6. TOXICITY ASSESSMENT

The toxicity assessment weighs the available and relevant evidence regarding the potential for constituents to cause adverse health effects in exposed individuals and provides a quantitative estimate of the relationship between the magnitude of exposure and the likelihood of adverse effects (USEPA 1989). A fundamental principle of toxicology is that the dose determines the severity of the effect. Accordingly, the toxicity criteria describe the quantitative relationship between the dose of a constituent and the type and incidence of the toxic effect. This relationship is referred to as the dose-response. Exposure to constituents can result in cancer or non-cancer effects, which are characterized separately. Essential dose-response criteria are the slope factor (SF) and unit risk factor (URF) used for assessing cancer risks, and the reference dose (RfD) and reference concentration (RfC) used for evaluating non-cancer effects.

The RfD is defined as the amount of COPC per unit body weight that can be taken into the body each day (e.g., mg/kg BW/day) without risk of adverse health effects. The RfC is defined as the amount of COPC (in milligrams per cubic meter) that can be taken into the body each day without risk of adverse healtheffects. A SF is the upper-bound estimate of the probability of a response-per-unit intake of material of concern over an average human lifetime. It is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime exposure to a particular level of arsenic. Upper-bound estimates conservatively exaggerate the risk to ensure that the risk is not underestimated if the underlying model is incorrect.

The OSWER hierarchy (USEPA 2003) was used to select toxicity values for the HHRA, which prioritizes toxicity values from the USEPA (2017) Integrated Risk Information System (IRIS) database.

Toxicity information is typically derived from laboratory studies, where dose-response information is extrapolated from animal test subjects to humans by applying uncertainty or safety factors. In most cases, uncertainty factors of 100 to 1,000 are applied to the laboratory-derived NOAELs. A NOAEL is the highest concentration used in a toxicity test that results in no observed or measured chronic health effects. These uncertainty factors account for interspecies extrapolation and the protection of the most susceptible individuals in the population (i.e., children and the elderly). Therefore, toxicity values based on animal studies generally have large margins of safety to ensure that the toxicity or risk of a substance to people is not underestimated.

The toxicity values used in the HHRA are presented in Table 6.1-1.

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TABLE 6.1-1: TOXICITY VALUES FOR CONSTITUENTS OF POTENTIAL CONCERN

COPC	Oral RfD (mg/kg BW/day)	Inhalation RfC (µg/m³)	Cancer Slope Factor (mg/kg BW/day)$^{-1}$	Inhalation Unit Risk Factor (µg/m³)$^{-1}$	Reference
Antimony	0.0004	0.2	NA	NA	USEPA (2017) (oral); ATSDR (1992) (inhalation) [a]
Arsenic	0.0003	0.015	1.5	0.0043	USEPA (2017)
Mercury	0.0003	0.3	NA	NA	USEPA (2017)
MeHg	0.0001	-	NA	NA	USEPA (2017)

COPC = constituent of potential concern

RfD = reference dose

RfC = reference concentration

BW = body weight

NA = not applicable; not classified as a human carcinogen by USEPA.

(-) = not available

[a] *The USEPA has not derived an inhalation RfC for antimony.*

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7. RISK CHARACTERIZATION

Risk characterization integrates the exposure assessment and the toxicity information. Non-cancer and cancer health risks were calculated for each COPC and for each media and potentially complete and significant exposure pathway. Non-cancer hazard quotients (HQs) represent the ratio of the estimated dose from exposure to a constituent in a particular medium to the dose that is not expected to result in adverse health effects, other than cancer. Cancer risk estimates represent the potential for cancer effects by estimating the probability of developing cancer over a lifetime due to site exposures (e.g., a risk of 1×10^{-6} indicates a one in 1 million chance of developing cancer due to exposures at the site). Equations for developing non-cancer and cancer risk estimates are described below, followed by a results and discussion section.

7.1. Non-carcinogenic Hazard Equations

The potential for adverse health effects other than carcinogenic effects (i.e., non-carcinogenic effects) is characterized by dividing estimated constituent doses (ADDs) or exposure concentrations (ECs) in air, by constituent-specific RfDs (for ADDs) or RfCs (for ECs). The resulting ratio is the HQ, which is derived as follows (USEPA 1992a):

$$HQ = \frac{ADD \text{ or } EC}{RfD \text{ or } RfC} \qquad \text{[Equation 7]}$$

where:

ADD = average daily dose (mg/kg BW/day)
EC = exposure concentration (µg/m^3)
RfD = reference dose (mg/kg BW/day)$^{-1}$
RfC = reference concentration (µg/m^3)$^{-1}$

A toxic effect can only occur if the metal is absorbed into the bloodstream. Not all constituents that are ingested or inhaled are subsequently absorbed. The fraction of constituents that are absorbed into the bloodstream by a receptor is termed the "bioavailable" fraction. USEPA (2007b, 2012) provided guidance on incorporating bioavailability estimates of metals in soil that might be ingested and subsequently absorbed via the gastrointestinal tract. The equation is:

$$HQ = \frac{ADD \times RBA}{RfD} \qquad \text{[Equation 8]}$$

where:

ADD = average daily dose (mg/kg BW/day)
RBA = relative bioavailability from the gastrointestinal tract (unitless)
RfD = reference dose (mg/kg BW/day)$^{-1}$

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Bioavailability was not specifically measured for samples in and around the proposed Project area. However, based on extensive literature review, USEPA (2012) has developed a recommended soil arsenic relative bioavailability factor of 0.6, a conservative, upper-end mean of measured relative bioavailability factors for different arsenic sources in soil. For antimony, mercury, and MeHg, bioavailability was assumed to be 100% (or a factor of 1.0), as no guidance is available for these constituents.

For a risk assessment that considers all potential exposure sources, doses and risk, USEPA's (1989) risk assessment guidelines consider the additive effects associated with simultaneous exposure to all constituents by summing HQs across exposure pathways and constituents to estimate the total hazard index (HI). This summation conservatively assumes that the toxic effects of all constituents would be additive or, in other words, that all constituents cause the same toxic effect and act by the same mechanism. For non-carcinogens, summing HQ values across different chemicals is properly applied only to compounds that induce the same effect by the same mechanism of action. Consequently, summation of HQ values for compounds that are not expected to include the same type of effects or that do not act by the same mechanisms could overestimate the potential for effects. Thus, all of the total HI values in this report, which sum HQ values across multiple metals, are likely to overestimate the true level of human health non-cancer risk.

7.2. Carcinogenic Risk Equations

The potential for carcinogenic effects is evaluated by estimating the probability of developing cancer over a lifetime based on exposure assumptions and constituent-specific toxicity criteria. The increased likelihood of developing cancer from exposure to a particular constituent is defined as the excess cancer risk (CR). Excess CR is the risk in excess of a background CR for subsistence residents.

Of the COPCs evaluated in this HHRA, arsenic (in the inorganic form) was recognized as a potential human carcinogen. CR estimates for ingestion exposures are the product of exposure assumptions (i.e., intake) and the constituent-specific SF, while CR estimates for inhalation exposures are the product of ECs and URFs. Excess lifetime CRs were estimated by multiplying the ingestion LADD or the inhalation EC by the cancer SF or URF, as follows:

$$\mathrm{CR} = \mathrm{LADD\ or\ EC} \times \mathrm{SF\ or\ URF} \qquad \text{[Equation 9]}$$

where:

LADD = lifetime average daily dose (mg/kg BW/day)
EC = exposure concentration (µg/m^3)
SF = cancer slope factor (mg/kg BW/day)$^{-1}$
URF = cancer unit risk factor (µg/m^3)$^{-1}$

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7.3. Risk Characterization Results and General Discussion

USEPA (1991) provides guidance on the role of the HHRA in supporting risk management decisions, and in particular, evaluating whether a response action is necessary. Though this guidance was originally developed for Superfund remedial investigations, it provides the frame of reference used by regulatory agencies to evaluate potential health effects of environmental impacts. An HQ or HI less than 1 indicates a noncancer effect is highly unlikely. For cancer risk, the "target range" is defined as "an excess upper-bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6}," or 1 in 10,000 to 1 in 1,000,000.

Additionally, the focus and purpose of this HHRA was to evaluate the potential incremental risk resulting from the proposed Project; therefore, the most relevant comparison was the incremental change in HQ, HI, and cancer probability estimates. For purposes of the HHRA, incremental changes in risk were considered substantial if:

- Non-cancer risks (i.e., HQs or HIs) changed from less than 1 to greater than 1 (based on USEPA (1989) guidance, HQs less than 1 indicate a an effect is very unlikely); or
- Cancer estimates increased in the risk assessment by 10-fold or more (for example, if baseline cancer risks are $1x10^{-6}$ and future cancer risks are $1x10^{-5}$ or greater).

Non-cancer and cancer risk estimates are shown in Tables 7.3-1 through 7.3-5.

As shown, even when incorporating the conservative assumptions of the HHRA, HQs and HIs were equal to or less than 1, indicating that noncancer effects are highly unlikely. The conservative assumptions were incorporated into this HHRA to prevent Type II errors, which are the elimination of a constituent, area, or activity from further consideration when, in fact, there should be a concern (i.e., false-negative conclusion). In the risk assessment process, uncertainties are handled conservatively (i.e., health-protective) estimates of exposure. This strategy is more likely to produce false-positive errors than false-negative errors. Particularly for this HHRA, a number of conservative estimates were made to assure that risk predictions erred on the side of Type I errors (i.e., false-positive conclusion) rather than Type II.

Non-cancer risks related to both THg and MeHg were low, with all HQs and HIs less than 1. Baseline risk estimates are consistent with a mercury hair study completed by the ADHSS. The ADHSS conducted MeHg testing of hair samples from pregnant women in selected communities including the Upper Kuskokwim River region communities, between May and September 2012. A total of 186 hair samples were obtained from eight Upper Kuskokwim River communities. Every study participant had a hair mercury level that was below the ATSDR NOAEL (15.3 ppm), as well as the Environmental Public Health Program cut-off for follow-up (5 ppm) (ADHSS 2013, as cited in the DEIS).

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In the case of cancer risk estimates for arsenic, both baseline and future risk estimates ranged between $2x10^{-5}$ and $5x10^{-5}$, which is within the USEPA risk management range of $1x10^{-4}$ to $1x10^{-6}$. Future estimates of cancer risk are estimated to increase from baseline results by less than $1x10^{-6}$, which is below the USEPA risk management range, and is considered a negligible incremental increase. Cancer SFs were used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to arsenic. Upper-bound estimates conservatively exaggerate the risk to ensure that the risk is not underestimated if the underlying model is incorrect.

Arsenic is a naturally occurring metal often found at elevated concentrations throughout Alaska, as described in Title 18 of the Alaska Administrative Code, Chapter 75.341(c). Baseline concentrations of arsenic are consistent with regional, state and nationally published studies of naturally occurring arsenic in soil (Gough et al. 1988). Due to the prevalence of naturally occurring arsenic in Alaskan soil, ADEC recommends that cumulative risk calculations do not include risk contributions from naturally occurring arsenic sources (ADEC 2009). Statistical comparisons of significance between baseline and future arsenic concentrations cannot be made because only a single future soil arsenic concentration was estimated. However, if baseline cancer risk estimates are subtracted from future cancer risk estimates, the resulting excess cancer risk for adults due to Project-related exposure would be $5x10^{-7}$, well below acceptable risk targets.

Other uncertainties related to the arsenic ingestion SF, as discussed in Section 8, likely result in an overprediction of potential cancer risk. Given that the overestimated potential current and future risks are within the risk management range and are related to naturally occurring levels of arsenic, unacceptable cancer risk due to arsenic exposure is unlikely.

7.4. Risks due to Fish Consumption

This HHRA evaluated risks due to northern pike fish consumption. EPCs were developed based on this resident fish species to estimate the incremental impact of proposed Project-specific COPCs. Resident fish were incorporated into the HHRA because of their relatively greater exposure period compared to migratory fish. While migratory fish may be exposed to proposed Project-related constituents while present in the study area, the exposure duration would be much less than for resident fish. As a result, migratory fish originating from the study area that are then harvested as returning adults pose a lower exposure of COPCs from the study area to consumers than harvesting resident fish. Further discussion on potential risks of salmon species are discussed in the next section.

Northern pike, a large, relatively long-lived top predator that is found in the Crooked Creek watershed, was used to estimate fish EPCs and assess risks associated with subsistence fishing. Top predators, such as northern pike, also bioaccumulate mercury to a greater degree than lower-trophic level species. Tissue burdens for northern pike were

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estimated by applying trophic transfer factors to sculpin data collected for the proposed Project. Risk estimates for northern pike indicated that both baseline and future HQs are at or less than 1, indicating that non-cancer effects are unlikely from northern pike consumption.

Estimates of future northern pike concentrations were also compared to the State of Alaska (2016) fish consumption advisory for northern pike that would be harvested from the Kuskokwim River in the vicinity of the proposed Project. The mean northern pike tissue mercury concentrations from different water bodies identified in the advisory range from 0.11 to 0.61 mg/kg ww. The mercury concentration in the one adult northern pike tissue sample collected within the study area falls within this range (0.421 mg/kg ww). When mean mercury concentrations in northern pike range from >0.34 to 0.46 mg/kg ww, the State of Alaska (2016) recommends limiting fish consumption to 0.068 kg/day. This consumption rate is higher than the estimated adult consumption rate for northern pike that was used in the HHRA (0.0091 kg/day), which is the average consumption rate of northern pike for eight communities in the Central Kuskokwim River Drainage (Brown et al. 2012). Therefore, subsistence populations currently consume northern pike at rates well below the northern pike consumption advisory level for the State of Alaska, and consumption would be expected to remain below the advisory level in the future.

As reflected in the DEIS comments, there is public concern regarding constituents in salmon species, which are a primary subsistence food in the region. Salmon species are present within the study area, including coho salmon. However, within the study area, only juvenile coho (and other species of) salmon reside within the Crooked Creek drainage system for extended periods. Adults, rather than juvenile species, are harvested for consumption. Juveniles migrate out of the freshwater system at an early age, spending the majority of their lifetimes in marine environments. These migratory fish gain the bulk of their body mass over a much larger exposure area (marine waters) that is entirely outside the study area. Because the exposure conditions of salmon are largely unrelated to the study area, the HHRA did not quantify risk due to salmon ingestion.

