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COMMISSION CANADIENNE DE SÛRETÉ NUCLÉAIRE

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Information relative aux radionucléides incluant le radon dans des mines d'uranium au Canada et leurs effets sur l'environnement

27 juin 2002

1. Concentrations of uranium and other radionuclides in groundwater dewatering waters (include information on %U of the ore body)

Key Lake (operating):

In 2001, the Key Lake deposit ore reserves were 80636 tonnes at an average grade of 0.53 % U_3O_8.

1998-2001 Concentration range of radionuclides measured in Dewatering water directed to Gaertner Pit:

Year	^{226}Ra Bq/L	U ug/L
1998	0.13	110.6
1999	0.19	154.3
2000	0.10	314.0
2001	0.14	160.8

1991 -2001 Concentration range of radionuclides measured in Dewatering Discharge to Horsefly Lake:

Year	^{226}Ra Bq/L	U ug/L
1991	0.06	64.5
1992	0.058	66.7
1993	0.063	55.3
1994	0.066	55.1
1995	0.066	61.8
1996	0.055	46.7
1997	0.043	44.9
1998	0.055	45.9
1999	0.054	8.9
2000	0.030	6.0
2001	0.012	3.0

McClean Lake (operating)

Information on the % uranium ore body was not provided

In 1997, dewatering well mean concentrations for uranium ranged from 0.5 – 9.9 ug/L and 0.01 – 0.05 Bq/L for ^{226}Ra.

In 1997, clean dewatering well discharge to Sink Reservoir uranium concentrations ranged from
< 0.5 – 1.1 ug/L. Radium 226 concentrations ranged from 0.005 – 0.04 Bq/L.

In 1997, contaminated dewatering well discharge to Sedimentation Pond uranium concentrations ranged from < 0.5 – 79900 ug/L. Radium 226 concentrations ranged from < 0.005 – 1710 Bq/L.

2. Radon in air measurements near mine sites:
a) Spatial gradient
b) At what distance does it become indistinguishable from background Include information on % U of the ore body and uranium and radium concentrations in tailings. Of particular interest are radon readings near areas where tailings are disposed of on land.

Radon is a gas under ambient environmental conditions and is a member of the uranium decay chain and has a half-life of 3.82 days. With an atomic weight of 222, radon is considerably heavier than normal air and therefore, tends to remain close to the ground surface. In stagnant air, diffusion is the major mechanism for the upward movement of radon. Radon-222 is released during ore handling and milling and through ongoing production of radon by inventories of its radium-226 parent. Radon is also released from the surface of ore stockpiles, areas of ore and waste exposed through mining activities, exposed tailings surfaces and from surfaces of waste rock piles. The radon release from exposed surfaces depends on radium content of the source material. The actual release rate is governed by factors such as pore space, moisture content, and climatic factors. There are no relevant air quality standards for radon.

Key Lake (operating)

In 2001, the Key Lake deposit ore reserves were 80636 tonnes at an average grade of 0.53 % U_3O_8.

Radiological composition of tailings placed in the Dielmann Tailings Management Facility from 1983-2001:

Year	U_3O_8 %	^{226}Ra (Bq/g)
2001	0.061	309
2000	0.108	248
1999	0.066	191
1998	0.044	174
1997	0.072	165
1996	0.038	180
1995	0.061	180
1994	0.080	220
1993	0.053	222
1992	0.041	252
1991	0.040	250
1990	0.036	213
1989	0.021	220

1988	0.024	NA
1987	0.026	NA
1986	0.029	NA
1985	0.082	NA
1984	0.050	NA
1983	0.1	NA

Dielmann Tailings Influent Porewater Quality

Year	**^{226}Ra Bq/L**	**U ug/L**
2001	251.33	79.28
2000	186.56	52.92
1999	87.99	56
1998	67.3	88.33
1997	119.33	37.5
1996	143.67	31.17
1995	155.59	410.33
1994	150.5	117.21
1993	199.15	130
1992	272.27	26.72
1991	269.18	28.09
1990	601.50	< 2

Background radon measurements (13 km north of site) ranged from < 0.0054 – 0.0135 Bq/L.

