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Memorandum

To: John Robison, Public Lands Director, Idaho Conservation League; Pete Dronkers, Southwest Circuit Rider, Earthworks; and Judy Anderson, private citizen
From: Ann Maest, PhD; Buka Environmental
Date: 27 October 2020
Re: Evaluation of the Draft Environmental Impact Statement (DEIS) for the Stibnite Gold Project, Idaho, and Related Water Quality Conditions, Predictions, and Effects

Summary of Major Findings

Modeling assumptions and characterization errors that affect water quality predictions

- The inputs to the site-wide water chemistry model from waste rock and pit wall leaching will underestimate the range of potential surface water quality results from new mining. The inputs were based on average, “steady state” release rates from longer-term leach tests (humidity cell tests, or HCTs), and higher “first flush” rates were excluded. The sensitivity analysis conducted by SRK Consulting in 2019 (SRK Consulting, 2019b) did not examine this most important variable – contaminant release rates or concentrations used as inputs to the models. As noted by ERM (2019), Nordstrom (2019), Zamzow (2020), and in the DEIS, excluding first flush releases is non-conservative and should be addressed in a supplemental sensitivity evaluation.
- Midas Gold’s consultants discarded results from two laboratory leach tests that were selected to represent the upper range of total arsenic concentrations in solid samples. The decision was made after the leach test results showed that arsenic and antimony releases from these samples were also high. I have never seen this approach taken at any other mine. Discarding these samples pushes the model input values to lower arsenic and antimony concentrations and underestimates the predicted higher end of the range of arsenic and antimony concentrations in groundwater and surface water. The full range of arsenic (and antimony and mercury, etc.) concentrations should be included to estimate realistic ranges of known contaminants in sources, pathways, and receptors.
- Before geochemical characterization began, geoenvironmental units (also called geochemical test units) with similar contaminant leaching and acid drainage potential should have been identified. Instead, the program used only broad distinctions based on lithology (rock types, such as granite or alaskite) to group samples. The geochemical testing results clearly show a wide variety in acid generation and contaminant leaching potential within a single rock type, indicating that further distinctions should have been applied. Without identified geoenvironmental units, the leaching potential from a given lithology could be underestimated and cause long-term environmental problems or the leaching potential within a given lithology could be overestimated and used safely as

construction materials. Mineralogic analysis could potentially be used to create geoenvironmental units at this stage, but the geochemical characterization results would then need to be broadly re-interpreted.

- The site-wide water chemistry (SWWC) model only estimates surface water quality during and after mining; site-wide groundwater quality was not predicted using the model. Groundwater plumes from existing contamination exist but were not drawn on any maps. Groundwater inputs to surface water represent an important contribution to existing poor surface water quality conditions. More information is needed on groundwater-surface water interactions under existing, operational, and post-closure conditions to help understand where mitigation measures can be most effectively applied. Although work has been conducted on groundwater-surface water interactions from a hydrogeologic perspective for many years (DEIS, p. 3.8-4), water quality effects of the groundwater-surface water interactions are less well understood. As suggested by Nordstrom (2019) and Etheridge (2015), a tracer injection study should be done to quantify loading of groundwater inputs to streams under existing conditions.
- The SWWC model was relatively successful in matching existing annual average water quality conditions, although sulfate, arsenic, and antimony were underpredicted at downstream assessment nodes by up to 48%, 62%, and 89%, respectively. These are the same constituents that were shown to be elevated in first flush HCT results. Extensive adjustments were made by adding contaminant loadings to improve the "calibration." The extensive use of averages in flow, concentrations, rates, and other parameters limits the usefulness of the SWWM model. In addition, the degree of success in matching current water quality conditions, after extensive adjustment, does not mean that the model can be used "as the basis for predicting future water quality concentrations at the site," as noted in the DEIS.
- One of the PHREEQC input files made available on October 19, 2020 was run (Hangar Flats DRSF cover). So many errors were found by the program that it only ran three of 80 simulations in the input file before shutting down. The results of the PHREEQC runs were not clearly discussed in the DEIS or any other document. It is unclear how output from the PHREEQC runs could be used in the SWWC model if so many simulations could not be successfully run using the code. The fact that antimony was not able to be included in the runs, even though it is one of the primary COCs, is an important shortcoming that was not discussed in the DEIS or any other document.

Missing environmental medium, contaminant pathway, and toxicity information

- The food chain/dietary pathway for fish (contaminated stream sediment to macroinvertebrates to fish) was not considered in the DEIS conceptual models, in the examination of existing conditions, or in current or future modeling efforts. It was also not considered when evaluating potential environmental improvements from planned legacy cleanup or mitigation measures. No information is provided in the DEIS on stream sediment metal/metalloid concentrations.
- Limited information from a USGS publication shows that sediment concentrations at three of five site locations for arsenic and four of five locations for mercury exceed Canadian sediment quality guidelines for the protection of aquatic life. Sediment arsenic concentrations exceed the probable effects level (PEL) by up to 400 times, and sediment

mercury concentrations exceed the PEL by up to 50 times. The food chain/dietary pathway for arsenic has been shown to adversely affect salmonids in laboratory experiments and at locations in Montana and Idaho, yet it was completely ignored in the DEIS. More sediment sampling is needed, and the results should be included in the design of conceptual models, mitigation and cleanup measures, and modeling efforts in a Supplemental DEIS.

- The toxicity of arsenic and antimony to humans via drinking water and to aquatic biota is highly dependent on their chemical form ("chemical speciation") in surface water and groundwater. No water samples have been analyzed for antimony speciation.
- Essentially no information is available in the literature on the potential food chain/dietary pathway for antimony, which is one of the most important contaminants from legacy and proposed mining activity. Further, little fundamental information is available on the aquatic toxicity of antimony, and arsenic cannot be used as a surrogate. Neither the State of Idaho nor the federal government have established antimony criteria for the protection of aquatic life. A reliable evaluation of the potential effects of the mine cannot be completed without site-specific information on chemical speciation and the toxicity of antimony to salmonids (O'Neal, 2020). If surrogate species are used, the results for all toxicity tests will need to be carefully interpreted. In addition, site-specific toxicity testing should be conducted using clean sediment and sediment with a range of elevated antimony concentrations. Such work is especially important for understanding the effectiveness of promised legacy cleanup measures.

Assumptions about acid-generation and contaminant leaching potential

- The neutralization potential (NP) of Stibnite waste and ore samples has been consistently overestimated. Overestimating the NP will make it appear as if fewer samples and waste types are potentially acid generating (PAG). The NP of all samples needs to be re-evaluated, as does the designation of whether samples are PAG. If PAG designations change, additional mitigation measures will be needed to prevent the formation of acid drainage from new mining activity.
- Regardless of the acid-generation potential of the proposed mined materials, arsenic and antimony are two of the three primary contaminants of concern (COC; the other being mercury) for historic and new mining and will be released under both acidic and alkaline pH conditions.
- Just one HCT sample was used to represent all PAG waste rock and pit wall rock in geochemical and water quality models. Using just one sample to represent all PAG rock across the three pits and the development rock storage facilities (DRSFs) will not represent the potential range of contaminant concentrations that can be released from mine wastes sources. Although the DEIS states repeatedly that acid drainage will not be a problem at the site, four surface samples from legacy mining had pH values below 4 and many had low paste pH values and net acid generating (NAG) pH values <4.5 with low sulfide sulfur percentages, indicating that secondary salts are responsible for the acid produced. Acidic surface samples from legacy wastes should have been used to represent PAG leaching from future mine wastes. As noted by Midas Gold, the legacy mine wastes were characterized "to provide an analogue of likely future geochemical

behavior." However, results from these samples were not used in the site-wide water chemistry model.

- The legacy Spent Ore Disposal Area (SODA) materials are proposed to be used to construct the tailings impoundment embankment (DEIS, Table 2.3-4) under the proposed action. The SODA samples were characterized to assess the "…suitability of spent ore for use in construction." The results found they have the highest release rates for arsenic and antimony and the highest initial concentration of mercury of any of the samples subjected to humidity cell testing. They should clearly not be used as construction materials.

Perpetual water treatment

- Discharge from several mine facilities are projected to require perpetual treatment, including toe seepage from the reclaimed Fiddle DRSF and the Hangar Flats pit lake overflow. Periodic treatment of the West End pit lake discharge could also be required in perpetuity. Because of large uncertainties in the water balance and the lack of consideration of climate change, additional sources could also require perpetual treatment. The perpetual treatment methods include iron precipitation for removal of arsenic, antimony, mercury, and copper; potentially sulfide precipitation for mercury removal; and a biochemical reactor/oxidation wetland passive treatment system that could produce methylmercury. Little technical information is provided on the active or the passive treatment methods, and pilot-scale testing has not been conducted. Active mine water treatment is only proposed for Alternative 2 in the DEIS, and no details on treatment or the need for perpetual treatment are provided for the other alternatives.

Shortcomings in Plans

- Adaptive management and development rock management plans do not currently exist. These plans need to be created as part of a Supplemental DEIS and included as appendices. A water quality management plan does currently exist, but it is not part of the DEIS or its appendices. Results from water quality predictions and geochemical characterization should feed into all three plans to create numeric thresholds, to help select effective mitigation measures, and to guide management of water and wastes during and after mining.

1. Introduction

The comments contained in this technical memorandum address water quality conditions, predictions, and effects related to development of the Stibnite Gold Project in central Idaho. My comments are submitted on behalf of the Idaho Conservation League (an Idaho-based nonprofit organization), Earthworks (a Washington DC-based nonprofit organization), and Judy Anderson (a private citizen). The comments are in response to the Draft Environmental Impact Statement (DEIS) released by the Forest Service in August 2020 (USDA Forest Service, 2020) and numerous related reports referenced in the DEIS. Additional information is also cited from published articles, environmental standards and criteria, and other technical information. All sources cited in my comments are listed in the References section (Section 9) of the memorandum.

The memorandum is organized in the following five sections (Section 2 through 6): existing and baseline water quality conditions and inclusion in modeling; water quality and geochemical modeling; perpetual water treatment; shortcomings in plans; and a listing of missing water quality geochemical data and information. A summary of my qualifications is contained in Section 7, and the references cited are contained in Section 8.

2. Existing and Baseline Water Quality Conditions and Inclusion in Modeling

Existing surface water, groundwater, and seep water quality conditions are presented in the DEIS (Section 3.9), SRK Consulting (2017a and 2018), Midas Gold (2019), and other reports and publications. SRK Consulting (2017a) includes a helpful table showing upstream legacy activities at water quality prediction nodes (Table 1) and maps with all historical mine features and all surface water, groundwater, and seep monitoring locations (Figure 1 to 3).

2.1 Baseline water quality and natural mineralization

Existing water quality has been affected by extensive historical gold, silver, antimony, and tungsten mining and processing activities (DEIS, p. ES-2) that ended approximately 25 years ago (Midas Gold, 2019). The DEIS claims that existing water quality has also been affected by natural mineralization (p. 3.9-18, 3.9-49), but no information is presented in the DEIS or other documents to quantify natural mineralization inputs in surface water, groundwater, or seeps. Some seeps are referred to as "natural," but the basis for this designation is not presented. The DEIS cites to Baldwin and Etheridge (2019) to support the natural mineralization claim, but these authors specifically state "…identification and characterization of the unmined mineralized areas was not the focus of this work." Inputs from natural mineralization cannot reasonably be remediated, and it is therefore important that natural background/baseline water quality absent historical mining impacts be carefully evaluated and the current effects of natural mineralization on water quality be assessed and presented in a Supplemental DEIS.

2.2 Surface water

Overall mean concentrations of antimony, arsenic, and mercury in surface water are presented in Section 3.9.3.1.1.2 of the DEIS. A better summary of minimum, maximum, mean, and median concentrations for these analytes is included in Midas Gold (2019), Tables 4-8 for surface water.[1] Mean antimony concentrations exceed the water quality standard (5.2 μg/L) at most locations in the East Fork of the South Fork of the Salmon River (EFSFSR) and at two locations in Meadow Creek and Sugar Creek. Minimum and mean arsenic concentrations exceed the water quality standard (10 μg/L) at more locations in all three drainages. Mean mercury concentrations exceed the water quality standard (12 ng/L) at two locations in Sugar Creek and nearly all locations in the EFSFSR. Table 4-6 in Midas Gold (2019) shows that 772 samples were analyzed for As(III).

