Ecological Risk Assessment for Use of Wet Scrubbers at Alcoa Fjarðaál Aluminum Plant in Reyðarfjörður, Fjarðabyggð, Iceland

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Acronyms and Abbreviations

AET	apparent effects threshold
B[a]P	benzo[a]pyrene
BCF	bioconcentration factor
CoPC	chemical of potential concern
EIA	environmental impact assessment
ERA	ecological risk assessment
ER-L	effects range-low
ER-M	effects range-median
JAMP	Joint Assessment and Monitoring Program
LD50	median lethal dose
LOAEL	lowest-observed-adverse-effect level
MFO	mixed function oxidase
MPC	maximum permissible concentration
NMFS	National Marine Fisheries Service
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effects concentration
OSPAR	Oslo and Paris Conventions for the Prevention of Marine Pollution
PAH	polycyclic aromatic hydrocarbon
PEL	probable effects level
SQG	sediment quality guideline
TEL	threshold effects level
tpy	metric tonnes per year
TRV	toxicity reference value

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Introduction

Alcoa is constructing the Fjarðaál aluminum smelter at the Hraun industrial area in Reyðarfjörður, Fjarðabyggð, East Iceland (Figure 1). When the smelter becomes operational in 2007, it will produce 346,000 metric tonnes per year (tpy) of aluminum. The original concept for an aluminum smelter at this location was developed by Hydro Aluminium AS, Hæfi hf., Landsvirkjun, and Reyðarál hf., who planned to construct a facility (hereafter referred to as Reyðarál) that would produce 240,000–280,000 tpy in Phase 1 with the possibility of increasing capacity to 360,000–420,000 tpy in a Phase 2 expansion. A detailed environmental impact assessment (EIA) was prepared in 2001 for both phases of the proposed plant to evaluate likely impacts on terrestrial and marine environments during plant construction and operation, and socioeconomic impacts during operation (Reyðarál 2001). The EIA was submitted to the Iceland Planning Agency in May 2001 and was approved in August 2001. The main structures of the original aluminum smelter included pot rooms, casthouse, anode production facilities, anode rodding plant, fume treatment facilities, power transmission substation, alumina silos, warehouses, and service buildings.

In the spring of 2002 when Hydro Aluminium AS declared that they would be unable to meet the project schedule, Alcoa and Landsvirkjun entered into negotiation with the Icelandic government to construct a facility at the same location. In November 2002, Alcoa purchased Reyðarál from Norsk Hydro and Hæfi hf. Also in November 2002, Alcoa prepared a report (Hönnun et al. 2002) to compare the environmental impacts of the 322,000 tpy Fjarðaál plant with the two-phase, 420,000 tpy original Reyðarál design. Other than the change in production capacity, currently determined to be 346,000 tpy, the main difference between the Fjarðaál plant and the original Reyðarál design is that Alcoa is not planning to install an anode production plant, using imported anodes in the production process instead. In addition, Alcoa will export spent pot liner instead of placing it in a landfill on the facility.

Hönnun et al. (2002) concluded that the Fjarðaál facility would be able to meet applicable air emission standards without the need for installing a wet scrubber system; thus, the Fjarðaál facility is expected to have no process water discharge to the sea. However, incorporation of wet scrubbers is one of the design options being considered by Alcoa if additional control of sulfur dioxide emissions should become necessary. Because the primary design does not use wet scrubbers, the environmental impacts associated with their use at Fjarðaál are not addressed in the comparison EIA (Hönnun et al. 2002).

Objective and Scope

For the reasons outlined above, Alcoa is interested in assessing the environmental risks if wet scrubbers are used as pollution control devices when the Fjarðaál plant becomes operational. Alcoa has tasked Exponent with conducting an ecological risk assessment (ERA) of wastewater discharges from seawater scrubbers at the smelter to receiving waters of Reyðarfjörður. The primary chemicals of potential concern (CoPCs) are polycyclic aromatic hydrocarbons (PAHs), fluoride, and sulfur dioxide, because these constituents may become incorporated into wet scrubber water. The focus of the risk assessment is on the maximally exposed key components of the marine ecosystem, which include phytoplankton, benthic macroinvertebrates, mussels, and fish, although effects on other ecological components, such as marine birds and mammals, are also evaluated.

Conceptual Site Model

This section describes the conceptual site model for potential ecological exposures to chemicals released in wet scrubber effluent. A conceptual site model is a planning tool used for identifying chemical sources, complete exposure pathways, and potential receptors on which to focus the risk assessment. The purpose of a conceptual site model is to ensure that all potential pathways are considered regardless of whether those pathways are complete. The following sections characterize the environmental setting, describe the physical characteristics of the ecosystem, and identify potential exposure pathways and receptors.

Environmental Setting

The smelter is being constructed on the Hraun industrial tract in the municipality of Fjarðabyggð, on the east coast of Iceland (Figure 1). Formerly an area of marshes, much of the land has been drained historically for agriculture, although that practice has now been largely abandoned (Reyðarál 2001). The area of the smelter consists of a mosaic of moss heath, dwarf shrub heath, and natural grasslands. The facility location lies immediately adjacent to the inner part of Reyðarfjörður, an inlet of the North Atlantic Ocean. Reyðarfjörður is a very long and narrow fjord that reaches a maximum depth of 200 m at its mouth. The sides of the fjord are steep and the water depth increases from 5 to 120 m over a distance of less than 1 km leading out from the shoreline. Details on bathymetry, tidal elevation, current patterns, temperature, and salinity are provided in the pollutant dispersion modeling report prepared by Vatnaskil Consulting Engineers (Vatnaskil 2001).

Marine Physical Environment

Studies of the marine physical environment, including measurements of water current, temperature, and salinity, were conducted in December 2000 and February 2001. Throughout Reyðarfjörður, the salinity ranges from 34 to 35 ppt. There is a constant flow of seawater into the fjord, a relatively small discharge of freshwater from area streams, and good mixing throughout the water column within the fjord. Fjord water temperatures range from 4 to 5°C in July and 6 to 8°C in August. Stratification occurs in July, with lower salinity, warmer water

flowing in from outside the fjord layering on the top 10–20 m. In August, the stratification breaks down and the water column becomes well mixed and stays well mixed throughout fall and the winter months (Reyðarál 2001). The average inflow current (measured at the mouth of the fjord) is 2.5 cm/s, and the average outflow is 4 cm/s. The average current off of the study site is 3 cm/s. Based on these flow rates, the renewal rate of seawater in the fjord is approximately 4 to 5 weeks (Reyðarál 2001).

Sediment at Reyðarfjörður was sampled in the summer of 2000. The average particle size ranges from 0.028 to 1.354 mm. The nearshore bottom sediment is composed of sand, and the sediment deeper and farther from shore (the innermost part of the fjord) is composed of silt [<0.063 mm]). The average sedimentation rate in Reyðarfjörður is similar to other fjords in Iceland, and is predicted to be 1 cm over a 10-year period (Reyðarál 2001).

Potential Exposure Pathways

An exposure pathway is the course a chemical takes from a source to an exposed receptor. Exposure pathways consist of the following four elements: 1) a source; 2) a mechanism of release, retention, or transport of a chemical to a given medium (e.g., water, sediment); 3) a point of contact with the medium (i.e., exposure point); and 4) a route of exposure at the point of contact (e.g., incidental ingestion, dermal contact). If any of these elements are missing, the pathway is considered incomplete (i.e., it does not present a means of exposure). Only those exposure pathways judged to be potentially complete are of concern for ecological receptors. Potential pathways by which ecological receptors may be exposed to chemicals associated with the wet scrubber effluent are illustrated in Figure 2.

Primary exposure pathways are those expected to contribute highest to risk estimates, while secondary exposure pathways are not expected to increase risk substantially. Primary exposure pathways for water-column dwelling organisms, including plankton, and pelagic invertebrates and fish include dermal contact with chemicals in water and ingestion of water or other pelagic fauna. Bottom-dwelling, or benthic, organisms are the components of aquatic life that are most directly exposed to contaminated sediments. Pathways by which they may be exposed include dermal contact and ingestion (including ingestion of other benthic fauna). Epibenthic fish may also be exposed to chemicals through the ingestion of prey such as invertebrates or other fish, direct contact with sediment, and possibly through the incidental ingestion of sediment that may occur during foraging.

Complete exposure pathways may exist for wildlife that are resident in Reyðarfjörður or that incorporate part of the fjord within their foraging area, either spatially or seasonally. These potential wildlife receptors include marine mammals, seabirds, and shorebirds that forage along the intertidal zone of the shoreline. Birds and aquatic mammals may be exposed to chemicals through the ingestion of prey items, such as shellfish and fish, and possibly through the incidental ingestion of sediment. The relative importance of different exposure pathways to ecological receptors depends upon their life history requirements and feeding habits as well as the physical and biochemical properties of the chemicals.

Potential Ecological Receptors

The most comprehensive information regarding potential ecological receptors in Reyðarfjörður comes from ecological surveys conducted as part of the Reyðarál (2001) EIA. The following sections briefly summarize key findings of those ecological surveys.

Coastal Flora and Fauna

Coastal fauna were studied in July 1999 as part of the Reyðarál (2001) EIA to document the species and communities that exist near the future plant site. The shallow subtidal and intertidal biotic community at the site, especially in the inner fjord, was found to be similar to the community in other areas in Reyðarfjörður. The gastropod species *Littorina saxatilis* and the barnacle *Semibalanus balanoides* were found in the greatest abundance. The mussel *Mytilus edulis* was found moderately distributed, but was not found in great amounts at each sampling station. The most prominent flora is marine algae. Species such as *Fucus distichus* and *Fucus vesiculosus* were abundant. The red algae *Mastocarpus stellatus* was found at only a single station within the study site. This species is typically found in warmer waters on the south, west, and north coasts of Iceland, indicating that Reyðarfjörður may be at the edge of the water temperature range for this species on the Icelandic coast.

Marine Flora and Fauna

Marine flora and fauna studies were conducted in the summer of 2000, also as part of the Reyðarál (2001) EIA. Results of the surveys indicated that there was little phytoplankton biomass in inner Reyðarfjörður. Dinoflagellates had a higher abundance than diatoms. A total of 28 diatom species were identified, with the most abundant being *Leptocylindrus danicus, Nitzschia closterium, Pseudonitzschia pseudodelicatissima,* and *Pseudonitzschia seriata*. The prominent autotrophic dinoflagellates were *Heterocapsa triquetra* and *Scrippsiella trochoidea*. The dominant coccolithophorid species were *Emiliania huxleyi* and *Coccolithus pelagicus* (Reyðarál 2001).

The biomass of zooplankton was greater in the middle of the fjord (approximately 4.5 g/m^3 dry weight) than in the inner and outer areas (approximately 2 to 3 g/m^3 dry weight). The most dominant zooplankton taxa were copepods (>80 percent) and cirripeds (10 percent). A total of 19 species and taxonomic groups were identified, with copepods *Acartia longiremis* and *Oithona* spp. the most abundant species. In general, the biomass and abundance of zooplankton in Reyðarfjörður are similar to other fjords and bays in Iceland (Reyðarál 2001).

Benthic flora and fauna were studied in the summer of 1998 offshore of the proposed plant site. The most abundant benthic species found were bristle worms. Less-abundant species included bivalves such as *Laminaria saccharina* and *Chorda filum*, green sea urchin (*Strongylocentrotus droebachiensis*), Iceland scallop (*Chlamys islandica*), sea cucumber (*Cucumaria frondosa*), starfish (*Asterias rubens* and *Solaster* sp.), gastropods (*Neptunea despecta*), flatfish (*Limanda limanda*), and starry ray (*Raja radiata*). In 2000, another study of benthos was undertaken. Results indicated that of the 101 species that were identified, bristle worms were again the most abundant (constituting 74.4 percent of the animals caught). The bristle worm *Chaetozone setosa* had the highest single species abundance of 17.6 percent (Reyðarál 2001). The number of

benthic species per station ranged from 23 to 53, and in total, 101 different species were identified.

In 2000, PAH concentrations were measured in blue mussels (*Mytilus edulis*) within Reyðarfjörður. The concentrations of individual PAHs in blue mussels were below the detection limit (0.5 µg/kg) in 9 of 14 samples. Concentrations in the other 5 samples were low, with a mean concentration of 1.8 µg/kg. In Norwegian fjords and coastal waters, PAH levels in the soft tissue of blue mussels below 50 µg/kg are classified as not polluted. Thus, levels seen in mussels reflect very low background levels of PAHs.

Most of the common shallow-water fish species that are known to be present in Icelandic waters have been observed in Reyðarfjörður. Stock assessment studies at Reyðarfjörður revealed that the main fish present are haddock (*Melanogrammus aeglefinus*), cod (*Gadus morhua*), and long rough dab (*Hippoglossoides platessoides*). The study determined that Reyðarfjörður is an important nursery ground for young cod, but not for Atlantic herrings (*Clupea harengus*). Stock assessment surveys indicate that shrimp (*Pandalus borealis*) and Iceland scallop populations are insufficient to sustain a commercial harvest (Reyðarál 2001).

Although there is no information available for occurrence of whales near the study site, killer whales (*Orcinus orca*) are known to follow herrings into fjords, which indicates that there is a possibility that they may use Reyðarfjörður if the whales winter in the area. Seals such as the common seal (*Phoca vitulina*), grey seal (*Halichoerus grypus*), bearded seal (*Erignathus barbatus*), and ringed seal (*Phoca hispida*) have been observed in Reyðarfjörður (Reyðarál 2001). Common seals breed in the fjord.

