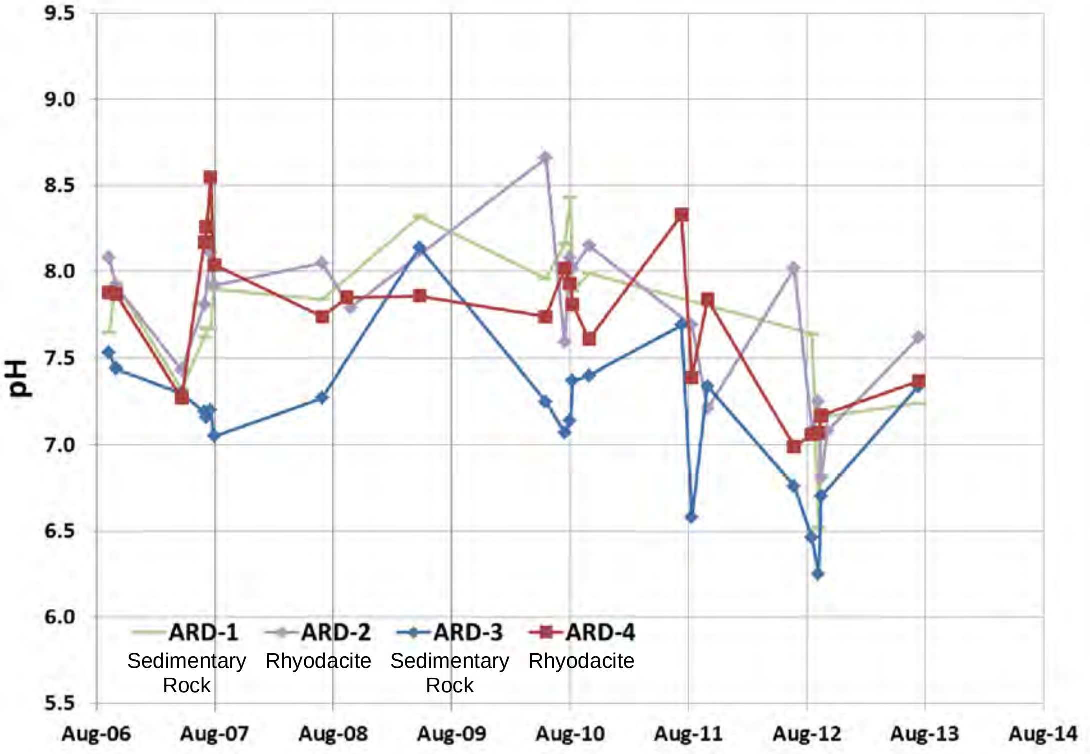

Figure 3.7-18A: Barrel Test Leachate pH: 2006 Barrel Tests

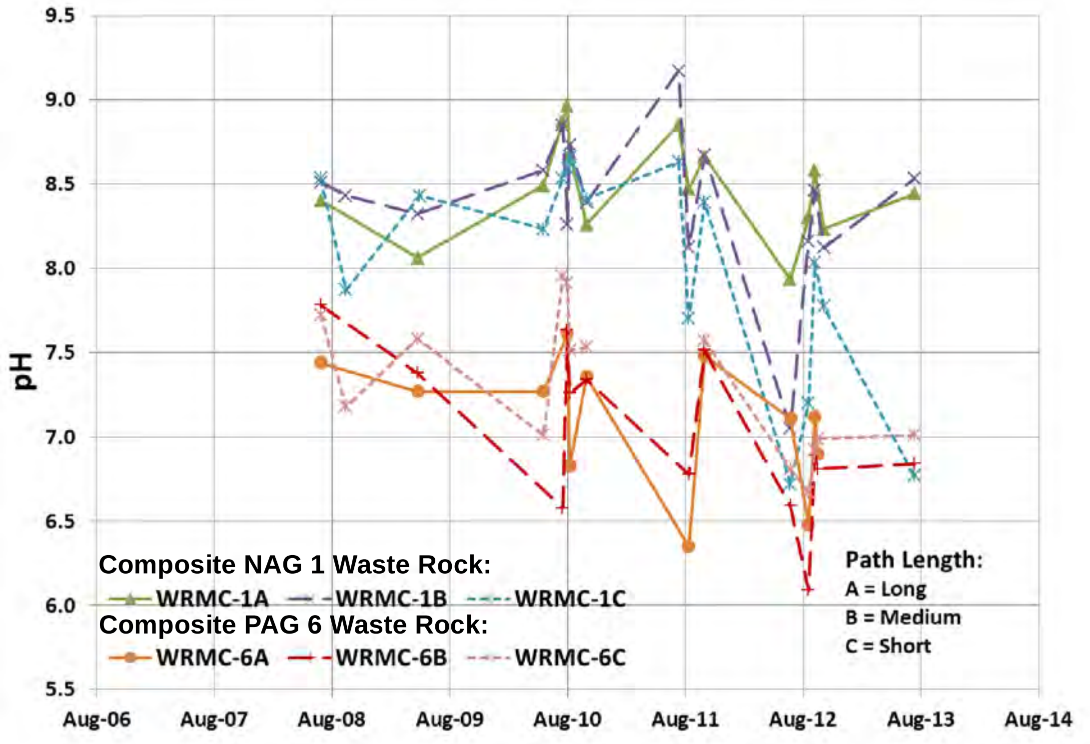

Figure 3.7-18B: Barrel Test Leachate pH: 2008 Flow Path Experiment

Data Source: SRK 2011

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BARREL TEST LEACHATE pH: 2006 BARREL TESTS AND 2008 FLOW PATH EXPERIMENT

JUNE 2017

FIGURE 3.7-18

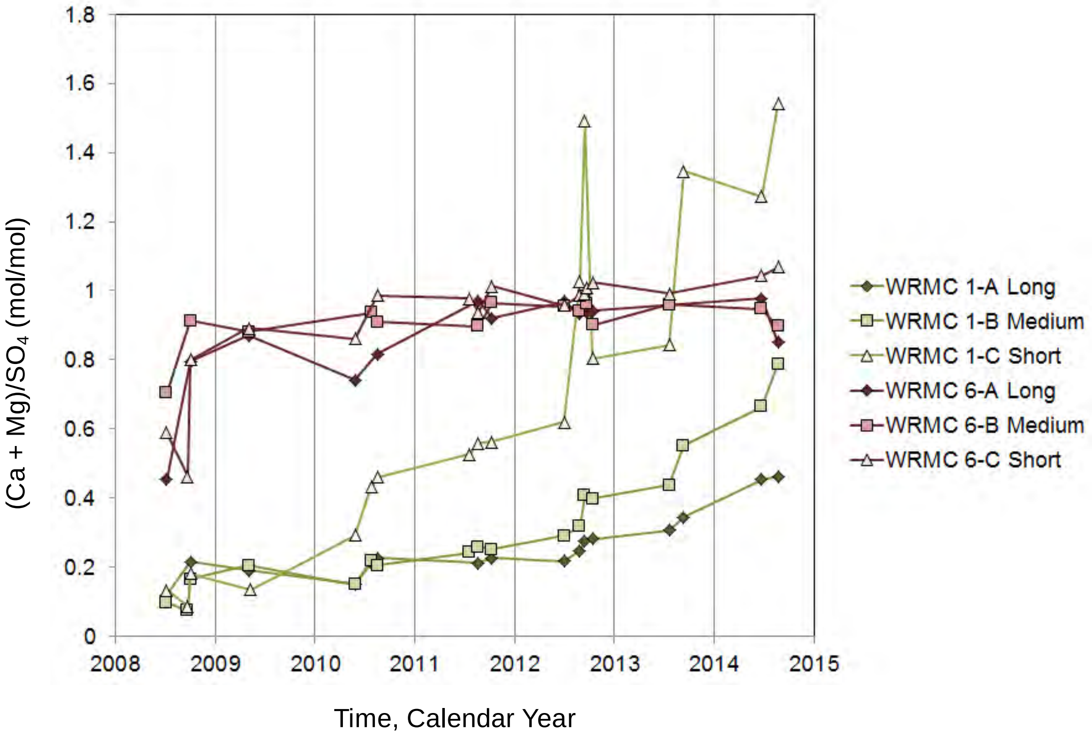

Figure 3.7-19A

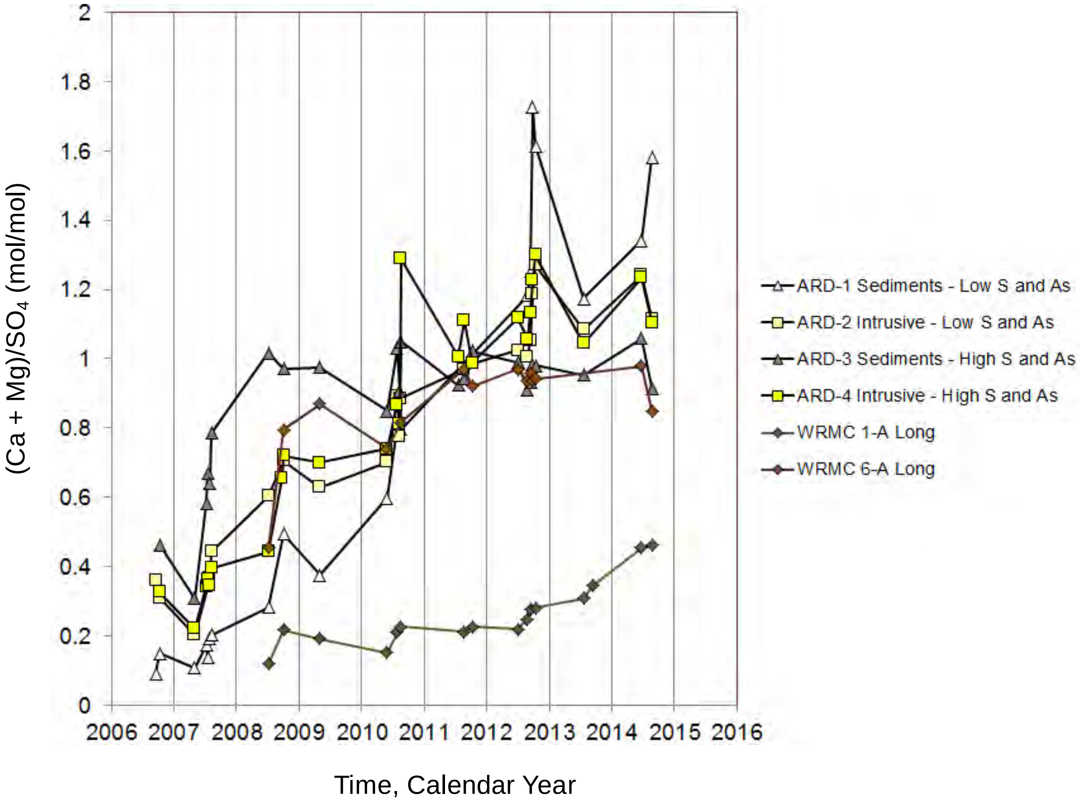

Figure 3.7-19B

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TEMPORAL VARIATION OF MOLAR RATIO OF (Ca +Mg)/SO_4 IN NAG 4 (ARD-4), PAG 5 (ARD-3) AND PAG 6 (WRMC-6) BARREL TESTS

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FIGURE 3.7-19

Data Source: SRK 2015d; Weglinski 2015e

Metals and Other Potential Contaminants

To investigate metals and other potential contaminants, three types of investigations were conducted. The first investigated the total concentrations of constituents; the other two investigated short-term and longer-term leaching potential of these constituents. To determine whole-rock concentrations, more than 2,200 samples were subjected to bulk geochemical analysis (Table 3.7-19). This analysis allowed comparison of concentrations of selected constituents with average crustal abundance data (from Price 1997) for similar rock types.

The bulk analyses indicated that the most enriched constituents were silver, arsenic, mercury, and antimony. Most base metals (cadmium, cobalt, copper, lead, nickel, zinc) are not enriched (SRK 2007). Arsenic was the main enriched element. It occurs mainly in arsenopyrite with some realgar, orpiment, and arsenian pyrite, and shows a similar bimodal distribution among rock types as sulfur. The rhyodacite rock type group appears mostly well mineralized; whereas the sedimentary greywacke and shale groups were largely weakly mineralized, although the average arsenic concentrations were well above global averages for these rock types. Antimony and mercury concentrations are strongly correlated with arsenic (Figure 3.7-20) (SRK 2007); this geochemical coherence allows arsenic distributions to be used as a proxy for the distribution of these two constituents as well.

To determine the potential for ML, a modification of the Nevada Division of Environmental Protection’s Meteoric Water Mobility Procedure (MWMP) was conducted on 20 composite samples for short-term leaching characteristics. The MWMP modification consisted of sequentially leaching a series of sub-samples of a given rock with the same leach water to allow assessment of the build-up of leachates along a flow path (SRK 2007). All samples but one produced basic leachate (Table 3.7-20). The basic leachates contained arsenic concentrations up to 10 mg/L and antimony up to 1.1 mg/L. Selenium concentrations tended to be higher in sedimentary (up to 0.12 mg/L) than intrusive rocks.

Data were collected on the leachates from the kinetic HCTs and barrel tests to investigate longer-term leaching. The four longer-term Phase 1 HCTs included samples with a range of arsenic concentrations—from a low of 812 mg/kg for intrusive sample 674322 to a high arsenic content of 2,690 mg/kg for the fault-zone sample 216377. As mentioned previously, the barrel test samples were specifically chosen to include a range of arsenic values. The HCT and barrel test results corroborated the MWMP results and indicated that arsenic has the potential to be leached at both acid and neutral-to-basic pH values, even from NAG rocks (Figure 3.7-21) (see also Table 3.7-18 regarding WRMC classification based on arsenic). Leachate arsenic tended to decrease with time initially in almost all HCTs. However, after some time, the arsenic concentration then tended to level out. In the fault-zone sample 216377, after an initial decrease, the arsenic concentrations again increased before leveling out (Figure 3.7-21). In the acidic rhyodacite PAG 7 sample 733611, arsenic concentrations varied almost cyclically with time.

Table 3.7-19: Distribution Statistics for Selected Elements in Waste Rock

Rock Type	Statistic	S %	As mg/kg	Cd mg/kg	Cu mg/kg	Hg mg/kg	Sb mg/kg	Se mg/kg	Zn mg/kg	AP kgCaCO$_3$/t	NP kgCaCO$_3$/t	NP/AP
GWK	n	1,543	1,488	1,488	1,488	1,488	1,487	1,488	1,488	1,543	1,543	1,543
	Min	0.01	1.4	0.01	0.6	0.01	0.03	0.4	53	0.15	3	0.28
	P5	0.01	5.1	0.1	15	0.06	0.56	1	72	0.15	15	2.3
	Median	0.07	32	0.25	33	0.25	3.4	2	100	2.2	58	26
	Average	0.26	290	0.34	36	0.68	16	2	110	8.3	71	9.6
	P95	1.3	1,400	0.68	65	2.1	31	3	150	43	150	180
	Max	5.5	10,000	58	430	90	3,300	10	5,900	170	470	870
MD	n	375	355	355	355	355	355	355	355	375	375	375
	Min	0.01	1.6	0.01	0.8	0.09	0.19	0.5	96	0.6	3.9	0.14
	P5	0.05	15	0.2	38	0.14	1.3	1	110	1.8	17	1.3
	Median	0.23	47	0.64	67	0.39	4.2	3	150	8.1	44	5.4
	Average	0.43	250	0.61	67	0.6	13	2.9	150	14	63	4.9
	P95	1.7	1,100	1.1	97	1.6	30	5	180	56	140	35
	Max	4.1	10,000	1.8	130	14	950	5	460	140	210	93
RDA	n	77	76	76	7	76	76	76	76	77	77	77
	Min	0.03	5.5	0.04	11	0.12	0.64	0.5	63	0.9	67	3
	P5	0.048	22	0.06	20	0.16	2.6	0.5	68	1.6	140	4.1
	Median	0.1	190	0.12	32	0.54	20	2	79	3.8	260	67
	Average	0.36	480	0.16	33	1.1	59	1.6	87	12	260	26
	P95	1.5	1,900	0.34	44	4.1	160	3	130	51	360	180
	Max	2.6	4,400	0.94	96	5.5	1,000	4	240	85	390	260
RDX	n	45	45	45	45	45	44	45	45	45	45	45
	Min	0.01	36	0.02	1.2	0.31	5.7	-0.2	20	0.3	2	0.17
	P5	0.01	57	0.02	1.4	0.35	8.6	-0.2	43	0.36	2.2	0.18

Table 3.7-19: Distribution Statistics for Selected Elements in Waste Rock

Rock Type	Statistic	S %	As mg/kg	Cd mg/kg	Cu mg/kg	Hg mg/kg	Sb mg/kg	Se mg/kg	Zn mg/kg	AP $kgCaCO_3/t$	NP $kgCaCO_3/t$	NP/AP
	Median	0.45	680	0.07	4.4	1	21	2	64	15	21	1.2
	Average	0.5	1,200	0.21	8.4	1.4	110	1.4	85	17	23	1.5
	P95	1.2	3,900	0.89	28	3.6	210	2	220	40	51	12
	Max	1.3	5,100	1.5	60	6.5	2,400	3	280	45	64	17
SHL	n	167	167	167	167	167	167	167	167	167	167	167
	Min	0.01	8.2	0.01	1.3	0.05	1.5	0.5	26	0.3	3.2	0.17
	P5	0.01	17	0.03	2.5	0.1	2	0.5	40	0.61	11	0.5
	Median	0.29	330	0.06	9.2	0.75	14	1	62	10	45	4.4
	Average	0.53	1,100	0.19	12	1.3	20	1.4	79	17	46	2.9
	P95	1.6	5,100	0.55	40	4.3	43	3	140	50	80	74
	Max	2.1	9,500	9.2	75	14	330	7	1,200	66	99	230

Notes:

Rock Types:
GWK = greywacke
MD = mafic dykes
RDA = aphanitic chyodacite
RDX = crowded crystalline rhyodacite
SHL = shale

Source: SRK 2011 (Table 8).

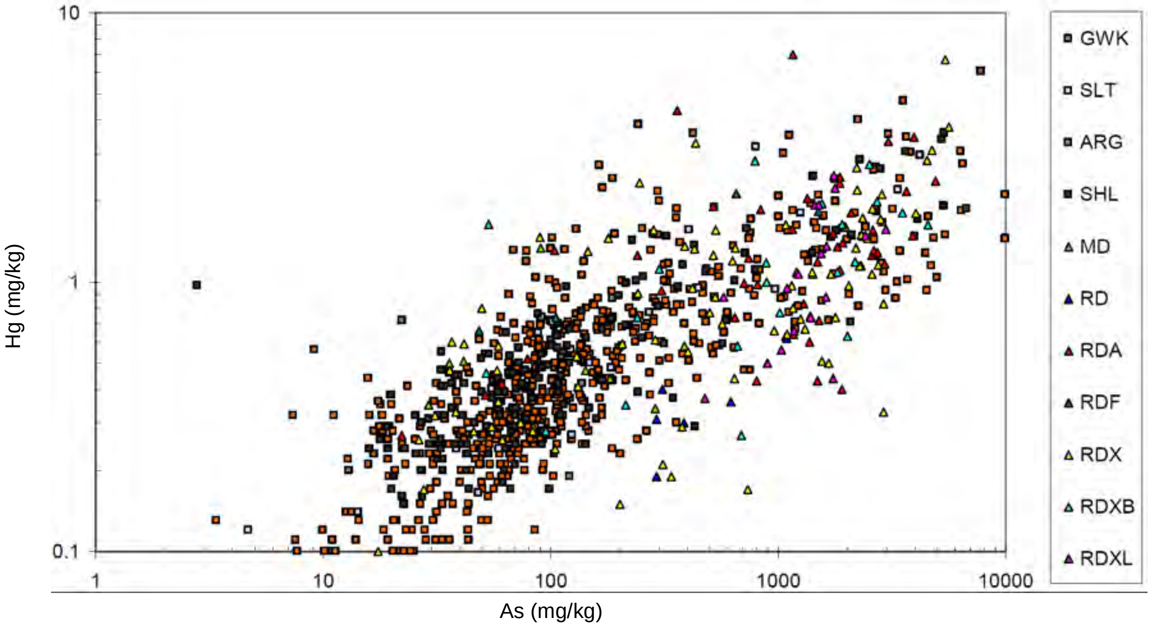

Figure 3.7-20 A: Arsenic correlation with mercury in waste rock.

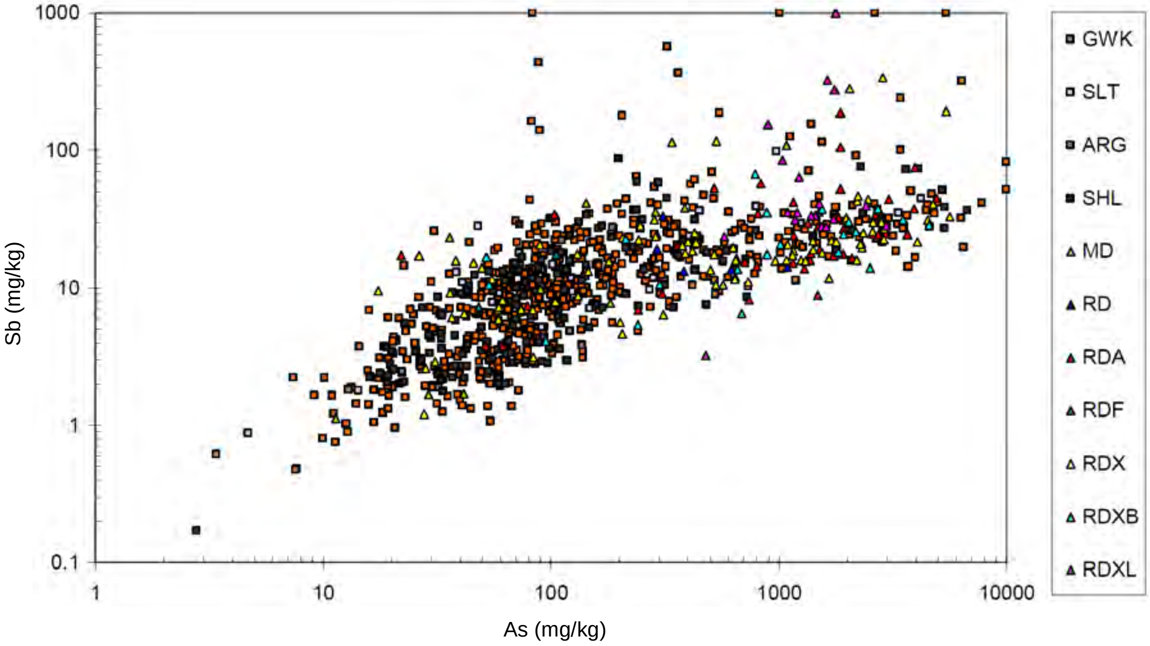

Figure 3.7-20 B: Arsenic correlation with antimony in waste rock.

Data Source: SRK 2007

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ARSENIC CORRELATIONS WITH MERCURY AND ANTIMONY IN WASTE ROCK

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FIGURE 3.7-20

Table 3.7-20: Waste Rock Leachate Concentrations in the Final Step of Sequential Meteoric Water Mobility Procedure

	Solids		**Leachate Chemistry**							
Rock Type	**Arsenic mg/kg**	**Sulfur %**	**Antimony µg/L**	**Arsenic µg/L**	**Cobalt µg/L**	**Nickel µg/L**	**pH s.u.**	**Selenium µg/L**	**Sulfate mg/L**	**Zinc µg/L**
ARG	68	0.18	220	220	<10	52	8.05	110	250	<10
ARG	76	0.19	240	7,700	17	54	7.62	51	450	<10
GWK	64	0.07	57	140	<10	12	8.11	43	70	10
GWK	81	0.07	170	150	<10	17	8.15	24	110	<10
GWK	1,745	0.80	160	1,400	35	170	7.73	7.7	330	57
MD	231	0.12	480	2,200	14	190	7.93	8.1	230	<10
MD	746	0.15	1,100	5,400	10	370	8.11	8.8	120	12
RD	110	0.25	220	150	<10	38	7.83	<5	210	11
RD	129	0.26	85	76	16	62	7.79	<5	730	<10
RD	1,244	0.38	200	530	<10	44	7.75	<5	260	28
RD	82	0.70	310	180	13	49	7.87	<5	600	20
RD	1,720	1.17	220	10,000	32	100	7.6	13	640	<10
RD	1,783	0.98	200	1,100	<10	36	7.74	12	480	13
SED	1,464	0.11	32	230	<10	20	7.06	<5	110	83
SED	68	0.85	170	100	<10	58	8.11	37	510	11
SED	2,034	0.99	120	2,400	18	77	7.7	37	820	10
SED	1,561	0.13	14	2,200	26	120	7.75	68	210	13
SED	117	1.50	410	310	29	140	8.23	120	770	<10
SHL	2,543	1.90	46	320	1,000	2,500	5.49	44	1,600	4,200
SHL	1,835	1.53	280	420	12	77	7.44	14	460	27

Rock Types:

ARG = Argillite
GWK = greywacke
MD = mafic dykes
RD = rhyodacite
SED = sedimentary
SH = shale

Source: SRK 2007.

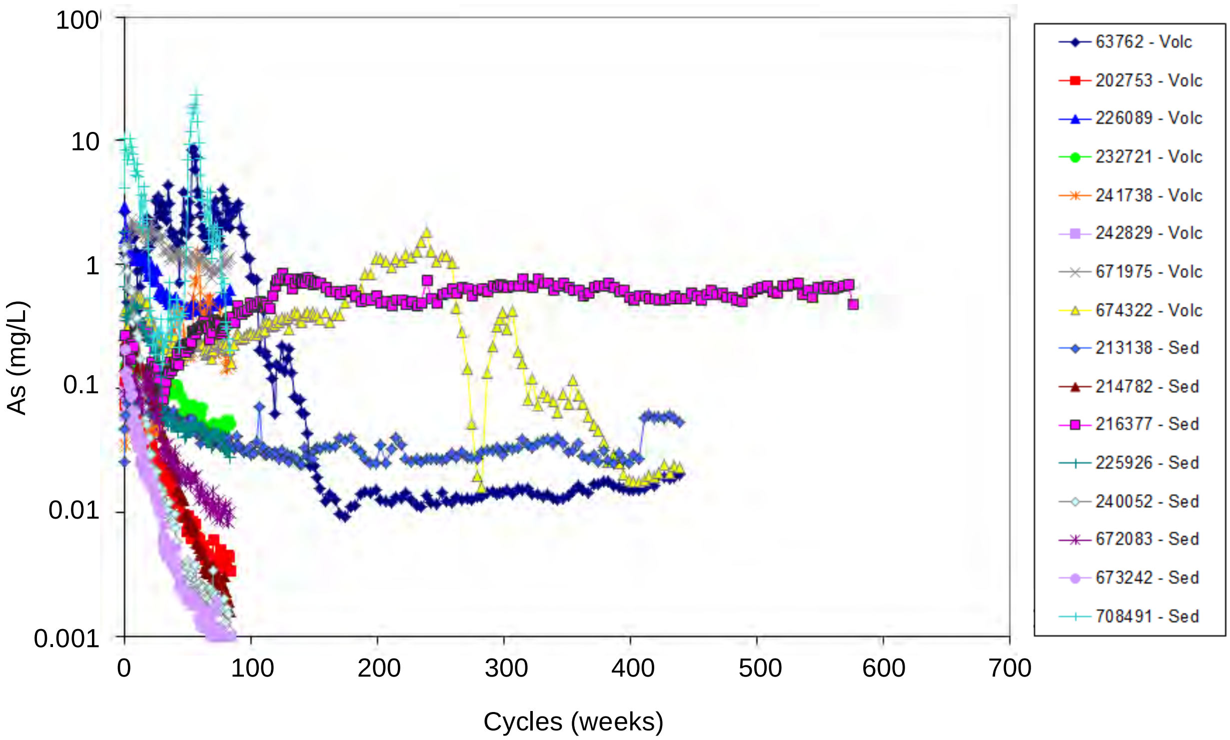

Figure 3.7-21A: Humidity Cell Test Arsenic Loading Results: Phase 1

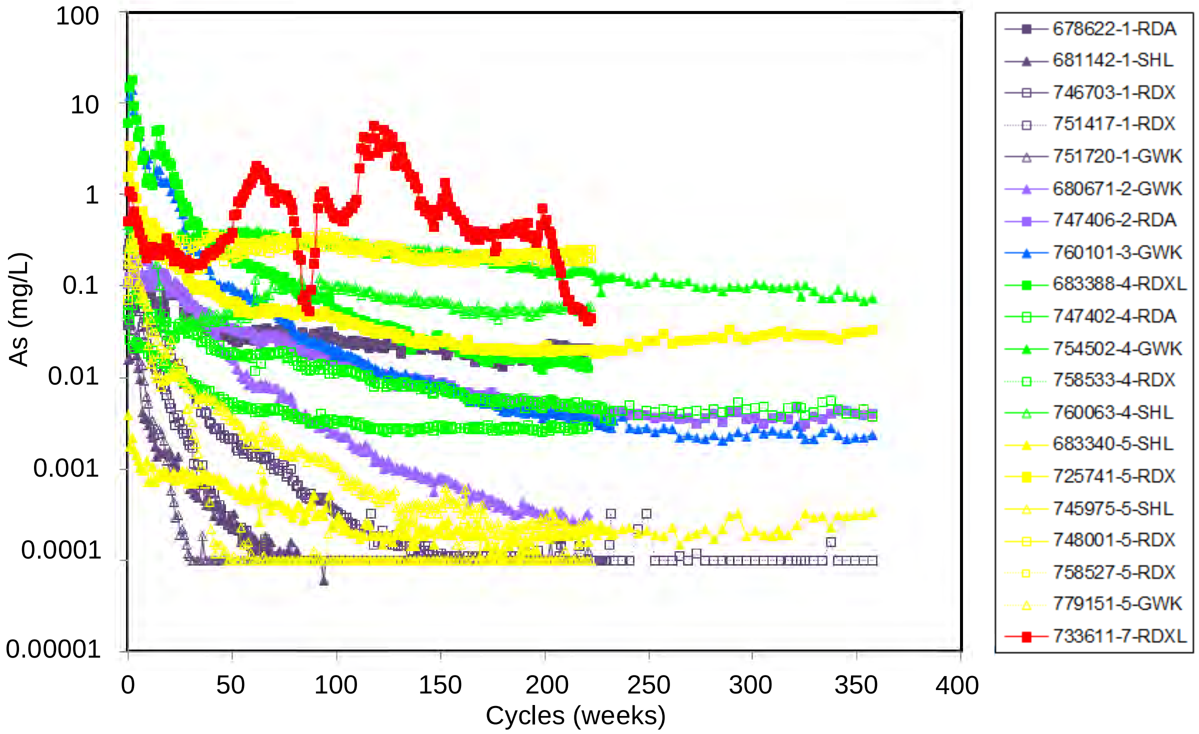

Figure 3.7-21B: Humidity Cell Test Arsenic Loading Results: Phase 2

Data Source: Weglinski 2015e

Note:
Scale change for cycles axis

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HUMIDITY CELL TEST ARSENIC LOADING RESULTS: PHASE 1 AND PHASE 2

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FIGURE 3.7-21

Table 3.7-21: Grouping of Major and Trace Element Leaching Trends with Time for Barrel Tests (2006-2014)

Decreasing	Stable	Increasing	Undetected
Nitrate, Chloride Sodium Arsenic, Antimony, Manganese, Molybdenum	Alkalinity, Sulfate Potassium Aluminum, Barium, Boron, Cadmium, Chromium, Cobalt, Copper, Fluoride, Lead, Lithium, Manganese (ARD 3, WRMC 6-A), Mercury, Nickel, Selenium, Zinc	Sulfate (ARD 3, WRMC 6-A) Calcium, Magnesium Iron, Zinc (ARD 3, WRMC 6-A)	Silver, Thallium, Cyanide, Vanadium, Beryllium

Source: SRK 2015b, Table 2.

In the ongoing barrel tests, calcium and magnesium have tended to increase with time in most barrels, suggesting that carbonate dissolution and acid neutralization has been an ongoing process in the barrels. Sulfate, iron, and zinc concentrations have tended to increase slightly with time in the PAG barrels ARD-3 and WRMC 6, suggesting continued pyrite oxidation over time. Leachate arsenic, antimony, and molybdenum concentrations have decreased with time (Table 3.7-21), whereas most other trace elements showed relatively stable behavior. In general, arsenic concentrations in the leachate tended to correlate with the concentration of arsenic in the solid (SRK 2011, 2015c). Decreasing arsenic leaching over time suggests that the initial oxidation of arsenopyrite is being hindered, perhaps through the formation of a protective oxidized coating on the arsenopyrite surface (SRK 2015b). Seasonal behavior has been observed for major ions, arsenic and most other constituents, as it was for pH. Concentrations tended to increase in summer, perhaps due to initial melting of ice and flushing of salts, with a subsequent drop later in the year as a result of rainfall and lower temperatures (SRK 2015b).

In the barrel flow path experiments, selenium, nitrate, chloride, and lithium concentrations have increased along the flow path in both the NAG 1 (WRMC 1) and PAG 6 (WRMC 6) barrels (Table 3.7-22). However, the NAG and PAG barrels have shown a clear difference in the major ion chemistry, likely due to oxidation of pyrite and concomitant dissolution of even the less reactive carbonates dolomite and dolomite-ankerite: Sulfate, calcium, magnesium, potassium, copper, and zinc concentrations have increased along the flow path in the PAG barrels, whereas the concentrations of all of these constituents except calcium have stabilized in the NAG barrels. Calcium concentrations have been highest in the shortest flow path for the NAG barrels, whereas arsenic and antimony concentrations have been highest in the shortest flow path for the PAG barrels. Leachate pH values have increased along the flow path in the NAG barrels, but have not shown consistent trends with time in either the NAG or PAG barrels. In general, the PAG barrels have produced a lower average pH (7.1) than the NAG barrels (pH 8.2).

Table 3.7-22: Grouping of Major and Trace Element Leaching Trends Along the Flow Path for Barrel Tests (2006-2014)

Waste Rock Management Category	Decreasing Along Flow Path	Stabilizing Along Flow Path	Increasing Along Flow Path	Highest in Shortest Flow Path	Undetected/ Inconclusive
WRMC1	--	Sulfate Potassium, Magnesium Arsenic, Copper, Zinc	pH, Alkalinity Sodium Antimony, Boron, Fluoride, Manganese, Molybdenum, Nickel	Calcium	--
WRMC 6	--	Sodium Boron, Fluoride, Molybdenum, Nickel, Manganese	Sulfate Calcium, Magnesium, Potassium Copper, Zinc	Arsenic, Antimony	pH, Alkalinity
WRMC 1 and WRMC 6	Barium	Aluminum, Cadmium, Cobalt, Iron, Lead, Mercury	Chloride, Nitrate, Selenium, Lithium	--	Beryllium, Silver, Thallium, Vanadium

Source: SRK 2015b, Table 3.

3.7.2.4.2 OVERBURDEN

Overburden is the soil and other material overlying a mineral deposit, which must be removed during mining. At the Mine Site, the overburden is composed mainly of coarser-grained materials, consisting of colluvium and terrace gravels, and finer-grained materials, consisting of organic material, loess, and alluvium. The coarser-grained material would be stored mainly in the South Overburden (SOB) Stockpile and the finer-grained material in the North Overburden (NOB) Stockpile (SRK 2016a). Thirty-three samples of overburden collected as part of exploratory drilling were subjected to ABA using the Sobek method (SRK 2013c). Twelve of these samples came from the pit area. The remaining samples were from other parts of the Mine Site. One of the locations in the pit area was sampled at multiple depths. Table 3.7-23 presents the data for individual samples from the pit area and summary statistics for the miscellaneous samples. The total sulfide sulfur increased with depth in the one set of samples from multiple depths in the pit area. The two deepest samples are PAG. These results suggest that the overburden at depth in the pit area may be PAG. The maximum measured carbonate was 0.52 percent carbon and the maximum measured sulfide was 0.24 percent sulfur. The overburden from miscellaneous areas around the Mine Site typically is pH-inert, with essentially no NP or AP. In other words, these results suggest that the overburden from the miscellaneous areas has low potential for ARD, but also has essentially no acid neutralizing, or buffering, potential. Arsenic concentrations were also analyzed for the ABA overburden samples. The results suggest two populations: Those samples from the pit area have substantially higher arsenic concentrations than the overburden samples from other parts of the Mine Site.

Six samples of overburden were collected as part of Mine Site feasibility studies and subjected to standard MWMP (SRK 2007). The leachate water pH ranged from 6.3 to 6.8 (Table 3.7-24). The concentrations of most metals were near or below the lower limit of detection. However, dissolved iron and aluminum were highly variable, with iron reaching 5.9 mg/L and aluminum 4.2 mg/L in the sample of coarse terrace gravels. Dissolved arsenic slightly exceeded 0.01 mg/L in the fine terrace gravels sample from the mineralized area, and in the coarse colluvium sample, in agreement with the higher arsenic concentrations in ABA overburden samples from the pit area (Table 3.7-23). These results suggest that arsenic leaching from overburden excavated from the pit area could exceed Alaska Water Quality Standards.

3.7.2.4.3 LOW-GRADE ORE

Dependent on the price of gold at any particular time, rock with a given percentage of gold could be considered either ore or waste rock. Ore with a relatively low percentage of gold is called low-grade ore and is planned to be placed in a low-grade ore stockpile between the pit and the Lower Contact Water Dam (CWD) (SRK 2017b). At Donlin, arsenopyrite is the predominant gold-bearing mineral. It is typically found associated in quartz veins with pyrite (Goldfarb et al. 2004). The veins often contain subordinant dolomite and ankerite, as well as younger stibnite (Sb_2S_3). According to SRK 2016d, this rock is PAG 7.

3.7.2.4.4 TAILINGS

Gold is removed from ore-bearing rocks in a process that includes crushing, grinding, flotation, pressure oxidation, and cyanide leaching. The finely ground rock and process water mixture remaining after the gold is removed is called tailings. In the proposed project, the tailings would be deposited in a large impoundment called the Tailings Storage Facility (TSF).

For the proposed project, the approach to studying gold recovery methods to be used in the process facilities was developed in multiple phases. Phase 1 was laboratory scale testing followed by an initial pilot-plant test that provided a preliminary understanding of the ore types, metallurgy, and necessary refinements to the beneficiation process. The results of Phase 1 justified a Phase 2 pilot plant test. Phase 2 was a stand-alone study and included what was learned during Phase 1 (Day 2013). The tails from Phase 2 are more representative of the combined ore types that would be processed through the process facilities. More comprehensive analysis for mercury was also performed on the different phases of the pilot plant.

Mineralogical studies of tailings samples produced by the phased pilot tests in the laboratory (SRK 2007) show them to be mainly silicates; carbonates; residual sulfides (mainly pyrite and arsenopyrite); and the oxidation products of pyrite, including jarosite, hematite, and gypsum (Table 3.7-25). The NP for the tailings was calculated with appropriate corrections for ankerite and dolomite; AP was calculated with appropriate corrections for jarosite (SRK 2007, 2011).

Table 3.7-23: Acid-Base Accounting Data for Pit-Area Overburden and Distribution Statistics for Other Miscellaneous Mine-Site Overburden

Hole ID	Depth		Paste pH	Total Sulfur % S	Sulfide Sulfur % S	Sulfate Sulfur % S	NP kg CaCO3/kt	AP kg CaCO3/kt	NP/AP	Total Carbonate % C	Arsenic mg/kg
	From	To									
Pit Area											
DC07-1597	6	12	6.8	0.01	0	0.01	3	0.3	9.6	0.025	419
DC07-1597	12	18	6.2	0.03	0.025	0.005	3	0.9	3.2	0.025	390
DC07-1597	18	24	5.4	0.04	0.035	0.05	3	1.3	2.4	0.16	354
DC07-1597	24	30	5.0	0.08	0.075	0.005	3	2.5	1.2	0.22	591
DC07-1597	30	35	5.2	0.26	0.24	0.02	3	8.1	0.37	0.08	854
DR97-339	0	3	5.0	0.03	--	--	0	0.9	0	0.025	163
DC98-483	0	4	7.4	0.08	0.08	0.0055	18	2.5	7.2	0.24	276
DGT06-1179	4	8	6.9	0.08	0.07	0.01	12	2.5	4.8	0.52	335
DC02-723	11	14	6.7	0.01	0.01	0.005	1	0.25	3.2	0.025	333
DC02-745	5	8	6.9	0.03	0.02	0.005	13	0.9	14	0.25	57.8
DC06-1249	4	9	8.2	0.03	--	--	15	1.3	12		57.8
Other Misc. Mine Site (Statistical Summary)											
P_{5}	0	1	5.4	0.01	0.005	0.005	2.0	0.2	5.8	0.025	6.4
P_{25}	0	2	5.9	0.01	0.005	0.006	6.3	0.2	20	0.025	10.3
P_{50}	0	3	6.5	0.01	0.005	0.010	8.0	0.3	30	0.025	13.9
P_{75}	1	5	7.0	0.01	0.010	0.010	9.8	0.3	51	0.025	19.8
P_{95}	3	6	7.4	0.02	0.020	0.020	14	0.6	70	0.025	105
Maximum	5	6	7.5	0.04	0.040	0.020	55	1.3	352	0.08	109

Source: SRK 2013c and associated documentation.

Table 3.7-24: Selected Meteoric Water Mobility Procedure Results for Overburden Samples

Parameter	Unit	Colluvium Coarse	Colluvium Medium	Colluvium Fine	Terrace Gravels Coarse	Terrace Gravels Medium Mineralized Area	Terrace Gravels Fine Mineralized Area
Major Constituents							
pH	pH Units	6.76	6.26	6.57	6.4	6.48	6.36
Calcium	mg/L	0.94	39	9.5	1.2	1.2	<0.5
Total Alkalinity	mg/L as $CaCO_3$	6	20	22	4	10	4
Magnesium	mg/L	<0.5	4.4	1.3	0.71	<0.5	<0.5
Potassium	mg/L	<0.5	4.2	0.72	0.68	<0.5	<0.5
Sodium	mg/L	3.2	11	8.8	5.9	2.2	1.7
Chloride	mg/L	1.9	4.6	3.1	1.9	<1	<1
Fluoride	mg/L	0.34	0.22	0.17	0.19	0.33	0.23
Sulfate	mg/L	3.4	67	11	3.2	<1	<1
Metals							
Aluminum[3]	µg/L	490	230	800	4,200	<45	<45
Antimony	µg/L	<2.5	<2.5	<2.5	<2.5	<2.5	<2.5
Arsenic	µg/L	11	<5	<5	<5	6	12
Barium	µg/L	13	380	58	57	<10	<10
Beryllium	µg/L	<1	<1	<1	<1	<1	<1
Cadmium	µg/L	<1	<1	<1	<1	<1	<1
Copper	µg/L	<50	<50	<50	<50	<50	<50
Iron	µg/L	460	280	1,200	5,900	17	45
Lead	µg/L	<10	<10	<10	<10	<10	<10
Manganese	µg/L	19	480	86	120	<5	6.3
Nickel	µg/L	<10	<10	<10	<10	<10	<10
Selenium	µg/L	<5	<5	<5	<5	<5	<5
Zinc	µg/L	<10	14	19	49	<10	<10

Notes:

1 All metals are dissolved concentrations.

Source: SRK 2007.

Table 3.7-25: Mineralogy and Acid-Base Accounting on Tailings Samples

Parameter	Unit	2006 Pilot Tails FT-1	2007 Phase 1 Pilot Tails FT-4
Pyrite	%, wt	1.19	0.45
Arsenopyrite	%, wt	0.05	0.26
Gypsum	%, wt	0.95	1.55
Jarosite	%, wt	0.57	0.17
Ankerite	%, wt	1.77	1.89
Dolomite	%, wt	0.41	1.10
Magnesite	%, wt	Not reported	0.23
Hematite	%, wt	0.02	1.00
Paste pH	s.u.	7.65	7.39
Total Sulfur	S,%	0.780	0.578
Sulfide	S,%	0.15	0.11
Sulfate	S,%	0.63	0.47
Estimated Sulfate as Jarosite	S,%	0.14	0.03
NP	kg $CaCO_3$/t	24.5	11.9
AP as Sulfide	kg $CaCO_3$/t	4.7	3.4
NP/AP (Sulfide AP)	ratio	5.2	3.5
AP as Jarosite	kg $CaCO_3$/t	3.4	0.8
NP/AP	ratio	3.0	2.8
Carbonate	%, CO_3	1.03	0.753

Source: SRK 2007.

In 2007 and 2011, SRK found that the Phase I pilot plant tailings have an uncertain potential for ARD, based on NP/AP values from samples ranging from 1.1 to 3.0. The same firm reported NP/AP values, for a Phase 2 final tailings sample analyzed in 2008, of 12.5 (SRK 2012e). The values show a high variation because both the NP and the AP were low. (The Phase 1 NP was 11.9 kg $CaCO_3$/t and the 2008 Phase 2 NP was 28.5 kg $CaCO_3$/t; whereas Phase 1 AP was 3.4 kg $CaCO_3$/t and the 2008 Phase 2 AP was 2.28 kg $CaCO_3$/t.)

