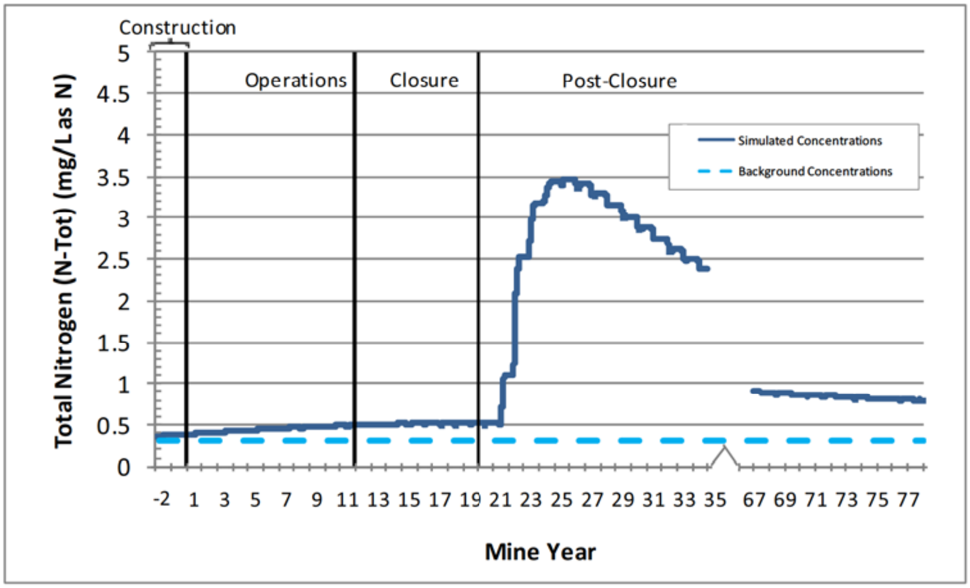
Figure 8.8-19 Predicted Total Nitrogen Concentrations in Area 8



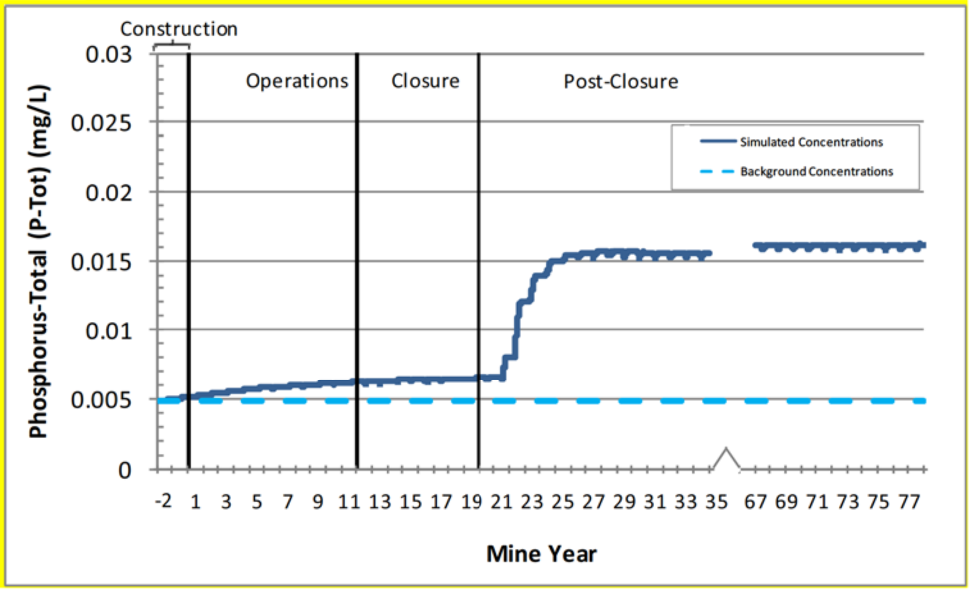
mg/L as N= milligrams per litre as nitrogen

Phosphorus

Concentrations of phosphorus are predicted to increase in Area 8 during post-closure for the reasons described in Section 8.8.4.1 (Table 8.8-15 and Figure 8.8-20). Based on the maximum predicted long-term total phosphorus concentrations (0.016 mg/L), Area 8 would be classified as mesotrophic (CCME 2004; Environment Canada 2004). There are no Canadian water quality guidelines for the protection of aquatic life for phosphorus.

The increase in total phosphorus concentration is expected to increase primary productivity, which is likely to exert a higher WODR within Area 8; the surface water depth zone of Area 8, particularly in areas where the depth is greater than 6 m, is expected to retain sufficient dissolved oxygen concentrations over winter for aquatic life.

Figure 8.8-20 Predicted Total Phosphorus Concentrations in Area 8 with Supplemental Mitigation Strategies



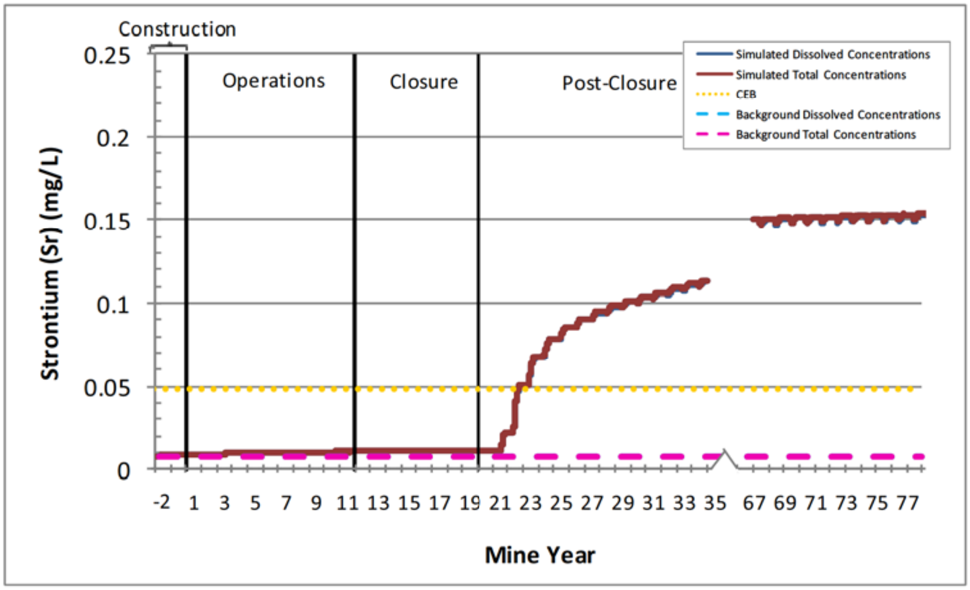
mg/L = milligrams per litre

Trace Metals

Concentrations of trace metals are predicted to follow the general trends described above for Area 8. After the initial period of approximately five years to approach Kennady Lake concentrations, trace metal concentrations are then predicted to decrease, remain relatively constant or decrease, for the reasons described for Kennady Lake in Section 8.8.4.1 for each metal. Representative time series plots are shown for strontium, which is predicted to increase following this five-year period; aluminum, which is predicted to remain relatively constant; and manganese, which is predicted to approach background conditions, in Figures 8.8-21 to 8.8-23, respectively.

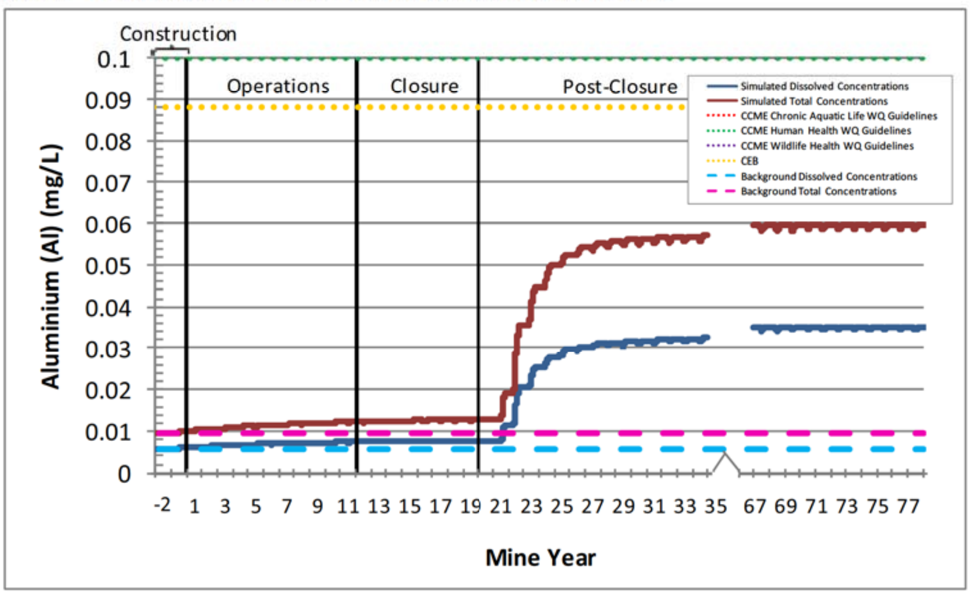
Of the 23 modelled trace metals, cadmium, chromium and copper are predicted to exceed guidelines in post-closure. These metals have been measured in Kennady Lake above guideline concentrations under existing environment conditions (Section 8.3.6).

Figure 8.8-21 Predicted Strontium Concentrations in Area 8



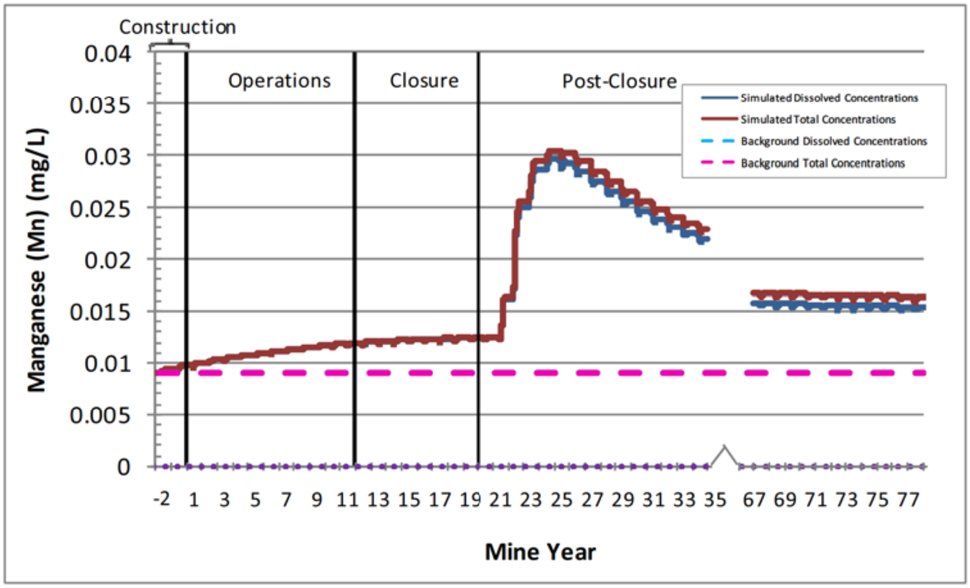
mg/L = milligrams per litre

Figure 8.8-22 Predicted Aluminum Concentrations in Area 8



mg/L = milligrams per litre

Figure 8.8-23 Predicted Manganese Concentrations in Area 8



mg/L = milligrams per litre

The potential health effects of all trace metals on aquatic life, including the three metals that are projected to have maximum concentrations above guideline concentrations (i.e., cadmium, chromium and copper), are assessed in Section 8.9.

8.8.4.2 Long-term Effects of Changes to Pit Water Quality on the Stability of Meromictic Conditions in the Tuzo Pit after Closure

Mining the kimberlite ore will result in three mine pits remaining at the end of operations (see Section 8.4). The first pit to be mined, 5034 pit, will be backfilled with waste rock to the sill between 5034 and Tuzo pit. Above the sill, 5034 will be partially backfilled, and the remaining unfilled portion of the pit will intersect with the portion of Tuzo pit above the sill. Therefore, analysis of Tuzo pit in this section also applies to the un-filled portion of 5034 pit.

Hearne pit will be partially backfilled with fine PK and process water, and the final water depth will be approximately 120 m. Because Hearne pit will not be receiving highly saline water, it is assumed that meromixis will not occur in Hearne pit, and that water in the pit will be fully mixed with water in Area 6. This assumption resulted in conservative predictions, because if meromixis does occur in Hearne pit, the Kennady Lake water quality model will predict higher concentrations from the bottom of Hearne pit (via diffusive flux from the fine PK) that can influence long-term water quality in Kennady Lake.

When Tuzo Pit and Kennady Lake are refilled during the closure phase, higher-salinity water from the remaining portions of Kennady Lake will first be transferred to Tuzo Pit. Local runoff and water from Lake N11 will then be used to fill the remaining pit and Kennady Lake, resulting in lower-density water overlying the initially placed water. The presence of a pycnocline (i.e., a density gradient) is expected to result in stratification of water within the pit, which will essentially isolate the underlying water from Kennady Lake.

As described in Section 8.8.2.3, the stability of stratification in Tuzo pit was evaluated using two methods: by hydrodynamic modelling, and by long-term mass balance analysis. The results of these analyses are presented in the following subsections.

8.8.4.2.1 Hydrodynamic Model Predictions

Because it is not known precisely how much mixing will occur in Tuzo pit during the transition from filling with higher- to lower-density water, the model was run iteratively to determine the elevation of a long-term pycnocline. This long-term elevation was determined by initializing the model with two distinct layers, each comprised of either 100% higher-density or lower-density water. The volume of water remaining isolated after being subjected to 100 years of wind-driven mixing was then set as the initial volume in the monimolimnion for the next iteration, as well as the volume that was assumed to be isolated from Kennady Lake in the Kennady Lake model (Section 8.8.4.1).

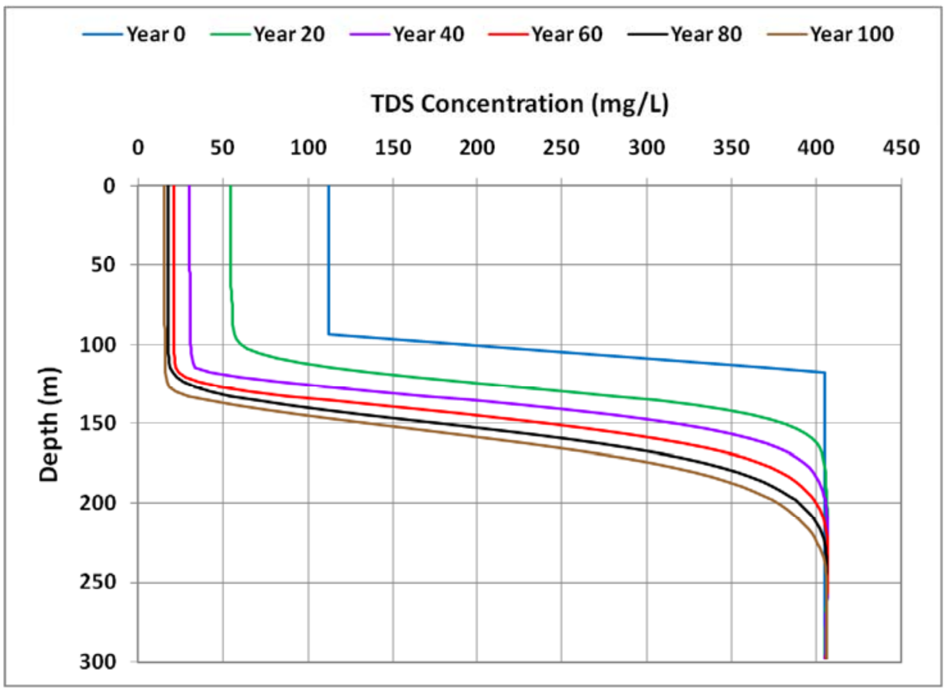
The second iteration was then used to determine the volume of monimolimnion water that might report back to the upper layers due to advection and diffusion. This simulation indicated that the pycnocline would move down slowly over a 100-year period (Figure 8.8-24). These elevations were then converted to volumes based on the storage-elevation curve for Tuzo Pit. A time series of monimolimnion volumes is shown in Figure 8.8-25.

As seen in Figure 8.8-24, the hydrodynamic model predicted a drop in pycnocline elevation of approximately 60 m over the 100-year time frame, which translates to a volume of 7.2 Mm^3 of water reporting to the upper layer of Kennady Lake. This volume of water was added back to the Kennady Lake model (Section 8.8.4.3) to account for the influx of underlying water.

The hydrodynamic results indicate that the rate of drop in pycnocline elevation will decline with time, which has two implications for water quality in Kennady Lake. First, it indicates that influences of Tuzo Pit water on Kennady Lake water quality will diminish with time, because the relative amounts of upward flux water will decrease accordingly. Second, it indicates a strengthening of the stratification as the pycnocline becomes deeper.

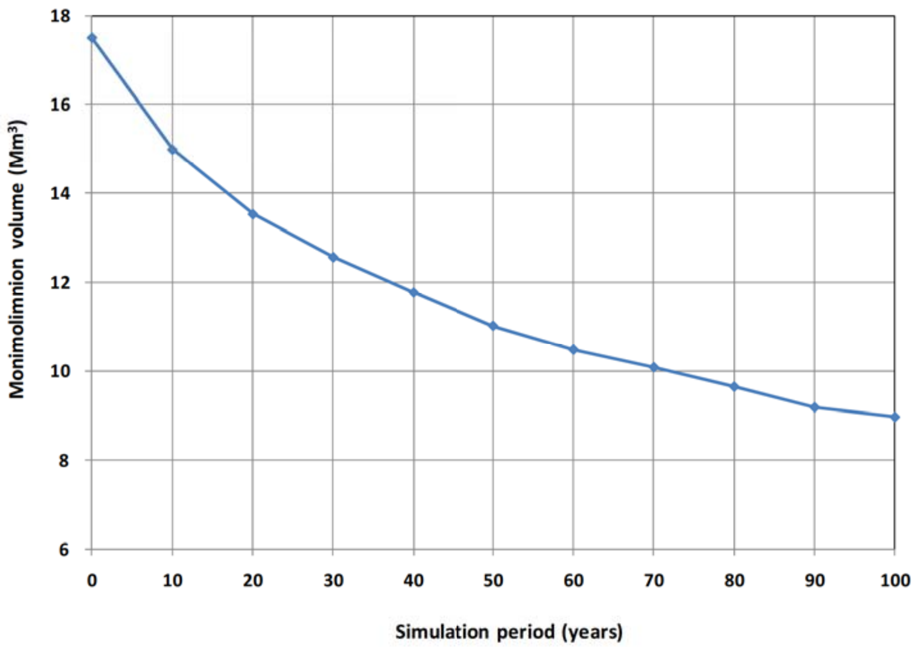
The strengthened stratification is predominantly due to two factors. First, a deeper pycnocline is inherently more stable because wind-driven forces are applied at the lake surface, so the energy required to perturb the system (i.e., the pit lake) increases with depth. Second, the gradual replacement of Kennady Lake waters with natural runoff will reduce the salinity of overlying water, thereby strengthening the pycnocline (i.e., increasing the difference in density between the surface and deep water zones) (Figure 8.8-24). A small influx of groundwater predicted by the groundwater modelling (see Section 11.6) is not predicted to increase salinity at depth over the modelled 100-year time frame.

Figure 8.8-24 Predicted Pycnocline Elevation over 100-year period after Refilling of Tuzo Pit



m = metres; mg/L = milligrams per litre

Figure 8.8-25 Predicted Monimolimnion Volumes over 100-year period after Refilling of Tuzo Pit



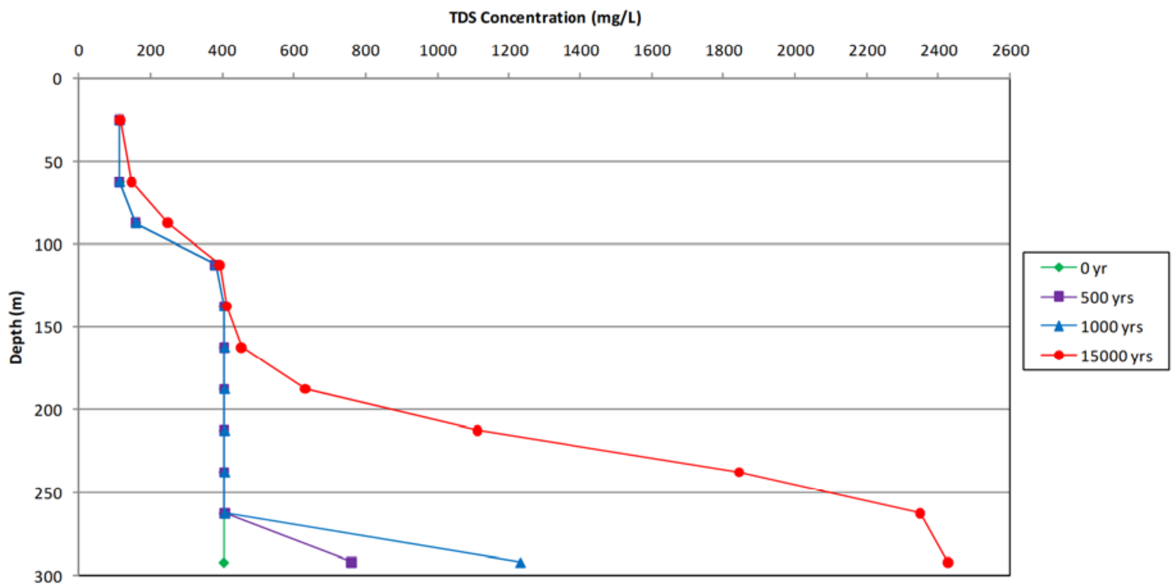
Mm^3 = million cubic metres

8.8.4.2.2 Long Term Modelled Total Dissolved Solids Concentrations in the Tuzo Pit

The mass-balance slice model predicted a rising and strengthening stratification in the long term. Although the hydrodynamic simulation indicated very little change in monimolimnion TDS in the first hundred years, the mass-balance slice model indicated that inflows would begin to change TDS at depth in the first thousand years. After 15,000 years, the model indicated that the monimolimnion would increase in TDS and expand upwards due to the slight net inflow (Figure 8.8-26). The deeper pit water will eventually, over the very long term, take on the characteristics of the surrounding deep, high TDS groundwater

While the general trend of increased TDS and upward expansion of the pycnocline is likely reliable, this model may over-predict the extent to which these phenomena may occur. The model did not account for upward diffusion due to a concentration gradient, and it extrapolated groundwater inflows beyond the timeframe modelled by hydrogeological modelling. Nevertheless, it may be concluded with some confidence from this modelling that stratification in Tuzo pit will strengthen with time.

Figure 8.8-26 Modelled Water Column Distribution of Total Dissolved Solids Concentration in the Tuzo Pit Projected Over Time



m = metres; mg/L = milligrams per litre

8.9 EFFECTS TO AQUATIC HEALTH

8.9.1 Introduction

This section assesses the potential for health effects to aquatic life (referred to herein as aquatic health) in the Kennady Lake watershed resulting from the modelled changes in water quality that were presented in Section 8.8.3.1 during construction and operation and in Section 8.8.4.1 during closure. Section 8.8 also evaluated potential changes in water quality resulting from deposition of dust and metals during construction and operation (Section 8.8.3.1). Effects of dust and metals deposition do not apply to closure, because mining activities that generate dust will ceased after operations end and closure and reclamation activities are complete.

Section 8.8 also evaluated potential changes in the water quality of lakes within the Kennady Lake watershed resulting from deposition of acidifying substances, namely sulphate and nitrate. However, water quality modelling results indicate that Project-related deposition of sulphate and nitrate is not predicted to result in acidification in the Kennady Lake watershed.

Summaries of the primary pathways by which changes to aquatic health could occur during construction and operations and during closure are presented in Table 8.9-1 (construction and operations) and Table 8.9-2 (closure).

Table 8.9-1 Valid Pathways and Effects Statements for Effects to Aquatic Health during Construction and Operation

Project Component	Pathway	Effects Statement	Effects Addressed
Construction and Mining Activity	deposition of dust and metals in the Kennady Lake watershed may change aquatic health [(a)]	effects of air emissions on aquatic health in the Kennady Lake watershed	Section 8.9.3.1

(a) Effects of dust emissions do not apply to closure, because mining activities that generate dust will ceased after operations end.

Table 8.9-2 Valid Pathways and Effects Statements for Effects to Aquatic Health during Closure

Project Component	Pathway	Effects Statement	Effects Addressed
Breaching Dyke A to reconnect Kennady Lake with Area 8	altered water quality in Kennady Lake and Area 8 resulting in changes to aquatic health to waterbodies within the Kennady Lake watershed	effects of water quality changes to aquatic health in waterbodies within the Kennady Lake watershed	Section 8.9.3.2

Based on the primary pathway, three closure scenarios were assessed:

- Initial closure discharge water quality in Kennady Lake. This scenario summarizes the maximum concentrations in Kennady Lake at the end of the closure period, that is, after refilling is complete and just after breaching of Dyke A, which is the dyke between Area 7 and Area 8.
- Long-term water quality in Kennady Lake. This scenario summarizes the maximum concentrations in Kennady Lake 100 years into the post-closure period.
- Post-closure water quality in Area 8. This scenario summarizes the maximum concentrations in Area 8 during the post-closure period, that is, from after refilling of Kennady Lake is complete and full flow is possible between Kennady Lake and Area 8 to 100 years into the post-closure period.

8.9.2 Methods

8.9.2.1 Effects of Air Emissions on Aquatic Health in the Kennady Lake Watershed

Results of the water quality assessment indicate that dust and metals deposition may result in predicted maximum concentrations of suspended solids and some metals exceeding water quality guidelines in two or more lakes (Section 8.8.3.2). These elevated concentrations in surface waters are only expected to occur in short-term pulses during snowmelt and after storm events. The predicted concentrations are based on highly conservative assumptions as used in the air quality modeling and mass balance analysis (Section 8.8.1.1.2). For example, the air quality modeling incorporated conservative assumptions such as not accounting for reductions in dust emissions due to precipitation in summer or snow in winter. The mass balance analysis did not consider total lake volumes as dilution factors in the dust input. Therefore, the predicted concentrations are likely conservative estimates of the maximum potential concentrations. There is no quantitative basis for assessing these predicted concentrations. For example, although elevated suspended solids are expected during freshet, there are no baseline data to indicate what aquatic organisms would be typically exposed to under freshet conditions and therefore, what concentration would be considered "elevated" in relation to baseline. Given the conservative nature of the predicted concentrations, and the limited scientific basis to assess potential effects to short-term changes in this case, the evaluation of potential effects to aquatic health in lakes affected by dust and metals deposition was conducted qualitatively.

8.9.2.2 Effects of Water Quality Changes to Aquatic Health in Waterbodies within the Kennady Lake Watershed

Predicted changes to water quality could affect aquatic health through two exposure pathways:

- direct exposure to substances in the water column; and,
- indirect effects related to possible accumulation of substances within fish tissue via uptake from both water and diet.

Both mechanisms were evaluated as part of the aquatic health assessment. Potential effects related to direct exposure were evaluated based on modelled water quality in Kennady Lake and Area 8 during closure (Section 8.9.2.1.1). Predicted water concentrations were compared with chronic effects benchmarks to evaluate the potential for aquatic health effects due to direct waterborne exposure. The analysis of indirect effects to fish tissue quality was conducted by using measured baseline water quality, modelled water quality, and measured fish tissue concentrations to predict tissue concentrations of chemicals within aquatic organisms (Section 8.9.2.2.2). Predicted tissue concentrations were compared with toxicological benchmarks to evaluate the potential for aquatic health effects related to tissue concentrations. The methods used for both evaluations are outlined in more detail below.

8.9.2.2.1 Direct Waterborne Exposure

Changes to water quality in Kennady Lake and Area 8 after the refilling of Kennady Lake is complete and Dyke A (the dyke between Area 7 and Area 8) is breached were predicted using a dynamic water quality model following the methods described in Section 8.8.2 and Appendix 8.I. The resulting modelled water quality results were passed through a screening procedure to identify substances of potential concern (SOPCs), which are substances for which the modelled concentrations were higher than those observed under baseline conditions and that were also higher than relevant and applicable water quality guidelines for the protection of aquatic life. To assess whether the SOPCs have the potential to affect aquatic health under the evaluated scenarios, modelled concentrations of these substances were compared to chronic effects benchmarks (CEBs), which were derived from a review of available toxicological literature.

The screening procedure used to identify an SOPC was a three-step process. The first step (Step 1) in the process involved assessing which of the modelled parameters had the potential to detrimentally affect aquatic health and which parameters could be excluded from further consideration for one of the following reasons:

- the parameter in question has been shown to have limited potential to affect aquatic health (i.e., innocuous substances);
- potential effects related to the parameter in question are assessed elsewhere in the EIS; and/or
- the parameter in question is a component of another parameter, which is a more suitable focus point for the analysis.

Parameters excluded during the first step of the screening process consisted of:

- sodium, based on work by Mount et al. (1997), which indicates that this substance has low toxicity to aquatic life;
- phosphorus and nitrogen compounds as nutrients, because potential effects related to any potential trophic changes are assessed in Section 8.10.2.1 (note that nitrate and ammonia were also screened for toxicity effects using water quality guidelines for the protection of aquatic life);
- calcium, chloride, magnesium, sulphate, and potassium, because they are individual ions for which Canadian protection of aquatic life guidelines have not yet been established and they are components of total dissolved solids (TDS), another modelled parameter included in the assessment; and
- the dissolved form of metals, metalloids, and non-metals[4], because they are a component of the corresponding total metal concentrations and total metal measurements are a more conservative basis for assessment than dissolved metals measurements.

The remaining substances, which included total metals, total suspended solids (TSS), and TDS, were subjected to a screening process, which involved comparing predicted maximum concentrations with:

- baseline water quality concentrations (Step 2); and,
- Canadian water quality guidelines for the protection of aquatic life (CCME 1999) (Step 3).

[4] Henceforth, metals, metalloids (e.g., arsenic), and non-metals (e.g., selenium) will be referred to as metals.

Step 2 recognized that existing concentrations may also exceed water quality guidelines. If the predicted concentration was less than or within 10 percent (%) of the long-term average concentration under baseline conditions, then the parameter was excluded from the assessment, because no incremental impact on aquatic health would be expected. A difference of less than or equal to 10% was not considered to be a change that would represent a potential effect to water quality, because:

- analytical uncertainty can be as high as, or higher than, 10%, depending on the individual parameter in question;
- a difference of less than 10% is unlikely to be statistically significant; for example, with a sample size of less than 200, the 95% confidence interval of the mean of a normally distributed variable with a typical coefficient of variation of 0.6 will be greater than 10%; and,
- effects to aquatic organisms are unlikely to be detectable for a change in a substance concentration of less than 10%.

Step 3 involved a comparison to water quality guidelines to determine whether substances with guidelines have the potential to affect aquatic health. For SOPCs with guidelines that were dependent on pH (i.e., aluminum) or hardness (i.e., cadmium, copper, lead, nickel), the predicted pH or hardness associated with those SOPC concentrations were used in the screening. For chromium, which has a guideline that is dependent on speciation, the most conservative guideline was used (i.e., hexavalent chromium) although it is assumed that most of the chromium will be present as trivalent chromium (see Section 8.8.4.1.1).

Water quality guidelines represent levels that, if met in any surface water, will provide a high level of protection to aquatic life. In this assessment, the *Canadian Water Quality Guidelines for the Protection of Aquatic Life* were used; these conservative guidelines are intended to ”protect all forms of aquatic life and all aspects of the aquatic life cycles, including the most sensitive life stage of the most sensitive species over the long term” (CCME 1999). That is, exceedance of a water quality guideline indicates the possibility of adverse effects, but not necessarily a likelihood. At this stage in the screening process, parameters without guidelines were identified as SOPCs, with the exception of those specifically excluded above.

For each SOPC, predicted concentrations were compared to chronic effects benchmarks (CEBs). The CEBs were developed using species sensitivity distributions (SSDs) whenever sufficient toxicity data were available. In the absence of sufficient data, CEBs were defined using the lowest chronic toxicity test value available for species relevant to the Gahcho Kué Project (Project) area. The toxicity database excluded non-resident species, which improved the relevance of the CEBs to the receiving environment of Kennady Lake and the downstream lakes.

The CEBs represent substance concentrations above which changes to aquatic health could occur on the scale of individual organisms. The benchmarks are less conservative (i.e., more realistic) than water quality guidelines, but retain a level of conservatism for the evaluation of population-level effects, which would require concentrations to be higher than the CEBs described herein. Consequently, the CEBs are considered to be conservative thresholds by which potential effects to aquatic health can be assessed. Further detail as to the methods used to derive the CEBs is provided in Appendix 8.IV.

8.9.2.2.2 Indirect Exposure - Changes to Fish Tissue Quality

In addition to assessing potential effects to aquatic health due to direct waterborne exposure, potential effects due to changes in fish tissue quality were assessed. Potential changes to fish tissue concentrations in Kennady Lake and Area 8 at closure were estimated by multiplying predicted maximum concentrations in water by parameter-specific bioaccumulation factors (BAFs). Only those parameters for which toxicological benchmarks could be defined were considered. These parameters, hereafter called substances of interest (SOI), were:

- aluminum
- antimony
- arsenic
- cadmium
- chromium
- copper
- lead
- mercury
- nickel
- selenium
- silver
- vanadium
- zinc

Site-specific BAFs for each SOI were derived for each lake and fish species using water quality concentrations and fish tissue concentrations measured during the baseline sampling programs. The lake- and species-specific BAFs were calculated using the following formula:

$$BAF_{(lake,\ species)} = C_{Fish} \div C_{Water}$$

where:

$BAF_{(lake,\ species)}$ = bioaccumulation factor for a specific lake and fish species

C_{Fish} = concentration of substance “x” in fish (milligrams per kilogram wet weight [mg/kg wet wt])

C_{Water} = concentration of substance "x" in water (mg/L).

The term C_{Water} was set to the median concentration observed in the water quality samples collected from the lake being considered. Given that water quality in the study lakes was similar among years, all available baseline water quality data were pooled and overall median water concentrations were calculated. The term C_{Fish} was similarly set to the median concentration observed in fish muscle tissue samples collected from either Kennady Lake, Lake N16, Kirk Lake, or Lake 410. All non-detectable tissue concentration results were set to the corresponding detection limit, which resulted in conservative multiplication factors.

Bioaccumulation factors were derived based on concentrations of substances measured in muscle tissue of lake trout and round whitefish. Only whole-body concentration data were available for slimy sculpin, and these were not included in BAF derivation based on the following rationale:

- The primary concern in terms of potential effects on fish health is large-bodied fish such as lake trout and round whitefish. These species are abundant in Kennady Lake, form a key component of the lake ecosystem, and are fished for consumption. Slimy sculpin are small-bodied, benthic feeding fish that are not abundant in the study lakes and are not fished. During the baseline sampling program in 2007, sculpin had to be collected from the outlet creeks of the lakes to obtain sufficient sample for tissue analysis.
- Analysis of whole body samples of sculpin unavoidably leads to the inclusion of gut contents in the analysis, and this can give unreliable measurements of the actual concentrations of substances in the tissues of the sculpin. Sculpin are benthic feeding fish that have a relatively high potential to ingest sediment with their prey. Thus, by including gut contents, whole body measurements can result in artificially inflated measurements of metals that are abundant in mineral sediments (e.g., aluminum), due to the inclusion of prey and incidentally-ingested sediment in the gut in the analysis.

The whole-body sculpin tissue concentrations of several metals, including aluminum and several other substances abundant in mineral sediments, were substantially higher than concentrations measured in lake trout and round whitefish (Annex J). The concentrations measured in sculpin whole body analyses are therefore considered most likely to be artefactual (i.e., reflecting the inclusion of sediment and prey in the gut), and not an accurate representation of the accumulation of these substances in fish tissue. Inclusion of the sculpin whole body concentration data in the BAF analysis would result in unrealistic estimates of tissue concentrations in fish. Therefore, the sculpin data were excluded and the BAF analysis was based on lake trout and round whitefish. The lake- and species-specific BAFs were categorized by level of reliability based on the frequency of detections in the water and tissue data. The BAFs calculated from water and tissue concentrations with high detection frequencies were considered the most reliable BAFs, and therefore were selected preferentially over less reliable BAFs. The reliability criteria were:

- If both water and tissue concentrations were frequently detected, then the resulting BAF was considered to be the most reliable;
- If water was detected frequently, but tissue was not, then the resulting BAF was considered to be less reliable, but still an acceptable upper-bound estimate (i.e., likely a conservative over-estimate) for the purposes of this assessment;
- If water was infrequently detected, and tissue was frequently detected, then the resulting BAF was considered less reliable and a potentially lower-bound estimate for the purposes of this assessment; and
- If both water and tissue were infrequently detected, then the resulting BAF was considered to be unreliable and was not used in this assessment.

The BAFs for each SOI used in the indirect exposure assessment are summarized in Table 8.9-3.

Table 8.9-3 Selected Bioaccumulation Factors for the Indirect Exposure Assessment

Substance of Interest	Selected Bioaccumulation Factor	Reliability Category
Aluminum	278	less reliable; upper-bound estimate
Antimony	2729	less reliable; upper-bound estimate
Arsenic	417	less reliable; upper-bound estimate
Cadmium	237	less reliable; lower-bound estimate
Chromium	78	most reliable
Copper	839	most reliable
Lead	80	less reliable; upper-bound estimate
Mercury	9450	less reliable; lower-bound estimate
Nickel	232	most reliable
Selenium	3000	less reliable; lower-bound estimate
Silver	2000	less reliable; upper-bound estimate
Vanadium	95	most reliable
Zinc	379	most reliable

Predicted fish tissue metal concentrations were compared to toxicological benchmarks that have been shown in laboratory studies to be associated with sublethal effects in fish. Jarvinen and Ankley (1999) provide a database linking effects on aquatic organisms and concentrations of inorganic and organic chemicals in various fish tissues. Both acute and chronic effect-endpoints for a range of species and trophic levels are provided in the database. Occasionally, only lethal endpoints were available. A summary of the Jarvinen and Ankley (1999) endpoints that were relevant to the current assessment is provided in Table 8.9-4

Table 8.9-4 Fish Tissue Effects Concentrations

Substance of Interest	Effects Concentration (mg/kg wet weight)	Endpoint	Tissue	Fish, Age/Size
Aluminum	20	survival – reduced	whole body	Atlantic salmon, alevin
	<8	growth – no effect		
	1.15	survival – no effect	muscle	rainbow trout, 171 g
Antimony	9.0	survival – reduced 50%	whole body	rainbow trout, fingerling
	5.0	survival – no effect		
Arsenic	11.2	survival – reduced	carcass	rainbow trout, juvenile
	6.1	survival, growth – no effect		
	3.1	growth – reduced		
Cadmium	2.8	survival, growth – no effect	muscle	rainbow trout, adult
	0.6	reproduction – reduced	muscle	
	0.4	reproduction – no effect	muscle	
Chromium	0.58	survival – no effect	muscle	rainbow trout, 150 to 200 g
Copper	3.4	survival, growth, reproduction – no effect	muscle	brook trout, embryo, adult, juvenile
	0.5	survival – no effect	muscle	rainbow trout, 138 g
Lead	4.0	survival – no effect	carcass	rainbow trout, under-yearlings
	2.5 to 5.1	growth – no effect	whole body	brook trout, embryo – juvenile
Mercury	5.8	survival – no effect growth – reduced	muscle	chum salmon, fry, juvenile
	5.0	growth, survival – no effect	whole body	rainbow trout, juvenile
	0.8	growth – no effect	whole body	fathead minnow, adult
Nickel	118.1	survival – reduced 50%	white muscle	carp, 15 g
	58.0	survival – no effect	white muscle	freshwater carp, 15 g
	0.82	survival – no effect	muscle	rainbow trout, 150 to 200 g
Silver	0.06	survival, growth – no effect	whole body	bluegill, young-of-the-year
	0.003	survival, growth – no effect	carcass	largemouth bass, young-of-the-year
Vanadium	5.33	survival – no effect	carcass	rainbow trout, juvenile
	0.41	growth – reduced		
	0.02	growth – no effect		
Zinc	60	survival, growth – no effect	whole body	Atlantic salmon, juvenile
	4.5	survival, growth – no effect	whole body	brook trout, embryo-larvae

Source: Jarvinen and Ankley (1999).

mg/kg = milligrams per kilogram; < = less than; g = gram; % = percent.

Benchmarks were selected from the Jarvinen and Ankley (1999) database to represent levels beyond which detrimental effects (e.g., reduced growth or reproductive success) may occur. However, for some SOIs, available information was limited to no observed effect concentrations (NOECs). The parameters for which only NOECs were available were arsenic, chromium, copper, lead, mercury, nickel, silver, and zinc. The tissue-based NOECs are similar to most water-based no-effect thresholds in that concentrations less than a NOEC are not considered likely to lead to detrimental effects, whereas the opposite is not necessarily true (i.e., concentrations in excess of NOECs will not necessarily result in detrimental effects). This resulted in benchmarks that were overly conservative estimates of effects thresholds, and predicted fish tissue concentrations were interpreted with this limitation in mind.

Although the Jarvinen and Ankley (1999) database includes information for selenium, the selenium threshold used herein originates from the United States Environmental Protection Agency (US EPA 2004), which represents a more up-to-date assessment of potential effects of selenium on fish health. The threshold derived from the US EPA (2004) data was evaluated by a review of more recent selenium toxicity studies with coldwater fish (Holm et al. 2005, Muscatello et al. 2006, Rudolph et al. 2008, McDonald et al. 2010) and was determined to be an appropriately protective benchmark for fish species that occur in the study area.

8.9.3 Results

8.9.3.1 Effects of Air Emissions to Aquatic Health in the Kennady Lake Watershed

Results of the water quality assessment indicate that dust and metals deposition may result in predicted maximum concentrations of suspended solids and some metals (aluminum, cadmium, chromium, copper, iron, mercury, and silver) that exceed water quality guidelines in some lakes, including both fish-bearing and non-fishing bearing lakes (Section 8.8.3.2). However, the predicted concentrations are likely conservative estimates of the maximum potential concentrations as they reflect the conservative assumptions used in the air quality modeling and mass balance analysis (Section 8.8.1.1.2).

The spatial extent of the dust and metals deposition is expected to be restricted to localized areas within the Project footprint. Most of the deposition will impact the affected lakes during the short period of freshet, when dust deposited to snow enters surface waters. The length of the freshet period is estimated to range from a few days in small lakes to 1 to 2 weeks in Kennady Lake. Therefore, the period of elevated suspended solids and metals in affected lakes is expected to be relatively short. Given the conservatism in the predicted concentrations, and the relatively short duration of the exposure to elevated concentrations, the potential for adverse effects from dust and metals deposition is considered to be low. Follow-up monitoring will be undertaken to confirm this evaluation.

8.9.3.2 Effects of Changes to Water Quality on Aquatic Health in Waterbodies within the Kennady Lake Watershed during Closure

8.9.3.2.1 Direct Waterborne Exposure

Based on the three-step screening process described in Section 8.9.2.2.1, 13 SOPCs were identified in Kennady Lake under the initial closure discharge water quality scenario (Table 8.9-5):

- TDS
- antimony
- barium
- beryllium
- cadmium
- chromium
- cobalt
- copper
- iron
- manganese
- strontium
- uranium
- vanadium

Based on the three-step screening process described in Section 8.9.2.2.1, 12 SOPCs were identified in Kennady Lake under the long-term water quality scenario (Table 8.9-6):

- TDS
- antimony
- barium
- beryllium
- cadmium
- chromium
- cobalt
- copper
- manganese
- strontium
- uranium
- vanadium

Based on the three-step screening process described in Section 8.9.2.2.1, 12 SOPCs were identified in Area 8 under the post-closure scenario (Table 8.9-7):

- TDS
- antimony
- barium
- beryllium
- cadmium
- chromium
- cobalt
- copper
- manganese
- strontium
- uranium
- vanadium

A summary of the SOPCs identified at each assessment point is presented in Table 8.9-8.

Table 8.9-5 Initial Screening Results for Kennady Lake under Initial Closure Discharge Water Quality Scenario

Parameter	Kennady Lake Background Concentrations (Long-term Average) (mg/L)	CCME Freshwater Aquatic Life Guideline (mg/L)[a]	Predicted Maximum Concentration (mg/L)	Screening: Higher than Predicted Background + 10%?	Screening: Higher than CCME Guideline?	Retained as Substance of Potential Concern?
Conventional Parameters						
Total Dissolved Solids	11	-	162	yes	-	**yes**
Total Suspended Solids	<2[b]	5[c]	1.0	no	no	no
Nutrients						
Ammonia as Nitrogen	0.018	4.5[d]	3.1	yes	no	no
Nitrate as Nitrogen	<0.007[b]	2.9	2.9	yes	no	no
Total Metals						
Aluminum	0.0094	0.1[e]	0.071	yes	no	no
Antimony	0.00014	-	0.0021	yes	-	**yes**
Arsenic	0.00013	0.005	0.0025	yes	no	no
Barium	0.0026	-	0.19	yes	-	**yes**
Beryllium	0.000048	-	0.00014	yes	-	**yes**
Boron	0.002	1.5	0.59	yes	no	no
Cadmium	0.00002	0.000032[f]	0.000042	yes	yes	**yes**
Chromium	0.00021	0.001[g]	0.0050	yes	yes	**yes**
Cobalt	0.000085	-	0.00048	yes	-	**yes**
Copper	0.0013	0.002[f]	0.0028	yes	yes	**yes**
Iron	0.042	0.3	0.44	yes	yes	**yes**
Lead	0.000039	0.002[f]	0.00038	yes	no	no
Manganese	0.0091	-	0.056	yes	-	**yes**
Mercury	0.0000066	0.000026	0.000017	yes	no	no
Molybdenum	0.000059	0.073	0.012	yes	no	no
Nickel	0.00048	0.065[f]	0.0031	yes	no	no
Selenium	0.000025	0.001	0.00084	yes	no	no
Silver	0.000043	0.0001	0.000076	yes	no	no
Strontium	0.0082	-	0.19	yes	-	**yes**
Thallium	0.000022	0.0008	0.00019	yes	no	no
Uranium	0.000024	-	0.0022	yes	-	**yes**
Vanadium	0.00021	-	0.0030	yes	-	**yes**
Zinc	0.0028	0.03	0.012	yes	no	no

(a) From CCME (1999).
(b) Median detection limit.
(c) Guideline is dependent on background concentration: predicted concentration must not be more than 5 mg/L higher than the background concentration.
(d) Guideline is dependent on temperature and pH. The value is based on pH = 7.0, temperature = 18°C.
(e) Aluminum guideline is dependent on pH; guideline shown is for pH ≥6.5, which corresponds to expected conditions in Kennady Lake.
(f) Guideline is hardness dependant; value shown based on a maximum predicted hardness of 97 mg/L as calcium carbonate ($CaCO_3$).
(g) Guideline is for chromium (VI), because it is more conservative than the chromium (III) guideline of 0.0089 mg/L.

mg/L = milligrams per litre; % = percent; < = less than; - = no guideline available or predicted concentration was less than the observed maximum background.

Table 8.9-6 Initial Screening Results for Kennady Lake under the Long-term Water Quality Scenario

Parameter	Kennady Lake Background Concentrations (Long-term Average) (mg/L)	CCME Freshwater Aquatic Life Guideline (mg/L)[a]	Predicted Maximum Concentration (mg/L)	Screening		Retained as Substance of Potential Concern?
				Higher than Predicted Background + 10%?	Higher than CCME Guideline?	
Conventional Parameters						
Total Dissolved Solids	11	-	83	yes	-	**yes**
Total Suspended Solids	<2[b]	5[c]	1.0	no	no	no
Nutrients						
Ammonia as Nitrogen	0.018	4.5[d]	0.021	yes	no	no
Nitrate as Nitrogen	<0.007[b]	2.9	0.037	yes	no	no
Total Metals						
Aluminum	0.0094	0.1[e]	0.070	yes	no	no
Antimony	0.00014	-	0.0019	yes	-	**yes**
Arsenic	0.00013	0.005	0.0024	yes	no	no
Barium	0.0026	-	0.19	yes	-	**yes**
Beryllium	0.000048	-	0.00014	yes	-	**yes**
Boron	0.002	1.5	0.59	yes	no	no
Cadmium	0.00002	0.000017[f]	0.000040	yes	yes	**yes**
Chromium	0.00021	0.001[g]	0.0013	yes	yes	**yes**
Cobalt	0.000085	-	0.00027	yes	-	**yes**
Copper	0.0013	0.002[f]	0.0027	yes	yes	**yes**
Iron	0.042	0.3	0.14	yes	no	no
Lead	0.000039	0.001[f]	0.00022	yes	no	no
Manganese	0.0091	-	0.015	yes	-	**yes**
Mercury	0.0000066	0.000026	0.000011	yes	no	no
Molybdenum	0.000059	0.073	0.012	yes	no	no
Nickel	0.00048	0.025[f]	0.0031	yes	no	no
Selenium	0.000025	0.001	0.00025	yes	no	no
Silver	0.000043	0.0001	0.000059	yes	no	no
Strontium	0.0082	-	0.19	yes	-	**yes**
Thallium	0.000022	0.0008	0.000038	yes	no	no
Uranium	0.000024	-	0.00085	yes	-	**yes**
Vanadium	0.00021	-	0.0029	yes	-	**yes**
Zinc	0.0028	0.03	0.0045	yes	no	no

(a) From CCME (1999).

(b) Median detection limit.

(c) Guideline is dependent on background concentration: predicted concentration must not be more than 5 mg/L higher than the background concentration.

(d) Guideline is dependent on temperature and pH. The value is based on pH = 7.0, temperature = 18°C.

(e) Aluminum guideline is dependent on pH; guideline shown is for pH ≥6.5, which corresponds to expected conditions in Kennady Lake.

(f) Guideline is hardness dependant; value shown based on a maximum predicted hardness of 47 mg/L as calcium carbonate ($CaCO_3$).

(g) Guideline is for chromium (VI), because it is more conservative than the chromium (III) guideline of 0.0089 mg/L.

mg/L = milligrams per litre; % = percent; < = less than; - = no guideline available or predicted concentration was less than the observed maximum background.

Table 8.9-7 Initial Screening Results for Area 8 Under Post-closure Scenario

Parameter	Kennady Lake Background Concentrations (Long-term Average) (mg/L)	CCME Freshwater Aquatic Life Guideline (mg/L)[a]	Predicted Maximum Concentration (mg/L)	Screening		Retained as Substance of Potential Concern?
				Higher than Predicted Background + 10%?	Higher than CCME Guideline?	
Conventional Parameters						
Total Dissolved Solids	11	-	94	yes	-	yes
Total Suspended Solids	<2[b]	5[c]	1.0	no	no	no
Nutrients						
Ammonia as Nitrogen	0.018	4.5[d]	1.6	yes	no	no
Nitrate as Nitrogen	<0.007[b]	2.9	1.5	yes	no	no
Total Metals						
Aluminum	0.0094	0.1[e]	0.060	yes	no	no
Antimony	0.00014	-	0.0016	yes	-	yes
Arsenic	0.00013	0.005	0.0020	yes	no	no
Barium	0.0026	-	0.15	yes	-	yes
Beryllium	0.000048	-	0.00013	yes	-	yes
Boron	0.0025	1.5	0.47	yes	no	no
Cadmium	0.00002	0.000019[f]	0.000039	yes	yes	**yes**
Chromium	0.00021	0.001[g]	0.0025	yes	yes	**yes**
Cobalt	0.000085	-	0.00034	yes	-	yes
Copper	0.0013	0.002[f]	0.0026	yes	yes	yes
Iron	0.042	0.3	0.24	yes	no	no
Lead	0.000039	0.001[f]	0.00022	yes	no	no
Manganese	0.0091	-	0.030	yes	-	yes
Mercury	0.0000066	0.000026	0.000013	yes	no	no
Molybdenum	0.000059	0.073	0.0098	yes	no	no
Nickel	0.00048	0.025[f]	0.0026	yes	no	no
Selenium	0.000025	0.001	0.00045	yes	no	no
Silver	0.000043	0.0001	0.000063	yes	no	no
Strontium	0.0082	-	0.15	yes	-	yes
Thallium	0.000022	0.0008	0.000096	yes	no	no
Uranium	0.000024	-	0.0011	yes	-	yes
Vanadium	0.00021	-	0.0024	yes	-	yes
Zinc	0.0028	0.03	0.0078	yes	no	no

(a) From CCME (1999).
(b) Median detection limit.
(c) Guideline is dependent on background concentration: predicted concentration must not be more than 5 mg/L higher than the background concentration.
(d) Guideline is dependent on temperature and pH. The value is based on pH = 7.0, temperature = 18°C.
(e) Aluminum guideline is dependent on pH; guideline shown is for pH ≥6.5, which corresponds to expected conditions in Kennady Lake.
(f) Guideline is hardness dependant; value shown based on a maximum predicted hardness of 54 mg/L as calcium carbonate ($CaCO_3$).
(g) Guideline is for chromium (VI), because it is more conservative than the chromium (III) guideline of 0.0089 mg/L.

mg/L = milligrams per litre; % = percent; < = less than; - = no guideline available or predicted concentration was less than the observed maximum background.

Table 8.9-8 Summary of Substances of Potential Concern Identified in Kennady Lake and Area 8 during Modelled Closure Scenarios

Parameter[a]	Kennady Lake		Area 8
	Initial Closure Discharge Water Quality	Long-term Water Quality	Post-closure Water Quality
Conventional Parameters			
Total Dissolved Solids	√	√	√
Total Suspended Solids			
Nutrients			
Ammonia			
Nitrate			
Total Metals			
Aluminum			
Antimony	√	√	√
Arsenic			
Barium	√	√	√
Beryllium	√	√	√
Boron			
Cadmium	√	√	√
Chromium	√	√	√
Cobalt	√	√	√
Copper	√	√	√
Iron	√		
Lead			
Manganese	√	√	√
Mercury			
Molybdenum			
Nickel			
Selenium			
Silver			
Strontium	√	√	√
Thallium			
Uranium	√	√	√
Vanadium	√	√	√
Zinc			

[a] Checkmark (√) indicates that the substance in question was identified as a substance of potential concern (SOPC).

For the direct waterborne exposure assessment, CEBs were derived for the SOPCs. For TDS, the CEB took the form of a range of concentrations, which were derived based on a review of the applicable literature. For the remaining SOPCs, single point benchmarks were identified, following the approach outlined in Appendix 8.IV. The predicted water concentrations summarized in Tables 8.9-5 through 8.9-7 were compared to the CEBs to conservatively evaluate the potential for adverse effects to aquatic health. The results of these comparisons are discussed below, beginning with TDS.

Total Dissolved Solids

Total dissolved solids was identified as an SOPC in Kennady Lake and Area 8, because of a projected increase in TDS concentrations over those that currently occur. The largest predicted increase occurs in Kennady Lake during the initial closure discharge phase of the Project, when TDS levels are predicted to increase from an existing maximum concentration of about 11 mg/L to a peak of 162 mg/L (Table 8.9-4). Long-term water quality in Kennady Lake and maximum post-closure TDS concentrations in Area 8 will be similar at maximum concentrations of 83 and 94 mg/L, respectively.

Total dissolved solids concentration (TDS) is a measurement of inorganic salts (e.g., sodium, potassium, calcium, magnesium, chloride, sulphate, and bicarbonate), organic matter, and other dissolved materials in water (Weber-Scannell and Duffy 2007). Toxicity can be caused by an increase in salinity, changes in ionic composition of the waters, or through toxicity of individual ions (Weber-Scannell and Duffy 2007). Sensitivity to TDS varies by species and is dependent on both the absolute concentration of all of the major ions contained in solution (effectively the absolute TDS concentration) as well as their relative abundance. In general, Mount et al. (1997) found that relative ion toxicity to freshwater species was potassium > bicarbonate = magnesium > chloride > sulphate, whereas calcium and sodium did not cause significant toxicity. However, ratios of particular TDS constituents, such as the ratio of calcium to sodium, may affect toxicity (Goodfellow et al. 2000). Species sensitivity may also vary with life stage; for example, fish embryos appear to be more sensitive if exposed before fertilization as opposed to after fertilization (Weber-Scannell and Duffy 2007). There is a very wide range of TDS and major ion concentrations in natural waterbodies. As a result of the significant variations in sensitivity of aquatic organisms and large range of concentrations in natural waterbodies, water quality guidelines have not been established in Canada for TDS or most major ions.

Background TDS in Kennady Lake is a mixture of calcium, chloride, magnesium, potassium, sodium, and sulphate, with calcium being slightly more abundant than the other ions. At the start of the post-closure phase, the ionic composition of the waters in Kennady Lake will be dominated by chloride, followed by calcium. During the post-closure phase, the three main constituents contributing to TDS in Kennady Lake and Area 8 will be chloride, sulphate, and calcium.

Toxicity data on the effects of TDS on freshwater species indicate that aquatic life in Kennady Lake or Area 8 will be largely unaffected by the projected increase in salinity. Beadle (1969), as cited in Bierhuizen and Prepas (1985), noted that freshwater species tend to be routinely found in waters with TDS levels of less than 1,000 mg/L, whereas they start to disappear when TDS levels exceed 3,000 mg/L (Hammer et al. 1975).

Adverse effects to fish are not expected at the predicted TDS concentrations in Kennady Lake and Area 8. Optimal habitat for northern pike (*Esox lucius*), one of the fish species present in Kennady Lake, includes TDS concentrations in the range of 80 to 800 mg/L (US Fish and Wildlife Service 1982). Northern pike and other freshwater fish species can be found in environments with higher TDS concentrations. For example, Buffalo Lake, which is located near Stettler, Alberta, has a moderate salinity (i.e., TDS concentrations around 1,500 mg/L) and contains northern pike, along with white suckers (*Catostomus commersonii*) and burbot (*Lota lota*) (University of Alberta 2008).

Most of the laboratory studies with fish embryos and swim-up fry have been conducted with TDS mixtures dominated by calcium and sulphate (e.g., Chapman et al. 2000, Stekoll et al. 2003, Brix et al. 2010). There were no adverse effects on early life stages of rainbow trout (*Oncorhynchus mykiss*) after seven days exposure to 2,000 mg/L TDS (Chapman et al. 2000). Brix et al. (2010) found no significant effects of elevated TDS on fertilization success and reported a 72-h EC20 of >2,782 mg/L for Arctic grayling (*Thymallus arcticus*) and a 24-h EC20 of >1,817 mg/L for Dolly Varden (*Salvelinus malma*). However, embryo water absorption was affected in 14-h exposures, with LOECs of 1,402 mg/L for Arctic grayling and 964 mg/L for Dolly Varden. Stekoll et al. (2003) found that salmonid embryos were most sensitive to TDS when exposed during fertilization: the 24-h LOECs ranged from 250 to 1,875 mg/L. Brannock et al. (2002) found that calcium chloride and sodium sulphate had the most detrimental effect on fertilization rates in king salmon (*Oncorhynchus tshawytscha*) and pink salmon (*Oncorhynchus gorbuscha*). As predicted closure concentrations in Kennady Lake and Area 8 are below these levels, negligible effects to fish health are expected.

Potential effects to pelagic invertebrates also are not expected to occur. Most of the TDS toxicity data are from studies with cladocerans, such as *Ceriodaphnia dubia*, and *Daphnia magna*, because these species are common laboratory test organisms. Predicted ion concentrations and TDS levels are lower than toxic thresholds identified by Cowgill and Milazzo (1990) for these species (i.e., 1,200 mg/L sodium chloride [NaCl]). Predicted concentrations are also lower than the 48-h LC50s reported by Mount et al. (1997) for *Ceriodaphnia dubia* for solutions containing a mixture of ions, including sodium, sulphate, bicarbonate, calcium, chloride and magnesium (i.e., 1,510 to greater than 5,700 mg/L). Although neither of these cladocerans may be present in Kennady Lake, they are recognized as being among the most sensitive invertebrates for a wide range of substances. For example, *Daphnia magna* and *Ceriodaphnia dubia* are more sensitive to calcium chloride than copepods (Baudouin and

Scoppa 1972). As the predicted TDS and major ion concentrations in Kennady Lake and Area 8 are expected to be below the levels associated with effects in the literature, negligible effects to pelagic invertebrates are expected.

Toxicity data specific to benthic invertebrates indicate that benthic invertebrate populations in Kennady Lake or Area 8 will be largely unaffected by the projected increase in salinity. Chapman et al. (2000) reported a 10-d LOEC of 1,750 mg/L for survival of *Chironomus tentans* exposed to synthetic TDS mixtures (TDS consisted mainly of calcium sulphate). Hynes (1990) described no effects on the benthic invertebrate community of a lake in northern Saskatchewan receiving treated uranium mill effluent where TDS levels increased from 76 to 2,700 mg/L. The major ions primarily responsible for this increase were calcium, sodium, chloride, and sulphate. No statistically significant decreases in abundance or species diversity were observed in the affected lake relative to reference conditions.

Based on the above, predicted changes to major ion levels and TDS concentrations in Kennady Lake and Area 8 are expected to have a negligible effect on aquatic health.

Remaining Parameters

In addition to TDS, 12 other SOPCs were identified in one or more of the assessment scenarios for direct waterborne exposure:

- antimony
- barium
- beryllium
- cadmium
- chromium
- cobalt
- copper
- iron
- manganese
- strontium
- uranium
- vanadium

During closure, maximum concentrations of total antimony, barium, beryllium, cadmium, chromium, cobalt, manganese, uranium, and vanadium are predicted to remain below the CEB identified for each substance, as shown in Table 8.9-9. As a result, the predicted increases in the concentrations of these nine substances are expected to have a negligible effect on aquatic health in Kennady Lake and Area 8 under closure conditions.

Maximum concentrations of the remaining three SOPCs, which include total copper, iron, and strontium are projected to be above their respective benchmarks at one or more points during closure (Table 8.9-9). The environmental relevance of these predictions is discussed below.

Table 8.9-9 Comparison of Maximum Concentrations to Chronic Effects Benchmarks for Selected Substances of Potential Concern

Substance of Potential Concern	Chronic Effect Benchmark (mg/L) [a]	Kennady Lake		Area 8
		Maximum Concentration in Initial Closure Discharge Water Quality (mg/L)	Maximum Concentration in Long-Term Water Quality (mg/L)	Maximum Concentration in Post-closure Water Quality (mg/L)
Antimony	0.157	0.0021	0.0019	0.0016
Barium	5.8	0.19	0.19	0.15
Beryllium	0.0053	0.00014	0.00014	0.00013
Cadmium	0.00026 [b]	0.000042	0.000040	0.000039
Chromium	0.0083 [c]	0.0050	0.0013	0.0025
Cobalt	0.0093	0.00048	0.00027	0.00034
Copper	0.002	**0.0028 (0.0022)** [d]	**0.0027 (0.0021)**	**0.0026** (0.0019)
Iron	0.3	**0.44** (0.36)	- [e]	-
Manganese	1.455	0.056	0.015	0.031
Strontium	0.049 [f]	**0.19 (0.19)**	**0.19 (0.19)**	**0.15 (0.15)**
Uranium	0.015	0.0022	0.00085	0.0011
Vanadium	0.0338	0.0030	0.0029	0.0024

Bolded concentrations are greater than corresponding chronic effects benchmark.

(a) Developed as outlined in Appendix 8.IV.

(b) The CEB for cadmium varies with hardness; the reported value is based on a hardness of 47 mg/L, which is the lowest predicted hardness of the three scenarios presented in this table.

(c) The CEB for chromium varies with speciation; the CEB for chromium (VI) is 0.0083 mg/L whereas the CEB for chromium (III) is 0.089 mg/L. Although it is anticipated that most chromium will be present as chromium (III) (Section 8.8.4.1.1), the more conservative CEB was used in the current assessment.

(d) Dissolved concentrations are shown in parentheses.

(e) - = parameter was not identified as a substance of potential concern (SOPC) at the scenario indicated.

(f) The available data did not support derivation of a CEB from a species sensitivity distribution. The adopted value of 0.049 mg/L is the lowest reported effects concentration, which is several orders of magnitude lower than the next-lowest value and therefore likely to be highly conservative.

mg/L = milligrams per litre.

Copper is a component of Kennady Lake bed sediment. Another source of copper to Kennady Lake as a result of the Project is from the PK, which will either be deposited in the Fine PKC Facility, or placed in the mined-out Hearne open pit. Predicted copper concentrations in Kennady Lake and Area 8 marginally exceed the CEB (Table 8.9-9).

Despite the predicted exceedances of the CEB, the potential for copper to cause adverse effects to aquatic life in Kennady Lake and Area 8 is considered to be low. The CEB for copper is based on the CCME guideline, which is intended to be conservative and protective of the most sensitive species. The predicted concentrations summarized in Table 8.9-9 are only slightly greater than the CEB, indicating the possibility (but not necessarily the likelihood) of effects to the most sensitive species. However, the CCME guideline does not consider the potential for other water quality characteristics (e.g., dissolved organic carbon) to reduce bioavailability and ameliorate copper toxicity. Furthermore, the CCME guideline is based on toxicity tests with naive organisms, whereas organisms inhabiting Kennady Lake potentially have some degree of acclimation or adaptation to copper, given that baseline sediment copper concentrations exceed the CCME interim sediment quality guideline (Section 8.3.6.2.1). Given the small magnitude by which predicted maximum concentrations exceed the CEB, and given the potential for ameliorating factors discussed above, the potential for adverse effects from copper is considered to be low. Follow-up monitoring will be undertaken to confirm this evaluation.

Iron is a common constituent of bed sediment in Kennady Lake, as outlined in Section 8.3.6.2.1 (Table 8.3-22). Bed sediment, entrained in site runoff and carried into the water management pond, is likely a source of iron in the Kennady Lake initial closure discharge scenario. Another source of iron in the predicted water quality is from groundwater. Most of the iron is predicted to exist in dissolved form (Table 8.9-9), and the dissolved concentrations of iron are predicted to be slightly above the corresponding CEB (Table 8.9-9).

The potential for iron to cause adverse effects to aquatic life in Kennady Lake is considered to be low. The CEB for iron is based on the CCME guideline, which is intended to be conservative and protective of the most sensitive species. The predicted concentrations summarized in Table 8.9-8 are only slightly greater than the CEB, indicating the possibility (but not necessarily the likelihood) of effects to the most sensitive species. As summarized in Section 8.IV.2.7, iron concentrations similar to the CEB have been reported by some authors to elicit sublethal effects on cladocerans (Dave 1984). However, other authors have reported effects thresholds for the same species more than an order of magnitude higher than the CEB (Biesinger and Christensen 1972). Lethal effects on cladocerans and effects on fish and other taxa have only been reported at much higher iron concentrations, greater than the CEB and greater than all predicted iron concentrations in Kennady Lake. Thus, the predicted iron concentrations summarized in Table 8.9-9 are not expected to result in adverse effects to aquatic life.

The source of strontium in Kennady Lake is likely from the PK and process water. Strontium is projected to be higher than the CEB in both of the Kennady Lake closure scenarios and in the Area 8 closure scenario (Table 8.9-9). However, the CEB is very low and likely highly conservative, and the actual likelihood of adverse effects to aquatic life is therefore highly uncertain. The CEB was based on a 28-d LC10 with rainbow trout embryos (Appendix 8.IV, Table IV-8) reported by Birge et al. (1979). This value was several orders of magnitude lower than any other reported toxicity datum, including studies with rainbow trout fry and other fish species. Given the high level of uncertainty in the toxicity data reported by Birge et al. (1979), and given that the maximum predicted strontium concentrations in Kennady Lake are orders of magnitude lower than all other effects concentrations in the toxicity dataset, the potential for adverse effects from strontium is considered likely to be low.

