

Environmental hazard/risk assessment of tailings storage facility overflow, Sorby Hills Project

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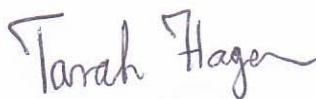
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Executive Summary

The Sorby Hills Silver-Lead-Zinc project in the Kimberley region of Western Australia will treat/store part of its waste stream in a proposed tailings dam. ToxConsult Pty Ltd (ToxConsult) was requested by Sorby Management Pty Ltd (Sorby Mgt) to conduct a desktop study investigating the potential environmental effects in the unlikely event of a significant overflow (e.g. in a large storm or flood event) from the tailing storage facility.

Ore at the mine is crushed and processed in a conventional flotation circuit. Flotation chemicals (e.g. frothers and collectors) are added along with water and air to separate the valuable minerals from the waste slurry (called gangue). The gangue, consisting of unwanted minerals, water, and flotation reagents, is pumped to the tailings storage facility. The solids are left to settle out, and the overlying supernatant is recycled back into the process to conserve flotation reagents. It is this supernatant, rather than the settled solids, that would be released from the tailings facility spillway after combining with large volumes of rainwater, if an overflow were to occur.

As the facility has not yet been constructed, this assessment is prospective in nature based on the results of tests conducted on pilot flotation liquids. These tests involved measuring the filtered concentrations of a wide range of metals/metalloids in the liquids and flotation leachate from separated solids. Filtered concentrations are assumed to represent concentrations in the supernatant that are biologically available, which is in line with current scientific knowledge.

The assessment broadly follows national and international guidance for conducting environmental risk assessments and was undertaken in two stages.

Stage 1: An initial screening assessment was carried out on the chemicals of interest in which anticipated chemical concentrations in the tailings overflow were compared with water quality guidelines (WQG) for the protection of freshwater organisms. If WQGs were not available from reputable national or international sources, and if there was sufficient information for a chemical, a site-specific 'preliminary aquatic screening value' was derived following the methodology used to derive WQGs in Australia.

Those chemicals whose predicted concentrations exceeded the WQG were designated chemicals of potential concern (CoPCs) and were subject to more detailed evaluation in Stage 2.

Stage 2: The potential for CoPCs to constitute an environmental risk at the Sorby Hills Project area was qualitatively evaluated by considering:

- their environmental fate,
- the applicability of the WQG for the anticipated exposures and organisms at the Sorby Hills Project area,
- and their toxicity to terrestrial plants and animals identified to be present in the area.

The chemicals considered in this risk assessment (i.e. the chemicals of interest) include:

- the flotation reagents (e.g. frothers, collectors) in the tailings supernatant. Their concentration has been estimated by Sorby Mgt and these estimates were used as provided.
- and the dissolved metals/metalloids in the flotation ultrafiltrates. The leachate testing was undertaken to simulate rainwater infiltration of damp solids in the tailings facility. The pH of rainwater is between 5 and 6 and the pH of the leachate was approximately 6. However, the pH of the supernatant in the facility is approximately 8.5. Most metals are more soluble in acid than alkaline conditions. It is uncertain whether or not supernatant fluid at pH 8.5, when diluted with rainwater at pH 5, will become acidic. To account for this uncertainty, the maximum concentration of dissolved metals in either flotation ultrafiltrate or leachate was used in this assessment. Nevertheless this is likely to be conservative because the pH of the supernatant will have a higher buffering capacity than rainwater, and hence the change of supernatant pH from alkaline to acid is unlikely to be great.

The mining operations for the Sorby Hills project will be located on a flat, clayey, alluvial floodplain with no defined natural drainage system. After an intense rainfall event, the clayey soils on the floodplain are likely to become saturated and swell forming an effective “clay-layer” across the floodplain (i.e. there is negligible water infiltration through the soil after it has swelled). Any additional water landing on the floodplain ponds or sheets on the surface and either:

- slowly flows to permanent watercourses Knox Creek (3.5 km to the south) and Keep River (4.6 km to the south-east) or
- evaporates

A minimum 7.5x dilution of the chemicals in the supernatant is required to fill the storage facility to the brim. Any overflowing supernatant/floodwater mix would be further diluted by water pooled or sheeted in the surrounding floodplains. If the storage facility overflow were to migrate to permanent waterways, it would be subject to additional dilution from other environmental flows and the water inside the waterway.

The assessment for potential environmental impact from storage facility overflow was therefore conducted for three scenarios, which consider assessment locations that are stepped away from the tailings facility according to designated dilutions.

- **Scenario 1 (S1).** Is for a hypothetical area in the immediate surrounds of the tailings storage facility that is represented by an environmental dilution of 10x of the overflow (i.e. in addition to the 7.5x required to fill the facility to the brim). Tailings facility overflow is expected to migrate away from the immediate surrounds of the facility.
- **Scenario 2 (S2).** As the diluted tailings facility overflow moves further away from the dam and mixes with the surrounding waters accumulated over the flood plain it will become progressively diluted. However as the overflow fans out, a larger area is exposed to the continuously diluted chemicals of interest. To enable an assessment of the potential impacts in the area between the immediate surrounds of the facility and the permanent waterways of Scenario 3, a minimum dilution of 1,000x of tailing overflow is assumed.
- **Scenario 3 (S3).** This scenario evaluates the potential impact of tailings facility overflow on the permanent waterways of Knox Creek and Keep River. By the time overflow from the tailings dam reaches the creek and river markedly more dilution is anticipated to have occurred. In addition these receiving waters will have significantly swelled, and indeed may have broken their banks and not be distinguishable as defined water courses from the surrounding flood plain. To represent this situation a dilution of 10,000x of tailings overflow has been assumed.

The CoPCs identified from Stage 1 were the following; the scenario(s) which resulted in an exceedence of a WQG are provided in brackets after the chemical name:

- Ethyl xanthate (S1, S2, S3)
- Butyl xanthate (S1)
- Sodium hydrogen sulphide (S1, S2, S3)
- Lead (S1)
- and Zinc (S1)

These chemicals were further evaluated in Stage 2. In light of:

- significant reductions in toxicity expected as a result of higher water hardness and levels of suspended material in the environment when compared to laboratory ecotoxicity tests;

- the inbuilt conservatism in the WQGs, especially those of low reliability which incorporate large assessment factors;
 - and the use of chemical concentrations that are likely overestimates,
- the potential risk of environmental effects from most of these chemicals is considered low.

Under the assumptions used in this risk assessment, there may be potential for toxicity to sensitive aquatic organisms (e.g. crustaceans or algae)¹ from process chemicals ethyl xanthate and sodium hydrogen sulphide in the tailings if an overflow were to occur. Such effects are likely to be confined to organisms inhabiting areas in the immediate vicinity of the storage facility (i.e. areas subject to only a 10x environmental dilution). Relative to the entire flood plain this is a very small area. An area in which the ecosystem is likely to be already affected by construction and maintenance activities of the storage facility. Other organism groups (troglofauna, stygofauna, amphibians, terrestrial plants, mammals, and birds) are not expected to be adversely impacted. There was insufficient toxicity information to evaluate the potential for adverse effects to reptiles.

If susceptible organisms are present, and if they are affected, the impact of substances in the tailings overflow water on the viability of populations within the wider floodplain ecosystem will be negligible compared to that which occurs with the periodic flooding of the plain.

¹ This assumes these organisms are present in pooled/sheeted water on the floodplains.

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1. Introduction and Scope

The Sorby Hills Silver-Lead-Zinc project in the Kimberley region of northern Western Australia will treat/store part of its waste stream in a proposed tailings storage facility. ToxConsult Pty Ltd (ToxConsult) was requested by Sorby Management Pty Ltd (Sorby Mgt) to conduct a preliminary desktop study for potential environmental impacts should there be a significant overflow (e.g. in a large storm or flood event) from the tailing dam. The study will inform the need (if any) to proceed to a more detailed assessment.

Ore is mined, crushed and processed in a conventional flotation circuit. Flotation chemicals (e.g. frothers and collectors) are added along with water and air to separate the valuable minerals from the waste slurry (called gangue). The gangue, consisting of unwanted minerals, water, and flotation reagents, is pumped to the tailings storage facility. The solids are left to settle out, and the overlying supernatant is recycled back into the process to conserve flotation reagents.

As the facility has not yet been constructed, this assessment is prospective and based on the results of tests conducted on pilot tailing materials.

2. Information provided

Sorby Management provided ToxConsult with several documents to aid the investigation. These consist of abstracts and reports describing the general ecology of the area, a geochemical characterisation study of the Sorby Hills region, a surface water flow study, and information on the types and concentrations of chemicals added to the flotation process. The names of the documents are provided in the reference list.

This information was used to identify the chemicals of interest, potential organisms that may be exposed in the event of a storage facility overflow and to estimate concentrations of flotation reagents in the tailings overflow.

The solids from pilot tailings were alkaline (pH > 9) and contained substantial levels of iron, manganese, lead and zinc (SWC 2011a).

A wide suite of total and dissolved metals/metalloids were measured in flotation liquids characteristic of the Sorby Hills project². This was achieved by separating the solids by filtration and subjecting them to leachate testing, the filtrate was further passed through an ultrafine filter (0.45 µm) to obtain fluid containing dissolved materials. The leachate from the solids was also ultrafiltered. Metals in

² E-mail correspondence with Sorby Management, received 07/07/2012.

both sets of ultra-filtrates were measured and the maximum concentration in either was used in the assessment (SGS 2012, Appendix A). The concentrations of metals in the filtrates are more likely to reflect metal concentrations to which organisms may be exposed and potentially be absorbed.

3. Conceptual model

The mining operations for the Sorby Hills project will be located on a flat, clayey, alluvial floodplain. There are no defined natural drainage systems on the flood plain. After an intense rainfall event, the clayey soils become saturated and swell forming an effective “clay-layer” across the plain (i.e. there is negligible water infiltration through the soil after it has swelled). Any additional water landing on the plain ponds on the surface and either:

- slowly flows to neighbouring Knox Creek (3.5 km to the south) and Keep River (4.6 km to the south-east, see Figure 3.1), or
- evaporates³

It is uncertain how long the ponded water is likely to remain on the floodplains after a storm event. Sorby Mgt (2011) indicates it is likely longer than five days due to the low permeability of the alluvial clays and low gradient towards Knox Creek and Keep River. This is an uncertainty which has been further discussed in Section 6.

During the wet season, the entire floodplain becomes waterlogged and there is 20-30 cm of ponded or sheeted water across the land surface (SWC 2011b). It takes approximately 350 mm of rain for the soils to become saturated and ponding to start to occur (SWC 2011b).

³ Evaporation is high with the annual average pan evaporation exceeding the annual average rainfall by a factor of 3.3, and exceeding rainfall for all months (AGE 2011).