Four HCTs were conducted on rinsed final tailings samples with the characteristics shown in Table 3.7-26 (SRK 2011). As was done for the waste rock samples, the NP was calculated based on the calcium, magnesium carbonate mineralogy. One of the HCTs was terminated after about two years. The other three have been ongoing for over four years. The leachate chemistry from the HCTs is mostly calcium and sulfate. Arsenic leaching increased initially in all HCTs, reached a peak, and declined thereafter. Leaching of most other constituents has been low, near the detection limits (SRK 2011). Leachate from the tests remained at near-neutral pH values throughout the tests (SRK 2011). However, calculations also predicted that tailings solutions could have concentrations of sulfate possibly in excess of 10,000 mg/L and arsenic concentrations on the order of 2 to 20 mg/L due to recycling of tailings water into the flotation and hydrometallurgical processes (SRK 2007). Sulfate is expected to increase beyond that expected in equilibrium with gypsum, because test work indicates that magnesium concentrations would exceed calcium concentrations in the tailings discharge (SRK 2011). Most of the reagents used in the metallurgical process would be expected to be oxidized during the metallurgical process. Breakdown of residual reagents would be expected during the neutralization step at the process facilities and in the process.

Phase 1 (2006) and Phase 2 (2007) final tailings samples were subjected to standard MWMP leaching. The resulting leachates were near neutral pH (Table 3.7-27). The leachate chemistries were dominated by sulfate, calcium, and magnesium, reflecting mainly gypsum dissolution. Arsenic leaching was in the range of 0.16 to 0.41 mg/L.

Table 3.7-26: Acid-Base Accounting for Tailings Samples Submitted for Humidity Cell Testing

Tailings Sample	Acid Potential Indicated by Different Mineralogical Forms			Neutralization Potential by Analysis and Mineralogy		$NP_{Ca,Mg}$ /AP
	Sulfide kg CaCO3/t	Jarosite kg CaCO3/t	Total AP kg CaCO3/t	Total kg CaCO3/t	Ca,Mg CO3 kg CaCO3/t	
FT Neutralized Prussian Blue Test	5.3	2.3	7.6	39.5	19.8	2.6
FT + CSS Neut. Slurry + CN Detox Tls (Air/SO_2)	4.7	4.5	9.2	25.9	10.4	1.1
Final Tails - CN Detox by Air/SO_2	4.7	3.4	8.1	24.5	10.4	1.3
FT Neutralized Slurry + CN Detox Tails	4.7	0.7	5.4	12.2	9.8	1.8

Source: SRK 2011.

Table 3.7-27: Meteoric Water Mobility Procedure Results for Tailings Samples

Parameter	Units	Most Stringent Applicable Water Quality Criterion	2006 Tailings Final Tailings With CN Detox By Air/SO_2	2007 Tailings FT Neutralized Slurry + CN Detox Tails ($S0_2$/Air)
Major Constituents				
pH	--	6.5-8.5[1]	7.06	7.47
Alkalinity	mg/L as $CaCO_3$	20 (min)[2b]	10	28
Calcium	mg/L	--	521	483
Magnesium	mg/L	--	188	304
Potassium	mg/L	--	25.2	28.1
Sodium	mg/L	--	49.9	31.0
Chloride	mg/L	230[2b]	9.8	4.2
Fluoride	mg/L	1[2e]	0.65	0.37
Sulfate	mg/L	250[1]	2400	2700
Nitrite	as N mg/L	--	<0.06	<0.6
Nitrate	as N mg/L	--	2.38	<0.5
Total Ammonia	as N mg/L	2.99[2g]	6.4	7.6

Table 3.7-27: Meteoric Water Mobility Procedure Results for Tailings Samples

Parameter	Units	Most Stringent Applicable Water Quality Criterion	2006 Tailings Final Tailings With CN Detox By Air/SO_2	2007 Tailings FT Neutralized Slurry + CN Detox Tails ($S0_2$/Air)
Total Cyanide	mg/L	--	<0.01	0.05
WAD Cyanide	mg/L	0.0052^{2b}	<0.01	0.04
Free Cyanide	mg/L	--	<0.02	<0.05
Cyanate	mg/L	--	<1	<1
Thiocyanate	mg/L	--	13	0.8
Silver	µg/L	34.9^{2ab}	<0.05	<5
Metals				
Aluminum	µg/L	$87/750^{2ab}$	<10	28.2
Antimony	µg/L	6^{2d}	130	58.8
Arsenic	µg/L	10^{2d}	162/168	406
Cadmium	µg/L	0.64^{2ab}	0.17	0.13
Cobalt	µg/L	50^{2e}	13.5	39.5
Copper	µg/L	29^{2ab}	10.6	4.6
Iron	µg/L	$1,000^{2b}$	20	120
Lead	µg/L	10.9^{2ab}	0.29	0.29
Manganese	µg/L	50^{2f}	5,250	8,250
Molybdenum	µg/L	10^{2e}	120	64.6
Nickel	µg/L	168^{2ab}	28.2	16.6
Selenium	µg/L	4.6^{2b}	5	14
Thallium	µg/L	1.7^{2f}	0.5	0.3
Zinc	µg/L	379^{2ac}	27.9	14.7

Notes:

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.

2a Aquatic life for fresh water hardness-dependent criteria. Per ADEC 2008, the maximum hardness value shall not exceed 400 mg/l even if the actual ambient hardness is greater than 400 mg/l as calcium carbonate. A hardness of 400 mg/L was used for all calculations. For aluminum, if pH≥7.0 and hardness ≥50, then 750 µg/L.

2b Aquatic life for fresh water (chronic) criteria. Free cyanide measured as weak acid dissociable (WAD) cyanide.

2c Aquatic life for fresh water (acute) criteria.

2d Drinking water primary maximum contaminant levels.

2e Irrigation water criteria.

2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).

2g Aquatic life for fresh water (chronic) criteria based on pH and temperature when early life stages of fish are present. Standard for ammonia is based on 85th percentile of pH (7.85) and temperature (6.83°C) measured at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).

Source: SRK 2007.

A 2007 Phase 2 final tailings sample was subjected to not only the standard MWMP, but also to a series of rinses to simulate the exposure of beach tailings to natural precipitation after deposition in the tailings storage facility. The TDS and sulfate concentrations decreased somewhat with the rinses; however, the dissolved arsenic concentrations stayed relatively constant (Tables 3.7-28 and 3.7-29).

Table 3.7-28: Meteoric Water Mobility Procedure and Subsequent Rinsing of Tailings, General Water Quality Parameters

Parameter	Unit	Most Stringent Applicable Water Quality Criterion	Leachate	1st DI Rinse	2nd DI Rinse	3rd DI Rinse	4th DI Rinse
Major Constituents							
pH		6.5-8.5[1]	6.85	6.86	6.83	6.65	7.06
Calcium	mg/L	-	528	512	555	326	378
Magnesium	mg/L	-	140	122	52	8.72	5.58
Potassium	mg/L	-	17.2	15.9	11.5	5.16	4.7
Sodium	mg/L	-	85.5	38.5	1.,76	0.34	0.69
Chloride	mg/L	230[3]	3.2	<2	<2	<2	<2
Fluoride	mg/L	1[2c]	0.45	0.50	0.47	0.30	0.27
Sulfate	mg/L	250[1]	2000	1900	1500	850	1600
Alkalinity	mg/L as $CaCO_3$	>20[2a]	28	32	35	20	27
Acidity	mg/L as $CaCO_3$	-	<2	<2	<2	<2	<2
Conductivity	µS/cm	-	2,960	2,770	2,350	1,390	1,440
Carbonate	mg/L as $CaCO_3$	-	<2	<2	<2	<2	<2
Bicarbonate	mg/L as $CaCO_3$	-	28	32	35	20	27
OH	mg/L as $CaCO_3$	-	<2	<2	<2	<2	<2
Total Ammonia	as N mg/L	2.99[2d]	0.8	0.4	0.1	<0.1	0.2
Nitrite	as N mg/L	1[2b]	8.63	3.54	1.27	0.38	1.84
Nitrate	as N mg/L	10[2b]	<0.5	<0.5	<0.5	<0.05	0.16
Total Dissolved Solids	mg/L	500[1]	2,900	2,820	2,370	1,160	1,440
Total Suspended Solids	mg/L	-	5	3	2	3	2
Cyanide							
Total Cyanide	µg/L	-	40	12	14	18	31
WAD Cyanide	µg/L	5.2[2a]	<10	<5	<5	<5	7

Notes:

Shaded cells exceed most stringent water quality standard.

Applicable Water Quality Criteria (AWQC):

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.
2a Aquatic life for fresh water (chronic) criteria. Free cyanide measured as weak acid dissociable (WAD) cyanide.
2b Drinking water primary maximum contaminant levels.
2c Irrigation water criteria.
2d Aquatic life for fresh water (chronic) criteria based on pH and temperature when early life stages of fish are present. Standard for ammonia is based on 85th percentile of pH (7.85) and temperature (6.83°C) measured at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).

Source: SRK 2007, Table 4-5, Phase 2 pilot transitional final tailings samples.

Table 3.7-29: Meteoric Water Mobility Procedure and Subsequent Rinsing of Tailings, Metals Results

Parameter	Unit	Most Stringent Applicable Water Quality Criterion	Leachate	1st DI Rinse	2nd DI Rinse	3rd DI Rinse	4th DI Rinse
Aluminum	µg/L	87[2a,b]	<10	<10	20	20	10
Antimony	µg/L	6[2d]	39.3	44.8	43.1	23.9	22.2
Arsenic	µg/L	10[2d]	463	506	500	412	402
Barium	µg/L	2,000[2d]	16	18.5	18.9	13.1	42.8
Beryllium	µg/L	4[2d]	<0.02	<0.02	<0.02	<0.02	<0.02
Boron	µg/L	750[2e]	160	158	73.6	20.7	31.6
Bismuth	µg/L	--	<0.01	<0.01	<0.01	<0.01	<0.01
Cadmium	µg/L	0.64[2a,b]	0.083	0.061	0.043	0.017	0.045
Cerium	µg/L	--	<0.07	<0.07	<0.07	<0.07	<0.07
Chromium, total	µg/L	100[2e]	<0.5	<0.5	<0.5	<0.5	<0.5
Cobalt	µg/L	50[2e]	6.65	5.84	3.62	1.5	1.89
Copper	µg/L	29[2a,b]	7.8	5.2	<3	1.5	3.6
Gallium	µg/L	--	0.02	0.05	0.04	0.05	0.06
Germanium	µg/L	--	0.15	0.15	0.08	0.04	0.05
Hafnium	µg/L	--	0.098	0.0048	0.061	0.018	0.02
Indium	µg/L	--	<0.01	<0.01	<0.01	<0.01	<0.01
Iron	µg/L	1,000[2b]	30	50	20	<10	30
Lanthanum	µg/L	--	<0.04	<0.04	<0.04	<0.04	<0.04
Lead	µg/L	10.9[2a,b]	0.22	3.6	0.8	0.56	0.2
Lithium	µg/L	2,500[2e]	2	2	2	2	2
Manganese	µg/L	50[2f]	3,270	3,140	2,060	870	966
Molybdenum	µg/L	10[2e]	36.8	30.9	11.5	2.68	4.77
Nickel	µg/L	168[2ab]	14.2	12.2	7.3	3.3	18.5
Niobium	µg/L	--	0.002	0.006	<0.001	<0.001	<0.001
Rubidium	µg/L	--	9.33	7.95	4.84	2.39	1.86
Rhenium	µg/L	--	<0.2	<0.1	<0.1	<0.1	<0.1
Selenium	µg/L	4.6[2b]	4	3	<1	<1	<1
Silicon	µg/L	--	2,870	3,560	3,520	1,830	2,030
Silver	µg/L	34.9[2ab]	<0.01	<0.01	<0.01	<0.01	<0.01

Table 3.7-29: Meteoric Water Mobility Procedure and Subsequent Rinsing of Tailings, Metals Results

Parameter	Unit	Most Stringent Applicable Water Quality Criterion	Leachate	1st DI Rinse	2nd DI Rinse	3rd DI Rinse	4th DI Rinse
Strontium	µg/L	--	1,900	1,860	1,820	94.1	0.3
Tantalum	µg/L	--	0.013	0.008	0.01	0.04	0.001
Tellurium	µg/L	--	<0.03	<0.06	<0.03	<0.03	<0.03
Thallium	µg/L	1.7[2f]	0.058	0.046	0.02	0.004	0.01
Thorium	µg/L	--	0.841	0.519	0.634	0.042	1.49
Tin	µg/L	--	0.14	0.32	0.35	0.35	0.45
Titanium	µg/L	--	0.3	0.6	0.3	0.3	0.3
Tungsten	µg/L	--	0.19	0.24	0.23	0.11	0.16
Uranium	µg/L	--	1.19	1.77	0.978	0.565	0.956
Vanadium	µg/L	100[2e]	0.25	0.41	0.3	0.6	0.59
Yttrium	µg/L	--	0.01	0.013	0.008	0.005	0.009
Zinc	µg/L	379[2ac]	8	7	4	3	10
Zirconium	µg/L	--	0.14	0.07	0.32	0.23	0.03
Mercury	ng/L	12[2b]	10	10	7	5	3

Notes:

Shaded cells exceed most stringent water quality standard.

AWQC:

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.

2a Aquatic life for fresh water hardness-dependent criteria. A hardness of 400 mg/L was used for all calculations. For aluminum, if pH≥7.0 and hardness ≥50, then 750 µg/L; otherwise, 87 µg/L.

2b Aquatic life for fresh water (chronic) criteria. Free cyanide measured as weak acid dissociable (WAD) cyanide.

2c Aquatic life for fresh water (acute) criteria.

2d Drinking water primary maximum contaminant levels.

2e Irrigation water criteria.

2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).

Source: SRK 2012b: Table 4-6, 2007 Phase 2 pilot transitional final tailings samples.

3.7.2.5 CLIMATE CHANGE

Climate change is affecting resources in the EIS Analysis area and trends associated with climate change are projected to continue into the future. Section 3.26.3, Climate Change, discusses climate change trends and impacts to key resources in the physical environment including atmosphere, water resources, and permafrost. Current and future effects to water quality are tied to changes in water resources (discussed in Section 3.26.3.2, Climate Change).

3.7.3 ENVIRONMENTAL CONSEQUENCES

This section describes potential impacts to surface water quality, groundwater quality, and sediment quality as a result of the project. Table 3.7-30 provides the impact methodology framework applied to assessing direct or indirect impacts to water quality based on four factors of intensity or magnitude, duration, extent or scope, and context (40 CFR 1508.27, described in Section 3.0, Approach and Methodology).

Table 3.7-30: Impact Methodology for Effects on Surface Water Quality, Groundwater Quality, and Sediment Quality

Type of Effect	Impact Factor	Assessment Criteria		
Changes to surface water quality, groundwater quality, and sediment quality	Magnitude or Intensity	Effects are below water quality regulatory limits or sediment quality guidelines. Effects may not be noticeable or measurable.	Not applicable	Effects are sufficient to exceed water quality regulatory limits and baseline ranges; mitigation measures are not effective.
	Duration	Water quality or sediment quality would be reduced infrequently but not longer than the span of the project construction and would be expected to return to pre-activity levels at the completion of the activity.	Water quality or sediment quality would be reduced through the life of the mine and for a duration of up to 100 years after the end of construction; however, would return to pre-activity levels sometime during that period.	Water quality or sediment quality would be reduced and would not be anticipated to return to previous levels or would take longer than 100 years to do so.
	Extent or Scope	Affects water or sediment quality within discrete portions of the project area affected. Impacts can be contained from moving downstream or throughout a waterbody.	Affects water or sediment quality potentially throughout the EIS Project Area or outside the project footprint. Impacts affect hydraulically connected waters.	Affects water or sediment quality beyond the EIS Analysis Area. Impacts affect hydraulically connected waters.
	Context	Affects areas of ordinary water or sediment quality or where there is an abundance of water or sediment resources.	Affects areas with high water or sediment quality or water or sediment resources that are considered valuable in the region.	Affects areas of high water or sediment quality that are protected by legislation.

3.7.3.1 ALTERNATIVE 1 – NO ACTION

Under the No Action Alternative, construction, operations, and closure activities associated with the proposed project would not take place and the required water quality-related permits would not be issued. The rocks at the Mine Site would continue to slowly weather in place as they are presently doing. Background water quality in the vicinity of the Mine Site would continue to contain certain constituents above regulatory levels due to natural mineralization and previous placer activities (Tables 3.7-2 through 3.7-4, 3.7-11, and 3.7-12). Water quality along the transportation corridor and pipeline would continue to contain naturally high levels of turbidity. Sediment quality in Crooked Creek watershed and Kuskokwim River would continue to contain certain metals above sediment quality guidelines (SQGs) (Tables 3.7-13 through 3.7-15). No project-related geochemical processes or impacts to surface water, groundwater, or sediment quality would occur under this alternative. Consequently, there would be no new effects, neither adverse nor beneficial, on these resources from implementation of the No Action Alternative.

3.7.3.2 ALTERNATIVE 2 – DONLIN GOLD'S PROPOSED ACTION

Based on comments on the Draft EIS from agencies and the public, one route option has been included in Alternative 2 to address concerns due to pipeline crossings of the Iditarod National Historic Trail (INHT):

- North Option: The MP 84.8 to 112 North Option would realign this segment of the natural gas pipeline crossing to the north of the INHT before the Happy River crossing and remain on the north side of the Happy River Valley before rejoining the alignment near MP-112 where it enters the Three Mile Valley. The North Alignment would be 26.5 miles long, with one crossing of the INHT and only 0.1 mile physically located in the INHT right-of-way (ROW). The average separation distance from the INHT would be 1 mile.

3.7.3.2.1 GEOCHEMISTRY OF MINE WATERS

Mine Site

Under Alternative 2, geochemical water-quality impacts are tied to several components of the Donlin Gold Project, specifically:

- The Waste Rock Facility (WRF);
- North Overburden (NOB) and South Overburden (SOB) Stockpiles;
- The Low-Grade Ore Stockpile and Ore Stockpile Berm;
- The Lower and Upper Contact Water Dam (CWD);
- Haul roads built with waste rock;
- The Tailings Storage Facility (TSF);
- The TSF Seepage Recovery System (SRS); and
- Open Pit and Pit Lake

Each of these components, except for haul roads, is discussed at length in Appendix H – Geochemistry. The haul roads are discussed in Section 3.7.3.2.2, Surface Water Quality, and Section 3.7.3.2.3, Groundwater Quality.

Summaries of the predicted chemistries of surface waters related to these components are given in the following subsections for mine water that would replace the natural environment (surface water). These include the Lower and Upper CWD, the TSF impoundment, and SRS during operations, and the pit lake post-Closure. Although these process-related waters would not be considered waters of the State of Alaska or subject to the APDES permitting program while on site, they may be treated and discharged to Crooked Creek, when necessary for water management, and they could represent a potential environmental impact, particularly for wildlife (see Section 3.12, Wildlife). Their expected water chemistries are presented in this section in comparison to most stringent Applicable Water Quality Criteria (AWQC) to provide the reader with a frame of reference for understanding the predicted concentrations of constituents.

Construction and Operations

Lower Contact Water Dam Pond

The Lower CWD is located in American Creek with the objective of managing runoff of contact water from the WRF and open pit. The Lower CWD would receive runoff from a variety of sources (BGC 2014b, 2015f; SRK 2017e):

- Surface and seepage runoff from the waste rock.
- Runoff from undisturbed ground upgradient of the waste rock.
- Surface runoff within the open pit footprint.
- Pit dewatering water not required for process or sent to the water treatment plant (WTP).
- Horizontal drains from the open pit.
- Runoff collected behind the ore stockpile berm.
- Runoff collected in a sediment pond downstream of the South Overburden (SOB) stockpile.

The Lower and Upper CWDs are designed to store water that will be used throughout the year as a source of fresh water for the process plant (SRK 2017b). Peak runoff is limited to the spring and summer months, with imperceptible runoff volumes between mid-October and the beginning of April. These variable flows are in contrast to the constant fresh water demand. During the former period, runoff volumes are in excess of fresh water requirements and this excess water will be stored. The stored water will be a useful source of fresh water during the fall and winter, when inflows are minimal. This water would be stored during construction and the Lower CWD would continue to receive water throughout the life of the mine. It would be monitored quarterly (SRK 2016e).

The water quality of the lower CWD would likely be variable, due to the variation in the input sources, both seasonally and over the longer term during the life of the mine. However, the quality of some of the source waters would be expected to decrease over time as weathering occurs in the PAG rock in the low-grade ore stockpile, the PAG 6 cells of the WRF, and the exposed surfaces of the pit and waste rock. Order-of-magnitude estimates of the water quality of the lower CWD under different input water assumptions are given in Tables 3.7-31 and H-6 (Appendix H). These estimates were developed using two approaches, as described below.

Table 3.7-31: Predicted Lower CWD Pond Concentrations

Units		AWQC1	95th Percentile Estimates (SRK 2017e, Appendix D, Table 6)			(AECOM 2018)				
						APDES Years 1- 5			APDES Years 11 - 26.25	
			APDES Years 1- 5	APDES Years 6 - 10	APDES Years 11 - 26.25	No Brine	Brine	APDES Years 6 - 10	Ore Stockpile same as PAG WRF	Ore Stockpile same as PAG 7
Major Constituents										
pH	S.U.	6.5-8.5[2]	8.1	8.1	8.1	8.1	8.2	7.0	6.4	5.0
Alkalinity as CaCO3	mg/L		107	105	98	110	120	88	74	76
Calcium	mg/L		168	218	218	68	140	130	440	250
Magnesium	mg/L		24	29	29	14	63	25	390	270
Potassium	mg/L		25	29	31	5.8	4.6	15	22	22
Sodium	mg/L		25	26	30	14	11	19	21	21
Chloride	mg/L	230 [2]	2.1	2.1	2.3	2.1	4.8	1.8	1.5	1.5
Sulfate	mg/L	250 [2]	513	659	670	190	520	470	2,800	6,500
Total Dissolved Solids (est)	mg/L	500 [2]	861	1,053	1,073	350	800	690	3,700	8,000
Metals										
Aluminum	µg/L	87 (T)	100	130	180	8,200	6,600	16,000	18,000	890,000
Antimony	µg/L	6 [2]	740	960	970	180	150	510	760	750
Arsenic	µg/L	10 [2]	2,900	4,400	4,400	1,100	920	3,200	4,400	4,600
Barium	µg/L	2000 [2]	190	190	200	190	150	130	89	89
Beryllium	µg/L	4 [2]	0.48	0.55	0.57	0.48	0.38	0.67	0.56	2.7
Boron	µg/L	750 [5]	140	160	170	50	40	96	130	130
Cadmium	µg/L	0.64 [1]	0.86	1.0	1.1	0.50	0.46	0.85	0.86	2.7
Chromium	µg/L	100 [2]	19	23	24	4.9	3.9	13	18	21

Table 3.7-31: Predicted Lower CWD Pond Concentrations

Units		AWQC1	95th Percentile Estimates (SRK 2017e, Appendix D, Table 6)			(AECOM 2018)				
						APDES Years 1- 5			APDES Years 11 - 26.25	
			APDES Years 1- 5	APDES Years 6 - 10	APDES Years 11 - 26.25	No Brine	Brine	APDES Years 6 - 10	Ore Stockpile same as PAG WRF	Ore Stockpile same as PAG 7
Cobalt	µg/L	50 [5]	44	62	61	16	12	36	49	74
Copper	µg/L	29.3 [1]	8.2	10	10	18	14	29	170	1,000
Iron	µg/L	1000 [4]	350	360	430	760	600	910	740	21,000
Lead	µg/L	10.9 [1]	68	99	98	22	21	59	260	190
Manganese	µg/L	50 [3]	1,800	2,500	2,500	690	560	1,600	18,000	12,000
Molybdenum	µg/L	10 [5]	170	230	240	46	46	120	190	180
Nickel	µg/L	168 [1]	270	380	380	78	62	210	300	320
Selenium	µg/L	4.6 [4]	200	270	270	73	67	190	1,200	2,900
Silver	µg/L	34.9 [1]	1.2	1.4	1.4	0.51	0.41	0.90	1.2	1.2
Thallium	µg/L	1.7 [3]	0.71	0.82	0.85	0.32	0.25	0.54	0.70	0.71
Zinc	µg/L	379 [1]	390	570	570	190	150	460	18,000	12,000
Mercury	ng/L	12 [3]	120	150	150	34	87	81	120	120

Notes:
Shaded cells exceed most stringent water quality standard.

1 Most stringent Alaska water quality criteria (AWQC), based on hardness values, where applicable. AWQC are in terms of dissolved, unless otherwise noted by (T) for total concentration.
2 Drinking water primary max. contaminant levels
3 Human health criteria for non-carcinogens (for consumption of water + aquatic organisms)4
4 Aquatic life criteria for fresh waters (chronic)
5 Alaska criteria for irrigation and stock water.
6 Source: AECOM (2018). For all chemistry calculations:
a) Undisturbed American Creek watershed, SOB and individual highwall rock category water qualities wre taken from SRK 2017e, App. A, Table 5;
b) The surface water drainage from the WRF has the concentrations used in PitMod (Lorax 2015);
c) Ratios of rock categories for the pit (averages over the time period), the NAG and PAG portions of the WRF (cumulative over the time period), and the ore stockpile were developed from the AP and NP schedule in Enos 2013c;
d) Average percentages for each modeled period of mined-pit area to total planned pit area and of ground covered, respectively, by active and reclaimed NAG and PAG WRF to the total planned WRF were estimated from BGC 2011, Appendix A end-of-period maps.
e) The third period ore stockpile drainage was assumed to be either well-mixed or poorly-mixed PAG drainage at closure.

Donlin Gold WTP Source Water Predictions for Lower CWD

The first approach was developed by SRK (2017b, 2017e, Appendix D) to predict source water concentrations for the design of the operations water treatment plant (WTP). It consists of using 75th percentile values of source terms for three LOM periods (APDES Years 1 through 5, Years 6 through 10; and Years 11 through 26.25). For all three cases, water quality estimates for the NAG and PAG waste rocks are from the early years of mining before the PAG cells are predicted to produce acidic drainage. The predicted results do not vary substantially for the three periods. The concentrations for Years 11 through 26.25 tend to be slightly higher than for the preceding years.

Independent Analysis of Lower CWD Water Quality

The second approach, an independent analysis completed by AECOM (2018), used a somewhat different method than that of SRK (2017e, Appendix D). In this approach, data from SRK (2017e) were used to estimate water quality of some flows to the lower CWD (e.g., from the SOB and undisturbed areas), but runoff from the pit and WRF were based on different ratios of rock categories for each modeled time period, rather than averages over the LOM. These ratios were developed from the mine AP and NP schedule (Figure 3.7-15) (Enos 2013c), mine production schedule (SRK 2016g, Appendix A), and end-of-period maps (BGC 2011b, Appendix A) as described in Appendix H. Drainage from the ore stockpile were based on ore and PAG 7 rock averages estimated from the same production schedule. Mixing of the predicted water chemistries from various sources was based on annual average relative flows within each modeling period, based on an interpolation of SRK (2017e, Figures 3-2 and 3-5). Runoff from reclaimed portions of the WRF was assumed to have the concentrations used in PitMod (Lorax 2012a, 2015); whereas runoff from active portions of the PAG and NAG WRF was assumed to be the same as their seepage chemistries. Areas not yet mined or covered in waste rock were assumed to have the drainage chemistries of undisturbed land.

Modeling in the second approach was based on the “what if” scenario that PAG rocks in the ore stockpile would produce acidic drainage during the third modeling period in the later mine life. While the SRK (2017e) approach assumed ore drainage would be similar to PAG 6 in the highwall, to be conservative, the second approach investigated two scenarios for ore in the stockpile during the latest APDES modeling period: 1) that drainage is equivalent to that of well-mixed PAG rock at Closure; and 2) that it is equivalent to poorly-mixed PAG rock at Closure. Two scenarios were also investigated for the first time period, assuming that: 1) no brine would be added to the Lower CWD; and 2) that 440 gpm of brine would be added, using a composition extrapolated from Hatch (2017).

Mixing in the second approach used PHREEQC, a public domain geochemical computer code developed by the U.S. Geological Survey (Parkhurst and Appelo 2013.) This type of modeling is based on known thermodynamic data, and is the standard method of estimating geochemical mixing and mineral precipitation across a spectrum of aqueous environments. The code allows both conservative mixing, which adjusts the final pH based on the acidities/alkalinities of the input waters, and mixing with mineral precipitation when the mineral is saturated and forming in the mixture. While no minerals were allowed to precipitate in the SRK (2017e) approach, the second approach considered variations caused by potential precipitation.

Comparison of Results

As shown in Table H-6, the two approaches to predicting water chemistry in the Lower CWD give similar results for several constituents for all three time periods, but deviate substantially for others, especially for the last time period, when use of PAG values for the ore stockpile in the second approach produces more acidic water than what is used in the first approach. The high concentrations of aluminum and iron in those waters are due to not allowing precipitation of aluminum and iron sulfate-oxyhydroxide minerals. If they were allowed to precipitate at saturation, one or both values would decrease substantially. Even if iron oxyhydroxide (ferrihydrite) precipitated, however, very little of the other constituents would adsorb onto mineral surfaces due to the relatively low pH values.

The results of both approaches suggest that several constituents would exceed the most stringent AWQC for most or all modeled scenarios. These include sulfate, TDS, aluminum, antimony, arsenic, cadmium, lead, manganese, molybdenum, nickel, selenium, and mercury. Cobalt, copper, and zinc also exceed standards for some scenarios. The pH is also outside the standard for the latest time period in the second approach, using the “what if” assumption (i.e., that PAG in the ore stockpile would produce acidic drainage), as is iron if not allowed to precipitate.

The major difference between the two approaches is that the “what if” scenario in the second approach predicts that Lower CWD water would likely eventually turn acidic as the PAG rock in the ore stockpile oxidizes, and the acid would trigger higher concentrations of sulfate, TDS, aluminum, iron, and certain metals. These results highlight the importance of monitoring the water quality of the various source areas over time. They also suggest that preferential processing of the PAG ore would likely result in better water quality for the Lower CWD over time.

Upper Contact Water Dam

The Upper CWD would receive water from the Lower CWD and runoff from undisturbed areas above the WRF. The relative volumes of the two types of water are predicted to be about 65 percent Lower CWD water and 35 percent runoff from undisturbed areas (SRK 2017e). Parallel to the Lower CWD, two approaches were used to predict the range of water chemistries for the Upper CWD. The results for both approaches are given in Tables 3.7-32 and H-7. Sulfate, TDS, antimony, arsenic, cadmium, lead, manganese, molybdenum, nickel, selenium, and mercury are predicted to exceed AWQC for both approaches. Aluminum, copper, iron, zinc, and pH are also predicted to not meet AWQC under the second approach that assumes that the ore stockpile would produce acidic drainage.

Table 3.7-32: Predicted Upper CWD Pond Concentrations

Units		[1]AWQC	1. Water Treatment Design Basis 95th Percentile Estimates (SRK 2017e, Appendix D, Table 7)			2. Independent Analysis[6] (AECOM 2018)				
						APDES Years 1- 5			APDES Years 11 - 26.25	
			APDES Years 1- 5	APDES Years 6 - 10	APDES Years 11 - 26.25	No Brine	Brine	APDES Years 6 - 10	Ore Stockpile same as PAG WRF	Ore Stockpile same as PAG 7
Major Constituents										
pH	S.U.	6.5-8.5[2]	8.4	8.2	8.5	8.2	8.3	7.8	7.5	5.1
Alkalinity as CaCO3	mg/L		113	97	118	110	120	99	89	91
Calcium	mg/L		128	147	151	57	100	100	300	180
Magnesium	mg/L		21	23	23	14	47	21	260	180
Potassium	mg/L		18	18	22	4.0	3.2	10	15	14
Sodium	mg/L		20	18	22	10	7.8	13	15	15
Chloride	mg/L	230 [2]	1.9	1.3	1.3	1.6	3.4	1.4	1.3	1.3
Sulfate	mg/L	250 [2]	365	423	437	140	360	320	1,800	4,300
Total Dissolved Solids (est)	mg/L	500 [2]	651	728	746	290	590	510	2,500	5,300
Metals										
Aluminum	µg/L	87 (T)	85	43	49	5,400	4,300	10,000	12,000	590,000
Antimony	µg/L	6 [2]	520	600	620	120	100	340	501	490
Arsenic	µg/L	10 [2]	2,000	2,600	2,700	760	610	2,100	2,900	3,000
Barium	µg/L	2000 [2]	160	88	89	140	120	110	77	77
Beryllium	µg/L	4 [2]	0.36	0.38	0.43	0.34	0.27	0.46	0.39	1.8
Boron	µg/L	750 [5]	100	100	120	36	29	66	89	89
Cadmium	µg/L	0.64 [1]	0.63	0.66	0.76	0.36	0.33	0.59	0.59	1.8
Chromium	µg/L	100 [2]	14	15	17	3.6	2.9	8.6	12	14
Cobalt	µg/L	50 [5]	31	38	39	11	8.4	24	32	49
Copper	µg/L	29.3 [1]	6.3	6.6	6.8	12	10	19	110	680

Table 3.7-32: Predicted Upper CWD Pond Concentrations

Units		[1]AWQC	1. Water Treatment Design Basis 95th Percentile Estimates (SRK 2017e, Appendix D, Table 7)			2. Independent Analysis[6] (AECOM 2018)				
						APDES Years 1- 5			APDES Years 11 - 26.25	
			APDES Years 1- 5	APDES Years 6 - 10	APDES Years 11 - 26.25	No Brine	Brine	APDES Years 6 - 10	Ore Stockpile same as PAG WRF	Ore Stockpile same as PAG 7
Iron	µg/L	1000[4]	290	140	140	500	400	600	490	14,000
Lead	µg/L	10.9[1]	47	59	61	14	14	39	170	120
Manganese	µg/L	50[3]	1,300	1,500	1,600	460	370	1,000	12,000	7,700
Molybdenum	µg/L	10[5]	120	150	150	31	31	82	120	120
Nickel	µg/L	168[1]	190	230	240	52	41	140	200	210
Selenium	µg/L	4.6[4]	140	170	170	48	44	120	800	1,900
Silver	µg/L	34.9[1]	0.90	0.90	1.00	0.39	0.32	0.65	0.84	0.84
Thallium	µg/L	1.7[3]	0.55	0.58	0.64	0.26	0.22	0.41	0.52	0.52
Zinc	µg/L	379[1]	280	340	350	130	110	300	12,000	7,900
Mercury	ng/L	12[3]	89	93	110	23	58	53	77	77

Notes:

Shaded cells exceed most stringent water quality standard.

1 Most stringent Alaska water quality criteria (AWQC), based on hardness values, where applicable.
AWQC are in terms of dissolved, unless otherwise noted by (T) for total concentration.

2 Drinking water primary max. contaminant levels

3 Human health criteria for non-carcinogens (for consumption of water + aquatic organisms)4

4 Aquatic life criteria for fresh waters (chronic)

5 Alaska criteria for irrigation and stock water.

6 Source: AECOM (2018). For all chemistry calculations:
a) Undisturbed American Creek watershed, SOB, and individual highwall rock category water qualities were taken from SRK 2017e, App. A, Table 5;
b) The surface water drainage from the WRF has the concentrations used in PitMod (Lorax 2015);
c) Ratios of rock categories for the pit (averages over the time period), the NAG and PAG portions of the WRF (cumulative over the time period), and the ore stockpile were developed from the AP and NP schedule in Enos 2013c;
d) Average percentages for each modeled period of mined-pit area to total planned pit area and of ground covered, respectively, by active and reclaimed NAG and PAG WRF to the total planned WRF were estimated from BGC 2011, Appendix A end-of-period maps.
e) The third period ore stockpile drainage was assumed to be either well-mixed or poorly-mixed PAG drainage at closure.

Tailings Storage Facility

The TSF is planned to be a fully lined facility constructed in six stages over the 27-year mine life. A SRS is planned immediately downstream of the main tailings dam, which would collect groundwater and any seepage from the lined TSF. This water is expected to be mainly groundwater and would be used as process water or pumped to the TSF pond. Reclaim water from the tailings pond would be recycled back to the process facility from a floating barge via a pipeline (SRK 2016a).

Tailings pond water is formed by recirculation as process water. It is assumed to have a concentration factor of three from initial tailings liquor, due to the recirculation (SRK 2015b). SRK (2015b) predicted the recirculated, concentrated TSF pond water using Geochemist's Workbench, a thermodynamic equilibrium code, and allowed precipitation of certain minerals if they had reached saturation in the concentrated water. The results are shown in Table 3.7-33 and H-8. The "Solubility Constrained" column is checked for those constituents whose concentrations were lowered due to mineral precipitation. SRK (2015b) also calculated concentrations of pore water deeper in the tailings based on reducing reactions by Dissolved Organic Carbon (DOC) due to the presence of residual process reagents. Predicted concentrations of TSF reclaim water used in an updated design basis for the operations WTP (Hatch 2017; SRK 2017b) are provided in the last column of Table 3.7-33.

Several constituents are predicted to exceed AWQC in all cases: sulfate, TDS, fluoride, antimony, arsenic, cadmium, manganese, molybdenum, selenium, and mercury. In addition, pH and iron are predicted to not meet AWQC in the deeper tailings pore water.

Along with the tailings pond water, two other types of water are likely to exist in the TSF during mine operations. One is pore water in contact with buried tailings. The other is surficial runoff of snowmelt and rainfall from the beach area. These are discussed in Appendix H, Geochemistry.

TSF Seepage Recovery System

A SRS is to be built at the downstream toe of the TSF that consists of a collection pond, groundwater monitoring and collection wells, and pumping system. It is designed to capture two sources of water: 1) potential seepage through the TSF liner and 2) surface water and groundwater that enters the rock underdrains (BGC 2011a). The SRS water may be treated at the Water Treatment Plant (WTP) and discharged to Crooked Creek, when needed, for water management.

Due to the TSF liner, the seepage is expected to be minimal, ranging from about 1.4 gpm in Year 1 to about 17.6 gpm in Year 27 (BGC 2016c). Flows from surface water and groundwater are expected to vary seasonally, with higher flow rates in summer than in winter. Flows are expected to increase over the first few years and then taper off again slowly by the end of mine life. Summer flows are predicted to range between about 550 gpm (at end of mine life) to about 1,030 gpm (around Year 4); whereas winter flows are predicted to range between about 380 to 690 gpm.

SRS water chemistry (Tables 3.7-34 and H-10) is predicted to be neutral pH. WAD cyanide, antimony, arsenic, iron, manganese, and mercury concentrations are expected to exceed AWQC (Hatch 2017; SRK 2017e).

Table 3.7-33: Tailings Pond Water and Pore-Water Quality in Buried Tailings

Parameter	Units	Most Stringent Applicable Water Quality Criterion	Solubility Constrained	Tailings Pond Water[4]	Buried Tailings – Process DOC[4]	WTP Design Basis – TSF Reclaim Water, Steady State[5]
Major Constituents						
Redox Potential	mV	--			300	-
pH	S. U.	6.5-8.5[1]	X	7.7	5.5	7.7
Calcium	mg/L	--	X	610	1,000	610
Magnesium	mg/L	--	X	440	1,000	1,733
Potassium	mg/L	--	X	120	120	120
Sodium	mg/L	--		1,100	1,100	1,100
Strontium	mg/L	--		7.9	7.9	7.9
Chloride	mg/L	230[2b]		26	25	26
Fluoride	mg/L	1[2e]	X	2	2	2
Sulfate	mg/L	250[1]	X	5,800	4,400	8,605
Silicon	mg/L	--		7	7	7
Alkalinity	mg/L	20 (min)[2b]		25	530	25
TDS	mg/L	500[1]		-	-7,779	11,550
Ammonia	mg/L	2.99[2g]		29	29	29
Cyanide						
WAD Cyanide	µg/L	5.2[2h]				1.6[6]
Metals						
Aluminum	µg/L	87/750[2a,b]	X	13	5.6	13
Antimony	µg/L	6[2d]	X	22	1,100	22
Arsenic	µg/L	10[2d]	X	3,300	15,000	3,300
Barium	µg/L	2,000[2d]	X	11	11	11
Beryllium	µg/L	4[2d]		<0.06	<0.06	0.03
Boron	µg/L	750[2e]		590	590	590
Cadmium	µg/L	0.64[2a,b]		0.73	0.73	0.73
Chromium	µg/L	100[2d]		12	12	12
Cobalt	µg/L	50[2e]		19	19	19
Copper	µg/L	29[2a,b]	X	18	18	18
Iron	µg/L	1,000[2b]	X	4.4	98,000	4.4
Lead	µg/L	10.9[2a,b]		3	3	3
Lithium	µg/L	2,500[2e]		<6	<6	3

Table 3.7-33: Tailings Pond Water and Pore-Water Quality in Buried Tailings

Parameter	Units	Most Stringent Applicable Water Quality Criterion	Solubility Constrained	Tailings Pond Water[4]	Buried Tailings – Process DOC[4]	WTP Design Basis – TSF Reclaim Water, Steady State[5]
Manganese	µg/L	50[2f]	X	2,000	2,000	2,000
Molybdenum	µg/L	10[2e]		230	230	230
Nickel	µg/L	168[2a,b]		62	62	62
Selenium	µg/L	4.6[2b]		42	42	42
Thallium	µg/L	1.7[2f]		0.41	0.41	0.41
Vanadium	µg/L	100[2e]		4.8	4.8	4.8
Zinc	µg/L	379[2a,c]		33	33	33
Mercury	ng/L	12[2a]		10,000[3]	10,000[3]	10,000[3]

Notes:

All tailings pond water and pore-water concentrations are “dissolved” and should be used as “average” annual.
Shaded cells exceed most stringent water quality standard.
AWQC:
1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012.
2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.
2a Aquatic life for fresh water hardness-dependent criteria. A hardness of >400 mg/L was used for all calculations. For aluminum, if pH≥7.0 and hardness ≥50, then 750 µg/L.
2b Aquatic life for fresh water (chronic) criteria.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water primary maximum contaminant levels.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life for fresh water (chronic) criteria based on pH and temperature when early life stages of fish are present. Standard for ammonia is based on 85th percentile of pH (7.85) and temperature (6.83°C) measured at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).
2h Aquatic life criteria (chronic) for free cyanide, measured as weak acid dissociable (WAD) cyanide.
3 Hatch (2015); Weglinski 2015a).
4 Source: SRK 2017e, Appendix C, Table C-8.
5 Source: SRK 2017e, Table 4-4; Hatch 2017, Table 4-2.
6 Source: SRK 2017e, Appendix D, Table D-8.

Source: SRK 2015a.

Table 3.7-34: Predicted TSF Seepage Recovery System Water Quality

Parameter	Units	AWQC	Estimated Water Quality		
			APDES Years 1- 5	APDES Years 6 - 10	APDES Years 11 - 26.25
Major Constituents					
pH		6.5-8.5[1]	7.8	7.8	7.8
Calcium	mg/L		38	46	90
Magnesium	mg/L		15	23	67
Potassium	mg/L		2.4	3.4	9
Sodium	mg/L		14	23	72
Chloride	mg/L	230[2b]	1.8	2	3
Fluoride	mg/L	1[2e]	0.1	0.12	0.2
Sulfate	mg/L	250 [1]	33	69	266
Ammonia	mg/L	2.99[2g]	1.2	1.4	2.6
Total Dissolved Solids	mg/L	500 [1]	210	273	615
Cyanide					
WAD Cyanide	µg/L	5.2[2b]	6.3	12	45
Metals					
Aluminum	µg/L	750[2a,b]	33	33	32
Antimony	µg/L	6[2d]	7.3	16	66
Arsenic	µg/L	10[2d]	99	220	890
Barium	µg/L	2,000[2d]	150	150	150
Beryllium	µg/L	4[2d]	0.065	0.065	0.065
Boron	µg/L	750[2e]	11	16	42
Cadmium	µg/L	0.64[2a,b]	0.079	0.085	0.11
Chromium	µg/L	100[2e]	0.63	0.73	1.2
Cobalt	µg/L	50[2e]	0.72	0.87	1.7
Copper	µg/L	29.3[2a,b]	1	1.2	1.9
Iron	µg/L	1,000[2b]	840	1,600	6,000
Lead	µg/L	10.9[2a,b]	0.23	0.25	0.38
Lithium	µg/L	2,500[2e]	1.6	1.6	1.6
Manganese	µg/L	50[2f]	440	450	520
Molybdenum	µg/L	10[2e]	3	4.9	15

Table 3.7-34: Predicted TSF Seepage Recovery System Water Quality

Parameter	Units	AWQC	Estimated Water Quality		
			APDES Years 1- 5	**APDES Years 6 - 10**	**APDES Years 11 - 26.25**
Nickel	µg/L	168[2a,b]	1	1.5	4.3
Selenium	µg/L	4.6 [2b]	1	1.4	3.2
Silver	µg/L	34.9[2a,c]	0.17	0.19	0.31
Thallium	µg/L	1.7[2f]	0.16	0.16	0.17
Vanadium	µg/L	100[2e]	3.1	3.1	3.2
Zinc	µg/L	379[2a,c]	10	11	12
Mercury	ng/L	12[2b,3]	84	170	610

Notes:
Values exceeding discharge criteria are highlighted.
1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels
2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.
2a Aquatic life for fresh water hardness-dependent criteria. A hardness of 400 mg/L was used for all calculations. For aluminum, if pH≥7.0 and hardness ≥50, then 750 µg/L; otherwise, 87 µg/L.
2b Aquatic life for fresh water (chronic) criteria. Free cyanide measured as weak acid dissociable (WAD) cyanide.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water primary maximum contaminant levels.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life for fresh water (chronic) criteria based on pH and temperature when early life stages of fish are present. Standard for ammonia is based on 85th percentile of pH (7.85) and temperature (6.83oC) measured at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).
Source: Hatch 2017; SRK 2017e, App. D, Table 9.