Range of radon measurements at surface of tailings (1986-2001):
a) North: 0.07 – 2.8622 Bq/L
b) South: 0.07 – 2.41 Bq/L
c) East: 0.1 – 1.3730 Bq/L
d) West: 0.011 – 0.9081 Bq/L

Range of radon measurements at Boundary Stations (1996-2001) located outside of the mine site:
a) Northeast Lake (about 4.2 km from site): < 0.005 – 0.08 Bq/L
b) Black Forest Lake (about 5 km from site): < 0.011 – 0.09 Bq/L
c) Northwest Lake (about 5.6 km from site): < 0.005 – 0.04 Bq/L
d) Kapesin Lake (about 5.6 km from site): < 0.0111 – 0.08 Bq/L
e) Douglas Lake (about 6 km from site): < 0.004 – 0.1 Bq/L

Elliot Lake (decommissioning):

Information on tailings characteristics and ore grade was not readily available.

In the 1980s, prior to remediation, data measured at the un-covered Lacnor TMA showed average radon fluxes of 20.8 pCi m^{-2} s^{-1}. Typical radon monitoring data for the dry Lacnor and Nordic tailings areas were as follows:

	Radon Bq/L
1976 Data	
- Nordic Lacnor upwind	- 0
- Nordic Lacnor Over Tailings	- 0.26 – 0.44
- Nordic Lacnor Downwind	- 0 - 0.22
1986 Data	
- Background Stations	- 0.002 – 0.019
- Over Nordic Tailings	- 0.006 – 0.255
- Average Nordic Tailings	- 0.002 – 0.255
- Former townsite	- 0.038 (mean)

Beaverlodge (decommissioning):

Information on tailings characteristics and ore grade was not readily available.

Background radon values ranged between 45 – 243 Bq/m^3. Ambient radon gas levels were between 37- 74 Bq/m^3. The following is a range of radon measurements from 1985-1998:

Within the Waste or Reclaimed Area:
Fay Waste (about 0.4 km from site): 14.8 – 125.8 Bq/m^3
Fookes Delta (about 3.7 km from site): 11.1 – 140.6 Bq/m^3
Ace Creek (about 1.4 km from site): 51.8 -510.6 Bq/m^3
Marie Delta (about 1.8 km): 25.9 – 240.5 Bq/m^3

Sites near Mining Area:
Northwest of Airport (about 0.75 km from site): 3.7 – 62.9 Bq/m^3
Airport Beacon (about 0.3 km from site): 7.4 – 55.5 Bq/m^3
Eldorado Townsite (about 1.1 km from site): 11.1 – 240.5 Bq/m^3

Regional Outside of Mining Area:
Uranium City (about 7.2 km from site): 3.7 – 62.9 Bq/m^3
Fredette Lake (about 6.9 km from site): 3.7 – 59.2 Bq/m^3
Donaldson Lake (about 5.3 km from site): 3.7 – 173.9 Bq/m^3

Cluff Lake (decommissioning)

The D ore body had an average grade of 3.0 % uranium. The Dominique-Janine extension, the Cluade Pit, and the Dominique-Peter deposit had ore grades ranging from 0.5 – 1.2 %. The following table represents Cluff Lake tailings solids concentrations since the start of operations:

Reference (year)	**U % or ug/g**	**Ra-226 (Bq/g)**
Meneley and Barbour , 1983	0.008 %	26
Amok Ltd. / Cluff Mining, 1986	NA	35 - 100
Amok Ltd. / Cluff Mining, 1987	96 - 190	45 -85
Amok Ltd. 1988	82 - 1080	15 - 450
Amok Ltd. 1989	133 - 1080	6 -390
Meneley 1990	NA	15 – 490
COGEMA, 1998a	613	73
Goulden 1998	127 -159	95 – 100

Mean Values for Tailings Solids Discharged to the Tailings Management Area in 1993-1999:

Year	**U (ug/g)**	**Ra-226 (Bq/g)**
1993	108	103
1994	180	88
1995	142	74
1996	146	75
1997	143	71
1998	131	91
1999	104	69