Assessment and Measurement Endpoints

U.S. Environmental Protection Agency (EPA) guidance states that ERAs should use site-specific assessment endpoints that address chemical-specific potential adverse effects to local populations and communities of plants and animals (U.S. EPA 1999). Following this guidance, assessment endpoints for this ERA were selected, taking into account their biological significance, their susceptibility to potential contact through direct or indirect exposure to CoPCs, and the availability of pertinent assessment models and toxicological information in the literature. The assessment endpoints for the ERA are:

- Aquatic invertebrate community abundance and population production
- Marine fish population abundance and community structure
- Aquatic wildlife population abundance.

Measurement endpoints provide the actual measurements used to evaluate attainment of each assessment endpoint. The measurement endpoints for this ERA (in relation to their respective assessment endpoints) are as follows:

Assessment Endpoint: Aquatic Invertebrate Community Abundance and Population Production

- **Measurement Endpoints**
  - Comparison of predicted PAH and benzo[a]pyrene (B[a]P) concentrations in sediment with available ecological screening criteria
  - Comparison of predicted PAH concentrations in sediment with no-effect and effect concentrations reported in studies of benthic invertebrate communities at other aluminum smelter sites
  - Comparison of predicted fluoride concentrations in water with effects thresholds reported in the scientific literature
  - Comparison of predicted pH of seawater with effects thresholds reported in the scientific literature.

Assessment Endpoint: Marine Fish Population Abundance and Community Structure

- **Measurement Endpoints**
  - Comparison of predicted PAH and B[a]P concentrations in water and sediment with effects thresholds reported in the scientific literature
  - Comparison of predicted fluoride concentrations in water with effects thresholds reported in the scientific literature.

Assessment Endpoint: Aquatic Wildlife Population Abundance

- **Measurement Endpoints**
  - Comparison of modeled dietary doses of PAHs extrapolated from predicted concentrations in water with toxicity reference values (TRVs)
  - Comparison of predicted fluoride concentrations in water with effects thresholds reported in the scientific literature.

Exposure Assessment

As indicated above, ecological receptors may be exposed to constituents in seawater scrubber effluent via direct contact, ingestion of constituents adsorbed to sediment or dissolved in seawater, and ingestion of contaminated prey items. This section presents estimates of exposure point concentrations for the receptors and pathways described above. These exposure estimates are used in the *Effects Assessment* section to evaluate risk to ecological receptors potentially exposed to the CoPCs.

Vatnaskil (2001) presents discussions of the transport and fate of constituents in effluent from the original Reyðarál plant, including modeling of 1) mass loading of PAHs to sediment, 2) water concentrations of PAHs and fluoride, and 3) pH of ambient water as a function of distance from the outfall. Vatnaskil (2005) provides an update of the modeling to reflect current Fjarðaál design. This section presents loading estimates using data for two other Alcoa facilities: Deschambault, Quebec, Canada, and Mosjøen, Norway, as a means of verifying the reasonableness of the predictions for Fjarðaál.

It is necessary to make several assumptions in developing loading predictions using the data sets for Deschambault and Mosjøen: 1) the production of emissions per ton of aluminum produced is constant through the typical range of production rates, 2) the relative concentration of constituents does not vary significantly among comparable modern facilities, and 3) the waste stream volumes for air emissions and seawater scrubbers are proportional to production rates. Assumptions 1 and 2 are reasonable given the standardization of production and emissions control practices at modern smelters operated by Alcoa. Assumption 3 was not independently verified by Exponent, but is consistent with the comparison EIA and both aerial and water dispersion modeling (e.g., Hönnun et al. 2002). Additional case-specific assumptions are detailed below.

Summary of Loading Estimates and Ambient Concentrations for Fjarðaál

To support the EIA for the original smelter design, Vatnaskil (2001) performed dispersion modeling to estimate transport and fate of wet scrubber effluent in the receiving waters of Reyðarfjörður. Vatnaskil (2005) presents a revision to their previous dispersion modeling in response to the final design production for Fjarðaál of 346,000 tpy. The main conclusions presented by Vatnaskil (2005) are as follows:

- Effluent is rapidly diluted in the receiving waters of the fjord, with at least 100-fold dilution occurring within 1 km of the shoreline.
- Fluoride concentration in the effluent is 1.51 mg/L, but that concentration is rapidly diluted, and typical background concentrations (1.3 mg/L) are exceeded in only a small area around the discharge outlet.
- Sulfur dioxide reacts with seawater to form sulfurous acid and sulfuric acid, which lower the pH of the sea below the background level of 8.2 in a small area around the discharge outlet.
- Wet scrubber effluent will discharge 15.8 kg/year of PAH-16[1] (0.17 μg/L) and 0.61 kg/year of B[a]P (0.006 μg/L).
- For particle size of 10 μm, the maximum predicted PAH-16 and B[a]P concentration in bottom sediments is 500 μg/kg.

[1] PAH-16 refers to the sum total concentration of benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, indeno[1,2,3-cd]pyrene, acenaphthene, acenaphthylene, anthracene, benzo[ghi]perylene, fluoranthene, fluorene, naphthalene, phenanthrene, and pyrene.

- The maximum expected PAH-16 concentration in water is 0.005 μg/L and the maximum B[a]P concentration is 0.0002 μg/L. In both cases, these concentrations are restricted to a small area around the outfall, and decline rapidly from extensive dilution in ambient seawater.

Loading and Concentration Estimates Based on Air Emissions from Deschambault Smelter

Air emissions from the Alcoa Deschambault smelter in Quebec, Canada, can be used to estimate theoretical concentrations of constituents in seawater scrubber water as a means of testing the reasonableness of the loading estimates presented by Vatnaskil (2005). For simplicity of analysis, it is assumed that constituents present in scrubbed air are transferred in their entirety to seawater during treatment (i.e., the scrubbers are 100 percent efficient). This simplifying assumption will tend to overestimate loading of constituents to the marine environment. Concentrations of constituents in air emissions (mg/m^3) are converted to concentrations in seawater scrubber effluent by the following procedure:

1. Calculate mass loading in emissions (mg/day) by multiplying the concentration in air emissions (mg/m^3) by the daily volume of air emissions (m^3/day)
2. Adjust for difference in production rate where the planned production rate at Fjarðaál plant (346,000 tpy) is 1.38 times the production rate at the Deschambault plant (250,000 tpy) at the time of the sampling
3. Wet scrubber discharge at the Fjarðaál plant will be at a rate of 260,400 m^3/day (2.604×10^8 L/day) (Vatnaskil 2005).

Table 1 presents Deschambault stack emissions data for August 2003 and corresponding theoretical seawater scrubber concentrations and loadings for Fjarðaál using the calculations described above. The extent to which these calculations represent conditions at the Fjarðaál smelter is affected by the following principal sources of uncertainty: 1) the Deschambault facility has anode baking furnaces, which are likely to contribute to PAH loads, 2) many PAHs were not detected in air emissions, yet it was assumed that they were present at concentrations equal to one-half the detection limit, and 3) it was assumed that all PAHs in air emissions would be entrained by the seawater scrubbers. All of these factors will tend to overestimate PAH releases to the marine environment from Fjarðaál. Using air data from the Deschambault smelter and the calculations presented above, PAH-16 and B[a]P concentrations in seawater scrubber effluent of the Fjarðaál plant are estimated at 0.19 μg/L and 0.0065 μg/L, respectively. Mass loading to the receiving environment is estimated at 18 kg/year for PAH-16, which includes 0.62 kg/year for B[a]P.

Loading and Concentration Estimates Based on Seawater Scrubber Effluent from Mosjøen, Norway, Smelter

Table 2 presents results of monitoring for both intake and discharge of seawater scrubber water for the Alcoa smelter at Mosjøen, Norway. This smelter is a pre-bake facility with a production capacity of 188,000 tpy. The facility did not have anode baking capacity at the time of the sampling; thus, the PAH characteristics and concentrations in the effluent should be a reasonable predictor of emissions characteristics for the Fjarðaál smelter. Using discharge data from the Mosjøen smelter, mass loading to the receiving environment is estimated at 32 kg/year for PAH-16, which includes 0.47 kg/year for B[a]P. As with the prior illustration using data for the Deschambault facility, this illustration is also likely to overestimate total PAH loading to the marine environment from Fjarðaál because many PAHs were not detected, yet they were assumed to be present at one-half the detection level and it was assumed that all of the PAHs in air emissions would become entrained in the scrubber water.

Exposure Assessment Conclusions

The loading estimates presented by Vatnaskil (2005) generally agree well with the calculations presented above. Estimated loadings of PAH-16 for Fjarðaál based on data for Deschambault and Mosjøen are 1.1 to 2.0 times higher than the estimate presented by Vatnaskil (2005). However, estimated loading of B[a]P for Fjarðaál based on data for Deschambault and Mosjøen are equivalent to or lower (0.76 for Mosjøen) than the estimate presented by Vatnaskil (2005). As noted above, PAH emissions are generally likely to be lower at Fjarðaál than either Deschambault or Mosjøen because these latter facilities contain sources of PAHs (e.g., anode baking and/or landfill leachate) that will not be present at Fjarðaál. All estimates of B[a]P emissions are very similar (i.e., within a factor of 1.3). These conclusions have implications for use of Fjarðaál dispersion modeling results, as discussed below.

Vatnaskil (2005) presents PAH-16 and B[a]P deposition rates (in mg/m^2-year) to sediment at the bottom of the fjord, based on settling of particles of either 1 or 10 μm in diameter. In the Reyðarál EIA, it is assumed that particles are initially small (approximately 1 μm) because this is the size range of particles that can penetrate the high efficiency fabric filters of the dry scrubbers. It is expected that these small particles will agglomerate into larger particles with higher settling velocities, resulting in less dispersal and therefore potentially higher PAH concentrations in localized bottom sediments of the fjord. In the EIA, a particle size of 10 μm is assumed to be a conservative estimate of the size of these larger particles. For purposes of this evaluation, results are presented for deposition rates of both 1 and 10 μm particles. This approach allows for risk estimates that encompass the most likely range of long-term PAH-16 and B[a]P concentrations in sediment. Figures 3 and 4, taken from the Vatnaskil (2005) report, show 1 and 10 μm particle size contour plots of PAH-16 sediment concentrations. Figures 5 and 6 show the sediment concentrations for B[a]P at 1 and 10 μm particle sizes. The area estimates presented in Figures 3–6 provide bounds on the extent of the fjord bottom sediments likely to fall into each concentration class if the Fjarðaál smelter operates wet scrubbers.

Effects Assessment

The following sections present the effects assessment to ecological receptors potentially exposed to predicted CoPC concentrations that were developed in the previous section. The effects assessment involves several components. First, predicted CoPC concentrations in sediments are compared to available screening criteria. Second, pertinent data from the scientific literature and investigations at other aluminum smelter sites are reviewed to provide an environmentally realistic perspective on the ecotoxicological relevance of predicted CoPC concentrations in environmental media. In addition, a simplified food-web model analysis is presented to evaluate risks of PAHs to wildlife.

Screening of PAHs

Initial evaluation to determine if effluent releases are of ecological concern is done by comparing concentrations of CoPCs in the waste stream with ecological screening values, as described below. Screening values are conservative benchmarks that are designed to be ecologically protective. The objective of a screening assessment is to identify potential chemical exposure pathways that may be present for ecological communities at the study area. These pathways are then evaluated to determine whether any of them can be eliminated from further evaluation on the basis of negligible potential for adverse ecological impacts. Two outcomes are possible from the screening assessment.

- Adequate information exists to conclude that ecological risks are negligible and, therefore, there is no need for further evaluation or risk management on the basis of ecological risk.
- The available information indicates that there is a potential for adverse ecological effects or is not adequate to determine whether such a potential exists. Consequently, a more thorough assessment is warranted.

The screening assessment does not result in quantitative risk characterization. Only the absence (not the presence) of elevated risk can be established by a screening assessment alone. If the possibility of elevated risk cannot be ruled out using screening approaches, further assessment is required for those exposure pathways and receptor communities.

Norwegian Screening Criteria

Iceland does not have ecological screening criteria that can be used to compare to chemical concentrations in marine sediment, therefore, comparisons must be made to other sets of national or international values. In the EIA for the Reyðarál smelter (Reyðarál 2001), estimated PAH-16 concentrations in sediments were compared to the Norwegian screening criteria for Classification of Environmental Quality and Degree of Pollution in fjords and coastal waters. Norwegian criteria are divided into the following five classifications:

- Class 1 (Good): <300 μg/kg
- Class 2 (Fair): 300–2,000 μg/kg
- Class 3 (Poor): 2,000–6,000 μg/kg
- Class 4 (Bad): 6,000–20,000 μg/kg
- Class 5 (Very bad): >20,000 μg/kg.

Class 1 represents background values based on samples collected in areas without point-source contamination, but not necessarily free of anthropogenic influence. Other classes do not appear to be determined based on toxicological thresholds, and it is not clear that the word descriptors associated with each category reflect any basis of severity of effects on ecological receptors. Figure 3 illustrates that long-term PAH-16 concentrations in sediment for particle size of 1 μm are expected to be within the range of Class 1 in all areas, including the area immediately adjacent to the outfall. These estimates can be considered as the lower bounds on predicted long-term PAH-16 concentrations in Reyðarfjörður. Figure 4 illustrates PAH-16 concentration estimates for particle size of 10 μm. These estimates can be considered as the upper bounds on predicted long-term PAH-16 concentrations in Reyðarfjörður. As indicated by Figure 4, approximately 0.12 km^2 (1,200 m^2) of sediment is predicted to have PAH-16 concentrations greater than 500 μg/kg, which exceed Class 1 (good) limits and fall within the range of the Class 2 (fair) limits. In addition, 0.19 km^2 of sediment will have concentrations between 200 and 500 μg/kg, the maximum of which falls within the Class 2 limits. Some indeterminate portion of this area would thus not exceed Class 1 limits. Sediment in the remaining areas of Reyðarfjörður is predicted to have PAH-16 concentrations well within the Class 1 classification.