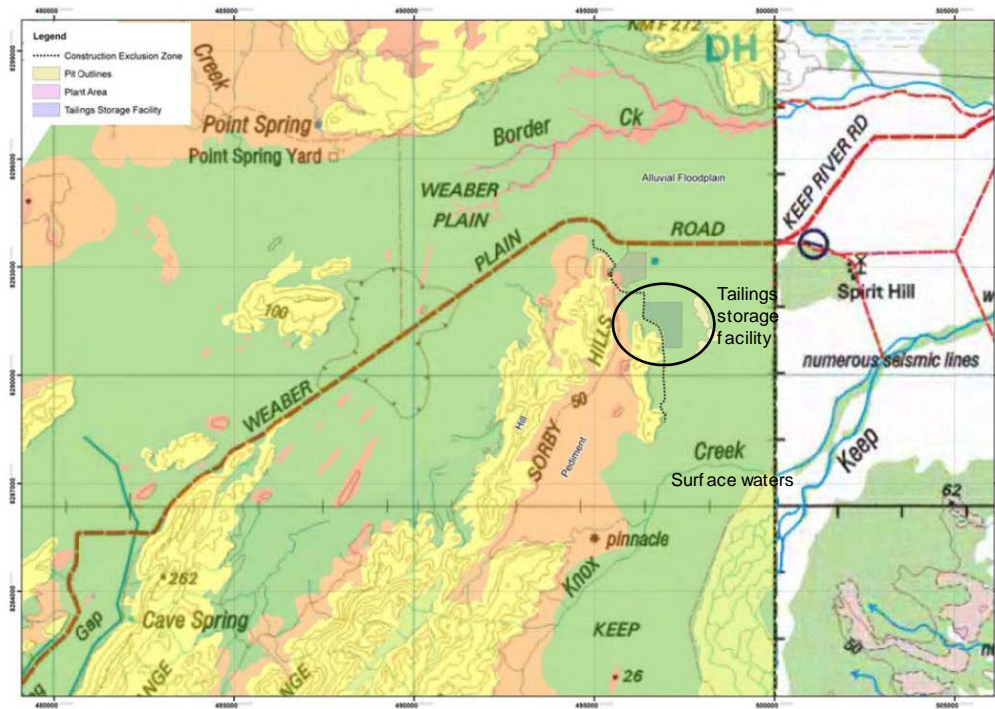


Figure 3.1: Location of the tailings storage facility in relation to closest surface waters (Soil Water Consultants 2011b).

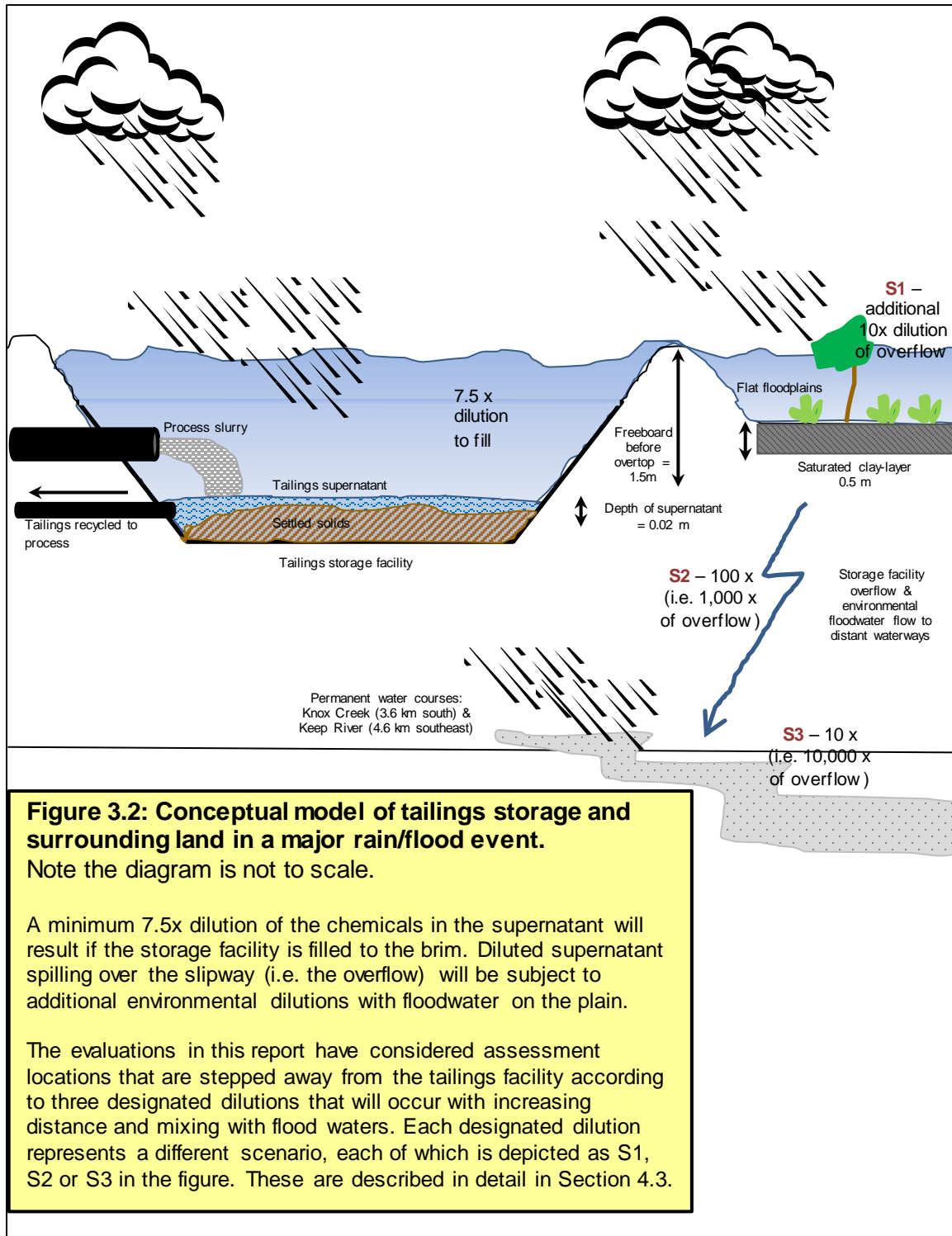
Tailings storage facility is circled. The closest surface waters to the facility are Knox Creek and Keep River, marked by blue lines south and southeast of the facility.

Using a conservative rainfall runoff scenario in which 50% of all rain that falls in the upstream catchments runs off onto the Sorby Hills floodplain, SWC (2011b) concluded all infrastructure needs to be raised at least 1 metre above the floodplain to prevent it from being flooded.

The tailings storage facility has a 1.5 metre freeboard (depth of empty space above the tailings supernatant line and the top of the tailings pond wall). The depth of the supernatant in the tailings storage facility at any one time is approximately 0.02 m (Sorby Mgt 2012f).

Figure 3.2 is a schematic representation of the tailings storage facility and the surrounding area.

The evaluations in this report have considered assessment locations that are stepped away from the tailings facility according to designated dilutions that will occur with increasing distance and mixing with flood waters; this concept is further described in Section 4.3.



4. Assessment methodology

4.1 Overview

The assessment broadly follows national and international guidance for conducting environmental risk assessments (WA DEC 2006, 2010; NEPC 1999; ANZECC 2000). It has been undertaken in two stages.

Stage 1: An initial screening assessment was carried out on the chemicals of interest (CoI)⁴ in which anticipated chemical concentrations in the tailings overflow were compared with water quality guidelines (WQG) for the protection of freshwater organisms (see Section 4.4). If WQGs were not available from reputable national or international sources, and if there was sufficient information for a chemical, a site-specific 'preliminary aquatic screening value' (PASV) was derived following the methodology used to derive WQGs in Australia (ANZECC 2000) (see Section 4.4).

Those chemicals whose predicted concentrations exceeded the WQG, i.e. giving a ratio of greater than one⁵, were designated chemicals of potential concern (CoPCs) and were subject to more detailed evaluation in Stage 2.

Stage 2: The potential for CoPCs to constitute an environmental risk at the Sorby Hills Project area was qualitatively evaluated by considering:

- their environmental fate,
- the applicability of the WQG for the anticipated exposures and organisms at the Sorby Hills Project area,
- and their toxicity to terrestrial plants and animals identified to be present in the area.

⁴ Chemicals of interest are those used in the flotation process and identified in laboratory testing of pilot tailings materials (Section 4.2).

⁵ This ratio is termed the hazard quotient (HQ).

4.2 Chemicals of Interest

The chemicals of interest (Col) considered in this risk assessment and the concentrations assumed to be in the tailings supernatant are summarised in Table 4.1. They are:

- The flotation reagents (e.g. frothers, collectors) in the tailings supernatant. Their concentration has been estimated by Sorby Mgt based on hourly inputs into the flotation circuit (Sorby Mgt 2012f). As the tailings supernatant is continuously recycled back into the process, the calculated concentrations represent the reagent chemicals when they have reached a steady-state in the ore extraction process. These estimates have been used as provided by Sorby Management.
- The dissolved metals/metalloids (i.e. assumed to be biologically available) in the flotation ultrafiltrates. The leachate testing was undertaken to simulate rainwater infiltration of damp solids in the tailings facility. The pH of rainwater is between 5 and 6 (SA EPA 2004, CSIRO 2011); the pH of the leachate was approximately 6. However, the pH of the supernatant in the facility is approximately 8.5. Most metals are more soluble in acid than alkaline conditions. It is uncertain if supernatant fluid at pH 8.5, when diluted with rainwater at pH 5, will become acidic. To account for this uncertainty, the maximum concentration of dissolved metals in either flotation ultrafiltrate or leachate has been used in this assessment. Nevertheless this is likely to be conservative because the pH of the supernatant will have a higher buffering capacity than rainwater, and hence the change of supernatant pH from alkaline to acid is unlikely to be great.

Despite the conservatism in the assumption that the pH of the supernatant in the tailings storage facility may be acidic and therefore result in higher concentrations of metals in the tailings overflow, most metals are still markedly below the concentrations that would cause concern (Section 5.1). There are two metals whose assumed concentrations from the leachate tests for Scenario 1 are above their respective guideline values (lead and zinc). The implications of this are discussed further in Section 5.2.3.

Table 4.1: Chemicals of Interest and concentrations estimated to be in undiluted supernatant ^a

Chemical	Concentration (µg/L) ^a
Flotation reagents	
Lime (CaO)	575,000
Methyl isobutyl carbinol (MIBC)	100,000
Ethyl xanthate	20,000
Butyl xanthate	15,000
Sodium hydrogen sulphide (NaHS)	280,000
Metals/metalloids	
Silver	0.5 ^b
Aluminium	80
Antimony	44
Arsenic	0.5 ^b
Barium	500 ^c
Beryllium	0.5 ^b
Bismuth	0.5 ^b
Boron	24
Cadmium	9.5 ^c
Chromium	0.5 ^b
Cobalt	25 ^c
Copper	39 ^c
Iron	86 ^c
Lead	630 ^c
Magnesium	18,000 ^c
Manganese	1,500 ^c
Molybdenum	17
Nickel	59 ^c
Phosphorus	25 ^b
Potassium	5,600
Selenium	1 ^b
Silicon	3,400 ^c
Sodium	90,000
Strontium	43
Sulphur	95,000
Thallium	0.5 ^b
Thorium	1
Tin	25 ^b
Titanium	0.5 ^b
Vanadium	0.5 ^b
Zinc	9,000 ^c
Mercury	0.05 ^b

^a Concentrations of flotation reagents have been estimated by Sorby Mgt based on hourly inputs into the flotation circuit (Sorby Mgt 2012f). These estimates have been used as provided. The maximum concentration of dissolved metals/metalloids in either flotation ultrafiltrate or leachate has been used in this assessment (SGS 2012).

^b These chemical concentrations were reported by the testing laboratory, SGS (2012), as less than their respective practical quantification limit (PQL). It was assumed in this assessment these concentrations are equal to half the PQL.

