



Technical Memorandum

To: Christie Kearney, Poly Met Mining, Inc.
From: Barr Engineering Cross-Media Analysis Team
Subject: Estimated Potential Concentrations of Arsenic, Cobalt, and Copper in a Wetland for a Representative Scenario for Sulfide Mineral Dissolution; Supplement to the Cross-Media Analysis to Assess Potential Effects on Water Quality from Project-Related Deposition of Sulfur and Metal Air Emissions (October 31, 2017)
Date: March 15, 2018
Project: 23690862.04
c: Stoel Rives LLP; MineraLogic LLC; Barr File

1.0 Introduction and Background

Poly Met Mining, Inc. (PolyMet) conducted a *Cross-Media Analysis to Assess Potential Effects on Water Quality from Project-Related Deposition of Sulfur and Metal Air Emissions (Analysis)* to evaluate potential water quality effects from air emissions associated with the NorthMet Project (Project) and from the potential cumulative consequences of these air-related effects in combination with the effects of traditional water quality-related discharges (such as from the Project's Waste Water Treatment System (WWTS)), to nearby streams and wetlands (herein collectively referred to as the "Analysis") (Reference (1)). More specifically, the Analysis assessed the potential water quality effects of estimated (modeled) atmospheric loading to the environment of sulfur and three indicator metals (arsenic, copper, and cobalt) from Project-related air emissions, as well as the implications of such potential water quality effects for methylmercury concentrations in fish tissue in the St. Louis River watershed. For the Analysis, arsenic, copper, and cobalt were identified as indicator metals such that the potential effects from other metals can be judged based on the estimated effects from arsenic, copper, and cobalt (Section 2.3.3 of Reference (1)).

As described in Reference (1), potential water quality effects from air emissions associated with the Project were calculated using a series of protective assumptions (i.e., assumptions that overestimated impacts) for each part of the Analysis. These protective assumptions are not additive, but similar to human health and ecological risk assessments, are multiplicative. Therefore, the calculated potential surface water quality changes were overestimated for the Project (Reference (1)).

To provide a measure of the overestimation of water quality effects reported in Reference (1), PolyMet asked Barr Engineering to facilitate development of an analysis using more realistic, or representative, assumptions (herein referred to as the "Representative Scenario") that included modifications, when appropriate, to some of the key initial assumptions to better represent the current and predicted environmental conditions in the Project area. In other words, the Representative Scenario was developed

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to evaluate environmental conditions more likely to result from Project operations in order to consider the extent of impact overestimation resulting from the protective assumptions used in Reference (1).

The purpose of this technical memorandum is to describe the modifications that were made to create this Representative Scenario, present the results of the Representative Scenario, and compare its results with the results presented in Reference (1) (herein referred to as the "Base Case Scenario"). The Representative Scenario focused on the potential water quality effects of estimated (modeled) atmospheric loading of sulfate and the indicator metals (arsenic, copper, and cobalt) to the Wetland of Interest (WOI) watershed, the watershed with the highest estimated potential sulfide mineral particle deposition and sulfate loading.

It is important to note that the evaluation of potential water quality effects related to the Project's air emissions for the Representative Scenario remains a protective evaluation. Many components of the Representative Scenario are the same as they were for the Base Case Scenario: for example, both scenarios used the same emissions sources (stacks/vents/mobile and fugitive sulfide mineral dust), emission estimates, air dispersion modeling, and sulfide particle deposition results. Revised assumptions are limited to those that PolyMet and its experts believe are so protective as to be very unlikely to occur in the actual environment.

Table 3-3 of Reference (1) provided a list of assumptions that qualitatively identified the protectiveness (i.e., overestimation) in the specific parts of the Base Case Scenario. Each assumption was assigned a level of protectiveness for the following parts: air emissions and modeling; release rates; sulfate loading and methylmercury concentration changes; and metals loading and wetland water quality changes.

In this Representative Scenario, most of the protective assumptions in Table 3-3 of Reference (1) were retained. Assumptions that were modified and components that were revised to make them more representative are summarized in Table 1-1 and discussed in Section 2.0 of this memorandum. In general, more representative assumptions were introduced regarding particle fate, environmental conditions, and geochemical reactions. These more representative assumptions, in turn, produced lower estimates of mineral dissolution, less release of sulfur and metals (arsenic, copper, and cobalt) from sulfide mineral particles, and lower Project-related metal concentrations in wetland water.

The only changes in assumptions for the Representative Scenario are those shown in Table 1-1. All other assumptions from the Base Case Scenario (Table 3-3 of Reference (1)) were retained for the Representative Scenario.

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Table 1-1 Summary of changes for the Representative Scenario and comparison to the Base Case Scenario

Component of Analysis	Base Case Scenario ^[1]	Representative Scenario	Summary of Rationale for the Representative Scenario
Area contributing atmospheric loading (sulfate and metals)	Total watershed (upland and wetland)	Wetland only	The contribution of water and parameter loads (sulfate, arsenic, copper, cobalt) from the upland portion of the watershed to the wetland will be limited based on studies of fate and transport in the environment. The effect of this change is to better represent the actual hydrology and chemistry of the upland-wetland complex. See Section 2.2.
Length of time for mineral dissolution in oxygenated conditions	365 days	30 days	Particle movement to anaerobic conditions occurs relatively quickly according to numerous studies. The effect of this change is to better represent the movement of small particles downward in the organic soil that limits mineral dissolution. See Section 2.3.
Temperature (annual average)	25°C	11°C	The annual average temperature for the Project area for snow-free months (~mid-April through October) when mineral dissolution can occur is ~11°C, based on long-term data from the National Weather Service. The effect is to decrease mineral dissolution. See Section 2.3.
Chalcopyrite dissolution rate ^[2]	2.1E-11 mol/m ² /s	7.03E-14 mol/m ² /s	Estimate of potential oxidative dissolution by oxygen, based on analogy to pyrite for higher pH conditions. The dissolution rate also decreases because the temperature was adjusted from 25°C to 11°C. The overall effect is to decrease the potential release of sulfur and metals from particles. See Section 2.4.
Pyrrhotite dissolution rate	Instantaneous (100% of sulfur and metals released)	1.74E-8 mol/m ² /s	The dissolution rate was calculated for each particle size for the 30-day time period based on laboratory studies rather than assumed to be an instantaneous dissolution. The effect is to decrease the potential release of sulfur and metals from particles. See Section 2.4.

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Component of Analysis	Base Case Scenario ^[1]	Representative Scenario	Summary of Rationale for the Representative Scenario
Removal/retention of metals in wetlands: Arsenic Copper Cobalt	0% 90% 70%	70% 90% 70%	The estimates of metal retention in natural wetlands are on the lower end of the potential range of values from studies of atmospheric loading of metals to boreal watersheds. Only arsenic is changed for the Representative Scenario. The effect of this change for arsenic is to reduce the potential concentration in water exiting the wetland. See Section 2.5.