Closure

Pit Lake

Inflow Sources and Gradient

In Year 22 of the mine life, the final limits of the pit would be reached. At this point, all PAG 6 and PAG 7 waste rock mined in the Lewis Pit would be placed as backfill into the ACMA Pit. Rock classified as NAG 1-4 and PAG 5 would also be backfilled into the ACMA Pit as it is mined from the Lewis Pit (SRK 2016d, g).

After closure, pit dewatering would stop and the pit would be allowed to fill with water from several sources (Lorax 2012a), including:

- Groundwater inflows;
- Highwall precipitation runoff;
- Direct precipitation on the surface of the pit lake;
- Excess tailings water in the impoundment at the end of operations;
- Tailings consolidation water and cover infiltration water;

- Tailings SRS water consisting of natural groundwater and a small component of tailings porewater;
- NAG seepage water from the WRF;
- PAG seepage water from the isolated cells of the WRF; and
- Undisturbed runoff from American Creek watershed and runoff from the WRF reclamation cover.

According to the numerical hydrogeologic model developed by BGC (2011d, 2015g), for about 8 years after pit dewatering concludes, water would flow into the pit from the groundwater at higher elevations and from the pit into the pore space of the waste rock placed as backfill and into the localized bedrock outside of and surrounding the pit from which bedrock water had been removed during mining. Once the localized bedrock had refilled with water, the direction of flow would reverse and groundwater would move towards the pit. Thereafter, the seepage rate from the lake to groundwater would gradually decrease to zero as the lake reaches its managed final elevation approximately 30 feet below the designed spillway overflow elevation of 359 feet AMSL (BGC 2015g; SRK 2017c). Additional description of the temporary localized flow reversal into bedrock as the pit fills is given in Section 3.6, Groundwater Hydrology. The pit outflow may result in an elevated input of sulfate and metals and decreased pH to the deep bedrock portion of the aquifer for the first few years. Pit lake levels would be managed in post-Closure to maintain a net inward groundwater gradient to the pit by pumping from the lake surface and treating it at the WTP for discharge to Crooked Creek (SRK 2012b, f; BGC 2014b).

Treatment and discharge of pit lake surface water are predicted to begin approximately 53 years after closure, based on the most likely pit filling rate obtained in the BGC (2015g) hydrogeological model. Hydrogeologic sensitivity runs suggested that the timing of pit lake filling could be as little as 30 years, assuming a future wet climate due to climate change or a higher hydraulic conductivity, or as long as 60 years. The uncertainty in the pit filling rate is discussed in more detail in EIS Section 3.6, Groundwater Hydrology.

Most WRF seepage would report to the WRF underdrain, and from there, would flow by gravity to the pit rim. Pumping would be required to get TSF and SRS water to the pit rim, where it would be combined with the WRF seepage, then flow via a gravity-fed pipe to the bottom of the pit lake. Groundwater seepage through the pit walls could possibly contain a small fraction of WRF seepage that infiltrates the WRF underdrain, but this is expected to be a very minor contribution to the pit lake in comparison to the piped flows and overall groundwater seepage from around the pit. The denser TSF/WRF water piped to the bottom of the pit lake would likely stay below the pycnocline. Groundwater seepage into the pit lake through the walls would likely be less dense than the TSF/WRF water. Section 3.6, Groundwater Hydrology further describes the inflows and water balance in relation to the pit lake.

Pit Lake Modeling

PitMod, a one-dimensional (vertical) model, was developed and used by Lorax (2012a) to investigate the evolution of the post-Closure pit lake. The model allowed assessment of the physics and geochemistry of the pit filling process. It provided justification for the assumption that the pit lake would stratify, and originally predicted the quality of the water that would be discharged from the lake for 100 years following closure (SRK 2012b, 2017b). A number of sensitivity runs were conducted. The initial modeling and sensitivity runs were done under assumptions of an operations water management plan that allowed no discharge of process or seepage waters. These runs were also done with an early version of water quality estimates for each of the NAG and PAG categories. This early dataset was developed while humidity cell tests (HCTs) were still ongoing (SRK 2007).

When the Advanced Water Treatment (AWT) option was introduced for the operations WTP as part of Alternative 2, a revised model was developed (Lorax 2015). In this model, an updated set of predictions from SRK 2012h and 2012g was used for the water quality to be expected from each of the NAG and PAG categories. In 2017, the revised AWT model was run for an additional 100 years, for a total of 200 years. (The 200-year run is discussed in Section 4.4.2.6 Temporal Changes in Surface Water Quality and Pycnocline Depth.)

Detailed discussion of the model is given in Appendix H, Geochemistry (Section 4.4.2, Pit Lake Modeling). It includes discussion of general physical and chemical lake processes, as well as the predicted chemistries of all input waters to the lake.

Results of Sensitivity Analyses

Several sensitivity runs were conducted, as shown in Table 3.7-35. They are discussed in some detail in Appendix H, Geochemistry. The results of the sensitivity analyses suggest that the pit lake would develop a density stratification with poor quality, highly saline bottom waters overlain by low TDS (112 to 142 mg/L) surface water under all likely scenarios. For the decreased salinity case, the TDS decreased somewhat from the base case, although the metals and sulfate concentrations remained essentially unchanged. In the base case, as well as the extreme winds and decreased salinity sensitivity cases, aluminum, antimony, arsenic, cadmium, lead, manganese, selenium, and mercury concentrations exceeded AWQC.

One sensitivity run forced the lake to mix completely in Year 55 (Table 3.7-35). This scenario provides an upper bound assessment of the concentrations of constituents predicted in the surface waters of the lake for that time period. (The chemistry of overturned lake water is expected to vary depending on how many years after filling the lake overturns, because the volume of water beneath the pycnocline is predicted to increase over time due to filling from the bottom, as is discussed in Appendix H, Section 4.4.2.6, Temporal Changes in Surface Water Quality and Pycnocline Depth.) Following the complete mixing event, salinity would begin to decrease in surface water due to precipitation and undisturbed runoff, and a new pycnocline would form that would vary in concentration seasonally from a low during spring snow and ice melt to a seasonal high in winter. Following an overturn event in Year 55, sulfate concentrations are predicted to decrease to less than 400 mg/L by year 99 (Lorax 2012a).

The complete mixing event would require strong physical forcing to overcome the stability of the density gradient predicted by expected conditions. Wind forcing is one possibility that could contribute to mixing, so an extreme wind event scenario was conducted by specifying a steady 125 mph wind over a two-day interval during the summer of year 55. The resulting surface water concentrations remained essentially unchanged from the base case scenario, due to the predicted depth of the pycnocline at that time.

Neither the decreased salinity nor the extreme wind scenario prevented formation of a stratified lake during the modeling period. Thus, the results of the sensitivity analyses suggest that the pit lake would develop a density stratification, with poor quality, highly saline bottom waters overlain by low TDS (112 to 142 mg/L) surface water under all likely scenarios for the first 99 years post-Closure (Table 3.7-35).

In the base case, as well as all the sensitivity cases except complete mixing, antimony, arsenic, cadmium, lead, manganese, mercury, and selenium concentrations exceeded AWQC. The complete mixing sensitivity run forced the lake to mix completely in Year 55. In that year, the TDS was calculated to be 2,350 mg/L throughout the water column. By Year 65, stratification had re-established, and the surface water TDS had decreased to 918 mg/L (Table 3.7-35, Complete Mixing).

Model Limitations

The base case and sensitivity runs used exhausted PAG runoff concentrations for PAG wall rock. However, pit walls are known to undergo periodic sloughing events that expose fresh rock to weathering processes (Filipek 2004). Accordingly, the PAG runoff is more likely to have concentrations over time that range between those used in the model and those given for peak oxidation rates. (See Appendix H, Geochemistry, Table H-13.) To test the effects of using peak PAG concentrations for runoff, Lorax (2014b, c) conducted two additional peak-PAG model runs for their decreased salinity case: 1) a conservative case, in which no reactions were allowed but PHREEQC was used to calculate pH of the mixed solutions; and 2) a reactive case allowing precipitation of low-temperature saturated aluminum, iron, and manganese oxyhydroxide minerals (Table H-36).

In both peak PAG, decreased salinity cases, the results were similar to the base case (within a factor of two for most constituents). Lake surface-water concentrations of aluminum, antimony, arsenic, cadmium, lead, molybdenum, selenium, and mercury exceeded AWQC in all three cases. Copper exceeded AWQC in both peak PAG cases, but not in the base case. Manganese concentrations exceeded AWQC when not allowed to precipitate. The predicted pH in both peak-PAG cases was predicted to be outside water quality criteria: The modeled pH was 5.2 for the no-reaction peak PAG case, and 5.1 with mineral precipitation.

The base case and sensitivity runs used predicted inflow water quality based on a no-discharge water management plan. When AWT water management option was added to Alternative 2 leading to the current operations WTP design, Lorax (2015) developed another pit lake model using the revised water quality values from Table H-12 in Appendix H (Geochemistry) for the major inflow sources. Table 3.7-37 shows a comparison of the chemistries used under the AWT assumption and the assumption of the initial no-discharge water management plan (labeled "base case") for sources whose predicted chemistries changed under the different assumptions.

The resulting predicted water quality for the uppermost 33 feet of the pit lake, shown in Table 3.7-36, is similar to those predicted for the original water management plan. Aluminum, antimony, arsenic, cadmium, copper, iron, lead, manganese, molybdenum, selenium, and mercury concentrations exceeded AWQC. Although not modeled, pH was estimated to be between 5.0 and 6.0, less than the lower AWQC.

Table 3.7-35: Surface Water Quality Estimates for Pit Lake at Closure Year 99—Sensitivity Analysis (assuming Exhausted PAG Pit-Wall Runoff and No-Discharge Water Management)

Parameter[1]	Units	Most Stringent Applicable Water Quality Criteria	Base Case	Groundwater Sensitivity	Complete Mixing[3]		Extreme Winds Sensitivity[4]	Decreased Salinity Sensitivity
					Year 55	Year 65		
Sulfate	mg/L	250[1]	31	48	1,607	658	31	30
Total Dissolved Solids	mg/L	500[1]	135	142	2,370	918	137	112
Aluminum	µg/L	87[2a,b]	310	311	31,900	12,700	316	311
Antimony	µg/L	6[2d]	67	120	382	216	68	67
Arsenic	µg/L	10[2d]	112	198	2,440	1,060	116	112
Boron	µg/L	750[2f]	202	356	1,740	880	205	199
Cadmium	µg/L	0.18[2a,b]	0.24	0.33	0.54	0.4	0.24	0.24
Chromium	µg/L	100[2e]	4	6.7	9.7	7.5	4	4
Copper	µg/L	6.2[2a,b]	1.4	1.7	90	32	1.4	1.4
Lead	µg/L	1.6[2a,b]	2.3	3.8	91	38	2.3	2.2
Manganese	µg/L	50[2f]	128	136	5,770	2,350	131	128
Nickel	µg/L	36[2a,b]	11	19	153	70	12	11
Selenium	µg/L	4.6[2b]	20	36	128	70	21	20
Zinc	µg/L	81[2a,c]	13	21	745	304	14	13
Mercury	ng/L	12[2b]	25	42	183	100	25	26

Notes:

Values are for dissolved metals, and represent depths between 0 and 33 feet in lake. Shaded cells exceed most stringent water quality standard. The pH was not modeled, but estimated to be between 5 and 6, based on input pH values.

AWQC:

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.

2a Aquatic life for fresh water hardness-dependent criteria. A value of 65 mg/L as CaCO3 was used for hardness, based on model predictions for calcium and magnesium (not shown). For aluminum, if pH≥7.0 and hardness ≥50, then 750 µg/L, otherwise, 87 µg/L.

2b Aquatic life for fresh water (chronic) criteria.

2c Aquatic life for fresh water (acute) criteria.

2d Drinking water primary maximum contaminant levels.

2e Irrigation water criteria.

2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).

3 Represents Years 55 and 65 (complete mixing induced in Year 55).

4 Represents Year 56 (hurricane winds induced in Years 55 and 56).

5. Source: Lorax 2012 (Table 4-2).

Table 3.7-36: Comparison of Surface Water Quality Estimates for Pit Lake-- at Closure Year 100 for Exhausted-PAG (Base Case) and Peak-PAG Pit-Wall Runoff (Decreased Salinity Case), and for Closure Years 100 and 200 for the AWT Model

Parameter[1]	Units	AWQC[2]	Base Case	Peak-PAG Case		AWT Year 100	AWT Year 200[6,7]
			No Reaction	No Reaction[3]	Mineral Precipitation[4]	No Reaction[5]	
pH	Units	6.5 – 8.5[1]	6 (est.)	5.2	5.1	5.0 – 6.0	
Sulfate	mg/L	250[1]	31	48	48	41	183
Total Dissolved Solids	mg/L	500[1]	139	143	97	125	340[7]
Aluminum	µg/L	87[2a,b]	310	590	97	1,570	5,080
Antimony	µg/L	6[2d]	67	75	76	67	148
Arsenic	µg/L	10[2d]	112	190	190	114	656
Boron	µg/L	750[2f]	200	230	200	31	50
Cadmium	µg/L	0.18[2a,b]	0.24	0.31	0.31	0.35	0.49
Chromium	µg/L	100[2e]	4.0	4.3	4.3	4.6	6
Cobalt	µg/L	50[2e]	3.0	5	5	6.5	
Copper	µg/L	6.2[2a,b]	1.4	7.1	6.7	10.5	20
Iron	µg/L	1000[2b]	830	960	180	1,200	
Lead	µg/L	1.6[2a,b]	2.3	2.7	2.7	2.6	21
Manganese	µg/L	50[2f]	128	170	15	176	439
Molybdenum	µg/L	10[2e]	17	19	18	17	
Nickel	µg/L	36[2a,b]	11	20	20	19	59
Selenium	µg/L	4.6[2b]	20	23	23	20	50
Zinc	µg/L	81[2a,c]	13	45	47	53	238
Mercury	ng/L	12[2b]	25	28	28	25	1,000

Notes:

Values are for dissolved metals, and represent depths between 0 and 33 feet in lake. Shaded cells exceed most stringent water quality standard.

AWQC:

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.

2a Aquatic life for fresh water hardness-dependent criteria. A hardness of 65 mg/L as $CaCO_3$ was used for hardness, based on model predictions for hardness. For aluminum, if pH≥7.0 and hardness ≥50, then 750 µg/L, otherwise, 87 µg/L.

2b Aquatic life for fresh water (chronic) criteria.

2c Aquatic life for fresh water (acute) criteria.

2d Drinking water primary maximum contaminant levels.

2e Irrigation water criteria.

Table 3.7-36: Comparison of Surface Water Quality Estimates for Pit Lake-- at Closure Year 100 for Exhausted-PAG (Base Case) and Peak-PAG Pit-Wall Runoff (Decreased Salinity Case), and for Closure Years 100 and 200 for the AWT Model

Parameter[1]	Units	AWQC[2]	Base Case	Peak-PAG Case		AWT Year 100	AWT Year 200[6,7]
			No Reaction	No Reaction[3]	Mineral Precipitation[4]	No Reaction[5]	

2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).

3 Lorax. 2014b. PitMod_conservative_monthly_means.xls

4 Lorax. 2014c. PitMod_mean monthly_PHREEQC.xls. Aluminum, iron, and manganese low-temperature oxyhydroxide minerals were allowed to precipitate if saturated.

5 Lorax 2015; PitMod_AWT2015.xlsx; and Lorax 2017a.

6 Lorax 2017a

7 Lorax 2017b

Temporal Changes in Surface-Water Quality and Pycnocline Depth

In all modeled cases, pit lake surface-water concentrations appeared to remain relatively constant for the first 50 years or so after the lake reached its maximum allowed elevation, so that the results for all but the Complete Mixing case for Year 53 are essentially identical to those shown in Table 3.7-35 for Year 99. In contrast, in all cases, the depth of the top of the pycnocline was found to decrease (move up) over time (Figure 3.7-22). As the pycnocline depth decreased, so did the concentration difference above and below the pycnocline.

For the three cases given in Table 3.7-36 (the base case; the peak-PAG, decreased salinity case; and the AWT case), the top of the pycnocline was at essentially the same depth in Year 99 after Closure (Figure 3.7-23). However, as evidenced by the relative slopes of the pycnocline, the strength of the pycnocline varied among the three cases, suggesting that the decreased-salinity and AWT systems could be more vulnerable to overturn and mixing than the base case.

The modeled pycnocline moves up through the water column with time in all cases, likely because each year additional water would be added below the pycnocline, due to ongoing groundwater influx below the pycnocline and the location of the feed pipe near the lake bottom, increasing the volume of water in the hypolimnion (i.e., below the pycnocline); while water would be removed and treated from above the pycnocline. Therefore, simple mass balance considerations suggest that eventually the pycnocline would be close enough to the surface that wind and/or winter overturn would cause water below the pycnocline to mix with the surface water.

In all three modeled cases shown in Appendix H (Figures H-4 through H-6), the depth of the pycnocline decreases by about 100 feet between Years 53 and 99, or about 100 feet per 50 years. In Year 99, the top of the pycnocline is at about 100 feet below the surface, and its base is at about 200 feet below the surface. Because the pit is essentially an inverted truncated cone, it would be expected that the rate of rise in the pycnocline elevation would slow over time, but not stop.

Table 3.7-37: Water Quality of Major Inflows to Pit at Closure, Comparison of AWT and Base Case Inputs

Parameter	Units	Most Stringent Applicable Water Quality Criterion	PAG Cell Seepage		Initial Pit Lake Water		Tailings Pore-water Seepage	
			AWT[3]	Base Case[4]	AWT[5]	Base Case[6]	AWT[7]	Base Case[8]
pH	Units	6.5-8.5[1]	3.5	4.1	7.7	7.7	7.7	5.5
Sulfate	mg/L	250[1]	180,000	42,450	3,811	10,537	4,400	15,900
Total Dissolved Solids (est.)	mg/L	500[1]	210,000	160,000	5,500	14,000	8,300	23,000
Aluminum	µg/L	87/750[2a,b]	30,000,000	30,470,000	221	18.14	5.6	1.9
Antimony	µg/L	6[2d]	2,800	2,800	1,630	920	1,100	1,160
Arsenic	µg/L	10[2d]	27,000	27,000	12,600	6,600	15,000	15,700
Cadmium	µg/L	0.64[2a,b]	65	65	1	0.7	7.3	0.93
Copper	µg/L	29[2a,b]	33,000	35,600	22	310	18	674
Chromium	µg/L	100[2d]	130	130	22	11	12	7.23
Lead	µg/L	10.9[2a,b]	460	99	680	350	3	82.5
Manganese	µg/L	50[2f]	170,000	370,300	5,600	43,200	2,000	45,600
Selenium	µg/L	4.6[2b]	80,000	18,790	470	330	42	220
Zinc	µg/L	379[2a,c]	190,000	3,991,000	4,000	2,100	33	79
Mercury	ng/L	12[2b]	180	150	34,900	720	73,000	1,440

Notes:

Shaded cells exceed most stringent water quality standard.

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column, except for aluminum and mercury, which are in terms of total recoverable metal.
  2a Aquatic life for fresh water hardness-dependent criteria. A hardness of >400 mg/L was used for all calculations based on modeled values for hardness. For aluminum, if pH≥7.0 and hardness ≥50, then 750 µg/L, otherwise, 87 µg/L.
  2b Aquatic life for fresh water (chronic) criteria.
  2c Aquatic life for fresh water (acute) criteria.
  2d Drinking water primary maximum contaminant levels.
  2e Irrigation water criteria.
  2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).

3 75th Percentile chemistry for poorly mixed PAG at year 2050 based on scaling of humidity cell leachate data (SRK 2015c).

4 75th Percentile chemistry for poorly mixed PAG at year 2050 based on scaling of humidity cell leachate data (SRK 2012h).

5 Conservative mixture of 8,920 acre-feet of backfill runoff and 24,000 acre-feet of tailings supernatant water (Lorax 2015).

6 Conservative mixture of 8,920 acre-feet of backfill runoff and 24,000 acre-feet of tailings supernatant water (Lorax 2012a).

7 Predicted concentrations of tailings pore water (SRK 2015a).

8 Predicted concentrations of tailings pore water (Lorax 2012a).

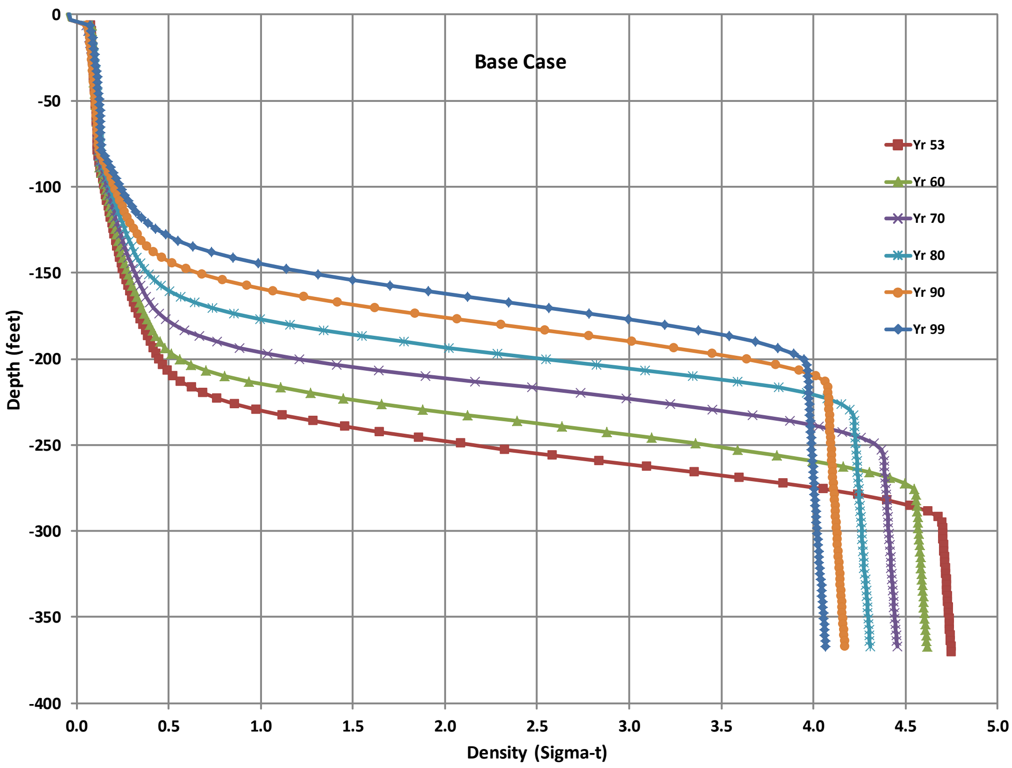
To test whether the modeled pycnocline would continue to move up the water column over time, Lorax (2017a, b, c) made an additional run extending the model to 200 years. Figures H-8a and H-8b (in Appendix H, Geochemistry) show that the rising pycnocline trend continues through the modeled year 200. Based on the complete mixing sensitivity run, after the lake overturns, the pit lake would likely eventually re-establish a pycnocline, but with higher constituent concentrations in the surficial layer, as occurred for the complete mixing case. At some future time beyond model Year 200, the base of the pycnocline would reach the pit lake surface. At that point, the pit lake would be completely mixed. A shallow seasonal pycnocline would likely form due to snow and ice melt, with lower surface-water concentrations in late-spring and summer, and higher concentrations in fall and winter. The actual concentrations of constituents in the resulting surface water would be mainly a function of the amount of sulfides remaining in the WRF and their oxidation rate at that time.

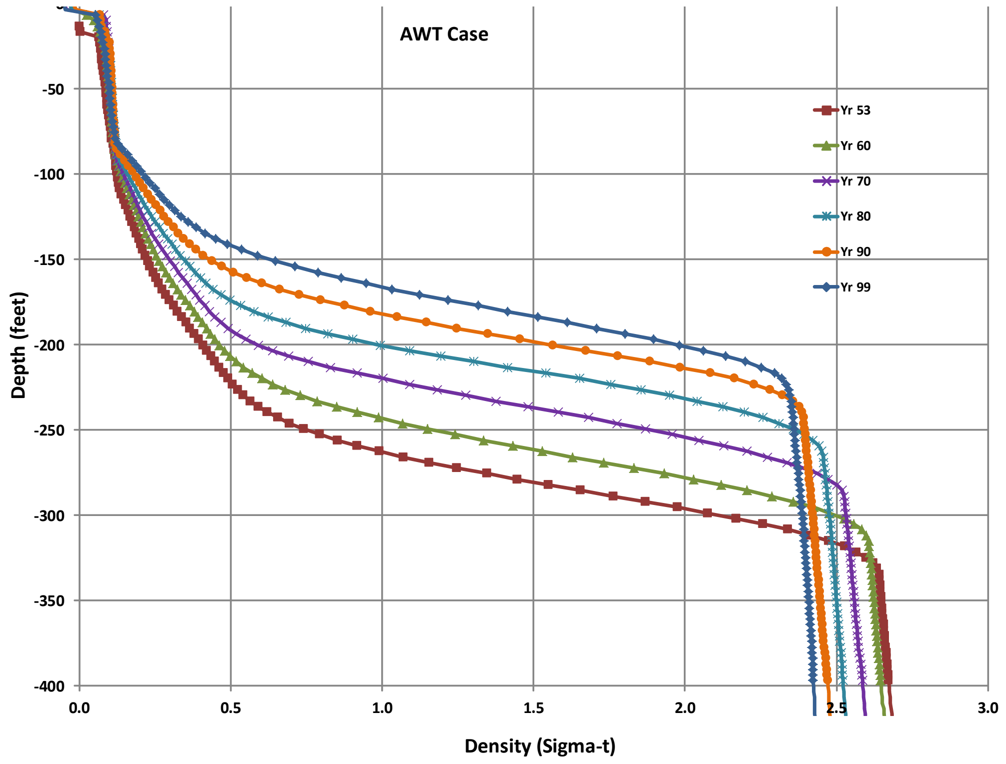
Summary of Pit Lake Modeling

The results of the various modeling efforts of the predicted pit lake suggest that the concentrations of several constituents in surface waters would exceed the most stringent AWQC throughout the 200-year modeling period. Additionally, the pycnocline is predicted to move upward toward the surface and become less intense over time, eventually reaching the surface at some point beyond the modeling period. Even before this time, as the pycnocline moves closer to the surface and becomes less intense, mixing by wind events becomes more likely. These conclusions are based both on the Lorax modeling and on simple mass balance considerations, because each year, most of the water entering the pit would be added below the pycnocline from the WRF and inflowing groundwater, and removed above the pycnocline. The actual concentrations of constituents in the resulting mixed surface water would be mainly a function of the amount of sulfides remaining in the WRF and their oxidation rate at that time. Pit-lake modeling contains a number of assumptions that can only be tested in real time after Closure. For these reasons, additional mitigation measures that would help manage lake stratification, surface water quality, and water treatment in post-Closure are provided in Chapter 5, Impact Avoidance, Minimization, and Mitigation.

Summary of Mine Site Geochemical Impacts

The water quality of drainages from the WRF and the isolated PAG cells, TSF, and SRS is predicted in all cases to exceed AWQC for several constituents, potentially for many tens of decades. The Lower and Upper CWD are also predicted to exceed AWQC for several constituents during operations. Drainage from the SOB may also exceed AWQC during operations. These drainages would be collected and stored in the Lower and Upper CWD for use in process water or for treatment and discharge to Crooked Creek, as needed for water management. In the post-Closure period, these drainages would be pumped to deeper layers of the pit lake, where they would be isolated from the surface for more than 200 years. Water from the pit lake would flow into backfill material, and also into the surrounding bedrock void space adjacent to the pit, during the period that the pit lake is filling towards its peak managed level. After the pit lake fills, groundwater (including the contact water from the pit lake) would flow back into the pit lake from all directions and at all times, as long as the pit lake level is maintained at approximately its design level.





DONLIN GOLD
PROJECT EIS

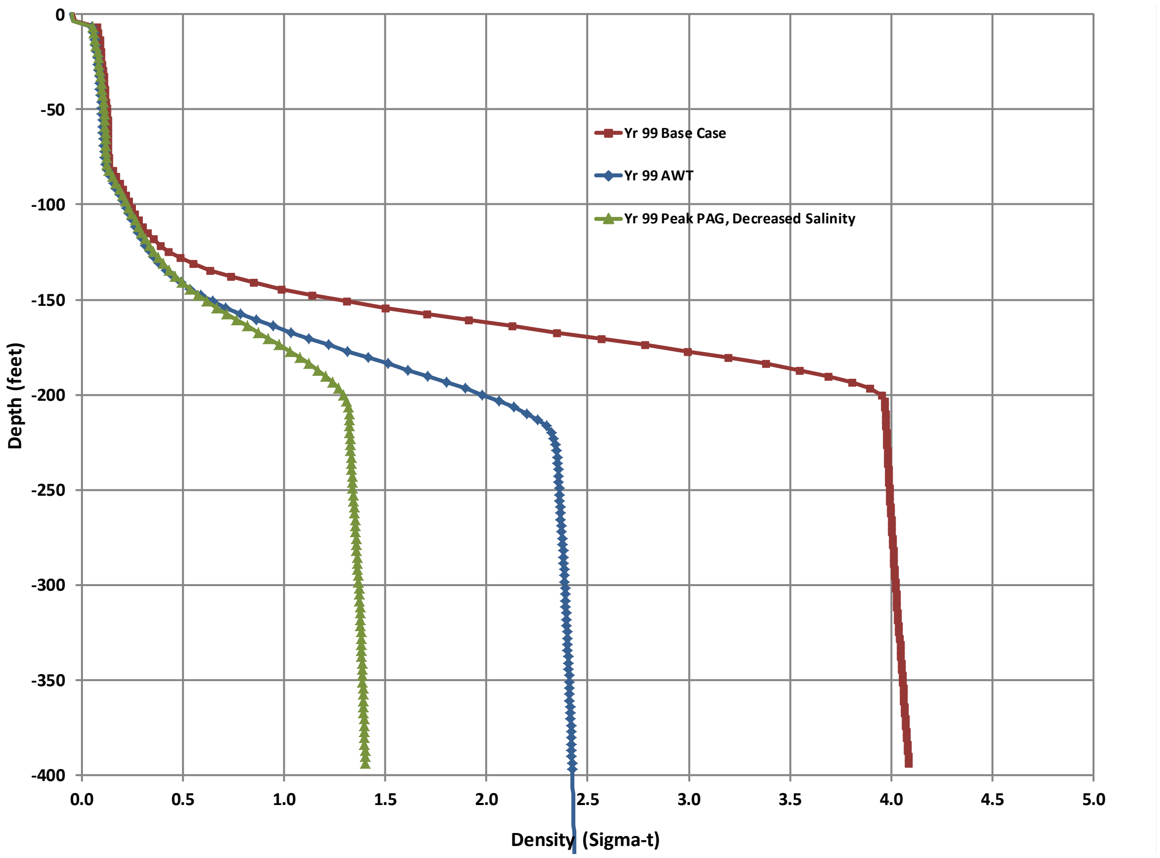
VARIATION OF PYCNOCLINE DEPTH WITH AGE OF PIT LAKE (55 TO 99 YEARS) BASE CASE AND AWT WATER MANAGEMENT PLAN CASE

JUNE 2017

FIGURE 3.7-22

Data Source: Lorax 2015

Note:
Scale change for Density



Data Source: Lorax 2015

DONLIN GOLD PROJECT EIS

COMPARISON OF PYCNOCLINE DEPTH AND STRENGTH FOR THE BASE CASE, PEAK-PAG, DECREASED SALINITY CASE, AND AWT CASE (YEAR 99)

JUNE 2017

FIGURE 3.7-23

The surficial pit lake water is also expected to exceed AWQC for several constituents. Once the pit lake reaches its maximum allowable elevation at or about Year 52 post-Closure, the surficial water would be treated to meet AWQC and then discharged. The results of the various modeling efforts of the predicted pit lake suggest that 1) the concentrations of several constituents in surface waters would exceed the most stringent AWQC throughout the 100-year modeling period and 2) the pycnocline is predicted to move upward toward the surface and become less intense over time, eventually reaching the surface and allowing complete mixing at some point beyond the modeling period. For these reasons, additional monitoring and adaptive management measures that would help maintain lake stratification as long as possible, manage surface water quality, and assure appropriate water treatment during post-Closure are provided in Chapter 5, Impact Avoidance, Minimization, and Mitigation.

Transportation Corridor

The geochemistry of rocks disturbed by mining would not impact water quality in the Transportation Corridor areas. Potential water quality impacts from rocks disturbed at material sites along the mine access road are described below in Section 3.7.3.2.2.

Pipeline

The geochemistry of rocks disturbed by mining would not impact water quality in the pipeline corridor. Potential water quality impacts from rocks disturbed at material sites along the pipeline are described below in Section 3.7.3.2.2.

3.7.3.2.2 SURFACE WATER QUALITY

The primary source of water quality issues for the Donlin Gold Project are related to rock geochemical processes described in the previous section and Appendix H, Geochemistry. The sections below address additional water quality and saturated sediment quality effects related to all three project components.

Mine Site

Water Treatment

Water would be produced by pit dewatering wells during both construction and operations. Additionally, under average precipitation conditions, the mine is expected to operate with a water surplus. Based on the geochemistry of the rocks and water at the site, some of this water is expected to exceed Alaska water quality standards. Accordingly, an APDES permit application has been submitted to treat and discharge excess water to Crooked Creek (Donlin Gold 2017d). The operations Water Treatment Plant (WTP) would use an advanced water treatment (AWT) process developed by Hatch (2015, 2017) to meet applicable water quality-based permit limitations associated with discharge from the following sources:

- Pit dewatering – Groundwater collected in the pit perimeter and in-pit dewatering wells.
- SRS – Water collected from the TSF underdrains at the SRS pond, which includes groundwater flows originating upslope of the areas covered by the TSF liner and a volume of seepage that may leak from the TSF liner (determined using EPA developed assumptions for potential seepage through synthetic liners).

- CWD – Water collected from the Upper and Lower CWDs. Sources of water in the CWDs include open pit drainage (direct precipitation falling on the pit walls and flows from horizontal pit drains); seepage and runoff from the WRF; seepage and runoff from the SOB; and undiverted runoff from undisturbed areas in the American Creek drainage. This water is predicted to have variable and increasing concentrations over time, with high concentrations of selenium, arsenic, and several metals (Tables 3.7-31 and 3.7-32). Water in the Lower CWD is expected to be worse than the Upper CWD, and would only go directly to the WTP if water quality permits.
- TSF Pond Water – Excess precipitation and tailings consolidation water collected in the TSF pond would be treated and discharged as provided in the NPDES New Source Performance Standards for gold mining operations (40 CFR 440.104). Accordingly, annual volume would not exceed the difference between: precipitation falling on the TSF and ore stockpile catchment areas, minus evaporation from the ponds. Based on an average annual precipitation of 19.6 inches and annual evaporation of 13.4 inches (BGC 2011f), these volumes could range from about 900 to 2,200 gpm on an annual basis depending on mine phase; however, the actual flow of TSF pond water to the WTP would be limited by mercury removal efficiency to a maximum of about 200 gpm (Hatch 2017).

Flow to WTP and Influent Chemistry

The ability to treat various mine waters in the WTP is based on both flow rates and chemical concentrations in each source. The amounts of these waters that could be treated and discharged were developed based on the site-wide water balance model that considers hydrology, quantities of water produced, and quantities of water required for process plant operations (BGC 2016c; SRK 2017b).

The intent of the WTP would be to enable the mine operation to discharge water which has been impacted by site activity, in compliance with applicable standards, when the water is not required for operational needs. In order to fulfill this objective, the water treatment technology must reliably meet all expected discharge limits. Other WTP objectives include controlling water accumulation in the TSF to maintain an acceptable inventory, and providing operational flexibility with respect to site water management.

The SRS water, in-pit runoff, water from the Upper CWD (assumed to be a mixture of pumped water from the Lower CWD water and runoff), and TSF pond water are predicted to contain concentrations of some metals and TDS above applicable standards (Table 3.7-38). Consequently, water treatment to actively manage metals and TDS in the water would be required to produce effluent that would conform to the water quality-based limitations of the APDES permit.

Under normal precipitation conditions, the WTP would operate seasonally. Treatment would primarily be required in the summer period (April through October) when there is expected to be more water in the CWD system than is required for process use. By contrast, in the winter period (November through March), CWD pond volumes are typically expected to be low, and dewatering well water and SRS water would be required as a source of make-up for the process plant. The WTP would operate at a reduced throughput or be dormant through the winter. An exception to the anticipated seasonal operation may arise, if needed, at the end of the construction period, when pumping from the dewatering wells and SRS would occur year-round, and the water could not be used in the process plant as it would not be operating yet. To address this need for water treatment during the construction period and to provide contingency for other needs that may arise, the WTP would be designed to be capable of year-round operation (Hatch 2017).

Table 3.7-39 presents the maximum and average flows for the four water sources during the life of mine. Flow to the WTP initiates when pit dewatering begins approximately two years prior to the process plant starting operations (Hatch 2017). The maximum flows from the CWD and TSF reclaim pipeline represent limitations based on the proposed pumping capacity at these locations. Actual rates of treatment for water from the CWD and TSF would be based on concentrations of parameters of concern and expected removal efficiency in the WTP, as well as flow and water quality from the other sources.

Mercury is the limiting parameter for treatment of TSF reclaim water, while selenium constrains the amount of CWD water that can be treated (Hatch 2017). The flow of TSF reclaim water that may be treated and discharged is further constrained such that the annual volume does not exceed the anticipated excess precipitation falling on the TSF. However, because excess precipitation (as defined in the NPDES Effluent Limitation Guidelines) is expected to exceed 220 gpm in each year, maximum TSF reclaim water flow to the WTP would likely be limited by the mercury removal efficiency in the WTP (Hatch 2017).

The average flow to the WTP from the CWD system in Table 3.7-39 represents the average available amount of water in excess of process water requirements from this location. The actual average amount of water from the CWD that is treated may be limited by selenium removal. Mercury removal efficiency by reverse osmosis (RO) is the factor which defines the average amount of TSF reclaim water sent to the WTP; the average TSF values in Table 3.7-39 are based on an assumed mercury removal of 96 percent (Hatch 2017).

Table 3.7-38: Estimated Operations WTP Influent Source Water Quality, to Year 10 of Discharge

Constituent[1]	Unit	Pit Dewatering, 95th Percentile		SRS	Upper CWD	TSF Reclaim
		Low Mineralized Wells	High Mineralized Wells	95th Percentile	95th Percentile	Steady State
Alkalinity	mg/L	151[a]	459[a]	127	113	25
Aluminum	mg/L	0.10	3.9	0.033	0.085	0.013
Ammonia	mg/L	0.37	1.1	1.4	0.66	29
Antimony	mg/L	0.00037	0.0058	0.016	0.60	0.022
Arsenic	mg/L	0.24	2.2	0.22	2.6	3.3
Barium	mg/L	0.87	1.5	0.15	0.16	0.011
Beryllium	mg/L	0.000065	0.00059	0.000065	0.00038	0.00003
Boron	mg/L	0.040	0.19	0.016	0.11	0.59
Cadmium	mg/L	0.000075	0.00020	0.000085	0.00066	0.00073
Calcium	mg/L	44[a]	64[a]	46	147	610
Chloride	mg/L	0.92[a]	6.5[a]	2.0	1.9	26
Chromium	mg/L	0.0015	0.0072	0.00073	0.015	0.012
Cobalt	mg/L	0.0006	0.0030	0.00087	0.038	0.019
Copper	mg/L	0.00066	0.011	0.0012	0.0066	0.018
Fluoride	mg/L	0.17[a]	2.3[a]	0.12	0.15	2
Iron	mg/L	5.7	1.8	1.6	0.29	0.0044
Lead	mg/L	0.00044	0.0045	0.00025	0.059	0.003
Lithium	mg/L	0.016	0.17	0.0016	0.073	0.003
Magnesium	mg/L	16[a]	27[a]	23	23	1733
Manganese	mg/L	1.4	0.13	0.45	1.5	2
Mercury	mg/L	0.0000023	0.000022	0.00017	0.000093	0.010[c]
Molybdenum	mg/L	0.0016	0.0081	0.0049	0.15	0.23

Table 3.7-38: Estimated Operations WTP Influent Source Water Quality, to Year 10 of Discharge

Constituent[1]	Unit	Pit Dewatering, 95th Percentile		SRS	Upper CWD	TSF Reclaim
		Low Mineralized Wells	High Mineralized Wells	95th Percentile	95th Percentile	Steady State
Nickel	mg/L	0.0018	0.0092	0.0015	0.23	0.062
pH	s.u.[b]	7.8	8.9	7.8	8.4	7.7
Potassium	mg/L	0.80[a]	8.1[a]	3.4	18	120
Selenium	mg/L	0.00075	0.0016	0.0014	0.17	0.042
Silica	mg/L	6.3	17	N/A	12	7
Silver	mg/L	0.00016	0.00016	0.00019	0.0009	0.00009
Sodium	mg/L	11[a]	235[a]	23	20	1,100
Strontium	mg/L	0.38	1.0	N/A	1.6	7.9
Sulphate	mg/L	27[a]	99[a]	69	423	8,605
TDS	mg/L	183[a]	690[a]	273	728	11,550
Thallium	mg/L	0.00016	0.00061	0.00016	0.00058	0.00041
TSS	mg/L	13	167	N/A	N/A	N/A
Vanadium	mg/L	0.0031	0.0084	0.0031	0.0043	0.0048
WAD Cyanide	mg/L	0.0039	0.0042	0.014	0.0041	0.14-0.73
Zinc	mg/L	0.014	0.042	0.011	0.34	0.033

Notes:

1 Total basis except where noted
a dissolved basis
b standard units
c Based on reductions observed at a Barrick facility using UNR reagent

N/A = not estimated
Highlighted parameters exceed the most stringent water quality objectives.

Source: SRK (2017b), Table 4-4; Hatch (2017)

Table 3.7-39: Average and Maximum Flows to WTP over Summer Treatment Season

LOM Year	APDES Discharge Year[1]	Dewatering Wells		SRS		CWD		TSF Reclaim Water		Total WTP Feed	
		Max.	Avg.	Max.	Avg.	Max	Avg.	Max.[2]	Avg.	Max.	Avg.
		gpm	gpm	gpm	gpm	gpm	gpm	gpm	gpm	gpm	gpm
-2	1	1471	1471	-	-	-	-	-	-	147	1471
-1	2	1449	1449	872	687	-	-	-	-	232	2135
1	3	1184	1167	872	669	-	-	-	-	205	1836
2	4	1391	1132	784	581	1100	612	220	44.5	349	2373
3	5	1814	757	713	277	1100	255	220	22.0	385	1316
4	6	1563	643	1162	445	1100	260	220	24.2	405	1374
5	7	1510	867	1136	608	1100	418	220	34.8	397	1933
6	8	1422	982	1088	691	1100	542	220	40.9	383	2259
7	9	1387	1066	1052	735	1100	643	220	44.5	376	2492
8	10	1726	1471	1017	784	1100	757	220	53.7	406	3069
9	11	1625	1418	982	775	1100	784	220	50.2	393	3034
10	12	1603	1396	969	762	1100	784	220	48.4	390	2998
11	13	1585	1413	933	753	1100	832	220	48.0	384	3056
12	14	2378	2259	907	775	1100	933	220	64.3	461	4038
13	15	2056	1986	876	753	1100	960	220	57.7	426	3765
14	16	1845	1788	845	731	1100	982	220	52.4	402	3562
15	17	1744	1717	819	718	1100	1030	220	49.8	389	3527
16	18	1845	1823	797	696	1100	1035	220	50.2	397	3615
17	19	1748	1730	771	674	1100	1030	220	47.1	385	3492
18	20	1761	1748	793	696	1100	1030	220	47.6	388	3536
19	21	1735	1726	771	674	1100	1026	220	45.8	383	3483
20	22	933	925	744	656	1100	995	220	29.1	301	2620
21	23	1052	1035	726	634	1100	837	220	29.9	311	2549
22	24	1114	1083	704	616	1100	740	220	29.5	315	2483
23	25	1149	1118	691	599	1100	682	220	28.6	317	2444
24	26	1167	1118	669	577	1100	630	220	27.3	317	2369
25	27	123	101	652	511	1100	520	220	4.8	211	1154
26	28	0	0	634	317	1100	207	220	0.0	197	542
27	29	0	0	621	247	1100	185	220	0.0	195	449

Notes:

1 APDES-regulated discharge from the WTP starts in Operational Year -2.

2 Maximum allowable based on net precipitation minus evaporation.

Source: Hatch (2017), Table 4-1; SRK (2017b), Tables 4-1 and D-1

Process Description

Several methods were initially investigated by Hatch (2015), and a conceptual water treatment process selected based on predicted water chemistries at the Mine Site and preliminary design work. The following description is based on preliminary design work which was further refined in Hatch (2017). Equipment quantities, capacities, and reagent use may change during detailed design. A flow schematic of the WTP is presented in Figure 3.7-24.