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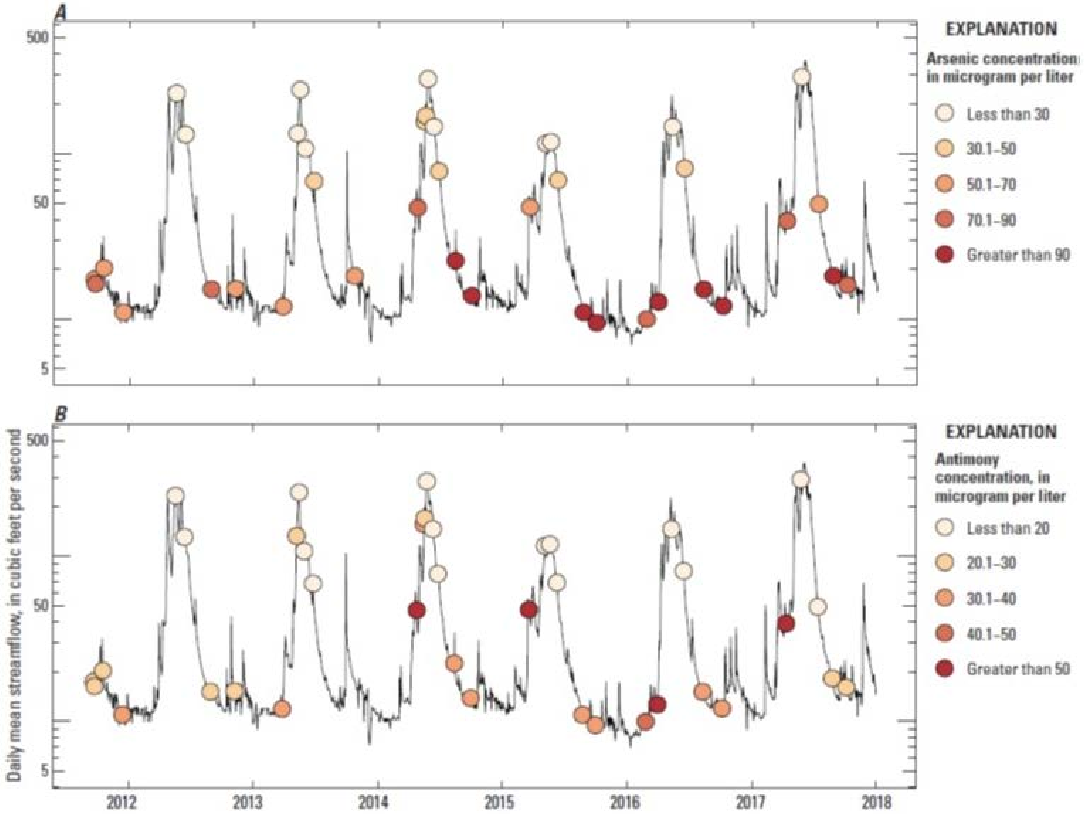
[1] Note that the antimony and arsenic standards are switched in this and other tables in the document.

One of the most distinctive features of site surface water quality is the temporal variability in concentrations associated with stream hydrographs. At sites affected by mine contaminants with notable temporal variability, it is important to know the maximum concentrations because they could cause acute, short-term toxicity to aquatic biota, depending on the element and the maximum concentrations. Knowledge of maximum concentrations is also important for assessing the effectiveness of cleanup and mitigation measures. Surface water monitoring is often not frequent enough or well-timed with the snowmelt hydrograph to identify temporal changes and maximum concentrations. As shown in Figure 1a, some samples associated with the rising limb of the snowmelt hydrograph (March – April, depending on the year) were collected, and those samples had the highest antimony concentrations and had high arsenic concentrations. Figure 1b shows conductivity values, which can be measured more frequently in, for example, automatic samplers, and obvious peaks in conductivity also occur on the rising limb of the snowmelt hydrograph.

Because the SWWC model is based on annual average conditions, these temporal changes are not well represented in the model. A “patch job” was done using monthly Seasonal Adjustment Factors (SAFs) to account for the temporal variability, but these adjustments only operate on a monthly level and would not capture peaks that could occur over shorter timeframes (see Section 5.2.2 of this memorandum). The same monthly adjustment factor is used for all years, regardless of interannual peak or other seasonal flow variability. The SWWC model report (SRK Consulting, 2018) states that concentrations were typically lowest during the spring snowmelt when flow rates were highest, and most exceedances occurred during low flow conditions. Figures 1 A and B show that antimony and specific conductance values are often higher during rising flows than high or low flows. Arsenic concentrations are highest during low flow, but not enough sampling was conducted during the rising limb; specific conductance was measured much more often. The high values for specific conductance during the rising limb are an indication that more sampling would have revealed higher contaminant concentrations during the rising limb.

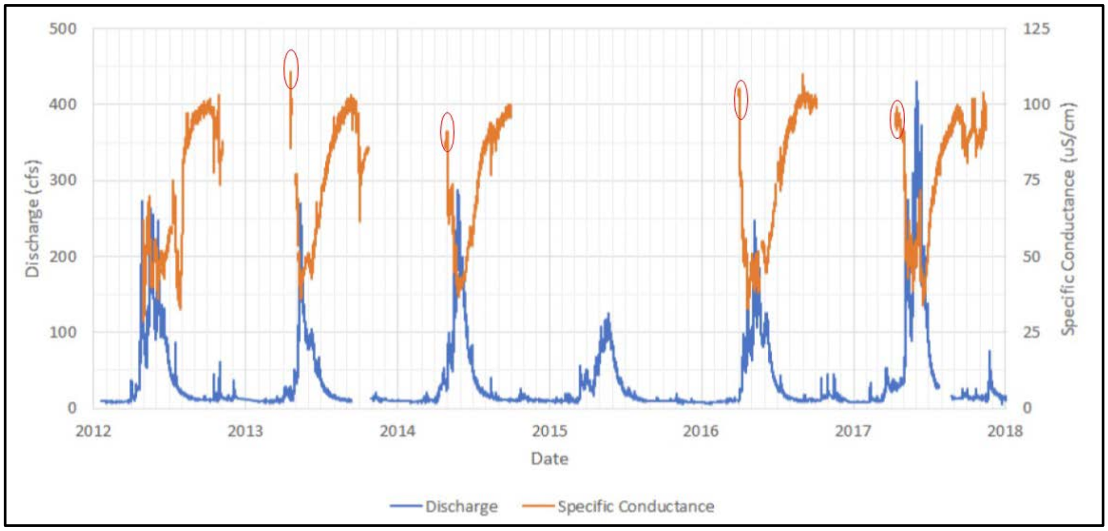
2.3 Groundwater

Existing groundwater concentrations are presented in Section 3.9 of the DEIS and are summarized and presented in more detail in HDR (2016) and Midas Gold (2019). Arsenic speciation was determined in many groundwater samples, including 323 measurements of As(III) in bedrock groundwater and 489 measurements in alluvial groundwater samples (Midas Gold 2019, Tables 5-4 and 5-3). Groundwater quality has been monitored in alluvial and bedrock aquifer locations across the site. Sample IDs with an A indicate alluvial wells, while sample IDs with a B indicate bedrock wells.



Source: Baldwin and Etheridge, 2019. Figure 4.

Figure 1a. Variability in (A) arsenic and (B) antimony concentrations with streamflow, 2012 to 2018 at USGS location EF3, East Fork South Fork Salmon River upstream of Sugar Creek. Higher arsenic (50->90 μg/L) and highest antimony concentrations (>50 μg/L) are associated with the rising limb of the snowmelt hydrograph.



Source: SRK 2018, Appendix D, Graph 1.

Figure 1b. Temporal variability in specific conductance as a function of stream discharge, USGS Gage 13311000, 2012-2018, EFSFSR near YP-SR-10, downstream of Meadow Creek confluence and upstream of Garnet Creek. Red circles show peaks associated with the rising limb of the snowmelt hydrograph.

Groundwater quality exceeds relevant standards for arsenic and antimony in many locations, but mercury concentrations in general do not exceed the groundwater standard. However, the groundwater standard for mercury is 2 μg/L (2,000 ng/L), while the surface water standard is 12 ng/L. As noted in Zamzow (2020), all groundwater samples had total mercury concentrations exceeding the surface water standards, and wells MWH-A04 and MWH-A05 (alluvial wells near historic heap leach pads in the Meadow Creek valley) had dissolved mercury concentrations exceeding the surface water standard. The higher mercury concentrations in these wells could be related to strong complexing of mercury with cyanide, which would cause the release of mercury to groundwater. Summary statistics for antimony, arsenic, and mercury are presented in Midas Gold (2019), Table 5.5. Graphs showing groundwater quality over time are presented in Midas Gold (2019), Figures 5-8 to 5-18. Some notable results in Midas Gold (2019) are presented as follows:

- Upper Meadow Creek Valley: In bedrock well MWH-B01 all samples exceed for arsenic and one samples exceeds for antimony. Alluvial well MWH-A01 has only one exceedance for arsenic (Fig. 5-8). Concentrations are higher in bedrock at this location (Fig. 5-9).
- Meadow Creek Valley near middle of airstrip: Alluvial well MWH-A05 and bedrock well MWH-B05 (Figs. 5-12 and 5-13) have large exceedances of arsenic and antimony. Arsenic concentrations are higher in the alluvial well. Antimony concentrations are in a somewhat similar range in alluvial and bedrock wells, but concentrations decrease over time in the alluvial well and increase over time in the bedrock well.
- EFSFSR Valley near Garnet Creek: Alluvial well MWH-A09 (Fig. 5-14) and bedrock well MWH-B09 (Fig. 5-15) also have large exceedances of arsenic and antimony (there are more exceedances and higher concentrations of both in the alluvial well).
- EFSFSR valley near Scout Prospect: Alluvial and bedrock wells (MWH-A12 and MWH-B12) have large arsenic and antimony exceedances. Arsenic concentrations are higher in the bedrock well, while antimony concentrations are similar in both wells.
- EFSFSR valley near Homestake Road: Alluvial well MWH-A-19 has the highest arsenic concentrations of any well (~800-900+ μg/L), and there are also some antimony exceedances. There appears to be no bedrock well at this location.

No groundwater plume map for existing conditions is presented in any document. A plume map with legacy mine features should be created to help produce a conceptual model of existing and future groundwater contamination.

2.4 Adit seeps and seeps

Existing seep concentrations are presented in Section 3.9 of the DEIS and are summarized and presented in more detail in HDR (2017) and Midas Gold (2019). Arsenic speciation was determined in 449 seep samples (Midas Gold, 2019, Table 4-7). Summary statistics for arsenic, antimony, and mercury concentrations in seeps are presented in Table 4-9 (Midas Gold, 2019). The seeps are split into adit seeps and seeps, but the distinctions between the two are not discussed. The location descriptions in Table 4-5 in Midas Gold (2019) suggest that most seeps are related to historical mining activity, although some appear to be related to faults and other features.

Concentrations of antimony, arsenic, and mercury are very high at most seep locations (Midas Gold, 2019, Table 4-9). Even minimum concentrations exceed the standards at nearly all locations for antimony and arsenic. Mean mercury concentrations also exceed the surface water quality standard at nearly all seep sampling locations. These results indicate that existing seep water has been severely affected by historical mining activity.

Unlike most surface water and groundwater sampling results, concentrations of cadmium, copper, cobalt, total cyanide, lead, manganese, nickel, selenium, and vanadium are elevated in some seep samples (Midas Gold, 2019, Table 4-7). The potential transport of these constituents to groundwater and surface water is not discussed in the DEIS or elsewhere, although seeps and springs from unconsolidated deposits are said to sometimes flow only a short distance over the surface before infiltrating back into the unconsolidated materials (SRK Consulting, 2018, p. 16). The SWWC model assumes that any existing seeps in the footprint of proposed future pits will be removed and no longer exist during post-closure. However, these will still likely feed into the pit and affect pit lake water quality during and after mining. Certain seeps are included in the SWWC model during the operational period for five or six years and others are included for the entire operational modeling period (SRK Consulting, 2018, Table 8-2). During the post-closure period, all seeps that were not excluded after five or six years are included in the SWWC model (SRK Consulting, 2018, Table 8-3).

No conceptual models exist to help explain the fate and transport of contaminants from seeps to groundwater and surface water. It is not clear how the SWWC model handles inputs of seeps to groundwater or surface water, but because the concentrations are so high in many seeps, this information is needed for the Supplemental DEIS.

2.5 Stream sediment samples

The chemistry of stream sediment samples was not discussed at all in the DEIS. Limited sediment quality data from five stream locations in June 2016 were presented in Holloway et al. (2017) (Table 1). The results show that sediment concentrations at three of five locations for arsenic and four of five locations for mercury exceeded Canadian sediment quality guidelines for the protection of aquatic life (CCMS, 2014). Sediment arsenic concentrations exceeded the probable effects level (PEL) by up to 400 times, and sediment mercury concentrations exceed the PEL by up to 50 times. The food chain/dietary pathway for arsenic has been shown to adversely affect salmonids in laboratory experiments and using stream sediment from mined areas in Montana and Idaho (Hansen et al., 2004; Kiser et al., 2010; Woodward et al., 1994 and 1995), yet it was completely ignored in the DEIS.