^c This concentration is for the dissolved metal/metalloid in the leachate. It is recognised the use of leachate data is likely to overestimate the concentration in the tailings supernatant and is probably conservative (see text in Section 4.2); however it was done to account for the uncertainties in what the actual concentrations are.

4.3 Exposure Scenarios

The volume of tailings supernatant in the facility is approximately 56,000 m³, and approximately 420,000 m³ of rain- or runoff- water is required to fill the tailings storage facility to its brim (Sorby Mgt 2012f). This would result in a minimum 7.5x dilution of the chemicals in the supernatant before any overflow occurs⁶. Any overflowing supernatant/water mix would be further diluted by water pooled or sheeted in the surrounding floodplains. If the storage facility overflow were to migrate to permanent waterways 3.5-4.6 km away, it would be subject to additional dilution from other environmental flows and the water inside the waterway. The evaluations in this report have considered assessment locations that are stepped away from the tailings facility according to designated dilutions. The assessment for potential environmental impact from storage facility overflow was conducted for three scenarios.

- **Scenario 1 (S1).** Is for a hypothetical area in the immediate surrounds of the tailings dam that is represented by an environmental dilution of 10x of the overflow (i.e. in addition to the 7.5x required to fill the facility to the brim). Tailings facility overflow is expected to migrate away from the immediate surrounds of the facility.

Scenario 2 (S2). As the diluted tailings facility overflow moves further away from the dam and mixes with the surrounding waters accumulated over the flood plain it will become progressively diluted. However as the overflow fans out, a larger area is exposed to the continuously diluted chemicals of interest. To enable an assessment of the potential impacts in the area between the immediate surrounds of the dam and the permanent waterways of Scenario 3, a minimum dilution of 1,000x of tailing overflow is assumed.

- **Scenario 3 (S3).** This scenario evaluates the potential impact of tailings facility overflow on the permanent water bodies of Knox Creek and Keep River. By the time overflow from the tailings dam reaches the creek and river markedly more dilution is anticipated to have occurred. In addition these receiving waters will have significantly swelled, and indeed may have broken their banks and not be distinguishable as defined water courses from the surrounding flood plain. To represent this situation a dilution of 10,000x of tailings overflow has been assumed.

Appendix E provides the concentrations of each chemical of interest for these scenarios.

⁶ As per the information provided by Sorby Mgt (2012f).

4.4 Water Quality Guidelines

If water quality guidelines (WQGs) for the chemicals of interest were available from reputable national agencies (e.g. ANZECC 2000, WA DEC 2010), these WQGs were used in the assessment. If not, WQGs were sourced from international reputable agencies, if available (e.g. US EPA 2012a, CCME 2012). No international WQGs were used in this assessment because:

- national WQGs were available or
- for those chemicals without a national WQG, there was also no international guideline.

If WQGs were not available from national or international sources, and there was sufficient information, a site-specific guideline was derived following the methodology used to establish WQGs in Australia (ANZECC 2000) if it was deemed the nature of the compound warranted derivation of a guideline value; in this report these are termed 'preliminary aquatic screening values' (PASV) to distinguish them from other WQGs. PASVs were derived for four chemicals; methyl isobutyl carbinol, barium, strontium and titanium; using an assessment factor⁷ approach. Their derivations are described in Appendix D.

The remaining WQGs sourced for this project are 'trigger values' for the protection of aquatic organisms established by the Australian and New Zealand Environment and Conservation Council (ANZECC 2000). If the data permits, ANZECC (2000) derived trigger values using a statistical approach which protects a given percentage of species from chronic exposures to chemicals; these trigger values are designated as being of moderate or high reliability. Otherwise, a traditional assessment factor approach is used which results in low reliability trigger values. The ANZECC (2000) methodology and an explanation of how trigger values should be used are further described in Appendix F.

Table 4.2 shows the chemicals of interest along with the WQGs used to perform the assessment.

Table 4.2: Chemicals of interest and their water quality guidelines used in the assessment

Chemical	WQG (µg/L) ^a	Comment
Lime (CaO)	-	
Methyl isobutyl carbinol (MIBC)	4,200 ^c	No WQG could be located for MIBC. However, there was sufficient information to derive a preliminary aquatic screening value (PASV). See Appendix D for its derivation.
Ethyl xanthate	0.05 ^b	
Butyl xanthate	5 ^b	

⁷ The assessment factors are safety or uncertainty factors applied to ecotoxicity values to derive a WQG considered to be protective for the majority of species. The magnitude of these assessment factors depends on whether acute or chronic toxicity data are available and the confidence in whether the experimental data reflects the field situation. Most of the factors are multiples of 10 and larger factors are applied when there is more uncertainty in the data (e.g. the largest factor of 1,000 is applied when there are very limited acute and no chronic data).

Chemical	WQG (µg/L) ^a	Comment
Sodium hydrogen sulphide (NaHS)	1	WQG is for hydrogen sulphide (H ₂ S).
Silver	0.05	
Aluminium	55	
Antimony	9 ^b	
Arsenic	13	Two WQGs are available for As: one for As ^{III} and the other for As ^V . The lower of the two (As ^V) was used in this assessment.
Barium	580 ^c	No WQG could be located for barium. However, there was sufficient information to derive a PASV. See Appendix D for its derivation.
Beryllium	0.13 ^b	
Bismuth	0.7 ^b	
Boron	370	
Cadmium	0.2	
Chromium	1	WQG is for Cr ^{VI} .
Cobalt	1.4 ^b	
Copper	1.4	
Iron	300 ^b	
Lead	3.4	
Magnesium	15,000	No WQG was set for the environment by ANZECC (2000). This guideline is used for protection of freshwater aquaculture species but was also sourced from ANZECC (2000).
Manganese	1,900	
Molybdenum	34 ^b	
Nickel	11	
Phosphorus	50	No aquatic toxicity guideline could be located for phosphorus. The value used corresponds to the long-term trigger value for phosphorus in irrigation water from ANZECC (2000), which has been set partially to prevent algal growth in irrigation water. Presumably this concentration would also prevent eutrophication of other water-bodies.
Potassium	-	
Selenium	11 ^b	
Silicon	-	
Sodium	115,000	No WQG has been set for aquatic toxicity of sodium. This value is for sodium in irrigation water in order to prevent foliar injury in sensitive crops, and is considered applicable to this assessment.
Strontium	31.5 ^c	No WQG could be located for strontium. However, there was sufficient information to derive a PASV. See Appendix D for its derivation.
Sulphur	-	
Thallium	0.03 ^b	
Thorium	-	
Tin	3 ^b	
Titanium	2.7 ^c	No WQG could be located for titanium. However, there was sufficient information to derive a PASV. See Appendix D for its derivation.
Vanadium	6 ^b	
Zinc	8	
Mercury	0.6	

- no WQG could be located from the sources consulted.

WQG = water quality guideline; ANZECC = Australian and New Zealand Conservation Council

^a WQG is ANZECC (2000) trigger value (see text) unless otherwise stated.

^b Low reliability ANZECC trigger value (see text).

^c No WQG could be located for this chemical in the sources consulted. This value is a preliminary aquatic screening value (PASV) and was derived by ToxConsult using ecotoxicity data sourced from the literature and an assessment factor approach consistent with low reliability trigger value derivation by ANZECC (2000). A description of the derivation of each PASV used in this report is provided in Appendix D.

5. Risk Characterisation

5.1 Stage 1: Identification of chemicals of potential concern (CoPCs)

The ratio of the predicted concentration of the chemical of interest to the WQG, termed the hazard quotient (HQ), was calculated for each assumed dilution scenario (S1, S2 and S3). A HQ greater than one does not necessarily indicate that adverse impacts are imminent. On the other hand, it can be stated with a high degree of certainty that a HQ less than one indicates adverse impacts are unlikely. The hazard quotients for each exposure scenario are provided in Table 5.1.

If a HQ was greater than one, it is bolded and shaded in the table, and the chemical designated a Chemical of Potential Concern (CoPC). CoPCs have been further evaluated in Stage 2 (Section 5.2).

The following were identified as CoPCs; the scenario(s) which resulted in an exceedence of a WQG are provided in brackets after the chemical name:

- Ethyl xanthate (S1, S2, S3)
- Butyl xanthate (S1)
- Sodium hydrogen sulphide (S1, S2, S3)
- Lead (S1)
- and Zinc (S1)

Table 5.1: Hazard quotients for each Scenario and identification of chemicals of potential concern (CoPCs)

Chemical	S1 HQ ^a	S2 HQ ^a	S3 HQ ^a	Comment
Lime (CaO)	-	-	-	No WQG could be located for lime. Lime reacts with water to form hydrated lime. Hydrated lime readily reacts with carbon dioxide or carbonate ions in the environment, forming sparingly soluble calcium carbonate (calcite), a mineral widely present all over the World. This mineral acts as a pH buffer in water, and increases water hardness. Increasing water hardness often reduces the toxicity of other chemicals such as metals. Therefore lime has not been considered further in this assessment.
Methyl isobutyl carbinol (MIBC)	0.3	0.003	0.0003	
Ethyl xanthate	5,333	53	5	CoPC. Further evaluated in Stage 2 (Section 5.2.1).
Butyl xanthate	40	0.4	0.04	CoPC. Further evaluated in Stage 2 (Section 5.2.1).
Sodium hydrogen sulphide (NaHS)	3,733	37	3.7	CoPC. Further evaluated in Stage 2 (Section 5.2.2).
Silver	0.1	0.001	0.0001	
Aluminium	0.02	0.0002	0.00002	
Antimony	0.1	0.001	0.0001	
Arsenic	0.001	0.00001	0.000001	
Barium ^e	0.01	0.0001	0.00001	
Beryllium	0.05	0.0005	0.00005	
Bismuth	0.01	0.0001	0.00001	
Boron	0.0009	0.000009	0.0000009	
Cadmium ^e	0.6	0.006	0.0006	
Chromium	0.007	0.00007	0.000007	
Cobalt ^e	0.2	0.002	0.0002	
Copper ^e	0.4	0.004	0.0004	
Iron ^e	0.004	0.00004	0.000004	
Lead	2.5	0.02	0.002	CoPC. Further evaluated in Stage 2 (Section 5.2.3).
Magnesium ^e	0.02	0.0002	0.00002	
Manganese ^e	0.01	0.0001	0.00001	
Molybdenum	0.007	0.00007	0.000007	
Nickel ^e	0.07	0.0007	0.00007	
Phosphorus	0.007	0.00007	0.000007	
Potassium	-	-	-	This is an essential element that has not been evaluated further ^b
Selenium	0.001	0.00001	0.000001	
Silicon ^e	-	-	-	No WQG could be located for silicon. Due to its essentiality, propensity in the environment, and low toxicity, silicon was not considered a CoPC and was not evaluated further ^c .
Sodium	0.01	0.0001	0.00001	
Strontium	0.02	0.0002	0.00002	

Chemical	S1 HQ ^a	S2 HQ ^a	S3 HQ ^a	Comment
Sulphur	-	-	-	Sulphur is an essential element. Elemental sulphur is not toxic, but some simple sulphur derivatives can be (e.g. H ₂ S, which has been evaluated separately, see NaHS). Sulphur is a ubiquitous component of the environment and there is a natural cycle of oxidation and reduction which transforms sulphur into organic and inorganic products. Many of the metals present in the tailings supernatant may combine with sulphur to produce less soluble metal sulphide species, for example. Thus sulphur itself was not further evaluated in this report.
Thallium	0.2	0.002	0.0002	
Thorium	-	-	-	No WQG could be located for thorium. However at the low concentrations of thorium assumed to be in the supernatant, the element is unlikely to be a cause for environmental concern, and thus has not been evaluated further in this report ^d
Tin	0.1	0.001	0.0001	
Titanium	0.002	0.00002	0.000002	
Vanadium	0.001	0.00001	0.000001	
Zinc ^e	15	0.2	0.02	CoPC. Further evaluated in Stage 2 (Section 5.2.3).
Mercury	0.001	0.00001	0.000001	

S1 = Scenario 1; S2 = Scenario 2; S3 = Scenario 3; HQ = hazard quotient; WQG = Water Quality Guideline

^a The hazard quotient (HQ) represents the ratio of the environmental concentration of a chemical (Appendix E) to its water quality guideline (Section 4.4). A ratio of greater than one does not necessarily indicate that adverse impacts are imminent. On the contrary, with certainty a HQ less than one indicates adverse impacts are unlikely. If a HQ is greater than one, it has been bolded and shaded, and the chemical designated a Chemical of Potential Concern (CoPC). CoPCs have been further evaluated in Stage 2 (Section 5.2).