[1] Base Case Scenario details are from Reference (1).

[2] For the Base Case Scenario, two potential chalcopyrite dissolution rates were estimated (Section 3.2 and Appendix B of Reference (1)). The dissolution rate of 2.1E-11 mol/m²/s is the higher estimated rate of potential chalcopyrite dissolution and provided an estimate of higher concentrations of copper in the Base Case Scenario (Table 4-9 of Reference (1)). Results for the Base Case Scenario presented in this evaluation are for the higher chalcopyrite dissolution rate.

2.0 Discussion of Revised Assumptions for the Representative Scenario

The following discussion is a summary of information primarily discussed in Section 2.3.6 of Reference (1). More detailed information can be found in Section 2.3.6 and Appendix D of Reference (1).

Several terms are defined here for reference in the following subsections. "Wetland" or "wetlands" is used as a generic term that encompasses all wetland types in the Project area. "Peatland" is used interchangeably with the term "wetland" in this technical memorandum. "Peat" is a general term for the organic soil that is present in the various wetland types in the Project area and is a heterogeneous mixture of more or less decomposed plant (humus) material that has accumulated in a water-saturated environment and in the absence of oxygen.

2.1 Overview

Atmospheric deposition of particles is mainly by dry deposition processes. A large majority (~90%) of particles initially deposited to leaf and vegetative surfaces are washed from those surfaces to either the upland soils (having a litter layer that overlies the mineral soil that is composed of leaves/needles and woody materials in various stages of decomposition) or to the peat surface by subsequent rainfall events (Reference (2)).

The potential release of sulfur and metals from sulfide mineral particles depends on the environment of the aquatic and terrestrial receptors to which the particles are deposited. Deposition alone of a particle does not necessarily result in release of sulfur or metals or affect water quality; release of sulfur and metals is a function of the physical, chemical, and biological processes that act on the particle. These processes, referred to as weathering, determine which portion of the constituents in the particles are released to the environment and which portion remains immobile within the particle.

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In Reference (1), the protective assumption in estimating an average loading to a watershed was that all particles deposited to the surface of the WOI watershed, whether in the wetland itself or the surrounding upland, would contribute sulfate and metals to the wetland water. This means that all parts of the watershed were assumed to contribute sulfate and metals for mixing in the WOI water and transport to downstream surface waters.

The following discussion summarizes research findings from a variety of disciplines which on the whole indicate that particles deposited in boreal wetland environments will likely be immobilized, and that these particles will undergo very little weathering. In turn, very little weathering of particles will reduce the amounts of sulfur and metals from Project deposition that would be released from the particles to be available for potential impacts on surface waters.

2.2 Limited Contribution from Atmospheric Loading in Uplands to the Wetland Portion of the Watershed

The protective approach in Reference (1) was that all particles deposited to a watershed, whether on the wetland itself or the surrounding upland, were used to estimate the average loading of particles to the watershed and thereby all particles and all parts of the watershed contributed sulfate and metals to wetland water. However, the weight-of-evidence from numerous research studies suggest that a significant portion of the Project atmospheric loading of dust (and associated sulfate, arsenic, copper, and cobalt) will be sequestered in the uplands surrounding the WOI and will never reach the wetland portion of the watershed.

The soil surface in forested uplands in northern Minnesota generally consists of a forest floor, which is a thick, highly porous organic soil horizon containing materials in various stages of decomposition. The stages of decomposition range from newly deposited bark, leaves, and twigs to partially decomposed older leaves and twigs to well decomposed organic matter at the mineral soil boundary. Consequently, it would be expected that mineral particles deposited to this surface would also move downward through larger pores until they were trapped in smaller pores, most likely associated with the underlying mineral soil horizons, where pore diameters are <10 times the diameter of the particles (Reference (3)). As discussed for wetlands further below in this subsection and in Section 2.3 of this memorandum, the downward movement of atmospherically deposited metals through organic material can occur relatively rapidly (within days to weeks) (Reference (4); Reference (5)).

Particles deposited to the forest floor of mineral upland soils are typically in an oxygenated environment for a portion of the year during which microbial decomposition of organic materials is occurring. Spring snowmelt and large rainfall events may decrease oxygen availability for short to moderate periods of time in these aquic and oxyaquic soils as water accumulates in the organic layer and upper portion of the mineral soil. Overall, the forest floor and upper 6 to 12 inches of the mineral soil are considered an oxygenated environment (Reference (6)). In this oxygenated environment, any deposited sulfide mineral

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particles would be expected to weather and release sulfur, copper, cobalt, and arsenic. Copper, cobalt, and arsenic, like most metals, have strong affinities for organic matter and for iron oxides and hydroxides that are common to the upland soils in the Project area (Reference (7), Reference (8), Reference (9), Reference (10), Reference (11), and Reference (12)).

Sulfur released from particles would be oxidized to sulfate, with the majority of the new sulfate being taken up by microbes and vegetation as a nutrient and would not be available for transport to wetlands or downstream surface waters. Biological sulfur requirements for temperate forests are less than 0.5 g sulfur/m²/yr (1.5 g sulfate/m²/yr equivalent) and trees will often take up 2 or 3 times that amount of sulfur (luxury consumption) if available (Reference (13)). For example, sulfur deposition in deciduous and coniferous forests in the Panola Mountain Research Watershed in Georgia was 3.87 and 3.81 g sulfate/m²/yr (Reference (14)). These rates are consistent with the uptake of sulfur found in northern Minnesota (~3.9 g sulfate/m²/yr; Reference (15)). Although these deposition rates at the Panola Mountain Research Watershed are well above sulfur growth requirements (and more than 7 times the background sulfur deposition rate of 0.482 g sulfate/m²/yr in the Project area), these forests retained more than 95% of atmospheric sulfur deposition (Reference (14)).

Consequently, in the Project area, the majority of sulfate deposited to uplands by background atmospheric deposition or released from sulfide particles would be expected to be taken up by the vegetation. Sulfate not taken up by vegetation or microbes would still be in soluble form and could move with soil water (vertically and/or laterally depending on soil characteristics), but would only constitute a very small fraction of the total amount of sulfate deposition.

In addition, in a typical year in northern Minnesota, sulfide mineral particles deposited to the snowpack would be retained in the snowpack from approximately November through mid-March. During mid-winter thaws and the onset of snowmelt (late March/early April) the fine particles (2.5 microns and smaller) would be moved downward in the snowpack with melt water, while larger particles (larger than 2.5 microns) would be retained in the snowpack (Reference (16)). Particulate matter typically remains behind during snowmelt, while soluble components are washed through or out of the snowpack (Reference (17)). Fugitive sulfide mineral dust particles are expected to behave in a similar fashion to other particulate matter and remain behind during snowmelt. While particle transport (2.5 to 30 micron particles) is possible for surficial runoff, particle transport through soil pores is essentially negligible, as these particles would be trapped similar to clay particles (Reference (18)).