No information is provided on the effects of temporal variability in arsenic, antimony, mercury, or other contaminant concentrations on the fish life cycle. Very general information on the effects of chemical contaminants on aquatic biota is contained in the DEIS (Section 3.12, p. 3.12-91), and general information about the fish life cycle is included in the DEIS (Section 3.12, e.g., Table 3.12-1), but the temporal variability in contaminant concentrations is never linked to fish life stages.

Excluding stream sediment from the contaminant pathway analysis is a major, fundamental problem with the conceptual model for the site. More sediment sampling is needed, and the results should be included in the design of conceptual models, mitigation and cleanup measures, and modeling efforts in a Supplemental DEIS.

Table 1. Water and sediment arsenic and mercury, East Fork South Fork Salmon River gage sites, June 2016, compared to Idaho surface water quality criteria and Canadian sediment quality guidelines.

USGS gaging station	pH	As(T),F µg/L	As(III),F µg/L	Hg,F ng/L	THg,U ng/L	As,S µg/g	Hg,S µg/g
EFSFSR above Meadow Ck	6.78	9.0	0.6	6.37	8.18	5.20	1.14
Meadow Ck	7.51	1.1	<0.5	5.26	7.27	8.48	0.04
EFSFSR below Meadow Ck	6.95	23.6	2.8	3.44	5.33	42.8	2.42
EFSFSR above Sugar Ck	7.85	64.1	13.2	3.00	4.24	7176	1.31
Sugar Ck	7.49	10.0	<0.5	9.38	29.1	51.8	26.4
Water quality standards[1]		10.0			12.0		
Canadian sediment quality criteria[2]							
ISQG						5.9	0.170
PEL						17	0.486

Source for water and sediment quality data: Holloway et al., 2017, Table 1.
F filtered (0.45 μM), U unfiltered, S sediment
EFSFSR East Fork South Fork Salmon River; As(T) total dissolved arsenic; THg total (unfiltered) mercury
1 USDA Forest Service, 2020. Table 3.9-2
2 CCME, 2014. ISQG Interim Sediment Quality Criteria; PEL Probable Effects Level
Yellow highlight indicates exceedence of water quality standard or PEL

3. Water Quality and Geochemical Modeling

3.1 Conceptual models

As noted in Zamzow (2020), many conceptual site models are missing, including very basic conceptual models for water and contaminant movement under existing and post-closure conditions. Conceptual models are needed to help define the modeling system and must include all geochemical and hydrologic processes that could alter future water quality for constituents of interest (NDEP, 2018). With the exception of a conceptual model for the tailings storage facility (TSF; DEIS, Figure 4.9-7), no geochemical/water quality conceptual models are presented in the DEIS.

Conceptual models are presented in SRK Consulting (2018) for the DRSFs (Figures 4-1, 4-2, 4-3). However, the only processes depicted are those that would *reduce* contaminant concentrations, including mineral precipitation and contaminant adsorption. Leaching of contaminants from waste rock and formation and dissolution of soluble salts is not depicted or described. The same is true of conceptual models for the pits (SRK Consulting, 2018, Figures 7-1, 7-2, 7-3). The dissolution of soluble salts is described in the pit lake modeling text (p. 119), but the wrong inputs are used to characterize their effects in the pit models. Instead of using first-flush release rates or short-term leach test leachate concentrations, the models use average, steady-state rates from limited HCT weeks.

Importantly, no conceptual model is described or depicted that includes stream sediment, which as explained in Section 3.4 of this memorandum is an important but completely ignored source of contaminant loading to fish. A conceptual model showing the food chain/dietary pathway for impacts to fish from consuming macroinvertebrates residing in contaminated stream sediment is sorely needed.

No conceptual models exist for the fate and transport of the high concentrations of contaminants of concern (COCs) other than arsenic, antimony, and mercury that are found in many seep samples. These COCs include cadmium, copper, cobalt, total cyanide, lead, manganese, nickel, selenium, and vanadium. What happens to these high concentrations as seep water moves into groundwater and surface water? Improved conceptual models are needed for all elements of the geochemical and SWWC modeling effort and should be included in a Supplemental DEIS.

3.2 Underlying Geochemical Characterization and Interpretation Problems that Affect the Reliability of the Models

The primary sources for geochemical characterization information are SRK Consulting (2017b) and the updated geochemical characterization report (SRK Consulting, 2019a). Very little information on geochemical characterization is included in the DEIS; what little information exists is in DEIS Section 4.9. As noted by Zamzow (2020), a geochemical appendix to the DEIS is needed.

The following geochemical characterization information (or lack thereof) and interpretation of geochemical characterization results will affect the reliability of geochemical and water quality models that depend on the information. In general, these issues result in the underestimation of predicted groundwater and surface water concentrations.

3.2.1 Sample representation

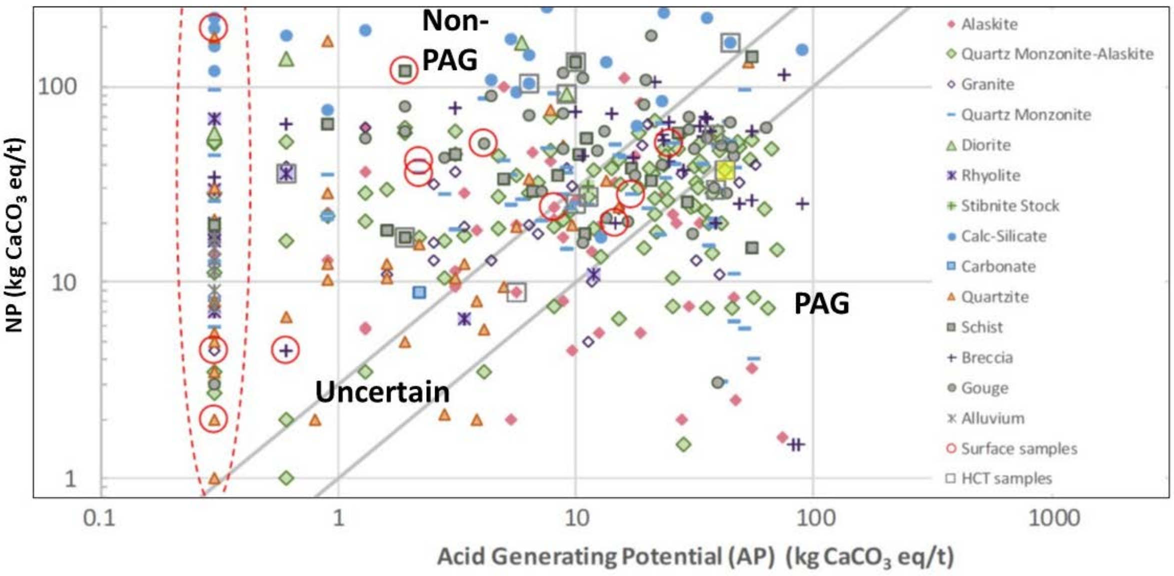
Geochemical/geoenvironmental test units were not defined

The geochemical testing program assumed that leaching characteristics are the same across a given lithology. As noted below, the geochemical testing results contradict this assumption. Geochemical test units should have been identified early in the program and used to select samples for static and kinetic testing before the program begins (Maest et al., 2005; Price, 2009). Many practitioners and guidelines used by the mining industry emphasize the importance of identifying geochemical test units that represent distinct compositional characteristics that can affect leachate quality.

- INAP (2009, Section 4.3.2.1) Sample Selection “Compositional Representation – Sample selection should include all major material types and cover the range of pertinent characteristics for each material type (e.g., pH, carbonate, sulphur, and neutralizing potential content). Personnel tasked with sample selection must be familiar with the geological characteristics of the deposit, including rock types, fracture patterns, weathering, alteration, and mineralization."

- NDEP (2018, p. 6): "A model that assumes geochemical homogeneity among lithologic units, or zones of alteration, mineralization, or weathering rather than documenting the actual range of variation, is unacceptable because it fails to demonstrate that the characterization is representative as required in NAC 445A.396. In the case of geochemical modeling, adequate characterization data are required to illustrate the full range in geochemical characteristics representative of each major lithologic, alteration, and mineralization unit and zone across the site that is identifiable and discrete."
- NDEP (2018, p. 5): "In the experience of the Division, it is common for geochemical modeling exercises to reach advanced stages without adequate representative characterization data…For a pit lake model, these sources would include all distinct groundwater aquifers and a thorough sampling of the spatial distribution of all lithologic, alteration, and mineralization units that will remain in the immediate vicinity of the ultimate pit walls. The geochemical characteristics of interest for these distinct potential solute sources could include mineralogy, leachable quantities of constituents (e.g., determined using the meteoric water mobility procedure [MWMP] and humidity cell tests [HCTs]), and acid-base accounting (ABA)."
- Parbhakar-Fox and Lottermoser (2015, p. 113): "For ARD prediction, this concept must be adapted to define geoenvironmental units (previously referred to as mesotextural groups in Parbhakar-Fox et al., 2011). Such units must be defined based on similar textural, mineralogical and chemical properties from which a similar ARD forming and metal leaching potential can be predicted. Furthermore, identifying geoenvironmental units must not be restricted to ore, but must extend to waste. Through defining geoenvironmental units, representative samples can be chosen based on a sampling interval dictated by site-based definitions and abundances of local rock types." And p. 120: "…geoenvironmental units should be identified and ARD risk-domains of the deposit should be established."
- Price (2009): Price refers to the units as "geological units", which reflects the separation of geologic *materials* into geological *units* based on geochemical alteration, weathering, and other factors. (p. 6-9).

Figure 2 shows that categorizing waste rock based solely on lithology is inadequate. Notice that alaskite, quartz monzonite-alaskite, quartz monzonite, schist, gouge, and breccia samples span from non-PAG to Uncertain to PAG. This is a good indication that more divisions should have been made within the lithologic units (geochemical or geoenvironmental test units) to ensure the units had at least somewhat similar acid-generation and contaminant leaching potential. Quartz monzonite is the most common rock type in the Yellow Pine pit (25%), the Hangar Flats Pit (54%), and the Fiddle DRSF (64%) (DEIS, Tables 4.9-1, 4.9-3).



Source: SRK Consulting (2017b) Figure 3-6.

Figure 2. NP vs. AP results for different lithologies showing surface and selected HCT samples. The one PAG sample selected to represent all PAG rock in SRK Consulting's water quality models (SRK Consulting, 2018) is shown in yellow highlight. The only other PAG sample selected for HCT testing is near this sample (shown in a square) but was excluded from use in the SWWC models.

Mineralogy and latent reaction potential

Mineralogy is used to identify geoenvironmental test units. The presence and characteristics of known acid-producing and acid-neutralizing minerals can affect a sample's leaching ability. Appendix D of the Baseline Geochemical Characterization Report (SRK Consulting, 2017b, Table 3-22) contains information on the mineralogy of some of the HCT samples before and after kinetic testing, and Appendix C contains information on mineralogy of the four Spent Ore Disposal Area (SODA) samples. Mineralogy results are also summarized in Table 3-22 for the HCT samples and Table 3-31 for the SODA samples (SRK Consulting, 2017b). Stibnite was only identified in one HCT sample analyzed for mineralogy but in two of the four SODA spent ore samples and four of nine legacy Bradley waste rock samples (SRK Consulting, 2017b, Appendix D). Because no stibnite was identified in the core HCT samples, it brings into question how representative they are of waste rock samples for the proposed mine. Stibnite weathers rapidly to form secondary stibnite-containing minerals (Klimko et al., 2011) and can produce acid when oxidized by oxygen (Plumlee, 1999). The kinetics of antimony oxidation are enhanced by the presence of calcium and magnesium in basic solutions (Biver and Shotyk, 2012), and these cations are present in all water samples tested. Antimony has been shown to be bacterially oxidized in mine sediments and produce secondary antimony oxide minerals (Terry et al., 2015). The release of COCs from waste rock and ore samples is often attributed to dissolution of soluble salts (e.g., for the Yellow Pine Pit, SRK Consulting, 2017b, p. 3-22). Some secondary minerals related to the breakdown of stibnite (schafarzikite, senarmontite and cervantite – antimony-containing oxide minerals) were identified in samples from the Bradley Dumps (SRK Consulting, 2017b, p. 3-132) but not in any of the HCT samples analyzed for mineralogy (SRK Consulting, 2017b, Appendix D). Much is made of the "encapsulation" of sulfide minerals throughout SRK Consulting (2017b). The encapsulation will lengthen the time to release of acid from acid-producing sulfide minerals, but the encapsulation is more common in the larger sized particles, while sulfides in more fine-grained particles were well-liberated (available for reaction) (SRK Consulting, 2017b, Appendix D). Even though the sulfide minerals were generally unreacted in the HCT samples, the mineralogy appendix states that their presence in HCT samples "points to latent reaction potential still present within the HCT samples." The HCT procedure calls for reducing the grain size to <6.3 mm (Maest and Nordstrom, 2017), which includes all particle sizes from gravel to clay. The first-flush concentration peaks for several parameters could indicate that the finer-grained particles were releasing arsenic and antimony (and other COCs), even in the fresh core samples. The focus in the geochemistry report is on the abundance of NP remaining at the end of the HCTs (between 79 to 99%; SRK Consulting, 2017b, p. iii), but abundant acid-generation potential also remains at the end of the tests. The latent potential for reaction of the sulfides indicates that the finer-sized particles in the DRSFs and pits will produce acid and leach other contaminants faster and that the larger particles will be able to leach COCs for a very long period of time.