^b Potassium's presence is of great importance for soil health, plant growth and animal nutrition. Its main role in plants is the maintenance of osmotic pressure and cell size, thereby influencing photosynthesis and energy production as well as stomatal opening and carbon dioxide supply, plant turgor and translocation of nutrients. Potassium has no known deleterious effects on the quality of natural waters and does not induce eutrophication in rivers and lakes.

^c Silicon is the most abundant element on Earth after oxygen, and can be found in various minerals. It is never found in nature in free form. Silicon dioxide is essential in the diet of various organisms (e.g. diatoms and sponges require silicon for strengthening their skeletons). Thus it can be reasonably expected to be relatively non-toxic to aquatic organisms. Ecotoxicity information for silicon minerals could not be located in the literature reviews consulted (Markich et al. 2002, CSIRO 2009, US EPA 2012b). Toxicity of silica (SiO₂) nanoparticles to freshwater green algae (*P. subcapitata*) has been determined, with a 72-hour EC₂₀ value (i.e. growth inhibition in 20% of the test population) of 20,000 µg/L (Van Hoecke et al. 2008). Similarly, SiO₂ bulk particles were found to be non-toxic up to 200,000 µg/L in another species of green algae (*Scenedesmus obliquus*) (Wei et al. 2010).

^d In the environment, thorium exists in various combinations with other minerals, such as silica (ATSDR 1990). Soluble thorium ions will hydrolyse at a pH above 5 forming hydroxy precipitates or complexes, which easily adsorb to particulate matter in water (ATSDR 1990). Any toxicity associated with exposure to thorium is largely attributable to radiation (i.e. activity) of its decay products, rather than the chemical *per se*. Hence at such low concentrations of thorium assumed to be in the supernatant, the element is unlikely to be a cause for environmental concern, and thus has not been evaluated further in this report.

^e The concentrations of these metals/metalloids in the supernatant of the tailings storage facility were assumed to be the concentrations of the dissolved metal/metalloid in the leachate (SGS 2012). It is recognised the use of leachate data is likely to overestimate the concentration in the tailings supernatant and is probably conservative (see text in Section 4.2); however it was done to account for the uncertainties in what the actual concentrations are.

5.2 Stage 2: Assessment of chemicals of potential concern

5.2.1 Ethyl and butyl xanthates

The predicted concentration of ethyl xanthate in all three Scenarios exceeded the ANZECC (2000) low-reliability trigger value of 0.05 µg/L (HQs of 5,333; 53; and 5 were calculated for Scenario 1, 2 and 3, respectively). The concentration of butyl xanthate in Scenario 1 also exceeded the low-reliability trigger value of 5 µg/L, with a hazard quotient of 40.

In neutral and alkaline water, xanthates hydrolyse to mono-, di-, and tri-thiocarbonates and carbon disulphide. The thiocarbonates further hydrolyse to carbonate and hydrogen sulphide. While some of the carbon disulphide will evaporate, the majority will remain in solution and be subject to hydrolysis to CO₂ and H₂S (NICNAS 1995).

The half-life of an ethyl xanthate (sodium ethyl xanthate, i.e. SEX) at a pH of 7 is approximately 260 hours (i.e. 11 days), increasing to over 500 hours (21 days) at pH 8-11. Decomposition rates increase with temperature and the presence of metal salts (NICNAS 1995).

As with ethyl xanthates, the half-lives of butyl xanthates decrease with decreasing pH and increasing temperature (Table 5.2).

Table 5.2: Half-lives (days) of butyl xanthates at different temperatures and pH ^a

		Isobutyl xanthate	n-butyl xanthate
25°C			
pH	9	49	43
	8	49	44
	7	20	21
	6	3	2
40°C			
pH	9	9	8
	8	8	7
	7	4	3
	6	1	1

^a Information from Masden and House (2006). Half-lives have been rounded to the nearest day.

Xu et al. (1988) investigated the degradation of several xanthates in a culture medium of salts growing duck weed at a temperature of 20°C. They found half-lives generally decreased with alkyl chain length. The media had an initial pH of 8.5 but dropped to pH 6.5 after a week. The half-life of

sodium isobutyl xanthate (SIBX) was 72 hours (3 days), comparable to the half-lives given in Table 5.2 at 25°C and a pH of 6.

The high average temperatures in the tropical region of northern Western Australia are conducive to decomposition of xanthates. Although the pH of the tailings is around 8.5, the mixing overflow water with large volumes of floodwater is likely to lower the pH of the overflow water to that of the floodwaters which is assumed to be approximately neutral. Thus half-lives of ~11 days for ethyl xanthates and 3-21 days for butyl xanthates (values for pH 7 in Table 5.2) are reasonable. Ethyl and butyl xanthates are therefore expected to completely decompose in the environment within 15-105 days (i.e. 5 half-lives).

Only limited ecotoxicity data are available for xanthates, covering only fish and invertebrates. Fish (lowest LC₅₀ is 0.01 mg/L) appear to be 10 times more sensitive to ethyl xanthate toxicity than crustaceans (lowest EC₅₀ is 0.1 mg/L), and ethyl xanthates are more toxic than butyl xanthates (ANZECC 2000).

The trigger value for ethyl xanthates is based on data (i.e. 50 µg/L) for the most sensitive species (fish) and application of an assessment factor⁸ of 1000. For butyl xanthates, crustaceans are more sensitive and the trigger value was derived using these data (5,000 µg/L ÷ assessment factor of 1000). Such large assessment factors embed a high level of conservatism in the trigger values which is one of the reasons they are regarded as being of low reliability.

Thus at the hazard quotients calculated for ethyl xanthates (5,333; 53; and 5 for Scenario 1, 2 and 3, respectively), toxicity to crustaceans and fish could be expected for Scenario 1, but is unlikely for Scenarios 2 and 3. Toxicity to aquatic organisms is unlikely to occur from butyl xanthates (HQ of 40 in S1).

The estimated xanthate intakes by mammals if consuming water in the immediate surround of the tailings storage facility from drinking overflow water are over 100 times lower than the respective wildlife NOAELs, i.e. the dose of xanthates at which adverse reproductive effects would not be anticipated. In reality, the estimated doses would be much lower, since most Australian mammals have a low requirement for free water because they obtain most of their water from the diet (Macroevolution undated, OFFW 2011, Blair-West et al. 1983). Therefore, toxicity of xanthates in overflow to eutherian mammals is not expected (see Appendix B). There was no information available

⁸ 10x for use of median lethal concentrations, i.e. LC₅₀, instead of No Observed Effect Concentrations, i.e. NOECs; 10x to account for potential increased sensitivities in other species not tested in the laboratory; and 10x for use of acute data to protect against effects from chronic exposure.

for marsupials to allow quantitation of potential effects. However, intuitively the calculated intakes of xanthate in drinking water for marsupials are also low (Appendix B), so an effect would probably not be expected.

Toxicity information for xanthates to birds was not available. However, information for a surrogate compound (Thiram) can be used for the evaluation. At the concentration of total xanthates in overflow in the immediate surround of the tailings storage facility (0.5 mg/L), a 0.1 kg bird would need to ingest 1.5-2 L (10-20 times its body weight) of overflow a day to reach the surrogate level at which no adverse effects can be expected. This volume would be greater for Scenarios 2 and 3. Therefore, toxicity of xanthates in tailings overflow to birds is not expected (Appendix B).

No toxicity information for xanthates could be located for reptiles. Thus the potential for adverse effects to reptiles from overflow cannot be evaluated.

5.2.2 Sodium hydrogen sulphide (NaHS)

The predicted concentrations of NaHS in Scenarios 1 (HQ = 3,733), 2 (HQ = 37), and 3 (HQ = 3.7) exceeded the ANZECC (2000) trigger value of 1 µg/L for hydrogen sulphide.

Sodium hydrogen sulphide (NaHS) in water dissociates into Na^+ and HS^- , the latter partially binding to H^+ to form un-dissociated (un-ionised) H_2S . Hydrogen sulphide in aqueous solution forms an equilibrium between un-ionised H_2S , bisulphide ions (HS^-) and sulphide ions (S^{2-}). At environmental conditions ($\text{pH} < 10$), the concentration of S^{2-} is negligible compared to the concentration of H_2S and HS^- . The concentration of H_2S dissolved in solution is inversely correlated with the concentration of dissolved O_2 , i.e. greater H_2S is available to aquatic organisms when less oxygen is present (ANZECC 2000). The toxicity of sulphides to aquatic organisms is mainly due to the un-ionised hydrogen sulphide rather than the HS^- or S^{2-} ions (ANZECC 2000).

Hydrogen sulphide's solubility in water varies with temperature from 5.3 g/L at 10°C to 3.2 g/L at 30°C (ATSDR 2006). Thus at the concentration expected in tailings overflow ($< 0.01 \text{ g/L}$)⁹, hydrogen sulphide is expected to be fully soluble. ANZECC (2000, Table 8.3.9) provide approximate percentages of un-ionised H_2S in total aqueous sulphide solutions at a range of pH and temperatures. The percentage of H_2S increases with decreasing temperatures and pH. Mean (since 1944) minimum and maximum temperatures in the Kimberley region during the rainy season months (December to March) range from 23.5 to 37.6 °C (Sorby Mgt 2011). At these temperatures and a near-neutral water

⁹ From Table E1, the highest concentration anticipated in tailings overflow (S1) is 3,733 µg/L (i.e. 0.004 g/L).

pH (7-7.5) as a result of mixing alkaline tailings overflow with slightly acidic rainwater, roughly 20-50% of total aqueous sulphide would be expected to be present as un-ionised H_2S .

Hydrogen sulphide can also form insoluble sulphide salts with various metals present in the tailings liquid (i.e. copper, zinc, nickel and iron), thereby potentially reducing the concentration of H_2S (ANZECC 2000). Therefore the concentration of NaHS used in this assessment is likely to be an overestimate of the actual concentration in the tailings.

In light of these considerations, adverse effects on sensitive organisms (i.e. crustaceans and algae) are likely to be confined to organisms inhabiting areas in the immediate vicinity of the storage facility (i.e. areas subject to only a 10x environmental dilution).