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TABLE 7.3-1. BASELINE NON-CANCER RISK CHARACTERIZATION FOR ADULTS IN SUBSISTENCE POPULATIONS

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.01	0.03	0.003	0.002	0.02	NA	0.07
Arsenic	0.03	0.01	0.001	0.01	0.1	NA	0.2
Mercury	0.0007	NA	NA	NA	NA	0.00003	0.0007
MeHg	0.000008	0.006	0.0005	0.09	0.005	NA	0.1
Final HI							**0.3**

COPC = constituent of potential concern

HQ = hazard quotient

HI = hazard index

NA = Not applicable (mercury in biota assumed to be 100% MeHg; antimony and arsenic are negligible in baseline air; MeHg is not present in air)

TABLE 7.3-2. BASELINE NON-CANCER RISK CHARACTERIZATION FOR CHILDREN IN SUBSISTENCE POPULATIONS

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.1	0.1	0.01	0.006	0.09	NA	0.3
Arsenic	0.3	0.03	0.002	0.05	0.4	NA	0.8
Mercury	0.007	NA	NA	NA	NA	0.00003	0.007
MeHg	0.00009	0.02	0.002	0.3	0.02	NA	0.4
Final HI							**1**

COPC = constituent of potential concern

HQ = hazard quotient

HI = hazard index

NA = Not applicable (mercury in biota assumed to be 100% MeHg; antimony and arsenic are negligible in baseline air; MeHg is not present in air)

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TABLE 7.3-3. FUTURE NON-CANCER RISK CHARACTERIZATION FOR ADULTS IN SUBSISTENCE POPULATIONS

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.01	0.03	0.003	0.002	0.03	0.000003	0.07
Arsenic	0.03	0.01	0.001	0.01	0.1	0.001	0.2
Mercury	0.0007	NA	NA	NA	NA	0.000032	0.0007
MeHg	0.000008	0.006	0.0006	0.1	0.005	NA	0.1
Final HI							**0.3**

COPC = constituent of potential concern

HQ = hazard quotient

HI = hazard index

NA = Not applicable (mercury in biota assumed to be 100% MeHg; MeHg is not present in air)

TABLE 7.3-4. FUTURE NON-CANCER RISK CHARACTERIZATION FOR CHILDREN IN SUBSISTENCE POPULATIONS

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.1	0.1	0.01	0.006	0.09	0.000003	0.3
Arsenic	0.3	0.03	0.002	0.05	0.4	0.001	0.8
Mercury	0.007	NA	NA	NA	NA	0.000032	0.007
MeHg	0.00009	0.02	0.002	0.3	0.02	NA	0.4
Final HI							**1**

COPC = constituent of potential concern

HQ = hazard quotient

HI = hazard index

NA = Not applicable (mercury in biota assumed to be 100% MeHg; MeHg is not present in air)

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TABLE 7.3-5. BASELINE AND FUTURE CANCER RISK CHARACTERIZATION

COPC	Oral - Soil CR	Oral - Beaver CR	Oral - Mallard Duck CR	Oral - Pike CR	Oral - Berries CR	Inhalation CR	Total CR
Baseline Arsenic - Adults	3.8E-06	1.2E-06	7.6E-08	2.0E-06	1.4E-05	NA	**2.E-05**
Baseline Arsenic - Children	1.1E-05	1.2E-06	7.6E-08	2.0E-06	1.4E-05	NA	**3.E-05**
Total Baseline Cancer Risk	1.5E-05	2.3E-06	1.5E-07	4.0E-06	2.9E-05	NA	**5.E-05**
Future Arsenic - Adults	3.8E-06	1.2E-06	7.7E-08	2.0E-06	1.5E-05	2.3E-08	**2.E-05**
Future Arsenic - Children	1.2E-05	1.2E-06	7.7E-08	2.0E-06	1.5E-05	6.6E-09	**3.E-05**
Total Future Cancer Risk	1.5E-05	2.4E-06	1.5E-07	4.0E-06	2.9E-05	3.0E-08	**5.E-05**

NA = Not applicable (arsenic concentrations negligible in air)

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8. UNCERTAINTY ANALYSIS

8.1. General Discussion

Estimating and evaluating health risks from exposure to environmental constituents is a complex process with inherent uncertainties. Uncertainty reflects limitations in knowledge, and simplifying assumptions must be made to quantify health risks. The uncertainty analysis provides decision makers with a broader understanding of the basis of the risk assessment, and how changes in key assumptions can change risk assessment results. Many uncertainties in selection of HHRA input parameters are identified throughout this report. This section summarizes several key areas of uncertainty in the analysis.

The process of evaluating human health risks from exposure to environmental media involves multiple steps, each containing inherent uncertainties that ultimately affect the final risk estimates. These uncertainties may be related to the collection of samples, laboratory analysis, estimation of potential exposures, and derivation of toxicity values, resulting in either an over- or under-estimation of risk. However, for the present assessment, where uncertainties existed, assumptions made lead to overestimating potential risks.

Uncertainty in the risk assessment produces the potential for two kinds of errors. A Type I error is the identification of a specific constituent, area, or activity as a health concern when, in fact, it is not a concern (i.e., false-positive conclusion). A Type II error is the elimination of a constituent, area, or activity from further consideration when, in fact, there should be a concern (i.e., false-negative conclusion). In the risk assessment process, uncertainties are handled conservatively (i.e., health-protective choices are preferentially made). This strategy is more likely to produce false-positive errors than false-negative errors and was applied in this HHRA.

8.1.1. Uncertainties in Media Concentrations

Primary media data collected in the study area were used to estimate baseline EPCs in soil and air, and in the development of BAFs for biota. Future EPCs rely on models to estimate future concentrations using certain assumptions that have some uncertainty.

In baseline estimates of EPCs, samples are assumed to be representative of concentrations found in the environment that receptors may be exposed to. Uncertainties are always present in the representativeness of samples. Sample sizes for soil, surface water, and sediment were considered to be adequate to represent conditions in the study area. This HHRA used only data that gave accurate chemical concentrations and had associated laboratory reports with which to verify sample qualifiers and reported detection limits. The datasets collected for the proposed Project included

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method detection limits and sample quantitation limits below ADEC and/or USEPA criteria for their media, and included explanations for qualified data.

Where possible, site-specific baseline data were compared to other regional data, and as noted throughout this report, baseline concentrations were generally consistent with other regional studies.

Uncertainties exist in estimating future EPCs. As noted throughout this report, uncertainties are associated with mechanisms of future fate and transport, including deposition onto surfaces, uptake into biota, and transformations in soil and aquatic systems. Simple, conservative models were preferred over complex models, both due to the volume of data that would be required in the latter (thus generating new uncertainties in data representativeness and robustness), as well as the lack of better predictive power demonstrated by the more complex models. In the case of mercury, key conservative approaches taken to estimate future EPCs in soil and biota included: estimating future soil/sediment EPCs based on an upper bound methylation rate of 1% (lower rates were calculated by ProUCL, 0.3 to 0.5%); estimating biota EPCs based on predicted methylmercury content in primary media; and use of the same factor for increases associated with sulfate deposition and the methylation rate.

Uncertainties are related to BAFs used to model biota concentrations, including the use of median concentrations to develop BAFs. It was also conservatively assumed that mercury in biota was in the form of MeHg (rather than THg, which has a lower toxicity and bioavailability), even though there are varying proportions reported in the literature for various species of plants and wildlife.

In estimating fish EPCs, using data for resident fish was preferred, because of their greater exposure period compared to migratory fish. For this reason, northern pike was preferred to estimate fish EPCs, because it is a resident, large, relatively long-lived top predator. However, the limited dataset for northern pike resulted in uncertainties in estimating baseline and future tissue concentrations. Because of this, available data for other residential fish species was used. As noted in Section 5.2.2, the sculpin dataset was appropriately large to develop BAFs. Using a method consistent with other regional studies (Ecology and Environment 2014), sculpin BAFs were derived using the project-specific dataset. Northern pike concentrations were then estimated using the sculpin-based BAF and applying a food chain multiplier (FCM) of 3. Although there is some uncertainty in this method, use of this FCM is accepted to account for biomagnification between sculpin and northern pike, as northern pike is one trophic level higher than sculpin (McGeer 2004).

In estimating future air exposures, several assumptions were made that likely overestimate the $PM_{2.5}$ concentrations of COPCs: The air quality model simulated impacts from the mine during the projected peak emission year (mine year 16); the receptor location in Crooked Creek with the highest average $PM_{2.5}$ concentration was

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used to estimate COPCs in the $PM_{2.5}$ material originating from the proposed Project; it was assumed that all exposure throughout the mine life was at these concentrations.

8.1.2. Uncertainties with Subsistence Population Assumptions

The HHRA evaluated subsistence populations because this receptor group is thought to have the highest exposure potential to COPCs. Subsistence population intake rates of locally caught food items are typically higher than the general population (USEPA 2011). These populations are also exposed via direct contact with abiotic media (e.g., incidental ingestion of soil).

For the purposes of the HHRA, a generalized subsistence group was developed to quantify risks. This generalized subsistence group is assumed to live in the study area (therefore evaluated for year-round inhalation and soil ingestion risks), and to hunt, fish, and gather only within the study area. Because not all individual species or wildlife trophic components of an ecological system are practical or potentially necessary in order to quantify incremental risk, representative species were chosen, which includes a level of uncertainty with respect to whether all substantial risks were identified and quantified.

Based on review of the subsistence food items harvested by the Upper Kuskokwim group (see Section 3.21 in the DEIS), and accounting for physiological considerations and residency in the study area, the HHRA quantifies the incremental risk of subsistence population consumption of items thought to best represent types of foods that could be frequently harvested within the study area.

The comprehensive study by Brown et al. (2012) was used to develop area-specific subsistence intake estimates, based on Upper Kuskokwim harvest patterns and rates. However, the study reported only harvest rates, rather than consumption rates, and it was conservatively assumed that all harvested food was consumed. Additionally, individual harvest patterns and annual variability were not considered for the HHRA; instead, a very conservative assumption was incorporated that all harvested items quantified in the HHRA would be harvested within the study area at the same annual rate. Because these groups also hunt, fish, and gather outside the study area to varying extents, these assumptions overestimate the exposure frequency and duration in the study area. However, by utilizing the harvested amount to estimate intake, this HHRA provides a "health-protective" estimate of potential exposures from this pathway: if no incremental risk is indicated by this approach, then groups ingesting only a portion of harvested foods are unlikely to be at risk.

Only two subsistence populations live within the study area: the village of Crooked Creek, and Georgetown. Inhalation risk was specifically quantified for air concentrations predicted for the Crooked Creek watershed, which includes the Crooked Creek village

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area. This village is within the watershed predicted to have the highest incremental increases in COPC concentrations related to the proposed Project. This assumption will over-predict exposures by other subsistence groups, but addresses populations with the highest potential exposure.

Other exposure rates, body weights, and the proportion of subsistence foods consumed by children as a function of adult intake, were based on USEPA and ADEC guidance. These parameters are based on national studies that may not be representative of subsistence populations specifically for the Upper Kuskokwim region.

8.1.3. Uncertainties in Toxicity Values

There is uncertainty associated with estimating toxicity values by extrapolating potential effects on humans from animal studies in the laboratory. In developing toxicity values, it is standard practice to assume that people are more sensitive to the toxic effects of a substance than laboratory animals. Therefore, the toxicity benchmarks for human health are set at much lower levels than the animal benchmarks (by applying a safety factor of 100 to 1,000 or more). This large margin ensures that doses less than the toxicity value are safe and that minor exceedances of these benchmarks are unlikely to cause adverse health effects.

8.1.3.1. Uncertainties in Evaluating Effects of Multiple Constituents

Toxicity values are derived for individual constituents. However, it is recognized that multiple constituents may be present within a food item and interactions between compounds may result in additivity (overall effect is the sum of the individual effects), antagonism (overall effect less than the sum of the individual effects), synergism (overall effect is greater than the sum of the individual effects), or potentiation (presence of one constituent results in toxicity of another constituent that otherwise would have been safe). Many of these interactions are poorly understood or remain unknown by modern science. Furthermore, in natural systems numerous physical variables (e.g., media temperature, pH, salinity, hardness) can accelerate or impede these constituent interactions. In most cases, available toxicity data are not sufficient to define what type of interaction is expected, so USEPA generally assumes effects are additive for non-carcinogens that act on the same target tissue and for carcinogens (all target tissues). Because documented cases of synergistic interactions between chemicals are relatively uncommon, this approach is likely to be conservative for most chemicals.

8.1.3.2. Uncertainties in Arsenic Cancer Toxicity Values

Cancer SFs were used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to arsenic through inhalation and ingestion. Upper-bound estimates conservatively overestimate the risk to ensure that the risk is not underestimated if the underlying model is incorrect. The arsenic ingestion slope factor is based on one affected population in Taiwan concerning non-fatal skin

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cancer incidence, age, and level of exposure to arsenic via drinking water rather than food or soil ingestion (USEPA 2017). The predominant form of arsenic in the drinking water was in inorganic form (arsenic trioxide), which is a highly toxic form of arsenic and not the predominant form of arsenic that occurs in the study area. The confidence in the oral slope factor is considered to be low overall. Studies that show the strongest link between ingestion of arsenic and cancer involve ingestion of inorganic arsenic at elevated levels in drinking water (Naujokas et al. 2013). Arsenic in soil is less bioavailable, and an adjustment is made to the soil exposure concentration accordingly. There are additional uncertainties estimating cancer risk from arsenic because the mechanism of action in causing human cancers is not known, and studies on arsenic mutagenicity are inconclusive (USEPA 2017). However, safety factors included in the slope factor provide a conservative estimate of risk.

8.1.4. Uncertainties in Risk Estimates

A number of uncertainties are present in the risk characterization approach for non-cancer and cancer health effects.

First, because risk estimates for a chemical are derived by combining uncertain estimates of exposure and toxicity (see above), the risk estimates for each chemical are more uncertain than either the exposure estimate or the toxicity estimate alone. However, even if the risk estimates for individual chemicals were quite certain, there is considerable uncertainty in how to combine risk estimates across different chemicals. As noted above, the effects caused by one chemical may or may not influence the effects caused by other chemicals. In some cases, the effects of one chemical may interact with effects of other chemicals, causing responses that are approximately additive, greater than additive (synergistic), or less than additive (antagonistic).

