SCIENCE INTEGRITY KNOWLEDGE

MARINE ECOLOGICAL RISK ASSESSMENT OF BRUNSWICK SMELTER, A GLENCORE FACILITY
BELLEDUNE, NB

FINAL REPORT

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In association with

minnow environmental inc.

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MARINE ECOLOGICAL RISK ASSESSMENT OF AREAS ADJACENT TO GLENCORE'S BRUNSWICK SMELTER, BELLEDUNE, NEW BRUNSWICK

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EXECUTIVE SUMMARY

Glencore Canada (Glencore) has been operating the Brunswick Smelter in Belledune, New Brunswick, since the mid-1960s. Several detailed risk assessment studies have been previously conducted to investigate the potential for human health risks associated with exposures from facility emissions in residential areas near the facility (*i.e.*, Shore Road Soil Study; Intrinsik Environmental Sciences Inc. et al., 2008), as well as potential ecological risks in the terrestrial and freshwater environments south of the facility (Intrinsik Environmental Sciences Inc., 2013). Glencore is now interested in examining the potential for ecological risks in the marine environments adjacent to the facility, associated with current and on-going operations. As such, Glencore commissioned Intrinsik Environmental Sciences Inc. (hereafter referred to as Intrinsik) to conduct an ecological risk assessment (ERA) of the marine areas, and species foraging in those areas, near the smelter. Intrinsik is conducting this study with Minnow Environmental Inc., who specializes in aquatic surveys, and have conducted monitoring associated with the facility for many years.

The primary releases of interest from the smelter relate to current lead smelter treated effluent discharge, former fertilizer plant gypsum-based effluent discharge, atmospheric discharges, and possible contributions related to erosion of the former slag disposal area on Belledune Point. The main receptor groups of interest include aquatic species (marine phytoplankton and pelagic invertebrates, benthic invertebrates, and marine fish species) as well as avian species living at or near the facility and foraging in the marine environment and the associated shoreline. Following the review of existing data and information, a field sampling program was implemented to conduct the following:

- A benthic invertebrate abundance and diversity study, including sediment chemistry and physical characterization;
- A shellfish health assessment, involving deployed mussels and assessment of survival, growth and condition endpoints, with body burden and marine water quality chemistry characterization;
- A fish health assessment, involving a benthic fish species, and survival, growth, condition and reproduction endpoints;
- Sampling of whole fish tissue and shoreline invertebrate chemistry analysis, as well as beach sand chemistry and bioaccessibility testing, for input into an exposure model to characterize exposure and risks to various avian species nesting and foraging in the area;
- Sampling of salvage chick organ tissue and eggs of the common tern, for the purposes of metal residue chemistry analysis.

Based on the data and assessments conducted, the following conclusions were drawn:

Marine Phytoplankton and Pelagic Invertebrates:

- Risks are considered to be negligible to low, based on comparison of measured water quality metals concentrations to marine water quality guidelines and reference, as well as other toxicology data and information.
- The exposure data are limited in terms of number of samples, and hence there is uncertainty in this conclusion. This uncertainty is reduced by knowledge that the area adjacent to the smelter is a highly dispersive environment, and while releases from the facility are measureable in the environment, exposure levels for transient mobile species are expected to be low, and hence would not be anticipated to result in population- or community-level effects.

Marine Benthic Community:

- Risks are considered to be low for benthos near the former fertilizer outfall location (FPO), and in an area distant to the final effluent discharge point (SST2), and are considered moderate for final effluent discharge area (FE), based on the existing chemistry data, and the benthic density, diversity and richness data. Evenness and diversity of the benthic community at the FE area suggested ecologically meaningful differences from reference. There was also reduced diversity in this area, relative to reference, albeit, to a lesser degree than that reported for evenness and diversity. In the current survey, increased sediment metals concentrations and lower benthic invertebrate density and differences in community structure, relative to surveys conducted in 2008 and 2004 at FE, were noted, which was not linked to effluent discharge quality or flow volume. Rather, these changes appear to be related to either erosion of the former slag disposal area at Belledune Point as a result of a large storm event in 2010, or the recently completed Belledune arbor dredging project.

Marine Shellfish:

- Risks are considered to be low, based on the available data and studies conducted. Survival was not considered to be influenced in the study area, relative to reference. Growth was actually greater in the study area mussels at several sites, than in reference areas, but condition was slightly lower. These results were attributed to higher allocation of energy use to growth in the smelter-exposed mussels compared to reference. Increased growth rates could be a function of slight temperature related differences, or higher levels of nutrients, such as nitrogen, iron and manganese in surface waters near the smelter, when compared to reference areas. While tissue metals were significantly higher in the exposure group for arsenic, cadmium, copper, lead, selenium, silver, strontium and zinc, the results of the survival, growth and condition endpoints indicate no adverse effects in blue mussels near the smelter.
- Uncertainties in the assessment include a lack of assessment of the reproductive endpoint, since the study was initiated outside of the season of reproductive tissue development (and hence reproduction endpoint could not be evaluated).

Nonetheless, numerous juvenile blue mussels were found adhering to the cages of the deployed mussels. While a quantitative assessment of reproductive endpoints was not undertaken, qualitative observations suggest presence of juveniles in all cage areas, with lower numbers being observed at the Study area station located furthest from the smelter (Station S4). These observations suggest juvenile mussels appear abundant in areas near the facility.

Marine Fish:

- Risks are considered to be low, based on assessment of water quality, survival, growth/condition, reproduction and tissue residue data. No critical effect sizes were exceeded for any endpoint with the exception of egg size. Smaller egg size in smelter-exposed fish was hypothesized to reflect natural variability in spawning timing between the exposure and reference fish populations.
- Male outcomes are uncertain due to limited sample size, but are not indicative of adverse effects, based on the existing dataset.

Avian Species Nesting and Foraging in the Area:

- Common tern nest on the smelter property annually, and forage in both the near shore and far shore areas adjacent to the smelter. Based on the weight of evidence, risk potential to the common tern colony is considered low. Modelled exposures suggest low risk potential to the common tern colony, with only iron having 95th percentile Hazard Quotients (HQs)> 1. Clutch counts from 2010 suggest the colony is within the range of clutch counts in other areas of New Brunswick. Fish tissue concentrations of mercury and selenium are well below thresholds associated with adverse effects in piscivores, and measured residues in eggs, kidney and liver are below toxicity thresholds (where they are available), with the exception of lead in a number of kidney and liver samples. While exceedance of toxicity thresholds for lead in some samples suggests a high potential for adverse effects in those individuals, a limited number of dead chicks were found following extensive daily surveys of the colony in 2014, and many of the metals residues within tissues were below toxicity thresholds suggestive of clinical or severe effect levels. Weighing the available information, some individuals within the colony have a high potential for adverse effects from exposures to lead, but there appears to be a low probability of effects on the colony as a whole, based on the numbers of chick tissue samples exceeding toxicity thresholds, relative to the number of eggs reported in previous colony counts. The colony has returned to nest at the smelter year after year, and anecdotal observations suggest it is increasing in size. There is uncertainty in this conclusion related to specific clutch size for 2014, and exposures to chicks which were not sampled.

- Black-crowned night heron forage on and near the smelter property (Belledune Point), but nesting pairs have not been observed in previous surveys conducted. Risk potential for this species is considered to be negligible to low, based on low probability of Hazard Quotients exceeding 1, with the exception of iron, lead, and to a lesser extent, strontium and thallium. Lead and zinc concentrations in beach sand along Belledune Point are elevated relative to concentrations of sediments considered to be protective of waterfowl in other areas, but beach sand metal concentrations are not elevated in Areas 2 or 3, down the shore. The dominant exposure pathway is diet, but considering that there would be a limited number of individuals present in this area, and hence, population level effects are considered unlikely near the Brunswick Smelter. Lead would be considered the Chemical of Concern (COC) with greatest risk potential, based on the available data.

- Sandpiper forage along the shore of the beach on the smelter property, and four nesting pairs were reported on Belledune Point in surveys conducted in 2009. This survey was updated in 2015, and a total of 6 nesting pairs were confirmed in Area 1 and 2, with 4 possible additional nesting pairs identified. Risk potential for this species is considered to range from low to moderate, depending upon proximity to the smelter. On Belledune Point, risks are considered to range from low to moderate based on the high probability of multiple Hazard Quotients exceeding 1 (aluminum, copper, iron, lead, selenium, thallium and zinc). Lead had the most elevated HQ in this area, and represents the substance of greatest concern. The HQs are likely biased high, due to assumptions that metals in dietary items are 100% bioavailable, and the TRV used is based on lead acetate, which is more bioavailable than the form present in the Belledune area (which would be a lead sulphate). Belledune Point is the area with highest exposure potential, due to the presence of slag along the beach/shoreline, and concentrations of lead and zinc in this area were also found to exceed concentrations reported as being protective of waterfowl in other published literature. Areas further down the shoreline to the east of the facility represent a low risk potential. The risk potential for the shoreline overall was considered to be low as diet was found to be the most important exposure pathway in all areas considered (and bioacccessibility in diet was assumed to be 100%). However, adverse effects in some individuals could be occurring on Belledune Point but are considered less likely in Areas 2 and 3. Depending on exposures and population size an effect on the local population could be possible, but is unlikely.

MARINE ECOLOGICAL RISK ASSESSMENT OF AREAS ADJACENT TO GLENCORE'S BRUNSWICK SMELTER, BELLEDUNE, NEW BRUNSWICK

1.0 INTRODUCTION

Glencore Canada (Glencore) has been operating the Brunswick Smelter in Belledune, New Brunswick, since the mid-1960s. Several detailed risk assessment studies have been previously conducted to investigate the potential for human health risks associated with exposures from facility emissions in residential areas near the facility (*i.e.*, Shore Road Soil Study; Intrinsik Environmental Sciences Inc. et al., 2008), as well as potential ecological risks in the terrestrial and freshwater environments south of the facility (Intrinsik Environmental Sciences Inc., 2013). Glencore is now interested in examining the potential for ecological risks in the marine environments adjacent to the facility, associated with current and on-going operations. As such, Glencore commissioned Intrinsik Environmental Sciences Inc. (hereafter referred to as Intrinsik) to conduct an ecological risk assessment (ERA) of the marine areas, and species foraging in those areas, near the smelter. Intrinsik is conducting this study with Minnow Environmental Inc. (hereafter referred to as Minnow), who specializes in aquatic surveys, and have conducted monitoring associated with the facility for many years.

There have been decades of environmental monitoring in the marine area near the facility [as part of Glencore's Environmental Effects Monitoring Program (or EEM), and their Certificate of Approval to operate], as well as several specialty studies conducted on key issues over the years. The purpose of this document is to present the background data and information which was used to determine the focus of the study, the data collected during the course of the study, and the findings of the ERA.

This marine ERA follows a contaminated sites assessment framework which was developed by Environment Canada for federal land holdings, and provides a robust and flexible assessment framework specifically designed for assessment of contaminated areas (FCSAP, 2012). While this assessment framework does not apply to the Glencore facility as it is not a federally-owned facility or property, the framework represents the most recent comprehensive national guidance for assessing the potential ecological implications of environmental contamination and is therefore considered appropriate to use as a tool to frame the study, and assess potential for effects. Using this guidance, a step-by-step approach was followed which included the development of a Problem Formulation document, which was used to determine data needs for the study. Based on the outcomes of the Problem Formulation, sampling of environmental media (*e.g.*, beach sand, surface water, sediment and biota) and biological studies were conducted within a defined study area, and using these data, the potential for ecological risks was assessed.

This document provides details of all stages of the marine ERA of the Brunswick Smelter in Belledune, the ERA process that was followed, and the outcomes of the assessment.

1.1 ERA Approach

The risk assessment framework used in this ERA is depicted in Figure 1-1 (FSCAP, 2012a) and follows the standard risk assessment paradigm comprised of four steps: Problem Formulation, Exposure Assessment, Effects Assessment and Risk Characterization. These steps are bounded by iterative feedback from study outcomes, and/or local consultation and discussions. Each of these steps of ERA are briefly described below, and discussed further in Appendix A and Sections 2.0 to 5.0.

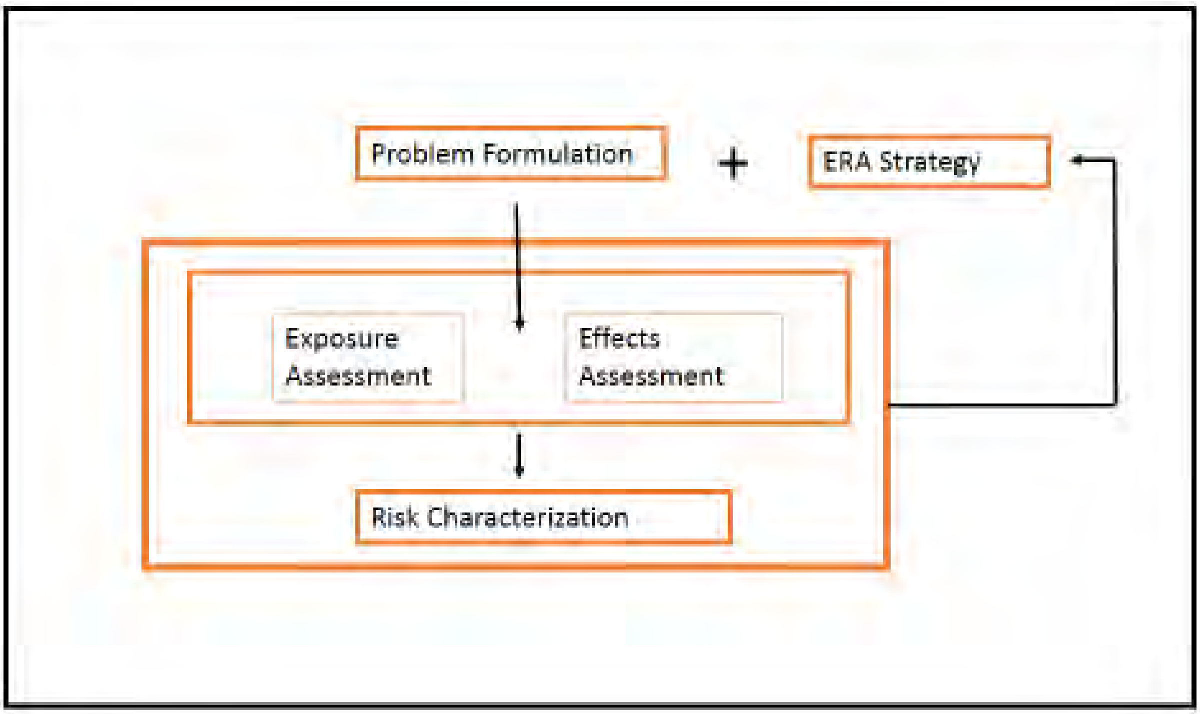

Figure 1-1 Steps of an Ecological Risk Assessment (taken from FCSAP, 2013)

Step I: Problem Formulation: The problem formulation of an ERA acts as an information-gathering and interpretation step, which serves to plan and focus the approach of the risk assessment on critical areas of concern for the site being evaluated. There are several components to the problem formulation stage including: establishing the objective of the ERA; site characterization; site management goals; selection of reference areas; identification of chemicals of potential concern (COPC) and chemicals of concern (COC); identification of receptors of concern (ROC) and relevant exposure pathways; identifying assessment and measurement endpoints; developing lines of evidence (LOE); and developing a conceptual model.

The outcomes of the problem formulation stage form the basis of the approach taken in the ERA and are summarized briefly in Section 2.0, and are presented in greater detail in Appendix A.

Step II: Exposure Assessment: The exposure assessment step of ERA involves estimating the amount of each chemical of concern that is potentially received by each selected ecological receptor. For quantitative assessments involving modelling, exposures are generally estimated using key receptor characteristics and parameters (*e.g.*, body weight, diet proportions, food intake rates, energy utilization, home ranges, amount of time spent in study area, *etc*.). Where modelling is not utilized, exposures are often assumed to be equal to the media in which the receptor occurs (*e.g.*, surface water concentrations for pelagic invertebrates; sediment concentrations for benthic invertebrates). Details of the exposure assessment step for the various receptor groups are provided in Sections 4.0 and Section 5.0, and analytical data characterizing metals levels in the environment are presented in Section 3.0, and Appendix B.

Step III: Effects Assessment: In the effects assessment (which is also referred to as hazard or toxicity assessment), toxicity reference values (TRVs) or other types of toxicity benchmarks (such as water quality guidelines, sediment quality guidelines, etc.) are identified for each receptor or receptor group evaluated, for each chemical of concern. Toxicity reference values are estimates of an exposure level that is not likely to cause unacceptable adverse effects. Details of the effects assessment are provided in Section 4.0 and Section 5.0.

Step IV: Risk Characterization: Risk characterization is comprised of several steps which include: evaluating / interpreting each LOE; summarizing data for the LOEs; and applying a weight of evidence (WOE) approach to make conclusions on the potential for risk and / or potential magnitude of effect. Uncertainties and limitations of the ERA are considered before rendering final risk characterization conclusions. Where required, the risk characterization step may recommend further actions or study. Risk characterization methods and results are provided in Sections 4.0 (aquatic receptors) and 5.0 (avian receptors), respectively.

1.2 Report Organization

The report provides a summary of the Problem Formulation, including the field program that was implemented in August and October of 2014 (Section 2.0). Analytical data collected in 2014 are summarized in Section 3.0, with Section 4.0 presenting the methods, data analysis and conclusions related to potential risks to aquatic receptors. Section 5.0 presents the methods, data analysis and conclusions related to potential risks to avian receptors. Section 6.0 presents uncertainties and limitations of the study, and Section 7.0 provides the summary of conclusions. There are a series of Appendices, which present details related to background history on the site, analytical reports, separate supplementary studies, statistics, and quantitative modelling.

2.0 PRÉCIS OF PROBLEM FORMULATION

The Problem Formulation was undertaken to frame the assessment and data needs, based on what was known about potential releases from the facility, existing and historical metals levels in the environment near the facility, and aquatic and terrestrial-based receptors living in and/or foraging in the area. The Problem Formulation was prepared in June-July of 2014, prior to the collection of samples. It has since been updated to reflect the sampling plan that was implemented in August and October of 2014. This document is provided in Appendix A, with a précis of key information presented here.

2.1 Site Management Goal

The site management goal of the ERA from the Brunswick Smelting operations is to determine whether COCs present in the marine environment related to past or current operations have the potential to adversely affect ecological receptors inhabiting, or foraging in the area. Glencore is not required to conduct this study, but rather, has elected to undertake it in preparation for either retrofitting of the facility, or closure.

2.2 Facility Overview, Historical Studies and Environmental Monitoring

The Brunswick Smelting facility is located on the Baie des Chaleurs in the Village of Belledune, NB, which is approximately 220 km north of Fredericton and 35 km northwest of Bathurst, NB. Figure 2-1 shows the location of the smelter and surrounding area. The facility has operated since 1966, and is currently a lead smelter, but formerly included a zinc smelter and fertilizing plant (which closed in 1995). The fertilizer plant produced a di-ammonium phosphate product, using by-products from the smelting process. Adjacent to the site is the NB Power Belledune Thermal Generating Station, which burns coal, and opened in 1993. In addition, a Canadian Gypsum Company facility has operated in the area since 1996, as has a battery recycling facility, which is owned by Glencore.

Figure 2-1 Brunswick Smelting Facility, and Adjacent Port of Belledune and New Brunswick Power Facility

The primary releases to the environment from the Brunswick smelting facility include atmospheric stack emissions and fugitive dusts, as well as direct effluent discharge, and storm water drainage/runoff. Predominant metals in the atmospheric or effluent emissions profile include lead (Pb), zinc (Zn), cadmium (Cd), arsenic (As) and thallium (Tl), as well as sulphur dioxide (SO_2) and nitrogen oxides (NO_x)(the latter two being restricted to stack emissions). Predominant wind directions in the area are largely to the east, with the next most significant direction being to the southeast, with some seasonal winds also in a westerly direction. With this in mind, atmospheric deposition (of both stack and fugitive emissions) over Baie des Chaleurs would be a direct contribution to the marine environment. Effluent release from the historical fertilizer plant, as well as the smelter processing facility, are / were also prime sources of contaminants. The fertilizer plant, when it was operating, produced a gypsum-based (calcium sulphate) slurry which was released into Baie des Chaleurs via a conveyer system (see Figure 2-1). Due to limited dispersion at this location, gypsum accumulated at this location, creating a hard-pack which affected sediment habitat in the immediate area.

Between 1966 and 1980, waste products from the smelting facility were discharged directly into a slag disposal lagoon that drained into Belledune Harbour. A leak of processing water discovered in the late 1970s resulted in significant metal contamination of sediments (especially Cd and Pb and Zn), particularly in Belledune Harbour, which resulted in closing the lobster fishery within the harbour. Glencore (then known as Noranda), identified the source of leak and with modified treatment of the effluent, Cd levels decreased by 97% by the mid-1980s, and the lobster fishery in the outer harbour area was re-opened in 1985. While there were a number of studies in the marine environment in response to this issue, more recent work conducted by Parsons and Cranston (2006) helps to understand metals transport throughout the Baie des Chaleurs, and the contributing sources. Details related to this study are provided in Appendix A, and are summarized as follows:

- In areas close to the smelter (within 2-3 kilometers), metals concentrations in most cores were noticeably elevated at 15 – 30 cm below the surface of the core, and reached peak values at 5 – 10 cm below the surface, and then decreased substantially in the upper cm of the core. This suggests industrial inputs in more recent years are lower than those from earlier years.
- Some of the highest concentrations of metals in the Belledune area surface sediments were within 1 – 2 km of the facility. The authors defined background levels of As, Cd, mercury (Hg), and Pb in marine sediments as 19, 0.26, 0.04 and 7.3 mg/kg (respectively), based on the 95thpercentile of each element within the pre-industrial sediment core bottoms.
- Nickel in sediments in the Belledune area is not considered to be elevated. Arsenic and copper contaminated sediments near the smelter are restricted to within 1 – 2 km (relative to background). Cadmium levels near the smelter decrease rapidly outside of Belledune Harbour, but appear elevated, relative to background up to 15 km away. Lead and Zn appear to affect a wider area, based on the analysis conducted. Zinc and Pb appear to influence sediments as much as 20 km away from the smelter, relative to background levels. Hg levels are complicated by the multiple sources, and the difficulty in determining source contributions (since both a coal-fired power plant and smelter are located in Belledune, and other sources are present in the bay area).

This analysis found that it was difficult to identify dominant sources of Pb in surface sediments throughout Baie des Chaleurs. Lead levels in surface sediments were summarized as most likely being related to historical combustion of fuels and smelter emissions (particularly in downwind areas), but it was not possible to determine the relative importance of these sources (Parsons and Cranston, 2006).

Environmental Effects Monitoring

Under the Certificate of Approval (C of A) for the facility, Glencore undertakes numerous types of monitoring programs in the marine environment which have been conducted since the early years of operations. This monitoring has included the following:

- Effluent sampling
- Salt water outlet sampling
- Outfall sampling at the east and west diversion ditch outlets
- Native mussel sampling
- Beach sand sampling

- Native mussel culture sampling
- Lobster sampling
- Benthic community and sediment sampling

Recent data (2008 – 2012) from each of these monitoring programs are presented in Appendix A and these data were used to determine sampling areas and needs for the current study.

Port of Belledune Harbour Dredging Project

In 2009 to 2011, a major harbour dredging project was undertaken by the Port of Belledune, to expand the port. The project involved dredging of approximately 170,000 cubic metres of sediment from the harbour (see Figure 3-9; Appendix A). The excavated sediments were placed within 3 cells adjacent to Glencore's property, which were formerly part of the harbour (see Figure 3-9). In total, 16 hectares of land were formed with the filling of these cells, and the inner harbour area which was formerly part of the Environmental Effects Monitoring Program for benthic community impacts was eliminated. Since marine habitat was lost in this project, a habitat compensation was undertaken, which involved the creation of 23,000 artificial lobster reefs, and release of 100,000 larval lobster (Gemtec, 2011). In conjunction with this project, silt curtains were set up to minimize potential dispersion of sediments while cells were being filled. Turbidity was monitored as an indicator of potential sediment release. Despite these safe guards, this dredging project could be a source of recent sediment dispersion into areas east of the harbour, due to the direction of prevailing ocean currents.

Other Studies of Interest

In 2011, a study was initiated to relocate a colony of common terns (*Sterna hirundo*), which is listed as sensitive in New Brunswick. This species had set up roosting and nesting sites in the active industrial area of the facility over the previous several years, and continued to return to the smelter annually. This nesting began during seasonal shut downs which occurred in 2002 through to 2005, wherein shut downs of 2 to 4 months provided an opportunity for terns to establish nesting areas on the roofs of several buildings, and in low lying areas around the facility. The terns usually arrive on-site in mid-May and leave in mid- to late-August. As a result, Glencore began a Tern Management Program in 2009, and decided to actively pursue re-locating the terns to an alternative nesting area, in consultation with Canadian Wildlife Service (CWS). The Tern Management Program included placement of rigid plastic mesh on buildings, installation of a water sprinkler, etc. Considerable effort was undertaken to encourage the terns to relocate to Belledune Point, away from the active industrial areas of the facility, but these efforts were unsuccessful. CWS conducted a nest and clutch survey in 2010, and Glencore collected observational data related to foraging zones, as well as fish tissue concentrations, and dead chicks for metals analysis.

Summary

Based on the information available related to facility emissions, environmental monitoring, and assessments conducted to date, the following was concluded:

- Predominant pathways of release to the marine environment include atmospheric deposition of air emissions, direct effluent, and outfall discharge. In addition, historical erosion of the slag disposal area (which was relocated to areas south of Highway 134 in 2013), as well as groundwater releases are likely contributing factors.
- The predominant chemicals of interest are a variety of metals and metalloids, which will be referred to as metals in this report. Lead, zinc, cadmium, thallium and arsenic are reported in previous studies as being of primary interest (in light of elevated concentrations in various media), but some other metals also exhibit increased concentrations in some areas.
- Recent harbor dredging at the Port of Belledune could be a supplemental source of contaminated sediments.
- Sediment contamination appears to have been more pronounced in early years of operation, as surface sediments have lower metals concentrations than deeper profiles, and the highest levels of contamination have been found in areas close to the facility. Differences between historical background levels and current surface sediment metals vary with metal, but range from 1 to 2 km to up to 20 km, and in some cases, include contributions from other sources (Parsons and Cranston, 2006).
- Environmental releases are predominantly carried in an easterly direction, down the shoreline, due to prevailing winds and ocean currents.
- In general, there is a decreasing trend of metals concentrations in beach sands and native mussels in an easterly direction from the facility, with the most elevated concentrations being within 3 km of the facility. Contamination levels to the west of the facility decrease rapidly along the shore, and are only slightly above reference levels within 2 km.
- Avian species nest on the smelter property and forage in the marine environment and shoreline.

2.3 Receptors of Concern, Conceptual Model, Protection Goals and Assessment and Measurement Endpoints

Appendix A provides details related to the selection of receptors of concern (ROCs), exposure pathways, and the development of the assessment and measurement endpoints and lines of evidence for the risk characterization. To summarize, Table 2-1 provides the rationale used for selection of ROCs, and Figure 2-2 presents the overall conceptual model for the assessment.

Table 2-1	Receptor of Concern (ROC) Selection for the Marine Ecological Risk Assessment			
Aquatic Receptor Group	**Aquatic Receptor Type**	**Included in ERA?**	**Rationale**	**Surrogate ROC**
Primary Producers	Phytoplankton	Yes	Phytoplankton would be expected to be found within the study area.	Assessed as a group
	Macrophyte	No	The heavy wave action and unstable substrate in the vicinity of the site does not make habitat suitable for emergent vegetation. As such, aquatic macrophytes were not included.	Not applicable
Pelagic Invertebrates	Zooplankton	Yes	Zooplankton would be expected to be found within the study area.	Assessed as a group
Benthic Invertebrates	Epifauna / Infauna	Yes	Benthic invertebrates would be expected to be found within the study area in and on sediments.	Assessed as a group
Fish	Benthivorous	Yes	Benthivorous fish could be exposed to Site COCs via eating benthic invertebrates from contaminated sediments or via the incidental ingestion of sediments.	Specific species to be selected under the Fish Health assessment
	Piscivorous	No	Exposures to piscivorous fish are expected to be low given these fish and their food tend to be mobile thereby limiting their exposures related to the sites.	Not applicable
Aquatic Feeding Mammals	Herbivorous	No	Aquatic marine vegetation is not expected to be plentiful in the near-shore area due to poor habitat and wave action; exposures to marine herbivorous mammals from site COCs are expected to be low.	Not applicable
	Piscivorous	No	While piscivorous mammals could be exposed to site COCs via ingestion of contaminated fish, given their large home range, the amount of fish they would ingest from areas affected by smelter releases is expected to be limited, thereby limiting their exposures. In addition, the small size of the site would provide inadequate habitat for an entire population of piscivorous mammals. As such, population level effects to this receptor group would not be expected.	Not applicable
	Omnivorous	No	Aquatic vegetation is not expected to be plentiful in near-shore areas and the amount of fish that omnivorous mammals would ingest from areas affected by smelter releases would expected to be limited and hence exposures to omnivorous mammals from the site are expected to be low. In addition, the small size of the study area would not provide adequate habitat for an entire population of omnivorous mammals. As such, population level effects to this receptor group would not be expected.	Not applicable
Aquatic Feeding Birds	Herbivorous	No	Aquatic vegetation is not expected to be plentiful in the near-shore area due to poor habitat and wave action; exposures to herbivorous mammals from site COCs are expected to be low.	Not applicable
	Invertivorous	Yes	Invertivorous birds feeding in the nearshore were observed within the study area including	Spotted sandpiper

Table 2-1	Receptor of Concern (ROC) Selection for the Marine Ecological Risk Assessment			
Aquatic Receptor Group	**Aquatic Receptor Type**	**Included in ERA?**	**Rationale**	**Surrogate ROC**
			the black-bellied plover, the killdeer and spotted sandpiper. The killdeer and spotted sandpiper have also been observed nesting in Belledune Point (Morneau, 2010). These species are listed as not at risk on the Species at Risk Public Registry (Government of Canada, 2015) and are listed as secure in New Brunswick (NB DNR, 2015). These birds could be exposed to metals in their food and via the incidental ingestion of sediments. As such, invertivorous birds feeding in the nearshore area were included in the ERA. The diet of the killdeer is mainly terrestrial invertebrates, while the spotted sandpiper diet is comprised more of marine and freshwater invertebrates (BNA on-line, 2015). As such, the spotted sandpiper was selected as the surrogate receptor for this group.	*(Actitis macularius)*
	Piscivorous	Yes	A nesting colony of common tern is present on-site and could be exposed to site COCs via the ingestion of fish found within the study area. Double-crested cormorants were also observed feeding offshore of Belledune Point and the black-crowned night heron and great blue heron were observed hunting along the edge of the water, but neither were observed nesting in the area (LGL, 2011; Morneau, 2010). Piscivorous birds were therefore assessed in the ERA. The common term was selected as the surrogate receptor for bird species feeding on pelagic fish as it nests in the area. The common tern is listed as not at risk on the Species at Risk Public Registry (Government of Canada, 2015) but is listed as a sensitive species in New Brunswick (NB DNR, 2015).	Common tern (*Sterna hirundo*)
	Omnivorous	Yes	Aquatic vegetation are not expected to be plentiful in near-shore areas, and as such, was not included as a dietary item. The Black-crowned night heron was selected as a surrogate receptor for avian species that feed on a varied diet in the near-shore area, which could include fish, and near-shore benthic species. The black-crowned night heron is listed as not at risk on the Species at Risk Public Registry (Government of Canada, 2015) but is listed as a sensitive species in New Brunswick (NB DNR, 2015).	Black-crowned night heron (*Nycticorax nycticorax*)
Amphibians	Carnivorous	No	Not expected to be found within marine study area	Not applicable
Reptiles	Omnivorous	No	Not expected to be found within marine study area	Not applicable

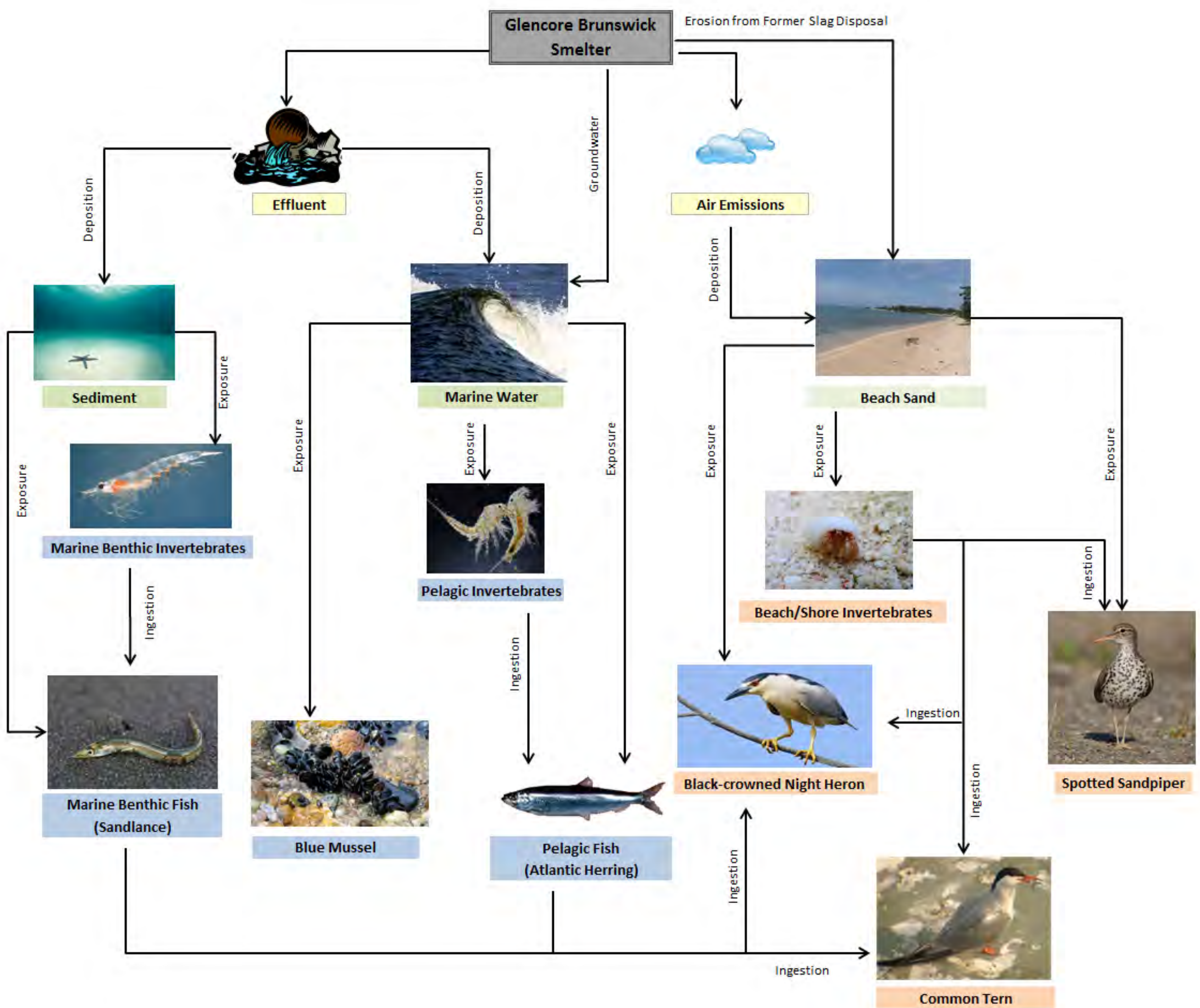

Figure 2-2 Conceptual Site Model for the Marine ERA for the Brunswick Smelter

The Protection Goal for this ERA is to maintain ROC communities / populations similar to background conditions for non-species at risk.

Therefore the protection goal for the common tern, black-crowned night heron and spotted sandpiper is focussed on populations. While the common tern is identified as sensitive in NB, it is not identified as a species at risk and as such, the focus is at the population level. For bird species, published TRVs have been selected as the acceptable effect levels. The TRVs selected are based on lowest-observed-adverse-effect levels (LOAELs) or some minimal level of risk (e.g., EC10 or EC20, where available). Risk is negligible if the estimated contaminant exposures for bird species on-site do not exceed the TRV (i.e., if Hazard Quotient ≤1). Multiple lines of evidence will be used, where available, to draw conclusions with respect to risks.

The protection goal and acceptable effect levels (AELs) for primary producers and invertebrates were at the community level while for fish, they were at the population level. Concentrations of metals in media below established surface water and sediment guidelines in addition to reference area concentrations would be indicative of negligible risk levels. Multiple lines of evidence were used, where available, to draw conclusions with respect to risks.

Assessment endpoints express the environmental value to be protected and include a receptor (what is being protected) and specific property or attribute of that receptor. Measurement endpoints describe (measure) the change in the attribute / property of the assessment endpoint or describe (measure) the exposure or effect for a ROC (FCSAP, 2012). Lines of evidence used to estimate risks to the ROC are based on the measurement endpoints. Table 2-2 presents the assessment and measurement endpoints, as well as the selected lines of evidence for each of the ROCs included in the ERA.

Table 2-2 Assessment Endpoints, Measurement Endpoints and Lines of Evidence

Receptor Group	Assessment Endpoint	Measurement Endpoints	Lines of Evidence
Marine Primary Producer and Pelagic Invertebrate Community	Survival, growth and reproduction of marine primary producer and pelagic communities	Concentrations of COCs in marine surface water Literature related to the toxicity of metal exposures in water on marine invertebrates	Outcomes of the comparison of marine surface water COC concentrations to marine water SWQGs and to reference area concentrations Consider toxicological / biological information from other (literature) studies and extrapolate where applicable to this study.
Marine Benthic Community	Marine benthic community diversity and abundance	Concentrations of COCs in marine sediments Benthic community abundance and diversity study (density; richness and diversity)	Outcomes of the comparison of site sediment COC concentrations to marine Sediment Quality Guidelines (SED QGs) and to reference area concentrations. Statistical analysis of benthic community abundance and diversity endpoints, relative to reference.
Marine Shellfish (i.e., mussel)	Survival and growth of marine shellfish populations	Concentrations of COCs in marine surface water Caged mussel survey: tissue metals analysis; survival (mortalities; age); growth (change in length between deployment/collection); and, condition	Outcomes of the comparison of marine surface water COC concentrations to marine water SWQGs and to reference area concentrations Assessment of caged mussel data relative to control/reference area, with respect to growth, condition and survival endpoints and relative to tissue metals residue data, if available
Marine Fish (pelagic and bottom dwelling)	Survival, growth, reproduction of marine fish populations	Concentrations of COPCs in marine surface water Fish survey (benthic species only; pelagic species not selected due to more limited exposure potential): survival (age; age structure); growth (length-at-age; weight-at-age); reproduction (gonad weight-at-length; fecundity; egg size); condition (weight-at-length; liver size) Fish tissue metals levels (whole fish); Relevant literature, where available	Outcomes of the comparison of marine surface water COC concentrations to marine water SWQGs and to reference area concentrations. Outcomes of fish survey study Consider toxicological / biological information from other (literature) studies and extrapolate where applicable to this study.

Table 2-2 Assessment Endpoints, Measurement Endpoints and Lines of Evidence

Receptor Group	Assessment Endpoint	Measurement Endpoints	Lines of Evidence
Piscivorous avian (i.e., common tern) Populations	Survival, growth, reproduction of piscivorous populations	Exposure modelling; Marine fish (whole fish; pelagic / benthic) tissue concentrations; Bioaccessibility testing of beach sand for selected metals	Predicted ERs from exposure modelling (*i.e.*, comparison of estimated or measured COC exposures via ingestion of fish to TRVs.
		Literature on fish tissue residue effects levels in upper trophic species (piscivores)	Comparison of fish tissue residue data to tissue effects literature for piscivores
		Tissue residue measurements in avian mortalities (e.g., chicks fallen from nests) and rejected eggs	Comparison of liver, kidney or egg tissue residues in avian mortalities to tissue effects literature
		Literature studies discussing effects of COCs on piscivorous avian species at other relevant sites.	Consider toxicological / biological information from other studies and extrapolate where applicable to this study
		Clutch counts	Compare clutch counts to those from other areas to determine if colony is within ranges reported in other areas of New Brunswick
Omnivorous avian (i.e., black-crowned night heron) Populations	Survival, growth, reproduction of piscivorous populations	Exposure modeling; Marine fish (whole fish; benthic / pelagic) tissue concentrations; Beach sand metals concentrations; Bioaccessibility testing of beach sand for selected metals	Predicted ERs from exposure modelling (*i.e.*, comparison of estimated or measured COC exposures via oral ingestion of fish, beach sand, near-shore invertebrates; mussels, etc., to TRVs.
		Literature on fish tissue residue effects levels in upper trophic species (piscivores); Literature studies discussing effects of COCs on similar avian species at other relevant sites	Consider toxicological / biological information from other studies and extrapolate where applicable to this study
		Tissue residues of possible food sources (*e.g.*, near-shore invertebrates, such as scuds) along beach, for input into food chain model (paired with beach sand samples)	

Table 2-2 Assessment Endpoints, Measurement Endpoints and Lines of Evidence

Receptor Group	Assessment Endpoint	Measurement Endpoints	Lines of Evidence
Invertivore avian (i.e., spotted sandpiper) Populations	Survival, growth, and reproduction of avian invertivore populations;	Exposure modelling; Beach sand metals concentrations; Bioaccessibility testing of beach sand for selected metals; Tissue residues of possible food sources (*e.g.,* near-shore invertebrates, such as scuds) along beach, for input into exposure model (paired with beach sand samples)	Predicted ERs from exposure modelling (*i.e.*, comparison of estimated or measured COPC exposures via oral ingestion of beach sand and invertebrates to TRVs.
		Literature studies discussing effects of COCs on similar avian species at other relevant sites	Consider toxicological / biological information from other studies and extrapolate where applicable to this study

Note: SWQGs = Surface water quality guidelines; SedQGs = Sediment quality guidelines; ERs = exposure ratios; TRVs = toxicity reference values

2.4 ERA Strategy and Associated Field Program

To address the data gaps, a field sampling program was developed by Minnow, in consultation with Intrinsik. The data collected through this program are used to characterize risk levels to the various receptor groups associated with releases from the smelter. The Brunswick Smelter field program included five primary components: a sediment quality assessment; a benthic invertebrate community survey; a shellfish health assessment; a fish health survey; and sampling to be used in the modelling of avian receptor in the ERA (fish tissue; shoreline invertebrate tissue; beach sand sampling), as well as supporting water quality and habitat measures required for data interpretation. Study areas, methodology, endpoints and study timing for each of these components are detailed in Appendix A, and the overall types of samples and layout of the sampling locations are provided in Figure 2-3.