The estimated NaHS intakes by mammals if consuming water in the immediate surround of the tailings storage facility from drinking overflow water are 20-110 times lower than the respective wildlife NOAELs, i.e. the dose of NaHS (i.e. H_2S) at which reproductive effects would not be anticipated. Therefore, toxicity of NaHS in tailings overflow to mammals (including marsupials) is not expected (Appendix C).

Although limited toxicity information for H_2S could be located for birds, it would be reasonable to expect toxicity similar to or lower than that in mammals.

No toxicity information for H_2S could be located for reptiles. Thus the potential for adverse effects to reptiles from overflow cannot be evaluated.

5.2.3 Lead and Zinc

The predicted concentrations of lead and zinc in Scenario 1 exceeded the respective trigger values ($\text{HQ} = 2.5$ and 15 , respectively), but the HQs for the other two scenarios were well below one.

Lead is strongly complexed by dissolved organic matter in most natural waters, and adsorbs to suspended clay, humic substances and other suspended material (ANZECC 2000). Zinc is an essential trace element required by most organisms for growth and development. Zinc also adsorbs readily on suspended material. This is an important attenuation process in the Sorby Hills because torrential rains are likely to disturb existing vegetation resulting in high levels of suspended material in pooled/sheeted floodwater. Thus the predicted impacts of these metals in Stage 1 are likely overestimated.

Furthermore, the aquatic freshwater toxicity of both lead and zinc is lower in hard than in soft water (ANZECC 2000). The ANZECC trigger values for these metals apply to waters with low hardness, i.e. 30 mg/L as CaCO_3 , thus the trigger values used in the assessment are likely to overestimate the toxicity of these metals. Using the hardness-dependent algorithm for lead provided by ANZECC (2000, Table 3.4.3) to adjust the trigger value, a slightly increased hardness of 70 mg/L CaCO_3 would be sufficient to reduce the HQ for S1 to less than 1, for example.¹⁰

In addition, the assumed concentrations of lead and zinc in the supernatant come from the leachate testing and are likely overestimates, since the pH change that would occur from alkaline supernatant (pH 8.5) mixing with acidic rainwater (pH 5) is not expected to be great (see Section 4.2). Furthermore, the concentrations of substances in the overflow water are likely overestimates because when full there will be minimal mixing of water in the tailings storage facility. Most of the water flowing over the spillway will be water from the top layer and will be mostly rainwater.

Therefore, it is unlikely that lead and zinc would present an environmental hazard at the concentrations found in the tailings overflow.

¹⁰ Using the hardness-dependant algorithm in ANZECC (2000, Table 3.4.3), the adjusted trigger value for lead is calculated as follows:

$$\text{Hardness-modified trigger value for Pb} = \text{Trigger value} (\text{Hardness}/30)^{1.27}$$

Thus:

$$\text{Hardness-modified trigger value for Pb} = 3.4 (70/30)^{1.27}$$

$$\text{Hardness-modified trigger value for Pb} = 10 \mu\text{g/L}$$

Therefore:

$$\text{HQ for S1} = \text{Predicted concentration of Pb in S1, see Table E1 (8 } \mu\text{g/L)} \div \text{Hardness-modified trigger value (10 } \mu\text{g/L)}$$

$$\text{HQ for S1} = 0.8$$

5.3 Consideration of specific organism groups

Any anticipated impact from overflow of the tailings storage facility is not expected to significantly contribute to environmental impacts over and above those expected from such a large flood event.

5.3.1 Troglafauna

These organisms are air-breathing terrestrial animals that live in underground caves and smaller air-filled voids beneath the ground. Typical troglafauna include spiders, pseudoscorpions, cockroaches, beetles, and woodlice. In a comprehensive subterranean fauna study, no troglafauna were collected from within the Sorby Hills mining tenements. Bennelongia (2012) concluded the lack of troglafauna records, together with the occurrence of very fine grained alluvial sediments unlikely to contain extensive interconnected voids, and the generally shallow water table and seasonal inundation, suggest troglafauna communities are unlikely to occur in the proposed mining area. For this reason, troglafauna were not further assessed in this report.

5.3.2 Stygofauna

These organisms are small, aquatic invertebrates that live below the groundwater table. Typical stygofauna include roundworms, oligochaete worms, seed shrimp, copepods, and isopods. In a comprehensive subterranean fauna study, nine out of a total of ten species identified were collected outside the proposed mining area. The single species collected was a seed shrimp and was actually considered to be a surface water species (Bennelongia 2012). Thus there is a limited presence of stygofauna species in the mining area. The geology of the floodplains is such that a 'clay-liner' is formed after heavy rainfall, which will inhibit significant seepage of tailing supernatant overflow into groundwater. This will significantly limit or eliminate potential exposure of stygofauna to tailings overflow. Therefore, it is considered unlikely stygofauna would be adversely impacted in the event of a hypothetical overflow. The results of the assessment for surface water quality could potentially be used to inform risk to stygofauna. It is however expected, that the dilution would be greater if tailings seepage to groundwater occurred.

5.3.3 Surface water invertebrates, algae and fish

It has been assumed in this assessment that in the event of heavy rainfall, surface water invertebrates and algae could potentially be present in slow-moving ephemeral pooled or sheeted floodwater, as well as the more permanent water-courses (i.e. Knox Creek and Keep River). Fish are only expected to be present in permanent water courses. They would be intimately exposed to chemicals in the tailings supernatant-floodwater mix, having constant direct contact with the water. This makes this group of organisms the most susceptible to the potential effects of the chemicals in the overflow. The WQGs sourced for the assessment (Section 4.4) are protective of a large number

of species. Thus if adverse impacts on these organisms are unlikely in the scenarios evaluated, impacts to terrestrial organisms are also considered unlikely.

5.3.4 Terrestrial flora

APM and Associates conducted a detailed flora and fauna survey of the Sorby Hills mining area; Sorby Mgt (2012c) provided summaries of the information. The area immediately surrounding the tailing storage facility is covered in woodland or open woodland of Bohemia, Gutta-percha, and Eucalyptus trees over open to sparse shrub-land of yellow and white-wood, over mixed or closed grassland/forbland. The screening assessment conducted will protect aquatic algae; because algae are likely to be more susceptible to the tailings overflow than terrestrial plants, it is considered the algae assessment is protective of terrestrial plants. It is also noted the effects of submerging terrestrial plants with no significant adaptations for dealing with high volumes of flood water is likely to overshadow any effect of chemical exposure.

5.3.5 Mammals, reptiles, amphibians, and birds

A large number of reptile, amphibian, ground dwelling mammal species, and birds were identified within the Sorby Hills project area (Sorby Mgt 2012c), of which *Ctenotus rimacola camptis* (olive-brown skink) is listed as 'vulnerable' under the *Territory Parks and Wildlife Conservation Act 2000* and a total of seven (out of 113 recorded) bird species are listed under the *Environment Protection Biodiversity Conservation Act 1999*.

Amphibians have intimate contact with water, since a portion of their life cycle is spent inhabiting it. The WQGs used in the screening assessment for aquatic organisms have been derived with consideration of toxicity to amphibians, and thus the assessment is considered protective of amphibians.

Most mammals, reptiles, and birds are primarily terrestrial, and are unlikely to come into intimate contact with tailings overflow. Limited exposure to tailings chemicals could potentially occur via ingestion of pooled/sheeted floodwater mixed with tailings overflow, ingestion of organisms (e.g. crustaceans) inhabiting the flood water, or dermal contact while swimming in the water. The aquatic screening assessment conducted is considered protective for these organisms. Some of these exposure routes were further considered in Stage 2 assessments for xanthates and NaHS (see Appendices B and C).

6. Uncertainty analysis

The risk assessment process involves a number of steps (e.g. exposure assessment, toxicity assessment and risk characterisation), each of which incorporates the use of assumptions and simplifications to manage uncertainty and lack of information. Without such assumptions and simplifications it would not be possible to quantitatively evaluate the potential for environmental impacts. Although uncertainties in the risk assessment may influence its accuracy, the assumptions used to cope with unknown data for specific parameters err on the side of safety and therefore bias the evaluation to over estimation of environmental risk. It must be realised however the conservatism embedded in any one value is at least additive, most times multiplicative, with conservatisms in other values such that the cumulative or compound conservatism incorporated into the assessment can be very large. This is especially so when gross, unrealistic default parameters are used in lieu of measured data.

This section contains a general qualitative discussion of the major uncertainties in the risk assessment and their potential influence on the conclusions. The uncertainties are primarily related to:

- 1) Concentrations of substances in the pilot tailings material.
- 2) Assumed dilutions.
- 3) Applicability of the water quality guidelines.
- 4) Length of time floodwater will remain on the plains.
- 5) Assumptions for mammals, birds and reptiles.

1) *The representativeness of measured concentrations in pilot tailings material.*

It was assumed in this assessment the concentrations of soluble metals/metalloids in the pilot tailings material are representative of those in the tailings storage facility supernatant. Although the pilot tailings material was carefully made by Sorby Management there is nonetheless uncertainty that can only be addressed once the mine is in operation. It is however unlikely that there will be differences of sufficient magnitude that will change the conclusions of this report.

2) *The assumed environmental dilutions of the tailings overflow.*

The initial dilution of tailings supernatant by 7.5x to form the overflow waters from the storage facility is simply the effect of filling up the 1.5m freeboard space with rainwater. This is likely to

overestimate concentrations of substances in the overflow water because when full there will be minimal mixing of water in the tailings dam. Most of the water flowing over the spillway will be water from the top layer and will be mostly rainwater.

For assessing the environmental impact of the tailings overflow three environmental dilutions have been assumed. They represent dilutions close to the facility (10x), far afield at the permanent water courses (10,000x), and for areas in between (1,000x). While it is not known precisely where the mobile boundaries of these dilutions may be it is reasonably certain such dilutions will occur. Indeed advice from Sorby Mgt¹¹ indicates the dilutions are likely conservative of what could be expected in the environment. In which case the assessment potentially overestimates the impact from substances in tailings overflow water.

3) *The applicability of the water quality guidelines (WQGs) used in the assessment.*

The ANZECC water quality guidelines are derived for protecting the viability of the majority of species populations in permanent water bodies that have reasonably stable ecosystems and food chains that need to be maintained. Their applicability to turbid flood waters invading a system that is regularly dry is debatable. Intuitively it would be expected the guidelines would be markedly over protective and hence the assessment over predictive of impacts.

4) *The length of time floodwater will remain on the plain after a heavy storm event.*

Exactly how long water is likely to remain on the floodplain is unknown. To address this uncertainty, WQGs have been used which are protective for chronic exposures, i.e. exposures for the length of an organism's life cycle or at critical reproductive stages.

5) *Assumptions made for mammals, birds and reptiles.*

The quantitative exposure assessments in Appendices B and C have been conducted to provide a reasonably conservative estimate of whether or not toxicity to these species could be expected. There are numerous uncertainties associated with the extrapolation of data from a mammalian test species to Australian mammals, not the least of which includes differences in physiology and behaviour.

¹¹ Telephone conversation with Ed Newman on 8/08/2012.

The allometric equations used to estimate drinking water intakes are likely to overestimate the water intakes of Australian mammals, since most obtain their water from their diets.