Norway also has determined screening criteria for B[a]P using the same five descriptive categories as developed for PAH-16. The five classes for B[a]P are:

- Class 1 (Good): <10 μg/kg
- Class 2 (Fair): 10–50 μg/kg
- Class 3 (Poor): 50–200 μg/kg
- Class 4 (Bad): 200–500 μg/kg
- Class 5 (Very bad): >500 μg/kg.

Figures 5 and 6 depict predicted long-term concentrations of B[a]P in sediment for particle size 1 μm (considered lower bound) and 10 μm (considered upper bound). For particle size 1 μm, long term B[a]P concentrations are predicted to exceed Class 1 in an area of 0.086 km^2 immediately adjacent to the outfall. All remaining areas of Reyðarfjörður would fall within Class 1. For particle size 10 μm, B[a]P is expected to fall within Class 5 in an area of 0.0092 km^2 (9,200 m^2) immediately adjacent to the outfall and within Class 4 in an area of 0.089 km^2 adjacent to the outfall. An area of 0.3 km^2 would fall within Class 3 and an area of

12.9 km^2 would fall within Class 2. The remaining area of Reyðarfjörður would fall within Class 1.

Additional Screening Criteria

The Norwegian sediment criteria have limited use as screening values for this ERA because their toxicological basis is not known. Screening criteria have been developed by various other organizations for PAHs and/or for B[a]P, as shown in Table 3. Several different approaches have been used for setting sediment quality guidelines (SQGs). In some cases, a background sediment chemistry approach is used where the concentrations of chemicals in sediments are compared to concentrations in local or regional background areas outside of the direct influence of sources. In other cases, screening values are set based on the results of toxicity tests and represent various effect thresholds to sediment-dwelling marine organisms. The screening criteria listed in Table 3 are briefly summarized below.

Effects Range and Effects Level Approach—Effects range and effects level approaches (Long et al. 1995; Long and MacDonald 1998) have been widely used, particularly in the United States to set SQGs. The effects range approach is based on two values, the effects range-low (ER-L) and the effects range-median (ER-M). The effects level approach is based on the threshold effects level (TEL) and the probable effects level (PEL). Although similar in concept, and based on the same data compilations, there are several important differences in the approach for the two sets of screening values. The ER-L is calculated as the lower 10th percentile concentration based on only those samples that were classified as toxic by original investigators; samples deemed to be nontoxic are excluded. Because it is at the low end of the concentration range, the ER-L is considered to be the concentration at which adverse effects would start to be seen in sensitive species. The ER-M is the median concentration based on only samples classified as toxic by investigators.

The ER-L/ER-M approach excludes data from all sediment samples that were not considered toxic when calculating effects ranges. As such, this approach omits useful information from samples where no toxicity was observed, but where the concentration of a specific chemical may be greater than either its ER-L or ER-M. The effects level approach incorporates data from all samples, whether effects were observed or not. The TEL is calculated as the geometric mean of the 15th percentile concentration of the toxic effects data set and the median concentration of the nontoxic effects data set. The TEL represents the concentration below which adverse effects are rarely predicted to occur. The PEL is calculated as the geometric mean of the 50th percentile concentration of the toxic effects data set and the 85th percentile concentration of the nontoxic effects data set. The PEL represents the concentration above which effects are frequently expected to occur.

Apparent Effects Threshold—The apparent effects threshold (AET) relates chemical concentrations in sediment to synoptic biological indicators of injury (i.e., sediment bioassays or diminished benthic faunal abundance). Individual AETs for a chemical are equivalent to the highest concentration of that chemical measured in a sediment sample considered to be nontoxic. As such, the AET represents the chemical concentration above which biological effects would always be expected to occur due to exposure to that chemical alone.

Canadian Environmental Quality Guidelines—Canadian SQGs are formulated on the basis of biological-effect data of sediment-associated chemicals and are intended to be used as nationally consistent benchmarks (CCME 2002). During their implementation, however, guidance notes that allowance must be made for the incidence of natural inorganic and organic substances in sediments. Therefore, the potential for adverse biological effects as indicated by the exceedances of SQGs must be evaluated in conjunction with other information such as the natural background concentrations of substances. In some management scenarios, it may also be necessary to consider concentrations of ubiquitous organic chemicals (i.e., the low level contamination of certain substances that are found throughout many environmental compartments) that are representative of reference or "clean" sites. Canadian guidelines provide two thresholds: the interim SQG, which is equivalent to a TEL, and the PEL.

Netherlands Environmental Quality Standards—The Netherlands uses a two-class system to develop SQGs based on ecotoxicological data or equilibrium partitioning (ICES 2003). The maximum permissible concentration (MPC) represents the concentration of a chemical in sediment that should protect all species in the ecosystem from adverse effects. The MPC represents the fifth percentile of a species sensitivity distribution of no-observed-effects concentrations (NOECs) based on data for four or more species from at least four different taxonomic groups. An equilibrium partitioning approach is used to calculate the MPC if there are insufficient toxicological data. The negligible concentration represents a concentration that causes negligible effects to ecosystems, and is derived by dividing the MPC by a factor of at least 100 to take into account potential synergistic effects.

OSPAR Background Concentrations—The Oslo and Paris Conventions for the Prevention of Marine Pollution (OSPAR) have set agreed-upon concentrations for chemicals in various marine locations that represent background levels for areas not directly influenced by sources (ICES 2003). As part of its Joint Assessment and Monitoring Program (JAMP), OSPAR has also developed agreed-upon ecotoxicological criteria for chemicals. These values do not represent background concentrations, and have no legal significance, but are used only for preliminary assessment of chemical monitoring data with the aim of identifying potential areas of concern.

Screening Comparison Results

The only criteria available to screen sediment PAH-16 concentrations are the effects levels and ranges. As shown in Table 3, the lowest value for this set of screening criteria is a TEL of 1,684 μg/kg. This value is greater than the maximum sediment concentration for PAH-16 (500 μg/kg) stated in the Vatnaskil (2005) modeling report. This assessment uses a conservative upper bound sediment concentration of 500 μg/kg. This value would indicate that there is no exceedance of the TEL. The maximum estimated sediment concentration is also lower than the PEL, ER-L, or ER-M. Because the TEL represents the concentration below which adverse effects are rarely expected to occur, these screening results suggest that predicted concentrations of PAH-16 in sediment are unlikely to be of ecotoxicological concern. There is some uncertainty associated with this conclusion because of imprecise understanding of the maximum sediment PAH-16 concentration. This uncertainty is discussed in greater detail in the *Risk Characterization* section.

The maximum sediment concentration for B[a]P presented by Vatnaskil (2005) is 500 μg/kg in a small area (0.0092 km^2) near the outfall. Concentrations ranging from 200 μg/kg to 500 μg/kg affect only 0.089 km^2. The maximum concentration of 500 μg/kg is lower than the PEL, ERM, and AET, but is greater than both the TEL and ERL. The maximum calculated concentration is also less than the Dutch MPC, which represents the chemical concentration that should protect all species from adverse effects. The maximum calculated concentration exceeds the OSPAR JAMP background concentration for the Arctic Ocean (100 μg/kg), that OSPAR has established for preliminary identification of areas of ecotoxicological concern. However, the majority of the area sediment (24 km^2) has a predicted concentration below 200 μg/kg. Together, these results indicate that predicted B[a]P concentrations in sediment are likely to be below concentrations that would be of ecotoxicological concern, with the possible exception of a small area in the immediate vicinity of the outfall.

Screening of Fluoride and Sulfur Dioxide

In addition to PAHs, fluoride and sulfur dioxide will also be discharged in effluent if wet scrubbers are used at the Fjarðaál smelter. As discussed above, heavy metals are not anticipated to be discharged in effluent because the only source of these chemicals, leachate from spent pot liner disposal areas, will not exist at Fjarðaál. Therefore, screening is not required for these chemicals. There are no suitable screening criteria to evaluate the ecotoxicological significance of fluoride or sulfur dioxide. Vatnaskil (2005) estimates that the fluoride concentration in discharge water will be 1.51 mg/L. However, as noted by Vatnaskil (2005), effluent discharge is rapidly diluted, such that beyond the immediate vicinity of the outfall, fluoride concentrations in ambient seawater are expected to reach background levels of 1.3 mg/L. Therefore, any exceedance of background concentrations is likely to be restricted to a very limited area immediately around the outfall. Norwegian screening criteria use the five-category procedure described above to evaluate fluoride concentrations in sediment. However, the information presented in the Vatnaskil modeling report is insufficient to estimate fluoride concentrations in sediment because no deposition rates are provided, and comparisons cannot be made with these criteria. The ecotoxicological significance of fluoride is discussed further in the next section of this report.

The main ecotoxicological concern related to sulfur dioxide discharges stems from its reaction with seawater to form sulfurous acid and sulfuric acid, resulting in a lowering of the marine water pH. Contour plots in the Vatnaskil (2005) modeling report indicate that pH will be depressed by only 0.1 pH unit below the background level in an area no greater than 0.0027 km^2 immediately around the discharge outlet. Therefore, although screening cannot be performed for this chemical, the likelihood of adverse ecotoxicological effects appears to be minimal. The ecotoxicological significance of changes in pH is discussed further in the *Risk Characterization* section.

Ecotoxicological Evaluation of PAHs and Fluoride

The screening comparisons presented above indicate that the likelihood of adverse ecotoxicological effects resulting from PAHs and fluoride in wet scrubber effluent discharged to

Reyðarfjörður is minimal. Screening criteria represent very conservative values that can be used to eliminate from further consideration chemicals and pathways where chemical concentrations are lower than their corresponding criterion. However, because of the conservative manner by which they are derived, exceedance of screening criteria does not necessarily imply that adverse effects to ecological receptors would occur. Therefore, in this section, findings from other sites and ecotoxicological investigations are reviewed to provide additional context to the environmental relevance of PAHs and fluoride in effluent discharges. Brief toxicological profiles of both CoPCs are also presented to provide an overview of the types of effects that have been observed in ecological receptors.

Polycyclic Aromatic Hydrocarbons

Toxicity Profiles

PAHs consist of hydrogen and carbon arranged in the form of two or more fused benzene rings. PAH compounds differ in the number and position of aromatic rings and in the position of substituents on the basic ring system. Environmental concern has focused on two-ring (naphthalene) to seven-ring (coronene) structures. Unsubstituted two- or three-ring PAHs are not carcinogenic but are the PAHs most commonly associated with acute toxicity. Four- to seven-ring unsubstituted PAHs are significantly less toxic than two- or three-ring PAHs, but some are carcinogenic, mutagenic, or teratogenic to a variety of organisms, including fish and birds.

Accumulation in aquatic organisms is largely related to the ability to metabolize the compounds. In general, PAHs show little tendency to biomagnify in food chains, primarily because they are rapidly metabolized. In aquatic food chains, fish have the best metabolic capacity to metabolize PAHs, crustaceans are intermediate, and molluscs have the poorest metabolic capacity (James 1989; Stegeman and Lech 1991). Species at lower trophic levels, such as plankton and molluscs, can accumulate PAHs in their tissues. However, even in contaminated areas, only low to moderate PAH concentrations are typically found in fish (Dawe 1990; Varanasi et al. 1990). Biomagnification to higher trophic levels does not occur to any practical extent.

The ability to biotransform PAHs is due to the cytochrome P-450 mixed function oxidase (MFO) system. The MFO system is well developed in many birds, mammals, and fish and allows PAHs to be readily metabolized (Kalf et al. 1995), and direct toxicity to these receptors is unlikely. MFO activity usually takes place in the liver, although activity has also been found in the heart and posterior kidney of rainbow trout (*Salmo gairdneri*), in the gills of brown trout (*Salmo trutta*), capelin (*Mallotus villosus*), and cunner (*Tautogolabrus adspersus*). The metabolites of PAHs include intermediates that can covalently bind to DNA, RNA, and proteins and become toxic, mutagenic, or carcinogenic. Metabolites include PAH diols, phenols, quinone, and PAH conjugates with sulfate, monosaccharide, glucoronate, and glutathione (Neff 1978). The toxicity of PAHs varies because PAHs and their metabolites exhibit different toxicodynamics. In some cases, the polar metabolites of PAHs are excreted more slowly than the parent compound (Kalf et al. 1995). For example, the metabolite of B[a]P, 7,8-diolepoxide, has a higher carcinogenic capacity than its parent (Kalf et al. 1995). Tests with Japanese

medaka (*Oryzias latipes*) showed that epoxides produced from the metabolism of B[a]P were ultimately carcinogenic (Hawkins et al. 1990).

Low levels of accumulated PAHs are found in fish because of mediation by the MFO system. A study by D'Adamo et al. (1997) showed that sea bass (*Dicentrarchus labrax*) increased its MFO activity as the presence of PAHs increased in muscle tissue. Varanasi and Stein (1991) reported that although B[a]P could not be detected in several test fish species, metabolites were detected. However, PAH metabolism varies with species and compounds. Amphipods such as *Rhepoxynius abronius* and *Eohaustorius washingtonianus* have the ability to metabolize B[a]P to intermediate compounds (Reichert et al. 1985). Tests with chironomids (*C. riparius*) demonstrated that B[a]P is rapidly and completely transformed (Giesy et al. 1983). English sole (*Parophrys vetulus*) collected from polluted sites in Puget Sound, Washington, were found to have metabolites of fluorene, dibenzofuran, phenanthrene, fluoranthene, pyrene, and B[a]P (Krahn et al. 1987). B[a]P, fluoranthene, and benzo[a]anthracene were metabolized in *Pimephales promelas, Poecilia reticulata*, and *Brachydanio rerio*, but naphthalene, anthracene, and phenanthrene were not (Kalf et al. 1995).