For non-carcinogens, summing HQ values across different chemicals is properly applied only to compounds that induce the same effect by the same mechanism of action. Consequently, summation of HQ values for compounds that are not expected to include the same type of effects or that do not act by the same mechanisms could overestimate the potential for effects. Thus, all the HI values in this report, which sum HQ values across multiple metals, are likely to overestimate the true level of human health non-cancer risk.

8.2. **Sensitivity Analysis**

To investigate the sensitivity of the risk calculations to input parameters, the following sensitivity analyses were conducted:

1. Modification of ingestion rates: the soil and subsistence food ingestion rates were decreased by 25% and 50% from those listed in Table 5.5-1 for both baseline and future scenarios.
2. Modification of EPCs: the soil, sediment, and air EPCs were increased by 25% and decreased by 25% from those listed in Tables 5.3-1 and 5.4-1. Changes in soil

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and sediment EPCs are reflected in biota EPCs modeled using these concentrations and BAFs.

8.2.1. *Modification of Ingestion Rates*

To develop biota ingestion rates, it was conservatively assumed that the entire harvest amounts reported in Brown et al. (2012) were consumed, and that the food items were harvested only within the study area; however, it is likely that both assumptions and, therefore, the consumption rates are overestimates. The consumption rate was amortized over an entire year to determine the daily ingestion rate. For the purposes of the analysis, only the HQs for were compared. Risk results for the child subsistence user (the more sensitive receptor) under baseline and future scenarios using these lower ingestion rates are shown in Tables 8.2-1 and 8.2-2.

The effect of reducing all food and soil ingestion rates by 25% (meaning 25% of ingestion occurs outside the study area) generally resulted in only a slight decrease in child subsistence HQ values and a similar overall HI. If it is assumed that 50% of ingestion occurs is associated with outside the study area, the overall HQ drops below 1.0. The effect on individual exposure pathway HQs depends on the constituent being evaluated and is shown in the tables.

TABLE 8.2-1. BASELINE RISK CHARACTERIZATION, CHILD SUBSISTENCE, DECREASED INGESTION RATES

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.12	0.1	0.01	0.006	0.09	NA	0.3
Arsenic	0.3	0.03	0.002	0.05	0.4	NA	0.8
Mercury	0.007	NA	NA	NA	NA	0.00003	0.007
MeHg	0.00009	0.02	0.002	0.3	0.02	NA	0.4
Final HI							**1**
							IR 25% lower
Antimony	0.09	0.10	0.009	0.005	0.07	NA	0.3
Arsenic	0.2	0.03	0.002	0.04	0.3	NA	0.6
Mercury	0.005	NA	NA	NA	NA	0.00003	0.005
MeHg	0.00006	0.02	0.002	0.3	0.014	NA	0.3
Final HI							**1**
							IR 50% lower
Antimony	0.08	0.07	0.006	0.004	0.06	NA	0.2
Arsenic	0.2	0.02	0.001	0.03	0.2	NA	0.5
Mercury	0.005	NA	NA	NA	NA	0.00003	0.005
MeHg	0.00006	0.01	0.001	0.2	0.011	NA	0.2
Final HI							**0.9**

COPC = constituent of potential concern
HQ = hazard quotient; HI = hazard index
IR = ingestion rate
NA = Not applicable

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TABLE 8.2-2. FUTURE RISK CHARACTERIZATION, CHILD SUBSISTENCE, DECREASED INGESTION RATES

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.1	0.1	0.01	0.006	0.09	0.000003	0.3
Arsenic	0.3	0.03	0.002	0.05	0.4	0.001	0.8
Mercury	0.007	NA	NA	NA	NA	0.0000282	0.007
MeHg	0.00009	0.02	0.002	0.3	0.02	NA	0.4
Final HI							**1**
							IR 25% lower
Antimony	0.1	0.1	0.01	0.005	0.07	0.000003	0.3
Arsenic	0.2	0.02	0.001	0.04	0.3	0.001	0.6
Mercury	0.005	NA	NA	NA	NA	0.0000282	0.005
MeHg	0.00007	0.02	0.001	0.3	0.01	NA	0.3
Final HI							**1**
							IR 50% lower
Antimony	0.1	0.1	0.01	0.003	0.04	0.000003	0.2
Arsenic	0.2	0.02	0.001	0.03	0.2	0.001	0.4
Mercury	0.004	NA	NA	NA	NA	0.0000282	0.004
MeHg	0.00005	0.01	0.001	0.2	0.01	NA	0.2
Final HI							**0.7**

COPC = constituent of potential concern
HQ = hazard quotient; HI = hazard index
IR = ingestion rate
NA = Not applicable

8.2.2. Modification of Exposure Point Concentrations

Determining EPCs, particularly for a future condition that does not yet exist, inherently contains uncertainties because it relies on models rather than empirical data. A number of conservative assumptions were included in this HHRA to estimate future concentrations. A sensitivity analysis was run where future concentrations of soil and sediment EPCs were increased and decreased by 25% to assess the effect on future risk calculations. Increasing or decreasing those primary media EPCs also affects estimated biota concentrations, which are derived from soil and sediment EPCs and BAFs. Table 8.2-3 compares the estimated baseline HQs for children when the baseline EPCs in Table 5.4-1 are increased and decreased by 25%, and Table 8.2-4 provides the same comparison for the future scenario. As shown, risk calculations are not very sensitive to changes in EPCs, and this indicates that if the models used to predict future EPCs are inaccurate by a large margin, the results do not change significantly.

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TABLE 8.2-3. BASELINE RISK CHARACTERIZATION, CHILD SUBSISTENCE, CHANGED EPCS

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.12	0.1	0.01	0.006	0.09	NA	0.3
Arsenic	0.3	0.03	0.002	0.05	0.4	NA	0.8
Mercury	0.007	NA	NA	NA	NA	0.00003	0.007
MeHg	0.00009	0.02	0.002	0.3	0.02	NA	0.4
Final HI							**1**
							EPC 25% lower
Antimony	0.09	0.10	0.009	0.005	0.07	NA	0.3
Arsenic	0.2	0.03	0.002	0.04	0.3	NA	0.6
Mercury	0.005	NA	NA	NA	NA	0.00003	0.005
MeHg	0.00006	0.02	0.002	0.3	0.014	NA	0.3
Final HI							**1**
							EPC 25% higher
Antimony	0.15	0.14	0.012	0.008	0.11	NA	0.4
Arsenic	0.4	0.04	0.002	0.06	0.5	NA	0.9
Mercury	0.009	NA	NA	NA	NA	0.00004	0.009
MeHg	0.00011	0.03	0.002	0.4	0.020	NA	0.5
Final HI							**2**

COPC = constituent of potential concern
EPC = exposure point concentration
HQ = hazard quotient; HI = hazard index
NA = Not applicable

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TABLE 8.2-4. FUTURE RISK CHARACTERIZATION, CHILD SUBSISTENCE, CHANGED EPCS

COPC	Oral - Soil HQ	Oral - Beaver HQ	Oral - Mallard Duck HQ	Oral - Pike HQ	Oral - Berries HQ	Inhalation HQ	Total HI
Antimony	0.12	0.1	0.01	0.006	0.09	0.000003	0.3
Arsenic	0.3	0.03	0.002	0.05	0.4	0.001	0.8
Mercury	0.007	NA	NA	NA	NA	0.0000282	0.007
MeHg	0.00009	0.02	0.002	0.3	0.02	NA	0.4
Final HI							**1**
							EPC 25% lower
Antimony	0.09	0.08	0.008	0.004	0.07	0.000002	0.3
Arsenic	0.2	0.02	0.002	0.04	0.3	0.0009	0.6
Mercury	0.005	NA	NA	NA	NA	0.0000281	0.005
MeHg	0.00007	0.02	0.001	0.3	0.01	NA	0.3
Final HI							**1**
							EPC 25% higher
Antimony	0.15	0.14	0.013	0.004	0.11	0.000004	0.4
Arsenic	0.4	0.04	0.002	0.04	0.5	0.0015	0.9
Mercury	0.009	NA	NA	NA	NA	0.0000282	0.009
MeHg	0.00011	0.03	0.002	0.4	0.02	NA	0.5
Final HI							**2**

COPC = constituent of potential concern
EPC = exposure point concentration
HQ = hazard quotient; HI = hazard index
NA = Not applicable

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9. SUMMARY CONCLUSION

This HHRA evaluated the likelihood of adverse effects to subsistence populations that could be exposed to constituents released into the environment by the operation of the proposed Donlin Gold Project. Risks from antimony, arsenic, and mercury were quantified in the HHRA. A health-protective approach was taken to quantify exposure to these constituents. Highly conservative exposure assumptions were used to calculate potential incremental risk to subsistence populations that may be hunting, gathering, or living in the area around the proposed Project.

This HHRA demonstrates that the small increases in constituent concentrations estimated to occur outside of the Core Operating Area due to Project-related activities will not result in unacceptable risks to human populations who would have the highest exposure. Based on the exposures to the conservatively defined receptors analyzed in this health-protective HHRA, other human populations, such as residents in the region, would not be expected to be exposed to unacceptable risk due to exposure to Project-related concentrations of mercury, arsenic, or antimony.

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APPENDIX A. SOURCES OF DATA IN THE HHRA

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Appendix A. Summary of Project-specific data sources used in the HHRA

Reference	Summary of data collected	Year Data Collected	Air	sediment	soil	surface water	berry plant	fish	intake rates
Air Sciences. 2016. Air Quality Analysis Report, Donlin Gold Project, Alaska. Prepared by Air Sciences, Inc., Golden, Colorado, for Donlin Gold LLC. February	air As, Sb	Future	x						
Air Sciences. 2017. Air Quality predictions contained in spreadsheet "Crooked Creek PM2.5 Annual Results (2017-05-17).xlsx" provided via e-mail to Mike Rieser, Donlin Gold, by Ejaz Memon, Air Sciences, Inc., May 17.	air As, Sb	Future	x						
ARCADIS. 2007a. Baseline Mercury in Vegetation, Soil, and Stream Sediment, Donlin Creek Project, Alaska. Ref. CO001194.0001.00001, March 2007.	soil, sediment, plant (Hg only)	2006		x	x		x		
ARCADIS. 2007b. DRAFT - Donlin Creek Project Stream Sediment Analysis Report. Prepared for Barrick Gold Corporation. November 2007.	sediment (As, Sb only)	2007		x					
ARCADIS. 2008. DRAFT - Mercury/Metals Sediment 2008 Report, Donlin Creek Project, Alaska. Prepared for Barrick Gold Corporation, Donlin Creek Joint Venture. September 2008.	sediment (Hg only)	2008		x					
ARCADIS. 2014. Assessment of Mercury Fate in the Environment from Changes in Atmospheric Deposition. Prepared by ARCADIS for Donlin Gold, LLC. June. 196 pp.	soil, sediment (Hg only)	2013		x	x				
Brown, C., J.S. Magdanz, D.S. Koster and N.M. Braem. 2012. Subsistence Harvests in 8 Communities in the Central Kuskokwim River Drainage, 2009. Alaska Department of Fish & Game, Division of Subsistence. Technical paper No. 365.	subsistence harvest data	2009							x

Appendix A. Summary of Project-specific data sources used in the HHRA

Reference	Summary of data collected	Year Data Collected	Air	sediment	soil	surface water	berry plant	fish	intake rates
Enos, Robert (Nick). 2013. Personal communication. Environmental & Permitting Manager. Donlin Gold. Remaining data/document needs for Donlin geochem (#15, 16, 27, 28); and file Data Request 79.zip. September 11, 2013. Nancy Darigo. URS	surface water data	2013				x			
Environ. 2013. Monitoring of Baseline Air Concentration and Deposition of Mercury for the Donlin Gold Project. Prepared for the Donlin Gold Project, Alaska. December 2013.	Air (Hg baseline)	2011 - 2013	x	x					
Environ. 2015. Modeling of Local Impacts of Mercury Air Emissions from Stacks and Fugitive Sources, Advanced Water Treatment Scenario: Donlin Gold Mine, Alaska, Final Report. September 10.	Air (Hg emission estimates)	Future	x						
Fernandez, E. 2014. Personal communication. Senior Environmental Coordinator. Donlin Gold. Donlin Soil Samples 20140825.xlsx. August 25, 2014. Dave Every. URS.	soil As, Sb	1988 - 2010			x				
Ottertail. 2009. 2008 Aquatic Biomonitoring Report For the Donlin Creek Project (2004, 2005, 2006, 2007, and 2008 data compilation). May 2009	fish (As, Sb, Hg)	2004 - 2008						x	
OtterTail. 2012a. Access Database: OtterTail_Fish_Tissue_Data_04-12_130829	fish (As, Sb, Hg)	2005 - 2012						x	
Rieser, M. 2016.Personal communication, Senior Environmental Engineer. Subject: SO2 and sulfate emissions. Attachments: Port Emissions - Extract from 28-02-2014 EIS.xlsx, and Donlin_EI_FSU2_2016-11-08-ReportADECr0-summary.xlsx. Email to Penny Hunter, ERM. December 29, 2016.	SO2 emission estimates	Future	x						

Appendix A. Summary of Project-specific data sources used in the HHRA

Reference	Summary of data collected	Year Data Collected	Air	sediment	soil	surface water	berry plant	fish	intake rates
Rieser, M. 2017a. Personal communication, Senior Environmental Engineer. Subject: Sulfate discharge at average discharge rate, with attachment: 2015-06-09 Most Recent Results Summary – SO2.pdf. Email to Penny Hunter, ERM. January 11, 2017.	WTP discharge estimates	Future				x			
Rieser, M. 2017b. Personal Communication, Senior Environmental Engineer. Subject: Question about the arsenic/antimony air data. Attachments: Arsenic and Antimony LOM Year 16-2015 Air Impacts Emissions.xlsx, Environ 2013.pdf	As & Sb emission estimates	Future	x						
Weglinski. 2016. Personal communication, Senior Permitting Coordinator. Subject: Mercury info for Donlin. Attachments: AK-Hg4-Rusak-revised.xlsx; Existing MeHg Data-SW+Sed_083016.docx; Table 3.7-6 revised_Baseline MeHg Crooked Creek SW_083016.docx; WQ_Analytical_Results_SW_Update_3Q2013-3Q2015.xlsx	surface water baseline (and references for soil/sediment data)	2013 - 2015				x			

APPENDIX B. DEVELOPMENT OF BIOACCUMULATION FACTORS FOR THE HHRA

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Appendix B

Development of Bioaccumulation Factors for the HHRA

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1. TYPES AND APPLICATION OF BIOACCUMULATION (UPTAKE) MODELS

In risk assessments, concentrations of a constituent in biota are commonly estimated using an uptake model that takes one of the following forms (U.S. Environmental Protection Agency [USEPA] 2005a):

Form 1:	$C_{tissue} = UF \bullet C_{environ}$	…Equation 1	(constant)
Form 2:	$C_{tissue} = a \bullet C_{environ} + b$	…Equation 2	(linear regression)
Form 3:	$\ln(C_{tissue}) = a \bullet \ln(C_{environ}) + b$	…Equation 3	(log-linear regression)

Where:

C_{tissue} = concentration in tissue of subsistence food (biota)

$C_{environ}$ = concentration in environmental medium (e.g., soil, water, sediment)

UF = uptake factor

a = slope

b = y-intercept

An uptake factor (UF) is a simple ratio of the concentration in the tissue divided by the concentration in a particular environmental medium (e.g., soil, sediment, water). Regression models (Forms 2 and 3) are developed by fitting a relationship to paired concentrations, i.e., co-located concentrations in environmental medium and concentrations in the tissue. Consistent with methods used to develop USEPA ecological soil screening levels (2005a), a regression model is considered to be acceptable when both a statistically significant and a reliably predictive model exist. Regression models are developed in a manner consistent with standard practice, regulatory guidance (USEPA 2010), and basic concepts/techniques described in USEPA (2012) Benchmark Dose Software.