Considering the embedded conservatism in the assessment (e.g. the use of NOAELs instead of effect levels for evaluation of reproductive effects and the likely overestimation of the dose of chemical delivered to each animal), these uncertainties are unlikely to alter the conclusions made.

7. Overall conclusions

Under the assumptions used in this risk assessment, there may be potential for toxicity to sensitive aquatic organisms (e.g. crustaceans or algae)¹² from ethyl xanthates and sodium hydrogen sulphide in the tailings if an overflow were to occur. Such effects are likely to be confined to organisms inhabiting areas in the immediate vicinity of the storage facility (i.e. areas subject to only a 10x environmental dilution). The risk decreases as the environmental dilution increases, and thus as the migratory distance from the tailings storage facility increases. Hence the likelihood of adverse impacts on permanent waterways is much lower than in the immediate vicinity of the facility.

Other low organism groups (troglofauna and stygofauna) as well as higher organisms (e.g. amphibians, terrestrial plants and mammals, birds) are not expected to be adversely impacted. There was insufficient toxicity information to evaluate the potential of adverse effects to reptiles.

The assessment suggests it is only the process chemicals xanthates and hydrogen sulphide that may impact organisms and then only when the overflow water is minimally diluted (10x) near the facility itself. Relative to the entire flood plain this is a very small area. An area in which the ecosystem is likely to be already affected by construction and maintenance activities of the storage facility. If susceptible organisms are present, and if they are affected, the impact of substances in the tailings overflow water on the viability of populations within the wider floodplain ecosystem will be negligible compared to that which occurs with the periodic flooding of the plain.

¹² This assumes these organisms are present in pooled/sheeted water on the floodplains.

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Appendix A

SGS (2012). Australian standard leaching procedure (ASLP) AS 4439 1997 for the 0264 FTI damp sample from AMML Pty Ltd. Plus heavy metal analysis for 0264 filtrate FTI float 1 and 2. SGS Report Number ENV 15873 (LIMS 109936). 16th July, 2012.

Attached under separate cover.

Appendix B

Toxicity of xanthates to mammals, reptiles and birds

There is limited information available on the toxicity of xanthates to mammals, and no information could be found for reptiles or birds. The information has been sourced primarily from literature reviews. No attempt has been made to verify this information, or to conduct a comprehensive literature review, as this was considered beyond the scope of this project.

NICNAS (1995) has published a priority assessment report for sodium ethyl xanthate (SEX), in which a brief review of the available toxicological data for SEX and other xanthates to mammals is provided. Since xanthates readily decompose to carbon disulphide (CS_2) in the stomach of animals and are metabolised to CS_2 in tissues, the toxicity of xanthates is largely the same as that of CS_2 (NICNAS 1995, 2000).

The oral LD_{50} of SEX to male mice is 730 mg/kg. As might be expected with CS_2 being the ultimate toxic metabolite, clinical observations were primarily related to the nervous system. Symptoms included depression followed by hyperexcitability, tremors, paralysis, exophthalmia, and clonic then tonic convulsions. Other observations included pinkness of the feet, preening and salivation. Gross pathology revealed consolidated lungs, pale granular livers, unusually small spleens and atonic intestines. Surviving animals showed no abnormalities. Thus the target organs for oral toxicity of SEX were the central nervous system, liver and spleen (NICNAS 1995).

The LD_{50} of various other xanthates (e.g. potassium ethyl, sodium isopropyl, potassium isopropyl, potassium n-butyl, sodium isobutyl, potassium isobutyl, sodium sec-butyl, potassium amyl, and potassium iso amyl xanthate) ranged from 411 to 583 mg/kg in mice and 1,000 to > 2,000 mg/kg in rats. Similar symptoms and pathology were seen in these studies as with SEX (NICNAS 1995).

A 4-month oral toxicity study with 10 mg/kg potassium butyl xanthate administered daily to rats resulted in tachypnoea, cyanosis, loss of hair and dermatitis observed from week 6 to week 7 of the study. Loss of weight and increase in blood sugar and cholesterol were observed later. Convulsions and paralysis of the extremities were observed in some animals from week 9 of administration. A No-Observed Adverse Effect Level (NOAEL) in this single-dose study was not established (NICNAS 1995).

The most sensitive effect with repeat dosing of xanthates is peripheral neurotoxicity, primarily manifested as decreased nerve conduction velocity (NICNAS 1995).

For an environmental risk assessment, the endpoints of most interest are those that could result in a significant change to the survivability of the population, e.g. reproductive impairment or death of large numbers of individuals (Sample et al. 1996). Therefore the nervous system effects observed in both acute and repeat-dose oral toxicity studies with laboratory rats and mice, while needed to be avoided, are not likely to impact on population viability. There was no information available on the reproductive toxicity of xanthates. However, since xanthate toxicity is directly mediated by its metabolite, CS₂, information for this chemical can be used to address the data gap that exists in the xanthate toxicological data base.

Comprehensive reviews of the mammalian toxicity of CS₂ have been produced by the World Health Organisation (WHO 2002) and the Agency for Toxic Substances and Disease Registry (ATSDR 1996). With regards to mortality, median lethal doses (LD₅₀) were 3,020 mg/kg/d CS₂ in rats, whereas no deaths occurred after gavage administration of lower concentrations (253 or 632 mg/kg/day) (ATSDR 1996).

Human data indicate inhalation exposure to CS₂ can affect the reproductive system; in males, sperm morphology, hormone levels and libido have been altered by exposure to CS₂. Data in animals support the reproductive effects seen in humans.

For the toxicological effects relevant for this assessment the available information indicates reproductive effects occur at lower concentrations than lethality. Therefore a reproductive endpoint in the rabbit, the most sensitive species when exposed to CS₂ for which there is information, was used in the assessment. At the lowest dose tested in rabbits (25 mg/kg/d given orally during days 9 to 19 of gestation) CS₂ caused severe toxicity to the foetus¹³.

During stomach degradation and tissue metabolism, one molecule of xanthate can give rise to one molecule of CS₂. Since the toxicity of xanthates is mediated by CS₂, the LOAEL for CS₂ in rabbits can be used as a LOAEL for xanthates if it is assumed all ingested xanthate is degraded/metabolised. It is conventional to apply a 10x adjustment factor to the LOAEL to account for uncertainty when using the LOAEL rather than the no observed effect level (NOAEL). Hence the assumed NOAEL in rabbits, had lower doses been used in the study, is 2.5 mg/kg/d. For wildlife risk assessments extrapolation of critical chemical doses from one species to another is achieved by scaling the dose according to body weight (US EPA 2009; Sample et al. 1996).

¹³ In rabbits gavaged with 25, 75, or 150 mg/kg/d CS₂ in oil during days 9 to 19 of gestation, embryo- and foetotoxicity were observed at all dose levels (increases in per cent resorptions, dead foetuses). The highest dose also induced significant increases in the frequency of malformed foetuses. A LOAEL for severe foetotoxicity effects was established at the lowest dose (25 mg/kg/d) (ATSDR 1996).

Numerous physiological functions such as metabolic rates, as well as responses to toxic chemicals, are a function of body size. Smaller animals have higher metabolic rates and usually are more resistant to toxic chemicals because of more rapid rates of detoxification (Sample et al. 1996). The US EPA (US EPA 2009; Sample et al. 1996) applies an equation in wildlife risk assessments to derive wildlife mammalian toxicity values from laboratory animal data by adjusting the dose according to differences in body size. For mammals, this relationship is expressed in terms of body weight as per Equation B1; the assumption is that the dose per body surface area for the test and wildlife mammalian species would be equivalent.

$$d_w = d_t \times (bw_t/bw_w)^{1/4} \dots\dots\dots \text{Equation B1}$$

Where:

d_w = dose (i.e. NOAEL or LOAEL, in mg/kg/d) for mammalian wildlife species

d_t = dose (i.e. NOAEL or LOAEL, in mg/kg/d) for mammalian test species

bw_t = body weight (kg) for mammalian test species

bw_w = body weight (kg) for mammalian wildlife species

Equation B1 was used to derive adjusted dose-based toxicity values for two Australian eutherian mammals (i.e. non-marsupials) which have been identified to inhabit the Sorby Management project area (Sorby Mgt 2012c) (Table B1).

Water consumption rates for the Australian mammals could not be readily sourced from the literature. Instead, water intakes were estimated from an allometric regression model based on body weight (Calder and Braun 1983); these are summarised in Table B1. This model is recommended for use by the US EPA (2009; Sample et al. 1996) in wildlife risk assessments.

If it is assumed an animal's sole source of drinking water is the tailings overflow, an animal's xanthate intake if consuming water in the immediate surround of the tailings storage facility (i.e. Scenario 1, which assumes 10x environmental dilution) is 0.02-0.09 mg/kg/d (Table B1). These intakes are at least 100 times lower than the respective wildlife NOAELs, i.e. the dose of xanthates at which adverse reproductive effects would not be anticipated. In reality, the estimated doses would be much lower, since most Australian mammals have a low requirement for free water because they obtain most of their water from the diet (Macroevolution undated, OFFW 2011, Blair-West et al. 1983). Therefore, toxicity of xanthates in tailings overflow to eutherian (i.e. non-marsupial) mammals is not expected.

Since the LOAEL used in the assessment to derive adjusted wildlife NOAELs is from a study identifying foetal toxicity in rabbits (a eutherian mammal), and marsupials do not have a placenta, the

NOAEL from the reproductive study is not applicable to marsupials. There was no information available for marsupials to allow quantitation of potential effects. However, intuitively the calculated intakes of xanthate in drinking water for marsupials are also low (Table B1), so an effect would probably not be expected.

Table B1: Estimated intake of xanthate in drinking water for mammals expected to be present in the Sorby Hills region

Mammal ^a	Body weight (kg)	Reference	Adjusted wildlife NOAEL (mg/kg/d) ^b	Estimated drinking water intake (L/d) ^c	Intake of xanthate in drinking water (mg/kg/day) ^d	Margin of Exposure ^f
Long-tailed planigale	0.004	TheWebsiteof Everything (undated)	- ^e	0.0007	0.09	-
Pygmy long-eared bat	0.009	Geiser and Brigham (2000)	11.1	0.001	0.06	185
Short-tailed mouse	0.025	WA DEC (undated)	8.6	0.004	0.08	108
Antilopine wallaroo	17.5-37	AWC (undated)	- ^e	1.3-2.6	0.02-0.04	-

^a The first three mammals in the table were identified in the fauna surveys conducted for the Sorby Project Area (Sorby Mgt 2012c). The antilopine wallaroo (*Macropus antilopinus*) was also included as it is one of the largest mammals likely to inhabit the region (AWC undated, Ritchie et al. 2008)

^b Calculated using Equation B1. The body weight of the gravid New Zealand white rabbits used in the study from which the laboratory animal LOAEL is derived was not provided in the abstract. The full study could not be sourced. At sexual maturity, a female New Zealand white rabbit weighs approximately 3.5 kg (Harlan 1995, Charles River undated, Masoud et al. 2005). This was used as 'bw_r' in Equation B1.