Several studies have demonstrated negative effects of PAH in water, especially B[a]P, on hatching, larval development, and viability, primarily with fish, but also with invertebrates in a few cases. Typical effects noted in fish were delayed or decreased hatching (Hose et al. 1981, 1982; Winkler et al. 1983; Hall and Oris 1991), increased incidence of larval malformations (Hose et al. 1981, 1982; Hannah et al. 1982; Winkler et al. 1983), cell and tissue lesions in yolk sac fry (Hose et al. 1984) or reduced larval growth (Hannah et al. 1982). In general, effects are seen only at environmentally unrealistic concentrations. For example, Hall and Oris (1991) found reductions in the number of eggs in fish exposed to anthracene at 6 μg/L for 6 weeks, and a lowering of hatching success at 12 μg/L. However, Hose et al. (1982, 1984) found effects at concentrations as low as 0.1 μg/L. These concentrations are still greater than the maximum concentrations likely to occur at Fjarðaál. Vatnaskil (2005) models show that the maximum expected PAH-16 concentration is 0.005 μg/L and the maximum B[a]P concentration is 0.0002 μg/L. In both cases, these concentrations are restricted to a small area around the outfall, but decline rapidly from extensive dilution in seawater.

A number of studies have been conducted on effects of sediment-bound PAHs on marine fish and other biota. These studies show that PAHs exhibit many of the same toxic effects in fish as they do in mammals, such as liver cancer and related lesions (Moore and Myers 1994, as cited in Johnson 2000). The National Oceanic and Atmospheric Administration (Johnson 2000) reviewed the literature on the effects of PAHs in the highly exposed bottom-dwelling species, English sole (*Pleuronectes vetulus*). Data from laboratory and field studies show that this highly exposed species is susceptible to the development of liver cancer and lesions as well as other adverse health effects such as reproductive abnormalities, immune dysfunction, and alterations in growth and development (Johnson 2000). Most of these data are from contaminated urban embayments including Elliot Bay and Commencement Bay in Puget Sound, Washington, and San Francisco Bay, California, USA. Although English sole do not occur in Iceland, several related species do, including plaice (*Pleuronectes platessa*), dab (*Limanda limanda*), long rough dab (*Hippoglossoides platessoides*), lemon sole (*Microstomus kitt*), and grey sole (*Glyptocephalus cynoglossus*).

Some of the effects that have been observed in English sole include neoplastic, preneoplastic, or unique degenerative liver lesions. In addition, reproductive effects have been reported in this species including lower plasma concentrations of 17β-estradiol, inhibited ovarian development, reduced egg production, suppression of estradiol-induced vitellogenin production. Exposure to PAHs has also been implicated in inhibited spawning ability and reduced egg and larvae survival in lab studies (Casillas et al. 1991, as cited in Johnson 2000).

Threshold concentrations of PAHs in sediment associated with such effects in English sole range from 54 μg/kg dry weight, for liver neoplasms, to about 2,800 μg/kg for hepatic preneoplastic foci of cellular alteration (Johnson 2000). Based on these data, the National Marine Fisheries Service (NMFS) suggests an SQG of 1,000 μg/kg to protect estuarine fish from adverse effects. However, it should be noted that this guideline is lower than PAH concentrations that are typically found in urban areas as a result of stormwater runoff and atmospheric deposition. The 1,000 μg/kg threshold also does not consider the contribution of co-occurring chemicals in the sediment from areas in which the fish were caught.

Evaluation of PAHs at Aluminum Smelters

A number of studies have evaluated the ecotoxicological effects of PAHs in marine waters and sediment at aluminum smelter sites. These studies are briefly reviewed here to provide some perspective on effects that could occur at Fjarðaál. However, differences between smelters with regard to production technologies and capacities must be considered, such that results observed at other smelter locations may not be directly applicable to Fjarðaál. This issue is addressed later in the uncertainty analysis.

Næs et al. (1995) conducted a comprehensive review of the occurrence of PAHs in organisms and sediments in Norwegian fjords and coastal waters where smelter discharges occurred. Most of the aluminum smelters they evaluated use Söderberg production technology and also produced anodes onsite. Neither of these features is part of the design for Fjarðaál. They reported high concentrations of PAHs in effluent. For example, at the smelter at Lista, the total PAHs were 7.9 μg/L and at the smelter at Vefsnfjord, the total PAHs were 4.8 μg/L. By comparison, the Reyðarál (2001) EIA estimated a maximum concentration of 0.32 μg/L for total PAHs in effluent. PAH concentrations in sediments ranged up to almost 800,000 μg/kg dry weight, which exceeds the maximum calculated concentration for the Fjarðaál smelter by more than 1,600-fold. Extensive accumulation occurred in some marine fauna. Total PAH and B[a]P concentrations in blue mussels, horse mussels, periwinkles, and limpets collected within 1–2 km of scrubber outfalls were as much as 3,000–5,000 times higher than concentrations in organisms collected at background locations. The authors also found that there was considerable variation in concentrations in littoral indicator species under constant load conditions, which could be due to fluctuating currents, variable waste discharges, spawning, or other biological factors.

Næs et al. (1995) did not report on the ecotoxicological relevance of PAHs measured in media and fauna. However, Knutzen (1995) performed a comprehensive review of effects on marine organisms from PAHs, fluoride, and pH in wastewater from aluminum smelters in Norway, including some of the smelters studies by Næs et al. (1995) and found that exposure does not necessarily imply adverse effects. For example, Knutzen noted that despite the extensive accumulation noted in mussels, there has been no indication of adverse effects on mollusc

populations themselves, or on populations of birds that prey on these species. Knutzen extensively investigated the smelter at Lista, which was briefly noted above, reviewing about 20 years of monitoring data for the site. The annual PAH load in the waste stream at Lista was estimated at approximately 4 tons. By contrast, the maximum PAH load for Fjarðaál is estimated to be 15.8 kg/year (Vatnaskil 2005). Soon after the Lista smelter became operational, snails and some benthic algae disappeared from stations close to the outfall, although some recolonization occurred in later years. However, Knutzen noted that differences in community composition did not appear to correlate with the waste stream concentration gradient, and he concluded that physical factors associated with the waste stream (smothering and scouring of shallow bottom sediments) were more important determinants of community composition than were chemical concentrations. Similarly at the Årdal smelter, Knutzen noted high tolerance of soft bottom fauna to PAH concentrations up to 400,000 μg/kg, which greatly exceeds the concentrations predicted for Fjarðaál. The lack of organisms in a zone approximately 0.5 km^2 around the outfall, where PAH concentrations in surface sediment reached 800,000 μg/kg, was concluded to be the result of physical factors, not chemical factors.

Unlike the studies stated above, Oug et al. (1998) did note changes in soft bottom fauna community composition along PAH gradients in four fjords receiving effluents from aluminum smelters. The study integrated benthic community data from monitoring programs in Norwegian fjords using canonical correspondence analysis, a multivariate direct gradient technique, to relate species patterns to environmental variables. PAH concentration gradients accounted for 5–10 percent of total variance in species patterns, which although low, was statistically significant. The PAH concentrations ranged from background levels up to 760,000 μg/kg at the most heavily polluted location. Species composition changed along the PAH concentration gradient, particularly the distribution of carnivorous polychaetes. At low PAH levels (approximately 5,000 μg/kg), the carnivores represented 15 to 20 percent of the individuals. The proportion began to increase at a PAH concentration in sediment of about 10,000 μg/kg, and by 100,000 μg/kg, the carnivores represented more than 50 percent of the community. The authors concluded that these results suggest that smelter effluents favor carnivorous species, or conversely, are more detrimental to deposit feeders. The authors stated that this could be because carnivorous polychaetes are mobile and may be less exposed than sessile deposit feeders, and that carnivores are also less dependent on sediments for food and shelter. The trend appeared to be most prominent in shallow waters and at moderate depths. There was also a significant inverse correlation between echinoderm abundance and PAH concentrations, suggesting that these species may generally avoid PAH-contaminated areas, whereas other groups were more evenly dispersed. The community changes occurred at concentrations below those where total number of species and community diversity were affected, and at which detrimental effects became obvious (i.e., concentrations at the most contaminated stations). The authors caution that patterns illustrated in the study are based on correlations, and do not necessarily indicate cause and effect, and that stressors other than PAH could be responsible for observed effects.

Paine et al. (1996) conducted a sediment quality triad study (sediment chemistry, sediment bioassay, benthic community analysis) to assess impacts from discharges at an aluminum smelter in Kitimat Arm, a coastal inlet in British Columbia, Canada. Total PAH concentrations in sediment at study locations ranged from 1,300 to 9,890,000 μg/kg, and B[a]P concentrations ranged from <140 to 1,100,000 μg/kg. No sediment toxicity was observed despite extensive

testing with echinoderm and amphipod species. No consistent effects were observed on benthic communities. Although invertebrate abundance was lower at stations near the smelter outfall than at a reference location, species richness (number of species) was actually higher, which runs contrary to trends generally observed at contaminated sediment sites. The study concluded that differences between site stations and reference area stations were probably not contaminant effects but may have resulted from differences in physical factors, particularly wave exposure.

Paine noted that PAH concentrations (up to approximately 10,000,000 μg/kg) were well in excess of concentrations that have been associated with significant effects in toxicity tests or benthic community structure at other PAH-contaminated sites. Paine concluded that the lack of toxicity or benthic community effects at Kitimat were probably attributable to limited bioavailability of PAHs, due to their being bound in pitch and coal particles. The low concentration of parent PAHs and metabolites in Dungeness crabs (*Cancer magister*) was also cited to support the hypothesis of limited bioavailability. Paine et al. (1996) concluded that high PAH concentrations at aluminum smelter sites may not be associated with adverse effects if availability is limited. Knutzen (1995) also noted the discrepancy between PAH toxicity thresholds from the literature and the minimal adverse effects observed at Norwegian smelter sites, and attributed this in part to limited availability of PAHs.

In summary, the aluminum smelter studies cited above have not shown a conclusive correlation between sediment PAH concentrations and effects on benthic invertebrate toxicity or community structure. The sediment PAH concentrations at some of the locations evaluated in these studies are substantially higher than the predicted maximum concentrations at Fjarðaál. These results support the screening evaluation described above, and provide further evidence that the likelihood of adverse effects from PAHs would be minimal if wet scrubbers were used at Fjarðaál.

Evaluation of Risks to Wildlife from PAHs

As discussed above, some invertebrates, such as molluscs, can accumulate PAHs because of their lesser ability to metabolize these compounds compared with other aquatic species. Wildlife species that forage on molluscs, such as blue mussels that occur in the fjord, could potentially be exposed to PAHs through their diet. For example, shorebirds foraging along the coast in the vicinity of the Fjarðaál smelter might be exposed to PAHs in prey items.

To evaluate risk to avian receptors that feed on marine organisms, a food-web exposure model was developed to predict the amount of PAHs to which birds might be exposed while feeding in the vicinity of the plant site. Two avian receptors were evaluated: the Eurasian oystercatcher (*Haematopus ostralegus*) and the herring gull (*Larus argentatus*). The oystercatcher is a summer breeder on the east coast of Iceland.[2] It feeds almost exclusively on shellfish and other marine invertebrates in intertidal areas by wading and stabbing at its prey (Nol and Humphrey 1994). The herring gull is a more opportunistic feeder, ingesting fish, squid, crustaceans, molluscs, worms, insects, small mammals and birds, duck and gull eggs and chicks, and garbage. Gulls forage on open water by aerial dipping and shallow diving in areas where prey are concentrated (U.S. EPA 1993).

[2] (http://www.birdguides.com/html/vidlib/species/Haematopus_ostralegus.htm#)

The food-web modeling approach that was used is a standard approach consistent with EPA's wildlife exposure guidance (U.S. EPA 1993; 61 Fed. Reg. 47552). The food-web model estimates dietary exposure as a body-weight-normalized total daily dose for each receptor species. The general structure of the food-web exposure model is described by the following equation:

$$IR_{chemical} = \frac{\sum_i \left(C_i \times M_i \times F_i\right)}{W}$$

where:

$IR_{chemical}$	=	total ingestion rate of chemical from all dietary components (mg/kg body weight-day)
C_i	=	concentration of the chemical in a given dietary component or environmental medium (mg/kg)
M_i	=	rate of ingestion of dietary component or medium (kg/day)
F_i	=	fraction of the daily intake of a given dietary component or medium derived from the outfall area (unitless area-use factor)
W	=	body weight of receptor species (kg).

The term $IR_{chemical}$ can be expanded to specify each ingestion medium, which includes one or more primary food items, drinking water, and incidentally ingested sediment:

$$IR_{chemical} = [\Sigma\ (C_{food} \times M_{food} \times F_{food}) + (C_{water} \times M_{water} \times F_{water}) + (C_{sed} \times M_{sed} \times F_{sed})]/W$$

This model provides an estimated total dietary exposure for PAHs and B[a]P resulting from the consumption of food and water and the incidental ingestion of sediment on a mg/kg body weight-day basis. For both avian species, the exposure calculation assumes that the entire diet for all receptors comes from the outfall area (F_i = 1) and that 100 percent of the chemical in ingested food is absorbed. The diet of the oystercatcher and gull was conservatively assumed to be 100 percent mussels obtained from the proposed outfall area. These conservative assumptions regarding avian receptors' exposure likely overestimate the potential for exposure, but the use of such conservative assumptions is appropriate in a predictive screening assessment such as this to reduce the possibility of drawing false negative conclusions regarding ecological risk.

