previously stated) and that in the LV rock, the oxygenated equilibrium resulted in the production of ARD. GRE also stated that kinetic analysis was not utilized due to the long travel time of water within the unsaturated pit backfill. These statements indicate that the solutions were equilibrium reacted with a set of minerals representing the waste rock under oxygenated conditions and, that phases were permitted to precipitate. Then, according to GRE (2014f), an anoxic equilibrium phase analysis was performed to simulate the de-oxygenated conditions deep within the pit backfill. Presumably this statement means that a different set of saturation indices were calculated, and phases were selected to precipitate and/or permitted to dissolve. The step at which an average solution was calculated is not clear. For the Arshak pit, GRE (2014f) states that the ARD-74C HC week 10 test solution was reacted with LV waste rock for 20 weeks using PHREEQC's kinetic function. Then an anoxic phase equilibrium analysis was performed, as for the backfill solution, to simulate the de-oxygenated conditions deep below the Arshak pit bottom.

The final simulated water qualities of Arshak seepage and Tigranes-Artavasdes seepage are presented (GRE, 2014f Table 3) with a statement that the water quality is consistent with water samples representing long-term ARD reactions that are occurring in unmitigated LV waste piles located in Sites 13 and 27. The simulated solutions are very different from each other, and pH of the solutions are in the range of the pH of Site 27 leachate. Otherwise, little similarity is observed. Especially noteworthy is the iron concentration of 3.2 mg/L in the Site 27 leachate (ESIA Appendix 4.6.2 Table/Appendix F-1), compared to simulated total iron concentrations (ferric and ferrous) of approximately 3.05×10^{-5} mg/L and 5×10^{-3} mg/L for Arshak and Tigranes-Artavasdes seepage, respectively. The simulated solutions have essentially no copper; silicon was not simulated, and nitrogen was not simulated for the Arshak seepage. The simulated results for iron and copper are indicative of the same suspect phases that were specified to precipitate as for the runoff modeling (Appendix C of this report shows the phases that were specified in the modeling).

2.1.3.3.3 Backfilled Erato Seepage

The geochemical modeling of the water quality of post-closure seepage from the partially-backfilled Erato pit (Golder, 2014e) is generally well done with respect to current, accepted methods, the procedures are well documented, and a range of results was produced based on uncertainty of inputs. However, there are some important concerns. The modeling uses the results of the Erato post-closure pit water balance (Golder, 2014c). Golder (2014e) include the following conflicting statements:

- Seasonal fluctuations in the inflows and outflows of the pit result in a shallow, strongly seasonally-dependent (ephemeral) water body within the backfill;
- Evaporation losses in the water balance model include both evaporation from the pit water surface and from the saturated backfill to a depth of 1 meter; and
- Water levels in the backfill are below the top of the backfill at all times.

More importantly, the water balance reported in Golder (2014c) does not incorporate (does not mention) backfill. Golder states an assumption for the water quality modeling that the pit will be backfilled to the level required to contain the water body. Unclear is how a water balance model based on a seasonal pit lake (with water surface evaporation) with no backfill was adapted for use with backfill. Golder (2014e) correctly states that the assumption of 40% of precipitation on the pit walls reporting to the pit bottom is non-conservative; yet, GRE (2014e) used that assumption to remain consistent with other ESIA studies.

Golder (2014e) subdivided the LV for modeling into two categories on the basis of paste pH and sulfide sulfur into Lower Volcanics – Pyrite and Lower Volcanics – Other. The Lower Volcanics – Pyrite is distinguished by high pyrite concentrations (> 1 wt. % sulfide sulfur and paste pH <4), acidic HC leachates, and considerable metals leaching. The Lower Volcanics – Other has more alunite and less pyrite, yielding HC leachate pH values ranging from 4.5 to 6.5. This subdivision is a good first step toward developing geochemical units, but HC and bucket test results for the LV (see Section 2.1.1.2.3.2 of this report), as well as the large range in pyritic sulfur (up to 9.5%), indicate further subdivision may be warranted (for the geochemical modeling also).

Table 1 of the Golder (2014e) report showing PHREEQC input parameters includes a long list of minerals as potential phases that could be specified to precipitate in the modeling (if supersaturated). With the exception of mentioning barite (Ba control) and schwertmannite (Fe control) in the results section of the report, the specific mineral phase controls on the chemistry of the solutions were not provided. In this context, the resultant range of iron concentrations is low, with a mean of 0.003 mg/L and a maximum of 0.3 mg/L (cf., iron concentration of 3.2 mg/L in the Site 27 leachate [(ESIA Appendix 4.6.2 Table F-1]). Silicon was omitted in the modeling. Otherwise, the simulated pH and concentration ranges encompass the observed pH and concentrations in the Sites 13 and 27 waste leachates and mine portal drainage.

2.1.3.3.4 Solute Transport Simulations

Solute transport calculations were performed with a spreadsheet (analytical) model for a 1,000-year time frame following closure based on particle paths determined from the simplistic regional groundwater flow model. Concentrations in surface water were calculated based on mixing of groundwater with the receiving waters. Use of a spreadsheet model to evaluate potential impacts is a screening level approach that is not appropriate for a project of this scope in such an environmentally-sensitive area. Given the complex geology of the GSA (faults, fractures, stratified rocks), a spreadsheet model cannot accurately represent the physical system (cf., INAP, 2009). Instead, solute transport simulation should have been integrated into and predicted using the regional numerical groundwater flow model.

Golder (2014d) states that the groundwater flow model represents a simplification of the complex geology (intensely-faulted rocks) surrounding the pits. Due to the uncertainty introduced by the simplification of the geology, a local area impacts scenario was developed based on the assumption that 100% of the mining influenced groundwater migrates to perennial springs in close proximity to the pits. This scenario is based on simple mixing/dilution of the solute mass released from the source areas with groundwater recharge estimated for each assumed local catchment contributing to a set of springs.

The text (Golder, 2014d) states that the understanding of potential solute migration pathways from the pit areas is substantially based on groundwater flow modeling (Golder, 2014a). The text states that five flow paths are defined which represent the majority of the infiltration from the closed pits as seen by the concentration in flow lines for these pathways. This statement gives a lot of weight to the model-determined pathways, yet for the spreadsheet model, assumptions are made about depths of transport pathways that are completely different than the groundwater flow model particle paths. The flow model advective transport times are voided and replaced with arbitrary, up to an order-of-magnitude shorter travel times to receptors and are justified on the basis of conservatism. The regional groundwater flow model cannot be both correct and incorrect. This approach is not good science. It is obvious that Golder does not have confidence in the groundwater flow model. Neither the spreadsheet model nor the existing groundwater flow model is suitable for the transport calculations. The regional model should be revised (see Section 2.1.2.5) and used for solute transport.

Elaborate schemes were devised for partitioning fluxes from the base and walls of each pit sub-area to various solute transport pathways and determining separate source concentrations for the pathways, based partly on division of the area around Amulsar Mountain into multiple sub-areas. The same approach was used for the local area scenario.

The spreadsheet calculations of solute transport and the local impacts scenario based on mixing are not conservative, with source concentrations determined by geochemical modeling that are too low. Furthermore, loading rates to groundwater are underestimated due to seepage rates from the pits that are too low, and the predicted concentrations in the spreadsheet model do not include the effects of the BRSF and HLF. With no alkalinity in the volcanic rocks, the acidic seepage water will leach additional metals on its pathways through the subsurface and exacerbate the ARD impacts.

2.1.3.4 Potential Impacts due to Initial Mine Construction Work

As part of the assessment of potential impacts of initial Amulsar Mine construction activities on water resources, ELARD-TRC Team reviewed monitoring data obtained by the "Environmental Monitoring and Information Center" of the MNP (SNCO, 2019).

Pursuant to a Lydian press release (Appendix C of this report), groundbreaking of the Mine occurred on August 19, 2016. However, there was no information available as to the schedule or sequence of the initial construction activities or when these activities were suspended.

SNCO has been monitoring the surface water quality at three observation points in the region of the Amulsar Mountain since 2006. The monitoring frequency varied from monthly to quarterly. The last monitoring event was in November 2018. The monitoring points are located on the Arpa River, Vorotan River, and Kechut reservoir. The Arpa River location is closest to the Amulsar and downgradient of the mine areas where most of the initial construction work occurred.

The SNCO groundwater monitoring program includes annual monitoring of the discharge at four springs, including:

- Spring 529 located near Gorhayq village (Syuniq Province) and Spandaryan reservoir
- Spring 650 located in Jermuk City (Vayoc Dzor Province)
- Spring 2048 located in Jermuk City (Vayoc Dzor Province)
- Spring 2060 located in Ketchut village (Vayoc Dzor Province)

SNCO has also been monitoring the groundwater quality of Spring 529 discharge semiannually since June 2015. The last monitoring event was in November 2018.

The provided data package did not include descriptions of sampling methodologies or analytical methods or standards. Moreover, no other information or details were provided about the monitoring points or about human or other activities or climatic conditions near the monitoring points. Therefore, only a preliminary comparative screening of data collected before and after the start of initial construction activities at the Amulsar Mine was conducted.

A review of the data (SNCO, 2019) indicates that there was an apparent increase in the concentration of nitrate in a water sample collected from Spring 529 in October 2016 (within two months post groundbreaking). However, the concentrations seemed to diminish rapidly by early 2017 but seemed to rebound by November 2018. No similar trend to that observed for nitrate could be discerned in other constituents or in the discharge rate of Spring 529. Sulfate concentrations exhibited a spike in June 2016 before the mine groundbreaking event, followed by a rapid and sustained decrease in concentrations. The annual average discharge data for Spring 529 exhibit a sustained decreasing trend since monitoring began in 2015.

The water quality data for Spring 529 were also compared to data obtained for the three surface water monitoring points on the Arpa River, Vorotan River, and Kechut reservoir. The comparison did not show any corresponding trend to that of nitrate at Spring 529, including in the Arpa River. These data suggest the nitrate trend at Spring 529 are localized and transient.

Given the absence of a corresponding trend in other constituents and in the discharge rate at Spring 529, and the paucity of data and details, the transient and localized nitrate trend cannot be attributed with a reasonable degree of scientific certainty to the initial mine construction activities. Accordingly, no discernable impacts to surface water or groundwater due to initial mine construction activities could be inferred from the SNCO (2019) data.

2.1.4 Project Water Balance

Golder (2018) presents analyses that utilize modeling to update the SWWB for the Project. Golder (2018) includes updates to calculations presented in the earlier SWWB (Golder, 2016a) based on review by GRE as well as based on design modifications and changes in production schedule and evolution of facility footprints. Some of the current assumptions are based on agreement between Golder and GRE.

The objective of the SWWB model is to estimate the volume of excess water generated and process make-up water demand over the construction period and operational life of the mine (LoM). Based on the model results, storm and process ponds and pipelines were sized for each of the major Project facilities (BRSF, HLF, and mine pits). The SWWB will be continually updated.

The current SWWB is based on historical climate data from both the Jermuk and Vorotan weather stations. Jermuk climate data were used for evaluations below 2,200 m (HLF).

The SWWB is based on a wide range of uncertainty evaluated with stochastic simulations of climate variability using the GoldSim software. Peak 24-hour events and maximum annual precipitation depths from the continuous time series produced by the simulator for the LoM were used to evaluate storage requirements that include the 100-year, 24-hour precipitation event. Minimum precipitation depths from the continuous time series were used to evaluate potential make-up water requirements.

The probabilistic models of weather for each station are a good basis for design. However, the source of the percentages of snow melting when temperatures are below freezing (95% for above 2,200 m and 92% for below 2,200 m) in the snow accumulation calculation are not provided. Likewise, references for sublimation of 10% of snowpack below 2,200 m and 20% above 2,200 m are not provided. No justification is given for the choice of melting coefficient (2.74 mm/°C/day) from the referenced range (1.6 to 6.0 mm/°C/day).

The water balances illustrated and outlined for each facility are logical. However, some parameters are questionable and/or have very high uncertainty, such as the annual volume of groundwater reporting to the pits. Also, for pit backfill and the BRSF, a soil evapotranspiration parameter was applied. If this parameter is derived from (GRE, 2014b and GRE, 2014c), evapotranspiration is much too high and infiltration and runoff are correspondingly low. Furthermore, the illustration for the BRSF indicates that a pan factor and a soil moisture adjustment were used to calculate evaporation of surface water on the backfill and BRSF. This calculation is not explained, and it is not clear whether there is a soil cover on the materials.

The results for the 95% probability of exceedance suggest sufficient capacities are calculated for pit pumping and pond sizing. Section 9.0 of the Golder report correctly states there is uncertainty in some of the water balance inputs such as evaporation rates from different surfaces, runoff coefficients, and infiltration rates. Addressing the issues raised in the previous paragraphs can reduce some uncertainty. Pond volumes are based on the 100-year, 24-hour precipitation event, which is consistent with International Finance Corporation (IFC) performance standards (PS).

Noteworthy is that the water balance conceptualizations for the pits and BRSF are more realistic and meaningful in the SWWB (Golder, 2018) than those presented in the Pit Seepage and BRSF Runoff and Seepage Sub-Models (GRE, 2014a/2014b/2014c).

2.1.5 Mitigation Measures

ESIA Section 6.0 presents an assessment of the sources of environmental impacts related to the Mine infrastructure and activities during construction, operations, and closure in the context of the regulatory framework, Lydian policy framework, and Project-specific criteria. The main sources of potential impacts to surface water and groundwater are the BRSF, the HLF, the Mine pits, and all infrastructure for collecting, channeling, impounding, and treating contact water. A mitigation measure is defined in the ESIA as the engineering design to reduce impacts to acceptable levels.

Generally, the design concepts used in the Amulsar ESIA/EIA for development of mitigation measures are reasonable and appropriate (e.g., low permeability liners, encapsulation, capping, drainage, and leachate treatment). However, a number of the measures and plans, are partial, not-sufficiently protective, and/or unreliable with a high degree of uncertainty, particularly due to deficient and questionable data, models, model simulations, design bases, and/or assessment.

2.1.5.1 Mine Pits

The mitigation measures that will be implemented for the mine pits are partial backfilling and emplacement of an evapotranspiration (ET) soil cover on backfill. Non-PAG VC waste rock is intended for backfilling the Erato pit, which will not be covered with an ET medium, and is expected to seasonally accumulate runoff that would develop a pit lake without backfill. The base of the Tigranes-Artavasdes pit is expected to be below groundwater levels, which would result in the development of a permanent pit lake without emplacement of backfill. PAG waste rock will be placed in this pit and overlain by an ET cover.

Backfilling pits that accumulate runoff and groundwater prevents evapoconcentration of dissolved constituents. Pit lakes can develop anoxic conditions in the deep water, especially a deep pit lake as expected for the western end of the Tigranes-Artavasdes pit. Anoxic conditions increase metals solubility, and this anoxic, evapoconcentrated water would seep to groundwater. The backfilling, therefore, mitigates the negative effects of pit seepage to groundwater. Backfilling also mitigates impacts to fowl and fauna.

The ET soil cover on the Tigranes-Artavasdes backfill limits infiltration of precipitation and snowmelt. The cover will also limit oxygen ingress to the LVA PAG backfill. The cover, therefore, mitigates generation of ARD from backfill and seepage of ARD-impacted water to groundwater. Infiltration of precipitation during prolonged wet conditions and snowmelt will occur, however, carrying oxygenated water into the pit backfill, and potentially generating ARD.

Appropriate mitigation measures are planned for the Tigranes-Artavasdes pit. Considering the potential that some of the non-PAG VC rock intended for backfilling the Erato pit could also be acid generating (see Section 2.1.1.2.3.2), an ET cover would also be appropriate. The Arshak pit will not be backfilled and will seasonally accumulate runoff in the base of the pit. Backfilling the Arshak pit would mitigate seepage of evapoconcentrated seasonal lake water.

Only complete backfilling and covering would be better options. Noteworthy is the unavoidable accumulation of runoff water in the backfill of the partially backfilled pits. This potentially acidic runoff water is oxygenated and could react with backfill in the base of the pits, leaching metals and oxidizing sulfides. The Arshak runoff accumulation is also in contact with the backfill. Anoxic conditions could also develop in saturated backfill due to depletion of oxygen by reaction with the backfill and microbial activity, increasing solubility of the metals in the seepage to groundwater. Any infiltration through backfill could contribute to the load. The seasonal Arshak pit lake will also contribute seepage to groundwater.

There is clearly potential for contamination of groundwater by ARD-impacted pit seepage water. There are no contingency plans to mitigate groundwater contamination originating during operation and post closure from the pits beyond monitoring, for which no details are provided.

2.1.5.2 BRSF

The BRSF mitigation measures include:

- Encapsulation of LVA PAG waste rock in non-PAG VC waste rock (post closure);
- An ET cover (post closure);
- A non-PAG VC rock drainage layer beneath the PAG rock;
- Compacted native clayey soil or constructed clay liner beneath the non-PAG rock drainage layer;
- Subgrade drains with piping in perennial stream channels where groundwater emerges beneath the BRSF;'
- A toe pond to hold contact water collected by the subgrade drains; and
- Runoff diversion channels.

These planned mitigation measures for the BRSF are generally appropriate. Note that neither of the provided versions of the ARD Management Plan (Geoteam, 2016c; GRE, 2017) discusses emplacement of soil liner fill material in areas where the subgrade native clayey soil does not meet the specifications for a soil liner. The constructed clay liner is mentioned in GRE (2014a), with details provided in Golder (2017).