^c Drinking water consumption rates were estimated using an allometric regression model based on body weight recommended for use by the US EPA (2009; Sample et al. 1996) in wildlife risk assessment in the absence of data. The equation is:

$$DW_m = 0.099(bw)^{0.9}$$

Where:

DW_m = drinking water intake of mammal (L/day)

bw = body weight of mammal (kg)

^d Xanthate intake calculated as follows:

Intake of xanthate in drinking water per animal (mg/animal/day) =

Estimated daily drinking water intake (L/d) x concentration of ethyl and butyl xanthate anticipated in overflow (mg/L) (S1)

Intake of xanthate in drinking water per kg per day (mg/kg/d) =

Intake of xanthate in drinking water per animal (mg/animal/day) ÷ Body weight of animal (kg/animal)

Thus for the long-tailed planigale:

Intake of xanthate in drinking water per animal (mg/animal/day) = 0.0007 L/d x 0.5 mg/L (rounded) = 0.00035 mg/animal/d

Intake of xanthate in drinking water per kg per day (mg/kg/d) = 0.00035 mg/animal/d ÷ 0.004 kg/animal = 0.09 mg/kg/d

^e The LOAEL used in the assessment to derive an adjusted wildlife NOAEL is from a study identifying foetal toxicity in rabbits (a eutherian mammal). Marsupials do not have a placenta, therefore the NOAEL from the reproductive study is not applicable to marsupials. There was no information available for marsupials to allow quantitation of potential effects.

^f The margin of exposure is the ratio of the adjusted wildlife NOAEL to the intake of xanthates in drinking water per day.

No toxicity information for xanthates (or CS₂) could be located for reptiles or birds; however there is information available for birds of a related dithiocarbamate, Thiram, whose toxicity is also mediated by CS₂. In animals, Thiram is rapidly converted to dimethyldithiocarbamate and then to dimethylamine and CS₂. CS₂ is converted to COS and carbonate. About 70-80% of the dose is metabolised through the CS₂ pathway (WHO/FAO 1985). Hence in the absence of direct avian toxicity information for xanthates, it is reasonable to extrapolate from information available for Thiram.

Knowing approximately how much CS₂ is produced by Thiram and ethyl or butyl xanthates the toxicological mass doses (mg/kg) of Thiram can be converted to equivalent molar doses (mmol/kg) of CS₂, and then this dose can be converted to an equivalent mass dose (mg/kg) of xanthate. The lowest bird LD₅₀ for Thiram is 100 mg/kg in the sparrow (WHO/FAO 1985). The molar equivalent dose¹⁴ is 0.42 mmol/kg. If it is conservatively assumed 70% of orally administered Thiram is metabolised to CS₂, and one mole of Thiram produces two moles of CS₂, the molar equivalent dose¹⁵ of CS₂ is 0.59 mmol/kg (i.e. 0.00059 mol/kg). One mole of xanthate can theoretically give rise to one mole of CS₂.

Therefore 0.00059 mol/kg of CS₂ = 0.00059 mol/kg x 144.1 (or 188.4)¹⁶ g/mol
= 85 – 110 mg/kg xanthate (rounded).

Thus the read-across toxicity information from Thiram for birds indicates for the most sensitive species of bird for which there is toxicity data, the equivalent LD₅₀ dose of ethyl or butyl xanthate giving rise to the same dose of active metabolite would be 85-110 mg/kg. Since the starting point for the calculations was the LD₅₀ for Thiram from a sensitive species a safety factor of 5.7 can be used to convert the derived value to one that can be used for risk assessment that corresponds to the hazardous dose for 5% of species for the no-observed effects level (NOEL) (Luttik and Aldenberg 1997). The toxicity reference values of ethyl and butyl xanthate that can be used to judge potential impact in birds is therefore 15-19 mg/kg (rounded).

At the concentration of total xanthates in overflow for Scenario 1 (0.5 mg/L), a 0.1 kg bird would need to ingest 1.5-2 L (10-20 times its body weight) of overflow a day to reach the surrogate NOEL. This volume would be greater for Scenarios 2 and 3. Therefore, toxicity of xanthates in tailings overflow to birds is not expected.

¹⁴ Concentration in mol/kg.bw calculated using a molecular weight of 240 g/mol for Thiram.

¹⁵ 0.42 mmol/kg.bw x 0.7 x 2 = 0.59 mmol/kg.bw

¹⁶ The molecular weight of SEX is 144.1, and of potassium butyl xanthate is 188.4 g/mol

As indicated previously, no toxicity information for xanthates (or CS₂) could be located for reptiles. Thus the potential for adverse effects to reptiles from overflow cannot be evaluated.

Conclusion

Since the estimated doses to animals via drinking tailings overflow water in the near vicinity of the tailings storage facility (an assumed 10x dilution of water within the tailings dam) are less than the doses estimated to impact eutherian mammalian reproduction and bird lethality, it is concluded it is very unlikely mammal and bird populations in the area will be impacted. In areas further away from the facility where dilutions of overflow water are higher the likelihood of adverse population effects are even lower. Due to a lack of information for marsupials and reptiles, the potential impact from overflow cannot be evaluated for these animals.

Appendix C

Toxicity of hydrogen sulphide to mammals, reptiles and birds

For an environmental risk assessment, the toxicological endpoints of most interest from animal studies are those that could result in a significant change to the survivability of the population, e.g. reproductive impairment or death of a large number of individuals (Sample et al. 1996).

High concentrations (≥ 500 ppm, i.e. ≥ 700 mg/m³)¹⁷ of hydrogen sulphide gas must be inhaled by laboratory animals to result in death. No deaths were found after 30 adult female mice inhaled 100 ppm hydrogen sulphide (140 mg/m³) over a period of 1-4 days (ATSDR 2006). Hydrogen sulphide is not regarded as a reproductive toxicant. No significant alterations in reproductive performance¹⁸ were observed in mice or rats exposed to 10-80 ppm hydrogen sulphide (14-112 mg/m³) for 60-90 days. The available animal data suggest H₂S is not a developmental toxicant at concentrations of 80 ppm or lower (ATSDR 2006). These studies established a No Observed Adverse Effect Level (NOAEL) of 80 ppm (112 mg/m³) in rats and mice. This is equivalent to an absorbed dose of 18 mg/kg/day in rats.¹⁹ One molecule of NaHS can theoretically give rise to one molecule of H₂S. Thus, using a similar rationale as for the xanthates in Appendix B, the NOAEL of 18 mg/kg/d for H₂S was used to assess the potential for NaHS to cause reproductive effects to Australian mammals. Because a wide range of reproductive parameters were evaluated in the study on which the NOAEL is based, including parameters that are not unique to eutherian mammals, the assessment is also applicable to marsupials.

The approach taken is the same as that for xanthates (Appendix B).

Equation B1 (in Appendix B) was used to derive adjusted dose-based toxicity values for several different Australian mammals (including marsupials) which may inhabit the Sorby Management project area using body weight information sourced within the public domain (Table C1). Anticipated water intakes were estimated from an allometric regression model based on body weight (Calder and Braun 1983). This model is recommended for use by the US EPA (2009; Sample et al. 1996) in

¹⁷ 1 ppm H₂S = 1.4 mg/m³

¹⁸ A wide range of end points were evaluated in these studies. No alterations were found in any of them which included histopathological changes in reproductive organs, sperm count or morphology, gestation length, viability, litter size, and fertility (e.g mating index, fertility index, post-implantation loss, late resorptions, still births).

¹⁹ Using an inhalation rate of 0.23 m³/d and a body weight of 0.35 kg for rats (US EPA 2002), and assuming 100% of the administered concentration is absorbed, a dose of 18 mg/kg/d can be calculated as follows:

$(112 \text{ mg/m}^3 \times 0.23 \text{ m}^3/\text{d} \times 6 \text{ hours exposure}/24 \text{ hours per day}) \div 0.35 \text{ kg} = 18.4$, rounded to 18 mg/kg/d.

wildlife risk assessments. However, these equations may result in overestimates of drinking water intakes for many Australian mammals, since most have a low requirement for free water because they obtain most of their water from the diet (Macroevolution undated, OFFW 2011, Blair-West et al. 1983).

If it is assumed an animal's sole source of drinking water is the tailings overflow water, a mammal's NaHS intake from the tailings overflow in the immediate surround of the tailings storage facility (i.e. Scenario 1, assumes 10x environmental dilution) can be calculated using the estimated drinking water intakes in Table C1.

The estimated NaHS intakes (0.3-0.7 mg/kg/d) are 20-110 times lower than the respective wildlife NOAELs, i.e. the dose of NaHS (i.e. H₂S) at which adverse effects would not be anticipated. Therefore, toxicity of NaHS in tailings overflow to mammals (including marsupials) is not expected.

Table C1: Estimated intake of NaHS in drinking water for mammals expected to be present in the Sorby Hills region

Mammal ^a	Body weight (kg)	Reference	Adjusted wildlife NOAEL (mg/kg/d) ^b	Estimated drinking water intake (L/d) ^c	Intake of NaHS in drinking water (mg/kg/day) ^d	Margin of Exposure ^e
Long-tailed planigale	0.004	TheWebsiteof Everything (undated)	54.7	0.0007	0.7	78
Pygmy long-eared bat	0.009	Geiser and Brigham (2000)	44.6	0.001	0.4	112
Short-tailed mouse	0.025	WA DEC (undated)	34.6	0.004	0.6	58
Antilopine wallaroo	17.5-37	AWC (undated)	5.6-6.7	1.3-2.6	0.3	20

^a The first three mammals in the table were identified in the fauna surveys conducted for the Sorby Project Area (Sorby Mgt 2012c). The antilopine wallaroo (*Macropus antilopinus*) was also included as it is one of the largest mammals likely to inhabit the region (AWC undated, Ritchie et al. 2008)

^b Calculated using Equation B1. The NOAEL was the same in four laboratory studies, two of which were conducted using Sprague-Dawley rats, one using Fisher-344 rats and one using B6C3F1 mice. Only one of the four studies could be sourced by ToxConsult, a study using Sprague-Dawley rats. The body weight of the female rats in the study (Dorman et al. 2000) ranged from 0.25 – 0.38 kg before parturition. As the body weights from the other studies are not known, the US EPA (1994) default Sprague-Dawley female rat body weight of 0.34 kg was used as 'bw_i' in Equation B1.

^c Drinking water consumption rates were estimated using an allometric regression model based on body weight recommended for use by the US EPA (2009; Sample et al. 1996) in wildlife risk assessment in the absence of data. The equation is:

$$DW_m = 0.099(bw)^{0.9}$$

Where:

DW_m = drinking water intake of mammal (L/day)

bw = body weight of mammal (kg)

The results are likely to be overestimates for Australian mammals, since most have a low requirement for free water because they obtain most of their water from the diet (Macroevolution undated, OFFW 2011, Blair-West et al. 1983).

^d NaHS intake calculated as follows:
 Intake of NaHS in drinking water per animal (mg/animal/day) =
 Estimated daily drinking water intake (L/d) x concentration of NaHS anticipated in overflow (mg/L) (S1)

Intake of NaHS in drinking water per kg per day (mg/kg/d) =
 Intake of NaHS in drinking water per animal (mg/animal/day) ÷ Body weight of animal (kg/animal)

Thus for the long-tailed planigale:

Intake of NaHS in drinking water per animal (mg/animal/day) = 0.0007 L/d x 4 mg/L = 0.0028 mg/animal/d

Intake of NaHS in drinking water per kg per day (mg/kg/d) = 0.0028 mg/animal/d ÷ 0.004 kg/animal = 0.7 mg/kg/d

^e The margin of exposure is the ratio of the adjusted wildlife NOAEL to the intake of NaHS in drinking water per day. These values are rounded.