8.9.3.2.2 Indirect Exposure - Changes to Fish Tissue Quality

Predicted fish tissue concentrations in Kennady Lake and Area 8 are above toxicological benchmarks for only one SOI: silver (Tables 8.9-10 to 8.9.12). All predicted concentrations for other SOIs were below their respective tissue benchmarks.

The predicted silver concentrations in fish from all three scenarios ranged from 0.12 to 0.15 milligrams per kilogram wet weight (mg/kg ww) (Tables 8.9-10 to 8.9-12), which are higher than the toxicological benchmark of 0.06 mg/kg ww. The benchmark was based on a 180-d NOEC for survival and growth of bluegill (*Lepomis macrochirus*). A low-effect tissue threshold could not be found for silver, either in Jarvinen and Ankley (1999) or during a literature search for concentrations of silver in muscle or whole body. Therefore, the selected tissue benchmark, which is based on a no-effect threshold, is likely a highly conservative basis for assessing the potential for predicted silver concentrations to cause effects to fish.

The predicted silver concentrations are similar to the maximum baseline tissue concentration in the dataset used to derive the silver BAF. The maximum baseline tissue concentration was 0.09 mg/kg ww. Therefore, fish tissue silver concentrations are predicted to increase only marginally above baseline conditions as a result of the Project. Given the modest predicted increase, and given that both baseline and predicted tissue concentrations only marginally exceed the available no-effects benchmark, the potential for the predicted silver concentration to cause effects to fish is concluded to be low.

Based on the above results, changes to concentrations of all substances considered in this assessment are predicted to result in negligible effects to aquatic health in Kennady Lake.

Table 8.9-10 Predicted Metal Concentrations in Fish Tissues in Kennady Lake under Initial Closure Discharge Water Quality Scenario

Metal	Predicted Maximum Concentration (mg/L)	Bioaccumulation Factor	Estimated Fish Tissue Concentrations (mg/kg ww) [(a)]	Toxicological Benchmark (mg/kg ww)[(b)]
Aluminum	0.071	278	19.7	20
Antimony	0.0021	2729	5.8	9
Arsenic	0.0025	417	1.0	3.1
Cadmium	0.000042	237	0.0098	0.6
Chromium	0.0050	78	0.39	0.58
Copper	0.0028	839	2.4	3.4
Lead	0.00038	80	0.030	4.0
Mercury	0.000017	9450	0.16	0.8
Nickel	0.0031	232	0.72	0.82
Selenium	0.00084	3000	2.5	2.58
Silver	0.000076	2000	**0.15**	0.06
Vanadium	0.0030	95	0.28	0.41
Zinc	0.012	379	4.6	60

(a) **Bolded** estimated fish tissue concentrations are greater than corresponding toxicological benchmark.

(b) Benchmarks originate from Jarvinen and Ankley (1999), with the exception of selenium; the selenium benchmark is based on data contained in US EPA (2004) expressed as wet weight assuming a moisture content of 76%.

mg/L = milligrams per litre; mg/kg ww = milligrams per kilogram wet weight.

Table 8.9-11 Predicted Metal Concentrations in Fish Tissues in Kennady Lake under Long-term Water Quality Scenario

Metal	Predicted Maximum Concentration (mg/L)	Bioaccumulation Factor	Estimated Fish Tissue Concentrations (mg/kg ww) [(a)]	Toxicological Benchmark (mg/kg ww) [(b)]
Aluminum	0.070	278	19.6	20
Antimony	0.0019	2729	5.2	9
Arsenic	0.0024	417	1.0	3.1
Cadmium	0.000040	237	0.0096	0.6
Chromium	0.0013	78	0.10	0.58
Copper	0.0027	839	2.3	3.4
Lead	0.00022	80	0.018	4.0
Mercury	0.000011	9450	0.10	0.8
Nickel	0.0031	232	0.71	0.82
Selenium	0.00025	3000	0.75	2.58
Silver	0.000059	2000	**0.12**	0.06
Vanadium	0.0029	95	0.27	0.41
Zinc	0.0045	379	1.7	60

(a) **Bolded** estimated fish tissue concentrations are greater than corresponding toxicological benchmark.

(b) Benchmarks originate from Jarvinen and Ankley (1999), with the exception of selenium; the selenium benchmark is based on data contained in US EPA (2004) expressed as wet weight assuming a moisture content of 76%.

mg/L = milligrams per litre; mg/kg ww = milligrams per kilogram wet weight.

Table 8.9-12 Predicted Metal Concentrations in Fish Tissues in Area 8 under Post-closure Water Quality Scenario

Metal	Predicted Concentration (mg/L)	Bioaccumulation Factor	Estimated Fish Tissue Concentrations (mg/kg ww) [(a)]	Toxicological Benchmark (mg/kg ww) [(b)]
Aluminum	0.060	278	17	20
Antimony	0.0016	2729	4.4	9
Arsenic	0.0020	417	0.83	3.1
Cadmium	0.000039	237	0.0093	0.6
Chromium	0.0025	78	0.19	0.58
Copper	0.0026	839	2.2	3.4
Lead	0.00022	80	0.018	4.0
Mercury	0.000013	9450	0.12	0.8
Nickel	0.0026	232	0.61	0.82
Selenium	0.00045	3000	1.4	2.58
Silver	0.000063	2000	**0.13**	0.06
Vanadium	0.0024	95	0.23	0.41
Zinc	0.0078	379	2.9	60

[(a)] **Bolded** estimated fish tissue concentrations are greater than corresponding toxicological benchmark.

[(b)] Benchmarks originate from Jarvinen and Ankley (1999), with the exception of selenium; the selenium benchmark is based on data contained in US EPA (2004) expressed as wet weight assuming a moisture content of 76%.

mg/L = milligrams per litre; mg/kg ww = milligrams per kilogram wet weight; < = less than.

8.9.4 Sources of Uncertainty

Key sources of uncertainty in this aquatic health assessment were the data used to estimate exposure and effects.

The predicted water concentrations are a source of uncertainty in this aquatic health assessment and Section 8.8 outlines the assumptions used in the water quality modelling. To address this uncertainty, maximum predicted water concentrations were used as conservative estimates of the exposure concentrations for aquatic life in the Kennady Lake watershed during the post-closure period.

The predicted tissue concentrations are a source of uncertainty in this aquatic health assessment. The predicted tissue concentrations were derived from predicted water concentrations and BAFs derived using baseline conditions. To address this uncertainty, maximum predicted water concentrations and the highest BAF for each SOI was used to calculate tissue concentrations, which provided a conservative estimate of predicted tissue concentrations.

A source of uncertainty in the effects assessment was that the potential for the predicted water concentrations to cause adverse effects on aquatic life in Kennady Lake could not be assessed with site-specific toxicity data. There are no toxicity data for populations of aquatic life in the Kennady Lake watershed and toxicity data from the scientific literature were used as surrogates. In general, these toxicity data were based on studies with naïve laboratory organisms tested under optimal culture conditions. Therefore, the use of literature-based data is a conservative approach to address this source of uncertainty. In the direct waterborne assessment, either the estimated hazard concentration above which 5% of the species would be affected or the lowest chronic toxicity value was used as the CEB. In the fish tissue quality assessment, the lowest tissue concentration related to an effect from waterborne exposure was used to assess effects. Finally, individual-level effects were used to judge the potential of effects on populations. These approaches provided conservatism to the effects assessment.

8.10 EFFECTS TO FISH AND FISH HABITAT

This section assesses the potential for effects to fish and fish habitat in Kennady Lake and in small lakes and streams in the Kennady Lake watershed resulting from physical changes, and changes to water quantity and quality. Summaries of the valid pathways for effects to fish and fish habitat are presented in Table 8.10-1 for construction and operations, and in Table 8.10-2 for closure.

The assessment of effects to water quality were assessed in Section 8.8 and resulting effects to fish health were assessed in Section 8.9; therefore, only conclusions of the aquatic health assessment are presented herein under Section 8.10.4.3.3. The recovery of the aquatic ecosystem in Kennady Lake for fish and lower trophic levels (i.e., phytoplankton and zooplankton) is addressed in Section 8.11.

Sections 8.10.1 and 8.10.2 provide an overview of the methodology used to analyze the effects to fish and fish habitat in the Kennady Lake and its watershed during construction and operations, and closure, respectively. The discussion of analysis results for construction and operations is provided in Section 8.10.3 and for closure and post-closure in Section 8.10.4.

For the purposes of the assessment, fish habitat is defined as the area required by fish for spawning, nursery, rearing, food supply (e.g., benthic invertebrates), overwintering, and migration.

Table 8.10-1 Valid Pathways for Effects to Fish and Fish Habitat in Kennady Lake and the Kennady Lake Watershed – Constructions and Operation

Project Activity	Pathway	Effects Statement
Project footprint (e.g., dykes, mine pits, Coarse PK Pile and Fine PKC Facility, mine rock piles, access roads, mine plant, airstrip)	project development in the Kennady Lake watershed will result in the loss of fish habitat	Effects of Project construction and operations activities to fish and fish habitat in Kennady Lake, and streams and lakes within the Kennady Lake watershed
Dewatering of Kennady Lake	dewatering of much of Kennady Lake and other small lakes may cause mortality and spoiling of fish, temporary loss in productive capacity, and the alteration of flows, water levels, and channel/bank stability in Area 8	
Isolation and diversion of upper Kennady Lake watersheds	change of flow paths and construction of retention and diversion dykes in the A, B, D and E watersheds may result in loss of stream habitat, alteration of water levels and lake areas, shoreline erosion, re-suspension of sediments and sedimentation, and changes to lower trophic levels, fish communities, and migration	
Construction and mining activity (air emissions)	deposition of dust and particulate matter may cause increases in suspended sediment, and changes to aquatic health	

Table 8.10-2 Valid Pathways for Effects to Fish and Fish Habitat in Kennady Lake and the Kennady Lake Watershed – Closure and Post-Closure

Project Activity	Pathway	Effects Statement
Removal and reclamation of Project infrastructure in Kennady Lake basin	development of fish habitat compensation works	Effects of Project closure and post-closure activities to fish and fish habitat in Kennady Lake, and streams and lakes within the Kennady Lake watershed
Removal of the temporary diversion dykes in the B, D and E watersheds	change of flow paths in the B, D, and E watersheds may result in alteration of water levels and lake areas, changes to lower trophic levels, fish communities and migration	
Refilling of Kennady Lake	continued isolation of Area 8 during refilling	
Post-Closure Activities	changes to nutrient levels may result in changes to lower trophic communities, dissolved oxygen levels, fish habitat, and fish communities	
	changes to aquatic health may affect fish populations and abundance	

8.10.1 Effects Analysis Methods – Construction and Operation

8.10.1.1 Effects of Project Footprint on Fish Habitat

Changes to fish habitat will occur in Kennady Lake and the Kennady Lake watershed due to the development of the Project, e.g., excavation of the mine pits, placement of mine rock, placement of PK, dykes, and other construction activities. The affected habitat areas include portions of Kennady Lake and adjacent lakes within the Kennady Lake watershed that will be permanently lost, portions that will be physically altered after dewatering and later submerged in the refilled Kennady Lake, and portions that will be dewatered (or partially dewatered) but not otherwise physically altered before being submerged in the refilled Kennady Lake. The methods for quantification included the following steps:

- Habitat area determination;
- Habitat suitability determination; and
- Calculation of Habitat Units.

A brief summary is provided below; more details can be found in the Conceptual Compensation Plan (CCP) (Section 3, Appendix 3.II).

The areal quantity of fish habitat permanently lost, physically altered, or dewatered as a result of the Project was determined using a Geographic Information System (GIS) to overlay the Project footprint over habitat classification maps of the affected waterbodies. Habitat was classified into categories of substrate type, gradient and depth. The area of each habitat category within the Project footprint was digitized using GIS for each waterbody and quantified in hectares. The area of the watercourse affected was determined by multiplying the length of each watercourse segment by an assumed width for permanently affected watercourses (3 m). Kennady Lake tributary streams are generally small and less than 3 m wide (Annex J).

The suitability of fish habitat permanently lost, physically altered or dewatered by the Project was quantified using a modified Habitat Evaluation Procedure (HEP). With a HEP approach, habitat suitability is assigned to discrete habitat types using Habitat Suitability Index (HSI) models developed for fish species known or assumed to be present in affected areas. The HSI models were used to quantify the suitability of habitat categories for various life-history stages, and for each fish species present on a scale of 0 (unsuitable) to 1 (optimal). Habitat suitabilities were determined for all permanently lost or affected waterbodies and for the eight fish species known to occur in Project area (lake trout, round whitefish, Arctic grayling, northern pike, burbot, lake chub, slimy sculpin, and ninespine stickleback).

The area and suitability of fish habitat permanently lost, physically altered, or dewatered by the Project were integrated into a single, dimensionless unit called a Habitat Unit (HU). For each species, HUs permanently lost or altered were calculated as the product of the area lost for each habitat category and the suitability of that habitat category for each life-history stage. For each permanently lost or altered waterbody, the HUs are then summed across all habitat categories and species life-history stages to calculate the total HUs lost for a species in a given waterbody.

8.10.1.2 Effects of Kennady Lake Dewatering

Effects of dewatering the main basins of Kennady Lake during mine operations included the direct effects of dewatering activities on the fish population of Kennady Lake, the temporary loss of fish habitat while Kennady Lake is dewatered, and the effects of the dewatering discharge on flows, water levels, and channel/bank stability in Area 8. The effects of the dewatering discharge on fish and fish habitat downstream of Area 8 are discussed in Section 9.10 (Key Line of Inquiry: Downstream Water Effects). The effects of isolation of Area 8 are discussed in a separate section (Section 8.10.1.4).

The quantification of changes to water levels in Area 8 resulting from the diversions is based on the data and results presented in the Effects to Water Quantity section (Section 8.7). The effects on fish and fish habitat were assessed qualitatively, taking into account the fish species present, their habitat use, and life history requirements.

8.10.1.3 Effects of Diversions

The quantification of changes to streamflows and water levels resulting from the diversions is based on the data and results presented in the Effects to Water Quantity section (Section 8.7). Effects to fish and fish habitat in lakes from these changes were assessed by considering the amount and type of habitat in the nearshore areas that will be flooded and the effects to fish based on the use of these habitat types by different life stages of fish present. Habitat use was based on results of baseline investigations and from the published literature. Effects in streams were assessed by calculating the amount of habitat that will be temporarily or permanently lost downstream of dykes and the known use of these streams by different fish species for spawning and rearing or as migration routes.

The effects of shoreline erosion and sedimentation on fish and fish habitat in the diversion lakes were assessed qualitatively following a review of the Effects to Water Quantity section (Section 8.7). Effects on lower trophic levels were assessed qualitatively, taking into account the above information.

Effects on fish migrations and communities in the diverted watersheds were assessed qualitatively, through consideration of the fish species present in each watershed and their habitat use, as well as their life history requirements.

8.10.1.4 Effects of Isolation on Fish and Fish Habitat in Area 8

The effects of isolation of Area 8 on fish migrations and communities in Area 8 were assessed qualitatively, taking into account the fish species present in Area 8 and their habitat use and life history requirements. The changes to flows downstream of Area 8 were quantified in the Effects to Water Quantity section for downstream effects (Section 9.7) and discussed in more detail in Section 9.10.

8.10.1.5 Effects of Dust Deposition on Fish and Fish Habitat

Windborne dust from Project facilities and exposed lake bed sediments, and air emissions from Project facilities, may result in increased deposition of dust in the surrounding area. Changes in total suspended solids (TSS) in lake water from deposition on the lake surface and within the watershed, for selected lakes in the Kennady Lake watershed, were quantified in the Effects to Water Quality section (Section 8.8.1.1). Predicted changes in TSS in local lakes are considered to be conservative (high) estimates of the maximum potential changes that could occur during construction and operations.

To provide an indication of the potential effects of increased TSS on fish in these waterbodies, the Newcombe and Jensen (1996) dose-response relationship was applied. This relationship estimates the magnitude of adverse effect expected when fish are exposed to a given concentration of sediment over a given period. Their dose-response relationship generated a severity of effect (SEV) value ranging from 0 to 14. An SEV value of zero implied no effect. SEV values of one to three indicated behavioural changes are expected, four to eight indicated sublethal effects ranging from increased respiration and coughing rates to major physiological stress. Lethal and paralethal effects are expected with SEV values of 9 to 14.

Potential effects to aquatic health from dust and metals deposition were evaluated in the Effects to Aquatic Health section (Section 8.9). The results of the Aquatic Health assessment were then used to describe and assess changes that relate to fish and fish habitat (i.e., fish populations and communities). A discussion of the methods, models, and assumptions used in the Water Quality and Aquatic Health assessments can be found in Sections 8.8 and 8.9.

8.10.2 Effects Analysis Methods – Closure and Post-closure

Areas 3 to 7 of Kennady Lake will remain a closed-circuited system until completion of refilling and breaching of Dyke A at the end of closure; during closure, the effects analysis for fish and fish habitat only includes Area 8 of Kennady Lake, as well as other lakes and streams in the Kennady Lake watershed. Post-closure includes the period when water quality is restored, the refilled Areas 3 to 7 are reconnected to Area 8, the natural flow path is re-established, and fish passage is resumed.

8.10.2.1 Effects of Habitat Enhancement to Fish and Fish Habitat

Compensation options have been developed and evaluated for the Project (see CCP, Section 3, Appendix 3.II). The methods for quantification of the habitat gains include the following steps:

- Preliminary Habitat Quantification;
- Planned Detailed Habitat Quantification;
- Habitat Area Determination;
- Habitat Suitability Determination; and
- Calculation of Habitat Units.

A brief summary is provided below; more details can be found in the CCP (Section 3, Appendix 3.II). Preliminary estimates of habitat gains potentially achieved from the compensation options under consideration were quantified using GIS. The footprint of each compensation option was overlaid on maps of the project area that included bathymetry of lakes in the Project area.

Detailed quantification of habitat gains potentially achieved by the selected compensation options will be included in the detailed compensation plan that is to be completed in 2011. The general strategy for quantification is equivalent to the approach taken for quantifying permanently lost, physically altered, or dewatered habitats (Section 8.10.1.1).

8.10.2.2 Effects of Rediverting B, D, E Watersheds to Kennady Lake

The quantification of changes to streamflows and water levels at closure is based on the data and results presented in the Effects to Water Quantity section (Section 8.7). Effects to fish and fish habitat in lakes from these changes were assessed qualitatively through understanding the amount and type of habitat in the nearshore areas that will have lowered water levels and estimating the effects to fish based on the use of these habitat types by different life stages of fish present. Habitat use was based on results of baseline investigations and from the published literature. Effects in streams were assessed by evaluating the potential use of these streams by different fish species for spawning and rearing or as migration routes. The effects of shoreline erosion and sedimentation on fish and fish habitat in the diversion lakes were assessed qualitatively following a review of the Effects to Water Quantity section (Section 8.7). Effects on lower trophic levels were assessed qualitatively, taking into account the above information.

Effects on fish migrations and communities in the diverted watersheds were assessed qualitatively, through consideration of the fish species present in each watershed and their habitat use, as well as their life history requirements.

8.10.2.3 Effects of Continued Isolation of Area 8 during Refilling of Kennady Lake

The effects on fish and fish habitat of the continued isolation of Area 8 were evaluated as described for operations in Section 8.10.1.4 above.

8.10.2.4 Effects of Changes in Nutrient Levels in the Refilled Kennady Lake

As discussed in the water quality assessment (Section 8.8.4.1), concentrations of phosphorus are projected to increase in post-closure. The predicted increases result from runoff waters coming into contact with the mine rock piles, Coarse PK Pile and the Fine PKC Facility.

The modelled phosphorus projections were developed assuming contact of seepage flows with materials located in the mine rock piles, the Coarse PK Pile and the Fine PKC Facility, including mitigation associated with the fine PK deposit in the Fine PKC Facility (Section 8.8.4.1.1).

The analysis of potential effects related to predicted changes in nutrient levels considered the following components of fish and fish habitat:

- lower trophic communities, including phytoplankton, zooplankton, and benthic invertebrates;
- fish production;
- changes to physical habitat, including the availability of spawning habitat and dissolved oxygen levels; and
- fish community structure.

Effects on lower trophic levels and fish and fish habitat from changes in nutrient concentrations were predicted using qualitative methods, relying largely on trophic classification of aquatic ecosystems based on nutrient concentrations (CCME 2004; Environment Canada 2004), experience from effects monitoring at operating northern diamond mines, and the scientific literature on the effects of nutrient enrichment on lakes.

Quantitative relationships between the physical/chemical features of sub-arctic lakes and lower trophic community characteristics are not available; however, the relationship between nutrient concentrations and aquatic productivity has been well studied (Wetzel 2001; Environment Canada 2004). Several studies (commonly referred to as “fertilization” studies or experiments) have documented changes in plankton and benthic community structure in sub-arctic and northern temperate lakes in response to nutrient additions (Morgan 1966; Smith 1969; Welch et al. 1988; Hershey 1992; Jorgenson et al. 1992; Clarke et al. 1997; Johnston et al. 1999). Aquatic effects monitoring programs at existing diamond mines in the NWT have also reported changes in lower trophic community characteristics with increasing levels of nutrients (De Beers 2010; Golder 2010), which is directly applicable to potential effects in Kennady Lake.

A qualitative assessment of effects related to closure and post-closure was also completed for lower trophic communities in Kennady Lake. That assessment used the results of the assessments of effects to water quantity (Section 8.7) and water quality (Section 8.8), and results of a literature review on the recovery of aquatic ecosystems following drawdown and establishment of aquatic ecosystems in newly-created reservoirs (see Section 8.11).

An estimate of winter oxygen demand for Kennady Lake was calculated in the Effects to Water Quality assessment (Section 8.8.4.1) based on an empirical relationship of phosphorus concentration and annual rate of primary productivity. Effects to fish habitat from potential changes to summer dissolved oxygen in Kennady Lake were assessed qualitatively.

Effects of changes in lower trophic communities and fish and fish habitat due to changes in nutrient levels and trophic status were qualitatively assessed based on published literature regarding trophic interactions, food web complexity, and known effects of nutrient additions on fish communities in sub-arctic oligotrophic lakes similar to Kennady Lake.

8.10.2.5 Effects of Changes to Aquatic Health

Fish populations and abundance can be affected by changes in water quality if they result in changes in aquatic health (i.e., fish and invertebrate health). Potential effects to aquatic health were evaluated in the Effects to Aquatic Health section (Section 8.9) through direct exposure to substances in the water column and indirect effects related to possible accumulation of substances within fish tissue via uptake from both water and diet. The assessment was based on modelled water quality in the main basins of Kennady Lake and Area 8 during closure and post-closure.

The results of the Aquatic Health assessment were then used to describe and assess changes that relate to fish and fish habitat (i.e., fish populations and communities). A discussion of the methods, models, and assumptions used in the Water Quality and Aquatic Health assessments can be found in Sections 8.8 and 8.9.

8.10.2.6 Long-Term Effects

Recovery of Fish Community

The recovery of the fish community in Kennady Lake in post-closure is described in Section 8.11. The recovery was qualitatively assessed using relevant information from a literature review, expected physical conditions and modelled water quality after the lake has been refilled and stabilized, and the ecological concepts of colonization, natural succession, and trophic interactions between plankton, benthic invertebrate, and fish communities. The duration of the predicted recovery was based on the expected timing of recovery of plankton and benthic invertebrate communities, the changes to the lake from the Project, and the life history attributes of the species expected to establish self-sustaining populations in the refilled Kennady Lake.

8.10.3 Effects Analysis Results – Construction and Operation

8.10.3.1 Effects of Changes to Fish Habitat from Project Footprint

Changes to fish habitat will occur due to the development of the Project, e.g., excavation of the mine pits, placement of mine rock piles, the Water Management Pond (WMP), Coarse PK Pile and Fine PKC Facility, dykes, and other construction activities. The affected habitat areas include the following:

- portions of Kennady Lake and adjacent lakes within the Kennady Lake watershed that will be permanently lost;
- portions of Kennady Lake that will be physically altered after dewatering and later submerged in the refilled Kennady Lake; and
- portions of Kennady Lake that will be dewatered (or partially dewatered) but not otherwise physically altered before being submerged in the refilled Kennady Lake.

These affected habitat areas are described below; more details are provided in the CCP (Section 3, Appendix 3.II).

Permanently Lost Areas

The permanently lost areas are those affected by the following:

- The Fine PKC Facility (Areas 1 and 2, Lake A1, Lake A2, Lake A5, Lake A6, Lake A7);
- The Coarse PK Pile (Area 4 and Lake Kb4);
- West Mine Rock Pile (Area 5 and Lake Ka1);
- South Mine Rock Pile (Area 6); and
- Dykes C, D, H, I and L.

The Project will result in the permanent loss of 194.56 ha of lake area (Table 8.10-3). Most of the losses will occur in Kennady Lake (154.61 ha), representing about 19% of the total pre-development Kennady Lake area of 813.57 ha. The remainder of the permanently lost areas includes the complete loss of Lakes A1, A2, A5, A7, Ka1, and Kb4, and partial losses of small portions of Lakes A3, A6, and N7. The largest category of habitat that will be permanently lost is deep lake bed covered by fine substrate, with additional habitat loss occurring in other areas dominated by fine substrates (which is typically of relatively low habitat quality). A considerable proportion of the remaining permanent losses will occur in areas dominated by boulder, which is typically of relatively high habitat quality. The Project will also result in the permanent loss of 0.51 ha of watercourse area in tributaries to Kennady Lake (Table 8.10-4).

Table 8.10-3 Lake Areas Permanently Lost as a Result of the Project

Mine Infrastructure	Area Permanently Lost (ha)										
	Kennady Lake	A1	A2	A3	A5	A6	A7	Ka1	Kb4	N7	Total[(a)]
Fine Processed Kimberlite Containment Facility	59.24	34.45	3.07	-	0.14	0.07	0.12	-	-	-	97.09
Coarse Processed Kimberlite Pile	1.05	-	-	-	-	-	-	-	1.03	-	2.08
West Mine Rock Pile	34.08	-	-	-	-	-	-	0.94	-	-	35.03
South Mine Rock Pile	52.71	-	-	-	-	-	-	-	-	-	52.71
Dyke C	-	-	-	0.09	-	-	-	-	-	-	0.09
Dyke D	-	-	-	-	-	-	-	-	-	0.04	0.04
Dyke H	0.62	-	-	-	-	-	-	-	-	-	0.62
Dyke I	2.25	-	-	-	-	-	-	-	-	-	2.25
Dyke L	4.67	-	-	-	-	-	-	-	-	-	4.67
Total	**154.61**	**34.45**	**3.07**	**0.09**	**0.14**	**0.07**	**0.12**	**0.94**	**1.03**	**0.04**	**194.56**

(a) Totals may not be exact due to rounding errors.

Table 8.10-4 Watercourse Areas Permanently Lost as a Result of the Project

Stream	Length (m)	Assumed Width (m)	Area (m^2)
A1	100	3	300
A2	20	3	60
A3	294	3	882
A5	115	3	345
A6	371	3	1,113
A7	31	3	213
B1	94	3	282
F1	168	3	504
Ka1	170	3	510
Kb4	309	3	927
Total Area (m^2) **Total Area (ha)**			**5,136** **0.51**

m = metre; m^2 = square metre; ha = hectare.

In the calculation of Habitat Units (HUs), fish-bearing lakes that are expected to be affected include the following:

- Lake A1 has a total of 110.34 HUs, all of which will be permanently lost due to the Project; most of the HUs are for lake trout (22.36 HUs), Arctic grayling (21.15 HUs), and burbot (20.34 HUs);

- Lake A2 has a total of 15.74 HUs, all of which will be permanently lost due to the Project; the largest amounts of HUs are for slimy sculpin (3.09 HUs), lake trout (2.84 HUs), and Arctic grayling (2.59 HUs);
- a total of 0.90 HUs will be permanently lost in Lake A3 due to the Project; the largest amounts of HUs are for slimy sculpin (0.28 HUs);
- a total of 0.08 HUs will be permanently lost in Lake N7 due to the Project; the largest amounts of HUs are for lake trout (0.02 HUs), Arctic grayling (0.02 HUs), and burbot (0.02 HUs);
- permanent habitat losses in Kennady Lake will total 1,157 HUs, which represents about 20% of the HUs currently in Kennady Lake (i.e., total of 5,826 HUs); the species most affected by the lost habitat include lake chub (206 HUs), lake trout (185 HUs), slimy sculpin (171 HUs), and burbot (170 HUs).

Lakes A5, A6, A7, Ka1, and Kb4 were assessed as being non-fish bearing in the baseline assessment (Section 8.3, Annex J, Addendum JJ), and, therefore, not considered further in the calculation of HUs permanently lost.

Physically Altered and Re-submerged Areas

Fish habitats that will be physically altered during operations and then submerged in the refilled Kennady Lake include the following:

- Part of Kennady Lake Area 3 (affected by Dyke B);
- Part of Kennady Lake Area 4 (affected by Tuzo Pit, Dyke B, Dyke J, and CP6 Berm);
- Part of Kennady Lake Area 6 (affected by Hearne Pit, 5034 Pit, Dyke K, Dyke N, Road between Hearne Pit and Dyke K, CP3 Berm, CP4 Berm, and CP5 Berm); and
- Part of Kennady Lake Area 7 (affected by Dyke A and Dyke K).

The Project will result in 83.32 ha of lake area being physically altered and re-submerged at closure. All of this area will be located in Kennady Lake (Table 8.10-5), representing about 10% of the total pre-mine Kennady Lake area of 813.57 ha. The largest category of habitat that will be physically altered and re-submerged is deep lake bed covered by fine substrate. Almost 70% of the habitats to be physically altered and re-submerged will occur in areas dominated by fine substrates, which is typically of relatively low quality. A considerable proportion of the remaining affected area will occur in areas dominated by boulder, which is typically of relatively high quality.

Table 8.10-5 Areas in Kennady Lake that are Physically Altered and then Re-Submerged at Closure

Mine Infrastructure	Area Physically Altered and Re-Submerged (ha)
Hearne Pit	13.87
Tuzo Pit	20.81
5034 Pit	19.8
Dyke A	0.35
Dyke B	16.13
Dyke J	0.41
Dyke K	2.89
Dyke N	3.99
Roads	3.96
Water Collection Pond Berms	1.12
Total[a]	**83.32**

[a] Total may not be exact due to rounding errors.

ha = hectares.

In terms of HUs, habitat losses in Kennady Lake from areas that will be physically altered and re-submerged at closure will total 610 HUs), which represents about 11% of the HUs currently in Kennady Lake. The amounts of habitat units lost will be highest for lake trout (104 HUs) and lake chub (97 HUs).

Dewatered and Re-submerged Areas

The areas that will be dewatered (or partially dewatered) but not otherwise altered before being re-submerged include the following:

- Portions of Kennady Lake Areas 3 through 7 (those parts that are not either permanently lost or physically altered);
- Lake D1; and
- Streams D1, D2, and E1.

The Project will result in approximately 435.90 ha of lake area being dewatered and re-submerged at closure but that will remain otherwise unaltered. This area includes 434.06 ha in Kennady Lake, which represents about 53% of the total pre-mine Kennady Lake area, and 1.87 ha in Lake D1. The largest category of habitat that will be physically altered and re-submerged is deep lake bed covered by fine substrate (46.96 ha). Almost 60% of the habitats that will be dewatered and re-submerged, but otherwise unaltered is deep lake bed covered by fine substrate (262.66 ha). The Project will also result in 0.23 ha watercourse area in tributaries to Kennady Lake (Streams D1, D2, and E1) being dewatered and re-submerged at closure, but that will remain otherwise unaltered.

The number of habitat units in Kennady Lake from areas that will be dewatered and then re-submerged at closure, but will remain otherwise unaltered, will total about 3,011 HUs, which represents about 52% of the HUs currently in Kennady Lake. The amount of habitat units lost will be highest for lake chub (502 HUs) and lake trout (495 HUs). Lake D1 has a total of 4.61 HUs, all of which will be unaltered but dewatered and then re-submerged at closure. The largest amounts of habitat units in Lake D1 are for burbot (1.65 HUs).

Compensation Plan

Where prevention of harmful habitat alteration or loss is not feasible, fish habitat of equivalent or higher productive capacity will be developed. The CCP (Section 3, Appendix 3.II) describes the various options considered for providing compensation, and presents a proposed fish habitat conceptual compensation plan to achieve no net loss of fish habitat according to DFO's Fish Habitat Management Policy (DFO 1986, 1998, 2006). The options include: construction of impounding dykes to raise lake levels; construction of finger reefs in Kennady Lake; construction of habitat structures on the decommissioned mine pits/dykes; and widening the top bench of pits to create shelf areas where they extend onto land. More information on compensation works is included in Section 8.10.4.1.

8.10.3.2 Effects of Dewatering on Fish and Fish Habitat

Dewatering Areas 2 to 7 of Kennady Lake is required to allow mining of the three diamond-bearing kimberlite pipes located under the lake-bed. Dyke A will be constructed at the narrows separating Areas 7 and 8 during the construction phase. Dyke A will allow the dewatering of Areas 2 through 7 while maintaining similar lake levels in Area 8. A portion of Area 1 (Lakes A1 and A2) will also be dewatered into Lake A3 after Dyke C is constructed.

Dewatering will result in the temporary loss of fish populations and lower trophic communities from the main basins of Kennady Lake. During operations, Areas 6 and 7 will be completely dewatered and Areas 2 through 5 will be partially dewatered. Kennady Lake is known to support eight species of fish, including, in order of abundance, round whitefish, lake trout, lake chub, Arctic grayling, northern pike, burbot, slimy sculpin, and ninespine stickleback. Lakes in the A watershed are known to support Arctic grayling, burbot, round whitefish, lake trout, and northern pike.

Removal of Fish

Fish salvage will be conducted to remove fish from Areas 2 to 7 before and during dewatering. The fish salvage is intended to minimize the waste of fish caused by the dewatering of Kennady Lake. The salvage would occur prior to and during dewatering of the lake and would also include removal of fish from Lakes A1 and A2 prior to partial dewatering and fine PK storage. Because Kennady Lake contains large-bodied and small-bodied fish species with a variety of habitat preferences, a combination of gear types would be used to maximize capture efficiency. These gear types could include gill nets, trap-nets, minnow traps, boat and backpack electrofishing, and angling. The fish salvage will be designed and implemented in consultation with Fisheries and Oceans Canada (DFO) and local Aboriginal communities, and may follow the draft General Fish-out Protocol for Lakes and Impoundments in the Northwest Territories and Nunavut (Tyson et al no date), as appropriate. Project-specific protocols for fish salvage will be developed prior to initiating the salvage.

The fish salvage at the Diavik Diamond Mine (McEachern et al. 2003) showed a survival rate of approximately 50% for fish captured during the salvage. Therefore, the possibility exists that fish could be moved to other lakes near Kennady Lake. This option would depend on the availability of barren lakes (i.e., those containing no large fish species) in the Project area and the approval of DFO and input from local Aboriginal communities. Release of captured fish to other fish-bearing lakes is not recommended because of the possibility of negative effects to the lake receiving the fish. These negative effects could include disease, parasites, genetic implications, and inter-species and intra-species density-dependent interactions (i.e., predation and competition). Based on the project-specific protocols developed, salvaged fish may be provided to Aboriginal communities to avoid wasting of fish. Capture techniques and salvage protocols for Lake A1 and Lake A2 will be similar to those for Kennady Lake.

Temporary Habitat Loss

Dewatering will result in the temporary loss of productive capacity of fish habitat within Areas 2 to 7 of Kennady Lake. Although Areas 2 to 5 will only be partially dewatered and will serve as the WMP for the Project, the depth, habitat, suspended sediment and water quality conditions in these areas will not be suitable to support a fish community. As described in Section 8.10.3.1 and the CCP (Section 3, Appendix 3.II), 434.06 ha of Kennady Lake will be dewatered and re-submerged at closure but will remain otherwise unaltered, representing about 53% of the total pre-mine Kennady Lake area. Similarly, 1.87 ha in Lake D1 will be dewatered and then re-submerged at closure. The majority of the habitats that will be unaltered but dewatered and re-submerged will occur in areas dominated by fine substrates, which is typically of low quality. However, since they are not being altered, habitat losses incurred during dewatering will be offset by equivalent habitat gains during refilling.

The loss of the fish community and the productive capacity of the fish habitat will last for the 13 years of construction and operations, the estimated 8 year refill period, and an additional period until the lower trophic communities and fish populations have re-established after closure. However, it is expected that a self-sustaining fish community will be present in Kennady Lake post-closure. The recovery of fish and lower trophic communities, and the productive use of fish habitat, are described in Section 8.11.

Changes to Lake Levels and Lake Areas

As described in the Effects to Water Quantity section (Section 8.7), estimated water levels in Area 8 will be slightly augmented relative to baseline conditions during Kennady Lake dewatering. However, discharges into Area 8 will be limited to ensure that 2-year flood conditions are not exceeded in Area 8 or its outlet channel (Stream K5); no effects to shoreline stability would be expected (Section 8.7.3.2). The estimated increase in the maximum depth of 0.03 m and surface area of less than 1% would not have any effect on fish habitat, as it would be well within the natural variability of the basin. Effects to fish and fish habitat from alteration of flows in the Area 8 outlet channel (Stream K5) are addressed in Section 9.10.

8.10.3.3 Effects of Watershed Diversions on Fish and Fish Habitat

To reduce the volume of runoff entering the controlled areas of Kennady Lake, the A, B, D and E watersheds will be diverted to the adjacent N watershed (see Figure 8.4-3). The watersheds will be diverted by constructing earth-fill dykes in their respective outlet channels to increase lake elevation and by constructing diversion channels to carry backed-up water away from Kennady Lake.

In the A watershed, a permanent dyke C will be constructed at the south end of Lake A3 to increase the water level in Lake A3 and divert its flow north to Lake N9 through a constructed channel.

In the B watershed, dyke E will be constructed between Lake B1 and the north end of Kennady Lake to prevent inflow to Kennady Lake and to divert all watershed B flow north to Lake N8 thorough a constructed channel. Near the end of operations, dyke E will not be removed, but will be partially breached to allow the flow from Lake B1 and upstream lakes to return to Kennady Lake.

In the D watershed, a temporary dyke F will be constructed between lakes D1 and D2 to increase water levels in lakes D2 and D3, resulting in one raised lake, D2-D3. The waters from the raised lake, together with flow from upstream lakes D4 and D7, will be diverted to the northwest shore of Lake N14 thorough a constructed channel. Lake D1, the lowermost lake in the D watershed, will continue discharging to Kennady Lake; however, its recharge area will be greatly reduced by dyke F. Near the end of operations, dyke F will be removed and the flow paths will be returned to pre-Project conditions (i.e., through Lake D1 to Kennady Lake).

In the E watershed, a temporary dyke G will be constructed to increase the water level in Lake E1 and divert the flow to the south shore of Lake N14 thorough a short constructed channel. Near the end of operations, dyke G will be removed and the flow paths will be returned to pre-Project conditions (i.e., through Stream E1 to Kennady Lake).

The diversions in the A and B watersheds will connect to lakes and streams in the east part of the N watershed (lakes N6 to N2), which drain into Lake N1. The diversions in the D and E watersheds will connect to lakes and streams in the west part of the N watershed, which also drains into Lake N1, but through lakes N14, N17, N16, and N11 (in that order). The two diversion pathways converge in Lake N1, which in turn drains into Lake 410.

Watershed diversions will result in raised water levels in lakes A3, D2, D3, and E1 during operations. Pre-diversion (baseline) and post-diversion (operations) lake areas, maximum depths and known fish species in these lakes are shown in Table 8.10-6. The fish species recorded in N watershed lakes downstream of the diversions are shown in Table 8.10-7.

Table 8.10-6 Pre-Diversion (Baseline) and Post-diversion (Operations) Lake Areas and Depths in Diverted Lakes of the A, B, D and E Watersheds and Fish Species Known to Inhabit the Lakes

Lake	Lake Area (ha)		Maximum Depth (m)		Fish Species Recorded
	Baseline	Operations	Baseline	Operations	
A3	23.8	46.6	12.4	15.9	ARGR, BURB, LKTR, NRPK
B1	8.2	8.2	4.1	4.1	ARGR, LKTR, NNST, SLSC
B2	6.6	6.6	1.1	1.1	none
B3	1.5	1.5	-	-	not sampled
D1	1.9	1.9	3.8	3.8	BURB, NRPK
D2	12.5	104 [a]	1.0	4.6 [a]	NRPK
D3	38.4		3.0		BURB, LKTR, NRPK
D7	40.2	40.2	4.5	4.5	ARGR, BURB, NRPK
E1	20.2	27.0	3.9	4.7	NRPK, SLSC
E2	3.0	3.0	0.4	0.4	none
E3	1.1	1.1	0.7	0.7	none

[a] Raised water levels will result in one lake D2-D3.

ha = hectare; m = metre: ARGR = Arctic grayling; BURB = burbot; LKTR = lake trout; NRPK = northern pike; LKCH = lake chub; NNST = ninespine stickleback; SLSC = slimy sculpin; - = not sampled for depth.

Table 8.10-7 Fish Species Recorded in the N Watershed Lakes Downstream of the Diversions

Diversion	Lake	Fish Species Recorded
Downstream lakes along the A and B watersheds diversion	N2	ARGR, LKCH, LKTR, NNST, RNWH, SLSC
	N3	ARGR, BURB, LKCH, RNWH
	N4	ARGR, LKCH
	N5	ARGR, LKCH, LKTR, NNST, RNWH, SLSC
	N6	ARGR, BURB, LKTR, NNST, RNWH
Downstream lakes along the D and E watersheds diversion	N11	not sampled
	N14	ARGR, LKCH, LKTR, LNSC, NNST, SLSC
	N16	BURB, CISC, LKCH, LKTR, LNSC, NNST, RNWH, SLSC, WHSC[a]
	N17	BURB, LKCH, LKTR, SLSC
410	410	BURB, CISC, LKCH, LKTR, NRPK, RNWH, SLSC

[a] The reported presence of white sucker in Lake N16 may potentially be a misidentification.

ARGR = Arctic grayling; BURB = burbot; LKTR = lake trout; NRPK = northern pike; CISC = cisco; RNWH = round whitefish; LNSC = longnose sucker; WHSC = white sucker; LKCH = lake chub; NNST = ninespine stickleback; SLSC = slimy sculpin.

The streams connecting the diverted lakes are generally short in length (between 63 and 538 m). Most feature fish passage potential during the entire open water period; however, even some of the larger streams (e.g., N2 between lakes N2 and N1) can present barriers to fish movement during low flow periods in the fall. Fish species recorded in streams between the diverted lakes, as well as in streams downstream of the diverted watersheds are shown in Table 8.10-8.

Table 8.10-8 Channel Length, Fish Passage Potential and Fish Species Known to Inhabit the Streams between Diverted Lakes of the B, D, E and N Watersheds

Stream	Channel Length (m)	Fish Passage Potential[a]	Fish Species Recorded
B1	94	spring to fall	ARGR
B2	169	spring only	not sampled
B3	332	spring only	not sampled
D1	118	spring to fall	ARGR, BURB, NNST
D2	228	spring to fall	ARGR, BURB, NRPK, SLSC
D3	97	spring to fall	not sampled
D4	428	spring to fall	SLSC
D7	206	spring to fall	SLSC
E1	426	spring to fall	ARGR, BURB, NNST, NRPK
E2	290	spring only	not sampled
N1[b]	70	spring to fall	BURB, LKCH, SLSC
N2[b]	228	spring/summer	ARGR, BURB, LKCH, LNSC, NNST, SLSC
N3[b]	65	spring to fall	ARGR, BURB, LKCH, LKTR, LNSC, SLSC
N4[b]	63	spring/summer	ARGR, BURB, LKCH, NNST, SLSC
N5[b]	73	spring/summer	ARGR, BURB, LKCH, NNST, SLSC
N6[b]	155	spring/summer	ARGR, BURB, LKCH, NNST, SLSC
N11[c]	174	spring to fall	BURB, LKCH, SLSC
N14[c]	500	spring/summer	ARGR
N16[c]	538	spring to fall	ARGR, BURB, LKCH, LKTR, LNSC, SLSC
N17[c]	348	spring to fall	ARGR, BURB, LKCH, LKTR, LNSC, NNST, SLSC

(a) Seasons of potential fish passage estimated during habitat assessments.
(b) Streams downstream of the A and B watersheds diversion.
(c) Streams downstream of the D and E watersheds diversion.
m = metre: ARGR = Arctic grayling; BURB = burbot: LKTR = lake trout; NRPK = northern pike;
LNSC = longnose sucker; LKCH = lake chub; NNST = ninespine stickleback; SLSC = slimy sculpin.

Eleven fish species in total have been recorded in the lakes and streams potentially affected by the diversions (Tables 8.10-6 to 8.10-8). Some species, such as Arctic grayling, lake trout, burbot, ninespine stickleback, and slimy sculpin, have been recorded in most of the affected watersheds. Other species, such as longnose sucker, white sucker, cisco and lake chub, have been recorded only in the N watershed and not in the A, B, D and E watersheds. Conversely, northern pike have not been recorded in the N watershed, although they are common in the A, D, and E watersheds.

Potential effects to fish and fish habitat in the diverted watersheds include the following:

- loss of stream and lake habitat downstream of dykes;
- increased lake levels and lake areas upstream of dykes;
- changes to erosion, resuspension of sediments, and sedimentation;
- changes to fish migrations; and
- changes to fish communities.

These effects are discussed separately in the following subsections.

Loss of Stream Habitat Downstream of Dykes

Habitat downstream of the dykes will be dewatered and lost to fish residing in upstream lakes, which will include Stream A3 in the A watershed, Stream B1 in the B watershed, Streams D1 and D2 in the D watershed, and Stream E1 in the E watershed. The loss of fish habitat resulting from the placement of the dykes and the dewatering of downstream stream segments, and Lake D1, is described in Section 8.10.3.1 and included in the CCP (Section 3, Appendix 3.II) to ensure that no net loss in fish habitat is achieved for the Project. Fish species and habitat use in each of the diversion watersheds is summarized below.

A Watershed

Arctic grayling and northern pike use Stream A3 as a movement corridor and for juvenile rearing; however, the spawning habitat quality for these species has been assessed as low. Ninespine stickleback had also been confirmed to use Stream A3 and, based on captures in the neighbouring waterbodies, burbot and slimy sculpin may also be present.

B Watershed

Loss of Stream B1 downstream of dyke E is likely to affect Arctic grayling as it will eliminate natural spawning habitat for this species in the B watershed. The persistence of Arctic grayling in the B watershed will depend on Arctic grayling using habitat constructed in the diversion channel and immigration of Arctic grayling from the N watershed. Lake trout, slimy sculpin, and ninespine stickleback, the only other fish species besides Arctic grayling captured in the B watershed, spawn and rear in lakes and will not be affected by the loss of Stream B1. Small numbers of round whitefish and burbot may also be intermittently present in Lake B1 but do not require access to Stream B1 for spawning, rearing, or foraging.

D Watershed

Northern pike, burbot, Arctic grayling, lake trout, and slimy sculpin are known to use lakes and streams in the D watershed upstream of the proposed dyke F. Although not documented by fish captures, these waterbodies are also likely to contain ninespine stickleback. Arctic grayling in Kennady Lake use streams within the D watershed for spawning, but the numbers of fish using these streams is small in comparison to the numbers of Arctic grayling using streams downstream of Kennady Lake. Lakes in the D watershed appear to be one of the primary northern pike spawning locations as large numbers of adult northern pike have been captured moving upstream into Lake D2 in spring. Use of lakes in the D watershed by lake trout is minimal and likely limited to seasonal foraging.

Loss of Lake D1 (through the reduction of recharge area) is expected to have a small effect on fish populations in the D watershed. Lake D1 is a small (1.9 ha) lake with a maximum depth of 3.8 m, which is relatively deep in comparison to most lakes of this size in the Kennady Lake watershed. The nearshore habitat is comprised mostly of boulder/cobble substrates covered with fine sediments. This type of habitat is generally of low value to most fish species. However, areas of higher-quality clean boulder/cobble substrates and areas of submerged and emergent vegetation exist in this lake. Loss of vegetation is expected to have a small effect on northern pike and ninespine stickleback populations because much larger areas of aquatic vegetation exist in lakes upstream of the dyke, specifically in lakes D2, D3, and D7. Similarly, areas of clean boulder/cobble substrates used by lake trout, burbot and slimy sculpin exist in the other larger, upstream lakes. The amount of overwintering habitat lost in Lake D1 is small in comparison to the amount that will continue to be available, principally in Lake D7 (lake area of 40 ha and maximum depth of 4.5 m) and in the raised Lake D2-D3 (lake area of 104 ha and maximum depth of 4.6 m).

Loss of streams D1 and D2 will result in the loss of two of the three streams in the D watershed with habitat suitable for Arctic grayling spawning. These losses, combined with the loss of suitable spawning habitat in Stream D3 when the water level in Lake D2 is raised, will eliminate all natural spawning habitat for Arctic grayling in the D watershed.

Persistence of Arctic grayling in the D watershed during operations will depend on the spawning use of habitat in the diversion channel constructed between Lake D3 and Lake N14 and immigration of Arctic grayling from Lake N14 and waterbodies farther downstream. An artificial stream constructed at the Ekati Diamond Mine was found to allow fish migration and provide spawning and nursery habitat for Arctic grayling, albeit with 63% lower standing stock than natural streams (Jones et al. 2003). The diversion channel between Lake D3 and Lake N14 will be designed and constructed to allow fish passage while also incorporating lessons learned from the Ekati Diamond Mine experience to increase Arctic grayling production.

E Watershed

Loss of Stream E1 downstream of dyke G is likely to affect Arctic grayling as it will eliminate natural spawning habitat for this species in the E watershed. The persistence of Arctic grayling in the E watershed will depend on Arctic grayling using habitat constructed in the diversion channel between lakes E1 and N14 and immigration of Arctic grayling from the N watershed.

Northern pike, burbot, slimy sculpin and ninespine stickleback, the only other fish species besides Arctic grayling captured in the E watershed, spawn and rear primarily in lakes and will not be substantially affected by the loss of Stream E1.

Changes to Lake Levels and Lake Areas Upstream of Dykes

Raising water levels in Lakes A3, D2, D3, and E1 will result in increased lake habitat area. This increase will be permanent in Lake A3, but water levels in the remaining lakes will be lowered to pre-Project (i.e., baseline) conditions after dykes F and G are removed at the end of operations. Water level in Lake A3 will be raised by about 3.5 m, resulting in a 95% increase in the surface area of the lake. In lakes D2 and D3, the water level will be raised by 2.8 m and 1.6 m, respectively, creating one lake with a surface area approximately twice as large as the combined pre-Project area of the two lakes. Water level in Lake E1 will be raised by about 0.8 m, resulting in a 34% increase in the surface area of the lake.

Raised water levels may create a benefit to fish residing in these lakes during mine construction and operations. These benefits will be manifested largely from the additional space and increased amount of overwintering habitat for all resident species. Populations of northern pike and ninespine stickleback may also benefit from the increased spawning and rearing habitat in areas with flooded vegetation.

Shoreline Erosion, Resuspension of Sediments and Sedimentation

Raising lake levels in Lakes A3, D2, D3, and E1 will create new shorelines at higher elevations than the existing shorelines, which will expose new soils, often on steeper slopes than the existing shorelines, to wave erosion and potential instability due to permafrost disturbance. This can result in shoreline erosion and an increased sediment load into the lakes. Total suspended sediment (i.e., TSS) can affect fish directly and settling of the sediment (i.e., sedimentation) can affect nearby habitats.

The nature and extent of adverse effects of increased TSS is influenced by both the TSS concentration and the duration of exposure. Fish can tolerate low TSS concentrations for long periods and high concentrations for short periods without suffering adverse effects. The effects of sediment deposition can include infilling of interstitial spaces between substrate particles that provide habitat for rearing of fry or incubation of eggs, covering aquatic plants, which can provide habitat for juvenile rearing or incubation of eggs, and potential shifts to benthic communities. The severity of the effect depends on the type of habitat and its use by fish.

The shorelines in the diversion lakes are currently dominated by boulders and large cobble substrates. Nearshore habitats in Lakes A3, B1, D2, and D3 are dominated by boulder substrates, with some areas of cobble or bedrock. The presence of these large substrates will promote long-term stability of the shorelines. Rates of shoreline erosion are related to composition of bank material (Newbury and McCullough 1984), and boulder shorelines reduce erosional forces, compared to fine sediment (Fitzpatrick 1995). As described in the Effects to Water Quantity section (Section 8.7.3.3), increases in TSS concentrations in the raised lakes are expected to be low due to the armouring action of morainal materials and the rapid settling of its coarse fractions from the water column, along with the location of organic soils in low-gradient locations. A baseline monitoring program will be established and mitigation measures will be applied if areas of substantial erosion are identified during construction and operations. It is also expected that any increases in TSS concentrations due to shoreline erosion would occur during spring freshet or storm events. Fish are routinely exposed to higher TSS levels during these periods and would tolerate the levels in the short-term. Fish would also show a behavioural response, moving away from any shoreline areas with a high sediment load. As a result, negligible effects on fish and fish habitat are expected from shoreline erosion, resuspension of sediments, and sedimentation.

Changes to Lower Trophic Levels

Changes in water levels and lake areas in Lakes A3, D2-D3, and E1 are expected to increase habitat area available for plankton and benthic invertebrates, once new lake areas are fully colonized. This will result in overall increased total biomass of plankton and benthic invertebrates in these lakes, after a period of adjustment to the new water levels. Based the topography of land around Lakes D2-D3 and E1, the enlarged lakes will have relatively large shallow areas suitable for development of benthic algae, which will in turn provide food for benthic invertebrates. The increased lake levels are also expected to result in reduced benthic invertebrate biomass in deeper areas of these lakes, as their benthic fauna becomes more typical of deep-water areas, which are usually characterized by lower invertebrate density and richness.

Production of plankton and benthic invertebrate communities in the A, B, D and E watersheds is not expected to be negatively affected with the raising of the lake levels and their diversion to the N watershed, but will require a period of adjustment to the new water level. As explained above, diversions are not expected to substantially increase TSS concentrations in the water column or appreciably alter other water quality parameters upon which invertebrate production is dependent (i.e., phosphorus, nitrogen, carbon). However, some initial changes in water quality are expected after flooding of the new areas, as fine sediments are redistributed and any residual organic material is used by bacteria. These changes are anticipated to be small after the first year of elevated lake levels; the baseline monitoring program will also identify areas where mitigation would be required to minimize sources of suspended sediments.

Development of aquatic vegetation will likely be limited by the type of substrates that will be flooded (i.e., mostly coarse materials) and low nutrient concentrations. Over time (i.e., anticipated as less than five years), existing lower trophic communities will colonize new habitat created by raising water level. The initial colonizers in newly-flooded areas will likely be midges, followed by non-insect groups, such as fingernail clams and mollusks. Once established, the benthic community of newly-flooded areas is predicted to be one of low density and diversity, which is typical of lakes in the region.

Changes to Fish Migrations

Dykes in streams A3, B1, D2 and E1 will interrupt the movements of fish between Kennady Lake and waterbodies upstream of the dykes. This effect will be permanent for the A watershed, but will be limited to the period of mine operations for the B, D and E watersheds. The effect of the dykes on fish migrations is mainly limited to the potential interruption of obligatory migrations that a particular fish species would need to make to and from Kennady Lake to fulfill its life history requirements. This situation would occur only if there was some unique habitat available in Kennady Lake or in the streams downstream of the dykes that were unavailable in watersheds A, B, D or E.

Loss of access to the lowermost streams in the A, B, D and E watersheds is likely to affect Arctic grayling, which currently use these stream habitats for spawning and rearing. As natural spawning habitats for Arctic grayling do not currently exist in these watersheds upstream of the dykes, persistence of this species will depend on whether Arctic grayling use habitat constructed in the diversion channels and any immigration of Arctic grayling from the N watershed. Arctic grayling are common in the N watershed and it is likely that they will move into the upstream watersheds (A, B, D and E) after they are connected through the newly constructed diversion channels.

The diversion channels connecting B1, D2-D3, and E1 to the N watershed and Lake A3 to the L watershed will be designed to provide spring spawning and rearing habitat for Arctic grayling and allow the seasonal passage of fish between lakes that approximates natural conditions. Physical features of these channels will include the following:

- bank and bottom substrates will consist predominantly of cobble, boulder and gravel to allow Arctic grayling spawning and to limit erosion;
- riffle and pool sequencing will be included; and
- slopes, channel depths, and widths will be sufficient to allow fish passage throughout the open-water season; designs will ensure that water velocities in spring will be low enough to avoid creating barriers and that sufficient flow is present in late summer/fall to allow fish to move to overwintering habitat downstream, if necessary.

Northern pike have been documented to use lake and stream habitat in the A, D, and E watersheds and suitable spawning, rearing, and overwintering habitats exist in these watersheds upstream of the dykes. The dykes will preclude the annual spring spawning migrations of adults from Kennady Lake and prevent potential recruitment from this system. Although the dykes will in effect isolate the northern pike populations within their respective watersheds for the duration of mine operations (and permanently in Lake A3), it is likely that the isolated populations will be self-sustaining. Unlike Arctic grayling populations that can be augmented in these watersheds through potential immigration from the N watershed, the presumed absence of northern pike in the N watershed would preclude similar recruitment. During baseline sampling, northern pike have not been captured in lakes and streams in the N watershed, although they are present in Kennady Lake and downstream to Lake 410; therefore, it appears that northern pike are absent from the N watershed, or are present at extremely low numbers. As a result of the diversions, it will be possible for northern pike from Kennady Lake to move into the upper part of the N watershed, where suitable spawning and rearing habitat exists in shallow bays of downstream lakes. It should be noted, however, that the lower part of the N watershed is already well connected to Lake 410 (i.e., Lake N16 is about 15 km upstream from Lake 410) and northern pike have not taken advantage of this connection to disperse into the N watershed. Although habitat conditions in the Kennady and N watersheds are generally similar, differences in the abundance and distribution of aquatic vegetation may have contributed to the apparent difference in northern pike use of the two watersheds. As such, the probability of northern pike dispersing into the N watershed via the proposed diversion channel in the upper part of the N watershed (i.e., from D and E watersheds to Lake N14) is expected to be low, and no substantial changes to the resident fish communities in the N watershed are anticipated.

Small populations of burbot, slimy sculpin, and ninespine stickleback will likely continue to spawn in the diverted watersheds. These species are not known to undergo extensive migrations between waterbodies and the loss of connectivity to Kennady Lake is not likely to affect their abundance. The lakes upstream of the dykes are deep enough (exceed 3.4 m and will be even deeper after the water levels are raised) to provide suitable overwintering habitat. Nearshore habitats in lakes A3, B1, D2, D3, D7 and E1 (all upstream of the dykes) include clean boulder/cobble substrates used by slimy sculpin and burbot for spawning and rearing, and submerged and emergent aquatic vegetation used by northern pike and ninespine stickleback for spawning, rearing, and foraging. As such, all life history requirements for these species can be fulfilled in the diverted watersheds, without the need to access Kennady Lake.

A second effect of the dykes on fish migrations is the prevention of out-migrations of juvenile and young-of-the-year fish to Kennady Lake. Although not specifically documented, it is expected that some proportion of each year class migrate out of lakes in the A, B, D and E watersheds down to Kennady Lake each year. These emigrations are most likely in response to density-dependent competition for food and space but may also be in response to increased predation in the smaller lakes.

Prevention of downstream emigration to Kennady Lake is expected to have a minor effect on fish populations in lakes upstream of the dykes. These lakes have a carrying capacity which, like all lakes in the Kennady Lake area, is limited by low nutrient availability. The lakes can be assumed to be at their natural carrying capacity and will remain at or near this carrying capacity during mine operations, regardless of whether fish can emigrate to Kennady Lake. If the carrying capacity is exceeded, the fish will be able to disperse to lakes in the N watershed through the constructed diversion channels.

Changes to Fish Communities in the A, B, D and E Watersheds

Persistence of fish populations in the diverted A, B, D and E watersheds will be dependent on the following:

- suitable water quality;
- the continued production of plankton and benthic invertebrate communities; and
- the continued availability of and access to habitat necessary to complete their life histories.

As noted above, water quality is expected to remain suitable for aquatic life in the A, B, D and E watersheds during diversions, and plankton and benthic invertebrate communities are expected to remain viable in the lakes that are predicted to increase in size.

Populations of small-bodied fish, such as ninespine stickleback and slimy sculpin, are likely to persist in diverted watersheds during mine operations because suitable spawning, rearing, and foraging habitat for each species will be available and there is no critical habitat in Kennady Lake that any of these species require to complete their life histories.

Northern pike, like ninespine stickleback, require aquatic vegetation for spawning and rearing (Scott and Crossman 1973; Casselman and Lewis 1996; Richardson et al. 2001). This type of habitat does not exist in the B watershed and if any northern pike are present in Lake B1 (none have been reported to date), they would be unlikely to reproduce successfully. Aquatic vegetation exists in lakes A3, D2, D3, D7, and E1, and these lakes will continue to provide suitable habitat for northern pike and ninespine stickleback throughout mine operations. Increasing the depth and area of lakes D2, D3 and E1 may actually create a benefit to northern pike and ninespine stickleback residing in these lakes during mine construction and operations because of the increased amount of riparian vegetation flooded when raising these lakes.

Few lake trout have been captured in the A, B, and D watersheds and none have been reported in the E watershed. This is most likely because the smaller size and shallower depths of these lakes are generally unsuitable for lake trout, which prefer lakes with deeper water that have low water temperatures in summer and high levels of dissolved oxygen year round. Lake trout that have been captured in lakes in the B and D watersheds are likely using the lakes seasonally for rearing and feeding, e.g., juvenile lake trout that move out of Kennady Lake in the summer to feed and escape predation from adults. Lake trout are fall spawners that use boulder/cobble substrates at depths exceeding 2 m along the shorelines of lakes for spawning. These lakes will likely continue to provide the same amount of habitat for lake trout that currently exists. However, as it is unlikely that these lakes currently support self-sustaining lake trout populations, it is not expected that this species will persist in these lakes during operations.

Small numbers of burbot have been captured in lakes A3, D3 and D7. Burbot have not been reported in the B watershed. Although they were captured in Stream E1 near the Kennady Lake confluence, it is not known if they are present in Lake E1. Burbot spawn in similar habitat as lake trout; however, they do so in late winter under the ice (Richardson et al. 2001). Lakes A3, D3 and D7 will likely continue to provide the same amount of habitat for burbot that currently exists.

All available spawning and rearing areas used by Arctic grayling in the A, B, D and E watersheds are located under the footprint or downstream of the proposed dykes. As such, the construction of the dykes will negatively affect Arctic grayling reproduction. The persistence of Arctic grayling in the diverted watersheds, therefore, will be dependent on the suitability of spawning and rearing habitat constructed in the diversion channels and the use of this new habitat by Arctic grayling. The diversion channels between lakes A3 and N9, lakes B1 and N8, lakes D1 and N14, lakes D2-D3 and N14, and lakes E1 and N14 will be designed and constructed to allow fish passage and to provide suitable substrate and habitat conditions to allow Arctic grayling spawning and rearing. Lessons learned at the Ekati Diamond Mine and other places where artificial channels were constructed will be used to maximize the potential for Arctic grayling production.

In addition to creating appropriate spawning and rearing habitats in the constructed diversion channels, the persistence of Arctic grayling in the diverted watersheds will likely be influenced by potential immigration of Arctic grayling from the neighboring lakes in the N watershed. These lakes and interconnecting streams are known to support a large number of Arctic grayling, some of which may migrate to the diverted watersheds if the newly constructed channels provide adequate fish passage conditions.

8.10.3.4 Effects of Isolation of Area 8 on Fish and Fish Habitat

This section assesses the ability of Area 8 to support a fish population and effects to fish migration in and out of Kennady Lake while isolated.

Changes to Lower Trophic Communities

Isolation of Area 8 during operations and closure from the remainder of Kennady Lake was predicted to result in a slight increase in nutrient concentrations due to evaporative concentration of solutes in lake water (Section 8.8.4.1.2). Between construction and the end of operations, total phosphorus was predicted to gradually increase from a mean background concentration of 0.005 mg/L to 0.007 mg/L, along with a proportional increase in concentrations of nitrogen compounds. This change is not expected to alter the trophic status of Area 8 from oligotrophic (i.e., total phosphorus range of 0.004 to 0.010 mg/L) (CCME 1999). However, this increase in nutrient concentrations is expected to result in a slight increase in productivity of plankton and benthic invertebrate communities, without notable changes in community composition or dissolved oxygen concentration.

Changes to Fish Community

Area 8 is unique in comparison to the other basins of Kennady Lake because it is long (about 4 km), narrow (typically less than 500 m wide), and shallow (generally less than 4 m deep). Two deep areas (greater than 8 m deep) exist in Area 8; however, habitat greater than 4 m deep represents less than 8% of the total surface area of Area 8. Results of the radio-telemetry program showed that lake trout and Arctic grayling migrating from the Kennady Lake outlet (Stream K5) to Areas 2 to 7, moved quickly through Area 8 presumably because habitat conditions were more suitable in Areas 2 to 7. Similar data are unavailable for round whitefish; however, similar avoidance of Area 8 is expected. However, habitat in Area 8 is relatively diverse in comparison to habitat in the other basins of Kennady Lake, and nearshore habitat includes clean boulder/cobble substrates, as well as bedrock slopes, and shallow, silt-covered embayments with aquatic vegetation.

Arctic grayling are the only fish species in Kennady Lake known to make extensive migrations between the main basins of Kennady Lake and the Kennady Lake outlet (Stream K5). These fish migrate downstream in spring to spawn in the streams immediately downstream of the Kennady Lake outlet (Stream K5). Most of these fish then return to Kennady Lake soon after spawning. Lake trout and northern pike are also known to migrate out of Kennady Lake in spring but in small numbers; lake trout presumably migrate to feed on congregations of spawning Arctic grayling, and northern pike migrate to spawn in weedy bays and flooded riparian tundra in downstream lakes.

Fish are known to use sub-optimal habitat (Birtwell and Korstrom 2002; Birtwell et al. 1999; Jones and Tonn 2004) and, therefore, individuals of each species may persist, but it is uncertain whether residual populations of round whitefish and lake trout will persist in Area 8 during mine operations. The main habitat factors that make the persistence of residual populations of these fish species in Area 8 uncertain include the following:

- Area 8 is shallow and does not provide the same cover or refuge from higher summer water temperatures that is available in other, deeper basins of Kennady Lake; and
- the shallower depth and lower dissolved oxygen levels suggests that the volume of overwintering habitat in Area 8 is smaller than in the other basins of Kennady Lake.

Under baseline conditions, winter dissolved oxygen concentrations were lower in Area 8 compared to Areas 3, 5, and 6. Lake trout and round whitefish are salmonids that generally require higher dissolved oxygen concentrations than non-salmonid species. For example, optimal dissolved oxygen habitat for lake trout is greater than 6 mg/L (MacLean et al. 1990; Clark et al. 2004; Ryan and Marshall 1994; Marshall 1996; Evans 2005).