According to Geoteam (2015 Section 3.2.3), the BRSF will be founded on a low permeability liner, which will be comprised of native clayey soils (subgrade) with a maximum hydraulic conductivity of $1x10^{-6}$ cm/sec. Geoteam (2015 Section 3.4) specifies a thickness of 0.3 m and a hydraulic conductivity of $1x10^{-5}$ cm/sec as the design criteria (vs. $1x10^{-6}$ cm/sec in Golder (2015 Section 3.2.3) and in Golder (2017)). Geoteam (2015 Section 3.4) further states that based on the modeling results "*the Project is not required to conduct extensive confirmatory hydraulic conductivity testing in order to ensure that the $1x10^{-5}$ cm/sec target is achieved.*" The clay liner would be placed in 0.15 lifts and be compacted by vehicle traffic. In areas, of exposed scoria and basalt, a compacted clay liner will be constructed using borrow material from local areas.

The ET soil cover on the BRSF limits infiltration of precipitation and snowmelt. The cover will also limit oxygen ingress to the waste rock pile. The cover, therefore, mitigates generation of ARD from PAG waste rock and potential seepage of ARD-impacted water to groundwater. However, infiltration of precipitation during prolonged wet conditions and snowmelt will occur, carrying oxygenated water into the BRSF rock pile.

The intent of the non-PAG layer of waste rock under the soil cover and overlying the LVA PAG waste rock is to deflect infiltration water laterally at a presumed permeability contrast, preventing infiltration into the PAG waste rock. The design is conceptually appealing but deceiving. The LVA PAG waste rock is not an impermeable pile of fat clay. The LVA waste rock will consist of blocks, chunks, and pieces of rock with a range of grain sizes. Pore space among these various rock particles will permit infiltration. Consequently, some ARD will occur further adding to the mass flux reporting to the PTS.

The design concept of using non-PAG drainage layer at the base of the waste rock pile and an underlying low permeability clay liner to deflect seepage through the BRSF to the subgrade drains and mitigate infiltration to groundwater is adequate. However, the key liner design criteria are unreliable and raise concerns about the long term integrity, performance, and protectiveness of the liner including:

- The effectiveness of vehicle traffic in compacting native soil to develop a uniformly low permeability liner and the plasticity and homogeneity of the native soil across the BRSF area. Appropriate soil compaction equipment should be used.
- The small thickness of 0.3 m[15] and the relatively high hydraulic conductivity of $1x10^{-5}$ cm/sec (vs.$1x10^{-6}$ cm/sec specified in Golder (2107) and Geoteam (2015 Section 3.2.3)), particularly given the questionable modeling results and variability in subgrade conditions.
- The high degree of uncertainty associated with limiting the confirmatory hydraulic testing, particularly given the variability in subgrade conditions.

The toe pond size is based on the 100-year, 24-hour precipitation event consistent with IFC PS.

2.1.5.3 HLF

The mitigation measures included in the design of the HLF are generally appropriate. The design includes a composite liner system consisting of geomembrane underlain by compacted low permeability soil or geosynthetic clay liner in the steep terrain. The pregnant leach solution

[15] Typical design criteria for constructed clay liners in the USA are: thickness of 2 feet to 3 feet (0.6 m to 1 m) and hydraulic conductivity of $1x10^{-7}$ cm/sec to $1x10^{-6}$ cm/sec.

(PLS) will percolate through the ore to a drainage collection system above the liner consisting of perforated drain pipes embedded in a durable granular medium. The PLS will drain laterally from the HLP through transfer pipes to the process pond.

Diversion embankments and channels will be constructed upslope of the various HLF phases to divert storm and snowmelt runoff away from the pad and collection ponds. Underdrains consisting of trenches with pipes will be constructed in existing drainages and seeps within the leach pad and collection pond footprints for discharge to a collection sump, where the discharge water quality will be monitored for release or use as process water. The leach pad will have a toe berm and perimeter berms to prevent applied solution and rainfall/snowmelt water within the pad from overflowing the pad. The solution and storm flows will be routed to the process pond.

The HLF collection ponds include the process pond and storm water ponds to contain overflow from the process pond. The collection ponds were sized according to Project design criteria using the results of the HLF water balance calculations for wet year climatic conditions. Pond volumes are based on the 100-year, 24-hour precipitation event consistent with IFC criteria.

The process pond will have a composite double-geomembrane liner system underlain by a compacted low permeability soil liner. A leak collection and recovery system (LCRS) layer will separate the two geomembranes. The storm water ponds will have a composite liner consisting of geomembrane underlain by compacted low permeability soil. The collection ponds will have floating "bird balls" to prevent birds from contacting cyanide-bearing solutions.

Closure measures include an ET soil cover on the HLF to limit infiltration of precipitation and snowmelt. The cover will also limit oxygen ingress to the spent ore.

A fence with locking gates will be constructed around the perimeter of the HLP. This fence will bar entrance to livestock and wildlife, as well as unauthorized human access.

2.1.5.4 Contact Water Treatment Systems

Treatment of the contact water discharged from the mine operations is important to ensure that surface water quality is not impacted above applicable Armenian water quality standards. The ESIA focuses on and proposes PTS for the Mine contact water. Two PTS's are proposed - one for the heap pile leachate after mine closure and the second for the BRSF leachate (i.e., ARD) both during and after closure of mine operation. The heap pile leachate treatment system is addressed in a separate section, while the discussion below focuses on the PTS for the BRSF.

The ESIA (Section 6.10, page 22) and ARD Management Plan – V. 3 (Geoteam, 2016c Section 1.1 Commitments) indicate that if treatment trials indicate that a PTS will not meet the discharge criteria (MAC II standards) then a conventional packaged active water treatment system will be used. There are no descriptions of the decision-making process or details about the active treatment processes or requirements. Noteworthy, however, is that the commitments made in the ARD Management Plan – V. 3 (Geoteam, 2016) including the commitment to use active treatment in case of PTS inability to meet MAC II Standards, have been omitted in the updated ARD Management Plan – V. 4 (GRE, 2017). Therefore, this option cannot be assessed.

Furthermore, Lydian during the March 28, 2019 presentation and the June 27, 2019 conference call indicated that it will adopt an adaptive management approach for the mitigation and treatment of the Amulsar Mine impacts on water resources. However, Lydian provided no details or protocols for this approach. Therefore, the adaptive management approach cannot be assessed.

2.1.5.4.1 ARD - BRSF

2.1.5.4.1.1 Overview

The PTS for the BRSF leachate is described in Appendix 8.19 of the ESIA (GRE, 2017) and in a design basis memorandum (Sovereign, 2015). The PTS will treat water from the BRSF and excess water from the HLF operation during the time the mine is active, and will treat the BRSF leachate after the mine closes. The water is planned to be collected in pond PD-8 and then sent to the PTS. Sovereign (2015) shows the PTS will consist of the following units:

- Pond PD-8;
- Nitrate Reducing Biochemical Reactor (BCR);
- Aerobic Polishing Wetland (APW) No. 1;
- Sulfate Reducing BCR;
- Sulfide Scrubbing Unit;
- APW No. 2;
- Manganese Removal Beds (MRB); and
- A discharge pipe to the Arpa River tributary located downgradient from HLF ponds.

GRE (2017) has a similar sequence, with the exception that the first aerobic polishing unit has been replaced by an anoxic limestone drain.

The treatment system is based on projected water quality coming primarily from the BRSF toe pond, which is transferred to PD-8. The focus of the ESIA discussion of the PTS is on ARD from the BRSF, with less emphasis on the treatment of water coming from the mine pits to PD-8 and on nitrate and ammonia from the blasting residues on the rock. The nitrate reducing bioreactor is obviously designed to remove nitrate from the blasting operation, but there is little discussion on the incoming nitrate concentrations, and no discussion of ammonia. The incoming water is inexplicably projected to have low iron and aluminum concentrations. This is a key and questionable assumption given that there will be ARD in the BRSF and the pits' water going to the HLF (GRE, 2014d), as will be discussed later.

There are three major concerns with the proposed PTS design bases as discussed below:

1. The system design using a PTS has been selected too early in the process and does not allow the flexibility needed to deal with such a complex water system. The design is based on simulated water quality that may or may not be valid. If the simulated water quality is not valid, then the system will most likely fail and not achieve the treatment objectives.
2. The water quality modeling has significant discrepancies that make the modeling results highly uncertain and raise concerns about the ability of PTS to meet treatment objectives.
3. Ammonia in the wastewater will most likely be present at concentrations well in excess of the discharge criterion, but the treatment process for the ammonia is not discussed except in brief comments. Nitrate treatment is discussed in a little more detail, but is not given the focus that it should have and the nitrate concentrations appear to have been underestimated. Nitrate and ammonia are likely to be major contaminants that require

treatment while the mine is operating along with the products of ARD. The system as designed may not be able to achieve the treatment criteria for either nitrate or ammonia.

2.1.5.4.1.2 *PTS Approach*

The approach has been to develop a model of the water requiring treatment, select and design a system to treat that water, and then build the treatment system early in the life of the mine. The emphasis has been on a passive biological system rather than active system with no discussion of why a passive system was selected. As shown in the Table 2.1.1 below, the GARD Guide (INAP, 2009; Table 7-1) suggests that a passive system is most appropriate for closure and post-closure phases, while an active system is more appropriate for operational phases.

Table 2.1.1: Qualitative Comparison of Different Categories of Treatment (INAP, 2009; Table 7.1)

Feature/ Characteristic	Active Treatment	Passive Treatment	In Situ Treatment
1. Application to phase of mining	Most appropriate to exploration and operational phases because it requires active control and management. Closure and post-closure applications mainly associated with large flows	Most attractive to the closure and post-closure phases, because it requires only intermittent supervision, maintenance, and monitoring of self-sustaining properties.	Appropriate to the exploration and operational phases because it requires ongoing operation and maintenance

However, in contrast with INAP (2009) recommendations for using an active treatment system during mine operations, the ARD Management Plan for the Amulsar Mine (GRE, 2017) considers only a PTS for the BRSF during both operational and post-closure phases. GRE (2017) focused on the water quality and volume during the post closure phase and did not elaborate the rationale or feasibility for using a passive system during mine operation.

The use of a passive (vs. active) treatment system was decided early in the design of the mine, as illustrated by Sovereign (2015). There is no indication that any review has been given since then as to whether an active system should be used (except in response to the comments by Bronozian as discussed below) despite the update of the site-wide water balance (Golder, 2018).

Further, the selected PTS depends on very low iron and aluminum concentrations in the incoming water, as projected in the geochemical model. The major concern is that the whole system depends on the accuracy of the initial water model, both in terms of volumes and water quality, and on the biological systems behaving as predicted. There are many places where this model could be off, both in terms of flow and more importantly in terms of water quality. The amount of water seeping through the LV rock in the BRSF and the pits and generating ARD (including iron, aluminum, sulfate, acidity) could be off as noted in Golder (2018), and the modeling of the water quality could be off (refer to Sections 2.1.1.2; 2.1.2; and 2.1.3 of this report). To select a PTS that requires low metals content for mine water coming from an area of known ARD, with very little flexibility once the system is constructed, seems imprudent. The ARD in the area has high aluminum concentrations, and ARD is known to have iron concentrations as seen in the humidity cell leachates and as discussed in Sections 2.1.1.2 and 2.1.3 of this report. To design and implement the system based on incoming water having low iron and aluminum concentrations requires a high degree of certainty in the accuracy of the water quantity and quality, which the current models do not have. Even if the water quality modeling is correct (and even if concerns noted below are nonexistent), changes in mining operations or projections thereof may alter the inputs to the model, and hence the projected metals concentrations. At this stage, it is prudent for the design to be flexible to account for uncertainties in the water quality modeling projections and results.

Even if the model provides a good projection of the incoming water quality, there is no certainty that the proposed system would work. Biological systems are subject to numerous influences that can prevent or disrupt operation, and the system may not work as designed. Bench scale[16] and field scale pilot treatability studies are needed before the design is finalized. The success of studies will require good understanding and accurate representation of the incoming water quality, and of the range of concentrations of the key parameters in the incoming water. The water quality of the incoming water is subject to too many influences to be accurately modeled ahead of time. Rather, the water quality needs to be determined after the mine is in operation and the water quality can be directly measured. Moreover, the actual time as to when this PTS system should be operational may occur earlier than the timing (*i.e.*, Year 4/5) projected by the site-wide water balance calculations (Golder, 2018). Therefore, a flexible (*i.e.*, active) water treatment system may need to be considered for treating any discharged water prior to developing the final system.

The GARD Guide (INAP, 2009) similarly states in Section 7.3 (Mine Drainage Treatment):

> The approach adopted for mine drainage treatment will be influenced by a number of considerations related to the following: ...Different stages of mining and how the mine water system and water balance will change over the life of the mine. A mine drainage treatment facility must have flexibility to deal with increasing and decreasing water flows, changing water qualities, and regulatory requirements. This may dictate phased implementation and modular design and construction of a treatment facility.

As noted above, the ESIA (Golder, 2016) and older ARD Management Plan – V.3 (Geoteam, 2016c) states that if treatment trials indicate that a PTS will not meet the discharge criteria, an active water treatment system will be used. However, the bench scale treatability tests currently being conducted are not aimed at evaluating the PTS performance under varying conditions or determining under what conditions the PTS will fail. Instead, the bench scale tests are focused on a specific set of treatment processes and on demonstrating that the PTS is successful for a very limited set of input water quality conditions. If the tested water quality conditions are not representative of the actual water quality from the Site throughout both the active phase of mine operation and after the mine closes, then the testing does not address the question of whether an active or passive system should be used.

Wardell Armstrong (2017) provided responses to comments provided by Buka (2017a/b), Clear Coast (2017), Blue Minerals (2107), and Blue Minerals *et al.* (2018) about the Mine's water quality and treatment issues. A typical response is given below (Wardell Armstrong, 2017 page 8):

> 3.6.3 Passive Water Treatment
> Passive treatment of mine-impacted water is a standard method for treating and managing water quality concerns from metals mines (INAP, 2009).

[16] A limited laboratory bench scale testing by Lydian is ongoing.

> Passive treatment is an effective method to mitigate mild ARD and drain down from a spent HLF and rapidly becoming the industry-standard for all but the most severe ARD. Please see (A.M. Moderski, 2013), [(INAP, 2009), Section 7.5.2.]. Passive treatment was successfully applied, for example, at the Santa Fe Mine in Nevada, USA to treat HLF drain-down (R. Cellan, 1997). Predictions performed to-date and reviewed by IESC confirm that the predicted ARD and HLF drain down fall well within the range of acceptable chemistry that is treatable with passive treatment technology.
>
> The passive treatment system outlined in the ARD Management Plan is consistent with successful designs world-wide. Furthermore, a detailed programme of studies will confirm the efficacy of the design of the passive treatment system. The treatment system will be assessed using laboratory and field scale trials, which have been discussed with, and reviewed by, independent consultants. The testing will be completed by August 2018.

The above response along with the omission of the commitment to use active treatment in the updated ARD Management Plan (GRE, 2017) indicate that active treatment is not seriously considered. The GARD Guide (INAP, 2009 Table 7-1) specifically indicates (Table 2.1.1 above) that passive treatment systems can be used for ARD after mine closure, but that an active system is most appropriate during mine operations. The inference in Wardell Armstrong (2017) responses that the GARD Guide affirms that passive treatment is effective at all times in the mine operation is untenable and is contrary to INAP (2009 Table 7-1) recommendations. Passive treatment can work and has worked in a number of cases, and it may be appropriate for the Amulsar Mine. But to select a PTS for an active mine, and even for post-closure strictly based on questionable modeling data (see discussion below) without a definitive analysis and actual measurements of the influent water quality is incorrect. It is essential at this point to have a plan for collecting representative and necessary data and treatability testing to design treatment systems for operation and post-closure phases. Such activities should include pilot scale tests to assess the effectiveness of the cap on the BRSF for minimizing ARD generation, and the dynamics of acid generation, metals leaching, and nitrate and ammonia leaching from the rock after blasting.

2.1.5.4.1.3 Geochemical Modeling

The ARD Management Plan (GRE, 2017; v.4) states that "*Geochemical modelling has predicted that the mine contact water quality that **[sic]** can be treated with passive treatment methods. Table 14 shows the anticipated average water quality post-closure.*" Table **2.1.2** below presents excerpts of Table 14 (GRE, 2017):

Table 2.1.2: Projected PTS Influent water quality (GRE, 2017; Table 14)

Quality Indicators	Unit	Detention Pond (PD-8)
pH		3.92
Acidity	mg/L $CaCO_3$	157.2
Aluminum	mg/L	27.2
Calcium	mg/L	12.5
Chloride	mg/L	0.215
Iron, total	mg/L	5.66E-07
Magnesium	mg/L	5.11
Manganese	mg/L	0.0016
Nitrate	mg/L N	2.35
Potassium	mg/L	6.39
Sulfate	mg/L	97.3

There are discrepancies between modeled water quality shown in Table **2.1.2** and the water quality model given by GRE (2017) and that given by Sovereign (2015) as noted below.

A. <u>Iron concentrations are too low – key parameter for system as designed</u>

The treatment approach adopted is described in the GARD Guide (INAP, 2009) and is for influent water with low metals concentration, namely iron (Fe) < 2 mg/L and aluminum (Al) < 2 mg/L and dissolved oxygen (DO) <1 mg/L. Higher Fe and Al concentrations can cause problems in some systems as the metals precipitate and form solids that can clog the treatment system. Water coming from pyrite oxidation can have elevated concentrations of iron and aluminum (if the acid water passes through aluminum-bearing solids). The water in the humidity cells have maximum iron concentrations of approximately 125 mg/L and aluminum concentrations of 85 mg/L (GRE 2014d). Thus, the influent water of the PTS needs to have significantly lower metal concentrations than those concentrations in order to justify the proposed low metal PTS layout, and the current design may not be able to treat the ARD generated at the Site.