Background radon levels were not measured at Cluff Lake prior to mining. Regional background radon levels measured in the Wollaston Lake area range from less than the detection limit (3.7 Bq/m^3) to 210 Bq/m^3. Summary statistics on the ambient radon measurements collected during 1994-1998 ranged between 15-160 Bq/m^3. The highest radon concentrations were measured in the vicinity of the tailings areas and mill facility at 40 -150 Bq/m^3. The lowest concentrations were measured to the south and east of the tailings and mill facility (Germaine camp) where typical levels were about 15 Bq/m^3. The higher radon concentrations near the TMA are likely associated with radon emissions from tailings. The levels at the mill are associated with processing ore and radon from nearby stockpiles. Other elevated radon concentrations are associated with radon from underground workings. The following is a range of radon measurements at various sites:

a) 1500 m northwest of Mill: 11. 1 – 114. 7 Bq/m^3
b) 500 m north of Mill: 11.1 – 99.9 Bq/m^3
c) 500 m east of Mill: 11.1 – 199.8 Bq/m^3
d) 500 m south of Mill: 7.4 – 62.9 Bq/m^3
e) 500 m west of Mill: 11.1 – 81.4 Bq/m^3
f) 1.75 km southeast of mill: 7.4 – 59.2 Bq/m^3
g) 3 km southeast of Dominique-Janine-X: 7.4 – 55.5 Bq/m^3
h) 2 km northeast of Dominique-Janine-X: 11.1 – 37 Bq/m^3
i) 1.5 km northwest of Dominique-Janine-X: 7.4 – 62.9 Bq/m^3
j) 1 km northwest of Dominique-Janine-X: 14.8 -62.9 Bq/m^3

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[4] Environmental Assessment Screening Report: Possession and Management of Waste nuclear Substances Situated at Spanish American, Milliken, Lacnor, Nordic/Buckles and Pronto Historical Properties in the Elliot Lake Area, Ontario. May 2001

[5] Cameco. Beaverlodge Mine-Mill Decommissioning Annual Report (Year 14 – Transition Phase). Period July 1, 1998 – June 30, 1999.

[6] Cogema Resources Inc. Cluff Lake Project: Comprehensive Study for Decommissioning

[7] Cogema Resources Inc. Cluff Lake Project: 2000 Status of the Environment Report

[8] Cogema Resources Inc. McClean Lake Project. Annual Report for 1997 and Appendices

2) Radon:

At what distance does radon become indistinquishable from background.

Radon-222 is an inert gas with a half-life of 3.82 days. Rn-222 is produced from the radioactive decay of ^{226}Ra. Because ^{222}Rn is an inert gas it is able to escape during the various stages of mining and milling. Rn-222 comprises most of the radioactivity released from a uranium mining operation. For example, at Key Lake about 1.4 x 10^8 $Bq \cdot s^{-1}$ of ^{22}Rn and 3.2 x 10^3 $Bq \cdot s^{-1}$ of ^{238}U are released. Both monitoring data and atmospheric model calculations demonstrate that Rn concentrations drop to natural background concentrations within a few kilometers of the mines in northern Saskatchewan, because of the relatively large amounts of Rn naturally present in the environment (Garisto et al. 1997). At ore bodies with low concentrations of U, less ^{222}Rn will be released and the distance before Rn concentrations reach natural levels may be less than that recorded for high level U deposits, i.e., less than a few kilometers. The concentration of Rn is predicted to reach background concentrations at about 10 km from the Key Lake site. Concentrations of ^{210}Pb, a daughter radionuclide of ^{22}Rn, in lichen show no trend with distance.

Work by EC and HC (2002) have demonstrated that a small mammal (mouse) exposed to ^{222}Rn emissions at mining sites in northern Saskatchewan, including near a mine ventilation shaft, would not receive a harmful radiation dose.

3) **Doses to small mammals at Key Lake near the surface tailings area.**

Small mammals (voles) living near the surface tailing management area at Key Lake are calculated to receive a radiation dose of 3.8 $mGy \cdot d^{-1}$(1.4 $mGy \cdot a^{-1}$) in comparison to a dose of 0.19 (0.07 $Gy \cdot a^{-1}$ at a reference location (EC and HC 2002)

4) **Doses to Humans**

The concentration of U in fish fillets from 15 lakes in the Elliot Lake area were well below the consumption benchmark of 53.8 $mg \cdot kg^{-1}$ and no different than the concentrations measured in fillets from four reference lakes. Ra-226 concentrations in fish fillets were slightly elevated in four lakes by a factor of ≤ 2 and were well below the bench mark of 194 $Bq \cdot kg^{-1}$. In northern Saskatchewan, the radiation dose to humans (first nation) from eating fish and drinking water at potentially impacted water bodies (0.11 to 0.194 $mSv \cdot a^{-1}$ depending on age group) was similar to that from reference areas and were below the dose criteria of 1 $mSv \cdot a^{-1}$ (CanNorth 2002).