In Areas 3, 5, and 6, minimum winter dissolved oxygen concentrations were generally higher than 6 mg/L at depths down to 8 to 9 m. In comparison, in Area 8, minimum winter dissolved oxygen concentrations were generally higher than 6 mg/L at depths down to 6 m, resulting in more limited overwintering habitat at shallower depths. Although water with dissolved oxygen concentrations exceeding 6 mg/L does exist at depths less than 4 m, ice thickness is typically 2 m in winter and the volume of suitably oxygenated water for salmonids in Area 8 is less than other basins in Kennady Lake. The change in lake levels in Area 8 during operations is small (i.e., less than 0.1 m) and no change in trophic status is predicted based on increased nutrient concentrations; as a result, overwintering habitat conditions are not expected to be affected. However, as a result of the existing overwintering limitations in Area 8 and the elimination of alternative overwintering refugia in Areas 2 through 7, lake trout and round whitefish may not continue to persist in Area 8 throughout the operational period.

Northern pike are more likely to persist in Area 8 than round whitefish or lake trout because they can tolerate lower dissolved oxygen concentrations and because aquatic vegetation is relatively common in Area 8 compared to other basins of Kennady Lake. Northern pike are tolerant of low dissolved oxygen concentrations (less than 3 mg/L) (Ford et al. 1995; Casselman and Lewis 1996) and will be able to overwinter successfully in Area 8. Vegetation for northern pike spawning and rearing is typically found in shallow embayments along the southern shoreline and near the Kennady Lake outlet (Stream K5). Weedy areas of lakes downstream of Kennady Lake may also provide spawning and feeding habitat for Kennady Lake northern pike. Northern pike residing in Kennady Lake currently use aquatic vegetation in lakes of the D watershed for spawning and rearing; this habitat will be unavailable to northern pike in Area 8 during operations. Although northern pike residing in Area 8 during operations will be able to use aquatic vegetation in Area 8 and in lakes and streams downstream for spawning and rearing, it is likely that the isolation of Area 8 from the D watershed will affect northern pike. There may also be a reduction in potential prey availability in Area 8 compared to that of the entire Kennady Lake (including round whitefish and juvenile lake trout). Therefore, there may a reduction in the growth and overall production of the northern pike population in Area 8.

Arctic grayling show considerable low oxygen tolerance for salmonids (Eriksen 1975 as cited in Hubert et al. 1985); although the overwintering habitat in Area 8 will be more limited than currently exists in Kennady Lake, it is expected that

Arctic grayling will be to persist in Area 8 during isolation. Although some of Kennady Lake Arctic grayling spawning and rearing occurs in the tributaries upstream of Kennady Lake, most takes place in streams downstream of Area 8. Burbot are also expected to persist in Area 8 because they can forage successfully on the bottom or among boulders along the shoreline and can use limited overwintering habitat more effectively than the salmonids.

Populations of small-bodied fish species such as lake chub, ninespine stickleback, and slimy sculpin are more likely to persist in Area 8 than the larger-bodied fish species. This is due to their ability to find suitable cover in the boulder substrates present along the shoreline and their greater tolerance for lower dissolved oxygen concentrations than salmonids. The diversity of habitat in Area 8 is expected to provide the entire habitat necessary for all of the smaller-bodied fish species; as a result, the lack of access to small lakes within the Kennady Lake watershed is not expected to affect the small-bodied fish populations in Area 8.

Changes to Fish Migrations from Downstream Fish Communities

There will be flow changes in the Area 8 outlet channel (Stream K5), which will affect fish migration into and out of Area 8 during operations. Effects to fish and fish habitat from alteration of flows in Stream K5 are assessed in Section 9.10.

8.10.3.5 Effects of Dust Deposition on Fish and Fish Habitat

Total Suspended Particulate Deposition

The increased deposition of dust may enter surface waters, particularly during spring freshet, and could result in increased concentrations of suspended sediments in lake water. The spatial extent of dust and metal deposition is anticipated to be restricted to localized areas within and close to the Project footprint, with maximum deposition expected to occur near haul roads along the southern, western, and eastern boundary of the development area, and primarily reflect winter fugitive road dust emissions (Section 11.4 Subject of Note: Air Quality). The concentrations of TSS in nearby lakes may be elevated during and after freshet (Section 8.8.3.1, Table 8.8-11). The predicted maximum TSS concentrations for fish-bearing lakes range from 10 to 69 mg/L. The largest predicted maximum TSS concentrations are for lakes D2 (69 mg/L) and I1 (64 mg/L), with all of the other lakes being less than 30 mg/L.

The nature and extent of adverse effects of increased TSS on fish is influenced by both the TSS concentration and the duration of exposure. Fish can tolerate low TSS concentrations for long periods and high concentrations for short periods without suffering adverse effects.

The period of elevated TSS in affected lakes is expected to be short, where the largest load of suspended sediments to surface waters during the year will occur during spring freshet, when dust deposited to snow during winter and eroded materials enter surface waters. Sediment inputs during other times of the year are anticipated to be sporadic and too small to result in measurable changes in TSS concentrations in lakes. The length of the freshet period is estimated to range from approximately a few days to a few weeks, depending on lake size. The particles would be expected to settle fairly quickly, within less than a month.

Based on the Newcombe and Jensen (1996) dose-response relationship, the Severity of Effect (SEV) values suggest that exposure to peak TSS concentrations such as those estimated to occur could cause responses ranging from moderate to major physiological stress (i.e., reduction in feeding rate and feeding success). No lethal and paralethal effects would be anticipated. This is likely an overestimation for the following reasons:

- Predicted changes in TSS are considered to be conservative (high) estimates of the maximum potential changes that could occur during construction and operations (See Section 8.8.1.3).
- The period of exposure in the dose-response relationship is to peak concentrations; however, the peak levels are transitory, with the particles settling fairly quickly after snowmelt. As a result, the model likely overestimates the true duration period.

Nevertheless, the overestimation was used as a worse-case scenario for the dose-response relationship; the actual response is expected to be less. Furthermore, fish are routinely exposed to higher TSS levels during spring freshet periods and would tolerate the levels in the short-term.

The increases in sediment would be too small to produce measurable effects on fish habitat. Most of the increased suspended sediment will occur during spring freshet. Although it will settle out of the water column fairly quickly, the high water levels, wave action, and currents will move the sediment off any sensitive habitat areas in the nearshore areas of lakes (e.g., spawning shoals or vegetation) into the deeper main basin of the lake.

In summary, effects of TSS from dust and particulate deposition are expected to be localized in the immediate vicinity of the Project and temporally restricted to the period during and after freshet.

Aquatic Health

Potential effects to aquatic health from dust and metals deposition were evaluated in the aquatic health assessment (Section 8.9.3.1). The maximum concentrations of some metals (aluminum, cadmium, chromium, copper, iron, mercury, and silver) were predicted to exceed water quality guidelines in some lakes. However, similar to TSS, the predicted maximum metal concentrations are likely conservative estimates (Section 8.8.1.1.2); the spatial extent of the dust and metals deposition is also expected to be restricted to localized areas and occur primarily for a short period during spring freshet. Given the conservatism in the predicted concentrations, and the length of the exposure to elevated concentrations, the potential for adverse effects to aquatic health from dust and metals deposition was considered in the aquatic health assessment to be low (Section 8.9.3.1), with follow-up monitoring being undertaken to confirm. As a result, no effects to fish populations or communities would be expected to occur from changes in aquatic health.

8.10.4 Effects Analysis Results – Closure and Post-closure

8.10.4.1 Effects of Development of Fish Habitat Compensation Works on Fish and Fish Habitat

To compensate for habitat permanently lost or altered due to proposed mine development (as described in Section 8.10.3.1), and eliminate potential adverse effects due to changes in habitat area, the Project includes a habitat compensation plan designed to create new fish habitat (see CCP, Section 3, Appendix 3.II). The objective of the plan is to provide compensation habitats to offset predicted habitat losses so that there is no net loss of fish habitat according to DFO's Fish Habitat Management Policy (DFO 1986, 1998, 2006).

Several of the identified compensation options focus on the construction of habitat structures within specific areas of Kennady Lake. Others focus on opportunities for habitat compensation in adjacent areas. Although some of the habitat compensation works may potentially be developed during the operations phase of the Project, most of the compensation habitat will be developed at closure. Compensation features will be permanent structures designed to provide habitat for the fish community that will be re-established in the Kennady Lake watershed after closure. The following options have been identified:

- Option 1a: raising the water level of some lakes to the west of Kennady Lake (in the D watershed) to a level greater than what would be required only for development of the Project through construction of impounding dykes.

- Option 1b: raising the water level of some lakes to the west of Kennady Lake (in the D, E, and N watersheds) to the same level as in Option 1a, but creating more habitat than Option 1a by involving more lakes and land area.
- Option 1c: additional raising, after mine closure, the water level in the flooded area created by Option 1b.
- Option 2: raising Lake A3 to a greater elevation than would be only for development of the Project.
- Option 3: constructing finger reefs in Areas 6 and 7.
- Option 4: developing habitat enhancement structures in Area 8.
- Option 5: constructing shallow littoral and reef habitat structures on the shallow portions of the backfilled Hearne Pit within Kennady Lake.
- Option 6: constructing shallow littoral and reef habitat structures on the shallow portions of the backfilled 5034 Pit within Kennady Lake.
- Option 7: developing some shallow habitat structures within Kennady Lake around the rim of the Tuzo Pit.
- Option 8: developing a Dyke B habitat structure within Kennady Lake after closure.
- Option 9: constructing impounding dykes to the south of Area 7 to raise Area 8 and Lakes L2, L3, and L13 (would also raise water levels in the remaining portions of Kennady Lake at closure).
- Option 10: widening the top bench of the Tuzo and 5034 mine pits to create shelf areas where they extend onto land.

The proposed fish habitat compensation plan consists of a combination of the compensation options listed above. The preferred options include Options 1b and 1c (raising the water level in lakes to the east of Kennady Lake), Option 2 (raising the level of Lake A3), and Option 10 (widening the top bench of mine pits where they extend onto land. Also included in the proposed compensation plan are Options 3 and 4 (construction of habitat enhancement features in Areas 6, 7 and 8) and Option 8 (the Dyke B habitat structure).

The amount of compensation habitat, in terms of surface area, provided by the proposed compensation plan is summarized in Table 8.10-9. Quantification of habitat gains in terms of HUs, and determination of compensation ratios based on HUs, will be completed as part of the development of a detailed compensation plan to be completed in 2011. More details on the various options and the proposed fish habitat compensation plan are provided in Section 3, Appendix 3.II.

Table 8.10-9 Summary of Fish Habitat Compensation Achieved with the Proposed Conceptual Compensation Plan

Compensation Description	Compensation Habitat Area (ha)	
	During Operations	After Closure
Newly Created Habitat		
Option 1b – Construction of Impounding Dykes F, G, E1, and N14 to the west of Kennady Lake to raise Lakes D2, D3, E1, and N14 to 428 masl elevation	149.7	–
Option 1c – After closure, further raise the water level in Lakes D2, D3, E1, and N14, and the surrounding area, to 429 masl and reconnect the flooded area to Kennady Lake through Lake D1	–	195.9
Option 2 – Construction of Impounding Dyke C between Area 1 and Lake A3, Dyke A3 to the north of Lake A3, and Dyke N10 between Lakes A3 and N10 to raise Lake A3 to 427.5 masl elevation	31.1	31.1
Option 10 – Widening the top bench of pits (to create shelf areas) where they extend onto land	–	13.7
Altered Areas Reclaimed and Submerged at Closure		
Hearne Pit [(a)]	–	16.0
5034 Pit [(a)]	–	35.0
Tuzo Pit [(a)]	–	35.2
Dykes A, B, J, K and N	–	23.8
Road in Area 6	–	4.0
Water Collection Pond Berms CP3, CP4, CP5, and CP6	–	1.3
Mine rock areas [(b)]	–	25.3
Total	**180.8**	**381.3**
Compensation Ratios (gains:losses)[(c)]	**0.65**	**1.37**

(a) Areas for these options are entire pit areas, including habitat features along the edges and deep-water areas.
(b) Mine rock piles with final surface elevations between 410.0 and 418.0 masl are considered as compensation habitat.
(c) Calculated based on total area of permanently lost habitat and physically altered and re-submerged habitat (Section 8.10.3.1).

masl = metres above sea level; ha = hectares.

8.10.4.2 Effects of Re-diverting B, D, and E Watersheds to Kennady Lake

At closure, the natural drainage of the B, D, and E watersheds to Kennady Lake will be restored. Dykes F and G will be breached and flow from these watersheds will be re-diverted to Kennady Lake through D1. In the A watershed, Dyke C will be permanent and Lake A3 will continue to flow to the L watershed.

Changes to Lake Levels and Lake Areas

At closure, water levels in the raised Lakes D2-D3 and E1 will decrease relative to operations when the D and E watersheds are re-diverted to Kennady Lake (Table 8.10-10), and D2 and D3 will once again form separate lakes. These changes in lake levels will lead to a decrease in littoral area and lake volume; as a result, there may be corresponding decreases in the availability and suitability of fish habitat in the lakes. However, as the water levels will be returning to pre-Project (i.e., baseline) conditions, the fish and benthic invertebrate communities within the lakes will adjust to the lowered lake levels. As described in Section 8.7.4.3, the restored baseline lake shorelines are expected to remain stable. Habitat conditions for spawning, rearing and overwintering will be similar to pre-Project conditions. As a result, the change would not be expected to have a substantive effect on fish populations within the D and E watersheds. Water levels in Lake A3 will remain the same as during operations. No changes in B1, D1 and D7 will occur from the Project (Table 8.10-10).

Table 8.10-10 Lake Areas and Depths in Diverted Lakes of the A, B, D and E Watersheds by Project Phase

Parameter	Project Phase	Lake						
		A3	B1	D1	D2	D3	D7	E1
Lake area (ha)	Baseline	23.8	8.2	1.9	12.5	38.4	40.2	20.2
	Operations	46.6	8.2	1.9	104		40.2	27.0
	Closure	46.6	8.2	1.9	12.5	38.4	40.2	20.2
	Post-Closure	46.6	8.2	1.9	12.5	38.4	40.2	20.2
Maximum depth (m)	Baseline	12.4	4.1	3.8	1.0	3.0	4.5	3.9
	Operations	15.9	4.1	3.8	3.8	4.6	4.5	4.7
	Closure	15.9	4.1	3.8	1.0	3.0	4.5	3.9
	Post-Closure	15.9	4.1	3.8	1.0	3.0	4.5	3.9

ha = hectare; m = metre.

Changes to Lower Trophic Levels

Total biomass of plankton and benthic invertebrate communities in some lakes in the B, D and E watersheds will decrease due the decreased habitat areas compared to operations. Although productivity of lower trophic communities in these lakes is not expected to be negatively affected by the diversions back to the Kennady Lake watershed, there will be a period of adjustment to the new water level.

Changes to Fish Migrations

In the B, D and E watersheds, the dykes, diversion channels, and other associated infrastructure will be decommissioned. Where possible, the watersheds will be reconnected to Kennady Lake along previous connecting streams. Additional cobble and boulder placement will occur to reduce erosion potential where necessary. These streams will provide fish habitat similar to that currently present in these connecting channels, including Arctic grayling spawning habitat. Fish salvage will be conducted as appropriate during decommissioning the diversion channels (i.e., the connections between the B, D, and E watersheds and the respective lakes in N basin).

Until the water quality in Kennady Lake is deemed suitable for fish, exclusion measures will be taken to limit the initial migration of large-bodied fish from the upper B, D, and E watersheds into Kennady Lake. Mitigation measures will be designed to target large-bodied fish, such as northern pike, burbot, lake trout, and Arctic grayling. However, benthic invertebrates, small forage fish and some juvenile life stages would be expected to pass through the exclusion measures into Kennady Lake. It is anticipated that during the initial period of refilling, some mortality of the incoming small-bodied fish is likely to occur, because of insufficient water depths and possibly elevated levels of turbidity.

During the refilling period, there will be a period of time where the B, D, and E watersheds will be not be connected for fish migration to a large lake (e.g., Kennady Lake or the N lakes). However, similar to Section 8.10.3.3, the stock of most large-bodied fish species is expected to be maintained in the B, D, and E watersheds over this period. Based on lake areas and depths (Table 8.10-10), it is expected that the lakes within the B, D, and E watersheds will provide suitable habitat for fish species, such as Arctic grayling, burbot, and northern pike (i.e., spawning, rearing and overwintering habitat). When the water quality is suitable to support aquatic life, and stable plankton, benthic invertebrate, and forage fish communities have become established, large-bodied fish from the B, D, and E sub-watersheds will be able to freely immigrate to Kennady Lake and become brood stock for recolonization.

The reconnection may also allow fish species not previously present in Kennady Lake to become introduced to Kennady Lake from the N basin. For example, cisco and sucker species from N16 could enter the D and E lakes during operations and then move into Kennady Lake after the connection is restored. More information on the expected re-establishment and recovery of the fish community in Kennady Lake is provided in Section 8.11.

8.10.4.3 Effects of Continued Isolation of Area 8 During Refilling on Fish and Fish Habitat

During refilling of Kennady Lake (i.e., closure), Area 8 will remain effectively isolated from the remainder of Kennady Lake. Water quality in Area 8 is predicted to remain stable during refilling the remainder of Kennady Lake (Section 8.8.4.1.1). Discharges from Kennady Lake will not occur until water quality conditions are deemed acceptable for release. As a result, effects to the fish and fish habitat in Area 8 will be similar to those identified in Section 8.10.3.4 above.

8.10.4.4 Effects to Fish and Fish Habitat in Kennady Lake during Post-Closure

The following section describes the effects on fish and fish habitat from reconnection of Kennady Lake to Area 8 and associated changes in water quality. Recovery of the Kennady Lake fish community after refilling is discussed in Section 8.11.

8.10.4.4.1 Effects of Changes in Nutrient Levels

As discussed in the water quality assessment (Section 8.8.4.1), phosphorus concentrations in Kennady Lake are projected to gradually increase to steady state concentrations during post-closure due to seepage from materials located in the mine rock piles, Coarse PK Pile and the Fine PKC Facility. The Fine PKC facility is the largest contributing source of phosphorus. Using a combination of mitigation strategies, De Beers is committed to incorporating additional mitigation to achieve a long-term maximum steady state total phosphorus concentration of 0.018 mg/L in Kennady Lake.

After reconnection of the refilled Kennady Lake to Area 8, concentrations of nutrients are predicted to be higher than during pre-development conditions. Because refilled areas of Kennady Lake will have elevated nutrient concentrations from loadings associated with the Project, reconnection will result in a rapid increase in both total phosphorus and total nitrogen concentrations in Area 8 over a period of about five years (Section 8.8.4.1.1). Thereafter, total phosphorus will continue to gradually increase to the long-term steady-state concentration, reflecting inputs from other lake areas and basin hydrology, while total nitrogen will decline from a maximum of 6.4 mg/L to about double the pre-development concentration over a period of about 50 years. The predicted long-term concentration of total phosphorus in Kennady Lake is 0.018 mg/L and total nitrogen is 0.8 mg/L (Section 8.8.4.1.1). Slightly lower long-term total phosphorus (0.016 mg/L) and total nitrogen (0.76 mg/L) concentrations are predicted in Area 8, which will receive additional runoff from its watershed. While Kennady Lake is oligotrophic under baseline conditions, based on these values, the post-closure trophic status of Kennady Lake is predicted to be mesotrophic.

Phosphorus generally is the limiting nutrient for primary production in most Canadian Shield lakes (Schindler 1974), because of its scarcity in bedrock and overburden in Shield watersheds and efficient retention by upland forests and wetlands (Allan et al. 1993; Devito et al. 1989 cited in Steedman et al. 2004). Studies have shown that total phosphorus is the nutrient limiting primary productivity and also fish production in lakes (Dillon et al. 2004). Productivity is especially low in arctic waters, as arctic soils are shallow and frozen for most of the year, so the weathering rate and hence production of dissolved nutrients is slower than farther south.

Under baseline conditions, Kennady Lake is a phosphorus-limited system, as indicated by an N to P molar ratio of about 150, well above the range of values where the shift to N limitation may occur (10 to 30; Wetzel 2001; Environment Canada 2004). Despite the predicted changes in total nitrogen and total phosphorus concentrations along different trajectories during post-closure, the N to P molar ratio is predicted to remain greater than or equal to about 100 throughout post closure and over the long-term. Therefore, Kennady Lake is expected to remain phosphorus limited.

Changes to Lower Trophic Communities

Phytoplankton

Some differences in phytoplankton community structure were documented between Area 8 and Areas 2 through 7 during baseline studies (Section 8.3 and Addendum JJ). In Area 8, Cryptophyta dominated the phytoplankton biomass. Chrysophyta and diatoms accounted for a large proportion of the phytoplankton biomass in Areas 2 through 7. Chlorophyta and cyanobacteria were present in all basins of Kennady Lake, but accounted for a small amount of the biomass. The differences in the phytoplankton community composition between Area 8 and other basins of Kennady Lake may be a reflection of variation in physical conditions among basins.

There is abundant evidence in the scientific literature that increased concentrations of the limiting nutrient result in increased primary productivity and phytoplankton biomass. For example, LeBrasseur et al. (1978) found that during annual fertilizer treatment in Great Central Lake, British Columbia, mean summer primary production increased five-fold. Hyatt and Stockner (1985) found that addition of fertilizer to several British Columbia coastal lakes increased autotrophic (phytoplankton) and heterotrophic (bacteria) production. Whole lake fertilization experiments (i.e., two to three-fold increases in total phosphorus concentration) performed at small oligotrophic lakes in Saqvaqjuac in the Canadian central arctic from 1978 to 1983 found increases in primary and secondary production (Jorgenson et al. 1992; Welch et al. 1988; Welch et al. 1989). Increases in summer phytoplankton biomass documented by some of these studies were several-fold relative to background biomass estimates. Based on these studies, the predicted change in the trophic status of Kennady Lake from oligotrophic to mesotrophic is expected to result in an increase in summer phytoplankton biomass, with the magnitude of the increase depending on nutrient concentrations and physical factors (e.g., temperature, wind-induced mixing).

The predicted change in trophic status of Kennady Lake will also result in altered species composition of phytoplankton and shifts in dominance at the level of major phytoplankton group. The observed Cryptophyta dominance in Kennady Lake under baseline conditions is likely to change to dominance by other groups, such as diatoms, Chlorophyta, and possibly other groups, which usually dominate mesotrophic and eutrophic lakes (Wetzel 2001; Watson et al. 1997; Kalff 2002).

Although an increase in the proportion of cyanobacteria may occur as a result of increased nutrient concentrations in Kennady Lake, a shift to strong cyanobacteria dominance is unlikely. In nitrogen limited lakes, cyanobacteria are capable of directly fixing nitrogen, which can result in their dominance of the phytoplankton community (Environment Canada 2004). However, Kennady Lake is expected to remain phosphorus-limited. In addition, the prediction of total phosphorus concentrations was based on conservative assumptions regarding sources, suggesting that actual concentrations may be lower than the predicted long-term value of 0.018 mg/L. A total phosphorus concentration of 0.03 mg/L has been suggested as a threshold, below which the risk of cyanobacteria dominance is only 0 to 10% (Downing et al. 2001).

Zooplankton

Despite differences in sampling methods, Copepoda consistently dominated the Kennady Lake zooplankton community between 2001 and 2010 (Section 8.3 and Addendum JJ). Species richness for Copepoda has been low during this time, which was expected (Williamson and Reid 2001). Cladocera accounted for a high proportion of the zooplankton biomass, particularly in Area 8.

Zooplankton biomass is frequently enhanced by nutrient enrichment of lakes. For example, LeBrasseur et al. (1978) found that during annual fertilizer treatment in Great Central Lake, British Columbia, mean summer primary production increased five-fold and zooplankton standing stock increased nine-fold. Other studies have also found that phosphorus enrichment can increase zooplankton biomass indirectly, through enhanced food availability (Hanson and Peters 1984; Shortreed and Stockner 1986).

The predicted increase in primary productivity in Kennady Lake is expected to result in increased secondary productivity and biomass of the zooplankton community, reflecting the increased amount of available food for zooplankton. However, because energy transfer between trophic levels is inefficient (McCauley and Kalff 1981; Kalff 2002), the proportional increase in zooplankton biomass caused by increased nutrient concentrations will likely be lower than that in phytoplankton biomass.

Changes in zooplankton community composition are also possible, although difficult to predict, because zooplankton species composition is frequently more strongly controlled by predation (i.e., top-down) than food availability (i.e., bottom-up) (McQueen et al. 1986; Carpenter 1989; Carpenter et al. 2001).

Benthic Invertebrates

The benthic invertebrate community in Area 8 consisted mostly of midges (Chironomidae), fingernail clams (Pelecypoda: Sphaeriidae), and roundworms (Nematoda) during baseline studies conducted in 2004 (Section 8.3) and in 2007 (Addendum JJ). Communities in other basins of Kennady Lake were similar, but also included greater numbers of snails (Gastropoda) and aquatic worms (Oligochaeta). Invertebrate abundance was low in all basins, consistent with the oligotrophic status of Kennady Lake under baseline conditions.

Nutrient enrichment has been found to result in increased benthic invertebrate biomass (Rasmussen and Kalff 1987; Jorgenson et al. 1992; Clarke et al. 1997), but no response to enrichment has also been documented (Dinsmore et al. 1999). The predicted increase in nutrient concentrations and primary productivity in Kennady Lake are expected to result in an increase in benthic invertebrate abundance and biomass, reflecting the increased food supply. The response by invertebrates may be delayed by several years relative to that by phytoplankton, as observed by Hershey (1992) in an experimentally fertilized lake. In addition, declining late winter dissolved oxygen concentrations may limit the response of benthic invertebrates to nutrient enrichment, by eliminating organisms sensitive to low oxygen concentrations (Rosenberg and Resh 1993).

A shift in composition of the benthic invertebrate community is also expected during post-closure, as observed by studies of artificially fertilized lakes. For example, Jorgenson et al. (1992) observed different responses by different invertebrate taxa to enrichment, ranging from two-fold increases in caddisfly biomass, to five to ten-fold increases in the biomass of amphipods. Some groups (e.g., snails) are able to take advantage of the increased food supply in fertilized lakes in the absence or predators, while abundances of others (e.g., chironomids) may be effectively controlled by predation (Hershey 1992), which may also contribute to shifts on community composition. Lower late winter dissolved oxygen concentration than that under baseline conditions may also alter community composition by eliminating sensitive taxa, thereby favouring certain species of chironomids and aquatic worms, which are tolerant of low dissolved oxygen concentrations (Rosenberg and Resh 1993).

Effects in other unproductive sub-arctic lakes undergoing nutrient enrichment related to operating diamond mines, such as Snap Lake and Lac de Gras, also provide an indication of expected benthic community changes in Kennady Lake during the early stages of enrichment. In both Snap Lake and Lac de Gras, total phosphorus has increased slightly (remains <0.01 mg/L, on average) and total nitrogen has increased moderately during the first decade of operations. This resulted in increases in total invertebrate density and densities of fingernail clams (Pisidiidae), snails (*Valvata*) and a number of chironomid genera (Snap Lake: *Microtendipes*, *Corynocera*, *Procladius*; Lac de Gras: *Procladius*, *Heterotrissocladius*, *Micropsectra*) in affected areas of these lakes, without apparent changes in densities of other invertebrates, or in other benthic community variables (richness, diversity, dominance, evenness) (Diavik 2011; De Beers 2011). The changes observed in these lakes can be expected during the early stages of enrichment in Kennady Lake, because in terms of total phosphorus concentrations, both Snap Lake and Lac de Gras have remained oligotrophic to date, whereas projected nutrient concentrations in Kennady Lake are in the mesotrophic range.

Changes to Fish and Fish Habitat

Many studies have examined the effects of nutrient enrichment on lakes from domestic sewage, clearcut logging, and urban or agricultural land drainage. There have been concerns related to the threat posed to cold-water fish populations, especially lake trout, from nutrient enrichment. Lake trout populations generally require large volumes of cold, well-oxygenated water to thrive (Martin and Olver 1980). Lake trout are often considered vulnerable to environmental changes due to their slow growth, late maturity, and low reproductive potential, as well as their habitat limitations (cold, well-oxygenated water).

Increases in phosphorus concentration and change in trophic status may have the following potential effects on fish and fish habitat:

- increase in fish production through increased primary and secondary productivity;
- elevated nutrient concentrations in lake sediments and the water column causing excessive plant and algae growth and subsequent degradation of fish spawning and nursery habitat;
- loss of dissolved oxygen in the deep water areas of the lake; and
- changes to fish community structure.

Fish Production

The effect of increased primary and secondary production on fish in lakes is complex and is dependent on several factors. These factors include the physical and chemical conditions of the water and lake sediments (Schindler 1974), the complexity of the food web (Schindler 1974; Carpenter et al. 1985; Elser et al. 1990), the efficiency of energy transfers between trophic levels (McQueen 1990; Micheli 1999), and the relative importance of “bottom-up” (i.e., resource availability) or “top-down” (i.e., predation) control of lake productivity (Power 1992; Carpenter et al. 1985; McQueen et al. 1986). Top-down and bottom-up controls both operate in aquatic ecosystems (McQueen et al. 1986; Power 1992) but the relative importance of top-down effects increases in oligotrophic systems (McQueen et al. 1986).

As Kennady Lake is currently a phosphorus-limited system, it is expected that an increase in phosphorus concentration will increase fish production due to increases in primary and secondary productivity. As described above, phosphorus is the nutrient that controls algal biomass in almost all lakes in the Boreal ecozone of Canada (Dillon et al. 2004). Phytoplankton is a very important component of the food web in all lakes, and algal abundance may strongly influence the biomass and productivity of higher trophic levels, including fish.

Studies have shown that nutrients, and in particular TP, control the rate of fish production in lakes (Colby et al. 1972; Hanson and Leggett 1982; Plante and Downing 1993), including cold-water fish production (Dillon et al. 2004). As research showed that phosphorus is the nutrient limiting primary productivity and also fish production in lakes, a number of investigations were conducted on the experimental fertilization of oligotrophic lakes to increase salmonid production. For example, LeBrasseur et al. (1978) found that during annual fertilizer treatment in Great Central Lake, British Columbia, mean summer primary production increased five-fold, zooplankton standing stock increased nine-fold, the percentage survival from estimated potential egg deposition to juvenile sockeye salmon (*Oncorhynchus nerka*) increased 2.6 times, and the mean stock of adult sockeye salmon increased from less than 50,000 to greater than 360,000 fish. Hyatt and Stockner (1985) found that addition of fertilizer to several oligotrophic British Columbia coastal lakes, which approximately doubled total phosphorus concentrations, increased autotrophic (phytoplankton) and heterotrophic (bacteria) production and led to larger standing stocks of zooplankton and increased in-lake growth of juvenile sockeye salmon. The researchers suggested that this may also lead to increases in harvestable surplus sockeye adults and provided evidence to support predictions of gains in sockeye stock returns due to the application of lake fertilization as an enhancement technique (Hyatt and Stockner 1985). In fertilization experiments in Norway, Johannessen et al. (1984, cited in Dillon et al. 2004) found that the length and weight of brown trout (*Salmo trutta*) increased during the three seasons of fertilization (Johannessen et al. 1984 cited in Dillon et al. 2004). Johnston et al. (1990) found that whole-river fertilization of the Keough River, British Columbia, which increased total phosphorus concentration from <1 µg/L to 10-15 µg/L, increased the size of steelhead trout (*Oncrorhynchus mykiss*) and coho salmon (*Oncorhynchus kisutch*) fry. Nutrient additions to the North Arm of Kootenay Lake, BC, increased the biomass of phytoplankton, zooplankton, and kokanee salmon (*Oncorhynchus nerka*) in the lake (Ashley et al. 1997 cited in Dillon et al. 2004).

Whole lake fertilization experiments were also performed at small oligotrophic lakes in Saqvaqjuac, in the Canadian central arctic, from 1978 to 1983. During these studies, total phosphorus concentration was raised two to three-fold depending on lake, starting from 0.005 to 0.007 mg/L. The studies found increases in primary production and secondary production (i.e., macrobenthos) (Jorgenson et al. 1992; Welch et al. 1988; Welch et al. 1989). Fish populations increased in P&N Lake as a result of the fertilization, but in keeping with their relatively long life cycles, had not stabilized at the time of the last sampling in 1983 (Jorgenson et al. 1992). O'Brien et al. (2005) found increased phytoplankton growth during nutrient addition, but mixed responses in macrozooplankton and benthic invertebrates, indicating that other factors (e.g., predation) also play a role in the arctic food web. During whole-lake experimental fertilization of a small, oligotrophic arctic lake in Alaska, the researchers found increased primary productivity, chlorophyll *a,* and snail density (Lienesch et al. 2005). Lake trout density was not affected by the manipulation, but growth and average size increased. The increased growth during the period of fertilization was considered to be due to the increased food availability (Lienesch et al. 2005). Other researchers found increased fish growth and yield in lakes that underwent cultural eutrophication in the 1970s (i.e., from sewage, detergents, fertilizers, agricultural runoff, etc.) (Nümann 1972; Gascon and Leggett 1977; Hartmann and Nümann 1977; Näslund et al. 1993).

As a result of the increased nutrients in Kennady Lake during post-closure, it is expected that there will be increases in the food base for fish (zooplankton and benthic invertebrates), as well as in the small-bodied fish community (e.g., lake chub, slimy sculpin, and ninespine stickleback). Because of the increased food base, there may also be increased growth and production in the large-bodied fish species of Kennady Lake (e.g., northern pike, burbot, round whitefish, lake trout, Arctic grayling); however, this is more difficult to predict, as other factors associated with the change in trophic status will also play a role in the response of the fish population.

Habitat Changes

Increased eutrophication of the lake through increased phosphorus levels may affect fish habitat through the following:

- degradation of spawning shoals, or increased growth of aquatic vegetation, potentially affecting fish spawning and nursery habitat; and
- declines in dissolved oxygen may limit habitat availability or suitability for species that are intolerant of low dissolved oxygen levels.

Spawning Habitat

Increased nutrients in lakes may lead to increased algal growth or hypoxia on spawning areas used by fish species, such as lake trout and round whitefish. An increase in attached algae on spawning shoals can potentially affect the recruitment of lake trout or round whitefish, as reproduction relies upon the presence of suitable spawning shoal habitat for successful egg incubation and fry emergence. For example, researchers have been concerned with algal growth on lake trout spawning reefs, and increased deposition of detrital organic matter on reefs in early fall. Decay of these materials could increase ammonia and hydrogen sulphide levels and decrease dissolved oxygen if little water circulation occurs at the site (Sly 1988; Marsden et al. 1995). Oxygen depletion of interstitial water within the substrate, due to decomposition of organic material, is a concern where excessive nutrient inputs occur; dissolved oxygen concentrations less than 4.5 mg/L have been found to retard development, delay hatching, and to cause malformation of embryos (Evans et al. 1991). During whole-lake fertilization experiments in an arctic lake, hypoxic conditions that developed near the sediments were implicated in the loss of lake trout recruitment, through the mortality of overwintering embryos and decreased habitat availability (Lienesch et al. 2005). Similar concerns have also been raised for whitefish species. For example, Nümann (1972) suggested that the eggs of coregonines in the Bodensee suffered from oxygen deficiency due to eutrophication. Similarly, eutrophication and associated degradation of spawning habitat was implicated in the decline of the lake whitefish population in Lake Simcoe, Ontario (Evans et al. 1988).

Lake trout spawn in September and October in northern areas, with most spawning taking place over cobble and large gravel substrate in shallow nearshore areas of lakes. Wind and wave action is the primary mechanism keeping the spawning areas clean (Martin and Olver 1980). Wave action and currents maintain the incubating eggs free of detritus and remove metabolic wastes. Eggs remain in the substrate until hatching in early spring (i.e., March and April) (Scott and Crossman 1973; Richardson et al. 2001). Round whitefish spawn in October in northern regions, with spawning typically taking place over gravel and rubble substrates. Eggs are broadcast over the substrate and incubate until hatching some time from March to May (Scott and Crossman 1973; Richardson et al. 2001). Water currents and wave action can scour and clean reefs in the fall and maintain adequate water quality over the winter; however, too much wave action can damage eggs by blanketing them with suspended sediments or by dislodging them from the substrate (Marsden et al. 1995).

Eutrophication was reported to have resulted in increased growth of algae on lake trout spawning shoals in Lake Ontario (Evans et al. 1991; Manny et al. 1989) and degradation of shoal spawning habitat related to eutrophication was implicated as a contributing factor in the recruitment failure of lake trout and lake whitefish in Lake Simcoe, Ontario (Evans and Waring 1987; Evans et al. 1988; McMurtry et al. 1997).

In the Great Lakes, the degradation of traditional spawning habitats and its effects on spawning success by stocked lake trout has been discussed, but poorly documented (Marsden et al. 1995). Christie (1972) suggested that excessive shoal and shoreline growth of the alga *Cladophora* in response to eutrophication may have been detrimental to lake trout reproduction in Lake Ontario. Egg incubation studies at a reef in southern Lake Huron where environmental conditions were considered to be degraded by nutrient enrichment and contaminants, found that burial by sediments resulted in increased egg mortality (Manny et al. 1995); however, as there was still survival-to-hatching occurring within the incubators, suitable habitat conditions required to survival of lake trout eggs was present. Studies in Lake Ontario by Casselman (1995) found that fry survival at degraded spawning sites was inversely affected by temperature, especially if the spawning substrate is degraded by organic sedimentation, which causes increased biological oxygen demand and reduced oxygen concentrations. However, in the Great Lakes, habitat quality is only one of many factors that may affect lake trout reproduction in these lakes. Marsden et al. (1995) indicated that Lake Ontario is meso-eutrophic and yet, under certain conditions, supports relatively high lake trout egg survival. In studies of interstitial water quality at lake trout spawning sites, Sly (1988) found that although some of the spawning habitats studied were degraded, the extent to which this affected reproductive success was uncertain.

In Lake Ontario, the quality of the spawning habitat is related to the substrate size, where ideal substrate size was considered to be from 10 to 20 cm in size and angular to subangular in shape (Fitzsimons 1996). Smaller material with correspondingly reduced porosity was considered to result in the irreversible accumulation of sediment and organic material that could lead to poor water quality and low egg survival; material that was too large would provide little protection for washout from currents and allow increased access by predators (Fitzsimons 1996). Also, lake trout eggs in the Great Lakes have hatched on reefs with “poor” water quality and considerable summer growth of filamentous algae (Marsden et al. 1995).

Lake trout also were proven to be highly adaptable to spawning habitat disturbances and repeatedly selected new sites in Whitepine and Helen lakes in northern Ontario when previous spawning sites were covered with opaque plastic sheeting. The new sites were also found to produce alevins (i.e., newly hatched fish with yolk sac) indicating that recruitment was not eliminated by the habitat disturbances (Gunn and Sein 2004).

As described in Annex J (Section J4.4.7.2), fall field studies have identified potential lake trout spawning locations in Kennady Lake. The data suggest that the nearshore area of the island separating Areas 3 and 4 is an important spawning location for lake trout in Kennady Lake. This area has characteristics considered optimal for lake trout spawning including the following:

- cobble, rubble and large gravel substrates (Richardson et al. 2001);
- depths less than 5 m near steep drop-offs (Martin and Olver 1980; MacLean et al. 1990); and
- areas kept clean by wind-generated wave action (Richardson et al. 2001).

Like most of Kennady Lake, the island has clean/boulder substrates; however, the island is directly adjacent to deep (greater than 10 m) areas on both sides and is exposed to the largest fetch (greater than 1.5 km) in the lake, Other nearshore areas with suitable habitat for lake trout spawning have also been identified in all basins of Kennady Lake, including the shorelines of Areas 3 and 4, Round whitefish spawning habitat was also present along the shorelines of the lake, including Areas 3 and 4. Both species spawn at depths greater than 2 m to avoid freezing of eggs and ice scour damage.

Due to increased nutrient levels, there may be an increase in algae or sediment on spawning shoals in Kennady Lake. However, most of the high quality spawning habitat in Kennady Lake is in the 2 to 4 m depth range, which is kept clean of silt and fine organic debris by wave-generated currents. It is expected that the current and wave action will lessen the growth of algae on these exposed shoals. This wind and wave action would also minimize the accumulation of sediment and organic matter in the interstices of the shoals. The island separating Areas 3 and 4 is considered to be an important lake trout spawning area in Kennady Lake. It is expected that due to the fetch, drop-off, and wind-generated wave action at this site, the potential for algal growth or accumulation of sediment in the interstices would be minimized and that the habitat would remain suitable for lake trout spawning.

As well, as indicated in the CCP (Appendix 3.II), additional habitat enhancement structures may be constructed, including finger reefs in Kennady Lake and habitat structures on the decommissioned mine pits/dykes. Lake trout have been shown to use secondary spawning sites when traditional spawning grounds become unavailable (McAughey and Gunn 1995; Gunn 1995; Gunn and Sein 2000) and artificial structures have been successfully used by lake trout for spawning and rearing (Fitzsimons 1996). A review of lake trout spawning areas in the Great Lakes found consistently higher abundance of eggs, fry, and young-of-the-year lake trout associated with artificial structures than natural spawning areas (Fitzsimons 1996).

As described in Section 8.8.4.1.1, the refilled Kennady Lake may be subject to a higher winter oxygen demand than under baseline conditions. However, the surface zone of the water column (i.e., under ice to 6 m) is expected to remain well oxygenated through the period of egg incubation; the majority of lake trout and round whitefish spawning in Kennady Lake occurs within this zone (i.e., 2 to 4 m in depth). There may be localized oxygen deficiencies in deeper interstitial water within the substrate due to decomposition of organic material, in areas that are not as exposed to wind and wave action; this may affect the suitability and availability of spawning habitat for lake trout as their eggs incubate in interstitial spaces. However, it is expected that there will continue to be areas of suitable spawning and egg incubation habitat for lake trout and round whitefish in the refilled Kennady Lake, and that natural recruitment will occur. Although spawning habitat conditions are expected to be suitable for lake trout, spawning and egg incubation habitat may be a limiting factor for this species, which may affect the abundance of the population. This may, however, be alleviated through the construction of additional spawning structures.

On the other hand, northern pike are dependent on aquatic vegetation for spawning and rearing. The presence of aquatic vegetation in Kennady Lake is currently limited by physical factors, such as rocky substrates and wave action. However, existing macrophyte beds in sheltered areas may benefit from the increased nutrient concentrations, which would be reflected in increased plant abundance and productivity.

Dissolved Oxygen

Summer

Increased primary productivity also has the potential to decrease dissolved oxygen concentrations due to increased oxygen demand by the bacterial decomposition of organic matter (Colby et al. 1972; Cornett and Rigler 1979; Welsh and Perkins 1979; Molot et al. 1991; Wetzel 2001). Oxygen depletion can limit the production and survival of certain fish species, particularly species intolerant of low dissolved oxygen concentrations, such as lake trout (Ryan and Marshall 1994; Marshall 1996; Evans 2005; Shuter and Lester 2004). Lake trout are intolerant of dissolved oxygen concentrations below about 5 to 6 mg/L (Sellers et al. 1998). Two indices have been used by lake managers to define lake trout habitat: optimal habitat (the volume of water 10°C or colder, with dissolved oxygen of 6 mg/L or higher); and usable habitat (the volume of water 15.5°C or colder, with dissolved oxygen of 4 mg/L or higher) (MacLean et al. 1990; Evans et al. 1991; Ryan and Marshall 1994; Dillon et al. 2003). Schindler et al. (1996) used 10°C and 4 mg/L to define summer refugia for lake trout in the Experimental Lakes Area; however, it was noted that there can be considerable lake-to-lake variation in the preferred temperature of individual species. Sellers et al. (1998) found that lake trout in the Experimental Lakes Area inhabit only highly oxygenated regions of lakes, with oxygen concentrations usually in excess of 6 mg/L, and tended to avoid regions having less than 5 mg/L, even when large volumes of appropriate thermal habitat were available.

For many lakes affected by increasing eutrophication due to phosphorus enrichment, the main concern is the limitations in summer habitat for cold-water fish species (especially lake trout), primarily in regards to hypolimnetic oxygen depletion in thermally stratified lakes (Evans et al. 1996; Brown and Krygier 1971; Cornet and Rigler 1979; Welsh and Perkins 1979; Molot et al. 1991; Steedman and Kushneriuk 2000; Dillon et al. 2004). As water temperatures rise in summer, adult lake trout seek the deeper, cooler (~10°C) water below the thermocline (Scott and Crossman 1973). With increasing eutrophication, the volume of suitable habitat may be further restricted through depletion of dissolved oxygen in the hypolimnion during the period of summer stratification from the increased algal biomass in the lake. In extreme situations, eutrophication can result in lost habitat for cold-water species (Evans et al. 1996). However, studies in the Experimental Lakes Area found that lake trout utilized a wide range of temperatures up to 21°C and that lake-to-lake variation in water temperatures where lake trout occurred was considerable (Sellers et al. 1998).

For southern lakes that stratify in summer, depletion of hypolimnetic dissolved oxygen is a major stress on cold-water fish species. However, arctic lakes do not warm up very much in summer because by the time the ice disappears in July, the peak of solar radiation has passed. In southern lakes, ice-off occurs earlier, so the lakes stratify with the warmer, surface water (15 to 25°C) floating on top of the colder, bottom water (4°C). In arctic lakes, the surface warming is not very great, typically only a few degrees; thermal stratification is weak, if at all, and the lakes tend to be mixed vertically. By late August, cool water and fall storms are frequent, so the lakes tend to lose heat quickly.

For the most part, Kennady Lake remains well mixed and does not typically stratify in summer due to wind-driven circulation (Section 8.3.6). Dissolved oxygen concentrations were generally uniform throughout the water column of Areas 2 to 8 of Kennady Lake during open water conditions, ranging from 9 to 16.5 mg/L. In the summer, large volumes of cool water are available throughout the lake. A weak thermocline was periodically observed in Area 6 during summer sampling, with slight decreases in dissolved oxygen below the thermocline (Section 8.3.6.2.1); however, the wind action generally keeps Kennady Lake well-mixed in summer. Mixing in large lakes is determined by topography, lake morphometry, and water colour (Fee et al. 1996). Although decreased water clarity may occur as a result of increased dissolved organic carbon (DOC) and nutrients, it is unlikely to affect lake stratification (Steedman and Kushneriuk 2000).

Kennady Lake is expected to remain cool and generally well-mixed in summer (Section 8.8.4.1.1), due to its climate and northern location; therefore, it not expected that summer oxygen depletion would become a limiting factor for cold-water fish species in the lake.

Winter

Changes to phosphorus levels may also affect winter habitat conditions within lakes. For example, Danylchuk and Tonn (2003) suggested that phosphorus enrichment, among other factors, may increase the frequency or the severity of winterkill in fathead minnows brought about by increased primary production and winter oxygen depletion. Although eutrophication can cause oxygen reductions in midwinter, these effects are usually not as critical to lake trout as are midsummer effects (Evans et al. 1991). However, in arctic lakes, there may be additional challenges with respect to oxygen depletion during winter due to the length of the season with ice cover, the depth of the ice, and the lack of inflows. For example, Lienesch et al. (2005) found that winter hypoxia became more pronounced during whole-lake fertilization experiments, although winterkill of the lake trout population in the lake did not occur during the study. In 1993 and 1994, measurements taken before ice out showed low dissolved oxygen levels (i.e., less than 6 mg/L) throughout the water column, indicating that respiration occurring throughout the winter was further depleting oxygen (Lienesch et al. 2005).

In Section 8.8.4.1.1 and Appendix 8.V, baseline data and published empirical models were used to estimate the winter oxygen depletion rates for Kennady Lake based on the predicted total phosphorus concentration of 0.018 mg/L. Due to the increase in nutrients, the winter oxygen demand in the refilled Kennady Lake is predicted to be greater than under baseline conditions. The mid-depth and bottom depth zones are subject to lower oxygen levels. The anoxic conditions at the lower levels of the refilled lake will be likely to occur earlier in the winter season and potentially extend higher up into the water column than under baseline conditions. However, it is expected that the surface zone of the water column (i.e., under ice to 6 m) should remain well oxygenated over the winter period (i.e., >8 mg/L). Under these conditions, Kennady Lake would be expected to retain sufficient levels of dissolved oxygen during winter to support fish, including lake trout. Under worse case conditions predicted using the empirical models (Appendix 8.V), dissolved oxygen levels in the upper level of the lake at the end of winter are estimated to be less than 5 mg/L (4.9 mg/L). Based on published literature on lake trout, this would be outside of the optimal (or preferred) range of dissolved oxygen levels for this species (i.e., less than 5 to 6 mg/L), but would above their tolerance limit and, therefore, would be considered useable habitat (i.e., greater than 4 mg/L). However, as the upper levels of the open Hearne and Tuzo pits are likely to remain well-oxygenated through the winter due to their depths (i.e., greater than 100 m), oxygen levels may be higher than predicted to depths that exceed the maximum depth of the other areas of Kennady Lake. It is expected that these pits will provide additional overwintering refugia for fish due to the large volumes of cold, well-oxygenated water.

Studies of winter respiration of experimentally fertilized arctic lakes found that phytoplankton production increased by 77%, whereas winter respiration rate showed a significant but relatively small increase of 19%, indicating that winter respiration lags behind lake production, and may not increase to the same extent (Welch and Bergman 1985a). Plankton respiration may be a less important component of winter respiration in arctic than temperate lakes (Welch and Bergmann 1985a). Agbeti and Smol (1995) also found that dissolved oxygen in meso-eutrophic lakes remained near saturation at the surface but declined to zero at the sediment-water interface. Studies have also shown that arctic lakes tend to circulate continually under the ice (Welch and Bergmann 1985b). Although studies have shown that most lake trout lakes are oligotrophic (Dillon et al. 2004), sensitive species, such as lake trout, are able to survive in some mesotrophic systems, as factors other than just trophic status play a role. For example, lake trout habitat is also considered to be controlled by morphometry, with lake trout lakes typically being deep, and consequently large in area (Dillon et al. 2004).

As a result, it is expected that due to the change in trophic status to mesotrophic, overwintering habitat in Kennady Lake at post-closure may become more limited for some fish species than pre-Project. Fish species that are more tolerant of low dissolved oxygen levels (e.g., lake chub, slimy sculpin, ninespine stickleback, Arctic grayling, northern pike, and burbot) will likely be able to overwinter successfully in the refilled Kennady Lake. Cold-water fish species, such as lake trout and round whitefish, are less tolerant of low dissolved oxygen levels. Although little information is available on oxygen tolerances for round whitefish, they have been captured in waters with oxygen concentrations as low as 2.6 mg/L (Hale 1981 reported in Steinhart et al. 2007); as such, round whitefish will also likely be able to overwinter in the refilled Kennedy Lake. Lake trout being the most sensitive fish species to low dissolved oxygen levels may be affected by the reduced suitability and availability of overwintering habitat in Kennady Lake. Although overwintering habitat conditions are expected to be suitable for lake trout in the refilled lake, overwintering habitat may be limiting, which may affect the abundance of the population. Lake trout may not return as the most abundant predatory fish in the lake, due to overwintering habitat limitations from reduced under-ice dissolved oxygen levels; habitat conditions may also be more favourable to other predatory fish species, such as northern pike and burbot

Change in Fish Community

Change in trophic status can change fish species richness and composition of fish assemblages. Studies have shown changes in fish communities with respect to increasing eutrophication, but many other contributing factors (e.g., introductions, exploitation, etc.) also played a role (Christie 1972; Nümann 1972; Colby et al. 1972; Evans and Waring 1987; Tammi et al. 2003). For example, reported effects of eutrophication on fish species in lakes include increased numbers of cyprinids, and reduction in salmonids and other predatory species (Colby et al. 1972; Nümann 1972; Hartman and Nümann 1977; Persson et al. 1991). Studies to investigate fish species abundance and community structure in Canadian boreal forest lakes in response to environmental variables, found that with increasing trophic status, there is an apparent shift from a community dominated by cold-water species to cool-water species, i.e., gradient shift from salmonines (trout), to coregonines (whitefish), to percids (walleye, perch), to esocids (pike); however, other variables, especially mean depth of the lake, also were important factors in species presence and abundance (Marshall and Ryan 1987). Similarly, in south Swedish lakes, Persson et al. (1991) found that with increasing productivity, salmoniformes were replaced by percids, which in turn were replaced by cyprinids; however, these authors suggested that environmental factors, such as increased water transparency and structural complexity (i.e., in the form of submerged littoral vegetation) with increasing productivity, may be the cause of the observed pattern of succession. Colby et al. (1972) suggested that with increased nutrient loading of a very oligotrophic ecosystem, species that are more tolerant of the changed conditions are likely to become dominant over time.

In a fish status survey of Nordic lakes, lakes with total phosphorus levels greater than 0.025 mg/L were found in Finland and Sweden, in areas with mainly agricultural and forestry land-use practices (Tammi et al. 2003); the fish communities were found to dominated by cyprinids in these eutrophic lakes. In this study, no fish species extinctions were found to be associated with eutrophication, with the number of fish species present found to be changed more by stocking and species introductions than through environmental changes, such as acidification and eutrophication (Tammi et al. 2003).

Due to the change in trophic status (i.e., from oligotrophic to mesotrophic) and associated habitat conditions in the refilled Kennady Lake, the fish community structure may be different from what was present pre-Project. The effect of increased primary and secondary production on fish will be partially dependent on which fish species are likely to persist in Area 8 during mining, which fish species are most likely to move into Kennady Lake from downstream and the reconnected B, D, and E watersheds, and the habitat preferences of the fish species. As discussed in Section 8.10.3.4, populations of salmonids (lake trout and round whitefish) may not persist in Area 8 during mine operations; Area 8 is relatively shallow, and will provide limited overwintering habitat to these fish species. Species more likely to persist include lake chub, slimy sculpin, ninespine stickleback, burbot, Arctic grayling, and northern pike.

In lake systems, fish community structure is determined by both abiotic (e.g., physical and chemical conditions), and biotic factors (e.g., predation, inter-species competition), as well as spatial factors (e.g., habitat, heterogeneity, and connectivity) (Jackson et al. 2001). Physical factors can include climatic conditions, temperature, and lake morphometry, whereas the principal chemical factors affecting community composition are dissolved oxygen levels and the acidity of the system (Jackson et al. 2001). Small differences in the development of winter anoxia result in very different community composition in adjacent lakes (Jackson et al 2001).

The final fish community of Kennady Lake will likely be characterized by a small-bodied forage fish community (e.g., lake chub, slimy sculpin, ninespine stickleback) and large-bodied species, such as Arctic grayling, northern pike, burbot, round whitefish, lake trout, and possibly longnose sucker. Northern pike and burbot are generally tolerant of low dissolved oxygen levels; as a result, one or both of these species may be able to re-establish as top predators within Kennady Lake. Arctic grayling show considerable low oxygen tolerance for salmonids (Eriksen 1975 as cited in Hubert et al. 1985) and have been shown to tolerate dissolved oxygen levels down to 1.5 mg/L for fish acclimated to 5°C and held at 10°C (Stewart et al. 2007). Dissolved oxygen levels of 0.6 to 4.8 mg/L have been observed in Arctic grayling overwintering areas (Alt and Furniss 1976; Kreuger 1981 as cited in Hubert et al. 1985). Based on these known habitat requirements, it is likely that Kennady Lake would continue to support Arctic grayling populations. Round whitefish will also likely re-establish in the lake, but may not recover to their previous level of abundance.

Although lake trout are expected to re-establish in the lake, the limitations on overwintering habitat from lower under-ice dissolved oxygen levels may affect the population. Lake trout may not return as the most abundant predatory fish in the lake, due to overwintering habitat limitations, and the fact that conditions may be more favourable to other predatory fish species, such as northern pike and burbot. Lake trout populations generally require large volumes of cold, well-oxygenated water to thrive (Martin and Olver 1980). Lake trout are often considered vulnerable to environmental changes due to their slow growth, late maturity, and low reproductive potential, as well as their habitat limitations (cold, well-oxygenated water).

The increase in primary productivity may result in greater densities of forage fish species, such as lake chub, ninespine stickleback, and slimy sculpin, in the refilled Kennady Lake and Area 8. This increased abundance of lake chub, ninespine stickleback, and slimy sculpin will likely be tempered by intra- and inter-species competition for food (i.e., density-dependent responses) and by predation from burbot and possibly northern pike (i.e., the “top-down” response). Juvenile burbot feed mainly on small benthic invertebrates (McPhail 2007) but switch to a diet primarily of fish by their third or four year (Thornburgh 1986, cited in Ford et al. 1995). Northern pike are found in low numbers in Kennady Lake, but are known to use wetland areas near the Kennady Lake outlet (Stream K5) and weedy areas in lakes downstream. Northern pike fed almost exclusively on fish and should benefit from the increased density of forage fish. Northern pike are tolerant of low dissolved oxygen concentrations (less than 3 mg/L) (Ford et al. 1995; Casselman and Lewis 1996) than most other large-bodied fish species and will likely be able to overwinter in Kennady Lake.

Piscivory by northern pike during recolonization of Kennady Lake may also affect the species composition. Studies have shown that predation is an important factor in determining overall species composition (Tonn and Magnuson 1982;

Tonn et al. 1990); Jackson et al. 2001). For example, Robinson and Tonn (1989) found that predation by northern pike to be the dominant process structuring species assemblages in small lakes in central Alberta. Pike show wide tolerances for abiotic factors and environmental conditions (Spens and Ball 2008) and are keystone piscivores that are important in "top-down" structuring of fish communities (Casselman and Lewis 1996). Analysis of distribution patterns of coexistence of pike and salmonids by Spens and Ball (2008) indicated that salmonid introductions often fail to become self-sustaining in lakes if pike are present.

As a result, it is expected that the fish species assemblage within Kennady Lake will be similar to pre-Project conditions, but that due to biotic and abiotic factors, the community structure may differ.

8.10.4.4.2 Effects of Changes to Aquatic Health

Potential effects to aquatic health in Kennady Lake and Area 8 were evaluated for closure and post-closure in the aquatic health assessment (Section 8.9) based on predicted changes in water quality and sediment quality.

For the direct waterborne exposure assessment, total dissolved solids (TDS) was identified as a substance of potential concern (SOPC); however, adverse effects to fish and aquatic invertebrates are not expected at the predicted TDS concentrations in Kennady Lake and Area 8 (Section 8.9.3.2.1). At closure, predicted maximum concentrations of SOPCs in Kennady Lake and Area 8 are below chronic effects benchmarks (CEBs), with the exception of total iron, copper, and strontium. The predicted iron concentrations are not expected to result in adverse effects to aquatic life, and the potential for copper and strontium to cause adverse effects to aquatic life in Kennady Lake and Area 8 was considered to be low (Section 8.9.3.2.1).

For the indirect exposure pathway, predicted fish tissue concentrations in Kennady Lake were projected to be above toxicological benchmarks for only one substance of interest (SOI): silver. However, the potential for the predicted silver concentration to cause effects to fish was considered to be low (Section 8.9.3.2.2).

Based on the aquatic health assessment (Section 8.9), predicted changes to concentrations of all substances considered were projected to result in negligible effects to fish tissue quality and, by association, aquatic health in Kennady Lake. As a result, no effects to fish populations or communities would occur from changes in aquatic health.

8.10.4.4.3 Long-Term Effects

Recovery of Fish Community

The recovery of the fish community in Kennady Lake in post-closure is described in Section 8.11. Physical conditions in the lake at closure include habitat losses due to excavation of the mine pits and habitat enhancement structures built to replace lost habitat in Kennady Lake. The assessment of effects to aquatic health concluded that modelled changes in chemical constituents of water quality will have a negligible effect on the health of aquatic life in the refilled Kennady Lake. The nutrient enhancement will cause a shift in the lake trophic status, and the resulting change in habitat conditions may cause the re-established fish community structure to differ from that prior to the Project.

8.11 RECOVERY OF KENNADY LAKE AND ITS WATERSHED

With the physical environment, hydrology and water quality of Kennady Lake returning to stable conditions after Project closure, it is expected that an aquatic ecosystem will develop within Kennady Lake. The uncertainty lies in how long the recovery will take and how similar the aquatic ecosystem will be to baseline conditions.

Under baseline conditions, Kennady Lake ecosystem consists of various aquatic biota, including aquatic plants, phytoplankton, zooplankton, benthic invertebrates, and fish. A three-step process was adopted to evaluate and assess how each of these components of the aquatic ecosystem may develop in Kennady Lake after refilling. The first step involved the completion of a literature review. The literature review was undertaken to develop a summary of the published information relevant to the recovery of lakes after flooding or refilling and to identify, to the extent possible, the main drivers that control the rate and direction of recovery. The specific objectives of the literature review were as follows:

- to summarize the key findings that other researchers have observed on other systems;
- to identify the main drivers that were responsible for the observed changes in each system (to the extent possible); and
- to highlight the management options that have been applied to aid in lake recovery (if presented and/or identified in the reviewed literature).

The second step in the assessment process involved evaluating how the results of the literature review applied to Kennady Lake, given its location and physical structure. The final step in the process consisted of taking the information obtained from the literature review and the evaluation of its suitability to Kennady Lake and using it to project how the aquatic ecosystem in Kennady Lake will likely recover.

A more detailed discussion of the methods used to complete each step of the assessment process is outlined below in Section 8.11.1. The results of the assessment are presented in Section 8.11.2.

8.11.1 Effect of Project Activities on the Long-term Recovery of Kennady Lake

8.11.1.1 Background

As noted in the Key Line of Inquiry: Water Quality and Fish in Kennady Lake (Section 8), some of the aquatic habitat in Kennady Lake disrupted or disturbed by Project activities will be replaced, and the long-term hydrology of Kennady Lake is expected to return to a stable state similar to current conditions. Water quality in the lake is similarly expected to return to existing conditions over time with the potential exception of nutrients and some components of total dissolved solids (TDS). This also takes into account the negligible effects predicted to aquatic health related to potential changes in the chemical constituents of water quality in the refilled Kennady Lake after mine closure. With the physical and chemical environment of Kennady Lake returning to stable conditions, it is reasonable to conclude that an aquatic ecosystem will develop within Kennady Lake. There is uncertainty in how long the recovery may take and what the final aquatic ecosystem will consist of particularly when colonization and trophic change are considered.

Similar to most lakes, Kennady Lake currently contains phytoplankton, zooplankton, benthic invertebrates, aquatic plants, and fish. As outlined in Section 8.11, a three step process was adopted to evaluate and assess how each of these components of the aquatic ecosystem may develop in Kennady Lake after refilling.

A more detailed discussion of the methods used to complete each step of the assessment process for the long-term recovery of Kennady Lake is outlined in Section 10.5.2, and the results of the assessment are presented in Section 10.5.3; the information presented in these two sections is virtually identical to that which appears in Sections 8.11.1 and 8.11.2 of the Key Line of Inquiry: Water Quality and Fish in Kennady Lake. The information is being presented in both locations to ensure that the Terms of Reference (Gahcho Kué Panel 2007) requirement that each key line of inquiry must be a comprehensive stand-alone analysis with only minimal cross-referencing with other parts of the environmental impact statement (EIS) is met.

8.11.1.2 Effects Analysis Methods

8.11.1.2.1 Literature Review

Search Methods

The literature search was conducted with a focus on the following topics:

- impacts of damming on upstream environments;
- flooding of new land and the development of aquatic ecosystems in previously terrestrial habitats (e.g., development of off-stream reservoirs);
- recovery of previously-drained systems; and
- management of lake recovery.

The databases searched included Agricola, Arctic & Antarctic Regions, BIOSIS Previews, Environment Complete, Environmental Abstracts, Genie Catalog, Google Scholar, Scopus, SpringerLink, Web of Science, and Wildlife & Ecology Studies.

The search terms included primary keywords, such as lake recovery, increasing lake volume, impoundments, environments upstream of dams, downstream environments of removed dams, ecosystem establishment, dam formation (focusing on flooding of terrestrial environments), and lake formation. Secondary keywords included cold climates, arctic, subarctic, tundra, and oligotrophic systems. Additional terms were added during the search process. They included “recovery and disturbance and aquatic systems”, “impoundments not dams”, recovery and lentic systems, reservoir aging, and turbidity.

Document tracking was completed using a spreadsheet that outlined the databases searched, the date the searches were conducted, the keywords used, the number of hits and the number of hits sourced for further short-listing.

In addition, Niemi et al. (1990) completed a review of articles on the recovery of aquatic systems after disturbance. The authors focused their efforts on retrieving articles published from 1970 to 1986. As part of the present literature review, the search completed by Niemi et al. (1990) was repeated using similar databases and search terms with a focus on articles published since 1986. A cited references search was also completed using Niemi et al. (1990) as the focus article, and references in all reviewed articles were examined for additional sources relevant to the topic of the current literature review.

Review Procedure

The citations for all retrieved articles were entered into a spreadsheet, and the titles were reviewed for applicability to the topic of lake recovery. A scoring system was applied to short-list the articles to be reviewed. Each citation was scored as 1, 2, or other. A score of 1 meant that the article appeared to be directly applicable to the topic of lake recovery in northern climates, whereas a score of 2 meant that the article contained some relevant material.

Articles that were rated as "other" contained information related to lakes or reservoirs in southern or tropical locations, discussed lake recovery following a spill or other short-term input, or were focused on the recovery of flowing systems (rather than lakes or reservoirs). Articles that provided general background information (e.g., limnology studies that did not necessarily discuss lake recovery) were also generally considered to be non-relevant to the main topic of the literature review. Articles with scores of 1 or 2 were further short-listed by scanning the abstract or, in some cases, the body of the text of short articles. Priority for article review was given to the articles rated as 1, followed by those assigned a rating of 2. The key findings of the articles that were considered relevant to Kennady Lake were brought forward and are discussed in Section 10.5.3

8.11.1.2.2 Assessing Applicability to Kennady Lake

Following the completion of the literature review, key findings were evaluated with reference to their applicability to Kennady Lake. The evaluation involved looking at how the systems described in the reviewed literature compared in size and location to Kennady Lake, as well as assessing whether the drivers identified in the reviewed articles would have equal application to an arctic lake. Of particular focus was the potential role of flooded terrestrial vegetation. Flooded terrestrial vegetation was identified in a number of studies as a key driver that influences initial nutrient dynamics and primary productivity in flooded or refilled systems. An evaluation was, therefore, completed to examine the potential extent of vegetative in-growth into the drawn-down sections of Kennady Lake during the operational life of the Project.

8.11.1.2.3 Forecasting Recovery Rates and the Nature of the Final System

The information obtained from the literature review, balanced by its applicability to Kennady Lake, was used to evaluate how the aquatic ecosystem in Kennady Lake would recover. The evaluation was completed using professional judgement, with due consideration given to the ecological concepts of colonization, natural succession, and trophic interactions, particularly as they apply to small arctic lakes.

Re-establishment of self-sustaining fish populations is the ultimate end-point of the Kennady Lake ecosystem recovery. To assess the re-establishment of fish, it was first necessary to predict the composition, abundance, and distribution of plankton and benthic invertebrate communities expected to re-establish in the lake, because these lower trophic communities form the basis of the food web upon which fish in the lake will depend. Once the predicted recovery of the lower trophic levels was complete, attention was focused on predicting the recovery of the fish community, including both forage and sport fish. As part of the analysis, consideration was given to the potential for restocking Kennady Lake with lake trout and/or round whitefish.

8.11.1.3 Effects Analysis Results

8.11.1.3.1 Summary of Key Findings from Literature Review

Each of the following sub-sections contains a summary of the key information obtained from the literature review with reference to a particular part of the aquatic ecosystem. The first three sub-sections are focused on nutrient dynamics, erosion and turbidity, and the potential release of metals from newly flooded areas. Key findings related to the establishment and growth of bacteria and phytoplankton, zooplankton, benthic organisms, and fish are then outlined in the next four sub-sections.