The projected iron concentration in Table **2.1.2** (5.66×10^{-7} mg/L) for PD-8 water, which is presumably coming from an ARD process, is unrealistically low as summarized below:

- This iron concentration in the modeled PD-8 water is significantly lower than iron concentrations in the Amulsar groundwater and surface water and even rain water, including water not affected by ARD, which range between approximately 0.001 and 300 mg/L (Lydian, 2018; Golder, 2019).
- GRE (2017) suggested that iron is in the ferric oxidation state, with the concentration controlled by Schwertmannite, a ferric hydroxysulfate commonly found in ARD water, along with Jarosite. Schwertmannite is only stable under low pH conditions and will slowly transform to goethite under acid conditions (Vithana *et al.*, 2015). Studies on Schwertmannite solubility in actual ARD have found iron concentrations of approximately 10^{-5} to 10^{-6} M at pH values between 3 and 4 (Yu *et al.*, 1999). Vithana *et al.* (2015) give ferric iron (Fe^{3+}) solubility lines for Schwertmannite[17], which give iron concentrations of 10^{-5} to 10^{-8} M at pH values between 3 and 4. These correlate to iron concentrations of 5×10^{-1} to 5×10^{-4} mg/L, approximately 3 to 6 orders of magnitude higher than the predicted iron concentration in the PD-8 water. Measured iron concentrations in ARD water in contact with Schwertmannite are at the higher end of the solubility calculations, suggesting that the projected values for PD-8 are underestimated by a factor of 10^6.
- Snoeyink and Jenkins (1980) give an average iron concentration of around 0.05 mg/L for iron in terrestrial waters (covering a wide range of Eh and pH conditions) with values ranging up to around 5 to 10 mg/L, or five to seven orders of magnitude higher than the water coming from pyrite oxidation in the site water.
- The GRE (2017) analysis does not include iron and aluminum concentrations typical of ARD, which can contain much higher iron concentrations. The GARD Guide (INAP, 2009) states that iron concentrations in ARD can range from 1,000s to 10,000s mg/L.

[17] $[Fe^{3+}] = -2.582\ pH + 2.996$ and $-2.582\ pH + 1.946$ (Vithana *et al.*, 2015)

- Sovereign (2015) recognized this discrepancy in the water quality projection and modified the PTS influent water quality shown in Table **2.1.2** (GRE, 2017; Table 14) by increasing the iron concentration to be more realistic and representative of ARD, albeit still lower than expected levels for ARD and in some Amulsar groundwater samples (Golder, 2019).
- GRE (2017) assumed the iron to be in the ferric oxidation state. However, pyrite oxidation first generates iron in the ferrous oxidation state[18], which is much more soluble at low pH than is ferric iron. Ferrous iron will oxidize to ferric iron and after oxidation contributes to the acidity of ARD. But, it has to oxidize first. Under conditions of somewhat limited oxygen supply, pyrite will oxidize to yield a low pH water with high ferrous iron content. Once this water encounters more oxic conditions, iron will oxidize and precipitate as Schwertmannite, Jarosite, or ferrihydrite to form the "yellow boy" seen in the drainage from many old mines. If there is pyrite oxidation, the question of where the ferrous iron oxidizes becomes very important in determining how to treat the ARD.

The discrepancy and high uncertainty in iron concentrations does not give confidence in the modeled water quality and raises concerns about the certainty and reliability of other parameters.

B. Charge Balance

The projected water quality results have significant inconsistencies in the cation-anion charge balance. The charge for each ion is calculated from the concentration by dividing the concentration (in mg/L) by the equivalent weight (the atomic weight of the ion divided by the charge on ion) to obtain the concentration of charge from that ion (in mequiv/L). The total charge (the sum of the cations and anions) in solution has to be zero, so as the total cation charge must equal the total anion charge. The charge balance calculations are shown below in Table 2.1.3 below (excerpted from Table 14 of GRE, 2017):

Table 2.1.3: Charge balance of major ions given for PTS influent (GRE, 2017; Table 14)

Cations				Anions			
Parameter		Concentration		Parameter		Concentration	
ID	Equiv. Weight	mg/L	mequiv/L	ID	Equiv. Weight	mg/L	mequiv/L
H^+	1	(pH 3.92)	0.12	Cl^-	35.5	0.215	0.61
Al^{3+}	9	27.2	3.02	SO_4^{2-}	48	97.3	2.03
Ca^{2+}	20	12.5	0.63				
Mg^{2+}	12	5.11	0.42				
K^+	39	6.39	0.18				
Total Cation Charge			4.37	Total Anion Charge			2.63

Charge Balance Error (CBE) = (total cations-total anions) / (sum of anions and cations) = 24.9%. This error is higher than the acceptable CBE of (less than ±5%) (Standard Methods, 1999). Possible causes for electrical imbalance are: 1) laboratory errors; 2) some species

[18] Refer to Section 2.1.1.2 of this report for descriptions of ARD processes.

(major ions) are not measured; and/or using unfiltered samples that contain solids which dissolve during sample preservation in acid.

There is clearly much more cationic charge than anionic charge in the Amulsar water samples. In ARD, the cationic charge comes primarily from the H^+ and Fe^{2+} (and other metals but to a lesser degree), while the anionic charge comes from SO_4^{2}. The oxidation and subsequent precipitation of ferric oxides results in the generation of H^+, which carries the positive (cationic) charge. Reactions between the acid and aluminum bearing rocks transfers the positive charge to aluminum. However, sulfate remains as the primary anionic charge. Therefore, the problem is that the charge from the sulfate concentrations in the PTS water do not balance the charge from the aluminum. Golder (2014f) states that sodium and fluoride are used to balance slight differences in the charge balance during the modeling, but to have fluoride account for the charge difference in Table 2.1.3 would require a fluoride concentration of 33 mg/L, which is unrealistic as demonstrated by the Amulsar surface water, groundwater and rain water monitoring results (Lydian 2018; Golder 2019). Also, Table 2.1.3 shows no sodium in the incoming water to the PTS. Sodium is usually a major cation in water, and if there is much sodium in the water, the charge balance would become even worse. The charge balance discrepancies further raise concerns about the reliability of the model projections and water quality.

C. *Aluminum concentrations inconsistent*

The water quality model predictions (GRE, 2017; Table 14) and data used in the PTS design basis (Sovereign, 2015) are different as shown in the comparison in Table 2.1.4 below.

Sovereign (2015) states that they have modified the PTS incoming water quality given in Table 14 (GRE, 2017) by increasing the iron concentration to be more realistic, but do not mention that they lowered the aluminum concentration by an order of magnitude. The nitrate concentration has been increased to account for blasting residue as estimated by Golder (2014f). (This last point will be discussed in more detail later.) Since the design of the PTS depends on having aluminum concentrations below 2 mg/L, the decrease in aluminum concentration from 27.2 mg/L to 2.27 mg/L is significant. The value of 27 mg/L (GRE, 2017 Table 14) indicates the selected PTS system (with a design criterion of less than 2 mg/L for aluminum) was not appropriate for this water quality.

Table 2.1.4: PTS Influent quality predictions (GRE, 2017) vs. PTS design basis (Sovereign 2015)

Parameter (*)	GRE (2017)	Sovereign (2015)
pH	3.92	3.5
Aluminum, mg/L	**27.2**	**2.27**
Calcium, mg/L	12.5	Not given
Chloride, mg/L	0.215	Not given
Iron, total, mg/L	**5.66E-07**	**3.22**
Magnesium, mg/L	5.11	Not given
Manganese, mg/L	0.0016	0.002
Nitrate, mg/L	**2.35**	**42**
Potassium, mg/L	6.39	Not given
Sulfate, mg/L	97.3	105

(*) Shaded numbers reflect parameters with discrepancies between water quality model projections (GRE, 2017) and data used for the PTS design (Sovereign, 2015)

This discrepancy is further elaborated by comparing the PTS influent data (Sovereign, 2015) to water quality from various test results and data sets including HC and modelled concentrations of key parameters for the BRSF seepage and underdrain (GRE, 2014g). Concentrations for the key parameters are given in mg/L in Table 2.1.5 and in mequiv/L units in Table 2.1.6.

Table 2.1.5: Comparison of water quality for BRSF leachate in mg/L

Location	Water Quality				
	pH	Acidity mg/L as $CaCO_3$	Fe mg/L	Al mg/L	SO_4 mg/L
Measured Values					
Humidity Cell–74-C (week 20)	2.5	960	125	38	980
Humidity Cell–76-C (week 20)	2.8	470	115	18	440
Modeled Values					
BRSF Seepage - post	3.02	962.8	0.5	164	412.3
BRSF Underdrain - post	3.88	171.6	0.0	30	105.4
PTS Input – GRE (2017)	3.91	159.6	5.88 E-07	27.6	98.8
PTS Input – Sovereign (2015)	3.5	Not given	3.22	2.27	105

(GRE, 2017; Sovereign, 2015; GRE, 2014g)

Table 2.1.6: Comparison of water quality for BRSF leachate in mequiv/L

Location	Water Quality				
	H^+	Acidity	Fe (as Fe(III))	Al	SO_4
	mequiv/L				
Measured Values					
Humidity Cell–74-C (week 20)	3.2	19.2	6.68	4.2	20.4
Humidity Cell–76-C (week 20)	1.6	9.4	6.1	2.0	9.2
Modeled Values					
BRSF Seepage - post	0.95	19.3	0.03	18.2	8.6
BRSF Underdrain - post	0.13	3.43	0	3.3	2.2
PTS Input – GRE (2017)	0.12	3.19	0	3.1	2.1
PTS Input – Sovereign (2015)	0.31	-	0.17	0.25	2.2

(GRE, 2017; Sovereign, 2015; GRE, 2014g)

Acidity at these pH values and in ARD comes from the sum of H^+, Fe^{3+}, and Al^{3+}. Since these ions should be the major cations in the leachate, they also add up to the cation charge. The major anion is sulfate, and so the acidity should be equal to the sulfate concentration (in mequiv/L units). For the HC leachates, these relationships hold. Sulfate concentrations for the HC leachates in Table 2.1.6 are close to the acidity values, while the H^+, Fe and Al concentrations are close in Cell 76-C, and relatively low in Cell 74-C. (The charge balance for Cell 74-C is much better if the Fe is assumed to be Fe(II), in which case the concentration is 10.02 mequiv/L). The modeled results show a good correlation between the acidity and the cations, but significantly lower sulfate concentrations than either the acidity or the cation concentrations. This discrepancy in sulfate concentrations raises concerns and further increases uncertainty in the system design.

The modeled water quality is dependent on a number of factors that could be different from that originally modeled. For example, the Site-Wide Water Balance (Golder, 2018) indicates that changes in design of BRSF have changed the leachate water volumes estimated in the previous model (Golder, 2016a). Further, estimates of the water going through the BRSF may be low (see Sections 2.1.2.2 and 2.1.2.3 of this report). Presumably, this will alter the proposed water quality projections. These changes have not been incorporated in water quality modeling since the model was reported in 2014. The BRSF leachate is not the only water going into PD-8. In addition, water from the mine pits will be discharged into the pond and will interact with the water from the BSRF. The water quality from the mine pits has also been modeled, but there is no indication that the water quality of the mixed water (which is what will go into the PTS) has been modelled, nor has there been much effort made to evaluate the changes in water quality after the mine pits are closed. Such changes highlight the uncertainty in the water quality modeling, and the risk in design treatment systems based on projections that may change.

2.1.5.4.1.4 PTS Design

GRE (2017) provides an updated ARD Management Plan while Sovereign (2015) provides the basis of the PTS design. The design is based on incoming water with low iron and aluminum concentrations. Water with higher iron and aluminum concentrations is treated using a different sequence of steps according to the GARD Guide (INAP 2009). While the modeled water has low metals concentrations, there is no guarantee that the model is correct and no ability to adjust the system for high iron and aluminum concentrations if the modelling is not correct.

The unit operations considered for the PTS (*i.e.*, nitrate reduction and sulfate reduction) are well established technologies and are used for mining-influenced water (ITRC, 2013; USEPA, 2014). Sulfide removal is less common. However, nitrate and sulfate bioreactors have been used on old mines with fairly constant flow and water quality, whereas, the proposed PTS is for an operating mine and for the first years after closure when both flow and water quality will vary. A key discrepancy is that the design (Sovereign, 2015) does not elaborate what happens to the acid, or why the two manganese removal beds are necessary.

Gusek *et al.* (2018) presented the results of bench scale testing on some parts of the proposed system in a paper at the Tailings and Mine Waste Conference in Colorado, USA, in 2018. The testing initially used simulated mine water spiked with nitrate and sulfate with pH adjusted to a representative value, then later used locally sourced acidic water with low metals concentrations (from a former mine in Armenia). The test reportedly demonstrated that the nitrate and sulfate reducing bioreactors were effective, provided that there was sufficient limestone to keep the pH neutral. The sulfide removal reactor was less successful, but could be improved in future testing. While it is valuable to have the bioreactors demonstrated, the focus again is on post-closure water with low metals concentrations. The testing did not address what would happen if the iron and aluminum concentrations were higher than projected in the water quality models.

2.1.5.4.1.5 Nitrate and Ammonia

The ARD Management Plan and PTS design basis focused on water quality at one point in the mine life, namely post closure. But the BRSF PTS is intended to treat water during the mine operation phase (during the last five years[19]) and during the post-closure phase. The water quality during these two phases will be significantly different. During mine operation, Golder (2014f) estimated that water coming from the mined rocks (both the pits and the BRSF) will

[19] Based on the updated SWWB (Golder, 2018), all contact water from the BRSF and pits during the first five years of mine operation, will be used in the HL process, and there will be no water to treat. However, this 5-year period may be overestimated due to the incorrect water fluxes from the BRSF and the pits.

have significant concentrations of both ammonia and nitrate from the explosive residue. Golder (2014f; Table 2) estimated the concentrations as shown in Table 2.1.7

Table 2.1.7: Estimated nitrate & ammonia levels in pits & BRSF water (Golder, 2014f; Table 2)

Area	Nitrate Concentration (mg/L N)		Ammonia Concentration (mg/L N)	
	Minimum	Maximum	Minimum	Maximum
Pit Sumps	12-30	>1000[*]	12-30	>1000[*]
Pit Backfill	70	440	70	440
BRSF Fluids	13	420	13	420

[*] Significant uncertainty in the high concentration, low volume sump water

The estimated average concentration in the pit sump water is between 70 and 180 mg/L N each for nitrate and ammonia for the time period that the excess pit water will be sent to the PTS. These concentrations will decrease rapidly after the mine is closed, since both nitrate and ammonia will be rapidly leached from the mined rock.

The estimated flow from the pit sumps is 250,000 m^3/year for years five through nine, while the seepage from the BRSF is estimated at 63,000 m^3/year (2 L/sec). It is not clear from the reports how much of the pit water will be sent to the PTS, but if it constitutes a significant portion, the water coming into the PTS could contain on the order of 100 mg/L nitrate and 100 mg/L ammonia. These values are significantly over the criteria for surface water (0.4 mg/L NH_4 and 2.5 mg/L NO_3 for Type II water in Arpa River basin, which is intended as the discharge water body). Thus, the water will need to be treated.

The projected influent PTS nitrate concentration during operations in the incoming water for the PTS used by Sovereign (2015) is 2.35 mg/L, while the post closure nitrate concentration is projected to be 42 mg/L. These numbers are questionable since the highest concentrations should be during operation not after, and are contradictory to the projected nitrate concentrations from Golder (2014f) estimates.

Furthermore, there is no projected concentration for ammonia in the influent water for the PTS. Sovereign (2015) states that ammonia will be oxidized in PD-8. However, the projected water quality indicates that this water will be acidic (pH 3.5-3.92, depending on stage), and nitrifying bacteria require neutral conditions of between pH 6.5 to 8.5 (USEPA, 2002). Further, it is not clear that the pond will be oxic, If the incoming water contains significant ferrous iron (as it might form the ARD), then the water could be anoxic. Also note that the requirements for the PTS system chosen is for a DO content of < 1.0 mg/L. If the DO is this low, then ammonia will not be oxidized in the pond, since nitrification is an aerobic process.

Moreover, treatment of the nitrate and ammonia during the last five years of the mine operation when the PTS is treating the water from both the BRSF and the excess water from the pit sumps needs to be addressed. The focus of the PTS discussion has been on treating ARD water from the BRSF, while the need to treat the two nitrogen species has been less highlighted. Obviously, the nitrate reduction ponds are designed to treat the nitrate, but the incoming concentrations would appear to be off, and there is no discussion of how ammonia will be addressed.

2.1.5.4.1.6 *Summary*

- The system design using a passive system has been selected much too early in the process and does not allow for the flexibility needed to deal with such a complex water treatment system. The design is based on modelled water quality that may or may not be valid. If the modelled water quality is not valid, then the system will likely fail.

- The GARD Guide (INAP, 2009) indicates active water treatment system is most appropriate during mine operations. A passive system is more appropriate for treatment of water with low chemical concentrations and steady water flux after the mine has closed. Since water will need to be treated during the last five years of mine operation, an active system will be more appropriate.

- The water quality modeling has significant discrepancies and uncertainties that raise significant concerns about the reliability of water quality projections and ultimately the feasibility and effectiveness of the proposed PTS.

    a. The modelled iron concentrations are much too low for natural waters, especially waters impacted by ARD.

    b. The charge balance for cations and anions in the incoming water for the PTS has a large error that significantly exceeds the acceptance criterion.

    c. The aluminum concentrations are inconsistent in different descriptions of the incoming water for the PTS.

    d. The water quality modeling was done early in the mine planning. Changes to the site-wide water balance and design, especially as related to water in the BRSF may impact the projected water quality.

    e. During the years of mine operation, the water coming into the PTS will contain both water from the BRSF and excess water for the pits that is not used in the heap leach operation. The influence of the pits water on the overall water quality has not been included in the model. This is important, especially after the updated site-wide water balance.

- Ammonia in the wastewater will be present at concentrations in excess of the regulatory discharge criterion, but the treatment process for the ammonia is not discussed except in brief and extraneous comments. Although discussed, Nitrate treatment requires further and more robust evaluation. Nitrate and ammonia are likely to be the major contaminants along with the products of ARD that require treatment while the mine is operating.