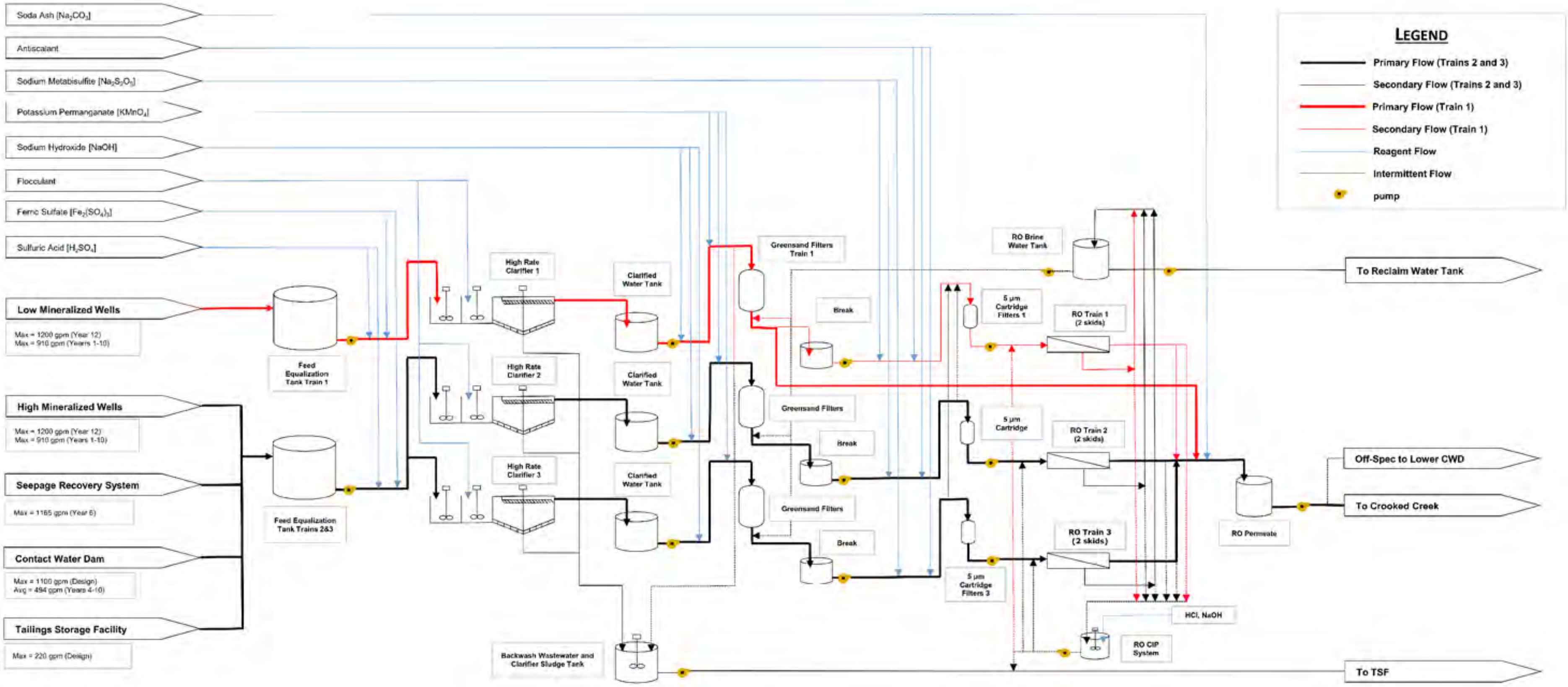
Trains and Feed Equalization Tanks. The WTP consists of three identical treatment trains each rated for a maximum design flow of 1,580 gpm. This configuration provides treatment capacity for operational reliability, the ability to treat a wide range of flow from different sources, and performance. The three parallel treatment trains are expected to operate at the same time (Hatch 2017).

The treatment process includes two feed equalization tanks. The first tank exclusively receives feed from low-mineralized dewatering wells (i.e., those with lower concentrations of minerals that can be treated to meet discharge objectives without RO). The second tank collects incoming feed from the high-mineralized dewatering wells, CWD, SRS and TSF sources. The first tank feeds Train #1, and the second feeds Trains #2 and #3 (Figure 3.7-24).

The design volume of both tanks would be about 53,000 gallons. This provides approximately 15 minutes of hydraulic retention time (HRT) at the design treatment rate for Trains #2 and #3. The HRT for well water feeding Train #1 is approximately 30 minutes, allowing greater level fluctuation in response to the more highly variable flow coming from this source. Both feed equalization tanks are mixed by a common set of air blowers (one duty, one stand-by) to promote oxidation of iron in the feed.

Two pumps installed at the first feed equalization tank supply the high-rate clarifier (HRC) in Train #1, and three pumps feed the HRCs in Trains #2 and #3 from the second feed tank. One of the pumps at each tank would be on stand-by to be used during maintenance of the duty pump. The HRC feed pumps have a design flow of 1,580 gpm.

High-Rate Clarifiers. Sulphuric acid and ferric sulphate are added just ahead of the HRC to adjust pH for the iron co-precipitation process and optimize arsenic and antimony removal. After chemical dosing, the influent enters the HRC step. Removal efficiency for antimony in the HRC is expected to be 75 to 80 percent, and for arsenic is anticipated to be greater than 90 percent. Aluminum, molybdenum and suspended solids removal also occurs at this step.



Data Sources: Hatch 2017, SRK 2017b

DONLIN GOLD PROJECT EIS

FLOW DIAGRAM OF OPERATIONS WATER TREATMENT PLANT

JUNE 2017

FIGURE 3.7-24

There is one HRC for each treatment train, each designed to treat a maximum of 1,580 gpm. The HRCs in Trains #2 and #3 are operated in parallel at equal flows of water from the second feed tank. A single HRC is capable of operation at 150 percent of the design flow during periods when the other HRC unit is out of service.

In the HRC, a polymeric flocculant is added to assist with agglomeration of precipitated ferric hydroxide and co-precipitates. The solids are separated in the clarification step, and the sludge sent to the backwash wastewater / clarifier sludge receiver tank, which also receives wastewater from the greensand filters. Sludge flow from the clarifiers is estimated at 3 percent of the peak flow, and does not vary with HRC throughput.

Each train has a clarified water transfer tank to receive flow from the HRC. These tanks have a capacity of 16,000 gallons and an HRT of 10 minutes at peak flow. The clarified water transfer tanks for Trains #2 and #3 are hydraulically connected to allow a single on-line HRC to feed the downstream greensand filters in Trains #2 and #3.

Greensand Filters. Water is pumped to the greensand media filters from the clarified water transfer tank in each train. Two pumps (one duty, one stand-by) are installed at the clarified water transfer tank in Train #1, while a common set of three pumps (two duty, one stand-by) are used by Trains #2 and #3. As with the other process steps, the pumping capacity would be 1,580 gpm per pump.

Each treatment train features a set of four 12-foot diameter greensand media filters, which would operate in parallel and be fed by a dedicated pump. The overall filtration rate allows one of the filters to be off-line for backwashing. The greensand filters are dual-media: the top layer consists of anthracite intended for TSS removal, and the bottom layer is the greensand media. Potassium permanganate ($KMnO_4$) is injected upstream of the greensand filters to oxidize manganese. The pH at the inlet of the greensand filters may be as low as 5, depending on the operating pH needed in the HRC to achieve target antimony removal. At this low pH, oxidation kinetics would be affected; thus, sodium hydroxide (NaOH) is added to raise the pH to 7 before the injection of $KMnO_4$.

The greensand filters would be backwashed with air and water. The backwash system is comprised of two blowers and two backwash pumps, one duty and one stand-by for each. A common backwash system is used for all three treatment trains. Backwash would occur daily at maximum flow. Typical backwash times are 2 to 3 minutes for air backwash, followed by 10 minutes for water backwash. Concentrate from the RO is used for backwash.

Wastewater from filter backwash would be sent to the backwash wastewater / clarifier sludge receiver tank. This tank would have a capacity of about 21,000 gallons, and would be able to store one filter backwash while receiving the continuous sludge flow coming from the HRC. The tank is equipped with a 5-horsepower mixer intended to keep solids in suspension. The combined wastewater from the HRCs and the greensand media filters would be sent to the TSF via the final tails collection pumpbox.

The filtrate from each set of greensand filters would be collected in a break tank before feeding the RO step. The break tank accommodates variation in treatment flow and, if RO treatment is required, allows the RO units to be started and stopped in response to the level in the tank. The break tank for each treatment train would have a capacity of about 16,000 gallons in order to provide 10 minutes of HRT at the design flow.

Reverse Osmosis. RO pre-treatment is required to protect the membranes from oxidation, fouling, and scaling. Since the RO membranes are susceptible to degradation by residual oxidants, sodium metabisulphite ($Na_2S_2O_5$) is used to remove any trace of $KMnO_4$. Antiscalant addition is also required to maintain soluble salts in solution, preventing precipitation on the membranes. In order to protect the RO membranes from fouling, polypropylene disposable cartridge filters are installed ahead of the RO system. Cartridge replacement would occur every two to three months, or whenever the pressure drop across the cartridge filter housing exceeds 15 psi.

Each treatment train would contain two parallel RO systems rated for 790 gpm each (including Train #1 which may not require RO for treatment of low-mineralized well water). Each RO system would be fed by a single high pressure pump (two duty pumps per train, no stand-bys). The second RO system in each treatment train would cycle on and off in response to the level in the break tank (RO feed tank) when the treatment rate is below the design flow. RO permeate may be recycled to the break tank for a short duration whenever an RO system is restarted until operations have stabilized.

Installation of a total of six 790-gpm RO systems does not include any redundant RO treatment capacity at the LOM design flow of 4,614 gpm (Table 3.7-40). However, the peak flow in most years is below the 3950-gpm capacity at which one RO system could be on stand-by with five operational. Further justification for the decision not to install redundant RO capacity includes the following:

- Treatment objectives can be achieved without use of RO in Train #1 if it only receives low-mineralized well water. Installed RO capacity in Train #1 may then be allocated to provide back-up for Trains #2 and #3.
- Storage capacity (tankage) in the treatment system may be used to accumulate water during peak WTP feed flow conditions in order to bring the feed rate to the RO system closer to the average flows listed in Table 3.7-40, which are below 3,950 gpm in all but APDES Year 14.
- It is possible to retain water in the CWD system while an RO unit is being repaired, and to then operate at maximum capacity in order to lower the CWD pond levels.

The RO systems operate at 75 percent recovery. RO permeate is discharged to a final storage tank (RO permeate water tank). Before entering the tank, the RO permeate receives a dose of NaOH if necessary to adjust pH to within the target range for discharge (6.5 to 8.5). Soda ash may be added as an alternative or in addition to NaOH for pH adjustment, and to also increase the alkalinity of the treated water.

During conditions when only low-mineralized well water is directed to Train #1, the application of RO for this water is discretionary. Based on the well water qualities specified in Table 3.7-39, it would be possible to meet overall treatment objectives for the WTP without operating the RO polishing step in Train #1. In this case, the Train #1 greensand filters would discharge directly to the RO permeate water tank, and mixing of water from the three trains in this tank may reduce the need to add soda ash to meet alkalinity discharge standards.

Transfer to Crooked Creek Discharge. There would be three transfer pumps (two duty, one stand-by) to move treated water from the RO permeate water tank to the discharge point at Crooked Creek. The flow for the treated water transfer pumps is 1,800 gpm each. If the treated water is out of specification, then it would be transferred to the Lower CWD until the problem is resolved.

Treated Water Quality

Table 3.7-40 presents the projected treated water quality from Hatch (2017), with comparisons to most stringent water quality criteria applicable for discharge to Crooked Creek. Table 3.7-41 also lists water quality for Category 2 sites, which represent background water quality in waters draining areas of defined mineralized zones with no placer mining activities.

The projections are based on 95th percentile water quality of each feed source (Table 3.7-40), average flows from the individual sources for the first 10 years of WTP operation (Table 3.7-41), and an assumption that 50 percent of the dewatering well water would be characterized as low-mineralized. The projections are also based on removal efficiencies for individual chemicals and expected treatment performance (reductions in chemical concentrations) at each WTP step based on both published information and unpublished Hatch experience at other sites (Hatch 2017).

Several parameters represent the greatest challenge or limiting factors for treatment, including antimony, arsenic, mercury, selenium, and WAD cyanide. As described above, arsenic and antimony would primarily be removed by iron co-precipitation at the HRC step with removal efficiencies ranging from 75 to 95 percent under certain pH conditions, followed by further reductions at the greensand filtration and RO steps. Mercury, selenium, and cyanide removal would primarily be accomplished by the RO system at assumed removal efficiencies of 96, 91, and 80 percent, respectively. The treatment basis for each of these parameters is further analyzed in Hatch (2017).

Waste Streams

Each of the steps of the water treatment process produces a waste stream that would need to be disposed. The sludge from the HRC would be sent to the TSF. The greensand filter would be backwashed intermittently and the wastewater sent to the TSF. Chemical characteristics and flow rates of these wastes are listed in Table 3.7-41 (Hatch 2017).

Brine from the RO process would be collected in a 24,000-gallon RO brine water tank (Figure 3.7-24). The tank capacity includes 17,000-gallons of water storage for one greensand media filter backwash. Two sets of pumps would be installed to draw from this tank: one for greensand media filter backwash, and the other to transfer RO brine to the reclaim water tank. The design flow for the RO brine transfer pumps is 1,100 gpm each.

RO membranes can become fouled by mineral scale, biological matter, colloidal particles, and insoluble organic constituents. RO elements would be cleaned quarterly; or when the flow decreases, salt passage increases, or the pressure drop across membranes increase past certain thresholds (ranging from 5 to 15 percent).

After collection, the brine from the RO process would be returned to the process plant as a source of reclaim water, except during construction, when the volume of brine is expected to be less. During that period, it would report to the Lower CWD.

Table 3.7-40: Predicted Treated Water Quality from Operations WTP

Parameter	Units	Treated Water	Background Category 2 (averages[3])	Most Stringent Applicable Water Quality Criterion
Aluminum	µg/L	< 50	302	750[2ab]
Ammonia	mg/L	<0.5	0.0402	2.99[2h]
Antimony	µg/L	<5	1.85	6[2d]
Arsenic	µg/L	<6	6.91	10[2d]
Barium	µg/L	<400	60.6	2,000[2d]
Boron	µg/L	<50	81.9	750[2e]
Cadmium	µg/L	<0.1	0.11	0.23[2a]
Chloride	mg/L	<1	1.60	230[2b]
Chromium, total	µg/L	<2	3.88	100[2d]
Cobalt	µg/L	<1	6.63	50[2e]
Copper	µg/L	<1	0.922	7.7[2a]
Iron	µg/L	<50	679	1,000[2b]
Lead	µg/L	<1	0.564	2.4[2a]
Manganese	µg/L	<50	107	50[2f]
Mercury	ng/L	<12	6.11	12[2b]
Molybdenum	µg/L	<5	17.0	10[2e]
Nickel	µg/L	<5	1.45	43[2a]
pH	units	6.5-8.5	7.51	6.5-8.5
Selenium	µg/L	<5	0.804	5[2b]
Sulfate	mg/L	<60	21.3	250[1]
TDS	mg/L	<240	141	500[1]
TSS	mg/L	<1	15.9	20
WAD Cyanide	µg/L	<5	1.05	5.2[2g]
Zinc	µg/L	<20	16.0	100[2a]

Notes:

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012, maximum drinking water levels.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column.
2a Aquatic life for fresh water hardness-dependent criteria using a hardness value of 80.55 mg/L as $CaCO_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d). For Al, if pH≥7.0 and hardness ≥50, then 750 µg/L.
2b Aquatic life for fresh water (chronic) criteria.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water primary maximum contaminant levels.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life criteria (chronic)for free cyanide, measured as weak acid dissociable (WAD) cyanide.
2h Aquatic life for fresh water (chronic) criteria based on pH and temperature when early life stages of fish are present. Standard for ammonia is based on 85th percentile of pH (7.85) and temperature (6.83°C) measured at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).

3 Calculated for all observations using detected values and ½ of the method detection limit value when analyte not detected. Metals are total values from Table 3.7-3.

Source: Hatch 2017, Table 4-5; SRK 2017b.

Table 3.7-41: Waste Streams from Operations WTP

Waste Stream	Type	Flow or Volume	Main Characteristics	Destination
HRC sludge	Continuous	130 gpm	• TSS: 700 mg/L • TSS: 1,400 mg/L • Al (total): ~10 mg/L • As (total): ~15 mg/L • Fe (total): ~450 mg/L • Mn (total): ~7 mg/L • Mo (total): ~0.1 mg/L • Sb (total): ~2 mg/L • SO_4: ~350 mg/L • pH: 5-6 • All other concentrations similar to feed water quality	TSF
Greensand media filter backwash wastewater	Intermittent	1,540 gpm	• TSS: 300 mg/L • As (total): ~3 mg/L • Fe (total): ~30 mg/L • Mn (total): ~15 mg/L • pH: 7-8 • All other concentrations similar to RO brine	TSF
RO brine	Continuous	370 gpm	• TDS < 3,500 mg/L • NH_4: <5 mg/L • Ca: <400 mg/L • Cl: <15 mg/L • Mg: <250 mg/L • Se: <0.25 mg/L • SO_4: ~1,800 mg/L • Total metals: <3 mg/L • WAD CN: <0.05 mg/L • pH: 7-8	Reclaim Water Tank
RO CIP waste	Intermittent	26,000 gal/cleaning cycle/ RO train	• pH: 1-12	TSF

Notes:

1 RO brine calculated assuming no RO treatment for Train #1.

Source: Hatch 2017, Table 4-6.

WTP Summary

A water treatment process has been developed to the preliminary design stage for Donlin Gold. The process, consisting of iron co-precipitation, greensand filtration, and a RO polishing step, would achieve each of the design objectives. The WTP would treat water that has been impacted by site activity to meet APDES permit requirements. The design allows water from the dewatering wells, SRS, CWD, and TSF that is not needed for process plant feed to be treated and discharged, rather than stored onsite during operations. The WTP is an integral component of the site water management system that provides operational flexibility to manage and control water accumulation in the TSF. The WTP design incorporates a number of redundancies (e.g., multiple trains, pumps, tanks) to allow continuous operation during maintenance, as well as flexibility to accommodate the anticipated water quality of multiple sources, including the ability to achieve discharge water quality requirements if source water quality varies over time, as predicted.

Construction

Under Alternative 2, construction at the Mine Site would take place over a 3-year period. The American Creek drainage into the CWDs, pit dewatering, overburden stockpiles, construction camp potable water wells, Snow Gulch fresh water dam (FWD), and TSF (Anaconda Creek runoff) are the primary components of the Mine Site that could potentially impact surface water quality during the construction phase. In order to address potential impacts to surface water quality, water management objectives during construction include:

- Treat all pit dewatering groundwater prior to discharge into Crooked Creek;
- Minimize build-up of contact water in the lower CWD during construction; and
- Eliminate the need to store water in the TSF facility until immediately before mine start-up.

Besides geochemical processes, three primary mechanisms would be responsible for potential impacts to surface water quality at the Mine Site during Construction. Those three mechanisms would be:

- Discharges of treated water to Crooked Creek,
- Runoff from construction materials, and
- Erosion and sedimentation.

Discharges of Treated Water to Crooked Creek

A modular Construction WTP would be commissioned to coincide with site earthworks, which would be used to treat stormwater where necessary and discharge it to Crooked Creek during the 3- to 4-year construction period (SRK 2017a, 2017b). Pit dewatering would start at LOM Year -2, about 1 to 2 years into construction period, at which point construction of the Operations WTP would be complete and ready to receive dewatering water for treatment.

As contact water would not be discharged during the early construction period, impacts to surface water quality attributable to contact water would be limited in geographic extent to the areas used for contact water storage. Water collected at the Lower CWD is not expected to meet AWQC for discharge and would not be released to the environment unless it is first treated; the contact water would be stored through the early construction period, then treated as necessary after the Operations WTP is complete (BGC 2011d; SRK 2017b). Any excess non-contact water captured by the American Creek fresh water diversion dam (FWDD) located upstream of the Lower CWD would be expected to meet AWQC for discharge, and would be pumped to Crooked Creek (SRK 2017b).

The main source of water that would be treated and discharged to Crooked Creek in the last two years of the construction period would be from pit dewatering (up to about 1,500 gpm) (Table 3.7-39) (SRK 2017b). Concentrations of aluminum, arsenic, copper, iron, lead, manganese, mercury, fluoride, TDS, and TSS in the pit dewatering water would be expected to be greater than the concentrations specified in the most stringent AWQC (Tables 3.7-37 and 3.7-38) (SRK 2017b, Hatch 2017). In order to meet effluent limitations for discharge, groundwater from the pit perimeter wells would be treated to meet water-quality -based effluent limitations of an APDES permit (based on AWQC) prior to discharge to Crooked Creek using the treatment system discussed in the previous section (Donlin Gold 2017d). The proposed treatment process is expected to provide effective treatment for all parameters of concern, and the effluent would be compliant with the most stringent effluent limitations. The water quality of the low-mineralized dewatering well water is estimated to be substantially better than the high-mineralized well water, SRS, CWD, or TSF pond water, so it is assumed that 50 percent of the dewatering well water will not require RO treatment (Hatch 2017). Brine from treatment of the high-mineralized well water during Construction would be stored in the Lower CWD during this period. It is not anticipated that in-pit dewatering wells would be required during the construction period (SRK 2017b).

Based on available data, stream temperature downstream of the Mine Site is anticipated to remain relatively constant during construction (ARCADIS 2013a). Although the treated groundwater discharged to Crooked Creek during construction could have a temperature higher than that of the surface water in Crooked Creek, the larger contribution of surface water inputs relative to treated groundwater inputs would be expected to attenuate changes to water temperature within Crooked Creek during construction. Existing surface water temperatures downstream of the Mine Site vary between 0° and 9.16°C depending on the time of year, and construction activities are not expected to have any substantial impacts on surface water temperatures (ARCADIS 2013a).

There would be no discharge of power plant cooling water at the Mine Site, as the cooling system would be closed-circuit. Boiler replacement water would be pumped to the mill (Weglinski 2015f).

As a result of the effective water management and treatment processes proposed under Alternative 2, the intensity of impacts to water quality in Crooked Creek resulting from discharges of treated pit dewatering water would be below AWQC (Table 3.7-40). The wintertime treatment of water is not planned; however, the capability for wintertime operations is present. Any impacts to temperature in Crooked Creek resulting from discharge of treated water would likely be greatest during the winter when the discharged treated groundwater would be warmer than the ambient water in Crooked Creek and the discharged water would represent a greater proportion of the overall water in Crooked Creek. Conservatively assuming that the discharged treated water represents 10 percent of the total flow in Crooked Creek, if the temperature of the discharged water is 9°C and the temperature of the receiving water is 0.5°C, the effect would be an increase of 0.9°C, and the final water temperature would be 1.4°C.

The duration of impacts would extend over the construction period on a discontinuous basis. In terms of context, impacts would affect areas of ordinary water quality or where there is an abundance of water resources.

Runoff from Construction Materials

Non-acid generating (NAG) rock, including waste rock and material from borrow sites, would be used for construction of various facilities associated with the Mine Site including roads, berms, and dams. There is potential for impacts to surface water quality to occur as a result of metal leaching from NAG rock used for construction. In order to minimize potential impacts, the materials collected from borrow sites for the construction of roads and other mine infrastructure would be tested for metal leaching before use (URS 2013a) (Table 5.2-1, Design Features). To determine metal leaching in waste rock (WR) samples, rock material would be subjected to three tests: 1) bulk geochemical analysis; 2) meteoric water mobility procedure (MWMP); and 3) acid-base accounting (ABA). If sample materials from a particular borrow site show potential for metal leaching or acid generation, then those materials would not be used for applications where surface water quality could be adversely impacted.

NAG waste rock with metal leaching potential could be used for construction of the Lower CWD. Under Alternative 2, there would be potential for the generation of seepage and runoff with elevated metals concentrations derived from metal leaching from the NAG used for construction of the lower CWD. Runoff and seepage from the Lower CWD would be collected either in the ACMA pit dewatering system, or at a proposed ore stockpile berm designed to minimize surface runoff to the pit. The ACMA pit would intersect American Creek downstream of the dam during construction, so that runoff and seepage from the lower CWD would not migrate off site and would only affect surface water at discrete portions of the project area in the American Creek watershed.

Erosion and Sedimentation

Construction of the Mine Site facilities would result in increased erosion and potential increases in concentrations of suspended sediment in surface waters in the vicinity of the Mine Site. Clearing, grading, and excavation work during the construction of the mine's processing and support facilities would expose large areas to erosion. Use of heavy construction equipment would cause disturbance of near-surface soils that could locally result in increased runoff and subsequent increased sedimentation at downstream locations (see Section 3.2, Soils). In order to minimize such impacts to water quality from erosion, runoff and sedimentation, an important part of the water management strategy for construction involves diversion structures that would direct surface water and runoff from precipitation around and away from the exposed areas (see Section 3.2, Soils). By minimizing the rates of flow over the cleared areas, impacts from erosion and sedimentation would be controlled so that surface water quality would be expected to comply with all AWQC during the construction phase. Stormwater collected in diversion ditches and areas not in contact with waste rock, mine pit, or process solutions (non-contact water) would be suitable to discharge to the environment without treatment, or would be directed to the Construction WTP where necessary for stormwater treatment (SRK 2017b). Energy dissipating and erosion control features would be installed or modified as required to meet APDES Multisector General Permit (MSGP) discharge requirements. BMPs detailing sediment and erosion control measures would be developed and implemented as part of the SWPPP (Section 3.2, Soils).

As described in Section 3.7.3.2.1, several overburden stockpiles would be required to store material that would ultimately be used to reclaim the TSF and WRFs. These stockpiles would lay downgradient of areas that drain into proposed dams and, therefore, would require sediment control structures to prevent sediment-laden runoff from affecting surface water quality. Runoff from these overburden stockpiles would be managed by intercepting and directing surface runoff toward settling ponds sized to contain a 10-year return period, 24-hour duration storm (SRK 2017b). Upslope diversions would limit runoff to the stockpiles, while channels on the downslope sides would direct surface runoff to the settling ponds. The runoff from the SOB stockpile, which could contain leached metals, would report to the Lower CWD. The effects of potential arsenic leaching and ARD from the SOB stockpile are described in Section 3.7.3.2.1, and Appendix H, and recommended mitigation for reducing these effects are provided in Chapter 5, Impact Avoidance, Minimization, and Mitigation. Runoff from the NOB and other overburden stockpiles is considered suitable for discharge without treatment, other than stormwater and sediment runoff control (SRK 2017b).

The greatest potential impacts to surface water quality at the Mine Site during construction would likely result from suspension of sediment caused by in-stream construction and erosion of cleared stream banks in the American Creek and Anaconda Creek watersheds. In-stream construction could cause dislodging and transport of channel bed sediment and the alteration of stream bottom contours, resulting in increased suspended sediment concentrations in surface water. Changes in the bottom contours could alter stream dynamics and increase downstream erosion or deposition. Surface discharges to the local drainages during construction would potentially result in increased erosion and sedimentation, which could adversely affect surface water quality. Treated water from the pit dewatering system would be discharged to Crooked Creek below Omega Gulch. The outfall structural design would include energy dissipators and erosion control measures to reduce downstream impacts from erosion and sedimentation, and to meet AWQC (SRK 2017b).

Operations

Alternative 2 would involve operation of Mine Site facilities in the American Creek and Anaconda Creek drainages, including the mine pit, WRF, TSF and associated SRS. A dam would also operate in Snow Gulch to serve as a source of freshwater for the mine process circuit and other applications. The American and Anaconda Creek FWDDs would be removed early in operations following construction of the Lower CWD, initial lifts of the WRF, and TSF starter dam.

Five primary mechanisms would be responsible for potential impacts to surface water quality at the Mine Site during the operational phase:

- Discharges of treated water to Crooked Creek;
- Reduced flows in Crooked Creek due to diversion of water to support mine operations;
- Impacts to water quality resulting from wetlands alteration;
- Inputs of mercury to surface water from runoff and atmospheric sources, and potential changes to mercury speciation and mercury methylation rates in the project area; and
- Water quality at facilities containing contact water (i.e., TSF and CWD ponds).

Diversion structures to limit the amount of freshwater entering the Mine Site are an important part of the water management strategy. Synthetic liners, runoff collection systems, and seepage collection and pump back facilities would prevent the discharge of untreated contact water.

Mine water from several sources (pit dewatering groundwater, CWD pond water, SRS water, and limited amounts of TSF reclaim water) would be treated and discharged to Crooked Creek during Operations when surpluses are available, in order to manage impacts to surface water resources, minimize build-up of water in the TSF, and build flexibility into the overall water management system.

Although water within the mine facilities would not be considered waters of the State of Alaska or waters of the U.S., and would not be subject to regulation under the Clean Water Act or the APDES permitting program, receiving waters downstream from the proposed mine would be subject to Clean Water Act regulations. Thus, their expected water quality is discussed in this section for comparison to baseline water quality conditions.

Discharges to Crooked Creek

Water that would be discharged under Alternative 2 would be treated prior to discharge to meet AWQC and permit limitations associated with the discharge (see above discussion under “Water Treatment”). The intent of Alterative 2 is to maximize the total amount of water that can be treated and discharged from the project in order to minimize the amount of water that must be stored and managed during operations and at closure, while still supplying water required by the processing facility. This would enable operational flexibility with regard to the Mine Site’s water balance and could potentially prevent problems associated with accumulation of too much water within the mine facilities during periods of heavy precipitation.

The SRS water, in-pit runoff, water from the Upper and Lower CWDs, and TSF pond water are predicted to contain concentrations of some metals and total dissolved solids (TDS) that would be elevated relative to the most stringent AWQC (Tables 3.7-32 through 3.7-35). Consequently, additional water treatment measures to actively manage metals and TDS in the water would be required in order to produce effluent that would conform with AWQC, as discussed previously.

Treatment and discharge of pit dewatering water to Crooked Creek would occur mostly during spring, summer, and fall, when excess water in the Upper and Lower CWDs would be sufficient to meet the freshwater demand for the process plant. In contrast, during the winter months, CWD pond volumes would be low and the pit dewatering water would be used in the process plant rather than discharged to Crooked Creek. The average estimated discharge flow from the treated pit dewatering well water over the operational phase of the mine is about 1,360 gpm, although there would be periods of no discharge when all pit dewatering water is used in the process (BGC 2016c). The volumes of pit dewatering water that would be treated and discharged and the temporal dynamics of the discharges under Alternative 2 are discussed in Section 3.5, Surface Water Hydrology.

Based on 95th percentile concentrations of dissolved and total constituents in water from wells completed in the mineralized zone, groundwater derived from the pit perimeter dewatering wells would be expected to require treatment prior to discharge (Table 3.7-38) (BGC 2014b; Hatch 2015). The monitoring wells summarized in Table 3.7-38 were completed within the mineralized zone and are therefore considered to represent a reasonable characterization of baseline groundwater quality in the vicinity of the pit (BGC 2014b).

Reduced Flows to Crooked Creek

As described in Section 3.5, Surface Water Hydrology, and Section 3.6, Groundwater Hydrology, water management and requirements for process water during operations would result in the disruption of the natural flows of water in Snow Gulch and in the American Creek and Anaconda Creek drainages. These drainages are tributaries to Crooked Creek, and the disruptions would result in reduced flows that could potentially influence the water quality in Crooked Creek. Additionally, pit dewatering would result in direct streamflow losses from Crooked Creek into groundwater and then into the pit dewatering system, which could also influence the quality of water in Crooked Creek.

During mining operations, surface water runoff and groundwater seepage in some parts of the Crooked Creek watershed would be diverted and captured for use in milling processes and power plant operations. For example, precipitation that falls on the WRF and does not evaporate or infiltrate into the ground would be captured in the lower CWD and then used in the milling process; the water would be deposited in the TSF as tailings slurry, then the water would be recycled through the process circuit. Because the facilities have been designed to capture water affected by the mining operations, contact water would be managed so it is not released to surface water streams. Regardless of their final use or consumption, the diversion and storage of waters in the Crooked Creek watershed would result in reduced rates of runoff and base flow that would normally reach surface waters in the proposed project area. During the operational phase of the mine, surface water would be removed from the American Creek and Snow Gulch drainages and diverted to the process facilities. This would result in reduced discharge in Crooked Creek relative to baseline conditions.

Correlation analysis performed as part of a water quality loading study for the Donlin Gold Project indicates that relatively strong relationships exist between levels of flow and the concentrations of chemical constituents in Crooked Creek (Tetra Tech 2013). Concentrations of total aluminum, total iron, and total manganese generally increase with increasing stream flow because these metals are primarily associated with sediments which consist of naturally occurring minerals and organic material. Higher stream discharge is usually associated with higher flow velocity which entrains sediments from the substrate, increasing the total concentrations of metals in the water column. In contrast, concentrations of sulfate in Crooked Creek generally decrease with increasing rates of flow as a result of dilution (Tetra Tech 2013).

Inversely, the lower flow rates in Crooked Creek that would occur as a result of surface water diversion and reduced base flows associated with Alternative 2 may be expected to result in decreased total concentrations of aluminum, iron, and manganese in the water, and increased concentrations of sulfate. However, considering the relatively small portion of the Crooked Creek drainage area in Anaconda and American creeks that would be affected by operations, it is unlikely that reduced flows would measurably affect water quality in Crooked Creek downstream of the Mine Site. During high runoff periods, sediments would continue to be mobilized to Crooked Creek from undisturbed areas outside the mining area, and concentrations of aluminum, iron, and manganese in the water would remain similar to those observed during baseline studies. Likewise, overall inputs of sulfate would decrease in proportion to the fraction of the Crooked Creek drainage area that would be affected by operations, thereby offsetting any effect of decreased dilution; as a result, concentrations of sulfate in the water would be expected to remain similar to those observed during baseline studies.

Wetlands Impacts on Water Quality

Another factor that could affect surface water quality during mine operation involves the in-filling, and removal of wetlands in the American Creek, Omega Gulch, Anaconda Creek, Snow Gulch, and Crooked Creek watersheds. Wetlands naturally serve to maintain water quality through a variety of mechanisms; debris and suspended solids may be removed by physical processes such as filtering and sedimentation. Nutrients and dissolved solids may be removed or degraded by biological processes, or incorporated into biomass. Similarly, microbial activity occurring in anoxic wetland environments may result in the chemical reduction of certain iron and sulfate species facilitating their removal from the water as insoluble precipitates. Many dissolved metals are removed by wetlands; for example, arsenic (in association with iron) has been shown to accumulate in wetlands in areas influenced by mining activities. Wetlands can reduce the solubility and mobility of iron, manganese, and zinc by retaining these metals within the sediment. Wetlands systems in regions of permafrost generally have elevated DOC, organic nitrogen, and relatively low concentrations of dissolved minerals (SRK 2012b).

As a result of the wetlands removal that would occur under Alternative 2 (see Section 3.11, Wetlands), total concentrations of elements that are associated primarily with the particulate fraction, such as arsenic, mercury, lead, and zinc, could increase within the proposed project area, specifically at Category 3 sample locations that drain the proposed project area (see Section 3.7.2.1.1). Increased particulate loading in watersheds draining the operational area would not necessarily translate into an increase in dissolved concentrations of these elements, although there may be a slight increase in dissolved arsenic, mercury, lead, and zinc due to desorption and dissolution from sediments (SRK 2012b). However, it is likely that any effects of wetlands removal on water quality would be offset by the capture of the water that comes into contact with the disturbed areas.

Inputs of Mercury from Runoff and Atmospheric Sources

Predicted concentrations of mercury in surface water were calculated based on two different approaches as described in the following sections:

1) Concentrations were estimated for the two watersheds closest to the Mine Site which exhibit the highest predicted mercury deposition rates, to identify an upper range of potential changes in mercury concentrations in surface water. This approach is based on a simplified extrapolation from air modeling results; and
2) Concentrations were estimated based on an area-weighted average of deposition rates for watersheds within a 20-mile radius of the Mine Site, for use in determining human exposure concentrations for a Human Health Risk Assessment (HHRA) (described in Section 3.22 and Appendix AB). This approach takes several factors that control methylation rates into account.

Predicted Mercury in Surface Water for Watersheds Closest to Mine Site

Naturally Elevated Baseline Concentrations. Elevated concentrations of mercury are a natural feature of surface water in the vicinity of the Mine Site. As described in Section 3.7.2.1.1, concentrations of total mercury detected in 545 of 564 water samples collected between June 2005 and June 2015 ranged from 0.518 to 260 nanograms per liter (ng/L), with a mean of 7.81 ng/L (SRK 2017b; Rieser 2017). The applicable numeric criteria for mercury for CWA purposes are the EPA-approved aquatic life criteria: 2,400 ng/L acute and 12 ng/L chronic, both as total recoverable mercury (EPA 2013k). Total mercury concentrations did not exceed the acute criterion in any samples; about 80 samples exhibited total mercury concentrations in excess of the chronic criterion. Samples with total mercury concentrations greater than 12 ng/L were distributed relatively evenly between all three categories of sites defined by the Donlin Gold water quality characterization program. These data suggest that existing concentrations of total mercury in surface water are sometimes elevated above the chronic criterion at locations throughout the Mine Site area, with occasional spikes more than 10 times the criterion, possibly due to precipitation and localized rock weathering conditions.

Predicted Changes from Mine Site Activities. The activities proposed under Alternative 2 would result in additional inputs of mercury to surface water from both atmospheric and aqueous sources, which would likely cause an increase in exceedances of the 12 ng/L chronic criterion. Predicting changes in mercury concentrations in aquatic systems is challenging because a portion of mercury that is deposited to surface waters will be transported downstream, such that estimates of mercury deposition by watershed do not necessarily correspond to direct increases in mercury content in surface waters (ARCADIS 2014). Additionally, the importance of atmospheric deposition relative to runoff inputs to streams is another source of uncertainty. Studies of mercury mass balances in forest-dominated catchments have shown that mercury inputs to aquatic systems are more heavily dominated by contribution from wetland runoff (St. Louis et al. 1996; Selvendiran et al. 2008; Berndt and Bavin 2012) than by atmospheric deposition; however, it should be noted that in some instances, most of the mercury in the wetland runoff originally came from the atmosphere.

A conservative estimate of the potential changes in mercury concentrations in surface water due to atmospheric deposition resulting from Mine Site activities was developed by ARCADIS (2014) using watershed-based deposition rates predicted by Environ (2015). The study area for the model included the Hydrologic Unit Code (HUC) 12 watershed in which the Mine Site would be located, as well as surrounding watersheds in the area where potential for deposition would occur. The Mine Site lies within the Crooked Creek HUC 12 watershed (which includes Crooked Creek Village), and is surrounded by the following HUC 12 watersheds: Donlin Creek, Flat Creek, Grouse Creek, West Juningguira Mountain, Getmuna Creek, and Bell Creek (Figure 3.2-14). Within these watersheds there are a number of smaller drainages, including small tributaries within the Crooked Creek drainage located along the west side of the Mine Site.

The baseline rate of mercury deposition within the two closest HUC 12 watersheds to the Mine Site, Crooked Creek and Donlin Creek, is estimated to range from 7.8 to 8.4 micrograms per square meter per year (µg/m^2/yr) (Environ 2015). Mercury deposition rates are projected to increase in these watersheds by 2.3 to 4.7 µg/m^2/yr, with an average increase of about 3.5 µg/m^2/yr; this represents an increase of about 40 percent as a result of the activities proposed under Alternative 2. In this analysis, the rates of change in mercury transformation and transport in upland/wetland systems and aquatic sediments were assumed to be minimal as a result of project activities, and a linear response between atmospheric deposition rates and mercury concentration in surface water was assumed (ARCADIS 2014). This assumption is conservative because it precludes consideration of phase partitioning of mercury deposited from atmospheric sources, and the possibility that a large fraction of the mercury deposited from atmospheric sources would partition into soils and sediments in the project area and only a small fraction would be present as aqueous mercury in surface water. The model simply assumes a linear relationship between atmospheric deposition rates of mercury and concentrations of mercury in surface water. Following this methodology, the activities proposed under Alternative 2 could cause an increase in the average concentration of total mercury in surface water to approximately 11 ng/L, which is close to or below EPA-approved aquatic life criteria of 2,400 ng/L (acute) and 12 ng/L (chronic) (EPA 2013k), and the Alaska water quality standard of 50 ng/L for total recoverable mercury.

The average and 95th percentile baseline mercury concentrations and the range of estimated potential increases resulting from the project are shown in Table 3.7-42. With the projected changes applied to current concentrations, the resulting surface water concentrations of total mercury would, in some instances, exceed the applicable chronic criterion of 12 ng/L. However, as described above, existing concentrations of total mercury in surface water exceeded the 12 ng/L chronic criterion in about 80 of the 564 samples collected between 2005 and 2015, indicating that concentrations of mercury above the applicable chronic criterion are a widespread natural feature of surface water in the vicinity of the Mine Site. Atmospheric deposition of mercury to surface water resulting from the activities proposed under Alternative 2 could result in increases in the concentrations of total mercury in surface water, which may be sufficient to exceed AWQC at some locations.

Table 3.7-42: Baseline and Predicted Total Mercury Concentrations in Surface Water for Watersheds Closest to Mine Site

Statistic	Baseline Total Mercury (ng/L)	Estimated Total Mercury in Operations[3] (ng/L)
Average	7.81[1]	11
95% UCL[2]	23.5	33

Notes:

State of Alaska most stringent water quality criterion: 50 ng/L. EPA-approved State of Alaska aquatic life criteria: 2,400 ng/L acute, 12 ng/L chronic.

1 Based on data from 2005-2015 (SRK 2017b,e; Rieser 2017).

2 Statistics generated from data collected 2005-2011 (ARCADIS 2014). 95% UCL = 85 percent Upper Confidence Limit on the arithmetic mean.

3 Based on an average 40% increase in modeled mercury deposition rates for the two HUC 12 watersheds closest to the Mine Site: Crooked Creek and Donlin Creek (Arcadis 2014; Environ 2014a, 2015).

The evaluation of estimated impacts to concentrations of total mercury in surface water shown in Table 3.7-42 is considered conservative because the majority of mercury potentially deposited as a result of the activities proposed under Alternative 2 would be particulate mercury (ARCADIS 2014). Considering the high density of particulate mercury and the depositional environments in area streams, a substantial fraction of the particulate mercury from atmospheric deposition would be quickly removed from active cycling through burial in stream sediments (ARCADIS 2014). The above evaluation is intended to identify an upper range of potential changes in total mercury concentrations in surface water resulting from the activities proposed under Alternative 2.

Predicted Mercury in Surface Water for Area-Wide Watersheds

Estimates of the increase in total mercury and methylmercury for a wider area surrounding the Mine Site were conducted for the purpose of determining concentrations that surrounding residents could be exposed to from project-related activities (see Section 3.22, Human Health). Prediction of future increases in methylmercury in surface water, which is the most toxic form of mercury to humans and biota, considered the potential for changes in methylation rates over the life of the project, as described below.

Factors Affecting Mercury Methylation and Demethylation Rates

Methylation is the process by which inorganic mercury is transformed to methylmercury by the action of microbes that live in aquatic environments. Methylmercury is a transient species in wetlands and other aquatic systems. In most aquatic systems, including wetlands and anoxic sediments in lakes and rivers, in situ production (methylation of inorganic mercury to methylmercury), rather than input from runoff water or atmospheric deposition, is the major source of methylmercury (Li and Cai 2013). Net rates of mercury methylation are typically greater in wetlands compared to river sediments. However, site-specific rates of methylmercury production and decomposition in wetlands, relative to those in rivers, vary between systems depending on landscape and hydrological variables.

Decomposition of methylmercury, or demethylation, occurs simultaneously with methylation in the natural environment, such that the steady-state concentrations of methylmercury measured in environmental media depend upon the rates of both formation and decomposition processes at a particular location, as well as upon import and export rates of methylmercury between locations. Natural demethylation occurs primarily via photochemical reactions driven by sunlight (Zhang and Hsu-Kim 2010; Naftz et al. 2011). However, demethylation also occurs via microbiological and non-photochemical abiotic pathways under a wide range of environmental conditions (Khan and Wang 2010; Johnson et al. 2015).

Experiments using isotope-labeled methylmercury compounds added to lake sediments determined that the half-life of methylmercury in sediments was 1.7 days, suggesting a rapid turnover and low persistence of methylmercury in lake sediments (Hintelman et al. 2000). Similar experiments conducted using contaminated lagoon sediments determined apparent first-order demethylation rate constants of about 0.6 / day, which were much greater than the observed methylation rate constants, which ranged from near zero to 0.054 / day, indicating rapid mercury cycling in sediments with elevated porewater concentrations of sulfate and mercury, and overall net demethylation of mercury in contaminated sediments (Hines et al. 2012).

Sulfate-Reducing Bacteria (SRB). SRB facilitate both methylation and demethylation processes (Bridou et al. 2011). Experiments using species-specific isotopic tracers for inorganic and methylmercury to simultaneously determine the methylation and demethylation potentials of specific SRB strains showed that all of the methylating SRB strains also demethylate methylmercury (Bridou et al. 2011). Rates of mercury methylation and demethylation were related to the bacterial activity, but were not dependent on the type of metabolism used by the tested bacteria (sulfate reduction, fermentation, or nitrate respiration). The results demonstrated that SRB contribute to methylmercury demethylation under anoxic conditions, leading to the most common mercury oxidation state, Hg(II) or Hg(2+), as the end product. Based on the observed transformation rate constants for microbial methylation and demethylation processes, the authors concluded that methylation is the slower process, mainly controlled by the assimilation of Hg(II), whereas demethylation is faster and not dependent on the methylmercury concentration (Bridou et al. 2011). The slowness of methylation relative to demethylation in the environment explains the extremely low steady-state concentrations of methylmercury observed in environmental matrices, even under the anoxic circumneutral-pH conditions requisite for SRB activity.