Each sub-section is organized in a similar fashion, beginning with an introductory paragraph that contains an overall summary of the sub-section contents. The remaining portion of each sub-section is then devoted to a more detailed discussion of the key findings, with relevant examples and citations included in the text.

Nutrient Dynamics

The information obtained from the literature review suggests that nutrient dynamics in a refilled reservoir or flooded lake are driven primarily by the flooding of terrestrial vegetation and, to a more limited extent, soil. Although nitrogen, phosphorus, and carbon are released, not all forms may be equally bioavailable. In particular, phosphorus may be released in a non-bioavailable form, which can lead to the preferential growth of bacteria over that of phytoplankton. The type and quantity of terrestrial vegetation that is flooded can affect the amount and duration of the initial nutrient pulse. Another potential source of nutrients is flooded soil. The benefits of removing terrestrial vegetation or soil prior to flooding to reduce the initial nutrient surge are dependent on site-specific conditions, and can, in some cases, result in some unforeseen detrimental effects, as outlined in greater detail below.

Flooding terrestrial vegetation can result in rapid and dramatic changes in water quality, including a surge in nutrient concentrations (Northcote and Atagi 1997). Following the flooding of a sedge meadow in Sweden, large quantities of terrestrial plant material dominated the detritus pool and supported a system dependent on allochthonous[5] organic matter for five to six years (Danell and Sjoberg 1982). Similarly, following the creation of a new reservoir, Thouvenot et al. (2000) noted that flooding and decomposition of existing terrestrial vegetation released nitrogen and carbon into the system.

Paterson et al. (1997) observed that the concentrations of phosphorus, nitrogen, and dissolved organic carbon in a newly impounded lake increased as a result of vegetative decay. Concentrations of most nutrients were higher in the shallower, flooded peat areas, relative to those measured in the open water areas of the lake — a pattern attributed to the dilution provided by upstream water input from an oligotrophic[6] system.

Although nitrogen, carbon, and phosphorus are released from decaying vegetation, their relative bioavailability can differ. The released forms of carbon and nitrogen are typically bioavailable, whereas the released phosphorus can be in a non-bioavailable form bound to organic particles (Paterson et al. 1997; Thouvenot et al. 2000). This lack of phosphorus or bioavailable phosphorus may initially cause an increase in bacterial biomass over phytoplankton, because phytoplankton require immediately bioavailable phosphorus. In contrast, bacteria can obtain phosphorus from more resilient materials during the decay process.

Geraldes and Boavida (1999) measured higher concentrations of different forms of phosphorus in a newly created reservoir, compared to an old reservoir that had been drained and refilled. The observed variation in phosphorus concentrations between the two systems was attributed to the abundance of terrestrial vegetation in the newer reservoir that was undergoing decay. However, phosphorus concentrations declined after the initial spike post-impoundment, potentially due to an increase in sedimentation rates, the uptake of nutrients by phytoplankton, and/or a reduction in the amount of flooded terrestrial vegetation.

5 Allochthonous organic matter refers to organic matter that did not originate in the place it was found. In comparison, autochthonous organic matter is derived from sources found within the system, such as plankton debris.

6 Oligotrophic systems are characterized as being poor in nutrients and plant life, and rich in oxygen.

The type and quantity of vegetation entering a system can determine the magnitude and duration of the nutrient surge. The decay of moss-peat in limnocorral[7] experiments initially increased then lowered primary productivity and biomass, because it initially released nutrients (soluble phosphorus and nitrogen) and then lowered some other factor (such as iron or some other essential metal by binding to the increased humic matter) (Guildford et al. 1987). After one year, the moss-peat material no longer released humic material, phosphorus, or nitrogen in sufficient quantities to influence concentrations in the water column (i.e., concentrations in the experimental cells were similar to those in the control cells).

In general, leaves, needles, and other soft parts of trees decay faster than shrubs, brush, and other non-woody vegetation, all of which decay faster than the woody components of large trees (Northcote and Atagi 1997). For example, conifer needle litter can decompose completely within a year, initially by leaching and microbial activity, and then by macroinvertebrate feeding (especially by midge [Chironomidae] larvae) (Crawford and Rosenberg 1984), whereas trees take longer.

Flooded soil can be a source of nutrients to a newly created lake or reservoir, although its input may be minor in comparison to that of flooded vegetation. In experiments with eroded clays, the release of phosphorus from the introduced clays was small, and the increase in phosphorus availability as mainly due to decreased primary productivity, rather than an increase in available phosphorus (Guildford et al. 1987). In contrast, Northcote and Atagi (1997) noted that topsoil removal greatly lowered carbon, nitrogen, and phosphorus levels and reduced phytoplankton growth by over half in a newly flooded system, suggesting that the flooded soil was a major contributor to nutrient release. Hecky et al. (1984) similarly noted an increase in phytoplankton and zoobenthos biomass as a result of the release of nutrients from flooded soils.

McGowan et al*.* (2005) investigated the effects of lowering the water level during the winter on a shallow lake (Wascana Lake) in southern Saskatchewan. They hypothesized that reductions in water level and exposure of sediments to freezing temperatures would increase nutrient release rates. In particular, the rate at which phosphorus was released from the sediment was expected to increase upon refilling, because, based on the work of James et al*.* (2001), oxidation and mineralization of organic phosphates would occur while the sediments were exposed to atmospheric conditions. Sediment desiccation was also expected to increase aerobic nitrification, leading to a build-up of nitrate in

[7] Large experimental cylinders enclosing a column of water in a lake.

the exposed sediments, as previously noted by Kadlec (1962). However, it was acknowledged that nitrate accumulation rates would be moderated by the increased rates of denitrification that could occur in the underlying anoxic sediments, as per De Groot and Van Wijck (1993).

Water levels in Wascana Lake were reduced by about 50 percent (%) in October, which resulted in most shallow upstream reaches of the lake being completely dry. The following spring, natural runoff restored lake levels. During the subsequent growing season, the authors did not find any evidence of increases in phosphorus release from sediments (McGowan et al. 2005). They speculated that phosphorus was not released, because phosphorus concentrations were naturally high in Wascana Lake and this would have limited diffusion from the sediment. Alternatively, the cold winter temperatures may have limited the rate at which the oxidation and mineralization of organic phosphate occurred.

Changes in the concentrations of nitrogen compounds in the water column were also limited and less than expected. Ammonia levels in Wascana Lake were elevated following lake refilling, but only minor variations were observed in nitrate levels. Overall, McGowan et al. (2005) attributed the lack of a more appreciable response to the winter drawdown to the resilience of the system and the fact that the lower winter water level was within the range of hydrological fluctuations Wascana Lake already experiences.

The benefits of removing terrestrial vegetation prior to flooding to limit the initial nutrient surge are not clear. For example, in Maltañski Reservoir in mid-western Poland, terrestrial vegetation and topsoil were removed prior to refilling the reservoir that had been dry for 10 years. However, post-flooding phosphorus concentrations in the water were higher than expected, suggesting that the preparation step was not effective (Goldyn et al. 2003). The higher than expected phosphorus levels were attributed to external loading of nutrients from incoming water and frequent emptying of the reservoir in the years after the initial refill period[8]. In contrast, Campbell et al. (1975) (as cited in Northcote and Atagi 1997) found phytoplankton densities were 5 to 100 times higher in flooded systems where soil had not been stripped, in comparison to those that had.

Stripping vegetation and topsoil may yield side effects that negate the potential gains to limiting nutrient surge. Soon-to-be-flooded areas around Southern Indian Lake were cleared of timber prior to impoundment. However, the cleared zone was entirely eroded within the first year of impoundment, which resulted in increased suspended sediments in the lake (Hecky et al. 1984). The authors

[8] The Polish reservoir was also completely emptied between years.

also noted that the only effective clearing done on Southern Indian Lake was on bedrock shorelines and on protected shorelines without permafrost. In their review, Northcote and Atagi (1997) stated that, from a range of preparation procedures from nothing to complete clearing, the most appropriate procedure(s) for a given site are dependent on the final use of the new reservoir. For example, the purpose of the preparation could range from improving water quality to improving angler and boating access.

There are other benefits to maintaining terrestrial vegetation prior to flooding. Keeping woody debris and other forms of terrestrial vegetation on the flooded lake bed may aid in macrophyte growth; Northcote and Atagi (1997) cite a few studies that found macrophyte colonization was enhanced by inundated trees and brush, because of protection from wave action and erosion of soils. The authors also summarized studies on the Campbell River impoundments on Vancouver Island where high benthic diversity and abundance were associated with flooded trees, brush, and shrubs, compared to lower rates of diversity and abundance in systems where the terrestrial vegetation was nearly completely cleared prior to flooding.

Another possible management option available to limit the initial nutrient surge in newly flooded systems includes the repeated draining and refilling of the lake or reservoir in question. This action will remove the nutrients released from the decaying vegetation and allow for a more rapid establishment of conditions that are in equilibrium with upstream water sources. For example, successive reservoir emptying sped up the change from allochthonous to autochthonous organic matter, which likely caused the changes in unicellular plankton of a newly flooded reservoir (Thouvenot et al. 2000). Nursall (1952) also noted that a flooded oligotrophic lake in the Rocky Mountains was maintained as fundamentally oligotrophic partly because of the rapid replacement of water. Periodic fluctuation of water level (common to hydroelectric reservoirs) and deposition of sediment also contributed to this condition.

Erosion and Turbidity

Site-specific factors (e.g., permafrost, fine-grained sediments) affect shoreline erosion rates in newly created lakes and reservoirs, with the eroded sediment/soil potentially leading to changes in water-column turbidity. Although turbidity can have a profound effect on algae and vegetative growth within a lake or reservoir, the majority of the studies reviewed did not discuss turbidity as being notably higher after impoundment, or having an undue influence on biological development within the newly created systems. There were, however, some notable exceptions (e.g., Southern Indian Lake).

Marzolf (1990) stated that the dominant feature of reservoirs created on the Great Plains of North America is turbidity. However, these reservoirs are typically created by damming a river such that upstream riverbanks are flooded and scoured. Suspended sediment is often kept in the water column by the inflow current and the shallowness of the reservoir relative to its fetch.

Similarly, the distribution of erodible, fine-grained shorelines was the critical difference between the resultant turbidity levels that were measured in a northern Saskatchewan lake (Reindeer Lake) after impoundment and Southern Indian Lake, which is located in northern Manitoba (Hecky et al. 1984). Reindeer Lake is similar to Southern Indian Lake in latitude, magnitude of water level increase, surface area, operating regime, climate, bedrock geology, and pre-impoundment fishery; however, Reindeer Lake did not have turbidity issues, because fine-grained deposits were sparse and erosion was minimal (Hecky et al. 1984).

Southern Indian Lake is located on the Precambrian Shield in a zone of discontinuous permafrost. It is surrounded by boreal forest and experiences high wave action (Newbury and McCullough 1983). A dam constructed at the natural outlet to the lake caused flooding beyond the sub-lake thawed zone into the permafrost-affected upland. Extensive shoreline erosion subsequently altered the sedimentation regime and water quality of Southern Indian Lake (Hecky et al 1984). Pre-impoundment sediment input was about 200,000 tonnes/year compared to a post-impoundment rate of greater than 4,000,000 tonnes/year (Newbury and McCullough 1983).

Shoreline erosion along the edges of Southern Indian Lake continued until bedrock was exposed. The time required for shoreline stabilization was estimated from the frequency of bedrock encounters and the pre-impoundment estimate of how much of the shoreline was bedrock-controlled. From these calculations, it was estimated that it would take 35 years to restore 90% of the shoreline (Newbury and McCullough 1983).

As a consequence of the shoreline erosion, Southern Indian Lake was, on average, a darker, less transparent lake than it was prior to its expansion through impoundment (Hecky et al. 1984). However, primary productivity increased in well-illuminated regions of Southern Indian Lake, due to increased nutrient concentrations resulting from the decay of terrestrial vegetation[9]. Primary productivity did not increase in other areas of the lake, because of low light intensity and increased turbidity. Light extinction in Southern Indian Lake was a

[9] Phosphate concentrations increased in the water column of Southern Indian Lake after impoundment (C. Anema, unpublished data, cited by Hecky et al. 1984).

linear function of suspended sediments (Hecky 1984), and algal production was limited when mean water column light intensity fell below 5 micro-Einsteins per square metre per minute ($mE/m^2/min$) (Hecky and Guildford 1984). Limnocorral experiments with suspended clays showed that light reduction due to suspended clay and silt depressed primary productivity and phytoplankton biomass (Guildford et al. 1987).

Decreased zooplankton biomass and changes in species composition in major regions of the lake were attributed to lower water temperatures and reduced predation due to poorer transparency (Patalas and Salki 1984). Zoobenthos alternately benefited from, and were depressed by, increased suspended sediments; in some areas, the high concentration of suspended sediments negated the benefit of high input of organic substrate as a source of nutrients and habitat (Hecky et al. 1984).

Fish were negatively affected by the impoundment of Southern Indian Lake. The increased turbidity reduced the ability of fish to locate food, and it may have resulted in reduced embryo survival due to increased rates of sedimentation (Hecky et al. 1984). In addition, the introduced sediments contained mercury, and increased mercury concentrations were observed in fish tissue (Bodaly et al. 1984).

Metals Release

With the exception of mercury, metal release from sediment as a consequence of flooding was not observed or commented upon in the majority of the reviewed literature. Low oxygen conditions in flooded sediment, however, can result in the release of dissolved manganese and iron, as observed in a refilled reservoir in Germany (Nienhuser and Braches 1998).

In contrast to other substances, mercury appears to be a common concern following flooding or impoundment. Increased methyl mercury contamination in fish has been noted in a number of studies, and factors affecting methyl mercury production and its uptake by biological organisms have been identified. This phenomenon is likely due to the inundation and subsequent decomposition of organic material that promotes the microbial methylation of inorganic mercury to organic methyl mercury. It has been well established that (1) mercury in pristine and flooded soils is predominately bound to organic matter, and (2) that mercury methylation is related to organic carbon content of the flooded soil/sediment. Methyl mercury is the most toxic form of mercury and readily accumulates in aquatic organisms.

Three processes can be targeted to mitigate the effect of methyl mercury generation, bioaccumulation through the food chain, and associated environmental and health risks: mercury methylation, mercury bioaccumulation, and fish consumption. Available mitigation options include partial or complete stripping or capping of organic materials and soils, high temperature burning of vegetation and leaf litter, liming, selenium additions to newly created reservoirs and lakes, intensive fishing, fish barriers (screens), restricted access, and/or fish consumption advisories. Of the options available, selenium additions and liming are not widely recommended, and consumption advisors serve only to protect human health without directly addressing mercury concentrations in fish.

Key findings from these studies, as well as that completed by Nienhuser and Braches (1998), are discussed in more detail below.

Iron and Manganese

Nienhuser and Braches (1998) published an account of their difficulties in refilling a drinking water reservoir in Germany. Just after the beginning of refilling, the reservoir became ice-covered for two months. Dissolved oxygen levels in the lower section of the reservoir declined, due to the mineralization of organic matter in the flooded soils/sediments. While the bulk of the terrestrial vegetation was removed prior to refilling, the soil was not, and terrestrial organic matter remaining in flooded soil/sediment decayed. The change in oxygen conditions over the flooded soils/sediment resulted in the development of anoxic conditions within the sediments, which led to the release of dissolved manganese and iron.

Artificial mixing was introduced into the reservoir in an attempt to remedy the situation and prevent further increases in the concentrations of iron and manganese. However, the opposite occurred. Manganese and iron concentrations increased, because the artificial mixing system resulted in the suspension of the bottom sediments/soils in the water column.

Nienhuser and Braches (1998) noted that the rate of oxidation of reduced divalent manganese is slower than that of reduced divalent iron, perhaps because manganese needs a higher redox potential to be oxidized. Therefore, aeration acts faster or more effectively to remove dissolved iron than it does to remove or control dissolved manganese. Manganese oxidation is also influenced by microbial processes and takes between 1 and 100 days at high oxygen levels. Based on their experience, Nienhuser and Braches (1998) concluded that the refilling phase was a critical step in regards to the prevention of future problems with water quality, and they recommend that sediment and all fish are removed prior to refilling. They also suggested that artificial mixing systems be used.

Mercury

Background

Mercury (Hg) concentrations in fish have increased after impoundment in almost all reservoirs in North America and northern Europe. In Canada, various authors (e.g., Verdon et al. 1991; Bodaly and Fudge 1999; St. Louis et al. 2004) reported that reservoir creation often results in fish mercury concentrations that exceed the Canadian human consumption guideline of 0.5 micrograms of mercury per gram (μg Hg/g). In addition, reservoir creation may cause mercury problems in fish in downstream waterbodies (Johnston et al. 1991).

Methyl mercury is the most toxic form of mercury, and it accumulates readily in aquatic organisms (Ullrich et al. 2001). Fish mercury concentrations are thought to increase after flooding in reservoirs because the decomposition of soils and vegetation promotes the microbial methylation of inorganic mercury to organic methyl mercury (Bodaly et al. 1984; Jackson 1988; Hecky et al. 1991; Porvari and Verta 1995). The main pulse of methyl mercury production appears to occur in the first few years following impoundment (St. Louis et al. 2004; Heyes et al. 1998), but even this relatively short period of methyl mercury production has the potential to raise fish mercury concentrations for 20 to 30 years (Hall et al. 2005).

Although all inundated soils and vegetation are sources of organic carbon and potential sites of methyl mercury production, inundated conifer trees, needles, and boughs may be especially important methylation sites (Hecky et al. 1991; Heyes et al. 1998; Hall et al. 2004). Total methyl mercury production is also related to total organic carbon stored in the reservoir. In a study at the Experimental Lakes Area in northern Ontario, the reservoir with the most organic carbon (a flooded wetland) produced the most methyl mercury (Hall et al. 2005). Wetlands store a lot of organic carbon in the form of peat, and in the flooded wetland in the Experimental Lakes Area, 97% of methyl mercury production in the first two years occurred in flooded peat (St. Louis et al. 2004). Peat may cause additional problems in reservoirs, because it can acidify the water (St. Louis et al. 2003), which increases bacterial uptake of mercury for methylation (Kelly et al. 2003). Peat can also float to the surface and create floating peat mats (St. Louis et al. 2004), which can affect turbidity and light penetration. In a reservoir in northern Quebec, the highest concentration of dissolved methyl mercury in water was found under a floating peat mat (Montgomery et al. 2000).

Once methylated in the bottom sediments of a reservoir or lake, methyl mercury may be transferred to biota in a number of ways. There may be passive diffusion of methyl mercury into the water column (Morrison and Thérien 1991) and subsequent uptake into phytoplankton (Plourde et al. 1997). More commonly, flooded soil particles are suspended into the water column by wind or ice-driven erosion. Once in the water column, the soil particles are ingested by zooplankton and benthic invertebrates (Louchouarn et al. 1993; Grondin et al. 1995; Mucci et al. 1995). Plourde et al. (1997) have suggested that passive diffusion of methyl mercury is important in new reservoirs, whereas suspension of flooded soil particles is the most important transfer process in the long term. Ingestion of mercury-contaminated particles by burrowing benthic invertebrates may also be an important route for methyl mercury to enter the food chain. Once in zooplankton and benthic invertebrates, methyl mercury biomagnifies up the food chain to higher concentrations in fish.

Because suspension of sediment is a primary vector by which methyl mercury becomes available to biota, the amount of erosion in a reservoir can significantly impact methyl mercury concentrations in biota. St. Louis et al. (2004) suggested that, if there is little erosion, methyl mercury produced in peat will likely be demethylated. If there is erosion, however, methyl mercury will be rapidly transferred up the food chain. Littoral areas of reservoirs (i.e., the shallow zone where light penetrates to the bottom) are often particularly vulnerable to erosion and, therefore, maybe important sites of methyl mercury transfer. Littoral areas are also usually warmer than pelagic areas (i.e., the deep zones of the lake where light does not penetrate to the bottom), and the rate of mercury methylation increases with increasing temperature (Bodaly et al. 1993).

Once mercury levels are elevated as a result of impoundment, the time required for fish mercury concentrations to return to background levels is dependent on species, fish size, and fish diet. Planktivorous and omnivorous species return to background concentrations sooner than piscivorous species, and smaller fish return to background concentrations sooner than larger fish (Brouard et al. 1990; Verdon et al. 1991; Andersson et al. 1995). Most researchers have estimated that, for piscivorous species, the time to return to background mercury concentrations is 15 to 30 years (Verdon et al. 1991; Andersson et al. 1995; Porvari 1998), whereas those for non-piscivorous species range from 2 to 20 years (Verdon et al. 1991). Northern pike often show the longest recovery time (Jackson 1991; Andersson et al. 1995). The recovery period may be influenced by fish harvesting and reservoir discharges. Higher rates of fish harvesting and high flushing rates reduce the time to return to background concentrations in fish (Verdon et al. 1991).

Management Options

It has been well established that (1) mercury in pristine and flooded soils is predominately bound to organic matter (e.g., Louchouarn et al. 1993; Dmytriw et al. 1995; Tremblay and Lucotte 1997), and (2) mercury methylation is related to organic carbon content of the flooded soil/sediment. To mitigate the effect of methyl mercury generation, bioaccumulation through the food chain and associated environmental and human health risks, there are three processes that can be targeted: mercury methylation, mercury bioaccumulation, and fish consumption. The following identified mitigation options target one or more of these processes (as noted in parentheses):

- partial or complete stripping of organic material and organic soils within the footprint where a new lake or reservoir is to be established, with priority given to those materials/soils that have the greatest potential to contribute to methyl mercury production (e.g., peat and leaf litter) (methylation);
- partial or complete capping of organic material and organic soils within the lake/reservoir footprint, with priority given to those materials/soils that have the greatest potential to contribute to methyl mercury production (methylation);
- high temperature burn of vegetation and leaf litter within the lake/ reservoir footprint (methylation);
- adding lime to the newly created lake/reservoir (methylation);
- adding selenium to the newly created lake/reservoir (bioaccumulation);
- intensive fishing (bioaccumulation and fish consumption);
- fish screens (bioaccumulation and fish consumption); and/or
- restricted access/fish consumption advisories (fish consumption).

Each of these mitigation options are discussed in more detail below.

Stripping or Burning

The potential for mercury methylation can be reduced by burning or stripping vegetation and organic soils prior to flooding (Morrison and Thérien 1991). Complete stripping may also reduce methyl mercury transport into the water column. For both burning and complete stripping, it is important that all vegetation, including mosses, lichens, shrubs, leaf litter, and deadfall be burned/stripped, because all of these materials are capable of enhancing mercury methylation (Morrison and Thérien 1991). If partial stripping is used as a mitigation strategy, areas with high organic matter content (e.g., peat) should be targeted, because these materials have a higher potential for methyl mercury production.

A recent investigation of burning before flooding found that burning vegetation and soils (high-temperature burn) reduced post-flood mercury concentrations in water, but not in zooplankton and benthic invertebrates (Mailman and Bodaly

2004). It appeared that zooplankton and benthic invertebrate mercury concentrations were lower in the control treatment than the burned treatment, because high dissolved organic carbon concentrations inhibited mercury bioaccumulation in the control. The effects of pre-flood burning on mercury concentrations in fish tissues are not yet known.

High temperature burning would likely only be effective in areas having leaf litter overlying mineral soils. If burning is employed as a mitigation strategy, high-intensity burning (leaf litter and vegetation) would be more effective at reducing mercury concentrations in water than low-intensity burning (vegetation only) (Mailman and Bodaly 2004).

Capping

Capping of sediment is usually restricted to small areas that have received extensive anthropogenic inputs of heavy metals or organic pollutants (e.g., Zeman 1994; Wiener and Shields 2000). Capping of soils before flooding would isolate organic material and inorganic mercury, and it would relocate the most biologically active part of the sediment to the clean cap materials (Zeman 1994). This should result in reduced mercury methylation, and, if it is a complete cap, reduced methyl mercury transport into the water column.

Both sandy and fine-grained materials have been used for capping contaminated sediment; sandy caps are more resistant to erosion, while fine-grained caps are better at isolating underlying contaminants (Zeman 1994). Problems with capping include erosion, transfer of contaminants through the cap during consolidation, bioturbation[10], diffusion and the impacts on benthic invertebrate colonization (Zeman 1994). These factors are often investigated under laboratory conditions before and during the capping operation, and field monitoring is part of a comprehensive capping program (e.g., Zeman 1994; Wiener and Shields 2000). Similar to partial stripping, partial capping should target areas with high organic matter.

Liming

Mercury concentrations in fish tend to increase with decreasing pH (McMurtry et al. 1989; Grieb et al. 1990; Greenfield et al. 2001). In other words, the more acidic the lake water, the higher the mercury concentrations tend to be in the fish. Flooding of peatlands and soils rich in organic acids may promote more acidic conditions in a given lake or reservoir. Liming involves the addition of high-pH

[10] Bioturbation is the process by which aquatic organisms modify the physical and chemical properties of the substrate in which they live. Bioturbation includes mixing of sediment, and solute flux and suspension of sediment into the water column.

limestone mixtures to the lake and/or surrounding catchment. This mitigation strategy has been successful used to reduce fish mercury concentrations in acidic waters in Sweden (Hakanson et al. 1988; Lindqvist et al. 1991; Andersson et al. 1995). Post-liming pH increases should reduce uptake of mercury by methylating bacteria, reduce direct uptake of mercury across fish gills, increase mercury volatilization, and increase demethylation rates (Ponce and Bloom 1991; Ullrich et al. 2001; Kelly et al. 2003). All of these mechanisms should work to decrease mercury concentrations in fish.

As a mitigation strategy, liming should only be considered if fish mercury problems are being exacerbated by low pH, because not all lakes with fish mercury problems are acidic. Liming requires repeated treatments.

Selenium Additions

Selenium additions can mitigate elevated levels of mercury in fish by disrupting mercury bioaccumulation and methylation. Selenium occurs in both natural and industrial settings, and, at low concentrations, it is an essential nutrient for many organisms. Research has shown that there is a negative relationship between mercury and selenium concentrations in fish tissue and that high selenium concentrations in lakes (either intentional or accidental) can result in low fish mercury concentrations (Turner and Swick 1983; Paulsson and Lundberg 1991; Chen and Belzile 2001). The mechanism through which selenium interferes with mercury bioaccumulation is not fully understood. Selenium may affect mercury biomagnification from food sources (Turner and Swick 1983), the availability of inorganic mercury for methylation (Bjornberg et al. 1988), or the activity of methylating bacteria (Oremland and Capone 1988).

Selenium has been successfully added to lakes in Sweden to reduce fish mercury concentrations (Paulsson and Lundberg 1991). Lakes in the Sudbury area with high selenium levels contain fish with low levels of mercury (Chen and Belzile 2001). However, selenium additions are not often the best mitigation strategy, because high selenium concentrations can be toxic to fish and cause a variety of diseases and deformities (Lemly 1993; Chen and Belzile 2001). Most researchers agree that further studies are required before selenium additions are considered a widely applicable mitigation strategy.

Intensive Fishing

Intensive fishing targets mercury bioaccumulation processes by removing fish with potentially high mercury burdens from the system and reducing mercury bioaccumulation in remaining fish. The removal of the mercury-laden fish should decrease the risk to human health, because sportfish catch-per-unit-effort and mercury concentrations will decrease. Wildlife and raptor mercury concentrations may also decrease, but there is also potential for these organisms to be negatively affected by a reduced food supply. Intensive fishing has proven effective at reducing mercury concentrations in predator fish in Scandinavian and Canadian lakes, and it has been evaluated as a mitigation tool by the Collaborative Mercury Research Network (COMERN) in partnership with Hydro Québec (Gothberg 1983; Verta 1990; Surette et al. 2003; Tremblay et al. 2004).

There are three possible mechanisms through which intensive fishing reduces predator fish mercury concentrations. They consist of the following:

- increased growth rates of fish due to decreased competition (fish that grow faster have lower mercury);
- changes in feeding to lower-mercury prey; and
- a reduction in the total amount of mercury cycling in the ecosystem (Verta 1990; Surette et al. 2003; Tremblay et al. 2004).

Results indicate that increased growth rates and changes in diet account for most fishing-induced mercury decreases (Verta 1990; Surette et al. 2003). To be effective, a large proportion of the predator fish biomass (about 30 to 50%) must be removed from the lake (Verta 1990; Lindqvist et al. 1991). Also, more than one intensive fishing event may be required. Verta (1990) reported that fish growth rates had declined to pre-fishing levels within five years, and Lindqvist et al. (1991) predicted that fishing-induced decreases in fish mercury levels would last for six to eight years.

Fish Screens, Restricted Human Access, and Fish Consumption Advisories

Increased concentrations of mercury in fish can be avoided by excluding fish from the lake system with fish screens. Fish screens would effectively reduce the risk of both human and wildlife mercury consumption, but they would negate the purpose of creating fish habitat and reduce wildlife and waterfowl presence. To achieve an appropriate balance, fish screens can be used on a short-term basis or as a back-up option, if monitoring results indicate that another mitigation option has not been fully successful.

Fish screens could be employed until mercury concentrations in water and invertebrates decreased to baseline concentrations (about three to five years after impoundment), after which fish could be re-introduced into the system. If installed in response to post-flooding monitoring results, fish screens may be especially useful in decreasing mercury concentrations in fish if combined with an intensive fishing program.

Restricted human access and fish consumption advisories are intended to reduce human consumption of mercury-contaminated fish. Fish consumption advisories would address only human health risks. They would be developed by determining sportfish mercury concentrations after impoundment and relating them to the Canadian human consumption guideline. The efficacy of consumption advisories, however, is not always high. A study in Maine showed that only 25% of anglers who were aware of fish consumption advisories changed their fishing behaviour (MacDonald and Boyle 1997). Also, it has been shown that knowledge and understanding of fish consumption advisories can vary greatly with ethnicity, age, income, education, and whether the person is a resident of the area (Burger 1998; MacDonald and Boyle 1997).

A fish advisory approach could be effective for addressing human health risks if access to the lake can be effectively controlled. Restricted access does not mitigate the risk to wildlife health or the time required for mercury concentrations in fish to return to background concentrations.

Bacteria and Phytoplankton

Nutrient dynamics in a refilled reservoir or flooded lake are driven primarily by the flooding of terrestrial vegetation and, to a more limited extent, soil. Although nitrogen, phosphorus, and carbon are released, not all forms may be equally bioavailable. In particular, phosphorus may be released in a non-bioavailable form, which can lead to the preferential growth of bacteria over that of phytoplankton (i.e., algae). However, a predominance of bacteria does not always occur, since sufficient bioavailable phosphorus may be released to support active phytoplankton growth. The type and quantity of terrestrial vegetation that is flooded can affect the magnitude and duration of the initial nutrient pulse; it can also affect the potential for bacteria to initially dominate the lowest trophic level.

As the source of organic matter switches from allochthonous to autochthonous, a shift in dominance typically occurs, with phytoplankton replacing bacteria as the dominant planktonic organism. In other words, when an area is first flooded, the organic matter in the system or entering the system tends to originate from primarily external sources, such as flooded terrestrial vegetation or other materials entering the system through runoff or tributary inflow. The characteristics of these materials tend to favour bacterial growth over phytoplankton. However, over time, the external material decays and is replaced with organic matter that originates primarily from within the reservoir, be it in the form of dead and decaying phytoplankton, zooplankton, fish, or other internal material (e.g., fish feces). The characteristics of the internal materials tend to favour phytoplankton growth over that of bacteria, which leads to a shift in dominance between these two groups. The period over which the shift from bacterial to phytoplankton dominance occurs is dependent on site-specific conditions. However, in some studies, plankton community structure returned to that of oligotrophic environments within three years of flooding. Key findings from studies that document the factors affecting the dominance of bacteria and phytoplankton are discussed in more detail below.

Paterson et al. (1997) observed a brief increase in phytoplankton photosynthesis after the flooding of an experimental lake. Phytoplankton biomass and total phosphorus concentrations were highest nearest the shore over the flooded peat, suggesting that the higher nutrient concentrations in this area of the lake contributed to the increase in phytoplankton biomass. However, the increased rate of phytoplankton growth was short-lived.

In the same system, Paterson et al. (1997) also observed a large sustained increase in bacterial biomass. Bacterial biomass was higher over the flooded peat areas than the open waters, consistent with the patterns observed with phytoplankton. Bacterial size structure and morphology were also affected in that mean individual biovolume increased from 0.1 to 0.3 cubic micrometres (μm^3). Large increases in methane and CO_2 production that occurred after flooding also indicated increased bacterial production (Kelly et al. 1997). The dominance of bacteria in the plankton community for the first year after flooding was also observed in a newly-created, oligo-mesotrophic reservoir in France (Thouvenot et al. 2000).

Both studies (i.e., Paterson et al. 1997 and Thouvenot et al. 2000) found that high bacterial growth coincided with low phytoplankton growth. Paterson et al. (1997) noted that the proportion of bacteria to phytoplankton changed from pre-flood conditions. Before flooding, average bacterial biomass was 27% of average phytoplankton biomass; after flooding, bacterial biomass exceeded that of phytoplankton by nine times. The proportional change resulted both from increasing bacterial biomass and changes in phytoplankton abundance. For example, phytoplankton biomass declined by 70% from the pre-flooding average of 0.45 milligrams per litre (mg/L) to a post-flood average of 0.13 mg/L. The same parameter was 0.40 mg/L in the upstream lake at the same time (Paterson et al. 1997). The observed decline in phytoplankton biomass was not due to dilution by deepening of the water column, because the low levels of algal biomass persisted throughout the first summer of impoundment until drawdown in the fall.

The high phosphorus concentrations, low phytoplankton concentrations, and high bacterial biomass observed by Paterson et al. (1997) suggest that the majority of the phosphorus released after impoundment was not in a bioavailable form that could be easily used by phytoplankton. The authors suggested that the majority of the phosphorus present immediately after impoundment may have been bound to organic complexes that were unavailable to phytoplankton but usable by bacteria. Phytoplankton growth rates following impoundment may also have been affected by limiting amounts of other essential elements. For example, high levels of dissolved organic carbon (DOC) could have resulted in essential metals being bound within organic complexes that rendered them inaccessible to phytoplankton (Guildford et al. 1987). Finally, bacteria may have out-competed phytoplankton for phosphorus. Bacteria predators, such as ciliates and nanoflagellates, were grazed down by *Daphnia rosea*, and the DOC released after flooding was likely highly labile and readily available to the bacteria. These mechanisms may have removed any competitive advantage previously experienced by the phytoplankton (Paterson et al. 1997).

Throughout the four-year study by Paterson et al. (1997), the algal community consisted of small (less than 30 micrometres [µm]) cryptophytes, chrysophytes, and chlorophytes that could be consumed by zooplankton. Whereas species composition changed after flooding, proportions of edible algae did not.

Similar to Paterson et al. (1997), Thouvenot et al. (2000) suggested that the low abundance of autotrophic algae and cyanobacteria observed in the newly created Sep Reservoir in France was the result of restrictions in phosphorus bioavailability, which favoured bacteria growth over that of phytoplankton. No relationship was observed between the concentrations of dissolved organic matter in the water column, which were elevated one year after flooding, and phytoplankton biomass, as estimated from chlorophyll *a* concentrations (Jugnia et al. 2007). The lack of a notable relationship suggests that the dissolved organic matter did not come from autochthonous photosynthetic sources (Jugnia et al. 2007).

In the same system, higher bacterial biomass production after the first year of flooding was correlated to higher concentrations of dissolved free carbohydrate. Bacterial biomass decreased in the second year after flooding, coincidental with an observed decrease in the concentrations of allochthonous dissolved organic matter (Jugnia et al. 2007). Jugnia et al. (2007) also determined that bacterial production was controlled by the type of substrate available, in that it depended on sources of substrate other than phytoplankton exudates and most likely on allochthonous dissolved organic matter. The ratio of bacterial production to primary production (as measured by photosynthesis activity) was also higher than expected, suggesting that bacteria were out-producing phytoplankton (Jugnia et al. 2007).

The shift between allochthonous to autochthonous sources of organic matter and its effect on bacterial versus phytoplankton dominance was also observed in the

Sep Reservoir (Thouvenot et al. 2000). In the first year after flooding, the reservoir's plankton community was dominated by bacteria, gradually shifting to mixotrophic flagellates (algae with whip-like appendages), heterotrophic flagellates, and finally autotrophic phytoplankton, which are species similar to that found in humic lakes where allochthonous organic matter dominates (Jansson et al. 1996). The high initial abundance of bacteria suggested that high allochthonous organic matter stimulated bacterial production (Thouvenot et al. 2000). As the allochthonous organic matter decreased, bacteria became dependent on autochthonous carbon (e.g., carbon produced by phytoplanktonic excretion, cell lysis, and sloppy feeding). As such, they were no longer in direct competition for phosphorus with other planktonic organisms, such as autotrophic algae and cyanobacteria, which accounted for 67% of the total unicellular plankton biomass by the end of the study. While the structure of the plankton communities immediately after flooding resembled that of environments rich in allochthonous matter, the community structure returned to that of oligotrophic environments within three years (Thouvenot et al. 2000).

Thouvenot et al. (2000) determined that the low abundance of autotrophic algae and cyanobacteria observed immediately after the creation of a new reservoir in France was not caused by heavy predation by large zooplankton, because zooplankton biomass was low. The authors also suggest that the high proportion of mixotrophic organisms present after flooding may have been indirectly due to the high dissolved organic matter content of the water, which was exploited either directly by adsorption of dissolved organic matter or indirectly through the consumption of bacteria. Pigmented flagellates, which feed on bacteria, were also commonly observed after flooding. The dominant species of pigmented flagellates (Chrysophyceae and Cryptophyceae) were similar to those occurring in humic lakes, further suggesting that the high dissolved organic matter contributed to the observed species composition immediately after flooding. Pigmented flagellates, such as *Dinobryon* sp., and *Cryptomonas ovata,* could have been affected directly by competition for phosphorus and obtained phosphorus by ingestion of bacterial prey (Thouvenot et al. 2000).

Zooplankton

Information obtained from the reviewed studies suggests that an increase in zooplankton abundance is likely to occur after flooding. Bacterial growth, fuelled by decomposition of terrestrial vegetation and the release of nutrients from flooded soil, will likely stimulate production of rotifers and cladocerans that graze on bacteria. Rotifers typically dominate the zooplankton community initially, because they can colonize new environments faster than cladocerans and copepods. Lower fish predation, due to lower fish densities and more hiding places in the flooded vegetation, will also encourage this production. As the allochthonous organic matter dissipates, there will be a shift to larger-sized cladocerans and copepods. If persistent turbidity depresses bacterial and phytoplankton growth, then zooplankton biomass likely will decline and changes in species composition will be more strongly influenced by fish predation, as outlined in the following section. Key findings from studies that document these changes in the zooplankton community are discussed in more detail below.

According to Northcote and Atagi (1997), zooplankton biomass generally increases in newly flooded reservoirs, with flooded herbaceous vegetation or grasses supporting large populations of cladocerans and rotifers. The large observed increase in zooplankton biomass and productivity was one of the most dramatic responses noted by Paterson et al. (1997) following the impoundment of an experimental lake in northern Ontario. High food availability and low mortality rates contributed to the high zooplankton abundance post-impoundment. Thouvenot et al. (2000) similarly noted an increase in zooplankton biomass in a newly-created reservoir in France.

In the study completed by Paterson et al. (1997), bacteria were likely the primary food source for the zooplankton, because the production of zooplankton was frequently equal to or exceeded that of phytoplankton. Based on the available zooplankton, phytoplankton, and bacteria data, Paterson et al. (1997) mapped the dominant pathway of carbon flow in their system. Prior to impoundment, they postulated that the zooplankton fed primarily on phytoplankton and not bacteria. In the year after impoundment, zooplankton switched to feeding on bacteria, because the high biomass and cell size of bacteria increased their availability. In the second year after impoundment, bacterial biomass declined, although larger cell size was maintained. At the same time, phytoplankton production increased, leading to uncertainty as to the relative importance of bacteria and phytoplankton to zooplankton.

Low mortality rates of zooplankton post-impoundment were either due to lower fish densities or to the increased occurrence of refugia (hiding places) where zooplankton could escape predation. The refugia included the flooded peat sections of the expanded lake where dissolved oxygen concentrations were low (Paterson et al. 1997). The authors noted a shift in dominance from small species (e.g., *Bosmina longirostris*) to larger species (e.g., *Daphnia rosea*). This pattern is consistent with decreased predation by fish, which preferentially select larger prey species, after impoundment.

In general, rotifers initially dominate the zooplankton community in the first year after impoundment (Pinel-Alloul et al. 1989; Thouvenot et al. 2000). Rotifers have a short generation time, but high fecundity. They can, as a result, colonize new environments faster than cladocerans and copepods, which have lower fecundity but longer generation times (Thouvenot et al. 2000). Rotifers also benefit from the increased level of bacterial production commonly observed following impoundment of new reservoirs (Knauer and Buikema 1984, as cited in Pinel-Alloul et al. 1989).

Polyarthra sp. was the most abundant rotifer observed by Thouvenot et al. (2000). It accounted for 24 to 50% of the zooplankton biomass measured immediately after the flooding of a reservoir in France. *Hexarthra mira* was the next most abundant, accounting for 18 to 20% of the total zooplankton biomass.

In contrast, *Conochilus* sp. and *Kellicottia* sp. were the co-dominant species of rotifers observed in a newly created subarctic reservoir in northern Quebec in the first year after impoundment (Pinel-Alloul et al. 1989). *Conochilus* are detritus-bacteria feeders, while *Kellicottia* sp., are microfilter-feeders that consume nanoplankton and, to some extent, bacteria and detritus. Therefore, the relative abundance of these two rotifers immediately after flooding is reflective of the high availability of detritus and bacteria in the newly created system. Species that feed more on nanoplankton and microflagellates, such as *Polyarthra vulgaris*, became more abundant after several years once phytoplankton production increased (Pinel-Alloul et al. 1989).

Cladocerans typically become the dominant species of zooplankton after rotifers (Paterson et al. 1997; Thouvenot et al. 2000; Pinel-Alloul et al. 1989). Thouvenot et al. (2000) observed a switch from rotifers to cladocerans two years after the initial flooding of a reservoir, with larger-sized species, such as *Daphnia longispina* and *Eudiaptomus gracilis*, being evident.

In a subarctic reservoir, Pinel-Alloul et al. (1989) noted that the "inefficient" microfilter-feeder, *Bosmina longirostris*, dominated the cladoceran community in the first two years after impoundment; "inefficient" referred to the optimum food particle size of this species as being below 2 to 5 μm, which is why bacteria and detritus dominate its diet. After the initial two year period, more efficient microfilter feeders, such as *Daphnia longiremis*, *Skistodiaptomus oregonensis*, and *Leptodiaptomus minutes*, became more prominent. The shift in the composition of the cladoceran community coincided with a shift in the dominance of phytoplankton over bacteria.

The performance of the zooplankton community in Southern Indian Lake was notably different from those outlined above. In this lake, zooplankton biomass decreased by 30 to 40% after flooding, with cladocerans and small cyclopoid copepod species being most heavily affected (Patalas and Salki 1984). Calanoid copepods were less affected, with larger species actually being more abundant and widespread after impoundment. *Mysis relicta* went from rare to common

(Hecky et al. 1984), a change attributed to this species' preferences for lower water temperatures and to reduced whitefish predation due to poorer transparency (Patalas and Salki 1984). As previously noted, Southern Indian Lake experienced high rates of shoreline erosion following impoundment, which resulted in high levels of turbidity and reduced water temperatures. These changes to turbidity levels and water temperature were most likely the agents responsible for the observed performance of the zooplankton community.

Benthic Invertebrates

Similar to zooplankton, benthic invertebrates tend to be abundant in new impoundments, because the flooded terrestrial vegetation provides structural habitat and a food source. Generally, midges (Chironomidae) colonize new impoundments first, and are usually more abundant than other benthic organisms. However, other groups of benthic invertebrates can initially dominate, depending on how the new lake or reservoir is formed. For example, early colonizers in a reservoir created by damming a river are riverine species, which are gradually replaced with species that prefer standing water. The succession of benthic invertebrate species varies among case studies, due to differences in the quality and quantity of terrestrial vegetation contained within the newly flooded systems and the dispersal abilities of the local benthic species, as discussed in greater detail below.

Increased benthic invertebrate abundance in newly created lakes and reservoirs was reported in virtually all of the reviewed studies that included a benthos component (Nursall 1952; Danell and Sjoberg 1982; Voshell and Simmons 1984; Hecky et al. 1984; Northcote and Atagi 1997). Hecky et al. (1984) attributed the increased post-impoundment densities in Southern Indian Lake primarily to the input of nutrients and organic material via the flooded shorelines, although reduced predation by adult whitefish also may have been a contributing factor. The coincidental shoreline erosion that occurred in Southern Indian Lake resulted in increased suspended sediment concentrations, which negated the benefits of the nutrient and organic inputs in some areas of the lake. As a result, benthic invertebrate densities were not consistent across the lake, and it was not until the third year following impoundment that benthic invertebrate densities returned to pre-impoundment levels in some regions of the lake (Wiens and Rosenberg 1984).

In most other studies, including Northcote and Atagi (1997), the presence of flooded terrestrial vegetation was identified as the key driver that led to the increased abundance of benthic invertebrates. The flooded vegetation provided habitat and increased food availability.

Midges were commonly reported as one of the earliest and most abundant colonizers of new impoundments, due in large part to the availability of decomposing terrestrial vegetation (e.g., Nursall 1952; Danell and Sjoberg 1982; Hecky et al. 1984; Voshell and Simmons 1984; Northcote and Atagi 1997). Midges and the amphipod *Hyalella azteca* were the most abundant benthic invertebrates detected in Lake Anna a year after it was created (Voshell and Simmons 1984). Abundant genera of midges in Lake Anna included *Glyptotendipes*, *Dicrotendipes,* and *Chironomus*. High midge populations were similarly observed in Southern Indian Lake in areas containing flooded terrestrial vegetation (Hecky et al. 1984). Black spruce (*Picea mariana*) needles were shown to be readily used as a major source of organic substrate by midges (Crawford and Rosenberg 1984).

In a flooded sedge meadow in Sweden, midges were the early invertebrate colonizers, likely because many species of this group could tolerate the rather extreme habitat conditions present in the newly created lake (Danell and Sjoberg 1982). For example, the new lake would experience regular bottom-freezing during the winter, drastic variations in water temperature and low oxygen levels. Danell and Sjoberg (1982) note that species possessing haemoglobin and having large-size larvae that are able to survive on a wide range of food items tend to be more successful that other species of benthic invertebrates in newly created lakes and reservoirs.

Cantrell and McLachlan (1977) studied midge competition and distribution in a newly flooded lake in Northumberland, England. The lake basin lacked any terrestrial vegetation prior to filling, with the bottom substrate consisting primarily of boulder clay. It only received water from rainfall, effectively isolating the basin from invertebrate drift. The study focused on two species of midges: *Chironomus plumosus* and *Tanytarsus gregarious*.

C. plumosus is often the first species to colonize temperate impoundments, because of its extended period of emergence during the summer months and the presence of haemoglobin in its body fluids; the haemoglobin allows this organism to withstand low concentrations of dissolved oxygen for extended periods of time (Cantrell and McLachlan 1977). In contrast, another early colonizer midge species, *T. gregarious*, is unable to withstand low dissolved oxygen levels and is phototaxic (i.e., attracted to light) (Cantrell and McLachlan 1977).

Cantrell and McLachlan (1977) found that *T. gregarious* was more abundant toward the edges of the lake, because the larger size of *C. plumosus* effectively excluded it from the sediment in offshore areas. *T. gregarious* reacted positively to the light and moved into the shallower water present along the edge of the lake. *T. gregarious* was able to effectively colonize these shallower areas, because of a lower amount of sediment compared to the lake centre and a preference of deeper sediment by the competitively superior *C. plumosus*.

In Barrier Reservoir, which was created by damming the Kananaskis River in the Rocky Mountains of Alberta, *Pentapedilum* sp. was the first among the midges to become established, mainly because it was among the first benthic invertebrates to arrive in the newly created reservoir (Nursall 1952). The dominance of *Pentapedilum* sp. and other lotic species in the reservoir, however, was short lived. The inability of these organisms to survive in the standing waters of the reservoir resulted in a notable decline in their abundance by the end of the first summer.

The following spring, the leaf litter present along the bottom of Barrier Reservoir was completely buried with riverine sediment that was washed in during a heavy spring runoff event. As a result, the eutrophic conditions that were previously present in the reservoir disappeared, and a shift in the benthic invertebrate community occurred. *Tanytarsus* sp., a midge commonly found in oligotrophic systems, became dominant by the following year (Nursall 1952).

A similar pattern was observed by Voshell and Simmons (1984) following the creation of Lake Anna in the south-eastern United States. Initially, the benthic invertebrate community was dominated by facultative species originating from the upstream river. By the second year of impoundment, most of the terrestrial vegetation that had been flooded was gone, and the bottom substrate of the lake had changed. The fertile topsoil that was originally flooded was being replaced or covered with finer silty sediments that washed in from upstream sources. Plankton debris and feces was also accumulating on the lake bottom. The change in bottom substrate and the shift from allochthonous to autochthonous organic matter facilitated the replacement of the first colonizers with species that prefer standing water, such as aquatic worms (Oligochaeta).

Midges and *H. azteca* were the most abundant benthic invertebrates detected in Lake Anna a year after it was created (Voshell and Simmons 1984). Snails (Gastropoda) were the third most abundant organisms in the first year after impoundment, with the genera *Physa*, *Ferrissia* and *Helisoma* being abundant. Other abundant organisms included the mayfly (Ephemeroptera) *Caenis amica*, and caddisflies (Trichoptera) of the genus *Oecetis*.

Benthic invertebrate density and composition were similar in the second and third years after impoundment, likely because of the uniformity of the pre-impoundment basin and the use of artificial substrates for sampling (Voshell and Simmons 1984). However, for the reasons outlined above, the fauna in these years was considerably different compared to that present in the first year following impoundment. Midges overtook *H. azteca* as the most common benthic invertebrate, with *Glyptotendipes* sp., *Procladius* sp., the tribe Tanytarsini, *Ablabesmyia* sp., *Chironomus* sp., and *Cryptochironomus* sp. making up most of this group. *Dicrotendipes* sp. and *Endochironomus* sp. became less common. Aquatic worms appeared in the second year, as well as flatworms (Turbellaria) and the phantom midge *Chaoborus* sp. Different species of caddisflies and dragonflies/damselflies (Odonata) became established in the second year. In contrast, abundances of *H. azteca*, *C. amica*, and snails declined substantially. In the third year, caddisflies became notably more abundant, particularly *Cernotina spicata* and *Cyrnellus fraternus*, along with a burrowing mayfly, *Hexagenia munda*. *Hexagenia* species are considered to be a major benthic macroinvertebrate in lentic habitats in temperate areas (Edmunds et al. 1976, as cited in Voshell and Simmons 1984).

Danell and Sjoberg (1982) studied benthic invertebrate succession in a lake created by flooding meadowland that had been dry and dominated by sedges for approximately 20 years. They reported the dates of occurrence and relative biomass of different benthic groups starting from the third year after flooding to the eighth year. As observed in other studies, midges were dominant in the flooded lake and occurred in every sample, although they declined in abundance during the study period. Other early colonizers included some species of aquatic beetles (Coleoptera), aquatic sow bugs (Isopoda), and water bugs (Heteroptera), whereas caddisflies, mayflies, and dragonflies/damselflies became more prevalent six to seven years after flooding. Other invertebrates, such as molluscs (Mollusca; mainly the snail *Gyraulus* sp.), did not appear until the sixth year after flooding, likely due to their lower capacity for dispersal. Danell and Sjoberg (1982) noted that the shift from allochthonous to autochthonous production of organic matter occurred five to six years after flooding, which is longer than that noted in other studies. The prolonged period over which the flooded terrestrial vegetation exerted an influence on the lake system resulted in a longer than expected succession of benthic species, compared to that reported by others (e.g., Voshell and Simmons 1984).

The effects of draining a shallow prairie lake over the winter months were studied by McGowan et al. (2005). The study was completed on Wascana Lake, which is located in southern Saskatchewan. It was expected that the complete exposure and desiccation of bottom sediments would reduce the diversity and abundance of benthic invertebrates. However, no substantial effects were observed during the next summer after spring melt refilled the lake. The results of this study suggest that benthic invertebrate populations living in northern climates may be resilient to variation in water level and subsequent exposure to harsh winter conditions.

Fish

Summary articles highlight the positive and negative effects of new reservoir development on fish (i.e., outline general effects to fish). However, few articles obtained in the literature review specifically discussed effects to fish (i.e., fish were not typically included as one of the test organisms under detailed study). Fish were either removed prior to reservoir development or were not part of the study design. When studied, the abundance and diversity of the resident populations are dependent on the fish species and the extent to which the species can take advantage of the new environment. Upstream and downstream fish populations helped re-establish populations in newly-created or enlarged lake systems. Juvenile fish populations recovered quickly, because of the availability of new habitat and food sources. In contrast, older and larger adult age classes generally were reduced. Fish abundance tended to be higher in lakes containing flooded vegetation compared to those with little or no vegetation. Fish abundance in new impoundments with increased concentrations of suspended sediments tended to be lower than those in systems without high turbidity, because high turbidity levels prevented effective feeding and resulted in reduced growth and egg survival rates. Methyl mercury accumulation in fish tissues tend to be higher in these highly turbid systems, as discussed in more detail below.

According to O'Brien (1990), the creation and operation of reservoirs can affect fish through several mechanisms. Fluctuations in water level and changes in substrate composition can affect spawning success. Spawning success can also be negatively affected by the presence of unstructured shoreline and a scarcity of littoral zone vegetation, particularly in steep-sided reservoirs. Development of littoral zone vegetation may be hindered by wave action, turbidity, or large fluctuations in water level. Increased turbidity levels that may result from shoreline erosion can limit feeding success. They can also directly affect growth and survival rates if turbidity levels are sufficiently high to result in gill abrasions and loss of spawning habitat through sedimentation.

Conversely, the increases in zooplankton productivity that typically occur after impoundment can provide a larger food source to resident fish. Flooded terrestrial vegetation can also provide spawning and rearing habitat to certain fish species. The net effect of reservoir creation on the abundance and diversity of the resident fish community, therefore, is dependent on the make-up of the fish community and to what extent it can adapt to, or take advantage of, the new environment created within the reservoir.

For example, Lindström (1973) observed that when the littoral zone in a reservoir is lost, the fish community shifts from one dominated by littoral fish to one dominated by largely pelagic fish (i.e., fish that prefer to reside in deeper water).

From the fisherman’s point of view (at least as defined by Lindström [1973]), there may be a general decline in the numbers and size of "good fishing" species and an increase in undesirable fish. However, it should be noted that Lindström (1973) focused on systems that experience severe fluctuations in water levels, which tend to destroy littoral habitat.

In north-eastern Belgium, fish densities were drastically reduced after draining and refilling a lake, as detailed by Van de Meutter et al. (2006). However, the studied lake was connected to a number of neighbouring lakes through a series of streams and channels. The connections among the lakes provided a migratory conduit, so that fish could rapidly re-colonize the lake. Within two years of refilling, the number of juvenile fish had recovered to pre-disturbance levels due to a combination of dispersal and increased survival of young-of-the-year fish. Overall fish density, however, had not fully recovered within two years of refilling, with older fish being under-represented in the lake.

Paller (1997) observed a similar trend in a South Carolina reservoir. Within nine months of refilling, the fish community had re-established itself. The number of species present, their relative abundance, and overall fish biomass was similar to pre-disturbance conditions. However, the age-class structure was skewed. The lake contained more juvenile and fewer older fish compared to the age-class structure present prior to disturbance.

The faster recovery observed by Paller (1997), relative to that observed by Van de Meutter et al. (2006), may have been due to the presence of seed organisms in the refilled reservoir Paller (1997) studied. In addition, the reservoir Paller (1997) studied was not completely drained prior to refilling, unlike the Belgian lake that Van de Meutter et al. (2006) examined.

Some studies have found fish populations to be numerous in the first years of the existence of a reservoir, potentially from increased reproduction rate brought about by secure spawning grounds, production of fry afforded by flooded vegetation, or increased food availability (Legault et al. 2004). At the time the La Grande complex reservoirs were impounded in Quebec, overall abundance for all species dropped substantially but then rose over the next three years before decreasing slightly up to the tenth year (DesLandes et al. 1995).

Northcote and Atagi (1997) reviewed the importance of submerged vegetation to fish in reservoirs. In terms of providing spawning habitat, the reproductive success of species, such as lake whitefish, lake trout, northern pike, and walleye, is generally unaffected by the flooding of terrestrial vegetation. However, inundated terrestrial vegetation is often associated with higher abundance of young-of-the-year fishes, because it directly provides rearing habitat and indirectly provides increased food supply through coincidental increases in zooplankton abundance. If the flooded vegetation is structurally complex (i.e., contains lots of branches, leaves and cross-connections), then it can provide valuable habitat for adult fish, it terms of providing rich feeding sites and providing shelter and protection from ambush predators, such as bass and northern pike. Overall fish abundance also tends to be higher in areas with flooded vegetation, compared to that generally observed in lakes or reservoirs that were cleared prior to flooding or that contain little to no vegetation.

In the case of Southern Indian Lake, the abundance of young-of-the-year northern pike (*Esox lucius*) was initially high in the first year following impoundment (Bodaly and Lesack 1984); however, this trend did not persist, and spawning success declined in subsequent years. The initial year-class was also slower growing and in poorer condition than other year-classes (Bodaly and Lesack 1984). Northern pike is also a species that generally does well in reservoirs (Legault et al. 2004). In the first few years of a reservoir's creation, the abundance and growth of northern pike usually increase (Machniak 1975 cited in Legault 2004).

As previously noted, Southern Indian Lake is located in northern Manitoba on the Precambrian Shield in a zone of discontinuous permafrost. It is surrounded by boreal forest and experiences high wave action (Newbury and McCullough 1983). A dam constructed at the natural outlet to the lake caused flooding beyond the thawed zone under the lake into the permafrost-affected upland, which was cleared of timber prior to flooding. Extensive shoreline erosion subsequently occurred, leading to elevated turbidity levels and increased rates of sedimentation within the lake (Hecky et al. 1984).

The increased turbidity levels present in the lake post-impoundment reduced light penetration and visibility. Observed declines in the catch-per-unit-effort of lake whitefish (*Coregonus clupeaformis*) on traditional fishing grounds was attributed to reduced visibility affecting feeding success, either by triggering a redistribution of the fish stocks within the lake or by affecting schooling behaviour. Lake whitefish continued to spawn on their old spawning grounds, but in situ experiments suggested that increased sedimentation of clay and silt negatively affected egg survival (Fudge and Bodaly 1984). Lake trout populations generally are low in Quebec's reservoirs despite the fact that reservoirs present abiotic and biotic factors considered suitable for the species; however, drawdown effects may be a factor (Legault et al. 2004).

The flooding of Southern Indian Lake and the subsequent introduction of large quantities of soil and sediment coincided with a notable increase in mercury concentrations in the muscle tissue of all commercial fish, including northern pike and walleye (*Sander vitreus*) (Bodaly et al. 1984). Similar increases in mercury tissue concentrations were not observed in surrounding, undisturbed lakes over the same period. The source of the mercury was determined to be bacterial methylation of naturally occurring mercury in the flooded soils and the suspension of sediments (Bodaly et al. 1984).

Once methylated in the bottom sediments of a reservoir or lake, methyl mercury may be transferred to biota in a number of ways. There may be passive diffusion of methyl mercury into the water column (Morrison and Thérien 1991) and subsequent uptake into phytoplankton (Plourde et al. 1997). More commonly, flooded soil particles are suspended into the water column by wind or ice-driven erosion. The soil particles are ingested by zooplankton and benthic invertebrates (Louchouarn et al. 1993; Grondin et al. 1995; Mucci et al. 1995). Once in zooplankton and benthic invertebrates, methyl mercury biomagnifies up the food chain to higher concentrations in fish.

Because suspension of sediment is a primary vector by which methyl mercury becomes available to biota, the amount of erosion in a reservoir can substantially impact methyl mercury concentrations in biota. St. Louis et al. (2004) suggested that, if there is little erosion, methyl mercury produced in peat will likely be demethylated. If there is erosion, however, methyl mercury will be rapidly transferred up the food chain. In Southern Indian Lake, mercury levels in fish increased quickly after impoundment, and continued to be high eight years after the lake was flooded (Bodaly et al. 1984).

Once mercury levels are elevated as a result of impoundment, the time required for fish mercury concentrations to return to the background level is dependent on species, fish size, and fish diet. Mercury levels in tissues of planktivorous and omnivorous species return to background concentrations sooner than in piscivorous species, and levels in smaller fish return to background concentrations sooner than in larger fish (Brouard et al. 1990; Verdon et al. 1991; Andersson et al. 1995). Most researchers have estimated that, for piscivorous species, the time to return to background mercury concentrations is 15 to 30 years (Verdon et al. 1991; Andersson et al. 1995; Porvari 1998), whereas those for non-piscivorous species range from 2 to 20 years (Verdon et al. 1991). The persistence of high mercury levels in fish tissues in Southern Indian Lake, as reported by Bodaly et al. (1984), is consistent with these projections.

Summary and Conclusions

Flooding terrestrial vegetation can result in a surge in nutrient concentrations, particularly of nitrogen, carbon, and phosphorus. However, the released phosphorus may be in a non-bioavailable form, which encourages the growth of bacteria over that of phytoplankton. Herbaceous vegetation generally decomposes faster than woody vegetation, and thus the type and amount of flooded vegetation affects the magnitude and duration of the nutrient surge.

Flooded soil can also be a source of nutrients to a newly created lake or reservoir. However, this input may not be as substantial as that originating from flooded vegetation. Removing terrestrial vegetation and soil prior to impoundment may limit the magnitude and duration of the nutrient surge, but the overall net benefits that result from this management option are dependent on site-specific conditions. Keeping the vegetation in place can enhance macrophyte growth, zooplankton abundance, and benthic invertebrate diversity. The removal of the terrestrial vegetation can also lead to increased shoreline erosion in the newly created lake or reservoir. Another management option available to limit or stabilize nutrient levels in a newly created system involves the repeated draining and refilling of the system.

Most of the reviewed studies did not discuss turbidity as a major driver for ecosystem recovery. Site-specific factors, however, can lead to erosion and increased levels of turbidity in newly created lakes and reservoirs. In a well-studied lake impoundment in northern Manitoba, extensive and ongoing erosion of fine-grained shorelines contributed to a sustained increase in turbidity levels, which has had a notable effect on the aquatic ecosystem.

With the exception of mercury, release of metals from sediment as a consequence of flooding was not observed or commented upon in the majority of the reviewed literature. Low oxygen conditions in flooded sediment, however, can result in the release of dissolved manganese and iron, as observed in a refilled reservoir in Germany (Nienhuser and Braches 1998).

In contrast to other substances, it is common for mercury concentrations in fish to increase following impoundment. This phenomenon is likely due to the inundation and subsequent decomposition of organic material that promotes the microbial methylation of inorganic mercury to organic methyl mercury. It has been well established that (1) mercury in pristine and flooded soils is predominately bound to organic matter, and (2) that mercury methylation is related to organic carbon content of the flooded soil/sediment. Methyl mercury is the most toxic form of mercury and readily accumulates in aquatic organisms.

To mitigate the effect of methyl mercury generation, bioaccumulation through the food chain and associated environmental and health risks, there are three processes that can be targeted: mercury methylation, mercury bioaccumulation, and fish consumption. Available mitigation options include partial or complete stripping or capping of organic materials and soils, high temperature burning of vegetation and leaf litter, liming, selenium additions to newly created reservoirs and lakes, intensive fishing, fish barriers (screens), restricted access, and/or fish consumption advisories. Of the options available, selenium additions and liming are not widely recommended, and consumption advisories serve only to protect human health without directly addressing mercury concentrations in fish.

The surge in nutrients that typically occurs following the creation of a new lake or reservoir generally leads to a brief increase in phytoplankton growth and photosynthesis, but bacterial growth quickly dominates. High bacterial growth coincides with low phytoplankton growth, suggesting that bacteria out-compete phytoplankton for nutrients.

As the source of organic matter switches from external to internal, a shift in dominance typically occurs, with phytoplankton replacing bacteria as the dominant planktonic organism. In other words, when an area is first flooded, the organic matter in the system or entering the system tends to originate primarily from external sources, such as flooded terrestrial vegetation or other materials entering the system through runoff or inflow. The characteristics of these materials tend to favour bacterial growth over phytoplankton; over time, the external material decays and is replaced with organic matter that originates primarily within the lake, from dead and decaying phytoplankton, zooplankton, fish, or other material (e.g., fish feces). The characteristics of the internal materials tend to favour growth of phytoplankton over that of bacteria, which leads to a shift in dominance between these groups of organisms. The period over which the shift from bacterial to phytoplankton dominance occurs is dependent on site-specific conditions. However, in some studies, plankton community structure returned to that characteristic of oligotrophic environments within three years of flooding.

Zooplankton biomass is generally high initially in new impoundments, with the possible exception of systems that experience notable shoreline erosion and turbidity. The high biomass is due to high food availability, in the form of abundant bacterial or phytoplankton growth, and low mortality rates, due to low fish densities and the availability of refugia in flooded vegetation. Initially, rotifers typically dominate the zooplankton community, because they are able to colonize new environments faster than cladocerans and copepods. Rotifers also benefit from the initial high level of bacterial production commonly observed following impoundment of new reservoirs.

Cladocerans typically become the dominant species of zooplankton after rotifers, possibly within two to three years of impoundment. Changes in the zooplankton community generally coincide with the shift from bacteria to phytoplankton dominance in the lowest trophic level. If persistent turbidity occurs following impoundment, then zooplankton biomass will likely decline and changes in species composition will be more strongly influenced by fish predation.

Similar to zooplankton, benthic invertebrates tend to be abundant in new impoundments, because the flooded terrestrial vegetation provides structural habitat and a food source. Generally, midges colonize new impoundments first, and they are usually more abundant than other benthic organisms. However, other groups of benthic invertebrates can initially dominate, depending on how the new lake or reservoir is formed. For example, early colonizers in a reservoir created by damming a river tend to originate from the river. They are then gradually replaced with species that prefer standing water. The rate at which succession within the benthic community occurs is dependent on the time required for flooded terrestrial vegetation to decay and dissipate, as well as the dispersal abilities of the local benthic populations. In general, succession will occur more quickly in systems where the flooded vegetation quickly dissipates and invertebrate drift from nearby standing waters occurs.

Key findings of the literature review for fish suggest that the net effect of reservoir/lake creation on the abundance and diversity of the resident fish population is dependent on the make-up of the fish community and the extent to which it can adapt to, or take advantage of, the new environment created within the lake. Flooded or refilled systems that are connected to surrounding waterbodies can experience rapid colonization and/or recovery, although the age-class structure post-impoundment tends to be biased towards a greater abundance of juvenile fish relative to older fish. Fish abundance also tends to be higher in lakes or reservoirs containing flooded vegetation, compared to that generally observed in lakes or reservoirs that were cleared prior to flooding or that contain little to no vegetation.