  The current PTS is not designed to treat ammonia during mine operations and will not be able to treat concentrated and complex ARD during and shortly after cessation of mine operations. If the PTS fails, high nitrate and ammonia concentrations and ARD could be discharged to the Arpa River with potential impacts to the surrounding water bodies.

2.1.5.4.2 HLF

2.1.5.4.2.1 Overview

Metals are extracted from the ore at the HLF by spraying an alkaline cyanide (sodium cyanide - NaCN) solution on the heap leach (HL) pile of ore, allowing it to percolate through the ore. The pregnant leach solution is collected in a pond next to the pile, then sent to a processing facility - adsorption/desorption/recovery (ADR) plant - to remove the gold and silver cyanide complexes onto activated carbon. The water (the barren leach solution) is replenished with fresh cyanide and base and then returned to the HL pile for further leaching. The metals are extracted from the carbon with hydrochloric acid (HCl), which is then neutralized with NaOH. Silver and gold are precipitated by adding zinc metal, and the solids removed for processing. The solution is returned to the barren solution tank for reuse.

Water in the process is recycled, with no discharge during the HL operation. The ADR plant uses 1,050 tons per year NaCN, 531 tons per year HCl and 190 tons per year of sodium hydroxide solution, with the liquid effluent going to back to the HL operation (Table 3.14 in Section 3 of the ESIA). In addition, water from the BRSF and the mine pits is discharged into the HL water. After year 5 of the mine operation, there may be excess water from the BRSF and mine pits, which is discharged to the PTS for the BRSF.

At the end of mine operation, the ore pile continues to be leached until the gold is extracted, then it is rinsed with fresh water with hydrogen peroxide added to destroy residual cyanide. This water is collected separately. Following rinsing, the pile is capped and any leachate after capping is treated in a PTS designed just for the HL water. Lydian states that it is difficult to predict what will be in the heap pile leachate, that it is difficult to design a treatment system for this water at this point, and that the design will be done after a better understanding of the water quality of the HL water is available. The deferral of a final detailed design of the PTS until after mining operations start is a reasonable approach. What could be added is a plan for how to and who will monitor the HLS and design the system so that it is not neglected as mining operations are shut down.

There are, however, several concerns about the HLS characterization, treatment, and discharge.

2.1.5.4.2.2 Water Quality Projections

Table 2 of the Hydrogeologic Risk Assessment - Proposed HLF (Golder, 2014b) presents water quality data for both the final HL solution and for the solution after detoxification. The concentrations for key parameters are given in Table 2.1.8 below.

Table 2.1.8: Key Water Quality Parameters - HLF Solutions (Golder, 2014b; Table 2)

Parameter	Units	Final Barren Solution		Final Detoxified Solution	
		Test 61781	Test 61790	Test 61781	Test 61790
Alkalinity, total	mg/L as $CaCO_3$	490	330	360	170
Bicarbonate	mg/L as $CaCO_3$	83	<1	130	120
Carbonate	mg/L as $CaCO_3$	260	190	160	43
Hydroxide	mg/L as $CaCO_3$	<1.0	3.4	<1.0	<1.0
Aluminum	mg/L	1.1	6.6	0.38	2.4
Calcium	mg/L	1.6	7.8	4.2	13
Chloride	mg/L	41	27	28	27
Cyanide (total)	mg/L	42	67	0.66	0.61

Parameter	Units	Final Barren Solution		Final Detoxified Solution	
		Test 61781	Test 61790	Test 61781	Test 61790
Fluoride	mg/L	1.8	2.9	1.9	2.8
Iron	mg/L	0.24	0.91	0.2	0.12
Magnesium	mg/L	<0.50	<0.50	<0.50	<0.50
Nitrate	mg/L	1.4	0.96	3.0	2.5
TKN	mg/L	76	80	20	29
pH		9.99	9.74	9.91	9.23
Potassium	mg/L	14	45	15	48
Sodium	mg/L	310	408	260	340
Sulfate	mg/L	45	390	140	590
TDS	mg/L	770	1200	720	1200

There are several issues with this table and the projected water quality for the spent HLS.

Firstly, the water quality is taken from a study by Kappes, Cassidy and Associates (2012) on the effectiveness of cyanide to recover gold from the ore. The testing included looking at whether peroxide could be used to destroy cyanide once gold extraction was complete. The testing was not intended to evaluate how the water quality of the HLS would be after ten years of operation with continuous recycling and cannot be used for that purpose, for several reasons:

- The reagents used in the laboratory testing are relatively different from those to be used in the full-scale operation, namely the laboratory testing used lime (CaO) to raise the pH whereas the full-scale operation will use sodium hydroxide (NaOH). The calcium and sodium concentrations in the laboratory testing are not representative of field solutions.
- More importantly, the HL water will be recycled for ten years in the HLF operation with no discharge. NaCN, NaOH, and HCl will be added at each pass through the pile (with a 60-day cycle, this corresponds to around 60 cycles). The concentrations of the continually added soluble ions (sodium, chloride) will increase, as will sulfate from the rock. Yet, the table shows the concentrations of sodium based on the concentration added in the initial wash of the HL pile. This is unrealistic, particularly when considering mass accumulation due to continued water loss due to evaporation in the HLF and contact water ponds. For the soluble ions such as sodium, chloride, nitrate, and sulfate the concentrations given are obviously too low, which would result in underestimating corresponding loading to and treatment requirements at the PTS. In addition, for the other metals that may be extracted from the ore itself (many of the trace metals), the concentrations after ten years of use in leaching the ore and recycling likely to be much higher than the concentrations in the original solution.
- Attachment 1 of Golder (2014f) estimated that the blasted ore in the heap pile is expected to contain between 234,058 and 585,156 kg of nitrogen (both as nitrate and ammonia) from the blasting operation over the life of the mine. This nitrogen (N), both as nitrate and ammonia, will rapidly leach into the HL solution. Unless there is a loss in the ADR discharge, this will give a very high concentration of nitrogen (probably as nitrate) in the HLS. If the heap pile contains over 1,000,000 m^3 of water (Golder, 2018) after rinsing, this gives a final concentration of 200 to 600 mg/L N in the water from the blasting residuals. This does not include the nitrate and ammonia coming from the mine pits and from the BRSF facility, the water from both of which goes completely to the HL water for the first five years. The values of 1.4 and 0.96 mg/L in Table 1a for nitrate in

the final barren solution likely underestimate the actual nitrate concentrations by orders of magnitude. Golder (2014f) did not estimate the concentration of nitrogen in the HLS due to other sources of nitrogen, saying the analysis was outside the scope of the memorandum.

- The water being sprayed on the pile is at a pH of 11-12. Water at such a high pH will scrub carbon dioxide (CO_2) from the air to form bicarbonate and carbonate. After ten years of use and recycling, the bicarbonate and carbonate concentrations in the HL water should be quite high. Yet the bicarbonate concentrations in the barren heap solutions in Table 1a above are relatively low. Some carbonate will be removed by reaction with the lime in the rock pile, however, the calcium concentrations given do not reflect the solubility of calcium carbonate, so it is not clear that such precipitation was considered.

The alkalinity values in Table 2.1.8 are inconsistent. Total alkalinity is the sum of bicarbonate, carbonate, and hydroxide alkalinities by definition and by the way they are measured. However, the total alkalinities shown in Table 2.1.8 are considerably higher than the sum of the three components. A comparison of the alkalinities is presented in Table 2.1.9 below.

Table 2.1.9: Comparison of Alkalinities – HLF Solutions (Golder, 2014b; Table 2)

Parameter	Units	Final Barren Solution		Final Detoxified Solution	
		Test 61781	Test 61790	Test 61781	Test 61790
Alkalinity, total	**mg/L as $CaCO_3$**	**490**	**330**	**360**	**170**
Bicarbonate	mg/L as $CaCO_3$	83	<1	130	120
Carbonate	mg/L as $CaCO_3$	260	190	160	43
Hydroxide	mg/L as $CaCO_3$	<1.0	3.4	<1.0	<1.0
Bicarbonate + carbonate + hydroxide		**343**	**193.4**	**290**	**163**
pH		9.99	9.74	9.91	9.23

These values are simply incorrect as presented. The hydroxide alkalinity is measured from the amount of acid required to bring the pH down to 10.3. At a pH of 9.74, the oxide alkalinity is 0.0 by definition. Therefore, one cannot have a hydroxide alkalinity of 3.4 mg/L $CaCO_3$ at pH 9.74 as shown for Test 61790.

Alkalinity is a measure of the acid-neutralizing ability of the solution. The most common sources of alkalinity in natural waters or wastewater are the carbonate species, hence, the divisions in measurement. If the carbonate species are the primary sources of acid-buffering, then the bicarbonate alkalinity will always be higher than carbonate alkalinity, since any carbonate present in the sample is measured in both sections of the titration. However, other bases can contribute to alkalinity, such as cyanide and ammonia. These would show up primarily as carbonate alkalinity rather than as bicarbonate alkalinity. As discussed above, alkaline water that has been in contact with the atmosphere for ten years should have very high concentrations of both carbonate and bicarbonate, and the alkalinity results should reflect these high concentrations. The values shown in the Table 2 are questionable.

Moreover, the laboratory results sheet for the Kappes-Cassidy (2012) study presents a charge balance for the water quality. However, the charge balance results are inconsistent with the water quality, since in some samples the charge from sodium is greater than the total cationic charge.

Kappes-Cassidy (2012) study was performed to evaluate the effectiveness of cyanide at recovering gold and the ability of peroxide to destroy the cyanide, where water quality results may arguably be peripheral to the point of the study. However, the use of its questionable water quality data to model the water quality of the final leach solution, design the PTS, and assess the potential environmental impacts and compliance is problematic and untenable.

2.1.5.4.2.3 HLS Treatment

At the end of the heap leaching operation, it is not clear what happens to the HLS used in the HL operation. GRE (2014d) states the following (Section 2.1.2.2; page 8-9):

> GRE indicates the following processes occur following cessation of ore deposition on the heap:
>
> - For a period of six to ten months, "rinsing" of the heap occurs. This comprises continued irrigation of the heap with sodium cyanide solution and circulation of leach solutions to the processing plant to recover any remaining precious metals from the ore. No source term attenuation is anticipated during this period. It is assumed that active evaporation to reduce solution volumes may be undertaken toward the end of the period.
> - Following the rinsing period, a detoxification process is undertaken where the heap is leached with hydrogen peroxide solution to destroy the cyanide in the heap and solution. This process will continue for six months to one year until cyanide concentrations are sufficiently reduced to permissible levels to discharge, and
> - Following rinsing and detoxification, the facility is covered and passive drawdown of the leach solution occurs. Closure management continues for a further five years during which drainage from the heap is sent to a passive treatment system and is monitored prior to discharge.

A flow chart of the water management during the closure phase presented as Figure 6.10.3 in Chapter 6.10 of the ESIA is shown below as Figure 2.1.2. The water from the residual leaching and rinsing are shown going to the ADR plant, and then inexplicably disappearing. GRE (2014d; page 71) gives an estimate of 2 million m^3 of water in the HLF after rinsing. The water by this time has been in circulation for ten years and will have elevated levels of sodium, chloride, sulfate, nitrate and probably other ions as well. Assuming the volume of HL solution is in the same range as that remaining in the HLF after rinsing (*i.e.*, >1,000,000 m^3), the management and disposition of such large volume of contaminated water should be addressed.

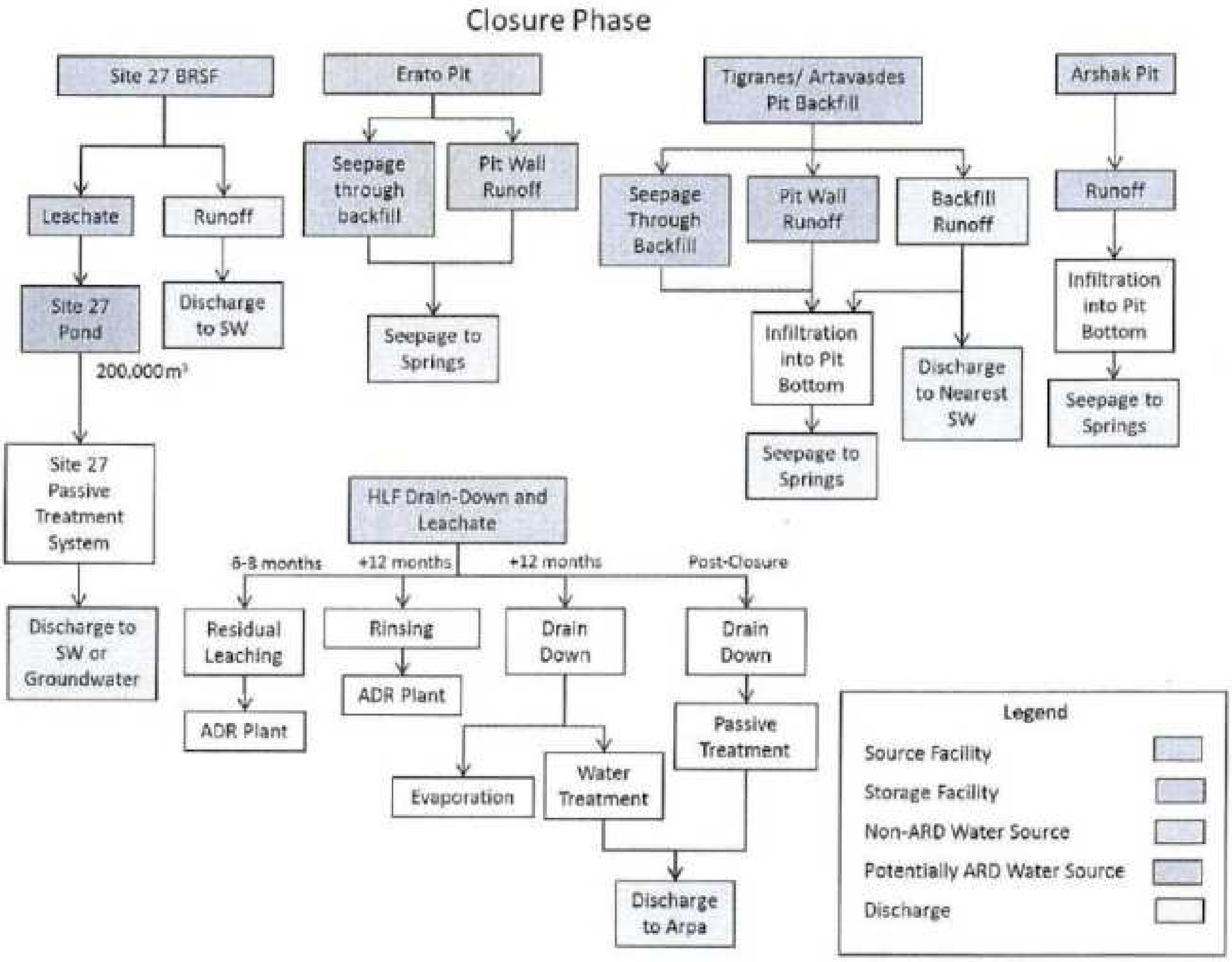

Figure 2.1.2: Flow Chart - Water management during closure phase (Figure 6.10.3 of the ESIA)

The ESIA proposes a dedicated PTS to be designed and implemented upon closure based on post-closure water quality monitoring and present generalized descriptions of processes for treating the leachate from the HLF post closure. This may be a reasonable approach; however, the ESIA should include plans for laboratory treatability and pilot testing to evaluate and confirm the feasibility and effectiveness of the PTS in treating the HLS and leachate and in achieving the discharge criteria. Furthermore, there are no contingency plans in case of the PTS failure to effectively treat the HLF wastewater or in future cases of PTS failure or emergency.

2.1.5.4.2.4 Summary

The discussion of the heap leaching operations solutions has two major issues:

- The projected water quality at the end of operation, both before and after cyanide treatment is unrealistic. The water quality used for modeling the system comes from tests that were not designed and not appropriate for assessment of environmental impacts, treatment, and compliance of the wastewater, and the water quality results have internal inconsistencies indicating that some of the results are incorrect. Further, at the end of the mine operations, the water will have been in circulation for ten years, and will have elevated concentrations of soluble constituents (sodium, nitrate, chloride) added to the water from operations and elevated concentrations of trace constituents leached from the ore (sulfate, trace metals) that will require treatment prior to discharge. There is no good modeling of these concentrations to know what to treat and how to treat them.

- There is no indication of how the HLS (the barren solution after the pass through the ADR) will be managed and treated and only a limited discussion of how the rinse water will be treated. This water may be on the order of 1 million m^3 and may contain high concentrations of ions that can be difficult to treat (sodium, chloride, nitrate...), so how it is handled is important to prevent contamination of the receiving surface water.

  The water coming from the HL ore pile after it is covered will be treated in a PTS to be designed at some point in the future after obtaining actual water quality data. Although PTS is a potentially applicable treatment technology for the post-closure HLF solution and leachate, there are no plans for laboratory treatability or pilot testing to assess the feasibility and effectiveness of PTS. Furthermore, there are no discussions of contingent or supplemental plans in case of the PTS failure to effectively treat the HLF solution and leachate and no contingency plans for future PTS failure or emergency.

The reports are not clear on how the HLS will be treated immediately after mining operations cease, thus it is difficult to determine whether treatment will be successful or what the impacts are if treatment is not successful.

2.1.5.5 Catastrophic Events

River flood risk is extremely low. The Mine facility closest to a river in distance and in elevation is the HLF, which is at least 200 m above the Arpa River. The Arpa River is a managed watercourse, with mitigation of flood provided by the Kechut Reservoir.

Consistent with the IFC standards, the current design of the contact water ponds includes free-board for the 100-year, 24-hour storm.