The exposure parameters for oystercatcher and gull are provided in Table 4. The mean adult body weight was used in the exposure calculations. Food and water ingestion rates were calculated using allometric equations by Nagy (1987) and Calder and Braun (1983) (Table 4). The proportion of sediment incidentally ingested was estimated based on the analysis of acid-insoluble ash in waterfowl scat by Beyer et al. (1994). In the absence of measured sediment in the diet of the two receptor species, the estimated value for woodcock (10.4 percent) was used for the sediment fraction of the oystercatcher diet and the estimated value for ducks (2 percent)

was used for the sediment fraction of gull diet. Oystercatchers and woodcocks both feed on invertebrates by probing in the substrate, and gulls and ducks are open-water feeders. It was conservatively assumed that the entire prey component of the receptors' diet consisted of mussels and that the birds are residents of coastal Iceland for the entire year.

To estimate exposure to these receptors, it is first necessary to estimate concentrations of PAHs in prey items resulting from bioaccumulation. Bioaccumulation of PAHs from water by marine organisms is directly proportional to the K_{ow} of the individual PAH. K_{ow} for PAHs increase with increases in molecular weight, with log K_{ow} ranging from 3.33 to about 7.0 (Neff 2002). Bioaccumulation of PAHs is well-studied for bivalve molluscs. For bivalves, estimated bioconcentration factors (BCFs) based on regression analysis range from 65.1 for naphthalene to 226,000 for indeno[1,2,3-cd]pyrene (Neff 2002). BCFs as high as these estimated values are rarely attained under natural field conditions because they are based on long-term exposures to pure compounds in solution (i.e., ideal laboratory situations where equilibrium can be reached). Under natural conditions, measured BCFs are usually found to be much lower than predicted values. For this assessment, the average bivalve BCF from those reported by Neff (2002), 34,057 (Table 5), was used as the BCF for total PAHs. The BCF for B[a]P was reported as 26,800. BCFs were used to predict the mussel concentrations of PAHs using the following formula:

$$BCF \times C_{sw} = \text{Mussel Tissue Concentration}$$

C_{sw} is the predicted surface water concentration, which is 0.005 μg/L for total PAHs and 0.0002 μg/L for B[a]P. These concentrations are the highest ones predicted by Vatnaskil (2005) for the area around the outfall that extends to the shoreline where exposure may occur to avian receptors.

Avian Toxicity Reference Value—The exposure estimates from the food-web models are compared to threshold, or benchmark, TRVs for each receptor to develop quantitative risk estimates or hazard quotients. Hazard quotients are the ratio of the predicted exposure concentrations to the TRV:

$$HQ = \frac{IR_{chemical}}{TRV}$$

where:

HQ	=	hazard quotient (unitless)
$IR_{chemical}$	=	ingestion rate of the chemical (mg/kg body weight-day)
TRV	=	toxicity reference value.

Hazard quotients less than 1.0 indicate that the chemical is unlikely to cause adverse ecological effects. Hazard quotients exceeding 1.0 indicate some potential for adverse ecological effects. Hazard quotients exceeding 1.0 do not necessarily signify unacceptable risk. Other pieces of

information, such as sources of uncertainty and site-specific exposure information, are weighted in the risk evaluation and interpretation of hazard quotients.

TRVs are derived from the primary toxicology literature. TRVs are expressed as a daily dietary dose, and are calculated from dietary exposure endpoints according to the following general formula:

$$TRV = ([diet] \times IR)/BW$$

where:

TRV	=	toxicity reference value for no-observed-adverse-effect level (NOAEL) or lowest-observed-adverse-effect level (LOAEL) endpoints (mg/kg body weight per day)
[diet]	=	dietary concentration (mg/kg food or mg/L drinking water) associated with a given endpoint
IR	=	daily ingestion rate (kg food/day or L drinking water/day)
BW	=	body weight (kg).

Uncertainty factors are applied only when necessary to estimate a chronic NOAEL from other reported endpoints:

- LOAEL to NOAEL; uncertainty factor = 0.1
- Subchronic (e.g., < 10 weeks and through a non-critical lifestage) to chronic exposure; uncertainty factor = 0.1
- Acute (e.g., single dose, LD50) to chronic exposure; uncertainty factor = 0.01.

A study by Hough et al. (1993) on the effects of B[a]P in pigeons was used to develop the TRV for birds. This study evaluated atherosclerosis, hepatic biomarkers, and reproductive effects over a 5-month period where the birds were injected with B[a]P in corn oil. A dose of 10 mg B[a]P per kg body weight per week was shown to cause reproductive effects (cessation of egg-laying and gross deleterious ovarian changes) in exposed birds. This weekly dose equates to a daily dose of 1.43 mg/kg-day, or the LOAEL for reproductive effects. Applying the LOAEL to NOAEL uncertainty factor, the resulting TRV is 0.143 mg/kg-day.

Table 6 provides the model results for oystercatcher and herring gull exposure to total PAHs and B[a]P. Average daily ingestion estimates of these chemicals for both avian species were less than the NOAEL and LOAEL doses for reproductive effects (hazard quotients less than 1.0), indicating that even under very conservative exposure assumptions, PAHs are unlikely to cause adverse effects in avian species exposed through their diet.

Fluoride

Fluoride is highly reactive and does not occur in its elemental state in nature. Inorganic fluorides are much more abundant in nature than organic fluorides. In water, inorganic fluorides usually remain in solution as fluoride ions. Aquatic organisms living in soft water may be more affected by fluoride than organisms living in hard water or seawater because fluoride ion bioavailability decreases with increasing water hardness (Camargo 2003). Fluoride concentrations in unpolluted marine waters generally range from 1.2 to 1.5 mg/L, although higher levels can be found in areas of geothermal or volcanic activity.

The mechanism of fluoride toxicity to algae and aquatic plants results from fluoride ions affecting metabolism of nucleotides and nucleic acids that govern cell division. Some algal species, however, show growth stimulation in the presence of fluoride, indicating a requirement of fluoride for optimal growth. Oliveira et al. (1978) exposed 12 species of marine phytoplankton to fluoride at concentrations of 0 to 100 mg/L for exposure times up to 25 days. Three species showed a 25–30 percent inhibition in growth at 100 mg/L, one species showed stimulated growth at concentrations of 25–100 mg/L, but eight species showed good growth without evidence of growth enhancement or inhibition. Antia and Klut (1981) exposed five marine phytoplankton algae to fluoride at concentrations of 50–200 mg/L for up to 36 days. Two species showed growth inhibition ranging from 25 to 90 percent at fluoride concentrations greater than 150 mg/L. Klut et al. (1981) found that while the marine dinoflagellate *Amphidium carteri* was initially unable to grow on seawater with 200 mg/L, gradual increases in fluoride concentrations resulted in the species becoming adapted to this concentration.

For fish and aquatic invertebrates, exposure via water is the main route of fluoride uptake, and uptake via food represents a minor exposure pathway. Although some fluoride is excreted, much can become accumulated in the exoskeletons of invertebrates or the bone tissue of fishes. Wright and Davison (1975) found that marine crustaceans inhabiting an area close to an aluminum smelter, where the fluoride concentration in seawater was 3.4 mg/L, accumulated fluoride in their exoskeletons. Estuarine species can accumulate relatively high amounts of fluoride in their bodies. For example, Hemens and Warwick (1972) found concentrations ranging from 1,414 to 7,743 μg/g dry weight in crustaceans and fish exposed for 72 days to 52 mg/L.

Fluoride toxicity to aquatic animals results from the chemical acting as an enzyme inhibitor, eventually resulting in interruption of metabolic processes such as glycolysis and protein synthesis (Camargo 2003). Fluoride appears to cause effects by interacting with calcium, which is required as a cofactor for proper functioning of many enzymes. Studies in fish have shown that fluoride toxicity decreases with increasing calcium concentrations in water. Estuarine and marine organisms may, therefore, be more tolerant to fluoride toxicity than freshwater organisms because of the higher concentration of calcium in marine waters (Hemens and Warwick 1972). For example, on the basis of acute mortality data, a safe concentration of fluoride in water for some freshwater organisms is around 1 mg/L. However, some marine species appear to be tolerant to concentrations of 100 mg/L, even after a 96-hour exposure.

The toxicity of fluoride to marine and estuarine fishes has not been well studied. In general for fish, toxicity appears to increase with increasing fluoride concentration, exposure time and

water temperature. Increasing intraspecific fish size and water hardness appear to decrease toxicity. Among freshwater fishes, the rainbow trout (*Oncorhynchus mykiss*) appears to be among the most sensitive species tested. For this species, Pimentel and Bulkley (1983) have proposed maximum chronic exposure levels of between 2.5 mg/L, at a water hardness of 17 mg $CaCO_3$/L, and 9.6 mg/L, at a water hardness of 385 mg $CaCO_3$/L. Evidence exists that physiological and genetic adaptation can occur in the wild to elevated fluoride concentrations. Damkaer and Dey (1989) found that upstream migrating salmon in the Columbia River were adversely affected by water concentrations as low as 0.2 mg/L. However, Sigler and Neuhold (1972) found that fish living in a river in Yellowstone National Park with naturally elevated fluoride concentrations were tolerant of concentrations as high as 14 mg/L. Limited studies on marine and estuarine fish indicate that they are much less sensitive to fluoride than freshwater fish. Hemens and Warwick (1972) indicated that three estuarine fish species exhibited no mortality after 96-hour exposure to concentrations as high as 100 mg/L. Hemens et al. (1975) found that striped mullet *Mugil cephalus* tolerated an exposure of 5.5 mg/L for 113 days without mortality or physical deterioration of juveniles. Also, Heitmuller et al. (1981) found a NOEC of 226 mg/L for juvenile sheepshead minnow (*Cyprinodon variegatus*) after a 4-day exposure.

Marine mammals and birds are not likely to be highly exposed to fluoride in seawater because ingestion of saltwater is minimal and usually incidental to food ingestion. Furthermore, high concentrations in water are required before adverse effects are observed. Merkley and Sexton (1982) provided hens drinking water with 100 mg/L fluoride and found no reduction in egg fertility or hatching success. Eggshell thickness was not affected in quail exposed to 50 mg/L fluoride in drinking water. Taylor et al. (1961) estimated the 30-day LC_{50} for rats to be 205 mg/L. Boulton et al. (1994) noted that 80 mg/L caused 50 percent mortality in field voles (*Microtus agrestis*) after a 25-day exposure, but no mortality in laboratory mice. All these concentrations are environmentally unrealistic and at least 20- to 25-fold higher than concentrations predicted in the waters of Reyðarfjörður.

Marine wildlife may be exposed to fluoride in prey, for example accumulated in the bones of fish. As with water exposure, high concentrations are required to cause effects. For example, Patee et al. (1988) found that screech owls (*Otus asio*) showed decreased productivity when fed a diet supplemented with 200 mg/kg fluoride. However, starlings (*Sturnus vulgaris*) consuming up to 360 mg/kg showed no reproductive effects (Fleming 1996).

In summary, studies cited show that fluoride can elicit adverse effects in marine organisms, including plankton, aquatic plants and invertebrates, and fish. However, the majority of adverse effects in marine organisms are observed only at environmentally unrealistic concentrations, typically 100 mg/L or higher. Similar concentrations would potentially be required to cause effects in marine mammals and birds. The maximum fluoride concentration in waters of Reyðarfjörður is much lower than this concentration, indicating that the likelihood of adverse effects from fluoride would be minimal if wet scrubbers were used at Fjarðaál.

Risk Characterization

The evaluation presented above examines the likelihood that chemicals in wet scrubber effluent from the Fjarðaál smelter would produce adverse effects in populations or communities of

ecological receptors inhabiting the waters or sediments of Reyðarfjörður. The risk assessment used information from the Reyðarál EIA (2001) and effluent dispersion modeling (Vatnaskil 2005) to estimate the concentrations of chemicals (PAHs, fluorides, and sulfur dioxide) emitted in effluent and their ultimate distribution within the fjord. In addition, emission data from other aluminum smelters was used, when possible, to ground-truth estimated loadings for Fjarðaál based on values estimated using the original Reyðarál smelter design. Predicted chemical concentrations in sediment were then screened against available SQGs. However, because these guidelines represent very conservative thresholds for the onset of adverse ecological effects, sediment concentrations were also contrasted with biological exposure and effects data from marine environments receiving effluent from other aluminum smelters. This comparison provides a more realistic evaluation of the likely ecological effects that might be associated with the operation of wet scrubbers at Fjarðaál. In this section, the results of these comparisons are presented to characterize the risk to ecological receptors of Reyðarfjörður. In addition, major uncertainties that may affect the risk characterization are also presented and their potential effects on risk estimates are described.

Evaluation of Risk to Aquatic Receptors and Wildlife from Polycyclic Aromatic Hydrocarbons

The predicted sediment concentrations of PAH-16 and B[a]P were compared to available sediment quality criteria for marine or coastal waters. The most stringent criteria are the Norwegian values for Classification of Environmental Quality and Degree of Pollution. Under this scheme, a total PAH concentration of 300 μg/kg is considered to represent the threshold for degradation in sediment quality, because this value represents the upper limit on background concentrations. Predicted concentrations of PAH-16 in sediment (500 μg/kg), based on the maximum cited in Vatnaskil (2005; for particle size 10 μm), exceed this most conservative criterion. The exact spatial extent of the exceedance cannot be determined. There is an area of approximately 0.12 km^2 (> 500 μg/kg) where this threshold is exceeded, and an indeterminate extent of area of 0.19 km^2 where predicted concentrations (200–500 μg/kg) straddle the threshold, but for which the Vatnaskil model does not permit more precise delineation of sediment concentrations. The predicted maximum sediment concentration (500 μg/kg), if accurate, would mean that an indeterminate extent of sediment no greater than 0.31 km^2 would be classified as "fair" as per the generic descriptors applied to the Norwegian criteria. The Norwegian classification system considers 10 μg/kg to represent the threshold B[a]P concentration, above which sediment degradation is assumed to occur. Based on Vatnaskil (2005), the maximum predicted sediment B[a]P concentration (for particle size 10 μm) is 500 μg/kg, which exceeds this threshold over an area of approximately 13.2 km^2, but the majority of this area (12.9 km^2) falls within Class 2 (fair). In conclusion, modeling of sediment PAH-16 and B[a]P concentrations indicates some exceedance of the most stringent sediment criteria.