Thus, in order to meaningfully estimate potential changes in mercury methylation that could occur in response to the proposed action and alternatives, one must consider net mercury methylation, which includes the combined effect of changes to both methylation and demethylation processes, both of which often involve the activity of SRB and occur simultaneously in wetlands and other aquatic systems.

Environmental Setting. In freshwater aquatic ecosystems, shallow sediment catchments and the anoxic bottom waters of stratified lakes are generally believed to be the most important zones for net methylation, and research from the Marcell Experimental Forest in northern Minnesota suggests that methylation may also occur in peatlands (Mitchell et al. 2008). Methylation is less prevalent in environments with higher flow and low hydraulic retention (St. Louis et al. 1994). In-river methylation is typically a negligible component of the methylmercury budget for creeks, whereas wetlands and peatland pore waters are frequently the most important contributors of methylmercury to downstream aquatic ecosystems (St. Louis et al. 1994, Mitchell et al. 2008; Berndt and Bavin 2012).

Wetlands can be either sources or sinks of methylmercury. Any changes to rates of mercury methylation and demethylation in wetland systems as a result of activities proposed under Alternative 2 would depend upon several variables, including: prevalence of anoxic environments requisite for mercury methylation; prevalence of sulfate-reducing or iron-reducing bacteria to facilitate the methylation process; availability of mercury to participate in methylation reactions; the nutrient status of the wetland systems (the concentrations and bioavailability of organic carbon and inorganic nitrogen and sulfur); the pH of the sediments or soils; and the rates of the photochemical and biological processes that drive demethylation in the environment (Marvin-Dipasquale and Oremland 1998; Zhang and Hsu-Kim 2010).

Field data collected at the Mine Site in 2013 were used to evaluate methylation potential and predicted project-related changes in mercury concentrations and speciation in natural waters and wetland systems (ARCADIS 2014). The 2013 field program samples show that the total mercury concentration in upland soil (average of 260 µg/kg) is slightly lower than in wetland soil (average of 320 µg/kg), although the variance in the soil concentration data was high in both cases.

Upland soils tend to be more oxygenated than lowland/wetland soils and consequently, upland soils generally have a lower methymercury generation potential than wetland soils (ERM 2017d). However, measured differences in methylmercury concentrations between upland soils and wetland soils within the project area were very small, suggesting that wetland systems in the study area do not have a large methylation potential relative to uplands (ARCADIS 2014). The 2013 field program results show that the percentage of total mercury that is methylmercury in soils currently ranges from 0.23 and 2.59 percent in upland systems, and 0.09 to 1.83 percent in wetland systems, with an overall average of less than 1 percent in each system (only detected values were included in these calculations). These ratios of methylmercury to total mercury are typical of boreal wetland and upland soils (ARCADIS 2014). Similar fractions (approximately 1 percent methylmercury) were detected in a similar system evaluated in northern Minnesota (a boreal system dominated by spruce/moss/shrub wetlands), which was characterized as having low methylmercury production (Hines et al. 2004).

Nutrients. Recent studies suggest that the nutrient status of sediments and soils (organic carbon and inorganic nitrogen and sulfur) and their pH play key roles in determining mercury methylation potential in the environment (Tjerngren et al. 2012; Braaten et al. 2014). In a study of mercury methylation processes in subarctic and boreal lakes, Braaten et al. (2014) found that concentrations of both total organic carbon (TOC) and total nitrogen were correlated to the fraction of mercury present as methylmercury in the subarctic and boreal lake systems. The authors concluded that total nitrogen was the variable most strongly correlated to the size of the methylmercury fraction (a proxy for methylation potential), and suggested that nitrogen availability may exert a positive contribution on concentrations of methylmercury and the fraction of mercury present as methylmercury (Braaten et al. 2014). Tjengren et al. (2012) found that maximum methylmercury yield was obtained in wetlands with an intermediate soil acidity (pH of about 5.0) and a carbon-to-nitrogen (C:N) ratio of about 20 (Tjengren et al. 2012). The authors hypothesize that an intermediate C:N ratio of about 20 is required for methylation rates to exceed demethylation rates, whereas higher or lower C:N ratios may promote demethylation rates relative to methylation rates, leading to net loss of methylmercury from the system (Tjengren et al. 2012).

Geochemical data from the Mine Site suggest low rates of net mercury methylation in project area wetlands. The average C:N ratio of the upland and wetland soils is high (41 and 36 in uplands and wetlands, respectively), and the average pH is acidic (4.7 and 4.5 in uplands and wetlands, respectively) (ARCADIS 2014). Wetland systems studied elsewhere that have similar C:N ratios and lower soil pH also show relatively low rates of net methylmercury production. For example, Tjerngren et al. (2012) showed that systems with a C:N ratio of approximately 35 to 38, and a soil pH of approximately 4.5, were characterized as “nutrient poor” and were less productive in terms of net methylation activity relative to systems with lower C:N ratios. Under conditions observed at the Mine Site, it is likely that relatively rapid mercury demethylation processes are at least partially responsible for the low steady-state concentrations of methylmercury observed in environmental media.

New vs. Old Mercury. The relative bioavailability of pre-existing mercury in the area, compared to that of newly deposited mercury from Mine Site activities, is an important consideration that could influence mercury methylation rates. Typically only a small fraction of inorganic mercury is available for methylation (Marvin-Dipasquale et al. 2009). Results from several studies suggest that newer inputs of inorganic mercury are generally more available for biological methylation processes than older mercury, and that mercury deposited to aquatic ecosystems becomes less available for uptake by biota over time (Harris et al. 2007; Orihel et al. 2008). The METAALICUS (Mercury Experiment To Assess Atmospheric Loading In Canada and the US) project, a whole ecosystem experiment designed to study the activity, mobility, and availability of atmospherically deposited mercury, found that newly deposited mercury was more reactive than native mercury with respect to volatilization and methylation pathways (Hintelmann et al. 2002). In addition, Moreau et al. (2015) found that the bioavailability of new mercury depends on the size and composition of natural organic matter (NOM) in the environment, and also on the concentration and bioavailability of aged mercury (e.g., ambient mercury) already associated with NOM. Studies conducted using isotopically labeled mercury compounds incubated with certain NOM isolates found that differences in availability for methylation between old and new mercury depend on interactions between mercury and NOM in the system (Moreau et al. 2015). A natural environment likely harbors more than one “pool” of mercury bioavailable for methylation (i.e., new vs. aged Hg(II), or Hg(II) that is labile or tightly bound to NOM or other functional groups), for which the bioavailability, and the methylation rate, of each Hg pool depend partially on that of one or more others (Moreau et al. 2015).

Sulfate. Sulfate concentrations and methylation rates can be both positively and negatively correlated (ERM 2017d). Low concentrations of sulfate can limit reduction reactions by SRB, while high concentrations can slow down reactions due to production of waste products which compete for available oxidized mercury (Hg[2+]), the form that is more likely to be methylated (Environment and Climate Change Canada 2016). While a lower limit of sulfate concentration has not been identified, the optimal sulfate concentration for Hg(2+) methylation to occur ranges from 1 to 29 mg/L (Environment and Climate Change Canada 2016; Drott et al. 2008; Benoit et al. 1999, 2001). Up to about 50 mg/L, sulfate can stimulate SRB (Houben et al. 2016), indicating that until that concentration is reached, the availability of sulfate is a rate-limiting factor for the formation of methylmercury in the environment (ERM 2017d). At sulfate concentrations greater than 100 mg/kg, the formation of sulfide phases may also limit the metabolic rate of SRB, and sulfide buildup results in decreased methylation of mercury (Fitzgerald and Lamborg 2014).

Available literature suggests that increases to sulfate concentration in natural systems with low concentrations of sulfate result in increases in methylmercury at a factor increase 0.5 to 1.0 times that of sulfate (ERM 2017d). Studies involving enhancing sulfate concentrations in a wetland/peatland environment demonstrate that the methymercury response to enhanced sulfate concentrations was approximately proportional; i.e., a 10-fold increase in porewater sulfate concentration was accompanied by a 5- to 7-fold increase in porewater methylmercury (Jeremiason et al. 2006). Background sulfate concentrations were initially low (at or below 0.05 mg/L) until the initial spike, and subsequent spikes in the summer did not increase sulfate or methymercury concentrations due to warmer temperatures enhancing sulfate reduction rates.

Building on previous work performed at the Marcell Experimental Forest in northern Minnesota, Coleman Wasik et al. (2012) investigated the effect of enhanced sulfate concentrations on methylmercury concentrations in peat/wetland porewater. Artificially increasing the sulfate concentration from approximately 2 to 2.9 mg/L in 2006, and from less than 0.25 to 3.8 mg/L in 2008, resulted in an increase of methylmercury from approximately 2ng/L to 4.3 ng/L in 2006, and to 3.6 ng/L in 2008. The study concluded that increasing the sulfate concentration in porewater by four times led to a methylmercury increase of a similar magnitude in both the peat porewater and peat solid.

Data presented in Mitchell et al. (2007) indicate that the consistent positive relationship between sulfate and methylmercury concentrations is not observed at higher sulfate concentrations. High methylation rates occurred at sulfate concentrations below 5 mg/L, while above 7 mg/L, the methylation rate decreased to near zero at 47 mg/L sulfate, supporting the assertion that high concentrations of sulfate can slow down or inhibit the reactions due to the production of sulfide metabolic waste products (Environment and Climate Change Canada 2016).

Although a number of environmental variables contribute to the overall effect of sulfate methylmercury concentrations, overall the literature suggest that in systems where sulfate is initially at low concentrations, methylation rates may increase up to a similar magnitude as the increase in sulfate concentrations (ERM 2017d).

Review of Mercury Models

Several existing mercury models were assessed for potential use in evaluating effects of the proposed project (ERM 2017d). In a mercury report to Congress, EPA (1997a, b, c), modified a generalized watershed and water body model, IEM-2M, to predict environmental media concentrations resulting from atmospheric mercury deposition. The IEM-2M model is mainly soil-based, representing a watershed with untilled upland soils, and is not intended to be representative of wetlands, nor designed to model transformations among mercury species.

EPA's analysis relies heavily on computer modeling because no monitoring data has been identified that conclusively demonstrates a relationship between individual anthropogenic sources in emissions and increased mercury concentrations in environmental media.

Uncertainties inherent in the modeled estimates arise from many individual assumptions needed for input to the IEM-2M model, and that it has not been validated with site-specific data; yet a significant input to the model is the estimate of existing mercury concentrations in the environment from all sources. EPA (1997c) also noted there is a general lack of conclusive data characterizing methylation rates, and that the rate of movement of mercury from watershed soils to water bodies is prohibitive to modeling. This lack of information caused poor predictability in the model results; because of the uncertainties, EPA interpreted the results qualitatively rather than quantitatively.

The EPA (2008) SERAFM model was developed for application in lake systems, not wetlands, uplands, or streams. It was designed to simulate lakes in a watershed, and to be used at contaminated sites, but not to predict future impacts on a watershed. Essential data to run the model include lake, water, sediment, and fish tissue concentrations of total mercury and methylmercury collected at three sample points in summer with replicates. Future predictions of concentrations in Mine Site media would not have the necessary data to satisfy the spatial and temporal requirements to run this model. More importantly, the data used to develop the model are not applicable to the ecosystem at the Mine Site.

D-MCM, a proprietary model published by Electric Power Research Institute (EPRI) (2013), is another lake-focused model that is not applicable to the Mine Site, because there are no lakes in the proposed project area. Additionally, the complexity of the D-MCM model is best utilized when extensive information is known about speciated mercury in the food web (ERM 2017d).

Because none of the available models are applicable to the project, and because of the numerous uncertainties inherent in the model inputs and results, a site-specific calculation approach was used to predict project-related effects, as described below.

Predicted Total Mercury and Methylmercury in Surface Water

Changes in the availability of mercury for methylation processes due to atmospheric deposition of mercury from point and fugitive sources associated with the proposed project would vary by watershed. The project is estimated to increase mercury deposition over baseline values in the nearest watersheds, generally decreasing with distance from the project. The area-weighted average deposition rate (including wet and dry deposition) of watersheds within a 20-mile radius of the Mine Site is estimated at 0.58 $\mu g/m^2/yr$ (Environ 2015; ERM 2017d). This is an overestimate of the deposition rate in the study area in that the Environ (2015) approach did not account for the pit retention of emissions.

Atmospheric mercury that would be deposited from Mine Site sources would consist of gaseous elemental mercury Hg(0), gaseous oxidized mercury Hg(2+), and the majority in particulate form Hg(p). Oxidized mercury, Hg(2+), the form that is more likely to be methylated, would be a minor component. Environ (2015) estimates that the deposition rate of Hg(2+) from project sources would be approximately 0.0116 µg/m2/yr, or 2 percent of total project-related mercury deposition. The rate of methylation for the newly deposited Hg(2+) is not known for the study area. As described above, different forms of mercury may have different methylation rates, and newly deposited mercury may be methylated more rapidly; thus, it was assumed that all of the newly deposited Hg(2+) would be rapidly methylated to methylmercury (ERM 2017d).

The dominant mercury species in the global atmospheric environment is Hg(0), constituting more than 90 percent of total atmospheric mercury (Lyman et al. 2007). Atmospheric transport and deposition of mercury is complex, as there are chemical, diurnal, and seasonal trends that affect the redox state of mercury. Atmospheric deposition flux has been estimated globally and regionally, and can vary by two or more orders of magnitude. Annual total mercury deposition rates from the six Mercury Deposition Network (MDN) stations located in Alaska vary between 1.2 and 5.7 µg/m2/yr, with annual total mercury precipitation weighted mean concentrations varying between 1.5 and 10 ng/L. Estimated global background deposition near the Mine Site is 8.4 µg/m2/yr. The Hg(2+) fraction of background deposition (wet and dry) is estimated at 49 percent, or 4.1 µg/m2/yr (Environ 2015; ERM 2017d).

As discussed above, the increase in methylation rate is of a similar magnitude to the increase in sulfate concentration in systems where sulfate is initially at low concentrations. The average annual background atmospheric sulfur dioxide (SO_2) concentration is approximately 1.3 micrograms per cubic meter (µg/m3), and annual project emissions would result in an additional 0.04 µg/m3 at the Core Operating Area boundary, which equates to a 3 percent increase in atmospheric SO_2. A conservative assumption would be that the SO_2 concentration throughout the 20-mile radius study area increased by this same amount, and that all SO_2 produced was deposited locally as aqueous sulfate, increasing the annual load and average sulfate concentration in local soils and streams by approximately 3 percent (ERM 2017d).

Sulfate and mercury loading from the Mine Site WTP discharge to Crooked Creek was also evaluated for potential effects on surface water concentrations and mercury methylation rates (Donlin Gold 2017o). The discharge is predicted to slightly increase sulfate and mercury concentrations in Crooked Creek (from 14.0 to 14.3 mg/L sulfate, and from 9.1 to 9.4 ng/L total mercury) at a point immediately downstream of the proposed water treatment plant outfall, and recover within a short distance downstream. Loadings are predicted to decrease slightly within this zone, because overall flow would be reduced along this reach due to dewatering activities. These estimates were produced using conservative (95th percentile, higher than average annual) estimates of sulfate and mercury concentrations in the WTP discharge. Because overall sulfate loading would decrease downstream of the WTP and recover within a short distance, these effects were not considered in the calculation of methylation rate applied to the wider study area.

The project is not expected to significantly impact the pH, DOC, temperature, oxygen content, or redox potential of the receiving environment. Therefore, no additional adjustments to methylation rates for items such as carbon loading were incorporated.

Based on the above, the following calculation-based approach was employed to predict increases in total mercury and methylmercury in the Mine Site area from project-related atmospheric deposition effects (ERM 2017d). The steps of the calculation are as follows:

1. Project effects are expected to increase the total mercury load to the receiving environment, primarily due to deposition of Hg(p) from fugitive dust and point source emissions. Of the estimated project-related deposition rate of 0.58 µg/m^2/yr of mercury, 0.0116 µg/m^2/yr, or about 2 percent of the total, would consist of Hg(2+). Because this fraction represents the newly deposited mercury that could be more rapidly methylated, it was assumed that all of it would be converted to methylmercury instantly.

2. The remaining mercury (98 percent of total mercury deposition) was assumed to be non-Hg(2+), which would undergo methylation at a rate of 1 percent. This is twice the median rate of all project baseline data (ERM 2017d), but is similar to the mean of paired soil samples collected in 2013 (ARCADIS 2014). An EPA (2005a) mass fraction equation (see Appendix F, Section F-5.2.1) was used to calculate future soil Hg(2+) and non-Hg(2+) concentrations based on these deposition rates.
3. The reviewed literature and Mine Site data suggests that the increase in methylation rate would be of a similar magnitude to the increase in sulfate concentration in the system. The estimated SO_2 emissions from the project would result in a 3 percent increase in atmospheric SO_2, which was conservatively assumed to translate to a similar increase in deposited sulfate. This would result in increasing the methylation rate of baseline total mercury and newly deposited non-Hg(2+) from the current estimate of 1 percent to a future rate of 1.03 percent (ERM 2017d).
4. Based on the above mercury fractionation, compounded by the increase in methylation rate, total mercury in soil is predicted to increase by 0.2 percent, and methylmercury by 4 percent (ERM 2017d).
5. To estimate future concentrations in aquatic systems, the same percent increase computed for soil systems was used to derive total mercury and methylmercury in sediment, and changes in sediment mercury content were assumed to be proportionally reflected in surface water concentrations (ERM 2017d).

Applying the 0.2 percent increase to baseline total mercury concentrations shown in Table 3.7-2 yields future surface water concentrations within a 20-mile radius of the Mine Site of 7.83 ng/L (mean) and 23.55 ng/L (95% UCL), resulting from both global background and project-related mercury deposition. While the mean value is below the chronic AWQC of 12 ng/L, the range of baseline data reflected in the 95% UCL value indicates that this criteria would be exceeded in some areas within the 20-mile radius of the Mine Site some of the time.

Applying the 4 percent predicted increase in methylmercury to the mean of baseline methylmercury data in Mine Site surface water of 0.074 ng/L (Table 3.7-6) yields a future concentration of methylmercury in surface water, from both atmospheric deposition and increases in methylation rates, of 0.077 ng/L. As described in Section 3.7.1.1, the applicable water quality criterion for methylmercury is expressed as fish tissue concentration (0.3 mg/kg), not a surface water concentration. For the purposes of the HHRA, ERM (2017d) estimated an increase in northern pike tissue from a baseline of 0.0829 to 0.0871 mg/kg based on the following: assuming 100 percent of mercury in tissue samples is methylmercury; a bioaccumulation factor (BAF) of 55 from sediment to fish tissue; and a food chain multiplier (FCM) of 3 from sculpin to pike. This approach accounts for both the greater bioaccumulation expected in pike, and the larger Mine Site dataset for resident sculpin tissue. The predicted concentration represents an increase of about 5 percent in fish tissue methylmercury concentration from project-related effects, but is well below the methylmercury criterion. Additional assessment of predicted fish consumption impacts are discussed in Section 3.22, Human Health.

ERM (2017) also evaluated the concentrations of total mercury and methylmercury in Crooked Creek under baseflow conditions (i.e., the portion of streamflow fed by shallow groundwater), in order to assess effects from direct atmospheric deposition to surface water without contributions from stormwater runoff. Time periods with baseflow conditions were identified by separation analysis at four Mine Site stations where flow was continuously monitored. Baseline concentrations for all Mine Site monitoring stations during baseflow conditions were then calculated for these same time periods (TetraTech 2013; ERM 2017d). The average of the calculated baseflow concentrations for all stations is 2.6 ng/L total mercury and 0.07 ng/L methylmercury. Applying the predicted 0.2 percent increase to the total mercury concentration yields a future surface water concentration of 2.61 ng/L total mercury and 0.073 ng/L methylmercury, resulting from both global background and project sources (ERM 2017d).

Summary of Predicted Mercury Increases in Surface Water

In terms of intensity, estimates predict that the chronic total mercury surface water quality criteria may or may not be exceeded as a result of project-related atmospheric deposition of mercury. Impacts would depend upon sample location, season, and existing natural concentrations. The estimated levels of impacts are predicated upon several conservative assumptions with regard to phase partitioning, environmental transport, and methylation rates. Total mercury concentrations for watersheds closest to the Mine Site could rise approximately 40 percent, to an average concentration that is close to or below water quality criteria. Effects decrease to negligible within 15 to 20 miles of the Mine Site. Based on an area-weighted average over a 20-mile radius from the Mine Site, total mercury in surface water is estimated to increase about 0.2 percent. In either case, some surface water concentrations would range above criteria some of the time in both background areas and the project-impacted area. Methylmercury concentrations within the 20-mile radius are predicted to rise about 4 to 5 percent in surface water and fish tissue to levels that would be below applicable methylmercury criteria for fish consumption. (Potential impacts to fish and other aquatic organisms are described in Section 3.13, Fish and Aquatic Resources, and impacts from human consumption of fish residing in area waters are described in Section 3.22, Human Health).

The duration of impacts to mercury concentrations in surface water would be such that concentrations would be expected to last throughout Operations and return to pre-activity levels at some time after the completion of the project. The geographic extent of such impacts would be considered regional due to the potential for impacts outside of the immediate Mine Site area. In terms of context, impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high water quality or water resources that are considered valuable in the region.

Water Quality at Water Management Facilities

The Donlin Gold Project includes numerous structures and operating rules designed to ensure availability of sufficient freshwater for ore processing while minimizing the accumulation of excess water in the system (SRK 2017b).

During the operations phase of the mine, water from the process plant would be recycled through the TSF. The Upper and Lower CWDs, the pit dewatering system, and the Snow Gulch Freshwater Reservoir would also supply water to the process circuit under certain conditions. The water balance model and operating rules for water management are discussed in Section 3.5, Surface Water Hydrology.

The TSF and upper and lower CWDs included under Alternative 2 are not considered waters of the State of Alaska or waters of the U.S. These facilities are not subject to regulation under the Clean Water Act or the APDES permitting program; the quality of water in the TSF and CWDs is discussed in this section in order to provide a means for comparison with the baseline water quality conditions that presently exist in the American Creek, Anaconda Creek, and Snow Gulch watersheds; and to provide an understanding of potential impacts to ecological receptors that may encounter mine water bodies (see Section 3.12, Wildlife).

Tailings Storage Facility

Tailings deposited in the TSF during the operations phase would consist of flotation tailings, wash thickener overflow, and the tailings from the carbon-in-leach (CIL) circuit.

The configuration and chemistry of the TSF, and the resulting impacts to the quality of water contained in the TSF, would evolve over the operational period of the mine. The tailings would be comprised initially of about 64 percent water (36 percent solids) by weight. Predicted concentrations of constituents in the liquor component of the tailings generated through metallurgical pilot testing (Appendix H, Geochemistry, Table H-4) suggest that the final plant tailings liquor would be elevated relative to the most stringent water quality criteria for a number of constituents. The concentrations of constituents in the TSF pond and pore water could increase beyond the results summarized above by as much as a factor of three, as discussed in Appendix H, Geochemistry, due to recirculation of tailings water through the process circuit (SRK 2015b). Assuming the three-fold concentration, the predicted concentrations are as follows: sulfate (5,800 mg / L), arsenic (3,300 µg / L), mercury (10,000 ng / L), manganese (2,000 µg / L), molybdenum (230 µg / L), antimony (22 µg / L), and selenium (42 µg / L) (SRK 2015b). The water quality criteria are mentioned in this section to provide the reader with a frame of reference for interpreting the predicted concentrations of constituents in the TSF water; however, surface water in the TSF is not subject to regulation under the CWA or the APDES permitting program.

Under Alternative 2, fugitive dust would be generated from drilling, blasting, material handling, ore loading and unloading, waste loading and unloading, ore and waste hauling, maintenance equipment (dozers, graders, water trucks), and wind erosion of exposed surfaces including the TSF tailings beach, haul roads, access roads, waste dumps, and stockpiles. As described in Section 3.8, Air Quality, during operations, 90 percent fugitive dust control efficiency would be expected for dust generated from unpaved roads (haul roads and access roads), material handling (ore and waste), and maintenance equipment (dozers, graders, water trucks). No dust controls would be applied to the fugitive emissions resulting from drilling, blasting, or wind erosion of the TSF tailings beach. Additional best practical methods to suppress dust from other dust generating sources within the Mine Site would minimize the potential for water quality impacts associated with dust from the TSF and other mine site components.

Contact Water Dam Ponds

The lower and upper CWDs would be located in the American Creek watershed, adjacent to the WRF, with the objective of managing contact water and runoff from the WRF and pit. The dams would contain water that would be used throughout the year as a source of water for the process plant (SRK 2017b).

As discussed in Section 3.7.3.2.1, Geochemistry, and Appendix H (Section H-3.4 and Table H-6), the Lower CWD would receive runoff from a variety of sources:

- Surface and seepage runoff from the WRF (bare and reclaimed);

- Runoff from undisturbed ground upgradient of the WRF;
- Surface runoff within the open pit footprint;
- Horizontal drains from the open pit;
- Runoff collected within the ore stockpile berm; and
- Runoff from the SOB stockpile.

The Upper CWD would receive runoff from undisturbed areas and would also serve to provide additional capacity for contact water storage.

There would be considerable variability in the quality of the water that would be contained behind the lower and upper CWDs, as shown in Table H-6, Appendix H. As the volumes of water stored behind the Lower CWD increases, the waste rock would become progressively inundated. Subsequent use of the CWD water in the process circuit would decrease the volumes of water stored behind the CWDs, allowing the waste rock to dry. Intermittent drying and inundation of the waste rock could result in increased rates of oxidation and dissolution of sulfide minerals, leading to ARD and potential for leaching of metals and metalloids (e.g., arsenic) and salts (e.g., sulfate), which could degrade the quality of water stored behind the CWDs. Water quality issues in the CWD reservoirs would be managed during operations by limiting storage volumes in the Lower CWD not to exceed 405 acre-feet more than 5 percent of the time (SRK 2017b).

The objective of treating water from the CWDs at a maximum rate of 1,100 gpm is to build flexibility into the water management system such that TSF pond volumes are minimized to the extent practical during operations (BGC 2015f, SRK 2017b). Water from the upper and lower CWDs would be managed as mine contact water and would not be discharged to the environment. The operation of the Lower CWD as described under Alternative 2 would cause acute and obvious adverse changes to the quality of water in the CWDs' portion of the American Creek watershed relative to baseline conditions, but the Lower CWD would prevent the contact water from reaching the lower part of the American Creek watershed and Crooked Creek.

Closure

Pit Lake and Post-Closure WTP Effluent

The pit lake would serve an important water management function during closure. As described in Appendix H, Geochemistry, water from the TSF and seepage from the WRF would be directed to the bottom of the pit lake. Pumping would be required to get TSF and SRS water to the pit rim, where it would be combined with WRF seepage, then flow by gravity-fed pipe to the bottom of the pit lake. Runoff would be directed to the surface of the pit lake. The difference in densities of these waters and their vertical delivery locations would result in a stratified lake with lower quality waters at the bottom and higher quality waters at the surface. As the pit fills, water level and quality (at different depths) would be monitored, and the pit lake model would be re-calibrated as data become available.

Treatment and discharge of lake water would be required as the pit nears complete filling. A post-Closure WTP would be constructed 5 years prior to the pit completely filling, and treatment would begin 2 to 3 years before filling is complete and discharge commences to allow adequate freeboard (SRK 2017b). Long-term management of mine contact water by pit lake containment and groundwater flow management is a standard and proven mine management strategy used at similar sites around the world. Long-term pumping and treatment of pit lake water is necessary to maintain containment, and would be funded by financial assurance instruments set up by the applicant with ongoing state oversight. This would be the largest mine in Alaska to employ these techniques.

The parameters of concern predicted in pit lake water requiring treatment prior to release are discussed in Section 3.7.3.2.1 and Appendix H, Geochemistry. Pit lake modeling results (Lorax 2012) indicated that the chemistry of the pit lake surface water would not likely change substantially from the time when water treatment begins (approximately 52 years following closure) through the final model year (99 years after closure). However, as discussed in Section 3.7.3.2.1 and Appendix H, assumptions regarding the weathering of highwall rock do produce substantial differences in metals concentrations. These model results, along with predicted pit lake water treatment effluent and the most stringent AWQC, are summarized in Table 3.7-43.

The post-Closure WTP would use conventional chemical precipitation technology, specifically a high-density sludge (HDS) system with a silica gel polish, to target elements such as aluminum, arsenic, manganese, antimony, cadmium, copper, selenium, and mercury (SGS-CEMI 2008, SRK 2017b). It is possible that certain constituents, such as selenium and TDS, would not be removable with such a system. Hence, the final WTP configuration would be updated closer to the end of mine life to incorporate advances in treatment technologies (SRK 2017b).

Tailings Storage Facility and Seepage Recovery System Water Quality

At closure, the tailings pond water would be pumped to the open pit. In addition, tailings water from beach runoff and tailings consolidation would be pumped to the open pit as reclamation of the TSF begins. As the TSF is drained, it would be reclaimed with an engineered cover overlain by a peat/mineral growth media mix. The engineered cover would promote runoff and reduce the potential for runoff or precipitation to contact the consolidated tailings. The reclaimed TSF would be covered and revegetated, with intercepted precipitation draining to a lined settling pond. Once this water meets AWQC (expected in approximately Year 5 of closure), it would be directed to Crevice Creek through a spillway (SRK 2017b).

Closure of the proposed TSF cover is expected to take approximately four years, with an overall goal of minimizing meteoric water that comes into contact with the tailings. The underlying tailings would continue to drain and consolidate over a period of approximately 52 years, with water from the consolidation process and any infiltrated water through the cap being captured in a capillary break between the cover and the tailings and directed to the pit, along with water that flows to the seepage recovery system (SRS). It is assumed that by approximately Year 52 post-Closure, consolidation would be complete and only minimal pumping from the capillary break would be required. Quarterly monitoring to demonstrate seepage water quality would continue for both the SRS pond and collection wells until analytical results indicate acceptable chemistry for discharge. If the seepage water is not suitable for discharge, it would continue to be pumped to the pit lake (SRK 2017b).

Table 3.7-43: Predicted Pit Lake and Treated WTP Effluent Water Quality

Parameter[1]	Units	Most Stringent Applicable Water Quality Criteria	Treated Effluent Water Quality (Base Case)[5]	Pit Lake Surface Water Quality at Closure Year 99		
				Base Case	Peak-PAG Case	
				No Reaction[2]	No Reaction[3]	Mineral Precipitation[4]
pH	units	6.5 - 8.5	6.5-8.5	6 (est.)	5.2	5.1
Sulfate	mg/L	250	31	31	48	48
Total Dissolved Solids	mg/L	500	139	139	143	97
Aluminum	µg/L	750[a,c]	1.3	310	590	97
Antimony	µg/L	6	0.040	67	75	76
Arsenic	µg/L	10	0.18	112	190	190
Boron	µg/L	750	194	200	230	200
Cadmium	µg/L	0.23[a,b]	0.030	0.24	0.31	0.31
Chromium	µg/L	100	4.2	4.0	4.3	4.3
Cobalt	µg/L	50	<3	3.0	5	5
Copper	µg/L	7.7	0.0235	1.4	7.1	6.7
Iron	µg/L	1,000	<1,000	830	960	180
Lead	µg/L	2.4[a,b]	2.0	2.3	2.7	2.7
Manganese	µg/L	50	0.397	128	170	15
Molybdenum	µg/L	10	<0.473	17	19	18
Nickel	µg/L	43[a,b]	0.40	11	20	20
Selenium	µg/L	5	1.41	20	23	23
Zinc	µg/L	100[a,b]	7	13	45	47
Mercury	ng/L	12	11[6]	25	28	28

Notes:

1 Values are for dissolved metals, and represent depths between 0 and 33 feet in lake. Shaded cells exceed most stringent water quality standard.

2 Lorax Environmental, 2015. Donlin Creek Gold Project Pit Lake Modeling Assessment in Support of Project Permitting, October; PitMod_base case 1012. xls

3 Lorax. 2014b.

4 Lorax. 2014c. Aluminum, iron, and manganese low-temperature oxyhydroxide minerals were allowed to precipitate if saturated.

5 ARCADIS 2013a, Donlin Gold Environmental Evaluation Document

6 Hatch 2015.

a Value shown corresponds to the chronic standard.

b Acute and chronic aquatic life numeric criteria for some metals (Cd, Cr, Cu, Pb, Ni, Zn) are hardness dependent. Values contained in this table were calculated using a hardness value of 80.55 mg/L as $CaCO_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d).

c Where the pH of receiving water is greater than or equal to 7.0 and the hardness is greater than or equal to 50 ppm as CaCO3, the chronic aluminum standard will then be equal to the acute aluminum standard, 750 µg/L as total recoverable aluminum.

Based on plans to capture, pump and ultimately treat all contaminated surface and porewater from the TSF, the impacts to surface water quality outside of the Mine Site resulting from the reclaimed tailings facility would be imperceptible.

Avoidance of impacts to surface water quality under Alternative 2 would depend upon the reliability of the pumps used to convey captured seepage to the pit lake during the approximately 52-year period during which the covered tailings would drain and consolidate. If the SRS pumping system were to fail during this period, it is estimated that it would take approximately ½ week to 1 week under high to average precipitation conditions, respectively, for the SRS to overflow to Anaconda Creek, which could subsequently impact Crooked Creek (Section 3.5, Surface Water Hydrology); and roughly 2 weeks for the cone of depression beneath the SRS to diminish to the extent that potentially contaminated groundwater can migrate downstream through Anaconda Valley. Unlike the WRF, the TSF would not naturally drain to the pit lake, and impacts to surface and groundwater quality could occur in the event of an SRS pump failure greater than 1/2 to 2 weeks in duration.

The SRS water would be comprised of seepage from the TSF and groundwater flow to the SRS. The ratio of seepage to groundwater flow in the SRS water would vary over time. Based on estimated rates of seepage from the TSF and estimated rates of groundwater flow reporting to the TSF SRS during the post-Closure period, approximately 96 percent (431 gpm) of the SRS water would be natural groundwater, and only 4 percent (18 gpm) would be seepage from the TSF (SRK 2017b, see Figure 3.5-26: Schematic Water Balance During Closure, in Section 3.5, Surface Water Hydrology). Thus, potential TSF seepage would be diluted by a factor of about 24:1 with groundwater before reporting to the TSF SRS. Several constituents in the SRS water are predicted to exceed AWQC in spite of this dilution factor, including TDS, sulfate, antimony, arsenic, cadmium, cobalt, copper, iron, manganese, selenium, and mercury (Table 3.7-44. However, the release of SRS water to the environment during the approximately 52-year period during which the covered tailings drain and consolidate would only occur in the event of a pump failure greater than 2 weeks in duration, and such an event is considered unlikely under Alternative 2.

Table 3.7-44 shows the 95th percentile concentrations of selected constituents in natural groundwater samples from the mineralized area, the concentrations of constituents in the tailings pore water, and the potential concentrations in the SRS water, which would be comprised of approximately 96 percent groundwater and 4 percent tailings pore water. Actual concentrations of constituents in SRS water may be lower than shown in Table 3.7-44 because the groundwater in the vicinity of the TSF may have lower mineral concentrations than water in the proposed pit area, and actual seepage through the liner may be below predicted amounts 52 years after closure.

Because water treatment and discharge would continue after mine closure, in-stream constituent concentrations in Crooked Creek would be expected to be essentially the same as those occurring during mine operations. An increase in the dilution effect that occurs during low stream flows could be expected because the predicted annual discharge is more than twice as high after closure (2916 gpm) than during active operations (1,293 gpm) (Figures 3.5-22 and 3.5-29).

Table 3.7-44: Potential SRS Water Quality

Parameter	Units	95th Percentile Concentrations in Groundwater[3]	Tailings Pore Water[4]	SRS water (estimated based on 431 gpm groundwater and 18 gpm tailings pore water)	Most Stringent Applicable Water Quality Criteria
Aluminum	µg/L	160	1.9	154	750[2a]
Antimony	µg/L	33	1,300	83.8	6[2d]
Arsenic	µg/L	1,950	17,000	2,550	10[2d]
Barium	µg/L	1300	10	1,250.0	2,000[2d]
Boron	µg/L	19		18.2	750[2e]
Cadmium	µg/L	0.5	0.91	0.5	0.23[2a]
Chromium	µg/L	3	7.1	3.2	100[2d]
Cobalt	µg/L	4	710	32.3	50[2e]
Copper	µg/L	3	680	30.1	7.7[2a]
Iron	µg/L	5,500	120,000	10,090	1,000[2b]
Lead	µg/L	2	2.3	2.0	2.4[2a]
Manganese	µg/L	1,440	450,00	3,186	50[2f]
Mercury	ng/L	19	1,400	74.4	12[2b]
Nickel	µg/L	19	320	31.1	43[2a]
Selenium	µg/L	5	220	13.6	5[2b]
Sulfate	mg/L	83.1	19,000	841	250[1]
Thallium	µg/L	1	2.5	1.1	1.7[2f]
TDS	mg/L	577	19,000	1,316	500[1]
Zinc	µg/L	30	78	31.9	100[2a]

Notes:

1 18 AAC 70. ADEC, Alaska Water Quality Standards. Amended as of April 8, 2012.

2 Alaska Water Quality Criteria Manual for Toxic and Other Deleterious Organic and Inorganic Substances. Amended through December 12, 2008. Most stringent criteria used. Criteria are expressed in terms of dissolved metal in the water column.
2a Aquatic life for fresh water hardness-dependent criteria. Values calculated using a hardness value of 80.55 mg/L as $CaCO_3$ based on 15th percentile of hardness data measured in receiving water at Crooked Creek Station CCBO 1996-2015 (Donlin Gold 2017d). For Al, if pH≥7.0 and hardness ≥50, then 750 µg/L.
2b Aquatic life for fresh water (chronic) criteria.
2c Aquatic life for fresh water (acute) criteria.
2d Drinking water primary maximum contaminant levels.
2e Irrigation water criteria.
2f Human health criteria for non-carcinogens (for consumption of water + aquatic organisms).
2g Aquatic life criteria (chronic)for free cyanide, measured as weak acid dissociable (WAD) cyanide.

3 Source: BGC (2014c); 95th percentile concentrations for groundwater in the mineralized area. Dissolved values listed when available; total values listed when dissolved values unavailable. Detection limits used for values reported as non-detect.

4 Source: SRK (2007); highest concentration of the three redox conditions included in the tailings porewater model.

Summary of Mine Site Impacts

Surface water in the American and Anaconda Creek watersheds would be influenced by the creation and perpetual maintenance of the pit lake and TSF impoundment; however, due to planned water treatment and water management practices, water from the TSF and pit lake would not leave these watersheds, and would be restricted to facilities within discrete portions of the project area. In terms of intensity, effects from all project–related discharges to Crooked Creek would be treated to meet the most stringent AWQC prior to discharge. These include treated waters from pit dewatering during Construction; pit dewatering, CWDs, SRS, and TSF pond water during Operations; and pit lake and SRS during post-Closure. There is a low risk that impacts to Anaconda and Crooked creeks could result in the event of SRS pump failure and overflow in post-Closure thereby exceeding AWQC. In terms of context, there are many relatively similar watersheds in the region, and water quality is governed by regulation.

The intensity of impacts to surface water quality resulting from atmospheric deposition of mercury would vary. Effects are sufficient to exceed AWQC at some locations within the Crooked Creek and Donlin Creek watersheds, where the inputs of mercury to the water are expected to be the greatest. Water quality is likely to be within regulatory limits on average, but could exceed baseline conditions and EPA chronic criteria in some areas (based on 95 percent upper confidence limit values). The duration of such impacts to mercury concentrations in surface water would persist through the life of the mine, but water quality would be expected to return to pre-activity levels at some time after the completion of the project. The extent or scope of such impacts may be inside and/or outside of the immediate project area due to mercury deposition from atmospheric sources.

Transportation Corridor

Construction

Roads

Material sites are proposed along the mine access road to supply appropriately sized and crushed gravel for road construction and continuing maintenance. As each site is no longer needed, it would be reclaimed. Stockpiles of non-acid generating (NAG) rock used for road construction could act as sources of arsenic, selenium, antimony, and possibly other constituents of concern, to area surface water resources, including Jungjuk Creek (see Appendix H, Geochemistry). Eight material sites have been identified along the lower mine access road corridor. Six material sites have been identified along the central mine access road corridor. In order to mitigate potential impacts to surface water resources that could result from materials stockpiles, Donlin Gold would test materials for the potential to act as sources of constituents of concern prior to stockpiling materials. Materials that could act as sources of contamination would not be used for road construction, and other material sources would be identified (URS 2013a).

Runoff and Sedimentation from Construction Activities

Runoff and sedimentation associated with the construction of transportation facilities (mine access road, airstrip, ports, and tank farms) would potentially impact the quality of surface water resources. The intensity of impacts resulting from runoff of sediment-laden waters would be such that BMPs and erosion and sedimentation control (ESC) measures would mitigate effects (described in Section 3.2, Soils) so that receiving waters would comply with ADEC water quality criteria. (Turbidity levels in freshwater streams may not exceed 5 NTU above natural conditions when the natural turbidity is 50 NTU or less, and may not have more than 10 percent increase in turbidity when the natural turbidity is more than 50 NTU, not to exceed a maximum increase of 15 NTU.) Turbidity levels and concentrations of suspended sediment in the water would be expected to return to pre-activity levels immediately following the cessation of the construction activities. The extent or scope of impacts would affect water quality within discrete sections of the project area. The context of impacts would affect areas of ordinary water quality or where there is an abundance of water resources.

Ports and Tank Farms

The Angyaruaq (Jungjuk) Port would serve as the node between barge transport from Bethel and road transport to the mine. A dock would also be constructed at the Bethel Port, and fuel storage facilities at Dutch Harbor would be expanded. The Jungjuk dock facility would be constructed both above and below the high water line and would include barge berths on the dock face. An access road would be constructed against the front of the bluff to allow access from the berth area to a terrace storage yard. Hazardous materials and fuel storage facilities would be constructed in concrete- and geosynthetic-lined and bermed containment structures at the back of the gravel pad, furthest from the river, and would be fully covered under required spill contingency and response plans. BMPs and ESC structures would be constructed and maintained at the dock and tank farm facilities as needed to prevent degradation to adjacent wetlands or waters of the U.S. (see Section 3.2, Soils). The potential for impacts to water quality from fuel spills at ports and tank farms is described in Section 3.24, Spill Risk, and Section 3.7.3.8. Any actions that would occur at Dutch Harbor or the Port of Bethel at the Bethel Yard Dock are not part of the proposed action, and are considered connected actions (see Section 1.2.1, Connected Actions, in Chapter 1, Project Introduction and Purpose and Need).

Operations and Closure

Barge operations on the Kuskokwim River would add roughly twice as many barge trips as are currently operated on the Kuskokwim (Section 3.23, Transportation), and would be restricted to an ice-free season of approximately 120 days per year. Barge operations are expected to have imperceptible impacts on surface water quality in the Kuskokwim River. Although concentrations of suspended sediments may increase in some shallow locations at low water levels as a result of increased barge traffic (see Section 3.5, Surface Water Hydrology), these impacts would be such that the disturbed sediments are expected to settle within a short distance, and are unlikely to exceed natural levels of suspended sediment in this river (Table 3.7-7) (Wang 1999). The duration of any changes to surface water quality resulting from increased barge traffic would be intermittent (low water conditions only), and water quality would be expected to return to pre-activity levels within hours of a barge passing.

During reclamation of the Angyaruaq (Jungjuk) Port, all support facilities (buildings, tanks, fencing, and equipment) would be removed. Reclamation activities could result in infrequent impacts to surface water quality in the areas immediately adjacent to the reclamation activities. It is possible that levels of turbidity and concentrations of suspended solids would increase as a result of Closure Phase activities. BMPs and stormwater pollution controls would limit the intensity of such impacts, and water quality would be expected to return to pre-activity levels at the completion of the activities.

Summary of Transportation Corridor Impacts

In terms of intensity, effects to surface water quality under Alternative 2 would be below water quality regulatory limits. Impacts would result from occasional barge–induced suspended sediment, or erosion effects at construction sites, due to high natural conditions (Section 3.7.2.1.2) and/or planned mitigation measures designed to limit erosion effects. Potential impacts resulting from runoff of water from rock obtained from material sites along the road alignment and used for road construction could include inputs of arsenic, selenium, antimony, and possibly other constituents of concern, to area surface water resources, including Jungjuk Creek. The extent or scope of such impacts would be limited to areas in the immediate vicinity of roads and road construction material stockpiles, and could be mitigated through implementation of a materials testing program prior to the beginning of road construction activities. Impacts would likely persist for the duration of the project, and water quality would return to baseline levels at some time following the completion of mining activities. In terms of context, impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high water quality or water resources that are considered valuable in the region.

Pipeline

Construction

Under any Alternative 2 option, the primary impacts to surface water quality would be associated with installation of the natural gas pipeline at river and stream crossings, and the use of local water sources for hydrostatic testing. Impacts at material sites would be the same as those described above under Transportation Corridor.