Only one HCT sample was used to simulate all PAG rock

The HCT leachate results from a single PAG rock sample (HC-14, a quartz monzonite-alaskite sample; SRK Consulting, 2018b) was used to represent all PAG rocks across all pits and DRAFs. This sample did not produce acid, it has some of the highest NP values of all the PAG samples, and it is very close to the Uncertain range (see Figure 2). Therefore, it can barely be considered a PAG sample. In fact, using NNP[2] rather than NPR for ABA interpretation, this sample and the other PAG sample selected (HC-3; see Figure 2) were in the uncertain category (SRK Consulting, 2017b, Table 3-12). Therefore, the selection of PAG samples for water quality modeling was not representative of the range or even middle tendency of all the PAG samples. A true representative sampling for the HCTs would have included other PAG samples with lower NP values and a range of AP values. Some of the lack of representation was remedied in Phase 2 geochemical testing (SRK Consulting, 2019a), but none of those samples produced acid in the shorter testing period. As noted in the ERM (2019) review of SRK Consulting's modeling effort, and in the DEIS, Section 4.9.8, other samples did produce acid but were not used to calculate the PAG source term. ERM's conclusion is that there is a risk that the SWWC model predictions do not adequately capture PAG contributions to surface water quality. The issue is discussed further in Section 5.2.1 of this memorandum.

In contrast, several of the surface samples[3] did produce acid in static testing results. Several surface samples had low paste pH values and NAG pH values <4.5 (SRK Consulting, 2017b, Appendix A, Figure 3-24, Figure 3-37). Note that SRK Consulting (2017b) uses a NAG pH cut-off value of 4 as indicating a sample is net acid generating (i.e. any sample with a pH values <4 is NAG). The GARD Guide (INAP, 2009, Section 5.4.10) uses a cut-off pH of 4.5, which means that more samples would be considered net acid generating than identified. All surface samples with

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[2] NNP = net neutralization potential, or NP-AP.

[3] Surface samples were collected from existing waste rock dumps and pit walls from previous mining operations and have been exposed to oxygen for over 50 years (SRK Consulting, 2017b)

NAG pH values <4.5 had low sulfide S percentages (< 0.1% sulfide sulfur, SRK Consulting, 2017b, Figure 3-31), indicating that the acidic conditions were produced by dissolving soluble salts. The following waste rock lithologies had some samples with NAG pH values ≤4.5: alaskite, quartz monzonite, granite, quartz monzonite-alaskite, quartzite, schist, and gouge (SRK Consulting, 2017b, Table 3-9).

ABA results for the pulp samples[4] showed that many had high % pyritic sulfur values (>0.3%), low NPR values, and NAG pH values <4.5, indicating they are PAG (SRK Consulting, 2017b, Appendix A, pdf pgs 248-258). Many of the core samples are also considered PAG based on their static test results (SRK Consulting, 2017b, Appendix A, pdf pgs. 206-211 and 278). The short-term leach testing results using the meteoric water mobility procedure (MWMP) from acid-generating surface, pulp, or core samples – or even first flush results from HCT samples – could have been used to represent PAG development rock samples. The historical surface samples were characterized “to provide an analogue of likely future geochemical behavior,” (Midas Gold, 2018) yet the results were not used as source terms for the modeling efforts.

3.2.2 Use of averages and underestimating variability

Use of average values as model inputs and average steady-state release rates

The SRK Consulting (2019b) sensitivity analysis report did not evaluate the most important model input for predicted water quality: concentrations and rates. Instead the report evaluated more minor effects related to the parameters used for scaling (% fines, % of rock contacted, temperature) and the ABA cut-off value. The variability in source term concentrations should have been evaluated in a sensitivity analysis. According to Price (2009, p. 4-8), “Unlike ore analysis, where the concern is with the average and whole sample composition, drainage chemistry depends on the range in composition, and composition of different size fractions (e.g. < 2 mm fraction of waste rock or tailings slimes and sand that segregate during deposition).” The SRK Consulting (2019b) sensitivity analysis evaluated the effects of varying the percent of fines in DRSFs, but it is unclear what was used to come to the conclusion that increasing the percentage of fines would increase arsenic, antimony, and mercury concentrations (p. 9). No information on the chemistry of different size fractions is presented in SRK Consulting (2017b) except for calcium in different size fractions of SODA material. Therefore, the effects of using a range in drainage composition and in the composition of different size fractions was not accomplished.

Average, steady-state contaminant release rates are used as the basis for source terms for the DRSFs and the pit walls (see SRK Consulting, 2018, Tables 4-1, 5-1, 7-1), sometimes using multiple samples from the same lithology. However, no information is provided on how the steady-state rates were summarized to create the single rates for each lithology. Did they use the 95th percentile of means for samples from each lithology, the median, the mean? The statistical approach should be discussed.

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[4] Pulps are from the exploration program representative of ore and waste to improve the spatial distribution of the dataset.

The SWWC existing conditions model used Seasonal Adjustment Factors to derive average monthly stream concentrations (SRK Consulting, 2018). Annual or even monthly stream concentrations are not the important measure for fish toxicity. According to results from existing conditions (see Section 3.2 of this memorandum), concentrations often peak during the rising limb of the snowmelt hydrograph, during low flows, and during summer thunderstorms. The peaks associated with early snowmelt and summer thunderstorms can last a matter of days or hours and potentially create acutely toxic conditions for aquatic biota (e.g., Nordstrom, 2009; Maest et al., 2004). Therefore, monthly averages are not helpful for defining critical exposure times for fish.

Average annual temperatures were used to scale the average steady-state release rates from laboratory to supposed field conditions (SRK Consulting, 2018). The scaling used a laboratory temperature of 25°C and an average annual temperature of 2.6°C (SRK Consulting, 2018, p. 54). Different temperatures were considered in the sensitivity analysis (SRK Consulting, 2019b), but the results were not included in a revised SWWC model and incorporated into the DEIS.

Excluding samples with higher release rates

The variability in arsenic concentrations is not well represented in the modeling effort. Arsenic is one of if not the most important water quality concern at the site. The results from two HCT samples from Phase 1 were excluded from use in the model because they were higher than the 95th percentile in total (solid) arsenic concentrations (HC-3 and HC-8). However, it appears that this decision was made only after the HCT leachate results were examined. HC-3 is a quartz monzonite sample from Hangar Flats and HC-8 is a carbonate from the West End. HC-8 was replaced with a sample that reflected "average" arsenic concentrations (25th to 75th percentile; SRK Consulting, 2019a, Table 3-2). HC-8 had high release rates for sulfate, arsenic, and antimony (SRK Consulting, 2017b, Figures 3-48, 3-51, 3-52), and HC-3, which was the only other PAG sample included in Phase 1 testing, had the highest release rate for arsenic after week 40 (SRK Consulting, 2017b, Figure 3-51) and about the highest antimony release rates as well (SRK Consulting, 2017b, Figure 3-52). Therefore, excluding HC-3 lowers input arsenic values for PAG wastes in the models. Excluding HC-8 lowers the source term values for the West End DRSF. The following statements by Price (2009) demonstrate the shortcomings of this decision:

- "Sufficient numbers of samples should be taken to accurately characterize the variability and central tendency (e.g., average, median and 10th and 90th percentiles) of the different waste materials, project components and geologic units. This includes characterization of localized areas of material with differences in physical, geochemical, mineralogical, weathering and leaching conditions that alter drainage chemistry." (p. 8-6)
- "It is critical that kinetic tests are run on the samples of the project component that will produce problematic drainage chemistry….Just as it is important that static testing covers the entire range in geological, geochemical, physical, weathering and leaching conditions, it is important that kinetic testing cover the entire range for the portion of the project component that is a concern." (p. 9-12)
- "Only a small portion of the material may be sufficient to produce significant ARD or metal leaching. Consequently, variability and distribution of parameters such as NPR (NP/AP) and metal concentrations are typically more important than central tendency or average compositions." (p. 21-12)

Excluding samples with higher release rates will underestimate concentrations used as source terms in the models. Results from HC-3 and HC-8 should have been included to better represent the range in arsenic and antimony concentrations. SRK Consulting (2019b) did not evaluate the effect of using ranges in concentrations or rates in their sensitivity analysis.

Steady-state assumptions

"Steady-state" conditions related to HCT release rates are defined in SRK Consulting (2018, p. 28) as "stable effluent sulfate concentrations (within ±5 milligrams per kilogram per week [mg/kg/week])."[5] In the same report, SRK Consulting (p. 46) makes the following statement: "Arsenic and antimony release rates in the HCTs are strongly correlated to sulfate release (Figure 4-15 and Figure 4-16), therefore the attainment of steady-state release for sulfate occurs at the same rate and at approximately the same time as for arsenic and antimony." The trends in arsenic and antimony release rates are not at all well correlated with sulfate release rates, and the rates are not the same. As shown in Figure 3, while sulfate release rates for the one PAG sample used in site modeling (HC-14) are relatively steady in the timeframe selected by SRK Consulting (see SRK Consulting, 2018, Table 4-5 for weeks and samples chosen to represent model source terms), arsenic and antimony release rates are not. Using this time period also eliminates early-week first-flush releases, which should be used to represent the variability in release rates. Table 4-6 (SRK Consulting, 2018) lists release rates used for the HCT development rock samples that were weighted "to reflect the lithologies contained within each block modeled solid." However, the document does not clearly or specifically describe how they arrived at the rates in Table 4-6. Figures 4-13 and 4-14 show the single source terms for Yellow Pine and Hangar Flats intrusives, again showing that the variability in arsenic and sulfide values are not represented.

3.2.3 Overestimating neutralization potential

The ABA test results are used to determine whether samples are PAG or non-PAG. The SRK Consulting (2017b) geochemical characterization report has many confusing discussions about which methods were used to determine NP. Some efforts were made to ensure NP was not overestimated, but generally the NP values of most samples are overestimated because the method also counts silicates as contributing to NP (which they do not substantially) or iron carbonates in many cases were counted as contributing to NP (they can actually lower the pH of a sample).

SRK Consulting (2017b, p. 3-12) states that the modified Sobek method was used, but they give an incorrect reference for it (Sobek et al., 1978). This is the *original* Sobek method, which is known to overpredict NP (Maest et al., 2005; INAP, 2009). Even the modified Sobek method can overpredict NP (SRK Consulting, 2017b, p. 3-16). The SODA samples were analyzed using another method, the Nevada modified Sobek method (NDEP, 2015; SRK Consulting, 2017b, p. 3-99), but the "regular" modified Sobek appears to have been used for all other samples (p. 3-15). Another method used was total inorganic carbon (TIC), which will count iron carbonates that do not

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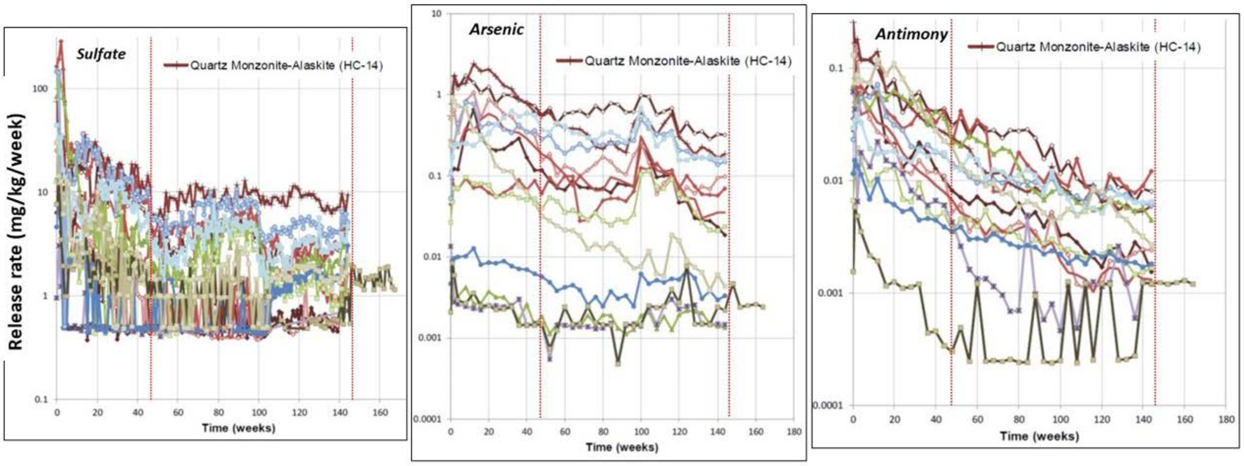
[5] Note that the units reflect rates, not concentrations.