Briefly, the majority of the sampling was undertaken in two field trips by Minnow, as follows:

August 2014:

- Caged mussels were deployed at the 2 reference stations, and 4 study area stations (see Figure 2-3), and baseline tissue metals body burdens, as well as all growth and condition parameters were measured. Marine water samples were taken at each station to characterize baseline metals concentrations at time of deployment. Water analysis was conducted by Maxxam Analytics in Burnaby, BC, whereas tissue analysis was undertaken by Research and Productivity Council (RPC) Laboratories, in Fredericton, NB;
- Pelagic and benthic fish were sampled for the avian exposure modelling. These samples were collected in the inshore area at Belledune Point (see Figure 2-3), and included both Atlantic herring, and sand lance. Observations of active avian foraging in the inshore area confirmed that the fishing locations were representative, albeit additional observations also indicated substantial foraging in offshore areas, further out (See Appendix A; Section 2.3.2). These fish samples were analyzed by RPC Laboratories for available metals, based on whole fish composite samples.

October, 2014:

- Caged mussels were collected, 66 days post-deployment. Marine water samples were taken at each of the two reference mussel stations, and at the 4 Study area stations. Mussels were assessed for the various growth, survival and condition endpoints, and tissues were analyzed for available metals (RPC Laboratories);
- Shoreline invertebrates were collected, in conjunction with beach sand samples, along the intertidal zone of the shoreline from the top of Belledune Point to approximately 3 km east of the smelter (see Figure 2-3). For beach sand, composite sampling was undertaken, such that a surface sample of the high-tide, mid-tide and low-tide areas was taken, and combined to form a representative composite for each station. Shoreline invertebrates were collected at the same station by examining the underside of rocks, etc. and collecting a large enough

mass to enable tissue analysis. Both sample types were submitted to RPC Laboratories for available metal analysis. Beach sand samples were sieved using a 2 mm sieve, as per standard soil analysis procedures for metals. For beach sand, a total of 21 separate samples were analyzed, whereas 19 samples were collected for shoreline invertebrates. For assessment purposes, samples were grouped into 3 areas. Area 1 includes samples from Stations 1 to 7 (on Belledune Point; SBS 1 – SBS-7; Figure 2-3); Area 2 includes samples from Stations 8 to 14 (SBS8 – SBS14; Figure 2-3); and Area 3 includes samples from Stations 15 to 21 (SBS 15 – SBS 21; Figure 2-3).

- Sediment and benthic invertebrate samples were collected from 2 reference areas, as well as the Fertilizer Plant Outfall (FPO) and the Final Effluent (FE) discharge area (See Figure 2-3). Sediment samples were submitted to RPC Laboratories for metals characterization and particle size, as well as total organic carbon. Benthic community samples were submitted to Zeas Inc., Nobleton, ON, for abundance and diversity evaluation. An additional series of sediment samples were taken in a transect radiating south east from the smelter, to enable an examination of chemistry with increasing distance from the facility (See Figure 2-3).
- Fishing was undertaken to identify the candidate species for the fish health survey, and to collect adequate fish samples to undertake the survey. Fishing locations are identified in Figure 2-3 for both reference and study area zones.

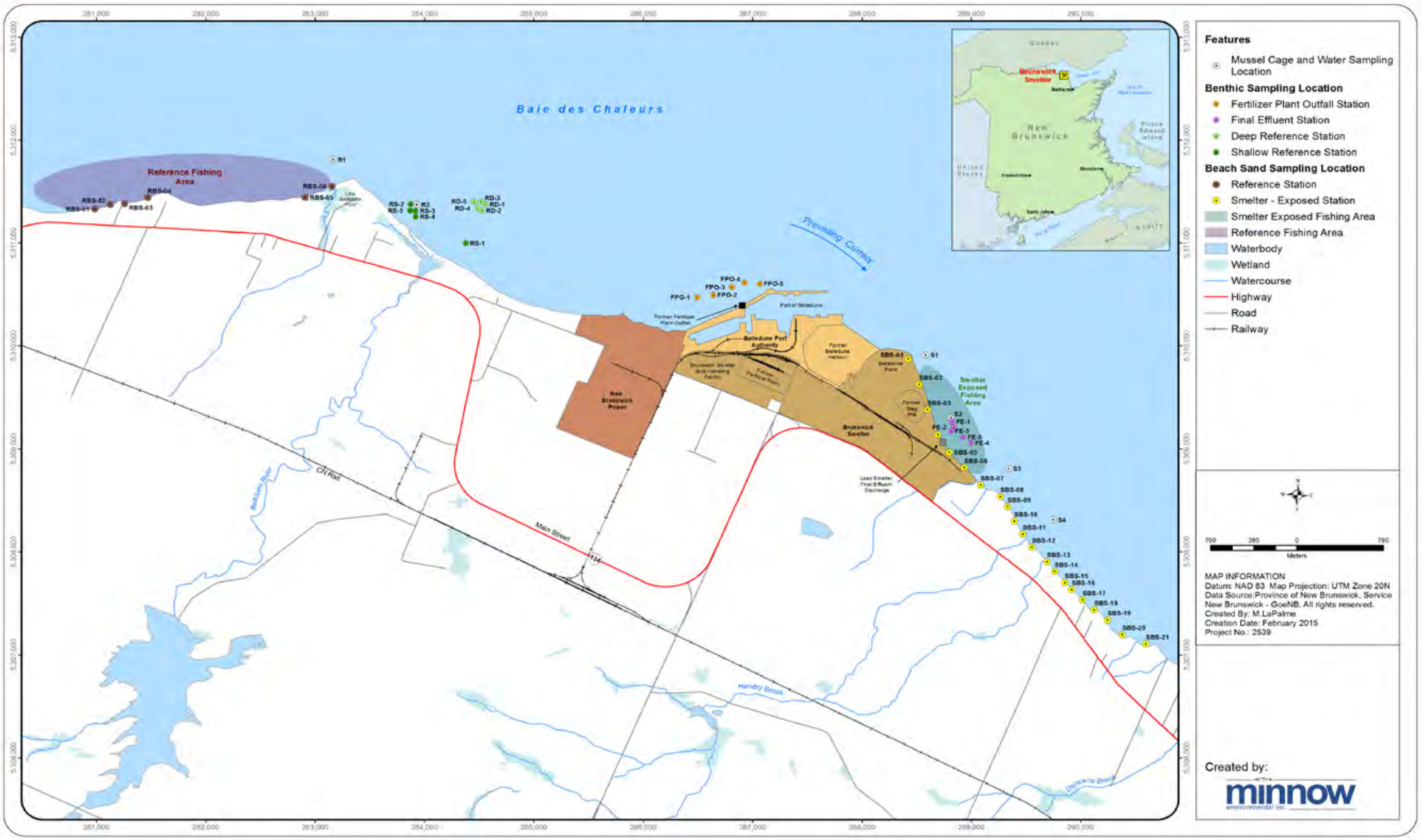

Figure 2-3 Study Area Sampling Stations and Zones included in the 2014 Field Program

Additional sampling was undertaken by Glencore Environment staff, related to the common tern chicks and eggs. Briefly, dead chicks or rejected eggs of common terns nesting on the facility were collected by Glencore staff, to obtain egg metals residue data and chicken internal organ metals residues. Prior to collecting the samples, a permit was obtained from CWS (Canadian Wildlife Services; Scientific Permit #SS2791). Sampling was conducted by Glencore staff listed within the permit, and observations were made to identify and collect rejected eggs or deceased chicks, with minimal disturbance to the colony. Specific locations where chicks and eggs were collected are outlined in Figure 2-4. Upon collection, egg samples were rinsed with distilled water, decanted into sterile plastic vials, and frozen. Chick samples were bagged in plastic baggies, and frozen. Samples were shipped to Research and Productivity Council (RPC) Laboratories for dissection, and analysis of trace metal concentrations, and percent moisture, where possible. Due to the small size of some chick samples, and the condition of some samples upon thawing, composites of organs had to be undertaken in some cases.

Figure 2-4 Locations where Deceased Chicks and Rejected Eggs were Collected in Summer, 2014

Note: Site A = South of lab near fence; Site B = Roof of lab; Site C = Roof of change house; Site D = Small area north of security near fence; Site E = South side of CRP pond; Site F = South of slag settling pond; Site G = Island inside CRP pond; Site H = North parking lot; Site I = Main Office (lawn in front)

3.0 ANALYTICAL CHEMISTRY RESULTS

Summaries of analytical chemistry results for the reference area and study area as a result of the 2014 field program are provided in Sections 3.1 and 3.2, respectively. These data provide statistical summaries of the data, including minimum, maximum, mean, median and 95th percentile of the data for the given sampling areas. Reference data presented are based on combined reference areas, whereas study area data are presented either by sampling zones (Area 1, 2 and 3 for beach sand and shoreline invertebrates) or as a combined study area (for marine waters, sediments, deployed mussels and fish tissues). QA/QC of the data was conducted and results are provided in Section 3.3. Raw analytical data are presented in Appendix B. A map of sampling locations was provided in Figure 2-3. Additional maps are provided in Appendix C.

3.1 Reference Area

The reference area sediment, marine water, beach sand, shoreline invertebrates, marine fish species (Atlantic herring and sand lance) as well as deployed mussels, are characterized by low levels of metals such as cadmium, lead, thallium and zinc, which are associated with facility releases. Statistical analysis of reference data, relative to study area are presented in Section 4.0 for marine waters, sediments and marine mussels. The remaining media concentrations were used in the exposure models to characterize exposures to avian species and are discussed in Section 5.0.

3.1.1 Sediment Data

Table 3-1	Reference Area Sediment Concentrations (mg/kg)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	9200	11500	10597	10900	11500
Antimony	<0.1	0.1	0.1	0.1	0.1
Arsenic	4	6	5.1	5	6
Barium	17	127	72.7	77	123
Beryllium	0.5	0.7	0.58	0.6	0.7
Bismuth	<1	<1	NC	NC	NC
Boron	5	8	6.6	7	8
Cadmium	0.13	0.38	0.282	0.305	0.38
Calcium	2880	31400	16802	15635	30770
Chromium	20	26	24	24.5	26
Cobalt	7.5	9.8	8.75	9	9.7
Copper	7	9	8.4	8.5	9
Iron	13800	18000	16400	16600	17865
Lead	14.8	26.6	21.03	19.95	26.4
Lithium	16	20.4	18.58	18.9	20.2
Magnesium	6930	8510	7858	8030	8474
Manganese	211	370	289.4	311.5	359
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.2	0.7	0.32	0.3	0.6
Nickel	23	29	26.1	26.5	29
Potassium	1060	1420	1297	1310	1402
Rubidium	6.1	8	7.36	7.45	8.0
Selenium	<1	<1	NC	NC	NC
Silver	<0.1	<0.1	NC	NC	NC
Sodium	1870	3090	2444	2405	3000
Strontium	11	25	19.6	20.5	25
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	0.2	0.4	0.3	0.3	0.4
Tin	<1	<1	NC	NC	NC
Uranium	0.5	0.8	0.63	0.65	0.8
Vanadium	28	34	31.5	31.5	34
Zinc	42	56	51.7	53	56

Notes:
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 4
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

3.1.2 Marine Water Data

Table 3-2 Reference Area Marine Water Concentrations (Summer)

Analyte	Units	Min	Max	Mean	Median	95th Percentile
Dissolved Aluminum	µg/L	<10	<10	NC	NC	NC
Dissolved Antimony	µg/L	<0.5	0.5	NC	NC	NC
Dissolved Arsenic	µg/L	1.25	1.33	1.30	1.30	1.33
Dissolved Barium	µg/L	10.4	11.6	10.9	10.9	11.5
Dissolved Beryllium	µg/L	<1	<1	NC	NC	NC
Dissolved Bismuth	µg/L	<1	<1	NC	NC	NC
Dissolved Boron	µg/L	3190	3230	3207.5	3205	3227
Dissolved Cadmium	µg/L	0.055	0.061	0.05875	0.0595	0.061
Dissolved Chromium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Cobalt	µg/L	<0.1	<0.1	NC	NC	NC
Dissolved Copper	µg/L	<0.50	0.96	0.81	0.88	0.96
Dissolved Iron	µg/L	<2	<2	NC	NC	NC
Dissolved Lead	µg/L	<0.10	0.15	0.11	0.10	0.14
Dissolved Lithium	µg/L	120	124	121.75	121.5	123.7
Dissolved Manganese	µg/L	<0.50	0.79	0.57	0.50	0.75
Dissolved Mercury	µg/L	NA	NA	NA	NA	NA
Dissolved Molybdenum	µg/L	9.1	9.7	9.425	9.45	9.685
Dissolved Nickel	µg/L	0.25	1.03	0.56	0.49	0.96
Dissolved Phosphorus	µg/L	<50	<50	NC	NC	NC
Dissolved Selenium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Silicon	µg/L	188	199	195.5	197.5	198.85
Dissolved Silver	µg/L	<0.05	<0.05	NC	NC	NC
Dissolved Strontium	µg/L	6260	6370	6292.5	6270	6355
Dissolved Thallium	µg/L	<0.1	<0.1	NC	NC	NC
Dissolved Tin	µg/L	<1	<1	NC	NC	NC
Dissolved Titanium	µg/L	<10	<10	NC	NC	NC
Dissolved Uranium	µg/L	2.25	2.33	2.28	2.28	2.32
Dissolved Vanadium	µg/L	<10	<10	NC	NC	NC
Dissolved Zinc	µg/L	<1	<1	NC	NC	NC
Dissolved Calcium	mg/L	322	331	327	327	331
Dissolved Magnesium	mg/L	882	910	900	904	909
Dissolved Potassium	mg/L	290	302	297	298	302
Dissolved Sodium	mg/L	7230	7340	7298	7310	7340
Dissolved Sulphur	mg/L	744	785	763	761	782
Total Phosphorus	mg/L	<0.020	<0.020	NC	NC	NC
Dissolved Hardness (CaCO3)	mg/L	4460	4560	4520	4530	4557
Total Nitrogen	mg/L	0.221	0.282	0.253	0.255	0.280

Notes:
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 4
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-3 Reference Area Marine Water Concentrations (Fall)						
Analyte	**Units**	**Min**	**Max**	**Mean**	**Median**	**95^{th} Percentile**
Dissolved Aluminum	µg/L	54	59	58	59	59
Dissolved Antimony	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Arsenic	µg/L	1.27	1.61	1.45	1.46	1.60
Dissolved Barium	µg/L	3.3	7.4	5.2	5.0	7.1
Dissolved Beryllium	µg/L	<1	<1	NC	NC	NC
Dissolved Bismuth	µg/L	<1	<1	NC	NC	NC
Dissolved Boron	µg/L	3280	3540	3420	3430	3527
Dissolved Cadmium	µg/L	0.054	0.077	0.065	0.064	0.075
Dissolved Chromium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Cobalt	µg/L	<0.1	<0.1	NC	NC	NC
Dissolved Copper	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Iron	µg/L	2.6	3.7	3.1	3.1	3.7
Dissolved Lead	µg/L	<0.10	0.16	0.13	0.12	0.16
Dissolved Lithium	µg/L	156	162	160	160	162
Dissolved Manganese	µg/L	1.76	3.09	2.54	2.66	3.07
Dissolved Mercury	µg/L	NA	NA	NA	NA	NA
Dissolved Molybdenum	µg/L	9.7	11.5	10.3	10.1	11.3
Dissolved Nickel	µg/L	0.29	1.23	0.56	0.37	1.11
Dissolved Phosphorus	µg/L	<50	<50	NC	NC	NC
Dissolved Selenium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Silicon	µg/L	<100	<100	NC	NC	NC
Dissolved Silver	µg/L	0.05	0.05	0.05	0.05	0.05
Dissolved Strontium	µg/L	6540	7010	6733	6690	6962
Dissolved Thallium	µg/L	<0.10	0.16	0.12	0.11	0.15
Dissolved Tin	µg/L	<1	<1	NC	NC	NC
Dissolved Titanium	µg/L	<10	<10	NC	NC	NC
Dissolved Uranium	µg/L	2.63	2.74	2.69	2.69	2.74
Dissolved Vanadium	µg/L	<10	<10	NC	NC	NC
Dissolved Zinc	µg/L	1.6	1.9	1.7	1.7	1.9
Dissolved Calcium	mg/L	344	369	353	350	367
Dissolved Magnesium	mg/L	979	1040	1007	1005	1036
Dissolved Potassium	mg/L	319	343	331	330	341
Dissolved Sodium	mg/L	8000	8690	8305	8265	8644
Dissolved Sulphur	mg/L	812	1050	894	857	1022
Dissolved Hardness (CaCO3)	mg/L	4890	5210	5028	5005	5183
Total Nitrogen	mg/L	0.152	0.182	0.169	0.171	0.182

Notes:
Data collected by Minnow in October, 2014 (phosphorus was not analyzed in October, 2014 as phosphorus was not detected in all in August, 2014); Raw data are provided in Appendix B
N = 4
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95^{th} percentile calculated assuming non-detectable samples were equal to the reportable detection limit

3.1.3 Beach Sand Data

Table 3-4	Reference Area Beach Sand Concentrations (mg/kg)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	6670	9270	8025	8040	9110
Antimony	<0.1	0.2	0.12	0.1	0.18
Arsenic	2	5	3.3	3	4.8
Barium	9	17	11.5	10.5	15.8
Beryllium	0.4	0.5	0.42	0.4	0.48
Bismuth	<1	<1	NC	NC	NC
Boron	3	4	3.2	3	3.8
Cadmium	0.04	0.09	0.1	0.045	0.1
Calcium	6690	29700	14080	12500	26350
Chromium	11	18	16.2	17.5	18.0
Cobalt	5.3	7.3	6.08	6	6.98
Copper	5	9	6.8	6.5	8.8
Iron	10800	14100	12250	12150	13950
Lead	5.1	7.1	6.1	6.3	7.0
Lithium	11	16.5	14.1	14.6	16.4
Magnesium	4270	6930	5852	5950	6833
Manganese	224	279	247.2	245.5	274.3
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.1	0.2	0.17	0.2	0.2
Nickel	13	21	17.8	18	20.8
Potassium	810	1090	953	945	1078
Rubidium	4.2	6.5	5.65	5.95	6.5
Selenium	<1	<1	NC	NC	NC
Silver	<0.1	<0.1	NC	NC	NC
Sodium	1200	1950	1485	1355	1908
Strontium	10	29	16.8	13	28
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	<0.1	<0.1	NC	NC	NC
Tin	<1	<1	NC	NC	NC
Uranium	0.3	0.5	0.33	0.3	0.45
Vanadium	19	26	22.0	21.5	25.5
Zinc	23	36	31.7	33	36.0

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 6
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

3.1.4 Biota

Table 3-5	Reference Area Marine Shoreline Invertebrate Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	79.4	232	133.83	111.95	220.25
Antimony	<0.01	0.02	0.01	0.01	0.0175
Arsenic	0.6	0.8	0.7	0.7	0.8
Barium	10.1	42.8	28.73	28.85	41.65
Beryllium	<0.01	<0.01	NC	NC	NC
Bismuth	<0.1	<0.1	NC	NC	NC
Boron	7.2	9.7	8.73	9.2	9.7
Cadmium	0.029	0.066	0.04	0.0415	0.062
Calcium	12200	16100	14550	14750	16000
Chromium	0.2	0.4	0.28	0.25	0.4
Cobalt	0.04	0.1	0.07	0.06	0.095
Copper	2	5.6	3.45	3.15	5.3
Iron	96	226	150.17	140	220
Lead	0.3	0.8	0.52	0.47	0.79
Lithium	0.21	0.35	0.25	0.235	0.325
Magnesium	887	1360	1191.17	1240	1337.5
Manganese	5.4	10.6	7.15	6.2	10.025
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.05	0.06	0.06	0.06	0.06
Nickel	0.2	0.3	0.23	0.2	0.3
Potassium	592	1580	1032.00	873	1562.5
Rubidium	0.43	0.68	0.55	0.52	0.6775
Selenium	0.1	0.2	0.15	0.15	0.2
Silver	0.03	0.07	0.05	0.05	0.0675
Sodium	3770	7070	6201.67	6685	7045
Strontium	186	228	212.50	215.5	227
Tellurium	<0.01	<0.01	NC	NC	NC
Thallium	<0.01	<0.01	NC	NC	NC
Tin	0.02	0.09	0.05	0.04	0.08
Uranium	0.02	0.03	0.02	0.02	0.0275
Vanadium	0.3	0.6	0.43	0.4	0.6
Zinc	6	10.6	7.73	7.2	10.1

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 6
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-6	Reference Area Atlantic Herring Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	0.9	3.3	2.04	1.6	3.24
Antimony	<0.005	<0.005	NC	NC	NC
Arsenic	0.4	0.6	0.532	0.55	0.594
Barium	<0.05	0.07	0.062	0.06	0.07
Beryllium	<0.005	<0.005	NC	NC	NC
Bismuth	<0.05	<0.05	NC	NC	NC
Boron	0.6	1.31	1.01	1.06	1.27
Cadmium	0.0438	0.095	0.0658	0.0624	0.0927
Calcium	3460	7480	5802	6100	7300
Chromium	<0.05	<0.05	NC	NC	NC
Cobalt	<0.005	0.008	0.007	0.007	0.008
Copper	0.54	0.89	0.754	0.79	0.87
Iron	12	21	17.4	19	20.8
Lead	0.045	0.17	0.0956	0.072	0.163
Lithium	0.024	0.043	0.0378	0.042	0.0428
Magnesium	369	623	555.2	600	622
Manganese	1.19	2.9	2.17	2.61	2.86
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.01	0.018	0.0152	0.016	0.0176
Nickel	<0.05	0.07	0.064	0.07	0.07
Potassium	2670	3880	3390	3600	3864
Rubidium	0.539	0.808	0.703	0.752	0.804
Selenium	0.32	0.5	0.418	0.44	0.488
Silver	<0.005	0.005	0.005	0.005	0.005
Sodium	1300	2400	2018	2130	2356
Strontium	8.06	16.1	13.3	15.2	16
Tellurium	<0.005	<0.005	NC	NC	NC
Thallium	<0.005	<0.005	NC	NC	NC
Tin	<0.005	0.017	0.0076	0.005	0.0148
Uranium	<0.005	<0.005	NC	NC	NC
Vanadium	<0.05	<0.05	NC	NC	NC
Zinc	15.7	29.4	24	25.2	28.6

Notes:
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 5
Percent Moisture ranged from 73.9% to 78.9% with an average % moisture concentration of 76.4%
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-7	Reference Area Sand Lance Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	1.4	2.3	1.85	1.75	2.3
Antimony	<0.005	<0.005	NC	NC	NC
Arsenic	0.64	1.12	0.809	0.79	1.07
Barium	0.1	0.26	0.162	0.155	0.242
Beryllium	<0.005	<0.005	NC	NC	NC
Bismuth	<0.05	0.05	0.05	0.05	0.05
Boron	0.61	1.04	0.798	0.79	0.995
Cadmium	0.0363	0.102	0.0709	0.0708	0.0964
Calcium	3960	6670	5286	5360	6477
Chromium	<0.05	0.05	0.05	0.05	0.05
Cobalt	0.007	0.008	0.0075	0.0075	0.008
Copper	0.64	0.77	0.719	0.735	0.77
Iron	16	23	18.6	18	23
Lead	0.016	0.039	0.026	0.025	0.0377
Lithium	0.042	0.091	0.0578	0.055	0.0807
Magnesium	460	572	506	496	559
Manganese	1.84	3.47	2.52	2.54	3.25
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.013	0.019	0.0154	0.0155	0.0186
Nickel	0.05	0.07	0.06	0.06	0.07
Potassium	3450	3900	3774	3830	3891
Rubidium	0.774	0.998	0.943	0.974	0.997
Selenium	0.52	0.61	0.575	0.575	0.61
Silver	<0.005	<0.005	NC	NC	NC
Sodium	1420	2080	1776	1795	2062
Strontium	12.8	21.6	17.1	17.3	21.1
Tellurium	<0.005	<0.005	NC	NC	NC
Thallium	<0.005	0.007	0.0052	0.005	0.0061
Tin	<0.005	0.024	0.0111	0.009	0.0227
Uranium	<0.005	0.01	0.0056	0.005	0.0082
Vanadium	<0.05	<0.05	NA	NA	NA
Zinc	25.9	38	30.04	29.5	35.2

Notes:
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 10
Percent Moisture ranged from 73.5% to 76.7% with an average % moisture concentration of 75.0%
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-8	Pre-Deployment (Reference) Marine Mussel Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	25.2	99	53.6	47.6	89.4
Antimony	<0.005	0.01	0.01	0.005	0.008
Arsenic	1.31	1.62	1.45	1.40	1.62
Barium	0.22	0.61	0.36	0.30	0.59
Beryllium	<0.005	<0.005	NC	NC	NC
Bismuth	<0.05	0.07	0.05	0.05	0.061
Boron	4.24	5.44	4.84	4.95	5.37
Cadmium	0.252	0.317	0.28	0.281	0.31
Calcium	606	2460	902	731	1769
Chromium	0.13	0.28	0.19	0.165	0.262
Cobalt	0.084	0.126	0.10	0.101	0.124
Copper	1.26	1.64	1.49	1.50	1.62
Iron	34	91	56	51	84
Lead	0.211	0.418	0.31	0.315	0.394
Lithium	0.073	0.12	0.09	0.089	0.118
Magnesium	491	602	543	546	585
Manganese	2.41	3.75	3.06	2.94	3.65
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.088	0.19	0.13	0.12	0.17
Nickel	0.29	0.45	0.36	0.36	0.44
Potassium	1810	2120	1977	2010	2111
Rubidium	0.92	1.08	1.00	0.996	1.08
Selenium	0.59	0.8	0.69	0.72	0.77
Silver	0.018	0.027	0.02	0.024	0.027
Sodium	2200	3100	2733	2740	3060
Strontium	4.56	8.9	5.53	5.18	8.02
Tellurium	<0.005	<0.005	NC	NC	NC
Thallium	<0.005	<0.005	NC	NC	NC
Tin	0.006	0.074	0.02	0.009	0.048
Uranium	0.021	0.071	0.05	0.043	0.064
Vanadium	0.2	0.4	0.28	0.25	0.40
Zinc	13.9	20.4	17.31	18.05	20.18

Notes:
Data obtained by Minnow in August, 2014 (pre-deployment sample concentrations); Raw data are provided in Appendix B
N = 10
Percent Moisture ranged from 79.6% to 84.1% with an average % moisture concentration of 82.1%
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-9	66-Day Post Deployment (Reference Area) Marine Mussel Concentrations (mg/kg dw)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	122	693	298	202	613
Antimony	<0.1	0.1	0.1	0.1	0.1
Arsenic	5	12	8.4	8	11.6
Barium	2	17	7.6	6	15.7
Beryllium	<0.1	<0.1	NC	NC	NC
Bismuth	<1	<1	NC	NC	NC
Boron	18	30	20.6	19.5	26.85
Cadmium	0.56	2.15	1.30	1.245	2.01
Calcium	1280	5150	2477	2120	4471
Chromium	<1	2	1.1	1	1.55
Cobalt	0.2	0.6	0.36	0.4	0.56
Copper	4	9	6	6	8.1
Iron	130	650	294	220	574
Lead	1.1	3.3	2.0	1.7	3.1
Lithium	0.4	1.2	0.6	0.55	0.97
Magnesium	2300	4830	3007	2750	4349
Manganese	5	23	11	8.5	20
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.2	0.6	0.39	0.35	0.56
Nickel	<1	2	1.4	1	2
Potassium	9470	13300	10660	10500	12715
Rubidium	3.9	6.1	4.8	4.6	5.8
Selenium	2	4	3	3	4
Silver	<0.1	0.1	0.1	0.1	0.1
Sodium	13000	32400	19480	17550	30150
Strontium	16	47	24.6	21.5	39.8
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	<0.1	<0.1	NC	NC	NC
Tin	<0.1	<0.1	NC	NC	NC
Uranium	<0.1	0.3	0.12	0.1	0.2
Vanadium	<1	3	1.8	1.5	3
Zinc	32	146	79	78	137

Notes:
Data obtained by Minnow in October, 2014 (66 days post-deployment); Raw data are provided in Appendix B
N = 10
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; dw = dry weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

3.2 Study Area

Samples collected from the study area show influences from the operations of the Brunswick smelting facility, particularly for predominant metals of interest, which include cadmium, lead, thallium and zinc. Assessment of the data, relative to reference and appropriate marine quality guidelines is presented in Section 4.2 (marine water quality data), Section 4.3 (sediment quality data), Section 4.4 (mussel tissue residues), and Section 4.5 (marine fish tissue residues). In addition, data collected for the exposure modelling aspects of the ERA, including beach sand, marine fish and shoreline invertebrates, are evaluated in Section 5.0. Similarly, common tern chick organ metal residues and egg metal residues, are evaluated in Section 5.0, relative to toxicity thresholds.

3.2.1 Sediment data

Table 3-10 Study Area Sediment Concentrations in the Vicinity of the Fertilizer Plant Outfall (FPO Area)(mg/kg)

Analyte	Min	Max	Mean	Median	95th Percentile
Aluminum	1530	13900	6782	5180	13160
Antimony	0.1	1.1	0.5	0.5	1
Arsenic	1	8	4.6	5	7.8
Barium	12	69	45.6	54	66.8
Beryllium	0.1	0.7	0.38	0.4	0.66
Bismuth	<1	<1	NC	NC	NC
Boron	2	17	7.8	5	16
Cadmium	0.11	0.83	0.578	0.7	0.828
Calcium	16400	158000	54020	19900	138100
Chromium	4	31	16.8	19	29.4
Cobalt	0.5	10.4	4.4	3.5	9.66
Copper	5	21	12	13	20
Iron	710	20400	8348	5970	18840
Lead	13.8	192	79.5	66.3	170.18
Lithium	0.5	22.7	8.8	4.8	21.08
Magnesium	530	9310	3768	2100	8656
Manganese	8	294	123.4	72	275.8
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.3	1.2	0.8	0.8	1.18
Nickel	2	33	13.2	8	30.6
Potassium	360	1940	1022	940	1852
Rubidium	1	9.9	4.84	4.3	9.44
Selenium	<1	<1	NC	NC	NC
Silver	<0.1	0.2	0.12	0.1	0.18
Sodium	1990	5230	3534	3700	5012
Strontium	34	504	177.2	65	447
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	<0.1	0.4	0.28	0.3	0.4
Tin	<1	3	1.4	1	2.6
Uranium	3.1	70.8	25.06	18.5	61.26
Vanadium	2	37	17	14	34.4
Zinc	30	556	178.4	108	469.6

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B;
N = 5

TOC (total organic carbon): Min = <1%; Max = 1.3%; Mean = 0.48%
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; FPO = fertilizer plant outfall
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-11 Study Area Sediment Concentrations in the Vicinity of the Final Effluent (FE Area) (mg/kg)

Analyte	Min	Max	Mean	Median	95th Percentile
Aluminum	9850	10500	10170	10200	10440
Antimony	0.2	1	0.52	0.4	0.92
Arsenic	12	34	20.8	19	32
Barium	56	183	130	133	176.4
Beryllium	0.4	0.4	0.4	0.4	0.4
Bismuth	<1	4	2	2	3.6
Boron	6	8	7	7	7.8
Cadmium	1.47	2.64	2.14	2.02	2.638
Calcium	7860	12000	10284	11100	11860
Chromium	31	35	33.6	34	34.8
Cobalt	9.4	13.4	11.18	10.9	13.08
Copper	20	77	44	37	72.8
Iron	16000	20000	17820	17500	19620
Lead	206	860	474	374	806.8
Lithium	13.6	14.8	14.22	14.3	14.76
Magnesium	8150	8750	8554	8610	8734
Manganese	220	249	240	243	249
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.2	0.4	0.3	0.3	0.38
Nickel	27	29	28.6	29	29
Potassium	940	1180	1036	1040	1156
Rubidium	5.1	6.4	5.56	5.5	6.24
Selenium	<1	<1	NC	NC	NC
Silver	<0.1	0.3	0.16	0.1	0.28
Sodium	1870	2430	2168	2090	2426
Strontium	19	24	20.8	20	23.4
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	1.5	3.1	2.08	1.7	2.98
Tin	<1	7	3.2	3	6.2
Uranium	0.4	0.6	0.48	0.5	0.58
Vanadium	35	39	36.6	36	38.8
Zinc	326	1840	970.6	844	1722

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 5
TOC (total organic carbon): Min = 0.3%; Max = 0.5%; Mean = 0.38%
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; FE = final effluent
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-12	**Study Area Sediment Concentrations in the Vicinity of the Smelter Sediment Transect (SST2 Area) (mg/kg)**				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	9370	9980	9624	9590	9924
Antimony	<0.1	0.2	0.14	0.1	0.2
Arsenic	8	10	9.2	9	10
Barium	37	228	99.2	74	202.8
Beryllium	0.3	0.4	0.38	0.4	0.4
Bismuth	<1	<1	NC	NC	NC
Boron	5	6	5.8	6	6
Cadmium	0.7	1.57	0.904	0.76	1.41
Calcium	5360	7280	6092	6000	7048
Chromium	29	31	30	30	30.8
Cobalt	8.6	9.3	8.92	8.9	9.26
Copper	12	21	15.4	15	20
Iron	14700	15500	15000	14900	15440
Lead	98.1	162	128.0	117	159
Lithium	13.1	14.5	13.7	13.7	14.36
Magnesium	7890	8280	8082	8070	8270
Manganese	217	229	221.4	220	227.8
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.1	0.4	0.24	0.2	0.38
Nickel	27	28	27.4	27	28
Potassium	970	1070	1020	1010	1064
Rubidium	5.2	5.8	5.52	5.5	5.76
Selenium	<1	<1	NC	NC	NC
Silver	<0.1	<0.1	NC	NC	NC
Sodium	2170	2650	2388	2320	2642
Strontium	16	19	16.8	16	18.6
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	0.8	1.7	1.02	0.9	1.54
Tin	<1	<1	NC	NC	NC
Uranium	0.4	0.6	0.56	0.6	0.6
Vanadium	34	37	34.6	34	36.4
Zinc	101	238	177.4	182	235.8

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 5
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; SST = smelter sediment transect
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

3.2.2 Marine Water Data

Table 3-13 Study Area Marine Water Concentrations (Summer)

Analyte	Units	Min	Max	Mean	Median	95th Percentile
Dissolved Aluminum	µg/L	13	19	15	15	18
Dissolved Antimony	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Arsenic	µg/L	1.37	2.27	1.58	1.46	2.10
Dissolved Barium	µg/L	10.2	11.3	10.6	10.4	11.2
Dissolved Beryllium	µg/L	<1	<1	NC	NC	NC
Dissolved Bismuth	µg/L	<1	<1	NC	NC	NC
Dissolved Boron	µg/L	3260	3470	3353	3340	3463
Dissolved Cadmium	µg/L	0.058	0.989	0.207	0.096	0.696
Dissolved Chromium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Cobalt	µg/L	<0.1	<0.1	NC	NC	NC
Dissolved Copper	µg/L	<0.50	2.49	0.92	0.70	1.93
Dissolved Iron	µg/L	<2	4	2	2	3.7
Dissolved Lead	µg/L	0.24	1.60	0.62	0.46	1.40
Dissolved Lithium	µg/L	130	138	134	133	137
Dissolved Manganese	µg/L	1.14	6.00	2.79	2.47	5.32
Dissolved Mercury	µg/L	NA	NA	NA	NA	NA
Dissolved Molybdenum	µg/L	8.7	9.3	9.0	9.0	9.3
Dissolved Nickel	µg/L	0.29	0.63	0.44	0.46	0.58
Dissolved Phosphorus	µg/L	<50	<50	NC	NC	NC
Dissolved Selenium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Silicon	µg/L	138	201	156	154	187
Dissolved Silver	µg/L	<0.05	<0.05	NC	NC	NC
Dissolved Strontium	µg/L	5950	6250	6086	6070	6240
Dissolved Thallium	µg/L	0.10	3.30	0.66	0.28	2.39
Dissolved Tin	µg/L	<1	<1	NC	NC	NC
Dissolved Titanium	µg/L	<10	<10	NC	NC	NC
Dissolved Uranium	µg/L	2.38	2.54	2.45	2.45	2.53
Dissolved Vanadium	µg/L	<10	<10	NC	NC	NC
Dissolved Zinc	µg/L	1.2	8.4	3.2	2.6	6.9
Dissolved Calcium	mg/L	304	362	329	327	352
Dissolved Magnesium	mg/L	866	896	879	882	893
Dissolved Potassium	mg/L	272	296	289	293	295
Dissolved Sodium	mg/L	7130	7340	7228	7220	7337
Dissolved Sulphur	mg/L	699	774	739	746	766
Total Phosphorus	mg/L	<0.02	<0.02	NC	NC	NC
Dissolved Hardness (CaCO3)	mg/L	4380	4490	4439	4450	4487
Total Nitrogen	mg/L	0.190	0.285	0.233	0.231	0.277

Notes:
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 8
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-14 Study Area Marine Water Concentrations (Fall)						
Analyte	**Units**	**Min**	**Max**	**Mean**	**Median**	**95^{th} Percentile**
Dissolved Aluminum	µg/L	56	64	59	59	62
Dissolved Antimony	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Arsenic	µg/L	1.46	1.78	1.66	1.68	1.76
Dissolved Barium	µg/L	6.9	8.0	7.6	7.6	8.0
Dissolved Beryllium	µg/L	<1	<1	NC	NC	NC
Dissolved Bismuth	µg/L	<1	<1	NC	NC	NC
Dissolved Boron	µg/L	3270	3560	3424	3445	3539
Dissolved Cadmium	µg/L	0.100	0.256	0.174	0.162	0.252
Dissolved Chromium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Cobalt	µg/L	<0.1	<0.1	NC	NC	NC
Dissolved Copper	µg/L	<0.5	1.1	0.6	0.5	0.9
Dissolved Iron	µg/L	2.0	15.4	4.7	3.2	11.9
Dissolved Lead	µg/L	0.43	1.10	0.70	0.61	1.09
Dissolved Lithium	µg/L	156	163	160	160	163
Dissolved Manganese	µg/L	2.70	4.33	3.71	3.95	4.28
Dissolved Mercury	µg/L	NA	NA	NA	NA	NA
Dissolved Molybdenum	µg/L	9.1	10.2	9.8	9.9	10.2
Dissolved Nickel	µg/L	0.30	1.95	0.73	0.59	1.60
Dissolved Phosphorus	µg/L	<50	<50	NC	NC	NC
Dissolved Selenium	µg/L	<0.5	<0.5	NC	NC	NC
Dissolved Silicon	µg/L	<100	124	106	104	119
Dissolved Silver	µg/L	<0.05	<0.05	NC	NC	NC
Dissolved Strontium	µg/L	6530	6930	6731	6760	6902
Dissolved Thallium	µg/L	0.31	3.44	1.40	0.79	3.39
Dissolved Tin	µg/L	<1	<1	NC	NC	NC
Dissolved Titanium	µg/L	<10	<10	NC	NC	NC
Dissolved Uranium	µg/L	2.59	2.88	2.74	2.75	2.85
Dissolved Vanadium	µg/L	<10	<10	NC	NC	NC
Dissolved Zinc	µg/L	3.1	5.6	4.4	4.0	5.5
Dissolved Calcium	mg/L	348	364	356	357	363
Dissolved Magnesium	mg/L	960	1040	1002	1007	1033
Dissolved Potassium	mg/L	322	340	333	335	340
Dissolved Sodium	mg/L	8180	8690	8469	8465	8687
Dissolved Sulphur	mg/L	795	878	854	863	876
Dissolved Hardness (CaCO3)	mg/L	4840	5160	5011	5030	5139
Total Nitrogen	mg/L	0.136	0.223	0.166	0.161	0.210

Notes:
Data collected by Minnow in October, 2014 (phosphorus not analyzed in October, 2014 as it was not detected in all samples in August, 2014); Raw data are provided in Appendix B
N = 8
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95^{th} percentile calculated assuming non-detectable samples were equal to the reportable detection limit

3.2.3 Beach Sand Data

Table 3-15	Beach Sand Concentrations on Belledune Point (Area 1) (mg/kg)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	12500	15200	14043	13700	15200
Antimony	3.1	12.3	8.2	8.9	12.3
Arsenic	76	400	220.1	207.0	378.4
Barium	38	209	122.9	131.0	205.1
Beryllium	0.5	0.7	0.56	0.5	0.67
Bismuth	1	30	8.1	5.0	24
Boron	6	14	9.4	10.0	13.4
Cadmium	5.32	20.8	12.03	10.9	20.38
Calcium	8810	35900	24701	28200	35360
Chromium	37	57	46.1	48.0	55.2
Cobalt	25.1	89.9	55.97	56.8	87.98
Copper	168	999	611.3	783.0	977.1
Iron	36300	102000	67986	68600	100380
Lead	1830	19600	7500	7300	16519
Lithium	16.1	20.7	18.44	18.8	20.61
Magnesium	9310	12900	10979	10900	12870
Manganese	403	597	507.9	490.0	593.7
Mercury	<0.01	0.03	0.02	0.01	0.027
Molybdenum	1.9	13.7	7.44	7.4	13.22
Nickel	25	38	31.7	33.0	37.4
Potassium	920	1470	1240	1260	1422
Rubidium	4.7	7.7	6.71	7.0	7.67
Selenium	<1	4	2.3	2.0	4
Silver	0.2	2	1.04	1.1	1.85
Sodium	1280	2050	1684	1680	1981
Strontium	22	61	43.0	47.0	60.4
Tellurium	<0.1	0.9	0.33	0.2	0.75
Thallium	2	33	9.29	3.5	27.24
Tin	20	164	89.9	97.0	158.6
Uranium	0.4	1	0.73	0.7	1
Vanadium	47	61	52.1	52.0	58.9
Zinc	5570	36700	21096	22900	36370