For wetland watersheds in the Project area, the expectation is that considerably less than 2% of sulfide mineral particles (2.5 to 30 microns) deposited to uplands would be transported to the wetland. Based on research findings for upland-peatland systems (e.g., Reference (19)), uplands retain essentially all particles (and particle-bound metals). Dissolution of minerals in cold temperatures is minimal; therefore, a minimal amount of sulfate and metals would be released from fugitive dust from the Project during the snow

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accumulation (approximately November through mid-March) and snowmelt time periods (approximately late March through April). Any soluble sulfate or metals would have the potential to be transported to the lagg or surface channels of the wetland during snowmelt (discussed further below).

Hydrologic flow from the forested uplands to the wetland occurs mainly during the snowmelt period and usually represents only a small fraction of annual precipitation to the uplands (Reference (20)). In the uplands, snowmelt water tends to move downward into the soil (Reference (17)). Data from the Marcell Experimental Forest for 1993 to 1995 indicates that approximately 2% of annual precipitation becomes surficial runoff, even on relatively steep slopes (Reference (21)). In the same study at the Marcell Experimental Forest, subsurface lateral flow amounted to 10% to 16% of the annual mean precipitation (Reference (21)). While a large volume of runoff occurs during snowmelt events, only negligible, if any, runoff occurs during summer or fall (Reference (19)). During snowmelt in early spring, the peatland and upland contribute equally to streamflow (Reference (20)). However, for the other parts of the year (e.g., late spring and summer), the wetland is the major source of outflow water and for most of the summer and winter months there is very little, if any, contribution of water from the uplands to the wetlands (Reference (20)).

For the WOI, water flowing from the uplands towards the peatland is expected to enter the lagg, a narrow zone at the peatland margin that receives water from both the upland and the peatland. Water from the upland would not flow out onto the surface of the peatland (Reference (15)). Instead, it would accumulate in the lagg area and then either seep laterally away from the peatland or flow through the lagg to the stream exiting the wetland (Reference (20)). As shown in Appendix C of Reference (1), the wetland area of the WOI can be separated into a northern half and southern half; uplands form a pinch-point that narrows the southern portion of the wetland area. Aerial photography identifies there is a visible surface channel in the very southern portion of the wetland area of the WOI that feeds into a small pond. Water channels in wetlands are expected to deliver excess water (excess in this case meaning water that cannot be stored by the wetland area) to the outflow area. Snowmelt waters, which may contain dissolved constituent such as calcium, magnesium, and sulfate, are expected to be quickly routed to the wetland outflow area (Reference (15)). As identified in Reference (22), these snowmelt waters usually have very limited interaction with the peat, particularly when the peat is frozen at the start of snowmelt.

Therefore, for the several reasons summarized, and contrary to the protective assumption in the Base Case Scenario that all particles deposited in a watershed will contribute sulfate and metals to the waterbody or watershed outlet (Reference (1)), it is expected that there will be very little contribution of sulfate or individual metals to a majority of the wetland area from the uplands around the WOI. Instead, any sulfate and metals in snowmelt water is expected to move out of the watershed where it will mix with waters either in a downgradient wetland complex (such as occurs for the WOI) or in first or second order streams. In both cases, the water leaving the watershed would be diluted with downgradient water. Because the majority of this flow occurs during snowmelt, the temperature of these waters would be just

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above freezing (just above 0°C (32°F)) and would minimize or negate any interaction with microbes or other organisms.

To better account for the limited contribution from uplands to the loading of sulfate and individual metals to the WOI, this Representative Scenario assessed the Project atmospheric loading of dust (and sulfate, arsenic, copper, and cobalt) to only the wetland portion of the watershed. To estimate Project atmospheric loading to only the wetland portion of the WOI watershed, the atmospheric loading calculations were adjusted to only select modeling results for those receptors located in the wetland area, and to exclude receptors located in the upland area.

2.3 Limited Dissolution (Weathering) of Atmospherically Deposited Particles

For the Base Case Scenario, all particles deposited within a watershed were assumed to remain in an oxygenated environment for 365 days (1 year). Studies indicate, however, that atmospherically deposited particles that arrive at the peat surface (as well as in a lagg area where a small amount of particles from the uplands accumulate) can move downward in the peat relatively quickly to an anaerobic (oxygen depleted) environment. Therefore, there is only a short time period when particle weathering can occur that results in the release of sulfate and individual metals.

Section 2.3.1 below further discusses wetland soil characteristics and functions that create and sustain an anaerobic environment, even during the warmer summer months, that sequesters particles and greatly reduces or stops particle weathering. A short discussion on how rapidly particles can be moved downward in a wetland soil and the associated adjustment made for the Representative Scenario is provided in Section 2.3.2. Because temperature plays an important role in the weathering of particles (e.g., chemical reactions and biological activity), Section 2.3.3 provides a discussion on the Project area climate and temperature and the adjustment in temperature for the Representative Scenario.

2.3.1 Expected Particle Fate in a Wetland Environment

As discussed in Reference (1), the research findings identify that particles reaching the peat surface will be transported downward. The peat surface is very porous, with total porosity ranging from 71% to 95%. Pores are larger (commonly 5 mm in size) in the upper part of the peat, and smaller at depth in the peat profile (Reference (23)). Particles tend to move through larger pores without entrapment, which does not occur until pore diameters are <10 times the diameter of the particles (Reference (3)). Most mineral particles deposited from the atmosphere (2.5 to 30 microns in size; 0.0025 to 0.03 mm) can move downward in the peat profile through the larger pores, becoming entrapped lower in the peat where pores become smaller and the bulk density increases. The downward movement of particles can occur relatively rapidly (within days to weeks) (Reference (4)).

Water tables in wetlands fluctuate in approximately annual cycles, with 12 to 18 inches of fluctuation being typical for northern wetlands or peatlands (Reference (24); Reference (25); Reference (20)). Below

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the free water table, the peat environment is anaerobic and therefore has low redox (Eh) values (<300 mV and usually <100 mV), with 330 mV considered a lower limit of aerobic activity (Reference (26); Reference (27)). It is often assumed that above the water table peat is unsaturated and, therefore, aerobic. However, this is usually not the case. A capillary fringe is commonly present in peatlands, and can extend 30 to 40 cm (12 to 16 inches) above the free water table surface (Reference (20); Reference (28)). The capillary fringe is a zone of saturation or near saturation between the free water table surface and the unsaturated zone above it, where matric forces (the attraction of water to the solid phase) hold water above the free water table surface against the force of gravity. As long as the water pressure is lower than the air pore entry pressure, water will be held above the free water table surface and free air will not displace water from the pores.

The capillary fringe is usually anaerobic (Eh values are typically <200 mV) due to slow rates of oxygen diffusion from the unsaturated zone into the capillary fringe, even during summer months (Reference (29)). For example, when the rate of oxygen consumption in the capillary fringe is moderate to high, which would be common during warmer portions of the year due to enhanced microbial activity at higher temperatures, the capillary fringe is depleted of oxygen, with very little penetration (a few millimeters) of oxygen from the unsaturated zone above (Reference (29)).