Consistent with USEPA (2005a) guidance, a constituent-specific regression model can be developed if both of the following performance criteria are not met:

1. The model equation is significant — i.e., the slope differs significantly from zero ($p < 0.1$); and
2. The model reliably predicts concentrations — i.e., the coefficient of determination (r^2) is greater than 0.2.

Consistent with the general rule for regression analyses (USEPA 2010), at least eight detect pairs (detect-detect pairs of concentrations in both environmental media and

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subsistence food) are needed. If eight detect pairs were not available, then the median bioaccumulation factor (BAF) was used to estimate bioaccumulation.

2. DEVELOPMENT OF HHRA BAFS

In developing uptake factors (also referred to as BAFs) for the Human Health Risk Assessment (HHRA), Project-specific baseline tissue data that was temporally and spatially co-located with primary environmental media (soil, sediment or water) was preferred. In reviewing available data, most of the Project-specific data did not meet these criteria. However, in many cases, biota samples were collected in the vicinity of primary media and within the same season or sampling timeframe.

Given the need to evaluate potential changes in future methylation rates and differences in bioaccumulation[1] between total mercury (THg) and methylmercury (MeHg), BAFs for mercury were derived in two ways:

- THg data for primary media and THg[2] data for biota; and
- MeHg data for primary media and THg data for biota.

The first type of BAF was developed to allow for comparisons to other regional data and the general literature. The second type of BAF was derived to account for potential future changes in bioavailable forms of mercury in the environment.

Derivation of each BAF is described below.

2.1. Plant (Berry)

In the case of berry data, biota samples were matched with soil samples collected for mercury analysis on an area-by-area basis. Although samples may not have been collected co-temporally, it was assumed that soil concentrations were relatively stable during the 2006-2007 timeframe and that soil samples collected in 2006 can be related to biota samples collected in 2007 to develop soil-to-berry BAFs. Project-specific soil data include measurements of both THg and MeHg. However, Project-specific plant tissue burden data (blueberry and cranberry samples) were reported for THg only. Project-specific median BAFs for THg for both blueberry and cranberry samples are reported and were found to be similar.

---

[1] Biomagnification is the process by which a constituent increases its concentration in the tissues of organisms at successively higher levels in a food chain. MeHg can biomagnify to levels that are many times greater than levels in the surrounding environment; whereas, inorganic mercury does not biomagnify up the food chain.

[2] Only (THg concentrations in tissues are reported for Project-specific data.

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Project-specific biota and soil empirical data for THg were plotted and regression analyses conducted (Attachment A). Although sufficient detect pairs were available, the resulting regression model did not meet performance criteria; specifically, the model slope was not significant and did not meet the criterion for reliably predicting concentrations in biota (see model *p* and r^2 in Figure 1). Therefore, the median soil-to-berry BAF for Hg was selected to estimate tissue burdens in berries (Table 1).

TABLE 1: PROJECT-SPECIFIC BAFS FOR BERRY PLANTS

	THg to THg[1] ($kg_{soil,dw}/kg_{tiss,ww}$)	**MeHg to THg**[2] ($kg_{soil,dw}/kg_{tiss,ww}$)	**Arsenic** ($kg_{soil,dw}/kg_{tiss,dw}$)	**Antimony** ($kg_{soil,dw}/kg_{tiss,dw}$)
Cranberry	0.020	2.9	No data	No data
Blueberry	0.015	2.0		

1. *BAFs (median) based on Project-specific empirical data (blueberry n = 29, cranberry n = 25) – plant part not specified.*
2. *BAFs (median) are based on Project-specific methylmercury data for soils and THg data for plants (blueberry n = 3, cranberry n = 3) – plant part not specified.*

$kg_{soil,dw}$ = *kilograms soil as dry weight*
$kg_{tiss,ww}$ = *kilograms tissue as wet weight*

Project-specific empirical data were available for MeHg in soil and THg in biota, but detected paired data were limited (n = 3) and did not meet the minimum dataset requirement to confidently use a regression model. Hence, a regression analysis was not conducted for MeHg. No Project-specific data for arsenic (As) and antimony (Sb) concentrations in blueberry or cranberry plants were available. Therefore, literature-based models were used to estimate As and Sb concentrations in subsistence plants (see Section 3).

2.1.1. Comparison to Other Datasets

Regional studies and regulatory sources were reviewed to provide perspective on the BAFs derived using Project-specific data (Table 2). For the Red Devil Mine[3] (Ecology and Environment, Inc. [E&E] 2014), blueberry tissue concentrations in unimpacted areas were measured separately for leaf and stem (leaf-stem) samples and for fruit samples. Non-detect concentrations in berry leaf-stem and soil samples were assigned the surrogate value of one-half the method detection limit (½ DL). For consistent comparisons to Donlin project-specific BAFs, only detect pairs were considered and

[3] The Red Devil Mine is an abandoned cinnabar mine and mercury production facility in the same region as the Project. The mine operated from 1933 to 1971, before more rigorous current-day environmental regulations and standards, and is now the subject of a number of environmental restoration projects.

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berry leaf-stem data were adjusted for moisture content.[4] Eight blueberry (fruit) samples were analyzed — one collected in 2011 and seven collected in 2012. Although the sample number of blueberry fruit is appropriate, the E&E (2014) report notes that statistical analysis for background levels in blueberry fruit was not performed due to the high rate of non-detections. Given the limited data for fruit, the authors decided to use soil-to-plant (above ground produce) BAFs from Baes et al. (1984) (Table 2).

TABLE 2: REGIONAL STUDY, USEPA, AND USDOE SOIL-TO-PLANT BAFS[1]

Constituent	Red Devil Mine[3,4]		USEPA		Baes et al. (1984)[5]
	#Detect Pairs	*BAF*[2]	(2005b,c)[3]	(2005d)[4]	
THg[6]	8	0.034	-	0.015	0.20
Arsenic	5	0.0014	0.0375	0.0063	0.006
Antimony	—	No data	$\ln(C_{plant}) = 0.938 \bullet \ln(C_{soil}) - 3.233$	0.032	0.03

1. *Soil-to-plant BAFs in units of* $kg_{soil,dw}/kg_{tiss,dw}$.
2. *Geometric mean background soil-to-blueberry (stem/leaf composite) BAFs – from Table 6-78 of the E&E (2014) report.*
3. *Generic (not species-specific) soil-to-plant uptake.*
4. *Soil-to-plant BAF for mercuric chloride* ($HgCl_2$) *from USEPA (2005d) Hazardous Waste Companion Database – no value for THg.*
5. *Baes et al. (1984) BAFs also used in Red Devil Mine (Table 6-24b of E&E 2014).*
6. *THg to THg.*

ln = lognormally transformed
(C_{plant}) *= plant tissue concentration*
(C_{soil}) *= soil concentration*

Combining the Red Devil Mine study data and the Donlin Project baseline data was considered in order to increase sample size and improve the robustness of the soil-to-blueberry uptake model. When examining these two datasets (adjusted for moisture content and only using detect pairs), the datasets appeared markedly different (Figure 2). Although the combined dataset resulted in a significant slope ($p < 0.1$), the slope is 'driven' by an extreme data point (upper right point) (Figure 2). When datasets are examined separately, there is no significant slope ($p > 0.1$) and there is poor predictability ($r^2 < 0.2$). When removing the outlier (upper right point), the slope is significant, but the predictability is poor ($r^2 < 0.2$). Given the apparent differences in the two datasets and the above findings, the two datasets were not combined. The sample sizes ($n > 25$) of the Donlin project dataset were considered to provide a robust estimate of the median BAF. In addition, other factors (e.g., method of collection, sample

[4] E&E (2014) reported tissue concentrations in dry weight. To facilitate comparisons with Project-specific BAFs, Red Devil plant concentrations were adjusted assuming a moisture content of 69.8 percent (median moisture content of Donlin blueberries).

processing, and analytical methods) increase the uncertainties associated with pooling these two datasets. No other regional datasets (e.g., Matz 2012) were available to compare berry BAFs.

USEPA's soil-to-plant (above ground produce) BAFs reported in USEPA's *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facility* (USEPA 2005d) are derived from uptake slope factors provided in USEPA (1992a) or from Baes et al. (1984). This guidance document's BAF for Hg is comparable to the Project-specific soil-to-berry BAF (Table 2).

Differences in soil-to-plant BAFs among studies are not remarkable due to the following possible sources of differences (including, but not limited to):

- Portions of plants that were analyzed (e.g., fruit vs. leaf-stem);
- Degree of co-spatial / co-temporal correspondence[5] between soil and berry samples in the study ;
- Soil attributes affecting bioavailability (e.g., pH, grain size, total organic carbon); and
- Methods of collection, sample processing, and analytical methods and limits.

Nonetheless, regional and other sources of soil-to-plant BAFs appear to suggest that Project-specific BAFs are relatively consistent with existing regional information.

2.1.2. Berry BAF Uncertainty Discussion

Stems and leaves of the blueberry/cranberry plants were measured for Project-specific Hg concentrations. However, humans typically consume the fruit (berry) of blueberry or cranberry plants. In reviewing the Red Devil Mine remedial investigation (RI) findings, the fruits of the plants typically contained lower concentrations of constituents than the stems and leaves of the plants. Therefore, it is likely that THg concentrations estimated for berries are overestimated.

In addition, Project-specific concentrations measured in soil and in berries are neither precisely co-temporal nor co-located. All soil and berry plant samples were not collected in the exact same location. Therefore in these cases, concentrations measured in plants may be less related to concentrations in soils in which the plants were growing. In addition, soil samples were collected in 2006, while berry tissue burdens were collected in 2006-2007. Hence, this evaluation presumes that 2006 soil concentrations are representative of those related to plant samples collected in 2007. Soil concentrations are

[5] The lack of co-temporal correspondence in Project-specific soil-plant data may be mitigated in part because soils tend to be more temporally stable (lower variability) and hence likely to be more representative of soil concentrations at the time plant samples were collected.

considered to be relatively stable over time as compared to other environmental media (e.g., water). The lack of co-location (spatial and temporal) introduces uncertainty in characterizations of soil-to-berry bioaccumulation.

2.2. Fish

Regulatory agencies consider the consumption of fish an important exposure pathway for human exposure to mercury (Agency for Toxic Substances and Disease Registry [ATSDR] 2013). The following resident fish have been observed in Crooked Creek (Ottertail Environmental, Inc. [Ottertail] 2009):

- Slimy sculpin[6]
- Northern pike
- Arctic grayling
- Dolly Varden
- Burbot
- Ninespine stickleback
- Longnose sucker
- Alaskan brook lamprey
- Round whitefish

The Project-specific data for these species were limited; specifically, the dataset primarily contains tissue burdens for juveniles or fish of 'unspecified' age.

In the HHRA, northern pike were initially selected to evaluate fish consumption exposure. Northern pike is a large (relatively long-lived) top predator and is the preferred fish for characterizing uptake of metals (particularly Hg) given the existence of an Alaskan fish advisory (tissue target level). However, the northern pike has only recently been observed[7] in Crooked Creek (Ottertail 2009). Moreover, the Project-specific tissue burden dataset for adult northern pike consists of only one sample.[8]

Given the limited dataset for adult northern pike, other fish species were considered. Although sculpin are not large, top predators, and they are not widely harvested for subsistence consumption, Project-specific tissue data are abundant, and sculpin data have been used in the Red Devil RI to estimate Hg concentrations in subsistence fish (E&E 2014). Therefore, estimates of subsistence fish tissue burdens using sculpin data (with a food chain multiplier [FCM]) were developed. Consistent with USEPA guidance,

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[6] Among the most abundant species observed, but not a popular game fish.

[7] Four individuals were caught at the Crooked Creek weir and although not previously observed, their presence is not unexpected (Ottertail 2009).

[8] Other mercury data for the northern pike in the Project's database were apparently for matrix spike/matrix spike duplicate (MS/MSD) that are used for laboratory quality assurance purposes.

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an FCM was applied to account for food chain biomagnification of MeHg into subsistence fish (McGeer et al. 2004, E&E 2014).

First, scuplin BAFs were derived. To derive these BAFs, both sediment-to-fish and surface water-to-fish BAFs were considered. Project-specific fish tissue burden data were reported for THg only, while sediment and surface water data included both THg and MeHg data. Project-specific sculpin samples were co-located by reach (specifically for Donlin and Crooked Creek reaches), but were not collected from the exact location or within the same timeframe as surface water or sediment samples. Fish samples were matched with sediment and surface water samples on a reach-by-reach basis. Surface water data are available spanning from 2005 and 2015, and indicate daily, seasonal, and annual variability (Tetra Tech 2013). Sediment data, on the other hand, appear relatively similar from one year to the next, though like surface water, concentrations vary spatially by reach.

Sediments are considered to be 'sinks' for metals in aquatic systems (USEPA 1992b) and it is reasonable to assume that sediment chemistry would be relatively stable over time (as compared to water chemistry). With this in mind, sediment samples collected during 2007-2008 were related to sculpin samples collected during 2004-2006 to develop Project-specific sediment-to-fish BAFs. Surface water-to-fish and sediment-to-fish BAFs derived from Project-specific data (Ottertail 2009) are shown in Tables 3 and 4, respectively.