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 7
< = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-16	Beach Sand Concentrations in the Vicinity of Belledune Point (Area 2) (mg/kg)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	12100	16200	14371	14500	15840
Antimony	<0.1	0.2	0.1	0.1	0.17
Arsenic	11	25	16.0	14.0	23.5
Barium	11	35	18.6	15.0	30.5
Beryllium	0.4	0.4	0.40	0.4	0.4
Bismuth	<1	<1	NC	NC	NC
Boron	4	5	4.6	5.0	5
Cadmium	0.23	0.47	0.30	0.3	0.428
Calcium	16600	32100	23986	25200	30810
Chromium	34	53	43.4	41.0	52.4
Cobalt	11	15.1	13.26	13.3	14.77
Copper	12	18	14.9	14.0	17.7
Iron	18300	24900	22200	22800	24780
Lead	37.3	90.2	56.27	51.2	82.61
Lithium	16	19.1	17.67	18.1	19.07
Magnesium	11100	15400	13214	13400	14950
Manganese	301	426	362	362	411
Mercury	<0.01	0.01	0.01	0.01	0.01
Molybdenum	<0.1	0.2	0.14	0.1	0.2
Nickel	30	43	36.3	35.0	42.4
Potassium	730	890	793	780	881
Rubidium	3.8	5.1	4.33	4.3	5.01
Selenium	<1	<1	NC	NC	NC
Silver	<0.1	<0.1	NC	NC	NC
Sodium	1410	1930	1757	1830	1918
Strontium	32	41	37.1	38.0	40.7
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	0.5	0.9	0.60	0.5	0.84
Tin	<1	<1	NC	NC	NC
Uranium	0.2	0.3	0.29	0.3	0.3
Vanadium	46	63	52.9	52.0	62.1
Zinc	77	246	116	101	205

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 7
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-17	Beach Sand Concentrations in the Vicinity of Belledune Point (Area 3) (mg/kg)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	14200	18500	16157	16000	18230
Antimony	<0.1	0.4	0.2	0.1	0.34
Arsenic	12	28	19.9	21.0	27.7
Barium	15	251	70.7	39.0	201.2
Beryllium	0.3	0.4	0.39	0.4	0.4
Bismuth	<1	<1	NC	NC	NC
Boron	5	14	7.3	6.0	12.2
Cadmium	0.28	1.35	0.59	0.5	1.179
Calcium	20400	37100	27429	25600	36200
Chromium	46	62	52.4	50.0	61.1
Cobalt	14.6	18.1	16.01	15.7	17.83
Copper	14	28	21.4	22.0	27.7
Iron	23400	27100	25300	25400	26890
Lead	34.7	165	81.30	67.9	149.1
Lithium	15.5	20.6	18.50	18.6	20.57
Magnesium	12300	17100	14986	14800	17070
Manganese	373	458	408	407	451
Mercury	<0.01	0.02	0.01	0.01	0.02
Molybdenum	<0.1	0.3	0.14	0.1	0.27
Nickel	34	45	40.4	40.0	44.4
Potassium	620	840	754	750	834
Rubidium	3.3	4.5	3.87	3.8	4.38
Selenium	<1	<1	NC	NC	NC
Silver	<0.1	<0.1	NC	NC	NC
Sodium	1290	2760	1814	1760	2520
Strontium	29	65	48.7	49.0	64.7
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	0.3	0.8	0.51	0.5	0.77
Tin	<1	<1	NC	NC	NC
Uranium	0.2	0.3	0.29	0.3	0.3
Vanadium	54	74	62.4	61.0	72.8
Zinc	92	564	219	178	474

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 7
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

3.2.4 Biota Data

Table 3-18	Shoreline Invertebrate Concentrations on Belledune Point (Area 1)(mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	55.7	157	108.5	112	153.3
Antimony	0.07	0.15	0.11	0.1	0.15
Arsenic	1	3.6	1.8	1.2	3.4
Barium	12	26.5	20.0	22.3	25.5
Beryllium	<0.01	<0.02	NC	NC	NC
Bismuth	<0.1	0.2	0.13	0.1	0.2
Boron	5.7	14.2	9.1	7.6	14.0
Cadmium	0.298	2.3	0.87	0.689	1.923
Calcium	15100	49500	26367	17550	47925
Chromium	<0.2	0.4	0.30	0.3	0.4
Cobalt	0.08	0.18	0.12	0.11	0.17
Copper	6.2	49.2	21.0	13.9	44.5
Iron	92	190	139.8	145.5	185.3
Lead	13.8	37.5	23.0	20	36.2
Lithium	0.23	0.34	0.29	0.295	0.335
Magnesium	1160	3040	1803	1395	2945
Manganese	8.3	13.5	9.7	8.9	126
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.05	0.15	0.09	0.07	0.15
Nickel	0.2	0.4	0.25	0.2	0.375
Potassium	766	4260	2231	1625	4173
Rubidium	0.44	1.82	0.98	0.71	1.76
Selenium	0.2	0.7	0.38	0.25	0.7
Silver	0.37	2.44	0.86	0.57	2.02
Sodium	6370	16000	9638	7205	15600
Strontium	205	733	367	237	695
Tellurium	<0.01	0.01 [2]	0.01	0.01	0.02
Thallium	0.17	1.72	0.64	0.55	1.4
Tin	0.08	0.34	0.15	0.11	0.29
Uranium	<0.01 [1]	0.02	0.02	0.02	0.02
Vanadium	0.3	0.7	0.45	0.45	0.65
Zinc	18.8	59.9	36.40	31.5	59

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 6
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit
1. One sample had a non-detectable concentration, but the reportable detection limit (<0.02 mg/kg) was greater than the lowest detected concentration of 0.01 mg/kg. As such, the minimum value is reported as 0.01 mg/kg.
2. Two samples were not detected at a reportable detection limit of <0.02 mg/kg. This value was greater than the highest detected value. As such, the maximum concentration was reported as the highest detected value of 0.01 mg/kg

Table 3-19	Shoreline Invertebrate Concentrations on Belledune Point (Area 2)(mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	16.1	430	98.3	52.6	308.9
Antimony	0.01	0.14	0.04	0.02	0.105
Arsenic	0.6	3.1	1.26	1.15	2.47
Barium	2	20.5	5.44	3.5	15.25
Beryllium	<0.01	<0.02	NC	NC	NC
Bismuth	<0.1	<0.2	NC	NC	NC
Boron	3.4	19.8	8.51	7.35	16.545
Cadmium	0.077	0.81	0.25	0.171	0.629
Calcium	12400	53400	21563	16900	41955
Chromium	<0.1	0.9	0.26	0.2	0.69
Cobalt	0.02	0.27	0.07	0.05	0.193
Copper	4.2	22.4	9.81	9.4	18.2
Iron	23	545	126.00	76.5	393.8
Lead	1.25	22.3	5.83	3.47	16.60
Lithium	0.09	0.78	0.26	0.22	0.591
Magnesium	804	3720	1541	1345	2936
Manganese	2	21.5	9.1	7.3	18.6
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.03	0.19	0.07	0.05	0.14
Nickel	<0.1	0.7	0.21	0.15	0.53
Potassium	591	5320	1910.8	1485	4319
Rubidium	0.22	2.49	0.77	0.55	1.91
Selenium	0.1	0.7	0.26	0.2	0.56
Silver	0.16	1.05	0.32	0.23	0.77
Sodium	4380	14300	7418	6875	12235
Strontium	162	722	289.4	242.5	563.1
Tellurium	<0.01	0.02	0.01	0.01	0.017
Thallium	0.03	0.35	0.10	0.07	0.27
Tin	0.03	0.84	0.21	0.145	0.616
Uranium	<0.01	0.04	0.02	0.01	0.033
Vanadium	<0.1	1.5	0.38	0.25	1.08
Zinc	7.5	45.5	16.18	12.4	35.4

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 9
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-20	Shoreline Invertebrate Concentrations near Belledune Point (Area 3)(mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	43.9	135	78.4	71.2	128
Antimony	0.01	0.03	0.02	0.015	0.03
Arsenic	0.6	3.3	1.1	0.7	2.6
Barium	2.5	6	3.09	2.65	4.95
Beryllium	<0.01	<0.02	NC	NC	NC
Bismuth	<0.1	<0.2	NC	NC	NC
Boron	6.8	12	9.7	9.7	11.7
Cadmium	0.066	0.245	0.12	0.10	0.21
Calcium	13100	44200	18813	15300	35345
Chromium	0.1	0.6	0.23	0.2	0.5
Cobalt	0.04	0.15	0.07	0.05	0.1255
Copper	3.2	25.6	7.8	5.2	20.2
Iron	57	184	98.5	82	168.3
Lead	2.85	7.19	4.11	3.44	6.43
Lithium	0.23	0.39	0.27	0.25	0.35
Magnesium	1290	2990	1633.75	1460	2503.5
Manganese	5.9	22	10.63	9.95	18.15
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.03	0.15	0.06	0.05	0.12
Nickel	0.1	0.4	0.19	0.15	0.365
Potassium	552	4900	1415.13	822	3846.5
Rubidium	0.22	1.84	0.56	0.34	1.455
Selenium	0.1	0.7	0.24	0.15	0.56
Silver	0.15	0.47	0.22	0.185	0.397
Sodium	5900	14900	8265.00	7570	12639
Strontium	179	644	261.13	207.5	509.6
Tellurium	<0.01	0.02	0.01	0.01	0.02
Thallium	0.02	0.09	0.04	0.035	0.0795
Tin	0.03	0.15	0.08	0.08	0.1395
Uranium	<0.01	0.01 [1]	NC	NC	NC
Vanadium	0.2	0.6	0.31	0.25	0.53
Zinc	6.8	33.3	12.20	8.65	26.27

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 8
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit
1. One sample had a non-detectable concentration, but the reportable detection limit (<0.02 mg/kg) was greater than the lowest detected concentration of 0.01 mg/kg. As such, the minimum value is reported as 0.01 mg/kg.

Table 3-21	Study Area Atlantic Herring Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	2.5	45.1	10.95	6.25	31.2
Antimony	<0.005	0.015	0.0083	0.0075	0.0137
Arsenic	0.49	0.61	0.533	0.53	0.588
Barium	0.08	0.21	0.127	0.12	0.183
Beryllium	<0.005	<0.005	NC	NC	NC
Bismuth	<0.05	0.06	0.051	0.05	0.0555
Boron	1.62	2.08	1.91	1.94	2.08
Cadmium	0.0624	0.109	0.0832	0.0831	0.101
Calcium	5480	6850	6012	5940	6688
Chromium	<0.05	0.11	0.056	0.05	0.083
Cobalt	0.008	0.03	0.0136	0.0115	0.0242
Copper	0.8	1.02	0.888	0.885	0.998
Iron	19	71	29.6	23.5	56.15
Lead	0.769	1.71	1.26	1.27	1.67
Lithium	0.052	0.102	0.0644	0.059	0.0903
Magnesium	655	750	701	695	741
Manganese	1.82	3.11	2.26	2.21	2.84
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.014	0.019	0.0167	0.017	0.0186
Nickel	0.07	0.11	0.081	0.08	0.101
Potassium	3020	3680	3407	3400	3676
Rubidium	0.675	0.828	0.755	0.762	0.815
Selenium	0.45	0.49	0.474	0.475	0.49
Silver	0.01	0.02	0.0145	0.0135	0.0191
Sodium	2940	3220	3073	3090	3198
Strontium	16.2	20.3	17.7	17.2	20.3
Tellurium	<0.005	<0.005	NC	NC	NC
Thallium	0.239	0.358	0.290	0.275	0.356
Tin	0.005	0.04	0.0149	0.013	0.0324
Uranium	<0.005	0.005	0.005	0.005	0.005
Vanadium	<0.05	0.21	0.071	0.05	0.152
Zinc	22.2	27.2	24.4	24.5	26.6

Notes:
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 10
Percent Moisture ranged from 78.9% to 81.2% with an average % moisture concentration of 80.0%
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-22	Study Area Sand Lance Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	2.8	265	88	41.1	239
Antimony	<0.005	0.375	0.18	0.157	0.37
Arsenic	0.62	3.3	1.66	1.31	3.16
Barium	0.16	2.24	0.955	0.755	2.09
Beryllium	<0.005	0.007	0.00533	0.005	0.0065
Bismuth	<0.05	<0.05	NC	NC	NC
Boron	1.05	1.21	1.11	1.08	1.21
Cadmium	0.0728	0.205	0.133	0.114	0.203
Calcium	5870	7160	6390	6395	7020
Chromium	<0.05	0.78	0.295	0.175	0.718
Cobalt	0.013	0.895	0.332	0.224	0.826
Copper	0.94	8.56	4.24	4.42	7.96
Iron	21	1030	412	333	952
Lead	0.936	59.7	26.0	20.3	58.2
Lithium	0.056	0.31	0.130	0.088	0.275
Magnesium	570	694	608	593	677
Manganese	2.7	8.32	4.57	3.71	7.75
Mercury	<0.01	<0.01	NC	NC	NC
Molybdenum	0.011	0.249	0.085	0.0765	0.208
Nickel	0.07	0.47	0.173	0.115	0.4
Potassium	3610	3720	3677	3695	3720
Rubidium	0.897	1.17	1.02	0.998	1.16
Selenium	0.55	0.62	0.58	0.575	0.615
Silver	0.006	0.05	0.0215	0.021	0.0433
Sodium	2200	2300	2252	2255	2298
Strontium	20.8	23.8	22.6	22.65	23.7
Tellurium	<0.005	<0.005	NC	NC	NC
Thallium	0.347	0.549	0.448	0.444	0.542
Tin	0.018	2.05	0.852	0.628	1.97
Uranium	<0.005	0.012	0.00683	0.005	0.0113
Vanadium	<0.05	0.92	0.347	0.12	0.893
Zinc	31.7	299	145	128	286

Notes:
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 6
Percent Moisture ranged from 75.7% to 78.2% with an average % moisture concentration of 77.4%
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-23	66-Day Post Deployment (Study Area) Marine Mussel Concentrations Site S1 (mg/kg dw)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	109	348	248	278	341
Antimony	<0.1	0.1	0.1	0.1	0.1
Arsenic	8	15	10.2	9	14
Barium	5	13	8.6	8	12.4
Beryllium	<0.1	<0.1	NC	NC	NC
Bismuth	<1	<1	NC	NC	NC
Boron	20	21	20.6	21	21
Cadmium	2.25	5.42	3.72	3.43	5.25
Calcium	2060	4220	2638	2170	3910
Chromium	<1	1	1	1	1
Cobalt	0.2	0.5	0.38	0.4	0.5
Copper	5	9	7	7	8.8
Iron	140	350	266	280	350
Lead	32.6	105	63.4	62	96.6
Lithium	0.4	0.6	0.54	0.6	0.6
Magnesium	2830	3160	2978	2940	3152
Manganese	7	13	10.4	10	12.8
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.4	0.6	0.48	0.5	0.58
Nickel	<1	2	1.4	1	2
Potassium	10800	12600	11540	11400	12380
Rubidium	4.5	5.4	5.06	5.2	5.38
Selenium	3	5	4	4	5
Silver	0.1	0.2	0.12	0.1	0.18
Sodium	17200	21600	18700	18500	21000
Strontium	19	27	23	23	26.6
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	<0.1	<0.1	NC	NC	NC
Tin	<0.1	<0.1	NC	NC	NC
Uranium	<0.1	0.1	0.1	0.1	0.1
Vanadium	<1	2	1.4	1	2
Zinc	99	210	149	152	205

Notes:
Data obtained by Minnow in October, 2014 (66 days post-deployment); Raw data are provided in Appendix B
N = 5
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; dw = dry weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-24	66-Day Post Deployment (Study Area) Marine Mussel Concentrations Site S2 (mg/kg dw)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	133	730	336.4	287	655.2
Antimony	0.1	0.3	0.18	0.2	0.28
Arsenic	8	12	10	10	11.8
Barium	2	9	6.8	8	9
Beryllium	<0.1	<0.1	NC	NC	NC
Bismuth	<1	<1	NC	NC	NC
Boron	19	25	22.8	23	25
Cadmium	1.97	5.59	3.834	3.65	5.338
Calcium	2010	12300	5038	2630	11000
Chromium	<1	2	1.2	1	1.8
Cobalt	0.3	0.7	0.5	0.5	0.68
Copper	7	10	8.4	8	9.8
Iron	200	760	416	380	704
Lead	32.3	98.9	60.04	54.4	92.92
Lithium	0.4	1.1	0.72	0.7	1.04
Magnesium	2600	3980	3378	3410	3912
Manganese	12	27	16.6	14	25
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.3	0.6	0.48	0.5	0.58
Nickel	<1	3	2	2	2.8
Potassium	10500	12800	11780	11800	12760
Rubidium	4.2	5.8	5.22	5.4	5.78
Selenium	4	5	4.4	4	5
Silver	0.1	0.2	0.14	0.1	0.2
Sodium	16700	26400	22100	22300	25880
Strontium	20	61	34	26	56.4
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	<0.1	<0.1	NC	NC	NC
Tin	<0.1	<0.1	NC	NC	NC
Uranium	<0.1	0.1	0.1	0.1	0.1
Vanadium	1	3	2.2	2	3
Zinc	62	202	122.4	115	186.4

Notes:
Data obtained by Minnow in October, 2014 (66 days post-deployment); Raw data are provided in Appendix B
N = 5
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; dw = dry weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-25	66-Day Post Deployment (Study Area) Marine Mussel Concentrations Site S3 (mg/kg dw)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	99	798	328.6	174	732.4
Antimony	<0.1	0.2	0.14	0.1	0.2
Arsenic	10	14	11.8	11	14
Barium	2	14	7	6	12.6
Beryllium	<0.1	<0.1	NC	NC	NC
Bismuth	<1	<1	NC	NC	NC
Boron	21	26	23	22	25.6
Cadmium	2.54	5.06	3.486	3.28	4.752
Calcium	2500	12000	4858	3020	10444
Chromium	<1	2	1.4	1	2
Cobalt	0.3	0.8	0.54	0.5	0.78
Copper	7	9	8.4	9	9
Iron	160	870	404	240	812
Lead	35.6	66.8	52.48	50.7	66.26
Lithium	0.5	1.1	0.74	0.6	1.06
Magnesium	2870	4360	3650	3730	4260
Manganese	12	29	17.4	13	27.4
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.4	0.7	0.52	0.5	0.68
Nickel	<1	4	2.4	2	3.8
Potassium	11900	13000	12520	12400	12980
Rubidium	5.1	6.4	5.6	5.4	6.28
Selenium	4	6	5	5	5.8
Silver	<0.1	0.3	0.14	0.1	0.26
Sodium	17200	30500	24640	25100	29720
Strontium	27	55	38.2	38	52.2
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	<0.1	<0.1	NC	NC	NC
Tin	<0.1	<0.1	NC	NC	NC
Uranium	<0.1	0.2	0.12	0.1	0.18
Vanadium	<1	4	2.4	2	3.8
Zinc	92	210	140.6	132	200.2

Notes:
Data obtained by Minnow in October, 2014 (66 days post-deployment); Raw data are provided in Appendix B
N = 5
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; dw = dry weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-26	**66-Day Post Deployment (Study Area) Marine Mussel Concentrations Site S4 (mg/kg dw)**				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	68	311	214	237	304
Antimony	0.1	0.2	0.14	0.1	0.2
Arsenic	9	11	10	10	10.8
Barium	4	9	7	8	8.8
Beryllium	<0.1	<0.1	NC	NC	NC
Bismuth	<1	<1	NC	NC	NC
Boron	21	28	24.2	25	27.4
Cadmium	3.38	5.81	4.58	4.29	5.69
Calcium	2410	4330	3118	2810	4146
Chromium	1	1	1	1	1
Cobalt	0.5	0.9	0.58	0.5	0.82
Copper	7	10	8.8	9	10
Iron	140	390	294	320	382
Lead	71.5	100	82.22	81.2	96.62
Lithium	0.5	0.7	0.64	0.7	0.7
Magnesium	3220	4200	3754	3750	4130
Manganese	8	17	12.8	12	16.6
Mercury	NA	NA	NA	NA	NA
Molybdenum	0.4	0.6	0.54	0.6	0.6
Nickel	2	4	2.4	2	3.6
Potassium	10800	15300	12660	12700	14880
Rubidium	4.9	5.9	5.48	5.8	5.88
Selenium	4	5	4.6	5	5
Silver	<0.1	0.3	0.18	0.2	0.28
Sodium	21000	29200	25540	25700	28600
Strontium	27	37	32.2	32	36.6
Tellurium	<0.1	<0.1	NC	NC	NC
Thallium	0.2	0.3	0.22	0.2	0.28
Tin	<0.1	<0.1	NC	NC	NC
Uranium	<0.1	0.1	0.1	0.1	0.1
Vanadium	2	2	2	2	2
Zinc	129	232	181	176	225

Notes:
Data obtained by Minnow in October, 2014 (66 days post-deployment); Raw data are provided in Appendix B
N = 5
NA = not analyzed; NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; dw = dry weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit

Table 3-27	Common Tern Chick Egg Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	<0.1	0.1	0.1	0.1	0.1
Antimony	<0.01	0.01	0.01	0.01	0.01
Arsenic	<0.1	0.2	0.1	0.1	0.2
Barium	<0.1	0.1	0.1	0.1	0.1
Beryllium	<0.01	<0.01	NC	NC	NC
Bismuth	<0.1	<0.1	NC	NC	NC
Boron	<0.1	<0.1	NC	NC	NC
Cadmium	0.001	0.015	0.005	0.003	0.010
Calcium	469	2530	955	626	2250
Chromium	<0.1	0.1	0.1	0.1	0.1
Cobalt	<0.01	0.02	0.01	0.01	0.01
Copper	0.7	1.0	0.9	0.8	1.0
Iron	24.0	37.0	30.2	30.0	36.2
Lead	0.14	0.75	0.35	0.32	0.64
Lithium	<0.01	<0.01	NC	NC	NC
Magnesium	84	182	115	108	174
Manganese	0.3	1.0	0.6	0.6	0.8
Mercury	0.05	0.11	0.09	0.09	0.11
Molybdenum	0.01	0.03	0.02	0.02	0.02
Nickel	<0.1	<0.1	NC	NC	NC
Potassium	1200	1960	1461	1385	1833
Rubidium	0.35	0.65	0.47	0.46	0.64
Selenium	0.5	0.9	0.7	0.7	0.8
Silver	<0.01	0.02	0.01	0.01	0.01
Sodium	1430	1930	1635	1610	1879
Strontium	0.6	4.6	2.0	1.7	3.8
Tellurium	0.01	0.06	0.03	0.03	0.05
Thallium	0.03	0.54	0.17	0.10	0.49
Tin	<0.01	0.09	0.02	0.01	0.07
Uranium	<0.01	<0.01	NC	NC	NC
Vanadium	<0.1	<0.1	NC	NC	NC
Zinc	9.4	20.6	14.3	14.0	18.4

Notes:
Data collected by Glencore in June and July, 2014; Raw data are provided in Appendix B
N = 18; egg contents represented various stages of development ranging from orange liquid, ½ formed chick and formed chicks.
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit
Percent moisture ranged from 73.8 to 81.2 with a mean of 78.1

Table 3-28	Common Tern Chick Kidney Concentrations (mg/kg ww)				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	0.4	10.2	3.4	1.5	9.2
Antimony	0.12	2.26	0.74	0.22	2.05
Arsenic	0.5	4.8	1.6	0.8	4.2
Barium	<0.1	0.2	0.1	0.1	0.2
Beryllium	<0.01	<0.01	NC	NC	NC
Bismuth	<0.1	0.2	0.1	0.1	0.2
Boron	<0.1	0.1	0.1	0.1	0.1
Cadmium	0.213	2.040	0.866	0.410	2.008
Calcium	222	755	454	375	751
Chromium	<0.1	0.1	0.1	0.1	0.1
Cobalt	0.02	0.07	0.04	0.04	0.07
Copper	3.1	16.1	6.3	3.9	13.9
Iron	58	151	82	64.5	137.25
Lead	3.63	28.30	13.25	7.70	27.95
Lithium	<0.01	<0.01	NC	NC	NC
Magnesium	129	348	232.5	193	346.5
Manganese	0.8	3.8	1.7	1.3	3.3
Mercury	0.02	0.09	0.05	0.05	0.09
Molybdenum	0.07	1.03	0.28	0.15	0.81
Nickel	<0.1	<0.1	NC	NC	NC
Potassium	2510	5480	3388	2930	5065
Rubidium	0.85	3.15	1.51	1.13	2.82
Selenium	0.8	3.2	1.7	1.6	2.9
Silver	0.02	0.07	0.04	0.04	0.07
Sodium	1460	2220	1645	1540	2075
Strontium	0.3	1.1	0.6	0.5	1.0
Tellurium	0.05	0.46	0.30	0.33	0.46
Thallium	0.18	3.01	0.78	0.30	2.44
Tin	0.01	0.11	0.04	0.04	0.09
Uranium	<0.01	<0.01	NC	NC	NC
Vanadium	<0.1	<0.1	NC	NC	NC
Zinc	17.0	38.7	25.2	22.8	36.1

Notes:
Data collected by Glencore in July, 2014; Raw data are provided in Appendix B
N = 6
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit
Percent moisture not provided.

Table 3-29	**Common Tern Chick Liver Concentrations (mg/kg ww)**				
Analyte	**Min**	**Max**	**Mean**	**Median**	**95th Percentile**
Aluminum	<0.1	20.6	3.4	0.5	12.7
Antimony	0.04	1.69	0.41	0.19	1.20
Arsenic	0.40	3.50	1.08	0.60	2.72
Barium	<0.10	0.20	0.12	0.10	0.20
Beryllium	<0.01	<0.01	NC	NC	NC
Bismuth	<0.10	0.10	0.10	0.10	0.10
Boron	<0.10	<0.10	NC	NC	NC
Cadmium	0.08	1.81	0.51	0.26	1.71
Calcium	127.00	702.00	386.92	380.00	615.60
Chromium	<0.10	<0.10	NC	NC	NC
Cobalt	0.01	0.06	0.04	0.03	0.06
Copper	4.50	22.70	9.22	5.80	22.46
Iron	50.00	647.00	143.00	105.00	371.60
Lead	0.77	22.50	7.48	2.78	21.24
Lithium	<0.01	0.02	0.01	0.01	0.01
Magnesium	105.00	264.00	179.08	167.00	253.20
Manganese	0.90	7.00	2.28	1.80	5.62
Mercury	0.02	0.13	0.07	0.08	0.12
Molybdenum	0.06	0.61	0.23	0.18	0.58
Nickel	<0.10	0.10	0.10	0.10	0.10
Potassium	1770.00	3280.00	2672.31	2660.00	3262.00
Rubidium	0.60	1.88	1.11	1.01	1.84
Selenium	0.60	1.80	1.04	1.00	1.50
Silver	0.02	0.80	0.24	0.14	0.79
Sodium	1120.00	1910.00	1434.62	1400.00	1730.00
Strontium	0.10	1.70	0.46	0.40	1.16
Tellurium	0.06	0.46	0.20	0.16	0.42
Thallium	0.04	2.00	0.42	0.25	1.57
Tin	<0.01	0.15	0.04	0.01	0.12
Uranium	<0.01	<0.01	NC	NC	NC
Vanadium	<0.10	<0.10	NC	NC	NC
Zinc	18.40	98.40	32.61	24.50	69.36

Notes:
Data collected by Glencore, June-July 2014; Raw data are provided in Appendix B
N = 13
NC = not calculated as all data were not detected; < = concentration was less than the reportable detection limit; number presented is the detection limit; ww = wet weight
Mean and 95th percentile calculated assuming non-detectable samples were equal to the reportable detection limit
Percent moisture ranged from 69.8 to 74.2 with an average of 72.5

3.3 Quality Assurance/Quality Control (QA/QC) of Data

Laboratory analyses of samples were conducted by Maxxam (marine waters) and RPC (beach sand, sediments, shoreline invertebrates, mussel tissues, chicks and eggs, Atlantic herring and sand lance).

The percent (%) recoveries provided by Maxxam for laboratory-spiked samples were reviewed to ensure that the MDL achieved by the laboratory was appropriate. For samples where the percent recovery fell outside an acceptable range (*i.e.*, 75 – 125% for metals or 80-120% for phosphorus and nitrogen), lab comments and data were reviewed prior to deciding whether to reject the affected sample(s). A review of laboratory and field duplicate data was also undertaken to ensure that analyses were within acceptable ranges (*i.e.*, cases where duplicates yield relative percent differences (RPD) of more than 25% are discussed). All data supplied by Maxxam were determined to be within acceptable ranges.

The percent (%) recoveries were not calculated by RPC for laboratory-spiked samples. However, a review of field duplicate data was undertaken to ensure that analyses were within acceptable ranges (*i.e.*, RPDs were calculated for field duplicates, and assessed against an acceptability value of 30%). The acceptable RPD range of was adopted from the Ontario Ministry of the Environment, as an acceptable range was not specified by RPC. While some duplicate data were found to be outside the 30% range, in some cases this was due to sample heterogeneity (presence of slag in beach sand samples, for example). Based on the QA/QC review, data were considered acceptable for use in the assessment. Details of the QA/QC outcomes are provided in Appendix D.

4.0 MARINE LIFE ASSESSMENT

Potential risks to marine life including primary producers, pelagic invertebrates and benthic invertebrates, fish and shellfish were assessed. Methods used to assess potential risks are provided in Section 4.1, while results for receptors are provided in Sections 4.2 to 4.5 and overall conclusions for aquatic life in Section 4.6.

4.1 Methods

To evaluate potential risks to marine species many different lines of evidence were used which were presented in Section 2.0 and summarized in Table 4-1.

Table 4-1 Lines of Evidence for Marine Life

Receptor Group	Lines of Evidence
Marine Primary Producers and Pelagic Invertebrates	- Outcomes of the comparison of marine surface water COC concentrations to marine water SWQGs and to reference area concentrations - Consider toxicological / biological information from other (literature) studies and extrapolate where applicable to this study.
Marine Benthic Community	- Outcomes of the comparison of site sediment COC concentrations to marine SedQGs and to reference area concentrations. - Statistical analysis of benthic community abundance and diversity endpoints, relative to reference.
Marine Shellfish (e.g., mussel)	- Outcomes of the comparison of marine surface water COC concentrations to marine water SWQGs and to reference area concentrations - Assessment of caged mussel data relative to control/reference area, with respect to growth, condition and survival endpoints and relative to tissue metals residue data
Marine Fish (pelagic and bottom dwelling)	- Outcomes of the comparison of marine surface water COC concentrations to marine water SWQGs and to reference area concentrations. - Outcomes of fish health survey study - Consider toxicological / biological information from other (literature) studies and extrapolate where applicable to this study.

Note: SWQGs = Surface water quality guidelines; SedQGs = Sediment quality guidelines

4.1.1 Primary Producers and Pelagic Invertebrates

For evaluation of potential risks to primary producers and pelagic invertebrates, marine water data collected off the coast / to the east of Belledune Point were compared to marine surface water quality guidelines and to the 95th percentile of reference marine water concentrations collected off the coast of Little Belledune Point (See Figure 2-3). Where available, U.S. EPA, rather than CCME marine water quality guidelines (for the protection of aquatic life) were used for comparison purposes, since the water quality data are based on dissolved metals, as opposed to total metals, and U.S. EPA has guidelines based on dissolved concentrations.

In addition to the comparison to guidelines and reference area concentrations, toxicological / biological information from other (literature) studies were reviewed and extrapolated where applicable to this study.

In summary:

- Measures of Exposure: Dissolved water concentrations collected in reference and in the Study Area, over 2 sampling intervals (Summer and Fall);
- Measures of Effect: Dissolved water quality guidelines established to protect marine pelagic species and/or toxicological literature related to effects on marine species, where available.
- Characterization of Risk: Based on comparison of measured water quality concentrations to water quality guidelines or available toxicology literature.

The outcomes of the primary producer and pelagic invertebrate assessment are in Section 4.2.

4.1.2 Marine Benthic Community

For evaluation of potential risks to the benthic community, sediment data were collected in the marine areas off Belledune Point (i.e., by the Fertilizer Plant Outfall; the Final Effluent location and a transect located close to the final effluent location). Sediment data collected from these three areas were compared to marine sediment quality guidelines and to reference area sediment concentrations (95th percentile reference area concentrations). Where available, CCME marine sediment quality guidelines were used for comparison purposes. Both the ISQG (Interim Sediment Quality Guideline) and the PEL (Probable Effect Level) are provided, although assessment of potential for risk focused more on the PEL than the ISQG. If no CCME marine sediment quality guideline was available, guidelines from other sources were used where available.

Results of the Minnow (2015a) benthic community diversity and abundance report (See Appendix E) was an additional line of evidence with respect to determining potential risks to the benthic community.

In summary:

- Measures of Exposure: Sediment chemistry data collected in reference and in the Study Area, in 3 separate areas (FPO: Fertilizer Plant outfall; FE: Final Effluent; SST2: Surface Sediment Transect 2; as well as reference areas (deep and shallow; near Little Belledune Point; See Figure 2-3);
- Measures of Effect: Effect level sediment quality guidelines, such as ISQGs and PEL guidelines, as well as benthic community abundance and diversity data collected in October of 2014.
- Characterization of Risk: Based on comparison of measured sediment metal concentrations to sediment quality guidelines, and statistical analyses relative to reference, and consideration of benthic community outcomes.

Outcomes of the comparison of site sediment COC concentrations to marine SedQGs and to reference area concentrations in addition to consideration of the Minnow (2015a) benthic community assessment outcomes are provided in Section 4.3.

4.1.3 Marine Shellfish

To evaluate potential risks to marine shellfish, the outcomes of the comparison of marine water concentrations in the vicinity of Belledune Point to marine water quality guidelines and reference area concentrations (described in Section 4.1.1) were examined. In addition, bivalve tissue concentrations, relative to reference tissue concentrations, and survival and growth endpoints from the caged mussels in the vicinity of the site and reference areas were evaluated (See Appendix F; Minnow, 2015b). In summary:

- Measures of Exposure: Bivalve tissue concentration data, relative to reference, as well as marine water quality data;
- Measures of Effect: caged bivalve survival (mortalities; age); growth (change in length between deployment/collection), and condition assessment;
- Characterization of Risk: Based on analysis of growth, condition and survival endpoints, as well as outcomes of water quality data compared to marine water quality guidelines.

The outcomes of these evaluations are presented in Section 4.4.

4.1.4 Marine Fish

Potential risks to marine fish were evaluated using the outcomes of the comparison of marine water concentrations in the vicinity of Belledune Point to marine water quality guidelines and reference area concentrations (described in Section 4.1.1), as well as the fish health survey (Minnow, 2015b; See Appendix F), and whole fish tissue analysis. In addition, toxicological and biological information from other studies and literature were extrapolated where applicable to this study. In summary:

- Measures of Exposure: Marine water quality data for metals, as well as fish tissue data;
- Measures of Effect: Fish health assessment, including survival, growth/condition, reproduction and energy stores for males and females. In addition, tissue residue guidelines established to protect health (where available).
- Characterization of Risk: Based on analysis of survival, growth, reproduction and energy stores data, as well as comparisons of water quality and fish tissue data to guidelines set to protect fish health, or aquatic life.

Results of this assessment are presented in Section 4.5.

4.1.5 Assessment of Risk

Assessment of potential risk for each of the aquatic receptor groups was based on the approach outlined in Table 4-2.

Table 4-2 Risk Characterization Approach

Receptor Group	Risk Potential			
	Negligible	**Low**	**Moderate**	**High**
Marine Primary Producers and Pelagic Invertebrates	No effect on individuals expected	Possible effect on some individuals expected, but effects on communities unlikely	Potential adverse effect on individuals; effect on some populations within the local community possible, but a self-sustaining, persistent, local community is expected to remain	Potential adverse effect on, and possible loss of, the local community
Marine Benthic Community	No effect on individuals expected	Possible effect on some individuals expected, but effects on communities unlikely	Potential adverse effect on individuals; effect on some populations within the local community possible (altered density; richness; evenness or diversity, relative to +/-2 standard deviations (SD) Critical Effect Size of reference community), but a self-sustaining, persistent, local community is expected to remain	Potential adverse effect on, and possible loss of, the local community
Marine Shellfish (e.g., mussel)	No effect on individuals expected	Possible effect on some individuals expected, but effects on population unlikely	Potential adverse effect on individuals; effect on the local population possible (altered survival, growth or condition, relative to reference), but a self-sustaining, persistent, local population is expected to remain	Potential adverse effect on, and possible loss of, the local population
Marine Fish (pelagic and bottom dwelling)	No effect on individuals expected	Possible effect on some individuals expected, but effects on population unlikely	Potential adverse effect on individuals; effect on the local community possible [altered survival, growth, condition, energy storage use fecundity, egg size, egg number, relative to Critical Effect Sizes of +/- 25% for growth, reproduction and relative liver size endpoints, and +/- 10% for condition, compared to reference, but a self-sustaining, persistent, local community is expected to remain	Potential adverse effect on, and possible loss of, the local population

4.2 Marine Primary Producers and Pelagic Invertebrates Outcomes

4.2.1 Assessment of Chemistry Data Relative to Surface Water Quality Guidelines and Statistical Differences from Reference

One of the lines of evidence for evaluating potential risks to marine primary producers and pelagic invertebrates was to compare the maximum concentration of each metal in the study area to marine surface water quality guidelines and to the 95th percentile of reference area concentrations. Results of this comparison are provided in Table 4-3 (summer dataset) and Table 4-4 (fall dataset). Within each table, there are a limited number of samples (reference data N = 4; Study area N = 8, which include 2 water samples at each of 4 sampling areas; See Figure 2-3). Further statistical summaries are presented in Table 3-2 and 3-3 (reference marine water quality; summer and fall, respectively), and Tables 3-13 and 3-14 (Study area marine water quality; summer and fall, respectively). The study area data were combined to enable a conservative assessment of the data, rather than a spatial assessment, relative to guidelines.

Table 4-3 Comparison of Study Area Marine Water Concentrations (Summer) to Marine Surface Water Quality Guidelines and 95th Percentile Reference Area Concentrations				
Analyte	**Units**	**Max Study Area**	**95%ILE Reference**	**Marine Water Quality Guideline [1]**
Dissolved Aluminum	µg/L	19	<10 [2]	NGA
Dissolved Antimony	µg/L	<0.5	<0.5 [2]	NGA
Dissolved Arsenic	µg/L	2.27	1.33	36
Dissolved Barium	µg/L	11.3	11.5	NGA
Dissolved Beryllium	µg/L	<1	<1 [2]	NGA
Dissolved Bismuth	µg/L	<1	<1 [2]	NGA
Dissolved Boron	µg/L	3470	3227	NGA
Dissolved Cadmium	µg/L	0.989	0.061	8.8
Dissolved Chromium	µg/L	<0.5	<0.5 [2]	50
Dissolved Cobalt	µg/L	<0.1	<0.1 [2]	NGA
Dissolved Copper	µg/L	2.49	0.96	3.1
Dissolved Iron	µg/L	4	<2 [2]	NGA
Dissolved Lead	µg/L	1.60	0.14	8.1
Dissolved Lithium	µg/L	138	123.7	NGA
Dissolved Manganese	µg/L	6.00	0.75	NGA
Dissolved Mercury	µg/L	NA	NA	NA
Dissolved Molybdenum	µg/L	9.3	9.7	NGA
Dissolved Nickel	µg/L	0.63	0.96	8.2
Dissolved Phosphorus	µg/L	<50	<50 [2]	NGA
Dissolved Selenium	µg/L	<0.5	<0.5[2]	71
Dissolved Silicon	µg/L	201	199	NGA
Dissolved Silver	µg/L	<0.05	<0.05 [2]	NGA
Dissolved Strontium	µg/L	6250	6355	NGA
Dissolved Thallium	µg/L	3.30	<0.1 [2]	NGA
Dissolved Tin	µg/L	<1	<1[2]	NGA
Dissolved Titanium	µg/L	<10	<10 [2]	NGA
Dissolved Uranium	µg/L	2.54	2.32	NGA
Dissolved Vanadium	µg/L	<10	<10 [2]	NGA
Dissolved Zinc	µg/L	8.4	<1 [2]	81
Dissolved Calcium	mg/L	362	331	NGA
Dissolved Magnesium	mg/L	896	909	NGA
Dissolved Potassium	mg/L	296	302	NGA
Dissolved Sodium	mg/L	7340	7340	NGA
Dissolved Sulphur	mg/L	774	782	NGA
Total Phosphorus	mg/L	<0.020	<0.020	NGA
Total Nitrogen	mg/L	0.285	0.280	NGA

Notes:
Shading indicates maximum study area concentration is greater than 95th percentile of reference concentration or marine water quality guideline.
Data collected by Minnow in August, 2014; Raw data are provided in Appendix B
N = 8 study area; N = 4 for reference
NA = not analyzed; 95%ILE = 95th percentile; < = concentration was less than the reportable detection limit; number presented is the detection limit; NGA = no guideline available
1. USEPA dissolved marine water quality guideline unless otherwise indicated (URL: http://water.epa.gov/scitech/swguidance/standards/criteria/current/index.cfm#D)
2. 95th percentile not calculated because all samples were not detected; value presented is the detection limit

Table 4-4	Comparison of Study Area Marine Water Concentrations (Fall) to Marine Surface Water Quality Guidelines and 95th Percentile Reference Area Concentrations			
Analyte	**Units**	**Max Study Area**	**95%ILE Reference**	**Marine Water Quality Guideline [1]**
Dissolved Aluminum	µg/L	64	59	NGA
Dissolved Antimony	µg/L	<0.5	<0.5 [2]	NGA
Dissolved Arsenic	µg/L	1.78	1.60	36
Dissolved Barium	µg/L	8.0	7.1	NGA
Dissolved Beryllium	µg/L	<1	<1 [2]	NGA
Dissolved Bismuth	µg/L	<1	<1 [2]	NGA
Dissolved Boron	µg/L	3560	3527	NGA
Dissolved Cadmium	µg/L	0.256	0.075	8.8
Dissolved Chromium	µg/L	<0.5	<0.5 [2]	50
Dissolved Cobalt	µg/L	<0.1	<0.1 [2]	NGA
Dissolved Copper	µg/L	1.1	<0.5 [2]	3.1
Dissolved Iron	µg/L	15.4	3.7	NGA
Dissolved Lead	µg/L	1.10	0.16	8.1
Dissolved Lithium	µg/L	163	162	NGA
Dissolved Manganese	µg/L	4.33	3.07	NGA
Dissovled Mercury	µg/L	NA	NA	NA
Dissolved Molybdenum	µg/L	10.2	11.3	NGA
Dissolved Nickel	µg/L	1.95	1.11	8.2
Dissolved Phosphorus	µg/L	<50	50	NGA
Dissolved Selenium	µg/L	<0.5	<0.5 [2]	71
Dissolved Silicon	µg/L	124	<100 [2]	NGA
Dissolved Silver	µg/L	<0.05	0.05	NGA
Dissolved Strontium	µg/L	6930	6962	NGA
Dissolved Thallium	µg/L	3.44	0.15	NGA
Dissolved Tin	µg/L	<1	<1 [2]	NGA
Dissolved Titanium	µg/L	<10	10	NGA
Dissolved Uranium	µg/L	2.88	2.74	NGA
Dissolved Vanadium	µg/L	<10	<10 [2]	NGA
Dissolved Zinc	µg/L	5.6	1.9	81
Dissolved Calcium	mg/L	364	367	NGA
Dissolved Magnesium	mg/L	1040	1036	NGA
Dissolved Potassium	mg/L	340	341	NGA
Dissolved Sodium	mg/L	8690	8644	NGA
Dissolved Sulphur	mg/L	878	1022	NGA
Total Nitrogen	mg/L	0.223	0.182	NGA

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
Shading indicates maximum study area concentration is greater than 95th percentile of reference concentration or marine water quality guideline.
N = 8 study area; N = 4 for reference
NA = not analyzed; 95%ILE = 95th percentile; < = concentration was less than the reportable detection limit; number presented is the detection limit; NGA = no guideline available
1. USEPA dissolved marine water quality guideline unless otherwise indicated (URL: http://water.epa.gov/scitech/swguidance/standards/criteria/current/index.cfm#D)
2. 95th percentile not calculated because all samples were not detected; value presented is the detection limit

Marine water concentrations in the study area were all less than available marine dissolved water quality guidelines (Tables 4-3 and 4-4). Several metals (i.e., aluminum, barium, boron, iron, lithium, manganese, silicon, thallium, uranium, magnesium, sodium, calcium and nitrogen) had no guideline available and were greater than the 95th percentile reference concentration based on the summer and / or fall sampling. As such a statistical comparison of study area marine water concentrations during August (summer) and October (fall) for these metals to reference area water concentrations was conducted to see if there was a statistical difference between the study area and reference. Results of the statistical comparison are provided in Table 4-5, while details are provided in Appendix G.