Doses to a large mammal

MacLaren Plansearch (1987) and SENES Consultants Ltd. (1996) have demonstrated that that radiation effects on terrestrial biota are minor.

Atmospheric emissions of Rn from U mines do not cause a significant radiation dose to caribou or to humans (Garisto et al. 1997). This also applies to the radiation dose from other decay chain radionuclides. Most of the estimated radiation dose to biota is from sediment and therefore the radiation dose is higher for organisms associated with sediment. At the Stanleigh Mine, Elliot Lake area, the radiation dose to biota is estimated to range from 0.0024 to 0.06 $mGy \cdot d^{-1}$ with a value of 0.014 $mGy \cdot d^{-1}$ for the moose. At the Beaverlodge satellite mines, a radiation dose of 0.017 $mGy \cdot d^{-1}$ was calculated (Conor

Pacific and SENES 1999). The radiation dose calculated to wildlife at three lakes in the Elliot lake area ranged from 0.004 to 1.06 mGy·d $^{-1}$ which is not harmful (minnow and beak 2001). However, the estimated dose to a mallard and muskrat ranged from 2.4 to 6.25 mg·kg·d^{-1}, which is at level where U toxicity may occur.

5) Data from Port Hope studies (and other sources of data)

Soil to garden vegetable transfer factors

Sheppard et al. (1989) studied the uptake of u by field and garden crops. They reported an overall geometric mean CR of 0.013 for U between soil and plant. This value agrees with an overall mean value of 0.0045 from the literature (Sheppard and Evenden 1988) and with a value of 0.0085 recommended by Baes et al. (1984). Blueberry leaves and spinach had the highest CR (0.018), whereas corn grain had the lowest (0.00036). Concentration ratios tend to be higher in roots and low in fruit. U typically shows an acropetal gradient. CR values in potato peelings were 16 to 37 fold higher than the potato flesh. Overall peels (artichoke, beet, carrot, potato and turnip) had 1.5 to 14 fold higher radionuclide concentrations than flesh samples. In the Port Hope garden experiment, U CRs were 0.02 for leafy crops, 0.003 for root crops and 0.0005 for fruit crops. Barley and wild rice grain had lower CRs than stems. In general, CR values were lowest for organic soil and highest for sand. The Cr for U was dependent on soil U concentration with the CR increasing four fold with a 100 fold increase in soil U concentration. In a Port Hope soil lysimeter study leafy crops, especially lettuce (0.025) had the highest CR followed by root crops (e.g., raddish 0.014, carrots 0.0056 and beets 0.0024). Cucumber fruit had the lowest Cr (0.00086). Sheppard and Evenden (1988) calculated CRs of 0.002 for trees and 0.007 for annual native species growing in fine textured soils.

Since the dominant species of U in soil, $UO^{2+}{}_{2}$, is cationic and since soil solids have a negative charge U is strongly absorbed (Sheppard and Evenden 1988). Organic complexes and colloids can increase the mobility of U in soil Mobility in plants is restricted largely because of absorption on cell wall materials.

6) **Uranium behaviour**

In carbonate rich water, U exists as the uranyl-hydroxyl –carbonate and is poorly sorbed (Higgo 1987). In soft-water shield lakes carbonate concentrations are low or absent, uranyl ion is the dominat form of U and sorption to sediment occurs. In stream valleys, U is largely associated with organic matter as a result of sediment deposition and is associated with the acid-soluble fraction in sequential extractions. In groundwater, U is associated with colloids and suspended solids. In lakes, U tends to remain in the water column slowly partitioning to sediment. Loss of U from water to sediment is affected by sediment type and is in the order organic sediment> clay > sand (Bird and Schwartz 1994). The transfer rate for U from water to sediment has a GM of 0.5 a^{-1} with a GSD of 3, whereas the GM partition coefficient to organic sediment is 410 L·kg^{-1} dry sediment (Bird et al. 1992). Geometric mean CRs (L·kg^{-1} dw) are 1,780 for algae, 1750 for macrophytes, 950 for freshwater invertebrates and 6.2 for freshwater fish (Bird and Schwartz 1996).