TABLE 3: PROJECT-SPECIFIC SURFACE WATER-TO-FISH BAFS [1]

Constituent	# Samples	BAF ($L_{surface\ water}/kg_{tiss,ww}$)	Age Class
Northern Pike			
THg[2]	1	271,613	Adult
Sculpin			
THg[2]	44	21,032	Juvenile (1+ yrs and juvenile)
THg[2]	27	24,323	Unspecified
Arsenic	44	136	Juvenile (1+ yrs and juvenile)
Arsenic	27	138	Unspecified
Antimony	44	8.7	Juvenile (1+ yrs and juvenile)
Antimony	27	8.7	Unspecified

1. *Given temporal variability and lack of temporal co-location, surface water-to-tissue BAFs were not derived.*
2. *BAFs were based on Project-specific THg data for surface water and THg data for fish.*

* *Surface water data had a non-detect rate of 54% and fish tissue data had a non-detect rate of 100%; therefore, no BAFs were derived.*

L = liter

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TABLE 4: PROJECT-SPECIFIC SEDIMENT-TO-FISH BAFS[1]

Constituent of Interest	# Samples	BAF ($kg_{sediment,dw}/kg_{tiss,ww}$)	Age Class
Northern Pike			
THg[2]	1	1.7	Adult
Sculpin			
THg[2]	44	0.23	Juvenile (1+ yrs and juvenile)
THg[2]	27	0.26	Unspecified
THg[2,4]	105	0.22 – 0.68	Unspecified
MeHg (to THg)[3]	44	55	Juvenile (1+ yrs and juvenile)
MeHg (to THg)[3]	27	49	Unspecified
Arsenic	44	0.016	Juvenile (1+ yrs and juvenile)
Arsenic	27	0.011	Unspecified
Antimony	44	0.018	Juvenile (1+ yrs and juvenile)
Antimony	27	0.016	Unspecified

1. *Given regressions did not achieve the model reliability performance criterion of $r^2 > 0.2$, median BAFs were used to estimate fish tissue burdens.*
2. *BAFs were based on Project-specific THg data for sediments and THg data for fish.*
3. *BAFs were based on Project-specific MeHg data for sediments and THg data for fish.*
4. *ARCADIS (2014) data included for comparison. All other data from Ottertail 2009.*

* *There were no detect pairs (0 of 11 pairs) for antimony data, so no BAF was calculated.*

$kg_{sediment,dw}$ = *kilograms sediment as dry weight*

A regression model-based uptake model was considered for mercury BAFs (see Attachment B). However, although there were a number of fish samples, they were related to only two to three reaches (for which surface water or sediment sample results have been averaged). Hence, Project-specific regression models were composed of 'repeated' measures for two to three surface water or sediment concentrations (Figure 3, Attachment B). Hence, the resulting regression models (with 'repeated measures') had limited data to detect a trend and did not meet the criterion for reliably predicting concentrations in fish tissue (e.g., the regression model r^2 were less than 0.2). Therefore, median sediment-to-fish BAFs were selected to estimate tissue burdens in subsistence fishes (Table 4).

2.2.1. *Comparison to Other Datasets*

Despite numerous regional studies that have measured fish concentrations, few studies have developed BAFs. A brief discussion of each study comparison is included below. Tables 5 and 6 summarize the BAFs derived from regional studies or other guidance.

THg measured in fish for Project-specific studies were consistent with U.S. Geological Survey samples of fish collected from the region (Gray and Sanzolone 1996, Gray et al. 1997, Gray et al. 2000, as discussed in ARCADIS 2014), and was consistent overall with Bureau of Land Management (BLM) samples collected from the Kuskokwim River and tributaries within the middle Kuskokwim Region (Matz 2012). The BLM data indicated slightly higher average THg concentrations in whole body fish tissue samples (average 240 micrograms per kilogram [µg/kg] wet weight) compared to Project data. However, the fish data presented in the BLM study did not separate upstream and downstream reaches. Samples collected from some reaches where there was historical mining impact from the Red Devil Mine showed, at times, higher Hg content in fish than in other reaches (Matz 2012). BLM is working with USEPA and the Alaska Department of Environmental Conservation to investigate the extent of contamination and develop and implement remediation methods for the Red Devil Mine.

TABLE 5: SURFACE WATER-TO-FISH BAFS FROM REGIONAL STUDIES AND USEPA

Constituent	Red Devil Mine (E&E 2014)			USEPA (2005d)
	Sample Size	BAF[1,2]	Fish / Age Class	BAF[3]
THg[4]	45	5,715	Sculpin (adult)	NA
MeHg (to MeHg)[5]	7	1,656,000	Sculpin (adult)	NA
Arsenic	45	97	Sculpin (adult)	114
Antimony	21	126	Sculpin (adult)	40

1. *BAFs in units of $L/kg_{tiss,ww}$.*
2. *BAFs (= $tissue_{95UCL}$ / $surface\ water_{95UCL}$) in units of $L/kg_{tiss,ww}$ – data from Table O-3 and O-5 of E&E (2014) report.*
3. *Hazardous Waste Companion Database (USEPA 2005e).*
4. *BAFs were based on THg data for surface water and THg data for fish.*
5. *BAFs were based on MeHg data for surface water and MeHg data for fish.*

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TABLE 6: SEDIMENT-TO-FISH BAFS FROM REGIONAL STUDIES

Constituent	Red Devil Mine (E&E 2014)		
	Sample Size	BAF[1]	Fish / Age Class
THg[2]	45	0.028	Sculpin (adult)
MeHg (to MeHg)[3]	7	0.051*	Sculpin (adult)
Arsenic	45	0.015	Sculpin (adult)
Antimony	21	0.013	Sculpin (adult)

1. BAFs were (= $tissue_{95UCL}$ / $sediment_{95UCL}$) in units of $kg_{sed,dw}/kg_{tiss,ww}$ – data from Table O-2 and O-5 of E&E (2014) report.

2. BAFs were based on THg data for sediments and THg data for fish.

3. BAFs were based on MeHg data for sediments and MeHg data for fish.

** Derived from the 95% Upper Confidence Limit (95UCL) for seven fish samples for MeHg and 95UCL for 36 sediment samples for MeHg.*

2.2.1.1. Red Devil Mine RI Report (E&E 2014)

In general, the Red Devil surface water-to-fish and sediment-to-fish BAFs for As are similar to Project-specific BAFs for As. However, surface water-to-fish and sediment-to-fish BAFs for Hg reported in the Red Devil Mine RI report are lower than Project-specific BAFs for Hg. Modeled concentrations of As and Hg in the Red Devil Mine study are also lower than measured concentrations in subsistence fish (Table 6-24a of E&E 2014).

In the Red Devil Mine RI report, concentrations of metals in fish are estimated using slimy sculpin whole-fish tissue concentration data from Red Devil Creek, multiplied by an FCM.[9] Using sculpin data to estimate subsistence fish was considered by the authors to be health-protective because slimy sculpin are a resident fish. For MeHg, an FCM of 3 is assumed to account for biomagnification[10]; for other metals, an FCM of 1 is assumed.[11]

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[9] Game fish concentration is set equal to three times the concentration in sculpin (= 3 • EPC_{fish}). If modeled, this would be equivalent to the following general model: $C_{fish} = 3 \cdot C_{water} \cdot BAF$.

[10] This approach is supported by the fact that the biomagnification of MeHg typically is three-fold with each trophic transfer (McGeer et al. 2004).

[11] This approach is defensible because biomagnification of metals (other than MeHg) in aquatic organisms is rare.

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2.2.1.2. *Kuskokwim River Region (Matz 2012)*

During 2010-2011, fish[12] were collected in the Kuskokwim River and its tributaries, and analyzed for Hg, As, and Sb. As and Sb concentrations were higher in fish collected from Red Devil Creek compared to all other tributaries.

Northern pike, burbot, and arctic grayling had variable Hg concentrations across the study area. Significant, positive correlation between age and size of fish to Hg burdens were observed in northern pike, but not in burbot, arctic grayling, or sheefish. MeHg to THg ratios were near 1.0, suggesting chronic, long-term exposure to Hg via ingestion of prey (Guilherme et al. 2010).[13] This study's findings were used to issue fish consumption advisories based on Hg by the State of Alaska.

Note that this study reported only concentrations in fish; no corresponding surface water and sediment concentrations were reported. Given the lack of co-located water or sediment concentrations, data could not be used to derive surface water-to-fish or sediment-to-fish BAFs.

2.2.1.3. *HHRA Protocols for Hazardous Waste Combustion Facilities (USEPA 2005d)*

No surface water-to-fish BAFs for THg, mercuric chloride ($HgCl_2$), or MeHg are provided in this guidance document. Surface water-to-fish factors for As and Sb are provided and shown in Table 5. Literature for surface water-to-fish BAFs is often unclear whether the water concentrations are dissolved or total concentrations[14]; nonetheless, USEPA (2005d) recommends multiplying concentration (total or dissolved) in water and the surface water-to-fish BAFs.

No surface water-to-fish or sediment-to-fish BAFs for THg, $HgCl_2$, MeHg, As, or Sb are provided in USEPA 2005d.

2.2.2. ***Fish BAF Uncertainty Discussion***

Co-location and co-temporal collection of concentration data are preferred when deriving relationships between surface water and fish burden. Neither the sediment-to-fish nor the surface-water-to-fish datasets were exactly co-temporal or spatially co-located.

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[12] Adult salmon were not sampled because they spend most of their lives at sea (Matz 2012).

[13] MeHg-to-THg ratios much less than 1.0 indicate acute, short-term exposure to mercury in water (Guilherme et al. 2010).

[14] Most of the literature reviewed indicate that laboratory experiments were conducted using filtered or distilled water (USEPA 2005d).

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However, given temporal variability often observed in water chemistry data, it is uncertain whether and to what degree concentrations in fish tissue are related to concentrations in surface water. Therefore, Project-specific sediment-to-fish BAFs were used (in preference to surface water-to-fish) to estimate constituent tissue concentrations.

BAFs reflect the interactions of numerous biotic and abiotic factors, and are site-specific, reflecting both abiotic media geochemistry and area populations. Constituent content in sediment may not be the predominant exposure medium for the fish species (exposure via food pathways in uncontaminated sites typically dominates fish exposure), but sediment metal contents are often used to estimate bioaccumulation in prey items (plants, sediment-dwelling invertebrates), and sediments are a primary source of mercury to overlying water. Therefore given available data, Project-specific sediment-to-fish BAFs provide a defensible approximate measure of bioaccumulation rates to fish in aquatic systems of interest.

2.3. Waterfowl (Mallard Duck)

No Project-specific empirical data exist that relate metals concentrations in the environment to tissue burdens of birds. Similarly, few data exist in the peer-reviewed literature or regulatory guidance that relate metals concentrations in the environment to tissue burdens of subsistence birds.

A general query was conducted to search several peer-reviewed literature databases including, but not limited to:

- PubMed
- Google Scholar
- Science Direct

Additionally, regional studies were reviewed for relevant information. While there is literature documenting the accumulation of metals in birds, many studies examined uptake as function of concentration in food, not in environmental media (e.g., soil). Further, the remaining studies found to-date did not include both co-located environmental and bird tissue concentrations. Accordingly, no appropriate literature-based studies were found that could be used to develop quantitative uptake models.

Similarly, no relevant data or soil-to-bird BAFs are presented in USEPA (2005d). However, USEPA (2005d) provides a method to develop a biotransfer factor to estimate Hg burdens in subsistence birds for Hg. Biotransfer factors for birds are expressed as the ratio of the constituent concentration in the fresh weight tissue to the constituent intake from food. Biotransfer factors for birds are derived by dividing the uptake slopes (from

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USEPA 1992a) by a daily consumption rate,[15] and then converted to fresh weight by using a moisture content.[16] If using a biotransfer factor, tissue burdens in wild game birds may be estimated using the following two-step calculation (USEPA 2005d):

Step 1: Estimate concentrations in food using plant BAFs.

Step 2: Estimate concentrations in the subsistence game bird (e.g., mallard) tissues using biotransfer factors and the subsistence game bird's ingestion rate.

There are a number of uncertainties involved in deriving a biotransfer factor using this method. First, tissue concentrations of plants consumed by mallard ducks were not measured in the study area, and so would have to be estimated by developing additional BAFs. Additional uncertainties would be 'compounded' because of estimates needed for intake rates by mallard ducks, the types of plants consumed, and the proportion of plants consumed by mallards in the study area versus outside the study area.

Given the lack of both Project-specific empirical data and available widely recognized literature-based models/factors, no soil-to-bird BAFs were derived. The absence of numerical soil-to-bird BAFs is the state-of-the-practice and is not unique to this Project.

2.3.1. Comparison to Other Datasets

No other regional or relevant datasets for the mallard were found to conduct appropriate comparisons.

2.3.2. Waterfowl BAF Uncertainty Discussion

Currently, there are insufficient Project-specific empirical data to reliably characterize the bioaccumulation of metals (including Hg) into game bird tissue. Further, there is a lack of widely accepted bioaccumulation models to estimate tissue burdens in representative game birds (e.g., the mallard).

2.4. Small Mammals (Beaver)

No Project-specific empirical data were collected that relate metals concentrations in the environment (e.g., soils) to tissue burdens of wild mammal game. Given the lack of Project-specific data, literature-based models were reviewed to estimate tissue burdens in subsistence game mammals (e.g., beaver).

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[15] For chickens, a daily feed consumption rate of 0.2 kg $feed_{dw}$ per day) was used (USEPA 2005d).

[16] For chickens, an assumed tissue moisture content of 75 percent was used (USEPA 2005d).

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Soil-to-mammal BAFs are obtained from two widely accepted sources for the following metals of interest:

- Hg — Development and Validation of Bioaccumulation Models for Small Mammals (Sample et al. 1998)
- As —Ecological Soil Screening Levels for Arsenic (USEPA 2005b)
- Sb — Ecological Soil Screening Levels for Antimony (USEPA 2005b)

These soil-to-mammal BAFs are presented in Table 7.

Regional studies and regulatory sources were used to provide context for the BAFs selected for this Project. The Red Devil Mine RI did not propose soil-to-mammals BAFs for Hg, and proposed to use the same soil-to-mammals BAFs for As and Sb as shown in Table 7. No other regional dataset contains data with which to derive soil-to-mammal BAFs. No relevant data or soil-to-mammal BAFs are presented in USEPA (2005d).