Fish abundance in new impoundments with increased concentrations of suspended sediments tend to be lower than those in systems without turbidity issues. High levels of turbidity can negatively affect fish through reduced feeding success. High levels of suspended sediment can also cause gill abrasions and the associated sedimentation can reduce egg survival rates. In addition, mercury levels in fish tissues tend to be higher in more turbid systems.

8.11.1.3.2 Applicability of Literature Review Findings to Kennady Lake

There are some important differences between Kennady Lake and the systems that have been reported in the available literature. Key areas of difference include the following:

- the potential influence of terrestrial vegetation on the refilled lake;
- the potential for erosion;
- the amount of organic matter present initially after refill that may influence methyl mercury production;
- increased nutrient concentrations in the refilled lake; and
- the rate at which recovery may occur.

These key areas of difference are discussed in more detail below.

Terrestrial Vegetation

The presence of terrestrial vegetation was identified in a number of studies as a key driver that influences initial nutrient dynamics, methyl mercury production, and primary productivity in flooded or refilled systems. Unlike many of these studies, Kennady Lake refilling does not involve flooding of surrounding terrestrial vegetation. During refilling, the water level of Kennady Lake will return to its baseline elevation and not higher. However, during the operations phase, terrestrial vegetation could colonize the dewatered lake bed. Given the physical characteristics of Kennady Lake and its geographical location, notable in-growth of terrestrial vegetation is unlikely. Kennady Lake is situated in the sub-arctic and is surrounded by tundra vegetation, which consists largely of dwarf, upland woody vegetation interspersed with grasses, sedges, moss, and lichen. The lake itself contains three categories of aquatic habitat, which include the following (as outlined in Section 8.3):

- shallow, nearshore habitat within the zone of freezing and ice scour (i.e., less than 2 metres [m] deep);
- nearshore habitat deeper than the zone of ice scour where wave action prevents excessive accumulation of sediment (i.e., greater than 2 m but less than 4 m); and
- deep, offshore habitat with substrate usually consisting of a uniform layer of loose, thick organic material and fine sediment (i.e., greater than 4 m).

The lack of fine sediment around the periphery of the lake, and the consistent presence of boulder and cobble through the shallow areas of the lake, will effectively limit colonization of the lakebed by terrestrial vegetation through vegetative propagation (i.e., root growth). Vegetation is more likely to be established through seed dispersal and subsequent germination, with the seeds being dispersed across the nearshore rocky habitat to colonize the fine sediments that are currently located in the deeper sections of the lake. Vegetation is expected to establish slowly and coverage would be patchy. Initial colonizers are thought to be graminoids (grasses and sedges).

The size of the boulder/cobble barrier that separates the tundra vegetation from the fine lake sediments is expected to increase as a result of dewatering. During the latter part of the dewatering process, re-suspension of the bed sediments is projected to occur. Re-suspension of the bottom sediments will occur, because, as water levels in Kennady Lake decline, the portion of the lake affected by wind-driven scouring will change. It will move from the current shallow nearshore, boulder/cobble habitat into what is now the deeper fine sediment areas. The fine sediment located in these areas will become entrained in the water column and discharged to the Water Management Pond (WMP) in Area 3, where it will either settle out of solution or be removed. The net result will be an extension of the boulder/cobble habitat that currently separates the surrounding tundra vegetation from the fine sediments in Kennady Lake, which will increase the difficulty for terrestrial vegetation that relies on vegetative propagation to become established in the dewatered portions of Kennady Lake.

Based on the above, the degree to which terrestrial vegetation becomes established within the dewatered sections of Kennady Lake is expected to be limited.

Erosion and Associated Turbidity

In some refilled lakes or flooded areas, extensive erosion of the shoreline was noted. A similar trend is not expected to occur when Kennady Lake is refilled, because water levels are going to return to existing elevations. The surrounding tundra is not going to be inundated, and the existing shoreline consists almost entirely of rocky substrate, which is resistant to erosion. Consequently, effects associated with erosion are not expected to occur in Kennady Lake, such as prolonged periods of poor water clarity from increased turbidity, and the associated limitations on phytoplankton development.

Methyl Mercury Production

Results of the literature review indicate that the presence of organic matter is a key driver that controls methyl mercury production in a flooded or refilled area. The decomposition of the organic matter promotes the microbial methylation of inorganic mercury to organic methyl mercury, with the main pulse of methyl mercury production generally occurring in the first few years following flooding or refilling. Although all inundated soils and vegetation are sources of organic carbon and potential sites of methyl mercury production, inundated conifer trees, needles, and boughs appear to be especially important methylation sites, as noted in the literature review.

Conifer trees, needles, and boughs will not be present in Kennady Lake upon refilling. Terrestrial vegetation that may be present will likely consist of grasses and other weedy species, and the abundance of the terrestrial vegetation in Kennady Lake is expected to be limited. As noted above, the shallow zones of Kennady Lake consist almost exclusively of rocky substrate, which will prevent the vegetative propagation of the surrounding tundra vegetation. The size of the rocky zone is also expected to increase during the latter phases of drawn-down, further increasing the distance between the existing tundra vegetation and the fine lake bottom sediments.

Although flooded soils can provide the organic material necessary to support methyl mercury production, this mechanism is of little relevance to Kennady Lake. Refilling activities will be limited to the lake itself. The surrounding soils will not be inundated, and the bed sediments in the refilled lake will effectively be the same as those that currently exist, with one possible exception. Organic materials in the bed sediment exposed to the atmosphere during the operational life of the Project may undergo degradation. In other words, the organic content of the bed sediments may decline over the life of the Project, because of increased exposure to aerobic conditions.

The potential for methyl mercury production to occur in Kennady Lake after refilling, therefore, is expected to be limited, because a new source of organic matter is unlikely to be present. The in-growth of terrestrial vegetation will likely be limited by the rocky substrate that dominates the shallow zones of Kennady Lake, and the organic materials contained in the existing bed sediments may experience a greater level of aerobic decay than currently occurs while the lake is dewatered. Without a large organic carbon source to support the process, it is unlikely that methyl mercury production will be of concern in Kennady Lake once refilled.

Increased Nutrient Levels

Following the creation of a new lake or reservoir, there is typically a surge in nutrients, which leads to a brief increase in phytoplankton growth and photosynthesis. Water quality model results currently indicate that a long-term increase in nutrient levels may occur in the refilled Kennady Lake, which could result in a more prolonged influence on phytoplankton growth and photosynthesis than has been observed in new lakes and reservoirs. The long-term increase in nutrient levels in the refilled lake is expected to increase lake trophic status to mesotrophic (Sections 8.8.4.1.1 and 8.8.4.1.2). This change in trophic status may influence the recovery of aquatic communities, and result in differences in lower trophic and fish community structure in the refilled lake relative to pre-development conditions.

Recovery Time

In several of the reviewed studies, recovery times for different components of the aquatic ecosystem were noted, with some being as short as two to three years. Most of the lakes described in the literature review are located at lower latitudes and have different species composition, greater species richness, higher productivity, and more complex food-webs than Kennady Lake under baseline conditions. These factors have a direct bearing on recovery rates and suggest a slower recovery in Kennady Lake.

Summary

Although some of the key findings from the literature review are not directly applicable to Kennady Lake, the overall trends documented in the reviewed studies provide evidence that an aquatic ecosystem will re-establish itself within Kennady Lake after refilling. The expected trajectory of recovery is outlined below.

8.11.1.3.3 Predicted Recovery of Kennady Lake

At the end of operations, some of the aquatic habitat in Kennady Lake that was physically altered during Project operations will be re-submerged. Habitat enhancement structures (e.g., finger reefs and habitat structures on the decommissioned mine pits/dykes) will be constructed. Refilling of Areas 3 through 7 will begin, using natural runoff from the upland watershed and waters diverted from Lake N11. During this time, Area 8 will continue to receive only natural runoff from the surrounding area.

After approximately eight years, Areas 3 to 7 of Kennady Lake will be full of water. Thereafter, once water quality meets regulatory requirements, dyke A will be removed. Kennady Lake will once again consist of five interconnected basins.

The hydrology of the reconnected system is expected to be similar to existing conditions as soon as dyke A is removed and pumping from Lake N11 ceases. The natural drainage of the B, D, and E watersheds to Kennady Lake will be restored; however, in the A watershed, Lake A3 will continue to flow to the N watershed. Within a short period of time, water quality in the refilled lake is expected to return to conditions suitable to support aquatic life. Therefore, the physical and chemical environment in Kennady Lake will be in a state that will allow re-establishment of an aquatic ecosystem, although the re-established communities may differ from pre-development communities.

The predicted recovery of the aquatic ecosystem in Kennady Lake is outlined below, with reference to the key components of the system. These components include lower trophic communities (phytoplankton, zooplankton, benthic invertebrates) and fish.

Lower Trophic Levels

Development of lower trophic communities is expected to reflect the key factors identified in the literature review and the quality of water used to refill the lake. Because Kennady Lake does not support a substantial aquatic plant community due to physical factors and climate, it is unlikely to do so in the future. Therefore, the discussion of lower trophic levels is restricted to the development of plankton and benthic invertebrate communities.

Growth of terrestrial vegetation within Areas 3 through 7 during the period of exposure will be limited, based on the anticipated low rate of seed dispersal and slow vegetative growth that currently occurs around the margins of the lake in the tundra region. The dry lakebed will be surrounded by a margin of boulder and cobble, which will act as a barrier to vegetative propagation into the lake from the surrounding tundra. In addition, the amount of dried up organic sediments present on the exposed lake bottom will likely be reduced by oxidation during the period of exposure. Overall, the amount of organic material present on the exposed lake bottom upon refilling is expected to be lower than typically present when creating new reservoirs in the more southern locations described in the literature review, and potentially lower than currently exists in Kennady Lake.

Nutrient supply is expected to be higher than under baseline conditions, and should result in rapid development of phytoplankton in re-filled areas of Kennady Lake. Because the amount of flooded organic material in Kennady Lake is anticipated to be low, its contribution to an initial trophic upsurge will likely be minor and short-lived. Nevertheless an initial period of upsurge may occur due to additional nutrient inputs from re-flooded sediments. Phytoplankton productivity likely will peak during this period, although the magnitude of the peak compared to the long-term level of productivity may be reduced in comparison to those observed in other systems. The initial upsurge may be followed by a period of bacterial dominance, until allochthonous (i.e., external) sources of dissolved organic matter are exhausted. In Kennady Lake, external sources are primarily represented by the terrestrial vegetation (e.g., grasses and weedy species) that will have invaded the lakebed. The shift from allochthonous to autochthonous (i.e., internal) carbon sources is expected to occur during the filling period (eight years) and may be complete by the time the lake is fully refilled.

The rapid shift from external to internal sources of carbon will likely facilitate the development of the phytoplankton community in Areas 3 through 7, with the community becoming established during the refilling period. The development of the phytoplankton community in Areas 3 through 7 will also be facilitated by the arrival of phytoplankton from upstream sources (i.e., B, D, and E watersheds) and from Lake N11, which will be used as a water source to shorten the filling time. As a result, a phytoplankton community characteristic of the lake's new trophic status is expected to develop in Areas 3 through 7 by the end of the refilling period or shortly thereafter (i.e., within five years after refilling is complete).

The phytoplankton community of Area 8 will remain similar during the refilling period. Breaching of dyke A will result in greater flow-through in Area 8 and a shift in phytoplankton community structure, as water from Areas 3 through 7 containing higher nutrient levels, mixes with Area 8 water. As nutrient concentrations increase to the long-term levels, primary productivity will also increase, and the phytoplankton community is expected to shift to one characteristic of mesotrophic systems.

Overall, the expected time frame for recovery of the phytoplankton community is estimated to be approximately five years after refilling is complete, taking into consideration that the phytoplankton community will begin to develop during the eight year refilling period.

Zooplankton community development is predicted to closely follow recovery of the phytoplankton community. During the initial upsurge, zooplankton biomass is expected to peak, although this peak would also be less pronounced compared to that reported for zooplankton communities in other newly created reservoirs. It would then gradually decline to the level characteristic of the long-term nutrient concentrations in Kennady Lake. Colonization sources will be the same as those for phytoplankton and include the upstream watershed, Lake N11, and the WMP. The zooplankton community of the refilled lake is expected to be different from the existing community, as it will become characteristic of mesotrophic lakes. The expected time frame for the development of the zooplankton community is longer than that of phytoplankton (i.e., likely within about five to 10 years of Kennady Lake being completely refilled).

Recovery of the benthic invertebrate community is expected to be slower than that of plankton communities. In studies of newly-created reservoirs, benthic invertebrate community development was frequently found to be related to the availability of structural habitat and food provided by flooded terrestrial vegetation. As noted above, establishment of terrestrial vegetation on the exposed lakebed of Areas 3 through 7 is anticipated to be minimal. Limited dispersal abilities of non-insect benthic invertebrates (e.g., worms, mollusks), which lack a winged adult life stage, represent another factor accounting for the prediction of slower community development.

Benthic invertebrate density is anticipated to be higher than existing levels during the first few years of refilling, provided that turbidity does not limit phytoplankton productivity; however, as the shoreline is primarily rocky and not susceptible to erosion, prolonged periods of poor water clarity are not expected. Following this initial refilling period, benthic invertebrate densities are predicted to stabilize at a level characteristic of the new, higher trophic status of Kennady Lake.

The succession of benthic invertebrate species varies among case studies, largely due to differences in the amount and type of terrestrial vegetation present within newly flooded areas and the dispersal abilities of local benthic invertebrate species. As in the case of plankton, upstream surface waters (i.e., B, D and E watersheds), N11, and the WMP will represent sources of colonization via drift to Areas 3 through 7. However, aquatic insects can also colonize from adjacent watersheds by deposition of eggs by winged adults. Most colonization studies found that midges (Chironomidae) were the initial colonizers, followed by slower establishment of other groups. Under existing conditions, Kennady Lake benthic communities consist mostly of midges, fingernail clams (Pisidiidae), aquatic worms (Oligochaeta), and roundworms (Nematoda), with about half of the total abundance contributed by midges. A greater dominance by midges and aquatic worms is likely after refilling, reflecting the elevated nutrient levels predicted, which may result in reduced dissolved oxygen concentrations during winter.

The Area 8 benthic invertebrate community is likely to change after reconnection to the other basins, consistent with predicted changes in water quality and the plankton community. A gradual increase in concentration of the limiting nutrient (phosphorus) to the long-term level is expected in this basin, which in turn will result in increased plankton abundance and biomass, and ultimately, higher benthic invertebrate abundance and biomass compared to pre-development conditions.

The estimated time to recovery for the benthic community in Kennady Lake is about 10 years after refilling is complete, consistent with accounts of lake recovery and lake creation described by previous studies. At the end of the recovery period, the benthic invertebrate community will be different from the currently existing community in Kennady Lake and surrounding lakes. The new community will likely be of higher abundance and biomass, reflecting the mesotrophic nature of the refilled lake. In terms of composition, it will likely consist largely of the same groups that exist under pre-development conditions (i.e., midges, fingernail clams, aquatic worms, and roundworms). However, their relative proportions will differ from baseline, reflecting the different abilities of invertebrate species in these groups to take advantage of the increased food supply, interactions among invertebrate species, the influence of fish predation, and the potential for a greater reduction in dissolved oxygen concentration during winter than before development. In particular, a greater dominance of the benthic community by midges and aquatic worms is anticipated due to the likelihood of reduced late winter dissolved oxygen concentrations.

Fish

Kennady Lake is currently a nutrient limited, oligotrophic lake containing a simple food web. The dominant food chain consists of lake trout as the top predator, feeding primarily on round whitefish. Round whitefish, in turn, feed on zooplankton and benthic invertebrates. Burbot and northern pike are the only other piscivorous fish species present in Kennady Lake, but they are found in much lower abundance than lake trout. Arctic grayling, lake chub, ninespine stickleback, and slimy sculpin (i.e., other fish species present in Kennady Lake) are typically benthivorous and serve as secondary prey items for lake trout. Young-of-the-year lake trout feed almost exclusively on zooplankton, but undergo an ontogenetic diet shift to fish and benthic invertebrates as they grow (Martin and Olver 1980). Therefore, zooplankton are an important link in the dominant Kennady Lake food chain, in that they support the growth and recruitment of lake trout.

The re-establishment of the fish community within Kennady Lake, and the speed at which it will occur, will depend on the ability of fish to re-colonize the refilled lake, the habitat conditions within the refilled lake, and how succession among the top predators takes place within the refilled system after it has been fully connected to the surrounding environment. Predicted nutrient enrichment and change in trophic status to mesotrophic will also influence the rate of re-establishment and overall structure of the fish community. These aspects of recovery are outlined below, as well as a discussion of the potential for non-resident fish species to enter Kennady Lake, and of the time required for a stable fish community to become established within the refilled lake.

Location of Potential Migrants and Initial Re-colonization

Fish are present in the upper Kennady Lake watershed, including in the A, B, D, and E sub-watersheds. These sub-watersheds will be diverted away from Kennady Lake during the operational life of the Project. However, the B, D, and E watersheds will be reconnected to the lake once refilling begins. Fish populations, including Arctic grayling, northern pike, burbot, lake chub, slimy sculpin, and ninespine stickleback, are expected to persist in these watersheds during Project operations.

At closure, the B, D, and E sub-watersheds will be re-diverted back to Kennady Lake. During refilling, exclusion measures will be used to limit the initial migration of fish from the upper sub-watersheds into Kennady Lake. The mitigation measures will target large-bodied fish, such as northern pike, burbot, lake trout, and Arctic grayling. Small-bodied fish, such as lake chub, slimy sculpin, and ninespine stickleback, will likely pass through the exclusion devices, as will the young-of-the-year of large-bodied fish. It is anticipated that the smaller fish species will move quickly into Kennady Lake. The smaller cyprinids (i.e., minnows) and younger age classes of large-bodied fish species tend to be more mobile and prone to downstream immigration in comparison to larger fish (Binns 1967; Avery 1978; Olmsted and Cloutman 1978).

During the initial period of refilling, some mortality of the incoming small-bodied fish is likely to occur, because of insufficient water depths and possibly elevated levels of turbidity. As conditions improve, and water depths increase, the early migrants will become permanently established, feeding on the plankton and benthic invertebrate communities that are themselves becoming established in the refilled lake. Nutrient levels in the refilled Kennady Lake are predicted to be higher than under existing conditions. The increase in primary productivity from nutrients may also result in increased growth and production of these small-bodied forage fish species.

Development of self-sustaining populations of the small-bodied fish species in Kennady Lake may occur during the refilling process; however, prior to complete refilling, access to suitable spawning and rearing habitat may be limited, as it is primarily in the upper 3 to 4 m of the lake, where coarse sediments are present. Based on the current refill scenario, which assumes that remaining space in the Tuzo pit will be refilled first, the upper 3 to 4 m of Kennady Lake will likely only contain sufficient water depths to support spawning and rearing near the end of the refill period. Small-bodied forage fish species (e.g., lake chub, slimy sculpin, and ninespine stickleback), are less specific with respect to habitat and are more tolerant of low dissolved oxygen levels; as such, they will likely recolonize and establish self-sustaining populations within Kennady Lake over time.

Area 8 will be a second potential source of migrants. Area 8 will be separated from the dewatered basins of Kennady Lake during the operational life of the Project by dyke A, and it is anticipated that Area 8 will contain residual populations of lake chub, slimy sculpin, ninespine stickleback, Arctic grayling, northern pike and burbot, all of which are relatively hardy fish species. Area 8 is not expected to contain residual populations of lake trout or round whitefish because of overwintering habitat limitations. Area 8 is shallow, and will provide limited overwintering habitat to these cold-water fish species.

Lake I1 is another potential source of migrant fish. Lake I1 drains into Area 8; lake trout, Arctic grayling, slimy sculpin and ninespine stickleback have been captured in the lake. Due to the depth of Lake A1 (maximum depth of 11 m), the ephemeral nature of the outlet stream, and the presence of both adult and juvenile lake trout, it is likely that this lake has a self-sustaining population of lake trout. Juvenile fish in Lake I1 disperse downstream to Area 8 in spring, likely to alleviate density-dependent factors, such as competition for limited food resources and to escape predation.

Migrants from Area 8, and potentially from Lake I1, will complement downstream migration from the upper B, D, and E sub-watersheds after Kennady Lake has been completely refilled and dyke A has been removed. Prior to the removal of dyke A, the colonization of Kennady Lake by small-bodied fish will occur exclusively through the downstream migration of fish from the upper B, D, and E sub-watersheds. Once dyke A has been removed, the migration of fish from Area 8 into the rest of Kennady Lake is expected to be rapid, due to proximity and the increased productivity that is expected from nutrient enrichment in Kennady Lake and Area 8.

Large-bodied fish from the B, D, and E sub-watersheds will be allowed to freely immigrate to Kennady Lake once conditions in the lake are acceptable, i.e., water levels have been re-established, water quality is suitable to support aquatic life, and stable plankton, benthic invertebrate, and forage fish communities have become established. Immigration of fish from downstream lakes may also occur, although to a lesser extent. Lakes in the L watershed generally are too shallow to support populations of large-bodied fish species, but lakes in the M watershed, particularly Lake M4 and Lake M3, are deep enough and large enough to support populations of lake trout, round whitefish, Arctic grayling, northern pike, and burbot. Both Arctic grayling and northern pike have been documented migrating upstream from these lakes. The upstream migration occurs in spring when Arctic grayling are migrating into streams to spawn and northern pike are migrating into streams to access flooded riparian areas. It is unclear how far upstream these fish currently travel, but some may eventually find their way into Kennady Lake.

Immigration of lake trout and round whitefish from downstream lakes is less likely to occur. Lake trout and round whitefish are fall spawners and typically spawn in lakes. They do not, as a result, have an inherent need to migrate into streams to complete their life cycle. However, these species may make movements into streams for feeding or rearing; for example, lake trout have been observed moving into the Kennady Lake outlet (Stream K5) in spring to feed on spawning Arctic grayling. In addition, there are numerous barriers present in the streams that connect Kennady Lake to the lakes in the M watershed. Upstream passage over these barriers is only possible during the spring freshet, not during the fall spawning period when lake trout and round whitefish may be traveling through the streams.

The partially dewatered areas of Kennady Lake, including the WMP, will not be a source of migrant fish. Turbidity levels are expected to be high and habitat will be unsuitable for all fish species present in the existing lake.

Establishment of Large-bodied Fish in the Refilled Kennady Lake

Northern pike, Arctic grayling, and burbot are likely to be large-bodied fish species that readily re-establish in the refilled Kennady Lake, as they are hardy species and are found to inhabit a wide range of lake conditions. These species are likely to enter the refilled Kennady Lake initially as juveniles from the D and E sub-watersheds.

Burbot will also likely enter Areas 3 to 7 of Kennady Lake from Area 8 once dyke A is removed. Burbot are expected to become established early in the recovery of Kennady Lake as they are tolerant of turbid water (Chen 1969; Hatfield et al. 1972 cited in McPhail 2007) and are omnivorous and voracious feeders (Kahilainen and Lehtonen 2003). As a result, burbot are likely to be able to quickly take advantage of the forage fish and benthic invertebrates available in the lake, particularly in the absence of other large predators (e.g., northern pike and lake trout).

Arctic grayling are also expected to establish earlier than northern pike, because, like burbot, they are relatively tolerant of turbid water (McLeay et al. 1983, 1984, 1987), and have very broad feeding preferences (McLeay et al. 1984; Scott and Crossman 1973; Birtwell et al. 2005). Arctic grayling show considerable low oxygen tolerance for salmonids (Eriksen 1975 as cited in Hubert et al. 1985). Spawning habitat for Arctic grayling will be available in streams in the reconnected B, D, and E watersheds and downstream of Area 8.

Northern pike is expected to re-establish self-sustaining populations later in the refilled Kennady Lake. Northern pike are dependent on aquatic vegetation for spawning and rearing. Although aquatic vegetation is expected to eventually become re-established in the lake, re-colonization of aquatic vegetation is expected to be slow especially as the presence of aquatic vegetation in Kennady Lake is currently limited. As a result, recruitment of northern pike in Kennady Lake will occur for some time primarily through migration from lakes in the D and E sub-watersheds, and to a lesser extent from downstream of Area 8. However, as a result of the increased nutrients in the refilled lake, there may be an increase in aquatic macrophyte growth, which would improve the availability of suitable spawning and rearing habitats for northern pike in Kennady Lake. Piscivory by northern pike during recolonization of Kennady Lake may also affect the overall species composition in the lake.

Round whitefish and lake trout are expected to be the last fish species to re-establish in the refilled lake, as these species are less likely to migrate between lakes, in comparison to burbot, Arctic grayling, and northern pike. Barriers present in the streams between the lakes in the M watershed and Kennady Lake will also impede migration of these species. Due to the change in trophic status to mesotrophic, there may be reduced suitability and availability of overwintering and spawning/egg incubation habitat in the refilled Kennady Lake for cold-water fish species, such as lake trout and round whitefish; this may limit the abundance of lake trout and round whitefish relative to other large-bodied fish species.

Other Species

The possibility exists for species not currently residing in the lake to become established in Kennady Lake. Kennady Lake does not currently contain cisco or sucker species. One longnose sucker was captured moving downstream out of Kennady Lake in the spring of 2000. However, longnose sucker have never been captured in Kennady Lake, despite the extensive gillnetting efforts that have occurred since 1996. Cisco are present in Lake 410 and Lake M4, which are located downstream of Kennady Lake, and in Lake N16 in the adjacent N watershed. Longnose sucker[11] are similarly present in Lake N16 and in the small lakes and streams immediately north of Kennady Lake in the N watershed. Furthermore, cisco and longnose sucker from N16 could enter the D and E lakes during operations and then move into the refilled Kennady Lake after the connection is restored.

It is unclear why these species do not currently reside in Kennady Lake. Cisco co-exist with round whitefish in Lake N16 and Lake 410, indicating that these two closely-related coregonid species occupy niches different enough to allow them

[11] White sucker was also recorded in Lake N16 in 1999. This is the only reported instance of white sucker in the watershed upstream of Kirk Lake and may potentially be a misidentification.

to co-exist in the same lake. Physical habitat is unlikely to preclude cisco from Kennady Lake, because they spawn over rocky substrates in nearshore areas (similar to round whitefish). Cisco differ from round whitefish in their food preferences and feeding behaviour; cisco are pelagic planktivores, whereas round whitefish are typically bottom-feeding benthivores (Scott and Crossman 1973; Richardson et al. 2001). Plankton communities in Lake N16 and in Kennady Lake are currently similar; therefore, food availability is unlikely to be the reason for the absence of cisco from Kennady Lake.

Known populations of longnose sucker are located farther from Kennady Lake than cisco, but are more likely to eventually access Kennady Lake than cisco. Longnose sucker have an inherent instinct to migrate into streams in spring to spawn. Longnose sucker were found to move between all lakes and streams between Lac de Gras and Kodiak Lake during springtime (Low 2002). This finding suggests that longnose sucker can make extensive migrations between lakes and could eventually move from lakes in the N watershed to Kennady Lake in time. It is unclear whether longnose sucker could become established in Kennady Lake, should they be able to access it. Longnose sucker feed on benthic invertebrates and prefer cold, oligotrophic lakes (Scott and Crossman 1973; Richardson et al. 2001). Currently, habitat and food availability appear to be suitable for longnose sucker in Kennady Lake; however, based on catch data, it appears that a longnose sucker population does not exist in the lake. Conditions in the refilled Kennady Lake will be different than what currently exists, and it is possible that due to changes in prey abundance, habitat conditions, and predators, that a population of this species will become established.

The Effect of Habitat Modifications on Fish Recovery

The mine rock piles in Areas 5 and 6 will be designed and constructed in a way to ensure long-term stability. The Fine PKC Facility in Areas 1 and 2 will be reclaimed during mine operations, including covering with coarse PK and mine rock and grading. The mine pits will be reclaimed once mining has been completed. 5034 will be backfilled with mine rock and Hearne will be backfilled with fine PK. The Tuzo Pit will not be backfilled with material, but instead will be allowed to flood following operations. There will be some permanent losses of habitat in Kennady Lake due to mine rock piles, PK storage, and mine pits; however, compensation habitats will be constructed in the Kennady Lake watershed to offset losses. As identified in the Conceptual Compensation Plan (CCP, Section 3, Appendix 3.II), options for fish habitat compensation within Kennady Lake itself include construction of finger reefs, construction of habitat structures on the decommissioned mine pits/dykes, and widening of bench pits.

The refilled sections of the mine pits are not expected to hinder or prevent the recovery of the Kennady Lake fish community. Although the refilled areas will be in the order of 300 m deep, the loss of productive lake bottom within the pit is expected to be small (less than 10% of the lake area). There is no evidence to suggest that fish avoid deep-water pits where they contain sufficient dissolved oxygen. The upper level of the Tuzo pit will likely provide additional summer thermal refuge for fish. As observed during the 2010 hydroacoustic study, when Areas 4 and 6 are thermally stratified in summer, lake trout seek deeper water areas with cooler water.

Restocking Program

As previously noted, lake trout and round whitefish are expected to persist in Lake I1 throughout the operational life of the Project. These fish species may eventually immigrate into the refilled Kennady Lake, although it may take a considerable amount of time for the populations to establish. As a result, restocking may be considered as an approach to supplement the natural migration process and speed the time to recovery for lake trout and round whitefish. Restocking is a biologically viable and technically feasible option to re-establish more sedentary fish species, such as lake trout and round whitefish. Stocking is a well-established fisheries management technique that has been used in a variety of locations in Canada, particularly in Ontario (Powell and Carl 2004). It has been used to develop “put-and-take” sport-fisheries, to introduce new species to barren lakes, to supplement existing stocks, and to rehabilitate extirpated or reproductively suppressed fish stocks.

If stocking is determined to be appropriate, lake trout would not be transplanted into Kennady Lake until a self-sustaining population of round whitefish or other suitable prey species was established, which, in-turn, would require the re-establishment of the lower trophic levels. As such, monitoring of the lower trophic communities and forage fish populations in the refilled lake would be undertaken to determine if and/or when a restocking program should be undertaken. Kennady Lake would be stocked with lake trout from lakes within the Lake 410 or Kirk Lake watersheds to maximize the likelihood of transferring fish with similar genetic make-up to those currently residing in Kennady Lake. Stocking success is increased if the source population has genetic traits that have adapted it to habitat similar to habitat present in the lake to be stocked. Any stocking program proposed for Kennady Lake would require acceptance and input from local Aboriginal communities and from federal and territorial agencies.

Prediction of the Re-established Fish Community in Kennady Lake

It is expected that a fish community will become re-established in Kennady Lake. The physical habitats in the reconnected lake are expected to be similar to those that currently exist. Water quality will also be somewhat similar, with the exception of nutrient-related parameters. Density and biomass of zooplankton and benthic invertebrates are expected to increase in Kennady Lake.

The final fish community of Kennady Lake will likely once again be characterized by low species richness (less than 10 species), containing a small-bodied forage fish community (e.g., lake chub, slimy sculpin, ninespine stickleback) and large-bodied species, such as northern pike, Arctic grayling, burbot, round whitefish, lake trout, and possibly longnose sucker. Total lake standing stock and annual production may be increased over what currently exists in the lake.

Predicted Duration of the Recovery

Recovery of aquatic ecosystems after “press”[12] disturbances that last for some time is difficult to estimate (Niemi et al. 1990). It is clear from the available data that recovery times from “press” disturbances are longer than those from “pulse”[13] disturbances, and that the longest recovery times are typically associated with “press” disturbances that include long-term alterations of physical habitat (Niemi et al. 1990). Project activities at Kennady Lake constitute a “press” disturbance.

There are dependent and independent factors that can reduce the recovery time of fish communities following “press” disturbances (Niemi et al. 1990). The independent factors that can reduce recovery times of fish communities include the following:

- persistence of fish in the disturbed area;
- persistence of fish in refugia upstream or downstream of the disturbed area; and
- absence of barriers to fish movement.

Kennady Lake will have the benefit of having persistent fish populations in refugia upstream and downstream, and there will be no barriers to immigration of fish to the lake once dykes and fish exclusion measures are removed. Habitat

[12] “Press” disturbance is a disturbance of long duration and often involving changes in the watershed or stream channel (e.g., timber harvesting, mining, channelization, drought).

[13] “Pulse” disturbance is a disturbance of limited and easily definable duration, which has little effect on the surrounding watershed (e.g., chemical spills, floods).

enhancement features will also be constructed to replace habitat disturbed during Project operations. Potential increases in nutrient levels may also increase primary productivity within the lake. These attributes should minimize recovery times.

Dependent factors that can reduce recovery times of fish communities include the following (Niemi et al. 1990):

- colonizing fish species with quick generation times;
- fish species with resistant life stages;
- fish species with a high propensity to disperse; and
- influx of species with minimal competition and predation interactions.

While these dependent factors influence recovery times, there is little that can be done to manipulate these factors. They are inherent to the characteristics of the fish species located in the area.

Overall, it is the life history attributes of Arctic grayling, northern pike and burbot that will ultimately determine the duration of the primary recovery of the Kennady Lake aquatic ecosystem.

Burbot and Arctic grayling are expected to establish self-sustaining populations in the refilled Kennady Lake earlier than northern pike. The northern pike population in Kennady Lake is currently small, likely limited by the lack of suitable habitat (i.e., lack of aquatic plants for spawning and rearing). Re-establishment of a stable, self-sustaining northern pike population is expected to take a long time (i.e., approximately 50 to 60 years following the complete refilling of Kennady Lake). This is based on the following rationale:

- The time frame for recovery of the plankton communities and benthic invertebrate communities after refilling is complete is expected to be between 5 and 10 years.
- The time frame for recovery of the forage fish community after recovery of the lower trophic communities is expected to be an additional five years.
- Kennady Lake will have cool summer water temperatures (maximum of 17 degrees Celsius [°C]), and a short growing season (less than four months).

Although data are limited, recovery times range from 5 to greater than 52 years for fish communities recovering from “press” disturbances in studies compiled by Niemi et al. (1990). Most (greater than 70%) of these studies dealt with disturbances to streams, and few were located in cold climate zones. Streams can be expected to recover faster than lakes, because streams are more dynamic than lakes. Arctic systems usually recover slower than temperate or tropical systems, because of colder temperatures, shorter growing seasons, and low nutrient availability. Although nutrient availability may not be a limiting factor for the recovery of Kennady Lake, a longer recovery compared to temperate zone lakes remains likely due to Arctic climate-related factors. As a result, the estimated time to full recovery is expected to fall between 50 to 60 years following the complete refilling of Kennady Lake, or 60 to 76 years from the end of Project operations. The average life span of northern pike is 10 to 12 years in fast-growing southern Canadian populations and in slow-growing Arctic populations as high as 24 to 26 years (Scott and Crossman 1973). Maximum age for burbot is probably between 10 and 15 years (Scott and Crossman 1973). Allowing fifteen years for development of the supporting food web, the estimate of 50 to 60 years is expected to allow for the completion of two life cycles of these slower growing predators.

It is expected that lake trout will require the longest time to re-establish a stable, self-sustaining population. Lake trout are slow-growing, have the longest time to first maturity (eight to nine years) of any fish in the area, and typically spawn only once every two years. Restocking may also be required to supplement the natural migration process. Lake trout may not return as the most abundant predatory fish in the lake, due to overwintering and spawning habitat limitations from reduced under-ice dissolved oxygen levels; habitat conditions may also be more favourable to other predatory fish species, such as northern pike and burbot. The re-establishment of a stable, self-sustaining lake trout population is anticipated to take approximately 60 to 75 years following the complete refilling of Kennady Lake.

8.11.1.4 Summary

An aquatic ecosystem will develop within Kennady Lake after refilling and reconnection of its basins. There will be some permanent losses of habitat in Kennady Lake due to mine rock piles, PK storage and mine pits; however, compensation habitats will be constructed within the Kennady Lake watershed to offset losses. The long-term hydrology of Kennady Lake is expected to return to a state similar to current conditions and water quality in the refilled lake is expected to return to conditions suitable to support aquatic life over time. The physical and chemical environment in Kennady Lake, therefore, will be in a state that will allow re-establishment of an aquatic ecosystem. It is expected that the fish species assemblage (i.e., fish species present) within Kennady Lake will be similar to pre-development conditions, but that due to biotic and abiotic factors, the community structure (i.e., relative abundances of the species) may differ. The overall fish production in Kennady Lake is expected to be higher than under existing conditions due to the expected long-term increase in nutrients resulting from the Project.

The expected time frame for recovery of the phytoplankton community is estimated to be approximately five years after refilling is complete, taking into consideration that the phytoplankton community will begin to develop during the eight year refilling period. The increase in nutrient levels in the refilled Kennady Lake will also facilitate community development and will result in a more productive phytoplankton community in the refilled lake compared to the pre-development community.

Zooplankton community development is predicted to follow recovery of the phytoplankton community. Colonization sources will be the same as those for phytoplankton and include the upstream watershed (i.e., the B, D, and E watersheds), Lake N11, and the WMP. The zooplankton community of the refilled lake is also expected to be more productive than the existing community. The expected time frame for the development of the zooplankton community is longer than that of phytoplankton (i.e., likely within five to 10 years of Kennady Lake being completely refilled).

Recovery of the benthic invertebrate community is expected to be slower than that of the plankton communities. The estimated time to recovery for the benthic community in Kennady Lake is about 10 years after refilling is complete. At the end of the recovery period, the benthic invertebrate community in Kennady Lake will be different from the community that currently exists in Kennady Lake and in surrounding lakes. The community will likely be of higher abundance and biomass, reflecting the more productive nature of the lake, and will likely be dominated by midges and aquatic worms.

The re-establishment of the fish community within Kennady Lake, and the speed at which it will occur, will depend on the ability of fish to re-colonize the refilled lake, the habitat conditions within the lake, and how succession takes place within the refilled system after it has been fully connected to the surrounding environment. It is expected that a fish community will become re-established in Kennady Lake. However, due to changes in trophic status and associated habitat conditions, the fish community structure may be different than what exists currently. For example, there will likely be increased productivity in the small-bodied forage fish community. Due to the increased food base, there may also be increased growth and production in large-bodied fish species. Mesotrophic conditions are likely to be more favourable to northern pike, burbot and Arctic grayling, than cold-water species, such as lake trout and round whitefish. As such, the relative abundances of the large-bodied fish species are likely to change from baseline conditions. Lake trout may not return as the most abundant predatory fish in the refilled Kennady Lake due to overwintering and spawning habitat limitations from reduced under-ice dissolved oxygen levels; northern pike or burbot may become the top predatory species in the lake.

Fish populations, including Arctic grayling, northern pike, burbot, lake chub, slimy sculpin, and ninespine stickleback, are expected to persist in the B, D, and E watersheds during Project operations. These watersheds are likely to be the primary source of initial migrants into Areas 3 to 7 of Kennady Lake. During refilling, exclusion measures will be used to limit the initial migration of large-bodied fish, such as northern pike, burbot, lake trout, and Arctic grayling, from entering the lake. It is anticipated that during the initial period of refilling, some mortality of the incoming small-bodied fish is likely to occur, because of insufficient water depths and possibly elevated levels of turbidity. As conditions improve, and water depths increase, the early migrants will become permanently established, feeding on the plankton and benthic invertebrate communities that are themselves becoming established in the refilled lake. Nutrient levels in the refilled Kennady Lake are predicted to be higher than under existing conditions, which may result in increased primary productivity and increased growth and production of these small-bodied forage fish species.

Following the removal of dyke A, migrant fish will also enter Areas 3 through 7 of Kennady Lake from Area 8, which is expected to contain residual populations of lake chub, slimy sculpin, ninespine stickleback, Arctic grayling, northern pike, and burbot. The migration and establishment of fish from Area 8 into the rest of Kennady Lake is expected to be rapid, due to proximity and the increased productivity that is expected from increased nutrient levels in Kennady Lake and Area 8.

The final fish community of Kennady Lake will likely once again be characterized by low species richness (less than 10 species), containing a small-bodied forage fish community (e.g., lake chub, slimy sculpin, ninespine stickleback) and large-bodied species, such as northern pike, Arctic grayling, burbot, round whitefish, lake trout, and possibly longnose sucker. Total lake standing stock and annual production will be increased over what currently exists in the lake.

8.12 RELATED EFFECTS TO WILDLIFE AND HUMAN USE

8.12.1 Overview

This section presents a summary of the effects of changes to water quantity, water quality, and fish in the Kennady Lake watershed on wildlife and human health. The summary of residual effects is based on assessments presented in other sections of the environmental impact statement (EIS). The assessment of effects to wildlife for all pathways, including changes in water quantity, water quality, and fish are provided in the following other sections of the EIS:

- Key Line of Inquiry: Caribou (Section 7);
- Subject of Note: Carnivore Mortality (Section 11.10);
- Subject of Note: Other Ungulates (Section 11.11); and
- Subject of Note: Species at Risk and Birds (Section 11.12).

Potential pathways for effects to wildlife associated with changes in water quality, water quantity, and fish in the Kennady Lake watershed include:

- effects to wildlife health resulting from changes in water quality and fish tissue quality;
- effects of dewatering Areas 2 to 7 of Kennady Lake on riparian vegetation and related effects to wildlife; and
- effects to wildlife resulting from a decrease in lake area resulting from the dewatering of Areas 2 to 7 of Kennady Lake, isolation of Area 8, and changes to small lakes in the watershed.

The only potential pathway for effects to human health relevant to Section 8 is associated with changes in water quality and fish tissue quality.

A summary of the residual effects for each of these pathways is provided below.

8.12.2 Summary of Residual Effects

8.12.2.1 Wildlife

8.12.2.1.1 Effects of Changes in Water Quality and Fish Tissue Quality to Wildlife Health

An ecological risk assessment was completed to evaluate the potential for adverse effects to individual animal health associated with exposure to materials released from the Project. The result of the assessment indicated the potential for effects to occur to aquatic-dependant birds (i.e., waterfowl and shorebirds) as a result of boron levels in Kennady Lake after refilling. No other impacts were predicted to birds or other wildlife, including caribou, muskoxen and moose.

The ecological risk assessment was completed using water quality predications that were developed assuming that there was no isolation of the fine PKC material located at the base of the Fine PKC Facility, and that all waters travelling over the facility would come into contact with this material, which is the predominate source of boron to the refilled lake. Processes that would modify the degree of contact between the fine PK and the runoff waters were not considered, including the aggradation of permafrost and/or the application of cover material to limit infiltration. In addition, the water quality predications used in the risk assessment were developed by setting parameters concentrations in the runoff waters to the maximum concentrations observed in the geochemical investigations completed in support of the EIS. Consequently, the results of the risk assessment correspond to an extreme condition that has a low likelihood of occurring.

De Beers is committed to further study of this potential issue in 2011, and will incorporate mitigative strategies into the Project design to the extent required to maintain boron levels in Kennady Lake below those that may be of environmental concern, including the potential application of less permeable cover material to limit infiltration through the Fine PKC Facility. Given these commitments and the low likelihood of the assessed situation actually occurring, overall potential effects to wildlife were deemed to be environmentally insignificant. However, the predictions of environmental significance with respect to water birds are dependent on the execution of further study of the ingestion pathways discussed in Section 11.2 and the commitment that mitigative strategies will be incorporated into the Project design to the extent required to invalidate these pathways.

8.12.2.1.2 Effects of Dewatering Areas 2 to 7 on Riparian Vegetation and Related Effects to Wildlife Habitat

Riparian vegetation around the edges of Kennady Lake is currently limited, and primarily restricted to sheltered bays and streams. Most of this riparian vegetation will be lost when Areas 2 to 7 are dewatered. However, dewatering of Kennady Lake will result in the exposure of a portion of the lake-bed. There is the potential for vegetation to establish on the exposed lake-bed sediments. This type of habitat would likely be favoured by grasses, sedges, and possibly, invasive weedy species, and would create habitat for wildlife. In Area 8, higher concentrations of nutrients may increase riparian habitat.

Changes in abundance and composition of vegetation associated with dewatering will be localized, and will have a minor influence on the quantity of forage available for wildlife, relative to baseline conditions. Consequently, changes to forage quantity, resulting from the dewatering of Areas 2 to 7, are expected to be a minor pathway that would not contribute to effects to wildlife.

8.12.2.1.3 Effects of a Decrease in Open Water Area to Wildlife Habitat

During operations, a reduction in the surface area of open water in the Kennady Lake watershed will result primarily from the dewatering of Areas 2 to 7, and to a lesser extent from the loss of small lakes. This will be partially offset by the raising of lakes in the A, D and E watersheds, and resultant increase in the surface area of lakes A3 (22.8 hectares [ha]), D2 and D3 (53.1 ha combined), and E1 (6.8 ha).

After closure, once Kennady Lake has been refilled and the lakes in the D and E watersheds have returned to their pre-disturbance levels, a reduction in the surface area will persist due to the loss of Kennady Lake area through the development of the West and South Mine Rock Piles and Fine Processed Kimberlite Containment (PKC) Facility, and several small lakes (e.g., Lakes A1 and A2). This will be offset to a small degree due to the permanent raising of Lake A3.

The overall decrease in the surface area of open water in the Kennady Lake watershed through operations and closure will primarily affect habitat for water birds (e.g., waterfowl, loons, and grebes) and shore birds whose important habitats include vegetation communities with a wetter moisture regime including shallow and deep water, sedge wetlands, and riparian habitats. Approximately 68% of the Project footprint is aquatic habitat and 32% is terrestrial habitat. The baseline Local Study Area (LSA) is approximately 200 square kilometres (km^2) (Section 11.12.2.1, Figure 11.12-2), centered on Kennady Lake; it was selected to assess direct effects (e.g., habitat loss) to individuals from the Project footprint. The baseline Regional Study Area (RSA) is much larger at 5,700 km^2 (Section 11.12.2.1, Figure 11.12-2); it was selected to capture indirect effects of the Project.

At the local scale, the Project footprint will alter 4.4% of the wildlife baseline LSA. Direct effects from the Project footprint are expected to decrease the surface area of open water in the wildlife baseline LSA by 2.2%. However, there will be a less than 1% decrease in sedge wetland and riparian shrub relative to baseline values in the wildlife LSA. These local changes are expected to influence individuals that occupy or travel through habitats within and adjacent to the Project.

At the population level, the Project is expected to affect less than 1.4% of highly suitable habitat for water birds and shore birds (i.e., deep water, shallow water, sedge wetland, riparian habitat) in the wildlife baseline RSA. The greatest reduction in highly suitable habitat is to deep water (446 ha). The magnitude of the incremental decrease in habitat quantity caused by the footprint was predicted to be low. A less than 1.4% loss of habitat is well below the 40% threshold value for habitat loss associated with predicted declines in bird and mammal species (Andrén 1994, 1999; Fahrig 1997; Mönkkönen and Reunanen 1999). Therefore, the direct effect of the Project on the population size and distribution of water birds and shore birds is predicted to be low in magnitude.

8.12.2.2 Human

8.12.2.2.1 Effects of Changes in Water Quality and Fish Tissue Quality to Human Health

A human health risk assessment was completed to evaluate how the predicted changes to air and water quality in the Kennady Lake watershed could potentially affect human health. Emission sources considered in the assessment included fugitive dust, air emissions, site runoff and seepage and exposed lakebed sediments. Potential exposure pathways included changes in air, water, soil, vegetation and fish tissue quality.

The results of the assessment indicate that individuals living at the Project site could experience health issues should they consume fish, as predicted changes in metal levels in water could affect fish tissue quality. However, individuals working at the Project site will not be allowed to fish and, therefore, will not consume fish from the Kennady Lake watershed. In addition, individuals do not currently live at the Project site, and it is unlikely that non-workers would do so in the future. This exposure scenario was used to provide a conservative evaluation of potential effects to individuals using the area for traditional purposes, because traditional purposes typically involve a temporary presence on the land near the Project site. The human health assessment was also completed using the conservative water quality predictions described herein, which included the free and complete contact between site runoff waters and the materials contained in the mine rock piles, the Coarse PK Pile and the Fine PKC Facility.

De Beers is currently evaluating a variety of environmental design features and mitigation measures to limit contact between site runoff waters and the fine PK located within the Fine PKC Facility and other potential sources. The effectiveness of these environmental design features and mitigation measures is uncertain and requires further analysis. This analysis is expected to be completed in 2011. Once complete, De Beers will update the human health assessment to reflect the effects of these measures. De Beers is also committed to implementing additional environmental design features and mitigation measures to the extend required to protect human health.

As a result, human health is not expected to be detrimentally affected by Project activities, in the Kennady Lake watershed or in downstream systems. However, this statement is contingent on the results of further study and the implementation of mitigation strategies to the extent required to maintain exposure levels below those that would be of concern.

8.13 RESIDUAL EFFECTS SUMMARY

The potential environmental effects related to the valid pathways identified for water quality and fish in Kennady Lake is provided below for the following components:

- hydrology;
- water quality;
- aquatic health;
- fish and fish habitat; and
- recovery

8.13.1 Hydrology

8.13.1.1 Construction and Operations

The dewatering of Kennady Lake Areas 2 through 7 will begin after the construction of Dyke A. All discharge during construction will be by direct discharge to Lake N11 and Kennady Lake Area 8. A Water Management Pond (WMP) will be established in the dewatered Areas 3 and 5. Plant makeup water will be withdrawn from the WMP and fresh water supply for potable water (60,000 cubic metres per year [m^3/y] during construction and 28,000 m^3/y during operation) will be withdrawn from Area 8. It is expected during construction and operation that the dewatering process will not result in effects to natural channel or bank stability; however, the exposed lake-bed within the dewatered Kennady Lake may be subject to erosion, depending on the bed substrate. All runoff and water retained in collection ponds within the Kennady Lake watershed will be managed to prevent its release to the natural receiving environment if it does not meet specific water quality guideline criteria.

The diversion of drainage from watersheds A, B, D and E away from Kennady Lake will be achieved with the construction of saddle dykes on the outlets of lakes A3, B1, D2, and E1. The dykes will raise water level in lakes A3 (3.5 metres [m]), D2 (2.8 m), D3 (1.6 m) and E1 (0.8 m) and block the existing outlet of Lake B1 with no change in water levels, and cause the cessation of flows downstream of the dykes for most of the year. The lake surface areas will increase by 96% in Lake A3 (to 0.47 square kilometres [km^2]), 102% in lakes D2 and D3 (to 1.03 km^2) and 33% in Lake E1 (to 0.27 km^2); however, the mean annual water level variation is expected to be similar or reduced from pre-diversion conditions. The increase in water levels will result in the inundation of lake shoreline zones, but because of the natural armouring afforded by cobble and boulder substrate, and the preparation of the shoreline zone to be inundated if required, erosion potential and sediment sourcing will be minimized. The diversion channels constructed in watersheds A, B, D and E to convey the reversed flows to the N watershed will have similar annual hydrograph characteristics to the natural lake outflows, and will also be designed to prevent erosion and maintain stability in permafrost. Raised lake filling in the D watershed is expected to take 3 years, and in the A watershed is expected to take 11 years.

Project activities in the Kennady Lake watershed will include the development of project surface infrastructure (camp and plant site, processing facilities, sewage treatment plant, explosives management facilities, airstrip and site roads) as well as the West and South Mine Rock Piles, Coarse PK pile and Fine PKC Facility.

Watersheds A, Ka, Kb and Kd are tributaries to Areas 2 to 7 that include project surface infrastructure. All runoff from these watersheds will be conveyed to the WMP by the site water management system (e.g., the Project mechanism to which all elements of site contact and mine contact water, potable and plant water supply, pumped inflows and discharges, and natural inflows and outflows are managed and facilitated). Watersheds H, I and Ke are tributaries to Area 8 that include project surface infrastructure. All infrastructure within these watersheds will be free-draining and no measurable effect on the quantity of inflow to Area 8 of Kennady Lake is anticipated. Project surface infrastructure is not expected to have any measurable effect on natural channel or bank stability, because no natural lakes will be affected, and constructed ditches will incorporate erosion and sediment control measures.

Mine rock piles will be located entirely within the controlled area boundary and all drainage will be managed as part of the closed-circuit site water management system. Lake Ka1 will be covered by the West Mine Rock Pile and a portion of the tributary area to the Lake F1 outlet channel, downstream of the lake, will be occupied by the South Mine Rock Pile. No effects on natural channel or bank stability are anticipated, because runoff around the mine rock pile perimeters and in the diverted Lake F1 will be managed to prevent channel erosion.

The Coarse PK Pile will be located entirely within the controlled area boundary and all drainage will be managed as part of the closed-circuit site water management system. Construction and operation of the Coarse PK Pile will result in the permanent loss of Lake Kb4 as a waterbody. No effects on natural channel or bank stability are anticipated, because runoff from the Coarse PK Pile and upstream areas will be managed with internal and perimeter ditches to prevent channel erosion.

The Fine PKC Facility will be located entirely within the controlled area boundary and all drainage will be managed as part of the closed-circuit site water management system. Construction and operation of the Fine PKC Facility will result in the permanent loss of the northern portion of Kennady Lake (Area 2) as a waterbody, as well as lakes A1, A2, A5 and A7 and their outlet channels. No effects on natural channel or bank stability are anticipated, because runoff from the Fine PKC Facility and upstream areas will be managed with internal and perimeter ditches to prevent channel erosion.

The effects of altered discharge regimes to the water level in Area 8 will vary depending on the Project phase. During construction, dewatering flows through Area 8 from Area 7 will be generally increased from baseline conditions, with the duration of the flood flow extended through to September (with residual flows in extending into October). However, flows will be limited so that dewatering discharge will not exceed the 1:2 year flood discharge volume. During operation, flows through Area 8 will be generally decreased from baseline conditions, due to the closed-circuiting of the watershed upstream of Dyke A. The alterations in water levels in Area 8 will correspond with the flow changes, with the largest changes for open water discharge expected to occur in September for dewatering (+0.190 m) and in July during operation (-0.158 m). Because water levels in Area 8 and corresponding discharges in its outlet (Stream K5) will be managed not to exceed 1:2 year flood flows, no adverse effects to channels or bank stability are anticipated.

8.13.1.2 Closure

At closure, the dykes on the isolated and diverted upper watersheds (B, D and E) will be removed and surface flows restored to Kennady Lake. The diversion of Lake A3 to Lake N9 will be permanent. The restored watersheds will then contribute to the natural refilling of Kennady Lake, which will be supplemented by the diversion of water from Lake N11. The Kennady Lake refill time is expected to take approximately 8 to 9 years. Water levels in Kennady Lake will rise during refilling as a function of cumulative inflow less lake evaporation. No effects on channel or bank stability are expected during refilling, and erosion will be prevented at the refilling discharge points by armouring of the outfalls in Area 3, and the use of diffusers at the discharge points. No water from the refilled Areas 3 through 7 will be released to Area 8 until the water level is at the naturally armored shoreline elevation and water quality meets specific criteria.

During the refilling of Kennady Lake, flows at the Area 8 outlet will be reduced due to the removal of 77% of the upper natural drainage area (i.e., Areas 3 through 7 and their associated upper watersheds). The mean monthly water levels in Area 8 during refilling for June to October are expected to be 0.10 m to

0.16 m lower than median baseline flow conditions. No erosion effects are expected under these flow scenarios because flow levels will be below baseline values.

At closure, the Kennady Lake watershed will have been altered as a result of the Project development. That is, the watershed area will decrease by 2.6% (32.46 km^2 to 31.62 km^2), due to the permanent diversion of the A3 watershed, with a net decrease in lake surface area, including Kennady Lake tributaries, of 14.1% (11.29 km^2 to 9.70 km^2). The reduction in lake surface area will correspond to a decrease in lake proportion of the watershed from 34.8% to 30.7%. As a consequence, the water balance will change for the Kennady Lake watershed resulting in the increase of mean annual water yield by 8.9%. The reduction in the surface area of Kennady Lake of 14.1% (8.15 km^2 to 7.19 km^2) means that flood peak discharges will increase post-closure due to less storage in the lake.

8.13.2 Water Quality

8.13.2.1 Construction and Operation

Residual effects to water quality during construction and operations include effects of the deposition of dust and metals from air emissions and acidifying air emissions to waterbodies within the Kennady Lake watershed. Several pathways associated with deposition of Project air emissions in Area 8 and smaller waterbodies in the Kennady Lake watershed were assessed:

- deposition of total suspended particulates;
- deposition of trace metals; and
- deposition of acidifying emissions.

Effects of Deposition of Dust and Metals from the Project

The effects of dust and associated metal deposition on water quality were evaluated for 19 lakes within the Kennady Lake watershed, of which 12 are fish bearing. Changes to total suspended solids and trace metals concentrations in these lakes from deposition of total suspended particulates and metals will potentially exceed average baseline concentrations by greater than 100%. However, the spatial extent of dust and metal deposition is anticipated to be restricted to localized areas within and close to the active mine area. Maximum deposition is expected to occur near haul roads along the southern, western and eastern boundary of the development area, and primarily reflect winter fugitive road dust emissions. In general, no concentration of total suspended particulates (TSP) above the NWT air quality standard is predicted beyond approximately

2 km from the development area boundary (Section 11.4, Subject of Note: Air Quality).

Based on annual cumulative loading of TSS and metals, predicted maximum concentrations of aluminum, cadmium, chromium, copper, iron, mercury, and silver are anticipated to be above water quality guidelines in two or more lakes adjacent to the Project area during construction and operations. However, the estimated maximum changes in TSS and metal concentrations in lakes within the Kennady Lake watershed are based on air quality modelling results representing peak production periods during mine operation (i.e., Years 1 and 5). As a result, the level of conservatism built into the air quality assessment means that predictions of TSS and metal concentrations are likely to be higher than can be realistically expected.

The period of elevated TSS and metal concentrations in affected lakes is expected to be relatively short. During construction and operations, the largest load of suspended sediments to surface waters during the year will occur during spring freshet, when dust deposited to snow during winter and eroded materials enter surface waters. During the freshet period, elevated TSS and metals concentrations are naturally elevated above average baseline conditions due to the peak watershed runoff through the lakes. Sediment inputs during other times of the year are anticipated to be sporadic and too small to result in measurable changes in TSS and metal concentrations in lakes, except in localized areas near stream mouths during and immediately after precipitation events.

The length of the freshet period is estimated to range from approximately two days for small lakes to a maximum of one to two weeks based on the length of the freshet for Kennady Lake. This would be followed by a period of settling, estimated as less than a month based on observations at Snap Lake (De Beers 2010), by which time TSS concentrations in lake water are expected to be similar to background concentrations. Therefore, the effects on TSS and metal concentrations are expected to be localized in the immediate vicinity of the Project and temporally restricted to the period during and after freshet.

Effects of Acidifying Emissions from the Project

Predicted net PAI values representing peak emissions during construction and operations are below the critical loads for the 19 lakes included in the evaluation of Project-related effects. The annual deposition of nitrogen during construction and operations was less than 5 kilograms per hectare per year (kg/ha/y) for all lakes. Based on these results, Project-related deposition of sulphate (SO_4) and nitrate (NO_3) in the Kennady Lake watershed is not predicted to result in lake acidification.

8.13.2.2 Closure

Water quality in Kennady Lake is projected to vary over time as the lake is refilled after closure. At the end of the Project operations, Kennady Lake will be filled as quickly as possible (approximately 8 to 9 years) by restoring the upper diverted watersheds and augmenting natural watershed inflows with additional inflow from Lake N11.

To estimate the water quality in Kennady Lake and Area 8 through the closure phase, a dynamic, mass-balance water quality model was developed in GoldSim™. For this assessment, 1:2 year (median) wet conditions were assumed, which represents a close to average climate scenario, which is appropriate for assessing long-term water quality conditions in a lake environment.

Modelling of water quality in Kennady Lake during and after refilling was evaluated in two periods of time in the closure phase, which follow on from construction and operation:

- Closure – the Tuzo Pit will be filled and once Tuzo Pit is full, the dewatered Areas 3 to 7 will be refilled; and
- Post-closure – when Areas 3 to 7 are filled to the same elevation as Area 8, and water quality is acceptable, dyke A will be removed and the refilling of Kennady Lake will be completed and flow will occur between Areas 3 through 7 and Area 8.

The focus of this residual effects summary is on water quality in the refilled Kennady Lake and Area 8 after the removal of dyke A (post-closure) because this is when Kennady Lake will be physically restored and recovery of the aquatic ecosystem can begin.

After refilling, Tuzo and Hearne pits represent new waterbody features within the restored Kennady Lake. The bottom of Tuzo pit will be about 295 metres (m) below the surface of Kennady Lake, and Hearne Pit will be approximately 120 m deep, creating deep depressions within the lake. During and after refilling of Tuzo pit, saline groundwater inflow will collect in the bottom of the pit forming a higher density, more saline (TDS concentration of up to 400 mg/L) layer, which is referred to as a monimolimnion layer. The monimolimnion layer will be separated from the overlying freshwater layer in what is referred to as meromictic conditions. A long-term analysis evaluated the stability of meromictic conditions for 15,000 years, and concluded that the saline bottom layer will remain stable and will not overturn. The water quality in Kennady Lake above Tuzo Pit will, therefore, will be primarily determined by the upper 20 m of fresh water, which will be subject to temperature and wind-driven summer seasonal stratification.

Hearne pit will be partially backfilled with fine PK and process water, but will not be initially filled with saline water as will occur for Tuzo pit. Therefore, meromixis is assumed not to occur in Hearne Pit, and water in this pit will be fully mixed with water in Area 6. This assumption is a conservative prediction, because if meromixis does occur in Hearne Pit, the deeper water in contact with the fine PK will be isolated and the input of the diffusive flux of metals and nutrients from the bottom of Hearne pit to the water quality in Area 6 will be unlikely.

Water quality in Kennady Lake during refilling will be influenced by the following sources:

- natural watershed runoff with a background surface water quality;
- supplemental water pumped from Lake N11 with a background water quality;
- seepage from the Fine PKC Facility, and contact water from the Coarse PK Pile and the mine rock piles, and minor contribution from site runoff;
- contact water from the exposed pit walls during refilling of the Tuzo pit basin and Hearne pit; and
- diffusion from fine PK in the bottom of Hearne Pit.

After refilling is complete and the lake is restored to pre-mine levels, water quality in Kennady Lake will be influenced by:

- natural watershed runoff with a background surface water quality; and
- seepage from the Fine PKC Facility, and contact water from the Coarse PK Pile and the mine rock piles, and minor contribution from site runoff;
- contact water from the exposed pit walls during refilling of the Tuzo Pit basin and Hearne; and
- diffusion from PK material in the bottom of Hearne Pit.

Water quality in the surface water of Tuzo Pit will be influenced by water pumped from Lake N11 and contact water from the exposed pit walls. As water levels in the pit rise and exposed pit walls are flooded, mass loading from contact water will decrease.

During refilling of the Tuzo Pit, natural runoff and contact water will collect in the WMP and low points in the dewatered lake-bed of Areas 3 through 7. Once the Tuzo Pit is filled, the refilling of Areas 3 through 7 will begin and the Tuzo Pit basin becomes part of Area 4 and the Hearne pit will be part of Area 6.

Contact water from the exposed pit walls in the Tuzo Pit basin and Hearne pit will be negligible after flooding. Seepage from the Fine PKC Facility and contact water from mine rock piles are the primary source of mass loadings that will affect water quality in Kennady Lake during and after refilling. Water quality parameters in runoff that comes into contact with the mine rock and PK material will include TDS and major ions, metals and nutrients (e.g., phosphorus, nitrate, and ammonia).

Water quality in Area 8 will remain similar to background conditions during the refill period, before the removal of dyke A, because this Area will remain isolated from Kennady Lake. Water quality in Area 8 during the post-closure phase will be driven by the water flowing from Kennady Lake after Dyke A is breached, with additional dilution from the Area 8 sub-watershed.

Concentrations of all modelled constituents are predicted to increase when Dyke A is breached. In nearly all cases, concentrations are predicted to peak within five years of Dyke A being breached, as water in Area 8 is replaced with water from the refilled Kennady Lake. Concentrations are generally predicted to decline with time. In a few cases, concentrations are predicted to increase during post-closure and reach a long-term steady state concentration within a few decades.

8.13.2.2.1 Total Suspended Solids

There will be no influx of TSS above background concentrations to the refilling Tuzo Pit basin and Areas 3 through 7. Natural drainage from the restored upper watersheds and supplemental water pumped from Lake N11 will not be a source of additional TSS, with concentrations consistent with background water quality.

8.13.2.2.2 Total Dissolved Solids and Major Ions

Concentrations of TDS and major ions in Areas 3 to 7 are projected to increase during the operations phase (approximately 400 milligrams per litre [mg/L]), primarily due to saline groundwater discharged from the mining pits to the WMP. During the closure phase, TDS concentrations are predicted to decrease as higher TDS water is drained from the lake to Tuzo Pit and fresh water is imported from Lake N11 (approximately 150 mg/L). The main constituents of TDS during the two periods include calcium and chloride. This major ion dominance is consistent with the composition in background water quality.

In post-closure, concentrations are predicted to continue to decline as Kennady Lake receives fresh water inflows (i.e., natural drainage) from the basin and Dyke A is breached. In one to two decades of post-closure, concentrations are predicted to approach steady state at slightly less than 100 mg/L TDS. Calcium, chloride, magnesium and sodium are predicted to mirror the trends displayed by TDS.

The long-term results presented for post-closure reflect a reasonable degree of conservatism. Concentrations of TDS and major ions are predicted to remain elevated above background levels because loading of these constituents from the Fine PKC Facility, contact with mine rock and diffusion from PK material in the bottom of Hearne Pit are assumed to continue in the long-term. The loading of TDS from this facility to Kennady Lake is expected to reduce with the establishment of permafrost through the fine PK material.

Concentrations of TDS and major ions in Area 8 are predicted to follow the general trends described for Kennady Lake. All major ions follow this trend, except potassium and sulphate, which are predicted to increase following closure.

There are no CCME guidelines for TDS or any of the major ions. To put the predicted concentrations into context, TDS and all major ions are predicted to remain above background conditions but below levels that would affect aquatic health.

8.13.2.2.3 Nutrients

During the refilling of Tuzo Pit, ammonia and nitrate concentrations are projected to generally increase, primarily due to inputs from blasting residue. These are expected to decrease during the closure phase as higher concentration water is transferred to Tuzo Pit and fresh water is imported from Lake N11. By the time Dyke A is removed, modelled nitrogen and ammonia concentrations are expected to be at, or below, water quality guidelines and decline thereafter to near background levels. In Area 8, all forms of nitrogen are expected to peak in concentration in Area 8 within five years of breaching Dyke A, then return to near-background concentrations.

Concentrations of phosphorus are projected to increase in Areas 3 to 7 of Kennady Lake during the operations phase due to loading to the WMP from process water, runoff and seepage inputs from Project facilities, and groundwater inputs. Concentrations are then projected to decrease during the closure phase due to the refilling of Kennady Lake, and then gradually increase to steady state concentrations during post-closure due to seepage from materials located in the mine rock piles, Coarse PK Pile and the Fine PKC Facility. The Fine PKC facility is the largest contributing source of phosphorus. Using a combination of mitigation strategies, De Beers is committed to incorporating additional mitigation to achieve a long-term maximum steady state total phosphorus concentration of 0.018 mg/L in Kennady Lake.