U toxicity to aquatic organisms

Uranium does not biomagnify through the food web, probably because of its very low rate of uptake (i.e., <5%) through the gut of most organisms. As a result, the concentration of U in upper trophic levels is often much lower than in the bottom trophic levels. Older fish are more tolerant of U than early life stages. The chemical speciation of uranium, like that of other metals, may be influenced by water quality variables such as alkalinity, hardness, pH and natural organic matter (e.g., Hamelink *et al.*, 1994; Markich *et al.*, 2000). Hardness has a major influence on U toxicity. Elevated hardness (Ca or Mg) ameliorates U toxicity and chemical speciation modeling indicates that UO_2^{2+} and UO_2OH^+ are the U species most responsible for the toxicity of U.

Organisms are much more sensitive to U in soft water than hard water. Tarzwell and Henderson (1960) reported LC_{50} values for fathead minnow (*Pimephales promelas*) of 1.6 mg UAL^{B1} in water with hardness of 20 mgAL^{B1} and an LC_{50} of 77 mgAL^{B1} in 400 mgAL^{B1} hardness for uranyl sulfate.

Pickett *et al.* (1993) derived a chronic threshold effect value of 3 µgAL^{B1} for reproduction in *Ceriodaphnia dubia* in soft water (~ 4 mgAL^{B1} $CaCO_3$). In Collins Creek water (hardness of 27 mgAL^{B1} as $CaCO_3$ and pH 7.3), *C. dubia* had a 48-h LC_{50} of 430 µg UAL^{B1} (Liber and George 2000). In 7-d chronic toxicity tests using Collins Creek water adjusted to a hardness of 60 mgAL^{B1} as $CaCO_3$, the measured threshold effect concentration for reproduction was 19 ∀ 2 µg UAL^{B1}. The threshold effect concentration for reproduction was 218 µg UAL^{B1} at a hardness of 120 mgAL^{B1} $CaCO_3$.

Poston *et al.* (1984) reported a Lowest-Observed-Adverse-Effect Level (LOAEL) of 520 µgAL^{B1} for daphnid reproduction in water with a hardness of 66B73 mgAL^{B1}. Greater sensitivity was reported in a study by Trapp (1986), who reported a 48-h LC_{50} of 220 µgAL^{B1} for *Daphnia pulex* in soft water. Using an acute:chronic ratio of 10, this equates to a chronic toxicity value of 22 µgAL^{B1}.

Franklin *et al.* (2000) reported that the tropical alga *Chlorella* sp. has an 72-h EC_{50} for growth inhibition of 44 Φg≅L^{-1} at pH 6.5 and hardness of 2-4 mg≅L^{-1} as $CaCO_3$, whereas the minimum detectable effect concentration was 13 Φg≅L^{-1}.

EC and HC (in press) recommended an estimated no effect value (ENEV) for plankton for water hardness less that 100 mgAL^{B1} as $CaCO_3$ of 11 µgAL^{B1}. This value is the geometric mean of the ENEVs for *Ceriodaphnia* (Picket *et al.*, 1993; Liber and George, 2000), *Daphnia* (Trapp, 1986) and *Chlorella* (Franklin *et al.*, 2000). For water hardness greater than 100 mgAL^{B1} as $CaCO_3$, the threshold effect value of 218 µgAL^{B1} (Liber and George, 2000) was used for the ENEV.

Benthic invertebrates

There are few data on the toxicity of sediment-bound U to benthic invertebrates. BEAK International Inc. (1998) exposed both juvenile and adult *Hyalella azteca* to U-spiked sediment. Juvenile amphipods were much more sensitive to U than the adults. From their study, an LC_{20} of 15 mg UAkg^{B1} dw sediment and an LC_{50} of 57 mg UAkg^{B1} dw sediment were calculated for juvenile *Hyalella azteca,* whereas the LC_{50} and LC_{20} for the adult were 436 and 116 mg UAkg^{B1} dw respectively. In 10-d spiked sediment U toxicity tests performed by Liber (2001 in preparation) the LC_{50} and LC_{25} for *Chironomus tentans* were 10 551 mg UAkg^{B1} dw and 4849 mg UAkg^{B1} dw, respectively. In the same study effects on growth were observed with IC_{50} and IC_{25} values of 2695 mg UAkg^{B1} dw and 1440 mg UAkg^{B1} dw, respectively. Test results reported by Liber (2001 in preparation) for *Hyalella azteca* may not be reliable, because the NOEC value of 2551 mg UAkg^{B1} dw was higher than the LC_{50} value of 2442 mg UAkg^{B1} dw sediment.