Table 4-5 Summary of Statistical Comparison Reference Site Analyte Concentrations to Study Area Concentration (Summer and Fall, 2014)

Analyte	Significantly Different from Reference Site (Summer) ($p<0.05$)?	Notes (Summer)	Significantly Different from Reference Site (Fall) ($p<0.05$)?	Notes (Fall)
Dissolved Aluminum	NP	All samples were non-detect in reference area, therefore no statistical comparison carried out. Aluminum carried forward for further assessment since all study area samples (i.e., 8 out of 8 samples) had detectable concentrations	N	
Dissolved Barium	NA		Y	
Dissolved Boron	Y		N	
Dissolved Iron	NP	All samples were non-detect in reference area, therefore no statistical comparison carried out. Iron carried forward for further assessment since concentrations were greater than the detection limit (<2 µg/L) for 4 out of the 8 study area samples (detected at 2.4 µg/L in 3 samples and 4.4 µg/L in one sample)	N	
Dissolved Lithium	Y		N	
Dissolved Manganese	NP	The majority of samples (3/4) were non-detect in reference area, therefore no statistical comparison carried out. Manganese carried forward as a COPC since concentrations were greater than the detection limit for all samples from the study area (i.e., 8 out of 8 samples)	Y	
Dissolved Silicon	Y*	*concentration was less than reference, and therefore not evaluated further	NP	All samples were non-detect in reference area, therefore no statistical comparison carried out. Silicon carried forward for further assessment since concentrations were greater than the detection limit for the

Table 4-5 Summary of Statistical Comparison Reference Site Analyte Concentrations to Study Area Concentration (Summer and Fall, 2014)				
Analyte	**Significantly Different from Reference Site (Summer) ($p<0.05$)?**	**Notes (Summer)**	**Significantly Different from Reference Site (Fall) ($p<0.05$)?**	**Notes (Fall)**
				majority of samples from the study area (i.e., 5 out of 9 samples)
Dissolved Thallium	NP	All samples were non-detect in reference area, therefore no statistical comparison carried out. Thallium carried forward for further assessment since concentrations were greater than the detection limit in 7 if 8 study area samples	Y	
Dissolved Uranium	Y		N	
Dissolved Calcium	N		NA	
Dissolved Magnesium	NA		N	
Dissolved Sodium	NA		N	
Total Nitrogen	N		N	

Notes:
N = No statistical difference, analyte not carried forward; Y = Yes a statistical difference, analyte carried forward; NP = not performed due to non-detectable concentrations in reference; NA = not applicable – metal not carried forward for this sampling period since the maximum concentration was less than the 95th percentile of reference.
Analytes with study area concentrations significantly higher than those in the reference area and as such were carried forward for further evaluation in addition to analytes carried forward for additional evaluation based on qualitative considerations are shaded.

Based on the results of the statistical analysis, for the metals that did not have water quality guidelines and for which the maximum concentration was greater than the 95th percentile seasonal reference concentration, a number of metals were found to be different from reference (summer: boron, lithium, and uranium; fall: barium, manganese and thallium) in the study area marine waters, and were significant greater than concentrations in the reference area. A statistical comparison between the study area and reference could not be conducted with aluminum, iron, manganese, thallium (all in summer samples) and silicon (in fall samples), as all samples in the reference area were not detected. Given this, the following metals in marine waters were carried forward for further assessment:

- Aluminum
- Barium
- Boron
- Iron
- Lithium
- Manganese
- Silicon
- Thallium
- Uranium

While several of these metals may be associated with facility releases, many may not, and some metals may be within natural variability ranges, but due to the limited number of samples taken, this cannot be discerned. The data were assessed relative to available marine toxicity data, sourced from the U.S. EPA database (ECOTOX) (URL: http://cfpub.epa.gov/ecotox/), as well as other available literature.

Using this approach, the following information adds context with respect to the potential for toxicity of these substances:

- **Aluminum**: Aluminum was significantly different from reference in summer, where mean concentrations were 15 µg/L (Table 3-13), compared to non-detectable in reference in all samples (< 10 µg/L; Table 3-2). Aluminum toxicity data for marine species were evaluated, and found to be limited. Studies with rock oyster embryos yielded a 2-day NOEC (development) of 100 µg/L, and a 2-d LOEC of 150 µg/L (Wilson and Hyne, 1997), whereas a 3-day LC50 values for brine shrimp larvae was reported as 3,100 µg/L (Taneeva, 1973). All concentrations in the study area and reference sites, from both time periods, are below these toxicity values. In addition, although summer study area concentrations exceed reference concentrations, all concentrations are below the reference concentrations from the fall sampling. This information suggests limited concern with respect to aluminum toxicity in the study area.
- **Barium:** Barium concentrations in the study area were a maximum of 11.3 µg/L in summer (summer data are not significantly different from reference) and 8.0 µg/L in fall (fall data are significantly different from reference; 95th percentile = 7.1 µg/L). These values are likely a function of natural variability, as the study area data in the fall were within the range of reference values reported during the summer sampling event. A No-Observed-Effect Concentration (NOEC) of 100 µg/L and an Effect Concentration for 50% of the test population (EC50) of 189 µg/L were cited by Spangenberg and Cherr (1996) for mussels. In addition, Neff et al. (1995) suggest natural ocean concentrations of barium range from 4 – 20 µg/L, and the study area concentrations are well within that range. In addition, Neff et al. (1995) suggest that toxicity related to barium in seawater only occurs above the solubility of barium ions, and hence is not expected in the natural environment. Therefore, study area barium concentrations are not considered to represent a risk to pelagic aquatic life.
- **Boron:** Boron toxicity data for marine species were limited. Although no dissolved water quality guideline for boron in the marine environment could be located, BC MOE (2003) has a marine water quality guideline for boron (total metals) of 1.2 mg/L, which is based on a study by Thompson et al. (1976) on coho salmon (*Oncorhynchus kisutch*), that yielded a 283-hr LC50 of 12.2 mg/L. BC MOE (2003) added a safety factor of 0.1 to this value to generate the guideline. Study area concentrations of boron are above this value but below the LC50, even in reference (summer: 3.47 mg/L maximum value for study area; reference 95th percentile is 3.22 mg/L). Fall values were similarly elevated in both reference (95th percentile: 3.53 mg/L) and study area (maximum: 3.56 mg/L). However, summer study area concentrations fall within the range of fall

reference concentrations. In addition, boron concentrations in Canadian coastal waters have been reported to range between 3.7 and 4.3 mg/L (Health Canada, 1990), concentrations greater than those measured in the study area during both time periods. Boron also is not known to be associated with facility releases. Based on this information, boron concentrations are likely within the natural range of variability, and were not considered to be of concern.

- **Iron**: Iron is a required nutrient for phytoplankton growth and is generally considered to be limiting in marine waters (NOAA, 1999). Iron concentrations were not significantly different from reference in the fall, but could not be statistically evaluated in the summer dataset, due to reference concentrations being non-detect. Study area concentrations in the summer ranged from non-detect (< 2 µg/L) to 4.4 µg/L, with a mean and median value of 2 µg/L (equal to the detection limit) (See Table 3-13). Reference was non-detectable at < 2 µg/L. While the study area may contribute iron to the environment, these data do not suggest biologically significant differences between study area and reference. While no marine iron water quality guideline could be identified in the literature, BC MOE (2008) does have a freshwater guideline (dissolved; acute guideline of 350 µg/L). In addition, BC MOE (2008) states that while no marine guideline was developed, iron would tend to precipitate in the marine environment, due to elevated pH (approximately 8.2), and hence, it was not anticipated that iron toxicity would be a concern in marine environments. Based on this information, iron at the measured concentrations was not considered to be of concern to marine organisms.

- **Manganese**: As per iron, manganese is considered to be limiting in the marine environment (NOAA, 1999). Maximum study area concentrations of manganese are 6 µg/L (summer) and 4.3 µg/L (fall), relative to reference concentrations (0.75 and 3.07 µg/L). No marine toxicity data could be located in the literature reviewed. Manganese toxicity in freshwater systems is modified by hardness, and marine waters have high concentrations of calcium and magnesium, which would be expected to mitigate toxicity. Freshwater guidelines established in BC MOE (2001) for chronic exposures are 1,900 µg/L, at a hardness of 300 mg/L. Based on the fact manganese in marine environments tends to be limiting, and considering the chronic freshwater guideline is 300 to 400 times higher than measured concentrations in the study area, manganese concentrations are unlikely to be associated with any significant toxicity.

- **Lithium**: Lithium concentrations in summer were found to be significantly different from reference (maximum value of 138 µg/L, versus reference 95th percentile concentration of 123.7 µg/L), but were not significantly different in fall (maximum value of 163 µg/L versus reference 95th percentile of 162 µg/L). No marine or freshwater toxicity was identified in the literature reviewed. The summer concentrations within the study area are within the range of concentrations detected in reference throughout the year, and hence are unlikely to represent a toxicity concern to phytoplankton or pelagic invertebrates.

- **Silicon**: As per iron and manganese, silicon is also essential for phytoplankton growth, and is typically limiting in the marine environment (NOAA, 1999). Silicon concentrations in the study area were lower than reference areas in the summer sampling interval. In fall, concentrations were non-detect in reference samples (< 100 µg/L) and ranged from <100 µg/L to 124 µg/L (median = 104 µg/L) in study area samples. Marine or freshwater toxicity data could not be located in the literature reviewed. Considering the measured concentrations in summer (study area: 138- 201 µg/L; reference: 188 – 199 µg/L; See Section 3; Table 3-13 and 3-2) are higher than those reported in fall, these concentrations are unlikely to represent a toxicity concern.
- **Thallium**: Study area concentrations of thallium were a maximum of 3.3 µg/L in summer, and 3.4 µg/L in fall, compared to reference levels of < 0.1 µg/L and 0.15 µg/L, respectively. Thallium is associated with effluent releases from the facility. While no marine water quality guidelines were identified, CCME (1999) has a freshwater guideline of 0.8 µg/L. This is based on the lowest chronic value of 8 µg/L divided by a safety factor of 10. Chronic freshwater LOELs for plants, invertebrates and fish ranged from 8 to 181 µg/L. In the literature reviewed, some acute toxicity data for marine species were identified. Horne et al. (1983) cite 2-d LC50s for 3 marine species ranging from 2,500 µg/L to 5,600 µg/L (sand shrimp, scuds, and grass shrimp). McLeese (1976) cites a 4-d LC50 of 1,000 µg/L for lobster larvae. Because study area concentrations are lower than chronic freshwater toxicity test data and more than 100 times lower than acute marine toxicity data, study area concentrations are unlikely to represent a toxicity concern.
- **Uranium**: Uranium concentrations in summer were found to be significantly different from reference. However, concentrations in the study area in summer (2.38 to 2.54 µg/L; Table 3-13) were below the range of concentrations in reference samples from the fall (2.63 to 2.74 µg/L; Table 3-3). Uranium toxicity data for marine species was limited. There was a 41% decrease in respiration rate of a marine amphipod exposed to 100 µg/L (CCME, 2011). Uranium is not considered to be associated with releases from the facility, and total concentrations or uranium in seawater of the Atlantic and Pacific oceans has been reported to be 3.1 µg/L (CCME, 2011). This information suggests study area concentrations are unlikely to represent a toxicity concern to pelagic species.

4.2.2 Field Observations

During the marine bivalve study, cages were deployed into the marine environment in the study area, and in reference. As reported by Minnow (2015b), there was considerable bio-fouling on all cages. In the study area, juvenile blue mussels (*Mytilus edulis*) were very abundant on S1 – S3, as well as reference cages. Other invertebrate species were also noted, including amphipods, echinoderms, and polychaetes. Numerous algal species were also present on the cages, with red algae (Rhodophyta sp.) being most abundant, followed by rockweed (*Fucus sp*.). All cages had similar levels of bio-fouling. While these data are not quantitative, they do provide qualitative observations of algal and invertebrate species in the area of the facility.

4.2.3 Marine Primary Producers and Pelagic Invertebrate Weight of Evidence

Based on the information presented in Section 4.2.1, Table 4-6 outlines the Weight of Evidence evaluation for marine primary producers and pelagic invertebrates, with respect to potential risks. There are considerable uncertainties associated with the marine water quality data, since only a limited number of samples were taken in reference areas (N = 8), and study area locations (N = 16). These data only represent water quality characteristics on the days samples were taken, and metals concentrations could vary from these data. Nonetheless, they provide an indication of exposure potential in the areas sampled, for the time interval considered in this study.

Table 4-6 Weight of Evidence Evaluation for Marine Primary Producers and Pelagic Invertebrates

Area of Interest	Comparisons Water Quality Data to Marine Water Quality Guidelines	Comparisons to Marine Toxicity Studies	Potential Risks to Marine Primary Producers and Pelagic Invertebrates
S1 – S4	Maximum water quality concentrations less than marine water quality guidelines (As; Cd; Cu; Cr; Pb; Ni; Zn)	For metals lacking marine water quality guidelines, but found to be significantly different from reference, or where no statistical comparison could be completed, no potential risks related to toxicity were identified, based on reported concentrations being within natural and/or reference ranges, evaluation of limited toxicity data, or comparisons to freshwater quality guidelines.	Considered to be low, but uncertain, due to paucity of marine water quality data, marine toxicity data, and marine WQGs.

4.3 Benthic Community Outcomes

4.3.1 Assessment of Chemistry Data Relative to Sediment Quality Guidelines and Statistical Differences from Reference

One of the lines of evidence for evaluating potential risks to benthic species was to compare the sediment chemistry concentration of each metal in each area (i.e., FPO, FE and SST2) to marine sediment quality guidelines and to the 95th percentile of reference area concentrations. For these comparisons, the maximum and mean of the study area are included, but shading is indicated based on the maximum. Results of this comparison are provided in Tables 4-7 to 4-9.

Table 4-7	Comparison of Study Area Sediment Concentrations in the Vicinity of the Fertilizer Plant Outfall (FPO Area)(mg/kg) to Marine Sediment Quality Guidelines and 95th Percentile Reference Area Concentrations				
Analyte	**Mean**	**Max**	**95%ILE Reference**	**Marine Sediment Guideline** [1]	
				ISQG	**PEL**
Aluminum	6782	13900	11500	NGA	NGA
Antimony	0.5	1.1	0.1	NGA	NGA
Arsenic	4.6	8	6	7.24	41.6
Barium	45.6	69	123	NGA	NGA
Beryllium	0.38	0.7	0.7	NGA	NGA
Bismuth	<1	<1	<1 [3]	NGA	NGA
Boron	7.8	17	8	NGA	NGA
Cadmium	0.578	0.83	0.38	0.7	4.2
Calcium	54020	158000	30770	NGA	NGA
Chromium	16.8	31	26	52.3	160
Cobalt	4.4	10.4	9.7	NGA	NGA
Copper	12	21	9	18.7	108
Iron	8348	20400	17865	NGA	NGA
Lead	79.5	192	26.4	30.2	112
Lithium	8.8	22.7	20.2	NGA	NGA
Magnesium	3768	9310	8474	NGA	NGA
Manganese	123.4	294	359	NGA	NGA
Molybdenum	0.8	1.2	0.6	NGA	NGA
Nickel	13.2	33	29	30 [2]	50 [2]
Potassium	1022	1940	1402	NGA	NGA
Rubidium	4.84	9.9	8.0	NGA	NGA
Selenium	<1	<1	<1 [3]	NGA	NGA
Silver	0.12	0.2	0.1	1 [2]	2.2 [2]
Sodium	3534	5230	3000	NGA	NGA
Strontium	177.2	504	25	NGA	NGA
Tellurium	< 0.1	<0.1	<0.1 [3]	NGA	NGA
Thallium	0.28	0.4	0.4	NGA	NGA
Tin	1.4	3	<1 [3]	NGA	NGA
Uranium	25.06	70.8	0.8	NGA	NGA
Vanadium	17	37	34	NGA	NGA
Zinc	178.4	556	56	124	271

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B; N = 5
95%ILE = 95th percentile; < = concentration was less than the reportable detection limit; number presented is the detection limit; FPO = fertilizer plant outfall; NGA = no guideline available.
1. CCME Marine sediment quality guideline unless otherwise indicated (URL: http://st-ts.ccme.ca/en/index.html)
2. BC MOE (2014)
3. 95th percentile not calculated because all samples were not detected; value presented is the detection limit
Shading represents maximum value > 95%ile of reference or PEL; **bolding** represents maximum value > ISQG

Table 4-8	**Comparison of Study Area Sediment Concentrations in the Vicinity of the Final Effluent (FE Area) (mg/kg) to Marine Sediment Quality Guidelines and 95th Percentile Reference Area Concentrations**				
Analyte	**Mean**	**Max**	**95%ILE Reference**	**Marine Sediment Guideline [1]**	
				ISQG	**PEL**
Aluminum	10170	10500	11500	NGA	NGA
Antimony	0.52	1	0.1	NGA	NGA
Arsenic	20.8	34	6	7.24	41.6
Barium	130	183	123	NGA	NGA
Beryllium	0.4	0.4	0.7	NGA	NGA
Bismuth	<2	4	<1 [3]	NGA	NGA
Boron	7	8	8	NGA	NGA
Cadmium	2.14	2.64	0.38	0.7	4.2
Calcium	10284	12000	30770	NGA	NGA
Chromium	33.6	35	26	52.3	160
Cobalt	11.18	13.4	9.7	NGA	NGA
Copper	44	77	9	18.7	108
Iron	17820	20000	17865	NGA	NGA
Lead	474	860	26.4	30.2	112
Lithium	14.22	14.8	20.2	NGA	NGA
Magnesium	8554	8750	8474	NGA	NGA
Manganese	240	249	359	NGA	NGA
Molybdenum	0.3	0.4	0.6	NGA	NGA
Nickel	28.6	29	29	30 [2]	50 [2]
Potassium	1036	1180	1402	NGA	NGA
Rubidium	5.56	6.4	8.0	NGA	NGA
Selenium	<1	<1	<1 [3]	NGA	NGA
Silver	0.16	0.3	0.1	1 [2]	2.2 [2]
Sodium	2168	2430	3000	NGA	NGA
Strontium	20.8	24	25	NGA	NGA
Tellurium	<0.1	<0.1	<0.1 [3]	NGA	NGA
Thallium	2.08	3.1	0.4	NGA	NGA
Tin	3.2	7	<1 [3]	NGA	NGA
Uranium	0.48	0.6	0.8	NGA	NGA
Vanadium	36.6	39	34	NGA	NGA
Zinc	970.6	1840	56	124	271

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 5
95%ILE = 95th percentile; < = concentration was less than the reportable detection limit; number presented is the detection limit; FE = final effluent; NGA = no guideline available.
1. CCME Marine sediment quality guideline unless otherwise indicated (URL: http://st-ts.ccme.ca/en/index.html)
2. BC MOE (2014)
3. 95th percentile not calculated because all samples were not detected; value presented is the detection limit
Shading represents maximum value > 95%ile of reference or PEL; **bolding** represents maximum value > ISQG

Table 4-9	Comparison of Study Area Sediment Concentrations in the Vicinity of the Smelter Sediment Transect (SST2 Area) (mg/kg) to Marine Sediment Quality Guidelines and 95th Percentile Reference Area Concentrations				
Analyte	**Mean**	**Max**	**95%ILE Reference**	**Marine Sediment Guideline [1]**	
				ISQG	**PEL**
Aluminum	9624	9980	11500	NGA	NGA
Antimony	0.14	0.2	0.1	NGA	NGA
Arsenic	9.2	10	6	7.24	41.6
Barium	99.2	228	123	NGA	NGA
Beryllium	0.38	0.4	0.7	NGA	NGA
Bismuth	<1	<1	<1 [3]	NGA	NGA
Boron	5.8	6	8	NGA	NGA
Cadmium	0.904	1.57	0.38	0.7	4.2
Calcium	6092	7280	30770	NGA	NGA
Chromium	30	31	26	52.3	160
Cobalt	8.92	9.3	9.7	NGA	NGA
Copper	15.4	21	9	18.7	108
Iron	15000	15500	17865	NGA	NGA
Lead	128.02	162	26.4	30.2	112
Lithium	13.7	14.5	20.2	NGA	NGA
Magnesium	8082	8280	8474	NGA	NGA
Manganese	221.4	229	359	NGA	NGA
Molybdenum	0.24	0.4	0.6	NGA	NGA
Nickel	27.4	28	29	30 [2]	50 [2]
Potassium	1020	1070	1402	NGA	NGA
Rubidium	5.52	5.8	8.0	NGA	NGA
Selenium	<1	<1	<1 [3]	NGA	NGA
Silver	<0.1	0.1	0.1	1 [2]	2.2 [2]
Sodium	2388	2650	3000	NGA	NGA
Strontium	16.8	19	25	NGA	NGA
Tellurium	<0.1	<0.1	<0.1 [3]	NGA	NGA
Thallium	1.02	1.7	0.4	NGA	NGA
Tin	<1	<1	<1 [3]	NGA	NGA
Uranium	0.56	0.6	0.8	NGA	NGA
Vanadium	34.6	37	34	NGA	NGA
Zinc	177.4	238	56	124	271

Notes:
Data collected by Minnow in October, 2014; Raw data are provided in Appendix B
N = 5
95%ILE = 95th percentile; < = concentration was less than the reportable detection limit; number presented is the detection limit; SST = smelter sediment transect; NGA = no guideline available
1. CCME Marine sediment quality guideline unless otherwise indicated (URL: http://st-ts.ccme.ca/en/index.html
2. BC MOE (2014)
3. 95th percentile not calculated because all samples were not detected; value presented is the detection limit

Shading represents maximum value > 95%ile of reference or PEL; **bolding** represents maximum value > ISQG

None of the metals identified in Table 4-7 to 4-9, with the exception of lead and zinc in FPO and FE, and lead in SST2, exceeded PEL marine sediment quality guidelines. Maximum values exceeded ISQG for arsenic (FPO; FE; SST2), cadmium (FPO; FE; SST2), copper (FPO; FE; SST2) and nickel (FPO). Concentrations of several of the metals in FPO, FE and SST2, for which guidelines were not available, exceeded the 95th percentile reference area concentrations (See Table 4-10). As such, a statistical comparison of study area marine sediment concentrations (evaluated in each of the three areas separately) for these metals to reference area sediment concentrations was conducted. Results of the statistical comparison are provided in Table 4-11, while details are provided in Appendix G.

Table 4-10 Metals for Which No Guidelines were Available and Study Area Sediment Concentrations in Either FPO, FE or SST2 Exceeded the 95th Percentile Reference Area Concentrations			
Analyte	**FPO**	**FE**	**SST2**
Aluminum	√		
Antimony	√	√	√
Barium		√	√
Bismuth		√	
Boron	√		√
Calcium	√		
Cobalt	√	√	
Iron	√	√	
Lithium	√	√	
Magnesium	√	√	
Molybdenum	√		
Potassium	√		
Rubidium	√		
Sodium	√		
Strontium	√		
Thallium		√	√
Tin	√	√	
Uranium	√		
Vanadium	√	√	√

Notes:
FPO = fertilizer plant outfall; FE = final effluent; SST = smelter sediment transect
√ = metal had no marine sediment quality guideline available and the maximum concentrations was greater than the 95th percentile reference area concentration; these metals were carried forward for statistical analysis

Table 4-11 Summary of Statistical Comparison Results for Analytes for which no Suitable Guideline was Identified and that Exceeded the 95th Percentile Reference Concentration

Analyte	Significantly Different from Reference Site ($p<0.05$)?			Notes
	FPO	FE	SST2	
Aluminum	Y*	NA	NA	*Site FPO significantly lower than Reference
Antimony	NP	NP	NP	Antimony was not detected in 8 of 10 reference samples (detection limit of 0.1 mg/kg) and was present in the two detected samples at 0.1 mg/kg. As such, statistical analysis could not be performed. Antimony was detected in all samples at FPO (range = 0.1 mg/kg to 1.1 mg/kg) and FE (range = 0.2 mg/kg to 1 mg/kg). As such, antimony was carried forward for further assessment in FPO and FE sediments. At SST2, four of the five samples were detected at concentrations of 0.1, 0.1, 0.2 and 0.2 mg/kg. Given four of the five samples were detected at concentrations similar to the detection limit and similar to those in the reference area, antimony at SST2 was not carried forward for further assessment at this location.
Barium	NA	N	N	
Bismuth	NA	NP	NA	Bismuth was not detected in the reference area (detection limit of <1 mg/kg) and as such, statistical analysis could not be performed. Bismuth was detected in FE sediment at concentrations ranging from <1 to 4 mg/kg. Given four of the five FE samples were detected and at concentration greater than those in the reference area, bismuth was carried forward for assessment in FE sediments.
Boron	N	NA	N	
Calcium	Y	NA	N	
Cobalt	Y*	N	NA	*Site FPO significantly lower than Reference
Iron	Y*	N	NA	*Site FPO significantly lower than Reference
Lithium	Y*	N	NA	*Site FPO significantly lower than Reference
Magnesium	Y*	N	NA	*Site FPO significantly lower than Reference
Molybdenum	Y	NA	NA	
Potassium	N	NA	NA	
Rubidium	Y*	NA	NA	*Site FPO significantly lower than Reference
Sodium	Y	NA	NA	
Strontium	Y	NA	NA	
Thallium	NA	Y	Y	
Tin	NP	NP	NA	Tin was not detected in any reference area samples (detection limit of <1 mg/kg) and as such, statistical analysis could not be performed. Tin was detected in only 1 of 5 FPO samples at a concentration of 3 mg/kg. As such, tin was not carried forward for assessment in FPO. In FE tin was detected in 4 of the 5 samples with concentrations ranging from <1 to 7 mg/kg. Given four of the five FE samples were detected and at concentration greater than those in the reference area, tin was carried forward for assessment in FE sediments.
Uranium	Y	NA	NA	
Vanadium	Y*	N	N	*Site FPO significantly lower than Reference

N = No statistical difference, analyte not carried forward; Y = Yes a statistical difference, analyte carried forward; NP = not performed due to lack of detectable concentrations in reference; NA = not applicable as analyte not carried forward for statistical evaluation in this area; * = significantly lower than reference
Analytes with study area concentrations significantly higher than those in the reference area and as such were carried forward for further evaluation in addition to analytes carried forward for additional evaluation based on qualitative considerations are shaded.

Based on comparison to guidelines and reference area concentrations and statistical comparison between site and reference, the cells highlighted in Table 4-12 show which metals were considered to be either greater than PEL or ISQG sediment quality guidelines, or significantly different from reference, and therefore merited further consideration in the various marine sediment study areas (i.e., FPO, FE, and SST2).

Table 4-12 Metals Found to be Greater than PEL of ISQG Guidelines or Significantly Different from Reference in Marine Sediments

Analyte	Significantly Different from Reference Site (p<0.05)?		
	FPO	FE	SST2
Arsenic	Y**	Y**	Y**
Antimony	Y	Y	NA
Bismuth	NA	Y	NA
Cadmium	Y**	Y**	Y**
Calcium	Y	NA	NA
Copper	Y**	Y**	Y**
Lead	Y *	Y *	Y *
Molybdenum	Y	NA	NA
Nickel	Y**		
Sodium	Y	NA	NA
Strontium	Y	NA	NA
Thallium	NA	Y	Y
Tin	N	Y	NA
Uranium	Y	NA	NA
Zinc	Y *	Y *	Y**

Notes:
Sites for which analyte concentrations were significantly higher than those at the reference site are shaded
NA = not applicable, metal did not exceed 95th percentile reference area concentrations in the stated area
* indicates that maximum metal concentration exceeded Probably Effect Level sediment quality guideline
** indicates that maximum metal concentration exceeded Interim Sediment Quality guideline

These outcomes are discussed in conjunction with the sediment chemistry outcomes from the benthic community assessment completed by Minnow (2015a; Appendix E). The Minnow evaluation provides additional statistical assessment of the data at FPO and FE, comparisons to ISQG, as well as long-term temporal chemistry trends for FPO and FE (since these sites have been assessed for the C of A for many years). For FPO, Minnow assessed the site data relative to the deep reference area, whereas FE was assessed relative to the shallow reference area, due to potential differences in benthic communities at these sites, relative to reference, based on depth. Chemistry data between these reference areas are similar and hence were combined for assessment purposes in Tables 4-7 to 4-11. Assessment of chemistry data, based on Tables 4-7 – 4-11, and interpretation from the Minnow report (Appendix E) is summarized as follows:

Sediment Chemistry Conclusions: FPO

- Mean calcium, strontium, uranium were 10-fold higher than the mean of the deep reference site, whereas antimony, cadmium, lead, molybdenum and zinc were 2- to 5-fold higher than the mean deep reference;
- For metals with sediment quality guidelines, mean metal concentrations did not exceed PEL guidelines, and only mean lead and zinc were above ISQG. Maximum values of lead and zinc exceeded PEL guidelines, but no other metal maxima exceeded PEL guidelines;
- Mean sodium concentrations were slightly above the 95^{th} percentile of combined shallow and deep reference;
- This area exhibits greater variability in terms of chemistry than the FE area, which is likely due to the substrate (gypsum deposit) and varying degrees of recovery in the area;
- Chemistry data suggest limited potential for biological effects in the area;
- With respect to temporal changes from 2004 – 2014, mean sediment chemistry for arsenic, copper, lead, molybdenum and zinc in 2014 were slightly higher than mean concentrations from this area in 2004 and 2008;
- Principle Component Analysis of the temporal chemistry data did not indicate any definitive directional differences in sediment metal concentrations over time;
- Overall, no significant changes in sediment chemistry were noted at this location, relative to 2004 – 2008 surveys.

Sediment Chemistry Conclusions - FE

- Mean concentrations of antimony, arsenic, cadmium, copper, lead, thallium, and zinc in the FE area were approximately 5-fold higher than the shallow reference means;
- Where metals had sediment quality guidelines, only mean metals concentrations of lead and zinc exceeded PEL guidelines, whereas mean arsenic, cadmium and copper were above ISQG;
- Chemistry data suggest lead and zinc would be likely causative metals for any biological effects;
- With respect to temporal changes from 2004 – 2014, mean sediment chemistry for antimony, arsenic, cadmium, copper, lead, and zinc in 2014 were compared to earlier data (2004 - 2008), and concentrations of these elements have increased substantially in 2014, relative to earlier years. Statistical analysis of the data suggest that all of these metals, except for antimony were significantly higher in 2014, relative to 2008 survey results;
- Principle Component Analysis indicated greater divergence in sediment metals concentrations between this area and the shallow reference, relative to earlier years;

- These outcomes suggest a substantial increase in several metals, since previous surveys. Minnow examined the final effluent discharge concentrations and effluent volume from 2006 – 2014, and no significant increases in either volume or concentration were evident in the data (see Appendix E). As a result, the two possible factors associated with these increases were hypothesized to be the Port of Belledune harbor dredging project in 2010, and/or slag pile erosion due to a large storm even in December 2010.

Sediment Chemistry Conclusions – SST2

With respect to SST2, the sediment samples in this area were taken in a transect, heading southeast of the smelter along the eastern shore (See Figure 2-3 and Appendix C for additional figures). These sampling locations are distant to FPO, but would represent a possible gradient from FE, as they are down-gradient from the FE area. The COPCs identified in Table 4-12 for SST2 (arsenic, cadmium, copper, lead, thallium and zinc), are further assessed in Table 4-13 to identify whether any benthic community concerns, relative to chemistry, are apparent. All other metals are excluded from these comparisons, in that they were either not significantly different from reference, or maximum detected values were less than the 95th percentile of reference, or they were less than ISQG values.

Table 4-13 Comparisons of SST2 to Sediment Quality Guidelines for Main COPCs

Analyte	Sediment Quality Guidelines [1]		SST1	SST2	SST3	SST4	SST5
	ISQG	PEL					
Arsenic	7.24	41.6	10	9	10	9	8
Cadmium	0.7	4.2	1.57	0.7	0.79	0.76	0.7
Copper	18.7	108	16	15	21	13	12
Lead	30.2	112	162	116	147	117	98.1
Thallium	NGA	NGA	1.7	0.8	0.9	0.9	0.8
Zinc	124	271	238	182	227	139	101

Note: Shading means value exceeds Probably Effect Level Guidelines
NGA = No guideline available
1. CCME Marine sediment quality guideline (URL: http://st-ts.ccme.ca/en/index.html)

Based on Table 4-13, the following can be stated:

- Arsenic: Concentrations only marginal exceed the ISQG, but are substantially less than the PEL. In addition, Parsons and Cranston (2006) identified a background level of arsenic in sediments in the Baie de Chaleur of 19.0 mg/kg, based on deep sediment coring (see Appendix A; Section 3.2.1). Therefore the measured concentrations of arsenic are well within typical background for the basin, and would not be anticipated to result in measurable change in the benthic community.
- Cadmium: Cadmium concentrations decrease with increased distance along the transect, and are at ISQG levels in SST2 and SST5. These concentrations would not be anticipated to result in measurable change in the benthic community.

- Copper: Copper concentrations at all stations except SST3 are less than the ISQG, and hence, would not be expected to result in measurable change in the benthic community.
- Lead: Lead is above PEL levels at all stations with the exception of SST5. Lead concentrations therefore could be associated with some change in the benthic community. Concentrations appear to decrease with increased distance from the facility.
- Thallium: While no sediment quality guideline is available, there is a decrease in concentration with increased distance from the facility. The 95% percentile of combined reference is 0.4 mg/kg, and with that, concentrations from SST2 to SST5 are approximately 2-fold the upper range of background. Thallium is unlikely to contribute significantly to toxicity of sediments at these concentrations, although this conclusion is uncertain due to the lack of sediment guidelines.
- Zinc: No samples were above the PEL, and there is a reduction in concentrations with increased distance from the facility. Concentrations at SST4 only slightly exceed the ISQG, and concentrations at SST5 are less than the ISQG. While the data suggest some potential for toxicity in the SST1 and SST3 areas, the reduction in concentration with increased distance suggests limited potential for effects.

4.3.2 Benthic Community Assessment

A benthic community monitoring program was implemented by Minnow for the FPO and FE areas of the study area (and reference areas), as these areas are required to be assessed under the current C of A for the facility (Appendix E). The study examined sediment chemistry, particle size, water chemistry, as well as benthic community indices, to draw overall conclusions with respect to current status of benthic community abundance and diversity, as well as potential causative factors for effects, where possible. In addition, the 2014 dataset was compared to chemistry and benthic surveys completed in earlier years, to examine potential trends. The Minnow (2015a) report is presented in detail in Appendix E, and key aspects are summarized here for each area of interest.

FPO Area of Interest

This area has been impacted by former fertilizer plant discharges which have created a gypsum mat in the vicinity of the outfall. The sediment quality guideline comparisons, and statistical analysis outcomes presented in Table 4-12 suggest that there are several potential metals of interest which could be factors in benthic health.

Benthic Community Outcomes (See Table 4-14):

- Invertebrate density was significantly lower than the deep reference area, and approached +/- 2 Standard Deviations, which is the Critical Effect Size (CES) considered to represent an ecologically relevant magnitude of difference.

- Taxonomic richness and indices of evenness and diversity largely did not differ significantly from reference, and suggest a relative healthy, well balanced benthic invertebrate community structure. The Bray Curtis index did show significant differences from reference, but since there were no significant differences in major taxonomic groups, this was considered to represent variability in organism density rather than taxonomic composition.
- When examined on a temporal basis, only minor changes were indicated in indices, relative to earlier years. A two-way factorial analysis of benthic endpoints suggests that taxonomic richness differed significantly from 2008 to 2014, but this was concluded to be due to increased richness at the FPO relative to reference in 2014.
- The analysis suggests a slow, continuous improvement in habitat recovery with time (e.g., continued erosion, and/or burial of gypsum) that has resulted in a commensurate improvement in benthic invertebrate community conditions.

Table 4-14 Benthic Invertebrate Community Statistical Comparison Results between the Fertilizer Plant Outfall (FPO) and Deep Reference (RD) Study Areas (from Minnow Environmental, Table 5-3; Appendix E).

Metric	Statistical Test Results				Summary Statistics					
	Significant Difference Between Areas?	*p* -value	Statistical Analysis[a]	Magnitude of Difference [b] (No. of SD)	Area	Mean	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	YES	0.017	α	-1.9	Deep Reference	8,782	2,819	1,261	5,905	12,654
					Fertilizer Plant Outfall	3,548	2,664	1,191	489	6,003
Richness (Number of Taxa)	NO	0.303	α	-	Deep Reference	26.6	5.7	2.5	21.0	36.0
					Fertilizer Plant Outfall	22.6	5.8	2.6	14.0	28.0
Simpson's Diversity	NO	0.902	α	-	Deep Reference	0.859	0.040	0.018	0.796	0.899
					Fertilizer Plant Outfall	0.863	0.053	0.023	0.788	0.910
Simpson's Evenness (E)	NO	0.695	α	-	Deep Reference	0.894	0.040	0.018	0.827	0.925
					Fertilizer Plant Outfall	0.906	0.050	0.022	0.826	0.955
Shannon-Weiner Diversity (H')	NO	0.929	α	-	Deep Reference	3.428	0.342	0.153	2.967	3.864
					Fertilizer Plant Outfall	3.402	0.546	0.244	2.700	3.849
Shannon-Weiner Evenness (J')	NO	0.517	α	-	Deep Reference	0.728	0.061	0.027	0.624	0.787
					Fertilizer Plant Outfall	0.764	0.099	0.044	0.605	0.876
Bray-Curtis Index	YES	0.007	β	5.1	Deep Reference	0.245	0.081	0.036	0.129	0.325
					Fertilizer Plant Outfall	0.660	0.244	0.109	0.370	0.930
Errantia (%)	NO	0.333	δ	-	Deep Reference	26.1%	10.4%	4.7%	12.9%	41.2%
					Fertilizer Plant Outfall	19.9%	6.0%	2.7%	10.8%	27.4%
Sedentaria (%)	NO	0.437	δ	-	Deep Reference	36.0%	9.2%	4.1%	24.1%	48.7%
					Fertilizer Plant Outfall	20.7%	18.6%	8.3%	0.0%	41.4%
Metal-Sensitive Crustaceans (%)	NO	0.597	δ	-	Deep Reference	14.7%	6.2%	2.8%	4.6%	21.5%
					Fertilizer Plant Outfall	17.9%	10.5%	4.7%	8.4%	35.0%
Gastropoda (%)	NO	0.256	γ	-	Deep Reference	1.5%	0.9%	0.4%	0.0%	2.2%
					Fertilizer Plant Outfall	3.2%	3.6%	1.6%	0.0%	8.4%
Bivalvia (%)	NO	0.116	δ	-	Deep Reference	19.1%	5.1%	2.3%	11.2%	25.5%
					Fertilizer Plant Outfall	34.3%	19.4%	8.7%	16.2%	65.7%

[a] Data analysis included: α - data untransformed, single factor ANOVA test; β - data untransformed, single factor ANOVA test results confirmed using t-test assuming unequal variance; γ - data logit tran single factor ANOVA test results confirmed using Mann-Whitney U-test; and, δ - data logit transformed, single-factor ANOVA test conducted.[b] Magnitude calculated by comparing the difference between the reference area and FPO-exposed area means divided by the reference area standard deviation.
Shaded value indicates significant difference between study areas based on ANOVA p-value less than 0.10.

FE Area of Interest

The final effluent receiving environment is influenced by effluent discharges from the facility, as well as some supplementary sources which could have been major contributing factors to the outcome of the 2014 benthic community analysis. Two substantive events occurred since the previous (2008) survey conducted by Minnow (2009). Belledune harbor underwent a major dredging project (2009 to 2011) to remove sediments and deepen the Port of Belledune (See Appendix A, Section 3.2.3). While silt curtains were used during this sediment dredging program, some dispersion of contaminated sediments likely occurred. These sediments would travel around Belledune Point and potentially settle in the smelter-exposed area as a result of prevailing ocean currents (See Figure 2-1). In addition, a major storm event in 2010 resulted in significant erosion of the former slag disposal area on Belledune Point, which may have resulted in deposition of slag in the shoreline area off of Belledune Point. The findings of the Minnow survey conducted in 2014, relative to reference, and other time frame of monitoring are summarized here, and are presented in detail in Appendix E.