Based on the long-term steady state total phosphorus concentration projected in Kennady Lake for post-closure, the trophic status in Kennady Lake may shift from oligotrophic to mesotrophic. With this change in trophic status, increased growth of phytoplankton and algae within the lake is anticipated, which will result in a larger amount of total organic carbon remaining in the lake after senescence each fall. The decomposition of this organic carbon will exert an oxygen demand over winter after the ice has formed and atmospheric re-aeration has been cut off. The winter oxygen depletion rate for three depth zones in Kennady Lake and a dissolved oxygen balance for Kennady Lake at the end of winter were estimated. The results indicate that the surface zone of the water column (i.e., under ice to 6 m) should remain oxygenated over the winter period to concentrations that will likely remain in excess of the lower dissolved oxygen concentration guideline of 6.5 for cold-water fish. The mid-depth and bottom depth zones will likely be subject to lower oxygen levels, which may fall below guideline concentrations. The deeper epilimnion regions of the open Tuzo and Hearne pits are not expected to be subject to the same winter oxygen demand as other shallower areas of Kennady Lake and are expected to remain well oxygenated. Under open-water conditions, Kennady Lake is expected to remain well mixed and near, or at, saturation with respect to dissolved oxygen (similar to existing conditions).

8.13.2.2.4 Trace Metals

Of the 23 trace metals that were modelled for this assessment, chromium, cobalt, iron, lead, manganese, mercury, selenium, silver, thallium, uranium and zinc are predicted to increase in concentration during the operations phase, then steadily decline in concentration as the lake is flushed during post-closure. With the exception of thallium, the primary loading source of these metals to Kennady Lake is groundwater from the active mine pits, hence the decline once pit dewatering is finished. Thallium has two primary loading sources, namely, groundwater and mine rock runoff. Because the concentrations of these metals will be mainly groundwater-driven, the dissolved fraction of these metals is predicted to comprise the majority of the total concentrations. Chromium and iron are projected to exceed water quality guidelines in the post-closure phase

Aluminum, antimony, arsenic, cadmium, copper, nickel and vanadium will be influenced by a combination of sources throughout the operations phase. These metals are predicted to increase mainly due to inputs from groundwater and mine rock runoff, with secondary loading sources through runoff infiltration and contact with fine PK and process water. These metals are predicted to increase in concentration relatively steadily throughout the operations phase, rise or fall during closure, then remain fairly constant throughout post-closure. The lack of reduction in post-closure concentrations of these metals is due to the geochemical loading through runoff contact that will occur from the remaining mine rock and fine PK in and near Kennady Lake. Because the primary loading sources of these metals is groundwater and geochemical flux, the majority of these metals will be in the dissolved form. Cadmium and copper are projected to exceed water quality guidelines in post-closure.

Barium, beryllium, boron, molybdenum and strontium are predicted to increase in post-closure. Concentrations of these metals will mainly be driven by loadings through runoff infiltration and contact with mine rock, coarse PK and fine PK. Because these storage facilities will be present in post-closure, concentrations of these metals are predicted to increase after closure, reach steady state conditions in Kennady Lake within about 40 years. Because geochemical sources are the primary contributors of these metals, the majority of total concentrations will be in the dissolved form. None of these five metals are projected to exceed water quality guidelines in the post-closure phase.

Concentrations of trace metals in Area 8 are predicted to follow the general trends described above for Kennady Lake. After the initial period of approximately five years to approach Kennady Lake concentrations, trace metal concentrations are then predicted to decrease, remain relatively constant or decrease. Of the 23 modelled trace metals, cadmium, chromium and copper are projected to exceed water quality guidelines in post-closure.

Comparison to water quality guidelines is provided for reference only. Effects of trace metal concentrations on human health have been evaluated and are discussed in Section 8.12. Effects of trace metal concentrations on the health of aquatic life are summarized in the following residual effects summary for Aquatic Health.

8.13.3 Aquatic Health

Potential effects to aquatic health could occur as a result of changes to water quality and/or the deposition of dust and metals. Aerial deposition of sulphate and nitrate could also lead to changes in aquatic health through acidification of waterbodies. However, Project-related deposition of sulphate and nitrate is not predicted to result in acidification in the Kennady Lake watershed.

During construction and operation, predicted maximum concentrations of suspended solids and some metals may increase above water quality guidelines because of dust and metal deposition in some lakes, some of which are fish-bearing lakes. However, the predicted concentrations were derived using very conservative assumptions, and hence are likely conservative estimates of the maximum potential concentrations. Most of the deposition will impact the affected lakes during the short period of freshet, when dust deposited to snow enters surface waters. The length of the freshet period is estimated to be relatively short; therefore, the period in which aquatic life will be exposed to the elevated suspended solids and metals concentrations will be short. Given the conservatism in the predicted concentrations, and the length of the exposure to elevated concentrations, the potential for adverse effects from dust and metals deposition is considered to be low. Follow-up monitoring will be undertaken to confirm this evaluation.

At the end of operations, the Project is no longer a notable source of dust and metal deposition. Therefore, the incremental effect of the Project on metals in the affected lakes is anticipated to cease with a consequential return to existing (i.e., post-Project) conditions.

As a result of Project activities, changes to water quality in Kennady Lake and Area 8 during closure and post-closure are expected, that is, after refilling is complete in Kennady Lake and after breaching of Dyke A. The potential effect of these changes on aquatic health was evaluated by considering both direct waterborne exposure and accumulation within fish tissues.

In regards to direct waterborne exposure, predicted maximum concentrations for most substances of potential concern (SOPCs) were lower than the corresponding chronic effects benchmark (CEB), with the exception of total copper, iron, and strontium.

Despite the predicted exceedances of the CEB, the potential for copper to cause adverse effects to aquatic life in Kennady Lake and Area 8 is considered to be low. The CEB for copper is based on the CCME guideline, which is intended to be conservative and protective of the most sensitive species. Predicted copper concentrations are only slightly greater than the CEB, indicating the possibility (but not necessarily the likelihood) of effects to the most sensitive species. However, the CCME guideline does not consider the potential for other water quality characteristics (e.g., dissolved organic carbon) to reduce bioavailability and ameliorate copper toxicity. Furthermore, the CCME guideline is based on toxicity tests with native and sensitive organisms, whereas organisms inhabiting Kennady Lake are unlikely to be highly sensitive to copper, given that baseline sediment copper concentrations exceed the CCME interim sediment quality guideline. Given the small magnitude by which predicted maximum concentrations exceed the CEB, and given the potential for ameliorating factors discussed above, the potential for adverse effects from copper is considered to be low. Follow-up monitoring will be undertaken to confirm this evaluation.

The potential for iron to cause adverse effects to aquatic life in Kennady Lake is considered to be low. Maximum total and dissolved iron concentrations in Kennady Lake after refilling and Dyke A is removed are predicted to be slightly above the corresponding CEB. The CEB for iron is based on the CCME guideline, which is intended to be conservative and protective of the most sensitive species. Iron concentrations similar to the CEB have been reported by some authors to elicit sublethal effects on cladocerans (Dave 1984). However, other authors have reported effects thresholds for the same species more than an order of magnitude higher than the CEB (Biesinger and Christensen 1972). Lethal effects on cladocerans and effects on fish and other taxa have only been reported at much higher iron concentrations, greater than the CEB and greater than all predicted iron concentrations in Kennady Lake. Thus, the predicted iron concentrations are not expected to result in adverse effects to aquatic life. Follow-up monitoring will be undertaken to confirm this evaluation.

Strontium is conservatively projected to be higher than the CEB in Kennady Lake and Area 8 during closure and post-closure conditions. However, the CEB is highly conservative, and the actual likelihood of adverse effects to aquatic life is therefore highly uncertain. The CEB was based on a single study of rainbow trout embryos (Birge et al. 1979) that reported effects at strontium concentrations several orders of magnitude lower than any other study, including studies with rainbow trout and other fish species. Given the high level of uncertainty in the toxicity reported by Birge et al. (1979), and given that the maximum predicted strontium concentrations in Kennady Lake are orders of magnitude lower than all other effects concentrations in the toxicity dataset, the potential for adverse effects from strontium is considered likely to be low. Follow-up monitoring will be undertaken to confirm this evaluation.

Predicted fish tissue concentrations are below toxicological benchmarks for all substances considered in the assessment except silver. However, fish tissue silver concentrations are predicted to increase only marginally above baseline conditions as a result of the Project. Also, the selected silver tissue benchmark is based on a no-effect concentration, and thus is a highly conservative basis for assessing the potential for predicted silver concentrations to cause effects to fish. Given the modest predicted increase, and that both baseline and predicted tissue concentrations only marginally exceed the available no-effect concentration, the potential for predicted silver concentrations to cause effects to fish is concluded to be low.

Based on the above results, changes to concentrations of all substances considered in this assessment are predicted to result in negligible effects to aquatic health in Kennady Lake.

8.13.4 Fish and Fish Habitat

Effects to fish and fish habitat are predicted to occur in Kennady Lake and its watershed during mine construction and operations, and closure (including post-closure), as a result of physical changes to habitat, and changes to hydrology and water quality. Flow changes in the Area 8 outlet channel (Stream K5) affecting fish migration into and out of Area 8 are assessed in Section 9.10.

8.13.4.1 Construction and Operations

Changes to Fish Habitat from Project Footprint

Changes to fish habitat will occur in Kennady Lake and the Kennady Lake watershed due to the development of the Project. The affected habitat areas include portions of Kennady Lake and adjacent lakes within the Kennady Lake watershed that will be permanently lost, portions that will be physically altered after dewatering and later submerged in the refilled Kennady Lake, and portions that will be dewatered (or partially dewatered) but not otherwise physically altered before being submerged in the refilled Kennady Lake. The affected habitat areas were quantified in the Conceptual Compensation Plan (CCP) (Section 3, Appendix 3.II).

The permanently lost areas are those affected by the following:

- The Fine PKC Facility (Areas 1 and 2, Lake A1, Lake A2, Lake A5, Lake A6, Lake A7);
- The Coarse PK Pile (Area 4 and Lake Kb4);
- West Mine Rock Pile (Area 5 and Lake Ka1);
- South Mine Rock Pile (Area 6); and
- Dykes C, D, H, I and L.

The Project will result in the permanent loss of 194.56 ha of lake area of 0.51 ha of watercourse area in tributaries to Kennady Lake.

Fish habitats that will be physically altered during operations and then submerged in the refilled Kennady Lake include the following:

- Part of Kennady Lake Area 3 (affected by Dyke B);
- Part of Kennady Lake Area 4 (affected by Tuzo Pit, Dyke B, Dyke J, and CP6 Berm);
- Part of Kennady Lake Area 6 (affected by Hearne Pit, 5034 Pit, Dyke K, Dyke N, Road between Hearne Pit and Dyke K, CP3 Berm, CP4 Berm, and CP5 Berm); and
- Part of Kennady Lake Area 7 (affected by Dyke A and Dyke K).

The Project will result in 83.32 ha of lake area being physically altered and re-submerged at closure.

The areas that will be dewatered (or partially dewatered) but not otherwise altered before being re-submerged include the following:

- Portions of Kennady Lake Areas 3 through 7 (those parts that are not either permanently lost or physically altered);
- Lake D1; and
- Streams D1, D2, and E1.

The Project will result in approximately 435.90 ha of lake area and 0.23 ha of watercourse area in tributaries to Kennady Lake being dewatered and re-submerged at closure but that will remain otherwise unaltered.

The CCP (Section 3, Appendix 3.II) describes the various options considered for providing compensation, and presents a proposed fish habitat conceptual compensation plan to achieve no net loss of fish habitat according to DFO's Fish Habitat Management Policy (DFO 1986; 1998; 2006).

Effects of Dewatering on Fish and Fish Habitat

Effects of dewatering the main basins of Kennady Lake during mine operations included the direct effects of dewatering activities on the fish population of Kennady Lake, the temporary effect of habitat loss, and the effects of the dewatering discharge on flows, water levels, and channel/bank stability in Area 8.

To minimize the waste of fish caused by dewatering activities, fish salvage will be conducted to remove fish from Areas 2 to 7 before and during dewatering. A combination of gear types would be used to maximize capture efficiency.

Dewatering will result in the temporary loss of fish habitat within Areas 2 to 7 of Kennady Lake. However, it is expected that a self-sustaining fish population will be present in Kennady Lake post-closure (Section 8.13.5). Estimated water levels in Area 8 will be slightly augmented relative to baseline conditions during Kennady Lake dewatering; however, this would not have any effect on fish habitat or shoreline stability, as it would be within the natural variability of the basin.

Effects of Diversions on Fish and Fish Habitat

To reduce the volume of runoff entering the controlled areas of Kennady Lake, the A, B, D, and E watersheds will be diverted to the adjacent N watershed. Habitat downstream of the dykes will be dewatered and lost to fish residing in upstream lakes. The loss of fish habitat resulting from the placement of the dykes and the dewatering of downstream stream segments and lakes is included in the CCP (Section 3, Appendix 3.II). Raising water levels in lakes A3, D2, D3, and E1 will result in increased lake habitat area. The raised water levels will likely create a benefit to fish residing in these lakes during mine construction and operations through additional space and increased amount of overwintering habitat. Populations of northern pike and ninespine stickleback are also likely to benefit from the increased spawning and rearing habitat in areas with flooded vegetation. Changes in water levels and lake areas are also expected to increase habitat area available for plankton and benthic invertebrates, which will result in increased total biomass of plankton and benthic invertebrates, after a period of adjustment to the new water levels.

Raising lake levels in Lakes A3, D2, D3 and E1 will create new shorelines at higher elevations than the existing shorelines, which can result in shoreline erosion and an increased sediment load into the lakes. However, increases in total suspended solids (TSS) concentrations in the raised lakes are expected to be low due to the composition of substrate materials; as a result, negligible effects on fish and fish habitat would be expected.

Dykes in streams A3, B1, D2, and E1 will interrupt the movements of fish between Kennady Lake and waterbodies upstream of the dykes. This effect will be permanent for the A watershed, but will be limited to the period of mine operations for the B, D, and E watersheds. Loss of access to the lowermost streams in the A, B, D, and E watersheds is likely to affect Arctic grayling which currently use these stream habitats for spawning and rearing. Persistence of this species will depend on whether Arctic grayling use habitat constructed in the diversion channels and any immigration of Arctic grayling from the N watershed. Although the dykes will isolate the northern pike populations within the B, D, and E watersheds for the duration of mine operations (and permanently in Lake A3), it is likely that the isolated populations will be self-sustaining. Life history requirements for small populations of burbot, slimy sculpin and ninespine stickleback can be fulfilled in the diverted watersheds, without the need to access Kennady Lake. Prevention of downstream out-migration of juvenile and young-of-the-year fish to Kennady Lake is expected to have a minor effect on fish populations in lakes upstream of the dykes.

Populations of small-bodied fish, such as ninespine stickleback and slimy sculpin, are likely to persist in diverted watersheds during mine operations because suitable spawning, rearing, and foraging habitat for each species will be available and there is no critical habitat in Kennady Lake that any of these species require to complete their life histories. Aquatic vegetation exists in lakes A3, D2, D3, D7, and E1 and these lakes will continue to provide suitable habitat for northern pike and ninespine stickleback throughout mine operations. Although lake trout have been captured in the lakes of the diversion watersheds, it is likely that they are using them seasonally for feeding. The lakes in the B, D, and E watersheds likely do not currently support self-sustaining lake trout populations; therefore, it is not expected that lake trout will persist in these lakes during operations. Lakes A3, D3 and D7 will likely continue to provide the same amount of habitat for burbot that currently exists. The persistence of Arctic grayling in the diverted watersheds will be dependent on the suitability and use of spawning and rearing habitat constructed in the diversion channels and the use of this new habitat by Arctic grayling, as well as by potential immigration of Arctic grayling from the neighboring lakes in the N watershed. The diversion channels will be designed to provide spring spawning and rearing habitat for Arctic grayling and allow the seasonal passage of fish between lakes that approximates natural conditions.

Effects of Isolation on Area 8 on Fish and Fish Habitat

Isolation of Area 8 during operations and closure from the remainder of Kennady Lake was predicted to result in a slight increase in nutrient concentrations. Although the change is not expected to alter the trophic status of Area 8 from oligotrophic, it is expected to result in a slight increase in productivity of plankton and benthic invertebrate communities, without notable changes in community composition or dissolved oxygen concentration.

The residual fish community in Area 8 of Kennady Lake is anticipated to consist of small-bodied fish species (i.e., lake chub, ninespine stickleback, and slimy sculpin), as well as Arctic grayling, northern pike and burbot. As a result of the existing overwintering limitations in Area 8 and the elimination of alternative overwintering refugia in Areas 2 through 7 during operations, lake trout and round whitefish may not continue to persist in Area 8 throughout the operational period, as they are less tolerant of low dissolved oxygen concentrations.

Effects of Dust Deposition on Fish and Fish Habitat

Windborne dust from Project facilities and exposed lake bed sediments, and air emissions from Project facilities, may result in increased deposition of dust in the surrounding area. Effects of TSS from dust and particulate deposition on fish and fish habitat are expected to be localized in the immediate vicinity of the Project and temporally restricted to the period during and after freshet. The potential for adverse effects to aquatic health from dust and metals deposition was considered in the aquatic health assessment to be low (Section 8.13.3) therefore, no effects to fish populations or communities are expected to occur from changes in aquatic health.

8.13.4.2 Closure and Post-Closure

Effects of Development of Fish Habitat Compensation Works to Fish and Fish Habitat

To compensate for habitat permanently lost or altered due to proposed mine development, and eliminate potential adverse effects due to changes in habitat area, the Project includes a habitat compensation plan designed to create new fish habitat (CCP, Section 3, Appendix 3.II). As per the CCP, the preferred options for the proposed compensation plan include:

- raising the water level of some lakes to the west of Kennady Lake (in the D, E, and N watersheds);
- additional raising of the water level in the flooded area created by the above option after mine closure;
- raising Lake A3 to a higher elevation than planned for the development of the Project; and
- widening the top bench of the Tuzo and 5034 mine pits to create shelf areas where they extend onto land.

Also included in the proposed compensation plan are:

- constructing finger reefs in Areas 6 and 7;
- developing habitat enhancement structures in Area 8; and
- developing a Dyke B habitat structure within Kennady Lake after closure.

The amount of compensation habitat, in terms of surface area, provided by the proposed compensation plan is 180.8 ha developed during operations and 381.3 ha developed after closure. This corresponds to a compensation ratio (gains:losses calculated based on total area of permanently lost habitat and physically altered and re-submerged habitat) of 0.65 for operations and 1.37 for closure.

Effects of Restoring the B, D, and E Watersheds to Fish and Fish Habitat

At closure, the natural drainage of the B, D, and E watersheds to Kennady Lake will be restored. Where possible, the watersheds will be reconnected to Kennady Lake along previous connecting streams. Water levels in the raised D2 and D3, and E1 lakes will return to baseline levels. The fish and lower trophic communities within the lakes will adjust to the new lake levels and the restored lake shorelines are expected to remain stable. Habitat conditions for spawning, rearing and overwintering will be similar to pre-Project conditions. As a result, the change would not be expected to have a substantive effect on fish populations within the D and E watersheds. Until the water quality in Kennady Lake is deemed suitable for fish, measures will be taken to limit the initial migration of large-bodied fish from the upper B, D, and E watersheds into Kennady Lake.

Effects of Continued Isolation of Area 8 During Refilling to Fish and Fish Habitat

During refilling of Kennady Lake, Area 8 will remain effectively isolated during this period; effects on Area 8 will be similar to those described above.

Effects to Fish and Fish Habitat in Kennady Lake during Post-Closure

After reconnection of the refilled Kennady Lake to Area 8, concentrations of nutrients are predicted to be higher than during pre-development conditions. While Kennady Lake is oligotrophic under baseline conditions, the post-closure trophic status of Kennady Lake is predicted to be mesotrophic. The predicted change in the trophic status of Kennady Lake is expected to result in an increase in summer phytoplankton biomass and altered species composition of phytoplankton and shifts in dominance at the level of major phytoplankton group. The predicted increase in primary productivity in Kennady Lake is expected to result in increased secondary productivity and biomass of the zooplankton community, reflecting the increased amount of available food for zooplankton. The predicted increases in nutrient concentrations and primary productivity in Kennady Lake are likely to result an increase in benthic invertebrate abundance and biomass, reflecting the increased food supply. A shift in benthic invertebrate community composition is also likely during post-closure.

It is expected that there will be increases in the food base for fish (zooplankton and benthic invertebrates), as well as in the small-bodied forage fish community. Because of the increased food base, there may also be increased growth and production in the large-bodied fish species of Kennady Lake. However, due to the change in trophic status to mesotrophic, overwintering habitat in Kennady

Lake at post-closure may become more limited for cold-water fish species, such as lake trout and round whitefish than under baseline conditions.

The Project is expected to have low or negligible effects on aquatic health in Kennady Lake and Area 8 from changes in chemical constituents of water quality (Section 8.13.3); therefore, no effects to fish populations or communities are expected to occur from changes in aquatic health.

8.13.5 Recovery of Kennady Lake

An aquatic ecosystem will develop within Kennady Lake after refilling and reconnection of its basins. There will be some permanent losses of habitat in Kennady Lake due to mine rock piles, PK storage and mine pits; however, compensation habitats will be constructed within the Kennady Lake watershed to offset losses. The long-term hydrology of Kennady Lake is expected to return to a state similar to current conditions and water quality in the refilled lake is expected to return to conditions suitable to support aquatic life over time. The physical and chemical environment in Kennady Lake, therefore, will be in a state that will allow re-establishment of an aquatic ecosystem, although predicted nutrient concentrations indicate that the re-established communities may differ from pre-development communities.

The expected time frame for recovery of the phytoplankton community is estimated to be approximately five years after refilling is complete, taking into consideration that the phytoplankton community will begin to develop during the refilling period (approximately eight to nine years). The potential increase in nutrient levels in the refilled Kennady Lake may also facilitate community development and will result in a more productive phytoplankton community in the refilled lake compared to the pre-development community.

Zooplankton community development is predicted to follow recovery of the phytoplankton community. Colonization sources will be the same as those for phytoplankton and include the upstream watershed (i.e., the B, D, and E watersheds), Lake N11, and the WMP. The zooplankton community of the refilled lake also is expected to be more productive than the existing community. The expected time frame for the development of the zooplankton community is longer than that of phytoplankton (i.e., likely within five to 10 years of Kennady Lake being completely refilled).

Recovery of the benthic invertebrate community is expected to be slower than that of the plankton communities. The estimated time to recovery for the benthic community in Kennady Lake is about 10 years after refilling is complete. At the end of the recovery period, the benthic invertebrate community in Kennady Lake will be different from the community that currently exists in Kennady Lake and in surrounding lakes. The community will likely be of higher abundance and biomass, reflecting the more productive nature of the lake, and will likely be dominated by midges and aquatic worms.

The re-establishment of the fish community within Kennady Lake, and the speed at which it will occur, will depend on the ability of fish to re-colonize the refilled lake, the habitat conditions within the lake, and how succession takes place within the refilled system after it has been fully connected to the surrounding environment. It is expected that a fish community will become re-established in Kennady Lake.

Fish populations, including Arctic grayling, northern pike, burbot, lake chub, slimy sculpin, and ninespine stickleback, are expected to persist in the B, D, and E watersheds during Project operations. These watersheds are likely to be the primary source of initial migrants into Areas 3 to 7 of Kennady Lake. During refilling, exclusion measures will be used to limit the initial migration of large-bodied fish, such as northern pike, burbot, lake trout, and Arctic grayling, from entering the lake. It is anticipated that during the initial period of refilling, some mortality of the incoming small-bodied fish is likely to occur, because of insufficient water depths and possibly elevated levels of turbidity. As conditions improve, and water depths increase, the early migrants will become permanently established, feeding on the plankton and benthic invertebrate communities that are themselves becoming established in the refilled lake. Nutrient levels in the refilled Kennady Lake are predicted to be higher than under existing conditions. The increase in primary productivity from nutrient enrichment may also result in increased growth and production of these small-bodied forage fish species.

Following the removal of dyke A, migrant fish will also enter Areas 3 through 7 of Kennady Lake from Area 8, which is expected to contain residual populations of lake chub, slimy sculpin, ninespine stickleback, Arctic grayling, northern pike, and burbot. The migration of fish from Area 8 into the rest of Kennady Lake is expected to be rapid, due to proximity and the increased productivity that is expected from increased nutrient levels in Kennady Lake and Area 8.

The final fish community of Kennady Lake will likely once again be characterized by low species richness (less than 10 species) consisting of a small-bodied forage fish community (e.g., lake chub, slimy sculpin, ninespine stickleback) and large-bodied species, such as Arctic grayling, northern pike, burbot, round whitefish, lake trout, and possibly longnose sucker. Total lake standing stock and annual production will be increased over what currently exists in the lake. It is expected that the fish species assemblage (i.e., fish species present) within

Kennady Lake will be similar to pre-Project conditions, but that due to biotic and abiotic factors, the community structure (i.e., relative abundances of the species) may differ. Lake trout may not return as the most abundant predatory fish in the refilled Kennady Lake, as mesotrophic conditions are likely to be more favourable to species such as northern pike and burbot.

The estimated time to full recovery of fish populations is expected to fall between 50 to 60 years following the complete refilling of Kennady Lake, or 60 to 76 years from the end of Project operations.

8.14 RESIDUAL IMPACT CLASSIFICATION

The Gahcho Kué Project (Project) activities will result in changes to the hydrology, water quality, and aquatic communities of the Kennady Lake watershed. As summarized in Section 8.13, the changes are projected to occur during construction and operations, and closure, with changes to water quality and the persistence of fish in Kennady Lake that will continue after closure for the long-term. To assess the environmental significance of the projected changes, a residual impact classification system was developed and applied to the VCs considered in the key line of inquiry. For this key line of inquiry, the VCs consist of water quality, and specific fish species, i.e., Arctic grayling, lake trout, and northern pike, and wildlife and human health (refer to Section 8.5).

In the EIS, the term “effect”, used in the effects analyses and residual effects summary, is regarded as an “impact” in the residual impact classification. Therefore, in the residual impact classification for this section, all residual effects are discussed and classified in terms of impacts to water quality and fish in Kennady Lake.

The residual impact classification focused on VCs, because they represent the components of the aquatic ecosystems in the Kennady Lake watershed that are of greatest interest or concern (as outlined in the Terms of Reference). Projected impacts to VCs also incorporate, or account for, changes to other important key components, such as groundwater quality, groundwater flow, hydrology, fish habitat, and aquatic life occupying lower trophic levels in the ecosystem (e.g., aquatic plants, plankton, zooplankton, benthic invertebrates, forage fish species). Notable changes in water flows, for example, will contribute to changes in water quality, and the quantity and quality of habitat available for Arctic grayling, lake trout, or northern pike. The classification of impacts to water quality and the three valued fish species, therefore, incorporates the classification of impacts to hydrology and key components, according to their influence on the VCs.

The classification was carried out on residual impacts (i.e., impacts with environmental design features and mitigation considered). The environmental design features and mitigation were incorporated in the engineering design or the management plans, and were incorporated in the Project as it evolved (i.e., as the engineers received input from various scientists and traditional knowledge holders, the design evolved).

8.14.1 Methods

The pathways to effects to VCs and assessment endpoints were analyzed in Section 8.6. The pathways that were identified as primary pathways (i.e., likely to result in a measurable environmental change that could contribute to residual effects on a VC relative to baseline or guideline values) were considered and aggregated under their respective biophysical environment (i.e., hydrology, water quality, aquatic health, or fish) in effects statements (e.g., changes to water quality as a result of Project activities during construction and operations). These effects statements set the direction for the residual effects analysis (Sections 8.7 to 8.12), which considered the key Project activities (e.g., diversion of the upper Kennady Lake watersheds, dewatering of Kennady Lake, close circuiting Areas 2 through 7, refilling Kennady Lake) during the phases of the Project (i.e., construction and operations, or closure), to determine the extent of the change to the biophysical environment, and ultimately to the VCs.

The objective of each effects analysis was to determine how Project activities would affect an individual measurement endpoint or a given set of measurement endpoints for a given biophysical environment, e.g., the amount of habitat available to lake trout during operations, or metals concentrations in Area 8 after reconnection with Areas 3 and 7 of Kennady Lake in closure. The measurement endpoints are, in turn, connected to the broader-scale assessment endpoints, which represent the ultimate properties of the system that are of interest or concern.

The residual impact classification focuses on the assessment endpoints because these are statements of what is most important to future generations. The four assessment endpoints relevant to the Key Line of Inquiry: Water Quality and Fish in Kennady Lake, as outlined in Section 8.5, include the following:

- suitability of water quality to support a viable aquatic ecosystem;
- persistence and abundance of desired population(s) of Arctic grayling;
- persistence and abundance of desired population(s) of lake trout; and
- persistence and abundance of desired population(s) of northern pike.

Residual effects to the fifth assessment endpoint, “suitability of water and fish for human and wildlife consumption”, through changes in water quality and fish tissue quality to human health and wildlife health are summarized along with the key findings of the terrestrial wildlife and habitat assessments, in Section 8.12.

The effects analyses (Sections 8.7 to 8.11) and residual effects summary (Section 8.13) presented the incremental changes from the Project on water quality and fish, including the key components of these VCs. Incremental effects represent the Project-specific changes relative to baseline conditions (i.e., 1996 and 2010), through construction and operation of the Project (and into the future, i.e., closure and beyond closure). For this key line of inquiry, the primary focus of Project-specific effects during each Project phase is to the Kennady Lake watershed, which is a requirement in the Terms of Reference. Therefore, the spatial boundary of the assessment is limited to the local study area for the Project. This approach was also adopted to achieve consistency in the scales used to evaluate geographic extent across the key lines of inquiry that focus on aquatic ecosystems.

Residual impacts to each assessment endpoint were classified based on the results of the effects analyses and their linkage to these endpoints. For example, the results of the water quality and aquatic health assessments completed in Sections 8.8 and 8.9 were used to classify residual impacts to the first assessment endpoint (i.e., suitability of water quality to support a viable aquatic ecosystem). Similarly, the results of the analysis of effects to fish and fish habitat, and the projection of the recovery of Kennady Lake described in Sections 8.10 and 8.11, respectively, were used to classify residual impacts to the abundance and persistence of desired population(s) of key fish species.

The residual impact classification describes the residual impacts of the Project on the water quality and fish in Kennady Lake using a scale of common words (rather than numbers and units). The use of common words or criteria is a requirement in the Terms of Reference for the Project. The following criteria are used to describe impacts of the Project on the VCs:

- direction;
- magnitude;
- geographic extent;
- duration;
- reversibility;
- frequency;
- likelihood; and
- ecological context.

Generic definitions for each of the residual impact criteria are provided in Section 6.7.2.

The predicted scales for the impact criteria are also considered in the impact classification. The scales used to assign values (e.g., high, moderate, or low) to each of the classification criteria are outlined in Tables 8.14-1 and 8.14-2. The rating system for magnitude is presented separately in Table 8.14-2, because the scales used to define magnitude are specific to each assessment endpoint, whereas the scales defined for the remaining classification criteria are common across all five assessment endpoints. The results from this impact classification are then used to determine environmental significance of impacts from the Project on water quality and fish (Section 8.14.2).

To provide transparency in the EIS, the definitions for these scales were ecologically or logically based on aquatic environments. Although professional judgment is inevitable in some cases, a strong effort was made to classify impacts using scientific principles and supporting evidence. The scale for the residual impact criteria for classifying effects from the Project are specifically defined for water quality and fish, and definitions for each criterion are provided in Table 8.14-1.

As existing and planned projects in the NWT are located outside of the Kennady Lake watershed, there is no opportunity for the releases of those projects to interact with those of the Project within the Kennady Lake watershed. Consequently, there is no potential for cumulative effects to fish or water quality in Kennady Lake or small lakes and streams in the Kennady Lake watershed.

Table 8.14-1 Definitions of Scales for Seven of the Eight Criteria Used in the Residual Impact Classification

Direction	Geographic Extent	Duration	Frequency	Reversibility	Likelihood	Ecological Context
Neutral: no measurable change to a VC from existing conditions **Negative:** the Project will result in an adverse effect to a VC **Positive:** the Project will result in a beneficial effect to a VC	**Local:** projected impact is confined to watersheds upstream of the outlet of Lake 410; small scale direct and indirect impacts from the Project (e.g., footprint, dust deposition, dewatering) **Regional:** projected impact extends beyond Lake 410 to the inlet to Aylmer Lake; the predicted maximum spatial extent of combined direct and indirect impacts from the Project that exceed local scale effects **Beyond Regional:** projected impact extends into Aylmer Lake and beyond; cumulative local and regional impacts from the Project and other developments extend beyond the regional scale	**Short-term:** projected impact is reversible by the end of construction **Medium-term:** projected impact is reversible upon completion of refilling Kennady Lake (i.e., end of closure) **Long-term:** projected impact is reversible some time after the refilling of Kennady Lake is complete (i.e., beyond closure) or not reversible	**Isolated:** projected impact occurs once, with an associated short-term duration (i.e., is confined to a specific discrete period) **Periodic:** projected impact occurs intermittently, but repeatedly over the assessment period **Continuous:** projected impact occurs continually over the assessment period	**Reversible:** projected impact will not result in a permanent change from existing conditions or conditions compared to 'similar'[(a)] environments not influenced by the Project **Not reversible:** projected impact is not reversible (i.e., duration of impact is unknown or permanent)	**Unlikely:** projected impact is likely to occur less than one in 100 years **Possible:** projected impact will have at least one chance of occurring in the next 100 years **Likely:** projected impact will have at least one chance of occurring in the next 10 years **Highly Likely:** Projected impact is very probable (100% chance) within a year	**High:** projected impact relates to a valued component of the aquatic ecosystem

(a) "similar" implies a waterbody that is similar in size, shape, location, and general characteristics to that affected by the Project (e.g., Kennady Lake).

Table 8.14-2 Definitions Used to Rate the Magnitude of Projected Residual Impacts

Scale	Assessment Endpoint				
	Suitability of Water Quality to Support a Viable Aquatic Ecosystem	Abundance and Persistence of Desired Population(s) of Key Fish Species			Suitability of Water and Fish for Human or Wildlife Consumption
		Abundance of Lake Trout	Abundance of Arctic Grayling	Abundance of Northern Pike	
Negligible	results of the aquatic health and productivity assessments indicate that no measurable change to the overall health of the aquatic ecosystem will occur	no measurable change to the abundance of lake trout, relative to existing conditions	no measurable change to the abundance of Arctic grayling, relative to existing conditions	no measurable change to the abundance of northern pike, relative to existing conditions	results of the human and/or wildlife health assessments indicate that the consumption of water and/or fish from the affected waterbody(ies) will result in no measurable effects to the health of human users and/or wildlife
Low	results of the aquatic health and productivity assessments indicate that a measurable change to the aquatic community may occur, but no notable changes in community structure or overall health of the system are expected	no measurable change in the abundance of lake trout, but population statistics (such as age-class structure) may differ from existing conditions	no measurable change in the abundance of Arctic grayling, but population statistics (such as age-class structure) may differ from existing conditions	no measurable change in the abundance of northern pike, but population statistics (such as age-class structure) may differ from existing conditions	-[(a)]
Moderate	results of the aquatic health and productivity assessments indicate that a measurable change to the aquatic community, including a notable shift in community structure may occur, but no effect to the overall health of the system is expected	projected decrease in abundance of lake trout; however, the species is expected to persist	projected decrease in abundance of Arctic grayling; however, the species is expected to persist	projected decrease in abundance of northern pike; however, the species is expected to persist	-[(a)]
High	results of the aquatic health and productivity assessments conclude that the overall health of the aquatic ecosystem could be affected	projected decrease in the abundance of lake trout is sufficient to result in a complete loss of the species in question (i.e., will not persist)	projected decrease in the abundance of Arctic grayling is sufficient to result in a complete loss of the species in question (i.e., will not persist)	projected decrease in the abundance of northern pike is sufficient to result in a complete loss of the species in question (i.e., will not persist)	results of the human and/or wildlife health assessments indicate that the consumption of water and/or fish from the affected waterbody(ies) will negatively affect the health of human users and/or wildlife

(a) - = not applicable.

8.14.2 Classification Time Periods

Due to the overall nature of how the Project will affect the Kennady Lake watershed, residual impacts were classified for two specific time periods. The first period extended from the initiation of the Project to 100 years later. This time frame incorporated the construction and operations, and closure phases of the Project, and the expected recovery period in which the aquatic ecosystem would be in a stable and productive state (i.e., taking into account the duration of the Project during construction, operations, and closure, and recovery during post-closure). The recovery period was conservatively based on the amount of time that northern pike will re-establish to a stable, self-sustaining population in Kennady Lake following the complete refilling of Kennady Lake. Northern pike are expected to require a long time to re-establish (i.e., 50 to 60 years). As well, once suitable habitat conditions develop for lake trout in the refilled lake, it is expected that this species would also require a long time to re-establish a stable, self-sustaining population (i.e., approximately 60 to 75 years following the complete refilling of Kennady Lake).

The second period focused on future conditions after 100 years from Project initiation. Rather than classifying one snapshot in time, the classification in this period focussed on the ability of the affected ecosystems to recover to a steady state.

8.14.3 Results

8.14.3.1 Residual Impacts to Suitability of Water Quality to Support Aquatic Life

In Section 8.8 and 8.9, the effects of the Project on water quality and aquatic health in the main basins of Kennady Lake (i.e., Areas 3 through 7, Area 2 is incorporated into the Fine PKC Facility during operations) and Area 8 resulting from the pathways of physical changes to Kennady Lake as a result of diversions, dewatering, and refilling activities were assessed for construction and operations, and for closure (including post-closure). The residual effects were summarized in Section 8.13.

During construction and operations, predicted maximum concentrations of suspended solids and some metals may increase above water quality guidelines because of dust and metal deposition in some fish-bearing lakes within two kilometres (km) of the Project. However, given the conservatism in the predicted concentrations, and the potential for exposure to elevated concentrations being limited to the peak watershed flows associated with the freshet, the potential for adverse effects from dust and metals deposition is considered to be low. At the end of operations, a return to existing (i.e., pre-development) conditions is anticipated.

Potential effects to aquatic health in the main basins of Kennady Lake and Area 8 were evaluated for closure and post-closure based on predicted changes in water quality. For the direct waterborne exposure assessment, total dissolved solids (TDS) and several metals were identified as substances of potential concern (SOPCs). A total of four metals in the main basins of Kennady Lake (and three in Area 8) are expected to exceed water quality guidelines for the long term. These metals are cadmium, chromium, copper, and iron, each of which has been measured above guideline concentrations during existing conditions.

With respect to predicted TDS concentrations in the main basins of Kennady Lake and Area 8, adverse effects to fish and aquatic invertebrates are not expected. Predicted maximum concentrations of substances of potential concern (SOPCs) in Kennady Lake and Area 8 are below chronic effects benchmarks (CEBs), with the exception of total iron, copper, and strontium (strontium does not exceed water quality guidelines, but does exceed a conservative CEB [Section 8.9]). The predicted iron concentrations are not expected to result in adverse effects to aquatic life, and the potential for copper and strontium to cause adverse effects to aquatic life in Kennady Lake and Area 8 was considered to be low. For the indirect exposure pathway, predicted fish tissue concentrations in Kennady Lake were projected to be above toxicological benchmarks for only one SOPC: silver. However, as the predicted increase in silver concentration is modest, and baseline and predicted tissue concentrations only marginally exceed the available no-effects benchmark, the potential for the predicted silver concentration to cause effects to fish was considered to be low. Therefore, predicted changes to concentrations of all substances considered were projected to result in negligible effects to fish tissue quality and, by association, aquatic health in Kennady Lake.

After reconnection of the refilled Kennady Lake to Area 8, long-term concentrations of nutrients are predicted to be higher than during pre-development conditions. Whereas Kennady Lake is oligotrophic under baseline conditions, the post-closure trophic status of Kennady Lake and Area 8 is predicted to be mesotrophic.

An increase in the trophic status is expected to result in an increase in summer phytoplankton biomass and altered species composition of phytoplankton and shifts in dominance at the level of major phytoplankton group. The predicted increase in primary productivity is expected to result in increased secondary productivity and biomass of the zooplankton community, and increases in benthic invertebrate abundance and biomass, reflecting the increased food supply. A shift in benthic invertebrate community composition is also likely during post-closure. It is expected that there will be increases in the food base for fish (zooplankton and benthic invertebrates), as well as in the small-bodied forage fish community. Because of the increased food base, there may also be increased growth and production in the large-bodied fish species of Kennady Lake.

Due to the change in trophic status, overwintering habitat in Kennady Lake at post-closure may become more limited for some fish species than during pre-development conditions. Fish species that are more tolerant of low dissolved oxygen levels (e.g., lake chub, slimy sculpin, ninespine stickleback, Arctic grayling, northern pike, and burbot) will be able to overwinter successfully in the refilled Kennady Lake. Cold-water fish species, such as lake trout and round whitefish, are less tolerant of low dissolved oxygen levels. Reductions in the suitability and availability of overwintering habitat in Kennady Lake for these species, may limit the abundance of lake trout and round whitefish relative to other large-bodied fish.

Based on the above, projected impacts of the Project on the suitability of water within the Kennady Lake watershed to support a viable and self-sustaining aquatic ecosystem are negative in direction and moderate in magnitude during the first 100-year period. The moderate magnitude rating is based primarily on the dewatering and subsequent loss of the aquatic ecosystem in Kennady Lake during mining operations, the refilling of Kennady Lake during closure and the recovery of water quality through post-closure once the lake is filled and reconnected with Area 8. These projected impacts are local in geographic extent, long-term in duration, and reversible.

After the initial 100 year period, the projected impacts of the Project were rated as negative in direction, low in magnitude, local in geographic extent, long-term in duration, and not reversible. The low magnitude rating is based primarily on the long-term increases in the productivity of the main body of Kennady Lake and Area 8 as a result of increased nutrients. There are both positive and negative effects to aquatic life within Kennady Lake from the increased productivity. The increased nutrients will be reflected in increased biomass of lower trophic communities, and may also increase fish productivity, e.g., growth and production of large-bodied fish species, such as northern pike and burbot, compared to the present nutrient-limited system. However, due to the change in trophic status to mesotrophic, overwintering habitat in Kennady Lake at post-closure may become more limited for some fish species than pre-development conditions. Kennady Lake would be expected to retain sufficient levels of under-ice dissolved oxygen during winter to support fish; however, there may be reduced suitability and availability of overwintering habitat for cold-water fish species, such as lake trout and round whitefish. However, there are no predicted long-term effects to aquatic health that would impair the suitability of the water quality to support aquatic life. By this time period, it is expected that the lake would have reached its new equilibrium. Under both time periods, the projected impacts are considered to be continuous and likely to occur.

8.14.3.2 Residual Impacts to the Abundance and Persistence of Desired Population(s) of Key Fish Species

In Section 8.10, the effects of the Project on fish and fish habitat in Kennady Lake and in small lakes and streams in the Kennady Lake watershed resulting from the pathways of physical changes, and changes to water quantity and quality were assessed for construction and operations, and for closure and post-closure. The expected recovery of Kennady Lake and the nature of the final ecosystem are described in Section 8.11. The residual effects for each assessed pathway were summarized in Section 8.13.

Changes to fish habitat will occur in Kennady Lake and the Kennady Lake watershed due to the development of the Project. However, the conceptual compensation plan (CCP) (Section 3, Appendix 3.II) will provide compensation habitats to offset fish habitat permanently lost due to the Project. Areas 2 to 7 of Kennady Lake will be dewatered or partially dewatered to allow mining to proceed, resulting in the temporary loss of productive capacity of fish habitat; however, it is expected that a self-sustaining fish population will be present in Kennady Lake post-closure. Raising water levels in lakes A3, D2, D3, and E1 from the A, B, D, and E watershed diversions will result in increased lake habitat area, which may create a benefit to fish residing in these lakes through additional space and overwintering habitat. A slight increase in nutrient concentrations was predicted in Area 8 during isolation, which could result in a slight increase in productivity of plankton and benthic invertebrate communities. The fish community of Area 8 is also expected to be affected by the isolation, due to existing overwintering limitations in Area 8 and the elimination of alternative overwintering refugia in Areas 2 through 7.

Table 8.14-5 Residual Impact Classification of Projected Impacts to Water Quality and Fish in Kennady Lake

Assessment Endpoint	Direction	Magnitude	Geographic Extent	Duration	Frequency	Reversibility	Likelihood	Ecological Context
Suitability of water within the Kennady Lake watershed to support a viable and self-sustaining aquatic ecosystem								
Construction to 100 years from Project start	negative	moderate	local	long-term	continuous	reversible	likely	high
Beyond 100 years from Project start	negative	low	local	long-term	continuous	not reversible	likely	high
Abundance and persistence of Arctic grayling within the Kennady Lake watershed								
Construction to 100 years from Project start	negative	high	local	long-term	continuous	reversible	likely	high
Beyond 100 years from Project start	negative	low	local	long-term	continuous	not reversible	likely	high
Abundance and persistence of lake trout within the Kennady Lake watershed								
Construction to 100 years from Project start	negative	high	local	long-term	continuous	reversible	likely	high
Beyond 100 years from Project start	negative	moderate	local	long-term	continuous	not reversible	likely	high
Abundance and persistence of northern pike within the Kennady Lake watershed								
Construction to 100 years from Project start	negative	high	local	long-term	continuous	reversible	likely	high
Beyond 100 years from Project start	neutral - positive	negligible	-	-	-	-	-	-

"-" = not applicable.

When mining is complete, the B, D, and E diversion systems will be decommissioned, the water levels in the raised lakes D2 and D3, and E1 will return to baseline levels, and Kennady Lake will be refilled.

An aquatic ecosystem is expected to become established in the refilled Kennady Lake. The expected time-frame for recovery of the phytoplankton community is projected to be approximately five years after refilling is complete. Zooplankton community development is projected to closely follow recovery of the phytoplankton community, with the recovery of the benthic invertebrate community expected to take up to 10 years after refilling is complete. During this time, the forage fish community will also develop, followed by a slower recovery of the large-bodied fish community. Due to changes in trophic status, habitat conditions, and potentially predator/prey interactions during recovery, the fish community structure may differ from pre-Project conditions. Overall biological productivity is expected to increase in comparison to the nutrient-limited pre-development conditions.

From the pathways assessed in Section 8.10, the classification of projected impacts of the Project on the abundance and persistence of the three highly valued fish species, namely Arctic grayling, lake trout, and northern pike, is outlined in more detail below. As described above, the projected impacts on the abundance and persistence of the three key fish species were classified over two time periods: from the start of the Project to 100 years later; and after the first 100 years.

8.14.3.2.1 Arctic Grayling

During the first 100 year time period, the projected impacts on the abundance and persistence of Arctic grayling are negative in direction, high in magnitude, local in geographic extent, long-term in duration, continuous in nature, reversible, likely to occur, and high ecological context (Table 8.14-3). The largest impact to Arctic grayling in Kennady Lake and its watershed during construction and operations will be the dewatering of Areas 2 through 7 of Kennady Lake and the associated temporary loss of habitat. However, it is expected that Arctic grayling will be able to persist in Area 8 during isolation, although the population may be affected by predicted flow changes in streams downstream of Kennady Lake (Section 9). Persistence of Arctic grayling in the diverted watersheds will be dependent on the suitability and use of spawning and rearing habitat constructed in the diversion channels, and by potential immigration of Arctic grayling from the neighbouring lakes in the N watershed. The impacts are considered reversible, as it is expected that Arctic grayling will re-colonize the refilled Kennady Lake during post-closure.

During the second time frame, projected impacts are negative in direction, low in magnitude, local in geographic extent, long-term in duration, and not reversible (Table 8.14-3). Although increases in the growth and production of Arctic grayling in the refilled Kennady Lake may occur as a result of the increased planktonic and benthic food base, the re-established Arctic grayling population may take time to recover, or may not recover to existing conditions (i.e., in terms of standing stock and annual production rates) because of predicted changes in habitat conditions in the refilled Kennady Lake and downstream watershed.

Arctic grayling will likely establish a self-sustaining population in the refilled Kennady Lake earlier than northern pike. Arctic grayling will be able to access Kennady Lake from the downstream M watershed, as well as the upper B, D, and E watersheds. The recovery of the planktonic community will provide a stable food source for Arctic grayling rearing in Kennady Lake, which will likely be enhanced by nutrient enrichment. Spawning habitat will be available in streams in the reconnected B, D, and E watersheds and downstream of Area 8. It is expected that this species will be able to overwinter and form a self-sustaining population within the refilled Kennady Lake.

Arctic grayling begin to reach maturity in about four years and have a life expectancy of 6 to 10 years. A self-sustaining population of Arctic grayling reared in Kennady Lake should be present about 5 to 10 years after the exclusion measures are removed, or about 50 years after the start of construction. At that time, the abundance of Arctic grayling is expected to be substantially less than current abundance. However, given the relatively short time to maturity, the opportunities for immigration, and the reduction in predation by lake trout, the population is projected to increase in the next 50 years, which represents 5 to 10 generations. A precise prediction of fish abundance cannot be developed for an equilibrium state that will develop after 100 years; however, it is expected that a self-sustaining population of Arctic grayling will be present.

8.14.3.2.2 Lake Trout

Projected impacts to the abundance and persistence of lake trout, from the start of Project activities to 100 years later, are rated as negative in direction, high in magnitude, local in geographic extent, long-term in duration, and reversible (Table 8.14-3). The largest impact to lake trout in Kennady Lake and its watershed during construction and operations will be the dewatering of Areas 2 through 7 of Kennady Lake and the associated temporary loss of habitat. As a result of the existing overwintering limitations in Area 8 and the elimination of alternative thermal and overwintering refugia in Areas 2 through 7, lake trout may not continue to persist in Area 8 throughout the operational period, as they are less tolerant of low dissolved oxygen concentrations. Few lake trout have been captured in the lakes of the diversion watersheds, and only in lakes A3, B1, and

D3. Lake trout that have been captured in lakes B1 and D3 are likely using the lakes seasonally for rearing and feeding; these lakes likely do not currently support self-sustaining lake trout populations due to their small size and shallow depths. As a result, it is not expected that lake trout will persist in the lakes of the B, D, and E diverted watersheds during operations. However, Lake I1 and other lakes in the downstream watershed are potential sources for lake trout during colonization; the impacts are considered reversible, as it is expected that lake trout will re-colonize Kennady Lake during post-closure.

During the second time frame, the projected impacts on the abundance and persistence of lake trout are rated as negative in direction, moderate in magnitude, local in geographic extent, long-term in duration, and not reversible (Table 8.14-3). Although lake trout are expected to re-establish in the lake, recovery of the population abundance is partially influenced by the ability of lake trout to re-colonize Kennady Lake. Immigration of lake trout from downstream lakes is less likely to occur than for other large-bodied fish species. Furthermore, as lake trout are sensitive to low dissolved oxygen levels, this fish species may be affected by the reduced suitability and availability of overwintering habitat in Kennady Lake resulting from the change in trophic status. Although overwintering habitat conditions are expected to be suitable for lake trout in the refilled lake, overwintering habitat may be limiting, and the population may not recover to previous levels. Lake trout may not return as the most abundant predatory fish in the lake, due to overwintering habitat limitations, and the fact that conditions may be more favourable to other predatory fish species, such as northern pike and burbot. The geographic extent of any projected impact was determined to be local, as the impact would not extend to lake trout populations in downstream lakes (e.g., Lake 410).

8.14.3.2.3 Northern Pike

During the first 100 year time period, the projected impacts on the abundance and persistence of northern pike were rated as negative in direction, high in magnitude, local in geographic extent, long-term in duration, and reversible (Table 8.14-3). The largest impact to northern pike in Kennady Lake and its watershed during construction and operations will be the dewatering of Areas 2 through 7 of Kennady Lake and the associated temporary loss of habitat. However, it is expected that northern pike will be able to persist in Area 8 during isolation, although the population may be affected by predicted flow changes in streams downstream of Kennady Lake (Section 9). Although the dykes will isolate the northern pike populations within the A, D, and E watersheds for the duration of mine operations (and permanently in Lake A3), it is considered likely that the isolated populations will be self-sustaining. The impacts are considered reversible, as is expected that northern pike will re-colonize Kennady Lake during post-closure.

During the second time period, projected impacts on the abundance and persistence of northern pike were rated as neutral to positive in direction and negligible (Table 8.14-3). Increases in the growth and production of northern pike may occur in the refilled Kennady Lake as a result of the increased food base resulting from increased primary and secondary productivity. Spawning and rearing habitat in the refilled Kennady Lake is expected to be similar to, or better than, what currently exists for this species. Northern pike are dependent on aquatic vegetation for spawning and rearing. The presence of aquatic vegetation in Kennady Lake is currently limited by physical factors, such as rocky substrates and wave action. However, existing macrophyte beds in sheltered areas may benefit from the increased nutrient concentrations, which would be reflected in increased plant abundance and productivity. The recovery of the population may also be enhanced by the lack of lake trout as the top predator at least initially in the recovery. Northern pike populations are expected to recover to similar levels to what currently exists by the second time period. It is expected that northern pike will establish a self-sustaining population in the refilled Kennady Lake. Although migrants will be located in nearby systems, including the B, D, and E watersheds, northern pike are dependent on aquatic vegetation for spawning and rearing. Currently, the abundance of aquatic vegetation in Kennady Lake is limited to small isolated pockets where fine substrates accumulate within the lake. The pockets commonly occur at the mouths of the small tributaries that flow into Kennady Lake. Although aquatic vegetation is expected to eventually become re-established in the lake, re-colonization of aquatic vegetation is expected to be slow. With the exception of the younger juveniles, northern pike feed almost exclusively on fish and will rely on the recovery of the forage fish base; however, they will likely benefit from the increased density of forage fish due to higher biological productivity associated with nutrient enrichment. As northern pike are more tolerant of low dissolved oxygen concentrations than other large-bodied fish species, they will likely not be as influenced by the potential reduction in overwintering habitat in Kennady Lake and form a self-sustaining population within the refilled lake.

Northern pike mature relatively quickly, with an average age to maturity of about three years. Their life span ranges from about 10 to 26 years, generally averaging around 25 years. Even though northern pike reach maturity relatively quickly, northern pike is expected to re-establish self-sustaining populations in the refilled Kennady Lake later than Arctic grayling or burbot, due to the need for re-colonization of aquatic vegetation in the lake for spawning and rearing, and the development of a forage fish food base. As a result, recruitment of northern pike in Kennady Lake will occur for some time primarily through migration from lakes in the D and E sub-watersheds, and to a lesser extent from downstream of Area 8.

The re-establishment of a stable, self-sustaining northern pike population Kennady Lake during post-closure is expected to take a long time (i.e., approximately 50 to 60 years following the complete refilling of Kennady Lake) and it may take additional time (i.e., greater than 100 years) for the abundance of northern pike to recover to current levels. A precise prediction of fish abundance cannot be developed for an equilibrium state that will develop after 100 years; however, it is expected that a self-sustaining population of northern pike will be present at levels similar to existing conditions.

8.14.4 Environmental Significance

Ultimately, significance will be determined by the Panel. In the Mackenzie Valley Environmental Impact Review Board (MVEIRB 2006) reference bulletin on interpretation of key terminology, the term "significant" means an impact that is, in the view of the MVEIRB, important to its decision. To determine significance, the MVEIRB (2006) "will use its own values and principles of good EIA. It will use its combined experience and knowledge". Presumably the determination of significance will be made in a similar manner by the Gahcho Kué Panel. However, the Terms of Reference require that De Beers provide its views on the significance of impacts. To that end, projected impacts were evaluated to determine if they were environmentally significant.

The evaluation of significance for this key line of inquiry considers the entire set of primary pathways that influence a particular assessment endpoint, but does not assign significance to each pathway. The relative contribution of each pathway is used to determine the significance of the Project on assessment endpoints, which represents a weight of evidence approach. For example, a pathway with a high magnitude, large geographic extent, and long-term duration would be given more weight in determining significance than pathways with smaller scale effects. The relative impact from each pathway is discussed; however, pathways that are predicted to have the greatest influences on changes to assessment endpoints would be assumed to contribute to most to the determination of environmental significance.

Environmental significance is used here to identify projected impacts that have sufficient magnitude, duration, and/or geographic extent that they could lead to fundamental changes to the VCs. For example, significance is determined by the risk to the persistence of fish populations within the aquatic ecosystem. The following definitions are used for assessing the significance of effects on the protection of surface water quality for aquatic and terrestrial ecosystems, and human use are as follows.

Not significant – impacts are measureable at the local scale, and may be strong enough to be detectable at the regional scale.

Significant – impacts are measurable at the regional scale and are irreversible. A number of high magnitude and irreversible effects (i.e., pathways) at the regional scale would be significant.

The following definitions are used for assessing the significance of impacts on the persistence of VC fish populations, and the associated continued opportunity for traditional and non-traditional use of these VCs.

Not significant – impacts are measurable at the individual level, and strong enough to be detectable at the population level, but are not likely to decrease resilience and increase the risk to population persistence.

Significant – impacts are measurable at the population level and likely to decrease resilience and increase the risk to population persistence. A high magnitude and irreversible impact at the population level would be significant.

Suitability of water within the Kennady Lake watershed to support a viable and self-sustaining aquatic ecosystem

During the first 100 year time period, the projected impacts of the Project on the suitability of water within the Kennady Lake watershed to support a viable and self-sustaining aquatic ecosystem are considered to be not environmentally significant. During the second time frame, projected impacts are also considered to be not environmentally significant. Water quality is predicted to change but the level of changes in Kennady Lake (including Area 8) include a few metals that are expected to exceed water quality guidelines for the protection of aquatic life. Those that do (i.e., cadmium, chromium, copper and iron), are metals that have been measured above guidelines during existing conditions. Chronic effects benchmarks for these metals, and other parameters that were identified as SOPCs, in the aquatic health assessment were not exceeded.

Phosphorus is projected to increase in long-term steady state concentrations that may shift the trophic status of Kennady Lake, including Area 8, up one trophic level (i.e., from oligotrophic to mesotrophic). The projected increases in phosphorus will not pose a health risk to a viable and self-sustaining aquatic ecosystem, though it will likely be different to the pre-development ecosystem (e.g., the lake will become a more productive aquatic ecosystem compared to the nutrient-limited baseline condition).

Abundance and persistence of Arctic grayling within the Kennady Lake watershed

The projected impacts on the abundance and persistence of Arctic grayling are considered to be not environmentally significant for both time periods. Arctic grayling will be affected by the loss of habitat in Kennady Lake during the life of the mine, but will continue to persist in Area 8 and the diverted watersheds. It is expected that a self-sustaining population will become established in the refilled lake.

Abundance and persistence of lake trout within the Kennady Lake watershed

The projected impacts on the abundance and persistence of lake trout are considered to be not environmentally significant for both time periods. Lake trout will be affected by the loss of habitat in Kennady Lake during the life of the mine. Although lake trout are not expected to persist in Area 8, due to the elimination of alternative thermal and overwintering refugia in Areas 2 through 7 of Kennady Lake, they will continue to be present in the downstream watershed. It is expected that the refilled Kennady Lake will provide suitable habitat conditions for a self-sustaining lake trout population to become established, although the population may not recover to current levels of abundance, due to biotic and abiotic factors.

Abundance and persistence of northern pike within the Kennady Lake watershed

The projected impacts on the abundance and persistence of northern pike are considered to be not environmentally significant for both time periods. Northern pike will be affected by the loss of habitat in Kennady Lake during the life of the mine, but will continue to persist in Area 8 and the diverted watersheds. It is expected that a self-sustaining population will become established in the refilled lake.

8.15 UNCERTAINTY

Key areas of uncertainty for the assessment of effects to water quality and fish in Kennady Lake include the following:

- the Gahcho Kué Project (the Project) site water balance;
- quality and quantity of groundwater inflow to the mined-out pits;
- water quality modelling and quality of assigned chemistry of source inputs;
- dust and metals deposition to lakes adjacent to the Project;
- time required to refill Kennady Lake; and
- time to aquatic ecosystem recovery in Kennady Lake.

Each area of uncertainty is discussed in more detail below. The following discussion also includes a description of the approaches used to account for uncertainty in the effects analysis, so that potential effects were not underestimated. Where relevant, the inherent advantages of the design of the Project are also discussed, in terms of how they influence uncertainty in the assessment of effects to water quality and fish in Kennady Lake.

8.15.1 Project Site Water Balance

The site water balance describes the movement of water through the Project site over the life of the Project. The water balance determines how much water will be discharged from the Project site to the receiving environment. The site water balance also identifies the sources of water entering and leaving the site.

The site water balance was developed through the use of a water balance model, and there is a high degree of confidence in the hydrological aspects of the project description that are considered in the water balance model. In most cases, the changes to the Kennady Lake watershed that will result from the Project are well-defined and subject to limits arising from environmental design features. For example, the volume of Kennady Lake is well-defined, and discharges during dewatering will be managed within specified limits. Similarly, the drainage areas of the diverted A, B, D, and E watersheds are well-defined, and discharges will be managed within specified limits.

There is a corresponding high degree of confidence in the meteorological inputs to the water balance model inputs (e.g., temperature, precipitation) for median conditions, due to the quality of the available regional dataset. The length of the available datasets, which span from 46 years for the regional dataset to 2 to 7 years for more site-specific information, results in a lower level of confidence in the prediction of events with longer return periods, such as 1-in-50 or 1-in-100 year events. However, lake dynamics are driven to a greater extent by average or median conditions than by extreme events. As such, confidence levels are highest around those elements of the water balance model of most importance.

8.15.2 Quality and Quantity of Groundwater Inflow

An important assumption underlying the prediction of water chemistry in Areas 3 through 7 during operations and closure throughout the life of the Project is the quantity and quality of groundwater inflows to the pits. Groundwater inflow to the pits will be pumped to the WMP, where it will mix with site contact water. While water in the WMP meets specific water quality criteria it will be discharged to Lake N11. At the end of operations, water in the WMP will be diverted to the Tuzo Pit. If these objectives are not met, it could result in a different water quality profile than presented herein for Areas 3 through 7 during operations and in Kennady Lake after refilling.

As with all other geologic and hydrogeologic studies, there is a level of uncertainty in all effects analysis results. These uncertainties are inherent in these studies due to uncertainties within the groundwater measurement database, and the requirement to extrapolate or interpolate properties to a continuum based on sparse measurements. The primary uncertainties with regard to groundwater component in the water quality analysis within this key line of inquiry are related to the analysis of:

- pit inflow volumes; and
- groundwater quality.

Pit Inflow Volumes

At existing diamond mines in the Northwest Territories (NWT), which are adjacent to large waterbodies, groundwater inflow tends to be the largest source of water entering the mine site. At the Snap Lake Mine, the underground mine is located beneath a lake, which creates a steep hydraulic gradient that induces water to flow from the bottom of the lake through the shallow bedrock and into the mine (De Beers 2002). At Lac de Gras, the Diavik Diamond Mine is constructed inside a ring dyke, and fractured rock associated with a vertical fault in the bedrock provides a more permeable pathway from the lake to the mine than initially anticipated (Diavik 2006). The degree to which groundwater flow rates can be accurately estimated, therefore, has a large influence on the overall mine site water balance at these facilities.

Other mine developments in the north have experienced significant under estimations of the volumes of water reporting to the pits or underground workings, particularly in the Snap Lake and Diavik diamond mines. This under estimation of groundwater inflow prior to mining has been due to the presence of enhanced permeability zones. Enhanced permeability zones are zones of greater fracturing or larger fracture apertures related to structures such as faults. These zones have been found at Diavik, Ekati and at Snap Lake; none of which were identified during extensive field investigations prior to mining. At Diavik, in addition to the 100 m wide enhanced permeability zone referred to as Dewey's Fault, similar but thinner zones have been found: one zone parallel to Dewey's Fault and the other two perpendicular to this fault.

The hydrogeological model developed for the Project assumes that enhanced permeability zones are present and associated with geologic faults identified in the geophysical surveys that intersect the proposed open pits. However, their presence has not yet been confirmed, and it is possible that the structures are less permeable, thinner and/or less lateral extensive than the ones represented in the base case hydrogeological model.

Based on past experiences at other mines in the north, increases in the overall mine inflow result in increased mass loading as more groundwater is moving upwards from the region where the deep-seated saline groundwater is present. At the Project site, this phenomenon is expected to be more pronounced, because of the presence of permafrost that nearly surrounds all three of the planned open pits (which will limit the dilution of the deep-seated saline groundwater by shallow fresher groundwater). The average model-predicted percentage of groundwater inflow that originates from the freshwater lakes at the Project site is about 40% to 50%. At the Diavik Diamond Mine, there is a continuous source of freshwater from Lac de Gras, and the percentage of groundwater inflow that originates from the lake is estimated to be greater than 70%.

Groundwater Quality

The results of groundwater quality monitoring were used to estimate the composition of groundwater that could upwell into the open pits during operations. The results of groundwater quality monitoring are discussed in Section 11.6 (Subject of Note: Permafrost, Groundwater, and Hydrogeology). Depth profiles were developed to evaluate the variability of groundwater composition with depth. TDS is known to vary with depth in groundwater in the Canadian Shield. The purpose of the depth profiles was to identify parameters that correlate with TDS relative to depth. Linear regression equations were developed based on the results in the groundwater quality dataset to estimate the concentrations of TDS (including calcium, chloride, potassium, magnesium, sodium and sulphate), arsenic, boron, copper, nickel and selenium with depth.

Concentrations of ammonia, phosphorus, aluminum, antimony, barium, beryllium, chromium, cobalt, iron, lead, manganese, mercury, molybdenum, silver, thallium, uranium, vanadium and zinc were estimated based on the range of results in the groundwater dataset. The groundwater quality dataset was used to develop input concentrations for groundwater inflows to the Hearne and 5034 pits. Input concentrations are equal to the maximum concentration measured in groundwater samples from each pit. This approach was developed based on a detailed review of the groundwater quality dataset. This approach is considered somewhat conservative because of the high variability in metal concentrations with depth and by location. Furthermore, the review of the results of groundwater quality monitoring identified concentrations of some parameters, such as chromium, that were anomalously elevated in select samples. These concentrations were not excluded from the statistical calculations used to define groundwater input water quality; however, the input concentrations will be re-visited after supplemental groundwater samples are collected from the groundwater monitoring wells at the Project.

As indicated by the review of the groundwater quality dataset, groundwater quality varies with location and depth within the Kennady Lake area. The variability in groundwater quality may be a function of several factors:

- Difficulties encountered during groundwater sampling could have resulted in mixing of groundwater samples with drilling fluids, which, depending on the groundwater quality and chemical composition of these fluids, could result in over- or under-estimates of actual TDS levels in the deep groundwater.
- Groundwater quality, particularly TDS, could be influenced by local variations in the vertical and horizontal components of the convective flux due to hydraulic gradients, density gradients, hydraulic conductivity and/or local variations in diffusive flux from the deep-seated saline groundwater resulting from the relative interconnection of pore space in the rock mass.

Despite the variability in groundwater quality, the TDS values of groundwater samples are generally consistent with the TDS of groundwater observed at other sites in the Canadian Shield (see Section 11.6, Subject of Note: Permafrost, Groundwater, and Hydrogeology).

Because the inflow and TDS mass are interdependent, it is likely that if reasonably highly conservative values of bedrock hydraulic conductivity were simulated together with a reasonably highly conservative TDS/depth profile the result would be an overly conservative TDS mass load. Therefore, in a model sensitivity which employs a more conservative TDS/depth profile, less conservative values of bedrock hydraulic conductivity are considered to be appropriate.

As a consequence of the above, two model sensitivities were undertaken:

- Sensitivity Run #1: In this model simulation, the enhanced permeability zones were removed from the model. All other parameters, including the TDS/depth profile, remained the same as the Base Case model. This simulation resulted in a lower bound estimate of inflow and TDS mass.
- Sensitivity Run #2: In this model simulation, the enhanced permeability zones were removed, but a conservative TDS/depth profile was used. The TDS concentrations in this profile are twice that used in the Base Case model. All other hydrogeologic parameters remain the same as those in the Base Case model.