Very limited toxicity information for H₂S could be located for birds, and no information was located for reptiles. Chickens inhaling 500 ppm H₂S (700 mg/m³) for 30 minutes did not exhibit changes in ventilation; inhalation of 2,000-3,000 ppm H₂S (2,800 – 4,200 mg/m³) for 30 minutes resulted in irregular and variable tidal volume, with 4,000 ppm (5,600 mg/m³) for 15 minutes resulting in death (Brown et al. 1997). By comparison, inhalation of 500-1,500 ppm H₂S (700-2,100 mg/m³) by mammals for long periods can lead to respiratory failure and death (Brown et al. 1997). This suggests chickens (and perhaps other birds) may be less sensitive than mammals to the toxic effects of H₂S.

Therefore, although limited toxicity information for H₂S could be located for birds, it would be reasonable to expect toxicity similar to or lower than that in mammals.

No toxicity information for H₂S could be located for reptiles. Thus the potential for adverse effects to reptiles from overflow cannot be evaluated.

Conclusion

Since the estimated doses to animals via drinking tailings overflow water in the near vicinity of the tailings storage facility (an assumed 10x dilution of water within the tailings dam) are less than the doses estimated to impact mammalian (including marsupial) reproduction, it is concluded it is very unlikely mammal populations in the area will be impacted. In areas further away from the facility where dilutions of overflow water are higher the likelihood of adverse population effects are even lower. Although limited toxicity information could be located for birds, it would be reasonable to expect toxicity similar to or lower than that in mammals. Due to a lack of information for reptiles, the potential impact from overflow cannot be evaluated.

Appendix D

Derivation of site-specific preliminary aquatic screening values (PASVs)

D.2 Methyl isobutyl carbinol (MIBC)

MIBC is predicted to photo-degrade rapidly by reaction with hydroxyl radicals in the atmosphere with a half-life of approximately 10 hours (calculated by UNEP 2005). When released into water, the chemical is unlikely to partition to soil/sediment. Low bioaccumulation potential is expected based on the partition coefficient ($\log K_{ow}$ is 1.68) and other physical/chemical parameters (UNEP 2005). MIBC is also readily biodegradable.

A PASV for MIBC was derived as follows. A 48-hour EC_{50} ²⁰ for *Daphnia magna* of 337 mg/L and a No Observed Effect Concentration (NOEC) of 288 mg/L were found in the literature (UNEP 2005). The 96-hour growth rate reduction EC_{50} for algae (*Pseudokirchneriella subcapitata*) is 334 mg/L and for biomass reduction it is 147 mg/L. NOECs for these same endpoints were 75.5 and 41.6 mg/L, respectively (UNEP 2005). A 48-hour LC_{50} ²¹ in amphibians (*Xenopus laevis*) was 656 mg/L (US EPA 2012b). A PASV of **4,200 µg/L** was derived for this assessment from the lowest NOEC in algae by applying an assessment factor of 10 to account for possible increased sensitivities in species that have not been tested in the laboratory.

D.2 Barium

The limited aquatic toxicity information available for barium suggests it is relatively non-toxic with a 96-hour EC_{50} of 28,000 µg/L in green algae (*Pseudokirchneriella subcapitata*) and a 48-hour LC_{50} of 410,000 µg/L in the water flea (*Daphnia magna*) (US EPA 2012b). A 21-day (chronic) *Daphnia magna* test identified an LC_{50} of 13,500 µg/L, and 16% impairment of reproduction at 5,800 µg/L (WHO 1990). Fish are much less sensitive, with a 96-hour No Observed Effect Level (NOEL) of 500,000 µg/L reported for the sheepshead minnow. The lowest of the values identified (5,800 µg/L) was used as a NOEC for reproductive effects in water fleas to derive a preliminary aquatic screening value for this report. An uncertainty factor of 10 was applied to account for greater sensitivities in other species not measured in the laboratory to give a PASV of **580 µg/L** for barium.

D.3 Strontium

A NOEC for strontium toxicity could not be located. LC_{50} s from chronic-duration toxicity studies indicate strontium is relatively non-toxic (e.g. 7-day LC_{50} for the scud *Hyalella azteca* was >3,150 µg/L)

²⁰ EC_{50} = test concentration resulting in an effect of concern in 50% of the test population

²¹ LC_{50} = test concentration resulting in lethality of 50% of the test population

(US EPA 2012b). This value was used to derive a preliminary aquatic screening value (PASV) of **31.5 µg/L** by applying an assessment factor of 100 (10x for use of a chronic LC₅₀ instead of a NOEC, and 10x to account for potential greater sensitivity in other organisms).

D.4 Titanium

Titanium is an essential metal for some organisms (CSIRO 2009). Very limited aquatic toxicity information for titanium could be located in the databases consulted (CSIRO 2009, Markich et al. 2002, US EPA 2012b). 7-day (chronic) LC₅₀s in the amphipod *Hyalomma azteca* were 272 µg/L and 3,150 µg/L (US EPA 2012b). Toxicity of titanium chloride and titanium oxide were also low, with NOECs and EC₁₀s for various organisms in the high thousand µg/L range. Thus the lowest value found (7-day LC₅₀ of 272 µg/L for an amphipod) was used to derive a PASV of **2.7 µg/L** for this report, by applying an assessment factor of 100 (10x for use of an LC₅₀ instead of a NOEC and 10x to account for potential higher sensitivities in other organisms not tested).

Appendix E

Table E1: Assumed concentrations of chemicals of interest for each exposure scenario considered

Chemical	S1 conc (µg/L), 10 x dilution ^a	S2 conc (µg/L), 1,000 x dilution ^a	S3 conc (µg/L), 10,000 x dilution ^a
Lime (CaO)	7,667	76.7	7.7
Methyl isobutyl carbinol (MIBC)	1,333	13.3	1.3
Ethyl xanthate	267	2.7	0.3
Butyl xanthate	200	2	0.2
Sodium hydrogen sulphide (NaHS)	3,733	37.3	3.7
Silver	0.007	0.00007	0.000007
Aluminium	1	0.01	0.001
Antimony	0.6	0.006	0.0006
Arsenic	0.007	0.00007	0.000007
Barium	7	0.07	0.007
Beryllium	0.007	0.00007	0.000007
Bismuth	0.007	0.00007	0.000007
Boron	0.3	0.003	0.0003
Cadmium	0.1	0.001	0.0001
Chromium	0.007	0.00007	0.000007
Cobalt	0.3	0.003	0.0003
Copper	0.5	0.005	0.0005
Iron	1	0.01	0.001
Lead	8	0.08	0.008
Magnesium	240	2.4	0.2
Manganese	20	0.2	0.02
Molybdenum	0.2	0.002	0.0002
Nickel	0.8	0.008	0.0008
Phosphorus	0.3	0.003	0.0003
Potassium	75	0.7	0.07
Selenium	0.01	0.0001	0.00001
Silicon	45	0.5	0.05
Sodium	1,200	12	1.2
Strontium	0.6	0.006	0.0006
Sulphur	1,267	12.7	1.3
Thallium	0.007	0.00007	0.000007
Thorium	0.01	0.0001	0.00001
Tin	0.3	0.003	0.0003
Titanium	0.007	0.00007	0.000007
Vanadium	0.007	0.00007	0.000007
Zinc	120	1.2	0.1
Mercury	0.0007	0.000007	0.0000007

conc = concentration; S1 = Scenario 1; S2 = Scenario 2; S3 = Scenario 3

^a The dilution factors were applied to the assumed concentrations of chemicals in the overflow (i.e. the 7.5x diluted chemical concentrations in the supernatant).

Appendix F

ANZECC (2000) trigger values

The Australian and New Zealand Environment and Conservation Council (ANZECC 2000) have derived guidelines for the protection of aquatic organisms termed ‘trigger values.’ Trigger values for chemical toxicants are concentrations deemed to be non-hazardous to aquatic organisms; these concentrations are normally compared with environmentally measured concentrations to determine if there may be a potential for harm to the environment. If the environmental levels do not meet the trigger values, it may indicate a *potential* environmental problem and prompt a management response to more accurately determine whether the water is unlikely to elicit adverse effects in aquatic organisms or to remedy the problem. Trigger values are intended to be used as starting points to initiate an investigation; such an investigation may lead to the development of more refined guidelines tailored to regional, local, or site-specific conditions. Exceedences do not signify adverse impacts to aquatic ecosystems will occur (or have occurred). Conversely, meeting the guidelines provides certainty that there will be no significant impact on aquatic ecosystems.

The trigger values for toxicants are risk-based in the sense that they are calculated to protect a pre-determined percentage of species from sustained (chronic) exposure to toxicants with a specified level of confidence they will not be adversely affected. Trigger values are provided for three levels of protection according to the condition of the aquatic ecosystem:

- *high conservation/ecological value systems* (99% of species protected)²²,
- *slightly to moderately disturbed ecosystems* (95% of species protected)²², and
- *highly disturbed ecosystems* (80% of species protected)²².

The scheme is most commonly applied to ecosystems that could be classified as *slightly to moderately disturbed*. In these, aquatic biological diversity may have been adversely affected to a small but measurable degree by human activity. Once the mine has been developed, the Sorby Management Project area would most likely fit this description.

Most trigger values have been derived using data from toxicity tests for a range of species.

Depending on the data quality and quantity they have been assigned high, moderate or low reliability.

²² Both high and moderate reliability trigger values were derived using a statistical distribution method and calculated at four different arbitrary “protection levels”, 99%, 95%, 90% and 80%. Protection level signifies the percentage of species expected to be protected, which corresponds to the respective percentile from the resulting distribution curve. Low reliability trigger values were derived using a traditional assessment factor approach.

High reliability trigger values are calculated from no observable effect concentrations (NOECs) determined from chronic toxicity tests. Moderate reliability trigger values are usually derived from acute toxicity data using an acute-to-chronic conversion factor²³. Both high and moderate reliability trigger values are derived using a statistical distribution approach for assessing species sensitivity (i.e. species sensitivity distribution, SSD, method). Adopting an effect concentration from the distribution that allows for the protection of the majority of species (usually 95%) provides the trigger value.

Low reliability trigger values are derived by the simple approach of applying assessment factors²⁴ to the lowest toxicity value (usually a chronic NOEC, if available) for a particular chemical; this approach is meant to be used when there is insufficient information available for the statistical distribution method. If there is large data uncertainty, the calculated trigger value may be called an environmental concern level (ECL). Low reliability trigger values and ECLs are considered interim, or indicative working guideline concentrations subject to more test data becoming available. It should be noted the assessment factor approach often results in highly conservative values.

For this assessment, freshwater trigger values have been used for the 95% (*slightly to moderately disturbed ecosystems*) species protection level. If these were not available, low reliability trigger values or ECLs were used with cognisance as to their potential uncertainty.

²³ There is no simple relationship between acute and chronic data. The use of default acute-to-chronic ratios (ACRs) introduces a degree of uncertainty. Where ACRs have not been experimentally derived, ANZECC (2000) uses a default ACR of 10, consistent with recommendations from the OECD (1992) and US EPA (1986).

²⁴ The magnitude of these assessment factors depends on whether acute or chronic toxicity data are available and the confidence in whether the experimental data reflects the field situation. Most of the factors are multiples of 10 and larger factors are applied when there is more uncertainty in the data (e.g. the largest factor of 1000 is applied when there are very limited acute and no chronic data).