Benthic Community Outcomes (See Table 4-15):

- Invertebrate density at FE was significantly lower than the shallow reference area, and was less than +/- 2 Standard Deviations, which is the benchmark considered to represent an ecologically relevant magnitude of difference. Taxonomic richness was not significantly different, but indices of evenness and diversity were reduced at FE, suggesting that the benthos is dominated by few taxa. These differences from reference were considered ecologically relevant, in that they exceeded the Critical Effect Size of +/- 2 SD (see Table 4-15 below). The Bray Curtis index did show significant differences from reference, but not at an ecologically relevant level (less than the CES). These differences were concluded by Minnow to indicate that the benthic community at FE is different from the selected reference site.
- When examined on a temporal basis, there were a greater number of indices which differed significantly from reference than in earlier surveys. The FE site was consistently lower in a 2-way factorial analysis in 2014 for Shannon-Weiner Diversity and Evenness and gastropod abundance. Differences were noted between the 2008 and 2014 datasets, which suggests that increase sediment metals chemistry may be a causative factor. Despite this, most benthic endpoints at FE in 2014 are within historical ranges, suggesting that changes could also be related to natural temporal or seasonal variation between the reference and FE areas.
- The analysis suggests a change in benthic community structure, which may be linked to either of the two named events, but is unlikely to be associated with effluent quality or quantity, which has not changed to any substantive degree since 2004 (see Appendix A of Appendix E).
- If the slag erosion has contributed to the chemistry changes at FE, the leachability of metals within the slag is likely very low, as the pH of marine waters is 8 to 9 (with the possible exception of arsenic). As a result of this, while the chemistry may indicate elevated levels of metals, there may be limited exposure to metals as a result of low leachability.

Table 4-15 Benthic Invertebrate Community Statistical Comparison Results between the Final Effluent (FE) – Exposed area and Shallow Reference (RD) Study Areas (from Minnow Environmental, Table 5-1; Appendix E).

Metric	Statistical Test Results				Summary Statistics					
	Significant Difference Between Areas?	*p*-value	Statistical Analysis [a]	Magnitude of Difference [b] (No. of SD)	Area	Mean	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	YES	0.027	α	-1.3	Shallow Reference	7,692	3,080	1,377	2,671	10,851
					Final Effluent Area	3,750	1,082	484	2,070	4,996
Richness (Number of Taxa)	NO	0.158	α	-	Shallow Reference	20.0	2.5	1.1	18.0	24.0
					Final Effluent Area	17.2	3.1	1.4	12.0	20.0
Simpson's Diversity	YES	0.018	β	-6.2	Shallow Reference	0.806	0.022	0.010	0.778	0.833
					Final Effluent Area	0.673	0.099	0.044	0.545	0.772
Simpson's Evenness (E)	YES	0.020	β	-5.0	Shallow Reference	0.849	0.027	0.012	0.812	0.882
					Final Effluent Area	0.715	0.100	0.045	0.577	0.812
Shannon-Weiner Diversity (H')	YES	0.018	β	-4.6	Shallow Reference	2.940	0.139	0.062	2.724	3.051
					Final Effluent Area	2.304	0.461	0.206	1.738	2.731
Shannon-Weiner Evenness (J')	YES	0.028	α	-2.8	Shallow Reference	0.682	0.043	0.019	0.627	0.732
					Final Effluent Area	0.562	0.091	0.041	0.455	0.668
Bray-Curtis Index	YES	0.050	α	1.2	Shallow Reference	0.219	0.167	0.075	0.086	0.509
					Final Effluent Area	0.423	0.105	0.047	0.340	0.598
Errantia (%)	YES	0.036	γ	-1.0	Shallow Reference	12.0%	6.9%	3.1%	7.4%	24.2%
					Final Effluent Area	5.4%	2.6%	1.2%	2.3%	8.3%
Sedentaria (%)	NO	0.313	δ	-	Shallow Reference	32.1%	7.2%	3.2%	26.3%	42.8%
					Final Effluent Area	26.2%	12.2%	5.4%	11.6%	41.8%
Metal-Sensitive Crustaceans (%)	NO	0.852	δ	-	Shallow Reference	3.6%	1.2%	0.5%	2.1%	5.0%
					Final Effluent Area	4.0%	2.1%	0.9%	2.3%	7.2%
Gastropoda (%)	YES	0.087	δ	-0.7	Shallow Reference	4.6%	3.6%	1.6%	1.6%	10.7%
					Final Effluent Area	2.1%	2.9%	1.3%	0.3%	7.2%
Bivalvia (%)	YES	0.057	δ	2.5	Shallow Reference	45.8%	6.2%	2.8%	39.4%	52.5%
					Final Effluent Area	61.5%	13.8%	6.2%	45.1%	81.5%

[a] Data analysis included: α - data untransformed, single factor ANOVA test; β - data untransformed, single factor ANOVA test results confirmed using t-test assuming unequal variance; γ - data logit tran single factor ANOVA test results confirmed using Mann-Whitney U-test; and, δ - data logit transformed, single-factor ANOVA test conducted.

[b] Magnitude calculated by comparing the difference between the reference area and effluent-exposed area means divided by the reference area standard deviation.

Shaded value indicates significant difference between study areas based on ANOVA p-value less than 0.10.

SST2 Area of Interest

With respect to the SST2 dataset, there are no benthic community data. The trends discussed in Table 4-13 suggest low potential for toxicity for arsenic (within Baie de Chaleur background range; Parsons and Cranston, 2006), cadmium, copper (both are either slightly above, equal to or below the ISQG), and thallium (present at 2-fold the upper estimate of background), whereas zinc and lead could be associated with some toxicity in this area. Based on the outcomes of the benthic community analysis in FE, and elevated metal concentrations in FE (see Appendix E; Figure 4.2), box and whisker plots were created to compare data from FE for key metals/metalloids to concentrations in the SST2 area (Figure 4-1). FPO is included in these figures for completeness.

In Figure 4-1, the top and bottom of each box indicate the 75th and 25th percentiles of the data, respectively. The middle line in each box indicates the median (50th percentile). The whiskers indicate the lowest datum that is within 1.5 times the interquartile range (IQR, which equals the 75th percentile minus the 25th percentile) from the bottom of the box and the highest datum that is within 1.5 IQR from the top of the box. Values that are greater than 1.5 IQR but less than or equal to 3 IQR from the box are indicated with asterisks. Values that are more than 3 IQR from the box are indicated by empty circles.

Based on Figure 4-1, concentrations at SST2 are noticeably lower than those reported at FE, but are slightly higher or within the range of those at FPO (for these particular metals of interest). Measurable benthic effects were noted at FE when compared to reference, and the predominant metals of interest related to these outcomes are thought to be lead and zinc. Examining Figure 4-1, and considering the differences in zinc and lead concentrations at FE versus SST2, and the concentrations relative to the ISQG and PEL, the potential for adverse effects which would result in community structural alterations (such as density, diversity or richness) at an ecologically significant level to benthic communities at the SST2 stations is considered to be low.

4.3.3 Benthic Community Weight of Evidence

Based on the chemistry and benthic community outcomes, a Weight of Evidence evaluation is presented in Table 4-16.

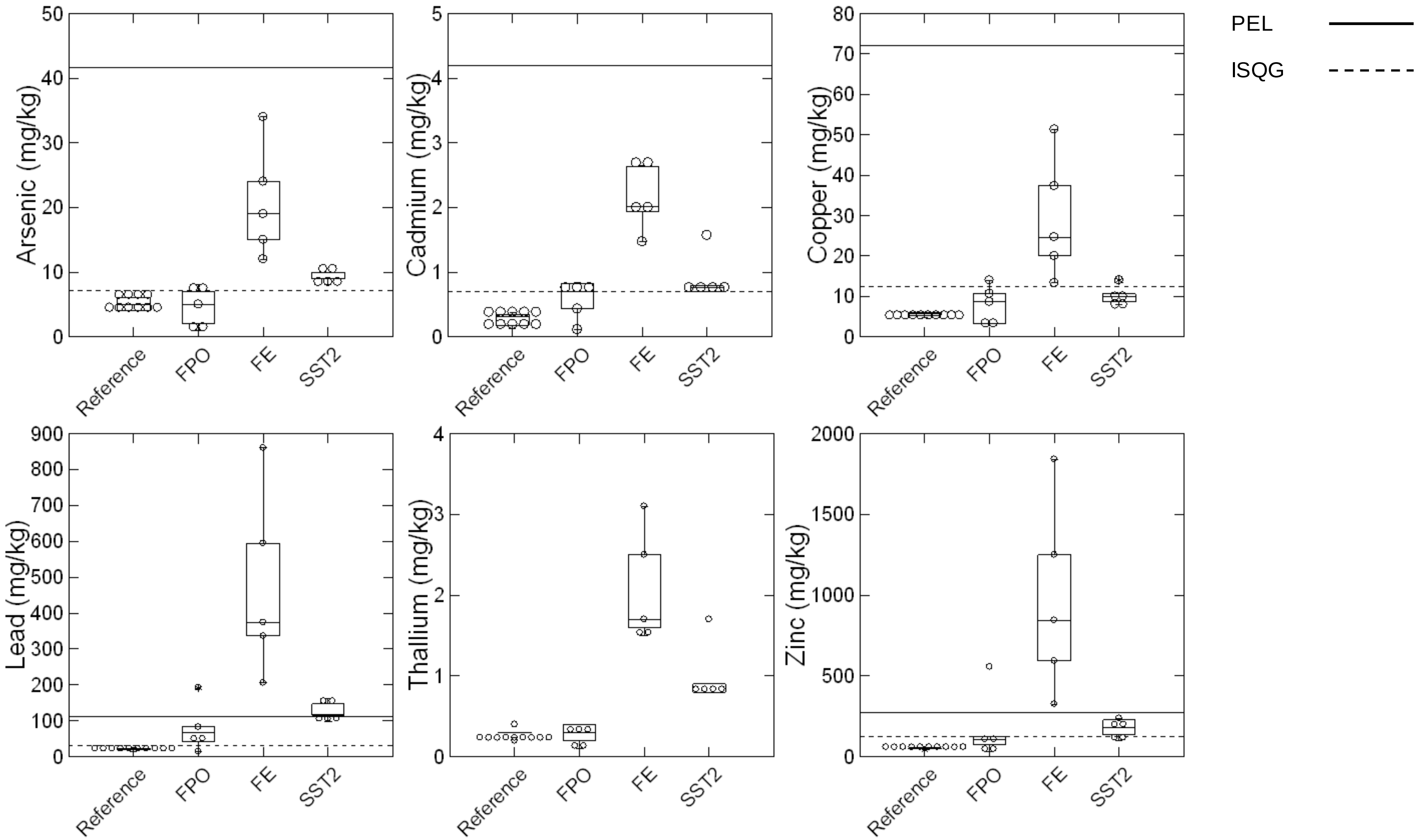

Figure 4-1 Box and Whisker Plots of SST2 relative to Reference, FPO and FE Areas of Interest for Selected Metals and Metalloids, Relative to Interim Sediment Quality Guidelines (ISQG) and Probable Effect Level (PEL) Sediment Quality Guidelines.

Table 4-16 Weight of Evidence Evaluation for Benthic Community

Area of Interest	Comparisons to Sediment Quality Guidelines and Statistical Outcomes	Benthic Community Diversity and Abundance Survey			Potential Risks to Benthic Invertebrate Communities
		Density	Richness	Evenness and Diversity	
FPO	Pb and Zn maximum concentrations exceed PEL guidelines, but mean concentrations do not; no other metals exceeded ISQG (based on mean concentrations); Ca, Sr and U were significantly different from reference; mean values were 10-fold reference levels; Other metals (Sb; Na; Mo) were significantly different from reference.	Significant reduction, relative to Reference (approaching but not > +/- 2 SD);	No significant difference, relative to reference	No significant difference, relative to reference, except Bray Curtis, which was considered to represent variability in organism density, rather than taxonomic differences	Historical benthic impacts related to physical habitat alteration; chemistry and benthic indices suggest slow but continuous recovery is in progress Risk Potential considered to be Low
FE	Pb and Zn maximum and mean concentrations exceed PEL guidelines; mean As, Cd, Cu were above ISQG; mean Sb, As, Cd, Cu, Pb, Tl, Zn were 5-fold reference means; Sb, Bi, Tl, Sn were significantly different from reference.	Significant reduction, relative to reference (not > +/- 2 SD);	No significant difference, relative to reference	Several indices significantly different from reference (> +/- 2 SD), suggesting ecologically meaningful effects	Minor effects indicated historically to benthos; 2014 suggests benthic impacts Risk Potential Considered to be Moderate
SST2	As marginally above ISQG, but within historical basin background levels; Cd and Cu are generally less than or equal to ISQG; Tl is approximately 2-fold 95th%ile of reference. Zn exceeds ISQG, but is less than PEL, while Pb exceeds PEL except for final station of transect. General reduction in concentration for all metals with increased distance from facility.	No Data	No Data	No Data	Benthic data unavailable. Comparisons of chemistry at SST2 to FE and FPO, statistical analysis relative to reference, and comparisons to ISQG and PEL suggest risk potential is low.

4.4 Marine Shellfish Outcomes

4.4.1 Assessment of Chemistry Data Relative to Surface Water Quality Guidelines and Statistical Differences from Reference

One of the lines of evidence to evaluate potential risks to marine shellfish was to compare marine water concentrations in the study area to marine surface water quality guidelines and to the 95th percentile reference area concentrations. The outcomes of this assessment were presented in Section 4.2.1 as follows:

- Toxicity-based marine water quality guidelines were available for arsenic, cadmium, chromium, copper, lead, nickel and zinc, and all water quality data collected during the marine bivalve study were less than these dissolved water quality guidelines, suggesting negligible potential for risk associated with these parameters (uncertainties due to limited data set are noted);
- For other metals/metalloids that were present in marine waters at concentrations exceeding the 95th percentile of reference, and found to be statistically different from reference (aluminum, barium, boron, iron, lithium, manganese, silicon, thallium, uranium), conclusions related to potential risk to aquatic life was based on toxicity literature (where available), comparisons to reference ranges of concentrations, or freshwater guidelines, where available. Conclusions from these comparisons indicated that toxicity potential is low related to these metals/metalloids.

4.4.2 Caged Bivalve Study Outcomes

Caged bivalves were deployed for 66 days at 4 stations adjacent to, and southeast of,Belledune Point(see Figure 2-3), for the purposes of examining the potential influence of smelter releases on bivalve health endpoints (Minnow , 2015b). Details of this study are presented in Appendix F, and this section presents a précis of the pertinent study outcomes.

4.4.2.1 Tissue Residues

The tissue residues of blue mussels were measured at deployment, and at 66 days post-deployment. The data are presented in Table 4-17. As discussed in Minnow (2015b), deployed mussels near the facility showed either similar or slightly higher tissue residues. Based on comparisons of two times the mean reference concentrations to study area, only cadmium, lead, (at all study area stations), thallium (at S4 only) as well as calcium (at S2) were considered to be elevated, relative to mean reference.

Table 4-17	Average Metal Concentration in Mussel Tissue 66 Days Post Deployment (mg/kg, dry weight)							
Analyte	**Units**	**Detection Limit**	**August Deployment[a]**	**Reference Mussel Samples**	**Site Mussel Samples**			
					S1	**S2**	**S3**	**S4**
Aluminum	mg/kg	<1	299	298	248.4	344.8	328.6	213.6
Antimony	mg/kg	<0.1	0.03	<0.1	<0.1	0.18	0.14	0.14
Arsenic	mg/kg	<1	8.1	8.4	10.2	10	11.8	10
Barium	mg/kg	<1	2.01	7.6	8.6	6.9	7	7
Beryllium	mg/kg	<0.1	0.028	<0.1	<0.1	<0.1	<0.1	<0.1
Bismuth	mg/kg	<1	0.28	<1	<1	<1	<1	<1
Boron	mg/kg	<1	26.9	20.6	20.6	22.9	23	24.2
Cadmium	mg/kg	<0.01	1.56	1.30	3.722	3.885	3.486	4.58
Calcium	mg/kg	<50	5,050	2477	2638	5083	4858	3118
Chromium	mg/kg	<1	1.06	1.1	<1	1.2	1.4	<1
Cobalt	mg/kg	<0.1	0.57	0.36	0.38	0.5	0.54	0.58
Copper	mg/kg	<1	8.32	6	7	8.4	8.4	8.8
Iron	mg/kg	<20	313	294	266	424	404	294
Lead	mg/kg	<0.1	1.72	2.0	63.36	60.95	52.48	82.22
Lithium	mg/kg	<0.1	0.525	0.6	0.54	0.73	0.74	0.64
Magnesium	mg/kg	<10	3,034	3007	2978	3396	3650	3754
Manganese	mg/kg	<1	17.2	11	10.4	16.9	17.4	12.8
Molybdenum	mg/kg	<0.1	0.69	0.39	0.48	0.49	0.52	0.54
Nickel	mg/kg	<1	2.01	1.4	1.4	2	2.4	2.4
Potassium	mg/kg	<20	11,045	10660	11540	11724	12520	12660
Rubidium	mg/kg	<0.1	5.57	4.8	5.06	5.23	5.6	5.48
Selenium	mg/kg	<1	3.85	3	4	4.4	5	4.6
Silver	mg/kg	<0.1	0.134	<0.1	0.12	0.15	0.14	0.18
Sodium	mg/kg	<50	15,263	19480	18700	22100	24640	25540
Strontium	mg/kg	<1	30.95	24.6	23	34.2	38.2	32.2
Tellurium	mg/kg	<0.1	<0.027	<0.1	<0.1	<0.1	<0.1	<0.1
Thallium	mg/kg	<0.1	<0.027	<0.1	<0.1	<0.1	<0.1	0.22
Tin	mg/kg	<0.1	0.095	<0.1	<0.1	<0.1	<0.1	<0.1
Uranium	mg/kg	<0.1	0.246	0.12	<0.1	<0.1	0.12	<0.1
Vanadium	mg/kg	<1	1.51	1.8	1.4	2.3	2.4	2
Zinc	mg/kg	<1	96.6	79	148.8	124.9	140.6	181

Notes:
Shading indicates average site blue mussel soft tissue metal concentration is greater than 2-fold the average reference concentration.
[a] Data were converted from wet weight to dry weight, using the mean mussel % moisture of 82.1%, and the equation: dw = ww/(1-% moisture)

Further statistical analysis was conducted to examine whether tissue metal concentrations from the study area mussels were greater than reference. This was done for all metals except those found to be non-detect in all samples (beryllium, bismuth and tellurium), as well as essential elements (calcium, magnesium, potassium, and sodium). Statistical analysis could not be conducted for those metals that were largely non-detect in reference, but detected in study area (antimony, thallium, tin and uranium). Boxplots were generated in order to compare metal concentrations in mussels from the reference area (sites R1 and R2, combined) with those collected from the smelter exposed area (sites S1, S2, S3, and S4). Additionally, for each analyte, a Dunnett's multiple comparison test was performed to compare the concentrations from the smelter exposed mussels from each of the sample sites back to the concentrations from the reference mussels. Note, for the purpose of all graphs and statistical comparisons, all concentrations reported as less than the detection limit were replaced with the full detection limit. Additionally, data were log transformed prior to analysis to improve data normality. The statistical analyses, and boxplots of the various datasets, are presented in Appendix G.

Based on this analysis, the tissue metals highlighted in Table 4-18 were found to be significantly different from reference data.

Table 4-18 Summary of Dunnett's Tests Comparing Reference Site Analyte Concentrations to Analyte Concentrations at Sites S1, S2, S3, and S4

Analyte	Significantly Different from Reference Site (p<0.05)?			
	S1	S2	S3	S4
Arsenic	N	N	Y	N
Cadmium	Y	Y	Y	Y
Copper	N	Y	Y	Y
Lead	Y	Y	Y	Y
Selenium	Y	Y	Y	Y
Silver	N	N	N	Y
Strontium	Y	N	N	N
Zinc	Y	N	Y	Y

Notes:
Shading indicates site tissue concentrations were statistically different from reference site.

For all other metals, no significant differences were noted between study area caged mussels and reference mussels. While this suggests that water and /or food chain exposures for those metals are similar to reference, it cannot be confirmed based on the current data set whether steady state had been achieved.

For the metals/metalloids outlined in Table 4-17, no bivalve specific tissue residue guidelines related to health were identified in the literature reviewed. Based on the outcomes of the survival, growth and condition endpoints in the caged mussel assessment (see Section 4.4.2.2), these metal tissue residues do not appear to be associated with adverse effects in caged bivalves for these endpoints.

4.4.2.2 Survival, Growth, and Condition Endpoints

Survival Outcomes:

Survival was not significantly impacted by the 66-day exposure period in the vicinity of the smelter. Survival rates in the two reference cages were 96.2% and 95.2%, whereas survival rates ranged from 94% to 100% in S1 to S4, near the facility.

Growth and Condition Outcomes:

Growth was assessed by examining differences in shell dimensions and weight, between deployment and cage retrieval. Condition was assessed by examining length-at-soft tissue weight relationships. Potential for causal links between tissue concentrations and growth endpoints were also evaluated.

For growth, the absolute change in shell dimensions (length, width and height) over the 66-day test period was greater at the smelter-exposed area, compared to reference. There was significantly greater growth in blue mussel shell height at Cages S1, S3, and S4, and significantly greater increase in whole animal wet weight at S4. Overall, the data suggest that bivalves grew more quickly at the exposed sites than reference areas, and did not suggest any adverse growth effects. Minnow attributed the higher growth rates at the smelter-exposed area to potential differences in nutrient input near the smelter, when compared to reference areas (e.g., nitrogen, iron, manganese), or natural differences in environmental variables (e.g., water temperature) compared to the reference area.

For the condition assessment, statistical analysis conducted by Minnow (2015b) between Whole Animal Wet Weight (WAWW) and soma dry weight (condition) is presented in Table 4-19. Table 4-19 suggests there are some significant differences between the smelter-exposed bivalves, and reference bivalves. The following conclusions were made on these differences (Table 4-19):

- Cage S1 was significantly different based on WAWW from reference at test initiation, and as such, Minnow concluded the statistical difference in Table 4-19 is likely an artifact of the study, in that differences at onset of study could have carried through the study. With respect to Cage S1 soma dry weight, condition did not differ at test termination.
- Cage S2 was not significantly different from reference based on either WAWW or dry soma weight condition assessments.
- Cage S3 WAWW and soma dry weight conditions were significantly different from reference at test termination. The magnitude of difference was small (approximately 2%), but suggests there may be energy usage differences between mussels deployed at Cage S3 and reference mussels.
- Cage S4 has no significant differences on condition based on WAWW, but was significantly different from reference based on soma dry weight condition factors. This was attributed to individuals in cage S4 having larger shell size, and lower condition.
- Minnow suggested that the average soft tissue metal concentrations between the exposed cages were similar, and hence, differences in energy usage (condition) may be related to other factors (as opposed to metals concentrations). Additional statistics were conducted by Intrinsik for cadmium, lead and zinc to test whether metals levels in each cage group within the study area were different from other cage groups in the study area. No significant differences in tissue metals residues was noted between S1 to S4, for these metals (i.e., S1 cadmium tissue levels were not significantly different than those in S2, S3 or S4, etc.; See Appendix G).
- Since some exposed mussels grew quicker than reference area mussels, lower condition in these groups may be related to greater allocation of energy to increase shell size.

Table 4-19 Caged Blue Mussel Condition (WAWW- and Soma Dry Weight – At-Shell Length) Comparison among smelter-exposed and pooled Reference Stations At Time of Test Termination (October) (from Minnow Environmental, 2015b; Table 3.3)

Condition Type	Smelter-Exposed Cage	Model[1]	Statistical Difference Between Areas (p-value)		Mean, Adjusted Mean or Predicted Value[2]		Sample Size		Mean Square Error	MoD (%)[3,4,5]	Power
					Reference	Exposed	Ref	Exp			
WAWW	Cage S1	ANOVA[6]	Yes	0.091	0.749 (NR)	0.721	104	39	0.002045	-3.7	0.519
	Cage S2	ANCOVA[6]	No	0.603	0.734	0.740	104	33	0.002416	-	0.145
	Cage S3	ANCOVA[6]	Yes	0.034	0.737	0.719	104	43	0.002161	-2.5	0.685
	Cage S4	ANCOVA[6,7]	No	0.731	0.739	0.742	104	27	0.002073	-	0.120
Soma Dry Weight	Cage S1	ANCOVA[6]	No	0.417	2.551	2.528	104	39	0.013965	-	0.209
	Cage S2	ANCOVA[6]	No	0.548	2.528	2.508	104	33	0.016741	-	0.160
	Cage S3	ANCOVA[6]	Yes	0.049	2.532	2.484	104	43	0.014724	-1.9	0.629
	Cage S4	ANCOVA[6,7]	Yes	0.050	2.531	2.476	104	27	0.014421	-2.2	0.627

[1] Statistical tests include Analysis of Variance (ANOVA), Analysis of Covariance (ANCOVA), Mann-Whitney U-Test (MW U-test) and Kolmogorov-Smirnov test (K-S Test).
[2] The mean is reported for ANOVA, adjusted mean is reported for ANCOVA, and predicted values of the regression line equations are reported for covariate min and max values in ANCOVA where slopes were unequal.
[3] Magnitude of difference between means for reference and exposure areas calculated as: [(exposed mean -reference mean) /reference mean] x 100.
[4] Magnitude of difference between adjusted means for reference and exposed areas calculated as: [(exposed adjusted mean - reference adjusted mean) /reference adjusted mean] x 100.
[5] Magnitude of difference between predicted minimum and maximum values for reference and exposed areas calculated as: [(exposed predicted value - reference predicted value) / reference predicted value] x 100.
[6] Studentized outlier removed (samples R2-M5-05)
[7] Studentized outlier removed (samples S4-M5-03)
[8] NRR indicates no regression relationship for both the reference and smelter exposed areas at p=0.05; NR indicates no regression relationship at p=0.05. No ANCOVA appropriate as a result of NRR/NR.
[9] Statistical comparisons for all endpoints were conducted using log-transformed data with the exception of age distribution.

Overall conclusions of the caged bivalve study were that while tissue body burdens for some metals were higher in smelter-exposed areas, survival was not affected by this. In addition, growth appeared more rapid in the smelter-exposed area, than in reference areas, but condition appeared to be affected in some smelter-exposed groups. These differences, while statistically significant, were small (approximately 2 – 3%), and were considered to likely be a function of the increased energy allocation to shell size, which may have influenced tissue mass. The outcomes were not considered to be indicative of adverse effects on survival or growth of bivalves.

4.4.3 Marine Shellfish Weight of Evidence

Based on the information presented in Section 4.4.1 and 4.4.2, Table 4-20 outlines the Weight of Evidence evaluation for marine shellfish, with respect to potential risks.

Table 4-20 Weight of Evidence Evaluation for Marine Shellfish

Area of Interest	Comparisons Water Quality Data to Marine Water Quality Guidelines	Caged Bivalve Study			Potential Risks to Shellfish Populations
		Survival	Growth/Condition	Tissue Residues	
S1	Maximum water quality data less than marine water quality guidelines (As; Cd; Cu; Cr; Pb; Ni; Zn); Other metals considered to have a low risk potential, based on comparisons to effects literature, where available	97.8% survival, relative to reference range of 95 – 96%	Growth significantly greater than reference; condition significantly different from reference at test termination (wet weight only); attributed to differences at test onset	Significantly greater than reference for Cd, Pb, Se, Sr and Zn	Risk potential considered to be low
S2		100% survival relative to reference range of 95 – 96%	No difference in growth between S2 and reference; condition was not significantly different from reference	Significantly greater than reference for Cd, Cu, Pb, and Se	Risk potential considered to be low
S3		94.1% survival relative to a reference range of 95-96%	Growth significantly greater than reference; condition was significantly different from reference (wet and dry weight)	Significantly greater than reference for As, Cd, Cu, Pb, Se, and Zn	Risk potential considered to be low
S4		96.9% survival relative to a reference range of 95-96%	Growth significantly greater than reference; WAWW was greater than reference; condition was significantly different from reference for dry weight only	Significantly greater than reference for Cd, Cu, Pb, Se, Ag and Zn	Risk potential considered to be low

4.5 Marine Fish Outcomes

4.5.1 Assessment of Chemistry Data Relative to Surface Water Quality Guidelines and Statistical Differences from Reference

As indicated in Section 4.2.1 and 4.4.1, marine water quality in the study area was not considered to be elevated relative to either marine water quality guidelines (where available), or, for those substances lacking marine guidelines, when assessed relative to reference ranges, available toxicity literature, or freshwater guidelines. As such, it is considered unlikely that water quality would be significantly impacting fish health in the study area. The water quality data are limited, and hence these conclusions are uncertain. The study area is a highly dispersive environment, which would suggest that aqueous exposures for mobile species would be variable, and while aqueous concentrations could be higher than those measured during the sampling intervals, it is expected that mobile species (such as fish) would not experience exposures that could result in adverse effects.

4.5.2 Assessment of Fish Tissue Concentrations

Atlantic herring (*Clupea harengus*) and sand lance (*Ammodytes*) were captured off of Belledune Point and in reference areas using seine nets. Composite whole fish samples were analyzed for metals and mean values for both study area and reference for each species are presented in Table 4-21. Supporting field information related to specific lengths and weights of samples are presented in Appendix H. Where the mean values in study area were two-fold greater than mean reference values, the metal is shaded. Sand lance data have higher metal levels than those of herring. Sand lance lack swim bladders, and much of their time is spent buried in substrate (Robards et al, 1999). In the process of burying themselves, some sand material may be ingested incidentally by sand lance. Since the study area sand lance were captured on Belledune Point, where sand metals levels are extremely elevated (relative to reference), the fish contained sand within their guts at time of capture. Notably, at RPC Laboratories, some sand was observed in sand lance tissue vials following the digestion stage. Hence, the measured concentrations of metals within sand lance are a function of systemic uptake, and ingested sand material.

Table 4-21 Whole Fish Tissue Metals Concentrations (mg/kg ww)

Analyte	Atlantic Herring (mean)		Sand Lance (mean)	
	Reference[a]	Study Area[b]	Reference[c]	Study Area[d]
Aluminum	2.04	10.95	1.85	88
Antimony	<0.005	0.0083	<0.005	0.18
Arsenic	0.532	0.533	0.809	1.66
Barium	0.062	0.127	0.162	0.955
Beryllium	<0.005	<0.005	<0.005	0.00533
Bismuth	<0.05	0.051	0.05	NC
Boron	1.01	1.91	0.798	1.11
Cadmium	0.0658	0.0832	0.0709	0.133
Calcium	5802	6012	5286	6390
Chromium	<0.05	0.056	0.05	0.295
Cobalt	0.007	0.0136	0.0075	0.332
Copper	0.754	0.888	0.719	4.24
Iron	17.4	29.6	18.6	412
Lead	0.0956	1.26	0.026	26.0
Lithium	0.0378	0.0644	0.0578	0.130
Magnesium	555.2	701	506	608
Manganese	2.17	2.26	2.52	4.57
Mercury	<0.01	<0.01	<0.01	<0.01
Molybdenum	0.0152	0.0167	0.0154	0.085
Nickel	0.064	0.081	0.06	0.173
Potassium	3390	3407	3774	3677
Rubidium	0.703	0.755	0.943	1.02
Selenium	0.418	0.474	0.575	0.58
Silver	0.005	0.0145	<0.005	0.0215
Sodium	2018	3073	1776	2252
Strontium	13.3	17.7	17.1	22.6
Tellurium	<0.005	<0.005	<0.005	<0.005
Thallium	<0.005	0.290	0.0052	0.448
Tin	0.0076	0.0149	0.0111	0.852
Uranium	<0.005	0.005	0.0056	0.00683
Vanadium	<0.05	0.071	<0.05	0.347
Zinc	24	24.4	30.04	145

Notes:
Shading indicates mean whole fish concentration in study area is greater than two-fold the mean reference concentration guidelines.
a N = 5; b: N = 10; c: N= 10; d: N = 6

Examination of the Atlantic herring data indicates that aluminum, barium, lead, silver and thallium are present at concentrations that are greater than twice the reference mean. Lead represents the largest difference from reference at 13 times reference levels, whereas aluminum concentrations are 5 times reference levels, and silver and barium are 3 and 2 times reference, respectively. Lead, silver and thallium could be related to facility emissions. Aluminum and barium also appear to be related to facility releases, based on examination of beach sand data for reference areas (Table 3-4: reference mean aluminum: 8,025 mg/kg; mean barium: 11 mg/kg; Table 3-15: Area 1 beach sand mean concentrations: aluminum: 14,043 mg/kg; barium: 123 mg/kg). The reference data size (N = 5) is smaller than study area (N= 10), and hence, some of the Atlantic herring data from the study area may have been found to be within reference ranges, had a larger sample size been available for reference fish. The sand lance data are clearly being influenced by metals present within sand, as numerous metals are present at concentrations twice the reference mean, including aluminum, arsenic, barium, chromium, cobalt, copper, iron, lead, lithium, molybdenum, nickel, thallium, tin and zinc. Both the sand lance and Atlantic herring were captured in the vicinity of the final effluent discharge (See Figure 2-3), and hence are considered to represent reasonable worst-case tissue concentrations.

With respect to causal associations between whole fish tissue data and effects in fish, whole body tissue guidelines are available for fish only for selenium toxicity. Selenium, while an essential element, has a unique characteristic of also imparting significant toxicity to fish and egg laying vertebrates (birds and amphibians and reptiles, in particular), at levels slightly above those considered to be essential (Janz et al, 2010). In fish, the toxicity endpoint is teratogenicity, which results in a series of deformities involving the spine, fins and craniofacial areas, as well as edema (Janz et al, 2010). There has been significant research conducted to better understand selenium toxicity in fish species (e.g., Chapman et al, 2010), with the majority of focus being in freshwater environments, as certain characteristics of freshwater environments (particularly those found in still water or lentic ecosystems) can convert selenium into a highly available and toxic form, which bioaccumulates in upper trophic levels, causing significant adverse effects. Toxicity in marine systems is less well understood, particularly related to fish. Recognizing this, comparison of marine fish tissues to concentrations of selenium in whole fish associated with a low likelihood of development of adverse effects which are based on freshwater toxicity data, is considered to have a high degree of uncertainty associated with it. Nonetheless, this type of comparison was undertaken for the sand lance and Atlantic herring tissue measurements in Table 4-22. Based on the information presented in Table 4-22, selenium whole fish tissue levels in Atlantic herring and sand lance do not appear to represent a concern with respect to toxicity in fish. An additional uncertainty related to these comparisons includes the small sample sizes.

Table 4-22 Comparison of Study Area and Reference Whole Fish Tissue Selenium Concentrations with Tissue Residue Guidelines for the Protection of Fish

Fish Species	Study Area Tissue Se Concentration (mean; maximum); µg/g dw[a]	Reference Area Tissue Se Concentrations (mean; maximum); µg/g dw[a]	BC MOE (2014) Se whole body guideline	Kentucky (2013)[b]	U.S. EPA (2014) Draft[c]
Atlantic Herring[d]	2.35; 2.45	1.77; 2.1	4 µg/g dw	8.6 µg/g dw	8.1 µg/g dw
Sand Lance[e]	2.52; 2.70	2.32; 2.44			

a Wet weight data were converted to dry weight using mean site specific moisture contents (80% for Atlantic herring study area; 76% for reference; 77% for sand lance study area; 75% for reference), and the equation dw = ww/(1-% moisture).
b http://water.ky.gov/Documents/Regulations/Proposed%20Se%20Criteria%204%202%202013.pdf
c http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/selenium/upload/External-Peer-Review-Draft-Aquatic-Life-Ambient-Water-Quality-Criterion-For-Selenium-Freshwater-2014.pdf
d N = 10 for study area; N = 5 for reference
e N = 6 for study area; N=10 for reference

4.5.3 Fish Health Assessment Outcomes

Minnow (2015b) conducted a fish health assessment in October of 2014. This report is provided in Appendix F, with a précis of information presented here. Briefly, gill netting conducted in the smelter-exposure and reference areas (see Figure 2-3) yielded a total of 9 different fish species, with slightly different species being found at the two areas. Salmonids (*Salvelinus fontinalis; Salmo salar*) were absent in the smelter-exposure area, but present in reference locations, and Atlantic tomcod (*Microgadus tomcod*) and Atlantic mackerel (*Scomber scombrus*) being present in both areas (amongst other species). Catch-per-unit effort was slightly lower in the smelter-exposed area, relative to reference. Minnow (2015b) concluded that fish species diversity and density differences were minor, and were likely attributable to habitat differences between the two areas. Based on the fish survey, Atlantic tomcod was selected as the best candidate species for the health assessment, due to numbers at both reference and smelter-exposure areas. Health outcomes were assessed separately for females and males, and are summarized below. The assessment included external condition evaluation, weight and length, aging of otoliths, gonad and liver evaluation, and fecundity and egg size determinations for females.

Female Atlantic Tomcod Population Evaluation:

Based on the numbers captured and age structures, there was no difference in mean age or age distribution between the exposure area and reference, suggesting similar survival between areas. Two year old females were selected for detailed assessment, and detailed statistical analysis is provided in Appendix F (Minnow, 2015b; Table 4.2). Based on the data collected and statistical analysis conducted, the following was concluded:

- No difference in mean weight of females was found between smelter-exposed and reference populations. Total length of Age-2 females was significantly shorter at the smelter-exposed area, when compared to reference, but the magnitude of difference was small (4.6%), and well within the Critical Effect Size (CES) of +/- 25%. This statistically significant outcome was therefore not considered ecologically significant, and growth differences between female tomcod at the smelter-exposed area versus reference were considered to be subtle, and within ecologically relevant thresholds.

- Both gonad size of females and egg size at the smelter-exposed area were significantly smaller than reference. While gonad size differences were slightly below the ecologically relevant CES of +/- 25% (magnitude of difference: – 19.1), the egg size was slightly above this threshold (magnitude of difference: – 31.7). There was no significant difference in fecundity between the exposure and reference areas, with females in both areas having egg counts well above averages for this species. Minnow hypothesized that the findings of smaller gonad and egg size in the smelter-exposed area, relative to reference, could reflect natural variability in spawning time between the two populations, with reference area females having slightly more advanced gonad development.
- Condition and relative liver weight were not significantly different between females in the smelter-exposed area and the reference area, which suggests that energy storage and usage between the two populations is similar. This suggests that food quality and quantity is similar between the study area and reference, and an examination of stomach contents indicated a similar dietary composition and relative mass of food items consumed between the areas.

Male Atlantic Tomcod Population Evaluation:

The male tomcod evaluation was handicapped by low catch rates. Only 8 males were caught in the exposure area, and 6 in reference, which limits the interpretation of the data. It was hypothesized that males may have started migration from the marine areas to river mouths for staging or spawning.

Based on the limited sample numbers, conclusions with respect to males are uncertain. Conclusions are as follows:

- No differences in mean age or age distribution between exposure and reference areas were noted, which suggest similar survival between areas. Age-2 males were selected for detailed assessment, based on maturity and available individuals.
- No differences in mean weight or total length between exposure and reference males were noted;
- No significant differences in relative gonad size, condition, or relative liver size between the two areas were noted;
- No abnormalities or deformities were noted in fish from either area
- While the available data suggest no smelter-related influences on male tomcod, the small sample size makes these conclusions uncertain.

4.5.4 Marine Fish Weight of Evidence

Based on the information presented in Section 4.5.1 - 4.5.3, Table 4-23 outlines the Weight of Evidence evaluation for marine fish, with respect to potential risks.

Table 4-23 Weight of Evidence Evaluation for Marine Fish

Area of Interest	Comparisons Water Quality Data to Marine Water Quality Guidelines	Fish Health Study			Tissue Residues	Potential Risks to Marine Fish
		Survival	Growth/Condition	Reproduction		
Smelter-exposed	Maximum water quality data less than marine water quality guidelines (As; Cd; Cu; Cr; Pb; Ni; Zn); Other metals considered to have a low risk potential, based on comparisons to effects literature, where available	Considered to be similar to reference population; male data uncertain due to low sample size	Females: weight not significantly different from reference; length was significantly shorter in smelter-exposure area, but not beyond CES threshold; Males: no weight or length differences between smelter and reference area; sample numbers are small, and conclusions uncertain.	Female: gonad size significantly smaller than reference, but not beyond CES threshold; egg size significantly smaller than reference and beyond CES threshold; no significant differences in fecundity , relative to reference population Males: relative gonad size not different from reference, but sample size is small, and therefore conclusions are uncertain	Average Pb concentrations in whole fish tissues (Atlantic herring) are 13 times average reference concentrations; Al and Ag are 5 times and 3 times average reference concentrations, respectively. Fish health tissue guidelines only available for Se; measured tissues well within guidelines	Risk potential considered to be low; while CES was exceeded for egg size, Minnow concluded that the small egg size could reflect natural variability in spawning timing between exposed and reference populations. Fecundity was above average, relative to literature based values. Male outcomes are uncertain due to limited sample size

4.6 Conclusions – Marine Life

Based on the data and assessments conducted, the following conclusions and uncertainties are noted for marine life.

Marine Phytoplankton and Pelagic Invertebrates:

- Risks are considered to be negligible to low, based on comparison of measured water quality metals concentrations to marine water quality guidelines and reference, as well as other toxicology data and information.
- The exposure data are limited in terms of number of samples, and hence there is uncertainty in this conclusion. This uncertainty is reduced by knowledge that the area adjacent to the smelter is a highly dispersive environment, and while releases from the facility are measureable in the environment, exposure levels for transient mobile species are expected to be low, and hence would not be anticipated to result in population- or community-level effects.

Marine Benthic Community:

- Risks are considered to be low for FPO and SST2, and moderate for FE, based on the existing chemistry data, and benthic density, diversity and richness data. Evenness and diversity of the benthic community at FE was > +/- 2 SD of reference, suggesting ecologically meaningful differences between the two areas. There was also a reduced diversity in this area, albeit, not beyond +/- 2 SD of reference. In the 2014 survey, increased sediment metals concentrations and lower benthic invertebrate density and differences in community structure, relative to surveys conducted in 2008 and 2004, were noted, which suggest recent events (either the harbor dredging project in 2010, and/or erosion of the slag pile at Belledune Point as a result of a major storm event) may have had a significant impact on sediments in the near-field area of the effluent discharge.
- Uncertainties in these conclusions are limited, based on the collection of site-specific benthic community data, with the exception of the SST2 area, which is lacking benthic community data. Comparisons of chemistry data between SST2 and FE and FPO, and consideration of benthic community outcomes at FE and FPO, relative to SST2 suggest risk potential at this area is low.