Source: SRK Consulting, 2017b, Figures 3-48, 3-51, 3-52.

Figure 3. Release rate trends for sulfate, arsenic, and antimony in HCT samples. The vertical red lines indicate the weeks chosen for the one PAG sample used in modeling, HC-14, as “steady state” conditions. While the sample has fairly steady sulfate release rates during the selected weeks, arsenic and antimony rates are not steady. Using average rates from these weeks ignores much higher release rates from early HCT week that represent first-flush conditions, as shown in all three graphs. Note that all y axes are logarithmic.

contribute to NP; the siderite (an iron carbonate mineral) correction method was applied to only a subset of the samples (SRK Consulting, 2017b, p. 3-16). Mineralogic analysis shows as much as 50% of total carbonate is siderite or ankerite in limited samples (SRK Consulting, 2017b, p. 3-16). In other places, SRK Consulting (2017b, p. 3-37) states that the calculated NP from calcium + magnesium concentrations from multi-element analysis was used to estimate NP; they admit that this measure would overestimate true NP (because it will count Ca and Mg from silicates).

Using a realistic measure of NP is important because the way wastes are handled are in part dependent on the ABA results, which rely on NP measurements. A lower (less protective) site-specific NPR is proposed to be used to determine which rocks and samples are PAG (DEIS, p. 4.9-9), which uses NP measurements. No scientific basis for this lowering is presented, and it is especially problematic because of the overestimation of NP.

3.2.4 Ignoring COCs and results from legacy surface samples

The primary COCs at the Stibnite site are arsenic, antimony, and mercury. As noted in Section 3.3 of this memorandum, additional COCs[6] were detected, often at concentrations above water quality standards, in seeps. Results from the short-term leach test used on waste rock and ore samples (MWMP) also showed elevated concentrations of aluminum, antimony, arsenic, beryllium, cobalt, copper, iron, manganese, mercury, nitrate, sulfate, and total dissolved solids (TDS). The MWMP samples are not identified by location or lithology, so it is not possible to determine if the selected samples are representative. The MWMP results in SRK Consulting (2017b, Appendix A) have sample identifiers, but they are not linked to lithology or

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[6] Cadmium, copper, cobalt, total cyanide, lead, manganese, nickel, selenium, and vanadium.

location/source. The parameters listed above generally had higher average concentrations in surface vs. core samples, as shown in Table 2. Many of the elevated concentrations were in samples from alaskite and quartz monzonite, some of which are ore samples[7] but which are also common development rock lithologies proposed to be present in pit walls or placed in DRSFs and backfilled pits (DEIS, e.g., Tables 4.9-1, 4.9-3, 4.9-8). Most of the surface samples had low percent sulfide sulfur values (SRK Consulting, 2017b, Appendix A), which indicates that the elevated COC concentrations likely derive from dissolving soluble salts.

Table 2. Comparison of selected *average* MWMP results (mg/L; mercury in μg/L) for core and surface samples.

Parameter (most stringent WQ criterion)	**Core or Surface Sample**	**Alaskite**	**Quartz Monzonite**	**Granite**	**Quartz Monzonite-Alaskite**	**Rhyolite**	**Calc-Silicate**	**Carbonate**	**Quartzite**	**Schist**	**Breccia**	**Gouge**
pH, s.u. (6.5-9)	Core	6.74	6.84	6.78	6.80	6.75	6.83	6.96	6.81	6.59	7.23	6.82
	Surface	**3.81**	**5.24**	--	--	--	7.85	7.39	6.82	7.35	6.99	7.36
Antimony (0.0056)	Core	0.060	0.026	0.130	0.053	0.003	0.004	0.013	0.003	0.003	0.45	0.014
	Surface	0.012	**0.339**	--	--	--	0.005	**0.018**	0.005	0.005	**0.53**	**0.26**
Arsenic (0.01)	Core	0.26	0.37	2.60	0.45	0.008	0.01	0.16	0.02	0.01	0.35	0.02
	Surface	0.2	**0.91**	--	--	--	**0.05**	0.032	**0.10**	**0.02**	**2.20**	**0.08**
Beryllium (0.004)	Core	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.003	<0.001	<0.001
	Surface	**0.09**	**0.003**	--	--	--	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Cobalt (--)	Core	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Surface	**0.28**	**0.018**	--	--	--	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Copper (0.009)	Core	<0.05*	<0.05*	<0.05*	<0.05*	<0.05*	<0.05*	<0.05*	<0.05*	<0.05*	<0.05*	<0.05*
	Surface	**0.41**	**0.032**	--	--	--	<0.003	<0.003	0.0042	<0.003	**0.011**	0.003
Iron (0.3)	Core	<0.01	0.01	<0.01	0.011	<0.01	<0.01	<0.01	<0.01	0.03	0.01	<0.01
	Surface	**0.65**	**3.01**	--	--	--	<0.01	<0.01	**0.016**	<0.01	**0.02**	<0.01
Manganese (0.05)	Core	0.030	0.007	0.022	0.013	0.005	0.008	<0.005	0.006	0.015	0.21	<0.005
	Surface	**9.2**	**1.18**	--	--	--	<0.005	<0.005	<0.005	<0.005	0.12	**0.016**
Mercury (0.012)	Core	0.2	0.1	0.2	<0.1*	<0.1*	<0.1*	<0.1*	0.19	0.2	<0.1*	<0.1*
	Surface	<0.11*	**0.3**	--	--	--	<0.1*	**0.2**	<0.1*	<0.1*	<0.1*	<0.1*
Sulfate (250)	Core	27.4	5.7	18.0	6.7	1.0	4.8	4.7	2.3	1.0	150	1.2
	Surface	**1800**	**243**	--	--	--	**31**	**4.8**	2.1	**160**	**290**	**400**
TDS (500)	Core	63	37	64	36	50	47	52	39	23	280	29
	Surface	**2300**	**341**	--	--	--	**110**	**77**	**80**	**300**	**600**	**630**

Source: SRK 2017b, Tables 3-10 and 3-11.
WQ Water quality, TDS total dissolved solids, -- no analysis or no WQ criterion
s.u. Standard pH units; < Below laboratory reporting limit; * Laboratory reporting limit > most stringent water quality criterion
Yellow highlight: concentrations exceed most stringent water quality criterion
Bolded values: concentrations in surface samples > concentrations in core samples (or pH lower)

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[7] Ore grade is defined in SRK Consulting, 2017b, p. 3-1, and in the DEIS (p. 3.9-58) as a gold concentration greater than approximately 0.5 g/t. Midas Gold elected to report its gold sulfide cut-off grade at 0.75 g/t and its oxide cut-off grade at 0.45 g/t (M3 Engineering and Technology Company, 2019); material below this grade will be considered waste rock. However, ore grade will vary throughout a mine's life (thus changing the designation of waste vs ore), and the amount of gold does not necessarily correlate with the amount of sulfide, which is one of the controls on leaching chemistry. In fact, the surface samples have low sulfide percentages (SRK Consulting, 2017b, Appendix A).

3.3 Shortcomings of the Water Quality/Geochemical Models and Implications for Environmental Impacts

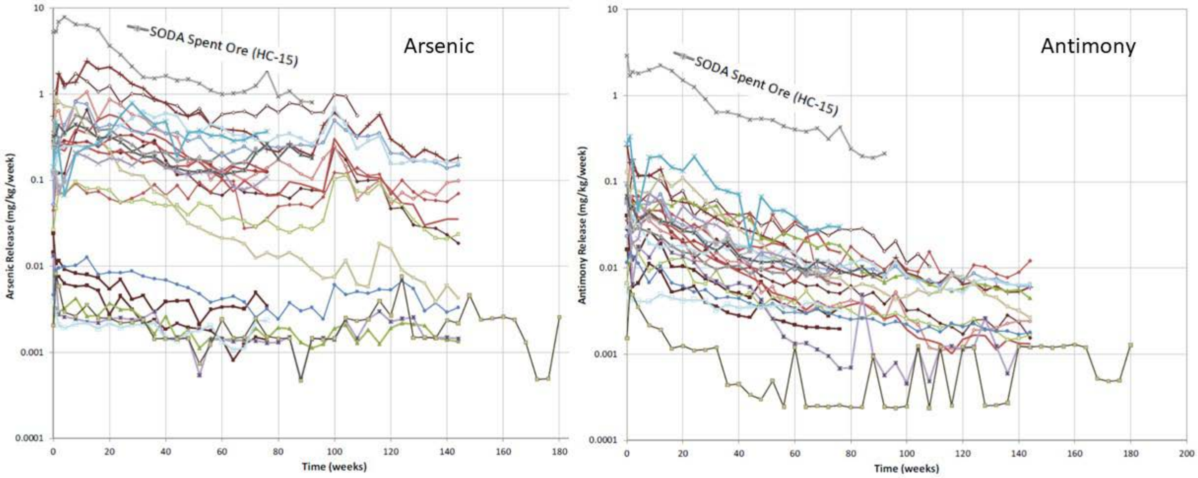
Some of the most important shortcomings of each of the geochemical models (for DRSFs, the TSF, and pit lakes) and the SWWC model are listed in this section. I have focused on shortcomings that will underestimate predicted impacts from the mine facilities to groundwater and streams or cause uncertainty in the predicted results from the models. Section 4.9 in the DEIS discusses the models, and Section 4.9.8 in the DEIS discusses uncertainty in the SWWC model. More detail on the models is provided in SRK Consulting (2018). ERM (2019) presents an independent review of groundwater and surface water models produced for the project related to for existing conditions and the proposed action. This section also highlights elements that were not evaluated in the models, such as the leaching of construction fill derived from development rock or legacy materials.

3.3.1 DRSF

- SODA spent ore HCTs (HC-15, -16, -17) were not used to estimate leaching rates, even though said these are good analogues for future mining and they are planned to be used for construction fill. Results from these samples could be used to represent the variability in releases from the DRSFs, but the SODA materials should not be used as construction fill because they produce very high release rates for arsenic and antimony, as shown in Figure 4.
- Development rock would be used to reconstruct the valley currently occupied by the Yellow Pine pit and the EFSFSR, to construct haul roads and pad areas for site facilities, and as road surfacing material or concrete aggregate (DEIS, p. 4.9-8). Initial HCT results and MWMP results suggest this would cause stormwater runoff that currently has no plan for capture and treatment.
- The DRSF source terms are not presented as concentrations, which form inputs to PHREEQC models – only rates are presented. Concentrations are presented for tailings (DEIS, Table 4.9-9).
- Average steady-state release rates are used for all analytes for the same weeks that sulfide releases reached steady state conditions. As shown in Figure 3 in this memorandum, steady-state rates were possibly reached for sulfate but not for arsenic or antimony.
- Results from the two HCTs with the highest arsenic release rates were eliminated from use in the model.
- Seepage source terms from the DRSFs are based on average steady-state release rates rather than using existing seep concentrations or using first-flush HCT rates (SRK Consulting, 2018, p. 54).
- ERM (2019, p. 17) notes the risk that the SWWC model predictions do not adequately capture PAG contributions to surface water quality, which use the same input values used for the DRSF models. Three waste rock samples had paste pH values <6 and multiple surface samples had NAG values <pH4. ERM (2019) states that a sensitivity analysis was not conducted "but has been promised by the proponent." However, an analysis was not conducted in SRK Consulting (2019b) as promised; the DEIS (Section 4.9.8) also noted that this issue. The selection of PAG rock for the models is inadequate

(only one sample, and it did not produce acid; see Section 5.2.1 of this memorandum). More representative PAG samples are needed, and these are available in legacy surface samples, including the SODA samples.

- First flush chemistry for contact water coming off the DRSFs was not considered in the model. ERM (2019, p. 17) states that this approach is non-conservative and highly recommends that results from first-flush rates be included in a sensitivity analysis, which was not conducted. This shortcoming is also noted in DEIS Section 4.9.8.
- The DRSF model assumes no leaching or release of nitrate from blasting during closure or post-closure, only during operations (SRK Consulting, 2018, p. 58). Two pages later they state that nitrate release could be an issue during closure. Nitrate releases should be included at least during closure and at reduced rates into post-closure.



Source: SRK Consulting, 2019a, Figures C 7 and C 8.