The seismic hazard risk is high for the Project Area. Seismicity was considered in the design of the HLF, BRSF, open pits, crushing plant, and overland conveyor system. However, an old construction standard was used for the analyses. Furthermore, known faults within the Project study area were not considered in the seismic hazards analysis. Movement on the seismically-active PSSF fault system could cause fault slip in the study area, potentially compromising the liner beneath the BRSF and the cover and destabilize the waste rock pile (Ꝣirak Fault beneath BRSF). This fault could conduct ARD-impacted seepage water from the BRSF toward the Kechut Reservoir and/or to the Vorotan River. Fault slip on the Agarakadzor Fault passing through the vicinity of the pits and BRSF could also impact the stability and integrity of the BRSF and pit backfill and cover systems. Ground motion could also impact the stability of the HLF, liner, piping, and cover and inflict damage on the contact water channels, the PTS, and system of process and stormwater ponds, resulting in contact water being released to surface water and groundwater.

Covers on the BRSF and pit backfill can be restored, if impaired by earthquakes. The repair of breached liners beneath the BRSF and the HLF will be challenging, requiring temporary or permanent relocation of the rock and spent ore. A destabilized BRSF or pit backfill could result in permanent loss of the non-acid generating VC layer of rock between the cover and the PAG rock. Any exposed PAG rock on pit walls or in the BRSF, HLF, or pit backfill due to earthquakes could impact the environment for hundreds, or possibly on the scale of a thousand or more years.

Historic landslides surrounding the Amulsar Mine are not documented in the ESIA, EIA, or supporting documents. Potential landslides are addressed in Section 2.4.4 of the EIA in the context of slope stability in the mine pits based on geotechnical data acquired through the exploration drilling program. The assessment is based on rock strength, RQD data, and orientation of discontinuities determined from rock core and includes the likelihood of earthquake-induced rock avalanches, slides, and slumps. The assessment does not pertain to the potential activation and consequences of landslides induced by blasting that would affect the environment and/or communities surrounding the Project. The Golder (2013) earthquake hazard assessment does not address the potential for landslides resulting from blasting. An assessment of potential landslide activation and the consequences is not possible with the available information.

The revised construction standard should be reviewed for compliance of all mining infrastructure including the need for reinforcement or double containment of piping.

2.1.5.6 Post Closure Cost

The Amulsar Mine closure cost bases and estimates are provided in Appendix 8.18 of the ESIA. The cost was reviewed for general consistency with standard practice. The cost estimates cover the major scope items. However, some cost items are questionable and the overall cost appears to be underestimated. Below are some of the key concerns:

- The post closure operation, maintenance & monitoring (OM&M) period is limited to <u>only five years</u>. In the US, regulatory requirements and guidelines for closure (e.g., RCRA 40 CFR Part 264.117; Nevada NAC 445A.446; USEPA, 2000) indicate post closure costs should be calculated for a revolving 30-year period[20] (minimum), especially when contamination sources remain. Post closure costs should include routine OM&M activities as well as periodic replacement, maintenance and repair actions that will be required after 5 years (e.g., treatment system components, liners, covers, containment systems, monitoring wells/points, piping, etc.), which can be significant. The shortened post closure monitoring period and the omission of the periodic replacement/maintenance costs will result in significantly underestimating the post closure costs. Clarifications of cost impacts of longer post closure monitoring duration are presented below:

    i. Appendix 8.18 of the ESIA provides a cost of $5,558,510 for monitoring and maintenance of the PTS. There is no breakdown for this cost. However, a footnote in the Cost Summary table (page 3 of 123 in Appendix A) indicates "*additional documentation required.*"

    Using the post closure period of five years assumed in the ESIA, the equivalent annual routine OM&M cost is estimated to be approximately $1.1M. Accordingly,

[20] The reference is to a revolving 30-year period whereby every year is year one, because the financial assurance should be reevaluated and updated annually.

the unadjusted cost (*i.e.*, without indirect cost) for the PTS OM&M alone for 30 years will be approximately $33.4M. The total cost will likely be higher due to the need to account for periodic maintenance and replacement items that occur after five years (not included in the ESIA costs) and the indirect and technical support costs. In fact, adjustment for the indirect cost referenced in the ESIA at a total of 21.3% (*i.e.*, 6% contingency, 10% contractor profit, and 5.3% contract administration) will result in a total cost for the PTS OM&M to be approximately $40.5M without adjustment for periodic replacement costs or realistic contingency (see comment below about contingency).

ii. Appendix 8.18 of the ESIA provides a cost of $410,576 for monitoring, which includes $286,252 for rehabilitation monitoring and maintenance and $124,324 for groundwater and surface water monitoring).

Similar to the PTS OM&M cost (based on a 5-year post closure monitoring period), the equivalent annual monitoring cost is estimated to be approximately $82,000. Accordingly, the unadjusted cost of monitoring for 30 years will be approximately $2.5M. The total cost will likely be higher due to the need to account for periodic maintenance and replacement of monitoring systems that occur after five years (not included in the ESIA costs) and the indirect and technical support costs. In fact, adjustment for the indirect cost referenced in the ESIA at a total of 21.3% will result in a total cost for monitoring to be approximately $3M (without adjustment for periodic replacement costs or realistic contingency).

iii. Combining items i and ii above with the other costs remaining unchanged indicates the total Mine rehabilitation and closure cost will increase from approximately $34M to approximately $70M (without adjustment for periodic replacement costs or realistic contingency).

- Contingency (scope and bid) is too low at 6%. The USEPA (2000) and AACE (2008a; 2008b; 2009) cost estimation guidelines indicate for this level of project development (pre-feasibility) and the high degree of uncertainty (*e.g.*, unreliable data and PTS and need for additional studies, *etc.*), the contingency on several items will likely exceed 20%. The Amulsar feasibility study (SGS, 2014 Table 21.5) used 16% for the initial capital phase. Clarifications of the effect of a more realistic contingency on the cost are presented below:
  - Using a more realistic contingency of 20%, the total indirect cost percentage would increase from 21.3% in the ESIA to 35.3%. Accordingly, the total rehabilitation and closure cost given in the ESIA will increase from approximately $34M for a contingency of 6% to approximately $38M for 20% contingency, without adjustment for the longer post-closure OM&M duration of 30 years.
  - Using the adjusted indirect cost with a 20% contingency and a 30-year post closure OM&M period, the total rehabilitation and closure cost would be estimated to increase to approximately $78M.

- Treatment scope and costs are unrealistic due to incorrectly assumed low leachate concentrations and mass loading and missing processes as discussed in Section 2.1.5.4.

- Professional/technical costs (design/engineering, project management/administration, and construction management) at approximately 3% of total construction/capital costs (on the order of $1M) are underestimated. USEPA (2000) indicates the professional/technical services are commonly greater than 15% of total construction costs for similar projects. The Amulsar feasibility study (SGS, 2014 Table 21.5) used 10% for the initial capital phase. Using 15% for total professional/technical services would result in an additional increase to the Mine rehabilitation and closure cost on the order of $4M to $5M.

- Many cost items are presented as lump sum without bases and cannot be assessed (*e.g.*, PTS maintenance and monitoring cost provided at $5,558,510 without basis; a footnote in the in Appendix A of the ESIA Appendix 8.18 states "*additional documentation required*").

2.1.6 Environmental Monitoring Program

2.1.6.1 Environmental Monitoring Plan

An environmental monitoring plan (EMP) was developed in June 2016 (Geoteam, 2016b; ESIA Appendix 8.12) for the pre-construction phase and was last updated in August 2018 for the construction phase and additional monitoring (Geoteam, 2018). The EMP includes acquisition of meteorological data at two stations in the Mine area (BRSF and HLF) and monitoring surface water flow and quality and groundwater levels and quality for compliance with RA regulatory standards and requirements, IFC Performance Standards, and of the European Bank for Reconstruction and Development Performance Requirements. The EMP refers to locations, parameters, and frequency for monitoring surface water, springs, and groundwater.

The EMP does not specify locations or details of future monitoring. The Project needs to develop a comprehensive plan for future monitoring (operations, closure, and post-closure). With respect to past monitoring locations, the following observations may be considered for future monitoring.

- Section 2.1.1.5 of this report addresses past surface water monitoring, including river flow and stage monitoring. Locations of continuous and spot flow measurements are posted on Figure 4.9.5 of the ESIA. Figures 4.9.6 and 4.9.7 of the ESIA show the locations of continuous monitoring on the three main rivers and tributaries, respectively. With respect to future monitoring requirements, the continuous flow monitoring stations established by the Project on the Arpa, Darb, and Vorotan Rivers would be generally adequate. A few apparent deficiencies in continuous flow monitoring are in the vicinity of Vorotan Pass. A station at the bend on the upper Darb River, where the course changes from northward to northwestward, would determine whether flow is perennial or ephemeral in that location (given the importance to groundwater flow modeling). Likewise, a station on the east side of Vorotan Pass on the upper reach of the (Porsughlu River?) flowing into Spandaryan Reservoir would serve the same purpose. A station should also be added to the stream below Benik's Pond to monitor potential effects of the Tigranes-Artavasdes pit.

- Surface water quality monitoring is addressed in Section 2.1.1.6 of this report. Surface water quality sampling locations are shown on Figure 4.10.1 of the ESIA. These locations would be generally adequate for future monitoring. A surface water quality monitoring location should be included on the main tributary of the Darb River downstream of station AW006, just before the confluence with the Darb River or just downstream of the tributary on the Darb River, to better assess the Mine impacts on surface water quality.

- Groundwater quality monitoring is addressed in Section 2.1.1.6 of this report. Groundwater and springs quality sample locations are shown on Drawing 4.8.2 of the ESIA. Few springs around Amulsar Mountain and the BRSF have been monitored. Given the importance of springs to livestock and the large number of springs that should be monitored, significantly increasing this number would offset the need for many additional groundwater monitoring wells. Assuming many more springs are added to the groundwater monitoring program, only a few more groundwater monitoring wells would be necessary near the mine pits. These wells should be located north-northwest of the BRSF, southwest of the Arshak pit, and east of the Tigranes-Artavasdes pit.

- Wells for monitoring groundwater levels and quality should also be installed near the Spandaryan-Kechut tunnel. Elevations of the tunnel would be required to determine whether discharge of groundwater is occurring. Tracer studies may be warranted to assess the seepage potential from groundwater into the tunnel.

According to the Cyanide Management Plan (Geoteam, 2016a; Appendix 8.11 of the ESIA), the HLF and process pond designs include sufficient measures to ensure that groundwater and surface waters will not be adversely affected under normal operating conditions. The Leak Collection and Recovery System (LCRS) of the HLF will enable the capture and diversion of any leaks in a closed system. The plan also states that monitoring wells will be installed beneath and close to the HLF location to establish baseline conditions and that additional wells will be required down-gradient of the HLF and solution ponds for pre-construction, construction, operation, closure, and post-closure monitoring of groundwater. Table 13 of the EMP indicates two sets of nested well pairs in the HLF footprint (GGDW013/GGDW013A and GGDW016/GGDW016A) and an additional well up-gradient of the HLF (GGDW011) will be sampled. Drawing 4.8.1 shows GGDW013/GGDW013A and some other wells in the footprint but not GGDW016/GGDW016A. The wells to be monitored down-gradient of the HLF and solution ponds are not specified.

2.1.6.2 Quarterly Environmental Monitoring Reports

Five quarterly environmental monitoring reports were available for review (Q1 – Q4 2017 and Q1 2018).

Nine surface water quality monitoring locations were sampled in Q1 2017. The number of sampling locations increased to a maximum of 16 in the most recent quarterly monitoring report (Q1 2018), which is much less than the number of stations shown on Figure 4.10.1 of the ESIA (39). Notably, no locations on the Darb River or north of the Kechut Reservoir (including Jermuk) were sampled. Most locations north of the BRSF, including the stream in the vicinity of the Madikenc springs, the Spandaryan Reservoir, two locations around Gorayk, and all locations east and west of Amulsar Mountain were omitted. These omissions are unjustified, especially for a deficient baseline dataset (see Section 2.1.1.6).

Five springs and groundwater quality monitoring locations were sampled in Q1 2017. A maximum of 21 locations was sampled in Q4 2017, far less than the number of stations shown on Drawing 4.8.2 of the ESIA (24 groundwater and 28 springs), which itself is deficient in springs sampling locations (see Section 2.1.1.6). There are some differences in locations sampled each quarter. Only one spring (SP83, Madikenc) in the GSA was sampled twice. This sampling program is unacceptable with respect to the number of locations and the deficiency in baseline data (see Section 2.1.1.6).

The monitoring reports do not include potentiometric surface contour maps or contour maps of key constituents in groundwater. There are no time-concentration graphs. There is no discussion of results with respect to previous results and no discussion of analytical methods.

2.2 Biodiversity

2.2.1 Baseline Characterization

2.2.1.1 Best Practice

The baseline characterization section needs to present clear bibliographical review enabling the identification of all habitats related to the project and all species occurring in the area and that could be potentially affected by the project. The section should also present an initial prioritization of the ecological significance of habitats and species.

All references cited should be made available for review and the ESIA/EIA report should clearly refer to date of consultation of the web references; moreover, the scientific literature and reports used to derive georeferenced data and maps should be properly cited and made available for review.

It is important that this section presents maps of main literature findings in terms of species reported and habitats. This should guide the preparation of an ecological significance map which would then guide the survey methodology (areas to be visited and intensity of survey) in view of available data and gaps. This could also serve as an Initial Environmental evaluation to orient project design in an attempt to reduce initial impact.

The baseline characterization section should present a clear methodology for data collection and baseline assessment. For each biological group (receptor) and for key species, the adapted methodology enabling the identification of key focus areas and protocols for field data collection to increase chances to confirm presence of a species should also be presented. This methodology should enable to confirm that all initial ecological significance of habitats and species have been duly considered.

In summary this section should present a synthetic table presenting for each receptor, who did the surveys, when, where, how, and survey field intensity. It should also display a methodological map showing the areas that were surveyed for each species and how the importance of the ecological significance was assessed.

2.2.1.2 Assessment

The Baseline Characterization in the ESIA/EIA presents a general habitat map highlighting nine habitat types but is missing a detailed habitat map showing the 30 habitat types mentioned in the report that would guide the identification of the ecological functionalities of the key species on site.

Information available in the baseline is not synthesized to enable a clear and direct understanding of the hierarchy of ecological significance; it also lacks reference to the status of those species in the Areas of Special Conservation Interest (ASCI) and the Bern Convention, as well as their local status in view of local evaluation of threats.

The baseline characterization section presents a description of species to be considered in the report while only distinguishing priority species (those reported in the Armenia Red Book) and the others. For instance, the report mentions 22 endemic plant species to be avoided where possible, though not in all cases, but refers to very few of those species by name.

Missing as well are quantified observations and locations of observations, and definitions of habitat for each species at stake, with relevant surface areas.

2.2.1.2.1 Habitats and plants assessment

The ESIA/ EIA highlight the importance of Armenia as a center of endemism for wild relatives of domesticated crops and center for breeding and selection of cultivated plants. Moreover, one of the aims of the ESIA/EIA is to protect plants and plant communities that have economic value and are used by others, specifically local people; however, efforts at identifying and conserving economically important plants at Amulsar, more specifically wild edible plants and crop wild relatives is lacking. The ethnobotanical survey is poor as translated, with vernacular, not Latin names of taxa. The ESIA summary explicitly states that the project will significantly change the rural landscape in which local people engage in traditional land management practices, but the project does not undertake efforts to survey and at least conserve the genetic diversity of economically important plants and crop wild relatives at Amulsar *ex situ* (through seed collection and preservation).

The distribution map of *Potentilla porphyrantha* shows all points that were sampled but does not clearly map the critical habitats for this species. The critical habitat of this endangered species is identified as "subalpine meadow with alpine elements" in which the species occurs on suitable rock substrate. The physical footprint of the project is estimated to be on 150.5 hectares (12.5% of the total area of critical habitat). This assumes that the species occupies the entire area of critical habitat (1200 hectares) when it occurs only on a subset of the area where suitable habitat occurs. Therefore, 12.5% is an underestimate of the area occupied by this species. The scientists involved in the assessment of this species seem to have undertaken a count of all individuals; it would have been assumed that such an effort would yield a precise map of where the species occurs in the study area, which would have significantly improved the estimate of project physical footprint and potential mitigation.

2.2.1.2.2 Insects

A clear description of the methodology used to justify the selection of the sampling points, which mostly fall outside the footprint area of the project, is lacking, thus seriously compromising the ability to properly understand the baseline situation.

A detailed map of the *Sedum album* host plant for the *Parnassius appolo* is needed to properly assess the habitat extent for the butterfly.

With regards to beetles, 14 species are reported in the Armenia Red book, of which *Dorcadion bistriatum* Motsch, *Dorcadion sisianum* Lazar and *Dorcadion scabricolle sevangense* are the most vulnerable endemic species of Armenia and should be considered as a conservation priority. *Dorcadion bistriatum* is reported in the vicinity of Ughedzor, Arpa river basin. A map of the habitat of these species of *Dorcadion* (endemic and vulnerable) is missing from the report to enable assessment of the species' potential presence on site.

2.2.1.2.3 Amphibians and reptiles

The report does not highlight the ecological significance of reptiles while two species of vipers *Montivipera raddei* and *Vipera eriwanensis* which are globally reported as vulnerable at International and National scale and reported in the red book and one additional species which is globally vulnerable and protected in the Red book of Armenia *Telescopus falax* occur and are reported in the direct footprint of the project. No further investigations were carried out to properly assess these priority species.

The baseline section lacks a list of key species of concern and a quantified estimation of the areas of suitable habitats that will be impacted.