The majority of rivers and streams on the pipeline route would be crossed by open cutting in the winter months when flows are lowest and disturbance of the river and streambanks can be held to a minimum. Select rivers and streams would be crossed using HDD technology. In all cases, pipe would be buried to a depth that is below the scour potential of the particular river or stream, and effects to the river or stream at the location would be imperceptible through the life of the mine. Potential effects on the overlying river and recommended mitigation related to HDD frac-out are described in Section 3.3, Geohazards and Seismic Conditions and Chapter 5, Impact Avoidance, Minimization, and Mitigation.

At river and stream crossings where the open-cut (trench) method is employed, the dredged or excavated material would be kept to a minimum, and the work would be conducted in a manner to minimize turbidity of the water in the work area and downstream to ensure compliance with applicable water quality standards. Water quality regulatory limits may be temporarily exceeded at some open-cut stream crossings, as well as in areas needing ditch maintenance, but such impacts would intermittent, and would influence only a very small fraction of the project area. Excavation of the pipeline trench would not result in the relocation of any existing stream or river channel or restrict stream flow.

ADEC water quality standards for fresh water streams specify that turbidity levels may not exceed 5 NTU above natural conditions when the natural turbidity is 50 NTU or less, and may not have more than 10 percent increase in turbidity when the natural turbidity is more than 50 NTU, not to exceed a maximum increase of 15 NTU (Section 3.7.2.1.3). Surface water quality at stream crossings along the pipeline corridor during construction would be expected to comply with this water quality standard. Isolated occurrences of impacts above this standard could occur (e.g., during periods of high precipitation along summer construction segments), but are expected to be reduced within a short timeframe due to planned redundancies in BMPs, ESC measures, and reclamation / cleanup crew functions.

The extent of potential impacts resulting from hydrostatic testing (for pipeline pressure testing, Section 3.2.2.3.1, Soils) would be limited because the amounts of water required for hydrostatic testing would be small compared to potential sources from rivers and small lakes along the route (see Chapter 2, Alternatives, Table 2.3-16, Susitna Valley Winter Access Potential Water Extraction Sites for Ice Road Construction, and Table 2.3-17, Potential Water Extraction Sites for Pipeline Construction). Impacts to surface water quality would likely be imperceptible. Discharges of hydrostatic test water would meet the requirements of the applicable APDES General Permit.

Clearing of riparian areas during pipeline construction activities could result in additional impacts to surface water quality. If vegetative cover is disturbed and bare soil is exposed, the potential for introduction of fine-grained sediments (silts and clays) to surface water is greatly enhanced. Surface water impacts during pipeline construction resulting from increased amounts of sediment in the water would be mitigated by using HDD to install the pipeline under selected crossings, using BMPs and ESC measures (Section 3.2, Soils), restoring banks at stream crossing sites, and initiating reclamation of disturbed areas as soon as practicable.

Thus, most impacts from pipeline construction on surface water quality would be mitigated to be within ranges specified by regulatory limits, with isolated occurrences of higher intensity effects. The effects would be limited to discrete locations (e.g., stream crossings) within the project area. Sediment control measures specified in the ESC Plan (SRK 2013b, Appendix H), would minimize the potential for erosion impacts to move downgradient from disturbed sites to waterbodies.

Operations and Closure

Impacts to surface water quality during the operational phase of the pipeline are expected to be imperceptible. The ESC Plan would be implemented to control erosion and storm water runoff. Continued ground surveillance and monitoring and corrective erosion control and vegetation maintenance would be employed throughout the life of the project. Existing drainage patterns would be maintained where practical.

If the pipe is abandoned in place, then any impacts caused by removal of the pipe would be avoided. Reclamation of the right-of-way (ROW) would include methods to recontour and revegetate disturbed areas with native vegetation to maintain surface drainage patterns.

Summary of Pipeline Impacts

The primary impacts to surface water quality resulting from the installation, operations, and closure of the natural gas pipeline as proposed under Alternative 2 would be associated with erosion and the introduction of fine-grained sediments (silts and clays) to surface water. These impacts would be such that water quality would return to baseline levels following the completion of the activity responsible for the erosion and subsequent stream bank stabilization activities. Wetlands disturbed during construction would be restored shortly after installation of the pipeline. Following restoration and revegetation, few effects on water quality would be expected to result from wetlands disturbance associated with construction, operation, and closure of the pipeline facilities. (Impacts to wetlands and their associated functions are described in Section 3.11, Wetlands). The intensity of the impacts to surface water quality resulting from the construction, operation, and closure of the natural gas pipeline would meet applicable regulatory standards due to planned BMPs and ESC measures designed to limit effects on water quality. The extent or scope of impacts would be limited to discrete portions of the pipeline corridor. In terms of context, impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high water quality or water resources that are considered valuable in the region.

3.7.3.2.3 GROUNDWATER QUALITY

Mine Site

Construction

During construction, non-acid generating (NAG) waste rock with potential for metal leaching would be used for construction of the Lower CWD. The generation of seepage and runoff with elevated metals concentrations derived from metal leaching from the Lower CWD construction material could potentially infiltrate shallow (alluvial) groundwater resources in the immediate vicinity of the Lower CWD. However, detailed groundwater modeling (BGC 2014c) (Section 3.6, Groundwater Hydrology) indicates that any groundwater affected by metal leaching from the construction materials would flow towards the pit, where it would be captured by the pit dewatering system, and subsequently treated before discharge to Crooked Creek. Therefore, the groundwater quality would be affected only within a discrete portion of the Mine Site. Concentrations of arsenic, antimony, and selenium are likely to exceed regulatory criteria in the solutions generated by interactions between meteoric water and NAG waste rock during the construction period, and the impacts would be likely to persist throughout the construction and operations periods and beyond.

Runoff from the initial pit sidehill cut during pre-stripping would be captured by a berm and sump pump system and directed to the Lower CWD (SRK 2017b). It is possible that this runoff could influence the quality of groundwater in the immediate vicinity of the pit during the construction period. However, any groundwater affected by runoff from the pit footprint would be captured by the pit dewatering system and treated to achieve compliance with the most stringent applicable water quality standards before release to the environment. Therefore, the geographic extent of such impacts to groundwater resources would be limited to areas immediately adjacent to, and underneath, the footprint of the ACMA pit. In terms of context, impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high water quality or water resources that are considered valuable in the region.

To achieve targets for depressurization of pit walls for slope stability, pumping from pit perimeter dewatering wells would begin six months before the start of pre-stripping, approximately 1.75 years prior to the start of operations (SRK 2017b). A description of bedrock aquifer depths affected by dewatering and their hydraulic connectivity to Crooked Creek is provided in Section 3.6, Groundwater Hydrology. Groundwater pumping during the construction period would result in localized depletion of groundwater resources within the project area, but would not be expected to affect groundwater quality. With the exception of localized inputs of metals and TDS from the use of waste rock in the construction of the Lower CWD, the quality (composition) of the groundwater would be expected to remain unchanged relative to background conditions during the construction period.

During the construction period, potable water for the construction camp would come from eight wells installed in the deep bedrock aquifer south of Omega Gulch (Figure 3.7-6). The water would pass through a WTP prior to distribution as potable water (SRK 2016a), and monitoring and testing would be conducted in accordance with the ADEC Drinking Water Program (ADEC 2017d). The existing exploration camp wells would be decommissioned in accordance with ADEC (2017e) guidance once the camp wells are operational. The new wells would be designed and constructed under the oversight of the ADEC (2017d) Drinking Water Program. ADEC provides design protocols for well siting and construction to ensure that drinking water wells do not create a potential conduit for surface contaminants to reach groundwater, through the use of BMPs such as minimum casing depths, grouting, and proper surface finish specifications (ADEC 2016, 2017d). Laydown areas and a portion of the plant site may lie upgradient of the construction camp wells (see Figures 2.3-7, 3.6-2, and 3.7-6). Lined secondary containment of fuel storage at the plant site (Section 2.3.2.1), and BMPs for general waste handling and spill management (SRK 2016b) which would apply to laydown areas, are expected to minimize the potential for introduction of contaminants to the drinking water supply wells. As such, impacts to drinking water quality from potential wellhead or surface contaminant sources are expected to be minimal.

Existing groundwater quality in the vicinity of the construction camp drinking water wells is described in Section 3.7.2.2.1 (Groundwater Quality, Mine Site). While groundwater is expected to exceed drinking water standards for several inorganic constituents (antimony, arsenic, barium, iron, and manganese), the potable WTP would reduce these to concentrations below drinking water standards prior to distribution (SRK 2016a). The location of the construction camp wells would be upgradient or side-gradient of potential new sources of shallow groundwater contamination in the Construction period, such as the South Overburden Stockpile or lower part of the WRF (Figures 3.6-2 and 3.7-6). As such, impacts to drinking water quality from mine contact water are not expected to occur.

Operations

Waste Rock Facility and Pit Area

During mine operations, the primary mechanism for impacts to groundwater quality would involve infiltration of meteoric water contacting the WRF to the local groundwater system. Seepage from the WRF is planned to be collected in a rock underdrain and directed to the lower contact water pond for use as process water during operations. However, the WRF and underdrain would not be lined and some water from these facilities would leak into the underlying groundwater system (Section 3.6.2.3, Groundwater Hydrology). Water from the WRF would have concentrations of several constituents that are predicted to exceed the most stringent AWQC, and therefore adverse impacts to groundwater quality would occur in areas underneath and immediately adjacent to the WRF. Arsenic, antimony, and selenium are constituents of particular concern in the seepage from the WRF, because these metals are soluble in the neutral to basic solutions likely to be generated by interactions between meteoric water and the NAG waste rock (Section 3.7.2.2.1).

The intensity of these direct adverse impacts would be such that concentrations of several constituents in the WRF seepage would exceed water quality regulatory limits. The duration of the effects would last throughout the operational period. Detailed groundwater modeling indicates that the spatial extent of impacts to groundwater quality resulting from infiltration of WRF seepage would be limited to areas immediately beneath the WRF and a small area between the WRF and the pit, because the affected groundwater would be intercepted by the pit and the pit dewatering system (BGC 2014c).

Surface water infiltration to the WRF would be controlled through concurrent reclamation of portions of the WRF during operations. A cover layer designed to minimize infiltration would be placed over portions of the WRF as the placement of waste rock in those areas is completed. The surface of this layer would be contoured to direct precipitation to the Lower CWD (BGC 2011b), and would be constructed from non-acid generating rock and growth medium. In addition, surface waters from the American Creek drainage would be directed around the WRF through diversion ditches. These design elements would reduce impacts to groundwater quality during operations.

As described in Section 3.6 (Groundwater Hydrology), pit dewatering wells would lower the water table in the vicinity of the mine pit, resulting in a cone of depression designed to direct groundwater flows towards the pit from alluvial deposits and bedrock underlying the project area (see Section 3.6.2.3). Groundwater from the pit perimeter wells and water from the in-pit dewatering wells would be treated to meet the most stringent applicable criteria prior to discharge to Crooked Creek (BGC 2014b). Thus, direct impacts to bedrock and alluvial groundwater quality resulting from infiltration of WRF seepage would be contained and would affect only a discrete portion of the project area due to the hydraulic control that would be exerted to prevent the flow of water out of the mined area of the pit (BGC 2014c).

Tailings Storage Facility

The TSF would be designed to reduce potential impacts to the quality of groundwater. Diversion structures and ditches would be constructed upgradient to reduce surface water deposition to the TSF, and an underdrain system would be constructed below the TSF. The TSF would be constructed with a synthetic liner, and a seepage recovery system would be constructed downgradient from the TSF dam to which water from the underdrain would report. The liner would substantially reduce connectivity between the TSF and groundwater resources. Any leakage through the liner would be captured by the seepage recovery system (SRS) and pumped to the process plant as part of the process freshwater requirement, effectively isolating the TSF from groundwater resources during operations. Four monitoring/interceptor wells (MIWs) would be located downgradient of the SRS to identify and capture any potential leakage from the TSF or SRS pond into the groundwater system (Section 3.6.2.2.1, Groundwater Hydrology, Environmental Consequences). Therefore, operation of the TSF would not be expected to result in impacts to groundwater quality.

Given the sub-arctic climate at the project site, ice is expected to form on the operating pond during the winter months. In contrast, ice is not expected to form on the tailings beach due to the high temperature of the tailings when discharged. An unfavorable situation could potentially develop if the frozen operating pond inundates portions of the liner where the tailings beach does not entirely cover it (SRK 2017b). Frozen pond water in direct contact with the liner could lead to potential liner damage due to ice-loading. Based on the 50th percentile water balance model, this situation could potentially occur along about 1.5 to 6.4 miles of the impoundment perimeter about half the time (SRK 2016c). Damage could result from either horizontal ice movement due to wind loading or thermal expansion, or vertical ice loads due to rising or lowering pond levels during the winter. Chemical impacts to groundwater resources could occur if the integrity of the TSF liner is compromised due to interactions with TSF ice and the tailings water permeates the liner. To address this risk, the TSF would be operated such that the beach above water has a slope of 0.5 percent, and the beach below water (subaqueous) has a slope of 1.0 percent (BGC 2013). By keeping TSF water within the inner cone, ice loading issues would be minimized (SRK 2016c). Additional planning is included in Chapter 5, Impact Avoidance, Minimization, and Mitigation, to identify further contingencies to reduce these effects.

Permanent Camp Wells and Plant Potable Wells

During the Operations period, potable water for the permanent camp would come from four wells drilled at the permanent camp site located about 1.3 miles west of Crooked Creek (Figure 2.3-12). The groundwater quality in this area can be inferred from geology and groundwater quality trends regionally and at the Mine Site. As described in Section 3.6.2.2.1 (Groundwater Hydrology, Mine Site, Operations, Permanent Camp Water Use), the wells would likely draw water from fractured granodiorite bedrock. The geology of this aquifer is different than that of the Mine Site, which consists of sedimentary rocks with intruded hydrothermal veins (see Section 3.1.2.1.2 and Figure 3.1-4). The granodiorite lies outside of the highly mineralized area at the Mine Site, but within the same central Kuskokwim Mineral Belt that hosts other deposits such as those at Red Devil and Flat (Miller et al. 2008). Thus, the groundwater at the permanent camp wells is likely to have similar or better water quality than those of the existing bedrock wells at the Mine Site, which exceed drinking water standards for several inorganic constituents (Table 3.7-11, Section 3.7.2.2.1). Regardless, the water would pass through a potable WTP prior to distribution (SRK 2016a), and monitoring and testing would be conducted in accordance with the ADEC Drinking Water Program (ADEC 2017d). As such, impacts to drinking water quality from natural sources of elevated inorganics in the groundwater are expected to be minimal.

Several of the construction camp water supply wells would remain in use during Operations to supply potable water to the plant site; these would require about one-fifth the flow volume as the construction camp (Section 3.6.2.2.1, Groundwater Hydrology, Mine Site, Operations, Permanent Camp Water Use). Impacts to drinking water quality from potential wellhead/surface contaminant sources would be the same as described above under Construction, and are expected to be minimal due to BMPs and water treatment. The plant supply wells would lie upgradient or side-gradient of potential new sources of shallow groundwater contamination that may develop during Operations, such as drainage from the WRF or TSF (Figure 3.6-10). Flow directions in the area of the plant supply wells would remain largely unchanged from pre-mining conditions (Figures 3.6-2 and 3.6-10). As such, impacts to drinking water quality from mine contact water are not expected to occur.

Other Effects

Disruption, in-filling, and removal of wetlands in the American Creek, Snow Gulch, Omega Gulch, Anaconda Creek, and Crooked Creek watersheds would influence groundwater recharge and discharge patterns, which could affect groundwater quality in the vicinity of the Mine Site. Currently the alluvial groundwater environment is reducing, and concentrations of total and dissolved aluminum, iron, manganese, and arsenic are elevated as compared to EPA maximum contaminant levels and applicable ADEC water quality criteria for aluminum, iron, and manganese, as discussed in Section 3.7.1.2. Reducing conditions are prevalent due partly to deposition of organic carbon from wetlands and infiltration of organic carbon during spring thaw. The redox state of the alluvium is not expected to change during mine operations; however, concentrations of metals in shallow groundwater may increase due to disruption of wetlands and increased sedimentation (SRK 2012b).

Closure

Concurrent and final reclamation of WRF, TSF, remaining overburden stockpiles, and associated disturbed areas are designed to manage stormwater runoff and reduce infiltration. At mine closure, contact water would be managed and low permeability covers installed so that release of leachate from the waste rock and tailing facilities is controlled, although some infiltration and seepage through the unlined WRF and then to the pit lake is expected to continue in post-Closure.

When pit dewatering is stopped, water would flow from the pit into the bedrock depressurized by dewatering wells underlying the proposed project area; this would result in inputs of sulfate and metals, and decreased pH, to the deep bedrock portions of the aquifer (SRK 2017b; see also Section 3.6, Groundwater Hydrology). Groundwater flow model analysis indicates that, although net groundwater flow during the time the pit lake is filling is toward the lake, there would be flow of contaminated water from the pit to the depressurized bedrock aquifer for the full period of time that it takes for the pit lake to fill; this water would ultimately return to the pit. The flow rate to bedrock would be highest during the approximately 8 years following the cessation of pit dewatering. The extent of contact groundwater migration into the bedrock during this period would be localized, and is not expected to reach further than the pit rim. Overall hydraulic containment of this water would be maintained due to strong topographic gradients outside of the localized pit area. After the pit lake fills to its highest managed level, groundwater would flow back into the pit lake during the remainder of the period of closure and throughout the post-Closure period, provided that pit lake levels are properly managed to maintain a net inward groundwater gradient to the pit (BGC 2014c; SRK 2017b).

The potable water wells remaining in use at the plant site during Operations would be decommissioned at Closure in accordance with ADEC (2017e guidance). As such, impacts to groundwater quality from potential wellhead/surface sources of contamination are not expected to occur. The permanent camp and associated potable water wells would remain during Closure to support continuing reclamation and water treatment activities. Impacts to these wells in the Closure period would be the same as described above under Operations.

Impacts to groundwater quality in the event of SRS pump failure during Closure are related to potentially concurrent surface water impacts, and are described under Section 3.7.3.2.2 (Surface Water Quality, Mine Site, Closure).

The potential for dewatering wells, in-pit drains, and monitoring wells to act as conduits for contamination to enter the aquifer as the pit fills bears examination. Dewatering and monitoring wells would be abandoned, when no longer required for dewatering or post-closure monitoring. Abandonment would be conducted in accordance with ADEC regulations to prevent the transmission of potential surface contaminants to the subsurface, and to prevent cross-aquifer contamination from one vertical level in the well to another. Abandonment would include plugging the well with bentonite or cement grout from the total depth to about 50 feet above the top of the screened interval, placing inert material (cuttings or alluvium) above that depth inside the unperforated casing which has an annular seal, and placing a 10-foot cement plug in the top of the well (SRK 2017f). In-pit drains would be designed to flow towards the pit, and as such, are not expected to provide conduits to the aquifer outside the pit area from infiltration. Once saturated, the in-pit drains would not act as conduits to areas outside the pit, because they would lie within the area of hydraulic containment formed by the maintained lowered lake level.

Summary of Mine Site Impacts to Groundwater Quality

Due to effective water management during all project phases, impacts to groundwater quality would be limited to discrete portions of the project area. The principal mechanisms responsible for effects to groundwater quality at the Mine Site would be inputs of seepage from the WRF to shallow groundwater resources underneath and immediately adjacent to the WRF, and the discharge of water from the pit to the surrounding deep bedrock groundwater. Groundwater that would be contaminated by inputs of WRF seepage would flow towards the pit, and the spatial extent of the impacts would be limited because the contaminated groundwater would be intercepted by the pit and the pit dewatering system. The intensity of impacts to local groundwater resources would be sufficient to exceed water quality regulatory limits and would persist through the life of the mine. However, the intensity of groundwater impacts outside of the cone of depression would be the same as background levels, as groundwater outside of the cone of depression would be unaffected by mine contact water. The intensity of impacts resulting from temporary inputs of contaminated water from the pit to the deep bedrock groundwater following the cessation of pit depressurization would be sufficient to exceed water quality regulatory limits; concentrations of sulfate, antimony, arsenic and selenium would be expected to increase in the deep bedrock groundwater surrounding the pit during the early Closure period following cessation of pit depressurization. These effects would be localized within the extent of the pit rim, and overall hydraulic containment would be maintained during this period, due to strong topographic gradients from outside the pit area. Following that period, flow modeling indicates that groundwater would flow towards the pit lake radially from all directions (BGC 2014c), thereby limiting the extent of migration and ultimately recapturing the contaminated groundwater and restricting the extent or scope of the impact.

The TSF would not be expected to result in impacts to downgradient groundwater quality due to design features such as the liner, SRS, and MIWs. Impacts to drinking water well use are not expected to occur due to planned well siting, construction, and treatment design features that would take place under the oversight of ADEC source water assessment and drinking water protection programs.

Overall, the duration of direct impacts to groundwater resulting from Alternative 2 at the Mine Site would vary and may persist through the life of the mine. Groundwater quality would be reduced during the life of the project, and is expected to improve following reclamation of the WRF, but some seepage from this unlined facility would continue into post-Closure and would be captured by the pit, provided that the pit lake would be managed during the post-Closure period to ensure that the direction of groundwater flow is towards the pit from all directions. The intensity of local impacts to groundwater resources would be such that groundwater would not meet regulatory criteria at certain discrete locations within the Mine Site (e.g., in groundwater underlying the WRF during the operations period, and in deep bedrock groundwater in the near vicinity of the pit during the period following the cessation of pit depressurization activities). However, there would be no impacts to groundwater quality outside the mine facilities area above background levels. In terms of context, there is an abundance of groundwater in the Project Area, the groundwater impacted by Mine Site facilities are not sources of drinking water, the impacted groundwater is not expected to be released to the environment outside of the pit lake, and the resource is governed by regulation.

Transportation Corridor

Construction

The construction of transportation facilities at Dutch Harbor, Bethel (connected action), and Angyaruaq (Jungjuk) ports proposed under Alternative 2 are not expected to have measurable effects on groundwater quality. It is possible that placement of sheet pile associated with the construction of transportation terminals at Bethel and Angyaruaq (Jungjuk) ports would have localized impacts on the rates of exchange between surface water and shallow groundwater deposits along the banks of the Kuskokwim River (see Section 3.7.2.2.2). These impacts would be limited to the areas immediately adjacent to the terminal facilities, and the effects on the composition of the groundwater and availability for beneficial uses would be imperceptible. The use of groundwater for drinking water supplies at the Angyaruaq (Jungjuk) Port would not adversely affect groundwater quality.

Operations and Closure

Operations proposed under Alternative 2 are not expected to have measurable effects on groundwater quality. Closure would not be expected to result in any additional impacts.

Summary of Transportation Corridor Impacts

In terms of intensity, impacts to groundwater quality resulting from changes in the rates of surface water-groundwater interactions during all project phases would be below water quality regulatory limits. Water quality would be reduced infrequently but not longer than the span of port construction. The extent or scope would be limited to discrete portions of the project area. In terms of context, impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high water quality or water resources that are considered valuable in the region.

Pipeline

Construction and Operations

Potential impacts to groundwater quality during construction and operation of the natural gas pipeline are expected to be minimal (SRK 2013b). The primary impacts would be associated with installation of the pipeline, which could result in alterations to shallow groundwater flow patterns and small changes in the composition of the groundwater. Trench breakers composed of closed-cell polyurethane foam sprayed around the pipe would be installed in sloped areas to minimize groundwater flow and internal erosion along the trench (SRK 2013b). Once cured after several minutes, the foam material is chemically inert and does not absorb water; thus it has little to no potential to leach chemicals into groundwater. Foam trench breakers last indefinitely as long as they are not exposed to sunlight (ultraviolet rays).

In terms of intensity, groundwater quality would not be expected to exceed applicable regulatory criteria. Impacts would be focused on the construction phase, and groundwater flow (drainage and recharge) patterns should reestablish after site reclamation has been completed. The extent or scope of groundwater impacts would be limited to particular areas, primarily in the vicinity of river and stream crossings (SRK 2013b).

Closure

Groundwater drainage patterns should reestablish after site reclamation has been completed (SRK 2013b). Under Alternative 2, the pipeline would be abandoned in place; therefore, no additional impacts to groundwater quality are expected to be associated with the termination phase, except perhaps for the production and mobilization of minor corrosion products from the steel pipe sections. In terms of intensity, groundwater would be below applicable regulatory criteria.

Summary of Pipeline Impacts

In terms of intensity, natural gas pipeline construction activities described under Alternative 2 would result in impacts below water quality regulatory limits. Water quality would be reduced infrequently but not longer than the span of the construction. The extent or scope would be limited to discrete portions of the project area. In terms of context, impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high water quality or water resources that are considered valuable in the region.

3.7.3.2.4 SEDIMENT QUALITY

Mine Site

Construction

Under Alternative 2, construction of the Mine Site and associated facilities could result in increased erosion that could produce increased sediment deposition in Crooked Creek and tributaries if uncontrolled. Extensive sediment control measures would be identified in the SWPPP, which is required to be developed and implemented as part of the mine permitting process. Active sediment controls used during construction would include silt fences, sediment control basins, cross bars and ditches, revegetation, and BMPs to reduce the intensity of surface runoff and sediment loading (see Section 3.2, Soils). Specific ESC design features at the overburden stockpiles would include diversion channels, berms, and sedimentation ponds.

Operations

The primary sources of impacts to sediment quality at the Mine Site during Operations would involve increased mercury concentrations in sediments as a result of atmospheric deposition. In order to understand the effect of additional mercury deposition from atmospheric sources on sediment quality in the vicinity of the Mine Site, a conceptual site model was developed that identified the sources, potential exposure routes, and transport mechanisms of mercury in the environment (Environ 2015; ARCADIS 2014; ERM 2017d). Based on the conceptual site model, baseline data, and results of deposition modeling, incremental increases in sediment concentrations as a result of project air emissions were estimated for two areas: 1) watersheds within about a 10 mile radius of the Mine Site (ARCADIS 2014; SRK 2014a); and 2) watersheds within about a 20-mile radius of the Mine Site (ERM 2017d), which is the approximate limit of predicted effects from project-related atmospheric deposition (Figure 3.8-7).

Estimates for six watersheds located within about 10 miles of the Mine Site were developed for Year 27 of the mine life, since this represents the maximum cumulative deposition and potential impact (SRK 2014a). If it is assumed that all of the mercury deposited from the atmosphere immediately adheres to stream sediment at the sediment surface (a conservative assumption, especially in areas of highest deposition), then estimated sediment mercury increases would be as shown in Table 3.7-45. While the predicted total concentrations at 27 years exceed SQGs, they would still be within the range of naturally occurring values presently found in the mine area (Table 3.7-12). The greatest estimated increase of mercury in sediment occurs in the Crooked Creek watershed, and the lowest estimated increase occurs in the Village Creek watershed (ARCADIS 2014). The model results demonstrate an estimated 2.1 percent increase in sediment total mercury concentrations closest to the Donlin Camp, and a 0.2 percent increase at the Bell Creek watershed. As described in Section 3.7.3.2.2 (Surface Water Quality), total mercury in sediment is estimated to increase within a 20-mile radius of the Mine Site by 0.2 percent based on an area-weighted average of deposition over the wider area (ERM 2017d). While these increases are small, the increased concentrations of mercury in sediment may result in increases in the mercury content of fish, wildlife, and plants.

Table 3.7-45: Estimated Changes in Sediment Concentrations as a Result of Changes in Mercury Deposition Rates

Watershed	Modeled Total Hg Deposition[1] (µg/m^2-y)	Baseline Sediment Average Total Hg Concentration (µg/kg)	Modeled Sediment Total Hg Increase at 27 Years (µg/kg)	Predicted Average Total Hg Concentration at 27 Years (µg/kg)	% Increase at 27 Years
Crooked Creek HUC 12	4.7	202	4.2	206	2.1%
Donlin Creek HUC 12	2.3	173	2.2	175	1.2%
Grouse Creek HUC 12	1.7	236	1.6	238	0.7%
Flat Creek HUC 12	1.0	238	0.8	239	0.4%
Bell Creek HUC 12	0.5	205	0.5	206	0.2%
Village Creek HUC 12	0.4	43	0.3	43	0.7%

Notes:

1 Modeled deposition rates from stack and fugitive sources associated with the project (Environ 2015).
The threshold effects level (TEL) and probable effects level (PEL) for mercury in sediment are 174 µg/kg and 486 µg/kg respectively. Values that exceed the 174 µg/kg TEL are shaded in the table.
N/A means not available

Sources: ARCADIS 2014, Environ 2015, SRK 2014a.

Although methylmercury generally represents a very small fraction of the total mercury content in sediments, it is of interest due to its ability to cross membranes in living organisms, high lipid solubility, and relatively high toxicity compared to other forms of mercury. For example, essentially all mercury in freshwater fish tissues is in the form of methylmercury, even though methylmercury typically accounts for less than 1 percent of the total mercury pool in a lake

(ATSDR 1999). Therefore, methylation of mercury is a key step in the entrance of mercury into food chains (EPA 2010a).

Factors that drive mercury methylation are discussed in Section 3.7.3.2.2. In-river methylation is typically a negligible component of the methylmercury budget for creeks, whereas mercury methylation in wetlands may be an important contributor of methylmercury to downstream aquatic ecosystems (St. Louis et al. 1994; Berndt and Bavin 2012). The geochemical data collected in the aquatic systems in the study area generally indicate that stream sediments are neither a sink for mercury nor a source of methylmercury production (ARCADIS 2014). The potential for methylation in these environments is generally limited by the availability of nutrients required to sustain heterotrophic microbiological activity that drives the methylation of mercury in anoxic environments (ARCADIS 2014). In sediments in the vicinity of the Mine Site, average TOC concentration is low (1.7 percent), pH is neutral (7.1), and average sulfate and sulfide concentrations are relatively low (13 and less than 5 mg/kg, respectively) (ARCADIS 2014). A study by Wang (1999) reported sediment TOC concentrations ranging from 3.6 to 5.4 percent in samples collected from a single location in Crooked Creek near its confluence with the Kuskokwim River, suggesting that concentrations of TOC in Crooked Creek sediment may be higher at locations near the confluence with the Kuskokwim River relative to upgradient locations in the vicinity of the Mine Site.

Based on consideration of several factors that cause methylation, including an increase in sulfate deposition from Mine Site activities, ERM (2017d) estimated that methymercury concentrations in sediment would increase approximately 4 percent over the life of the mine within a 20-mile radius. Increasing the mean and 95% UCL baseline methylmercury concentrations within this area (which are 0.503 and 0.627 ug/kg, respectively) by 4 percent yields predicted methylmercury concentrations at the end of mine life of 0.52 and 0.65 ug/kg, respectively (ERM 2017d). While there are no sediment quality criteria specific to methylmercury, the predicted concentrations are within the range of naturally occurring values at the Mine Site (Figure 3.7-11). The predicted sediment methymercury concentrations were incorporated into an estimate of fish tissue concentrations, and compared to an EPA fish tissue-based criterion for the purpose of assessing impacts to human health by fish consumption. These results are summarized in Section 3.7.2.2.2, Surface Water Quality, and described in more detail in Section 3.22, Human Health.

As described in Section 3.6, Groundwater Hydrology, pit dewatering as part of the mine operations would result in a large area around the pit becoming dryer. As these areas dry out, the breakdown of organic material could increase due to oxidation of organic material within the cone of depression. After mine closure, the water table would rise again, and previously dry soils would become rewetted. This could result in a pulse of bioavailable carbon into the system which could stimulate mercury methylation. Methylation of mercury requires the simultaneous occurrence of several conditions including anoxia, availability of carbon and other nutrients, presence of mercury-methylating bacteria, and pH values amenable for mercury methylation. An increase in any one of the factors necessary for mercury methylation would not necessarily result in increased methylation rates in sediments.

The potential for mercury methylation and bioaccumulation in the FWDD and CWD reservoirs would be insubstantial due to the low concentrations of dissolved organic carbon to support heterotrophic bacterial activity in these reservoirs, and the lack of higher trophic level biota, which would be requisite for bioaccumulation.

Additional impacts to sediment quality in the Crooked Creek watershed could result from streamflow alteration and changes in the surface water chemistry in Crooked Creek as a result of the activities associated with Alternative 2. However, such impacts would be unlikely to exceed Sediment Quality Guidelines (SQGs) recommended by ADEC (Buchman 2008).

Closure

After mine closure, the majority of the facilities including the WRF, the SOB stockpile, and TSF would be reclaimed. Surface runoff from the top of the WRF, as well as water that seeps through the reclamation cover, would be conveyed to the then-closed pit. Runoff from the SOB stockpile would be directed to control basins to allow sediment in the runoff to settle out, and then would be discharged directly to Crooked Creek. As a result of careful water management during closure and post closure activities, direct impacts to sediment quality at the Mine Site are expected to be imperceptible during the Closure Phase.

Impacts to sediment quality from mercury and effects on mercury methylation would gradually decline during the post-Closure period. Mercury methylation rates in stream sediments would likely continue to be limited by the availability of nutrients and organic carbon in aquatic systems in the vicinity of the Mine Site, and would not necessarily be expected to result in substantial impacts to concentrations of mercury and methylmercury in sediments.

Summary of Mine Site Impacts

The extent or scope of impacts to sediment quality would affect only discrete portions of the Mine Site and nearby watersheds. Although effects to sediment quality may result from altered streamflows and water chemistry in Crooked Creek and project-related atmospheric deposition of mercury, these effects would result in concentrations of mercury and methylmercury that are predicted to remain within the range of naturally occurring values presently found in the study area. Increases in total mercury from project-related atmospheric deposition are estimated to range from about 2 percent near the Mine Site to 0.2 percent about 10 to 20 miles from the Mine Site. Increases in methymercury from deposition and increased methylation are estimated to be about 4 percent within a 20-mile radius of the Mine Site at the end of Operations. Snow removal from roads may also impact sediment quality. In terms of context, sediment can be considered an abundant resource in most drainages, but impacts to sediment may also influence other resource types (e.g., fish and other biological resources). While sediment in the Donlin Gold Project area is governed by neither ADEC nor EPA regulations, it is the subject of Sediment Quality Guidelines (SQGs) recommended by these agencies.

Transportation Corridor

The primary project-related mechanisms of impact to sediment quality in the transportation corridor under Alternative 2 would be associated with the construction of shoreside facilities at the Bethel (connected action) and Angyaruaq (Jungjuk) ports, and occasional sediment disturbance by propeller wash during low water periods in the Kuskokwim River. In-stream construction has the potential to contribute to increased sediment loads at downstream locations.

Construction

As discussed in Section 3.2.2.2.1, Soils, there is potential for erosion and resuspension of fine-grained sediments during barging and construction of both the Bethel and the Angyaruaq (Jungjuk) ports. Construction of the Bethel Port would involve placement of approximately 1,150 linear feet of sheet pile at an average depth of approximately 20 feet below mean lower low water (MLLW) along approximately 850 feet of the Kuskokwim River shoreline. Similarly, construction of the Angyaruaq (Jungjuk) Port would involve placement of sheet pile bulkhead in the Kuskokwim River. The sheet pile cells would be backfilled to create a dock with an area of approximately 5 acres. There is also the potential for localized sediment erosion at certain critical sections of the river during low water periods (see Section 3.5, Surface Water Hydrology). Under Alternative 2, fine-grained sediments eroded or resuspended as a result of construction activities and barging could contribute to increased sediment loads and deposition. Any actions that would occur at Dutch Harbor or the Port of Bethel at the Bethel Yard Dock are not part of the proposed action, and are considered connected actions (see Section 1.2.1, Connected Actions, in Chapter 1, Project Introduction and Purpose and Need).

The eroded sediments deposited would be similar in composition to the naturally extant sediments. Due to the natural dynamics of sediment transport in the Kuskokwim River, the duration of any effects to sediment quality resulting from the redistribution of sediments would not last longer than the span of the project construction. Effects would be limited to the time period of construction of the port facilities, and sediment quality would be expected to return to pre-activity conditions at the completion of the construction activity. The effects would be limited to a discrete portion of the project area and the affected sediments would be considered ordinary in context. Temporary bank stabilization and/or mitigation measures would be required along or near construction areas bordering the Kuskokwim River shoreline for SWPPP compliance during terminal construction. Permanent erosion control at the port sites would include the installation of sheet pile and construction of sheet pile bulkheads at both the Angyaruaq (Jungjuk) and Bethel ports.

In general, the mine access road and airstrip would be constructed using conventional cut-and-fill techniques. At intervals not appropriate for cut-and-fill construction, for example, where snow-drifting and permafrost are a concern, an elevated fill section would typically be employed. The currently defined routes can be constructed with conventional equipment and methods. No excessively large cuts or fills would be required. A final surfacing layer would be applied, consisting of crushed and/or screened material that is suitable as a surface maintenance and wear course. Stormwater BMPs would be implemented according to a SWPPP (Section 3.2, Soils) to mitigate potential erosion effects on downgradient sediment quality.

Operations and Closure

Impacts to sediment quality during operations would be less than those during construction at the ports, road, and airstrip, because BMPs and ESC measures employed to control erosion and sedimentation effects are expected to stabilize in operations (Section 3.2, Soils). The potential for barge-induced resuspension of bottom sediment in operations would be the same as described above for construction.

The mine access road, Angyaruaq (Jungjuk) Port, and airstrip are required for post-mining reclamation and closure activities, and would remain as long-term assets after the end of mining. Therefore the impacts to sediment quality associated with Transportation Corridor facilities during the post-Closure period would be similar to those during operations.

Summary of Transportation Corridor Impacts

In terms of intensity, effects on sediment quality resulting from the construction, operations, and closure of the Transportation Corridor facilities are expected to be below sediment quality guidelines and effects may not be noticeable (e.g., resettled sediment would be of similar composition to the existing natural deposit). Sediment quality would be reduced infrequently but not longer than the span of the project construction. The extent or scope of impacts would be limited to areas within the immediate vicinity of the road, ports, and critical sections of the river. In terms of context, impacts would affect areas that range from ordinary sediment quality or abundant sediment resources, to areas with high sediment quality or sediment resources that are considered valuable in the region.

Pipeline

Construction

Under any Alternative 2 option, the primary mechanisms of impact to sediment quality resulting from construction of the natural gas pipeline would involve soil erosion associated with installation of the pipeline at river and stream crossings, and clearing of riparian habitat along the pipeline ROW. Streambank alterations at pipeline crossings could result in increased water velocity, increased erosion both upstream and at the location of the pipeline crossing, and increased deposition of sediment at downstream locations. Along the pipeline ROW, reduced permeability and infiltration associated with vegetation removal and compaction of near-surface soils would result in increased surface runoff and sediment input compared to natural conditions. Additionally, removal of vegetation from streambanks could result in changes to streambank stability, runoff patterns, and temperature regimes, potentially resulting in effects to sediment quality.

The majority of rivers and streams along the pipeline route would be crossed by open cutting in the winter when flows are lowest and disturbance of the river and streambanks can be held to a minimum (SRK 2013b). In general, the open-cut method would be used for three different types of water bodies:

- Intermittent streams, ditches, and non-sensitive water bodies where sedimentation is not a key consideration;
- Frozen rivers or streams in winter that have no surface flow; and
- Streams and rivers so large that no isolation method can be used.

Cuts would be excavated using chain trenchers, wheel trenchers, or backhoes, and crews would be equipped with both backhoes and trenchers to provide the technique best suited for each soil type (SRK 2013b). Trench width would vary, depending on what piece of equipment is used for excavation, but generally, trench width would be approximately 18 to 30 inches wide to allow for placement of pipe and backfill without excavating more material than necessary. At river and stream crossings, trench width at the ground surface may be wider to accommodate the sloughing action of soils in high groundwater tables (SRK 2013b). If necessary, sedimentation at stream crossing locations would be controlled during trench dewatering with the use of sedimentation basins, geofabrics, and silt fences, which would limit the intensity of impacts to sediment quality at downstream locations. For larger rivers, the trench would be excavated through the water body using backhoes operating from the banks or within the water body (SRK 2013b). Equipment operating within active stream channels would compact substrate and resuspend sediment, resulting in increased sedimentation and changes to sediment quality at downstream locations. For wide, braided rivers, backhoe operators would utilize channel diversion techniques to create places from which equipment could operate outside of active stream channels (SRK 2013b). In terms of intensity, concentrations of constituents of concern in sediments would be unlikely to exceed chemical SQGs recommended by ADEC, and sediment grain size in depositional areas downstream of the ROW are likely to be similar to that of the construction area.

BMPs and ESC measures that would minimize inputs of silt and suspended sediment to active stream channels are described in SRK (2013b) and Section 3.2 (Soils–Erosion). Excavated material from intermittent streams, trenches, and non-sensitive water bodies would be placed at least 10 feet from the water's edge on the construction ROW and would be contained as necessary using silt fencing. For water bodies other than non-flowing streams or drainage ditches, trench plugs would be placed between the upland trench and the in-stream activities to prevent diversion of water into upland portions of the pipeline trench and to keep sediment-laden trench water out of the water body. In order to control potential impacts to sediment quality resulting from trenching and excavation activities during pipeline construction, excavated mineral soils would be stored separately from windrowed, organic spoil piles. Subsoils would not be stored in flowing water bodies, dry drainages, or washes that cross the ROW. Subsoil would be placed on the banks of the drainage in such a manner as to prevent sedimentation from occurring. Temporary erosion control measures including stormwater control measures, such as rolled erosion control product sediment barriers (e.g., brush barriers or silt fences) and water interception or diversion ditches, would be installed as needed to contain disturbed soils on the construction ROW and to minimize the potential for sediment to enter wetlands or water bodies. After installation, erosion control measures would be regularly inspected and maintained in effective operating condition throughout the duration of construction, until soil sediment stabilization is achieved and reclamation is complete.

Select rivers and streams would be crossed using HDD technology (SRK 2013b). A Drilling Mud Disposal Plan would be prepared as part of the overall HDD Practices, Contingency, and Resource Protection Plan prior to work taking place on any stream crossing where HDD would be used. For the purposes of this analysis, it is assumed that the plan would prohibit discharges of HDD drilling fluids to the environment, and would include appropriate provisions for the disposal of drilling wastes. With the exception of a potential frac-out into the overlying river (see Section 3.3, Geohazards and Seismic Conditions), HDD techniques are expected to result in imperceptible disturbance to stream beds and active stream channels. Stream crossings using HDD practices are not expected to experience measureable direct impacts to sediment quality.

One of the most important erosion and sedimentation control considerations is the timing of construction activities. The time between initial disturbances and post construction stabilization is a critical element in minimizing adverse impacts to sediment quality (Corps 2014b). Under Alternative 2, pipeline construction would occur in two spreads, which would be subdivided into various sections. Each section would be completed during one construction season (either summer or winter depending on the location of the specific spread). In order to control the potential for impacts to sediment quality, ground disturbances would be reclaimed concurrent with construction or as soon as conditions allow after construction is completed. In the interim, the areas would be cleaned and stabilized, and any necessary erosion control measures would be implemented (SRK 2013b). The short timeframes between initial disturbance and final stabilization would limit both the duration and the intensity of impacts to sediment quality associated with pipeline construction.

Summary of Construction Effects

Sediment quality would be adversely affected temporarily by pipeline construction as a result of increased sedimentation at stream crossing sites. Under all options, the proposed pipeline corridor crosses more than 400 individual streams ranging in size from small, intermittent channels to large glacially fed river systems (CH2M Hill 2011b). Sedimentation can occur from trenching to lay pipeline beneath the stream channel; runoff at construction sites; and erosion resulting from construction of culverts, roads, bridges, or fords associated with the pipeline ROW. Silt or sand resuspended as a result of construction activities could fill interstices in gravel and reduce water flow through substrate. In addition, resuspension of sediments could result in localized increases in biological oxygen demand, and increased bioavailability of nutrients. In most cases, sediment quality would return to pre-activity levels upon completion of the construction activities at a given site. In terms of intensity, measureable increases in runoff and sedimentation could occur as a result of pipeline construction activities, but the resulting impacts to sediment quality would not exceed regulatory limits, and would be minimized through winter construction and the use of BMPs and ESC measures. The impacts would affect only discrete portions of the project area, and are expected to remain within the immediate vicinity of the ROW. In terms of context, impacts would affect areas that range from ordinary sediment quality or abundant sediment resources, to areas with high sediment quality or sediment resources that are considered valuable in the region.

Operations and Closure

Expected impacts to sediment quality resulting from the normal operation of the pipeline under Alternative 2 would be imperceptible. In all cases, the pipe would be buried to a depth that is below the scour potential of the particular river or stream (Section 3.5, Surface Water Hydrology), such that operation of the pipeline would not result in measureable impacts to sediment quality (SRK 2013b). Increased runoff from cleared riparian areas could result in inputs of sediment that would affect sediment quality in streams and rivers along the pipeline route. However, timely reclamation of the ROW would involve methods to recontour and revegetate disturbed areas with native vegetation to maintain surface drainage patterns. The Surveillance and Monitoring Plan and ESC Plan would address maintenance of surface drainage patterns and monitoring. BMPs, mitigation measures, and the requirements of the SWPPP would also be used to mitigate impacts to sediment quality within the project area.

If the pipe is abandoned in place at project closure, as may be authorized by the Pipeline Abandonment Plan, any new impacts to sediment quality that could be caused by removal of the pipe would be avoided.

Summary of Pipeline Impacts

Under Alternative 2 options, some infrequent but measureable impacts to sediment quality would occur as a result of increased sediment loads to area streams during construction of the natural gas pipeline. The intensity of the impacts would be below sediment quality guidelines, assuming adherence to construction practices, design features, and BMPs specified in the Pipeline Plan of Development (SRK 2013b). The extent or scope of impacts would affect only discrete portions of the project area. In terms of context, impacts would affect areas that range from ordinary sediment quality or abundant sediment resources, to areas with high sediment quality or sediment resources that are considered valuable in the region.