Nonetheless, comparison of predicted sediment PAH-16 concentrations to other sediment quality values indicates that the likelihood of adverse effects to benthic invertebrate ecological receptors is low, with the possible exception of a small area in the immediate vicinity of the outfall. Because the only exceedance is based on a conservative estimate of the maximum concentration, and because conservative no-effects thresholds for PAH-16 are never expected to

be exceeded and conservative no-effects thresholds for B[a]P are exceeded in only two cases, these results suggest that any adverse effects that might occur would be minimal in frequency and extent.

Other than in comparison to the most conservative screening benchmarks, there is little indication that predicted PAH concentrations in sediments of Reyðarfjörður would cause adverse effects in benthic invertebrate biota. This conclusion is supported by evaluations conducted at other smelter sites. Studies such as those conducted by Knutzen (1995) and Paine et al. (1996) have shown no definitive evidence of sediment toxicity or changes in benthic community structure at PAH concentrations ranging from 400,000 μg/kg up to almost 10,000,000 μg/kg. By contrast, the maximum PAH concentration predicted in sediment at Reyðarfjörður is 500 μg/kg. Also as noted by Knutzen (1995) and Paine et al. (1996), bioavailability appears to be low for PAHs in sediments near aluminum smelters, probably because they are tightly bound to soot carbon, coal, or pitch. This is further evidence that the minimal exceedances of screening values described above are very unlikely to translate into actual effects in the environment, even if predicted PAH concentrations should be realized. The large magnitude between the sediment PAH concentrations predicted for the Fjarðaál smelter and the effects thresholds for other smelters indicate with a high degree of confidence that no adverse effects are likely to occur to benthic organisms and marine invertebrates due to PAH discharges from Fjarðaál. Maximum sediment PAH concentrations predicted for the Fjarðaál smelter are also well below the SQG of 1,000 μg/kg suggested by NMFS for estuarine fish species. Accumulation of PAHs by bottom-dwelling fish is not anticipated given the low sediment concentrations and the well-developed ability of fish to metabolize PAHs; thus, it is very unlikely that there is a complete exposure pathway from fish to marine mammals. Predicted concentrations of PAH-16 and B[a]P in water appear to be well below the lowest concentrations that have elicited adverse effects in fish or other aquatic organisms, suggesting that effects on pelagic fish from the dissolved phase of these CoPCs is unlikely. Based on the multiple lines of evidence presented here, the risks to ecological receptors from PAH discharges are considered to be negligible. The only qualification on this conclusion is the recognition that effects of PAHs on marine plants, such as seaweeds, are poorly understood. There is very limited information in the scientific literature on effects thresholds or toxic effects. Therefore, the risks to seaweed growing along nearshore areas adjacent to the smelter cannot be fully evaluated.

Food-web modeling was conducted to evaluate risks to avian receptors (oystercatcher and herring gull) exposed to total PAHs and B[a]P. Average daily ingestion estimates of these chemicals for both avian species were less than the NOAEL and LOAEL doses for reproductive effects (hazard quotients less than 1.0), indicating that even under very conservative exposure assumptions, PAHs are unlikely to cause adverse effects in avian species exposed through their diet.

Evaluation of Risk to Aquatic Receptors and Wildlife from Fluoride

Fluoride concentrations in undiluted effluent are estimated to be 1.51 mg/L, which is only slightly higher than background concentrations. Rapid mixing of the effluent will result in dilution to background concentrations within a small area around the outlet. Marine organisms

appear to be relatively insensitive to fluoride. As described above, growth inhibition of algae and aquatic plants does not become pronounced until fluoride concentrations reach 100–200 mg/L. Therefore, risks to phytoplankton and aquatic plants, such as seaweed, from exposure to fluoride in effluent appear to be minimal. Marine fish can have short-term exposure to concentrations as high as 100 mg/L without exhibiting mortality. As these effects thresholds are about 66-fold higher than the maximum fluoride concentration before dilution, it is very unlikely that acute toxicity would occur to aquatic organisms. Less is known about the effects of chronic exposure to fluoride. Hemens et al. (1975) found that mullet could tolerate 5.5 mg/L fluoride for 113 days without mortality or deterioration of physical condition. This concentration is 3.6 times higher than the concentration in undiluted effluent, thus ambient concentrations of fluoride would not be expected to approach this level. The rapid dilution of fluoride to background concentrations, coupled with the ability of fishes to move away from areas of stress, suggest that adverse effects would be unlikely even in areas immediately around the outfall. Adverse effects to marine mammals and birds would be very unlikely at the fluoride concentrations predicted to exist in the fjord.

Evaluation of Risk to Aquatic Receptors from Sulfur Dioxide/pH

Sulfur dioxide reacting with seawater will produce acids that lower marine water below its typical pH of 8.2. Figures presented in Vatnaskil (2005) show that predicted minimum pH will be 8.1 or 0.1 pH unit below background in a highly localized area around the outfall. In total, other than an area of approximately 0.0027 km^2, the predicted pH in Reyðarfjörður will be at normal (background) levels. Knutzen (1981) states that tested marine algae and animals tolerate a pH reduction of 0.5–1.0 units, although also noting that few long-term studies have been performed. Kuwatani and Nishii (1969) and Bamber (1987) found indications of shell dissolution in bivalves at about pH 7.5. In conjunction, the highly localized area of decreased pH and the observations of Knutzen on tolerance of marine organisms, suggests that adverse effects due to interactions of sulfur dioxide and seawater are minimal. There is a low potential that effects could occur to molluscs in the immediate vicinity of the outfall, though such an effect would be localized to no more than 0.0027 km^2 in the immediate vicinity of the outfall.

Evaluation of Risk to Aquatic Receptors from Physical Stressors

Physical stressors are not directly evaluated in this risk assessment, but it should be noted that studies at other smelters have sometimes found physical stressors associated with effluent discharges to be more important than toxicological effects of chemicals in the effluent, particularly for benthic invertebrate communities. For example, as discussed above, both Knutzen (1995) and Paine et al. (1996) concluded that differences in benthic community composition were more strongly influenced by physical factors than by PAH concentrations in effluent. A number of physical impacts are possible. For example, scouring of soft-bottom sediments in the path of the discharge can eliminate habitat for benthic organisms. In addition, habitats elsewhere can be affected if they are smothered by resettlement of sediments suspended by the effluent flow. The spatial extent over which these effects could occur at Fjarðaál depends on factors such as the velocity of the effluent discharge, sediment composition in the vicinity of the outlet, bathymetry, and current velocity and direction. The information presented in the

comparative EIA (Hönnun et al. 2002) does not provide sufficient detail on characteristics of the seawater scrubber (outflow velocity, outfall placement, etc.) to quantify the potential magnitude of effects due to this potential stressor. Other physical factors that could affect ecological communities, such as pH and water temperature changes, appear to be less important because of the rapid dilution of discharged effluent in seawater.

Uncertainty Analysis

There are a number of factors that contribute a degree of uncertainty to risk conclusions presented here. This section briefly highlights some of the major factors.

The primary sources of uncertainty relate to incomplete knowledge regarding the precise concentrations of CoPCs released in the effluent stream, and their ultimate concentrations in sediments and water of Reyðarfjörður. The Fjarðaál EIA (Hönnun et al. 2002) was prepared on the assumption that wet scrubbers would not be used, so there are no data provided in that report on chemical concentrations in scrubber effluent. Therefore, concentrations had to be estimated from other sources, including data for other operating smelters that use similar production and pollution control technologies, but with differing production capacities. This approach introduces uncertainty, because it cannot be conclusively shown that waste stream volumes for air or water emissions are proportional across production rates or that the relative concentrations of constituents do not vary significantly among comparable facilities. The extent to which these uncertainties influence risk conclusions, however, appears to be minimal. For example, based on estimates by Vatnaskil (2005) and data for the Deschambault and Mosjøen smelters, PAH-16 loading for Fjarðaál is estimated to range from 15.8 to 32 kg/year. This limited range, despite large differences in production capacities between smelters suggests that predictions for PAH-16 loading presented by Vatnaskil (2005) are reasonable. The corresponding estimated range for B[a]P is 0.47 to 0.62 kg/year which is narrower than range than for PAH-16. Overall, the uncertainty associated with predicted PAH-16 and B[a]P loading is considered to be low, and has only a minimal degree of impact on risk conclusions, primarily because the upper end of the predicted sediment and water concentrations was used.

An additional uncertainty is that maximum chemical concentrations in sediments cannot be determined because the Vatnaskil (2005) report provides only ranges for concentrations of chemicals in sediment, and does not provide the upper bound. For purposes of this assessment, we had to assume likely upper bounds on PAH-16 and B[a]P concentrations as the maximum stated concentration (500 μg/kg for both PAH-16 and B[a]P) provided in the Vatnaskil (2005) report. If these conservative maximum concentrations (500 μg/kg for both PAH-16 and B[a]P) underestimate actual maxima, then risk to ecological receptors could also be underestimated. The spatial extent where these maxima could be exceeded is small, no more than 0.12 km^2 for PAH-16 and no more than 0.0092 km^2 for B[a]P. Therefore, the likelihood that this uncertainty results in a widespread underestimation of risk is minimal, and is further minimized by comparisons to screening criteria, as discussed below.

Several other uncertainties are associated with use of data from the Vatnaskil (2005) report. First, the model that was used assumes that 90 percent of PAH-16 and 10 percent of B[a]P released in effluent remains in the dissolved form and is flushed out of the fjord. These

estimates represent a simplification of partitioning between dissolved and particulate phases for PAHs. Most PAHs have partitioning characteristics that are closer to 70:30 or 60:40 than to 90:10. Thus, sediment concentrations of PAH-16 may be underestimated by a factor of 3–4 fold. If all PAH-16 were bound to particles, then the resultant sediment concentrations would be 10-fold higher than predicted here. The ecological effects of PAHs flushed from the fjord cannot be evaluated, because fate for this component of the effluent stream is not modeled in the Vatnaskil (2005) report. However, because adverse effects are not anticipated at the lowest deposition rates within Reyðarfjörður, it is unlikely that effects would be seen at areas beyond the fjord where ocean currents would produce greater dispersal and dilution. Therefore, the uncertainty that risk to ecological receptors outside the fjord has been underestimated is considered to be very low. The sediment calculations presented here are based on the assumption that settling particles have a 10 μm diameter. This represents a conservative upper estimate of particle size, and depending on the extent of agglomeration, the actual size could be smaller. If so, suspended particles would be dispersed more widely prior to settling, and resultant sediment concentrations would be lower than estimated. Thus, modeling based on a 10 μm particle size represents a conservative approach to risk estimation. This assumption is considered to add a medium level of uncertainty to the risk assessment, although the magnitude to which sediment concentrations, and thus risk, are overestimated is unknown.

There is less uncertainty with regard to the likelihood of adverse ecological effects than is associated with prediction of chemical concentrations in water or sediment. Predicted maximum sediment concentrations of PAH-16 and B[a]P just slightly exceed the most conservative screening benchmarks. Furthermore, these predicted concentrations are orders of magnitude lower than concentrations recorded near other aluminum smelters that were not associated with any adverse effects on sediment-associated organisms. Consequently, the likelihood of adverse effects due to PAHs in sediments is considered to be low.

Fluoride and sulfur dioxide-related effects on pH are other stressors that were evaluated in this assessment. For both these stressors, rapid dilution of effluent in seawater will return conditions to background levels rapidly, and areas of elevated fluoride concentrations or depressed pH are expected to occur only in a very small area around the outfall. Review of toxicology data suggests that neither of these stressors is likely to elicit adverse ecological effects, although there is some uncertainty in this conclusion in the immediate vicinity of the outfall.

Conclusions

This risk assessment was conducted to evaluate whether use of wet scrubbers at the planned Alcoa Fjarðaál smelter would result in unacceptable risks to ecological receptors inhabiting the waters and sediment of Reyðarfjörður. The primary CoPCs associated with use of the scrubbers are PAHs, fluoride, and sulfur dioxide. Data for the planned Reyðarál smelter and operating smelter facilities in Norway and Canada were used to estimate the range of likely loadings of these CoPCs to the fjord and subsequent concentrations in sediment. Estimated PAH and B[a]P concentrations in sediment were compared with all available marine SQGs, and were found to be lower than all but the most conservative of these screening values. Literature studies, including ecotoxicity reviews and surveys conducted at other aluminum smelter sites, were consulted to determine if predicted CoPC concentrations would be likely to elicit adverse

effects. Results of these comparisons indicate that maximum predicted PAH and fluoride concentrations at Fjarðaál are well below threshold effect levels, in some cases up to several orders of magnitude lower. Conservative estimates of PAH accumulation in aquatic invertebrates indicate that concentrations are very unlikely to be high enough to elicit adverse effects in wildlife populations that consume these prey items. It was also determined that depression of pH resulting from interactions with sulfur dioxide was unlikely to cause population-level effects in marine organisms.

In conclusion, there is minimal likelihood of adverse effects to marine organisms of Reyðarfjörður as a result of CoPC releases in wet scrubber effluent, although there is a possibility of limited effects in a small area (e.g., approximately 9,200 m^2) immediately around the outfall, primarily from B[a]P. There are several important uncertainties associated with this risk evaluation, particularly with regard to accurate estimation of CoPC loads in effluent and their ultimate distribution and accumulation in sediments of the fjord. This and other uncertainties are not considered to be serious enough to alter the risk assessment conclusions. The benthic marine community will be permanently altered to some extent in the immediate vicinity of the outfall, primarily because of physical alteration of substrates as a result of scouring from the effluent stream and the outfall structure itself.