Sulfide mineral particles that settle below the upper boundary of the capillary fringe would be subjected to an anaerobic environment, where mineral dissolution would be severely curtailed. Rausch et al. (Reference (30)) identified the near complete lack of aqueous metals measured in pore water in a 10-cm (4-inch) deep sample of a bog impacted by relatively recent sulfide mining in Finland. They found that only about 0.2% of the total atmospheric inputs of cobalt and copper from sulfide mine-related dust were present in the pore water and that chalcopyrite particles up to 25 microns in diameter were present even in the upper peat layers, indicating that dissolution of sulfide mineral particles was minimal under anaerobic conditions.

These findings support the conclusion that chalcopyrite and other sulfide mineral particles that are deposited to wetlands in the Project area, and specifically the WOI, would be transported downward in the peat and below the upper boundary of the capillary fringe to an anaerobic environment and therefore, would not be expected to undergo any further weathering of significance. Thus, the sulfide mineral dust particles deposited from the Project would not be expected to continue to release sulfur and associated metals to pore waters (or surface waters) once settled into the capillary fringe which is present just below the peat surface.

2.3.2 Particles in an Oxygenated Environment Only for a Short Time

As discussed above, atmospherically deposited small particles move downward in the organic soil profile relatively quickly, with porosity of the organic soil being an important factor in the depth of particle

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movement (Reference (4); Reference (5)). More rapid downward movement occurs in less decomposed peat while slower downward movement occurs in more decomposed organic soils (Reference (4)).

The range of potential values for particles remaining in oxidizing conditions in a wetland ranges from a very short time (about 1 to 2 days) to about 50 days (Reference (4)).

Based on these studies (Reference (4); Reference (5)), the Representative Scenario used a revised length of time for particles to remain in an oxygenated environment where dissolution could occur, which reduces the potential for further release of sulfur and metals to the environment. Specifically, a value of 30 days for particles to remain in oxygenated conditions and undergo dissolution, which represents a calendar month, the high-end of the approximate mid-point of the range of potential values in the literature, was used for the Representative Scenario rather than the 365 days used in the Base Case Scenario.

Mineral dissolution and release of sulfur and metals during the 30-day time period will be affected by the Project area climate and temperature, which are discussed below.

2.3.3 Climate and Temperature

In the Base Case Scenario, the calculations for estimating potential chalcopyrite particle dissolution used 25°C (77°F) as the reaction temperature based on the laboratory data used in the dissolution rate calculations (Appendix B of Reference (1)). However, annual average temperature and annual monthly temperatures in the Project area are much lower than 25°C (77°F) as shown by data from the National Weather Service.

The climate in northern Minnesota, including the Project area, is continental with warm, wet summers and cold, dry winters. Temperature data from the 50-year record at the Marcell Experimental Forest ranged from -45° to 38°C (-49 to 100.4°F). The mean annual temperature is 3.4°C (38.1°F) (Reference (31)). In comparison, based on National Weather Service data, the annual average temperature in the Project area (Hoyt Lakes/Babbitt) ranges from about 2°C (~36°F) to about 5°C (41°F) (Reference (32)). The monthly mean temperature is 18.9°C (66°F) during July and -15.1°C (4.8°F) in January (Reference (31)). Lakes begin to freeze in November and are usually ice-free by early May (Reference (31)).

Temperatures in the peatland surfaces tend to mirror, but temporally lag behind, air temperatures. For the 0- to 5-cm (0- to 2-inch) depth, the monthly mean high ($18.3 \pm 4.4^\circ\text{C}$; $64.9 \pm 8^\circ\text{F}$) occurs in August and the monthly mean low ($-0.9 \pm 1.3^\circ\text{C}$; $30.4 \pm 2.3^\circ\text{F}$) occurs in February (data from Junction Fen from 1989 to 2016). Temperatures at deeper depths follow similar trends, but the magnitude of differences between summer highs and winter lows becomes progressively smaller with increasing depths. About 50% of the year (November to April), temperatures are considered cool to cold and biological activity in soils, including organic soils (peat), is limited.

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For this Representative Scenario, to better reflect expected conditions in the Project area, the average temperature used in the calculations to estimate sulfide mineral particle dissolution was 11°C (52°F), the actual average temperature during snow-free months (approximately April through October). This adjustment has the effect of decreasing mineral dissolution rates as discussed further in Section 2.4.1 below.

2.4 Revised Assumptions about Geochemical Reactions

In the Base Case Scenario (Section 3.2 and Appendix B of Reference (1)), geochemical dissolution rates were applied to sulfide mineral fugitive dust deposited to wetlands. A number of assumptions were made in the application of these rates, and several of these assumptions were reevaluated and subsequently modified for this Representative Scenario, as described in the following subsections.

2.4.1 Temperature Adjustment

In the Base Case Scenario (Reference (1)), the dissolution rates were calculated assuming a temperature of 25°C (77°F or 298 K), a standard laboratory temperature. For this Representative Scenario, the rates were adjusted as discussed above to the average temperature in the Project area (Hoyt Lakes/Babbitt) during snow-free months of 11°C (52°F or 284 K; see Section 2.3.3 above).

The rates laws presented in the meta-studies of chalcopyrite and pyrrhotite dissolution used in this Representative Scenario are functions of temperature. As a result, a temperature-adjusted dissolution rate was calculated by substituting 284 K for 298 K into the rate equations.

Thermodynamic parameters, such as the solubility constant used to estimate ferric iron concentrations ($\log K_{sp}=4.89$), which were empirically derived at a temperature of 25°C (Reference (33)), were not adjusted for the Representative Scenario. This results in a continued overestimate of ferric iron concentrations, which in turn results in an overestimate of potential particle dissolution rates for chalcopyrite; however, because there is no single value in the scientific literature measured at a lower temperature for this purpose, no change was made to these thermodynamic parameters (see discussion in Section 2.4.3 below).

2.4.2 Pyrrhotite Dissolution Rate

In the Base Case Scenario (Reference (1)), it was assumed that all of the sulfur and associated metals were released from all sizes of pyrrhotite particles (and other sulfide minerals that were assumed to react the same as pyrrhotite) within the first year of deposition. Studies of pyrrhotite dissolution suggest, however, that sulfur and metals would not likely be completely released from all particle sizes within one year (Reference (34)). In this Representative Scenario, the amounts of sulfur, and therefore metals, released from pyrrhotite, and other minerals assumed to react at the same rate as pyrrhotite, were constrained by geochemical dissolution rates. These rates were derived from a meta-study by Chirita and Rimstidt of pyrrhotite dissolution rates (Reference (34)) that is analogous to the Kimball et al. study used for

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Subject: Estimated Potential Concentrations of Arsenic, Cobalt, and Copper in a Wetland for a Representative Scenario for Sulfide Mineral Dissolution; Supplement to the Cross-Media Analysis to Assess Potential Effects on Water Quality from Project-Related Deposition of Sulfur and Metal Air Emissions (October 31, 2017)
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estimating potential chalcopyrite dissolution (Reference (35)) in the Base Case Scenario (Section 3.2 and Appendix B of Reference (1)).