Marine Shellfish:

- Risks are considered to be low, based on the available data and studies conducted. Survival was not considered to be influenced in the study area, relative to reference. Growth was actually greater in the study area mussels at several sites, than in reference areas, but condition was slightly lower. These results were attributed to higher allocation of energy use to growth in the smelter-exposed mussels compared to reference. While tissue metals were significantly higher in the exposure group for arsenic, cadmium, copper, lead, selenium, silver, strontium and zinc, the results of the survival, growth and condition endpoints suggests no adverse smelter-related effects in blue mussels.

- Uncertainties in the assessment include a lack of assessment of the reproductive endpoint, since the study was initiated outside of the season of reproductive tissue development (and hence reproduction endpoint could not be evaluated). Nonetheless, numerous juvenile blue mussels were found adhering to the cages of the deployed mussels. While a quantitative assessment of reproductive endpoints was not undertaken, qualitative observations suggest presence of juveniles in all cage areas, with lowest numbers being observed at the smelter-exposed station furthest from the smelter (Station S4).

Marine Fish:

- Risks are considered to be low, based on assessment of water quality, survival, growth/condition, reproduction and tissue residue data. No critical effect sizes were exceeded for any fish survey endpoint with the exception of egg size. Minnow concluded the small egg size in smelter-exposed fish could reflect natural variability in spawning timing between the exposure and reference fish populations.
- Male outcomes are uncertain due to limited sample size, and no tissue residue guidelines related to fish health were located in the literature reviewed for metals found to be present at concentrations greater than 2 times the mean reference concentrations. Therefore, it is not known whether measured tissue residues are associated with adverse health outcomes.

5.0 AVIAN SPECIES ASSESSMENT

Potential risks to avian species feeding on the marine areas in the study area on Belledune Point were assessed. Methods used to assess potential risks are provided in Section 5.1, while results for receptors are provided in Sections 5.2 to 5.4 and overall conclusions for avian species in Section 5.5.

5.1 Methods

The assessment endpoints, measurement endpoints and lines of evidence used to evaluate avian species assessed in the ERA (i.e., common tern, black-crowned night heron, spotted sandpiper) were provide in Section 2.0, while a summary of the lines of evidence are provided below in Table 5-1.

Table 5-1 Assessment Endpoints, Measurement Endpoints and Lines of Evidence

Receptor Group	Lines of Evidence
Piscivorous avian species (e.g., common tern)	- Predicted Exposure Ratios (ER) from exposure modelling (*i.e.*, comparison of estimated or measured COC exposures via oral ingestion of fish to Toxicity Reference Values (TRVs)). - Comparison of fish tissue residue data to tissue effects literature for piscivores - Comparison of liver, kidney or egg tissue residues in avian mortalities to tissue effects literature - Consider toxicological / biological information from other studies and extrapolate where applicable to this study - Compare clutch counts to counts in other parts of New Brunswick to determine if numbers are different in the smelter colony
Omnivorous avian species (e.g., black-crowned night heron)	- Predicted Exposure Ratios (ER) from exposure modelling (*i.e.*, comparison of estimated or measured COC exposures via oral ingestion of fish, beach sand, near-shore invertebrates; etc., to Toxicity Reference Values (TRVs)). - Consider toxicological / biological information from other studies and extrapolate where applicable to this study
Invertivorous avian species (spotted sandpiper)	- Predicted Exposure Ratios from exposure modelling (*i.e.*, comparison of estimated or measured COPC exposures via oral ingestion of beach sand and invertebrates to Toxicity Reference Values (TRVs)). - Observational counts - Consider toxicological / biological information from other studies and extrapolate where applicable to this study

5.1.1 COPCs for Receptors Quantitatively Modelled in the ERA

To determine the chemicals of potential concern (COPCs) that would be quantitatively modelled in the ERA, beach sand concentrations were screened by comparing maximum site concentrations to relevant ecological-based guidelines and to the 95th percentile reference area concentrations. For metals with no applicable guidelines, and with maximum concentrations greater than the 95th percentile of reference, additional comparisons between reference and site concentrations were conducted to determine if study area beach sand concentrations were statistically different from the reference area.

Beach sand samples were collected along the shore of Belledune Point, to the east of Belledune Point, and in reference areas to the west of Belledune Point (close to Little Belledune Point).

The shoreline area in which these samples were collected was divided into three sampling zones and one reference area. The sampling zones in the study area were:

- Area 1: Data collected on Belledune Point, which is owned by Glencore and has been industrially altered with cleared trees, etc. (i.e., samples SBS-1 to SBS-7; Figure 2-3);
- Area 2: Data from areas that are owned by Glencore but not industrially altered (i.e., samples SBS-8 to SBS-14; Figure 2-3); and
- Area 3: Data collected from areas that are not industrially altered and not owned by Glencore (i.e., samples SBS-15 to SBS-21; Figure 2-3).

Details of the screening are provided in Appendix I and summarized in Table 5-2.

Table 5-2 COPCs Selected for Quantitative Modelling in the ERA in Each Study Area

COPC	Area 1	Area 2	Area 3
Aluminum	√	√	√
Antimony	√	x	√
Arsenic	√	√	√
Bismuth	√	x	x
Cadmium	√	√	√
Copper	√	x	x
Iron	√	√	√
Lead	√	√	√
Lithium	√	√	√
Mercury	√	x	√
Selenium	√	x	x
Strontium	√	√	√
Tellurium	√	x	x
Thallium	√	√	√
Zinc	√	√	√

Notes: √ = yes; x = no

While antimony, bismuth and tellurium were carried forward for quantitative modelling in the ERA, no TRV could be identified for these metals and there was a paucity of toxicity data from which to derive a TRV. Given this, antimony, bismuth and tellurium could not be quantitatively evaluated in the assessment. Bismuth is widely used in cosmetics and as a therapeutic agent, and has a low toxic potency. The oral LD50 in rats is reported to be greater than 2,000 mg/kg body weight (Sano et al, 2005). While not assessing these metals is an uncertainty in the assessment, the potential risks related to these metals is assumed to be less than metals that are being evaluated which are known to be highly toxic, and as such these metals would not be driving the overall risk assessment conclusions.

With respect to mercury, one of the key pathways of exposure for common tern, heron, and sandpiper is food ingestion. Fish tissue concentrations considered to represent a low potential for adverse effects in fish-eating wildlife species were sought from the literature. Values were only identified for methylmercury. Methylmercury toxicity in fish-eating wildlife is associated with reproductive effects and neurotoxicity (CCME, 2001). The CCME (2001) has developed a fish-eating wildlife tissue residue level to protect against possible effects associated with methylmercury in fish tissues. This guideline was derived based on the Wilson's storm petrel, which is a small ocean-foraging species with a high metabolic rate, which has a food intake almost equal to its entire body weight each day (CCME, 2001). While this guideline could be revised to account for species being assessed in the current study (such as common tern), which would result in a higher allowable fish tissue level, the storm petrel-based guideline was used for comparison purposes to be conservative. Table 5-3 presents the fish tissue data for Atlantic herring, sand lance, and shoreline invertebrates, relative to the mercury tissue residue guideline. In all cases, mercury levels were non-detectable at a concentrations of < 10 µg/kg (N = 39 samples in total for the study area), relative to the fish-eating wildlife guideline of 33 µg/kg. Based on these comparisons, mercury in aquatic foods was not considered to represent a concern for avian consumers, and was not quantitatively modelled. While incidental ingestion of beach sand could contribute to overall mercury exposures for avian species, mercury concentrations in beach sand were well below guidelines, and would be the inorganic form of mercury which is much less toxic, as opposed to methylmercury. Egg and chick mercury concentrations were assessed for possible effects in the common tern in Section 5.3, which provides site-specific exposure indications for this species.

Table 5-3 Comparison of Study Area and Reference Whole Fish Tissue and Shoreline Invertebrate Mercury Concentrations with Tissue Residue Guidelines for the Protection of Fish-Eating Wildlife

Dietary Species	Study Area Tissue Hg Concentration (maximum; µg/kg ww)	Reference Area Tissue Hg Concentrations (maximum; µg/kg ww)	CCME Tissue Residue Guideline CH_3Hg (µg/kg ww)
Atlantic Herring[a]	< 10	< 10	33
Sand Lance[b]	< 10	< 10	
Shoreline Invertebrates (Area 1)[c]	< 10	< 10	
Shoreline Invertebrates (Area 2)[d]	< 10		
Shoreline Invertebrates (Area 3)[e]	< 10		

Notes: ww = wet weight
a N = 10 for study area; N = 5 for reference
b N = 6 for study area; N=10 for reference
c N = 6 for Area 1 study area; N = 6 for reference
d N = 9 for Area 2 study area
e N = 8 for Area 3 study area

Based on the screening undertaken, the resulting COPCs which were quantitatively modelled in the ERA are:

- Aluminum
- Arsenic
- Cadmium
- Copper
- Iron
- Lead
- Lithium
- Selenium
- Strontium
- Thallium

- Zinc

5.1.2 Toxicity Reference Values for COPCs and Receptors

To evaluate potential risks to avian receptors, toxicity reference values (TRVs) were identified for the COPCs. Where possible, effects-based TRVs were used to evaluate potential risks. Effects based TRVs were derived for arsenic, cadmium, copper, lead, selenium and zinc based on toxicity data provided by the U.S. EPA EcoSSL (ecological soil screening levels) documents, while a U.S. EPA TRV for thallium was identified and used (See Table 5-4). For iron, lithium and strontium, no effects-based TRVs were identified in the literature reviewed. The TRVs for these COPCs were derived based on maximum tolerable levels (MTLs) derived by the NRC (2005) for poultry (Table 5-5). The effects associated with exposures greater than the MTLs generally include food aversion and decreased growth. Nevertheless, given a lack of other TRVs the MTLs were used for the ERA modelling of these COCPs.

Table 5-4 Effects-Based Toxicity Reference Values (TRVs) for Marine Bird Receptors Carried Forward for Assessment

COPC	Receptor	TRV (mg/kg/day)	Comment	Reference
Aluminum	Heron Sandpiper Tern	164	No effects-based toxicity reference values for aluminum could be found in the literature reviewed. A 4-month feeding study was conducted on Ringer Turtle-Doves, with doses of up to 1500 ppm in the diet (aluminum sulphate). These doses were not associated with any effect on egg production, fertility, or hatchability. Egg permeability was decreased initially, but recovered to normal levels and was not considered significant. Growth of juvenile Ringed Turtle- Doves (days 21 – 63) fed the same diets was not affected by the dietary levels of aluminum. A NOEL (growth and reproduction) TRV of 164 mg/kg/d was calculated based on the high dose of 1500 mg/kg-food, and on 0.166 kg body weight (Carrière et al, 1985) and 0.0181kg/day consumption rate for ringed turtle-dove (U.S. EPA, 1993; Equation 3-3).	Carrière et al, 1986
Arsenic	Heron Sandpiper Tern	14	No bounded reproduction or growth LOAELs for avian species were reported in the EcoSSL for Arsenic (U.S. EPA, 2005a). Three unbounded LOAELs for growth were identified (1.49 mg/kg/day, 3.55 mg/kg/day and 17.3 mg/kg/day). One of these growth LOAELs was less than the avian TRV of 2.2 mg/kg/day derived by the U.S. EPA (2005a). The U.S. EPA (2001a) derived an EC20 of 13.91 mg/kg/day for arsenic based on reproductive effects in mallard ducks exposed to sodium arsenate for greater than 10 weeks (Stanley et al, 1994). Given no bounded LOAELs were identified in the literature reviewed; the EC20 for arsenic of 13.91 (rounded to 14 mg/kg/day) was selected as the TRV for avian species.	U.S. EPA, 2001

Table 5-4 Effects-Based Toxicity Reference Values (TRVs) for Marine Bird Receptors Carried Forward for Assessment

COPC	Receptor	TRV (mg/kg/day)	Comment	Reference
Cadmium	Heron Sandpiper Tern	2.37	Several bounded avian LOAELS were reported in the U.S. EPA (2005b) EcoSSL document for cadmium (i.e., 5 reproductive studies; 6 growth and 3 survival). Toxicity tests were mainly conducted on chicken, duck and quail. The bounded avian reproductive LOAELs ranged from 2.37 to 21.1 mg/kg/day. The bounded growth LOAELs ranged from 7.08 to 37.6 mg/kg/day and survival from 14.3 to 44.6 mg/kg/day. Given the limited toxicity data available, the lowest bounded reproductive LOAEL of 2.37 mg/kg/day was selected for the heron, common tern and sandpiper TRV.	U.S. EPA, 2005b
Copper	Heron Sandpiper Tern	12	Seventeen bounded reproduction LOAELs for avian species were reported in the EcoSSL for Copper document (U.S. EPA, 2007a). Of these studies, one was approximately 6-fold the next highest bounded LOAEL. To provide a more conservative LOAEL-based TRV, this LOAEL (*i.e.*, 318 mg/kg/day) was excluded from the geometric mean of the LOAEL calculation (if this LOAEL was not excluded, the LOAEL-based geometric mean would be 37 mg/kg/day). The geometric mean of the 16 other bounded reproductive LOAELs was 32 mg/kg/day. Given all of the reproductive LOAELs identified were obtained from dietary exposure studies on the chicken to mainly one form of copper (*i.e.*, copper sulphate pentahydrate), the lowest bounded LOAEL was selected for assessment. The lowest bounded reproductive LOAEL for copper identified by the U.S. EPA (2007a) is 12.1 mg/kg/day (rounded to 12 mg/kg/day). No uncertainty factor was applied to this LOAEL to derive the TRV.	U.S. EPA, 2007a

Table 5-4 Effects-Based Toxicity Reference Values (TRVs) for Marine Bird Receptors Carried Forward for Assessment

COPC	Receptor	TRV (mg/kg/day)	Comment	Reference
Lead	Heron Sandpiper Tern	9.9	Five bounded reproductive LOAELs for avian species were identified in the EcoSSL for Lead document (U.S. EPA, 2005c). Three of these LOAELs were quite low (at 1.94, 3.26 and 4.04 mg/kg/day) while the two other LOAELs were much higher (126 and 135 mg/kg/day). All of these LOAELs were derived from dietary studies on chicken or quail by exposing them to lead acetate or lead oxide. The two lowest LOAELs (1.94 and 3.26 mg/kg/day) were from the same study that the U.S. EPA (2001a) used to derive an EC20 for lead (Edens and Garlich, 1983). In this study, lead acetate was given to domestic leghorn chicken hens and to Japanese quail hens. The study concluded that quail were more sensitive than chickens. U.S. EPA (2001a) estimated an EC20 of 9.9 mg/kg/d from the chicken reproductive data, because a dose-response model would not fit the quail data. The EC20 of 9.9 mg/kg/day was selected as the LOAEL-based TRV for avian receptors in this assessment. While this EC20 is slightly above the range of the three lowest avian LOAELs from lead reported in the U.S. EcoSSL document (U.S. EPA, 2005c), it is much lower than the 2 LOAELs that reported higher concentrations.	U.S. EPA, 2001
Selenium	Heron Sandpiper Tern	0.37	Several bounded LOAELS were reported in the EcoSSL document for selenium (i.e., 8 reproductive, 16 growth and 19 survival studies) (U.S. EPA, 2007b). Toxicity tests were mainly conducted on chicken and ducks. The bounded avian reproductive LOAELs ranged from 0.368 to 2.58 mg/kg/day. The bounded growth LOAELs ranged from 0.370 to 11.9 mg/kg/day and survival from 0.371 to 29 mg/kg/day. The lowest bounded LOAELs for reproduction, growth and survival were all very similar even though they were from 3 separate studies in chickens (i.e., 0.368 mg/kg/day in Thapar et al, 1969; 0.370 mg/kg/day in Jensen, 1986 and 0.371 in Arnold et al, 1973 for reproduction, growth and survival respectively). Given this, it was decided to select the lowest bounded reproductive avian LOAEL of 0.368 mg/kg/day (rounded to 0.37 mg/kg/day) as the TRV for the heron, sandpiper and tern. This value is lower than a bird TRV for selenium of 0.5 mg/kg/day derived by the U.S. EPA (1999) based on a chronic mallard study (Heinz et al, 1987). The value of 0.37 mg/kg/day was selected instead of the TRV of 0.5 mg/kg/day as effects on survival were noted in one study at concentrations lower than 0.5 mg/kg/day.	U.S. EPA, 2007b

Table 5-4	Effects-Based Toxicity Reference Values (TRVs) for Marine Bird Receptors Carried Forward for Assessment			
COPC	**Receptor**	**TRV (mg/kg/day)**	**Comment**	**Reference**
Thallium	Heron Sandpiper Tern	0.35	The avian TRV derived by U.S. EPA (1999) was based on an acute LD50 for the starling of 35 mg/kg/day (Schafer, 1972). An uncertainty factor of 0.01 was applied to this study to give a TRV of 0.35 mg/kg/day.	U.S. EPA, 1999
Zinc	Heron Sandpiper Tern	77	Thirty-four bounded LOAELs for avian species (6 reproductive, 21 growth and 7 survival) were reported in the EcoSSL for Zinc document (U.S. EPA, 2007c). Bounded LOAELs were from studies where chicken, turkey or Japanese quail were exposed to zinc via food. The lowest bounded LOAEL of 66.5 mg/kg/day (for reproductive effects in chickens via exposure to zinc acetate in food; Gibson et al, 1986) was based on a chemical form assumed not to be relevant to this ERA (*i.e.*, zinc acetate) and this LOAEL was excluded from further consideration. The next lowest bounded LOAEL identified by the U.S. EPA (2007c) is 76.7 mg/kg/day (rounded to 77 mg/kg/day) from a study of zinc oxide exposure in the diet of chickens (Stevenson et al, 1987). The LOAEL of 77 mg/kg/day was selected as the TRV for the heron, sandpiper and tern. This LOAEL is less than the U.S. EPA (2001a) EC20 of 135 mg/kg/day derived from a chicken reproduction study (Stahl et al, 1990).	U.S. EPA, 2007c

Table 5-5 Toxicity Reference Values (TRVs) Based on Maximum Tolerable Levels (MTL) for Marine Bird Receptors Carried Forward for Assessment

COPC	Receptor	TRV / MTL (mg/kg/day)	Comment	Reference
Iron	Heron Sandpiper Tern	37	Limited iron toxicity data were available. The NRC (2005) derived a maximum tolerable level for iron of 500 mg/kg-food for poultry. The NRC (2005) reported characteristic signs of chronic iron toxicity include a reduction in feed intake, growth rate, and efficiency of feed conversion. The MTL was converted to a dose based on 1.5 kg body weight and 0.11 kg-food/day consumption rate for poultry (Sample et al, 1996), yielding a value of 37 mg/kg/d.	NRC, 2005
Lithium	Heron Sandpiper Tern	2	Limited lithium toxicity data were available. The NRC (2005) derived a lithium MTL of 25 mg/kg-food for poultry. They reported that this amount would not be associated with food aversion and thus not result in decreased food intake. Similarly, this concentration was reported not to result in apparent toxicity signs not related to decreased food intake. The MTL was converted to a dose based on 1.5 kg body weight and 0.11 kg-food/day consumption rate for poultry (Sample et al, 1996), yielding a value of 2 mg/kg/d.	NRC, 2005
Strontium	Heron Sandpiper Tern	147	Limited strontium toxicity data were available. Strontium is reported to be of low toxicity when dietary calcium is adequate. At very high doses and when dietary calcium is inadequate, calcium metabolism may be affected. NRC (2005) reported that chicks can tolerate 3,000 mg/kg-food strontium (0.3%) when dietary calcium is adequate. The NRC (2005) derived a strontium MTL of 2000 mg/kg for poultry. The MTL was converted to a dose based on 1.5 kg body weight and 0.11 kg-food/day consumption rate for poultry (Sample et al, 1996), yielding a value of 147 mg/kg/d.	NRC, 2005

5.1.3 *Food Chain Modelling Methods*

The methods used to estimate the estimated daily intake (EDI) of COPC were based on food and sand exposures in each area of interest. The following equation was used to estimate EDI for each receptor and area of interest (FCSAP, 2012; U.S. EPA, 1996):

$$EDI = (C_{soil} \times SIR \times BA_{soil}) + \sum_{1}^{j} \frac{C_j \times P_j \times FMR \times BA_j}{ME_j}$$

Where

EDI = Estimated daily intake of COPC by receptor (mg/day)

C_{soil} = Concentration of metal in sand (mg/kg)
SIR = Sand ingestion rate (kg/day)
BA_{soil} = Bio-accessibility of COPC in sand (Unitless; %)
C_j = Concentration of metal in food item j (mg/kg-dw)
P_j = Proportion of food item j in diet (Unitless; %)
FMR = Free-living metabolic rate (FMR) (kcal/day)
BA_j = Bio-accessibility of food item j in diet (Unitless; %)
ME_j = Metabolizable energy of the food item j in diet (kcal/kg-DW)

The free-living metabolic rate (FMR) is defined as the total daily energy requirement for an animal in the wild and includes energy costs of basal metabolic rate (BMR), resting metabolic rate (RMR), thermoregulation, locomotion, feeding, predator avoidance, alertness, posture, and other energy expenditures to meet daily energy requirements. Combined with the metabolizable energy (ME) that is available in the receptor's forage or prey, the EDI can be predicted.

Predicted EDI values were normalized to the receptor body weight based on the following equation:

$$TDI = \frac{EDI}{BW}$$

Where
TDI = Total daily intake of COPC by receptor (mg/day)
EDI = Estimated daily intake (mg/kg/day)
BW = Body weight (kg)

The EDI was predicted for each area of interest, receptor and COPC on a probabilistic basis. Probabilistic assessment methods use a modelling technique called Monte Carlo simulation where parameter values are drawn at random from defined input probability distributions, combined according to a model equation, and the process repeated iteratively until a relatively smooth distribution of outcomes is predicted. Probabilistic rather than a deterministic (fixed parameter value) assessment was selected to provide a more informative assessment that incorporates spatial variability in exposure media and biological attributes of wildlife (e.g., body weight, feeding rate, diet, etc). The result of a probabilistic assessment is a distribution of possible outcomes calculated based on distributions of important biological, chemical, physical, and environmental parameters that are linked through mathematical equations. As such the natural variability in parameters can be acknowledged, characterized and incorporated with the avian exposure model. Probabilistic methods in this assessment provide more information on the range and likelihood of potential outcomes among individuals in a population from exposure to each area of interest. The model did not include correlation among variables as it was unnecessary. Exposures were assumed to be random in each area of interest and receptor ingestion rate was automatically correlated with body weight through the equation used to predict FMR.

The wildlife exposure modelling relied on the use of an EDI model. The primary focus of the model is on ingestion of prey and sediment, which are generally the most important exposure pathways for wildlife (Moore and Caux, 1997; Moore et al, 1999). Thus, the avian exposure model did not include the dermal or inhalation routes of exposure in the model calculations (Suter et al, 2000). In addition, the avian exposure model excluded water ingestion, as water typically represents a less important contribution to exposure.

Appendix J provides a description of the model input parameters used in the avian exposure model based on the following components:

- Media concentrations;
- Sand ingestion rates;
- Receptor variables;
- Free-living metabolic rate (FMR);
- Dietary apportionment;
- Miscellaneous variables; and
- Model Precision and Uncertainty.

5.1.4 Bioaccessibility

An important factor to consider in assessing risks related to metals in the environment is bioaccessibility of metals from both soils (in this study, beach sand) and food sources. While the application of bioaccessibility data in ERAs is not commonplace, there is growing consensus that the general default assumption that all metals bound to soils are 100% bioavailable is overly conservative. The use of bioaccessibility data for lead and arsenic in ecological risk assessments of waterfowl and mammals has been recently published (*e.g.*, Furman et al, 2006; Ollson et al, 2009; Saunders et al, 2011). Ollson et al, (2009) found that accounting for arsenic bioaccessibility in an ecological risk assessment of deer mice living on mine tailings resulted in an order of magnitude reduction in calculated risks. This study also found that exposure assessment results, when derived based on the bioaccessible-estimated daily intake of arsenic (in soil and vegetation), were not significantly different from results derived based on the actual daily intake (based on measured data from stomach contents), indicating that the incorporation of arsenic soil bioaccessibility data into an ERA provides a more realistic assessment of risk. Similarly, Saunders et al, (2011) found that the bioaccessible fraction of arsenic in soil was significantly less than total arsenic. The authors concluded that the use of site-specific bioaccessibility data in ERAs may result in a more realistic level of conservatism. Furman *et al*. (2006) examined lead bioaccessibility in sediments in the Coeur d'Alene River basin (an area impacted by historical mining and smelting activities), to waterfowl using the basin. Bioaccessibility of lead in sediments using a modified PBET (physiologically based extraction test) technique ranged from 27% to 12%, depending on the area tested. Amendment of sediments with Phosphorus (P) significantly reduced bioaccessibility (to < 1%).

Bioaccessibility testing was conducted on beach sands collected in this study, for the key COCs of interest, following some preliminary risk modelling. The preliminary modelling was used to examine the relative contribution of beach sand to overall exposure, which was found to be high for the sandpiper in particular (due to the elevated ingestion rate for sand in this species). Based on these findings, a total of 9 beach sand samples from the study area (3 from each shoreline area; i.e., Areas 1, 2 and 3) as well as 1 reference beach sand sample were sent to the Environmental Sciences Group lab at the Royal Military College in Kingston, for bioaccessibility testing, following an avian protocol (see Appendix K). Samples sent for testing were based on the initial metals analyses, such that those samples with the highest lead, arsenic, cadmium or zinc concentrations were selected and submitted for bioaccessibility testing. While the direct relevance of these estimates to the various species that are being assessed in the current ERA can be debated (Marshall et al, 2010), all available data strongly suggest that beach sand bioaccessibility of the metals of interest will be less than 100%.

Bioaccessibility outcomes from RMC (2014) are summarized in Table 5-6 and presented in full in Appendix K. Bioaccessibility of metals within both the gastric phase (Phase 1; wherein pH is dropped to 2.6) and intestinal phase (Phase 2; wherein pH is increased to 6.2) was assessed. For exposure modelling, a range of values was input, from the lowest to highest estimated values.

Table 5-6 Gastric (Phase 1) and Intestinal (Phase 2) Percent Bioaccessibility for Beach Sand in Areas 1, 2, 3 and Reference (RMC, 2014)

	Aluminum (Al)	Arsenic (As)	Cadmium (Cd)	Copper (Cu)	Iron (Fe)	Lead (Pb)	Selenium (Se)	Thallium (Tl)	Zinc (Zn)
	%BA	%BA	%BA	%BA	%BA	%BA	%BA	%BA	%BA
PHASE 1 (Gastric)									
Area 1	0.64	7.1	5.4	0.14	4.3	3.5	NC	12	8.1
	0.84	7.9	6.6	1.5	5.0	4.5	NC	17	9.2
	0.66	6.6	7.9	4.5	4.2	5.7	NC	12	9.3
Area 2	0.044	2.2	34	12	0.073	17	NC	19	2.5
	0.14	6.8	40	19	(<0.04)	38	NC	19	3.4
	0.079	6.7	62	16	0.090	49	NC	22	13
Area 3	0.13	13	77	24	0.28	77	NC	35	39
	(<0.06)	3.7	13	1.8	0.12	6.5	NC	15	4.3
	0.18	8.3	48	9.1	0.27	50	NC	36	26
Reference	(<0.12)	<7.4	NC	11	0.012	7.2	NC	NC	(<5.4)
PHASE 2 (Intestinal)									
Area 1	<0.06	7.3	5.0	0.74	0.31	0.18	NC	12	3.8
	<0.06	9.0	6.4	1.8	0.55	0.25	NC	20	5.4
	<0.05	6.7	6.4	2.9	0.46	0.45	NC	14	4.0
Area 2	<0.07	3.8	28	(<4)	(<0.04)	5.8	NC	19	3.8
	0.10	7.4	37	6.2	(<0.04)	19	NC	22	22
	0.10	11	55	25	0.016	20	NC	23	34
Area 3	<0.06	7.2	51	17	0.007	15	NC	28	16
	<0.06	3.9	13	1.8	0.052	2.3	NC	18	3.5
	0.073	6.8	31	5.8	(<0.03)	9.0	NC	29	13
Reference	<0.12	<7.4	NC	(<11)	(<0.06)	1.2	NC	NC	5.6

Notes:
NC = could not be calculated
(<#) indicates less than an approximate percent bioaccessibility, calculated from detection limit and total concentration

It is interesting to note that samples from Area 1, which is most heavily influenced by slag, have lower bioaccessibility for several metals (e.g., cadmium, copper, lead and zinc) than those from Areas 2 or 3. This is likely due to the fact that slag is formed after intense heating in the blast furnace of the smelter, and is in a glassy matrix. Metals within this matrix do not appear to be readily released and available for uptake within the gut or intestine. Hence, despite the very elevated concentrations of metals in these samples, exposures are reduced due to the lack of availability of the metals. Areas 2 and 3, which contain some slag, but are more dominated by natural sand, have higher bioaccessibility for several metals, such as cadmium, copper and lead (Phase 1) and thallium and zinc (Phase 2).

5.1.5 Risk Characterization and Weight of Evidence Evaluation Approach

The wildlife risk model was used to predict exposures for each COPC on a probabilistic basis. Exposure estimates then were compared to the point estimate TRV for specific metals and receptors. The risk is expressed as a Hazard Quotient (HQ) value and calculated as follows:

$$Hazard\ Quotient = \frac{Exposure\ Estimate}{Toxicity\ Reference\ Value}$$

Predicted HQ values are the ratio between the measure of exposure (numerator) and effect threshold (denominator). An HQ of < 1 is generally considered indicative of negligible risks for the endpoint considered, as conservative assumptions are used to derive the HQ values. If an HQ is >1, then there is a possibility for adverse effects but uncertainty in both the numerator and denominator need to be considered prior to determining risk potential. As the HQ is dependent upon the information used to parameterize the numerator and denominator of the equation, interpretation of the magnitude of a HQ (beyond the comparison to 1.0) is not universal, and requires careful consideration of the uncertainties and confidence in the various data inputs (FCSAP, 2012).

HQs are not directly proportional to the magnitude of risk. They do not contain information about the specific probability that an adverse effect will occur or the magnitude of that effect nor can they be scaled across different metals for risk ranking (FCSAP, 2012). However, when HQs are examined individually by chemical and by receptor with consideration of the degree of confidence and uncertainties associated with the input values used to derive the magnitude of the HQ can offer information with respect to potential risks.

The predicted HQ values were used to rule out unacceptable risk as opposed to predicting risks. The risk assessment results were first reviewed to determine the probability of the ER exceeding an HQ=1.0. The probabilities were interpreted as follows:

- 10% or greater probability of HQ value less than or equal to 1.0: signifies that most estimated exposures are less than the TRV (i.e., NOAEL, LOAEL, IC20, etc.), indicating that the likelihood of adverse effects is negligible.
- Greater than 10% probability of HQ value greater than 1.0: potential for adverse effects is not ruled out; however, the significance of this potential must be judged according to the uncertainty and degree of conservatism incorporated into the risk assessment, as well as site-specific information.

Presenting the probabilities of exceeding an HQ value of 1.0 does not include information regarding the magnitude of the exceedance. Therefore, where risks could not be ruled out, the average and 95th percentile HQ value was also presented.

With respect to the overall categorization of risk potential, Table 5-7 presents the approach taken to decision-making on risk potential. The risk potential ranking was based on consideration of exposure and hazard inputs into the modelled HQ value, as well as outcomes from other lines of evidence, such as clutch counts, tissue residues, relative to effect levels, etc.

Table 5-7	**Risk Characterization Approach**			
Receptor Group	**Risk Potential**			
	Negligible	**Low**	**Moderate**	**High**
Piscivorous Avian Species (common tern)	No effect on individuals expected	Possible effect on some individuals expected, but low probability of effects on colony	Adverse effects on individuals could occur; effect on the local colony possible, that may have an impact on the sustainability of the local colony	Significant probability of impacts on the sustainability of the local colony
Omnivorous Avian Species (Black-crowned night heron)	No effect on individuals expected	Possible effect on some individuals expected, but low probability of effects on population	Adverse effects on individuals could occur; effect on the local population possible, that may have an impact on the sustainability of the local population	Significant probability of impacts on the sustainability of the local population
Invertivorous Avian Species (spotted sandpiper)	No effect on individuals expected	Possible effect on some individuals expected, but low probability of effects on population	Adverse effects on individuals could occur; effect on the local population possible, that may have an impact on the sustainability of the local population	Significant probability of impacts on the sustainability of the local population

The overall risk potential for each ROC was determined by evaluating the HQ line of evidence (including evaluation of probability of exceeding an HQ of 1) as well as information from the literature, tissue residue data, and field survey data, as available. Field survey data generally are given more weight (that is, there is more confidence in the results) than other lines of evidence. The HQ line of evidence generally is the most conservative, and as mentioned earlier, is best used to rule out risks than to predict risks.

5.2 Exposure Model Outcomes

Table 5-8 to Table 5-10 present the predicted probability that HQ values for COPC are greater than 1.0 for the tern, heron, and sandpiper, respectively. Predicted probabilities provided in Tables 5-8 to 5-10 for Areas 1, 2 and 3 were derived using the full distribution of exposure point concentrations (e.g., full distribution of beach sand concentrations, fish, invertebrates).

In addition, Table 5-11 to Table 5-13 present the average and 95th percentile HQ values for the tern, heron, and sandpiper for Areas 1, 2 and 3. For Areas 1 to 3 (combined) an average HQ is presented which was calculated assuming each receptor spends 1/3 of their time in each of the three areas.

Finally, Table 5-14 to Table 5-16 present the COPC pathway contribution to the tern, heron, and sandpiper in Area 1, respectively.

For each of these outcome tables, bioaccessibility associated with beach sand ingestion for aluminum, arsenic, cadmium, copper, iron, lead, selenium, thallium and zinc was based on the site-specific bioaccessibility results from RMC (Appendix K). Bioaccessibility testing was not measured for lithium and strontium and as such, bioaccessibility of these metals via beach sand ingestion was assumed to be 100%. Bioaccessibility of metals in food items was assumed to be 100%, which would tend to overestimate exposures.

The assessment focused on presenting pathway contribution in Area 1 based on the following:

- Area 1 is the area of interest where the highest HQ values are predicted; and
- Pathway contributions for the remaining areas of interest (i.e., Reference, Area 2, and Area 3) were predicted to be similar to each other (see Table J-3 in Appendix J).

Table 5-8 Predicted Probability of HQ Values Greater than 1.0 for the Tern

COPC	Full Distribution of Exposure Point Concentrations			
	Reference	**Area 1**	**Area 2**	**Area 3**
Aluminum	0%	0%	0%	0%
Arsenic	0%	0%	0%	0%
Cadmium	0%	0%	0%	0%
Copper	0%	0%	0%	0%
Iron	0%	74%	73%	70%
Lead	0%	3%	2%	2%
Lithium	0%	0%	0%	0%
Selenium	3%	5%	5%	5%
Strontium	0%	0%	0%	0%
Thallium	0%	1%	0%	0%
Zinc	0%	0%	0%	0%

Note: probabilities of exceeding an HQ of 1> 10% are shaded red; UCLM: 95th upper confidence limit of the mean

Table 5-9 Predicted Probability of HQ Values Greater than 1.0 for the Heron

COPC	Full Distribution of Exposure Point Concentrations			
	Reference	**Area 1**	**Area 2**	**Area 3**
Aluminum	0%	0%	1%	0%
Arsenic	0%	0%	0%	0%
Cadmium	0%	0%	0%	0%
Copper	0%	7%	0%	0%
Iron	41%	90%	89%	82%
Lead	0%	28%	6%	3%
Lithium	0%	0%	0%	0%
Selenium	0%	3%	1%	1%
Strontium	1%	15%	8%	6%
Thallium	0%	16%	0%	0%
Zinc	0%	0%	0%	0%

Note: probabilities of exceeding an HQ of 1> 10% are shaded red; UCLM: 95th upper confidence limit of the mean

Table 5-10	Predicted Probability of HQ Values Greater than 1.0 for the Sandpiper			
COPC	Full Distribution of Exposure Point Concentrations			
	Reference	Area 1	Area 2	Area 3
Aluminum	61%	52%	57%	28%
Arsenic	0%	1%	0%	0%
Cadmium	0%	23%	1%	0%
Copper	6%	87%	61%	55%
Iron	100%	100%	93%	97%
Lead	0%	99%	56%	36%
Lithium	26%	42%	40%	41%
Selenium	32%	85%	64%	61%
Strontium	96%	94%	89%	88%
Thallium	0%	91%	19%	0%
Zinc	0%	83%	5%	3%

Note: probabilities of exceeding an HQ of 1> 10% are shaded red; UCLM: 95th upper confidence limit of the mean

Table 5-11	Predicted Average and 95th Percentile HQ Values for the Tern				
COPC	Full Distribution of Exposure Point Concentrations				UCLM
	Reference	Area 1	Area 2	Area 3	Areas 1-3
Aluminum	0.02(0.06)	0.12(0.25)	0.12(0.26)	0.11(0.24)	0.12(0.22)
Arsenic	0.02(0.03)	0.03(0.05)	0.03(0.05)	0.03(0.05)	0.03(0.05)
Cadmium	0.01(0.02)	0.03(0.05)	0.02(0.04)	0.02(0.03)	0.02(0.04)
Copper	0.03(0.06)	0.11(0.22)	0.08(0.15)	0.08(0.16)	0.09(0.17)
Iron	0.30(0.55)	1.7(3.6)	1.7(3.7)	1.7(3.6)	1.7(3.2)
Lead	0.00(0.01)	0.40(0.86)	0.37(0.81)	0.35(0.79)	0.38(0.71)
Lithium	0.01(0.03)	0.03(0.05)	0.03(0.05)	0.02(0.05)	0.03(0.04)
Selenium	0.53(0.90)	0.59(1.0)	0.59(0.99)	0.59(0.99)	0.59(0.99)
Strontium	0.08(0.14)	0.12(0.25)	0.10(0.22)	0.10(0.20)	0.11(0.21)
Thallium	0.01(0.01)	0.45(0.77)	0.40(0.69)	0.40(0.68)	0.42(0.71)
Zinc	0.14(0.24)	0.32(0.63)	0.31(0.61)	0.31(0.61)	0.31(0.55)

Note: HQs >1 are shaded

Table 5-12	Predicted Average and 95th Percentile HQ Values for the Heron				
COPC	Full Distribution of Exposure Point Concentrations				UCLM
	Reference	Area 1	Area 2	Area 3	Areas 1-3
Aluminum	0.19(0.41)	0.23(0.44)	0.28(0.65)	0.19(0.38)	0.23(0.45)
Arsenic	0.02(0.04)	0.05(0.09)	0.04(0.08)	0.04(0.08)	0.04(0.08)
Cadmium	0.01(0.02)	0.10(0.24)	0.04(0.09)	0.02(0.04)	0.05(0.11)
Copper	0.08(0.15)	0.48(1.1)	0.23(0.50)	0.23(0.54)	0.31(0.64)
Iron	1.0(2.0)	2.3(4.6)	2.5(5.3)	2.0(4.2)	2.3(4.2)
Lead	0.01(0.03)	0.84(1.7)	0.50(1.0)	0.41(0.86)	0.59(1.1)
Lithium	0.04(0.07)	0.05(0.09)	0.05(0.11)	0.05(0.09)	0.05(0.09)
Selenium	0.33(0.56)	0.49(0.91)	0.42(0.78)	0.42(0.78)	0.45(0.80)
Strontium	0.34(0.65)	0.61(1.4)	0.49(1.2)	0.45(1.1)	0.52(1.1)
Thallium	0.01(0.02)	0.66(1.4)	0.27(0.49)	0.22(0.37)	0.38(0.72)
Zinc	0.08(0.14)	0.34(0.66)	0.28(0.57)	0.27(0.54)	0.30(0.54)

Note: HQs >1 are shaded red

Table 5-13	Predicted Average and 95th Percentile HQ Values for the Sandpiper				
COPC	Full Distribution of Exposure Point Concentrations				UCLM
	Reference	Area 1	Area 2	Area 3	Areas 1-3
Aluminum	1.4(3.1)	1.2(2.4)	1.5(4.0)	0.83(1.8)	1.2(2.5)
Arsenic	0.09(0.17)	0.32(0.71)	0.17(0.39)	0.17(0.41)	0.22(0.46)
Cadmium	0.03(0.07)	0.73(1.8)	0.23(0.59)	0.10(0.22)	0.35(0.80)
Copper	0.49(1.05)	3.4(8.3)	1.4(3.4)	1.4(3.6)	2.1(4.4)
Iron	6.9(14)	10(21)	8.6(23)	4.7(10)	7.9(17)
Lead	0.09(0.19)	5.8(13)	1.4(3.7)	0.96(2.1)	2.7(5.8)
Lithium	0.82(1.7)	1.0(2.1)	1.0(2.1)	1.0(2.1)	1.0(2.1)
Selenium	0.90(1.8)	2.3(5.0)	1.5(3.6)	1.5(3.6)	1.8(3.6)
Strontium	2.5(4.7)	4.4(10)	3.5(8.4)	3.2(7.7)	3.7(7.8)
Thallium	0.07(0.14)	4.0(9.7)	0.66(1.6)	0.24(0.55)	1.6(3.6)
Zinc	0.17(0.35)	2.2(5.0)	0.42(1.0)	0.39(0.91)	0.99(2.1)

Note: HQs >1 are shaded

Based on the outcomes presented in Table 5-8 to 5-13, the following can be stated:

- The probability of HQs exceeding 1 is greatest for the sandpiper, followed by the heron, and tern;
- For tern in Areas 1, 2 and 3 only iron had an HQ>1 (Table 5-11). The probability of the iron HQ exceeding 1.0 in Areas 1, 2 and 3 were 74%, 73% and 70%, respectively (Table 5-7).
- For heron, HQs based on average concentrations were only greater than 1.0 for iron (Table 5-12). The probability of the iron exceeding an HQ of 1.0 in Area 1 was greatest for iron at 90% (Table 5-9). The only other COPCs in Area 1 which were predicted to have a >10% probability of the HQ exceeding 1.0 were: lead (28%), strontium (15%) and thallium (16%) (Table 5-9). In Areas 2 and 3, only iron was predicted to have a >10% probability of the HQ exceeding 1.0 (at 89% and 82%, respectively).
- For sandpiper, several COPCs had an HQ>1 based on average and / or 95th percentile concentrations (See Table 5-13). Iron, lead, strontium, thallium and copper have the highest probability of exceeding an HQ of 1 in Area 1 (Table 5-10).
- The number of COPCs exceeding an HQ of 1.0 were the greatest in Area 1, with Areas 2 and 3 showing a lower number of COPCs having elevated HQs (for the heron and sandpiper; See Tables 5-12 and 5-13).
- Average HQs are exceeded in reference for the sandpiper (aluminum; iron; strontium; Table 5-13), which speaks to the conservative nature of these TRVs;
- The TRVs for iron, lithium and strontium are based on a maximum tolerable level and the aluminum TRV is based on a NOEL (no observable effect level), neither of which are adverse effect levels. Above this concentration, some effects may be noted, but they would not be expected to affect reproduction or survival until substantially higher doses are received. For those substances based on MTLs, growth could be reduced, if intake levels were high enough to reduce food intake (NRC, 2005). In addition, other elements present in food, such as calcium, can

modify the uptake and toxicity of substances such as iron (NRC, 2005). None of these substances would be expected to be predominant contributors to overall risks for the species in any of the affected areas.