TABLE 7: BAFS /MODELS FOR SMALL MAMMALS

Constituent	BAF/Model ($kg_{soil,dw}/kg_{tiss,dw}$)	Source
THg to THg[1]	0.054	Sample et al. 1998[1]
Arsenic	$\ln(C_{mammal}) = 0.8188 \bullet \ln(C_{soil}) - 4.8471$	USEPA 2005b
Antimony	0.050	USEPA 2005c

All BAFs in units of $kg_{soil,dw}/kg_{tiss,dw}$.

1. *Recommended by Sample et al. 1998 because significant regression model fits were not obtained and because data for insectivore and omnivore BAFs were limited.*

2.4.1. *Comparison to Other Datasets*

No other regional or relevant datasets for beaver were found to conduct appropriate comparisons.

2.4.2. *Small Mammal BAF Uncertainty Discussion*

Currently, there is insufficient Project-specific empirical data to reliably characterize the bioaccumulation of metals (including Hg) into mammal tissue. Given the lack of Project-specific empirical data, soil-to-mammal BAFs presented in USEPA (2005b, c) and Sample et al. (1998) were used for the HHRA.

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3. SUMMARY OF BAF REVIEW

Selected BAFs for use in the HHRA are shown in Table 8.

For berry data, Project-specific BAFs for mercury are relatively consistent with other existing regional information. The USEPA (2005d) equation to estimate uptake into plants was used to derive a BAF for As and Sb. To estimate small mammal and waterfowl BAFs, linear regression model-based BAFs developed by USEPA and Sample et al. (1998) were used for these biota endpoints.

For fish data, sculpin tissue burdens were used to develop exposure estimates for fish for use in the HHRA given that: (1) sculpin is a resident fish (likely to spend much of its lifetime in the study area); (2) a large dataset is available (n > 40 samples); and (3) the precedent established by the Red Devil Mine RI in using sculpin tissue data to estimate tissue burdens in subsistence fish. Combining the Project's dataset with other regional studies to obtain a more robust dataset was also considered. However, given the differences and uncertainties associated with combining datasets from other studies, datasets could not be combined. Project-specific sediment-to-fish BAFs were used to estimate constituent concentrations in sculpin. Consistent with USEPA guidance and other assessments conducted in the region, an FCM of 3 was applied to sediment-to-fish BAFs to account for food chain biomagnification of MeHg into pike (McGeer et al. 2004) — for other metals, a FCM of 1 was used.

TABLE 8: BAFS USED IN THE HHRA

Biota	Environ. Medium	THg[1]	MeHg[2]	Arsenic	Antimony
Berry[3]	Soil	0.020	2.9	0.0375	$\ln(C_{plant}) = 0.938 \bullet \ln(C_{soil}) - 3.233$
Sculpin[4]	Sediment	0.23	55	0.016	0.018
Mallard[5]	Soil	0.054	—	$\ln(C_{mammal}) = 0.8188 \bullet \ln(C_{soil}) - 4.8471$	0.05
Beaver[6]	Soil	0.054	—	$\ln(C_{mammal}) = 0.8188 \bullet \ln(C_{soil}) - 4.8471$	0.05

1. *THg in medium to THg in biota.*
2. *MeHg in medium to THg in biota.*
3. *The soil-to-plant BAF for Hg is in units of* $kg_{soil,dw}/kg_{tiss,ww}$.
   The soil-to-plant BAF for As and Sb is in units of $kg_{soil,dw}/kg_{tiss,dw}$.
4. *All sediment-to-fish BAFs are in units of* $kg_{sed,dw}/kg_{tiss,ww}$.
5. *All soil-to-bird BAFs use soil-to-mammal BAFs as surrogate values given the lack of Project-specific and literature-based BAFs/models.*
6. *All soil-to-mammal BAFs are in units of* $kg_{soil,dw}/kg_{tiss,dw}$.

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Figures

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Figure 1. Scatterplot of Mercury Concentrations in Blueberry Plants and Soils

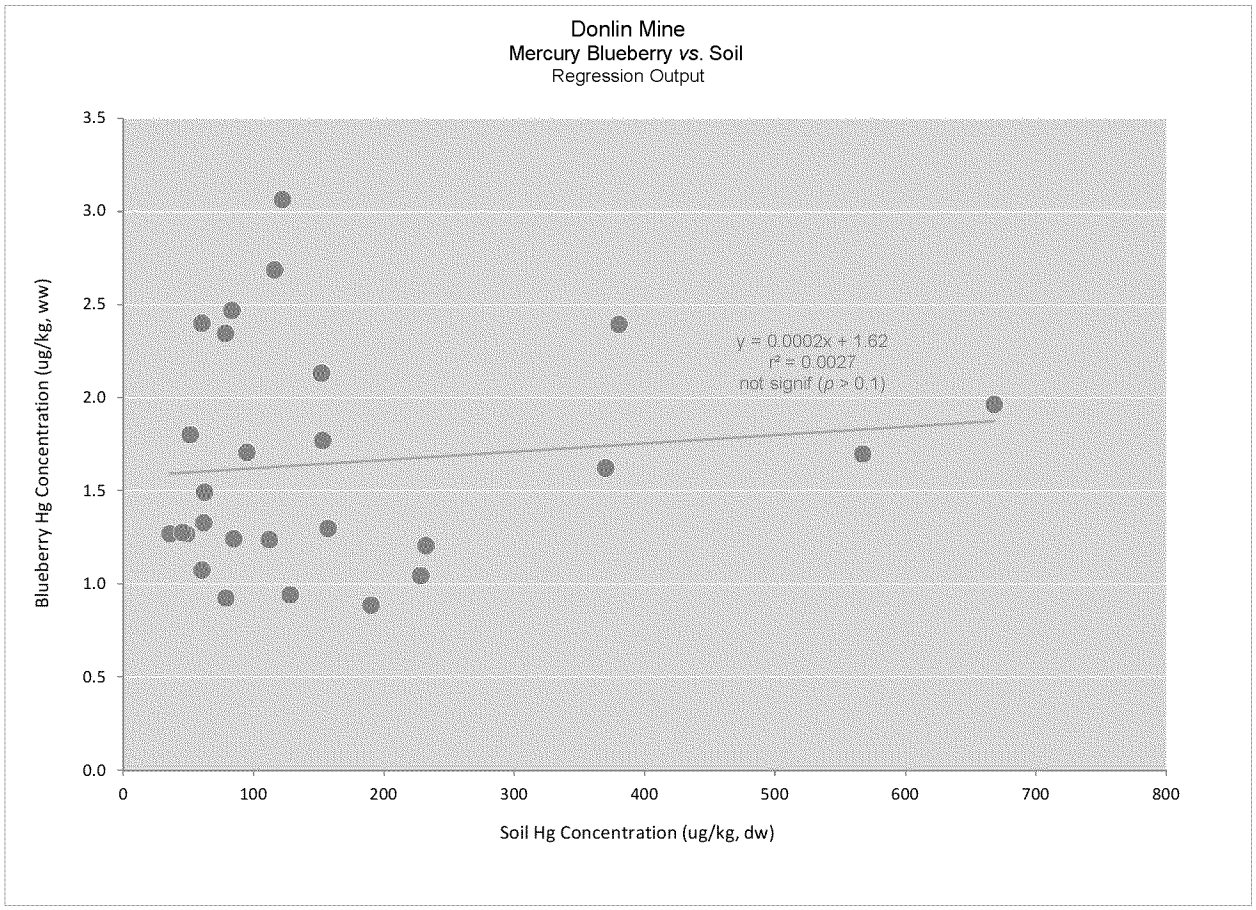


Figure 2. Scatterplot of Mercury Concentrations in Blueberry Plants and Soils with Red Devil Mine Data

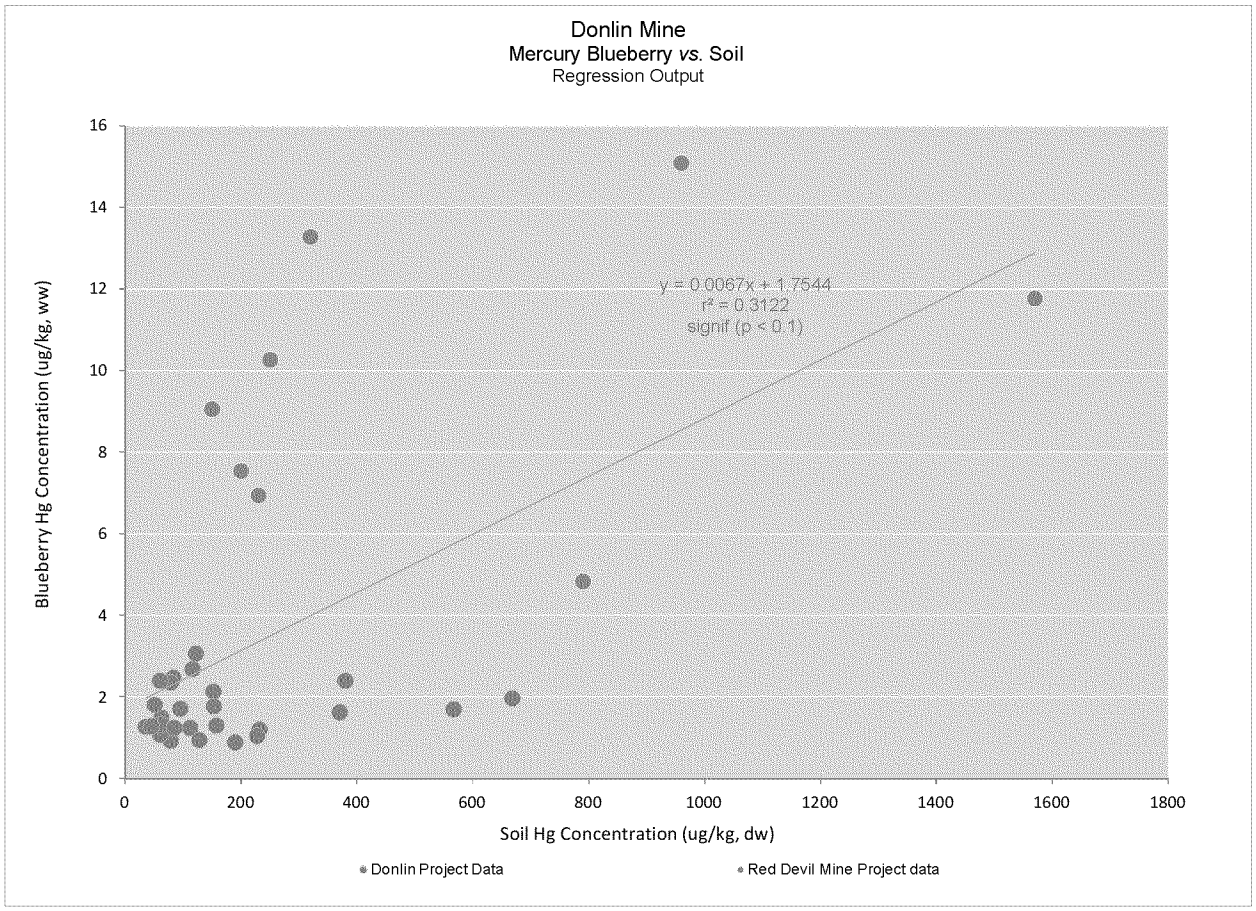
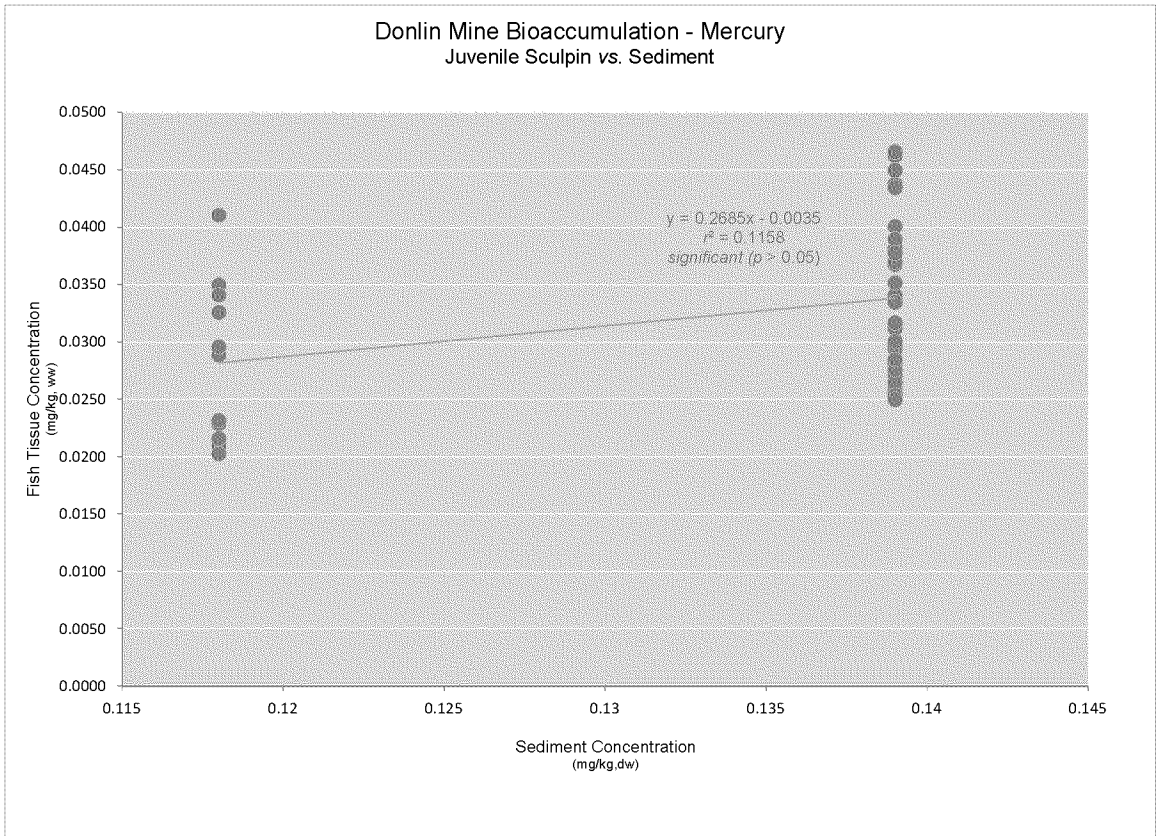
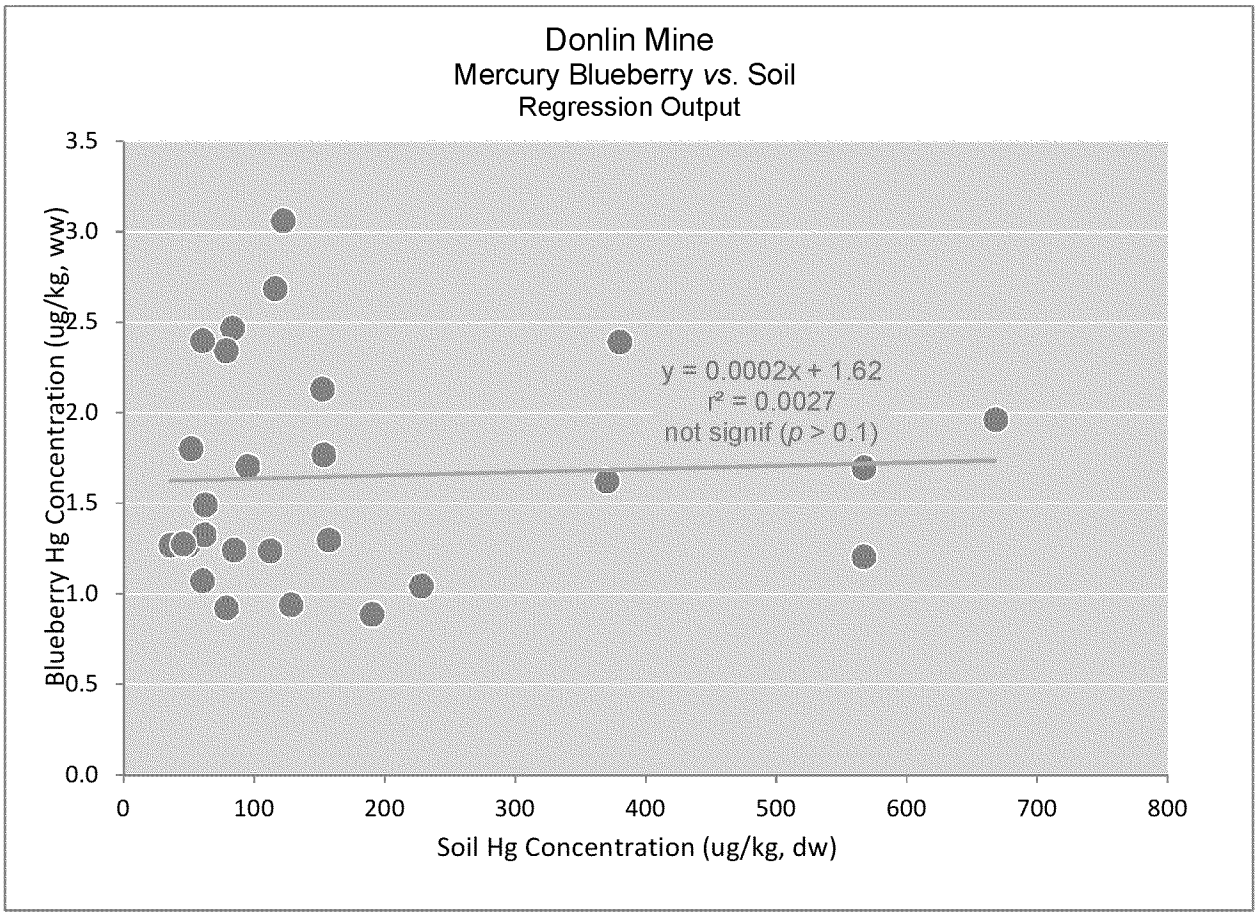