The pyrrhotite meta-study presents rate laws for the dissolution of pyrrhotite by non-oxidative dissolution and by oxidative dissolution by ferric iron and by oxygen. The rate laws are a function of temperature and the amount of certain chemical species that vary depending on the oxidation mechanism. These chemical species include the concentrations of hydrogen ions (non-oxidative dissolution) and ferric iron (oxidative dissolution by ferric iron) and the partial pressure of oxygen (oxidative dissolution by oxygen).

As described in Appendix B of Reference (1), different dissolution mechanisms predominate under different geochemical conditions. Based on the rate laws derived by Chirita and Rimstidt (Reference (34)), at pH values above approximately 3 and with atmospheric oxygen (partial pressure of 0.2 atmospheres), the dominant pyrrhotite dissolution mechanism is oxidation by oxygen. The rate law presented in the meta-study (Reference (34)) for oxidation by oxygen is not pH-dependent, so the rate at pH 6 (which is the condition expected to be present in the WOI) was assumed in this Representative Scenario to be the same as the rate in the acidic pH range encompassed by the meta-study. The rate law is:

$$r_{O_2}(\text{mol/m}^2/\text{s}) = 1.10 \times 10^{-2} e^{\left(\frac{-30,200}{8.314 T}\right)} P_{O_2}^{0.352}$$

The oxidative dissolution rate of pyrrhotite by oxygen at a partial pressure of 0.2 atmospheres and a temperature of 284 K (11°C or 52°F) yielded a rate of $10^{-7.76}$ mol/m²/s or 1.74E-8 mol/m²/s.

To estimate the amount of sulfur released in 30 days for purposes of the Representative Scenario, this calculated dissolution rate was applied to each particle size of model-estimated pyrrhotite fugitive dust deposition (2.5, 5, 10, 15, and 30 micron) at each receptor on the air emissions modeling grid in the WOI watershed. The mathematical application of this rate was the same as the application of the chalcopyrite dissolution rate in the Base Case Scenario (Reference (1)). For the Representative Scenario, we report only the atmospheric loading for those receptors within the wetland area of the WOI watershed (see Section 2.2 of this memorandum).

2.4.3 Chalcopyrite Dissolution Rate

Data from laboratory studies evaluating mineral dissolution in acidic conditions (pH 1 to 3) were used for the Base Case Scenario to derive estimates of rate constants for dissolution of chalcopyrite mineral particles (Section 3.2 and Appendix B of Reference (1)). Under acidic conditions, oxidative dissolution by ferric iron was considered the most important mechanism for chalcopyrite particle dissolution and was used to provide a protective estimate of potential mineral dissolution (Section 3.2 and Appendix B of Reference (1)).

However, the WOI is classified as an alder thicket wetland community type in the Partridge River watershed. The pH of the WOI is likely to range from 5.5 to 7.5 (circumneutral pH) based on data from the

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DNR (2003) and Swanson and Grigal (1991) (see Table 1 in Appendix D of Reference (1)). Field data collected by PolyMet indicates the pH of the WOI averages about 6.0 (Section 2.2.2 of Reference (1)).

In environmental conditions at circumneutral pH and with atmospheric oxygen (partial pressure of 0.2 atmospheres), the dominant chalcopyrite dissolution mechanism is expected to be oxidation by oxygen (Appendix B of Reference (1)). The activity of ferric iron (Fe^{3+}) is negligible at circumneutral pH values (Reference (36)). Hydrogen ion activity (non-oxidative dissolution) is low at circumneutral pH (Appendix B of Reference (1)).

As noted in the previous section on pyrrhotite dissolution rates, the rate of oxidative dissolution of sulfide minerals by dissolved oxygen does not have a significant relationship with pH. Therefore, an estimate of oxidative dissolution by oxygen for low pH conditions is applicable to higher pH conditions such as occur in the WOI (~pH 6). In other words, the rate of oxidative dissolution of chalcopyrite by dissolved oxygen was assumed for the Representative Scenario to be approximately the same at low pH as at the higher pH in the WOI (pH ~6).

In general, when chemical reactions can proceed via multiple mechanisms, as in the case of chalcopyrite, the mechanism that is the fastest will dominate the overall reaction rate. Under the acidic conditions considered in the Kimball et al. (Reference (35)) meta-analysis, and for which data are most available, the dissolution rate of chalcopyrite was dominated by either non-oxidative dissolution or oxidative dissolution by ferric iron (Fe(III)). Most of the study conditions considered in Kimball et al. (Reference (35)) included oxygen, and, therefore, oxidative dissolution with oxygen as the oxidant remained a potential mechanism; however, this oxidative dissolution by oxygen proceeded more slowly than either of the other two mechanisms (Appendix B of Reference (1)). This observation anchors the rate of oxidative dissolution by oxygen as being slower than the other two mechanisms under the low pH conditions studied (Appendix B of Reference (1)). Additional information provided in Appendix B of Reference (1) indicates that for circumneutral pH conditions, oxidative dissolution by oxygen could be similar to, or lower than, the rate estimated for oxidative dissolution by ferric iron at pH 3 conditions, which is the highest pH for which reliable data are available.

No rate values or laws (e.g., a pH-independent rate law for oxidative dissolution of chalcopyrite by oxygen, as described above for pyrrhotite) that would apply to dissolution of chalcopyrite under the expected WOI conditions (~pH 6) could be identified in the literature. Therefore, this Representative Scenario used the rate calculated for oxidative dissolution of chalcopyrite by ferric iron at pH 4 as a proxy for the rate of oxidative dissolution of chalcopyrite by dissolved oxygen at pH 6 (i.e., in the WOI). This approach is different from the approach used in the Base Case Scenario, which used the rate calculated for oxidative dissolution of chalcopyrite by ferric iron at pH 3 as a proxy for the rate of oxidative dissolution of chalcopyrite by dissolved oxygen at pH 6. The rate for the Representative Scenario was selected based on an analogy with pyrite. As noted in Appendix B of Reference (1), using pyrite as an

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analogue, at pH higher than approximately 4.2, the rate of oxidative dissolution by Fe(III) becomes slow enough that the overall chalcopyrite dissolution rate is expected to be controlled by oxidative dissolution by oxygen. Therefore, the rate at pH 4 represents a realistic, high-end rate for a range of neutral pH conditions.