Tables 5-14 to 5-16 present the relative contribution of each dietary item, and beach sand, to overall exposure for Area 1. For these tables, the highest percent bioaccessibility from the RMC report (Appendix K) was used. For the tern, sand ingestion rates are low (0.5%), and hence, Atlantic herring, sand lance and aquatic invertebrates generally dominate, in terms of exposure contribution (Table 5-14). For heron, aquatic invertebrates are generally the highest overall contributors to exposure (Table 5-15). Sand ingestion was an important contributor to exposure for zinc and lithium, relative to other metals (Table 5-15). Similarly, for the sandpiper (Table 5-16), aquatic invertebrates are contributing the greatest to total exposures. Sand ingestion is an important contributor to exposure for some metals, with zinc (66%), lithium (70%) and iron (47%) having the highest exposure contributions from sand. In addition, exposures related to sand lance include contributions from sand within their guts, which increased tissue concentrations in this species, relative to herring. The bioaccessibility of the sand in the sand lance would be lower than that assumed in the assessment (i.e., metals in dietary items were assumed to be 100% bioaccessible). In general, in Area 2 and 3, diet (e.g., shoreline invertebrates) is a more dominant exposure pathway than sand (see Appendix J).

Table 5-14 Predicted Average Pathway Contribution for the Tern in Area 1

COPC	Percentage of COPC Contribution to Each Pathway			
	Sand	Atlantic Herring	Sand Lance	Aquatic Invertebrate
Aluminum	0.3%	18.4%	48.9%	32.5%
Arsenic	2.1%	37.0%	38.2%	22.7%
Cadmium	0.5%	29.4%	15.5%	54.6%
Copper	0.7%	14.6%	23.1%	61.6%
Iron	2.7%	15.1%	69.5%	12.7%
Lead	4.5%	8.6%	58.9%	28.0%
Lithium	15.3%	34.4%	22.9%	27.5%
Selenium	0.5%	64.2%	26.0%	9.3%
Strontium	0.1%	19.5%	8.2%	72.2%
Thallium	0.1%	52.5%	26.8%	20.7%
Zinc	3.7%	29.8%	58.6%	7.9%

Table 5-15 Predicted Average Pathway Contribution for the Heron in Area 1

COPC	**Percentage of COPC Contribution to Each Pathway**			
	Sand	**Atlantic Herring**	**Sand Lance**	**Aquatic Invertebrate**
Aluminum	0.6%	2.1%	21.4%	75.9%
Arsenic	4.9%	5.5%	21.5%	68.1%
Cadmium	0.7%	2.5%	4.9%	91.9%
Copper	0.8%	1.1%	6.4%	91.7%
Iron	7.2%	2.6%	45.6%	44.5%
Lead	8.1%	1.0%	25.7%	65.2%
Lithium	26.0%	3.8%	9.5%	60.7%
Selenium	2.1%	18.1%	27.5%	52.3%
Strontium	0.1%	1.3%	2.1%	96.6%
Thallium	0.1%	9.2%	17.7%	72.9%
Zinc	12.2%	6.4%	47.2%	34.2%

Table 5-16	Predicted Average Pathway Contribution for the Sandpiper in Area 1			
COPC	**Percentage of COPC Contribution to Each Pathway**			
	Sand	**Atlantic Herring**	**Sand Lance**	**Aquatic Invertebrate**
Aluminum	3.8%	0.0%	0.0%	96.2%
Arsenic	28.0%	0.0%	0.0%	72.0%
Cadmium	3.8%	0.0%	0.0%	96.2%
Copper	4.5%	0.0%	0.0%	95.5%
Iron	46.7%	0.0%	0.0%	53.3%
Lead	40.1%	0.0%	0.0%	59.9%
Lithium	69.7%	0.0%	0.0%	30.3%
Selenium	17.7%	0.0%	0.0%	82.3%
Strontium	0.4%	0.0%	0.0%	99.6%
Thallium	1.0%	0.0%	0.0%	99.0%
Zinc	65.7%	0.0%	0.0%	34.3%

Note: the sandpiper was assumed to not consume Atlantic herring and sand lance, hence % contributions are zero

5.3 Literature Review

Relevant studies on the black crowned night heron and spotted sandpiper were not identified in the literature reviewed. Several studies on common tern in addition to other waterfowl species were identified and relevant studies are discussed below.

There have been a large number of studies of common terns nesting in contaminated areas, many of which have involved colonies in the US eastern seaboard. Custer et al. (1986) found that clutch size, reproductive success and growth of young from a colony nesting on a barge in a contaminated river in Rhode Island were either equal to or greater than those measured at less contaminated sites. The predominant sediment contaminants in the affected area included lead and other metals, such as silver, chromium and copper. Custer et al (1986) cite liver metal concentrations for 13 to 16 day old common tern chicks (Table 5-17), which are compared to data from the current study. These studies differ temporally by almost 3 decades, and could have used differing laboratory analysis techniques, and as such, caution should be used when comparing these datasets. While average copper concentrations in chick liver from the Brunswick Smelter study area appear to be lower than average levels reported in Providence Barge, mean iron and zinc levels are higher in the current study area. Comparisons of fish tissue data from Custer et al. (1986; limited dataset) to those measured in the current study suggest similar concentrations in Atlantic herring, but sand lance have higher concentrations (likely due to the presence of beach sand within their guts). The clutch size in Providence barge was reported as 2.22, whereas the clutch size in the Brunswick Smelter study area, based on CWS (2010), was 2.26 (see Appendix A, Section 3.2.5). Therefore, in summary, the concentrations measured in fish and tern chick livers may not be causing a decrease in tern clutch size, which, when compared to the results of the Custer et al. study, may suggest no impacts to tern reproductive success and hence to populations.

Table 5-17	Comparison of Study Area Common Tern Liver Metal Concentrations and Fish Tissue Concentrations to Those Measured in a Common Tern Colony in Providence RI				
Analyte	**Study Area Chick Liver (mean; mg/kg ww) N= 13**	**13-16 d old chick livers (mean; mg/kg ww[b]) Providence Barge[a] N = 14**	**Study Area Fish Tissue Concentrations (mean; mg/kg dw)[c]**		**Fish Tissue Concentrations [a] (mg/kg; dw) Rhode Island (N = 2)**
			Atlantic Herring (N=10)	**Sand Lance (N=6)**	
Copper	9.22	12.7	4.4	18.7	15.7
Iron	143	131.7	148	1823	153
Zinc	32.6	16.6	122	641.6	159
Lead	No data	No data	6.3	115	10.1

Notes: ww = wet weight; dw = dry weight; N = number

a. Source: Custer et al., 1986

b. Data were converted from dw, using the mean % moisture data of 78.5% cited by authors, and the following equation (ww = dw(1-% moisture)

c. Mean concentrations, converted to dw using site-specific % moistures outlined in Section 3

Other studies related to common tern and metal exposures have examined potential behavioural effects related to lead exposures. Burger and Gochfield (1985; 1988) ran a series of studies wherein lead nitrite was injected into common tern chicks, and behaviors such as locomotion, balance, righting response, feeding tasks (such as begging), and depth perception were examined. Lead-injected chicks performed poorer than control chicks for most behavioral indicators (except begging and movement on a stationary incline). The authors also tested chick feeding response, when presented with a novel fish positioning (reverse of fish position). Lead injected chicks took a significantly longer time to eat the same number of fish when challenged in this fashion (Burger and Gochfield, 1985). Injected doses in this study (0.2 mg/g, which is 16% of the LD50 for this species, or 1.2 mg/g) are difficult to compare to measured tissue data collected in the current study. Burger and Gochfield (1988) examined the effects of multiple doses of lead, injected into chicks, and determined that the initial dose of lead affected behavior more so than a second dose of lead. Later studies by Burger and Gochfield (2000) involved both common tern chicks and herring gull chicks, and included lead-injected chicks in the laboratory setting, as well as wild. This study found a dose-related increase in effects and that day of exposure (injection) affected outcomes. In general, low lead exposure affected growth, locomotion, balance, food begging capabilities, feeding, thermoregulation, depth perception and the ability of the chicks to recognize certain individuals. The authors found that lead-injected chicks placed in the wild had similar behavioral deficits, but recovered more completely than laboratory-raised chicks. This was attributed to wild parents providing additional care and food, such that the lead-exposed chicks actually fledged with similar weights to control chicks. Doses given are considered by the authors to be environmentally relevant, but since they are injected doses, it is not possible to compare to the exposure levels in the current common tern chicks and eggs.

Burger and Gochfeld have conducted a number of monitoring studies on common tern in New York state and New Jersey (e.g., Burger and Gochfeld, 1991a; 2003; Burger, 2002). These studies provide egg metals concentrations, but are not presented in detail here as effects outcomes related to the measured metal concentrations are not discussed in these papers.

With respect to available literature from other metals contaminated sites, the Coeur d'Alene basin in Idaho is an area which was heavily affected by mining and metallurgical releases associated with the Bunker Hill Complex. This area is a US Environmental Protection Agency Superfund site, and has been extensively studied for decades. It differs dramatically from the

Brunswick Smelter study area, in that it involved contamination of a vast area of land and watercourses, due to decades of historical activities with limited containment of wastes. Nonetheless, it is of merit to draw on some of the work completed in the Bunker Hill area, due to the extensive ecological risk assessment work completed in this area, relative to waterfowl exposures and risks. In this area, waterfowl such as tundra swan and Canada geese are the predominant species present, and the predominant pathway of concern was ingestion of contaminated sediments. The US EPA (2002a) Record of Decision (ROD) cites a Preliminary Remedial Goal (PRG) for lead of 530 mg/kg in sediments, as a threshold for subclinical effects. This concentration could result in measureable physiological responses, but these responses are not considered sufficient enough to result in severe biological impairment (Franson and Pain, 2011). A mortality threshold of 1,800 mg/kg of lead in sediments is cited in the ROD (U.S. EPA, 2002a). Lead was considered to be the critical metal of interest driving risks, via the sediment ingestion pathway. The Bunker Hill Complex is vast in comparison to the Brunswick Smelter study area, with over 18,300 acres of floodplain sediments with lead concentrations > 530 mg/kg, which represented over 95% of available habitat for waterfowl. Over 80% of available habitat contained surficial sediments at concentrations > 1,800 mg/kg lead, and avian mortalities were frequently occurring. Bioaccessibility of lead, and other metals/metalloids at this site could be substantially different from that present in the current study area. The ROD cites concentrations of various metals considered to be protective of aquatic birds and mammals using the area, based on the site-specific ecological risk assessment. These sediment concentrations are presented in Table 5-18, and are compared to beach sand concentrations present within Areas 1 to 3 within the existing study area. Direct comparisons of the sediment concentrations considered to be protective in Coeur d'Alene to the current area have considerable uncertainties, as the receptors of interest in each study are different (e.g., Tundra swan versus black-crowned night heron and spotted sandpiper), and potential differences in bioaccessibility.

Table 5-18 Concentrations of Metals in Sediments at the Bunker Hill Mining Area (Coeur d'Alene Basin, Idaho) Considered to be Protective for Aquatic Birds and Mammals Compared to Beach Sand Concentrations Associated with Brunswick Smelting[a]

	Bunker Hill Protective Concentrations (mg/kg)		Brunswick Smelter Beach Sand Concentration (mg/kg)		
Analytes	**Population LOAEL**	**Population/ED20**	**Area 1 (mean; 95th%ile)**	**Area 2 (mean; 95th%ile)**	**Area 3 (mean; 95th%ile)**
Arsenic	222	138	220; 378	16; 23	20; 28
Cadmium	173	664	12; 20	0.3; 0.4	0.6; 1.2
Copper	2157	2209	611; 977	15; 18	21; 28
Lead	249[b]	718[b]	7,500; 16,519	56; 83	81; 149
Mercury	2.5	7	0.02; 0.03	<0.01; 0.01	0.01; 0.02
Zinc	519	390	21,096; 36,370	116; 205	219; 474

Notes:
LOAEL = lowest observable adverse effect level; ED = effect dose; %ILE = percentile
a Shading of cells is based on mean concentration and indicates an exceedance over Bunker Hill protective concentrations.
b For lead, a PRG (preliminary remedial goal) of 530 mg/kg in sediments was selected for waterfowl, based on a LOAEL of 530 mg/kg; mortality was predicted to occur at concentrations > 1,800 mg/kg

Other studies from Bunker Hill Superfund area (Sample et al, 2011; Hansen et al, 2011), and the Southeast Missouri Lead Mining District (Beyer et al, 2013), as well as lead avian risks associated with small arms ranges (e.g., Johnson et al, 2007) were reviewed, and have focused on lead concentrations in soils and associated food items (such as soil invertebrates), and potential risks to songbirds, as opposed to shorebirds or waterfowl. No other appropriate shorebird or waterfowl studies were identified in the literature reviewed. In Johnson et al. (2007), blood lead levels were used as the effects metric (NOAEL-based TRV of 29.5 µg/dL), which makes comparisons to the current study difficult. Beyer et al. (2013) collected lead, zinc, copper and cadmium levels in blood, liver and kidney tissues of songbirds (soil insectivores), and used similar effects metrics for kidney and liver as those presented in Section 5.4, for common tern tissues collected in the current study. Sample et al. (2011) focused on lead exposures, based on site-specific blood, liver and ingesta lead concentration data in 3 ground-feeding songbird species (American robin, song sparrow, and Swainson's thrush). These authors also used similar effects metrics to those presented in Section 5.4 (Franson and Pain, 2011) for common tern tissues, and developed a site-specific PRG of 490 mg/kg lead for subclinical effects, and up to 7,200 mg/kg for severe clinical effects. Again, the Coeur d'Alene area involves a study area of over 18,000 acres, and hence, the probability of elevated lead concentrations eliciting a population-level effect in species is high in that area.

5.4 Common Tern Chick and Egg Data Assessment

Common tern chick and egg data were collected by Glencore smelter staff in the summer of 2014. The number of chicks and eggs collected during the program are presented in Table 5-19.

Table 5-19 Summary of Location of Samples of Common Tern Eggs and Chick Tissues

Tissue Collected	Location	Number of Samples
Egg	Main office lawn	1
Egg	Laboratory Roof	11
Egg	Changehouse Roof	7
Egg	East of polishing pond	1
Chick	Between Lab and Changehouse	1
Chick	Parking Lot	4
Chick	North Parking lot	5
Chick	Laboratory Roof	4
Chick	Ground beside Changehouse	1
Chick	Changehouse roof	4
Chick	Between Lab and Changehouse	3
Chick	West of Changehouse	1
Advanced chick	Beach	1

A selection of these samples was sent to RPC Laboratories for analysis of metal concentrations in chick eggs, liver and kidney. Due to the small size of several organs, compositing of some kidneys and livers was required to obtain a large enough tissue mass for analysis purposes. Analytical results for these samples were previously presented in Section 3.0. In addition to samples collected in 2014, livers from additional chick samples collected from the smelter site by smelter staff in July, 2010 were analyzed by CWS (2014). Kidney and egg data were not available from the 2010 collections. The CWS (2014) report is provided in Appendix L and a summary of liver data from this report is discussed in conjunction with the data collected in 2014.

To determine if metal concentrations in common tern chick eggs, liver and / or kidneys were at levels that could be of biological concern, tissue effect concentrations for these media were compiled and compared to site data. Common tern effects data were compiled where possible, but where no common tern effects data were available, effects data from other bird species were used.

Of note, 24 dead chicks were collected from the colony during the nesting season (Table 5-19). This was based on daily surveillance of the colony. This suggests a reasonably low mortality rate within the colony, albeit a clutch count was not undertaken in 2014. Based on previous observations by Glencore staff (which are anecdotal), mortalities did not appear more prevalent this year than other years. In common terns, egg and chick mortalities are not uncommon. In a two year study of common terns on Great Gull Island, 11% to 43% of eggs laid did not hatch, with chick survival being 13% to 36% per egg laid (LeCroy and Collins, 1972). Chick starvation, exposure to cold wet weather, predation, flooding, and human disturbance have been reported as major causes of common tern reproductive failure (Burger and Gochfeld, 1991b).

Avian tissue effect data were identified in the literature reviewed are summarized in Table 5-20.

Table 5-20 Tissue Effect Concentration Data Identified for Bird Eggs, Livers and Kidneys

Metal	Bird Eggs	Bird Livers	Bird Kidneys
Arsenic	√	NDA	NDA
Cadmium	NDA	√	√
Copper	√	√	NDA
Lead	NDA	√	√
Mercury	√	√	√
Selenium	√	√	√
Zinc	√	√	√

Notes:
√ = tissue effect data were available
NDA = not data available

The tissue effect thresholds identified in the literature ranged from background concentrations, to levels associated with reproductive effects, or teratogenicity. In addition, the terms "subclinical poisoning, clinical poisoning, and severe clinical poisoning" are also used in the assessment tables below, and these terms are defined by Franson and Pain (2011) as follows (specifics are related to lead intoxication);

- Subclinical Effects/Poisoning: concentrations which have been reported to cause physiological effects, but are considered insufficient to severely impair normal biological functioning. There are no external symptoms, and it is surmised that the bird would probably recover if lead exposure was removed;
- Clinical Poisoning: this threshold represents the onset of clinical symptoms of poisoning, such as anemia, microscopic lesions in tissue, weight loss, muscular incoordination, green diarrhea. Can result in mortality;
- Severe Clinical Poisoning: Threshold wherein effects may be life threatening.

A comparison of egg metal concentrations from the study area to tissue effect concentrations is provided in Table 5-21. Egg contaminant loadings in common terns are generally considered to represent more local exposures, rather than over-wintering exposures, since most birds spend several weeks in the nesting area before they lay eggs, and hence, are foraging and acquiring exposures from the nesting area (Burger, 2002). Based on the toxicity endpoints identified in the literature reviewed, common tern chick egg concentrations are less than those reported to be associated with adverse effects (Table 5-21). The maximum measured zinc concentration in eggs from within the study area was greater than the reported no effect concentration of 15 mg/kg, while the mean concentration (14.3 mg/kg) was below this value. Zinc concentrations in 6 of the 18 chick eggs from within the study area had concentrations greater than the no effect threshold of 15 mg/kg (ranging from 15.8 mg/kg to 20.6 mg/kg); however no adverse tissue effects thresholds were identified in the literature reviewed to determine whether / if these values would actually be associated with an effect (and if so, what type). Of the 6 eggs containing the elevated zinc concentrations, two were comprised of an orange liquid (early stage of development), one was a ½ formed chick and three were formed chicks (later stage of development). There was no clear gradient of zinc concentrations as one went from an early stage of development to the fully formed chick.

Given egg concentrations from the study area are below reported tissue effect concentrations and zinc concentrations in 1/3 of eggs exceeded a no effect level, concentrations of metals in common tern eggs are not considered to indicate a potential for development of adverse effects. There are uncertainties in these comparisons, in that the egg tissue thresholds are not specific to common tern and due to small sample sizes.

Tissue effects thresholds for egg data were not identified for either cadmium or lead, both of which are associated with the study area. Comparisons of the cadmium and lead concentrations in eggs in the study area to those reported in other areas is difficult, as recent data on tissue residues in common tern eggs could not be found in the literature reviewed. Studies conducted by Burger and Gochfeld (1991a) reported mean egg cadmium concentrations of 0.004 mg /kg ww (N = 24) in the New York Bight area, compared to a mean of 0.005 mg/kg ww (N = 18) in the current study (see Section 3). In Barnegat Bay, New Jersey, egg cadmium concentrations in common terns averaged 0.0013 mg/kg ww (N = 35; Burger, 2002). Egg lead concentrations in New York Bight were reported to average 0.089 mg /kg ww (N = 24; Burger and Gochfeld, 1991a), compared to those measured in Barnegat Bay, NJ, (mean of 0.055 mg/kg ww; Burger, 2002) and the current study (mean of 0.35 mg /kg ww; N = 18; See Section 3.0). While comparing egg concentrations from different years can be associated with some uncertainty, it nevertheless does provide an indication that concentrations of lead, and, to a lesser extent, cadmium, appear elevated in common tern eggs in the study area relative to other sites.

Table 5-21 Background and Tissue Effect Concentrations Reported for Various Metals in Bird Eggs (mg/kg wet weight; N= 18)

Species	Study Area Concentration (mg/kg ww) Range (Mean)	Literature Tissue Concentration (mg/kg ww)	Reported Effect in Literature	Comment	Literature Reference [c]	Source
Arsenic						
Birds	<0.1 to 0.2 (Mean = 0.1)	0.4 [a] 0.4 – 0.85 [a] >0.85 [a,b]	No effect Level of concern Toxicity threshold		J.P. Skorupa, unpublished data, 1996	US Department of the Interior, 1998 (pg. 11)
Copper						
Birds	0.7 to 1.0 (Mean = 0.9)	1.7 [a]	No effect		J.P. Skorupa, unpublished data, 1996	US Department of the Interior, 1998 (pg. 43)
Mercury						
Common tern	0.05 to 0.11 (Mean = 0.09)	1	Egg Hg concentration associated with no adverse effect		Eisler, 2004; Fimreite, 1974	Beyer and Meador, 2011 (pg. 618)
Common tern		3.5	Egg Hg concentrations associated with adverse effects on reproduction	Beyer and Meador (2011) indicated this value was the midpoint of Fimreite, 1974; Eisler, 2004 values	Eisler, 2004; Fimreite, 1974	Beyer and Meador, 2011 (pg. 618)
Non-marine birds		1.9 (geomean); 0.8 – 5.1 (range) >0.6 (HC5)	Reproduction	Authors proposed an indicative value for reproductive effects in bird eggs of >0.6 mg/kg ww which is a predicted HC5 value		Beyer and Meador, 2011 (pg. 620)
Selenium						
Birds	0.5 to 0.9 (Mean = 0.7)	<0.91 (typically 0.45 to 0.76) [a]	Mean background concentration	Concentrations < 0.2 mg/kg ww [a] could indicate inadequate Se in diet, while maximum levels for individual eggs were reported to be <1.5 mg/kg ww[a]; Authors reported concentrations may be higher in marine birds		Beyer and Meador, 2011 (pg. 676, 677, 695)
Birds		<1.5 [a]	Background			US Department of the Interior, 1998 (pg. 141)
Birds		<2.4 [a] >3.6 [a]	Reproductive impairment (low to increased probability)	<2.4 mg/kg ww [a] (low probability for ↓egg hatchability in sensitive species; >3.6 mg/kg ww [a] (↑ probability for ↓ egg hatchability in sensitive / moderately sensitive species); these values are recommended assessment values for effects of Se in birds		Beyer and Meador, 2011 (pg. 695)

Table 5-21 Background and Tissue Effect Concentrations Reported for Various Metals in Bird Eggs (mg/kg wet weight; N= 18)

Species	Study Area Concentration (mg/kg ww) Range (Mean)	Literature Tissue Concentration (mg/kg ww)	Reported Effect in Literature	Comment	Literature Reference [c]	Source
Birds		<6.1 [a] >10.6 [a]	Teratogenicity	<6.1 mg/kg ww [a] (threshold for statistically discernable incidence in sensitive species; low probability for teratogenic effects in most species); >10.6 mg/kg ww [a] (probability of teratogenic effects in species of average sensitivity); these values are recommended assessment values for effects of Se in birds		Beyer and Meador, 2011 (pg. 695)
Shorebirds (freshwater)		0.36 – 2.79 (mean: 0.82)[a]	No effect	Se concentrations in shorebird eggs at the Great Salt Lake ecosystem. Breeding success 94-97%; considered consistent with non-exposed populations	Ohlendorf et al, 2009	Ohlendorf et al, 2009
Birds using Great Salt Lake Ecosystem		1.9 – 4.85[a]	Se standard in eggs associated with minimal potential for adverse effects		Ohlendorf et al, 2009	Ohlendorf et al, 2009
Marine birds		1.2[a]	Average egg concentration of Se in marine birds		Janz et al, 2010	Janz et al, 2010
Waterbirds		<0.91 [a] 0.91 to 1.8 [a] >1.8 [a]	No effect Level of concern Toxicity threshold		No effect level from Skorupa and Ohlendorf, 1991; Toxicity threshold from Skorupa, 1998	US Department of the Interior, 1998 (pg. 142)
Zinc						
Birds	9.4 to 20.6 (Mean = 14.3)	15 [a]	No effect		J.P. Skorupa, unpublished data, 1996	US Department of the Interior, 1998

Notes:
Study area N = 18 eggs
dw = dry weight; ww = wet weight
a. Dry weight concentrations were converted to wet weight concentrations by using a conversion factor of 3.3 (i.e., dry weight ÷ 3.3 = wet weight) provided by Beyer and Meador, 2011 (pg. 695) assuming 70% moisture.
b. This value was reported by the US Department of the Interior (1998; pg. 11) as <2.8 mg/kg dw, but assumed it should be >2.8 mg/kg dw. Value converted to wet weight concentration of 0.85 mg/kg ww using conversion factor of 3.3.
c. Only the full reference for the source has been provided in the reference list.

A comparison of study area chick metal concentrations in kidneys to tissue effect concentrations is provided in Table 5-22. Based on the toxicity endpoints identified in the literature reviewed, common tern chick kidney concentrations are much lower than those reported to be associated with adverse effects, with the exception of lead (Table 5-22).

The maximum measured lead concentration in kidneys from within the study area (28.3 mg/kg ww) was greater than the severe clinical poisoning level of >15 mg/kg ww. Of the six study area kidney samples analyzed for lead, two had concentrations greater than the severe clinical poisoning level of 15 mg/kg ww (at 26.9 and 28.3 mg/kg ww), two had concentrations within the clinical poisoning level of 6 to 15 mg/kg ww (at 7.09 and 8.31 mg/kg) and two were within the sub-clinical poisoning level of 2 to <6 mg/kg ww (at 3.63 and 5.25 mg/kg ww). Based on these lead concentrations in chick kidneys from within the study area, kidney lead levels are elevated in some chicks, relative to available effect levels from the literature.

Table 5-22 Background and Tissue Effect Concentrations Reported for Various Metals in Bird Kidneys (mg/kg wet weight)

Species	Study Area Concentration (mg/kg ww) Range (Mean)	Literature Tissue Concentration (mg/kg ww)	Reported Effect in Literature	Comment	Literature Reference [b]	Source
Cadmium						
Marine-pelagic birds	0.213 to 2.04 (Mean = 0.866)	31.3 median; range: 1.2 – 137.0	Background concentration	Concentration reported as background cadmium concentration in pelagic marine birds	No specific reference provided	Beyer and Meador, 2011 (pg. 650)
Marine-coastal birds		12.7 median; range: 0.2 – 79.8	Background concentration	Concentration reported as background cadmium concentration in coastal marine birds	No specific reference provided	Beyer and Meador, 2011 (pg. 650)
Birds		65 - 100	Concentration associated with a 50% probability (95% CI: 22-79%) of alterations in energy metabolism or structural / functional damage to tissues	Concentration of 65 µg/g ww was suggested to be a conservative risk level for Cd-induced damage in adult birds, while a concentration of 100 µg/g ww is considered a level above which adverse effects are likely to occur	No specific reference provided	Beyer and Meador, 2011 (pg. 660)
Lead						
Anseriformes (ducks, geese and swans)	3.6 to 28.3 (Mean = 13.3)	<2	Background concentration of lead		No specific reference provided	Beyer and Meador, 2011 (pg. 583)
		2 to <6	Sub-clinical poisoning	Suggested concentration for interpretation of tissue concentrations of lead from Beyer and Meador (2011)	Dieter and Finley, 1979; Degernes, 1991	Beyer and Meador, 2011 (pg. 571; 583)
		6 to 15	Clinical poisoning		Longcore et al, 1974, Degernes, 1991, Beyer et al, 2000	Beyer and Meador, 2011 (pg. 571; 583)
		>15	Severe clinical poisoning		Cook and Trainer, 1996; Longcore et al, 1994; Mautino and Bell, 1986; Beyer et al, 1988, 2000; Pain and Rattner, 1988; Pain, 1989; Blus et al, 1991, 1999; Degernes, 1991; Kelly et al, 1998; Nakade et al, 2005; Degernes et al, 2006	Beyer and Meador, 2011 (pg. 571; 583)

Table 5-22	Background and Tissue Effect Concentrations Reported for Various Metals in Bird Kidneys (mg/kg wet weight)					
Species	**Study Area Concentration (mg/kg ww) Range (Mean)**	**Literature Tissue Concentration (mg/kg ww)**	**Reported Effect in Literature**	**Comment**	**Literature Reference [b]**	**Source**
Mercury						
Non-marine birds	0.02 to 0.09 (Mean = 0.05)	76 (geomean); 21 - 130 (range)	Death	Authors proposed an indicative value of >40 mg/kg ww which is a predicted HC5 value		Beyer and Meador, 2011 (pg. 620)
Selenium						
Birds	0.8 to 3.2 (Mean = 1.7)	No clearly defined / poorly understood	Background / effect concentrations of selenium in bird kidneys	Authors reported that background concentrations of selenium in bird kidneys have not been clearly defined. Similarly, the concentration of selenium in adult bird kidneys associated with harm to health or reproductive success is poorly understood.		Beyer and Meador, 2011 (pg. 686; 694)
Zinc						
Birds	17 to 38 (Mean = 25)	<64 [a] >636 [a]	No effect Toxicity threshold		J.P. Skorupa, unpublished data, 1996	US Department of the Interior, 1998 (pg. 186)

Notes:
Study area: N = 6 kidneys
ww = wet weight; dw = dry weight
a. A conversion factors was not provided for kidneys by Beyer and Meador (2011); a conversion factor of 3.3 was provided for eggs and liver (based on a moisture content of 70%), this conversion factor was applied to the kidney data in this table that was given in dry weight to convert to wet weight (ww = dw ÷3.3)
b. Only the full reference for the source has been provided in the reference list.

A comparison of study area chick metal concentrations in liver from the study area from 2014 in addition to data collected on-site in 2010 (See CWS Report, Appendix L) to tissue effect concentrations is provided in Table 5-23. The 2014 and 2010 data have very similar concentration ranges, which suggest similar exposure levels between the two sampling intervals. No tissue effects data were identified specifically for the common tern and as such, data from other species / groups of species were used and hence, these comparisons are associated with some degree of uncertainty. Based on the toxicity endpoints identified in the literature reviewed, common tern chick livers were below toxicity thresholds for cadmium, mercury and selenium. While a few livers had copper concentrations within the level of concern, they were not greater than the toxicity threshold. Similarly, one sample had zinc in the liver above the no effect level, but it was below the toxicity threshold.

Lead concentrations in liver ranged from background levels to levels associated with severe clinical poisoning (>10 mg/kg ww) (See Table 5-23). Of the 13 samples collected in 2014, 4 samples were within background levels, 4 samples were within the level indicative of sub-clinical poisoning, 1 sample was within levels associated with clinical poisoning and 4 samples were at concentrations associated with severe clinical poisoning ranging from 11.9 mg/kg ww to 22.5 mg/kg ww (See Appendix C for individual liver sample results). Of the 7 liver samples collected in 2010, 2 were within background ranges, 3 were within the range of sub-clinical poisoning levels and 2 were within levels associated with severe clinical poisoning. Lead tissue effects were reported for aniseriformes (waterfowl such as ducks, geese and swans), as opposed to common tern specific effect levels, and hence, are associated with some uncertainty (See Table 5-23). In summary, between the two sampling events, 13 of 20 liver samples had lead levels within the background and subclinical range (65%); 1 of 20 samples had lead levels within the clinical effects range (5%), and 6 of 20 samples had levels within the severe effects range (30%). These data suggest elevated levels of exposure are occurring within some individuals within the colony.

While several of the dead chicks analyzed had lead liver concentrations within / above sub-clinical poisoning / severe poisoning tissue effect concentrations, Franson and Pain (2011) reported that examining a tissue concentration alone is not adequate to provide a definitive diagnosis of lead poisoning causing death in an individual bird. Rather, tissue concentrations accompanied by an examination of the bird carcass for gross and microscopic lesions of lead poisoning is required.

In summary, egg residues for the metals of interest were not considered to be indicative of elevated exposures, based on the samples taken and measured concentrations reported. No toxicity thresholds are available for either lead or cadmium, and based on comparisons to other areas, concentrations of these metals in eggs appear elevated. Lead levels in some kidneys and livers suggest that some individuals had elevated exposures relative to available toxicity residue effect levels. Other metals were within acceptable levels, where threshold-based toxicity values were available. Elevated exposures to lead in the areas where these birds are nesting are not surprising, in that they are present in the active industrial area of the facility, and could experience increased exposure levels if food was dropped on site, and subsequently ingested (e.g., fish dropped in areas where there is lead particulate, and subsequently picked up and fed to chicks). This is a plausible situation, and could result in elevated exposures in some individuals.

There are a number of uncertainties associated with this conclusion, including that many of the effect levels are not specific to common tern, but rather, are based on more general avian species.

Table 5-23	Background and Tissue Effect Concentrations Reported for Various Metals in Bird Livers (mg/kg wet weight)					
Species	**Study Area Concentration (mg/kg ww) Range (Mean)**	**Literature Tissue Concentration (mg/kg ww)**	**Reported Effect in Literature**	**Comment**	**Literature Reference [c]**	**Source**
Cadmium						
Marine-pelagic birds	2014 Data: 0.08 – 1.81 (Mean = 0.51) 2010 Data[b]: 0.012 – 1.3	8.2 median; range: 0.4 – 42.5	Background concentration	Concentration reported as background cadmium concentration in pelagic marine birds	No specific reference provided	Beyer and Meador, 2011 (pg. 650)
Marine-coastal birds		2.9 median; range: 0.1 – 17.1	Background concentration	Concentration reported as background cadmium concentration in coastal marine birds	No specific reference provided	Beyer and Meador, 2011 (pg. 650)
Birds		45 to 70	Threshold effect level for adult birds	Authors indicated that a threshold for effects in liver of adult birds may lie between 45 and 70 µg/g ww	No specific reference provided	Beyer and Meador, 2011 (pg. 660)
Copper						
Birds	2014 Data: 4.5 – 22.7 (Mean = 9.2) 2010 Data[b]: 3.7 – 19.6	<18 [a] 7.6 – 91[a] >164 [a]	No effect Level of concern Toxicity threshold		Data for ducks from Puls, 1988; toxic concentrations in waterfowl diets are >200 mg/kg dry weight.	US Department of the Interior, 1998 (pg. 43)
Lead						
Aniseriformes (ducks, geese and swans)	2014 Data: 0.77 – 22.5 (Mean = 7.48) 2010 Data[b]: 0.12 – 25.4	<2	Background concentration of lead		Beyer and Meador, 2011	Beyer and Meador, 2011 (pg. 583)
		2 to <6	Sub-clinical poisoning	Suggested concentration for interpretation of tissue concentrations of lead from Beyer and Meador (2011)	Dieter and Finley, 1979; Degernes, 1991	Beyer and Meador, 2011 (pg. 571; 583)
		6 to 10	Clinical poisoning		Longcore et al, 1974; Degernes, 1991; Beyer et al, 2000; Beyer et al, 1998	Beyer and Meador, 2011 (pg. 571; 583)
		>10	Severe clinical		Cook and Trainer, 1996;	Beyer and Meador,

Table 5-23	Background and Tissue Effect Concentrations Reported for Various Metals in Bird Livers (mg/kg wet weight)					
Species	**Study Area Concentration (mg/kg ww) Range (Mean)**	**Literature Tissue Concentration (mg/kg ww)**	**Reported Effect in Literature**	**Comment**	**Literature Reference [c]**	**Source**
			poisoning		Longcore et al, 1994; Mautino and Bell, 1986; Beyer et al, 1988, 2000; Pain and Rattner, 1988; Pain, 1989; Blus et al, 1991, 1999; Degernes, 1991; Kelly et al, 1998; Nakade et al, 2005; Degernes et al, 2006	2011 (pg. 571; 583)
Waterfowl		≥6	Lead poisoning in waterfowl causing death	Author indicated that this concentration supported the diagnosis of lead-poisoning as a cause of death if the necropsy and pathological data are consistent with lead poisoning	Beyer et al, 1998	Beyer and Meador, 2011 (pg. 572)
Waterfowl		11.5 [a]	Lead poisoning in waterfowl	Concentration is a defensible criterion for identifying waterfowl suffering from lead poisoning	Beyer et al, 1998	Beyer and Meador, 2011 (pg. 572)
Mercury						
Non-marine birds	2014 Data: 0.02 – 0.13 (Mean = 0.07)	8.7 (geometric mean); 2 – 52 (range)	Reproduction	Authors proposed an indicative value for reproductive effects of >2 mg/kg ww in liver	No specific reference provided	Beyer and Meador, 2011 (pg. 620)
Non-marine birds	2010 Data[b]: 0.038 – 0.34	63 (geometric mean); 18.4 – 127 (range)	Death	Authors proposed an indicative value of >20 mg/kg ww which is a predicted HC5 value		Beyer and Meador, 2011 (pg. 620)
Selenium						
Freshwater and terrestrial bird species	2014 Data: 0.60 – 1.80 (Mean = 1.04)	<3 [a]	Background	Low probability for adverse effects		Beyer and Meador, 2011 (pg. 695); US Department of the

Table 5-23 Background and Tissue Effect Concentrations Reported for Various Metals in Bird Livers (mg/kg wet weight)

Species	Study Area Concentration (mg/kg ww) Range (Mean)	Literature Tissue Concentration (mg/kg ww)	Reported Effect in Literature	Comment	Literature Reference [c]	Source
	2010 Data[b]: 0.91 – 1.9	3 – 6 [a]	Elevated and potentially toxic	Tissue concentration suspicious of selenium toxicosis when accompanied by symptoms for toxic effects; sensitivity varies by species		Interior, 1998 (pg. 141)
Some marine species		6 to 23 [a]	Background	Low probability of adverse effects in some marine species		Beyer and Meador, 2011 (pg. 695)
Birds		>6 [a]	Toxic	Concentration diagnostic of selenium toxicosis when accompanied by emaciation, poor quality of shed nails, bilaterally symmetrical alopecia of the head and neck, hepatic lesions, and necrosis of maxillary nails; based on field observations and laboratory studies with mallards		Beyer and Meador, 2011 (pg. 695)
Zinc						
Birds	2014 Data: 18.4 – 98.4 (Mean = 32.6)	<64 [a] >636 [a]	No effect Toxicity threshold		J.P. Skorupa, unpublished data, 1996	US Department of the Interior, 1998 (pg. 186)

Notes:
Study area N = 13 liver; CWS (2011) = 7 livers
dw = dry weight; ww = wet weight
a. Dry weight concentrations were converted to wet weight concentrations by using a conversion factor of 3.3 (i.e., dry weight ÷ 3.3 = wet weight) provided by Beyer and Meador, 2011 (pg. 695); authors reported moisture range of 65% to 80% and recommended the use of 70% moisture.
b. 2011 data from CWS (2011). CWS provided data in dry weight. Data converted to wet weight using following equation: wet weight = dry weight (1 – % moisture), based on sample=specific moisture contents.
c. Only the full reference for the source has been provided in the reference list.

5.5 Clutch Counts and Avian Observations

In 2010, CWS conducted a survey of common tern colonies within New Brunswick (Table 5-24; See Appendix A for further discussion). The majority of nests had a clutch size of 3 (46%), followed by 2 (34%), 1 (20%) and 4 (<1%). The average clutch size in Belledune was calculated to be 2.26 based on the data provided. Clutch sizes in Belledune were on the lower end, but similar to, other colonies assessed in New Brunswick in 2010.

Table 5-24 Nest Counts and Clutch Sizes of Common Tern Colonies Surveyed by Ground along the Gulf of St. Lawrence Coast of New Brunswick (excluding colonies within Kouchibouguac National Park; CWS, 2010)

Location and Assessment Date	Clutch Size					Nests	Eggs	Clutch Size	
	1	2	3	4	5			Mean	SD
Belledune Smelter[1] (June 16)	54	92	123	1	0	276	611	2.26	0.78
Shediac Marina (June 14)	27	127	221	9	2	386	990	2.56	0.68
Tern Island, Tabusintac (June 18)	383	1013	1187	48	2	2633	6172	2.34	0.75
Tracadie (June 17)	209	407	918	21	0	1555	3861	2.48	0.74
Unnamed Island #1 near Val Comeau (June 16)	26	27	40	1	0	94	204	2.17	0.85
Total	699	1666	2489	80	4	4944	11838	2.40	0.75

[1]Clutch size could not be determined for six nests that were located on a small unreachable island (Figure 3-11 in Appendix A). The mean clutch size and standard deviation was calculated using 270 nests, only.

Anecdotal comments from Brunswick smelter staff suggest the colony has increased in size since 2010, but this has not been confirmed through a nest and clutch size survey.

In 2010, an additional observational avian survey was conducted on Belledune Point. In this survey, 4 nesting pairs of spotted sandpiper were reported, as well as numerous other species. As a result of elevated HQs for the sandpiper, a confirmatory nesting survey was conducted in June 2015 on Belledune Point, and Area 2, to confirm the number of nesting shorebirds using the area, relative to 3 different reference areas (Little Belledune Point and Jacquet River mouth, which are northwest of the facility, and Tetagouche Marsh, which is near Bathurst, New Brunswick). This study was conducted by Minnow and is presented in Appendix M (Minnow, 2015c). A summary of the findings is presented in Table 5-25. Only two species were identified, the spotted sandpiper and the killdeer. The habitat on Belledune Point is attractive for both species, and of high quality (physically optimal nesting and foraging habitat). This study was not designed to examine hatching success, but the number of eggs laid at the time the survey was conducted (3 to 4 eggs/nest, with the exception of 1 nest with 1 egg), is in keeping with expectations and reference counts (detailed egg count data is provided in Appendix M). This suggests that clutch size is not being affected by the metals exposure. The data cannot be used to speak to hatching success, or fledgling success of the young. The results of this study confirms that the habitat is being used for nesting and foraging of both spotted sandpiper and killdeer, and suggests that clutch is similar between reference and these areas.