Results of Model Sensitivity #1 indicate that groundwater inflows to the mines, if the enhanced permeability zones were not present, would be on average approximately 40% lower than predicted in the Base Case. Generally, predicted groundwater inflows in this sensitivity simulation are very close to those predicted in the Base Case when the pits are shallow and groundwater inflow occurs primarily through the till and exfoliated rock units; however, for the ultimate pit configurations predicted groundwater inflows are between 50% and 70% lower than in the base case predictions. The predicted groundwater load in this simulation is generally lower than that predicted for the base case.

Predicted TDS concentrations for Sensitivity Run #2 are, on average, 1.5 to 2 times greater than those predicted for the Base Case. However, because predicted groundwater inflow rates in this scenario are lower than those under the Base Case, the overall mass loading to the pits is similar in each of the two scenarios (i.e., Sensitivity Run #2 and the Base Case).

While considerable effort has been expended assessing the dynamics of pit dewatering, backfilling, and flooding, the assessments have simplified a highly complex and dynamic system and represent bounding conservative calculations that result in a reasonable degree of confidence that effects on groundwater and the potential for changes in groundwater to affect surface waters have not been underestimated.

De Beers is committed to complete monitoring and testing using standard field and laboratory procedures during the Project operation to evaluate groundwater quantity and quality. Where necessary, the water chemistry and quantity input profiles assigned to the loadings for groundwater will be revised and Project effects will be re-assessed, as appropriate. Where required, adaptive management strategies will be adopted.

8.15.3 Water Quality Modelling

Water chemistry in Kennady Lake and in Area 8 as a whole (after refilling) will be dependent on the quality of the influent streams entering the basin / lake. The predictions of water quality in Area 8 during construction, operation, and closure, and that in Kennady Lake during and after refilling, was completed using a dynamic, mass-balance model built within GoldSim™, which is widely used in environmental assessment. The GoldSim™ model was specifically used to simulate water quality outcomes in a receiving environment over time with multiple input variables.

The GoldSim™ water quality model was based on the site water balance and included inputs of material from the following sources:

- natural runoff to Areas 1 through 7, and Area 8, which were assigned mean baseline water quality;
- metals and other elements associated with the suspended solids in the WMP (the quality of which was defined by laboratory analysis of bed sediment from Kennady Lake [Appendix 8.I]);
- groundwater that will be pumped from open pits into the WMP (quality of which was derived from observed groundwater chemistry [Appendix 8.I]);
- contact runoff from Project areas to the WMP, including the input of:
  - mine rock and coarse PK contact water and seepage from the Fine PKC Facility (quality of which was defined based on geochemistry studies and loading calculations provided in the Metal Leaching and Acid/Alkaline Rock Drainage Report [Appendix 8.II]); and
  - blasting residue (quality of which was defined based on the nitrogen release assessment provided [Appendix 8.I]).

Baseline water quality data from the Project area provided the basis for estimation of the quality of natural runoff and inflows from unaffected areas. The prediction of water quality in Area 8 was based on modelling Project releases to mean baseline water quality conditions. Some uncertainty around these predictions results from the use of a mean baseline value assigned to each water quality parameter, when the dataset contains a naturally large degree of variability. The modelling was also focused on median climatic conditions. Although these areas of uncertainty exist, the selected approach is appropriate for lake systems, which are more strongly influenced by average conditions, rather than short-term extremes. In addition, the modelled water quality parameters were all treated as conservative substances; no chemical transformations, biological uptake, degradation, or precipitation was assumed. When deriving means for baseline water quality, individual data that were below reporting limits were replaced with a value equal to half the detection limit. This approach will likely yield a conservative estimate of the actual mean concentrations.

Projections of modelled water quality were based on the assumption that Area 8 will be completely mixed during open water conditions. This approach was adopted, because Area 8 has a short residence time, in the order of one year.

As described in Appendix 8.II, the composition of water that comes into contact with mine rock and processed kimberlite was estimated based on the results of geochemical characterization:

- Mine rock contact water chemistry were defined based on the results of humidity cell testing discussed in Appendix 8.II, Section 8.II.4.3.4. Water chemistry was defined for each mine rock type based on the concentrations measured during the initial flushing of the humidity cell tests, and the longer term, “steady state” results of humidity cell testing, respectively.
- The results of humidity cell testing and saturated column testing of fine and coarse PK, respectively, were used to define the major ion, trace metal and nutrient composition of PK runoff and seepage water quality. Fine PK and coarse PK exposed in the Fine PKC Facility and Coarse PK Pile, respectively, will undergo seasonal wet and dry cycles during the summer months as discussed in Appendix 8.II, Section 8.I.2.4.4.
  - The maximum major ion, trace metal and nitrogen-nutrient concentrations reported in the first five weeks of testing in the fine PK test programs was selected to represent the seepage water quality from fine PK materials during freshet (see Appendix 8.II). At the time of modelling, only five weeks of humidity cell test results were available from the supplemental fine PK humidity cell sample. Based on the available results, it was difficult to ascertain if steady-state conditions had been realized. As such, to determine the expected long-term concentration in the humidity cell tests, the 2008 fine PK humidity cell tests were compared to the water collected from the bottom of the saturated fine PK column tests. The expected

steady-state water chemistry, with respect to major ions, trace metals and nitrogen-nutrients, for fine PK was calculated as the maximum concentration reported in the last five weeks of testing from the AMEC (2008) humidity cell tests and the maximum concentration reported in the bottom leachate water of the saturated column test.

- The results of PK process water analysis were used to estimate the composition of the discharge from the Fine PK Facility. Process water, which is typically recycled multiple times through the plant, will be discharged to the Fine PKC Facility as a component of the fine PK slurry. As such, it is considered reasonable that the pond in Area 2, collecting drainage from fine PK runoff and seepage will have a composition similar to the process water quality. During operations, when a pond will be maintained in Area 2, Fine PKC Facility discharge was calculated as the maximum of the simulated Area 2 pond water quality and the process water quality.
- Saturated column tests were initiated to evaluate the effect of saturated fine and coarse PK in Kennady Lake. The major ion and trace metal composition of water in contact with saturated PK was defined based on the maximum results measured during the first five weeks of saturated column testing, as this was the only information available at the time of preparation of the water quality predictions.
- On-going humidity cell and saturated column testing will be used to refine the geochemical source terms for mine rock and PK material and update the water quality model.

Phosphorus concentrations in contact water tests with mine rock and PK material were measured using two methods: ICP-MS and colorimetry. Phosphorus loading from the contact water tests with mine rock was derived from early geochemical testing (AMEC 2008) that included both ICP-MS and colourimetric methods (Appendix 8.II). The bulk of the tests for phosphorus were measured by ICP-MS that were consistently below the analytical detection limit (0.15 mg/L). The detection limit of 0.15 mg/L for phosphorus was considered too high to use in the water quality model to reliably predict phosphorus concentrations in oligotrophic waters, which have phosphorus levels below 0.010 mg P/L (CCME 2004). To reduce the uncertainty around the prediction of phosphorus from mine rock drainage, kinetic test phosphorus concentrations used to calculate mine rock contact water loadings were revised to incorporate phosphorus concentrations measured by colorimetric methods, which had a detection limit of 0.01 mg/L (Appendix 8.I, Attachment 8.I.3; Appendix 8.II). For the coarse and fine PK, results of on-going additional testing initiated in 2010 (results from on-going humidity cell and saturated column tests of fine and coarse PK materials, respectively), were completed by ICP-MS and colorimetry with low detection levels (0.009 and 0.002 mg/L, respectively). Phosphorus concentrations from the geochemical testing were incorporated into water quality modeling to represent the quality of runoff and seepage water that contact with these materials that provide loading to Kennady Lake during operations and closure, and after closure (Appendix 8.I, Attachment 8.1.3).

- Mobilization of phosphorus from fine PK in the saturated column tests exhibited a rapid release from Weeks 10 to 26, reaching concentrations of 0.755 mg/L (Appendix 8.I, Attachment 8.I.3). After Week 26, phosphorus concentrations in leachate reduce to below 0.25 at Week 47. As steady state conditions are not yet achieved, input source terms for phosphorus are therefore conservative. On-going saturated column testing of current fine and coarse PK, and mine rock material, as well as new material sources, will refine phosphorus source terms for the water quality modelling.

Projections of end of winter dissolved oxygen concentrations in Kennady Lake using modelled long-term steady state phosphorus concentrations were derived from empirical relations that include Canadian lakes. The relationships provide a broad estimate of winter oxygen depletion rates (WODR), which vary in primary factors (i.e., summer phosphorus concentrations in the euphotic zone and average lake depth [Babin and Prepas 1985], annual primary productivity using an annual average total phosphorus concentration [Vollenweider 1979], and sediment oxygen demand based on trophic status [Mathias and Barica 1980]). There is some uncertainty around the application of empirical relationships to a lake system that has been disturbed and undergone a long period of recovery. However, as each approach accounted for baseline conditions, there is some confidence that the range of winter oxygen depletion rates estimated by the relationships provides an indication of the range of oxygen demand and end of winter dissolved oxygen levels for the projected phosphorus concentration (i.e., 0.018 mg/L). Higher winter oxygen depletion rates than estimated, or undersaturated dissolved oxygen conditions at the start of winter would lead to lower end of winter dissolved oxygen concentrations in Kennady Lake; this will result in lower dissolved oxygen levels in the surface waters and a greater extent of anoxia in the deeper waters at the end of winter. However, the presence of the open pits after closure will provide a region of the lake that will retain higher dissolved oxygen concentrations within their deeper epilimnetic zones than other areas of the lake through winter. On-going model development (i.e., mechanistic biological modelling) of Kennady Lake, including the open pits, and application of mitigation strategies to reduce the phosphorus loading, will be used to refine the dissolved oxygen predictions.

The approach and assumptions for contact water loading from mine rock and PK to runoff are consistent with the approaches used for other mine sites, such as at the Snap Lake Mine. However, the principal loading of a large number of dissolved metals for the modelling (including antimony, arsenic, cadmium, cobalt, chromium, copper, iron, lead, mercury, selenium, silver, and zinc) is based on contact water tests with a high number of results (>90%) below the detection limit (Appendix 8.II). Therefore, while some uncertainty exists around the predictions of these metals, there is a reasonable degree of confidence that the loading rate to the WMP will be lower than assumed.

Residual nitrogen loading in the water quality predictions are consistent with that used in other NWT diamond mines, and incorporates a mine production schedule as provided by the Project engineering team (Appendix 8.I). There is uncertainty in these loading predictions based on the possibility that blasting schedules, the amount of ammonium nitrate-fuel oil (ANFO) explosive required for blasting, and other factors (e.g., powder factors) may be altered during operation. In addition, other factors (e.g., powder factors) may be underestimated. De Beers is committed to updating modelled predictions of nitrogen loading to the WMP as blasting details are revised.

Simulated water quality in the downstream waterbodies did not include the development and persistence of permafrost conditions within the mine rock piles, the Coarse PK Pile, and the Fine PKC Facility. It was assumed that seepage quantities from these facilities would be representative of no permafrost conditions, and provide seasonal geochemical loading to Kennady Lake after closure. It is recognized that frozen layers may establish during the development of these facilities and that permafrost will likely continue to develop following closure, which will result in lower rates of seepage through the facilities and geochemical loading to Kennady Lake during operations and closure than simulated in this assessment. However, as the assessment of impacts to the suitability of the water quality to support aquatic life includes time periods that extend into the long-term (200 years), the assessment was designed to represent potential future climatic conditions where there would be no permafrost.

De Beers is committed to undertake regular monitoring and testing using standard field and laboratory procedures during the Project operation to evaluate water quality of components of the water management system (e.g., collection ponds), and the WMP. Where necessary, the water quality input profiles assigned to the loadings will be revised and Project effects will be confirmed. Where required, adaptive management strategies will be adopted.

8.15.4 Deposition of Dust and Metals to Lakes in the Kennady Lake Watershed

A simple mass balance calculation was used to predict changes in total suspended solids (TSS) and metal concentrations in lake water from deposition on the lake surface and within the watershed, for Area 8 and selected lakes in the Kennady Lake watershed. Changes in TSS and metal concentrations were calculated based on total suspended particulate (TSP) deposition rate and individual metal deposition rates, respectively, as predicted by air quality dispersion modelling (Section 11.4, Subject of Note: Air Quality).

A major source of uncertainty in the assessment of dust and metals deposition to lakes in and around the Project area relates to the air quality predictions (Section 11.4). The dispersion models used in the Air Quality assessment simplify the atmospheric processes associated with air mass movement and turbulence. This simplification limits the capability of a model to replicate discrete events and therefore introduces uncertainty. As a result of the uncertainty, dispersion models, coupled with their model inputs, are generally designed to conservatively model concentration and deposition values, so that practitioners can apply model results with the understanding that effects are likely to be over-estimated.

The following general comments are made with respect to air quality modelling results for this Project:

- Parameterization of emissions from diffuse area sources is difficult to simulate in dispersion models. Modelled results near mine pits and other sources of mechanically generated particulates are most uncertain. Most estimates of particulate emissions for mining activities are based on U.S. EPA emission factors. Many of these factors have limited applicability outside of the area in which they were developed (typically south-western United States coal mines). Based on experience, it is expected that emissions estimated using this approach would be conservative.
- The air quality and deposition rate predictions used the maximum emission rates from the Project during construction and operations associated with the development of the South and West Mine Rock Piles in Years 5 and 8. Predicted annual deposition rates were based on the maximum of the daily road dust emissions during summer and winter.
- Emissions of road dust from on-site haul roads, the primary sources of particulate matter and metal compounds, do not include potential mitigating effects of weather (such as precipitation or snow-covered

ground) which will result in an overestimate of annual air quality predictions and deposition rates.

- Geochemistry data used to estimate metal concentrations in dust included a large proportion of concentrations below the analytical detection limit for cadmium, mercury, selenium, and silver. Concentrations of these metals were set at the detection limit for air quality and deposition modelling.
- Based on a review of the particulate material monitoring data at the Snap Lake and Ekati mines, the elevated particulate matter deposition rates identified in this assessment are due in part to the conservative emission estimates.

The approach used to estimate incremental changes in concentrations of TSS and metals in surface waters using the modelled deposition rates was also conservative, for the following reasons:

- No retention of particulates or metals was assumed in lake watersheds, i.e., all deposited material was assumed to enter the lakes.
- Settling of suspended sediments in lakes was not incorporated.

As a result of these factors, predicted changes in TSS and metal concentrations in lakes are considered to be conservative estimates of the maximum potential changes that could occur during construction and operations.

De Beers is committed to undertake regular air quality testing using standard field and laboratory procedures during the Project operation to evaluate dust emissions and metals concentrations associated with dust. Where necessary, the water quality input profiles assigned to the loadings will be revised and Project effects will be confirmed. Where required, adaptive management strategies will be adopted to reduce the fugitive particulate matter emissions.

8.15.5 Time Required to Refill Kennady Lake

The time required to refill Kennady Lake has been estimated at 8 to 9 years. This estimate was derived from average flow conditions. If climatic conditions are drier than assumed at the time of refill, then the refill period may take longer, up to 20 years (Section 8.7). Conversely, if wetter conditions prevail during the refill period, it may be notably shorter, in the order of seven years.

A change in the filling time of Kennady Lake may alter the proportion of the different influent waters in the lake. Under drier conditions, the refilled system may contain a higher proportion of water originating from the upper watershed than from Lake N11, because the total water withdrawal from Lake N11 will be capped to ensure the maintenance of 1-in-5 dry year flows downstream of Lake N11.

Similarly, under wetter conditions, the proportions of the different influent waters may also vary from those that would occur under the assessed case. However, in both scenarios, the variation that may occur in the relative contribution of the different influent sources is unlikely to result in a change to the conclusions of the effects assessment. The water quality from both watersheds is similar. The time to full recovery would be longer, relative to the start of Project operations, if more than 12 years is required to refill the lake.

8.15.6 Time to Aquatic Ecosystem Recovery

A perfect analogue for Kennady Lake is not available, and the time required for the aquatic ecosystem in Kennady Lake to recover has been estimated from information presented in the available scientific literature. There is, as a result, some uncertainty in the estimated time quoted for full recovery (e.g., 50 to 60 years following the complete refilling of Kennady Lake for northern pike). Similarly, if habitat conditions are suitable for lake trout in the refilled lake, it is expected that this species will also require a long time to re-establish a stable, self-sustaining population (i.e., approximately 60 to 75 years following the complete refilling of Kennady Lake). The quoted range was developed using the longest recovery times noted in the literature (Section 8.11) and extending them to account for the fact that Kennady Lake is located in the sub-arctic. Arctic systems usually recover slower than temperate or tropical systems, because of colder temperatures, shorter growing seasons, and low nutrient availability. A longer recovery of Kennady Lake compared to temperate zone lakes remains likely due to Arctic climate-related factors. Because uncertainty is high, conservative assumptions were used in the estimation of the length of time for recovery, as described above. Consequently, there is a moderate degree of confidence that the length of time required for the ecosystem to recover is not underestimated. A moderate degree of confidence is the highest level that can be achieved in the assessment. The greatest uncertainty lies in the extent to which the abundance of each highly valued fish species returns to baseline values.

8.16 MONITORING AND FOLLOW-UP

8.16.1 Scope of Potential Monitoring Programs

Pursuant to the assessment approach outlined in the environmental impact statement (EIS) Section 6, three types of monitoring are planned, and they include the following:

- compliance inspection;
- follow-up monitoring; and
- effects monitoring.

Compliance inspection will consist of programs designed to confirm the implementation of approved design standards and the environmental design features described in the EIS.

Follow-up monitoring will consist of programs designed to verify key inputs to the effects analysis, such as the quality of the influent waters to the Water Management Pond (WMP; Areas 3 and 5), as well as monitoring compensation habitat to confirm the no net loss objective has been achieved. Results of follow-up monitoring will be used to reduce the level of uncertainty related to impact predictions.

Effects monitoring will involve programs focused on the receiving environment, with the objectives of verifying the conclusions of the EIS, evaluating the short-term and long-term effects on the physical, chemical and biological components of the aquatic ecosystem of Kennady Lake, estimating the spatial extent of effects, and providing the necessary input to adaptive management.

Follow-up monitoring and compliance inspection programs will be focused on the Gahcho Kué Project (Project) site, with little to no work occurring beyond the immediate Project area. Effects monitoring programs will encompass a larger area; however, they are unlikely to extend beyond Kirk Lake. Anticipated monitoring activities in the Kennady Lake watershed are described in this section.

There is no requirement for a cumulative effects monitoring program for aquatics, because the projected impacts of the Project on aquatics do not extend beyond the local study area. They do not, as a result, overlap with other regional projects (e.g., Snap Lake Mine).

8.16.2 Potential Monitoring Activities

8.16.2.1 Compliance Inspection

Compliance inspection by De Beers will verify that Project components are built to approved design standards and that environmental design features described in the EIS are incorporated. As each component of the Project is built, constructed features will be inspected to show that they comply with standard protocols, and that any variance from standard protocols has been completed with regulatory permission (as appropriate). A check list will also be developed to show that agreed-upon environmental design features are constructed as required. Compliance monitoring will extend throughout the life of the Project.

8.16.2.2 Follow-up Monitoring

Follow-up monitoring activities are expected to include water sampling in and around the South and West Mine Rock Piles, the Coarse PK Pile, the Fine Processed Kimberlite Containment (PKC) Facility, and other areas of the Project site to confirm the accuracy of the influent water quality profiles used to complete the effects assessment. Monitoring the progression of freezing within the external facilities will also be completed as part of this monitoring component.

8.16.2.3 Effects Monitoring

Effects monitoring programs will include a Surveillance Network Program (SNP) that focuses primarily on Project site operations as well as a more broadly focused Aquatic Effects Monitoring Program (AEMP). De Beers will develop the scope of the SNP and AEMP in consultation with regulators and interested parties. It is anticipated, however, that the AEMP will include water flow, water quality and sediment quality components, along with components focused on lower trophic communities (i.e., plankton and benthic invertebrates), fish and fish habitat. Sampling areas are likely to be located in the Kennady Lake watershed, potentially affected areas of the N watershed and the A, B, D, and E watersheds, Lake 410, and Kirk Lake, and a suitable reference lake. Components of the AEMP will be developed according to a common, statistically-based study design incorporating regulatory guidance and current scientific principles related to aquatic monitoring. Likely monitoring activities in the Kennady Lake watershed are described in this section.

Monitoring will also be conducted to evaluate the effectiveness of habitat compensation, and will include evaluation of both physical and biological characteristics. This monitoring will be critical to confirming that the no net loss objective has been achieved. The detailed monitoring plan will be included in the detailed No Net Loss Plan, and will be designed to meet all fish and fish habitat monitoring requirements included as conditions attached to regulatory authorizations, approvals or permits that may be issued for development of the Project.

The scope of the AEMP is expected to change over the life of the Project. In particular, monitoring in adjacent and downstream watersheds is expected to decline when operations cease. However, monitoring of Kennady Lake and the reference lake will be maintained during all phases of the Project.

Monitoring and sampling techniques, and analysis procedures, will be consistent with methods used during the baseline survey period to the extent possible. The field and laboratory processes will include the implementation of quality assurance/quality control measures for data acquisition, water and biota sampling, and analysis and reporting.

The assessment of data and information collected during the monitoring programs will be compiled into annual AEMP reports that will be submitted to the appropriate parties for review. Where necessary and appropriate, the results of other monitoring programs (e.g., groundwater monitoring) will be integrated into the AEMP reports.

8.16.2.4 Scope of the Aquatics Monitoring Programs

8.16.2.4.1 Construction and Operation

Potential monitoring in the Kennady Lake watershed during construction and operation is summarized below.

Hydrology

Monitoring of flows and water levels at key locations during construction and operation is considered necessary to determine actual runoff and discharge rates. Flow rates and water levels will be monitored during all phases of the Project at key lake outlets in the Kennady Lake watershed, specifically Area 8 and the A, B, D, and E watersheds. During construction and operation, continuous monitoring at the Area 8 outlet (Stream K5) will occur during dewatering over the open water period.

Hydrometric monitoring to provide measurements of lake levels and lake outlet discharges at key locations, including diversion channels at lake outlets, during open water conditions will be undertaken using hydrometric stations or gauging collection processes similar to those used as part of the baseline program (Section 8.3).

During the late season low flow period, in advance of the next season's spring thaw and freshet, observations will be undertaken to assess the integrity of the outlets and stream courses to monitor for the development of channel or bank erosion. Prior to the spring thaw (or snowmelt), snow surveys will be used to provide an early estimate of spring runoff. This is a reliable method to project annual watershed runoff volumes.

All piped and/or pumped discharges to lakes (e.g., to Area 8) will be monitored continuously.

Climate monitoring, including continuous measurements of rainfall and temperature, will be performed to allow validation of the hydrological model, assessment of seasonal conditions and to provide data for water management decision-making.

Water Quality

Water quality monitoring will focus on parameters monitored during baseline surveys and used as input variables through the modelling process, including pH, hardness, alkalinity, total organic carbon, total suspended solids, total dissolved solids, major ions, metals, nutrients (e.g., phosphorus), and selected organic parameters. Sampling points will include the WMP, the discharge zone in Area 8, the Area 8 outlet (Stream K5), the A, B, D, and E watersheds, and a suitable reference lake. Sediment sampling will be undertaken in the WMP and Area 8.

Sampling will occur on a seasonal basis (i.e., open water and under-ice conditions, at a minimum) to verify effect predictions related to changes in water quality and potential effects to aquatic health.

Fish and Fish Habitat

The fish and fish habitat monitoring program will be designed to obtain additional baseline information on watercourses and waterbodies that will be directly affected by the Project (i.e., permanent habitat losses), to determine if any effects to fish and fish habitat occur in watercourses and waterbodies directly (through changes in water quality) or indirectly (through changes in flow or water levels) affected by the Project, and to monitor the effectiveness of the development of compensation habitats. Fisheries data collected during fish salvages may also be used to complement data collected under the monitoring plan activities.

Monitoring will include phytoplankton, zooplankton, benthic invertebrates, and fish sampling of specific waterbodies in the Kennady Lake watershed and a reference lake. The frequency of sampling will be dependent on the trophic level.

8.16.2.4.2 Closure

The closure period is associated with the refilling of Kennady Lake, the re-connection of the B, D, and E watersheds and the removal of Dyke A. Throughout this period, the refilling of Kennady Lake will result in the continued reduction of downstream flows through Area 8. Natural refilling of Kennady Lake will be augmented by active pumping from Lake N11. Monitoring through this period is summarized below.

Hydrology

Flow rates and water levels will be monitored at lake outlets at key locations, specifically Area 8. Monitoring will occur on a seasonal basis at the Area 8 outlet (Stream K5).

During the drawdown of diverted lakes in the B, D, and E watersheds, lake water surface elevations and discharges from the lakes will be monitored until they are restored to pre-development levels. Re-established shorelines will be inspected on an annual basis until it is evident that shorelines are stable or until any required mitigation measures are implemented and shown to be effective.

Water Quality

Monitoring of Kennady Lake during refilling will test water quality predictions and once refilling is complete, provide a basis for measuring compliance with relevant applicable guidelines for the removal of Dyke A.

Water quality monitoring will focus on parameters monitored during the operation phase of the Project. Sampling points will include selected lakes in the upper watershed, the partially backfilled Hearne Pit and open Tuzo Pit, Areas 3 through 7, Area 8, and a reference lake. Additional physico-chemical water column profile monitoring in the Hearne and Tuzo pits will be conducted seasonally to monitor the extent of chemocline development.

Fish and Fish Habitat

Monitoring of phytoplankton, zooplankton, benthic invertebrates, and the fish community in the refilling Kennady Lake, in smaller lakes in the Kennady Lake watershed, and a reference lake will be required for the closure phase as summarized below:

- monitoring of spring spawning migrations and summer rearing densities of Arctic grayling in the Area 8 outlet (Stream K5);
- monitoring phytoplankton, zooplankton, benthic invertebrates, and forage fish in the refilling lake will be conducted to provide a basis to measure against the ecosystem recovery predictions for Kennady Lake. Monitoring will provide temporal trends and the information will also be useful to determine when Kennady Lake could support a piscivorous fish community to allow the removal of fish screens in the re-aligned B, D, and E watersheds, as well as Dyke A;
- monitoring fish migration in the channels of the restored B, D, and E watersheds will be conducted after the exclusion measures are removed to ensure fish movement between these watersheds and Kennady Lake. This program will include spring, summer, and fall sampling periods to document spring spawning migration, summer rearing success, and fall migration; and
- compensation habitats developed for the Project will be monitored for fish habitat and fish presence and abundance until the effectiveness of the compensation has been demonstrated.

8.16.2.4.3 Post-closure

After the removal of Dyke A, the upper Kennady Lake watershed and Areas 3 through 7 will be reconnected to Area 8 and downstream waterbodies. Anticipated post-closure monitoring is summarized below.

Hydrology

Hydrological monitoring of the reconnected watershed will occur at similar sites selected during the baseline surveys. Monitoring is expected to be less frequent than during operations or closure, and will only persist for several years after the removal of Dyke A. The primary purpose of this monitoring will be to determine that the post-closure watershed hydrology is consistent with pre-development conditions, taking into account the modified watershed and lake areas.

Water Quality

Water quality monitoring will focus on parameters monitored during the operation phase of the Project. Sampling points in Kennady Lake during post-closure will include the partially backfilled Hearne Pit and open Tuzo Pit basins, Areas 3 through 7, Area 8, and a reference lake. Additional physico-chemical water column profile monitoring will be conducted seasonally in the Hearne and Tuzo pits to monitor the seasonal regime of meromictic conditions and the extent of chemocline development.

Sampling may occur on a less frequent basis than during operations and closure, but will maintain a seasonal basis (i.e., open water and under-ice conditions). Monitoring would be expected to continue, until water quality conditions are consistent with the surrounding environment or are on a predictable trajectory to that endpoint. Sampling will also be conducted in a reference lake to provide a comparison with background temporal trends.

Fish and Fish Habitat

Monitoring will include phytoplankton, zooplankton, benthic invertebrates, and fisheries sampling in the refilled Kennady Lake, the Kennady Lake watershed, and a reference lake. Monitoring in the refilled Kennady Lake will focus on changes to fish and fish habitat resulting from changes in nutrient levels and trophic change.

These monitoring programs will persist until it is determined that fish and lower trophic communities of the Kennady Lake watershed have reached applicable recovery thresholds as determined by Fisheries and Oceans Canada (DFO) and other interested parties, and that the compensation habitats are considered to be effective and habitat compensation and confirming the predicted recovery processes and timing.

8.17 REFERENCES

Agbeti, M.D. and J.P. Smol. 1995. Winter Limnology: A Comparison of Physical, Chemical and Biological Characteristics in Two Temperate Lakes during Ice Cover. *Hydrobiologia*. 304: 221-234.

Alberta Environment. 2001. Guide to the Code of Practice for Watercourse Crossings, including Guidelines for Complying with the Code of Practice. Alberta Environment Pub. No. I/8422. 29 p.

Allan, C.J., N.T. Roulet and A.R. Hill. 1993. The Biogeochemistry of Pristine, Headwater Precambrian Shield Watersheds: An Analysis of Material Transport within a Heterogeneous Landscape. *Biogeochemistry.* 22: 37-79. Cited in Steedman, R.J., C.J. Allan, R.L. France and R.S. Kushneriuk. 2004. Land, Water, and Human Activity on Boreal watersheds. In Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment. Gunn, J.M., R.J. Steedman and R.A. Ryder, (eds). Lewis Publishers. CRC Press. 2004. *pp*. 59-85.

Alt, K. and R. Furniss. 1976. Inventory of Cataloguing of North Slope Waters. Alaska Department of Fish and Game. Federal Aid in Fish Restoration, Annual Report of Progress 17(F-9-8): 129-150.

AMEC (AMEC Earth & Environmental, A Division of AMEC Americas Limited). 2004a. Unpublished Water Chemistry Data Collected in Kennady Lake and Surrounding Watersheds (2004). Calgary, AB.

AMEC. 2004b. Unpublished Aquatic Resources Field Data Collected in Kennady Lake and Surrounding Watersheds (2004). Calgary, AB.

AMEC. 2005a. Unpublished Water Chemistry Data Collected in Kennady Lake and Surrounding Watersheds (2005). Calgary, AB.

AMEC. 2005b. Unpublished Aquatic Resources Field Data Collected in Kennady Lake and Surrounding Watersheds (2005). Calgary, AB.

AMEC. 2008. Gahcho Kué Project Description. Prepared for De Beers Canada Ltd.

Andersson, P., H. Borg and P. Karrhage. 1995. Mercury in Fish in Acidified and Limed Lakes. *Water, Air, and Soil Pollution*. 80: 889-892.

Andrén, H. 1994. Effects of Habitat Fragmentation on Birds and Mammals in Landscape with Different Proportions of Suitable Habitat- A Review. *Oikos.* 71: 355-366.

Andrén, H. 1999. Habitat Fragmentation, the Rand Sample Hypothesis, and Critical Thresholds. *Oikos.* 84: 306-308.

Avery, E.L. 1978. The Influence of Chemical Reclamation on a Small Brown Trout Stream in Southwestern Wisconsin. Wisconsin Department of Natural Resources Technical Bulletin N. 110, 35p.

Babin, J. and E.E. Prepas. 1985. Modelling Winter Oxygen Depletion Rates in Ice-Covered Temperate Zone Lakes in Canada. *Canadian Journal of Fisheries Aquatic Science*. 42: 239-249.

Baudouin, M.F. and P. Scoppa. 1974. Acute Toxicity of Various Metals to Freshwater Zooplankton. *Bulletin of Environmental Contamination and Toxicology.* 12:745-751.

B.C. (British Columbia) Ministry of Forests. 2002. Fish-stream Crossing Guidebook. Forest Practices Branch, BC Ministry of Forests, Victoria, BC.

Beadle, L.C. 1969. Osmotic Regulation and Adaptation of Freshwater Animals to Inland Saline Waters. *Verhandlungen des Internationalen Verein Limnologie.* 17: 421-429. Cited in Bierhuizen, J.F.H. and E.E. Prepas. 1985. Relationship Between Nutrients, Dominant Ions and Phytoplankton Standing Crop in Prairie Saline Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 42:1588-1594.

Bierhuizen, J.F.H. and E.E. Prepas. 1985. Relationship Between Nutrients, Dominant Ions and Phytoplankton Standing Crop in Prairie Saline Lakes. *Canadian Journal of Fisheries Aquatic Science.* 42: 1588-1594.

Biesinger, K.E. and G.M. Christensen. 1972. Effects of Various Metals on Survival, Growth, Reproduction, and Metabolism of *Daphnia magna*. *Journal of the Fisheries Resource Board of Canada*. 29:1691-1700.

Binns, N. A. 1967. Effects of Rotenone Treatment on the Fauna of the Green River, Wyoming. Fisheries Research Bulletin No. 1, Wyoming Game and Fish Commission, Cheyenne.

Birge, W.J., J.A. Black, A.G. Westerman and J.E. Hudson. 1979. *Aquatic Toxicity Tests on Inorganic Elements Occurring in Oil Shale*. In C. Gale (ed.). *Oil Shale Symposium: Sampling, Analysis and Quality Assurance*. United States Environmental Protection Agency. Cincinnati, OH, USA. pp. 519-534.

Birtwell, I.K. and J.S. Korstrom. 2002. A Commentary on Aquatic Degradation in Alberni Inlet, British Columbia, and Consequences to Juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) and Adult Sockeye Salmon (*Oncorhynchus nerka*). In *Cumulative Environmental Effects Management: Tools and Approaches.* A.J. Kennedy (Ed). A symposium held by the Alberta Society of Professional Biologists, Calgary, AB, November 2000. Alberta Society of Professional Biologists, Edmonton, AB. pp. 441-453.

Birtwell, I.K., R. Fink, D. Brand, R. Alexander and C.D. McAllister. 1999. Survival of Pink Salmon (*Oncorhynchus gorbuscha*) Fry to Adulthood Following a 10-d Exposure to Aromatic Hydrocarbon Water Soluble Fraction of Crude Oil and Release to the Pacific Ocean. *Canadian Journal of Fisheries Aquatic Science*. 56:2087-2098.

Birtwell, I.K., S.C. Samis and N.Y. Khan. 2005. Commentary on the Management of Fish Habitat in Northern Canada: Information Requirements and Policy Considerations Regarding Diamond, Oil Sands, and Placer Mining. Canadian Technical Report of Fisheries and Aquatic Sciences. No. 2606.

Bjornberg, A., L. Hakanson and K. Lundbergh. 1988. A Theory on the Mechanisms Regulating the Bioavailability of Mercury in Natural Waters. *Environmental Pollution*. 49: 53-61.

Bloom, N.S. 1992. On the Chemical Form of Mercury in Edible Fish and Marine Invertebrate Tissue. *Canadian Journal of Fisheries Aquatic Science*. Sci. 49:1010-1017.

Bodaly, R.A. and K.A. Kidd. 2004. Mercury Contamination of Lake Trout Ecosystems. In: *Boreal Shield Ecosystems: Lake Trout Ecosystems in a Changing Environment*. J.M. Gunn, R.J. Steedman, and R.A. Ryder (Eds). Lewis Publishers, Boca Raton, FL.

Bodaly, R.A. and L.F.W. Lesack. 1984. Response of a Boreal Northern Pike (*Esox lucius*) Population to Lake Impoundment: Wupaw Bay, Southern Indian Lake, Manitoba. *Canadian Journal of Fisheries and Aquatic Sciences*. 41: 706-714.

Bodaly, R.A. and R.J.P. Fudge. 1999. Uptake of Mercury by Fish in an Experimental Boreal Reservoir. *Archives of Environmental Contamination and Toxicology.* 37: 103-109.

Bodaly, R.A., R.E. Hecky and R.J.P. Fudge. 1984. Increases in Fish Mercury Levels in Lakes Flooded by the Churchill River Diversion, Northern Manitoba. *Canadian Journal of Fisheries and Aquatic Sciences*. 41: 682-691.

Bodaly, R.A., J.W.M. Rudd, R.J.P. Fudge and C.A. Kelly. 1993. Mercury Concentrations in Fish Related to Size of Remote Canadian Shield Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 50: 980-987.

Bodaly, R.A., V.L. St. Louis, M.J. Paterson, R.J.P. Fudge, B.D. Hall, D.M. Rosenberg and J.W.M. Rudd. 1997. Bioaccumulation of Mercury in the Aquatic Food Chain in Newly Flooded Areas. In *A. Sigel and H. Sigel (eds.) Metal Ions in Biological Systems. Vol. 34.* Mercury and Its Effects on Environmental Biology. Marcel Dekker, Inc. pp. 259-287.

Brannock, P., M.S. Stekoll, B. Failor and I. Wang. 2002. *Salt and Salmon: The Effects of Hard Water Ions on Fertilization*. Aquatic Sciences Meeting of the American Society of Limnology and Oceanography. Cited in Weber-Scannell P.K. and L.K. Duffy. 2007. Effects of Total Dissolved Solids on Aquatic Organisms: A Review of Literature and Recommendation for Salmonid Species. *American Journal of Environmental Sciences.* 3:1-6.

Brix, K.V., R. Gerdes, N. Curry, A. Kasper and M. Grosell. 2010. The Effects of Total Dissolved Solids on Egg Fertilization and Water Hardening in Two Salmonids – Arctic Grayling (*Thymallus arcticus*) and Dolly Varden (*Salvelinus malma*). *Aquatic Toxicology.* 97:109-115.

Brouard, D., C. Demers, R. Lalumiere, R. Schetagne and R. Verdon. 1990. Evolution of Mercury Levels in Fish of the La Grande Hydroelectric Complex, Quebec (1978-1989). Summary Report, Montreal, Quebec: Vice-Presidence Environnement, Hydro-Quebec and Goupe Environnement Shooner, inc. 97 p.

Brown, J.R. 1970. Permafrost in Canada. University of Toronto Press.

Brown, G.W. and J.T. Krygier. 1971. Clear-cut Logging and Sediment Production in the Oregon Coast Range. *Water Resources Research.* 7:1189-1198.

Burger, J. 1998. Fishing and Risk along the Savannah River: Possible Intervention. *Journal of Toxicology and Environmental Health*. 55:405-419.

Cabana, G. and J.B. Rasmussen. 1994. Modelling Food Chain Structure and Contaminant Bioaccumulation Using Stable Nitrogen Isotopes. *Nature*. 372:255-257.

Cabana, G., A. Tremblay, J. Kalff, and J.B. Rasmussen. 1994. Pelagic Food Chain Structure in Ontario Lakes: A Determinant of Mercury Levels in Lake Trout (*Salvelinus namaycush*). *Canadian Journal of Fisheries and Aquatic Sciences*. 51:381-389.

Campbell, P.G., B. Bobee, A. Caille, M.J. Demalsy, and P. Demalsy. 1975. Pre-impoundment Site Preparation: A Study of the Effects of Topsoil Stripping on Reservoir Water Quality. *Verhandlungen, Internationale Vereinigung fur Theoretische und Angewandte Limnologie*. 19: 1768-1777 (as cited in Northcote and Atagi 1997).

Canamera Geological Ltd. 1998. 1996 Environmental Baseline Studies: 5034 Diamond Project. Prepared by the Environmental Resources Division of Canamera Geological. Submitted to Monopros Ltd., Yellowknife, NT.

Cantrell, M.A. and A.J. McLachlan. 1977. Competition and Chironomid Distribution Patterns in a Newly Flooded Lake. *Oikos*. 29:429-433.

Carpenter, S.R. 1989. Temporal Variance in Lake Communities: Blue-green Algae and the Tropic Cascade. *Landscape Ecology.* 3: 175-184.

Carpenter, S.R., J.F. Kitchell and J.R. Hodgson. 1985. Cascading Trophic Interactions and Lake Productivity. *BioScience.* 35:634-639.

Carpenter, R.S., J.J. Cole, J.R. Hodgson, J.F. Kitchell, M.L. Pace, D. Bade, K.L. Cottingham, T.E. Essington, J.N. Houser and D.E. Schindler. 2001. Trophic Cascades, Nutrients, and Lake Productivity: Whole Lake Experiments. *Ecological Monographs.* 71: 163-186.

Casselman, J.M. 1995. Survival and Development of Lake Trout Eggs and Fry in Eastern Lake Ontario – *in situ* Incubation, Yorkshire Bar, 1989-1993. *Journal of Great Lakes Research*. 21(Suppl. 1): 384-399.

Casselman, J.M. and C.A. Lewis. 1996. Habitat Requirements of Northern Pike (*Esox lucius*). *Canadian Journal of Fisheries and Aquatic Sciences*. 53(Suppl. 1):163-174.

CCME (Canadian Council of Ministers of the Environment). 1999 (with updates to 2010). *Canadian Environmental Quality Guidelines*. Canadian Council of Ministers of the Environment. Winnipeg, MB.

CCME. 2002. Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Canadian Council of Ministers of the Environment, 1999, updated 2001, updated 2002. Winnipeg, MB.

CCME. 2003. Environmental Code of Practice for Above-Ground Storage Tanks Systems Containing Petroleum Products. Canadian Council of Ministers of the Environment. Winnipeg, MB.

CCME. 2004. Phosphorus: Canadian Guidance Framework for the Management of Freshwater Systems. Canadian Council of Ministers of the Environment. Winnipeg, MB.

CCME. 2006. Canadian Water Quality Guidelines for the Protection of Aquatic Life. Canadian Council of Ministers of the Environment. Winnipeg, MB.

CCME. 2007. Summary Table, Canadian Environmental Quality Guidelines for the Protection of Aquatic Life (Updated July 2006). Canadian Council of Ministers of the Environment. Winnipeg, MB.

Chapman, P.M., H. Bailey and E. Canaria. 2000. Toxicity of Total Dissolved Solids Associated with Two Mine Effluents to Chironomid Larvae and Early Life Stages of Rainbow Trout. *Environmental Toxicology and Chemistry.* 19:210-214.

Chen, L.C. 1969. The Biology and Taxonomy of the Burbot, *Lota lota leptura*, in Interior Alaska. Biological Papers of the University of Alaska. 11p.

Chen, Y.W. and N. Belzile. 2001. Antagonistic Effect of Selenium on Mercury Assimilation by Fish Populations near Sudbury Metal Smelters? *Limnology and Oceanography.* 46(7): 1814-1818.

Christie, W.J. 1972. Lake Ontario: Effects of the Exploitation, Introductions, and Eutrophication on the Salmonid Community. *Journal of the Fisheries Research Board Canada.* 29:913-929.

Clark, B.J., P.J. Dillon and L.A. Molot. 2004. Lake Trout (*Salvelinus namaycush*) Habitat Volumes and Boundaries in Canadian Shield Lakes. *In Boreal Shield Waters: Lake Ecosystems in a Changing Environment.* J.M. Gunn, R.J. Steedman, and R.A. Ryders (Eds). Lewis Publishers, New York. Pp 111-117.

Clarke, K.D., R. Knoechel and P.M. Ryan. 1997. Influence of Trophic Role and Life-cycle Duration on Timing and Magnitude of Benthic Macroinvertebrate Response to Whole-lake Enrichment. *Canadian Journal of Fisheries and Aquatic Sciences*. 54:89-95.

Colby, P.J., G.R. Spangler, D.A Hurley and A.M. McCombie. 1972. Effects of Eutrophication on Salmonid Communities in Oligotrophic Lakes. *Journal of the Fisheries Research Board Canada*. 29:975-983.

Cornet, J.J. and F.J. Rigler. 1979. Hypolimnetic Oxygen Deficits: Their Prediction and Interpretation. *Science*. 205:580-581.

Cott, P. and J.P. Moore. 2003. Working Near Water, Considerations for Fish and Fish Habitat. Reference and Workshop Manual. Northwest Territories Department of Fisheries and Oceans - Western Arctic Area. Inuvik, Northwest Territories. 92 p + appendices.

Cowgill, U.M. and Milazzo, D.P. 1990. The sensitivity of two cladocerans to water quality variables: salinity and hardness. *Archiv fur Hydrobiologie.* 120: 185-196.

Crawford, P.J. and D.M. Rosenberg. 1984. Breakdown of Conifer Needle Debris in a New Northern Reservoir, Southern Indian Lake, Manitoba. *Canadian Journal of Fisheries and Aquatic Sciences*. 41: 649-658.

Cumberland Resources Ltd. 2005. Aquatic Ecosystem/Fish Habitat Impact Assessment for the Meadowbank Gold Project.

Danylchuk, A.J. and W.M. Tonn. 2003. Natural Disturbances and Fish: Local and Regional Influences on Winterkill of Fathead Minnows in Boreal Lakes. *Transactions of the American Fisheries Society*. 132:289-298.

Danell, K. and K. Sjoberg. 1982. Successional Patterns of Plants, Invertebrates and Ducks in a Man-made Lake. *Journal of Applied Ecology*. 19:395-409.

Dave, G. 1984. Effects of Waterborne Iron on Growth, Reproduction, Survival and Haemoglobin in *Daphnia magna*. *Comparative Biochemistry and Physiology.* 78C:433-438.

De Beers (De Beers Canada Mining Inc.). 2002. Snap Lake Diamond Project Environmental Assessment Report. Prepared for De Beers Canada Mining Inc. by Golder Associates Ltd. Yellowknife, N.W.T. February 2002.

De Beers. 2010. 2009 Annual Report in Support of the Aquatic Effects Monitoring Program for the Snap Lake Project. Water License (MV2001L2-0002). Submitted to the Mackenzie Valley Land and Water Board.

De Groot, C.J. and C. Van Wijck. 1993. The Impact of Desiccation of a Freshwater Marsh (Garcines Nord, Camargue, France) on Sediment-Water-Vegetation Interactions. *Hydrobiologica*. 252: 83-94 (as cited in McGowan et al. 2005).

DesLandes, J-C., S. Guénette, Y. Prairie, D. Roy, R. Verdon and R. Fortin. 1995. Changes in Fish Populations Affected by the Construction of the La Grande Complex (phase 1), James Bay region, Québec. *Canadian Journal of Zoology.* 73:1860-1877.

Devito, K.J., P.J. Dillon and B.D. Lazerte. 1989. Phosphorus and Nitrogen Retention in Five Precambrian Shield Wetlands. *Biogeochemistry* 8:185-204. Cited in: Steedman, R.J., C.J. Allan, R.L. France and R.S. Kushneriuk. 2004. Land, Water, and Human Activity on Boreal Watersheds. In Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment. Gunn, J.M., R.J. Steedman and R.A. Ryder, (eds). Lewis Publishers. CRC Press. 2004.

DFO (Department of Fisheries and Oceans). 1986. Policy for the Management of Fish Habitat. Department of Fisheries and Oceans. Ottawa, ON.

DFO. 1995. Freshwater Intake End-of-Pipe Fish Screen Guideline. Communications Directorate, Department of Fisheries and Oceans, Ottawa, ON. 26 p.

DFO. 1998. Guidelines for the Protection of Fish and Fish Habitat: the placement and design of large culverts. A report prepared by Fisheries and Oceans Canada, Maritimes Region. Final draft April 1, 1998.

DFO. 2006. Practitioners Guide to Habitat Compensation for DFO Habitat Management Staff. Version 1.1 Updated December 4, 2006.

Diavik (Diavik Diamond Mines Inc.). 1998. Environmental Assessment Report. Diavik Diamond Mines Inc. Yellowknife, NT.

Diavik. 2006. Draft Ammonia Management Plan. Diavik Diamond Mines Inc. Yellowknife, NT.

Diavik. 2011. 2010 AEMP Annual Report for the Diavik Diamond Mine, NWT. Submitted to the Wek'èezhii Land and Water Board. Yellowknife, NWT

Dillon, P.J., B.J. Clark and H.E. Evans. 2004. The Effects of Phosphorus and Nitrogen on Lake Trout (*Salvelinus namaycush*) Production and Habitat. In Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment. Gunn, J.M., R.J. Steedman and R.A. Ryder, (eds). Lewis Publishers. CRC Press. 2004. *pp*. 119-131.

Dillon, P.J., B.J. Clark, L.A. Molot and H.E. Evans. 2003. Predicting the Location of Optimal Habitat Boundaries for Lake Trout (*Salvelinus namaycush*) in Canadian Shield Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*. 60:959-970.

Dinsmore, W.P., G.J. Scrimgeour and E.E Prepas. 1999. Empirical Relationships between Profundal Macroinvertebrate Biomass and Environmental Variables in Boreal Lakes of Alberta, Canada. *Freshwater Biology.* 41:91-100.

Dmytriw, R., A. Mucci, M. Lucotte and P. Pichet. 1995. The Partitioning of Mercury in the Solid Components of Dry and Flooded Forest Soils and Sediments from a Hydroelectric Reservoir, Quebec (Canada). *Water, Air, and Soil Pollution*. 80: 1099-1103.

Driscoll, C.T., C. Yan, C.L. Schofield, R. Munson and J. Holsapple. 1994. The Mercury Cycle and Fish in the Adirondack Lakes. *Environmental Science and Techoloogy.* 28:137-143.

Downing, J.A., S. Watson and E McCauley. 2001. Predicting Cyanobacteria Dominance in Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 58:1905-1908.

EBA (EBA Engineering Consultants Ltd.). 2002. Gahcho Kué Winter 2001 Water Quality Sampling Program, Gahcho Kué, NWT, Project No. 0701-98-13487.028. Prepared for De Beers Canada Inc. Yellowknife, NWT.

EBA. 2003. Kennady Lake Winter 2002 Water Quality Sampling Programme Kennady Lake NWT. Project # 0701- 98- 13487.035. Prepared for De Beers Canada Inc. Yellowknife, NWT.

EBA. 2004a. Kennady Lake Winter 2003 Water Quality Sampling Program, Project No. 0701-98-13487.048. Prepared for De Beers Canada Inc. Yellowknife, NWT.

EBA. 2004b. Faraday Lake Winter 2003 Water Quality Sampling Program, Project No. 0701-98-13487.048. Prepared for De Beers Canada Inc. Yellowknife, NWT.

EBA. 2004c. Kelvin Lake Winter 2003 Water Quality Sampling Program, Project No. 0701-98-13487-048. Prepared for De Beers Canada Inc. Yellowknife, NWT.

EBA. 2004d. Kennady Lake (Winter 2004) Water Quality Sampling Program, Project # 1740071.001. Prepared for De Beers Canada Inc. Yellowknife, NWT.

EBA (EBA Engineering Consultants Ltd.) and Jacques Whitford Environment Ltd. 2000. Gahcho Kué (Kennedy Lake) Environmental Studies 1999. Submitted to Monopros Ltd., Yellowknife, NT.

EBA and Jacques Whitford Environment Ltd. 2001. Gahcho Kué (Kennady Lake) Environmental Baseline Investigations (2000). Submitted to De Beers Canada Exploration Limited, Yellowknife, NT.

EBA and Jacques Whitford Environment Ltd. 2002. De Beers Canada Exploration Gahcho Kué Fisheries Studies (2001). Submitted to De Beers Canada Exploration Limited, Yellowknife, NT.

Edmunds, G.F. Jr, S.L. Jensen and L. Berner. 1976. The Mayflies of North and Central America. University of Minnesota Press. Minneapolis, 330 pp (as cited by Voshell and Simmons 1984).

Elser, J.J., E. Marzolf and C.R. Goldman. 1990. The Roles of Phosphorus and Nitrogen in Limiting Phytoplankton Growth in Freshwaters: A Review of Experimental Enrichments. *Canadian Journal of Fisheries and Aquatic Sciences*. 47:1468-1477.

English, M.C. 1984. Implications of Upstream Impoundment on the Natural Ecology and Environment of the Slave River Delta, Northwest Territories. *Northern Ecology and Resource Management*. Pages 311-339.

Environment Canada. 1997. Canadian Acid Rain Assessment, Volume Three. Aquatic Effects. Jeffries, D.S. (eds.). Aquatic Ecosystems Conservation Branch.

Environment Canada. 2004. Canadian Guidance Framework for the Management of Phosphorous in Freshwater Systems. Ecosystem Health: Science-based Solutions Report No. 1-8. National Guidelines and Standards Office, Water Policy and Coordination Directorate, Environment Canada, pp. 114.

Environment Canada. 2005. Historical Adjusted Climate Database for Canada. http://www.ccma.bc.ec.gc.ca/hccd/

Eriksen, C.H. 1975. Physiological Ecology and Management of the Rare “Southern” Grayling *Thymallus articus tricolor* Cope. *Verhandlungen des Internationalen Verein Limnologie.* 19:2448-2455.

Evans, D.O., J.J. Houston and G.N. Meredith. 1988. Status of the Lake Simcoe Whitefish, *Coregonus clupeaformis*. *Canadian Field-Naturalist*. 102:103-113.

Evans, D.O., J. Brisbane, J.M. Casselman, K.E. Coleman, C.A. Lewis, P.G. Sly, D.L. Wales and C.C. Willox. 1991. Anthropogenic Stressors and Diagnosis of their Effects on Lake Trout Populations in Ontario lakes. Lake Trout Synthesis Response to Stress Working Group. Manuscript Report, Ontario Ministry of Natural Resources. Toronto, Ontario. 115 p.

Evans, D.O., K.H. Nicholls, Y.C. Allen and M.McMurtry. 1996. Historical Land Use, Phosphorus Loading, and Loss of Fish Habitat in Lake Simcoe, Canada. *Canadian Journal of Fisheries and Aquatic Sciences.* 53(Suppl. 1):194-218.

Evans, D.O. and P. Waring. 1987. Changes in the Multispecies, Winter Angling Fishery of Lake Simcoe, Ontario, 1961-83: Invasion by Rainbow Smelt, *Osmerus mordax*, and the Roles of Intra- and Interspecific Interactions. *Canadian Journal of Fisheries and Aquatic Sciences.* 44 (Suppl. 2): 182-197.

Evans, C.L., J.D. Reist and C.K. Minns. 2002. Life History Characteristics of Freshwater Fishes Occurring in the Northwest Territories and Nunavut, with Major Emphasis on Riverine Habitat Requirements. *Canadian Manuscript Report of Fisheries and Aquatic Sciences*. 2614: xiii + 169 p.

Evans, D.O. 2005. Effects of Hypoxia on Scope-for-Activity of Lake Trout: Defining a New Dissolved Oxygen Criterion for Protection of Lake Trout Habitat. Ontario Ministry of Natural Resources, Aquatic Research and Development Section, Applied Research and Development Branch, Peterborough, ON. Tech. Rep. 2005-01.

Evans, R.D. 1986. Sources of Mercury Contamination in the Sediments of Small Headwater Lakes in South-central Ontario, Canada. *Archives of Environmental Contamination and Toxicology.* 15:505-512.

Fahrig, L. 1997. Relative Effects of Habitat Loss and Fragmentation on Population Extinction. *Journal of Wildlife Management.* 61:603-610.

Faulker, S.G., W.M. Tonn, M. Welz and D.R. Schmitt. 2006. Effects of Explosives on Incubating Lake Trout Eggs in the Canadian Arctic. *North American Journal of Fisheries Management.* 26:833-842.

Fee, E.J., R.E. Hecky, S.E.M. Kasian and D.R. Cruickshank. 1996. Effects of Lake Size, Water Clarity, and Climatic Variability on Mixing Depths in Canadian Shield Lakes. *Limnology and Oceanography.* 41:912-920.

Fitzpatrick, E.A. 1995. Arctic Soils and Permafrost. *Ecology of Arctic Environments.* Special Publication Series of the British Ecological Society. No. 13, pp 1-40.

Fitzsimons, J.D. 1996. The Significance of Man-made Structures for Lake Trout Spawning in the Great Lakes: Are they a Viable Alternative to Natural Reefs? *Canadian Journal of Fisheries and Aquatic Sciences.* 53(Suppl. 1):142-151.

Ford, B.S., P.S. Higgins, A.F. Lewis, K.L. Cooper, T.A. Watson, C.M. Gee, G.L. Ennis and R.L. Sweeting. 1995. Literature Reviews of the Life History, Habitat Requirements, and Mitigation/Compensation Strategies for Thirteen Sport Fish Species in the Peace, Liard, and Columbia River Drainages of British Columbia. *Canadian Manuscript Report of Fisheries and Aquatic Sciences* 2321.

Fritz, S.K. and P. Frape. 1987. Geochemical Trends for Groundwaters from the Canadian Shield. In: Fritz, P. and Frape, S.K. (eds.) *Saline Water and Gases in Crystalline Rocks.* Geological Association of Canada Special Paper 33 pp.

Fudge, R.J.P. and R.A. Bodaly. 1984. Postimpoundment Winter Sedimentation and Survival of Lake Whitefish (*Coregonus clupeaformis*) Eggs in Southern Indian Lake, Manitoba. *Canadian Journal of Fisheries and Aquatic Sciences*. 41: 701-705.

Gahcho Kué Panel. 2007. Terms of Reference for the Gahcho Kué Environmental Impact Statement. Mackenzie Valley Environmental Impact Review Board. Yellowknife, N.W.T. October 5, 2007.

Gascon, D. and W.C. Leggett. 1977. Distribution, Abundance, and Resource Utilization of Littoral Zone Fishes in Response to a Nutrient/Production Gradient in Lake Memphremagog. *Journal of the Fisheries Research Board Canada.*34:1105-1117.

Gazey, W.J. and M.J. Staley. 1986. Population Estimation from Mark-Recapture Experiments Using a Sequential Bayes Algorithm. *Ecology*. 67:941-951.

Geraldes, A.M. and M.J. Boavida. 1999. Limnological Comparison of a New Reservoir with One Almost 40 Years Old Which Had Been Totally Emptied and Refilled. *Lakes and Reservoirs: Research and Management*. 4:15-22.

GNWT (Government of the Northwest Territories). Working Group on General Status of NWT Species. 2006. NWT Species 2006-2010 - General Status Ranks of Wild Species in the Northwest Territories, Department of Environment and Natural Resources, Government of the Northwest Territories, Yellowknife, NT. III pp.

Godard, D.R., L. Peters, R. Evans, K. Wautier, P.A. Cott, B. Hanna and V. Palace. 2008. Development of Histopathology Tools to Assess Instantaneous Pressure Change-induced Effects in Rainbow Trout (*Oncorhynchus mykiss*) Early Life Stages. Environmental Studies Research Funds Report #164. Winnipeg. 93 p.

Golder. 2010. *Plankton Report in Support of the 2009 AEMP Annual Report for the Diavik Diamond Mine, NWT*. Prepared for Diavik Diamond Mines Inc. Yellowknife, Northwest Territories.

Goldyn, R., T. Joniak, K. Kowalczewska-Madura and A. Kozak. 2003. Trophic State of a Lowland Reservoir During 10 Years After Restoration. *Hydrobiologia*. 506-509: 759-765.

Golosov, S., O.A. Maher, E. Schipunova, A. Terzhevik, G. Zdorovennova, and G. Kirillin. 2007. Physical Background of the Development of Oxygen Depletion in Ice-covered Lakes. *Oecologia* 151: 331-340.

Goodfellow, W.L., L.W. Ausley, D.T. Burton, D.L. Denton, P.B. Dorn, D.R. Grothe, M.A. Heber, T.J. Norberg-King and J.H. Jr. Rodgers. 2000. Major Ion Toxicity in Effluents: A Review with Permitting Recommendations. *Environmental Toxicology and Chemistry.* 19:175-182.

Gordon, C., J.M. Wynn and S.J. Woodings. 2001. Impacts of Increased Nitrogen Supply on High Arctic Heath: The Importance of Bryophytes and Phosphorous Availability. *The New Phytologist*. 149: 461-471.

Gothberg, A. 1983. Intensive Fishing – a Way to Reduce the Mercury Level in Fish. *Ambio*. 13: 259-261.

Government of Canada. 1992. *Transportation of Dangerous Goods Act, 1992*. Transport Canada. S.C., 1992, c. 34. Current to July 12, 2009.

Greenbank, J. 1945. Limnological Conditions in Ice-covered Lakes, Especially as Related to Winter-kill of Fish. *Ecological Monographs.* 15: 343-392.

Greenfield, B.K., T.R. Hrabik, C.J. Harvey and S.R. Carpenter. 2001. Predicting Mercury Levels in Yellow Perch: Use of Water Chemistry, Trophic Ecology, and Spatial Traits. *Canadian Journal of Fisheries and Aquatic Sciences.* 58: 1419-1428.

Grieb, T.M., C.T. Driscoll, S.P. Gloss, C.L. Schofield, G.L. Bowie, and D.B. Porcella. 1990. Factors Affecting Mercury Accumulation in Fish in the Upper Michigan Peninsula. *Environmental Toxicology and Chemistry.* 9: 919-930.

Grondin, A., M. Lucotte, A. Mucci and B. Fortin. 1995. Mercury and Lead Profiles and Burdens in Soils of Quebec (Canada) Before and After Flooding. *Canadian Journal of Fisheries and Aquatic Sciences.* 52: 2493-2506.

Guildford, S.J., F.P. Healey and R.E. Hecky. 1987. Depression of Primary Production by Humic Matter and Suspended Sediment in Limnocorral Experiments at Southern Indian Lake, Northern Manitoba. *Canadian Journal of Fisheries and Aquatic Sciences*. 44:1408-1417

Gunn, J.M. 1995. Spawning Behaviour of Lake Trout: Effects on Colonization Ability. *Journal of Great Lakes Research.* 21(Suppl. 1): 323-329.

Gunn, J.M. and R. Sein. 2000. Effects of Forestry Roads on Reproductive Habitat and Exploitation of Lake Trout (*Salvelinus namaycush*) in Three Experimental lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 57(Suppl. 2):97-104.

Gunn, J.M. and R. Sein. 2004. Effects of Forestry roads on Reproductive Habitat and Exploitation of Lake Trout. In Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment. Gunn, J.M., R.J. Steedman and R.A. Ryder, (eds). Lewis Publishers. CRC Press. 2004. *pp*. 265-278.

Hakanson, L., A. Nilsson and T. Andersson. 1988. Mercury in Fish in Swedish Lakes. *Environmental Pollution*. 49: 145-162.

Hanson, J.M. and R.H. Peters. 1984. Empirical Prediction of Crustacean Zooplankton Biomass and Profundal Macrobenthos Biomass in Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 41:439-445.

Hale, S.S. 1981. Freshwater Habitat Relationships Round Whitefish (*Prosopium cylindraceum*). Alaska Department of Fish and Game, Contract No. 14-16-0009-79-119, Anchorage, Alaska. Reported in Steinhart, G.B., M. Mineau, and C.E. Kraft. 2007. Status and Recovery of Round Whitefish (*Prosopium cylindraceum*) in New York, USA. Final report to State Wildlife Grant T-3-1, NYSDEC, Bureau of Wildlife, Albany NY. Department of Natural Resources. 12 February 2007.

Hall, B.D., V.L. St. Louis and R.A. Bodaly. 2004. The Stimulation of Methylmercury Production by Decomposition of Flooded Birch Leaves and Jack Pine Needles. *Biogeochemistry*. 68: 107-129.

Hall, B.D., V.L. St. Louis, K.R. Rolfhus, R.A. Bodaly, K.G. Beaty, M.J. Paterson, and K.A. Peech Cherewyk. 2005. Impacts of Reservoir Creation on Biogeochemical Cycling of Methyl Mercury and Total Mercury in Boreal Upland Forests. *Ecosystem.* 8: 246-266.

Hammer, U.T., R.C. Haynes, J.M. Haseltine and S.M. Swanson. 1975. The Saline Lakes of Saskatchewan. *Verhandlungen des Internationalen Verein Limnologie.* 19:589-598.

Harriman, R., T.E.H. Allott, R.W. Baterbee, C. Curtis, J. Hall and K. Bull. 1995. Critical Load Maps for UK Freshwaters, in Critical Loads of Acid Deposition for UK Freshwaters, DOE Report, 19.

Harris, R.C. and R.A. Bodaly. 1998. Temperature, Growth and Dietary Effects on Fish Mercury Dynamics in Two Ontario Lakes. *Biogeochemistry* 40:175-187.

Hartmann, J. and W. Nümann. 1977. Percids of Lake Constance, a Lake Undergoing Eutrophication. *Journal of the Fisheries Research Board Canada.* 34:1670-1677.

Hatfield, C.T., J.N. Stein, M.R. Falk and C.S. Jessop. 1972. Fish Resources in the Mackenzie River Valley, Interim Report. Department of the Environment, Fisheries Service, Winnipeg, MB. *Cited in* McPhail, J.D. 2007. The Freshwater Fishes of British Columbia. The University of Alberta Press. Edmonton, AB.

HCI. 2005. Draft Predicted Hydrologic Effects of Developing Gahcho Kué Diamond Project. Prepared for SRK Ltd. HCI-1759. December 2005.

Health Canada. 2006. Summary of Guidelines for Canadian Drinking Water Quality. Prepared by the Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Environmental and Occupational Health.

Health Canada. 2007. Guidelines for Canadian Drinking Water Quality Summary Table. Prepared by the Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Health and the Environment.

Health Canada. 2008. Guidelines for Canadian Drinking Water Quality Summary Table, Prepared by the Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Health and the Environment, May 2008

Hecky, R.E. 1984. Thermal and Optical Characteristics of Southern Indian Lake Before, During and After Impoundment and Churchill River Diversion. *Canadian Journal of Fisheries and Aquatic Sciences*. 41:579-590.

Hecky, R.E. and S.J. Guildford. 1984. Primary Productivity of Southern Indian Lake Before, During, and After Impoundment and Churchill River Diversion. *Canadian Journal of Fisheries and Aquatic Sciences*. 41:591-604.

Hecky, R.E., R.A. Bodaly, D.J. Ramsey and N.E. Strange. 1991. Increased Methylmercury Contamination in Fish in Newly Formed Freshwater Reservoirs. In T.F.W. Clarkson, T. Suzuki, and A. Imura eds. *Advances in Mercury Toxicology*. Plenum Press, New York, NY.

Hecky, R.E., R.A. Bodaly, D.J. Ramsey, P.S. Ramlal and N.E. Strange. 1987. Evolution of Limnological Conditions, Microbial Methylation on Mercury and Mercury Concentrations in Fish in Reservoirs of Northern Manitoba. 1987 Summary Report. Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion.

Hecky, R.E., R.W. Newbury, R.A. Bodaly, K. Patalas, and D.M. Rosenberg. 1984. Environmental Impact Prediction and Assessment: the Southern Indian Lake Experience. *Canadian Journal of Fisheries and Aquatic Sciences*. 41:720-732.

Heginbottom, J.A. and M.A. Dubreuil (Compilers). 1995. National Atlas of Canada: Permafrost. Natural Resources Canada, Map MCR 4177, scale 1:7 500 000.

Henriksen, A. and M. Posch. 2001. Steady-state Models for Calculating Critical Loads of Acidity for Surface Waters. *Water, Air, and Soil Pollution: Focus* 1: 375-398.

Henriksen, A., J. Kämäri, M. Posch and A. Welander. 1992. Critical Loads of Acidity: Nordic Surface Waters. *Ambio*. 21: 356-363.

Henriksen, A., P.J. Dillon, and J. Aherne. 2002. Critical Loads of Acidity for Surface Waters in South-Central Ontario, Canada: Regional Application of the Steady-State Water Chemistry (SSWC) Model. *Canadian Journal of Fisheries and Aquatic Science*. 59:1287-1295.

Hershey, A.E. 1992. Effects of Experimental Fertilization on the Benthic Macroinvertebrate Community of an Arctic Lake. *Journal of the North American Benthological Society*. 11(2):204-217.