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Figures

Figure 1. Facility location

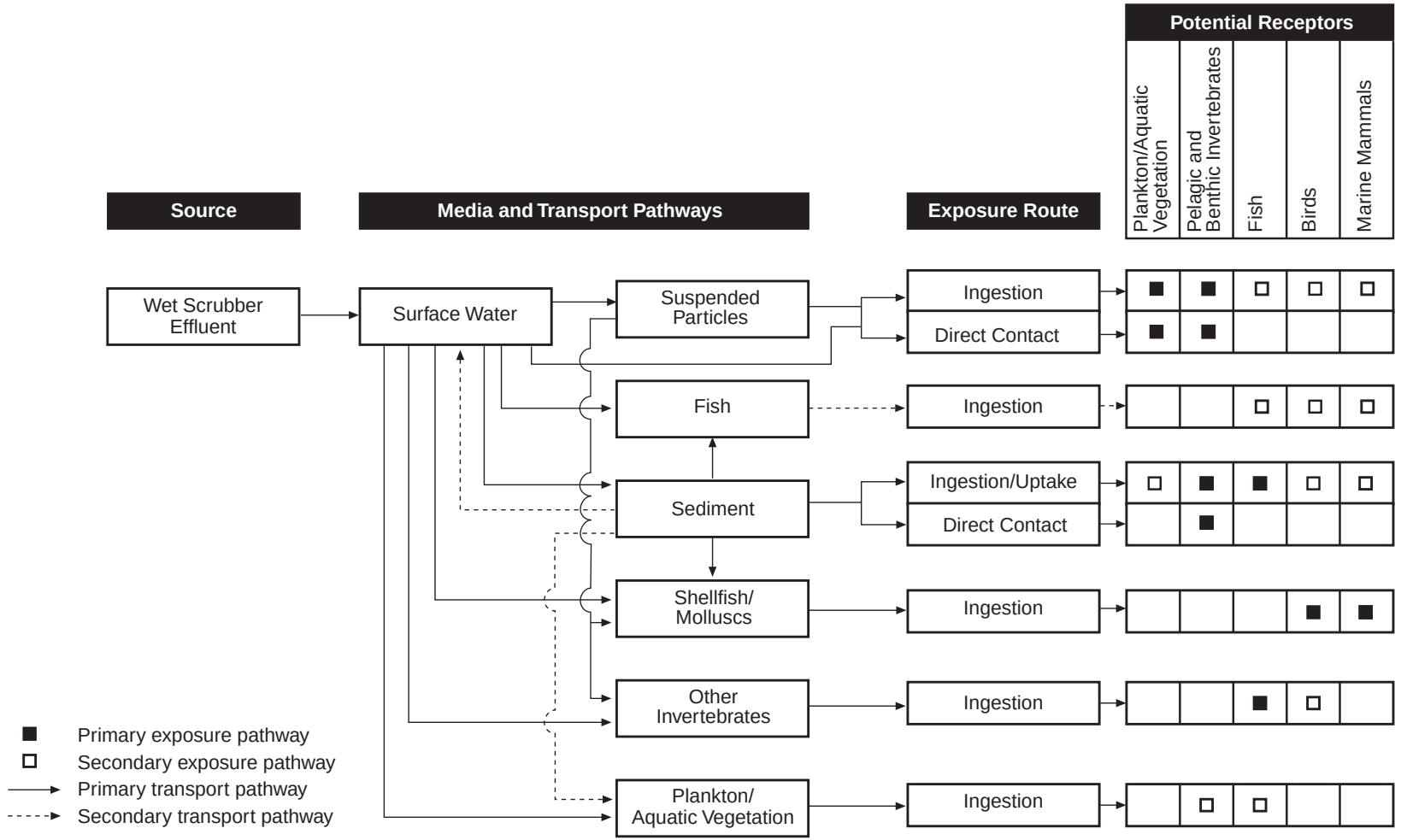

Figure 2. Conceptual site model

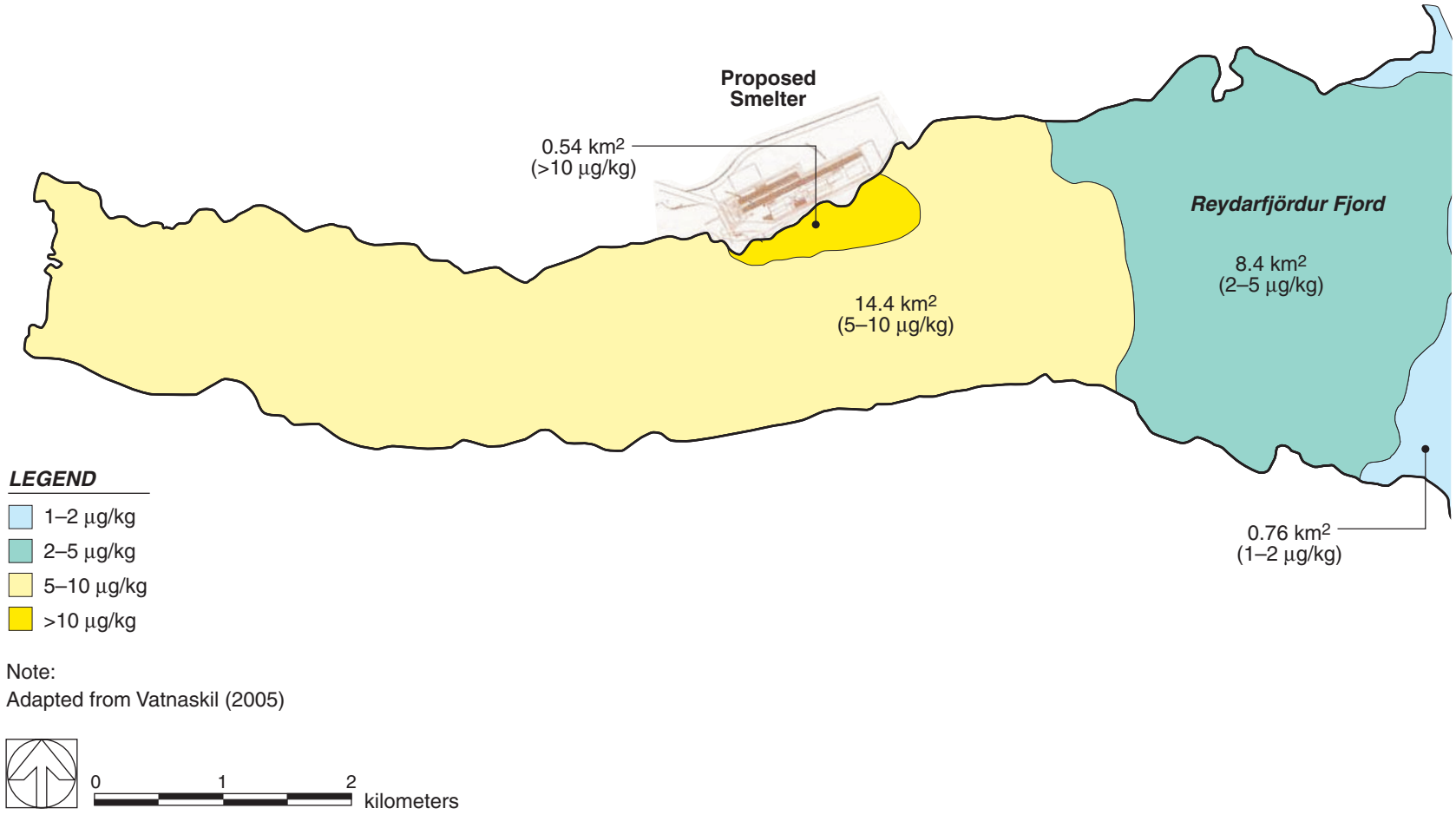

Figure 3. Approximate area (km2) associated with PAH-16 concentration ranges (1 µm particle size)

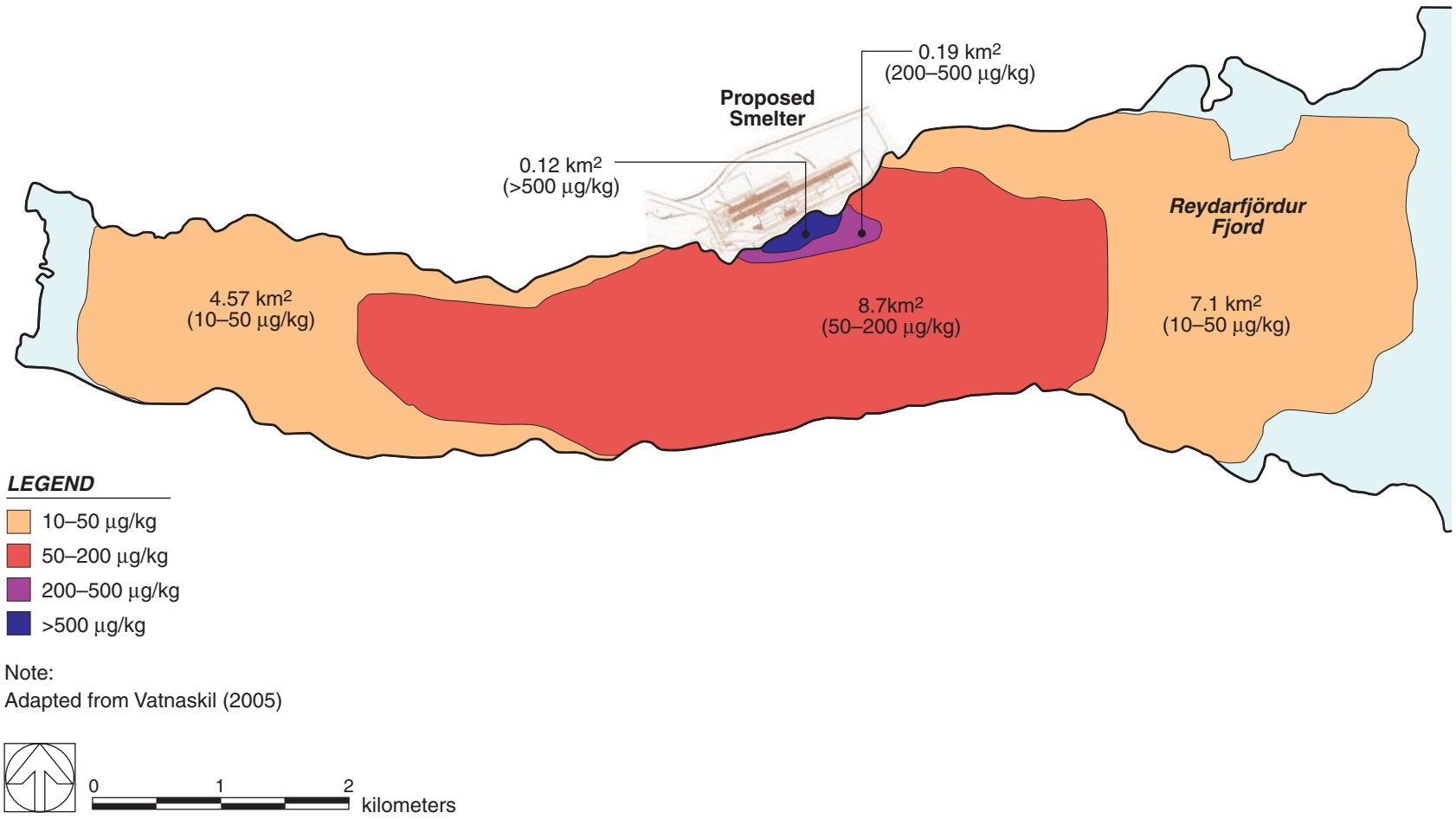

Figure 4. Approximate area (km2) associated with PAH-16 concentration ranges (10 µm particle size)