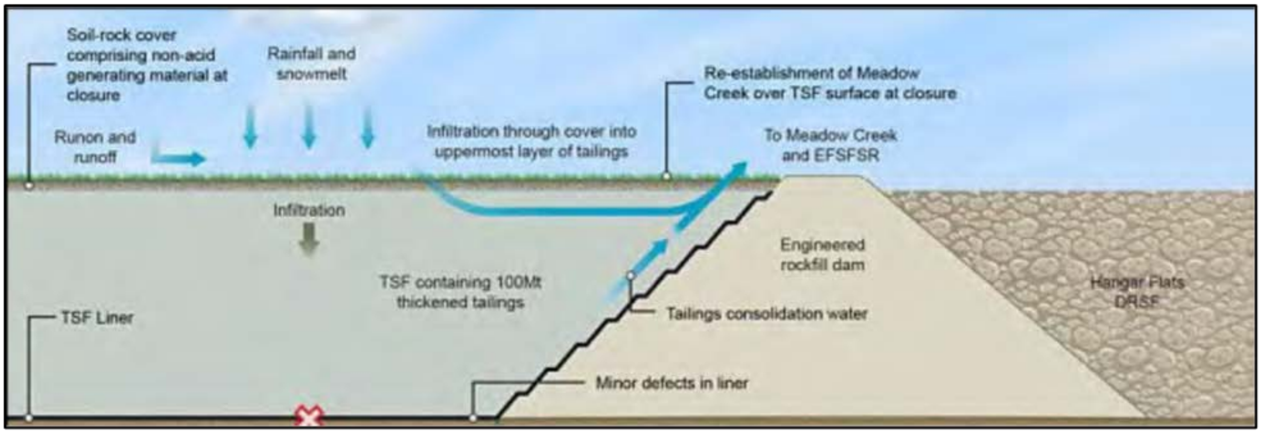
Figure 4. SODA spent ore HCT release rates for arsenic and antimony, Phase 1 and 2 HCT results.

3.3.2 TSF

- Flotation tailings are expected to generate neutral to alkaline pH drainage and require no special disposal considerations to prevent acidic drainage, and the mix of flotation and POX tailings expected in the TSF will be net acid neutralizing (90% flotation, 10% POX), but management to limit arsenic and antimony release will be required (SRK Consulting, 2017b, p. ii-iii). This statement contrasts with one in the DEIS (4.9-13) “Because of the small particle size, tailings can cause acid rock drainage (ARD) and metals leaching.” The POX leach residue tailings samples (A1-DX, A2-DX, A3-DX) had high percent sulfide sulfur values (1.5 to 2.1) and very low NPR values (0.21 to 0.33) (SRK Consulting, 201b7, Appendix A2). The NPR values of the few tailings samples were relatively high (SRK Consulting, 2017b, Appendix A2; SRK Consulting, 2019a, Table 5-5), but the methods used will overestimate NP and underestimate acid-production potential, as discussed in Section 5.2.3 of this memorandum.
- Only one sample of mixed flotation/POX tailings and one sample of bulk flotation tailings were subjected to ABA (SRK Consulting, 2017b, Fig. 3-61; both non-PAG). Only one bulk flotation tailings sample and one POX/bulk flotation tailings mixed sample were tested

for short-term leaching using the SPLP method, which uses a 20:1 water:solid ratio that causes too much dilution (compared to the MWMP test, which uses a 1:1 ratio); even with the dilution of the SPLP, leachate As and Sb concentrations exceeded water quality standards (SRK Consulting, 2017b, Table 3-26). More tailings samples need to be subjected to geochemical testing. No samples were subjected to long-term kinetic testing. A sensitivity analysis should be conducted to evaluate a range of pressure oxidation (POX)/bulk mixtures.

- During post-closure, the results from SPLP leachate are used to estimate the chemical composition of infiltration through the TSF cover. It is unclear from the discussion in SRK Consulting (2018; Section 5.4.1 and Tables 5-3 and 5-4) how these results will be mixed with the chemistry of seepage through the liner. A sensitivity analysis should be conducted that examines a range of assumptions of mixing to evaluate the resulting TSF leachate under the facility.
- Tailings water quality is represented by five tailings streams (DEIS, Table 4.9-9). During post-closure tailings water quality is represented only by the process water chemistry of West End oxide and sulfide tailings, which will be produced in the later stages of mining and will be therefore by on the top of the TSF (DEIS, p. 4.9-19). The TSF model assumes that most consolidation water would be derived from the upper parts of the TSF during post-closure. However, DEIS Figure 4.9-7, reproduced in part in Figure 5 below, shows consolidation water coming from *all* parts of the tailings and moving upward. Infiltration water may only affect the upper materials, but consolidation water comes from the entire column.



Source: DEIS, Figure 4.9-7.

Figure 5. Partial conceptual model of the TSF showing movement of consolidation water from all parts of the tailings.

3.3.3 Pit lakes and backfilled pits

- The surface water inflow source terms for the West End and Hangar Flats pit lake models during post-closure were averages from three locations with no upstream mining: Sugar Creek (YP-T-37), Meadow Creek (YP-T-33), and Meadow/Blowout Creek (YP-T-29), which just has reservoir for hydroelectric power (SRK Consulting, 2018, p. 131). The use of these locations assumes that no mining-influenced surface water will enter the pits during post-closure.

- Even though releases of contaminants from dissolution of soluble salts on the pit walls are discussed (DEIS, p. 4.9-16), those inputs are simulated using results from fresh core HCTs, which are unweathered and likely have little formation of soluble salts. Instead, MWMP results from surface samples or HCT results from the three SODA samples could have been used to estimate variability in releases. SODA samples high the highest arsenic and antimony release rates (see Figure 4) but very low sulfide sulfur percentages, indicating that the release derived from soluble salts.
- Pit lakes are expected to stratify seasonally and turnover each spring and fall, yet the pit lake models predict chemistry under well-mixed, annual average conditions (ERM, 2019, p. 16). The use of monthly or seasonal predictions would be industry-standard practice.
- The estimation of the amount of pit wall rock available for leaching is highly underestimated. SRK Consulting (2018, p. 136) assumed fracturing will only propagate to a depth of 3 ft in pit walls and the density of fracturing in that zone will average only 10%. Instead the fracture zone has been estimated to be up to 15 m thick, using measurements from open pit walls (Kempton et al. 2010; NDEP, 2018). Use of a shallower fracture zone will underestimate loadings and concentrations of contaminants from the pit walls during operations and closure/post-closure. Additionally, fracture zones can be highly variable, and ranges in concentrations of solutes entering the pit lake should be estimated (NDEP, 2018). SRK Consulting (2018) further assumed that water infiltration and reaction products will only be present up to 0.04 ft (one half inch!) on surfaces of rock fragment and assumed an oxidized rind of only 0.04 ft on pit wall surfaces. Sensitivity analyses related to fracture density (damaged rock zone) were conducted in SRK Consulting (2019b, p. 15), but only to a depth of 6 ft. Increasing the depth did increase antimony and arsenic concentrations in West End pit lake and resulted in arsenic and mercury exceedances in all modeled years and antimony exceedances in the first 20 years of post-closure; in the Midnight Area pit lake, concentrations of arsenic, antimony, sulfate, mercury, and aluminum increased slightly. For Yellow Pine pit backfill, concentrations of sulfate, mercury, manganese, and aluminum increased. Predicted concentrations in downstream surface water did not change notably. Additional sensitivity runs need to be completed with a much greater pit wall fracture depth.
- The “site-specific” NPR cutoff of 1.5 was used to distinguish PAG from non-PAG materials on pit walls. Because it is not site-specific and lower than the typical NPR of 2, less rock will be identified as PAG than the reality. A table should be produced showing the percentages of PAG rock in pit walls using an NPR cutoff values of 2 or 3.
- The DEIS (p. 4.9-13) states that submerging the PAG wall rock could help reduce surface water quality impacts by limiting further oxidation of the PAG material after it has been submerged.” Pit wall PAG rock is divided into areas above and below the water table (DEIS, Tables 4.9-5, 4.9-6, 4.9-7). No submerged column testing was conducted to examine this assumption. Submerged column leach tests conducted for the Pebble Project in Alaska showed that once wastes start producing acid, they will continue to release acid, metals, and sulfate even if submerged (Maest et al., 2020). Ignoring these

releases will underestimate pit lake concentrations, especially in a well-mixed pit lake. In addition, it appears that a single final water level is assume when the water level in pits during closure/post-closure will fluctuate by meters seasonally. Fluctuating water levels promote the formation of acidic and metal-rich drainage via the repeated formation of soluble salts. This fluctuation needs to be considered in pit water quality predictions.

3.3.4 SWWC model

- All the shortcomings listed above affect the reliability of the SWWC model, which is comprised of results from the DRSF, TSF, and pit lake models and surface water, groundwater, and seep inflows.
- The modeling efforts assume that capture of DRSF contact water runoff will be 100% effective and that no contact water will be discharged to the environment without treatment. ERM (2019, p. 12) also addressed this assumption and suggested doing a sensitivity model run to simulate potential impact of release of DRSF contact water runoff on surface water quality, which was not conducted. This shortcoming is also noted in the DEIS (Section 4.9.8).
- The SWWC model also assumed that the DRSF cover will be 100% effective during post-closure and runoff can be reliably represented by rain chemistry. This non-conservative assumption was noted by ERM (2019, p. 13) and the DEIS (Section 4.9.8). It excludes the possibility that surface runoff from or infiltration to DRSFs will contact underlying development rock and underestimates concentrations running off or infiltrating the DRSF.
- Predictions were made for surface water pH values in the SWWC model but should not have been. The spreadsheet model used was not integrated with the geochemical modeling results, so pH and concentrations do not reflect geochemical reactions.
- Seasonal adjustment factors (SAFs) were developed for each surface water prediction node using baseline water quality and streamflow data – one for each month, then multiplied by a predicted annual average to produce monthly estimates (SRK Consulting, 2018, Section 8.5 and Appendix D). The shortcomings with this approach are addressed in Sections 3.2 and 5.2.2 of this memorandum.
- Sulfate, arsenic, and antimony concentrations in surface water are underestimated by up to 48%, 62%, and 89% for sulfate, arsenic, and antimony, respectively (SRK Consulting, 2017a, p. 13) at certain locations. The underestimation was attributed to unaccounted for inputs from specific and diffuse sources and underpredicted loading from legacy facilities. However, these analytes are also the ones with the highest first-flush releases, and using those releases to estimate variability could improve the calibration but was not conducted.
- The assumption that an adequate calibration of existing conditions model (75%, then made many other adjustments) means it can be used to predict future water quality is unfounded.
- ERM (2019, p. 17) and the DEIS (Section 4.9.8) note that ammonia was not a modeled parameter but should have been, due to blasting. The same is true for nitrate, and no assumptions were made about high nitrate concentrations from blasting, at least during operations, closure, and early post-closure.

- Solubility controls were used in PHREEQC modeling to limit contaminant concentrations. PHREEQC modeling was completed for: (1) Yellow Pine pit and backfill, (2) Hangar Flats pit, (3) West End pit, (4) Fiddle DRSF, (5) Hangar Flats DRSF, (6) West End DRSF, and (7) the TSF and TSF embankment. ERM (2019, p. 17) states that this approach is generally non-conservative but highly variable and very sensitive to user-based inputs and assumptions. ERM recommended conducting a sensitivity analysis, which was not done. This issue was not addressed in the DEIS (Section 4.9.8). PHREEQC modeling should be redone without solubility controls, and the validity of the solubility controls should be examined as noted by Nordstrom (2019).
- ERM (2019, p. 17) noted that documentation regarding the approach and application of PHREEQC is insufficient to support data and document requirements for an EIS. This shortcoming was not noted in the DEIS. Further details should be included in an appendix to the main model report, or similar. PHREEQC input files and assumptions used should be included in a geochemistry appendix to the DEIS
- ERM (2019, p. 18) noted that seasonal predictions for surface water quality effects should be provided for value components (e.g., aquatics, fisheries, wildlife, etc.). This issue was not addressed in the DEIS. General implications and links between water quality and fish toxicity were also not addressed, including fish life cycle vs. water quality concentrations. Site-specific toxicity testing is needed on the effects of antimony to aquatic biota (O'Neal, 2020). If surrogate species are used, the results for all toxicity tests will need to be carefully interpreted.
- Sediment quality was not addressed at all in the SWWC model, the DEIS, or ERM (2019). As noted in Section 3.4 of this memorandum, more sediment sampling for chemistry is needed, and the results and the implications for sediment toxicity should be included in a Supplemental DEIS. Toxicity testing using sediment with a range of antimony, arsenic, and mercury concentrations representative of existing conditions should be conducted. Further, proposed mitigation measures include "reduce and prevent sediment delivery to streams by rehabilitating roads and mining sites" (e.g., DEIS p. 3.12-35). However, little baseline information exists on the concentration of mine-related contaminants in stream sediment against which to evaluate the effectiveness of the mitigation measures.

3.3.5 Brief comments on PHREEQC modeling

Some PHREEQE input files were released on the USDA Forest Service website on October 19, 2020, nine days before the end of the comment period.[8] According to the DEIS (Section 4.9.1.2, p. 4.9-5), numerical geochemical modeling using the U.S. Geological Survey software PHREEQC was completed for: (1) Yellow Pine pit and backfill, (2) Hangar Flats pit, (3) West End pit, (4) Fiddle DRSF, (5) Hangar Flats DRSF, (6) West End DRSF, and (7) the tailings storage facility (TSF) and TSF embankment. However, files for the West End DRSF, the TSF, and the TSF embankment were not included.