Heyes, A., Moore, T.R. and J.W.M. Rudd. 1998. Mercury and Methylmercury in Decomposing Vegetation of a Pristine and Impounded Wetland. *Journal of Environmental Quality*. 27: 591-599.

Holm, J., V. Palace, P. Siwik, G. Sterling, R. Evans, C. Baron, J. Werner and K. Wautier. 2005*. Developmental Effects of Bioaccumulated Selenium in Eggs and Larvae of Two Salmonid Species. Environmental Toxicology and Chemistry.* 24:2373-2381.

Holowaychuk, N. and R.J. Fessenden. 1987. Soil Sensitivity to Acid Deposition and the Potential of Soil and Geology to Reduce the Acidity of Acidic Inputs. Alberta Research Council. Earth Sciences Report 87–1. 38 pp. + Maps. Edmonton, AB.

Hubert, W.A., R.S. Helzner, L.A. Lee, and P.C. Nelson. 1985. Habitat Suitability Index Models and Instream Flow Suitability Curves: Arctic Grayling Riverine Populations. U.S. Fish and Wildlife Service Biological Report. 82(10.110). 34 p.

Hyatt, K.D. and J.G. Stockner. 1985. Responses of Sockeye Salmon (*Oncorhynchus nerka*) to Fertilization of British Columbia Coastal Lakes*. Canadian Journal of Fisheries and Aquatic Sciences.* 42:320-331.

Hynes, T.P. 1990. The Impacts of the Cluff Lake Uranium Mine and Mill Effluents on the Aquatic Environment of Northern Saskatchewan. Master of Science Thesis. University of Saskatchewan, Saskatoon, SK, Canada.

ICES (International Council for the Exploration of the Sea). 1995. Underwater Noise of Research Vessels: Review and Recommendations. Cooperative Research Report No. 209. Copenhagen, Denmark.

Illman, W., and D. Tartakovsky, 2006. Asymptotic Analysis of Cross-Hole Hydraulic Tests in Fractured Granite. *Groundwater*. 44: 555-563

Jackson, T.A. 1988. The Mercury Problem in Recently Formed Reservoirs of Northern Manitoba (Canada): Effects of Impoundment and Other Factors on the Production of Methyl Mercury by Microorganisms. *Canadian Journal of Fisheries and Aquatic Sciences*. 45: 97-121.

Jackson, T.A. 1991. Biological and Environmental Control of Mercury Accumulation by Fish in Lakes and Reservoirs of Northern Manitoba, Canada. *Canadian Journal of Fisheries and Aquatic Sciences.* 48: 2449-2470.

Jackson, D.A., P.R. Peres-Neto and J.D. Olden. 2001. What Controls Who is Where in Freshwater Fish Communities – the Roles of Biotic, Abiotic, and Spatial Factors. *Canadian Journal of Fisheries and Aquatic Sciences*. 58:157-170.

Jacques Whitford (Jacques Whitford Environment Limited). 1998. Water Quality Assessment of Kennady Lake, 1998 Final Report, Report BCV50016. Prepared for Monopros Ltd. Yellowknife, NWT.

Jacques Whitford. 1999a. Results of Water Sampling Program for Kennady Lake July 1999 Survey, Project 50091. Prepared for Monopros Ltd. Yellowknife, NWT.

Jacques Whitford. 1999b. Trip Report #1 and Data Assessment for Kennady Lake Water Quality - 1999 Survey Program. Prepared for Monopros Ltd. Yellowknife, NWT.

Jacques Whitford. 2002a. Baseline Limnology Program (2001), Gahcho Kué (Kennady Lake). Project No. ABC50254. Submitted to De Beers Canada Exploration Inc., Yellowknife, NWT.

Jacques Whitford. 2002b. Data Compilation (1995-2001) and Trends Analysis Gahcho Kué (Kennady Lake). Project No. ABC50310. Submitted to De Beers Canada Exploration Inc., Yellowknife, NWT.

Jacques Whitford. 2003a. Gahcho Kué (Kennady Lake) Limnological Survey of Potentially Affected Bodies of Water (2002) Project# NTY71008. Prepared for De Beers Canada Inc. Yellowknife, NWT.

Jacques Whitford. 2003b. Baseline Limnology Program (2002), Gahcho Kué (Kennady Lake). Project No. NTY71008. Submitted to De Beers Canada Exploration Inc., Yellowknife, NWT.

Jacques Whitford. 2004. Baseline Limnology Program (2003) Gahcho Kué (Kennady Lake), Project No. NTY71037. Prepared for De Beers Canada Inc. Yellowknife, NWT.

Jacques Whitford / EBA Consulting Engineering. 2001. Gahcho Kué (Kennady Lake) Environmental Baseline Investigation (2000), Project # 0701-99-13514. Prepared for De Beers Canada Inc. Yellowknife, NWT.

James, W.F., J.W. Barko, H.L. Eakin and D.R. Helsel. 2001. Changes in Sediment Characteristics Following Drawdown of Big Muskego Lake, Wisconsin. *Archiv für Hydrobiologie*. 151: 459-474 (as cited in McGowan et al. 2005).

Jansson, M., P. Blomqvist, A. Jonsson, and A.K. Bergström. 1996. Nutrient Limitation of Bacterioplankton, Autotrophic and Mixotrophic Phytoplankton, and Heterotrophic Nanoflagellates in Lake in Northern Sweden. *Limnology and Oceanology*. 41:1552-1559 (as cited in Thouvenot et al. 2000).

Jarvinen, A.W. and G.T. Ankley. 1999. Linkage of Effects to Tissue Residues: Development of a Comprehensive Database for Aquatic Organisms Exposed to Inorganic and Organic Chemicals. Pensacola FL: Society of Environmental Toxicology and Chemistry (SETAC) Workshop Proceedings. 364 pp.

Jeffries, D.S., and D.C.L. Lam. 1993. Assessment of the Effect of Acidic Deposition on Canadian Lakes: Determination of Critical Loads for Sulphate Deposition. *Water Science and Technology*. 28:183-187.

Jenkins, A., M. Renshaw, R. Helliwell, C. Sefton, R. Ferrier and P. Swingewood. 1997. Modelling Surface Water Acidification in the UK, Report No. 131, Institute of Hydrology, Oxfordshire.

Jensen, J. W. 1986. Gillnet Selectivity and the Efficiency of Alternative Combinations of Mesh Sizes for some Freshwater Fish. *Journal of Fish Biology*. 28:637-646.

Johannessen, M., A. Lande and S. Rognerud. 1984. Fertilization of 6 Small Mountain Lakes in Telemark, Southern Norway, International Association of Theoretical and Applied Limnology Proceedings. 22:673-678. Cited in Dillon, P.J., B.J. Clark and H.E. Evans. 2004. The Effects of Phosphorus and Nitrogen on Lake Trout (*Salvelinus namaycush*) Production and Habitat. In Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment. Gunn, J.M., R.J. Steedman and R.A. Ryder, (eds). Lewis Publishers. CRC Press. 2004. *pp*. 119-131.

Johnston, N.T., C.J. Perrin, P.A. Slaney and B.R. Ward. 1990. Increased Salmonid Growth by Whole-river Fertilization. *Canadian Journal of Fisheries and Aquatic Sciences.* 47:862-872.

Johnston, N.T., M.D. Stamford, K.I. Ashley and K. Tsumura. 1999. Responses of Rainbow Trout (*Oncorhynchus mykiss*) and Their Prey to Inorganic Fertilization of an Oligotrophic Montane Lake. *Canadian Journal of Fisheries and Aquatic Sciences*. 56:1011-1025.

Johnston, T.A., R.A. Bodaly and J.A. Mathias. 1991. Predicting Fish Mercury Levels from Physical Characteristics of Boreal Reservoirs. *Canadian Journal of Fisheries and Aquatic Sciences* 48:1468-1475.

Jones, N.E., W.M. Tonn, G.J. Scrimgeour and C. Katopodis. 2003. Productive Capacity of an Artificial Stream in the Canadian Arctic: Assessing the Effectiveness of Fish Habitat Compensation. *Canadian Journal of Fisheries and Aquatic Sciences.* 60:849-863.

Jones, N.E. and W.M. Tonn. 2004. Resource Selection Functions for Age-0 Arctic Grayling (*Thymallus arcticus*) and Their Application to Stream Habitat Compensation. *Canadian Journal of Fisheries and Aquatic Sciences.* 61:1736-1746.

Jorgenson, J.K., H.E. Welch and M.F. Curtis. 1992. Response of Amphipoda and Trichoptera to Lake Fertilization in the Canadian Arctic. *Canadian Journal of Fisheries and Aquatic Sciences.* 49: 2354-2362.

Jugnia, L.B., T. Sime-Ngando and J. Devaux. 2007. Relationship between Bacterial and Primary Production in a Newly Filled Reservoir: Temporal Variability Over 2 Consecutive Years. *Ecological Research*. 22: 321-330.

Kadlec, J.A. 1962. Effects of a Drawdown on a Waterfowl Impoundment. *Ecology*. 43: 267-281 (as cited in McGowan et al. 2005).

Kahilainen, K., H. Lehtonen. 2003. Piscivory and Prey Selection of Four Predator Species in a Whitefish Dominated Subarctic Lake. *Journal of Fish Biology.* 63:659-672.

Kalff, J. 2002. Phytoplankton Production in Char Lake, a Natural Polar Lake, and in Meretta Lake, a Polluted Lake, Cornwallis Island, Northwest Territories. *Journal of the Fisheries Research Board Canada.* 31:621-636.

Kämäri, J., M. Forsius and M. Posch. 1992a. Critical Loads of Sulfur and Nitrogen for Lakes II: Regional Extent and Variability in Finland. *Water, Air, Soil Pollution.* 66:77-96.

Kämäri, J.D., S. Jeffries, D.O. Hessen, A. Henriksen, M. Posch and M. Forsius. 1992b. Nitrogen Critical Loads and their Exceedances for Surface Waters. In: Proceedings Workshop on Critical Loads for Nitrogen. P. Grennfelt and E. Thornolof. Lokeberg, Sweden. 161-200 pp.

Kämäri, J., M. Amann, Y.-W Brodin, M.J. Chadwick, A. Henriksen, J.P. Hettelingh, J.C.I. Kuylenstierna, M. Posch and H. Sverdrup. 1992c. The Use of Critical Loads for the Assessment of Future Alternatives to Acidification, *Ambio*. 21, 377.

Kelly, C.A., J.W.M. Rudd, A. Furutani and D.W. Schindler. 1984. Effects of Lake Acidification on Rates of Organic Matter Decomposition in Sediments. *Limnology and Oceanography*. 29:687-694.

Kelly, C.A., J.W.M. Rudd and M.H. Holoka. 2003. Effect of pH on Mercury Uptake by Aquatic Bacterium: Implications for Hg Cycling. *Environmental Science and Technology.* 27: 2941-2946.

Kelly, C.A., V.L. St. Louis, K.J. Scott, B. Dyck, J.W.M. Rudd, R.A. Bodaly, N.R. Roulet, A. Heyes, T.R. Moore, S. Schiff, R. Aravena, B. Warner, R. Harris and G. Edwards. 1997. Increases in Fluxes of Greenhouse Gases and Methyl Mercury Following of an Experimental Reservoir. *Environmental Science and Technology*. 31:1334-1344 (as cited in Paterson et al. 1997).

Kidd, K.A., R.H. Hesslein, R.J.P. Fudge, and K.A. Hallard. 1995. The Influence of Trophic Level as Measured by δ15N on Mercury Concentrations in Freshwater Organisms. *Water, Air, and Soil Poll*. 80:1011-1015.

Knauer, G.W. and A.L. Buikema, Jr. 1984. Rotifer Production in a Small Impoundment. *Verhandlungen Internationale Vereinigung für Theoretische und Angewandte Limnologie*. 22: 1475-1481 (as cited in Pinel-Alloul et al. 1989).

Krueger, S.W. 1981. Freshwater Habitat Relationships – Arctic Grayling (*Thymallus arcticus*). Alaska Department of Fish and Game, Habitat Division, Anchorage, Alaska. 65 p. Cited in: Hubert, W.A., R.S. Helzner, L.A. Lee, and P.C. Nelson. 1985. Habitat Suitability Index Models and Instream Flow Suitability Curves: Arctic Grayling Riverine Populations. U.S. Fish and Wildlife Service Biological Report. 82(10.110). 34 p.

Kuchling, K., D. Chorley and W. Zawadzki. 2000. *Hydrogeological modelling of mining operations at the Diavik Diamonds Project.* In Proceedings of the Sixth International Symposium on Environmental Issues and Waste Management in Energy and Mineral Production, University of Calgary, Calgary AB.

Lasorsa, B. and S. Allen-Gil. 1995. The Methylmercury to Total Mercury Ratio in Selected Marine, Freshwater and Terrestrial Organisms. *Water, Air and Soil Pollution*. 80: 905–913.

LeBrasseur, R.J., C.D. McAllister, W.E. Barraclough, O.D. Kennady, J. Manzer, D. Robinson and K. Stevens. 1978. Enhancement of Sockeye Salmon (*Oncorhynchus nerka*) by Lake Fertilization in Great Central Lake: Summary Report. *Journal of the Fisheries Research Board Canada.* 35: 1580-1596.

Legault, M., J. Benoit and R. Berube. 2004. Impact of Reservoirs. *In:* Boreal Shield Ecosystems: Lake Trout Ecosystems in a Changing Environment. J.M. Gunn, R.J. Steedman, and R.A. Ryder eds. Lewis Publishers, Boca Raton, FL.

Lemly, A.D. 1993. Teratogenic Effects of Selenium in Natural Populations of Freshwater Fish. *Ecotoxicology and Environmental Safety.* 26: 181-204.

Lienesch, P.W., M.E. McDonald, A.E. Hershey, W.J. O'Brien and N.D. Bettez. 2005. Effects of a Whole-lake Experimental Fertilization on Lake Trout in a Small Oligotrophic Arctic Lake. *Hydrobiologia*. 548:51-66.

Lien, L., G.G. Raddum and A. Fjellheim. 1992. Critical Loads for Surface Water: Invertebrates and Fish. Acid Rain Research Report 21. Norwegian Institute for Water Research, Oslo, Norway.

Lindqvist, O., K. Johansson, M. Aastrup, A. Andersson, L. Bringmark, G. Hovsenius, L. Hakanson, A. Iverfeldt and M. Meili. 1991. Mercury in the Swedish Environment. Recent Research on Cause, Consequences, and Corrective Methods. *Water, Air, and Soil Pollution.* 55:1-262.

Lindström, T. 1973. Life in a Lake Reservoir. *Ambio*. 2:145-153.

Louchouarn, P., M. Lucotte, A. Mucci,and P. Pichet. 1993. Geochemistry of Mercury in Two Hydroelectric Reservoirs in Quebec, Canada. *Canadian Journal of Fisheries and Aquatic Sciences.* 50: 269-281.

Low, G. 2002. A Study of the Movements of Arctic grayling, *Thymallus arcticus*, Lake Trout, *Salvelinus namaycush* and Longnose Sucker, *Catostomus colostomies* between Kodiak Lake and Lac de Gras, Northwest Territories, June 10 to September 26, 2001. Manuscript report by BHP Diamonds, Yellowknife, NWT, Canada.

Lucas, A.E. and D.W. Cowell. 1984. Regional Assessment of Sensitivity to Acidic Deposition for Eastern Canada. In: Acid Precipitation Series 7. Bricker, O.P. (ed.), Butterworth. Boston, MA. pp. 113-129.

MacDonald, H.F. and K.J. Boyle. 1997. Effect of a Statewide Sport Fish Consumption Advisory on Open-water Fishing in Maine. *North American Journal of Fisheries Management*. 17: 687-695.

Machniak, K. 1975. The Effects of Hydroelectric Development on the Biology of Northern Fishes (Reproduction and Population Dynamics). IV. Lake Trout *Salvelinus namaycush* (Walbaum) A Literature Review and Bibliography, Fisheries and Marine Service Research and Development, Technical Report 530. Cited in Legault, M., J. Benoit and R. Berube. 2004. Impact of Reservoirs. *In:* Boreal Shield Ecosystems: Lake Trout Ecosystems in a Changing Environment. J.M. Gunn, R.J. Steedman, and R.A. Ryder eds. Lewis Publishers, Boca Raton, FL.

MacLean, N.G., J.M. Gunn, F.J. Hicks, P.E. Ihssen, M. Malhiot, T.E. Mosindy, and W. Wilson. 1990. Environmental and Genetic Factors Affecting the Physiology and Ecology of Lake Trout. Ontario Ministry of Natural Resources. Lake Trout Synthesis. Physiology and Ecology Working Group.

Mailman, M. and R.A. Bodaly. 2004. The Burning Question: Does Burning Before Flooding Lower MeHg and Greenhouse Gas Concentrations? 7th International Conference on Mercury as a Global Pollutant. Ljubljana, Slovenia, June 27-July 2, 2004.

Mann, D. A., P.A. Cott, B.W. Hanna and A.N. Popper. 2007. Hearing in Eight Species of Northern Canadian Freshwater Fishes. *Journal of Fish Biology*. 70:109-120.

Mann, D., P. Cott and B. Horne. 2009. Under-ice Noise Generated from Diamond Exploration on a Canadian Sub-arctic Lake and Potential Impacts on Fishes. *Journal of the Acoustical Society of America*. 126: 2215-2222.

Manny, B.A., D.J. Jude and R.L. Eshenroder. 1989. Field Test of a Bioassay Procedure to Assessing Habitat Quality on Fish Spawning Grounds. *Transactions of the American Fisheries Society.* 118:175-182.

Manny, B.A., T.A. Edsall, J.W. Peck, G.W. Kennedy and A.M. Frank. 1995. Survival of Lake Trout Eggs on Reputed Spawning Grounds in Lakes Huron and Superior: *in situ* Incubation, 1987-1988. *Journal of Great Lakes Research*. 21(Suppl.1): 302-312.

Marsden, J.E., J.M. Casselman, T.A. Edsall, R.F. Elliott, J.D. Fitzsimons, W.H. Horns, B.A. Manny, S.C. McAughey, P.G. Sly and B.L. Swanson. 1995. Lake Trout Spawning Habitat in the Great Lakes – A Review of Current Knowledge. *Journal of Great Lakes Research.* 21(Suppl. 1):487-497.

Marshall, T.R. 1996. A Hierarchical Approach to Assessing Habitat Suitability and Yield Potential of Lake Trout. *Canadian Journal of Fisheries and Aquatic Sciences*. 53(Suppl. 1):332-341

Martin, N.V. and C.H. Olver. 1980. The Lake Charr, *Salvelinus namaycush*. In Charrs: Salmonid Fishes of the Genus *Salvelinus*. Perspectives in Vertebrate Science. Vol. 1. E.K. Balon ed. Dr. W. Junk Publishers, The Hague. pp. 209-277.

Marzolf, G.R. 1990. Reservoirs as Environments for Zooplankton. Pages 195 to 208. In: *Reservoir Limnology: Ecological Perspectives*. Edited by Thornton, K.W., B.L. Kimmel and F.E. Payne. John Wiley & Sons, New York.

Mathias, J. and J. Barica. 1980. Factors Controlling Oxygen Depletion in Ice-covered Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*. 37:185-194.

McAughey, S.C. and J.M. Gunn. 1995. The Behavioural Response of Lake Trout to a Loss of Traditional Spawning Sites. *Journal of Great Lakes Research.* 21(Suppl 1): 375-383.

McCauley, E. and J. Kalff. 1981. Empirical Relationships between Phytoplankton and Zooplankton Biomass in Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 38:458-463.

McDonald, B.G., A.M.H. deBruyn, J.R.F. Elphick, M. Davies, D. Bustard and P.M. Chapman. 2010. Developmental Toxicity of Selenium to Dolly Varden Char (*Salvelinus malma*). *Environmental Toxicology and Chemistry.* 29:2800-2805.

McEachern, L.J., M.G. Kennedy and E. Madsen. 2003. Fish Salvage Activities Related to Diamond Mine Construction in the NWT. Report Prepared for Diavik Diamond Mines Inc., Yellowknife, NWT by Jacques Whitford Environmental Ltd, Yellowknife, NWT. Access on the World Wide Web on April, 29, 2008 at http://www.diavik.ca/PDF/Sudbury2003%20Diavik%20fish%20salvage.pdf

McGowan, S., P.R. Leavitt and R.I. Hall. 2005. A Whole-lake Experiment to Determine the Effects of Winter Droughts on Shallow Lakes. *Ecosystems*. 8:694-708.

McLeay, D.J., A.J. Knox, J.G. Malick, I.K. Birtwell, G. Hartman and G.L. Ennis. 1983. Effects on Arctic Grayling (*Thymallus arcticus*) of Short-term Exposure to Yukon Placer Mining Sediments: Laboratory and Field Studies. Canadian Technical Report of Fisheries and Aquatic Sciences. No. 1711.

McLeay, D.J., G.L. Ennis, I.K. Birtwell and G.F. Hartman. 1984. Effects on Arctic Grayling (*Thymallus arcticus)* of Prolonged Exposure to Yukon Placer Mining Sediment: A Laboratory Study. Canadian Technical Report of Fisheries and Aquatic Sciences. No.1241.

McLeay, D.J., I.K. Birtwell, G.F. Hartman and G.L. Ennis. 1987. Responses of Arctic Grayling (*Thymallus arcticus*) to Acute and Prolonged Exposure to Yukon Placer Mining Sediment. *Canadian Journal of Fisheries and Aquatic Sciences* 44:658-673.

McMurtry, M.J., D.L. Wales, W.A. Scheider, G.L. Beggs and P.E. Dimond. 1989. Relationship of Mercury Concentrations in Lake Trout *(Salvelinus namaycush*) and Smallmouth Bass (*Micropterus dolomieui)* to the Physical and Chemical Characteristics of Ontario Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 46:426-434.

McMurtry, M.J., C.C. Willox and T.C. Smith. 1997. An Overview of Fisheries Management for Lake Simcoe. *Lake and Reservoir Management.* 13: 199-213.

McPhail, J.D. 2007. The Freshwater Fishes of British Columbia. The University of Alberta Press. Edmonton, AB.

McPhail, J.D. and C. C. Lindsey. 1970. Freshwater fishes of Northwestern Canada and Alaska. Fisheries Research Board of Canada Bulletin. 173. 381 p.

McQueen, D.J., J. R. Post, and E.L. Mills. 1986. Trophic Relations in Freshwater Pelagic Ecosystems. *Canadian Journal of Fisheries and Aquatic Sciences*. 43:1571-1581.

MHBL (Miramar Hope Bay Ltd). 2005. Potential Shoreline Erosion Processes at Tail Lake, Doris North Project, Hope Bay, Nunavut, Canada. Prepared for Miramar Hope Bay Ltd. by SRK Consulting (Canada) Inc., October 2005, 32 p. + 1 appendix.

Micheli, F. 1999. Eutrophication, Fisheries, and Consumer-resource Dynamics in Marine Pelagic Ecosystems. *Science.* 285:1396-1398.

Molot, L.A., P.J. Dillon, B.J. Clark and B.P. Neary. 1991. Predicting End-of-summer Oxygen Profiles in Stratified Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*. 49:2363-2372.

Mönkkönen, M. and P. Reunanen. 1999. On Critical Thresholds In Landscape Connectivity: A Management Perspective. *Oikos.* 84:302-305.

Montgomery, S., M. Lucotte and I. Rheault. 2000. Temporal and Spatial Influences of Flooding on Dissolved Mercury in Boreal Reservoirs. *The Science of the Total Environment.* 260:147-157.

Morgan, N.C. 1966. Fertilization Experiments in Scottish Freshwater Lochs. II. Sutherland, 1954. 2. Effects on the Bottom Fauna. Freshwater Salmon Fish. Res. 36:1-19. Department of Agriculture and Fisheries for Scotland, Edinburgh.

Morrison, K.A. and N. Thérien. 1991. Experimental Evaluation of Mercury Release from Flooded Vegetation and Soils. *Water, Air, and Soil Pollution*. 56: 607-619.

Morrow, J.E. 1980. Freshwater Fishes of Alaska. Alaska Northwest Publishing Company, Anchorage, AK. 248 p.

Mount, D.R., D.D. Gulley, J.R. Hockett, T.D. Garrison and J.M. Evans. 1997. Statistical Models to Predict the Toxicity of Major Ions to *Ceriodaphnia dubia*, *Daphnia magna* and *Pimephales promelas* (Fathead Minnows). *Environmental Toxicology and Chemistry*. 16:2009-2019.

Mucci, A., M. Lucotte, S. Montgomery, Y. Plourde, P. Pichet and H.V. Tra. 1995. Mercury Remobilization in a Hydroelectric Reservoir of Northern Quebec, La Grande-2: Results of a Soil Resuspension Experiment. *Canadian Journal of Fisheries and Aquatic Sciences.* 52:2507-2517.

Muscatello, J.R., P.M. Bennett, K.T. Himbeault, A.M. Belknap and D.M Janz. 2006. Larval Deformities Associated with Selenium Accumulation in Northern Pike (*Esox lucius*) Exposed to Metal Mining Effluent. *Environmental Science and Technology.* 40:6506-6512.

MVEIRB (Mackenzie Valley Environmental Impact Review Board) 2006. Reasons for Decision and Report of Environmental Assessment for the De Beers Gahcho Kué Diamond Mine, Kennady Lake, NT. Released June 28, 2006.

Näslund, I., G. Milbrink, L.O. Eriksson and S. Holmgren. 1993. Importance of Habitat Productivity Differences, Competition and Predation for the Migratory Behaviour of Arctic Charr. *Oikos* 66:538-546.

Newbury, R.W. and G.K. McCullogh. 1983. Shoreline Erosion and Restabilization in a Permafrost-affected Impoundment. pp. 918-923. Proceedings, IV International Conference on Permafrost.

Newbury, R.W. and G.K. McCullough. 1984. Shoreline Erosion and Restabilization in the Southern Indian Lake Reservoir. *Canadian Journal of Fisheries and Aquatic Sciences*. 41:558-566.

Newcombe, C.P. and J.O.T Jensen. 1996, Channel Suspended Sediment and Fisheries: A Synthesis for Quantitative Assessment of Risk and Impact. *North American Journal of Fisheries Management.* 16:693-727.

Niemann, W.L. and C.W. Rovey, 2008. A Systematic Field-Based Testing Program of Hydraulic Conductivity and Dispersivity over a Range of Scales. *Hydrogeology Journal*. 17:307-320.

Niemi, G., N. Detenbeck, P. DeVore, D. Taylor, A. Lima and J. Pastor. 1990. Overview of Case Studies on Recovery of Aquatic Systems from Disturbance. *Environmental Management*. 14:571-587.

Nienhuser, A.E. and P. Braches. 1998. Problems and Practical Experiences during Refilling of the Kerspe-Talsperre Under Unfavourable Climatic Conditions. *Water Science and Technology*. 37:145-152.

Nilsson, J. and P. Greenfelt. 1988. Critical Loads for Sulphur and Nitrogen. Nordic Council of Ministers: Copenhagen, Denmark. Cited in Whatmough, S.A., Aherne, J., and Dillon, P.J. 2005. Effect of Declining Lake Base Cation Concentration on Freshwater Critical Load Calculations. *Environmental Science and Technology*. 39:3255-3260.

Northcote, T.G. and D.Y. Atagi. 1997. Ecological Interactions in the Flooded Littoral Zone of Reservoirs: The Importance and Role of Submerged Terrestrial Vegetation with Special Reference to Fish, Fish Habitat and Fisheries in the Nechako Reservoir of British Columbia, Canada. Skeena Fisheries Report SK-111. August 1997. Ministry of Environment, Lands and Parks.

Nümann, W. 1972. The Bodensee: Effects of Exploitation and Eutrophication on the Salmonid Community. *Journal of the Fisheries Research Board Canada*. 29:833-847.

Nünberg, G.K. 1996. Trophic State of Clear and Colored, Soft- and Hardwater Lakes with Special Consideration of Nutrients, Anoxia, Phytoplankton and Fish. *Lake and Reservoir Management* 12: 432–447.

Nursall, J.R. 1952. The Early Development of a Bottom Fauna in a New Power Reservoir in the Rocky Mountains of Alberta. *Canadian Journal of Zoology*. 30:387-409.

O'Brien, W.J. 1990. Perspectives on Fish in Reservoir Limnology. Pages 209 to 225. In: *Reservoir Limnology: Ecological Perspectives*. Edited by Thornton, K.W., B.L. Kimmel, and F.E. Payne. John Wiley & Sons, New York.

Olmsted, L.L. and D.G. Cloutman. 1978. Repopulation after a Fish Kill in Mud Creek, Washington County, Arkansas, Following Pesticide Pollution. *Transactions of the American Fisheries Society.* 103:79-87.

Oremland, R.R. and D.G. Capone. 1988. Use of Specific Inhibitors in Biogeochemistry and Microbial Ecology. *In:* K.C. Marshall (ed.) *Advances in Microbial Ecology*, *v. 10*. Plenum Press, New York.

Paller, M.H. 1997. Recovery of a Reservoir Fish Community from Drawdown Related Impacts. *North American Journal of Fisheries Management*. 17:726-733.

Park, S. and P. Jaffe. 1999. A Numerical Model to Estimate Sediment Oxygen Levels and Demand. *Journal of Environmental Quality*. 28:219-226.

Patalas, K. and A. Salki. 1984. Effects of Impoundment and Diversion on the Crustacean Plankton of Southern Indian Lake. *Canadian Journal of Fisheries and Aquatic Sciences*. 41:605-612.

Paterson, M.J., D. Findlay, K. Beaty, W. Findlay, E.U. Schindler, M. Stainton and G. McCullough. 1997. Changes in the Planktonic Food Web of a New Experimental Reservoir. *Canadian Journal of Fisheries and Aquatic Sciences*. 54:1088-1102.

Paulsson, K. and K. Lundberg. 1991. Treatment of Mercury Contaminated Fish by Selenium Addition. *Water, Air, and Soil Pollution*. 56: 833-841.

Persson, L., S. Diehl, L. Johansson, G. Andersson and S.F. Hamrin. 1991. Shifts in Fish Communities along the Productivity Gradient of Temperate Lakes – Patterns and the Importance of Size-Structured Interactions. *Journal of Fish Biology*. 38:281-293.

Pienitz, R., J.P. Smol, and D.R.S. Lean. 1997a. Physical and Chemical Limnology of 59 Lakes Located Between the Southern Yukon and the Tuktoyaktuk Peninsula, Northwest Territories (Canada). *Canadian Journal of Fisheries and Aquatic Sciences*. 54: 330-346.

Pienitz, R., J.P. Smol and D.R.S. Lean. 1997b. Physical and Chemical Limnology of 24 Lakes Located Between Yellowknife and Contwoyto Lake, Northwest Territories (Canada). *Canadian Journal of Fisheries and Aquatic Sciences*. 54:347-358.

Pinel-Alloul, E., G. Methot and M. Florescu. 1989. Zooplankton Species Dynamics during Impoundment and Stabilization in a Subarctic Reservoir. *Archiv für Hydrobiologie–Beiheft Ergebnisse der Limnologie*. 33:521-537.

Plante, C. and J.A. Downing. 1993. Relationship of Salmonine Production to Lake Trophic Status and Temperature. *Canadian Journal of Fisheries and Aquatic Sciences.* 50:1324-1328.

Plourde, Y., M. Lucotte and P. Pichet. 1997. Contribution of Suspended Particulate Matter and Zooplankton to MeHg Contamination of the Food Chain in Midnorthern Quebec (Canada) Reservoirs. *Canadian Journal of Fisheries and Aquatic Sciences.* 54:821-831.

Ponce, R.A. and N.S. Bloom. 1991. Effect of pH on the Bioaccumulation of Low Level, Dissolved Methyl Mercury by Rainbow Trout (*Oncorhynchus mykiss*). *Water, Air, and Soil Pollution.* 56:631-640.

Posch, M., M. Forsius and J. Kämäri. 1992. Critical Loads of Sulfur and Nitrogen for Lakes I: Model Description and Estimates of Uncertainty. *Water, Air, Soil Pollution.* 66:173-192.

Porvari, P. 1998. Development of Fish Mercury Concentrations in Finnish Reservoirs from 1979-1994. *The Science of the Total Environment.* 213:279-290.

Porvari, P. and M. Verta. 1995. Methylmercury Production in Flooded Soils: A Laboratory Study. *Water, Air, and Soil Pollution.* 80:765-773.

Powell, M.J. and L.M. Carl. 2004. Lake Trout Stocking in Small Lakes: Factors Affecting Success. Chapter 12 *in* Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment (J.M. Gunn, R.J. Steedman, and R.A. Ryder eds.). Integrative Studies in Water Management and Land Development Series. Lewis Publishers. New York. 501 p.

Power, M.E. 1992. Top-down and Bottom-up Forces in Food Webs: Do Plants Have Primacy? *Ecology.* 73:733-746.

Power, M., G.M. Klein, K.R.R.A. Guiguer and M.K.H. Kwan. 2002. Mercury Accumulation in the Fish Community of a Sub-arctic Lake in Relation to Trophic Position and Carbon Sources. *Journal of Applied Ecology*. 39:819-830.

Puznicki, W.S. 1996. An Overview of Lake Water Quality in the Slave Lake Structural Province Area, Northwest Territories. Water Resources Division, Natural Resources and Environmental Directorate. Prepared for the Department of Indian and Northern Affairs Canada. Gatineau, QC.

Rai, R., W. Maher and F. Kirkowa. 2002. Measurement of Inorganic and Methylmercury in Fish Tissues by Enzymatic Hydrolysis and HPLC-ICP-MS. *Journal of Analytical Atomic Spectrometry.* 17:1560 – 1563.

Rasmussen, J.B. and J. Kalff. 1987. Empirical Models for Zoobenthic Biomass in Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 44:990-1001.

Richardson, E.S., J.D. Reist and C.K. Minns. 2001. Life History Characteristics of Freshwater Fishes Occurring in the Northwest Territories and Nunavut, with Major Emphasis on Lake Habitat Requirements. Canadian Manuscript Report of Fisheries and Aquatic Sciences. 2569.

Rihm, B. 1995. Critical Loads of Acidity for Forest Soils and Alpine Lakes: Steady State Mass Balance Method. Published by the Federal Office at Environment. Forests and Landscapes. Berne, Switzerland.

RMCC (Research and Monitoring Committee of Canada). 1990. The 1990 Canadian Long–Range Transport of Air Pollutants and Acid Deposition Report. Part 4: Aquatic Effects. Federal–Provincial Research and Monitoring Committee. 151 pp. Ottawa, ON.

Robinson, C.L.K and W.M. Tonn. 1989. Influence of Environmental Factors and Piscivory in Structuring Fish Assemblages of Small Alberta Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*. 46: 81-89.

Rosenberg D.M. and V.H. Resh (eds.). 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman & Hall, New York, 488 pp.

Rosgen D.L. 1994. A Classification of Rivers. *Catena*. 22:169-199

Rouse, W.R., C.J. Oswald, C. Spence, W.M. Schertzer and P.D. Blanken. 2002. Cold Region Lakes and Landscape Evaporation. In: Proceedings of the 2nd GEWEX Asian Monsoon Experiment (GAME) – Mackenzie GEWEX Study (MAGS) Joint International Workshop, October 8-9, 2001. Institute of Low Temperature Science, Sapporo, Japan. Di Cenzo, P. and L.W. Martz (eds.), p. 37-42.

Rudd, J.W.M. 1995. Sources of Methylmercury to Freshwater Ecosystems: a Review. *Water, Air, and Soil Pollution*. 80:697-713.

Rudolph, B.L., I. Andreller and C.K. Kennedy. 2008. Reproductive Success, Early Life Stage Development, and Survival of Westslope Cutthroat Trout (*Oncorhynchus clarki lewisi*) Exposed to Elevated Selenium in an Area of Active Coal Mining. *Environmental Science and Technology.* 42:3109-3114.

Ryan, P.A. and T.R. Marshall. 1994. A Niche Definition for Lake Trout (*Salvelinus namaycush*) and Its Use to Identify Populations at Risk. *Canadian Journal of Fisheries and Aquatic Sciences.* 51:2513-2519.

Saffran, K.A. and D.O. Trew. 1996. Sensitivity of Alberta Lakes to Acidifying Deposition: An Update of Maps with Emphasis on 109 Northern Lakes. Water Management Division. Alberta Environmental Protection. 70 pp. Edmonton, AB.

Schindler, D.W. 1971. Light, Temperature and Oxygen Regimes of Selected Lakes in the Experimental Lake Area, Northwestern Ontario. *Journal of the Fisheries Research Board Canada*. 28:157-170.

Schindler, D.W. 1974. Eutrophication and Recovery in Experimental Lakes: Implications for Lake Management. *Science*. 184:897-899.

Schindler, D.W., S.E. Bayley, B.R. Parker, K.G. Beaty, D.R. Cruikshank, EJ. Fee, E.U. Schindler and M.P. Stainton. 1996. The Effects of Climatic Warming on the Properties of Boreal Lakes and Streams at the Experimental Lakes Area, Northwestern Ontario. *Limnology and Oceanography.* 41:1004-1017.

Schwartz, A.L. 1985. The Behaviour of Fishes in their Acoustic Environment. *Environmental Biology of Fishes.* 13:3-15.

Scott, K.M. 1978. Effects of Permafrost on Stream Channel Behavior in Arctic Alaska. United States Geological Survey Professional Paper 1068, 19 p.

Scott, W.B and E.J. Crossman. 1973. Freshwater Fishes of Canada. Fisheries Research Board of Canada, Bulletin 184.

Sellers, T. J., B. R. Parker, D. W. Schindler and W. M. Tonn. 1998. Pelagic Distribution of Lake Trout (*Salvelinus namaycush*) in Small Canadian Shield Lakes with Respect to Temperature, Dissolved Oxygen, and Light. *Canadian Journal of Fisheries and Aquatic Sciences.* 55:170-179.

Shortreed, K.S. and J.G. Stockner. 1986. Trophic Status of 19 Subarctic Lakes in the Yukon Territory. *Canadian Journal of Fisheries and Aquatic Sciences.* 43:797-805.

Shuter, B.J. and N.P. Lester. 2004. Climate Change and Sustainable Lake Trout Exploitation: Predictions from a Regional Life History Model. In Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment. Gunn, J.M., R.J. Steedman and R.A. Ryder, (eds). Lewis Publishers. CRC Press. 2004. *pp*. 281-291.

Sly, P.G. 1988. Interstitial Water Quality of Lake Trout Spawning Habitat. *Journal of Great Lakes Research*. 14:301-315.

Smith, M.W. 1969. Changes in Environment and Biota of a Natural Lake After Fertilization. *Journal of the Fisheries Research Board Canada*. 26:3101-3132.

Spens, J. and J.P. Ball. 2008. Salmonid or Nonsalmonid Lakes: Predicting the Fate of Northern Boreal Fish Communities with Hierarchical Filters Relating to a Keystone Piscivore. *Canadian Journal of Fisheries and Aquatic Sciences*. 65:1945-1955.

SRK (SRK Consulting [Canada] Inc.). 2004. Gahcho Kué Diamond Project - Mining Geotechnics. Prepared for De Beers Canada Ltd. November 2004.

St. Louis, V.L., A.D. Partridge, C.A. Kelly and J.W.M. Rudd. 2003. Mineralization Rates of Peat from Eroding Peat Islands. *Biogeochemistry.* 64:97-100.

St. Louis, V.L., J.W.M. Rudd, C.A. Kelly, R.A. Bodaly, M.J. Paterson, K.G. Beaty, R.H. Hesslein, A. Heyes and A.R. Majewski. 2004. The Rise and Fall of Mercury Methylation in an Experimental Reservoir. *Environmental Science and Technology.* 38:1348-1358.

Steedman, R.J., C.J. Allan, R.L. France and R.S. Kushneriuk. 2004. Land, Water, and Human Activity on Boreal watersheds. In Boreal Shield Watersheds: Lake Trout Ecosystems in a Changing Environment. Gunn, J.M., R.J. Steedman and R.A. Ryder, (eds). Lewis Publishers. CRC Press. 2004. *pp*. 59-85.

Steedman, R.J. and R.S. Kushneriuk. 2000. Effects of Experimental Clearcut Logging on Thermal Stratification, Dissolved Oxygen, and Lake Trout (*Salvelinus namaycush*) Habitat Volume in Three Small Boreal Forest Lakes. *Canadian Journal of Fisheries and Aquatic Sciences.* 57(Suppl. 2):82-91.

Stekoll, M.S., W.W. Smoker, I.A. Wang and B.J. Failor. 2003. Final Report for ASTF Grant #98-012, Project: Salmon as a Bioassay Model of Effects of Total Dissolved Solids. Prepared for the Alaska Science and Technology Foundation. Anchorage, AK, USA.

Stewart, D.B. 2001. Possible Impacts on Overwintering Fish of Trucking Granular Materials over Lake and River Ice in the Mackenzie Delta Area. Prepared by Arctic Biological Consultants, Winnipeg, Manitoba for Fisheries Joint Management Committee. Inuvik, Northwest Territories.

Stewart, D.B., N.J. Mochnacz, J.D. Reist, T.J. Carmichael and C.D. Sawatzky. 2007. Fish Life history and Habitat Use in the Northwest Territories: Arctic Grayling (*Thymallus arcticus*). Fisheries and Oceans Canada, Canadian Manuscript Report of Fisheries and Aquatic Sciences. 2797. 55 p.

Stober, I. and K. Bucher. 2007. Hydraulic Properties of the Crystalline Basement. *Hydrogeology Journal.* 15:213-224

Stumm, W. and J.J. Morgan. 1981. Aquatic Chemistry. An Introduction Emphasizing Chemical Equilibria in Natural Waters. (2^{nd} Ed) John Wiley and Sons, New York. 780 pp.

Sullivan, T.J. 2000. Aquatic Effects of Acidic Deposition. CRC Press LLC. Boca Raton, Fl. 373 pp.

Surette, C., M. Lucotte and A. Tremblay. 2003. Mercury Bioaccumulation in Fish: Effects of Intensive Fishing. Abstract from: COMERN congrès 2003, St. Michel des Saints, QC, November.

Tammi, J., M. Appelberg, U. Beier, T. Hesthagen, A. Lappalainen and M. Rask. 2003. Fish Status of Nordic Lakes: Effects of Acidification, Eutrophication and Stocking Activity on Present Fish Species Composition. *Ambio.* 32:98-105.

Thornburgh, K.R. 1986. Burbot Life History and Habitat Requirements: South, Central, Western, and Interior Regions. In Alaska Habitat Management Guide. Life Histories and Habitat Requirements of Fish and Wildlife. B. Burr ed. Alaska Dept. Fish. Game, Division of Habitat. Juneau, AK. pp. 377-388.

Thouvenot, A., D. Debroas, M. Richardot, L.B. Jugnia and J. Dévaux. 2000. A Study of the Changes Between Years in the Structure of Plankton Community in a Newly-flooded Reservoir. *Archiv für Hydrobiologie*. 149:131-152.

Tonn, W. M. and J. J. Magnuson. 1982. Patterns in the Species Composition and Richness of Fish Assemblages in Northern Wisconsin Lakes. *Ecology* 63:1149–1166.

Tonn, W. M. 1990. Climate Change and Fish Communities: A Conceptual Framework. *Transactions of the American Fisheries Society*. 119:337–352.

Tremblay, A. and M. Lucotte. 1997. Accumulation of Total Mercury and Methyl Mercury in Insect Larvae of Hydroelectric Reservoirs. *Canadian Journal of Fisheries and Aquatic Sciences*. 54:832-841.

Tremblay, A., M. Paterson, M. Lucotte, R. Schetagne and R. Verdon. 2004. Intensive Fishing as a Mercury Contamination Tool. *In:* www.unites. uqam.ca/comern.

Turner, M.A. and A.L. Swick. 1983. The English-Wabigoon River System: IV. Interaction Between Mercury and Selenium Accumulated from Waterborne and Dietary Sources by Northern Pike. *Canadian Journal of Fisheries and Aquatic Sciences.* 40:2241-2250.

Tyson, J.D., W.M. Tonn, S. Boss and B.W. Hanna. No date. General Fish-out Protocol for Lakes and Impoundments in the Northwest Territories and Nunavut - Draft. Department of Fisheries and Oceans, Yellowknife NWT. 33 p.

Ullrich, S.M., T.W. Tanton and S.A. Abdrashitova. 2001. Mercury in the Aquatic Environment: a Review of Factors Affecting Methylation. *Critical Reviews in Science and Technology*. 31:241-293.

University of Alberta. 2008. Atlas of Alberta Lakes. Buffalo Lake. http://sunsite.ualberta.ca/Projects/Alberta-Lakes/view/?region=South%20Saskatchewan%20Region&basin=Red%20Deer%20River%20Basin&lake=Buffalo%20Lake&number=99

US EPA (United States Environmental Protection Agency). 1997. Mercury Study Report to Congress. Office of Research and Development, Washington, DC. December, 1997.

US EPA. 2004. Draft Aquatic Life Water Quality Criteria for Selenium. EPA-822-D-04-001. EPA-822-D-04-001. National Technical Information Service, Springfield, VA.

US EPA. 2010. Regional Screening Level Fish Ingestion Supporting Table. May 2010. Available online: http://www.epa.gov/reg3hwmd/risk/human/index.htm.

US FWS (United States Fish and Wildlife Service). 1982. Habitat Suitability Index Models: Northern Pike. Washington, D.C. 48 p.

Van de Meutter, F., R. Stoks and L. De Meester. 2006. Rapid Response of Macroinvertebrates to Drainage Management of Shallow Connected Lakes. *Journal of Applied Ecology*. 43:51-60.

van Everdingen, R. (ed.). 1998. revised May 2005. Multi-language Glossary of Permafrost and Related Ground-ice Terms. Boulder, CO: National Snow and Ice Data Center/World Data Center for Glaciology.

Verdon, R., D. Brouard, C. Demers, R. Lafumiere, M. Laperle, and R. Schetagne. 1991. Mercury Evolution (1978-1988) in Fishes of the La Grande Hydroelectric Complex, Québec, Canada. *Water, Air, and Soil Pollution.* 56: 405-417.

Verta, M. 1990. Changes in Fish Mercury Concentrations in an Intensively Fished Lake. *Canadian Journal of Fisheries and Aquatic Sciences.* 47:1888-1897.

Vollenweider, R.A. 1979. Das Nährstoffbelastungskonzept als Grundlage für den externen Eingriff in den Eutrophierungsprozess stehender Gewässer und Talsperren. Z. Wasser-u. Abwasser-Forschung 12:46-56.

Voshell, J.R. and G.M. Simmons. 1984. Colonization and Succession of Benthic Macroinvertebrates in a New Reservoir. *Hydrobiologica*. 112:27-39.

Watson, S.B., E. McCauley and J.A. Downing. 1997. Patterns in Phytoplankton Taxonomic Composition across Temperate Lakes of Nutrient Status. *Limnology and Oceanography.* 42:487-495.

Weber-Scannell, P.K. and L.K. Duffy. 2007. Effects of Total Dissolved Solids on Aquatic Organisms: A Review of Literature and Recommendation for Salmonid Species. *American Journal of Environmental Sciences.* 3:1-6.

WRS (Western Resource Solutions). 2002. Analysis of the Water Quality of the Steepbank, Firebag and Muskeg Rivers During the Spring Melt (1989-2001). Prepared for: The Wood Buffalo Environmental Association. Fort McMurray, AB.

Welch, H.E. 1974. Metabolic Rates of Arctic Lakes. *Limnology and Oceanography*. 19: 65-73.

Welch, H.E. and M.E. Bergmann. 1985a. Winter Respiration of Lakes at Saqvaqjuac, N.W.T. *Canadian Journal of Fisheries and Aquatic Sciences.* 42:521-528.

Welch, H.E. and M.E. Bergmann. 1985b. Water Circulation in Small Arctic Lakes in Winter. *Canadian Journal of Fisheries and Aquatic Sciences.* 42:506-520.

Welch, H.E., J.K. Jorgenson and M.F. Curtis. 1988. Emergence of Chironomidae (Diptera) in Fertilized and Natural Lakes at Saqvaqjuac, N.W.T. *Canadian Journal of Fisheries and Aquatic Sciences.* 45:731-737.

Welch, H.E., J.A. Legault, and H.J. Kling 1989. Phytoplankton, Nutrients, and Primary Production in Fertilized and Natural Lakes at Saqvaqjuac, N.W.T. *Canadian Journal of Fisheries and Aquatic Sciences*. 46:90-107.

Welsh, H.B. and M.A. Perkins. 1979. Oxygen Deficit Phosphorus Loading Relation in Lakes. *Journal of the Water Pollution Control Federation*. 51:2823-2828.

Wetzel, R.G. 2001. Limnology 3rd Edition. Elsevier Science Academic Press, New York. 1,006 pp.

WHO (World Health Organization). 1994. Updating and Revision of the Air Quality Guidelines for Europe. Report on the WHO Working Group on Ecotoxic Effects. Copenhagen, Denmark . 22 pp.

White, D.M., H.M. Clilverd, A.C. Tidwell, L. Little, M.R. Lilly, M. Chambers and D. Reichardt. 2008. A Tool for Modeling the Winter Oxygen Depletion Rate in Arctic Lakes. *Journal of the American Water Resources Association*. 44:293-304.

Wiener, J.G. and P.J. Shields. 2000. Mercury in the Sudbury River (Massachusetts, U.S.A.): Pollution History and a Synthesis of Recent Research. *Canadian Journal of Fisheries and Aquatic Sciences.* 57:1053-1061.

Wiener, J.G., W.F. Fitzgerald, C.J. Watras and R.G. Rada. 1990. Partitioning and Bioavailability of Mercury in an Experimentally Acidified Wisconsin Lake. *Environmental Toxicology and Chemistry*. 9:909-918.

Wiens, A.P. and D.M. Rosenberg. 1984. Effect of Impoundment and River Diversion on Profundal Macrobenthos of Southern Indian Lake, Manitoba. *Canadian Journal of Fisheries and Aquatic Sciences.* 41:638–648.

Williamson, C.E. and J.W. Reid. 2001. Copepoda. In: J.H. Thorp and A.P. Covich (eds). Ecology and Classification of North American Freshwater Invertebrates. pp. 915-954.

Wismer, D.A. and A.E. Christie. 1987. Temperature Relationships of Great Lake Fishes: A Data Compilation. Great Lakes Fishery Commission Special Publication No. 87-3. 165p.

Wright, D.G. 1982. A Discussion Paper on the Effects of Explosives on Fish and Marine Mammals in the Water of the Northwest Territories. Canadian Technical Report of Fisheries and Aquatic Sciences. No. 1052.

Wright, D.G., and G.E. Hopky. 1998. Guidelines for the Use of Explosives In or Near Canadian Fisheries Waters. Canadian Technical Report of Fisheries and Aquatic Sciences. No. 2107.

Wright, D.R. and R.D. Hamilton. 1982. Releases of Mercury from Sediments: Effects of Mercury Concentration, Low Temperature, and Nutrient Additions. *Canadian Journal of Fisheries and Aquatic Sciences.* 39:1459-1466.

WRS (Western Resource Solutions). 2002. Analysis of the Water Quality of the Steepbank, Firebag and Muskeg Rivers During the Spring Melt (1989-2001). Prepared for Wood Buffalo Environmental Association. Fort McMurray, AB.

Zeman, A.J. 1994. Subaqueous Capping of Very Soft Contaminated Sediments. *Canadian Geotechnical Journal.* 31:570-577.

8.17.1 Personal Communication:

Horne, B. and G. Zhang. 2010. EBA Engineering Consultants Ltd. Personal Communication. Technical Memo to Dan Johnson (Wayne Corso, JDS), John Faithful (Golder Associates Ltd.) and Andrew Williams (De Beers Canada Inc.) on November 26, 2010.

8.18 ACRONYMS AND GLOSSARY

8.18.1 Acronyms and Abbreviations

ANC	acid-neutralizing capacity
ANFO	ammonium nitrate-fuel oil
API	American Petroleum Institute
ARD	acid rock drainage
BCF	bioconcentration factors
BOD	biological oxygen demand
$CaCO_3$	calcium carbonate
CCME	Canadian Council of Ministers of the Environment
CDWQ	Health Canada Guidelines for Canadian Drinking Water Quality
CEB	chronic effects benchmarks
CFU	coliform forming units
CO_2	carbon dioxide
COD	chemical oxygen demand
CP	collection pond
CWQG	Canadian Water Quality Guidelines
De Beers	De Beers Canada Inc.
DFO	Fisheries and Oceans Canada
DO	dissolved oxygen
DOC	dissolved organic carbon
DOM	dissolved organic matter
d/w	dry weight
e.g.	for example
EIS	environmental impact statement
EMS	environmental management system
ET	evapotranspiration
et al.	group of authors
Evap	evaporation
GIS	geographic information system
h	Hour
HEP	Habitat Evaluation Procedure
HSI	Habitat Suitability Index
HU	Habitat Unit
Hwy	Highway
i.e.	that is
ICP/MS	inductively coupled plasma/mass spectrometry

ISQG	Interim Sediment Quality Guidelines
L_N	Lake Number
LSA	Local Study Area
MDL	method detection limit
N	nitrogen
NAD	north American dataum
NAG	non acid-generating
NO_3^-	nitrate
NOEC	no observed effect concentrations
NO_X	oxides of nitrogen
NTU	nephelometric turbidity unit
NWT	Northwest Territories
P	phosphorous
PAG	potentially acid-generating
PAI	potential acid input
PK	processed kimberlite
PKC	processed kimberlite containment
PM	particulate matter
PMR	probable maximum rainfall
Project	Gahcho Kué Project
SE	standard error
SNP	surveillance net work program
SO_2	sulphur dioxide
SO_4^{2-}	sulphate
SOPC	substances of potential concern
SSD	species sensitivity distributions
SSWC	Steady-State Water Chemistry
STP	sewage treatment plant
SW	southwest
SWE	snow water equivalent
TCU	true colour unit
TDS	total dissolved solids
Terms of Reference	Terms of Reference for the Gahcho Kué Environmental Impact Statement
TKN	Total Kjeldahl nitrogen
TOC	total organic carbon
TP	total phosphorous
TPH	total petroleum hydrocarbon
TSP	total suspended particulates
TSS	total suspended solids

U.S. EPA	United States Environmental Protection Agency
U.S. FWS	United States Fish and Wildlife Service
ULC	Underwriters Laboratories Canada
UTM	Universal Transverse Mercator
V	volume
VC	valued components
WCB	Workers Compensation Board
WHU	weighted habitat units
WMP	Water Management Pond
WRD	mine rock drainage
WTP	water treatment plant
w/w	wet weight

8.18.2 Units of Measure

%	percent
~	approximately
<	less than
>	greater than
°	degree
°C	degree Celsius
µE/m^{-2}/min^{-1}	micro-Einsteins per square metre per minute
µg/g	microgram per gram
µg/L	microgram per litre
µg/L/y	microgram per litre per year
µg/m^{2}/s	microgram per square metre per second
µm	micrometre
µS/cm	microSiemens per centimetre
CFU/100 mL	coliform forming units per one hundred millilitres
cm	centimetre
cm/s	centimetre per second
cm^{2}	square centimetre
dB	decibel
g/m^{2}/y	grams per square metre per year
ha	hectares
Hz	Hertz
ind/m^{3}	individuals per cubic metre
keq/ha/y	kiloequivalents per hectare per year

$keqH^+$/ha/y	kiloequivalent hydrogen ions per hectare per year
kg	kilogram
kg N/ha/y	kilograms of nitrogen per hectare per year
km	kilometre
km^2	square kilometre
kPa	kiloPascals
L	litre
L/d	litre per day
L/ha/y	litre per hectare per year
m	metre
m/m	metre per metre
m^3	cubic metre
m^3/d	cubic metre per day
m^3/s	cubic metre per second
m^3/y	cubic metre per year
masl	metres above sea level
mbgl	metres below ground level
mg N/L	milligrams nitrogen per litre
mg P/L	milligrams phosphorus per litre
mg/kg	milligram per kilogram
mg/kg wet wt	milligrams per kilogram wet weight
mg/L	milligrams per litre
mg/L/m	milligram per litre per metre
mg/L/y	milligram per litre per year
MJ/m^2/day	mega joule per square metre per day
mL	millilitre
mm	millimetre
mm/h	millimetre per hour
mm/mo	millimetre per month
Mm^3	million cubic metres
Mm^3/y	million cubic metres per year
mpn/100 mL	most probable number per one hundred millilitres
m/s	metres per second
Mt	million tonnes
PM_{10}	particulate matter with particle diameter nominally smaller than 10 micrometres (µm)
$PM_{2.5}$	particulate matter with particle diameter nominally smaller than 2.5 µm
ppm	parts per million
W/D	width to depth (ratio)

8.18.3 Glossary

Acid Neutralizing Capacity (ANC)	The equivalent capacity of a solution to neutralize strong acids. Acid Neutralizing Capacity can be calculated as the difference between non-marine base cations and strong anions. This is the principal variable used to quantify the acid-base status of surface waters. Acidification is often quantified by decreases in ANC, and susceptibility of surface waters to acidic deposition impacts is often evaluated on the basis of ANC.
Acidification	The decrease of acid neutralizing capacity in water, or base saturation in soil, caused by natural or anthropogenic processes. Acidification is exhibited as the lowering of pH.
Acute	A stimulus severe enough to rapidly induce an effect; in aquatic toxicity tests, an effect observed in 96 hours or less is typically considered acute. When referring to aquatic toxicology or human health, an acute effect is not always measured in terms of lethality.
Alberta Environment (ANEV)	Provincial ministry that looks after the following: establishes policies, legislation, plans, guidelines and standards for environmental management and protection; allocates resources through approvals, dispositions and licenses, and enforces those decisions; ensure water infrastructure and equipment are maintained and operated effectively; and prevents, reduces and mitigates floods, droughts, emergency spills and other pollution-related incidents.
Alevin	A newly-hatched fish in the larval stage, dependent upon a yolk sac for nutrients while their digestive system develops.
Alkalinity	A measure of water's capacity to neutralize an acid. It indicates the presence of carbonates, bicarbonates and hydroxides, and less significantly, borates, silicates, phosphates and organic substances. Alkalinity is expressed as an equivalent of calcium carbonate. Its composition is affected by pH, mineral composition, temperature and ionic strength. However, alkalinity is normally interpreted as a function of carbonates, bicarbonates and hydroxides. The sum of these three components is called total alkalinity.
Anions	A negatively charged ion.
Anoxia	Little to no dissolved oxygen in the water sample. Waters with less than 2 mg/L of dissolved oxygen experience anoxia.
Anthropogenic	Pertaining to the influence of human activities.
Background	An area not influenced by chemicals released from the site under evaluation.
Base Case	The EIA assessment case that includes existing environmental conditions as well as existing and approved projects or activities.
Base Cation	An alkali or alkaline earth metal cation ($Ca2+$, $Mg2+$, $K+$, $Na+$).
Bathymetry	Measurement of the depth of an ocean or large waterbody.
Benthic Invertebrates	Invertebrate organisms living at, in or in association with the bottom (benthic) substrate of lakes, ponds and streams. Examples of benthic invertebrates include some aquatic insect species (such as caddisfly larvae) that spend at least part of their lifestages dwelling on bottom sediments in the waterbody. These organisms play several important roles in the aquatic community. They are involved in the mineralization and recycling of organic matter produced in the water above, or brought in from external sources, and they are important second and third links in the trophic sequence of aquatic communities. Many benthic invertebrates are major food sources for fish.
Biochemical Oxygen Demand (BOD)	An empirical test in which standardized laboratory procedures are used to determine the relative oxygen requirements of wastewaters, effluents and polluted waters.
Bioconcentration	A process where there is a net accumulation of a chemical directly from an

	exposure medium into an organism.
Bog	Sphagnum or forest peat materials formed in an ombrotrophic environment due to the slightly elevated nature of the bog, which tends to disassociate it from the nutrient-rich groundwater or surrounding mineral soils. Characterized by a level, raised or sloping peat surface with hollows and hummocks. Mineral-poor, acidic and peat-forming wetlands that receives water only from precipitation.
Buffering	The capability of a system to accept acids without the pH changing appreciably. The greater amounts of the conjugate acid-base pair, the more resistant they are to a change in pH.
Cations	A positively charged ion.
Chlorophyll *a*	One of the green pigments in plants. It is a photo-sensitive pigment that is essential for the conversion of inorganic carbon (e.g., carbon dioxide) and water into organic carbon (e.g., sugar). The concentration of chlorophyll a in water is an indicator of algal concentration.
Chronic	The development of adverse effects after extended exposure to a given substance. In chronic toxicity tests, the measurement of a chronic effect can be reduced growth, reduced reproduction or other non-lethal effects, in addition to lethality. Chronic should be considered a relative term depending on the life span of the organism.
Conductivity	A measure of the capacity of water to conduct an electrical current. It is the reciprocal of resistance. This measurement provides an estimate of the total concentration of dissolved ions in the water.
Dissolved Organic Carbon (DOC)	The dissolved portion of organic carbon water; made up of humic substances and partly degraded plant and animal materials.
Dissolved Oxygen (DO)	Measurement of the concentration of dissolved (gaseous) oxygen in the water, usually expressed in milligrams per litre (mg/L).
Electrofishing	A 'live' fish capture technique in which negative (anode) and positive (cathode) electrodes are placed in the water and an electrical current is passed between the electrodes. Fish are attracted (galvano-taxis) to the anode and become stunned (galvano-narcosis) by the current, allowing fish to be collected, measured and released.
Epilimnion	A freshwater zone of relatively warm water in which mixing occurs as a result of wind action and convection currents.
Esker	Long, narrow bodies of sand and gravel deposited by a subglacial stream running between ice walls or in an ice tunnel, left behind after melting of the ice of a retreating glacier.
Eutrophic	The nutrient-rich status (amount of nitrogen, phosphorus and potassium) of an ecosystem.
Eutrophication	Excessive growth of algae or other primary producers in a stream, lake or wetlands as a result of large amounts of nutrient ions, especially phosphate or nitrate.
Evaprotranspiration	A measure of the capability of the atmosphere to remove water from a location through the processes of evaporation and water loss from plants (transpiration).
Forage Fish	Small fish that provide food for larger fish (e.g., longnose sucker, fathead minnow).
Glaciofluvial	Sediments or landforms produced by melt waters originating from glaciers or ice sheets. Glaciofluvial deposits commonly contain rounded cobbles arranged in bedded layers.
Glaciolacustrine	Sediments that were deposited in lakes that formed at the edge of glaciers when the glaciers receded. Glaciolacustrine sediments are commonly laminar deposits of fine sand, silt and clay.

Groundwater	That part of the subsurface water that occurs beneath the water table, in soils and geologic formations that are fully saturated.
Hydraulic Gradient	A measure of the force of moving groundwater through soil or rock. It is measured as the rate of change in total head per unit distance of flow in a given direction. Hydraulic gradient is commonly shown as being dimensionless, since its units are metres/metre.
Hydrogeology	The study of the factors that deal with subsurface water (groundwater) and the related geologic aspects of surface water. Groundwater as used here includes all water in the zone of saturation beneath the earth's surface, except water chemically combined in minerals.
Hydrology	The science of waters of the earth, their occurrence, distribution, and circulation; their physical and chemical properties; and their reaction with the environment, including living beings.
Morphology	Morphology or fluvial geomorphology is the term used in the description of closure drainage designs that replicate natural analogues. It describes the process and the structure of natural systems that are to be replicated in constructed drainage channels, including regime relationships for various channel parameters such as width, depth, width/depth ratio, meander wavelength, sinuosity, bed material, gradient and bank slope.
Nitrogen Oxides (NO_X)	A measure of the oxides of nitrogen comprised of nitric oxide (NO) and nitrogen dioxide (NO_2).
Oligotrophic	Trophic state classification for lakes characterized by low productivity and low nutrient inputs (particularly total phosphorus).
Outliers	A data point that falls outside of the statistical distribution defined by the mean and standard deviation.
Peatlands	Areas where there is an accumulation of peat material at least 40 cm thick. These are represented by bog and fen wetlands types.
Pelagic	Inhabiting open water, typically well off the bottom. Sometimes used synonymously with limnetic to describe the open water zone (e.g., large lake environments).
Permafrost	Permanently frozen ground (subsoil). Permafrost areas are divided into more northern areas in which permafrost is continuous, and those more southern areas in which patches of permafrost alternate with unfrozen ground.
pH	The degree of acidity (or alkalinity) of soil or solution. The pH scale is generally presented from 1 (most acidic) to 14 (most alkaline). A difference of one pH unit represents a ten-fold change in hydrogen ion concentration.
Piezometre	A pipe in the ground in which the elevation of water levels can be measured, or a small diameter observation well.
Polygon	The spatial area delineated on a map to define one feature unit (e.g., one type of ecosite phase).
Potential Acid Input	A composite measure of acidification determined from the relative quantities of deposition from background and industrial emissions of sulphur, nitrogen and base cations.
Riparian	Refers to terrain, vegetation or simply a position next to or associated with a stream, floodplain or standing waterbody.
Runoff	The portion of water from rain and snow that flows over land to streams, ponds or other surface waterbodies. It is the portion of water from precipitation that does not infiltrate into the ground, or evaporate.
Sedge	Any plant of the genus Carex, perennial herbs, often growing in dense tufts in marshy places. They have triangular jointless stems, a spiked inflorescence and long grass-like leaves which are usually rough on the margins and midrib. There are several hundred species.

Sediment	Solid material that is transported by, suspended in, or deposited from water. It originates mostly from disintegrated rocks; it also includes chemical and biochemical precipitates and decomposed organic material, such as humus. The quantity, characteristics and cause of the occurrence of sediment in streams are influenced by environmental factors. Some major factors are degree of slope, length of slope soil characteristics, land usage and quantity and intensity of precipitation.
Solar Radiation	The principal portion of the solar spectrum that spans from approximately 300 nanometres (nm) to 4,000 nm in the electromagnetic spectrum. It is measured in W/m^2, which is radiation energy per second per unit area.
Thermokarst	Pock-marked topography in northern regions caused by the collapse of permafrost features.
Total Dissolved Solids	The total concentration of all dissolved compounds solids found in a water sample. See filterable residue.
Total Organic Carbon	Total organic carbon is composed of both dissolved and particulate forms. Total organic carbon is often calculated as the difference between Total Carbon (TC) and Total Inorganic Carbon (TIC). Total organic carbon has a direct relationship with both biochemical and chemical oxygen demands, and varies with the composition of organic matter present in the water. Organic matter in soils, aquatic vegetation and aquatic organisms are major sources of organic carbon.
Total Suspended Particulate (TSP)	A measure of the total particulate matter suspended in the air. This represents all airborne particles with a mean diameter less than 30 μm (microns) in diameter.
Total Suspended Solids (TSS)	The amount of suspended substances in a water sample. Solids, found in wastewater or in a stream, which can be removed by filtration. The origin of suspended matter may be artificial or anthropogenic wastes or natural sources such as silt.
Toxic	A substance, dose or concentration that is harmful to a living organism.
Trophic	Pertaining to part of a food chain, for example, the primary producers are a trophic level just as tertiary consumers are another trophic level.
Wetlands	Wetlands are land where the water table is at, near or above the surface or which is saturated for a long enough period to promote such features as wet-altered soils and water tolerant vegetation. Wetlands include organic wetlands or “peatlands,” and mineral wetlands or mineral soil areas that are influenced by excess water but produce little or no peat.
Young-of-the-year (fish)	Fish at age 0, within the first year after hatching.