The ESIA reports that the level of efforts invested for the field survey of reptiles and amphibians is not sufficient (7 days in total, for a total of 1800 ha, over 1 sampling season in non-optimal weather conditions) and yet no additional field work was done to complement this very preliminary assessment. This is especially of concern for the Ursini group of vipers, to which *Vipera eriwanensis* is affiliated. For instance, this viper inhabits very specific alpine steppes which could have been precisely mapped and specific counts could have been undertaken in order to establish the importance of the population. Throughout the report, this vulnerable species is said to be present in the general landscape without any precise scientific justification. Seven days is barely enough time to assess that the species is present without any notion of population quantities.

2.2.1.2.4 Birds

The methodology reported is exhaustive and comprehensive, covering all areas of concern. However, as no night surveys were conducted for the assessment of protected species of birds such as Eagle owl *Bubo bubo,* their overall presence is under evaluated in the baseline. As reported, night surveys were undertaken for the corncrake but surveys for the Eagle Owl should have been carried out earlier in winter when the species exhibit mating calls and territorial behavior.

Mapping and surface estimate of the functional areas for key species of concern (nearly 15) is missing from the ESIA report and should have been highlighted.

Furthermore, the presence of a breeding colony of Lesser Kestrel (*Falco neumanni*) in the Goryak IBA which uses part of the footprint of the project as a key area for hunting during breeding period is not precisely presented, and therefore impacts on this colony are underestimated throughout the report.

2.2.1.2.5 Bats

Functional analysis was performed in April, while activity was measured in May/ June. This enables the evaluation of early reproduction for bats; however, July and September activity measures are essential for assessing the real presence of bats at these altitudes.

Complete functional mapping of the bats is not possible using the given data.

2.2.1.2.6 Mammals

Especially for the Brown bear (*Ursus arctos*), the methodology is comprehensive but presentation of results could have been improved by adding a map with habitat location with respect to project components.

2.2.2 *Impact Assessment on Biodiversity*

2.2.2.1 Best Practice

To enable proper assessment of impacts on biodiversity (habitats and species), the ESIA/EIA Impact assessment section should provide a clear presentation of project location and geographical extent.

The detailed description of impacts should include the type of impacts (direct/ indirect), a quantified (surfaces of species habitat) estimation of the impact, its geographical extent as well as its duration and consequences. This should serve to evaluate the significance of the impact over the various phases of the project cycle (construction/operation/closure-post closure).

The section should also provide a clear method for the evaluation of the significance of the impacts and should cover the impact on species, habitats (especially key ecological species and critical habitats), as well as protected areas and areas of biodiversity importance.

The report should distinguish the initial impacts (before mitigation), the mitigation measures (avoidance and reduction) and residual impacts (after implementation of mitigation measures). Any residual impact should be addressed in the Biodiversity offset plan to reach a global NNL (no net loss from the project) or when possible a Net Gain.

2.2.2.2 Assessment

The section on impacts in the ESIA presents the mitigation measures instead of presenting the initial impacts and in a separate section the mitigation measures suggested and the resulting residual impacts. This is rather confusing for an evaluator.

The lack of consistent description/nomenclature in the baseline section has led to a confusing identification of the various receptors to be considered in the impact assessment section.

The section lacks a synthesis table showing for each receptor the initial impact, the suggested measures and the residual impact, and displaying figures on the extent and consequences including impact on ecological functionalities.

The impacts from accidental events are also missing from the report.

Monitoring and offset are considered in the ESIA report as mitigation measures. This is misleading to the evaluator as monitoring is part of the BMP (Biodiversity Management Plan) and offset is to be planned to address residual impacts on receptors.

The report includes different estimates of the footprint of the project on different biodiversity elements; for example, the impact is estimated to be 150.5 out of 1200 hectares for *Potentilla porphyrantha* but the total area of impact is 1766 ha + 160 ha which is confusing; for each receptor the report would benefit from a clear table with clearly annotated impacted surfaces.

The evaluation of the significance of the impacts is not based on quantified figures and therefore is not defendable.

As examples:

On habitats and plants

The impact on the subalpine meadow with alpine elements has been under-evaluated – in view of the extent of the areas that will be impacted. The importance of the impact should be "significant".

For *Potentilla porphyrantha* the report mentions an overall positive impact in the long term. This statement is too ambitious and is not based on conclusive findings in the report.

On reptiles and amphibians

For the vipers (especially in the urisini group), essential data is missing from the baseline to enable proper assessment of the impacts. The ESIA documents a non-significant residual impact without stating specific mitigation measures. The Ursini vipers are protected species, reported Vulnerable at global level, and the impact cannot therefore be considered 'Neutral" (as reported in the ESIA). Besides, the justification provided states that these vipers occur in wider landscapes without providing supporting data from the literature to confirm this finding.

On birds

For the Lesser Kestrel, the loss of feeding habitat is not to be considered neutral as the only colony feeding in Armenia is feeding on the project site. Besides, the hunting area of the species is usually close to its feeding area (which is convenient for the chicks and to teach them to hunt during their juvenile phase). In this case the predictive impact is to be considered "significant" both on the ecological functionality and on the Goryahk IBA.

The overall impact of the project on the Goryahk IBA is under evaluated especially in view of the project impacts on Lesser Kestrel and Egyptian vulture.

The Eastern rock nuthatch is a protected species. Residual impacts are not reported properly for this species; moreover, the species should be considered in the offset program.

The ESIA report considers "other birds" as a single group while they all use different habitats and different ecological functionalities for reproduction, nesting, hunting, feeding, resting. They should have been considered separately in view of their specific ecology in order to properly evaluate the impacts.

On Mammals

For the Brown Bear, the report mentions an overall positive impact of the project (presented as Net Gain). This is misleading and uncertain, as we have no guarantee that the Brown Bear will stay in the set aside areas; moreover, the report does not present a quantified estimate of the area of critical habitat of the Bear that will be impacted by the project. The predictive impact should remain "significant" on short and medium terms and offset measures should be considered.

2.2.3 *Mitigation Measures*

2.2.3.1 Best Practice

The mitigation section should present geolocalized, implementable measures and a map summarizing the measures. It should also describe, for each receptor, the residual impact and the eventual need to include the receptor in an offset program.

2.2.3.2 Assessment

Table 6.9 in the ESIA lists the mitigation measures. Those measures are too general and not always geolocalized to enable their implementation.

Some measures presented in the mitigation section cannot be considered as mitigation; the report suggests (Table 6.11) that monitoring would be carried out, and in case ongoing monitoring proves no residual impact, then in this case, the measures could be revised and reduced. The approach is misleading as mitigation measures should be proposed to address initial impacts, and in case of residual anticipated impact, then an offset program should be properly included. Without these, the IFC PS6 and EBRD PR6 are not properly addressed through this ESIA report.

As examples:

On habitats and plants

Translocation for *P. porphyrantha* should be considered an ongoing experimental measure and cannot be considered as mitigation, since success is not guaranteed.

The mitigation measures suggested for *Potentilla* state that:

> *« If research, monitoring and modelling suggest that pre-mining population size and the extent of the population cannot be restored, a comprehensive review of offsetting options will be undertaken».*

This review should consider protecting the remaining populations of RA if they are vulnerable or threatened. If not, what could possibly be considered for offsetting the loss of such an endangered species? Reintroduction of the species in the restored pits is not really to be considered given that the conditions will not be favorable (altitude will be drastically lower and being in a pit will induce quite different local conditions than summit conditions).

The storage of top soil (40ha) is recommended without clear location of the areas of storage. The report does not mention if these are accounted for in the footprint area calculation.

On reptiles and amphibians

The unique mitigation measure suggested for vipers is related to the reduction of the areas of direct impact; however no specific measures are suggested; moreover, the areas of direct impact for each protected species are not geolocated and their areas are not calculated.

On birds

Monitoring of Lesser Kestrel is a management measure and cannot be considered as mitigation. A possible mitigation for this bird could have been the identification in a close range of the breeding colony of degraded areas for hunting purposes in order to undertake restoration actions.

In addition, in table 6.11.13- it is stated that:

> *"species action plans have been produced but additional data is needed to develop mitigation measures".*

The ESIA should document clear measures and not suggest future (eventual) identification of measures. The precautionary principle should apply in case residual impact evaluation is not possible in view of available results and data.

2.2.4 Environmental Management plans

2.2.4.1 Best Practice

In line with IFC's Performance Standard 6 (2012) (PS6) and EBRD's Performance Requirement 6 (PR6), Lydian International aims to achieve 'no net loss' of natural habitat and a 'net gain' outcome for any residual impacts on critical habitat.

This section should include:

- Biodiversity management plan
- Biodiversity Action Plan and offset measures
- Mine closure and restoration plan
- Biodiversity monitoring plan

The offset is based on three main principles:

- Like for like
- Same geographical area
- Same time frame (or before) the project's impacts

2.2.4.2 Assessment

2.2.4.2.1 Biodiversity Management Plan (Appendix 8.21)

As presented, the biodiversity management plan is missing the operational section and map detailing the measures, their location, how to implement and who is responsible for implementation, and mostly how will those measures reduce the impact of each receptor (quantified estimates). As such the section is viewed as a "general recommendations" section rather than clear commitment from the project' owner.

In particular:

Bio 5 does not provide needed details on the surveys to be conducted nor period of the year.

> *"Pre-construction checks (surveys) will be carried out immediately prior to ground disturbance in order to confirm that the biodiversity baseline as reported in this ESIA has not changed significantly and that there are no additional features that should be avoided"*

Bio 8 provides a general statement that can hardly be translated into concrete actions.

> *"As a fundamental design principle, the footprint of Project infrastructure and the areas of land to be cleared will be minimized".*

Bio 9 should also include avoidance of priority/protected species with its habitat.

Recommendations/ commitments provided in Bio 24 should also be presented in georeferenced/ map form to enable proper evaluation of relevance and location of disturbance.

> *"Where practical, noisy construction-related activity will be avoided at dawn and dusk and during the night (see also noise & vibration impacts)".*

Bio 44 is missing a map to enable proper evaluation of relevance of the action.

Bio 46 is too vague to enable proper assessment of relevance; it is also not clear what is meant by good examples.

> *"Topsoil storage locations will be chosen to avoid "good" examples of natural vegetation types as well as rocks supporting Potentilla porphyrantha".*

Bio 50 is missing a map to illustrate the action in relation to the projects' components.

Bio 53 and Bio 54 are missing details on the monitoring measures.

2.2.4.2.2 Biodiversity Action Plan and Offset Measures (Appendix 8.20)

The establishment of the Jermuk National Park (JNP) is presented as the main offset measure for the project. However, the relevance of the added value of the JNP on biodiversity receptors affected by the project is yet to be demonstrated.

Natural habitats and plants

The "like-to-like" principle of the offset if not clearly demonstrated.

> *« An offset of 837 Habitat Impact Units (HIU) is required to achieve NNL of natural vegetation due to long term degradation and loss associated with Project development ».*

Of those 837 units, only 500 are reported from JNP.

The summary of residual impact highlights the need to offset 22 species of plant, however no evidence is provided on the fact that these 22 species potentially occur in the JNP area. Moreover, a list of the 22 endemic plants concerned is not included. Neither is a list of species occurring at JNP for comparison.

The list of plant species provided for each habitat suggests the presence of crop wild relatives in the area. One species of *Cicer* was also reported from the area in the past but was never found. It is surprising that no effort has been made to compile a list of crop wild relatives (and other economically important plant species in the area) and suggest measures to mitigate the impact of the project on these species.

As *P. porphyrantha* is not spontaneously found in the JNP, the conditions may not be suitable for the species. A statement such as « Research is ongoing on its ecology and growing conditions as outlined in the Species Action Plan, together with research on restoration techniques and searches for other populations in Armenia » cannot be considered valid for the JNP.

Figure 8 displays the distribution of *P. porphyrantha*; such a map would have been useful in the baseline and impacts sections and should have been developed for key species of concern.

Reptiles

Offset cannot be limited to protection from deliberate killing of snakes (where in the report the killing of snakes was evaluated as major impactful activities).

> *« Residual impacts are likely and can be offset through protection of reptiles and their habitats within the proposed Jermuk National Park, together with local awareness-raising about conservation importance to reduce levels of deliberate killing of snakes».*

The report does not clearly demonstrate the possible Net gain on vipers within the future Jermuk National Park. No supporting elements are provided to assess the validity of this statement.

The management actions suggested for viper by using prescribed fire, has proven drastic negative results in literature (http://www.vipere-orsini.com/fr/program-life-nature).

Moreover, the current conservation status of existing population of viper in the JNP is not properly assessed to justify eventual management actions for vipers.

> *« The residual impact on regional numbers of these three species is expected to be small since ample habitat is present outside of the Project-affected area. In the longer term, residual impacts may be detected through monitoring. Positive conservation measures may be needed to compensate for reduced habitat extent and quality in the longer term and to this purpose restoration measures could be undertaken within the proposed Jermuk National Park. »*

Birds

The Eastern rock nuthatch is missing from the offset program while it should be considered as this species is a protected species and impact on its population is significant (table 6.11).

The residual impacts anticipated on the Lesser Krestrel, are probable on the only nesting colony in RA.

Therefore, the statement

> *"Residual impact from the projects are possible but would be confirmed through monitoring"* and *"No specific conservation measures are currently proposed but in theory it may be possible to extend breeding range into JNP"*

cannot be considered satisfactory and an inclusion of the Lesser Krestrel in Offset program is required through active reintroduction program in the JNP.

The section presents some actions addressed to the "other birds" group.

> *« There are a number of actions that could be taken with respect to Project operations that might further reduce the risk of impacts to breeding birds in general ... At the moment these are presented as benefits for consideration, rather than required mitigation measures, but they may become more*

important depending on the results of the monitoring that will be ongoing during Project execution. »

This can hardly be considered in a Species Action Plan where every species has a specific ecology and consequently specific needs and cannot be grouped together, and because every possible effort could be considered. Some avoidance measures such as planning land preparation in view of the breeding seasons (cutting the bushes and earthworks would make the site inappropriate and direct destruction of nestlings (eggs and chicks) could be avoided.

Mammals

Set aside measures are suggested for the Brown Bear with no information as to their actual implementation and possible monitoring of efficiency to host the bears individuals.

« Surveys in 2015 confirmed the importance of the woodland north of Saravan, situated 1.5 km east of the HLF. This was used by at least 6 bears. Extending the Set-aside westwards to include this forest would make it more ecologically viable and suitable for bears. This possibility will be discussed when the boundary of the Set-aside and its proposed management are formalized with stakeholders in 2016 ".

As long as efficiency of set aside is not proven successful, a more in-depth program of offset should be planned.

Gorayk Important Birds Area & Key Biodiversity Area

The report confirms that no direct impact is to be foreseen on the Goryak IBA while the major impact is the one related to the Lesser Krestrel colonies feeding on the area of the project.

« The Project will not have a direct impact on the IBA. Measures to mitigate impacts on species originating from the IBA that use the Project-affected area - particularly Lesser Kestrel - are included in Table 6.11.11. No further mitigation measures are necessary. »

This statement is not based on solid considerations and indirect impacts on the IBA are likely to occur.

Most of the offset measures are postponed to eventual future results of monitoring of suggested mitigation measures.

Similarly, the Species Action Plans developed for *Potentilla porphyrantha* and *Ursus arctos* report:

« These have been produced for the two critical habitat species affected by the Project, Potentilla porphyrantha and Ursus arctos for which final analysis of survey data is needed before the Project mitigation strategy can be finalized. »

The ESIA reports many promises while it should document commitments; and the ESIA is supposed to be completed at date of submission and the latest statement undermines that the mitigation strategy is not yet fully developed.

The Biodiversity Offset Strategy (BOS) is missing a summary table summarizing for each receptor:

- Surface of critical habitat of this receptor and if possible number of individuals possibly affected by the project before mitigation,
- Mitigation measures
- Surface of critical habitat of this receptor and if possible number of individuals possibly affected by the project after mitigation (residual impact),
- Units lost due to the project after mitigation
- Offsetting measures
- Units gained by offsetting program
- Balance, NNL or Net Gain

2.2.4.2.3 Biodiversity Monitoring Plan (Appendix 8.12)

The Biodiversity monitoring is presented as part of the Environmental Monitoring plan (EMP), however no specific indicators for monitoring are provided for the natural habitats and biodiversity.

2.2.4.2.4 Mine closure and rehabilitation plan (Appendix 8.18)

This section is very general and in its current state is not directly implementable and operational. Actions are presented as objectives or aspirations and are built on experiments with no conclusive results. Ecological engineering techniques are not described in detail to enable proper assessment of relevance/adequacy.

The extraction of the turf, the storage as well as the propagation of *P.porphyrantha* are experimental measures and no guarantee related to the success of these interventions is provided.

It is current state, the evaluation of the applicability and adequacy of the post closure/ rehabilitation program remains uncertain.

2.3 Air Quality

2.3.1 Baseline Characterization

This part of the assignment assesses the methodology and results of the baseline conditions for air quality.

2.3.1.1 Expected Emissions Sources and Pollutants

The project consists of mining activities including blasting, loading and unloading of material, transport of material, crushing, but also the Gold processing including the auxiliary activities of electric power generation, organic liquid storage, and combustion of boilers. These activities will generate emissions into the air with the main pollutants being: CO, NOx, SO_2, TSP, PM_{10}, $PM_{2.5}$ but also some that are more specific to the gold processing like Hg, HCN, and HCl.

2.3.1.2 Air Quality Regulations

The air quality standards chosen by Lydian were those of the International Finance Corporation (IFC) (IFC, 2007). IFC adopts the World Health Organization (WHO) (2006) air quality guidelines in the absence of national air quality regulations. The IFC standards are generally the stricter in the world and are solely based on health impact without taking into consideration the socio-economic conditions of any country. The Maximum Permissible Concentrations (MPC) at the Republic of Armenia present generally higher standard values. From these IFC standards, the pollutants chosen to be monitored in order to determine the baseline are NO_2, SO_2, PM_{10}, and $PM_{2.5}$ and these are stricter than the Armenian standards. On the other hand, HCl and HCN are not considered in the WHO guidelines and Hg exhibits stricter values in the Armenian standards (Decision No 160-N of 2 February 2006) than those presented by the WHO (2000).