Figure 3. Scatterplot of Mercury Concentrations in Sculpin Juveniles and Sediments



Attachment A
Soil-to-Blueberry / Soil-to-Cranberry BAFs
Project-specific Empirical Data*

*Figure A-1 Blueberry vs. Soil: Mercury Concentrations**



Blueberry Regression

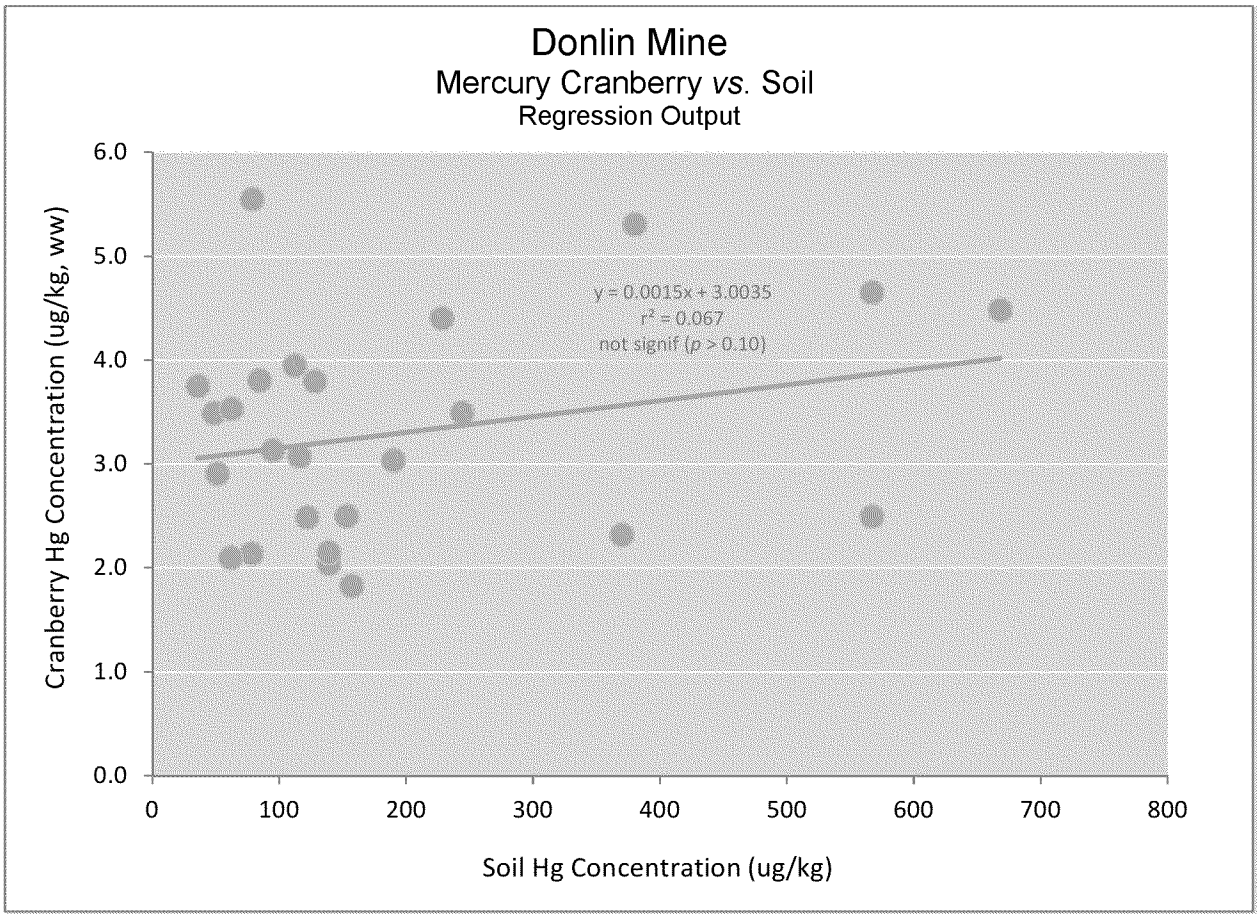
Regression Statistics	
R Square	0.003
Observations	27

ANOVA

	df	*SS*	*MS*	*F*	*Signif F*
Regression	1	0.025	0.025	0.07	0.80
Residual	25	9.22	0.37		
Total	26	9.24			

* Non-significant slope and poor predictability ($r^2 < 0.2$) – does not meet data quality objectives to support regression-based uptake model.

*Figure A-2 Cranberry vs. Soil: Mercury Concentrations**



Cranberry Regression

Regression Statistics	
R Square	0.067
Observations	25

ANOVA

	df	*SS*	*MS*	*F*	*Signif F*
Regression	1	1.7	1.7	1.7	0.21
Residual	23	24	1.0		
Total	24	26			

* Non-significant slope and poor predictability ($r^2 < 0.5$) – does not meet data quality objectives to support regression-based uptake model.

Table A-1 Soil-to-Berry BAF: Descriptive Statistics

Chem	*N*	*%Detect Pairs*	Min	Max	Median	Mean	Std Dev	CV
Blueberry								
Mercury	27	100%	0.0029	0.040	0.015	0.017	0.011	66%
$Mercury_{RD}$[1]	8	100%	0.0061	0.060	0.034	0.030	0.019	63%
MethlylMercury	3	100%	0.43	3.9	2.0	2.1	1.7	81%
Cranberry								
Mercury	25	100%	0.0044	0.10	0.020	0.030	0.025	82%
MethlylMercury	3	100%	0.63	3.1	2.9	2.2	1.4	62%

All uptake factors ($kg_{soil,dw}/kg_{tiss,ww}$)

1 Red Devil Mine blueberry data (Matz 2012)

Use of median uptake factors is recommended, given limited non-significant slopes ($p > 0.1$) and/or poor predictability ($r^2 < 0.2$)

Attachment B
Sediment-to-Fish/Surface Water-to-Fish BAFs
Project-specific Empirical Data

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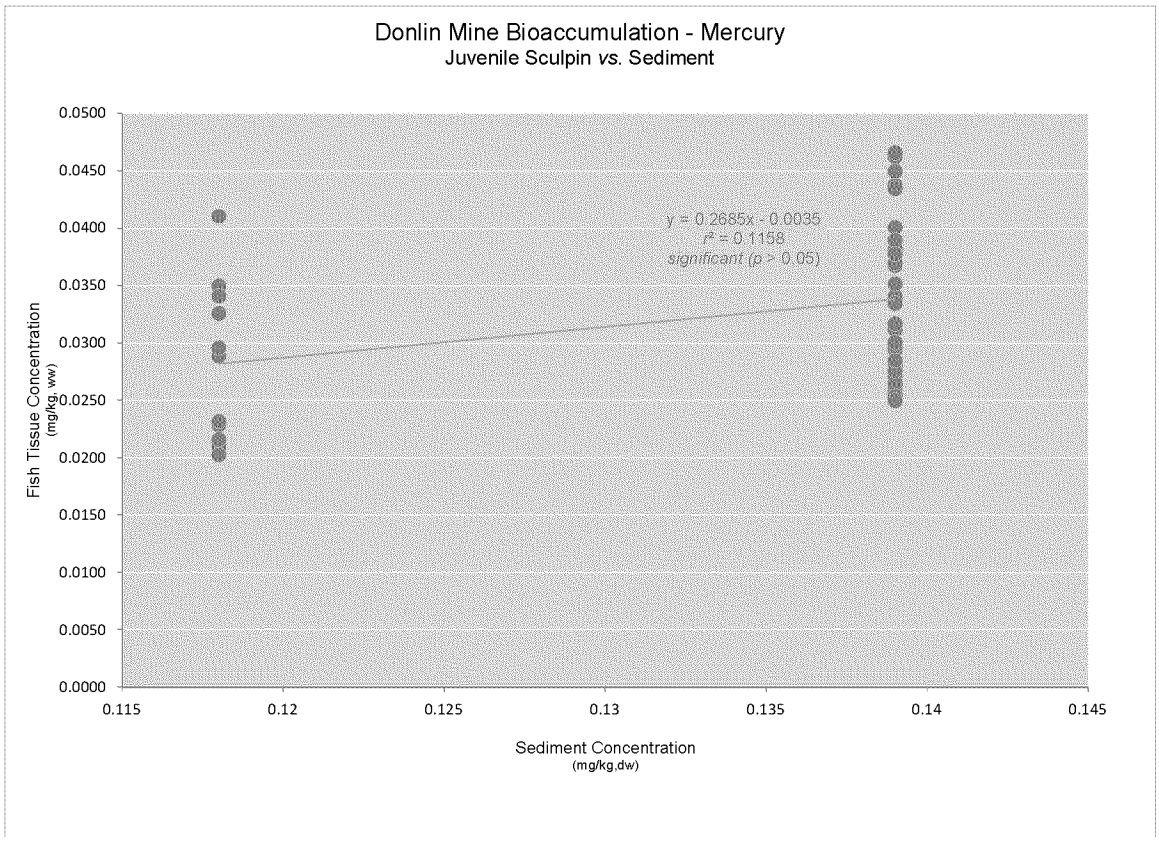
*Figure B-1 Mercury: Sculpin Juvenile vs. Sediment**

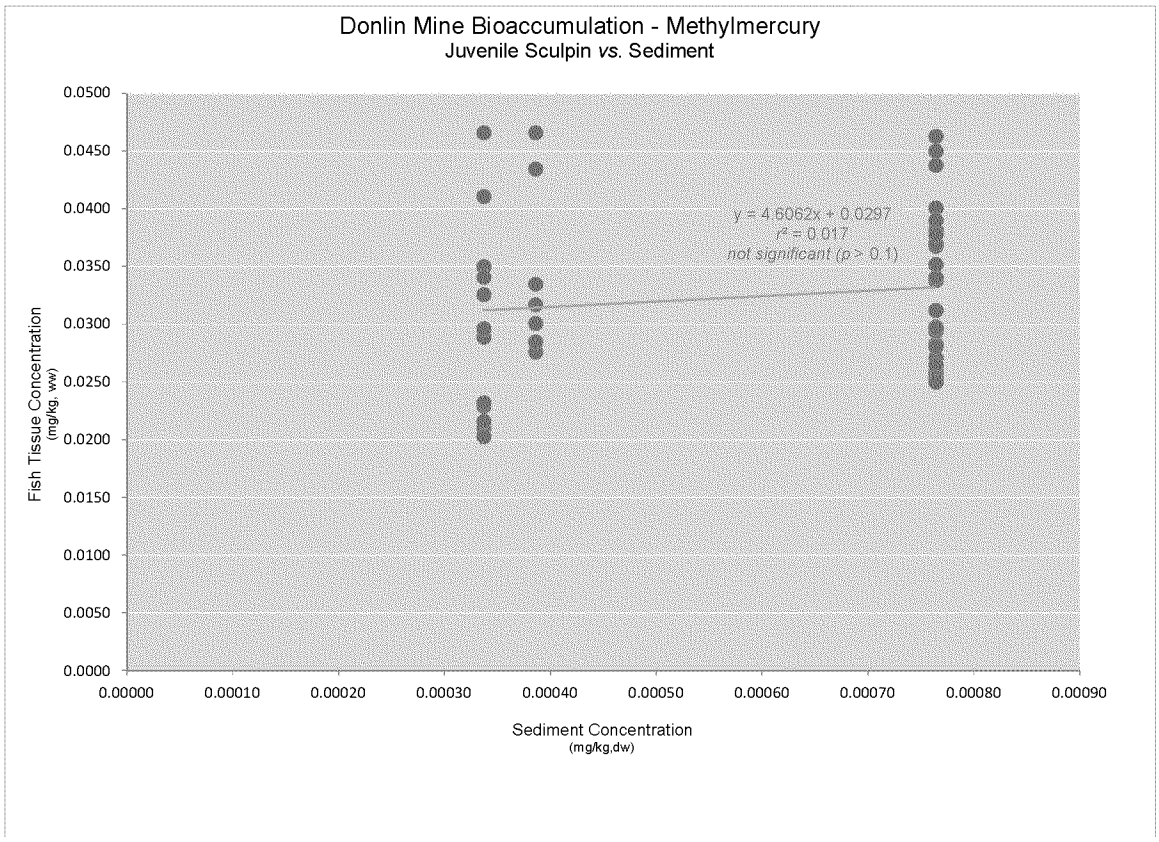


Regression Statistics	
Multiple R	0.34
R Square	0.12
Adjusted R Square	0.09
Standard Error	0.01
Observations	44

	df	*SS*	*MS*	*F*	*Signif F*
Regression	1	0.0003	0.0003	5.5	0.02
Residual	42	0.0020	0.00005		
Total	43	0.0023			

* Repeated measures at two surface water concentrations – does not meet data quality objectives to support regression-based uptake model.