In another unpublished study, Liber *et al.* (2001) assessed the toxicity of contaminated sediments from Horseshoe Pond, Horseshoe Creek and Hidden Bay at the Rabbit Lake uranium mine in comparison to a reference sediment from Tributary Creek. Uranium and molybdenum were the most elevated contaminants in the sediment, although As, Ni and Na were also elevated. Toxicity was assessed on the basis of survival and growth of *H. azteca* and *C. tentans* larvae in 10-d sediment bioassays. Toxicity was greatest in the Horseshoe Pond sediment and this was the only site where effects were consistently statistically different from the control location. Exposure of both species to a sediment U concentration of 512 mgAkg dw resulted in growth inhibition of 48% to 68% relative to the control. Similarly, exposure of both species to a U concentration of 328 mgAkg^{B1} resulted in 35% growth inhibition relative to the control.

The screening level concentration approach is an effects-based approach applicable to benthic organisms and is an estimate of the highest concentration of a contaminant that can be tolerated by a specific proportion of the benthic species; it may capture the combined effects of simultaneous exposure to several contaminants that may be present in an area. This approach was used by the Ontario Ministry of the Environment to derive their sediment quality guidelines for other contaminants (Persaud *et al.*, 1992). The lowest effect level (LEL) concentration for U was 104 ΦgAg^{B1} dw for the weighted average method and the severe effect level (SEL) value was 5874 ΦgAg^{B1} dw (Thompson *et al.*, 2002 in preparation).

From the above toxicity a geometric mean ENEV of105 mg UAkg^{B1} dw was derived. provide an adequate margin of safety for most benthic macroinvertebrate species.

The CTVs and ENEVs for aquatic biota are summarized in Table 1.

7) **Ra and U concentrations and retention in the Island Lake fen**

The mean U concentration in water at the outlet of Island Lake (ISL4000) in 1998 was 107 $\mu g \cdot L^{-1}$, whereas the mean U concentration downstream of the Island Lake fen (ISL5100 – Island Creek at Dolmites) was 1.75 $\mu g \cdot L^{-1}$. This is a 98.4% decrease in the U concentration. This decrease in U concentration as the water flows through the fen is due to a combination of dilution and uptake by both vegetation and sediments. Flows from Bridle Creek and Agnes Lake join Island Creek in the fen. Unfortunately discharge at the Island Lake outlet is not monitored, so it is difficult to calculate the dilution in the fen from hydrological data. However, dilution can be estimated from the change in the aqueous concentration of conservative elements. Ca concentrations below the fen were 23% of those at Island Lake outlet, whereas Cl concentrations were 20.3%, Na concentrations were 19% and K concentrations were 22% of those at Island Lake outlet. On the basis of the mean of these four values, 79% of the decrease in the U concentration is due to dilution. Hence, about 20% of the U is retained in the fen.

In the case of ^{226}Ra, the mean concentration in water was 0.0119 $Bq \cdot L^{-1}$ at Island Lake outlet and 0.0073 $Bq \cdot L^{-1}$ below the fen in Island Creek. This is a 38.7% decrease in the ^{226}Ra concentration, which can be accounted for by dilution alone. Note in this estimate of dilution, both U and ^{226}Ra concentrations in the diluting water were not accounted for. The fact that dilution can more than account for the decrease in ^{226}Ra concentration as the water travels through the fen suggests that the fen may be a source of ^{226}Ra to the water.

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Table 1: Summary of CTVs and ENEVs derived for uranium in aquatic biota.

Species	CTV	Application factor	ENEV
Fish	1.6 mgAL^{B1}	10	160 µgAL^{B1}
Plankton[1] *	11 µgAL^{B1}	1	11 µgAL^{B1}
Plankton[2]	218 µgAL^{B1}	1	218 µgAL^{B1}
Benthic invertebrates	116 mgAkg^{B1} dw 4849 mgAkg^{B1} dw 1440 mgAkg^{B1} dw 104 mgAkg^{B1} dw 512 mgAkg^{B1} dw 328 mgAkg^{B1} dw	10 10 1 1 10 10	105 mgAkg^{B1} dw sediment **

* Plankton includes both zooplankton and phytoplankton (algae); GM of four values.

** Geometric mean value.

1 hardness < 100 mgAL^{B1} as $CaCO_3$

2 hardness > 100 mgAL^{B1} as $CaCO_3$