3.7.3.2.5 CLIMATE CHANGE

Predicted overall changes in precipitation and patterns of runoff have the potential to influence the projected effects of the Donlin Gold Project on water quality. These effects are tied to changes in water resources as discussed in Section 3.26.4.2.2, Climate Change.

3.7.3.2.6 SUMMARY OF IMPACTS FOR ALTERNATIVE 2

Impacts to water and sediment quality under Alternative 2 are summarized below and in Table 3.7-46.

Mine Site. In terms of intensity, direct impacts to geochemistry, surface water quality, groundwater quality, and sediment quality at the Mine Site would range from impacts below applicable regulatory limits (e.g., water that meets AWQC and is eventually discharged) to impacts sufficient to exceed limits (e.g., water from the WRF and the isolated PAG cells, TSF, TSF cover, and possibly the overburden stockpiles that does not meet AWQC). Although potential impacts may increase with increased rock disturbance, real impacts to downgradient water quality may be reduced by mitigation efforts and material handling. Surface water from the pit lake would eventually be treated to meet AWQC and discharged. Additional mitigation measures are provided in Chapter 5, Impact Avoidance, Minimization, and Mitigation, that could help manage pit lake water quality efforts in post-Closure and reduce potential impacts from the overburden stockpiles.

Irreversible changes to surface water quality in the American Creek watershed would result from the creation and perpetual maintenance of the pit lake; however, due to water treatment and water management practices, contaminated water from the pit lake would not leave the American Creek watershed, and would therefore be restricted to a discrete portion of the project area.

Impacts to groundwater quality at the Mine Site may exceed regulatory limits as a result of activities conducted during the operations and closure periods; specifically, seepage from the WRF and flow of water from the pit lake into backfill and dewatered bedrock void spaces in the immediate vicinity of the pit lake would contaminate the local groundwater system; however, overall hydraulic containment would be maintained and this water would subsequently flow back into the pit lake for eventual treatment and discharge after meeting AWQC standards. Impacts to groundwater quality outside of the mine facilities area would be mostly below applicable regulatory criteria. There is a low risk that impacts to surface and/or groundwater could exceed regulatory limits in the event of SRS pump failure in post-Closure. Impacts to drinking water from the construction camp and permanent camp wells (from mine contact water, natural sources of elevated inorganics, or other wellhead/surface sources) are either not expected to occur due to siting upgradient or side-gradient from potential sources (e.g., WRF), or would be minimized through waste management BMPs and potable water treatment and testing.

The impacts to sediment quality of greatest concern would be associated with increased concentrations of mercury and methylmercury in sediments in the Crooked Creek watershed resulting from atmospheric deposition of mercury released by mine site facilities. However, detailed modeling has demonstrated that concentrations of mercury in sediments would increase by a maximum of 2.5 percent over existing levels (SRK 2014a). Therefore, the increased concentrations of mercury in sediment, while measurable, would be consistent with regional background concentrations, and would be unlikely to exceed regulatory guidelines for mercury concentrations in sediments. Such impacts would generally decrease with increasing distance from the mine facilities. The extent or scope of impacts would be limited to discrete portions of the project area.

The duration of impacts to surface water quality would be such that elevated mercury concentrations in surface water and sediments could potentially last through operations, and may not be anticipated to return to previous levels (e.g., pit lake). However, it is anticipated that mercury concentration levels would slowly return to pre-activity levels following completion of the project. The extent or scope of most water quality impacts would be limited to the immediate vicinity of specific project components. Impacts to surface water quality have the potential for mercury deposition from atmospheric sources to occur inside and outside of the immediate project area. The context of impacts would range from abundant water resources in the project area, to water quality released into the environment that is governed by regulation.

Transportation Corridor. For the Transportation Corridor facilities associated with Alternative 2, effects to surface water quality would be below regulatory limits (e.g., occasional barge–induced suspended sediment, or erosion effects at construction sites), due to high natural conditions and/or planned mitigation measures designed to limit erosion effects. The geochemistry of rocks disturbed by mining would not impact water quality in the Transportation Corridor areas. Potential impacts resulting from runoff of water from rock obtained from material sites along the road alignment and used for road construction could include inputs of arsenic, selenium, antimony, and possibly other constituents of concern, to area surface water resources, including Jungjuk Creek. The extent or scope of such impacts would be limited to areas in the immediate vicinity of roads and road construction material stockpiles, and would be mitigated through implementation of a materials testing program prior to the beginning of road construction activities. The duration of such impacts would persist for the duration of the project, and water quality would return to baseline levels at some time following the cessation of mining operations. Impacts to groundwater quality resulting from changes in the rates of surface water-groundwater interactions would be below applicable groundwater cleanup standards. Any effects on sediment quality resulting from Transportation Corridor facilities would be of similar composition to the existing natural deposit, and would be below applicable sediment quality guidelines. The duration of effects would be infrequent and limited to areas within the immediate vicinity of the road, ports, and critical sections of the river. In terms of context, impacts would affect areas that range from ordinary/sediment quality or abundant water/sediment resources, to areas with high water/sediment quality or water/sediment resources that are considered valuable in the region.

Pipeline. The primary impacts to surface water quality resulting from the natural gas pipeline as proposed under Alternative 2 would be associated with erosion and the introduction of fine-grained sediments (silts and clays) to surface water. The intensity of these impacts would be such that surface water quality would meet applicable regulatory standards due to planned BMPs and ESC measures. Some infrequent but measureable sediment quality impacts would also occur as a result of increased sediment loads to area streams during construction of the pipeline. Wetlands disturbed during construction would be restored shortly after installation of the pipeline. Following restoration and revegetation, few persistent effects on water quality would be expected to result from wetlands disturbance associated with the pipeline facilities. The geochemistry of rocks disturbed by mining would not impact water quality in the pipeline corridor. The extent or scope of impacts would be limited to discrete portions along the pipeline corridor. In terms of context, surface water, groundwater, and sediment resources are abundant in the project area, but are governed by regulation.

Table 3.7-46: Summary of Impacts[1] to Water Quality for Alternative 2

Impacts	Assessment Criteria			
	Magnitude or Intensity	**Duration**	**Extent or Scope**	**Context**
Mine Site				
Surface Water Quality	Treated water discharge to Crooked Creek would be below or meet AWQC. Atmospheric mercury deposition could result in AWQC exceedance, depending on watershed location and existing baseline mercury concentration.	Treated pit water quality would be reduced and would not be anticipated to return to previous levels or would take longer than 100 years to do so. Impacts to mercury concentrations in surface water would persist through the life of the mine, but water quality would be expected to return to pre-activity levels at some time after the completion of the project.	Impacts may be inside or outside of the immediate project area due to mercury deposition from atmospheric sources.	Impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high quality or resources that are considered valuable in the region. Water quality is governed by regulation.
Groundwater Quality	Effects would be below water quality regulatory limits or background levels outside the cone of depression. Effects would be sufficient to exceed water quality regulatory limits at locations within the Mine Site.	Impacts within the Mine Site would vary in duration, and would range from throughout the life of the mine, to groundwater quality impacts that would not be anticipated to return to previous levels or would take longer than 100 years to do so.	Effects on groundwater quality within discrete portions of the project area. Impacts would be hydraulically contained from moving downgradient or throughout aquifers.	Same as above.
Sediment Quality	Potential increases would be minimal as deposition concentrations would be consistent with regional background levels.	Impacts to sediment quality would persist through the life of the mine, but sediment quality would be expected to return to pre-activity levels at some time after the completion of the project.	Same as above.	Same as above.
Transportation Corridor				
Surface Water Quality	Barge-induced suspended sediment would be similar to high natural conditions	Impacts to surface water quality would persist through the life of the mine, but would be expected to return to pre-activity levels at some time after the completion of the project.	Same as above.	Same as above.

Table 3.7-46: Summary of Impacts[1] to Water Quality for Alternative 2

Impacts	Assessment Criteria			
	Magnitude or Intensity	**Duration**	**Extent or Scope**	**Context**
Groundwater Quality	Effects would be below applicable water quality criteria.	Impacts would be infrequent but not longer than the span of the construction, and would be expected to return to pre-activity levels at the completion of the activity.	Same as above.	Same as above.
Sediment Quality	Sediment deposition would be similar to natural material.	Same as above.	Same as above.	Same as above.
Pipeline				
Surface Water Quality	Effects would be below water quality regulatory limits, and increased turbidity would be controlled by appropriate BMPs.	Impacts to surface water quality would occur primarily during the construction phase, would be expected to return to pre-activity levels at some time after the completion of the activity.	Same as above. Approximately 10 more small streams impacted under North Option.	Same as above.
Groundwater Quality	Effects would be below water quality regulatory limits (e.g. from pipe erosion).	Impacts would be infrequent but not longer than the span of the construction and would be expected to return to pre-activity levels at the completion of the activity.	Same as above.	Same as above.
Sediment Quality	Effects would be below sediment quality guidelines and increased sedimentation would be controlled by appropriate BMPs.	Same as above.	Same as above. Approximately 10 more small streams impacted under North Option.	Same as above.

Notes:

1 The expected impacts account for impact-reducing design features proposed by Donlin Gold and Standard Permit Conditions and BMPs that would be required. They do not account for additional mitigation measures being considered.

3.7.3.2.7 MITIGATION AND MONITORING FOR ALTERNATIVE 2

These effects determinations take into account impact-reducing design features proposed by Donlin Gold (Table 5.2-1 in Chapter 5, Impact Avoidance, Minimization, and Mitigation), as well as Standard Permit Conditions and BMPs that would be implemented (Section 5.3). Design features important for reducing impacts to water quality include the following:

- Material sites at the Mine Site, mine access road, and pipeline would be evaluated prior to use for metals leaching and acid rock drainage potential in final design using bulk geochemistry analysis, meteoric water mobility procedure, and acid-base accounting methods. Such testing would also be conducted on waste rock intended for reuse in mine reclamation. Alternative materials would be selected if results indicate the potential for impacts to downgradient water resources.
- Monitoring activities of the WRF and tailings dam would be conducted to include water quality, biological resources, and vegetation. In addition, all dams on site would be monitored for mass stability to detect potential movement;
- The TSF will include a relatively flexible, textured geomembrane liner (60 mil or 1.5 mm) combined with an underdrain that is expected to withstand freezing temperatures, sharp rocks, and anticipated settlement scenarios, and a seepage recovery system (SRS) pond to minimize impacts from porewater seepage on groundwater quality;
- Numerous locations and combinations of locations were analyzed for TSF and WRF layouts during the alternatives development process. These are summarized in Appendix C. The layout of major mine facilities was designed to minimize wetland impacts and limit effects on water quality to the American and Anaconda Creek watersheds;
- Water management planning at the Mine Site would assist in controlling the flow of groundwater at the pit and other major facilities (WRF, TSF), as well as controlling the potential effects of groundwater flow on water quality downgradient of the mine. This would be accomplished through design elements such as dewatering wells, collection of groundwater infiltration through and around the TSF at the SRS pond, and lake level maintenance following closure. A variety of groundwater monitoring activities would also be planned (e.g., SRK 2016h);
- The project design at the Mine Site includes water management strategies that would maintain flow and storage within the design capacity of structures, provide flexibility for extra storage in high precipitation years, provide sufficient water supplies for processing in low precipitation years, and minimize storage if not needed through water treatment and discharge;
- Contact water would be collected and reused or treated and discharged. As applicable to the TSF, this process of minimizing water content of the TSF facility is aligned with specific recommendations of the Mount Polley Independent Review Board for application of Best Available Technology (BAT) and Best Applicable Practice (BAP) in TSF design and management;

- The project design includes maintenance of sufficient freeboard at the pit lake in post-Closure with several years of lead time to address pumping failures in order to prevent overflow to Crooked Creek;
- During tailings consolidation in closure, excess porewater would be captured in a capillary rock layer over the TSF, report to the lined pond at the southeast corner of the TSF, and pumped along with runoff water via pipeline to the pit until Water Quality Standards (WQS) have been met. This process of minimizing water content of the TSF facility is aligned with specific recommendations of the Mount Polley Independent Review Board for application of BAT and BAP in TSF design and management. Final closure details would be described in an updated reclamation and closure plan and would be subject to approval by ADNR;
- A detailed Mercury Management Plan would be developed that describes mercury control systems, storage areas, inspections, training, hazard communication, and procedures for off-site transport and disposal (Donlin Gold 2015d). Implementation of this plan would minimize the potential for release of mercury to the environment through normal ancillary activities;
- The Alaska Pollutant Discharge Elimination (APDES) five-year permit would be reevaluated as required, including water flow models and/or pit lake modeling as appropriate; the adequacy of post-Closure WTP technology would be reevaluated as pit lake water monitoring is conducted; and treatment technologies would be adjusted as necessary as a result of these evaluations;
- Regular inspections and maintenance of the SRS would be performed, and specific contingency/back-up plans would be in place, so that if failure of the SRS were to occur, the situation would be identified and response actions begun immediately;
- Construction camp and plant potable wells are sited on a groundwater interfluve (ridge) upgradient or side-gradient of potential future contaminant sources of mine contact water in American and Anaconda Creek drainages;
- Monitoring of bank erosion immediately upstream and downstream of Angyaruaq (Jungjuk) port would continue, with measures applied, as warranted, for streambank protection as part of adaptive management (as a Standard Operating Procedure). If warranted, this may include installation of geotextile matting, riprap armoring or methods from the ADF&G Streambank Revegetation and Protection Manual (Walter et al. 2005), such as willow staking, to reduce the effects of eddy formation, scour, and bank erosion during flood events (BGC 2014e);
- The project design includes use of BMPs at pipeline stream crossings (such as wattles, silt fence, brush berms, erosion control matting, etc.) to minimize alterations of the stream bed and bank erosion. It also includes design of pipeline depth of burial at stream crossings to avoid scour exposure of the pipe; and
- The project design includes installation of pipeline components (temporary roads and pipelines) at most water bodies and wetlands primarily in the winter months when frozen ground and snow are present, flows are lowest, and disturbance of the river, stream banks, and local groundwater would be minimized, or by using HDD technology to avoid flow impacts at major pipeline river crossings.

Standard Permit Conditions and BMPs related to water quality include:

- Potable well siting, construction, treatment, monitoring, and decommissioning in accordance with ADEC source water assessment and drinking water protection programs; and use of waste management BMPs under RCRA and ADNR solid waste programs (SRK 2016b) to minimize potential wellhead sources of contamination to drinking water wells;
- Implementation of SWPPPs and/or erosion and sediment control plans and use of industry standard BMPs for sediment and erosion control;
- Preparation and implementation of a Reclamation and Closure Plan (SRK 2017f);
- Use of BMPs such as revegetation planning, watering and use of dust suppressants to control fugitive dust; and
- Monitoring Mine Site facilities and associated surface water and groundwater, water in Crooked Creek, and discharge water from WTPs during all project phases; as established in State of Alaska permits to ensure the proper reclamation is completed for the protection of aquatic resources in Crooked Creek.

Additional mitigation, monitoring and adaptive management measures are being considered by the Corps and Cooperating agencies to further minimize project impacts, as reasonable and practicable, and are further assessed in Chapter 5 (Section 5.5 and Section 5.7). Examples of additional measures being considered that are related to water quality include the following:

- Complete a model run for the pit lake during post-Closure to confirm that containment will occur in the winter when there will be no pumping;
- Conduct an electrical leak detection survey of the TSF liner, using methods appropriate for the geomembrane type (TRI Environmental 2014), perform repairs prior to tailings placement, and update liner defect assumption in future WBM updates based on survey results and actual SRS flow and water quality data;
- Develop a sampling and analysis plan to ensure PAG rock and other sources of contaminants are not used for construction at the mine or for road surfacing (i.e., where such construction could lead to surface water quality impacts);
- Include detailed contingencies to mitigate the risk of ice damage and liner leakage in the TSF in an updated tailings deposition plan during final design;
- Reclaim lower portions of Snow and Ruby gulches, which have been disturbed by placer mining, to provide stable habitats for fish passage and shallow productive rearing (Owl Ridge 2017c; Donlin Gold 2018a);
- Include additional erosion and sediment control measures such as settling ponds, silt fences, or sediment barriers to minimize the amount of sedimentation from snowmelt;
- Implement pertinent BAP recommendations from the Mount Polley review panel for the tailings storage facility dam, design and tailings management, including participation in formalized tailings management program with audit functions, declaration of Quantitative Performance Objectives (QPOs) for tailings facility design and management, and use of independent tailings review boards;

- Apply one of the following to the South Overburden Stockpile (SOB); in either case, install downgradient monitoring wells, equip the sediment pond with redundant and freeze-protected pumping systems, and excavate and properly dispose of sediment at Closure:
  - Hydraulic containment (deep sump as part of sediment pond). Feasibility of digging a deep sump should be evaluated during design work; or
  - Additional studies during design work (fate and transport groundwater modeling) to demonstrate a lack of substantial groundwater volume that would result in no serious impact on the creek, as a result of natural attenuation of a small temporary slug of contaminated groundwater;
- Implement measures to minimize the risk of spills, such as avoiding operation of watercraft in fall and winter and in the presence of sea ice to the extent practicable, use of automatic identification system (AIS) for vessel tracking, and for barges transiting across the Cook Inlet maintaining a safe distance from the mean lower low water line of the Susitna Delta;
- Add an upstream monitoring site on Donlin Creek as a control point for monitoring water quality and discharge to enhance understanding of dewatering impacts on Crooked Creek habitat (monitoring site DCBO was specifically suggested as a location for background monitoring);
- Reexamine the groundwater flow model sooner than required by typical permit reevaluations, e.g., 3 years after the commencement of pit dewatering, to evaluate unexpected conditions (including impacts from faults and effects on WTP capacity), minimize uncertainty in the model, update and recalibrate the model as more groundwater level data are available, revise projections, and adjust management plans as needed;
- Conduct a reevaluation of the groundwater model and sensitivity analysis of potential contaminant migration from the pit lake after Year 15 of mining, when the ACMA pit is within a few 100 feet of its maximum depth;
- Collect relevant geotechnical and groundwater data (such as dewatering well testing, production rates, fault information, and water table levels around the pit) as mining progresses to refine interpretations and facilitate model revisions;
- Expand monitoring plans and data evaluation details to describe the proposed approach to facilitate comparisons with baseline data, and how it will be determined that water quality standards have been met and management activities can/should change. Baseline data should be evaluated using non-statistical means, such as spatial and temporal distribution, to allow a range of interpretive assessments;
- Conduct water quality monitoring during Operations in the sedimentation ponds downgradient of the North and South overburden stockpiles, as well as in Lewis Gulch for the North overburden pile. Monitoring results would form the basis for additional adaptive management measures (such as increased pumping or pond size) to reduce potential water quality effects;

- Include additional alluvial and/or bedrock groundwater monitoring wells at locations downgradient of mine facilities not already covered by the planned monitoring network (Figure 2.3-38, SRK 2016h) (e.g., overburden stockpiles), where sufficient alluvial aquifer material is present that could represent a pathway for contaminant migration to Crooked Creek, and bedrock groundwater is not captured by the pit cone of depression;

- Monitor drainages from the low-grade ore stockpile (which contains PAG 7 waste rock), the non-acid generating (NAG) WRF, and the isolated PAG cells within the WRF to allow evaluation of the effects of stockpile drainage and seasonal variability on CWD source water to the Operations WTP, and on pit lake stratification in Closure;

- Consider preferential processing of the PAG 7 material in the low-grade ore stockpile, if monitoring of WTP inflows and stockpile drainage indicates the need for improving CWD water quality;

- Conduct semi-annual pit lake monitoring (after spring melt and in late summer) for collection of in-situ temperature and total dissolved solids (TDS) (or the related measurement of specific conductance) at 20-foot depth intervals from the surface through the pycnocline, and then every 100 feet to the bottom, noting the depth of the discharge end of the pipe from the TSF and WRF. The goal would be to evaluate whether the pit lake is stratified; whether the water being delivered at depth is aiding or detracting from optimal pit lake stratification; and whether water from the TSF, WRF, or other source would improve stratification and long-term surface water quality (source water being treated in the Closure WTP), as well as minimize treatment cost, if delivered to an alternate depth;

- Rerun the pit lake model at regular intervals using the latest groundwater modeling results to predict the estimated duration of the pycnocline, the estimated source water quality going to the Closure WTP, and evaluate whether groundwater and reclaimed WRF runoff and seepage water delivered below the pycnocline would affect these changes;

- Based on pit lake monitoring and modeling results in post-Closure, 1) piped water from the reclaimed WRF and any other controlled water inputs to the lake should be delivered to the pit lake at such a depth so as to maximize the vertical salinity gradient; and 2) treating and discharging water from the bottom layer in the pit (instead of the top layer) should be considered to minimize adverse impacts to water quality at the surface of the lake;

- If substantially more dewatering water needs to be treated in Operations than the current WTP design basis allows, apply adaptive management measures such as extending the treatment season beyond April-November; storing more water in the TSF which would have excess capacity, reducing inflow from faults/fractures through grouting or sealing, and/or expanding the WTP which would take about 2 years;

- Develop a Pit Lake Groundwater Sampling and Monitoring Plan to focus on long-term water quality monitoring, sampling, and testing of the groundwater around the pit for the presence, abundance, and migration of contaminants such as mercury and arsenic;

- Based on performance of the Seepage Recovery System (SRS) in Operations, add an additional well field and/or pond that acts as a secondary containment system and/or supplemental storage to the SRS downgradient of the SRS. This measure may minimize the likelihood of an extended pumping failure in Alternatives 2 and 5A, if determined to be an issue through adaptive management;
- Perform testing of the SRS monitoring/pumping wells periodically throughout Operations and Closure to demonstrate that adequate hydraulic containment of TSF seepage is occurring;
- Monitor containment ponds and dams more frequently than quarterly or annually.
- Monitor the TSF cap for potential breaching by vegetation or animals and potential damage to the integrity of the cap;
- Construct one monitoring well to a depth equal to or deeper than the lowest elevation of the pit bottom, on the southwest side of the pit rim between Crooked Creek and the pit, prior to any pumping to dewater the pit. The primary purpose is to measure hydraulic head at the bottom of the hole and to confirm model predictions that water from the pit lake would not leak into a regional groundwater flow system. The well should be completed as water quality sampling well and incorporated into the groundwater monitoring program for the project in order to verify continuing protection of deep groundwater resources by the process of hydraulic containment through mining and post-mining periods. The well should be drilled at an elevation above Crooked Creek floodplain, if possible, to avoid having the well exhibit flowing artesian conditions;
- Extend pit lake pumping and treatment into winter months if necessary to maintain managed lake level, based on monitoring of lake and groundwater levels; and
- Monitor riparian crossing sites to identify areas that need additional restoration to prevent bank erosion which would be implemented after construction.

3.7.3.3 ALTERNATIVE 3A – REDUCED DIESEL BARGING: LNG-POWERED HAUL TRUCKS

3.7.3.3.1 GEOCHEMISTRY

Geochemical direct and indirect cumulative impacts under Alternative 3A would be the same as discussed for Alternative 2, as the Mine Site and associated facility footprints would be the same.

3.7.3.3.2 SURFACE WATER QUALITY

Mine Site

Alternative 3A would involve the construction of an LNG plant, LNG tanks, and LNG-fueling infrastructure for haul trucks at the Mine Site. These facilities would be located within the same facilities footprint proposed under Alternative 2, and are not expected to result in any changes in impacts to surface water quality at the Mine Site during construction, operations, or closure relative to Alternative 2. Potential impacts resulting from metals leaching from NAG used for construction of mine facilities would also be the same as those described under Alternative 2. The same design features and mitigation measures would apply to Alternative 3A as were discussed under Alternative 2.

Transportation Corridor

Alternative 3A would reduce peak annual Donlin Gold Project related barge traffic, both fuel and cargo, on the Kuskokwim River from an estimated 122 round trips to 83 (approximately 1.1 round trips per day to approximately 0.8 round trips per day). The decrease in barging activity would presumably reduce the risk for impacts to surface water quality resulting from spills. In addition, potential impacts to water quality in the Kuskokwim River resulting from increases in suspended sediment concentrations and turbidity at some shallow water locations would decrease by approximately 32 percent under Alternative 3A relative to Alternative 2, assuming a linear relationship between the number of barge trips and the potential impacts to water quality resulting from barging. Under Alternative 3A, the impacts to surface water quality in the Kuskokwim River resulting from normal barging operations would be below applicable water quality regulatory limits. The extent or scope of impacts would be limited to discrete portions of the project area. Such impacts would not last longer than the span of project construction, and would be expected to return to pre-activity levels at the completion of the activity. Impacts would affect areas of ordinary water quality or where there is an abundance of water resources.

Pipeline

Impacts to surface water quality associated with the pipeline component of Alternative 3A would be the same as discussed under Alternative 2. The primary impacts to surface water quality resulting from the installation, operations, and closure of the natural gas pipeline would be associated with erosion and the introduction of fine-grained sediments (silts and clays) to surface water. The intensity, duration, and extent or scope of impacts would be the same as described above for the Transportation Corridor. Impacts would affect areas that range from ordinary water quality or abundant water resources, to areas with high quality or resources that are considered valuable in the region. Water quality is governed by regulation.

3.7.3.3.3 GROUNDWATER QUALITY

Impacts to groundwater quality associated with the Mine Site, Transportation Corridor, and Pipeline components of Alternative 3A would be the same as discussed under Alternative 2. The reduced barging and operation of LNG-powered rock trucks at the Mine Site associated with Alternative 3A would not result in changes to any of the principal mechanisms of impact to groundwater quality at the Mine Site that were described under Alternative 2. The use of LNG-powered rock trucks instead of diesel-powered rock trucks would decrease the potential for spills associated with diesel fuel handling. However, under Alternative 3A, the WRF and pit lake, which are the facilities responsible for the principal mechanisms of impact to groundwater quality, would be constructed, operated, and closed in a manner identical to that described under Alternative 2.

3.7.3.3.4 SEDIMENT QUALITY

The types of sediment quality impacts and mitigative measures under Alternative 3A are expected to be the same as those described under Alternative 2 for the Mine Site, Transportation Corridor, and Pipeline. While less upland soils and riverbank areas would be subject to erosion at the transportation facilities under Alternative 3A, these areas are small compared to the project as a whole. The intensity, duration, extent, and context of impacts for sediment quality at remaining project components would be the same as described for Alternative 2.

3.7.3.3.5 SUMMARY OF IMPACTS FOR ALTERNATIVE 3A

Alternative 3A would involve the construction of an LNG plant, LNG tanks, and LNG-fueling infrastructure for haul trucks at the Mine Site. The Mine Site and Pipeline components would be located within the same facilities footprint proposed under Alternative 2, and are not expected to result in any changes in impacts to water quality relative to Alternative 2. The WRF and pit lake, which are the facilities responsible for the principal mechanisms of impact to groundwater quality, would be constructed, operated, and closed in a manner identical to that described under Alternative 2. Overall, impacts to water quality associated with the Mine Site and Pipeline would be the same as discussed under Alternative 2.

Effects on surface water quality, groundwater quality, and sediment quality under Alternative 3A associated with Transportation Corridor facilities would also be the same as those described for Alternative 2. The reduced barging and operation of LNG-powered haul trucks at the Mine Site associated with Alternative 3A would not result in changes to any of the principal mechanisms of impact to groundwater quality at the Mine Site that were described under Alternative 2. The use of LNG-powered haul trucks instead of diesel-powered trucks would decrease the potential for spills associated with diesel fuel handling. While less upland soils and riverbank areas would be subject to erosion at the transportation facilities under Alternative 3A, these areas are small compared to the project as a whole. In addition, potential impacts to water quality in the Kuskokwim River resulting from increases in suspended sediment concentrations and turbidity at some shallow water locations would decrease by approximately one-third relative to Alternative 2. Impacts associated with climate change would be the same as those discussed for Alternative 2. Overall, impacts to water quality associated with the transportation facilities would be the same as discussed under Alternative 2.

Design features, Standard Permit Conditions, and BMPs related to water quality are described under Alternative 2. Examples of additional measures being considered that are applicable to this resource are listed under Alternative 2.

3.7.3.4 ALTERNATIVE 3B – REDUCED DIESEL BARGING: DIESEL PIPELINE

Two options to Alternative 3B have been added based on Draft EIS comments from agencies and the public:

- Port MacKenzie Option: The Port MacKenzie Option would utilize the existing Port MacKenzie facility to receive and unload diesel tankers instead of the Tyonek facility considered under Alternative 3B. A pumping station and tank farm of similar size to the Tyonek conceptual design would be provided at Port MacKenzie. A pipeline would extend northwest from Port MacKenzie, route around the Susitna Flats State Game Refuge, cross the Little Susitna and Susitna rivers, and connect with the Alternative 3B alignment at approximately MP 28. In this option, there would be no improvements to the existing Tyonek dock; a pumping station and tank farm would not be constructed near Tyonek; and the pipeline from the Tyonek tank farm considered under Alternative 3B to MP 28 would not be constructed.

- Collocated Natural Gas and Diesel Pipeline Option: The Collocated Natural Gas and Diesel Pipeline Option (Collocated Pipeline Option) would add the 14-inch-diameter natural gas pipeline proposed under Alternative 2 to Alternative 3B. Under this option, the power plant would operate primarily on natural gas instead of diesel as proposed under Alternative 3B. The diesel pipeline would deliver the diesel that would be supplied using river barges under Alternative 2 and because it would not be supplying the power plant, could be reduced to an 8-inch-diameter pipeline. The two pipelines would be constructed in a single trench that would be slightly wider than proposed under either Alternative 2 or Alternative 3B and the work space would be five feet wider. The permanent pipeline ROW would be approximately two feet wider. This option could be configured with either the Tyonek or Port MacKenzie dock options.

3.7.3.4.1 GEOCHEMISTRY

Geochemical direct, indirect, and cumulative impacts under Alternative 3B would be the same as discussed for Alternative 2, as the Mine Site and associated facility footprints would be the same.

3.7.3.4.2 SURFACE WATER QUALITY

Mine Site

Under all options of Alternative 3B, the mine site infrastructure would be essentially the same as required under Alternative 2, with the exception of decreased capacity of diesel fuel storage tanks. Under Alternative 3B, approximately 10 Mgal of diesel would be stored at the Mine Site, compared to approximately 37.5 Mgal of diesel fuel storage capacity proposed under Alternative 2. The decreased requirement for diesel storage at the Mine Site is not expected to affect any changes to surface water quality at the Mine Site during the construction period relative to Alternative 2.

During mine operations, approximately 120 Mgal of diesel would be consumed at the Mine Site annually, compared to approximately 42.3 Mgal of diesel and 11.2 Bft^3 of natural gas consumed annually under Alternative 2. The additional use of diesel proposed under Alternative 3B would result in increased potential of adverse impacts to surface water resources resulting from diesel fuel spills; however, under normal operating conditions (i.e., no spills) the impacts to surface water quality at the Mine Site would be the same as those proposed under Alternative 2.

Transportation Corridor

Under Alternative 3B, with the exception of the Point MacKenzie Option, the existing dock at the Tyonek North Foreland Facility would need to be extended approximately 1,500 feet from shore in order to accommodate diesel tank vessels. Construction of the extended dock would presumably require some in-water construction and/or pile driving activities in Cook Inlet, which could potentially result in the resuspension of sediments and increased turbidity in the immediate vicinity of the in-water construction and/or pile driving activities. The intensity of impacts to surface water quality in upper Cook Inlet resulting from the construction of transportation facilities proposed under Alternative 3B would be such that marine water quality standards are likely to be met. The most stringent applicable water quality criterion for turbidity in marine waters specifies that turbidity may not exceed 25 NTU. Any impacts resulting from resuspension of sediments and increased turbidity near in-water construction and/or pile driving activities would cease immediately following the completion of the construction activities. High tidal current velocities and efficient water replacement in Cook Inlet due to tidal exchange would ensure rapid dissipation of local impacts to surface water quality resulting from the construction of transportation facilities in Cook Inlet. In terms of context, the activities would impact less than one percent of the water volume of Cook Inlet, and marine water quality is governed by ADEC regulations.

Increased operations at the Tyonek North Foreland Facility or at Port MacKenzie under the Port MacKenzie Option would increase the risk of spills associated with fuel handling. However, this increased potential for impacts to surface water quality in upper Cook Inlet would be offset by a decrease in potential impacts resulting from fuel handling activity at Dutch Harbor, Bethel, and the Angyaruaq (Jungjuk) ports. Under normal operations, the impacts to surface water quality that would result from the operations and termination of transportation facilities under any options of Alternative 3B would be essentially the same as those associated with Alternative 2. As a result of decreased barging activities on the Kuskokwim River under Alternative 3B, impacts to surface water quality resulting from localized sediment resuspension in critical shallow sections of the river would decrease.

Diesel Pipeline

The alignment of the diesel pipeline proposed under Alternative 3B and options, and its associated construction practices, schedule, and stream crossing techniques would be the same as those proposed for the natural gas pipeline discussed under Alternative 2. Impacts to surface water quality resulting from the construction of the diesel pipeline under Alternative 3B would be essentially the same as those described under Alternative 2; surface water quality at stream crossings would be expected to comply with applicable water quality standards throughout the construction phase, and impacts to surface water resources from hydrostatic testing would be limited because amounts of water required for hydrostatic testing would be small compared to potential sources from rivers and small lakes along the route (SRK 2013b).

Impacts to surface water quality resulting from operations and termination of the diesel pipeline under any option of Alternative 3B would be imperceptible. Relative to the natural gas pipeline proposed under Alternative 2, operations and termination of a diesel pipeline would result in increased potential for impacts to surface water quality as a result of spills or pipeline rupture. However, under normal operating conditions, erosion control measures including vegetation maintenance, BMPs, and mitigation measures would minimize impacts to surface water quality resulting from the operation of the pipeline. If the pipeline is cleaned effectively and abandoned in place, then any impacts to surface water quality caused by removal of the pipeline would be avoided.

3.7.3.4.3 GROUNDWATER QUALITY

Mine Site

The reduced diesel barging and construction of a diesel pipeline under Alternative 3B would not result in changes to any of the principal mechanisms of impact to groundwater quality at the Mine Site from those described under Alternative 2. Increased use of diesel at the Mine Site under Alternative 3B would result in greater potential for impacts to groundwater quality to result from spills associated with higher diesel throughput to the power plant. However, this mechanism of impact would be offset by the decreased tanker truck operation and transfers, and reduced fuel storage capacity at the Mine Site. Under Alternative 3B, approximately 10 Mgal of diesel would be stored at the Mine Site, compared to approximately 37.5 Mgal of diesel fuel storage capacity proposed under Alternative 2. The decreased requirement for diesel storage at the Mine Site is not expected to affect any changes to groundwater quality at the Mine Site relative to Alternative 2. Under Alternative 3B, the WRF and pit lake, which are the facilities responsible for the principal mechanisms of impact to groundwater quality, would be constructed, operated, and closed in a manner identical to that described under Alternative 2.

Transportation Corridor

Construction of Transportation Corridor facilities proposed under Alternative 3B would be substantially similar to that described under Alternative 2, with the exception of extension of the dock at the Tyonek North Foreland Facility. The extension of the dock is not expected to result in any measureable impacts to groundwater quality, and therefore the impacts resulting from the construction of transportation facilities under Alternative 3B would be the same as those described under Alternative 2.

Operations and closure activities associated with the transportation facilities proposed under Alternative 3B are not expected to have measurable effects on groundwater quality.

Diesel Pipeline

Construction

Under Alternative 3B, a diesel pipeline would be constructed instead of, or in addition to the natural gas pipeline proposed under Alternative 2. The alignment of the diesel pipeline would be dependent on the selected option (Alternative 3B, Alternative 3B Collocated Natural Gas and Diesel Pipeline Option, or Alternative 3B Port MacKenzie Option), but construction practices, schedule, and stream crossing techniques would be the same as those proposed for the natural gas pipeline under Alternative 2 and one option of Alternative 3B. Impacts to groundwater quality resulting from the construction of the diesel pipeline under Alternative 3B would be the same as those described under Alternative 2, with only minor variations in geographic impacts, depending on the selected option.

Operations and Closure

Like Alternative 2, impacts to groundwater quality resulting from operations and closure of the diesel pipeline are assumed to be imperceptible. Relative to the natural gas pipeline proposed under Alternative 2, operation and closure activities associated with a diesel pipeline would result in increased potential for impacts to groundwater quality as a result of spills or pipeline rupture. However, under normal operating conditions, erosion control measures including vegetation maintenance, BMPs, and mitigation measures would minimize impacts to groundwater quality resulting from the operation of the pipeline. If the pipeline is cleaned effectively and abandoned in place, then any impacts to groundwater quality caused by removal of the pipeline would be avoided.

3.7.3.4.4 SEDIMENT QUALITY

Mine Site

Under Alternative 3B, increased use of diesel at the Mine Site would result in increased risks to sediment quality relative to Alternative 2 as a result of diesel spills, which could occur during fuel handling. However, this potential for increased adverse impacts to sediment quality would be offset by the decreased capacity for diesel storage at the Mine Site under Alternative 3B relative to Alternative 2, which would theoretically decrease the intensity of potential adverse impacts to sediment quality at the Mine Site in the event of an undetected leak from a diesel storage tank. Under normal operational conditions, the expected impacts to sediment quality at the Mine Site under Alternative 3B would be the same as those described under Alternative 2.

Transportation Corridor

Construction

Under Alternative 3B, the existing dock at the Tyonek North Foreland Facility in upper Cook Inlet would need to be extended approximately 1,500 feet from shore in order to accommodate tank vessels of the size required to supply the diesel pipeline. As described above under Surface Water Quality, construction of the extended dock in Upper Cook Inlet would require some in-water construction and/or pile driving activities, which would potentially result in the resuspension of sediments in the immediate vicinity of the in-water construction and/or pile driving activities. The construction activity associated with the Tyonek Dock would not require dredging. The intensity of impacts to sediment quality in upper Cook Inlet resulting from the construction of transportation facilities proposed under Alternative 3B may or may not be measurable, and the overall impacts to sediment quality resulting from the construction of transportation facilities described under Alternative 3B would be the same as those described under Alternative 2.

Operations and Closure

The reduction in barging under Alternative 3B would reduce peak annual project barge traffic on the Kuskokwim River to an estimated 64 additional round trips per season from the estimated 122 additional round trips per season proposed in Alternative 2. This would reduce the level of impact to sediment quality during barging operations compared to those described under Alternative 2. The decrease in number of barge trips under Alternative 3B would reduce the amount of low water river travel through critical sections and result in comparatively fewer situations in which bottom sediment erosion would occur.

Diesel Pipeline

Construction

The alignment of the diesel pipeline, construction practices, schedule, and stream crossing techniques would be the same as those proposed for the natural gas pipeline under Alternative 2, with some geographic variation depending on the selected option (Alternative 3B or Alternative 3B Port MacKenzie Option). Impacts to sediment quality resulting from the construction of the diesel pipeline under any option of Alternative 3B would essentially be the same as those described for the natural gas pipeline under Alternative 2.

Operations and Closure

As described under Alternative 2, impacts to sediment quality resulting from operations and closure of the pipeline are assumed to be imperceptible. Relative to the natural gas pipeline proposed under Alternative 2, operation and termination of a diesel pipeline would result in increased potential for impacts to sediment quality as a result of spills or pipeline rupture. However, under normal operating conditions, erosion control measures including vegetation maintenance, BMPs, and mitigation measures would minimize impacts to sediment quality resulting from the operation of the pipeline. If the pipeline is cleaned effectively and abandoned in place, then any impacts to sediment quality caused by removal of the pipeline would be avoided.

3.7.3.4.5 SUMMARY OF ALTERNATIVE 3B IMPACTS

Under Alternative 3B, the primary mechanisms responsible for impacts to surface water quality, groundwater quality, and sediment quality at the Mine Site would be the same as discussed under Alternative 2. The decreased requirement for diesel storage at the Mine Site is not expected to affect any changes to water quality at the Mine Site during the construction period relative to Alternative 2. The additional use of diesel at the Mine Site under Alternative 3B would result in increased potential for diesel fuel spills; however, under normal operating conditions (i.e., no spills) the impacts to water quality at the Mine Site would be the same as those proposed under Alternative 2.

For the Transportation Corridor facilities, Alternative 3B would result in the potential for additional impacts to surface water quality in Upper Cook Inlet resulting from the extension of the Tyonek North Foreland Facility dock during the construction phase. There would also be a reduction in sediment quality impacts from reduced barging, but any impacts would be offset by small increases in sediment impacts for the additional port site at Tyonek. However, these impacts would not change the overall impacts associated with the Transportation Corridor for Alternative 3A from those described under Alternative 2.

The diesel pipeline proposed under all options of Alternative 3B would result in greater risk to surface water and groundwater resources from spills or pipeline rupture relative to the pipeline under Alternative 2. Impacts associated with climate change would be the same as those discussed for Alternative 2.

Design features, Standard Permit Conditions, and BMPs related to water quality are described under Alternative 2. Examples of additional measures being considered that are applicable to this resource are listed under Alternative 2.

3.7.3.5 ALTERNATIVE 4 – BIRCH TREE CROSSING (BTC) PORT

3.7.3.5.1 GEOCHEMISTRY

Geochemical direct, indirect, and cumulative impacts under Alternative 4 would be the same as discussed for Alternative 2, as the Mine Site and associated facility footprints would be the same.

3.7.3.5.2 SURFACE WATER QUALITY

Mine Site

Effects to surface water quality at the Mine Site under Alternative 4 would be the same as those described for Alternative 2.

Transportation Corridor

Alternative 4 would move the upriver port site from Angyaruaq (Jungjuk) to BTC. This would result in a shorter distance for barging on the Kuskokwim River, but would require a longer road to connect the port to the Mine Site relative to Alternative 2. Construction, Operations, and closure of the Transportation Corridor facilities would be the same as described under Alternative 2.

Alternative 4 would result in fewer impacts to surface water quality relative to Alternative 2 as a result of the decreased number of stream crossings associated with the BTC Road (40 stream crossings) relative to the mine access road of Alternative 2 (51 stream crossings). Material sites along the road would provide the material for road construction. NAG rock would not be used to construct the BTC Road. Impacts to surface water quality could result from leaching of arsenic, selenium, antimony and possibly other constituents of concern from the material used for road construction. The potential for such impacts would be controlled by testing materials from borrow sites for metal leaching potential prior to use for construction. Materials that could act as sources of contaminants of concern would not be used and alternative material sites would be identified.

In addition, runoff and sedimentation resulting from road construction could have impacts on surface water quality that exceed regulatory limits at stream crossing sites along the BTC Road corridor. The geographic extent of such impacts would decrease under Alternative 4 relative to Alternative 2 due to the decreased number of stream crossings associated with the BTC Road. Under Alternative 4, such impacts would be such that receiving waters would comply with most stringent applicable water quality criteria (AWQC), which specify that turbidity levels in freshwater streams may not exceed 5 NTU above natural conditions when the natural turbidity is 50 NTU or less, and may not have more than 10 percent increase in turbidity when the natural turbidity is more than 50 NTU, not to exceed a maximum increase of 15 NTU. Although the extent or scope of the impacts would increase under Alternative 4 relative to Alternative 2, in either instance the impacts would be limited to discrete portions of the project area. Impacts to surface water quality resulting from construction of the BTC Road would be such that water quality would return to pre-activity levels following the cessation of the construction activities. Impacts would affect areas that range from ordinary water/sediment quality or abundant water / sediment resources, to areas with high quality or resources that are considered valuable in the region. Water quality is governed by regulation.

Positioning the upper terminal at BTC instead of Angyaruaq (Jungjuk) would eliminate barge-related impacts to surface water quality in the Kuskokwim River upstream from BTC. As with Alternative 2, concentrations of sediment suspended in the water may increase in some shallow locations as a result of low-water barge traffic down-river from BTC port, but there would be fewer critical sections under Alternative 4. Such impacts would result in surface water quality below applicable regulatory limits. The extent or scope of impacts would be limited to discrete portions of the project area. The duration of any changes to surface water quality resulting from barge passage under normal operating conditions would be expected to return to pre-activity levels within hours of a barge passing.

Pipeline

Effects to surface water quality associated with the Construction, Operations, and Closure of the natural gas pipeline under Alternative 4 would be the same as those described for Alternative 2.

3.7.3.5.3 GROUNDWATER QUALITY

Mine Site

Effects to groundwater quality at the Mine Site under Alternative 4 would be the same as those described for Alternative 2.

Transportation Corridor

The overall impacts to groundwater quality resulting from the construction, operations, and closure of Transportation Corridor facilities would be considered imperceptible. The impacts to groundwater quality resulting from all phases of the transportation infrastructure proposed under Alternative 4 would be the same as those described under Alternative 2. The only difference between Alternative 4 and Alternative 2 is the location of the upriver terminal facility. Changing the location from Angyaruaq (Jungjuk) Port to the BTC Port would potentially transfer the location of impacts associated with alterations in the rates of exchange between surface water and shallow groundwater deposits along the banks of the Kuskokwim River (bank storage effects); however, in either instance the extent or scope of such impacts would be limited to the areas immediately adjacent to the terminal facilities and the effects on the composition of the groundwater and availability for beneficial uses would be imperceptible.

Pipeline

Effects to surface water quality associated with the construction, operations, and closure of the natural gas pipeline under Alternative 4 would be the same as those described for Alternative 2.