*Figure B-2 Methylmercury: Sculpin Juvenile vs. Sediment**

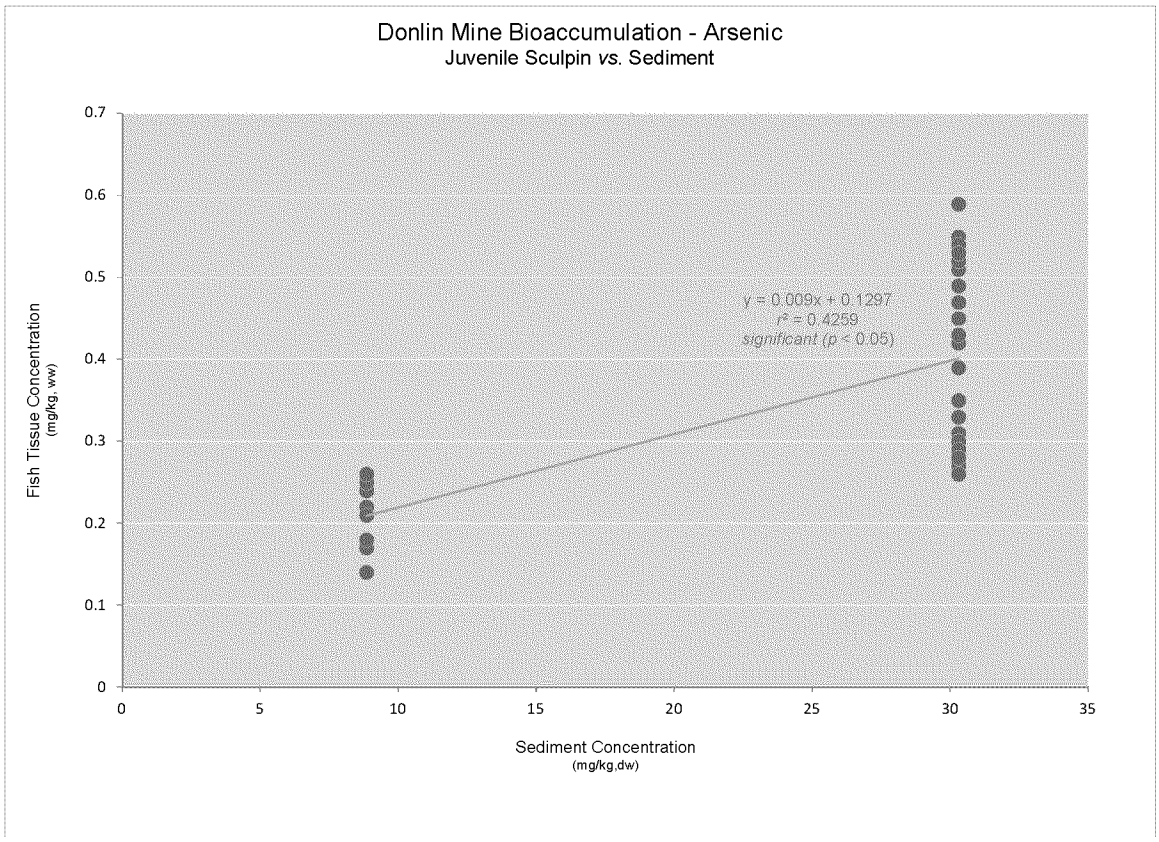


Regression Statistics	
Multiple R	0.13
R Square	0.02
Adjusted R Square	0.01
Standard Error	0.01
Observations	44

	df	*SS*	*MS*	*F*	*Signif F*
Regression	1	3.9E-05	3.9E-05	0.73	0.40
Residual	42	0.002	5.3E-05		
Total	43	0.002			

* Repeated measures at three surface water concentrations – does not meet data quality objectives to support regression-based uptake model.

*Figure B-3 Arsenic: Juvenile Sculpin vs. Sediment**

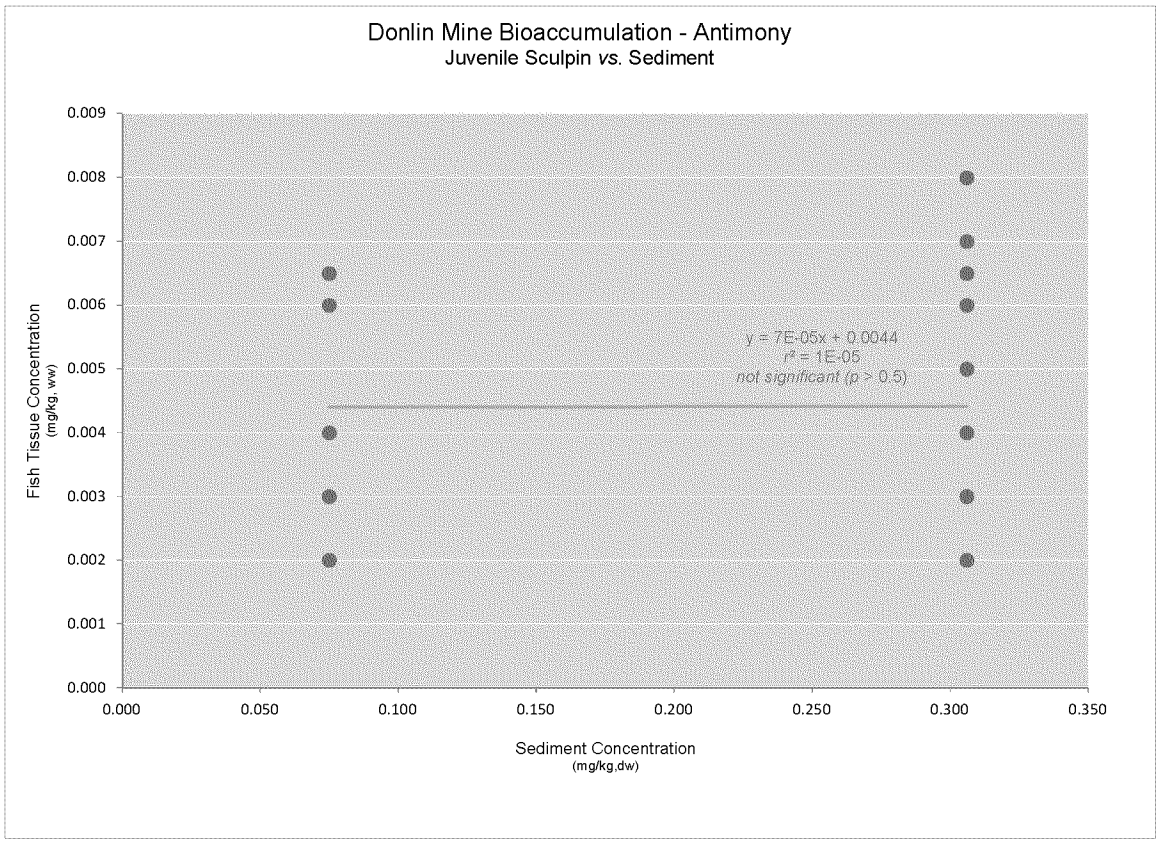


Regression Statistics	
Multiple R	0.65
R Square	0.43
Adjusted R Square	0.41
Standard Error	0.10
Observations	44

	df	*SS*	*MS*	*F*	*Signif F*
Regression	1	0.31	0.31	31	0.000002
Residual	42	0.41	0.01		
Total	43	0.72			

* Repeated measures at two surface water concentrations – does not meet data quality objectives to support regression-based uptake model.

*Figure B-4 Antimony: Juvenile Sculpin vs. Sediment**



Regression Statistics	
Multiple R	0.003
R Square	0.00001
Adjusted R Square	-0.024
Standard Error	0.002
Observations	44

	df	*SS*	*MS*	*F*	*Signif F*
Regression	1	1.89E-09	1.89E-09	0.0005	0.98
Residual	42	0.00016	3.7E-06		
Total	43	0.00016			

* Repeated measures at two surface water concentrations – does not meet data quality objectives to support regression-based uptake model.

Table B-1 Sediment-to-Fish BAF: Descriptive Statistics

Chem	N	%Detect Pairs*	Min	Max	Median	Mean	Std Dev	CV
Northern Pike - Adult[1]								
Mercury	1	100%	1.7	1.7	1.7	1.7	--	--
Methylmercury	1	100%	551	551	551	551	--	--
Sculpin - 1+ Yrs Juveniles[2]								
Arsenic	44	100%	0.0086	0.029	0.016	0.016	0.006	37%
Antimony	44	66%	0.0065	0.087	0.018	0.026	0.024	92%
Mercury	44	100%	0.17	0.35	0.23	0.24	0.051	21%
Methylmercury	44	100%	33	138	55	63	28	44%
Sculpin - Unspecified Age[3]								
Arsenic	27	100%	0.0089	0.026	0.011	0.015	0.005	35%
Antimony	27	66%	0.0098	0.160	0.016	0.030	0.039	130%
Mercury	27	100%	0.21	0.33	0.26	0.26	0.039	15%
Methylmercury	27	100%	39	120	49	71	33	46%

All uptake factors ($kg_{sed,dw}/kg_{tiss,ww}$) are:

- wet weight (ww) fish tissue burdens are estimated by the laboratory using a dry weight-to-wet weight conversion based on %moisture of tissue sample
- because sediment-fish data are not precisely co-located, the median sediment concentration for the corresponding creek reach is used to calculate the uptake factor

* %Detect Pairs is based on the lowest %detect of sediment or tissue data

1 Northern Pike Adult uptake factors are based on data from CR-R1 only
- all but one sample are lab MS/MSD (lab QA) samples

2 Sculpin Juvenile uptake factors are based on:
- fish tissue data from 3 reaches: CR-R1, CR-R2, and DO-R1
- given lack of sediment data for CR-R2 and DO-R1, data for CR-R3 and DO-R2 are used

3 Sculpin (unspecified age) uptake factors are based on:
- fish tissue data from 3 reaches: CR-R1, CR-R2, and DO-R1
- given lack of sediment data for CR-R2 and DO-R1, data for CR-R3 and DO-R2 are used

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*Mercury in Surface Water-Fish Regression Statistics**

Regression Statistics	
R Square	0.004
Observations	36

ANOVA

	df	*SS*	*MS*	*F*	*Signif F*
Regression	1	0.0006	0.0006	0.14	0.71
Residual	34	0.16	0.0047		
Total	35	0.16			

	Coeffs	*Stnd Err*	*t Stat*	*P-value*
Intercept	0.00	0.10	-0.01	0.99
Slope	26385	71114	0.37	0.71

* Replication at two surface water concentrations (n = 2) – does not meet data quality objectives to support regression-based uptake model.

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Table B-2 Surface Water-to-Fish BAF: Descriptive Statistics

Chem	*N*	% Detect Pairs*	Min	Max	Median	Mean	Std Dev	CV
Northern Pike - Adult[1]								
Mercury[1]	1	100%	271613	271613	271613	271613	--	--
Sculpin - 1+ Yrs Juveniles[2]								
Arsenic	44	65%	56	262	136	155	60	39%
Antimony	44	47%	4.0	17	8.7	9.4	4.1	43%
Mercury	44	100%	16129	33967	21032	22157	4701	21%
Sculpin - Unspecified Age[3]								
Arsenic	27	65%	64	284	138	154	71	46%
Antimony	27	47%	4.0	24	8.7	10	4.7	46%
Mercury	27	100%	19097	29806	24323	24217	3747	15%

All uptake factors ($L/kg_{tiss,ww}$) are:

- wet weight (ww) fish tissue burdens are estimated by the laboratory using a dry weight-to-wet weight conversion based on %moisture of tissue sample
- because surface water-fish data are not precisely co-located, the median surface water concentration for the corresponding creek reach is used to calculate the uptake factor
- all but one sample are lab MS/MSD (lab QA) samples

* %Detect Pairs is based on the lowest %detect of water or tissue data

1 Northern Pike Adult uptake factors are based on data from CR-R1 only
- all but one sample are lab MS/MSD (lab QA) samples

2 Sculpin Juvenile uptake factors based on:
- fish tissue data from 3 reaches: CR-R1,-R2, and DO-R1
- given lack of surface water data for CR-R1, CR-R2, and DO-R1, data for CR-R3 and DO-R2 are used

3 Sculpin (unspecified age) uptake factors based on:
- fish tissue data from 3 reaches: CR-R1,-R2, and DO-R1
- given lack of surface water data for CR-R1, CR-R2, and DO-R1, data for CR-R3 and DO-R2 are used

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