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[8] Available: https://usfs-public.app.box.com/s/y35kam707j0560hm9n5yjcsyeb3lye9y/folder/124615400518

Given the short amount of time available to evaluate the files, I ran one input file: Stibnite DRSF GW Under DRSF_OTM_HF Flats Cover – 5%Infiltration_Ave_v0.2. The wateq4f.dat database was used. A total of 80 simulations were included in the input file.

When the file ran, many errors were identified, including:

- Simulation 1:
    - WARNING: Did not find phases: Ferrihydrite, Al(OH)3(am), Dolomite (disordered), Dolomite (ordered), Lime, Hydroxylapatite, Hgmetal, Hgmetal(1), SiO2(am-ppt), and FeAsO4.H2O.
- Simulation 2:
    - WARNING: Could not find element in database: Be, Co, Cr, Hg, Mo, Sb, Tl, V. Concentrations were set to zero for these elements.
- Simulation 3:
    - WARNING: error: Phase not found in database: CdMoO4, Ferrihydrite, Hgmetal(1), Pyromorphite, SbO2.
    - ERROR: Calculations terminating due to input errors. Stopping.

As a result of the many errors, output information was only able to be produced for Simulations 1, 2, and 3 (3 of 80). The results of the PHREEQC runs were not clearly discussed in the DEIS or any other document. It is unclear how output from the PHREEQC runs could be used in the SWWC model if so many simulations could not be successfully run using the code. The fact that antimony was not able to be included in the runs, even though it is one of the primary COCs, is an important shortcoming that was not discussed in the DEIS or any other document.

In addition, all input values for temperature were set at 2.8^{o}C, and the only cover infiltration value evaluated was 5%. The variability in temperature and cover infiltration percentages was not evaluated in the available files. More time is needed to fully evaluate the available input files.

The many shortcomings listed for the geochemical and water quality models strongly suggest that the results of the models are overly optimistic in terms of predicting environmental impacts. Additional information on uncertainty of the SRK Consulting (2018) modeling effort can be found in Nordstrom (2019).

4. Perpetual Water Treatment

The need for perpetual water treatment is discussed in the DEIS (Sections 4.3, 4.4, 4.5, 4.7, and 4.9) and in Brown and Caldwell (2020). Figure 2-4 in Brown and Caldwell (2020) shows a water flow diagram for post-closure mine years 18 to 21. The two sources that will require perpetual treatment, according to the diagram, are toe seepage from the reclaimed Fiddle DRSF and the Hangar Flats pit lake overflow. However, because of large uncertainties in the water balance and the lack of consideration of climate change, additional sources could also require perpetual treatment.

To prevent predicted arsenic and mercury exceedances in Fiddle Creek, toe seepage from the Fiddle DRSF would be treated using a passive system in perpetuity using a biochemical reactor (BCR) and aerobic vertical-flow wetland (DEIS, p. 4.9-33). However, if the arsenic criterion cannot be met using passive treatment alone, the Fiddle DRSF toe seepage would be routed to the Centralized water treatment plant (WTP) for additional treatment (DEIS, p. 4.9-72). Hangar Flats pit lake discharge would be treated in perpetuity at the Centralized WTP to limit water quality impacts to Meadow Creek (DEIS, p. 4.9-71, p. 4.9-136). The DEIS (p. 4.9-75) also states that periodic treatment of the West End pit lake discharge could be required in perpetuity. Brown and Caldwell (2020, Table 2-2) also mentioned that enhanced evaporation would be another “treatment” method potentially used in perpetuity. Periodic discharge from the West End pit lake would be treated in perpetuity using enhanced evaporation or a temporary treatment system (DEIS, p. 4.9-134).

The Brown and Caldwell water management plan (2020, p. 1-1) states that specifics of the plan are “generally aligned” with ModPRO (Modified Plan of Restoration and Operations) – Alternative 2 in the DEIS – selected alternative but could apply to all alternatives in the DEIS. No specifics are provided for the other alternatives, but the report does note that Alternatives 1, 3, and 4 would require a larger flow capacity treatment plant for treatment of Hangar Flats pit lake water.

The proposed active treatment is iron coprecipitation to remove arsenic, antimony, and mercury; copper would also be removed to meet the surface water standard (DEIS, p. 4.9-71). If iron coprecipitation is not effective, mercury will be removed by sulfide precipitation (Brown and Caldwell,2020, p. ES-2; DEIS, p. 4.9-72). The WTP residuals (I assume this means the precipitated iron floc and the sulfide precipitate) will be sent to the TSF or to an off-site landfill; the type of landfill would depend upon the characteristics of the residuals (Brown and Caldwell, 2020, p. ES-2, ES-3). Brown and Caldwell (2020, p. 7-6) state that on-site pilot testing will be performed in Year –3 (three years before mining begins) for the active system. Pilot-scale testing should begin before the construction period and should be required for obtaining discharge permits.

Little technical information is provided on either the active or the passive treatment methods. A footnote in Figure 2-4 in Brown and Caldwell (2020) states “The passive treatment systems will require pilot testing for effectiveness to meet treatment objectives.” A possible passive system is described briefly in Section 6.7 of Brown and Caldwell (2020). No diagrams, chemicals needed, information on removal effectiveness, or maintenance requirements are included for the proposed passive system(s). The uncertainty extends to the production of methylmercury in the passive system (“It is unclear whether BCRs remove methylmercury or create it, so methylmercury would need to be monitored.”). No information is provided on methods to remove methylmercury if it is produced.

The TSF is proposed to be a zero-discharge facility during operations (Brown and Caldwell, 2020, p. 2-15), but a reliable water balance has not yet been put forth, so this premise is highly uncertain (Prucha, 2020). Brown and Caldwell (2020, p. 3-2) admit that uncertainties in the predicted mine water balance and predictive chemistry modeling remain, including those related to climate change. Climate change is not considered in the water balance model. As a result, they suggest adaptive management be used to allow for updating the design basis. However, not even a preliminary adaptive management plan exists. If the TSF is not a zero-discharge facility, treatment of TSF contact water will also be required. The summary of water treatment for each alternative (DEIS, Table 2.2-1, p. 2-10) lists *active* treatment for tailings runoff and consolidation water during reclamation and closure, but Brown and Caldwell (2020, Figure 2-4) shows only passive treatment of TSF consolidation water for mine years 21 to 42. This discrepancy needs to be resolved.

SRK Consulting conducted a water treatment evaluation, which is a very rough modeling effort to predict water quality concentrations on an annual timestep (Brown and Caldwell, 2020, Appendix A). Their tepid conclusion is that most predicted concentrations are within the range of existing conditions and several show an overall improvement relative to existing conditions (Brown and Caldwell, 2020, Appendix A, p. 11). Neither this effort nor the current water balance include an estimate of the volume and concentration of contact water that could escape capture. If contact water is not captured, it cannot be treated.

Perpetual mine water treatment would require approximately 34 annual truck trips for chemical deliveries and removal of residuals to and from the centralized WTP and a new transmission line to the mine site that would not be reclaimed under Alternative 2 (the only alternative with the treatment plant). The active treatment chemicals requiring transport include sodium hypochlorite, ferrous sulfate, and sulfuric acid.

5. Shortcomings in Plans

Adaptive management and development rock management plans do not currently exist. These plans need to be created as part of a Supplemental DEIS and included as appendices. A water quality management plan does currently exist, but it is not part of the DEIS or its appendices. Results from water quality predictions and geochemical characterization should feed into all three plans to create numeric thresholds, to help select effective mitigation measures, and to guide management of water and wastes during and after mining.

5.1 Water Quality Management Plan

A Water Quality Management Plan does currently exist (Brown and Caldwell, 2020). The plan is “generally aligned” with ModPRO (Modified Plan of Restoration and Operations) – Alternative 2 – but Brown and Caldwell state that the plan could apply to all DEIS alternatives with some adjustment of WTP capacity (increasing capacity) (p. 1-1). The assumptions and approaches outlined in the water quality management plan rely upon results and assumptions in the geochemical characterization program and water quality modeling, which as noted throughout this memorandum have a high degree of uncertainty. Issues related to water treatment are addressed in Section 4 of this memorandum (Perpetual Treatment).

The plan to route streams on top of waste piles after closure is unusual and ill-advised. Figure 2-1 shows streams diverted *around* the mine facilities, but the text states that Fiddle Creek will be "re-established" across the Fiddle DRSF, and Meadow Creek and tributaries will be "restored" across the top of the TSF (p. 2-10). At a minimum, this approach needs adaptive management to assess the survival of the streams over time and the longevity of the liners that are designed to prevent infiltration of streams (or overflow of streams) to wastes. If the liners fail, or even if they allow some infiltration to the piles, which all liners will, leaching of the underlying wastes will be enhanced. A sensitivity analysis should be conducted as part of future water quality modeling to assess the potential impact of liner failure or stream overflow to Fiddle Creek and Meadow Creek.

"Naturally mineralized conditions" are blamed, in part, for the difficulty in reducing overall surface water concentrations below Idaho water quality criteria (p. 2-16). They cite to Midas Gold (2016) for a statement about naturally mineralized conditions existing over several square miles within the vicinity of the site. Although Midas Gold (2016) does make this statement, but nothing in the document assesses the impact of natural mineralization on water quality. As noted in Section 2.1 of this memorandum, the claim that certain seeps are affected by natural mineralization is not supported by any document. A study of baseline (absent mining) water quality should be conducted to evaluate the potential effects of natural mineralization on site water quality.

5.2 Adaptive Management Plans

An adaptive management plan (AMP) does not currently exist. The DEIS (p. 2-76) mentions the possibility of including adaptive management planning in the Environmental Monitoring and Management Plan "if environmental outcomes may be uncertain." This memorandum highlights the uncertainty of many project elements related to water quality, geochemistry, and water quality predictions. Clearly, an AMP for water quality is needed. The water quality elements of the AMP must include numeric thresholds based on water quality predictions, which have not been finalized or included in the DEIS. A more robust and reliable water balance that quantitatively considers the effects of climate change must be developed to evaluate its effects on volumes and concentrations of mine water discharges and on water quality predictions.

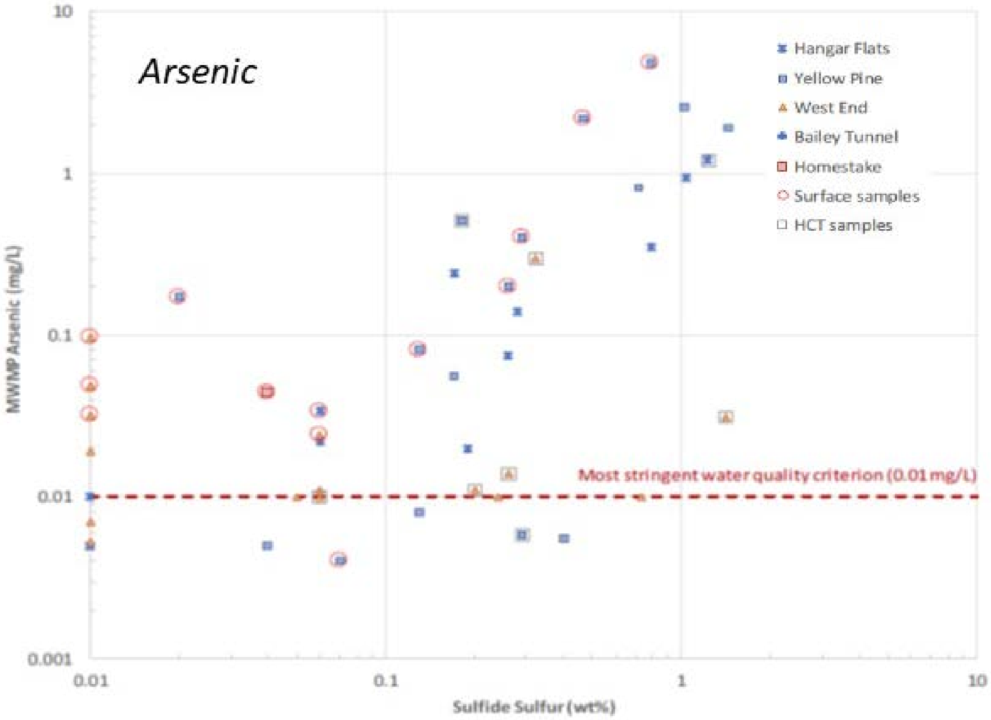
An AMP for water quality must also include an annual evaluation of the effectiveness of selected mitigation measures (e.g., water treatment, covers, mine water capture). The effectiveness of the mitigation measures can be evaluated based on water quality and quantity monitoring and monitoring of aquatic biological resources during construction, operations, reclamation, closure, and post-closure. Fish health and water quality should be linked in the AMP. The need for perpetual treatment of mine facility discharges should be evaluated from an adaptive management perspective with a goal of minimizing or eliminating the need for perpetual water treatment. A revised AMP for closure and post-closure will be needed because of the different project elements in these mine phases versus those in the construction and operation phases.