Moreover, the IFC indicates that emissions resulting from a project shall not contribute to more than 25% of the applicable air quality standards to allow additional, future sustainable development in the same airshed. This requirement was not considered by Lydian in the ESIA.

2.3.1.3 Meteorological Data Measurements

Meteorological data was considered for wind mainly at the Vorotan Pass from 1966 to 2013. Results show that a dominant wind comes from the East with an average wind speed of around 4 m/s. However, the dominant direction does not imply that lower wind direction frequencies cannot result in high pollutants concentrations since their dispersion is function of many other parameters like topography, land use and land cover.

2.3.1.4 Measurement Sites

Different measurement sites were chosen to monitor the concentrations of the ambient pollutants, especially at the settlements level regardless of the wind patterns: Gorayk (4.4 km south of Tigranes/Artavazdes pit), Saralanj (3.7 km west of Tigranes/Artavazdes pit), Gndevaz (1 km west of HLF), Gndevaz Livestock and Diary Farm (700 m west of truck loadout), Kechut (< 1 km).

2.3.1.5 Methods Used for the Measurement of Pollutants

For the measurement of NO_2 and SO_2, the passive sampling method was used which is acceptable (UK Environment Agency, 2011) along with the Light-scattering optical particle counter for PM_{10} and $PM_{2.5}$ which is also an acceptable method (UK Environment Agency, 2011).

The passive samplers were from Gradko and IVL known for this kind of measurements. The passive sampler traps targeted molecules which are extracted in the lab and their concentrations determined. It is generally exposed to ambient air for few weeks.

As for PM mass concentration, the Osiris Turnkey and EPAM 5000 were used for the measurements PM_{10} and $PM_{2.5}$. Osiris Turnkey is MCERTS certified (UK Environment Agency's Monitoring Certification Scheme for equipment, personnel and organisations). Osiris measures PM_{10} and $PM_{2.5}$ simultaneously while EPAM 5000 measure either PM_{10} or $PM_{2.5}$ depending on the configuration.

DustScan100 is a directional dust gauge used for directional dust monitoring and gives a qualitative assessment of fugitive 'nuisance' dust emissions and deposition (IAQM, 2012). It is generally installed over 7 to 14 days. The surface soiling method involves the measurement of the loss of surface reflectance, expressed as Effective Area Coverage (EAC%).

The methods for the assessment of the baseline are valid and acceptable for the above-mentioned pollutants. On the other hand, since contaminants from such project like heavy metals are potentially released and since soil metal concentrations for some elements determined by Lydian were found not to be negligible, chemical speciation of aerosols at the receptors should have been conducted to evaluate the impact the project might have during operation on the change in aerosol composition, ***only if the aerosols concentrations from the project were found to have a non-negligible impact on air quality at the receptors'***. Moreover, no baseline assessment was conducted for gaseous Hg, HCN, and HCl.

2.3.1.6 Results of the NO_2 Concentrations

NO_2 sampling was conducted over 4 weeks each time in the main receptors mentioned above from September 2011 to January 2012 and from January to December 2014. Six locations were added alongside the main receptors namely AQ1 to AQ6 and monitored from August to October 2015. The locations and period are considered acceptable (UK Environment Agency, 2011).

Results reflect an average over the entire sampling period, ie. 4 weeks. The highest recorded concentration was at Gorayk in November 2011 with 12.34 µg/m^3.

No NO_2 measurements were conducted with other instruments for shorter periods, ie. 1-hr to assess the compliance with the 1-hr averaging period as per the IFC guidelines (2007). However, this is a generally acceptable approach.

If one uses the empirical relationship to estimate the highest 1-hr concentration (Ontario, 2009) as per the IFC and WHO guidelines of 200 µg/m^3, a value of around 78 µg/m^3 is determined, way below the 200 µg/m^3 limit. This shows that the conclusion drawn by Lydian that with these concentrations it is highly unlikely to exceed the 1-hr is acceptable. The results show that annual average is compliant with the IFC and WHO of 40 µg/m^3.

2.3.1.7 Results of the SO_2 Concentrations

SO_2 sampling was conducted also over 4 weeks each time in the main receptors mentioned above from September 2011 to January 2012 and from January to December 2014. Six locations were added alongside the main receptors namely AQ1 to AQ6 and monitored from August to October 2015. The locations and period are considered acceptable (UK Environment Agency, 2011).

Results reflect an average over the entire sampling period, ie. 4 weeks. The highest recorded concentration was at Saravan in December 2011 with 4.77 $\mu g/m^3$.

No SO_2 measurements were conducted for shorter periods with other instruments, ie. 24-hr to assess the compliance with the 24-hr averaging period as per the IFC guidelines (2007). However, this is a generally acceptable approach.

If one uses the empirical relationship to estimate the highest 24-hr concentration (Ontario, 2009) as per the IFC and WHO guidelines of 20 $\mu g/m^3$, a value of around 12.5 $\mu g/m^3$ is determined, way below the 20 $\mu g/m^3$ limit. This shows that the conclusion drawn by Lydian that with these concentrations it is highly unlikely to exceed the 24-hr is acceptable. The results show that the 2014 monthly concentrations were also very low (<2 $\mu g/m^3$).

2.3.1.8 Results of the PM Concentrations

Dustscan samples were analyzed by DustScan in the UK. The results showed that dust deposition is low and has very limited impact.

PM_{10} and $PM_{2.5}$ were monitored with Osiris and EPAM at Gndevaz and Kechut. It is not clear why $PM_{2.5}$ was not measured along with PM_{10} or at least if measured the data should have been presented. However, conclusions reached by Lydian at the surrounding villages are acceptable and in line with the monitoring reports that followed. At AQ9 (West of Tigranes /Artavazdes) and at AQ10 (North of BRSF), results are also acceptable even though the number of samples at AQ10 is low (3 samples taken in July/August 2015) since one expects low concentrations of PM_{10} and $PM_{2.5}$.

2.3.2 Impact Assessment on Air Quality

Air quality at receptors' locations is modified through the transport of the pollutants emitted from the different project sources to these locations. It is linked to the quantity of pollutants released and the dispersion of these taking into consideration the topography that plays an important role and the meteorological parameters.

Different sources of pollutants in this project exist: the fugitive dust from the mining activity, the emissions from the road transport, the boilers emissions and the Gold ore processing including Heap Leach Facility and Adsorption-Desorption-Recovery (ADR) plant.

It is worth noting that electrical diesel generators are mentioned in the SOP for Air Quality Management and monitoring for the construction phase but these sources are not mentioned in any assessment of this section.

2.3.2.1 Fugitive Dust

The mining activities (excluding exhaust emissions from road transport) generate mainly fugitive dust but also negligible quantities of other pollutants like CO, NOx, etc. from blasting.

The sources considered in the ESIA are:

- Emissions from Overburden Removal
- Emissions from Boring / Blast Hole Drilling in Artavazdes Pit
- Emissions from Blasting in Artavazdes Pit
- Emissions from the crushing process
- Emissions from the screening process
- Emissions from Material Handling including loading and unloading of trucks
- Emissions Due to Wind Erosion of Stockpile Surfaces
- Emissions from haul roads

The pollutants considered are TSP and PM_{10}. Emission factors used in the ESIA for these pollutants are from the Australian NPI (2012) and the USEPA AP-42 as indicated in the ESIA document.

The EIA and ESIA considered only Artavazdes and according to these documents, the highest activity level (including construction and closure of mine) will take place in year 3 of the project that is when Artavazdes will be exploited. This approach is considered as a worst-case scenario and is acceptable. Moreover, the trucks that will be used for transport are solely considered in the fugitive emissions from disturbed unpaved surface, meaning that wheel loaders, etc. are not considered, however this approach is acceptable since the main emitter from the disturbance of unpaved surfaces are trucks.

Some concerns are raised in the emissions calculation: the PM_{10} emission factor for the low moisture secondary crushing operation taken from Australian NPI (2012) that is based on USEPA AP-42 chapter 11.24 is considered "0" whereas both references indicate that no data is available. Other sources should have been considered or TSP emission factor of 0.6 kg/Mg used as a worst-case scenario.

$PM_{2.5}$ was not considered in the assessment. The Australian NPI (2012) does not consider $PM_{2.5}$ in its emissions estimation techniques. As per the ESIA, the fine particles travel to distances over 1000 m and can therefore impact the human receptors. On the other hand, $PM_{2.5}$ is a regulated pollutant in the Armenian standards and the IFC guidelines, therefore it must be considered anyway.

The emission factors for $PM_{2.5}$ could not be taken from the Australian NPI (2012) but part of the above-mentioned sources have $PM_{2.5}$ emission factors in the USEPA AP-42. In addition to that, other references could have been considered like the Canadian Pits and Quarries Report guide or the Mojave Desert (2013) guidance that do have $PM_{2.5}$ emission factors..

In the EIA on the other hand, some additional sources were considered on the Crushing and sorting nodes. This increases the particles emissions by around 10%.

The impact assessment of these emissions on air quality at the receptors was conducted in the ESIA and EIA for TSP and PM10 with the same approach.

Nuisance dust:

Dust in the community is normally perceived as an accumulated deposit on surfaces such as washing, window ledges, paintwork and other light-colored horizontal surfaces, e.g. car roofs. When the rate of accumulation is sufficiently rapid to cause noticeable fouling, discoloration or staining (and thus decrease the periods between cleaning) then the dust is generally considered to be a nuisance. The point at which an individual makes a complaint regarding dust is highly subjective.

In the UK and Europe there are no definitive standards for deposited particulates, however, criteria and guidelines have been developed in many other countries. Studies undertaken in Australia, for example, have resulted in the adoption of a deposited dust criteria linked to the onset of loss of amenity of about 133 mg/m²/day, averaged over one month. In the UK, long term deposited dust nuisance criteria have been suggested for urban/semi-rural areas at, typically 200 mg/m²/day, averaged over a monthly period. The range around the globe varied from 133 to 350 mg/m²/day (Vallack and Shillito, 1998).

Customs and practice at quarries, coal, construction and demolition sites have used the figure of 200 mg/m²/day as a nuisance threshold for sites in the UK for dust deposition averaged over 1 month (IAQM, 2016).

The ESIA and the EIA proposed a model for the TSP deposition based on Arup report, Schmitz (1994) and ISO12013-1. The model is based on the decay of the different sizes of PM and their settling distance with a grid of 25 m x 25m. The levels considered in the ESIA and EIA are 133 and 350 mg/m²/day. The model is considered acceptable for TSP and shows that the impact of the deposition is negligible on the receptors.

PM_{10}:

As for the PM_{10}, the screening model AERSCREEN shows that 90% of the PM_{10} will be deposited at 500 m from the site and 99% at 1000 m. With sensitive receptors located at around 1000 m, the impact would be negligible.

In fact, the project emissions are not negligible and a conclusion giving only percentages is not reliable since the absolute concentration is needed to check whether the exceedances are expected at the settlements' locations or not.

However, in 2016, the Institute of Air Quality Management (IAQM, 2016) published the Guidance on the Assessment of Mineral Dust Impacts for Planning. It indicates that "it is commonly accepted that the greatest impacts will be within 100 m of a source and this can include both large (>30 µm) and small dust particles. The greatest potential for high rates of dust deposition and elevated PM_{10} concentrations occurs within this distance. Intermediate-sized particles (10 to 30 µm) may travel up to 400 m, with occasional elevated levels of dust deposition and PM_{10} possible. Particles less than 10µm have the potential to persist beyond 400 m but with minimal significance due to dispersion."

Moreover, the IAQM (2016) states that if no sensitive receptor is located within 1 km from the activity site, an assessment for nuisance dust and PM_{10} is screened out. This is in agreement with the ESIA and EIA conclusions on this matter.

On the other hand, the guidance does not provide a clear position as per the $PM_{2.5}$. Therefore, proper modeling should have been conducted to assess the impact of $PM_{2.5}$ especially that many parameters shall be taken into consideration like the complex wind field, the land use, and the different emissions locations.

2.3.2.2 Road Transport Combustion Emissions and their Impact

Road transport emits pollutants from the haul road and from the combustion of fuel. Fugitive emissions from haul roads were considered in paragraph 2.3.2.1 and showed that its impact is generally limited. This paragraph will consider the fuel combustion emissions from road transport.

The EIA presented the exhaust emissions of the different vehicles in the project and the impact assessment while the ESIA presented only the impact assessment.

The EIA presented only the total quantity of diesel expected to be used within a year and the emission factors adopted for the calculation of emissions without stating their reference. The emission factors were compared to the off-road values of the EMEP/EEA 2016 guidebook (EMEP/EEA, 2016) and were found to be higher which gives credibility to the emission factors values presented.

Then, the assessment is based on the UK Design Manual for Roads and Bridges (DMRB Volume 11, section 3, Part 1, HA 207/07) which indicates that receptors within 200 m from the road shall be considered in the assessment. Beyond that, impact is negligible.

Indeed, since all settlements are beyond 200 m, the impact of the combustion of fuel from road transport is considered negligible.

2.3.2.3 Boilers Emissions

Boilers are used in the project for the oven, the kiln but also the heating of the buildings.

The EIA presented the exhaust emissions of CO and NOx only from the combustion of natural gas. Neither PM nor SO_2 were considered even though their emissions are negligible.

The method used for the emissions calculation in the EIA is the one approved by the N268 A ordinance of the minister on 23 October 2012. The emission factors used are higher than the ones from the EMEP/EEA guidebook (2016). The difference is acceptable.

The assessment was considered by modelling using the Raduga model as described in paragraph 2.3.2.5.

2.3.2.4 Gold Ore Processing Emissions

The Gold ore processing exhibits different steps and a variety of techniques. At most facilities, the core step is extracting the gold from the crushed ore using cyanide and carbon adsorption.

The EIA indicated that based on the experience of international experts, the maximum percentage of emitted pollutants in an ADR plant is 0.5% of the input quantity while considering Cyanide and Hydrochloric acid only with an annual demand of 1000 tons and 505 tons respectively. On the other hand, the ESIA indicates that mercury emissions are to be mitigated.

The Australian NPI (2006) for Gold Processing presents a methodology for the estimation of the emissions from Gold ore processing based mainly on a mass balance approach. This includes the leaching and adsorption processes. The main potential emissions to air are HCN, and metals like Hg. Volatilisation is expected to be the most significant cyanide emission pathway from HLF but there is no reliable method for estimating emissions of cyanide because its pathway cannot be distinguished from other fates such as natural decomposition (Australian NPI, 2006). This mass balance approach needs very good knowledge of the process parameters to be implemented since for instance, a small variation of pH will affect significantly the emissions of HCN at the adsorption process. On the other hand, the Australian NPI (2006) considers that approximately 1% of the total cyanide added to the leach circuit is lost through HCN volatilization to the air from the leach/adsorption train. This is for example a deviation of what the EIA considered to be the maximum percentage which is 0.5%.

Another example that illustrates the complexity considered in the Australian NPI manual (2006) results in around 7% of cyanide input in the circuit in terms of HCN emissions through the different pathways of volatilisation at the leaching/adsorption steps but also the tailings storage facilities. Moreover, SGS application of the SART process to heap leaching of gold-copper ores at Maricunga in Chile considers 5% volatilization for HCN. This is to say that the emissions will be best estimated by the manufacturer of the process taking into account the right flow diagram tailored to the case of Amulsar project, the process type, and the different parameters' values that are designed for this Amulsar project.

In the EIA, HCN and HCl were considered to be mitigated with a water scrubber at 86% efficiency.

In addition to that, the US EPA 40 CFR Parts 9 and 63 related to the emissions of Hazardous Air Pollutants in Gold Mine Ore Processing published in 2011 considers mercury one of the main pollutants from such process and established emission standards to it.

It was only mentioned in the ESIA that Hg concentrations in the ore were at a maximum of 0.05 g/t and that since Hg was detected on the loaded carbon columns in leach tests, a potential exists for the volatilization of mercury.

However, since no local emission standards exist for this type of industry and since HCN, Hg, and HCl have local air quality standards, modeling should be conducted to assess the emissions allowed to be released (c.a. determination of the emission limit value) in order not to breach the air quality standards taking into consideration the background concentrations for these pollutants.

The assessment was considered by modelling only HCN and HCl using the Raduga model as described in paragraph 2.3.2.5. No modelling was considered for Hg.

2.3.2.5 Modeling of the Emissions Released by the Boilers and Gold Ore Processing

The modeling exercise using Raduga model was conducted for CO and NOx from the boilers and HCN and HCl from the ADR plant, and only in the EIA.

No documentation was found or provided for Raduga model to assess its adequacy for Amulsar case. It is the regulatory model in the Republic of Armenia. What can be indicated based on the input file provided:

- Raduga is only a screening model
- It is a Gaussian model
- It does not take into account building downwash

- It only gives the highest concentration calculated
- It has a simple coefficient to account for topography
- It does not include a complete terrain module
- It does not take into account observed meteorological data hourly to assess the resulting concentrations
- It calculates wind directions for every 10-degree sector
- Values calculated are 1-hour average
- It cannot handle complex wind field

In summary, the model is not adequate for this assessment even though it is requested in the Republic of Armenia.

When the model was run, the resulting concentrations were not added to the background values to assess the breach of the local air quality standards.