The rate law for this reaction presented by Kimball et al. (Reference (35)) is:

$$r_{\text{Fe(III)}} (\text{mol/m}^2/\text{s}) = 10^{1.88} e^{\left(\frac{-48,100}{8.314 T}\right)} [\text{H}^+]^{0.8} [\text{Fe(III)}]^{0.42}$$

A pH of 4 ($[\text{H}^+] = 10^{-4}$) and the same ferric iron ($[\text{Fe(III)}] = 10^{4.89} [\text{H}^+]^3$) assumptions used in the Base Case Scenario (Reference (1)) were used in this Representative Scenario to develop a protective proxy rate for sulfur release from chalcopyrite in the WOI (i.e., at pH 6 with oxygen as the dominant oxidizer). However, the temperature used to calculate the rate from the rate law was adjusted from 25°C to 11°C, as described above. The calculated rate was $10^{-13.15}$ mol/m²/s or 7.03E-14 mol/m²/s.

In addition, this calculated rate for the Representative Scenario was applied to each particle size (2.5, 5, 10, 15, and 30 microns) deposited at each receptor point on the air modeling grid in the WOI watershed to estimate the amount of sulfur released from chalcopyrite in 30 days rather than 1 year. An oxidation time of 30 days was used in this Representative Scenario as a more reasonable amount of time that sulfide particles deposited to a wetland would be exposed to surface weathering conditions before burial within the wetland, where dissolution rates are expected to be negligible in comparison (see Section 2.3 of this memorandum). The mathematical application of this rate was the same as the application of the chalcopyrite dissolution rate in the Base Case Scenario (Reference (1)). For the Representative Scenario, we report only the atmospheric loading for those receptors within the wetland area of the WOI watershed (see Section 2.2 of this memorandum).

2.5 Sequestering of Metals Released from Sulfide Mineral Dust in Wetlands

Section 3.6.2 and Appendix E of Reference (1) provided detailed information and discussion on metals retention/removal in wetlands. When particles deposited to wetlands undergo mineral dissolution and metals are released, copper, cobalt, and arsenic have strong affinities for organic matter in wetlands (Reference (26)). An additional mechanism for long-term removal/retention of metals in wetlands is the exclusion of solutes and colloidal particles that occurs during formation of ice. The formation of concrete frost in wetlands enhances the downward movement of particles and organic matter and their attached metals ahead of the freezing front in the wetland soil profile (Reference (20)). As described in Reference (20), *"The process of freezing moves colloidal organic matter and associated elements on nearly an annual basis from the acrotelm (upper layer of peat where organic matter regularly contributes to dissolved organic carbon and nutrients leached in streamflow) to the catotelm (deeper peat where they can be stored for millennia)"*.

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Studies have shown that the net retention of arsenic, copper, and cobalt in wetlands on a long-term basis can be greater than 99% and that the retention can extend over decades or centuries (Reference (37), Reference (38), and Reference (39)). Of particular note is that high percentages of the respective atmospheric loads of arsenic, copper, and cobalt to wetlands are retained/sequestered. The relative retention of copper, cobalt, and arsenic, measured in a number of studies of sites impacted by atmospheric deposition, with environmental conditions similar to northern Minnesota, range from 45% to >99% for copper, >90% for cobalt, and from 71% to >99% for arsenic (Reference (37), Reference (38), Reference (39)). While site-specific factors can and do affect the net retention of metals, the weight-of-evidence in the literature indicates that a high percentage of the atmospheric load to natural wetlands has been retained in the watershed and it is expected the same would occur in the WOI.

Based on the literature reviewed and the circumneutral pH conditions of the wetland area of the WOI discussed above, the expected removal/retention is in the upper range of the estimates for copper and arsenic (>90%). However, to maintain the protectiveness of the Representative Scenario and provide room for additional refinements to this analysis if needed, the following determinations were made:

- For arsenic, the removal/retention in the wetland was estimated at 70%. A value of 70% removal/retention for arsenic is at the low end of the potential range of values (the range reported in the literature is from about 70% to >99% for natural wetlands) that could be used in this Representative Scenario. However, there is some uncertainty regarding the mobility of arsenic in wetlands as some studies report removal/retention as low as 20% (Appendix E of Reference (1)). The use of 70% removal/retention of arsenic in wetlands reflects some of the uncertainty reported in the literature.
- For copper, the removal/retention in the wetland was estimated at 90%, the same percentage used in the calculations for the Base Case Scenario. A value of 90% removal/retention for copper is in the upper range of the potential values that could be selected (the range reported in the literature is from 45% to >99% for natural wetlands).
- For cobalt, the removal/retention in the wetland was estimated at 70%, the same percentage used in the calculations for the Base Case Scenario. In the Base Case Scenario, a value of 70% removal/retention for cobalt was identified as a protective value as it was slightly below the long-term average of 77% removal/retention found for treatment wetland W1D at Dunka (Appendix E-1 of Reference (1)). Based on data for natural wetlands, a value of 70% removal/retention is at the low end of the potential range of values, as the available information and studies consistently indicate low mobility of cobalt in the environment (Appendix E-2 of Reference (1)). However, there is limited removal/retention data for cobalt in natural wetlands. Therefore, the use of 70% removal/retention of cobalt for the Representative Scenario reflects both the data from treatment wetland W1D at Dunka (Appendix E-1 of Reference (1)) and the limited number of studies conducted for this parameter in natural wetlands (Appendix E-2 of Reference (1)).

3.0 Results

For the Representative Scenario, the calculations in the Cross-Media Analysis (Reference (1)) were replicated with no changes from the Base Case Scenario except for the revised assumptions discussed in Section 2.0 of this memorandum. The resulting outcomes are summarized below.

3.1 Atmospheric Sulfate Loading from Project

Table 3-1 summarizes the atmospheric sulfate loading to the WOI watershed from the Project for the Base Case Scenario (0.155 g/m²/yr) and for the Representative Scenario (0.0484 g/m²/yr). The lower estimate of sulfate loading for the Representative Scenario reflects the assumptions discussed in Section 2.0 of this memorandum. Existing conditions (background) sulfate loading was estimated for both the Base Case and the Representative Scenario to be approximately 0.482 g/m²/yr (Section 3.5 of Reference (1)). The potential Project sulfate loading of 0.0484 g/m²/yr for the Representative Scenario is small and only about 10% of existing conditions loading as compared to the Project sulfate load of 0.155 g/m²/yr (32% of existing conditions loading) for the Base Case Scenario.