5.3 Development Rock Management Plan

A Development Rock Management Plan (DRMP) does not currently exist. As noted in the DEIS, a DRMP would be developed after a preferred alternative has been identified. However, the preferred alternative has already been selected (Alternative 2). One of the purposes of the

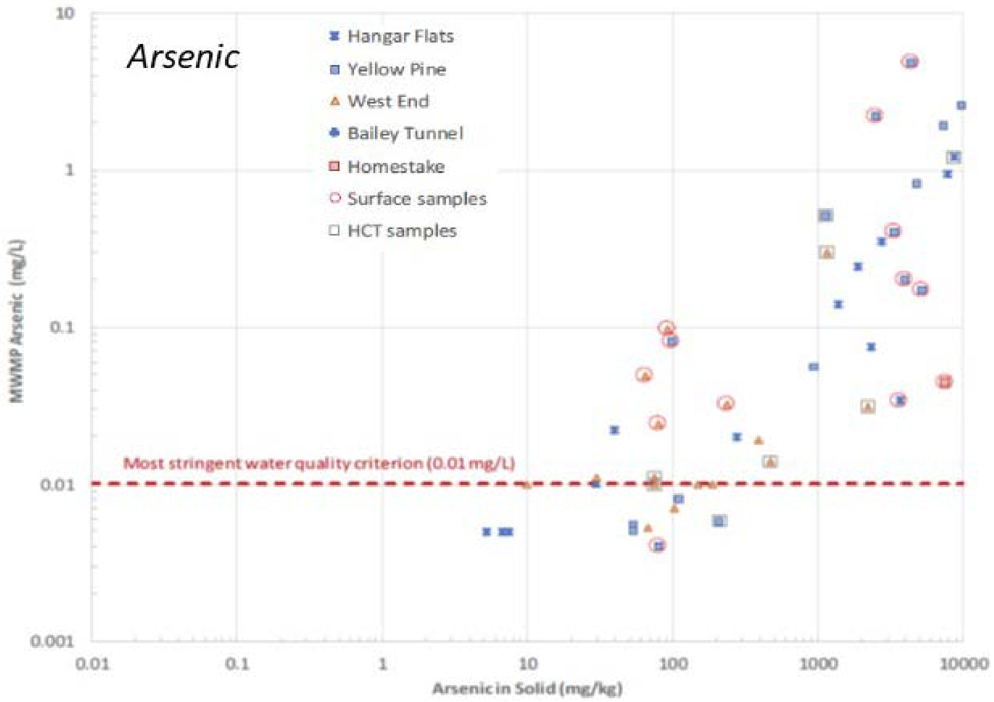
DRMP would be to determine which mined materials (from legacy or new mining) could be used for construction of the TSF embankment and other facilities (DEIS, Table 2.3-4, p. 2-26). The DEIS (p. 4.9-8) proposes to use development rock to reconstruct the valley occupied by the current Yellow Pine pit and the EFSFSR and as road surfacing material or concrete aggregate. Initial HCT and MWMP results suggest this would cause stormwater runoff that currently has no plan for capture and treatment. Because the development rock used for construction would be on the surface in perpetuity, oxidation of the material must be assumed. In that regard, leaching results from legacy mined materials are appropriate analogues.

The DRMP would include criteria to identify metal-leaching and non-metal leaching rock (DEIS, p. 4.9-8), but the geochemical characterization data show that, for example, measuring the percent sulfide sulfur would not necessarily identify rock with elevated contaminant leaching potential. High MWMP leachate concentrations have been produced from samples with low sulfide sulfur content and low total metal content in the solid, as shown in Figures 6a and b for arsenic. As noted in the figure captions, leachate antimony concentrations are also elevated in samples with low percent sulfide values and low total antimony concentrations in the solid. To distinguish samples with elevated contaminant leaching potential a test is needed that can be done quickly. The results in Figure 6 show that neither percent sulfide nor total arsenic (or antimony) content will be effective methods. Because so many of the samples tested show elevated arsenic and antimony leachate concentrations, not using development rock as construction fill or other uses at the site should be considered.



Source: SRK Consulting, 2017b, Figure 3-40.

Figure 6a. Arsenic MWMP leachate concentrations vs. percent sulfide sulfur. Arsenic MWMP leachate concentrations (SRK Consulting, 2017b, Figure 3.42) also show elevated concentrations (above the water quality criterion) at low percent sulfide sulfur values.



Source: SRK Consulting, 2017b, Figure 3-43.

Figure 6b. Arsenic MWMP leachate concentrations vs. concentration in solid. Antimony MWMP leachate concentrations (SRK Consulting, 2017b, Figure 3-44) also show elevated concentrations (above the water quality criterion) at low total concentrations in the solid.

6. Missing Water Quality and Geochemical Data and Information and Need for a Supplemental DEIS

Based on my review of the DEIS and associated technical documents, important water quality and geochemical data and information critical to evaluation of the DEIS are either missing, not publicly available, or presented in a confusing or incomplete manner in the DEIS. The DEIS appears to have been rushed, and a preferred alternative has not even been identified. According to the DEIS, Alternative 2 is now the proposed action for which Midas Gold is seeking approval, but the company has not submitted a plan of operations premised on Alternative 2 (DEIS, p. ES-13). One of the main differences in Alternative 2 compared to the other alternatives is the inclusion of a centralized active treatment plant and proposed perpetual water treatment for the project. However, little information is presented on the treatment options, and the site-wide water chemistry (SWWC) model upon which DEIS water quality predictions are based does not include the effects of mine water treatment. Although a revised SWWC model is included in Brown and Caldwell (2020, Appendix A), it is not part of the DEIS. In addition, findings from sensitivity analyses completed for the SWWC model are not presented in an updated modeling report. Certain mine plans that should be completed at least in full draft form at this stage are also not included in the DEIS. Because so much critical information is missing or incomplete, including the specific predicted effects of the preferred alternative, a Supplemental DEIS is required to understand the potential environmental effects and mitigation measures specific to the revised proposed action. Please see Sections 3.3 and 5 for more detail on shortcomings related to the water quality and geochemical models and to certain mine plans.

A listing of some of the most important missing, unavailable, or incomplete information related to water quality and geochemistry includes:

- No information is provided on geochemical test units (sometimes called geoenvironmental units). The separation of lithologies based on alteration mineralogy and characteristics (not just lithology) is needed to ensure the full range of leaching characteristics has been evaluated. These units should have been defined before geochemical characterization began.
- Different methods were used to calculate neutralization potential (NP) for different sample types, and it is unclear which method was used. The choice of NP methods makes a big difference in terms of identifying potentially acid generating (PAG) rock in the field and in their modeling assumptions.
- Use of a neutralization potential ratio (NPR)[9] of 1.5 vs 3. It is unclear which value is used for which purposes. Further, the selection of 1.5 has no scientific foundation, which reflects the rushed nature of the geochemical evaluation (DEIS, p. 4.9-9). No humidity cell tests (HCTs) have produced acid to date yet results from acid-producing HCTs are needed to select a site-specific NPR. On another site (Pebble Project in Alaska; Pebble Limited Partnership, 2018) SRK has provided a high level of detail (whether justifiable or not) on the selection of a site-specific NPR for mine waste management.
- No field or rapid laboratory method is currently proposed that could be used to separate waste rock based on contaminant leaching potential. Some general suggestions are included in the DEIS (p. 4.9-8), but because the Development Rock Management Plan does not exist - even though a preferred alternative has already been selected – the proposed method cannot be evaluated. The preliminary methods proposed in the DEIS can be shown to be ineffective based on the geochemical testing results. Further, no stormwater management plan currently exists. Leaching of waste rock in development rock storage facilities (DRSFs) or used as construction, remediation, or road materials could easily cause nonpoint source stormwater runoff that will be difficult to capture and that will require treatment.
- Source terms for the DRSFs and the pit walls are not included as concentrations in the DEIS. Model input concentrations are given for tailings but not for waste rock or pit wall source terms; instead only rates are provided for waste rock and pit wall leaching. To use in the PHREEQC code for the intended purposes, source terms must be in concentrations, and for mixing, relative flows of each source must be estimated. Additional information is needed that may be contained in the PHREEQC input files.
- PHREEQC input files. Some PHREEQE input files were released on the USDA Forest Service website on October 19, 2020, nine days before the end of the comment period.[10] According to the DEIS (Section 4.9.1.2, p. 4.9-5), numerical geochemical modeling using the U.S. Geological Survey software PHREEQC was completed for: (1) Yellow Pine pit and backfill, (2) Hangar Flats pit, (3) West End pit, (4) Fiddle DRSF, (5) Hangar Flats DRSF, (6)

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[9] NPR = NP/AP, where NP is the neutralization potential and AP is the acid-production potential determined by acid-base accounting tests.

[10] Available: https://usfs-public.app.box.com/s/y35kam707j0560hm9n5yjcsyeb3lye9y/folder/124615400518

West End DRSF, and (7) the tailings storage facility (TSF) and TSF embankment. However, files for the West End DRSF, the TSF, and the TSF embankment were not included.

- Baseline water quality absent historical mining effects is not evaluated. No information is presented in the DEIS or other documents to quantify natural mineralization inputs in surface water, groundwater, or seeps. The effect of natural mineralization on existing or predicted water quality at each prediction node is not assessed. A study of baseline (absent mining) water quality should be conducted to evaluate the potential effects of natural mineralization on site water quality.
- No sediment quality data are included in the DEIS, even though some limited data are available from the U.S. Geological Survey (Holloway et al., 2017). The Holloway et al. reference is cited in Section 3.9 of the DEIS but only for dissolved organic carbon concentrations.
- No groundwater plume maps were created for existing or future conditions that show exceedences of arsenic and antimony in alluvial and bedrock wells.
- No information is provided on the chemical speciation of antimony in site groundwater, seeps, or surface water. Because the toxicity of arsenic and antimony and their environmental behavior are dependent on chemical speciation, concentrations of the individual aqueous species are needed (i.e. As(III) and Sb(III); As(V) and Sb(V) can be estimated by subtraction from total dissolved arsenic and antimony concentrations). Many analytical results are available for As(III) (e.g., 323 samples in bedrock groundwater, 489 samples in alluvial groundwater, and 772 samples in surface water [Midas Gold, 2019, Tables 5-4, 5-3, and 4-6, respectively]).
- Little information in general is available on the toxicity of antimony to aquatic biota, and no site-specific information is presented on antimony or arsenic toxicity to resident and protected fish populations (or salmonids generally), macroinvertebrate populations, or aquatic plants.
- No information is provided on the relationship between fish life cycles (e.g., spawning, emergence) and the existing or future temporal variability in the concentrations of arsenic, antimony, mercury, or any other analyte in site surface waters.
- No information is provided on the exposure to fish to arsenic, antimony, mercury, or any other mine-related contaminant via the dietary or food chain pathway (sediment-macroinvertebrates-fish). This pathway has been shown to cause adverse effects to salmonids in laboratory experiments and from sediment at mine sites in Idaho and Montana and needs to be evaluated in a Supplemental DEIS.
- Pilot-scale testing of the proposed active and passive treatment systems for Alternative 2 should have begun, and the potential production of methylmercury in passive treatment wetlands needs to be evaluated and quantified.
- No information is provided on the effects of fluctuating water levels in the pits during post-closure. One final water level is assumed for each pit. The variability in post-mining water levels and the effects on water quality need to be assessed.

7. Qualifications

Ann Maest is an aqueous geochemist with Buka Environmental in Boulder, Colorado, USA. She has over 25 years of research and professional experience and specializes in the environmental effects of hardrock mining, baseline water quality, the fate and transport of natural and anthropogenic contaminants, and geochemical testing methods and modeling. She has evaluated more than 150 Environmental Impact Statements for large-scale mines in the United States, Latin America, Asia, and Africa and provides training to government agencies on EIS evaluation, the environmental effects of mining, and best practices. The results of her research have been published in peer-reviewed journals including Applied Geochemistry, Canadian Journal of Fisheries and Aquatic Sciences, Chemical Geology, Minerals, Applied and Environmental Microbiology, and Environmental Science and Technology. After completing her PhD, Dr. Maest was a research geochemist in the U.S. Geological Survey's National Research Program, where she conducted research on metal-organic interactions, metal and metalloid speciation, and redox geochemistry in surface water and groundwater systems. She has served on several National Academy of Sciences committees and a Board related to earth resources and has been an invited speaker at universities and national and international fora, including presenting on technical challenges and solutions for the mining sector at the United Nations. Ann holds a PhD in geochemistry and water resources from Princeton University.
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