Table 5-25 Shorebird Nesting Survey on Belledune Point and Area 2, relative to Reference (June 2015; Minnow, Appendix M)				
Species	**Location**	**Nests**	**Eggs**	**Adults**
Spotted Sandpiper	Belledune Point	4 confirmed; 1 possible	11	12.5 (average of two days)
	Area 2	2 confirmed; 3 possible	8	6 (average of two days)
	Little Belledune Point	1	0	5
	Jacquet River Mouth	2	5	6
	Tetagouche Marsh	0	0	0
Killdeer	Belledune Point	1 possible	0	5 (average of two days)
	Area 2	2 confirmed	8	4.5 (average of two days)
	Little Belledune Point	0	0	0
	Jacquet River Mouth	1	3	2
	Tetagouche Marsh	0	0	0

5.6 Conclusions – Avian Species

5.6.1 Weight of Evidence: Common Tern

The common tern colony resides on the active industrial part of the smelter property. While the majority of foraging appears to be further offshore, fish captured near the final effluent discharge (Atlantic herring and sand lance), as well as shoreline invertebrates along the coast, were used to model potential exposures, which results in a conservative evaluation (as exposures would be expected to be lower offshore). Modelled HQs and probability of exceeding an HQ of 1, clutch counts for the colony in 2010, egg, chick kidney and liver tissue residues from 2010 (liver only) and 2014, comparisons of prey items to tissue residue guidelines considered to be protective of adverse health outcomes (selenium and mercury), and literature from other areas, were used to characterize risk potential for the common tern. The tissue residue data are from rejected eggs and dead chicks, and hence may not be representative of exposure levels within the colony on average, but rather, could represent an upper bound of exposure levels for selected individuals which incurred higher exposures. A summary of the various lines of evidence for common tern is presented in Table 5-26.

Based on the weight of evidence, risk potential to the common tern colony is considered low. Modelled exposures suggest low risk potential, with only iron having 95th percentile HQs> 1. Clutch counts from 2010 suggest the colony is within the range of clutch counts in other areas of New Brunswick. Fish tissue concentrations of mercury and selenium are well below thresholds associated with adverse effects in piscivores, and measured residues in eggs, kidney and liver are below toxicity thresholds (where they are available), with the exception of lead in a number of kidney and liver samples. While exceedance of toxicity thresholds for lead in some samples suggests a high potential for adverse effects in those individuals, a limited number of dead chicks were found following extensive daily surveys of the colony this year, and many of the metals residues within tissues were below toxicity thresholds suggestive of clinical or severe effect levels. Weighing the available information, some individuals within the colony have a high potential for adverse effects from exposures to lead, but there appears to be a low probability of effects on the colony as a whole, based on the numbers of chicks exceeding toxicity thresholds, relative to the number of eggs reported in previous colony counts. The colony has returned to nest at the smelter year after year, and anecdotal observations suggest it is increasing in size. There is uncertainty in this conclusion related to specific clutch size for 2014, and exposures to chicks which were not sampled.

Table 5-26 Weight of Evidence Evaluation for Common Tern

Modelled HQs	Fish Tissues for Fish-eating Wildlife and other Literature	Measured Tissue Concentrations			Clutch Counts (CWS, 2010)	Potential Risks to Common Tern
		Egg	Chick Liver	Chick Kidney		
Fe was the only COPC with HQ >1 in any area. Probability of HQ> 1 was 70 to 74%; 95th percentile HQs ranged from 3.6 to 3.7 in Areas 1 to 3 and 3.2 in Areas 1 to 3 combined. Mean HQ was 1.7 in Area 1, 2 and 3. HQ for Fe considered to pose negligible risk given Fe is an essential nutrient and HQ is based on no effect level-TRV (i.e., an MTL)	Hg and Se in prey items are less than tissue residue guidelines established to protect fish-eating avian receptors; Other literature suggests exposure levels are elevated in study area	Concentrations of As, Cu, Hg, Se, Zn are below thresholds associated with effects; No effects thresholds available for Cd or Pb; Concentrations of Cd and Pb in eggs appear elevated, relative to other common tern egg data available in literature. Thresholds used are for other avian species at other contaminated sites, and hence, are associated with some degree of uncertainty.	Cd, Cu, Hg, Se, Zn measured at concentrations below toxicity thresholds; No As benchmarks available; Pb at concentrations above severe clinical effects thresholds in some livers (30% of samples); at clinical effects threshold range in 5% of samples; and at background or subclinical threshold ranges in majority of samples (65%). Thresholds used are for other avian species at other contaminated sites, and hence, are associated with some degree of uncertainty.	Cd, Hg, Se, Zn levels are below thresholds associated with adverse effects; No effects thresholds available for As or Cu. Pb within subclinical effects range in 2 samples; within clinical range in 2 samples, and above severe clinical threshold in 1 sample. Thresholds used are for other avian species at other contaminated sites, and hence, are associated with some degree of uncertainty.	Mean clutch count taken in 2010 (CWS, 2011) was 2.26, with other colonies in New Brunswick ranging from 2.17 – 2.56. Smelter counts are therefore in the range of those reported in other areas of NB. Counts for 2014 are not available, but anecdotal observations suggest colony is not decreasing in size.	Risk potential for the colony is considered low, based on HQs, clutch size compared to other areas and fish tissue concentrations of Hg and Se. Measured tissue levels in egg, liver and kidney suggest elevated risk in some individuals, based on exceedance of Pb clinical and several effect level thresholds in some samples. Pb appears to be the predominant COC, as opposed to other metals. Based on these data, adverse effects in some individuals could occur, but the weight of evidence suggests a low potential for population level effects on the colony. Foraging area (near shore versus off shore) would affect exposure levels, but additional exposures from nesting locations are likely contributors to overall exposures.

Notes:
HQ = hazard quotient; MTL = maximum tolerable level

5.6.2 *Weight of Evidence: Black - Crowned Night Heron*

The weight of evidence for the black-crowned night heron included exposure modelling results and probabilities of HQs exceeding 1.0 in addition to and literature from other sites, and is presented in Table 5-27.

Table 5-27 Weight of Evidence for Black-Crowned Night Heron

Area of Interest	Modelled HQs	Literature	Potential Risks to Heron
Area 1	Probability of HQ> 1 was 90% for Fe, 28% for Pb16% for Tl and 15% for Sr. All other COPC were predicted to have a 10% chance or less of the HQ exceeding 1.0 and were not considered a concern. The 95th percentile HQ = 4.6 for Fe, and = 1.7 for Pb. The 95th percentile HQs for Tl and Sr were 1.4l however the average HQs were less than 1.0. The Sr TRV is based on a MTL and 100% bioaccessibility was assumed for beach sand and food; Sr not considered of concern. The Tl TRV was based on an acute starling LD50 with a 100-fold uncertainty factor. Given the slight exceedance of the Tl HQ, the use of 100-fold uncertainty factor and other conservative assumptions, Tl was not considered to be of concern. Fe is an essential element and predicted HQ is based on a no-effect based TRV (i.e., an MTL) and therefore not considered to represent significant concern of adverse reproductive or survival effects (ranked as negligible). Lead 95th percentile HQ of 1.7 is not considered of concern given the TRV was based on an EC20 derived using lead acetate (Edens and Garlich, 1983) which would be expected to be more bioavailable than lead on this site.	Pb and Zn concentrations in beach sand are markedly elevated, relative to concentrations of sediments considered to be protective of waterfowl in Coeur d'Alene, Idaho. Arsenic beach sand concentrations in Area 1 are similar to those considered protective in Coeur d'Alene.	Risks to heron populations are considered to be low. Individual heron have been reported on Area 1, and while nesting has not been observed, it is possible that this could occur. There would be a limited number of individuals present in this area, and hence, population level effects are considered unlikely. Lead would be considered the COC with greatest risk potential, based on the available data.
Area 2	Probability of HQ> 1 was 89% for Fe; 95th percentile HQ = 5.3 for Fe and 1.2 for Sr; Fe is an essential element and predicted HQ is based on a no-effect TRV (i.e., and MTL); therefore not considered to represent significant concern of adverse reproductive or survival effects. Sr HQ slightly >1. Given Sr TRV based on a MTL and 100% bioaccessibility was assumed for beach sand and food; Sr not considered of concern.	Beach sand concentrations in Area 2 are well within concentrations of sediments considered to be protective of waterfowl in Coeur d'Alene, Idaho.	Risks to heron populations are considered to be negligible.
Area 3	Probability of HQ> 1 was 86% for Fe; 95th percentile HQ = 4.2 for Fe; Fe risks based on a MTL and therefore not considered to represent significant concern of adverse reproductive or survival effects.	Beach sand concentrations in Area 2 are well within concentrations of sediments considered to be protective of waterfowl in Coeur d'Alene, Idaho.	Risks to heron populations are considered to be negligible.

Table 5-27	Weight of Evidence for Black-Crowned Night Heron		
Area of Interest	**Modelled HQs**	**Literature**	**Potential Risks to Heron**
Area 1-3	Receptors were assumed to spend 1/3 of their time in each area. Predicted HQs were below 1.0 with the exception of Fe which had an average HQ of 2.3 and 95th percentile HQ of 4.2. As previously stated the Fe TRV was based on an MTL and as such, Fe was not considered to be of concern.	See conclusions for Areas 1, 2 and 3	Overall risks for heron populations along this shore are considered to be negligible.

Notes:
MTL = maximum tolerable level; TRV = toxicity reference value; HQ = hazard quotient

5.6.3 Weight of Evidence: Spotted Sandpiper

The weight of evidence for the Spotted sandpiper was limited to exposure modelling and literature from other sites, and an observational survey, and is presented in Table 5-28.

Table 5-28 Weight of Evidence for Spotted Sandpiper

Area of Interest	Modelled HQs	Literature	Observational Survey	Potential Risks to Spotted Sandpiper
Area 1	The majority of COPCs had mean and 95th percentile HQ values >1.0. Probability of HQ> 1 was >10% for all COPCs with the exception of As. Probability of HQ>1 was 100% for Fe with the next highest probabilities being (in decreasing order) Pb, Sr, Tl, Cu, Se and Zn at 99%, 94%, 91%, 87%, 85% and 83%, respectively. 95th percentile HQ = 21 for Fe and 13 for Pb. Fe is an essential element and predicted HQ is based on a no-effect based TRV (i.e., an MTL) and as such, it is not expected that Fe would be the driving metal for potential risks. Sr and Li TRVs are also based on an MTL. Beach sand is dominant exposure pathway for zinc, whereas diet is predominant for several metals. Lead 95th percentile HQ of 13 is based on a TRV (EC20) derived using lead acetate (Edens and Garlich, 1983) which would be expected to be more bioavailable than lead on this site (which would be a lead sulphide form). Given this, and the assumption that bioavailability of metals in foods is 100%, HQs are likely overestimated.	Pb and Zn concentrations in beach sand are markedly elevated, relative to concentrations in sediments considered to be protective of waterfowl in Coeur d'Alene, Idaho. Average arsenic beach sand concentration in Area 1 is similar to those considered protective in Coeur d'Alene.	4 nesting pairs reported in Area 1 in 2009 June 2015 survey indicates 4 confirmed spotted sandpiper nests, and 1 possible nest. Clutch size was 3 to 4 eggs/nest, with the exception of a single nest with 1 egg. These counts are similar to reference counts, and number of nests in the study area were higher than reference (which may be more related to habitat). Killdeer (another shore bird) were also found to be nesting in Area 1.	Risks to sandpiper populations considered low to moderate. Individual sandpiper and nesting pairs (4) have been reported in Area 1 historically, and were confirmed in 2015. Based on the outcomes of the assessment, risks to spotted sandpiper in Area 1 are low to moderate given the majority of COPCs had elevated HQs and the probability of the HQ being >1 was high for almost all COPCs. Since the avian TRV is based on a more bioavailable form of lead (lead acetate), and since dietary exposure was assumed to be 100%, these HQs are likely over estimated. Literature suggests Pb and Zn are elevated in beach sand, relative to protective concentrations in other studies. Lead would be considered the COC with greatest risk potential, based on the available data, but other COCs contribute to overall risk. Based on the available lines of evidence, adverse effects on individuals are possible and could be occurring in some individuals. While it is considered unlikely that adverse effects are occurring on the population, this remains a possibility.
Area 2	The majority of COPCs had mean and 95th percentile HQ values >1.0. Probability of HQ> 1 was >10% for all COPCs with the exception of As, Cd and Zn. Probability of HQ>1 was 93% for Fe with the next highest probabilities for Sr, Se, Cu, Al and Pb at 89%,	Beach sand concentrations of Cd, Cu, Pb, Zn in Area 2 are well within concentrations of sediments considered to be protective of waterfowl in	Based on the 2015 survey, sandpiper nesting was confirmed in Area 2, and killdeer (another shore bird) were also found to be nesting in Area 2.	95th percentile HQs and consideration of the basis upon which they were derived, indicate a low risk potential, based on mean HQ values. Literature suggests beach sand concentrations are well within protective levels, but diet is a predominant pathway in this area. While

Table 5-28 Weight of Evidence for Spotted Sandpiper

Area of Interest	Modelled HQs	Literature	Observational Survey	Potential Risks to Spotted Sandpiper
	64%, 61%, 57% and 56%, respectively. Fe is an essential element and predicted HQ is based on a no-effect based TRV (i.e., an MTL). Sr and Li TRV also based on an MTL. Se not considered to pose a risk based on concentrations < tissue residue guidelines. Diet is the dominant exposure pathway for most metals..	Coeur d'Alene, Idaho. No comparison could be made for Al or Fe.		exposures in this area could possibly effect some individuals, the available lines of evidence suggest there is a low probability of population level effects
Area 3	Cu, Fe, Se and Sr had mean HQ values >1.0. Probability of HQ> 1 was the greatest for iron at 97%, followed by Sr (88%), Se (61%) and Cu (55%). The 95th percentile HQ for Fe = 10 and Sr = 7.7. Fe and Sr risks based on a MTL and therefore not considered to represent significant concern of adverse reproductive or survival effects. Se not considered to pose a risk based on concentrations < tissue residue guidelines. The Cu HQ 1.4 (mean) and 3.6 (95th percentile) was derived based on the lowest bounded reproductive avian LOAEL for exposures to copper sulphate pentahydrate. This form of copper would be expected to be more bioavailable than copper in beach sand. Diet is the dominant exposure pathway for most metals..	Beach sand concentrations in Area 3 are well within concentrations of sediments considered to be protective of waterfowl in Coeur d'Alene, Idaho.	No data available; but habitat is present	Risks to sandpiper populations are considered to be low.
Area 1-3	Receptors were assumed to spend 1/3 of their time in each area. Predicted HQs were greater than 1.0 for the majority of COPCs.	See Area 1 - 3 conclusions	See above	Receptors assumed to spend equal amounts of time in each area. Risks are driven by Area 1 concentrations, but Area 1 is limited in size. Risks to sandpiper populations are considered to be low to moderate in Area 1 and low in Areas 2 and 3. Mean HQs in Areas 2 and 3 are < 2 for Pb, Overall risks for sandpiper populations along this shore are considered to be low, based on the exposure potential coming largely from diet.

6.0 UNCERTAINTIES AND LIMITATIONS

6.1 Overview

One component of ERA involves assigning numerical values to various input parameters in models to obtain estimates of exposure and risk. Numerical values are typically required to describe chemical concentrations in environmental media, their fate and transport, wildlife exposure and receptor parameters, and toxicity. The conclusions of any risk assessment are dependent on the data and assumptions that are evaluated within it, and are greatly influenced by the variability and uncertainty that are associated with these data and assumptions. Therefore, the key areas of variability and uncertainty and any major study limitations should be characterized and understood so as to avoid possible underestimating, or artificially overestimating risks, to the extent possible. Risk managers need this information to make informed decisions regarding whether or not risks need to be managed, to what extent, and how the risks can best be managed. By understanding variability and uncertainty, risk managers can identify situations where the use of more sophisticated approaches and/or further data collection can reduce or refine key sources of uncertainty and/or variability before making final risk management decisions.

Where variability and uncertainty are known to exist, it is standard risk assessment practice to make assumptions and select data that overestimate, rather than underestimate potential exposure and risk. Given the tendency for the numerous conservative assumptions used in the ERA to overestimate potential exposure and hazards for the COPCs, it is considered likely that the ERA has overestimated potential COPC exposures and risks in the receptors evaluated.

The inherent tendency of ERAs to overestimate exposures and toxicity to ecological receptors favours Type I errors (false positives; calculated ER>1.0 when in reality ER<1.0) and reduces the probability of Type II errors (false negatives; calculated ER<1.0 when in reality ER>1.0). For example, in the COPC identification approach used in this ERA, both simple comparisons of maximum beach sand concentrations collected within the Study boundary to soil guidelines and/or reference concentration statistics, and statistical comparison tests are prone to a high Type I error (Myers and Thorbjornsen, 2004; Leadon et al, 2007; CalEPA, 1997; U.S. EPA, 2001b; 2002b). Some reasons why these approaches tend to have a high rate of false positives are that trace element distributions in soils (or, in this case, beach sand) tend to have very large ranges (two or three orders of magnitude are not uncommon), and are highly right-skewed, often having, or resembling lognormal distributions. The accurate characterization of the upper tails of such skewed distributions requires a large number of background samples, which are often not available. The probability of false positives increases if the site dataset is larger than the background dataset (which is common, and was the case for all media and biota samples in the ERA Study). In addition, statistical comparison tests treat each COPC as an independently behaving entity, and do not consider the geochemical, ecological or biological contexts in which each chemical occurs (Myers and Thorbjornsen, 2004). The U.S. EPA (2001b) notes that a Type I error is less serious than a Type II error (false negative) when selecting COPCs, and the use of approaches that favour Type I errors are inherently more protective of the environment.

Uncertainty should not be confused with variability. Uncertainty is a lack of confidence in a result or estimate stemming from limited data, or missing information. Variability describes differences in parameter values such as metal concentrations at different locations within the Study boundary, or differences in body weight or food intake rates for individual animals (*i.e.*, population heterogeneity). In other words, variability is defined by the range or "spread" of values in a given population, and is influenced by sample size, repeated measures and area of coverage.

Gaining and maintaining an open acknowledgement and characterization of uncertainty and variability in an assessment is crucial to the success of the decision-making process (Moore and Bartell, 2000). The method used to assess the uncertainty surrounding the exposure estimates depends on the complexity of the model, the information available, and sources of uncertainty. Potential sources of uncertainty in the ERA can be divided into one of the following categories (U.S. EPA, 2001b):

- Parameter uncertainty;
- Model uncertainty; and,
- Scenario uncertainty.

One of the more difficult issues in assessing exposures and risks to ecological receptors, and characterizing the uncertainty and variability in the approaches used, is the establishment of *a priori* performance criteria for model results (Moore and Bartell, 2000). There are numerous complicating factors that can impede the efficiency and success of developing *a priori* criteria though, and all *a priori* approaches require at least some information on some variables from within the assessed area (which may not exist prior to initiating a study), and some assumptions must be made. Such requirements can make establishing *a priori* criteria impractical. This is especially the case when the assessors must design and conduct sampling programs for environmental and biological media, over a large and heterogeneous spatial area (which is the case in the current ERA).

In the evaluation of uncertainty and variability, what is ultimately most important is that one has reasonably high certainty that the ERA does not under-predict exposures and risks, and that the models used will rarely predict the absence of risk when there is indeed a risk (*i.e.*, avoid or minimize the occurrence of false negatives or Type II errors). Therefore, the objective for the analysis of variability and uncertainty in any ERA is to demonstrate the following:

- Model input variables reflect the natural variability in the environment; and,
- Model input variables are assigned conservative values in the face of uncertainty.

A key question when characterizing uncertainty and variability in relation to a particular model input parameter is: "Will the collection of more information dramatically improve the understanding of the variability, and/or reduce uncertainty?" At some point, the collection of additional data will reach the point of diminishing returns, when the effort and resources that are expended to further understand variability and reduce uncertainty are no longer producing meaningful improvements. For example, if additional beach sand or biota chemistry data collection were to occur, and the new data yielded concentrations that fell well within the range of existing data, with no substantial changes to values that measure the "spread" of the data (such as variance, standard error, standard deviation, coefficient of variation etc.), then the need for still further collection of the same type of data would be considered unnecessary and impractical, particularly if data collection efforts are time and resource (or cost) intensive. However, collection of data for a supporting line of evidence (e.g., biological survey) could be considered.

The variability and uncertainty related to each medium and parameter used in the assessment are characterized in this section.

6.1.1 Chemistry Data

Marine Water Data:

Water chemistry data are limited, and are comprised of sampling on 4 separate days during the course of the deployed mussel study. Water metals levels could range dramatically both within the water column, and seasonally, as well as with distance from the facility. The study area is an active open water area, and dispersion potential for effluent or other releases is considered to be reasonably high. The small number of samples means the data have a high degree of uncertainty associated with them, and it reduces the confidence in any conclusions related to potential impacts to pelagic species. But due to high dilution potential and the lack of exceedance of any dissolved marine water quality guidelines, the risk potential is considered low. Collection of additional data would reduce uncertainties associated with water chemistry, as it would improve the understanding of the temporal variability in concentrations.

Marine Sediment Data:

The collected data follow previous study designs, implemented in the same reference and study areas, with the exception of area SST2, which was a new sampling area. Since FE and FPO and the two reference areas have been sampled previously over the past decade (2004 and 2008), there is increased confidence that the data are representative of the area. For SST2, there is obviously lower confidence in that no historical data are available for comparison, but since identical sampling protocols were implemented to those used in historical studies, the uncertainty is considered moderate for this area. This suggests that collection of additional data east of the smelter (i.e., SST2) in this area may improve the understanding of the existing distributions of the data.

Beach Sand Data:

Beach sand chemistry is highly variable in Area 1, due to the presence of slag along the beach, which has created heterogeneity, in terms of metals concentrations. Since this situation creates high variability, additional sampling would not be expected to reduce variability. Variability is reduced in Areas 2 and 3, which is likely a function of reduced slag within the samples. Table 6-1 presents the means and standard deviations for each area, for selected metals of interest, which clearly indicates the high variability in Area 1, versus Areas 2 and 3.

Table 6-1	**Comparison of Mean and Standard Deviations for Selected Metals in Beach Sands in Areas 1, 2 and 3 (mg/kg)**		
Metals	**Area 1**	**Area 2**	**Area 3**
Arsenic	220.1 (122.0)	16.0 (4.9)	19.9 (6.4)
Cadmium	12.03 (6.0)	0.30 (0.08)	0.59 (0.37)
Lead	7500.0 (6249.1)	56.3 (17.6)	81.3 (45.4)
Thallium	9.29 (11.18)	0.60 (0.15)	0.51 (0.18)
Zinc	21095.7 (13174.5)	116.1 (58.7)	218.7 (166.0)

Note: Mean (Standard Deviation)
N = 7 for each area

Based on these data, the area is considered adequately characterized, and further sampling is unlikely to reduce uncertainty. Comparisons to EEM data collected by Glencore over the years are complicated by differing sampling protocols used. The protocol implemented in the current study focused on intertidal sediments, for the purposes of characterizing exposures to foraging shorebirds.

Fish Tissue and Shoreline Invertebrate Data:

Fish tissue and shoreline invertebrate data were used to characterize dietary exposures to avian receptors foraging in the area. The samples were collected in appropriate exposure areas (e.g., fish were netted near smelter effluent discharge, in areas where avian species were foraging; shoreline invertebrates were sampled along the shoreline tidal zone, where receptors have been seen foraging). While the number of samples is not large for sand lance or shoreline invertebrates in Area 1, it is reasonable for Atlantic herring, and shoreline invertebrates in Areas 2 and 3. Data from Glencore's EEM monitoring program for 2011 and 2012 are presented in Table 6-2, in comparison to data from the current program, to give perspective on other measured metals concentrations in potential food items in the area. The species sampled in the Glencore EEM program are native mussels of varying sizes. Concentrations in these mussels could be influenced by sand and adhered particulate matter, which is similar to both the shoreline invertebrate samples, and the sand lance samples (wherein the sand lance contained ingested sand from the beach areas). While the Glencore mussel data have higher concentrations of metals than the shoreline invertebrates and Atlantic herring, the sand lance data present similar concentration ranges, and have higher concentrations for zinc and thallium. Sandpipers were assumed to only feed on shoreline invertebrates, and because their preferred food is small arthropods and annelids, they would likely not forage extensively on mussels.

While uncertainties could be reduced through collection of more samples, and by sampling different species within the environment, the data provide a reasonable indication of variability in concentrations, and hence are considered to adequately characterize potential food intake items.

Table 6-2	**Concentrations of Selected Metals of Interest in Marine Biota (mg/kg ww)**												
Biota Type	Glencore EEM Data												
	Year	Area 1E				Area 2E				Area 3E			
		Pb	Zn	Cd	Tl	Pb	Zn	Cd	Tl	Pb	Zn	Cd	Tl
Small Mussels[a]	2011	22-46.1	22.3-37.5	1.6-2.42	0.05-0.06	14.6-18.3	13-18	0.84-1	<0.03-0.03	11-14.8	16.5-18.2	0.56-0.67	<0.03
	2012	51.9-73.7	27.4-41.4	1.67-2.19	<0.03	26.3-28.3	21.8-21.9	0.76-0.9	<0.03	19.7-26.3	20.5-23.3	0.56-0.59	<0.03
Large Mussels[a]	2011	24.1-53.7	19.1-36.7	2.22-4.19	0.05-0.08	17.1-26.4	11.2-13.5	1.19-1.55	<0.03	8.07-15.8	11-13.6	0.45-0.68	<0.03
	2012	41.5-74.1	20.9-27.4	1.51-2.45	<0.03-0.04	22.2-31.6	16.5-22.5	1.03-1.21	<0.03	18.3-29.3	13.9-19.4	0.67-1.03	<0.03
Biota Type	Current Study												
	Year	Area 1				Area 2				Area 3			
		Pb	Zn	Cd	Tl	Pb	Zn	Cd	Tl	Pb	Zn	Cd	Tl
Shoreline Invertebrates[b]	2014	13.8-37.5 (23.0)	18.8-59.9 (36.4)	0.298-2.3 (0.87)	0.17-1.72 (0.64)	1.25-22.3 (5.83)	7.5-45.5 (16.18)	0.077-0.81 (0.25)	0.03-0.35 (0.10)	2.85-7.19 (4.11)	6.8-33.3 (12.2)	0.066-0.245 (0.12)	0.02-0.09 (0.04)
Atlantic Herring[c]	2014	0.769-1.71 (1.26)	22.2-27.2 (24.4)	0.0624-0.109 (0.0832)	0.239-0.358 (0.290)	ND	ND	ND	ND	ND	ND	ND	ND
Sand Lance[d]	2014	0.936-59.7 (26.0)	31.7-299 (145)	0.0728-0.205 (0.133)	0.347-0.549 (0.488)	ND	ND	ND	ND	ND	ND	ND	ND

Concentration data is presented as a range (minimum – maximum). Mean is provided for the current study in parentheses ().

[a] A total of 3 samples of small mussels and large mussels were analyzed each year (n=3); Appendix A indicates that some data quality issues may be present in the 2012 EEM data. The data in this table represent the corrected data.

[b] Shoreline invertebrates N = 6 for Area 1; N = 9 for Area 2; N = 8 for Area 3

[c] Atlantic Herring… N = 10

[d] Sand Lance N = 6

Deployed Mussel Data:

The 66-day mussel chemistry data are considered an adequate representation of possible exposures in the marine environment. The study followed a standardized protocol, and sourced mussels from a reputable supplier, wherein mussels of a similar size were used to assess the growth and condition endpoints. It is uncertain whether the mussels reached steady state in the environment, and as such, concentrations in tissues could be lower than those that might be achieved if steady state were achieved. Exposures to the caged mussels would have been predominantly related to water and suspended particulate and food within the water column, as opposed to sediments, as the deployment cages were not suspended close to sediment.

Comparisons of native mussels, sampled along the shore during the Glencore EEM program suggest higher concentrations than those measured in the caged program, which could be a function of additional sediment exposures in native mussel colonies, adhered sand and particulate matter, or simply longer exposure periods which enabled the mussels to reach steady state, which could be increasing concentrations within the native mussel results. Tissue residues from the deployed mussel project were not used as food intakes for avian species. Based on the comparisons presented in Table 6-2, adequate variability in prey items has been captured due to the fact that 3 different categories of prey were analyzed, and hence, additional mussel data would not reduce uncertainties in exposure data.

Chick and Egg Data:

The chick liver (N = 13 from the current study, and N= 7 from CWS, 2014) and kidney tissue data (N = 7), and egg residue data (N = 18), are considered to have adequate sample sizes, with the possible exception of kidney samples, which were limited (due to the need to analyze composite samples due to their small size, and difficulty extracting kidneys from the body cavity). The data have uncertainties associated with them, in terms of characterizing exposure levels in the colony, as the samples were all either rejected eggs, or chicks that had died. As a result of these factors, the measured concentrations may be a biased high estimate of exposure, rather than a representative level of exposure for the colony. The chicks could have been rejected due to any number of factors, one of which may have been related to lead toxicity affecting the ability of the chick to thrive and compete with other nestlings. Lead exposure in these individuals could have been affected if food was dropped in the areas surrounding the nest, and adhered lead particles on the food were then ingested. This circumstance, while speculative, could result in higher exposure levels in some individuals than others.

Egg concentrations are likely related to maternal transfer of contaminant loadings. Egg contaminant loadings in common terns are generally considered to represent more local exposures, rather than over wintering exposures, since most birds spend several weeks in the nesting area before they lay eggs, and hence, are foraging and acquiring exposures from the nesting area (Burger, 2002). The specific reasons for egg rejection are not known, and hence it is difficult to confirm whether contaminant loading between rejected eggs versus non-rejected eggs would differ. Based on the available data, these eggs were considered to be reasonable representations of exposure potential within the colony.

6.1.2 Biological Measurements and Assessment Thresholds

Benthic Community Abundance and Diversity:

No benthic community data were available at SST2. Nonetheless, comparisons of chemistry data from other areas (FE and FPO), which are sufficiently characterized, as well as 2014 and historical benthic community diversity and abundance outcomes from FE and FPO, provide perspective on the possible effects measured metals at SST2 stations may have on benthic community metrics. Weighing this information suggests that there is a low potential for significant adverse effects on density, diversity or richness at SST2 stations, particularly with increasing distance from the facility. There is uncertainty in this conclusion, but the confidence in the conclusion is reasonable high, based on the available information.

Fish Health Assessment:

The low numbers of male tomcod represent a significant source of uncertainty in the fish health assessment. As discussed in Appendix F, it is currently unknown why male sample numbers were low at both the reference and smelter-exposed locations, but was hypothesized to reflect pre-spawning migration to breeding areas. Additional data, or repeating the study, would likely reduce uncertainties in the conclusions related to males. The sample size related to the female dataset appears within required numbers for these types of studies.

Deployed Mussel Study:

The deployed mussel study followed a standard protocol to assess survival, growth and condition factors, which requires that mussels be left in the environment for between 60 – 90 days (see Appendix F). The deployed time frame was 66 days, which is within the acceptable time frame. It is unlikely that extending the time frame of the study would have a significant effect on the outcome of the endpoints assessed in this study, since the protocol was followed. It may be that tissue residues increase slightly, if steady state had not been achieved.

Common Tern Chick and Egg Metal Assessment Thresholds:

The toxicity thresholds selected to assess metals residues in chick livers and kidneys, and in eggs, were not specific to the common tern, and hence are associated with some uncertainty. Many of these thresholds are from a compilation of literature conducted by experts in the field (Franson and Pain, 2011), for various avian species at different sites which could have differing speciation of metals associated with the data. Therefore, while not specific to the common tern, they likely represent reasonable benchmarks of comparison, but are associated with uncertainty, and require consideration of other lines of evidence in drawing conclusions. Coupling these data with clutch count information assists in understanding whether the colony is within expected colony sizes for New Brunswick. While the clutch counts are from 2010, and most of the chick liver and kidney data are from 2014, samples analyzed by CWS (chick livers) collected in 2010, show similar concentration ranges to those reported in 2014. This suggests similarities between the years, but there are uncertainties related to these comparisons e.g., different laboratories conducted the analyses; preparation of tissues may have varied, etc.).

Avian Species Assessment Model Parameters and Uncertainties and Variability:

Model uncertainties, variability and parameters are discussed in detail in Appendix J. The application of a probabilistic modelling approach considers the variability in the database, and the use of measured data for prey reduces uncertainties related to exposure parameters. Further discussions are presented in Appendix J.

With respect to avian TRVs, none of the selected TRVs are based on the species considered in this assessment, which is not unusual. Most of these TRVs are based on laboratory studies, as opposed to field studies, wherein additional stressors are present which could exacerbate effects. Therefore, there is uncertainty related to the application of these values in the current assessment. Where there are multiple lines of evidence (e.g., comparison of fish tissue residues to guidelines set for the protection of piscivores; assessment of egg and chick data, relative to toxicity thresholds; clutch counts; literature from other sites), uncertainties related to overall conclusions are reduced.

Bioaccessibility Testing Protocol:

The mobility of metals adhered to beach sand within the gut and intestine of avian species was tested using a protocol developed by the Royal Military College (See Appendix K). This protocol is based on sediment ingestion rates related to mallard duck, which is based on a 3.3% dry weight of food for the mallard duck. Based on the mallard's body size, gizzard size is estimated, as is sediment clearance/day to get an approximate 200:1 liquid to solid ratio (I. Koch, RMC, personal communication). These estimates are considered conservative, relative to the sandpiper (which has the highest sand ingestion rate of the three receptors), as the mallard is larger and eats less per body weight than the sandpiper. This suggests the residence time in the stomach and metabolism/clearance of food and sediment in gut and intestine may be faster in the sandpiper, relative to the mallard, which would shorten the exposure time. In addition, the sand ingestion rate for the sandpiper (18%) is higher than that assumed for the mallard (3.3 %), which would also reduce the liquid to solid ratio, relative to the mallard. This information suggests that the avian bioaccessibility protocol should over estimate, rather than underestimate bioaccessibility, for the sandpiper.

Bioaccessibility in dietary items was assumed to be 100% of all metals. This is likely an overestimate of exposure, as bioaccessibility of metals in diet have been reported to range in other studies (e.g., Ollson et al, 2009). The application of the assumption of 100% bioaccessibility in food items likely biases risk estimates high.

7.0 SUMMARY OF CONCLUSIONS

Glencore commissioned a study to examine the potential for ecological risks in the marine environments adjacent to the Brunswick Smelting facility, associated with current and on-going operations. This ecological risk assessment (ERA) focused on marine aquatic life and species foraging in areas near the smelter. The primary releases of interest from the smelter relate to current lead smelter treated effluent discharge, former fertilizer plant gypsum-based effluent discharge, atmospheric discharges, and possible contributions related to erosion of the former slag disposal area on Belledune Point. The main receptor groups of interest include aquatic species (phytoplankton and pelagic invertebrates, benthic invertebrates, and fish species) as well as avian species living and foraging in the marine environment and the associated shoreline at or near the facility. Following the review of existing data and information, a field sampling program was implemented to conduct the following:

- A benthic invertebrate abundance and diversity study, including sediment chemistry and physical characterization;
- A shellfish health assessment, involving deployed mussels and assessment of survival, growth and condition endpoints, with body burden and marine water quality chemistry characterization;
- A fish health assessment, involving a benthic fish species, and survival, growth, condition and reproduction endpoints;
- Sampling of whole fish tissue and shoreline invertebrate chemistry analysis, as well as beach sand chemistry and bioaccessibility testing, for input into an Exposure Model to characterize exposure and risks to various avian species nesting and foraging in the area;
- Sampling of salvage chick organ tissue and eggs of the common tern, for the purposes of metal residue chemistry analysis; and
- Shorebird population and nesting survey.

Based on the data and assessments conducted, the following conclusions were drawn:

Marine Phytoplankton and Pelagic Invertebrates:

- Risks are considered to be negligible to low, based on comparison of measured water quality metals concentrations to marine water quality guidelines and reference, as well as to other toxicology data and information.
- The exposure data are limited in terms of number of samples, and hence there is uncertainty in this conclusion. This uncertainty is reduced by knowledge that the area adjacent to the smelter is a highly dispersive environment, and while releases from the facility are measureable in the environment, exposure levels for transient mobile species are expected to be low, and hence would not be anticipated to result in population- or community-level effects.

Marine Benthic Community:

- Risks are considered to be low for benthos near the former fertilizer outfall location, and in an area distant to the final effluent discharge point, and are considered moderate for the final effluent discharge area, based on the existing chemistry data, and benthic density, diversity and richness data. Evenness and diversity of the benthic community at the final effluent area suggested ecologically meaningful differences from reference. There was also reduced diversity in this area, relative to reference, albeit, to a lesser degree than that reported for evenness and diversity. In the current survey, increased sediment metals concentrations, lower benthic invertebrate density and differences in community structure relative to surveys conducted in 2008 and 2004, were noted, which were not linked to effluent discharge, but rather, appear to be related to either erosion of the former slag disposal area at Belledune Point, or a recently completed harbor dredging project.

Marine Shellfish:

- Risks are considered to be low, based on the available data and studies conducted. Survival was not considered to be influenced in the study area, relative to reference. Growth was actually greater in the study area mussels at several sites, than in reference areas, but condition was slightly lower. These results were attributed to higher allocation of energy use to growth in the smelter-exposed mussels compared to reference. While tissue metals were significantly higher in the exposure group for arsenic, cadmium, copper, lead, selenium, silver, strontium and zinc, the results of the survival, growth and condition endpoints indicate no adverse smelter-related effects toblue mussels.
- Uncertainties in the assessment include a lack of assessment of the reproductive endpoint, since the study was initiated outside of the season of reproductive tissue development (and hence reproduction endpoint could not be evaluated). Nonetheless, numerous juvenile blue mussels were found adhering to the cages of the deployed mussels. While a quantitative assessment of reproductive endpoints was not undertaken, qualitative observations suggest presence of juveniles in all cage areas with lower numbers being observed at the smelter-exposed station located furthest from the smelter (Station S4).

Marine Fish:

- Risks are considered to be low, based on assessment of water quality, survival, growth/condition, reproduction and tissue residue data. No critical effect sizes were exceeded for any endpoint with the exception of egg size. Smaller egg size in smelter-exposed fish was hypothesized to reflect natural variability in spawning timing between the exposure and reference fish populations.
- Male outcomes are uncertain due to limited sample size, but are not indicative of adverse effects, based on the existing dataset.

Avian Species Nesting and Foraging in the Area:

- Common tern nest on the smelter property annually, and forage in both the near shore and far shore areas adjacent to the smelter. Based on the weight of evidence, risk potential to the common tern colony is considered low. Modelled exposures suggest low risk potential to the common tern colony, with only iron having 95^{th} percentile HQs> 1. Clutch counts from 2010 suggest the colony is within the range of clutch counts in other areas of New Brunswick. Fish tissue concentrations of mercury and selenium are well below thresholds associated with adverse effects in piscivores, and measured residues in eggs, kidney and liver are below toxicity thresholds (where they are available), with the exception of lead in a number of kidney and liver samples. While exceedance of toxicity thresholds for lead in some samples suggests a high potential for adverse effects in those individuals, a limited number of dead chicks were found following extensive daily surveys of the colony in 2014, and many of the metals residues within tissues were below toxicity thresholds suggestive of clinical or severe effect levels. Weighing the available information, some individuals within the colony have a high potential for adverse effects from exposures to lead, but there appears to be a low probability of effects on the colony as a whole, based on the numbers of chick tissue samples exceeding toxicity thresholds, relative to the number of eggs reported in previous colony counts. The colony has returned to nest at the smelter year after year, and anecdotal observations suggest it is increasing in size. There is uncertainty in this conclusion related to specific clutch size for 2014, and exposures to chicks which were not sampled.

  - Black-crowned night heron forage on and near the smelter property (Belledune Point), but nesting pairs have not been observed in previous surveys conducted. Risk potential for this species is considered to be negligible to low, based on low probability of Hazard Quotients exceeding 1, with the exception of iron, lead, and to a lesser extent, strontium and thallium. Lead and zinc concentrations in beach sand along Belledune Point are elevated relative to concentrations of sediments considered to be protective of waterfowl in other areas, but beach sand metal concentrations are not elevated in Areas 2 or 3, down the shore. The dominant exposure pathway is diet, but considering that there would be a limited number of individuals present in this area, and hence, population level effects are considered unlikely near the Brunswick Smelter. Lead would be considered the COC with greatest risk potential, based on the available data.

  - Sandpiper forage along the shore of the beach on the smelter property, and four nesting pairs were reported on Belledune Point in surveys conducted in 2009. This survey was updated in 2015, and a total of 6 nesting pairs were confirmed in Area 1 and 2, with 4 possible additional nesting pairs identified. Risk potential for this species is considered to range from low to moderate, depending upon proximity to the smelter. On Belledune Point, risks are considered to range from low to moderate based on the high probability of multiple Hazard Quotients exceeding 1 (aluminum, copper, iron, lead, selenium, thallium and zinc). Lead had the most elevated HQ in this area, and represents the substance of greatest concern. The HQs are likely biased high, due to assumptions that metals in dietary items are 100% bioavailable, and the TRV used is based on lead acetate, which is more bioavailable than the form present in the Belledune area (which would be a lead sulphate). Belledune Point is the area with highest exposure potential, due to the presence of slag along the beach/shoreline, and concentrations of lead and zinc in this area were also found to exceed concentrations reported as being protective of waterfowl in other published literature. Areas further down the shoreline to the east of the facility represent a low risk potential. The risk potential for the shoreline overall was considered to be low as diet was found to be the most important exposure pathway in all areas considered (and bioacccessibility in diet was assumed to be 100%). However, adverse effects in some individuals could be occurring on Belledune Point but are considered less likely in Areas 2 and 3. Depending on exposures and population size an effect on the local population could be possible, but is unlikely.

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