Table 3-1 Estimated Atmospheric Sulfate Loading from the Project to the Wetland of Interest (WOI) Watershed, Mine Year 13: Base Case Scenario and Representative Scenario

Parameter	Units	Base Case Scenario (loading to watershed; upland and wetland) ^[3]	Representative Scenario (loading to wetland area only) ^[4]
Gases, Aerosol, Fine Particles Stacks/Vents, Mobile ^[1]	g/m ² /yr	0.0053	0.00325
Fugitive Sulfide Mineral Dust - Chalcopyrite ^[2]	g/m ² /yr	0.00284	0.00000053
Fugitive Sulfide Mineral Dust - Pyrrhotite ^[2]	g/m ² /yr	0.147	0.0451
Total Sulfate Load	g/m ² /yr	0.155	0.0484
Background Sulfate Load	g/m ² /yr	0.482	0.482
Project Load as a Percent of Background Load	%	32%	10%

- [1] Potential atmospheric sulfate loading from building stacks/vents and heavy mine equipment (mobile) estimated using the Inference Method.
- [2] Potential atmospheric sulfate loading from fugitive sulfide mineral dust estimated for each sulfide mineral type (chalcopyrite and pyrrhotite). Sulfate loading based on the mass of sulfur released from each sulfide mineral type and the assumption that all released sulfur converts to sulfate.
- [3] For the Base Case Scenario, the release of sulfur was based on the higher chalcopyrite dissolution rate (pH 3 conditions, oxidative dissolution by ferric iron, and temperature of 25°C (77°F) and 100% dissolution of pyrrhotite (see Section 3.2 and Appendix B of Reference (1) for additional details).
Results for the Base Case Scenario are from Table 4-2 of Reference (1).
- [4] For the Representative Scenario, the potential release of sulfur from both chalcopyrite and pyrrhotite was estimated for pH 6 conditions, oxidative dissolution by oxygen, and temperature of 11°C (52°F) (for chalcopyrite, proxy data used to estimate dissolution). Sulfate loading reflects the assumptions described in Section 2.4 of this memorandum.

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3.2 Estimated Effects of Project Atmospheric Sulfate Loading on Methylmercury Concentrations

As shown in Table 3-1, the Base Case Scenario potential Project sulfate load of 0.155 g/m²/yr to the WOI watershed represented about 32% of the existing conditions (background) sulfate load. As described in Section 4.2 of Reference (1), this potential additional sulfate load to the WOI watershed resulted in no measurable change to methylmercury concentrations in surface waters (estimated change in concentration of 0.003 to 0.005 ng/L) (Table 4-5 and 4-6 of Reference (1)). The small potential change in methylmercury surface water concentration estimated for the Base Case Scenario also did not result in any measurable change in fish tissue mercury concentrations (estimated change of 0.014 mg/kg in the Embarrass River and 0.004 mg/kg in the Partridge River (Table 4-7 of Reference (1))).

The estimate of potential Project sulfate loading for the Representative Scenario (0.0484 g/m²/yr) (Table 3-1) represents an even smaller calculated change, about 10% of existing conditions (background) sulfate loading. The potential sulfate load of 0.0484 g/m²/yr estimated for the Representative Scenario is about a factor of 3 less than the sulfate loading estimated for the Base Case Scenario (~69% reduction in loading from the Base Case Scenario).

A potential Project sulfate load of 0.0484 g/m²/yr from the Representative Scenario would have even less of a possible effect on methylmercury concentrations in surface water and fish tissue mercury concentrations compared to the Base Case Scenario. In short, consistent with the Base Case Scenario, the lower sulfate loading for the Representative Scenario would result in no measurable change to methylmercury concentrations in surface water or in fish tissue in water bodies near to the Project or in downstream waters.

3.3 Atmospheric Loading of Arsenic, Copper, and Cobalt from the Project

Section 4.3 of Reference (1) identified that the potential loading of metals was highest in Mine Year 13, with lower results for Mine Year 8. The Base Case Scenario also estimated that more copper would be released when the higher chalcopyrite dissolution rate was used (Section 4.3 of Reference (1)). Therefore, the Mine Year 13 results for the higher chalcopyrite dissolution rate from the Base Case Scenario were the starting point to assess the potential changes associated with the Representative Scenario.

The estimated Project loading of arsenic, copper, and cobalt and wetland water concentrations for the Representative Scenario are provided in Table 3-2. As previously discussed in Section 2.0 of this memorandum, wetlands sequester metals. Therefore, the calculations conducted for the Representative Scenario account for the removal of arsenic, cobalt, and copper from wetland waters due to sequestration within the WOI (Table 3-2), and follow the same methodology as used for the Base Case Scenario.

As shown in Table 3-2, potential Project loading of these three metals for the Representative Scenario is one to two orders of magnitude lower than estimated for the Base Case Scenario. Estimated potential

increases to wetland water concentrations of these metals for the Representative Scenario were also one to two orders of magnitude lower than those estimated for the Base Case Scenario. For the Representative Scenario, the potential incremental changes in surface water concentrations in the WOI for arsenic, cobalt, and copper would not be measurable.

For arsenic, an additional consideration was the potential Project contribution to Colby Lake, downstream of the Project area, where the water quality standard is 2 µg/L because Colby Lake is used as a source of drinking water by the city of Hoyt Lakes. The existing conditions water concentration of arsenic in Colby Lake is approximately 1.0 µg/L based on baseline monitoring data for the Project, which is about 50% of the standard. The potential change in arsenic concentration in the WOI under the Representative Scenario was estimated to be 0.01 µg/L (Table 3-2), which results in no measurable change to the existing conditions (background) concentration of arsenic (~1.5 µg/L in the WOI). Overall, the small potential change in arsenic concentration due to the Project (~0.01 µg/L) would result in no measurable change in the arsenic concentration in the WOI or in Colby Lake, which is consistent with the results for the Base Case Scenario (Section 4.3 and Table 4-9 of Reference (1)).

Table 3-2 Comparison of Potential Release of Metals from Sulfide Mineral Particles for the Base Case and Representative Scenarios for the Wetland of Interest (WOI), Mine Year 13

Type	Parameter	Units	Base Case Scenario ^[1] (watershed basis; wetland and upland)	Representative Scenario ^[2] (wetland only)	Notes
Project in operation	Area	m ²	4.29E+05	8.26E+04	Wetland area, ~20.4 acres (8.3 hectares) Upland area, ~85.7 acres (34.7 hectares)
	Flow	L/yr	1.36E+08	1.36E+08	Flow estimated from hydrologic modeling
Estimated deposition	As	g/yr	49	3.3	Values obtained from air emissions modeling and post-processing
	Co	g/yr	432	26	
	Cu	g/yr	401	0.5	
Estimated deposition on a per-area basis	As	g/m ² /yr	1.15E-04	7.64E-06	Estimated deposition contributing to WOI divided by WOI watershed Area
	Co	g/m ² /yr	1.01E-03	5.98E-05	
	Cu	g/m ² /yr	9.35E-04	1.20E-06	
Estimated concentration increase	As	µg/L	0.36	0.02	Estimated deposition contributing to WOI divided by WOI Flow
	Co	µg/L	3.18	0.19	
	Cu	µg/L	2.95	0.004	
Concentration incremental; accounts for removal in wetland	As	µg/L	0.36	0.01	As: 70% removal (Representative Scenario only) Co: 70% removal Cu: 90% removal
	Co	µg/L	0.95	0.06	
	Cu	µg/L	0.29	0.0004	