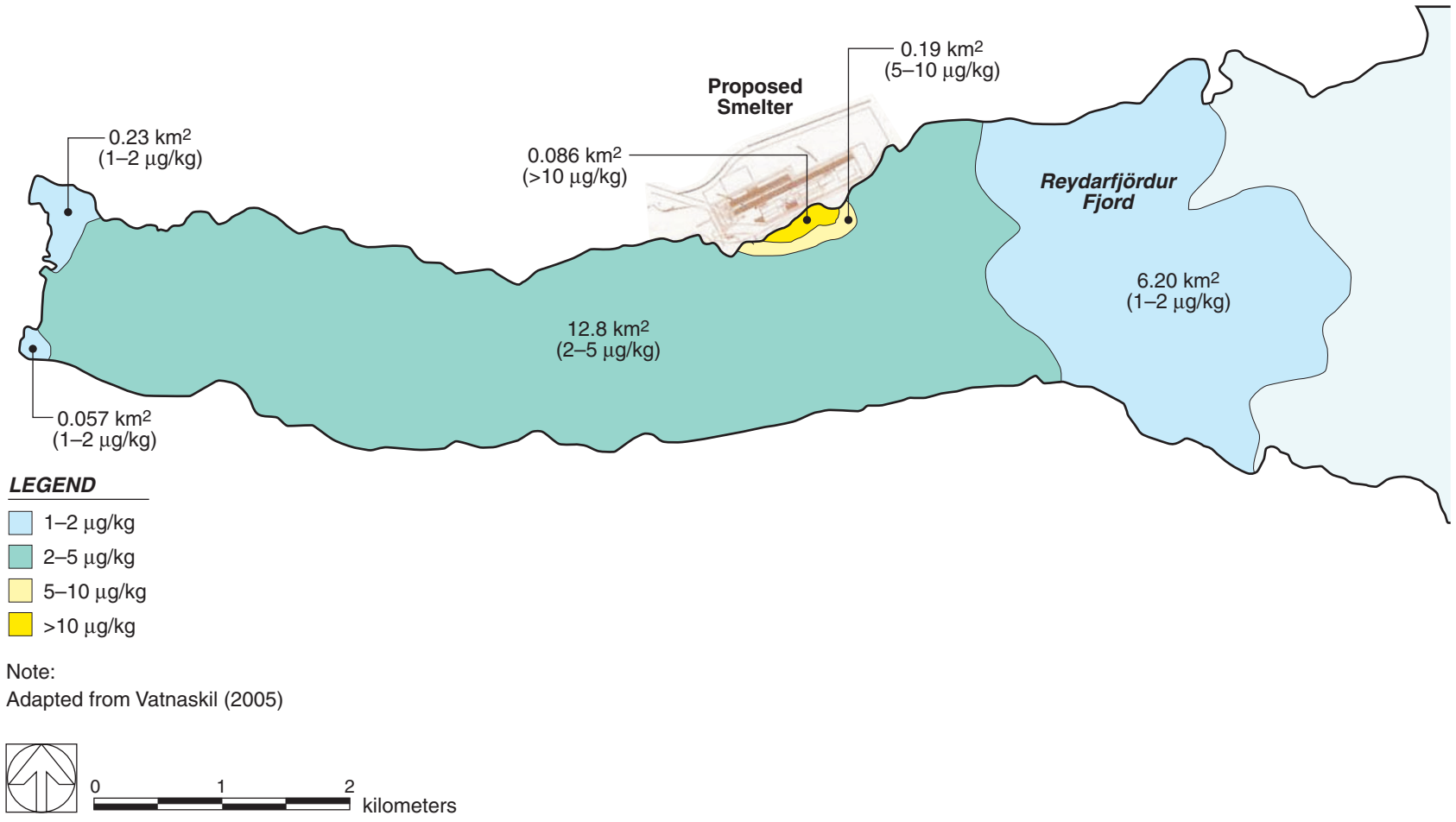

Figure 5. Approximate area (km2) associated with B[a]P concentration ranges (1 µm particle size)

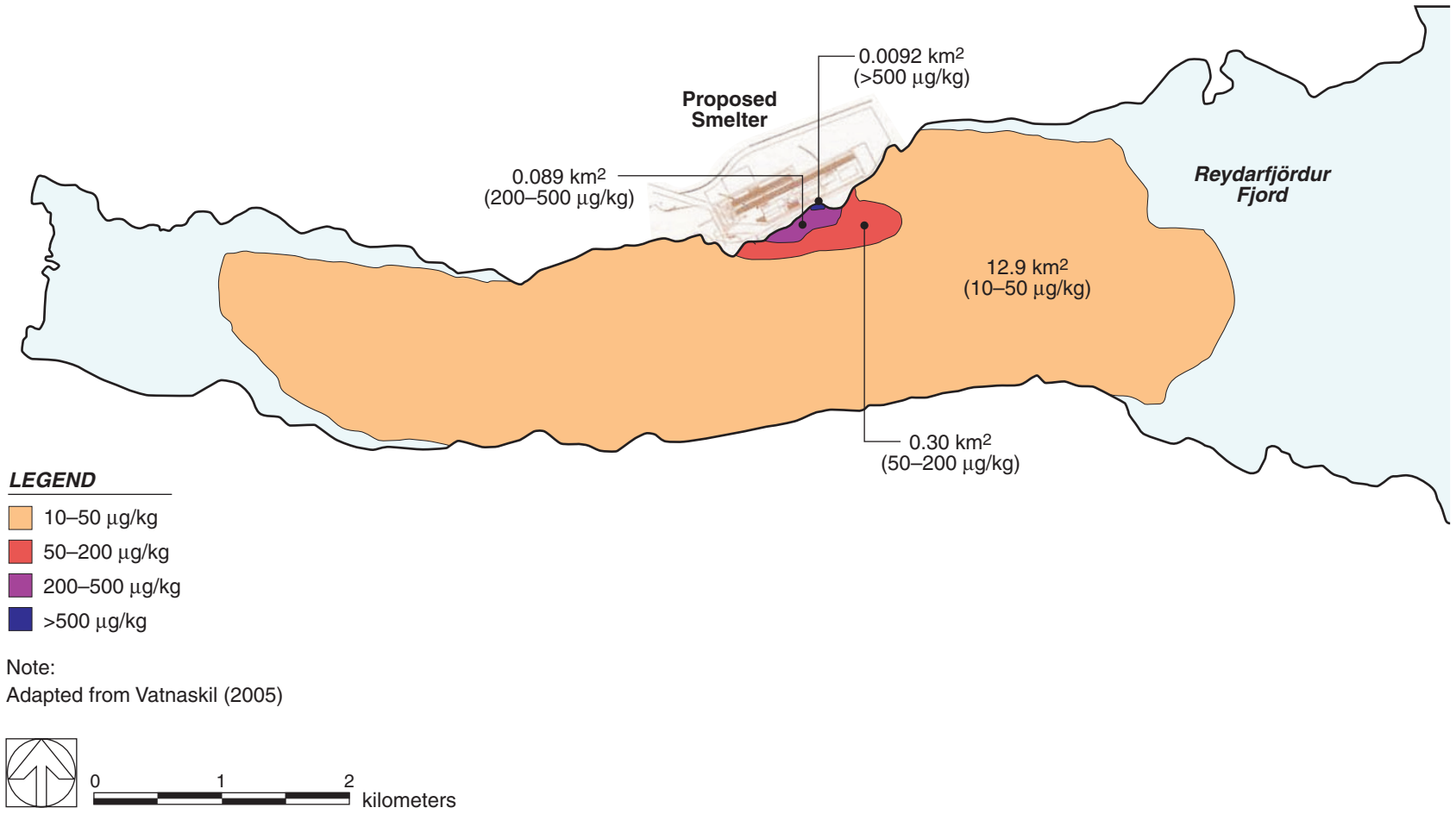

Figure 6. Approximate area (km2) associated with B[a]P concentration ranges (10 µm particle size)

Tables

Table 1. Emissions data for Deschambault smelter and conversion to theoretical seawater scrubber emissions for Alcoa-Fjarðaál plant

Analyte	Concentration in Air Emissions (mg/m^3)[a]	Concentration in Water Emissions ($\mu g/L$)[b]	Air/Water Loading (mg/day)[c]	Air/Water Loading (kg/year)
Naphthalene	0.00011	0.015502296	4,037	1.473
Acenaphthylene	0.00004	0.006513092	1,696	0.619
Acenaphthene	0.00012	0.017912888	4,665	1.703
Fluorene	0.00020	0.029008995	7,554	2.757
Phenanthrene	0.00030	0.04430516	11,537	4.211
Anthracene	0.00004	0.006513092	1,696	0.619
Fluoranthene	0.00007	0.010697369	2,786	1.017
Pyrene	0.00004	0.006513092	1,696	0.619
Benz[a]anthracene	0.00004	0.006513092	1,696	0.619
Chrysene	0.00004	0.006513092	1,696	0.619
Benzo[b,j,k]fluoranthene	0.00007	0.009769638	2,544	0.929
Benzo[b]fluorene	0.00004	0.006513092	1,696	0.619
Benzo[a]pyrene	0.00004	0.006513092	1,696	0.619
Indeno[1,2,3-cd]pyrene	0.00004	0.006513092	1,696	0.619
Dibenz[a,h]anthracene	0.00004	0.006513092	1,696	0.619
Benzo[ghi]perylene	0.00004	0.006513092	1,696	0.619
Total	0.00132	0.192327269	50,082	18.28

Note: PAH - polycyclic aromatic hydrocarbon

[a] Average of three measurements; concentrations of undetected analytes are presented at one-half the detection limit.

[b] Assumes all of the PAH becomes associated with scrubber water: PAH (mg/day) divided by seawater scrubber flow rate (2.604×10^8 L/day) times 1,000 μg/mg yields PAH equivalent (μg/L).

[c] Flow from the gas treatment center during testing was 1,144,382 m^3/hour. Includes adjustment factor of 1.38 to account for production rate difference between Deschambault (250,000 tpy) and Alcoa-Fjarðaál (346,000 tpy).

Table 2. Monitoring data from seawater scrubber at the Mosjøen smelter, Norway, and conversion to theoretical seawater scrubber emissions for Alcoa-Fjarðaál plant

	Elkem, Mosjoen Seawater Outlet (μg/L)	Predicted Loading at Alcoa-Fjarðaál Seawater Outlet (mg/day)[a]	Predicted Loading at Alcoa-Fjarðaál Seawater Outlet (kg/year)
Naphthalene	0.07	18,228	6.653
Acenaphthylene	0.13	33,852	12.356
Acenaphthene	0.02	5,208	1.901
Fluorene	0.04	10,416	3.802
Phenanthrene	ND	1,302	0.475
Anthracene	ND	1,302	0.475
Fluoranthene	0.01	2,604	0.950
Pyrene	ND	1,302	0.475
Benz[a]anthracene	0.01	2,604	0.950
Chrysene	0.01	2,604	0.950
Benzo[b]fluoranthene	ND	1,302	0.475
Benzo[k]fluoranthene	ND	1,302	0.475
Benzo[a]pyrene	ND	1,302	0.475
Indeno[1,2,3-cd]pyrene	ND	1,302	0.475
Dibenz[a,h]anthracene	ND	1,302	0.475
Benzo[ghi]perylene	ND	1,302	0.475
Total		87,234	31.840

Note: ND - not detected; one-half the detection limit (0.005 μg/L) used in loading calculations

[a] μg/L times 2.604×10^8 L/day divided by 1,000 μg/mg.

Table 3. Summary of environmental quality guidelines for PAH and B[a]P in marine sediment (all values in µg/kg, dry weight)

	Effects Levels and Ranges					Canadian Environmental Quality Guidelines		Netherlands		OSPAR Background Concentration[a]		
Analyte	TEL	PEL	ER-L	ER-M	AET	ISQG	PEL	NC	MPC	Min.	Max.	JAMP Value
B[a]P	88.8	763	430	1,600	1,100	88.8	763	3	3,000	1	3.8	100–1,000
Total PAHs	1,684	16,770	4,022	44,792								

Note: AET - apparent effects threshold
B[a]P - benzo[a]pyrene
ER-L - effects range-low
ER-M - effects range-median
ISQG - interim sediment quality guideline
JAMP - Joint Assessment and Monitoring Programme
MPC - maximum permissible concentration
NC - negligible concentration
OSPAR - Oslo and Paris Conventions for the Prevention of Marine Pollution
PAH - polycyclic aromatic hydrocarbon
PEL - probable effects level
TEL - threshold effects level

[a] For Arctic Ocean to Iceland Sea.

Table 4. Avian exposure parameters

Parameters	Units	Eurasian Oystercatcher		Herring Gull	
Mean adult body weight	kg	0.526 [a]	Dunning (1993)	0.999 [a]	U.S. EPA (1993)
Food ingestion rate (dw basis)	kg/day	0.0383 [b]	Nagy (1987)	0.0582 [b]	Nagy (1987)
Water ingestion rate	L/day	0.0384 [c]	Calder and Braun (1983)	0.0590 [c]	Calder and Braun (1983)
Proportion of sediment in diet	fraction	0.104 [d]	Beyer et al. (1994)	0.02 [e]	Beyer et al. (1994)
Proportion of diet that is mussels	fraction	1	Assumption	1	Assumption

[a] Mean adult body weight.

[b] Food ingestion rate = $0.0582\ BW^{0.651}$

[c] Water ingestion rate = $0.059\ BW^{0.67}$

[d] Value is for woodcock.

[e] Value is for ducks.

Table 5. Bivalve BCFs for PAHs as reported in Neff (2002)

Chemical	BCF
Naphthalene	65.1
1-Methylnaphthalene	216
1,4-Dimethylnaphthalene	656
2,3-Dimethylnaphthalene	701
1,3,5-Trimethylnaphthalene	2,260
Fluorene	430
Phenanthrene	1,020
1-Methylphenathrene	2,490
Dibenzothiophene	3,180
Pyrene	3,970
Fluoranthene	4,340
Chrysene	18,000
Benz[a]anthracene	20,100
Benzo[a]pyrene	26,800
Perylene	42,800
Indeno[1,2,3-cd]pyrene	226,000
Dibenz[a,h]anthracene	130,000
Coronene	130,000
Average BCF	34,057

Note: BCF - bioconcentration factor
PAH - polycyclic aromatic hydrocarbon

Table 6. Avian exposure models

	Surface Water (mg/L)	Sediment (mg/kg)	BCF	Mussel (mg/kg, wet)	Percent Moisture	Mussel (mg/kg, dry)	Intake via Food (mg/kg-day)	Intake via Surface Water (mg/kg-day)	Intake via Sediment (mg/kg-day)	Ingestion Rate (mg/kg-day)	Toxicity Reference Value (mg/kg-day)	Hazard Quotient
Eurasian Oystercatcher (*Haematopus ostralegus*)												
Total PAHs	0.0000050	0.50	34,057 [a]	0.17	82	0.95	0.036	0.00	0.0020	0.038	0.14	0.27
Benzo[a]pyrene	0.00000020	0.50	26,800	0.0054	82	0.030	0.0011	0.00	0.0020	0.0031	0.14	0.02
Herring Gull (*Larus argentatus*)												
Total PAHs	0.0000050	0.50	34,057 [a]	0.17	82	0.95	0.055	0.00	0.00058	0.056	0.14	0.39
Benzo[a]pyrene	0.00000020	0.50	26,800	0.0054	82	0.030	0.0017	0.00	0.00058	0.0023	0.14	0.02

Note: BCF - bioconcentration factor
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PAH - polycyclic aromatic hydrocarbon

[a] Mean of bivalve BCFs reported in Neff (2002), Table 5.