2.3.3 Mitigation Measures

The EIA and ESIA presented both the same mitigation measures and they are all relevant but their effectiveness in ensuring standards are not breached is uncertain given the identified deficiencies in the baseline and impact assessment. Some measures can be added to further minimize the impact:

- Use of high screens on roads and where relevant (next to stockpiles for example) as barriers to supplement the dust suppression. These can be added on one side or both sides of the roads depending on the location.
- Watering of the soil before blasting and the ore before loading into the trucks
- Trucks should be covered to minimize the wind erosion even though at low vehicle speed
- Use more efficient control equipment or increase the efficiency of planned control equipment on stationary sources especially the ADR plant where needed: a scrubber is already mentioned in the EIA to be installed for HCN and HCl but its efficiency shall be compared to the needed one so that the project complies with the air quality standards and the specifications shall be amended in case compliance is not attained (ex. addition of alkali solution concentrations, etc.), a retort furnace is also planned to be installed and a quantity of less than 60 kg of Hg is foreseen to be collected but the efficiency shall also be checked in a way to ensure compliance with the air quality standards and the specifications shall be amended in case compliance is not attained (ex. Use of scrubbers, activated carbon filters, etc.).

2.3.4 Environmental Monitoring Program

The proposed monitoring strategy considers a visual inspection by trained environmental staff, an intermittent monitoring of the flue gas emitted by the processing plant, the continuation of the measurements of NO_2 and SO_2 through passive samplers with a sampling time of 4 weeks, the continuation of the measurements of nuisance dust with DustScan 100 with a sampling time of 1 month, the addition of Frisbee pad to monitor deposition of nuisance dust with a sampling period of 1 month, the Osiris Turkey and EPAM 5000 for the measurement of $PM_{2.5}$ and PM_{10} will be rotated over different locations.

The monitoring plan presented by Lydian for air quality is relevant and acceptable but shall be augmented to include the following based on the assessment conducted above:

- Measurement of mercury at the settlements locations to be able to assess the ADR emissions impact taking into account the background concentrations and the national air quality standards. For the gaseous mercury: 4-week measurements shall be conducted every year over 2 consecutive months in winter and 2 consecutive months in summer. The method shall be in accordance with the latest EN 15852 Ambient air quality - Standard method for the determination of total gaseous mercury. For the particulate phase: 24-hour measurements shall be conducted once per week concurrently with the Hg gaseous measurements over 2 consecutive months in winter and 2 consecutive months in summer. The method used shall be in accordance with the latest EN 12341 and EN 14902.
- Continuous emission measurement of mercury at the stack of the ADR in case the emissions are above 50% from the emission limit value that will be determined based on the modeling to ensure low levels of mercury are compliant with the national air quality standards at the settlements' locations or only twice per year in case the emissions are below 50% of the emission limit value.
- Measurement of HCN, and HCl at the settlements locations to be able to assess the ADR emissions impact taking into account the background concentrations and the national air quality standards. Measurements shall be conducted during summer and winter according to international guidance, ex. UK Environment Agency (2011) Technical guidance note M8, Monitoring Ambient Air Version 2.
- Measurement of HCN, and HCl at the stack of the ADR only twice per year to ensure their compliance with the emission limit value determined for these based on the modeling exercise.
- Setting of the trigger value for deposition of nuisance dust for the Frisbee instrument at maximum of 200 mg/m²/day.
- Measurements for the EMP shall be carried out at the locations already considered for the baseline. As for the Frisbee and DustScan 100, they shall be co-located.
- Maintain measurement of meteorological parameters at PMS and BRSF stations.

3.0 Summary, Conclusions and Data Gaps

3.1 Water and Geology - Baseline Characterization

3.1.1 Geology

The baseline characterization of the geology of the Project study area is data deficient, and the interpretation and conceptualization of the geology across the area is too simplistic. Detailed surface geologic mapping was focused on the Amulsar Mountain ridge. Fault and fracture mapping throughout the rest of the study area was not conducted. Faults may be barriers and/or conduits of groundwater flow throughout the area.

The Amulsar Mine facilities are not entirely within the Amulsar Tectonic Block (ATB). The Tigranes/Artavazdes pit area and at least part of the Erato pit are south of the Agarakadzor Fault, the southern boundary of the ATB. Part of the Kechut Reservoir is within the ATB. The Zirak Fault appears to dissect the BRSF footprint, indicating that part of the BRSF is north/northeast of the ATB. The Zirak Fault and the Agarakadzor Fault are potential conduits of groundwater flow to the major rivers. These faults also represent potential seismic hazards. The ATB does not isolate potential Project impacts from the environment. Potential seepage to groundwater from the part of the BRSF north of the Zirak Fault could result in contaminated groundwater potentially reaching the Madikenc springs. Contaminated groundwater beneath the mine pits could potentially flow to and reach the Darb and Vorotan Rivers. The extent of the impacts to groundwater cannot be determined based on available information.

The seismic hazard risk is high for the Project Area. The seismic hazards assessment is generally thorough and conservative. However, the bounding faults of the ATB block were not considered in the assessment. If displacement occurs along major active faults in the vicinity of the Project Area, including the PSSF5a that potentially underlies the Vorotan River valley, movement could potentially occur along other faults in the Project area, including the Zirak Fault under the BRSF and the Agarakadzor Fault passing through the pit areas.

Even with part of the BRSF being north of the Zirak Fault, contaminated Amulsar Mine water will not impact the Jermuk springs. Surface water and groundwater moving northward from the BRSF follow northwest trajectories toward the Arpa River and Kechut Reservoir. Jermuk is at least 1,000 m higher than the Kechut Reservoir. The Arpa River flows southward from Jermuk. Groundwater potentials decrease along the river valley in the direction of river flow. Furthermore, there is a northeast-oriented tributary to the Arpa River between Jermuk and the Project Facilities, which is a probable hydraulic boundary. Finally, Jermuk is northwest of the trace of the Kechut fault, which may also be a barrier to groundwater flow.

3.1.2 Geochemistry

The broad categories of Upper Volcanics (VC) and Lower Volcanics (LV) are inadequate for acid rock drainage (ARD) characterization. These categories encompass a range of rock sub-types that are not defined as geochemical test units for specific characterization, resulting in insufficient characterization of each rock sub-type and the ARD potential of the rocks as a whole. Likewise, ore and colluvium may be insufficiently characterized. Acid-base accounting (ABA) and classification of the tested samples are incomplete, and maximum potential acidity (AP) is incorrectly calculated. The LV and at least part of the VC are potentially acid generating (PAG). The results of the characterization should be viewed with caution. The characterization was poorly planned and coordinated. The leachate from the Site 27 Soviet era waste pile has a pH of 3.3 and high acidity. These data are a clear indicator of the potential for ARD from the Amulsar Mine.

Subsequent to the characterization, an ARD block model was developed to determine the quantity of AP waste and its distribution. The model incorporates subdivision of LV based on the percentage of total sulfur. All VC is still considered non-PAG rock. Previously, all LV was assumed to be PAG and managed the same. The model is generally based on the conservative assumption that total sulfur is a proxy for sulfide sulfur and that total sulfur greater than 2 percent is strongly acid generating. Although this approach does not rectify the deficiencies in characterization, it improves ARD management of LV rock.

Bucket testing was initiated October 2017. A generally good correspondence is observed between ABA and bucket test results, with all 11 LV samples having pyritic sulfur greater than 4% producing pH less than 4. Lower pyritic sulfur percentages produced a higher pH range of 4.5 to 6.0. One of the VC samples attained a pH as low as 4.0 in May 2018, and two other VC samples generally show pH ranges from 4.5 to 6.0. The VC sample that produced pH as low as 4.0 (4.6 – 5.75 in May and June) has 0.13% pyritic sulfur. These results reinforce the need for sub-types of rocks (geochemical test units) and that VC has potential for acid generation even at the lower end of the pyritic sulfur range identified in the original ABA testing (VC pyritic sulfur up to and more than 5%). The oxidation observed in the core boxes provides an indication of the rapidity of acid generation (drilling dates 2010 – 2012).

ARD with pH in the range of 4 – 5 cannot be dismissed as unproblematic. Acid contributes to the rate of chemical weathering of rock, which contributes to accelerated physical weathering of the rock. Accelerated weathering contributes to the rate of exposure of more pyrite in all the rock types at Amulsar. With enough pyrite exposed, very low pH solutions develop that mobilize metals, as observed in the HC tests.

The ESIA discussion of ARD geochemistry is misleading because the ARD Management Plan:

- Ignores the importance of ferrous iron oxidation in generating acid and solids (metals).
- Postulates that the reaction of pyrite by ferric iron dominates the oxidation of pyrite, when in fact the ferric iron oxidation is just one of the two pathways for pyrite to be oxidized, and the two pathways cannot be distinguished based on the products generated.
- Postulates that there is some "natural suppression agent" inhibiting the oxidation of pyrite in the LV ores, when in fact there is no evidence for some suppression agent other than the slow reaction of pyrite. There is no evidence the Amulsar rocks have natural resistance to ferric iron oxidation of pyrite and ARD generation. Given the ample evidence of ARD generation in the area (the low pH springs and the waste from the old Soviet era mine), pyrite in these rocks is evidently susceptible to oxidation and generating ARD.
- Underestimates ARD generation, and corresponding water quality and environmental impacts and water treatment requirements.

3.1.3 Water Resources

Five hydrogeologic units were delineated in the groundwater study area (GSA), which is appropriately defined. The structural control of the boundary rivers ensures that flow and transport from the GSA do not traverse these hydraulic boundaries. However, the hydraulic properties of the units are inadequately characterized by a limited distribution of hydraulic tests across the GSA and a complete lack of pumping tests. Fractured rock has extremely heterogeneous and anisotropic hydraulic properties, which are dependent on rock type, fractures, and stratification (which are variable across the GSA) and proximity to structures. Only long duration pumping tests in the various hydrogeologic units at a variety of key locations can provide a good indication of the bulk hydraulic conductivity of fractured rock and the influence of structures. The water balance for the GSA, estimates of solute transport velocities, and assessment of potential impacts are dependent on good hydraulic characterization. These important objectives can only be attained with a well-constrained numerical groundwater model in this type of geologic setting.

Baseline data are lacking for many springs in the GSA. The flow rate at a number of springs in the vicinity of Kechut were not measured. In the south of the GSA, flow was not measured at the Pluskandyal springs or other community springs southeast of Ughezdor. The ESIA states that several potentially significant springs were not visited. There is a large number of springs in the GSA, especially in the vicinity of the Amulsar Mountain ridge. Given the importance of springs to the local communities and the potential for impacts to the springs from the mine pits, the springs flow characterization is inadequate.

The ESIA concludes that groundwater recharge ranges from 200 to 250 mm/year across much of the GSA, with the greatest rates of infiltration occurring at the higher elevations. This conclusion is subjective. A wide range of recharge rates was calculated from various approaches, demonstrating considerable uncertainty in the estimates. Recharge rates based only on flow of the major rivers are among the higher values of the estimates, ranging from 244 mm/year to 460 mm/year. The recharge rate is highly variable across the GSA.

The continuous flow monitoring stations established by the Project on the Arpa, Darb, and Vorotan Rivers are generally adequate. A couple of apparent deficiencies in continuous flow monitoring are in the vicinity of Vorotan Pass. A station at the bend on the upper Darb River, where the course changes from northward to northwestward, would determine whether flow is perennial or ephemeral in that location (given the importance to groundwater flow modeling). Likewise, a station on the east side of Vorotan Pass on the upper reach of the Porsughlu River flowing into Spandaryan Reservoir would serve the same purpose. A station should also be added to the stream below Benik's Pond to monitor potential effects of the Tigranes-Artavasdes pit.

The ESIA does not provide an explanation for the hiatus in continuous flow monitoring between May and December 2013. Termination of the continuous discharge monitoring after May 2014 is questionable. Good characterization of baseflow is necessary to understand the groundwater balance and to demonstrate that the Amulsar Mine is not impacting the environment.

Major ion characteristics of Jermuk geothermal water samples are completely distinct from all other sampling locations. The Jermuk and other geothermal waters have enriched carbon isotopic signatures ($\delta^{13}C$), in contrast to Amulsar Mountain springs and groundwater, surface water, and precipitation that have depleted ratios. The geothermal waters also have significantly enriched sulfur isotopic signatures ($\delta^{34}S$) (vs distinctly less enriched surface waters and depleted signatures in Amulsar Mountain groundwater and springs). Oxygen and hydrogen isotopic ratios of the geothermal waters are lower to much lower than those in GSA surface waters, springs, and groundwaters. These data indicate the source of the geothermal springs is old meteoric water that precipitated at historically lower temperatures (potentially Pleistocene age) than the Present. The data for the geothermal springs are consistent with completely separate sources, flow paths, and timeframes. Sulfur and strontium isotopic data support the interpretation of long, deep flow paths that pass through mafic rocks with enriched $\delta^{34}S$ and lower strontium isotopic signatures than more differentiated rock types like andesite.

The isotopic signatures of $\delta^{13}C$ and $\delta^{34}S$ for Amulsar Mountain groundwater and springs reflect considerably depleted isotopes relative to the Spandaryan-Kechut tunnel outfall. Oxygen and deuterium isotopes of the tunnel outfall are depleted relative to Amulsar Mountain springs and groundwater. This difference suggests the outfall water is entering the tunnel after a moderately long flow path (aged groundwater that originated at lower temperature). The strontium isotopic ratio ($^{87}Sr/^{86}Sr$) of the outfall is also lower than the Amulsar Mountain groundwater data, suggesting significant tunnel ingress is not occurring west of the mountain ridge. Isotopic data are lacking for the basaltic rocks to the north, but the low strontium isotopic signatures and relatively high/enriched $\delta^{34}S$ of the outfall suggest groundwater flow through mafic rocks. The outfall data suggest the relatively high sulfate concentration, the distinctly less depleted $\delta^{13}C$ (relative to Amulsar Mountain), the more depleted $\delta^{18}O$ and $\delta^{2}H$, low $^{87}Sr/^{86}Sr$, and the enriched $\delta^{34}S$ of the outfall are all consistent with the majority of the tunnel ingress occurring from the basaltic rocks to the north. This interpretation is consistent with shallower groundwater as the Kechut Reservoir is approached, with greater potential for the tunnel to intersect groundwater.

A surface water quality monitoring location should be included on the main tributary of the Darb River downstream of station AW006, before the confluence with the Darb River or downstream of the tributary on the Darb River, to better assess the Mine impacts on surface water quality.

Few springs around Amulsar Mountain and the BRSF are monitored. Given the importance of springs to livestock and the large number of springs that should be monitored, increasing this number would offset the need for many additional groundwater quality monitoring wells. Assuming many more springs are added to the groundwater monitoring program, only a few more groundwater monitoring wells would be necessary, located north-northwest of the BRSF, southwest of the Arshak pit, and east of the Tigranes-Artavasdes pit.

Baseline groundwater quality data for springs and wells are deficient. Likewise, baseline surface water quality data are far from sufficient. For comparison of future concentrations, meaningful statistics are necessary, requiring 30 to 50 data points for each analyte at each monitoring station.

3.2 Groundwater Flow and Solute Transport Modeling

The water fluxes from the pit seepage sub-model are incorrect. The major issue is too much evaporation from exposed rock and loose rock backfill (with no soil cover). Use of these fluxes in the regional groundwater flow model results in incorrect assessments of impacts to groundwater levels and springs. Furthermore, solute transport simulations would severely underestimate potential impacts to groundwater and springs from ARD.

The BRSF runoff and seepage sub-models have the same excessive evaporation problem as simulated for the pit backfill. The evaporation is inconsistent with coarse texture and high permeability of the exposed loose waste rock. The calculated volumetric fluxes that report to the base of the BRSF are greatly underestimated. Underestimated water fluxes translate to underestimated ARD mass fluxes, overestimated makeup water volume, and underestimated PTS influent volume, potentially delaying the timing when the PTS is required (i.e., water treatment will be required prior to year 5 estimated in the ESIA).

An analytical solution was used to assess the potential impacts to groundwater and surface water that could result from leakage through the membrane liners of the heap leach pad and pregnant solution pond during operations through post-closure. The application of the model to simulate transport in the saturated zone is questionable, especially with the existence of a numerical groundwater flow model. The source term for transport is poorly constrained. Simulated transport species omit chloride, the most mobile solute in groundwater. Furthermore, selenium should be included in transport simulations due to the detrimental impacts of this element to fish.

A three-dimensional groundwater flow model was constructed for the GSA. The model is inadequate due to incorrect specification of boundary conditions, insufficient and uniform recharge, oversimplification of geologic structure, homogeneous, too low, and poorly-constrained hydraulic conductivities, and a poor calibration for the intended predictive usage. The simplistic numerical representation of the subsurface in the model is inadequate for making the quantitative predictions that were performed, including estimates of pit inflow. The model does not correctly represent the water balance of the GSA. Less water is moving through the simulated rocks than the actual quantity, and the simulated rates of advective flow and transport (in the particle tracking simulations) are too low. A significant omission of the modeling is the performance of solute transport simulations for predicting chemical impacts to groundwater and surface water quality. Proper assessment of potential impacts to the environment and evaluation of uncertainty cannot be performed without a numerical model of the GSA that is adequately representative of the complex subsurface and that is hydraulically well-constrained.

3.3 Water Quality and Water Resources Impacts Assessment

Overall, the water quality modeling and solute transport model simulations are poor and deficient and conclusions made based on these simulations are unreliable.

The post-closure impacts analysis for the BRSF is flawed due to underestimated potential mass loading to groundwater. The simulated pH of the leachate is 3.0, similar to Site 27 waste rock leachate. Most of the simulated concentrations are much greater than the observed concentrations, which is consistent with a much longer vertical flow path than the existing waste piles. However, the simulated iron concentration is only 0.5 mg/L, compared to 3.2 mg/L in the Site 27 leachate, which has a similar pH of 3.3. This difference is indicative of inappropriate specification of iron phases in the equilibrium modeling. An assessment of impacts to groundwater was not performed, and transport was not simulated because the ESIA/EIA incorrectly concludes that no contaminated water will reach the groundwater.

The post-closure impacts analysis of the HLF suffers from inappropriate source concentrations for the solute transport modeling. Furthermore, the transport assessment does not integrate potential impacts to groundwater from the BRSF and the mine pits.