3.7.3.5.4 SEDIMENT QUALITY

Mine Site

Effects to sediment quality at the Mine Site under Alternative 4 would be the same as those described for Alternative 2.

Transportation Corridor

The primary project-related mechanisms of impact to sediment quality under Alternative 4 in the transportation corridor would be associated with the construction of shoreside facilities at Bethel (connected action) and the BTC port, and river bottom sediment erosion caused by propeller wash at critical (shallow) sections of the river, which would be fewer in number under Alternative 4. In-stream construction has the potential to contribute to increased sediment loads, which could potentially affect sediment quality at downstream locations. Under Alternative 4, the mechanisms and intensity of impacts to sediment quality at the port sites would be the same as those described under Alternative 2, except that the impacts associated with placement of the upstream facility would occur at BTC, instead of at Angyaruaq (Jungjuk). Impacts from propeller wash would be less than Alternative 2, but still the same level of intensity at critical sections downstream of the BTC Port.

Pipeline

Effects to sediment quality associated with the construction, operations, and closure of the natural gas pipeline under Alternative 4 would be the same as those described for Alternative 2.

3.7.3.5.5 SUMMARY OF ALTERNATIVE 4 IMPACTS

Effects to surface water quality, groundwater quality, and sediment quality at the Mine Site and along the natural gas pipeline under Alternative 4 would be the same as those described for Alternative 2. Like Alternative 2, the principal mechanisms of impact to water quality associated with the Mine Site (e.g., pit lake, TSF, WRF) do not change under Alternative 4. Impacts associated with climate change would be the same as those discussed for Alternative 2.

For the Transportation Corridor facilities under Alternative 4, impacts to surface water quality would increase slightly at stream crossings along the BTC Road, and would decrease slightly in the Kuskokwim River at locations between the BTC and Angyaruaq (Jungjuk) ports relative to Alternative 2. Impacts to groundwater quality resulting from the placement of the upriver terminal facility would occur at the BTC Port, instead of at the Angyaruaq (Jungjuk) Port as described under Alternative 2. However, these impacts would not change the overall intensity, duration, extent, and context of impacts associated with Transportation Corridor facilities for Alternative 4 from those described under Alternative 2.

Design features, Standard Permit Conditions, and BMPs related to water quality are described under Alternative 2. Examples of additional measures being considered that are applicable to this resource are listed under Alternative 2.

3.7.3.6 ALTERNATIVE 5A – DRY STACK TAILINGS

Alternative 5A would use the dry stack tailings method instead of the wet slurry tailings storage method that would be used under Alternative 2. Dry stack tailings can be described in the simplest terms as finely ground, dewatered, and processed tailings material, which, due to its low water content relative to conventional tailings slurry, can be stacked in stable layers.

Under Alternative 5A, the TSF would be located in the Anaconda Creek Valley, in the same general location as under Alternative 2. Tailings would be dewatered in a filter plant using specialized equipment to produce a partially saturated, compactable material (filter cake). This material would be delivered to the TSF by conveyor or truck and spread and compacted in layers using bulldozers. Process-affected water removed from the tailings would be transported to an operating pond via a pipeline. Reclaimed water from the operating pond would be pumped back to the process plant for reuse. The dry stack tailings material would be progressively reclaimed during the operations phase and would be covered at closure. The dry stack cover system would use the same material as Alternative 2 with the addition of an impermeable geomembrane liner incorporated into it. At closure, the operating pond water and any residual solids would be pumped to the open pit. The operating pond and main dam liners would be removed, the dam walls would be breached and graded back into the footprint, and the footprint reclaimed. Seepage recovery would continue as necessary and monitoring in closure and post-Closure would be the same as Alternative 2.

This alternative includes two options:

- Unlined Option: The tailings storage facility (TSF) would not be lined with a linear low-density polyethylene (LLDPE) liner. The area would be cleared and grubbed and an underdrain system placed in the major tributaries under the TSF and operating pond to intercept groundwater base flows and infiltration through the dry stack tailings (DST) and convey it to a Seepage Recovery System (SRS). Water collecting in the SRS pond would be pumped to the operating pond, lower contact water dam (CWD), or directly to the processing plant for use in process.
- Lined Option: The DST would be underlain by a pumped overdrain layer throughout the footprint, with an impermeable LLDPE liner below. The rock underdrain and foundation preparation would be completed in the same manner as the Unlined Option.

3.7.3.6.1 GEOCHEMISTRY

Alternative 5A has different effects associated with the TSF and the post-Closure pit lake at the Mine Site, discussed below. Impacts associated with the WRF and overburden stockpiles would be the same as those discussed under Alternative 2.

Mine Site

Construction and Operations

Alternative 5A includes a dry stack tailings disposal in contrast to the Alternative 2 conventional slurry tailings disposal within a lined TSF. This disposal method involves dewatering the tailings in a filter plant to produce a compactable, partially saturated material called a filter cake, with a moisture content of 19 percent by mass (BGC 2014a). The TSF of Alternative 5A would include an upper cell of filtered tailings, a lower fully lined operating pond, an SRS, fresh and contact water diversion channels, and overburden stockpiles. Return water from dewatering and filtration would either be pumped back to the plant process water or released into the operating pond. Any tailings that cannot be filtered to the specified moisture content would also be stored in the pond.

While detailed water quality modeling has not been conducted for Alternative 5A, it is anticipated that metals and TDS concentrations in dry stack porewater would be the same or higher than those of tailings porewater under Alternative 2 (BGC 2014a), likely due to less leaching and dilution by recirculated process water.

Issues involving the selection of dry stack design with regard to liners, seepage flow, and long-term water quality are complex and do not lend themselves to an obvious single option at a conceptual level of design. In light of the trade-offs associated with particular TSF design features, the following two options, as described by AECOM (2015e) and BGC (2015d) are considered:

- Unlined Option – with linear low-density polyethylene cover (LLDPE): The dry stack tailings would be placed on existing overburden material following removal of ice-rich or saturated overburden. The foundation material would typically consist of colluvium and loess, with alluvial deposits in the valley bottom. A rock underdrain would be placed in the major tributaries of the facility as described in BGC (2014a). As dry stack facilities usually do not require basal liners this option is consistent with the current state-of-practice.
- Lined Option – with Pumped Overdrain and LLDPE Cover: This design provides the advantage of minimizing (but not preventing) the potential for water quality impacts to groundwater and Crooked Creek, and the ability to optimize water management based on an ongoing understanding of seepage flow. After closure, the dry stack top and bottom surfaces would both be sealed by artificial liners in order to minimize or eliminate the flow of water through the dry stack over the long term.

Under either the Lined or Unlined Options, construction of the dry stack would be ongoing throughout the mine life as the tailings are produced. The operating pond size would be expanded in campaigns every 4 years to store the anticipated volume produced during the intermediate years (BGC 2014a). The SRS would be constructed downstream of the main dam and operating pond. The water collected would be pumped either to the operating pond, the Lower CWD, or to the process plant for use as process water.

Closure

Open Pit/Pit Lake

Alternative 5A is expected to result in a substantial increase in the volume of water that is stored in the operating pond and then pumped to the pit at closure. Accordingly, the pit lake would fill more rapidly than for Alternative 2, reaching its controlled discharge elevation in approximately Year 42 to 47 after closure, depending on the option (see Section 3.7.3.6.2, Surface Water Quality below).

Preliminary modeling of the effects of pumping the dry stack tailings water to the pit indicates that lake stratification would likely occur at an approximately 40 percent shallower depth, and surface water concentrations of metals would likely be higher than for Alternative 2. As shown in Appendix H, the modeled pycnocline in Alternative 2 becomes shallower and the chemical gradient weakens over time as the pit lake evolves after closure. Assuming the Alternative 5A modeling shows similar pit lake evolution over time to that observed for Alternative 2, the shallower pycnocline observed under Alternative 5A likely increases the chance of the lake overturning and mixing to the surface over a shorter period of time. Also, with higher surface water concentrations under Alternative 5A, a greater potential exists for ice melt to cause seasonal near-surface stratification and continued concentration of surface waters over time if more dilute ice melt is preferentially withdrawn for treatment and discharge each spring.

Tailings Storage Facility

At Mine Site closure, an LLDPE geomembrane liner would be incorporated into a soil cover for each of the options described above. A cover system of stripped overburden would be placed over the dams which would be flattened to 3H:1V. The operating pond and water from the SRS would be drained into the open pit and the liner removed. An additional SRS would be constructed downstream from the upper dam to collect underdrain seepage from the dry stack to be pumped to the pit. After Year 10 of closure, it is expected that the surface water runoff from the dry stack cover would be of suitable quality for discharge to Crevice Creek.

The water quality of seepage from the dry stack TSF of Alternative 5A is predicted to exceed AWQC for several constituents, and would be handled similarly to those in Alternative 2. At closure, the dry stack TSF drainage would be pumped to the pit lake. Once the pit lake reaches its maximum allowable elevation at or about Year 43 post-Closure, the surficial water is to be treated to meet AWQC and then discharged.

3.7.3.6.2 SURFACE WATER QUALITY

Mine Site

Construction

Under Alternative 5A, impacts to surface water quality resulting from construction phase activities at the Mine Site would be similar to those described under Alternative 2. The main source of water discharged to Crooked Creek during the construction period would be the pit dewatering wells; the groundwater derived from the pit perimeter and in-pit dewatering wells would be expected to require treatment prior to discharge in order to meet AWQC and expected effluent limits (SRK 2017b). The water would be treated using the same Operations WTP processes described in Section 3.7.3.2.2 for Alternative 2 (BGC 2015j; Hatch 2015, 2017). Under Alternative 5A, impacts to surface water quality resulting from runoff from construction materials and from erosion and sedimentation at the Mine Site would be similar to those described under Alternative 2. The construction of the dry stack TSF would occur within the Anaconda Creek watershed, and the footprint of the facility would largely overlap with that of the wet slurry TSF proposed under Alternative 2.

Operations

Under Alternative 5A, the volumes of water and slurry stored behind the tailings main dam during the operational period would be lower relative to Alternative 2, and progressive reclamation of the TSF would be conducted during the operational period. Runoff would be controlled by diversion of non-contact water away from the dry stack; contact water would be collected and transported to the operating pond via a pipeline, where it would be managed and treated similar to Alternative 2. According to BGC (2014a), while specific water quality modeling has not been carried out for the dry stack alternative, it is expected that the concentrations of metals and TDS in the porewater would be the same or higher than that under Alternative 2 (see Appendix H, Geochemistry).

Under Alternative 5A, a seepage recovery system (SRS) would be constructed downstream of the main dam and operating pond. The SRS would comprise a collection pond, diversion ditches, seepage recovery wells, and a pumpback system. The underdrains constructed under the liner and TSF would capture groundwater base flows from upgradient of the operating pond and dry stack, precipitation infiltrating through the dry stack, and potential seepage through the operating pond liner due to liner defects, and direct them to the SRS (BGC 2014a). The water in the collection pond would be pumped to the operating pond, Lower CWD, or directly to the process plant. Seepage from the dry stack would not be discharged without treatment under either the Lined or Unlined Options.

The WTP methodology described in Section 3.7.3.2.2 would be used to treat seepage SRS water to comply with the most stringent AWQC before discharge to Crooked Creek. Under Alternative 5A, sludge from the HRC and greensand water treatment processes would be placed in the TSF operating pond. During operations, brine generated as a byproduct of reverse osmosis (RO) water treatment would be sent directly to the process plant as a source of reclaim water for process. Additional details regarding flow rates of process and wastewater under Alternative 5A are provided in Section 3.5 (Surface Water Hydrology) and 3.6 (Groundwater Hydrology), as well as in the memorandum "Dry Stack Water Balance with Advanced Water Treatment" (AWT) (BGC 2015). Water from the tailings pond and SRS would be reclaimed and used as process water after treatment during the operational period. While water and tailings slurry volumes would be reduced under Alternative 5A and the dry stack would be progressively reclaimed, the intensity of impacts to downgradient surface water resources in the Crooked Creek watershed in operations would be the same as Alternative 2, because water captured by the TSF dams and SRS would be recycled through the process plant under either alternative.

The primary impacts to surface water quality associated with Alternative 5A would be related to fugitive dust from the dry stack tailings facility and potential runoff from precipitation to the surface of the tailings stack. The amount of fugitive dust generated by wind erosion would depend upon the frequency and intensity of disturbances of the erodible surface of the tailings material, and the rates of progressive reclamation of the dry stack tailings facility during the operational period, as each new lift would expose unreclaimed tailings material to wind and precipitation. Besides concurrent reclamation, the dry stack dust would be mitigated through the application of a polymer dust control at 3-foot lift intervals and windbreaks (BGC 2014a).

As described in Section 3.2, Soils, and Section 3.8, Air Quality, the dry stack would have a 60 percent greater surface area potentially exposed to wind erosion than the exposed tailings beach under Alternative 2. Fugitive dust scenarios associated with this alternative predict about a 6.6 percent increase in dust for the Mine Site as a whole compared to Alternative 2. The total increase in $PM_{2.5}$ and PM_{10} emissions under Alternative 5A relative to Alternative 2 would be 2.9 percent and 8.3 percent, respectively (Air Sciences 2015b, c). These estimates take into account the lack of a tailings beach under Alternative 2, as well as concurrent reclamation of the dry stack, which would be limited to south- and west-facing slopes. Dust estimates under Alternative 5A are based on an end-of-mine life source area of 1,500 acres for the exposed dry stack surface, out of a total final closure cover size of 2,500 acres. Given the small incremental increase in overall Mine Site dust above Alternative 2, the pattern of dust extent under Alternative 5A is expected to be largely similar to that of Alternative 2, depicted on maps in Section 3.8, Air Quality.

The composition of fugitive dust generated from the tailings stack would be similar to that of the tailings solids of Alternative 2, described in Section 3.2.3.2.4 (Soils, Quality) and shown in Table 3.2-4 (Soils) and Table 3.7-47. The tailings would have relatively high concentrations of arsenic, antimony, and mercury (Fernandez 2014e). There could be a small increase in these constituents in surface water above that predicted under Alternative 2, due to the small increase in fugitive dust. However, it is unlikely to translate to an increase in mercury methylation in wetlands, as the dust source would be primarily in an inorganic form that is unlikely to methylate.

There is some uncertainty with regard to the effectiveness of the dust control mitigation between polymer lift applications. Direct deposition of fugitive dust and indirect deposition from runoff to surface water within the area affected by dust could result in elevated concentrations of antimony, arsenic, and mercury in surface water. The increase in fugitive emissions due to the dry stack would be offset by the elimination of fugitive dust emissions from the TSF beach area. Under both Alternatives 2 and 5A, some mercury from the TSF would enter the air as a result of volatilization, and it is likely that the rate of volatilization from tailings would be greater from the dry stack (Alternative 5A) relative to wet slurry tailings (Alternative 2). The geographic extent of potential effects could range from the American and Anaconda creek watersheds, in which most runoff would be captured by the lower CWD and operating pond, to adjacent or other watersheds. In terms of intensity, depending on the rates of deposition and other factors, it is likely that concentrations of these constituents would be measurable and possible that they could exceed the most stringent AWQC in surface water, particularly for elements that are close to, or exceed, AWQC in baseline (e.g., arsenic, mercury; Tables 3.7-2 through 3.7-4). Water quality would potentially be reduced throughout the life of the project and return to background ranges sometime after the end of mine life, and effects could be felt throughout the Mine Site and potentially outside the project footprint. The context of impacts would be the same as described under Alternative 2.

Changes to equipment at the Mine Site under Alternative 5A relative to Alternative 2 include the addition of a pressure filter system to dewater the tailings prior to placement within the tailings disposal area. The operation of the pressure filter system is not expected to result in any differences in impacts to surface water quality relative to Alternative 2.

Table 3.7-47: Ore (Tailings) Metals Concentrations

Element	Concentration (mg/kg)	N
Arsenic	907	8625
Mercury	1.2	7439
Antimony	16	8624

Notes:

Based on compiled As, Sb, and Hg concentrations for the ore using the assay analysis from drill core samples in combination with the pit design model. The tailings material would be represented by the material designated as ore in the pit model. The As, Sb, and Hg concentrations are represented by the geometric mean of the individual values. The sample size, N, for each metal is also shown (Fernandez 2014e).

Source: Fernandez 2014e.

Closure

Surface Water Runoff

Under Alternative 5A, the dry stack tailings material would be progressively reclaimed during the operations phase and completed at closure. Progressive reclamation would be essential to reduce dust generation and resulting impacts to surface water quality (Davies and Rice 2002). After successful reclamation, fugitive dust and potential migration of contaminants to surface water would no longer occur. The dry stack cover, which is included in both the Lined and Unlined Options, would prevent fugitive dust releases, minimize erosion from runoff, and provide medium for vegetative recolonization of the site. The dry stack cover system would use the same material as Alternative 2 with the addition of an impermeable geomembrane liner incorporated into it. Progressive reclamation would occur on the south and west facing slopes of the dry stack as the lifts advance. It is anticipated that completion of dry stack reclamation in closure would occur over a 5-year period.

During the closure period, surface runoff would be directed to the relocated SRS, where it would be pumped to the open pit, and the cover surface would be graded to direct surface runoff to the southeast of the TSF (BGC 2014a). Surface runoff water to the SRS would be held and tested for water quality. It is assumed that a 5-year period would be necessary to demonstrate that water quality running off the cover is acceptable for discharge; during this period, all cover runoff would be pumped to the pit lake (BGC 2014a). It is further assumed that after Year 10 of closure, this water would be of suitable quality for discharge, and runoff from the pond would be permitted to drain to Crevice Creek from Year 11 of closure onwards via a spillway that would be excavated in the ridge dividing Anaconda and Crevice Creeks (BGC 2014a). Provided that runoff from the reclaimed dry stack meets permitted effluent limits prior to discharge during the post closure period, the impacts to surface water quality resulting from implementation of Alternative 5A would be similar to those described under Alternative 2.

Dry Stack Seepage

Under Alternative 5A-Unlined Option, runoff infiltrating the dry stack would report to the underdrain and mix with upstream runoff and groundwater baseflow also reporting to the underdrain. This combined flow would report to the downstream SRS and would then be pumped to the ACMA Pit via pipeline at closure. SRS water would be transferred to the pit in the closure and post-Closure periods until such time that seepage flow through the dry stack reduces to the point that the SRS may be able to be decommissioned. It is estimated that it would take roughly 200 years for seepage flow to reach the same rate as that predicted for the TSF under Alternative 2 (BGC 2015d). Water levels in the ACMA Pit would reach an elevation of 328 feet in about 42 years, which is the target elevation at which treatment of pit lake water would commence.

Under the Lined Option, runoff infiltrating the dry stack would be pumped directly from the overdrain to the pit lake at closure (Figure 3.5-38, in Section 3.5, Surface Water Hydrology). The SRS would still be required downstream of the Lined Option dry stack to capture underdrain flow and potential seepage through the dry stack liner. If there is no liner leakage beneath the Lined Option dry stack, it is estimated that underdrain flows to the SRS would be suitable for discharge to Anaconda Creek and the SRS would be able to be decommissioned after 10 years of closure. This would reduce the SRS discharge volume to the pit lake, and the lake elevation of 328 feet would be reached in about 47 years. It is possible, however, that the liner beneath the dry stack would leak at the same rate as that assumed for the Alternative 2 TSF. In this case, the time it would take for the SRS to be decommissioned would be similar to that of Alternative 2 (about 51 years).

Thus, the main difference between the Unlined Option, the Lined Option, and Alternative 2 with respect to surface water quality is the time it takes for the SRS water to clean up to the point that it can be discharged to Anaconda Creek, that is, roughly 200 years under the Unlined Option, and about 10 to 50 years under the Lined Option and Alternative 2. Under either option, SRS water would continue to be pumped until suitable for discharge, and effects on downgradient water quality in Crooked Creek would be the same as Alternative 2, as pit lake water would not be discharged to the environment without treatment.

Risk of SRS Pump Failure

The hydraulic containment system of the SRS would require monitoring, analysis, operation, periodic repair, and management to assure its continuing functioning and effectiveness in post-Closure. The likelihood of SRS pump failure and overflow under normal operating conditions would be very low. However, the pumping system is a mechanical system with many task-critical components (see Section 3.6, Groundwater Hydrology). Considering the long duration of pumping, the harsh climate, and the remote location, the consequences of a failure merit examination. The risk of SRS pump failure would be greater under the Unlined Option relative to the Lined Option or Alternative 2 because the SRS would need to remain operational and be maintained for a longer time period in post-Closure under the Unlined Option.

In either option, if the SRS pumping system were to go completely off-line, the SRS would likely fill to overflowing and/or lose hydraulic containment with respect to groundwater in approximately ½ week to 2 weeks, although there are many variables such as precipitation, time of year, and amount of drawdown at the start of the failure that could affect these calculations (see Sections 3.5, Surface Water Hydrology and 3.6, Groundwater Hydrology). In this event, overflow of SRS pond water would enter Anaconda Creek, with a high potential to subsequently reach Crooked Creek. If hydraulic containment of the groundwater system is lost, it is likely that contaminated groundwater would enter a topographically controlled flow path towards Crooked Creek, and it would be infeasible to retrieve because the water would migrate outside of the radius of influence of the SRS pond. In the event of a prolonged SRS pumping failure during post-Closure, the unpumped SRS water would migrate downgradient in the Anaconda Creek watershed, eventually potentially influencing the quality of surface water in Crooked Creek.

The intensity of the impacts to surface water quality in Anaconda Creek and Crooked Creek would depend upon the timing of the pump failure, connectivity between the seepage and surface water resources, and the rates of seepage flow relative to Crooked Creek discharge. Under both the Lined and Unlined Options, an event leading to the release of uncontained SRS water to the surface waters of Anaconda Creek or Crooked Creek would result in concentrations of several constituents in both tailings porewater and groundwater that would exceed AWQC. Under Alternative 2, concentrations of antimony, arsenic, cobalt, copper, manganese, molybdenum, nickel, selenium, thallium, mercury, and sulfate in tailings water would exceed the most stringent water quality standards for natural waters (see Section 3.7.3.2.2 - Closure, TSF and SRS Water Quality; and Table 3.7-44). While detailed water quality modeling has not been conducted for Alternative 5A, it is anticipated that metals and TDS concentrations in dry stack seepage would be the same or higher than those of tailings porewater described under Alternative 2 (BGC 2014a), likely due to less leaching and dilution by recirculated process water (AECOM 2015e).

The possible impact of pump failure on downstream water quality would be greater under the Unlined Option due to greater tailings seepage reporting to the underdrain in early post-Closure. For the Unlined Option, seepage flow at the beginning of the closure period represents about 16 percent of the total flow reporting to the SRS (about a 6:1 dilution), compared to 4 percent or less for the Lined Option and Alternative 2 (>24:1 dilution) (Section 3.7.3.2.2 - Closure, TSF and SRS Water Quality). Based on dilution alone, if dry stack porewater quality were similar to that of the wet tailings, predicted SRS concentrations under the Unlined Option would be about 1-1/2 to 4 times higher than those under the Lined Option or Alternative 2 in the early closure period, depending on background concentrations of individual constituents (AECOM 2015e). Downgradient water quality could be impaired under the Lined Option due to liner leakage and elevated baseline conditions, but the intensity of such potential impacts would generally be less under the Lined Option relative to the Unlined Option.

In terms of intensity, any unplanned release of SRS water under either option under Alternative 5A could result in impacts to surface water quality sufficient to exceed water quality regulatory limits. The duration of such impacts would likely be infrequent and not longer than the span of the project construction, provided that resources are readily available to repair or replace the failed pumping device in perpetuity. The extent or scope of such impacts would be limited to discrete portions of the project area in the Anaconda and Crooked Creek watersheds. In terms of context, surface water resources similar to those that could be impacted in the event of an SRS pump failure are considered relatively widespread in the region, but are protected by the CWA and other legislation.

Transportation Corridor

Under Alternative 5A, effects on surface water quality associated with the construction, operations, and closure of Transportation Corridor facilities would be the same as those described under Alternative 2.

Pipeline

Under Alternative 5A, effects on surface water quality associated with the construction, operations, and closure of the natural gas pipeline would be the same as those described under Alternative 2.

3.7.3.6.3 GROUNDWATER QUALITY

Mine Site

Construction

The primary sources of impact to groundwater quality at the Mine Site during Construction would be associated with localized inputs of dissolved metals and TDS from waste rock used in the construction of the Lower CWD. Although groundwater pumping to achieve pit depressurization would begin approximately 1.75 years before the beginning of operations, pit depressurization is not expected to have a substantial influence on groundwater quality during the construction period. These sources of impact would be identical under Alternative 5A and

Alternative 2; therefore, the impacts to groundwater quality at the Mine Site during Construction would be the same as discussed under Alternative 2.

Operations and Closure

Like Alternative 2, the dry stack would be designed to reduce potential impacts to the quality of groundwater. Construction of the dry stack would occur from the valley bottom up. As described above, two options described by AECOM (2015e) and BGC (2015d) are considered.

Under either the Unlined or Lined Options, diversion structures and ditches would be constructed upgradient to reduce surface water deposition to the TSF, and the surface of the dry stack would be graded towards the operating pond to shed contact runoff water and decrease infiltration into the stack.

Under both options, flows collected in the dry stack underdrains would be conveyed beneath the upper dam, the operating pond, and the main dam before discharging to the SRS. The potential for long-term degradation of the underdrains in post-Closure is not considered a concern, because they would be wrapped in geotextile to filter out tailings and native soils; the granular rock blanket filter underlying the upper dam would provide a drainage path for dry stack seepage through the dam; and infiltration through the dry stack would be minimized by progressively grading the dry stack throughout operations to facilitate surface runoff (BGC 2014a). Under both options, placement of the impermeable LLDPE cap at closure would limit infiltration to the dry stack facility during the Closure and post-Closure phases.

Under the Unlined Option, tailings seepage could potentially reach groundwater beneath the dry stack, although the underdrains would be expected to continue to capture some if not all of the tailings seepage. Seepage reaching groundwater would either 1) flow to the underdrains prior to reaching the SRS, or 2) flow through native material under the operating pond, and be captured by the cone of depression created by pumping the SRS and/or sentinel wells. Following removal of the operating pond and dam in closure, if contaminated groundwater is present in native materials beneath the dry stack or operating pond footprint, it would continue to migrate towards, and be captured by, the SRS and/or wells, and report to the pit lake. Meanwhile, the supply of tailings porewater that could potentially feed the contaminant plume would be reduced by the impermeable cover, and seepage flow through the dry stack would gradually reduce to the same as that predicted under Alternative 2 (and Alternative 5A, Lined Option) after 200 years. In other words, a contaminant plume, if present under the Unlined Option, would eventually improve in quality to that of the Lined Option and Alternative 2. Beyond 200 years, the amount of seepage flow under the Unlined Option is expected to continue its gradual decline as a result of the impermeable cover blocking infiltration of water to the flow system.

Under the Lined Option, the presence of a liner under the dry stack would limit tailings seepage from contaminating groundwater beneath it, although there could still be a small amount of liner leakage similar to that under Alternative 2. The groundwater quality beneath the dry stack and operating pond would likely be better than the Unlined Option for a period of time between early closure and 200 years, although in both cases groundwater would be captured by the SRS, pumped to the pit lake, and monitored until it can be shown to meet water quality standards.

As dry stack facilities usually do not require basal liners, the Unlined Option is consistent with the current state-of-practice. The Lined Option would provide the additional advantage of minimizing (but not preventing) the potential for water quality impacts to groundwater and Crooked Creek, and the ability to optimize water management based on an ongoing understanding of seepage flow into the rock overdrain.

Impacts to groundwater quality in the event of SRS pump failure during Closure under Alternative 5A are related to potentially concurrent surface water impacts, and are described under Section 3.7.3.6.2 (Alternative 5A, Surface Water Quality).

Transportation Corridor

Under Alternative 5A, effects on groundwater quality associated with the construction, operations, and closure of Transportation Corridor facilities would be the same as those described under Alternative 2.

Pipeline

Under Alternative 5A, effects on groundwater quality associated with the construction, operations, and closure of the natural gas pipeline would be the same as those described under Alternative 2.

3.7.3.6.4 SEDIMENT QUALITY

Mine Site

As discussed in Section 3.7.3.6.2 (Alternative 5A – Surface Water Quality), direct deposition of fugitive dust from dry stack wind and indirect deposition from runoff to surface water within the area affected by dust blown from the dry stack could result in elevated concentrations of antimony, arsenic, and mercury in surface water. BMPs and mitigation measures would be used to minimize adverse impacts associated with fugitive dust from the tailings dry stack under Alternative 5A. The majority of mercury potentially deposited as a result of fugitive dust would be particulate mercury. Some of the particulate mercury would be transported downstream, and some would rapidly settle out of the water column and become buried in stream sediments. Depending on the rates of deposition and other factors, it is likely that concentrations of these constituents in the sediments would be measurable, and possible that they could exceed applicable regulatory guidance levels for sediment in the region of the Mine Site. If the predicted 6.6 percent increase in dust under Alternative 5A (Section 3.7.3.6.2) were applied to the modeled mercury deposition rates under Alternative 2 (Table 3.7-46), the result would be a slight increase in predicted concentrations at the end-of-mine life of up to 1.3 percent (compared to 1.2 percent under Alternative 2). Thus, while the increase from fugitive dust would be slightly greater under Alternative 5A, the resulting concentrations would likely be within the range of naturally occurring mercury in the mine area.

In terms of duration, sediment quality would potentially be reduced throughout the life of the project, both throughout the Mine Site and potentially outside the project footprint. The context of impacts would be the same as described for Alternative 2. Under Alternative 5A, the dry stack tailings material would be progressively reclaimed during the operational period. After successful reclamation, fugitive dust deposition would no longer occur.

Transportation Corridor

Under Alternative 5A, effects on sediment quality associated with the construction, operations, and closure of Transportation Corridor facilities would be the same as those described under Alternative 2.

Pipeline

Under Alternative 5A, effects on sediment quality associated with the construction, operations, and closure of the natural gas pipeline would be the same as those described under Alternative 2.

3.7.3.6.5 SUMMARY OF ALTERNATIVE 5A IMPACTS

Alternative 5A (both options) includes a dry stack tailings disposal in contrast to the Alternative 2 conventional slurry tailings disposal within a lined TSF. The water quality of dry stack seepage and SRS water under both options is predicted to exceed AWQC for several constituents, and would be treated similarly to those in Alternative 2 prior to discharge. The main difference between the Unlined Option, Lined Option, and Alternative 2 with respect to surface water quality is the time it takes for the SRS water to clean up to the point that it can be decommissioned, that is, roughly 200 years under the Unlined Option, and about 10 to 50 years under the Lined Option and Alternative 2. The Lined Option would provide the additional advantage of minimizing (but not preventing) the potential for water quality impacts to groundwater. Under either option, effects on downgradient water quality in Crooked Creek would be the same as Alternative 2, provided that the SRS water is contained and conveyed to the open pit.

In terms of intensity, effects may be sufficient to exceed water quality regulatory limits in the event of a prolonged SRS pumpback failure during the post-Closure period under both options of Alternative 5A as well as Alternative 2, although the risk of failure and level of contaminants in SRS water would be greater under the Alternative 5A-Unlined Option than the Lined Option or Alternative 2. Like Alternative 2, the principal mechanisms responsible for high impacts to groundwater quality in other areas of the Mine Site would be inputs of seepage from the WRF to shallow groundwater resources underneath and immediately adjacent to the WRF, and the temporary migration of water from the pit to the surrounding deep bedrock groundwater in the early closure period.

Under both options, there could also be an increase in indirect impacts to surface water and sediment quality resulting from dry stack fugitive dust atmospheric deposition and terrestrial runoff from dust depositional areas. In terms of intensity, such indirect impacts could be sufficient to exceed water quality regulatory limits. Like Alternative 2, deposition of mercury to sediments and the potential for increased rates of mercury methylation would likely be measurable and could be sufficient to exceed regulatory guidance levels, although would likely be within the range of naturally occurring concentrations in the mine area. Impacts associated with climate change would be the same as those discussed for Alternative 2.

Effects on surface water quality, groundwater quality, and sediment quality along the transportation corridor and the natural gas pipeline under Alternative 5A would be the same as those described for Alternative 2. There would be no difference in the types of impacts or the amount of ground disturbance associated with these facilities.

Design features, Standard Permit Conditions, and BMPs related to water quality are described under Alternative 2. Examples of additional measures being considered that are applicable to this resource are listed under Alternative 2.

3.7.3.7 ALTERNATIVE 6A – MODIFIED NATURAL GAS PIPELINE ALIGNMENT: DALZELL GORGE ROUTE

3.7.3.7.1 GEOCHEMISTRY

Geochemical direct, indirect, and cumulative impacts under Alternative 6A would be the same as discussed for Alternative 2, as the Mine Site and associated facility footprints would be the same.

3.7.3.7.2 SURFACE WATER QUALITY

Mine Site

Under Alternative 6A, effects on surface water quality associated with the construction, operations, and closure of the Mine Site would be the same as those described under Alternative 2.

Transportation Corridor

Under Alternative 6A, effects on surface water quality associated with the construction, operations, and closure of Transportation Corridor facilities would be the same as those described under Alternative 2.

Pipeline

Under Alternative 6A, natural gas pipeline construction techniques and associated impacts to surface water quality would be substantially similar to those described under Alternative 2. The Dalzell Gorge alignment would cross the Happy River and the South Fork of the Kuskokwim River using HDD and resulting impacts to surface water quality would be below water quality regulatory limits at those locations. As described under Alternative 2, impacts to surface water quality resulting from the normal operation of the pipeline under Alternative 6A are expected to be imperceptible.

3.7.3.7.3 GROUNDWATER QUALITY

Mine Site

Under Alternative 6A, effects on groundwater quality associated with the construction, Operations, and Closure of the Mine Site would be the same as those described under Alternative 2.

Transportation Corridor

Under Alternative 6A, effects on groundwater quality associated with the construction, operations, and closure of Transportation Corridor facilities would be the same as those described under Alternative 2.

Pipeline

As described under Alternative 2, potential impacts to groundwater quality during construction and operations of the natural gas pipeline are expected to be minimal, and limited to a short period during construction. The primary impacts would be associated with installation of the pipeline at river and stream crossings. The impacts to groundwater quality resulting from the pipeline alignment proposed under Alternative 6A would be substantially similar to those described under Alternative 2; however, the specific locations of the impacts would be different as a result of the pipeline realignment proposed under Alternative 6A. Groundwater drainage and recharge patterns should reestablish after site reclamation has been completed, and any impacts to groundwater quality resulting from the normal operation of the pipeline would be reduced. The pipeline would be abandoned in place, and therefore, no additional impacts to groundwater quality would be associated with the closure phase.

3.7.3.7.4 SEDIMENT QUALITY

Mine Site

Under Alternative 6A, effects on sediment quality associated with the construction, operations, and closure of the Mine Site would be the same as those described under Alternative 2.

Transportation Corridor

Under Alternative 6A, effects on sediment quality associated with the construction, operations, and closure of Transportation Corridor facilities would be the same as those described under Alternative 2.

Pipeline

Like Alternative 2, the primary mechanisms of impact to sediment quality resulting from construction of the natural gas pipeline would involve soil erosion associated with installation of the pipeline at river and stream crossings, and clearing of riparian habitat along the pipeline ROW. Because the construction practices and stream crossing techniques employed under Alternative 6A would be substantially similar to those associated with Alternative 2, the resulting impacts to sediment quality would be similar. The specific locations of potential impacts to sediment quality would be different as a result of the pipeline realignment proposed under Alternative 6A. Expected impacts to sediment quality resulting from the normal operation of the pipeline would be imperceptible. If the pipe is abandoned in place, then any new impacts to sediment quality caused by pipe removal would be avoided.

3.7.3.7.5 SUMMARY OF ALTERNATIVE 6A IMPACTS

Effects on surface water quality, groundwater quality, and sediment quality at the Mine Site and along the Transportation Corridor under Alternative 6A would be the same as those described for Alternative 2. There would be no difference in the types of impacts or the amount of ground disturbance associated with these facilities. Impacts associated with climate change would be the same as those discussed for Alternative 2.

The specific locations of the impacts associated with the pipeline under Alternative 6A would be different as a result of the pipeline realignment from the route discussed under Alternative 2.

Like Alternative 2, the primary mechanisms of impact to groundwater quality and sediment quality resulting from construction of the natural gas pipeline would involve soil erosion associated with installation of the pipeline at river and stream crossings, and clearing of riparian habitat along the pipeline ROW. The construction practices and stream crossing techniques would be similar to those associated with Alternative 2, so the resulting impacts would be similar. The pipeline would be abandoned in place, and therefore, no additional impacts to groundwater quality would be associated with the closure phase.

Design features, Standard Permit Conditions, and BMPs related to water quality are described under Alternative 2. Examples of additional measures being considered that are applicable to this resource are listed under Alternative 2.

3.7.3.8 ALTERNATIVES IMPACT COMPARISON

A comparison between alternatives is presented below in Table 3.7-48. Although there are differences among alternatives in the project components that would affect water quality, they are relatively small. Overall there is little difference in the range of impacts to water resources for the various alternatives, as the scope and scale of the three project components are such that changes to a single mine structure, road, port, or pipeline route result in small changes to overall impacts.

Table 3.7-48: Comparison of Impacts by Alternative* for Water Quality

Impact-causing Project Component	Alternative 2 – Proposed Action (includes North Option)	Alternative 3A – LNG-Powered Haul Trucks	Alternative 3B – Diesel Pipeline	Alternative 4 – BTC Port	Alternative 5A – Dry Stack Tailings (Lined and Unlined Options)	Alternative 6A – Dalzell Gorge Route
Mine Site						
Geochemistry	Drainages from the WRF, TSF operating pond, and TSF cover drain layer are predicted to exceed AWQC for several constituents. Lower CWD and drainage from the SOB predicted to exceed AWQC for several constituents during operations. Surficial pit lake water expected to exceed AWQC for several constituents; about Year 52 post-Closure, the surficial water would be treated to meet AWQC and then discharged.	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2.	Impacts associated with the WRF and overburden stockpiles would be the same as Alternative 2. Pit lake stratification would occur at an approximately 40 percent shallower depth, and surface water concentrations of metals would likely be higher than Alternative 2. About Year 42 to 47 post-Closure (depending on the option), the surficial pit lake water would be treated to meet AWQC and then discharged.	Same as Alternative 2.
Surface Water Quality	Surface water in the American and Anaconda Creek watersheds influenced by the creation and perpetual maintenance of the pit lake and TSF impoundment. Pit dewatering discharge to Crooked Creek would be treated to meet AWQC prior to discharge. Atmospheric deposition of mercury could be sufficient to exceed AWQC and baseline ranges in some cases, depending on watershed location and existing baseline concentrations.	Kuskokwim River barge traffic would be reduced and the Bethel and Dutch Harbor ports would not require as much expansion. Changes would not affect overall impact levels from Alternative 2.	The additional use of diesel would result in increased potential of adverse impacts resulting from diesel fuel spills; however, under normal operating conditions (i.e., no spills), impacts would be the same as Alternative 2.	Same as Alternative 2.	Under both options, effects on downgradient water quality in Crooked Creek would be the same as Alternative 2, as SRS water would be contained and conveyed to the open pit. The main difference between the Unlined Option, Lined Option, and Alternative 2 is the time it takes for SRS decommissioning: 200 years under the Unlined Option; and 10-50 years under both the Lined Option and Alternative 2. A small increase in indirect effects could result from dry stack fugitive dust atmospheric deposition and terrestrial runoff from dust deposition; these impacts could exceed AWQC. Impacts that exceed AWQC could occur in the event of a prolonged SRS pumpback failure during post-Closure; risk and contaminant levels would be greater under the Unlined Option than either the Lined Option or Alternative 2.	Same as Alternative 2.

Table 3.7-48: Comparison of Impacts by Alternative* for Water Quality

Impact-causing Project Component	Alternative 2 – Proposed Action (includes North Option)	Alternative 3A – LNG-Powered Haul Trucks	Alternative 3B – Diesel Pipeline	Alternative 4 – BTC Port	Alternative 5A – Dry Stack Tailings (Lined and Unlined Options)	Alternative 6A – Dalzell Gorge Route
Groundwater Quality	Seepage from the WRF underdrain to groundwater between the WRF and Lower CWD (during Operations) and the pit lake (during closure) would occur. Net discharge of water from the pit lake to surrounding deep bedrock groundwater would occur during pit lake filling, primarily during the first 8 years following closure. Groundwater impacts would be limited in geographic extent.	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2.	Similar to Alternative 2, the Lined Option would provide an advantage over the Unlined Option of minimizing (but not preventing) the potential for impacts to groundwater quality.	Same as Alternative 2.
Sediment Quality	Impacts to sediment quality could result from altered stream flows and water chemistry in Crooked Creek and project-related atmospheric deposition of mercury. Impacts from dust deposition would likely exceed SQGs, but remain within the naturally occurring range presently found in the study area.	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2.	Impacts from fugitive dust would be slightly greater than Alternative 2 and likely exceed SQGs, but would likely remain within the naturally occurring range presently found in the study area. BMPs and mitigation would minimize adverse impacts associated with fugitive dust from the tailings stack.	Same as Alternative 2.
Transportation Corridor						
Geochemistry	No impact.	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2.
Surface Water Quality	Impacts from occasional barge-induced suspended sediment or erosion effects at construction sites would meet AWQC. Runoff of water from rock used for road construction could include inputs from constituents of concern; potential impacts could be reduced through mitigation.	Potential impacts related to surface water quality in the Kuskokwim River resulting from increases in suspended sediment concentrations and turbidity would decrease by approximately 32 percent from Alternative 2 due to reduced barging activity.	Same as Alternative 2, but with increased risk of spills associated with fuel handling at the Tyonek North Foreland Facility, and a decrease in potential impacts resulting from fuel handing at the ports.	Increased road length, but decreased number of stream crossings from Alternative 2 would result in fewer impacts. Material sites along the road would be used for road construction, which could result in leaching from constituents of concern; impacts could be reduced through mitigation.	Same as Alternative 2.	Same as Alternative 2.
Groundwater Quality	Placement of sheet pile associated with construction of port terminals could infrequently affect groundwater quality within discrete portion of the project area. Use of groundwater for drinking water supplies at the Angyaruaq (Jungjuk) Port would not impact groundwater quality.	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2; impacts would be transferred from Angyaruaq (Jungjuk) Port to the BTC Port.	Same as Alternative 2.	Same as Alternative 2.
Sediment Quality	Resettled sediment from barging and construction of the ports would be of similar composition to the existing natural deposit. BPMs and ESC measures would be employed to control erosion and sedimentation effects during construction.	Reduction in barging would reduce the amount of low water river travel by approximately 32 percent, resulting in fewer situations where sediment quality could be impacted.	Reduction in barging would reduce the amount of low water river travel, resulting in fewer situations where sediment quality could be impacted.	Impacts associated with sediment quality at the ports would be the same as Alternative 2, just located at the BTC Port instead of Angyaruaq (Jungjuk). Impacts from propeller wash would be less than Alternative 2.	Same as Alternative 2.	Same as Alternative 2.

Table 3.7-48: Comparison of Impacts by Alternative* for Water Quality

Impact-causing Project Component	Alternative 2 – Proposed Action (includes North Option)	Alternative 3A – LNG-Powered Haul Trucks	Alternative 3B – Diesel Pipeline	Alternative 4 – BTC Port	Alternative 5A – Dry Stack Tailings (Lined and Unlined Options)	Alternative 6A – Dalzell Gorge Route
Pipeline						
Geochemistry	No impact.	Same as Alternative 2.	Same as Alternative 2	Same as Alternative 2.	Same as Alternative 2.	Same as Alternative 2
Surface Water Quality	Under all options, potential erosion impacts and the introduction of fine-grained sediments to surface water associated with the pipeline would be mitigated to meet AWQC.	Same as Alternative 2.	The diesel pipeline under all options would increase risk to surface water resources from spills or pipeline rupture relative to Alternative 2. However, under normal operating conditions, mitigation measures would minimize impacts.	Same as Alternative 2.	Same as Alternative 2.	Impacts associated with installation of the pipeline at stream crossings would be the same as Alternative 2, but repositioned west as a result of the Dalzell Gorge alignment.
Groundwater Quality	Under all options, installation of the pipeline could result in alterations to groundwater flow patterns, minor pipeline corrosion, and small changes in groundwater quality. However, these changes would be infrequent and not last longer than the span of the project construction.	Same as Alternative 2.	The diesel pipeline under all options would increase risk to groundwater resources from spills or pipeline rupture relative to Alternative 2. However, under normal operating conditions, mitigation measures would minimize impacts.	Same as Alternative 2.	Same as Alternative 2.	Impacts would be the same as Alternative 2; specific locations would be different as a result of the Dalzell Gorge Route under this alternative.
Sediment Quality	Under all options, sediment quality would be impacted during pipeline construction as a result of increased sedimentation at the more than 400 stream crossing sites. However, impacts would not exceed regulatory limits, and would be minimized through mitigation.	Same as Alternative 2.	The diesel pipeline under all options would increase risk to sediment resources from spills or pipeline rupture relative to Alternative 2. However, under normal operating conditions, mitigation measures would minimize impacts.	Same as Alternative 2.	Same as Alternative 2.	Impacts would be the same as Alternative 2; specific locations would be different as a result of the Dalzell Gorge Route under this alternative.

Notes:

*Alternative 1 (No Action Alternative) would have no new impacts to water quality.