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Type	Parameter	Units	Base Case Scenario ^[1] (watershed basis; wetland and upland)	Representative Scenario ^[2] (wetland only)	Notes
Background concentration	As	µg/L	1.5	1.5	Average value at WP-1
	Co	µg/L	1.3	1.3	
	Cu	µg/L	3.6	3.6	
Estimated Project concentration	As	µg/L	1.9	1.5	Background concentration + incremental increase
	Co	µg/L	2.3	1.4	
	Cu	µg/L	3.9	3.6	
Applicable water quality standard	As	µg/L	53	53	Water quality standards for Class 2D waters ^[3]
	Co	µg/L	5	5	
	Cu	µg/L	6.0	6.0	

- [1] For the Base Case Scenario, the estimated metal concentrations are for the higher chalcopyrite dissolution rate (pH 3 conditions, oxidative dissolution by ferric iron, and temperature of 25°C (77°F) and 100% dissolution of pyrrhotite (see Section 4.3 of Reference (1) for additional details).
 Results for the Base Case Scenario are from Table 4-9 of Reference (1).
- [2] For the Representative Scenario, the potential release of metals from sulfide minerals was estimated for pH 6 conditions, oxidative dissolution by oxygen, and temperature of 11°C (52°F) (for pyrrhotite, Section 2.4.2 of this memorandum; proxy data used to estimate chalcopyrite dissolution, Section 2.4.3 of this memorandum).
- [3] The water quality standards are as follows:
 Arsenic: the water quality standard is for the Partridge River, upstream of Colby Lake; 53 µg/L.
 Cobalt: the water quality standard applies to the Partridge River, upstream of Colby Lake; 5 µg/L.
 Copper: the water quality standard is hardness based; the estimated average hardness of water in the WOI, based on surrogate data (Section 3.6.4 of Reference (1)) is 60 mg/L and results in a standard of 6.0 µg/L.

4.0 Summary and Conclusions

Potential atmospheric loading of sulfate, arsenic, copper, and cobalt from the Project to the WOI were evaluated in a Representative Scenario to better reflect the expected environmental conditions that will result from the operation of the Project. The results for the Representative Scenario were compared to the Base Case Scenario that were previously reported in Reference (1) and demonstrate the protectiveness of the Base Case Scenario. In particular, the results of this comparison confirm that the Base Case Scenario, while not showing any Project impacts to water quality or fish tissue mercury concentrations at levels of concern, does significantly overestimate those impacts due to the very conservative assumptions included in the Cross-Media Analysis (Reference (1)).

For the Representative Scenario, the estimated sulfate loading was 0.0484 g/m²/yr to the WOI from Project air emissions, about 10% of background (Table 3-1) and about 69% less atmospheric loading than estimated for the Base Case Scenario (0.155 g/m²/yr). Because the sulfate loading for the Base Case Scenario (0.155 g/m²/yr) did not result in a measurable change in surface water methylmercury concentrations or fish tissue mercury concentrations (Section 6.3 of Reference (1)), the lower sulfate loading for the Representative Scenario (0.0484 g/m²/yr) would likewise not have a measurable effect on those parameters.

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As shown in Table 3-2, the estimated atmospheric loading of metals (arsenic, copper, and cobalt) from the Project to the WOI for the Representative Scenario would be about one to two orders of magnitude lower than estimated for the Base Case Scenario. Table 4-1 identifies that potential incremental changes in arsenic, copper, and cobalt in the WOI for the Representative Scenario would not be measurable; there would be no change from existing conditions.

Arsenic, copper, and cobalt were identified as indicator metals such that the potential effects from other metals can be judged based on the estimated effects from arsenic, copper, and cobalt (Section 2.3.3 of Reference (1)). Because Project atmospheric loadings of arsenic, copper, and cobalt were estimated to result in concentrations that are well below their respective water quality standards in the WOI for both the Base Case Scenario and the Representative Scenario (Table 4-1), the other metals associated with Project emissions will not adversely affect any beneficial uses or result in any violations of water quality standards in the WOI or further downstream.

In summary, under the protective assumptions of the Base Case Scenario, it was concluded that existing uses of surface waters associated with sulfate, methylmercury, and metal concentrations will be maintained and protected at all evaluation points, and that the Project is not expected to cause or contribute to a violation of any water quality criteria in any surface waters (Section 6.3 of Reference (1)). The Representative Scenario demonstrates the protectiveness of the Base Case Scenario (i.e., overestimation of potential effects on water quality), and provides further support for the Base Case Scenario conclusions.

Table 4-1 Summary of Potential Change in Metal Concentrations at the Wetland of Interest (WOI): Base Case Scenario and Representative Scenario

	Units	Arsenic		Cobalt		Copper	
		Base Case Scenario	Representative Scenario	Base Case Scenario	Representative Scenario	Base Case Scenario	Representative Scenario
Background Average Concentration ^[1]	µg/L	1.5		1.3		3.6	
Measurable Change in Concentration ^[2]	µg/L	± 0.22		± 0.19		±0.54	
Potential Concentration Change from Project Atmospheric Loading ^[3]	µg/L	0.36 ^[3]	0.01 ^[4]	0.95 ^[3]	0.06 ^[4]	0.29 ^[3]	0.0004 ^[4]
Potential Concentration During Project Operations	µg/L	1.9	1.5	2.3	1.4	3.9	3.6
Is the Potential Change Measurable?	---	Yes	No	Yes	No	No	No
Is the Potential Post-Startup Concentration Less than the WQ Standard	---	Yes	Yes	Yes	Yes	Yes	Yes
WQ Standard	µg/L	53		5.0		6.0 ^[5]	

- [1] Background (existing conditions) metal concentrations from monitoring location WP-1; data from WP-1 used as surrogate data for the WOI.
- [2] Measurable change based on laboratory control sample acceptance criteria for USEPA Method 200.8 (USEPA; Reference (40)) is ±15% of background concentration.
- [3] For the Base Case Scenario, results are for the higher chalcopyrite dissolution rate (pH 3 conditions, oxidative dissolution by ferric iron, and temperature of 25°C (77°F) and 100% dissolution of pyrrhotite (see Section 4.3 of Reference (1) for additional details). Potential concentration change accounts for removal/retention of cobalt and copper in a wetland environment; 70% removal/retention for cobalt; 90% removal/retention for copper.
- [4] For the Representative Scenario, the potential release of metals from sulfide minerals was estimated for pH 6 conditions, oxidative dissolution by oxygen, and temperature of 11°C (52°F) (for pyrrhotite, Section 2.4.2 of this memorandum; proxy data used to estimate chalcopyrite dissolution, Section 2.4.3 of this memorandum). Potential concentration change accounts for removal/retention of metals in a wetland environment; 70% removal/retention for cobalt and arsenic; 90% removal/retention for copper.
- [5] The water quality standard for copper is hardness based; the estimated average hardness of water in the WOI, based on surrogate data (Section 3.6.4 of Reference (1)) is 60 mg/L and results in a standard of 6.0 µg/L.

5.0 References

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